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Doctoral Dissertation

Potentials to reduce environmental pressures by increasing the circularity of the global waste and wastewater sector: a model-based assessment

submitted by

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Affidavit

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Vienna, September 2021 Adriana Gómez Sanabria

Abstract

The increasing demand for natural resources and the rising quantities of output flows resulting from economic growth are among the main challenges of socioeconomic metabolism. Mitigating these problems requires an intervention in the society-nature interactions at different levels. With the current debate about climate change, circular economy, and sustainability, it is important to evaluate the role of waste and wastewater management systems as relevant sectors to meet the Paris Agreement and to achieve the Sustainable Development Goals. The objective of this doctoral dissertation is to evaluate to what extent the implementation of circular waste and wastewater management systems can contribute to reducing environmental pressures. I use the GAINS model to examine global waste and wastewater systems and flows, and identify the associated environmental burdens threatening the sustainability of cities/regions, and to identify concrete political, technical, and behavioural measures that could contribute to sustainable development. I develop future scenarios mitigation and baseline scenarios to evaluate the environmental co-benefits (e.g., reduction of greenhouse gases, energy generation, recycling) of implementing circular waste and wastewater management systems. This work shows that the challenges that hinder the development of waste management systems include lack of funds and expertise and, poor planning and implementation of law. I also find that the contribution of the waste and wastewater sources to global primary energy demand can increase from the current 2% to 9% by 2040 upon implementation of global circular waste and wastewater management systems. High technical methane abatement potentials of about 80% below the 2050 baseline are feasible from implementing circular waste management systems. Finally, I show that the environmental co-benefits of avoided waste combined with the speedy implementation of anaerobic digestion and the increase of recycling of materials, represented in the 'Sustainability – oriented' scenario, brings major and faster co-benefits in terms of reducing CH_4, CO_2, particulate matter, and air pollutants.

Kurzfassung

Die steigende Nachfrage nach natürlichen Ressourcen und die steigenden Mengen an Outputströmen infolge des Wirtschaftswachstums gehören zu den zentralen Herausforderungen des sozioökonomischen Stoffwechsels. Um diese Probleme zu mildern, ist ein Eingriff in die Gesellschaft-Natur-Interaktionen auf verschiedenen Ebenen erforderlich. Angesichts der aktuellen Debatte über Klimawandel, Kreislaufwirtschaft und Nachhaltigkeit ist es wichtig, die Rolle von Abfall- und Abwassermanagementsystemen als relevante Sektoren zu bewerten, um das Pariser Abkommen zu erfüllen und die Ziele für nachhaltige Entwicklung zu erreichen. Ziel dieser Dissertation ist es zu berechnen, inwieweit die Implementierung kreislauffähiger Abfall- und Abwassermanagementsysteme zur Reduzierung von Umweltbelastungen beitragen kann. Um diese Berechungen durchzuführen verwende ich das GAINS-Modell, um globale Abfall- und Abwassersysteme und -ströme zu untersuchen und die damit verbundenen Umweltbelastungen zu quantifizieren, die die Nachhaltigkeit von Städten / Regionen bedrohen, und um konkrete politische, technische und verhaltensbezogene Maßnahmen zu finden, die zu einer nachhaltigen Entwicklung beitragen können. Ich entwickle Zukunftsszenarien und Basisszenarien, um den ökologischen Zusatznutzen (z. B. Reduzierung von Treibhausgasen, Energieerzeugung, Recycling) der Implementierung von Kreislaufsystemen für Abfall und Abwasser zu berechnen. Diese Arbeit zeigt auf, dass der Mangel an Mitteln und Fachwissen sowie eine schlechte Planung und Umsetzung von Gesetzen die Entwicklung von Abfallwirtschaftssystemen behindern. Meine Arbeit zeigt weiterhin, dass der Beitrag der Abfall- und Abwasserquellen zum weltweiten Primärenergiebedarf von derzeit 2 % auf 9 % bis 2040 steigen kann, wenn weltweite Kreislaufsysteme für Abfall und Abwasser eingeführt werden. Hohe technische Methanvermeidungspotenziale von etwa 80 % unter dem Basisscenario von 2050 sind durch die Implementierung kreislauforientierter Abfallwirtschaftssysteme realisierbar. Schließlich zeige ich, dass der ökologische Zusatznutzen von vermiedenem Abfall in Kombination mit der zügigen Umsetzung der anaeroben Vergärung und der Zunahme des Recyclings von Materialien, dargestellt im Szenario „Nachhaltigkeitsorientiert", einen großen und schnelleren Zusatznutzen in Bezug auf die Reduzierung von CH4, CO2, Feinstaub und Luftschadstoffe mit sich bringt.

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Abbreviations

BC	Black carbon
BOD	Biological Oxygen Demand
CE	Circular Economy
CH_4	Methane
CLE	Current Legislation
CO	Carbon Monoxide
CO_2	Carbon Dioxide
COD	Chemical Oxygen Demand
EF	Emission Factor
EU	European Union
GAINS	Greenhouse Gas- Air pollution Interaction and Synergies model
GDP	Gross Domestic Product
GHG	Greenhouse gases
IAMs	Integrated Assessment Models
INW	Industrial solid waste
IPCC	Intergovernmental Panel on Climate Change
MEFA	Material and Energy Flow Accounting
MFA	Material Flow Analysis
MFR	Maximum technically feasible reduction
MSW	Municipal solid waste
NOx	Nitrogen Oxides
OC	Organic Carbon
$PM_{2.5}$	Particulate Matter less than 2.5 µm in diameter
SDGs	Sustainable Development Goals
SO_2	Sulphur dioxide
SSPs	Shared Socioeconomic Pathways
TCP	Temperature Correction Factor
UN	United Nations
VOCs	Volatile organic compounds

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1. Introduction

Pursuing sustainability requires efforts to alter interactions between society and nature (Kates et al., 2001). Society-nature relationships can be understood as systemic interactions in which intended or unintended interventions in any element will affect the entire socio-ecological system. Uncontrolled human appropriation of natural resources, such as land, material, water and energy, has already caused serious health and environmental impacts all over the world (Haberl et al., 2019). Global warming (Houghton, 2005), air pollution (Kampa and Castanas, 2008), water pollution (Schwarzenbach et al., 2010) and soil contamination are just some examples of environmental impacts at different levels and in different media caused by human activities. These impacts are in many cases trans-boundary due to the atmospheric transport of pollution, greenhouse gases that affect the world climate and international trade (Anenberg et al., 2012; Ramanathan and Feng, 2009; Selomane et al., 2019; Zhang et al., 2017).

The Vienna School of Social Ecology uses a conceptual model to analyse society-nature interactions. This model distinguishes a natural sphere of causation ("nature") and a cultural sphere of causation ("society") that overlap. In physical terms, this overlap contains the "biophysical structures of society" or socioeconomic "material stocks". The interrelation of these two domains can be empirically analysed by investigating social metabolism and society's colonization of natural systems (Fischer-Kowalski and Weisz, 2016). Social metabolism can be characterized by quantifying the interrelated flows of materials and energies associated with the reproduction of the biophysical structures of society. It includes the extraction of natural resources, their transformation through processing and consumption, stocks, and waste and emissions (Fischer-Kowalski and Erb, 2016; Fischer-Kowalski and Haberl, 2002; Haberl et al., 2021). Colonization includes a set of societal activities that purposefully alter natural processes in order to render them more useful to society, e.g. by delivering products, services and contributing to the well-being of society (Fischer-Kowalski and Erb, 2016; Fischer-Kowalski and Haberl, 2002; Haberl et al., 2021).

The coevolution of society and nature is determined by the level of mutual-dynamic interactions during which society intervenes in nature to meet its needs, and nature affects society through physical forces. Those interactions can be evaluated as being positive or negative for society by different stakeholders. Also, cultural processes (acting through communicative interactions) can shape nature which in turn can have effects in society (Fischer-Kowalski and Weisz, 2016). Metabolism and colonization strategies are characteristic of the type of society and population size. Different ideal types of societal organization, i.e., hunter-gatherer, agrarian, or industrial society, are distinguished by their mode of subsistence, including ways of production, technologies, lifestyles and environmental impacts (Fischer-Kowalski and Haberl, 2002). The transition from agrarian to industrial societies is marked by the broad spectrum of resources used such as fossil energy carriers, metals and non-metallic minerals in addition to biomass (Mayer et al., 2016). Agrarian societies rely mostly on solar energy, manpower and biomass for subsistence. In contrast, the energetic basis for industrial societies is mostly derived from fossil fuels (Sieferle, 2001). While in agrarian societies the energy source is renewable, in industrial societies the source of energy is non-renewable (Sieferle, 2001). These different ways of social interaction with nature directly impact resource availability as well as pollution levels.

The Sustainable Development Goals (SDGs) of the 2030 Agenda (United Nations Assembly, 2015) were designed to guide humanity on a path towards tackling major sustainability challenges, including poverty, inequality, biodiversity loss, climate change, poor sanitation, among others. The SDGs integrate social, environmental, and economic agendas. However, there is a large gap in understanding the trade-offs or contradictions when it comes to interlinkages of the different agendas as they are fragmented and largely sectoral (Selomane et al., 2019). Thus, when challenges are looked at in isolation, the success in one area can be detrimental for other areas (Haberl et al., 2019). Social ecology is a transdisciplinary scholarly field that aims to bridge different agendas by investigating society-nature interactions at all spatial and temporal scales (Fischer-Kowalski and Weisz, 2016) .

The continued increase in demand for natural resources and processing to satisfy society's lifestyle has profound impacts on the planet. Over the last 60 years the global resource use grew from 14 Gt/yr in 1950 to 70 Gt/yr in 2010 (Krausmann et al., 2017a) and it is estimated that solid waste generation has increased 15-fold, and that carbon, nitrogen, sulphur and methane emissions have increased 10-fold (Haberl et al., 2019). Environmental consequences of the metabolic transformation, such as depletion of natural resources, waste generation, air and water pollution and GHG emissions, are currently driving the global debate on economic growth and environmental change in which sustainable development plays the main role (Haberl et al., 2017; Steffen et al., 2018). In this regard, socio-metabolic research has a central role as it quantitatively links social, economic and environmental aspects (Haberl et al., 2019). In other words, socio-metabolic research links the problems of acquiring resources sustainably (input side of socioeconomic metabolism) and problems of eliminating the outflows of sociometabolic processes in a sustainable manner.

Economic growth aggravates the problems of socio-metabolic processes as they are directly related to increasing demand for material and energy and therefore associated with rising quantities of outflows. The rapid global economic growth and population increase after the World War II marked a period of massive increase in production and consumption of goods. In that period, many countries focused on the expansion of physical structures e.g., roads, motorways, cities, and on the increase of energy use and material consumption based on the (mis)conception that resources are unlimited. Resource scarcity and environmental impacts were not seen as an issue since 'technology can save us' and even now this view is quite widespread. It was with the oil crises in 1970s that the world became aware of the fact that resources are limited and the importance of avoiding/reducing environmental impacts (Moffatt and Kohler, 2008). One of the most used frameworks to quantify these flows in the society-nature system is the 'material and energy flow accounting'(MEFA) framework. MEFA is widely used to monitor resource use and to inform country governments about the impact of national policies on sustainable resource management. It provides a picture of historical and current turnover of materials and energy by means of the accounting of input and output flows, including undesirable outputs such as waste, wastewater and air pollution, among others (Martinez-Alier and Walter, 2016). MEFA is specially used in the development of economy-trade and technological policies, natural resource management, and environmental policies (OECD, 2008). Various studies have used MEFA as a framework to evaluate the degree of decoupling between materials, energy and pollution from economic growth, and together with other research tools (e.g., input-output analysis) it has provided evidence of the unsustainable metabolism of humanity (Haberl et al., 2019). Although in general MEFA methods focus mostly on ex-post observations, they are becoming an important tool in the modelling framework as a novel way to develop plausible future scenarios of GHGs emissions (Krausmann et al., 2020). The expansion of MEFA with a dynamic modelling approach allows one to systematically trace material and energy from the extraction throughout processing and consumption along with the wastes and emissions. Thus, MEFA can support the Integrated Assessment Models (IAMs)-tools to evaluate scenarios of climate change mitigation-, in the representation of material cycles (Pauliuk et al., 2017).

The circular economy (CE) concept has emerged as a strategy to cope with the high uncontrollable and unsustainable consumption rates of today's society through maximizing resource use efficiency with the aim of contributing to sustainability (Haas Willi et al., 2015; Krausmann et al., 2017b). The CE concept pledges to break the paradigm of make-use-dispose through the implementation of strategies aimed at prolonging the useful life-time of materials, increasing recycling and reducing material intensity, while lowering negative effects on the environment (Haas et al., 2020; Tisserant et al., 2017). However, the effectiveness of the CE as a strategy towards sustainable resource use is still unclear. The MEFA approach is a valuable tool to understand the global level of circularity. Haas Willi et al., 2020, applied this approach to assess the development of circularity during industrialization. He found that the CE could potentially change the socio-metabolic patterns towards sustainability if focused on tackling the growth of stocks, eliminating the production of unsustainable biomass, decarbonization of the energy system and reducing non-circular flows. Haas et al., (2020) also argues that the CE may be by far less beneficial than expected as, first, complete loop-closing is not achievable due to the law of thermodynamics, second, most of the CE strategies focus on waste recycling – downcycling, thus, in many cases the embedded value of the product is destroyed and third, CE is assumed to directly reduce demand for raw materials, which still needs to be proven.

In general, CE seeks to redefine growth while involving a gradual decoupling of economic growth from resource use and environmental degradation (Ellen MacArthur Foundation, 2013). However, whether decoupling is sufficient to achieve sustainable targets is still under discussion (Fischer-Kowalski and Swilling, 2011). Decoupling can be either relative or absolute. Relative decoupling happens when the growth rate of the environmental impact is less than the economic growth rate, while absolute decoupling occurs when the growth rate of the environmental impact decreases and the economic growth rate increases (OECD, 2001). Research on decoupling provides a framework to analyse the success of a policy and to highlight the need for a specific environmental policy (Hoffman and Ventresca, 2002).

In this broader context, the research presented in this thesis focuses on the outputs of socio-metabolic processes, specifically waste and wastewater, and the environmental burdens associated with them. Throughout, I assess the potential environmental co-benefits of implementing circular waste and wastewater management systems using the CE concept as a framework. This means that the reduction of waste generation, re-use, recycling and energy generation are prioritized. The overall objective of the research is to evaluate to what extent the implementation of circular waste and wastewater management systems brings environmental co-benefits. Herein, the social-metabolic approach is applied to, first, examine global waste and wastewater systems and flows, and identify the associated environmental burdens that threaten the sustainability of cities/regions, and, second, to identify concrete potential political, technical, and behavioural measures that could contribute to sustainable development. I use the Material-flow-analysis (MFA) method to examine the flows, i.e., inputs and outputs, in the waste and wastewater management systems. I expand this framework to develop plausible future scenarios to evaluate the environmental co-benefits (e.g., reduction of GHG, energy generation, recycling) of implementing circular waste and wastewater management systems. The future scenarios are then contrasted to their corresponding baselines to assess the impact of the adopted measures.

1.1 Waste and wastewater from the perspective of social metabolism

Scholars developed the concept of metabolism of nations/cities in analogy to the metabolism of living organisms, as all the energy and material flows to ensure the survival of organisms (society) must be considered to understand the system. Nutrients in a living organism are compared to resources in a city and metabolites are waste and pollution generated from the metabolic process in cities (Dinarès, 2014; Zhang, 2013; Fischer-Kowalski and Huttler, 1999). Outflows of the metabolic processes cause adverse effects in the cities as occurs in living organisms; however, unlike organisms, for cities it is much harder to purify/reuse those outputs (Zhang, 2013). The movement from linear to cyclical metabolism was proposed by Girardet, (1990) as a model to represent the ‘real’ influence of cities/urban settings on the

Earth's system (Earth's interacting chemical, physical and biological processes). The influence of societies on the Earth can be observed, for instance, in the transformation from an agrarian to an industrial society, which has massively increased the consumption of materials and energy, thus exacerbating problems such us global warming, air and water pollution and waste generation (Krausmann et al., 2017b; Mayer et al., 2016; Lehmann, 2011).

Although at a global level one can talk about industrial societies, different industrialization stages are seen in different countries. Factors such as population density, economic development, geographical location, labour and trade (Mayer et al., 2016) influence the 'metabolic rates', which refers to changes in material (and monetary) flows per capita (Fischer-Kowalski and Haberl, 2015) of each country (Demaria and Schindler, 2016). Regions in a mature state of industrialization rely more on fossil fuels as an energy source while regions as Africa still rely more on biomass (Fischer-Kowalski et al., 2018). Consequently, not just different quantities of wastes are generated but also the composition and impact of those outputs differ (Zhang, 2013).

Waste and resource management play a central role in the transition to a sustainable CE (Haberl et al., 2019; Velenturf and Purnell, 2017) and, therefore, transformation of practices are central when addressing sustainability issues (Velenturf and Purnell, 2017). Pursuing improvements on the waste and wastewater sector would, in addition to addressing sanitation (SDG 6) and waste reduction and management (SDG 12), add benefits to other Sustainable Development Goals (SDGs) of the 2030 Agenda (United Nations Assembly, 2015), such as good health and wellbeing (SDG 3), affordable and clean energy (SDG 7), sustainable cities and communities (SDG 11), climate action (SDG 13) and life below water (SDG 14).

While changing to sustainable patterns in the input flows i.e., resource use, directly influences output flows such as waste generation, the focus on developing appropriate waste and wastewater management practices is equally important to maximize benefits (Velenturf and Purnell, 2017). Estimates suggest that the world generated 1.9 Gt/yr of MSW in 2015 and is expected to generate about 3.5 Gt/yr MSW in 2050 (Chen et al., 2020). High income countries generate more waste per capita than low income countries and are responsible for 34% of the MSW generated, even though they account for just 16% of the global population (Kaza et al., 2018). The United Nations (2017) estimates that high-income countries treat around 70% of the waste generated and the ratio of treatment decreases in low-income countries. A similar situation is observed in terms of solid waste collection rates in which high-income countries have higher collection rates compared to low-income countries (Gómez-Sanabria et al., 2018; Hoornweg and Bhada-Tata, 2012). Global water demand is also projected to increase by between 20% and 30% per year by 2050 (Boretti and Rosa, 2019). The WaterGAP model estimated for the domestic and manufacturing sectors a global wastewater production of 450 km^3 in 2010, of which 70% was generated for the domestic sector and 30% for the manufacturing sector (Flörke et al., 2013). High income countries treat around 70% of the wastewater while the ratio of treatment declines as income level decreases (United Nations, 2017) – the same situation as with solid waste.

Air and water pollution and greenhouse gases emissions are some of the environmental consequences caused by the lack of proper waste and wastewater management practices. On the one hand, practices such as open burning and dumping of solid waste are potential sources of methane (CH_4), particulate matter (PM), black carbon (BC), organic carbon (OC), carbon dioxide (CO_2), carbon monoxide (CO), sulphur dioxide (SO_2), nitrogen oxides (NO_x), ammonia (NH_3), non-methane volatile organic compounds (NMVOCs), polyhydroxyalkanoates (PHAs) and dioxins (Sharma et al., 2019; Wiedinmyer et al., 2014; Young Koo, 2013). On the other hand, discharges of untreated domestic and industrial wastewater can potentially raise the content of nutrients in water bodies, thereby causing eutrophication (Khan and Mohammad, 2014; Naeem et al., 2014; Singh et al., 2017), and possibly increase the concentration of heavy metals (Akpor et al., 2014). In both cases, the aforementioned compounds cause adverse effects on the environment and human health (Andersson et al., 2016; Anenberg et al., 2012; Das et al., 2018).

Likewise, the lack of suitable waste and wastewater management systems results in high losses of potential secondary materials and energy sources (Gómez-Sanabria et al., 2018). In contrast, if appropriate management systems are in place, fertilizer derived from composting of food waste (Du et al., 2018) or biogas generation through anaerobic digestion (treatment) of organic waste and wastewater can replace fossil fuels for electricity and heat generation or as fuel in the transport sector (Woon et al., 2016). Other examples include the replacement of raw materials through recycling and recovery of solid waste (Huysman et al., 2015) and the recovery of phosphorus and nitrogen from wastewater (Di Iaconi et al., 2010; Yuan et al., 2012). Overall, the reduction of waste and wastewater generation and the improvement of waste and wastewater management systems can contribute to lower raw material extraction, reduce air and water pollution, mitigate greenhouse gases emissions as well as support the transition to the decarbonization of the energy system (Sitra et al. 2018 and Corsten et al. 2013).

Within this thesis, I analyse waste and wastewater flows in cities/regions with different patterns of social metabolism, geographical locations, and socioeconomic setups with the aim of understanding the historical trends and spatial elements in order to propose strategies that could transform management practices and therefore support the move towards sustainability. I adopt the material flow analysis (MFA) tool as a framework to represent and quantify the waste and wastewater flows in different social systems and to evaluate the adverse effects in the environment. The application of the MFA tool allows me to identify dynamics within and between regions, leading me to propose strategies to improve management systems and propose co-beneficial alternatives to reduce the environmental burdens associated with this sector.

1.2 Description of the current situation of waste and wastewater sectors within sustainability

Within the efforts to reach sustainability, the European Union has as a key objective the decoupling of environmental degradation from economic growth[1]. Instruments such as the Waste Framework Directive 2008/98/EC[2] and the Circular Economy Plan called “Closing the loop—a European Union (EU) action plan for the Circular Economy” have been adopted in various Member States to promote a climate neutral and circular economy and the enhancement of waste management strategies following the waste management hierarchy in which waste prevention is the preferred option followed by reuse, recycling, recovery, and disposal (Article 4 Waste Framework Directive 2008/98/EC and more recently the Directive 2018/850 amending Directive 1999/31/EC on the landfill of waste). Binding efforts to reduce waste generation at source are, however, still largely weak. Implemented regulations have focused on limiting the disposal of waste to landfills and promoting waste management operations alternative to landfill (Directive 2000/76/EC and Directive 1999/31/EC). Nevertheless, Member States were required to have national waste prevention programmes in place by the end of 2013. Overall in Europe, countries have improved their waste management strategies but are still facing problems with decoupling waste generation from economic consumption (FhG-IBP, 2014). For the EU, studies have shown some signs of relative decoupling of waste generation from economic growth, but no evidence of absolute decoupling (Montevecchi, 2016; Fischer-Kowalski and Swilling, 2011; Mazzanti and Zoboli, 2009). There is, however, strong evidence of relative decoupling between economic growth and declining volumes of municipal waste disposed of to landfills or incinerated (Mazzanti and Zoboli, 2009 and Mazzanti and Zoboli, 2008). A comprehensive review by Fell et al., (2010) of waste prevention and decoupling at a global level found that, in general, decoupling of economic growth and waste generation appears to be extremely weak, non-existent and highly ambiguous. Japan is currently the only country which has been able to decouple MSW generation from economic growth (Chen et al., 2020) through the implementation of the so-called 3R’s strategy (Reduce, Reuse, Recycle) in the municipal and industrial settings.

[1] http://ec.europa.eu/environment/eussd/escp_en.htm

[2] http://ec.europa.eu/environment/waste/framework/(European Council, 1999; European Parliament and European Council, 2008, 2000)

Although the general focus of CE has been on solid waste, the CE concept can also be applied to the wastewater treatment sector. Wastewater treatment should be looked at in a holistic manner as wastewater offers a huge potential for recovery of resources. In addition to energy generation, wastewater can also provide 'resources' such as bioactive compounds, antimicrobial agents, and natural chemicals (Federici et al., 2009). Furthermore, water recycling and reuse is also important when implementing wastewater treatment systems (Chen et al., 2019). These are relevant options that can reduce water use while reducing environmental impacts. Guerra-Rodríguez et al., (2020) points out some examples of legislation at a global level that focus on wastewater such as the 'Guidelines for water reuse' in the US (US EPA, 2004) with its revised version in 2012, the 'Guidelines for the safe use of wastewater, excreta and greywater' (WHO, 2006), and some European countries such as Spain and Portugal have implemented legislation to regenerate and reuse wastewater.

The implementation of measures such as reducing waste generation, avoiding landfilling, increasing recycling, generating energy from waste and wastewater, or recovering nutrients from wastewater, would then have positive direct effects on present but also future flows. On the contrary, if current waste and wastewater generation and management trends persist, it would further endanger the environment and human health and would hence be unsustainable. The development and enforcement of environmental policies targeting the waste and wastewater sectors can potentially contribute to tackling climate, health, pollution, and other environmental effects caused by poor waste and wastewater management systems (Ghisellini et al., 2016).

Within this thesis, I simulate different policy interventions targeted at the decoupling between societal welfare and resource use based on the circularity of the waste and wastewater management systems. Potential interventions include restrained landfilling of waste, increased material recycling rates, technological improvement, increased anaerobic wastewater treatment and behavioural changes such as reduction of food and plastic waste generation.

1.3 The role of governance in waste and wastewater management

Good governance is indispensable to move the world towards sustainability. Governance plays a critical role in implementing sustainable strategies and therefore reaching the different climate and sustainability objectives (e.g., Nationally Determined Contributions and SDGs). Although good governance is not sufficient for reaching sustainability, it is certainly a necessary condition (Fischer-Kowalski and Weisz, 2016; Kardos, 2012). More concretely, governance—the exercise of authority in the pursuit of one or more policy goals—is critical to making links between pollution and climate issues. It also influences whether agencies at multiple levels enforce regulations. It is finally related to whether this sufficient interagency coordination and networks enable the spread of successful solutions (Hewitt de Alcántara, 1998; Nanda, 2006; Stoker, 2018). Waste and wastewater management is an important component of public health. Yet due to the high costs of the systems, poor planning, lack of technology, low budget allocation and of governance, most of the developing world finds it difficult to cope with the rapid increase in quantities and changes in composition, thus causing detrimental health and environmental effects.

Although most countries around the world have some kind of regulations related to waste and wastewater management, their enforcement is often deficient. In general, high-income countries reach levels of waste and wastewater treatment of about 70%-80% on average while lower-income countries reach a maximum of 30% (Kaza et al., 2018; United Nations Environment Programme, 2015). Casiano Flores et al., (2017) enunciates that in many cases water problems are related to governance issues, however, inequalities regarding access to technologies and lack of knowledge make the implementation of solutions difficult. In general, waste and wastewater governance highly depends on the institutional arrangements in place, and the capability to develop and enforce legislation in a cooperative way with the private, public and also international sectors (Casiano Flores et al., 2017a; Wilson et al., 2015). Thus, governments should not look individually at waste generation but also to the supply chain from product design to production and consumption systems (Singh et al., 2014) as those factors influence the quantities and types of wastes and, therefore, predetermine management practices. Moreover, they might reduce or prevent waste generation, hence making current management/treatment systems obsolete.

Usually waste policies involve waste reduction, total coverage, resource recovery, adequate treatment, inclusion of informal actors, subsidiary principles, cost, among others (Wilson et al., 2015). Although the principles of governance apply for both waste and water management, water governance has a long history due to the fact that it is a main resource for human life in itself and socioeconomic development. Some of the relevant efforts to build a framework look back at the EUROWATER project in the 90's (Correira, 1997) and more recently the 12 OECD principles on water governance in which the main principles are effectiveness, efficiency, trust and engagement (Akhmouch and Correia, 2016).

Examples of governance include the EU adopted measures towards most sustainable MSW. Legislation includes the Waste Framework Directive 2008/98/EC and more recently the Directive 2018/850 amending Directive 1999/31/EC on the landfill of waste, the Circular Economy Action Plan[3] and the European Plastic Directive[4] and the further progressive enforcement by Member States. Moreover, the

[3] https://ec.europa.eu/environment/circular-economy/

[4] https://eur-lex.europa.eu/legal-content/EN/TXT/?qid=1516265440535&uri=COM:2018:28:FIN

European Commission has supported Member States on the implementation of the Waste Framework. Another important component is the interaction between stakeholders and associations at regional, national, and local scale[5]. Another example of successful governance is the achievement of Japan on the reduction of MSW generation and implementation of the so called 3R's strategy (Reduce, Reuse, Recycle) in the municipal and industrial settings. Yokohama is a showcase of this success. With the increase of waste generation and population Yokohama implemented the policy of reducing 30% of waste in 2010 compared to 2001. In addition to financial efforts, the involvement of citizens, industries, and the local government was necessary for the realization of the objectives (Jones, 2015). The strategy included different components such as the introduction of sanctions and quality inspections but also educational awareness and waste separation rules. Wilson et al., (2012) carried out a comparison of solid waste management in 20 cities, including high-, middle- and low-income levels. One of the indicators analysed was the level of governance including: user inclusivity, provider inclusivity, financial sustainability, and institutional coherence. One of the findings of Wilson et al. was that while in the high-income countries citizens receive waste collection services regardless of social status, in low-income countries that is a benefit of higher social status, leaving peri-urban, slums and rural areas out of the service. For provider inclusivity it was difficult to draw any conclusions. Concerning financial sustainability it was established that budget per capita rises sharply with income. Lastly, as is well known, a strong and transparent institutional framework is central in the governance of waste.

Concerning water governance, the lack of capacity at different levels is one of the main obstacles to ensuring safe drinking water and sanitation. Furthermore, as with the waste sector, insufficient funding also hinders the actions to replace and improve water infrastructures. To overcome those challenges integrated approaches at individual, institutional and social level based on systemic approaches are required (OECD, 2015). Lieberherr (2011) employed three governance dimensions to evaluate the performance of water utility in Zurich (Switzerland). The elements assessed include structural elements, regulatory style, and actors. Structural elements refer to horizontal and vertical coordination within the legal, political, and administrative frames. The regulatory refers to the tools used to implement the different measures and the actors refers to the involvement of the different stakeholders (Lieberherr, 2011). She concluded that although the system is efficient and effective and hence thrived, the legitimacy implemented tend to decrease democratic influence.

Based on the relevance of governance for the success (or not) of policies, I integrate this important factor in my modelling framework taking as an example a specific case study. While many models developing scenarios highlight the benefits of sustainable wastewater management on the environment

[5] https://www.municipalwasteeurope.eu/stakeholder-associations

and human health, few integrate the different levels of governance on the evaluation of the success of specific actions (Hourcade and Crassous, 2008). The case study looks at the current wastewater management in the fish processing industry in Indonesia, pinpointing the importance of the integration of actors at different levels in a system thinking context.

2. Objectives

In this dissertation, I aim to quantify the potentials for reducing waste and wastewater generation and related methane emissions. I focus on reductions that can be achieved through the implementation of circular waste and wastewater management systems achieved through the integration of various policy interventions and governance levels. I also analyse the changes on waste composition for different socioeconomic structures.

My research strategy is modelling. Specifically, I further develop the waste and wastewater sector of the IIASA's GAINS (Greenhouse Gas- Air Pollution Interaction and Synergies) model. This global model-based research aims to explore, and to provide scientific evidence of, the benefits but also the limitations of implementing circular waste and wastewater management systems with regards to material recovery (as recycling rates defined as the proportion of waste recycled from the total waste generated), decarbonization (reduction of carbon dioxide emissions) of the energy system, as well as a force to mitigate air and water pollution and limit global warming.

The specific research objectives of this project are:

1. To develop a new methodology in GAINS to project municipal solid waste generation and composition and increase the resolution of the existing GAINS database of waste and wastewater with regard to waste and wastewater generation quantities, composition, and treatment for each of the 184 country/regions in GAINS.
2. To simulate different policy interventions in the waste and wastewater sectors targeted at the decoupling between societal welfare and resource use based on the circular economy. Potential interventions include e.g., restrained landfilling of waste, increased material recycling rates, technological improvement, increased anaerobic wastewater treatment and behavioural changes such as reduction of food and plastic waste generation.
3. To quantify the carbon content and maximum theoretical energy potential that can be generated from the waste and wastewater sectors as supporters of the decarbonization of the energy system, as well as the potential limitations introduced by different waste and wastewater management regimes.

4. To quantify the global maximum theoretical methane and air pollution mitigation potential upon implementation of the different simulated policy interventions in the waste and wastewater management systems.
5. To integrate different governance levels (multi-level, multi-stakeholder governance) to the modelling framework to measure the success of the policies on reducing water pollution and greenhouse gas emissions in the fish processing industry in Indonesia.

The aforementioned objectives will help to identify initiatives that can foster the adoption of CE in the waste and wastewater sector at the global level. The initiatives or grand challenges include the quantification of waste and wastewater generation and composition as an indication of metabolic transitions (e.g., generation and composition of wastes is different in African countries versus European countries), political interventions fostering the implementation of CE, environmental co-benefits of CE and the integration of governance as an important factor for the success or not of the adopted policies.

3. Methods

This section provides an overview of the IIASA-GAINS model with a detailed explanation of how the waste and wastewater sector is further developed to make it suitable for this research but also to increase its scope to represent detailed waste management systems, thus moving from CH_4 quantification to air pollution and water pollution. Furthermore, the new structure also allows one to evaluate political interventions and analyse co-benefits of implementing circular waste and wastewater management systems.

3.1 The GAINS model

This research builds on the existing Greenhouse Gas- Air Pollution Interaction and Synergies – GAINS model. The GAINS model (Figure 1) explores cost-effective strategies to tackle simultaneously air and greenhouse gas emissions, maximizing the benefits at different scales and integrating synergies between policies (Amann, 2009). The GAINS model is a tool that brings scientific development and political processes together (Amann et al., 2011). GAINS integrates socio-economic development, emission control options and costs, atmospheric dispersion and health, environmental and climate impacts for 9 air pollutants ($PM_{2.5\text{-}10\text{–}BC\text{-}OC}$, SO_2, NO_x, VOCs, NH_3, CO) and 6 GHGs (CO_2, CH_4, N_2O, HFCs, PFCs, SF_6) (Amann et al., 2011). In general, the model can be operated in a 'scenario analysis' mode and in an 'optimization' mode. The 'scenario analysis' mode allows for calculation of emissions from sources to impacts while the 'optimization' mode allows one to identify the most cost-effective strategies to reduce emissions to meet specific targets e.g., concentration targets (Amann et al., 2011). The most recent version of GAINS is implemented at a global level, currently differentiating 184 regions with inventories covering the period from 1990 to 2015 and future projections up to 2050 in five-year steps. Economic activities in GAINS include agriculture, energy, transport, industrial processes, domestic /residential and waste and wastewater sectors.

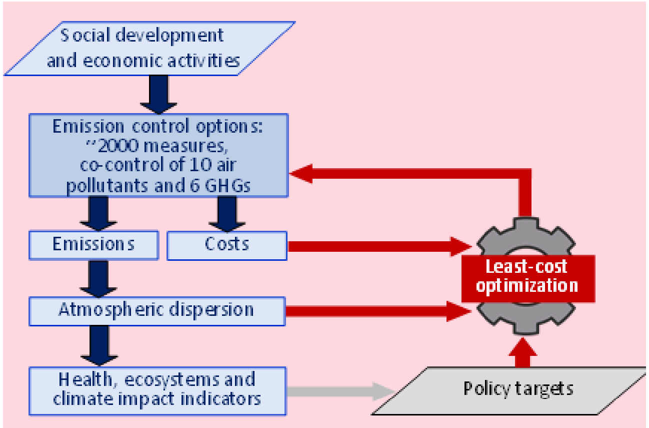

Figure 1. GAINS framework. Source: Amann et al, 2011.

3.2 Waste and wastewater sector in GAINS: What is new?

The following section explains the direct contributions to the further development of the waste and wastewater sector in GAINS. A deeper explanation of the methodology to project MSW and expansion of GAINS can be found in the supplement of **Gómez-Sanabria et al., (2018) – PAPER II.** The updates for the wastewater sector can be found in Höglund-Isaksson et al., (2018) Section 3.4.2.

3.1.1 Solid Waste

My research builds upon the waste and wastewater sector to quantify CH_4 emissions developed in the GAINS model by Höglund-Isaksson and Mechler (2004) and further developed by Höglund-Isaksson (2012). CH_4 emissions estimates were carried out on the biodegradable part of municipal and industrial solid waste and therefore the representation of a complete solid waste management system was not necessary.

Additionally, MSW projections were derived on a single global elasticity of MSW generation growth on GDP per capita and urbanization rate (Höglund-Isaksson, 2012). INW projections are based on INW generation growth on value added by industry (Höglund-Isaksson, 2012). Furthermore, GAINS also included a sector identified as open burning of MSW with the aim of calculating emissions to air derived from default values for developed and developing countries (Klimont et al., 2017).

Based on the importance of the waste sector for reducing GHG and air pollutant emissions and its relevant role in achieving sustainability, it was essential to develop a consistent MSW sector in the

GAINS model. After discussions with colleagues in the AIR program at IIASA, I took the lead to further develop the MSW sector. The development includes a consistent representation of MSW flows by stream using the material flow accounting. The representation of the MSW flows has been carried out at urban and rural areas for each of the 184 regions included in GAINS. This separation is meaningful as it highlights the big gaps between the different socio-economic settings. In line with this, I also developed a model to project MSW generation in which MSW elasticities to GDP per capita are estimated by income-group. Furthermore, the changes in MSW composition are also estimated by income-group based on the elasticity of food waste generation to GDP per capita. This approach enables the tracing of MSW from its generation to its management and to outflows of emissions and also allows one to model measures to improve MSW management aimed at reducing waste generation and pollution. This provides a comprehensive picture to recognize where actions are required for pursuing strategies towards a more sustainable resource use. **Error! Reference source not found.** shows the waste and wastewater sector structure before, and the current structure after the updates I have carried out, which are reflected in Gómez-Sanabria et al., (2018).

Table 1. Comparison between the old and new structure of solid waste in GAINS.

Sector	Item	Old structure (Höglund-Isaksson., 2012)	New structure (Gómez-Sanabria et al., 2018)
Municipal solid waste	Waste composition	Food Paper Plastic Wood Other municipal solid waste	Food Paper Plastic Wood Textile Glass Metal Other municipal solid waste
Industrial solid waste	Waste composition	Food manufacturing Industry Pulp and paper manufacturing industry Textile manufacturing industry Rubber manufacturing industry Other manufacturing industry	No update
Municipal solid waste/Industrial solid waste	Fraction of collected/uncollected waste	Not included	Explicit representation of waste collected and uncollected in urban and rural areas
Municipal solid waste/Industrial solid waste	Type of treatment (includes calculation of emission factors for the new technologies)	Covered solid waste disposal site (covered landfill) Landfill flaring Landfill energy use Anaerobic digestion	Unmanaged solid waste disposal site/dumpsite Unmanaged solid waste disposal site/dumpsite Compacted solid waste disposal site (compacted landfill) Covered solid waste disposal site (covered landfill)

		Household scale composting Large scale composting Incineration of waste with energy recovery Recycling	Landfill flaring Landfill energy use Anaerobic digestion Household scale composting Large scale composting Collected waste open burned Incineration of waste Incineration of waste with energy recovery Recycling Uncollected waste open burned Uncollected scattered waste
Municipal solid waste	Projections	Projections of MSW are carried out on one elasticity estimation of MSW generation to GDP per capita and urbanization rate. No projections for waste composition	Projections of MSW generation and composition are carried out on elasticities of MSW generation per capita to GDP per capita by income group

3.1.2 Wastewater

As with the solid waste sector, the wastewater sector has been developed by Höglund-Isaksson and Mechler (2004) and further improved by Höglund-Isaksson (2012). The updates I have done in this sector are related to the inclusion of a temperature correction factor (TCF) to improve the CH_4 emission estimations from the domestic wastewater sector. The methodology to calculate TCF is described in more detail in Höglund-Isaksson et al. (2018) Section 3.4.2 Wastewater. A shorter explanation is provided below.

Due to the fact that the methanogenic process is sensitive to temperature variations, temperature is an important factor that influences the microbiological community and therefore the degradation process of organic matter in wastewater (Dhaked et al., 2010). In GAINS a country-specific TCF was included to derive the CH_4 emission factors for domestic wastewater. The development of the TCF is derived by weighting the methanogenesis rate at different temperature intervals with the number of days per year in the respective temperature interval, as follows :

$$TCFi = \frac{\Sigma_j^4 \propto jDij}{365} \tag{1}$$

where αj are the rates of methanogenesis (0, 0.1, 0.6, and 0.9) at the four respective temperature intervals ≤ 5ºC, 5 to 15ºC, 15 to 30ºC and > 30ºC, and *Dij* are the average number of days (over years 2000, 2005 and 2010) when the maximum temperature in a country falls within the respective temperature intervals.

Data on the rates of methanogenesis at different temperature intervals are adopted from Lettinga et al., (2001), whilst daily data of the maximum temperature for years 2000, 2005 and 2010 at 25km resolution was taken from the Agri4 Cast Data Portal (JRC, 2015).
As a result, the new CH_4 emission factor is calculated as follows:

$$ef = BOD_i * B_0 * MCF_0 * TCF_i \quad (2)$$

Where, BOD_i is amount of biochemical oxygen demand per person in country i, *B0* is maximum CH4 producing capacity, MCF_0 is the methane correction factor, i.e. the fraction of *BOD* converted to CH4, and TCF_i is the temperature correction factor in country *i*.

3.1.1 Framework to simulate policy interventions.

The GAINS model is a scientific tool that explores cost-effective co-beneficial strategies to tackle air pollution and greenhouse gases at all scales. The GAINS model is recognized as an important tool for policy analyses under e.g., The Convention on Long-range Transboundary Air Pollution (CLRTAP) and, for negotiations under the United Nations Framework Convention on Climate Change (UNFCCC), among others (https://iiasa.ac.at/web/home/research/researchPrograms/air/GAINS.html).

The new waste structure boosts the capacity of the GAINS model, moving it from the computation of pure non-CO_2 greenhouse gases to the exploration of the co-benefits of implementing circular waste and wastewater management systems at global, regional, and local levels. This model architecture allows addressing the impacts of societal development on the environment with regards to waste and wastewater (Figure *2*). Environmental problems cannot be solved by just applying technical measures. As is well known, the integration of social and natural sciences, together with governance approaches offers the best framework to tackle environmental issues (Virapongse et al., 2016).

Within this framework it is possible to incorporate the socio-economic development, simulate behavioural measures such as reduction of MSW generation (SDG 12), and to simulate technical measures to address sanitation (SDG 6 and SDG 12), to reduce GHG emissions (SDG 13), and air and water pollution (SDG 3 and SDG 14).

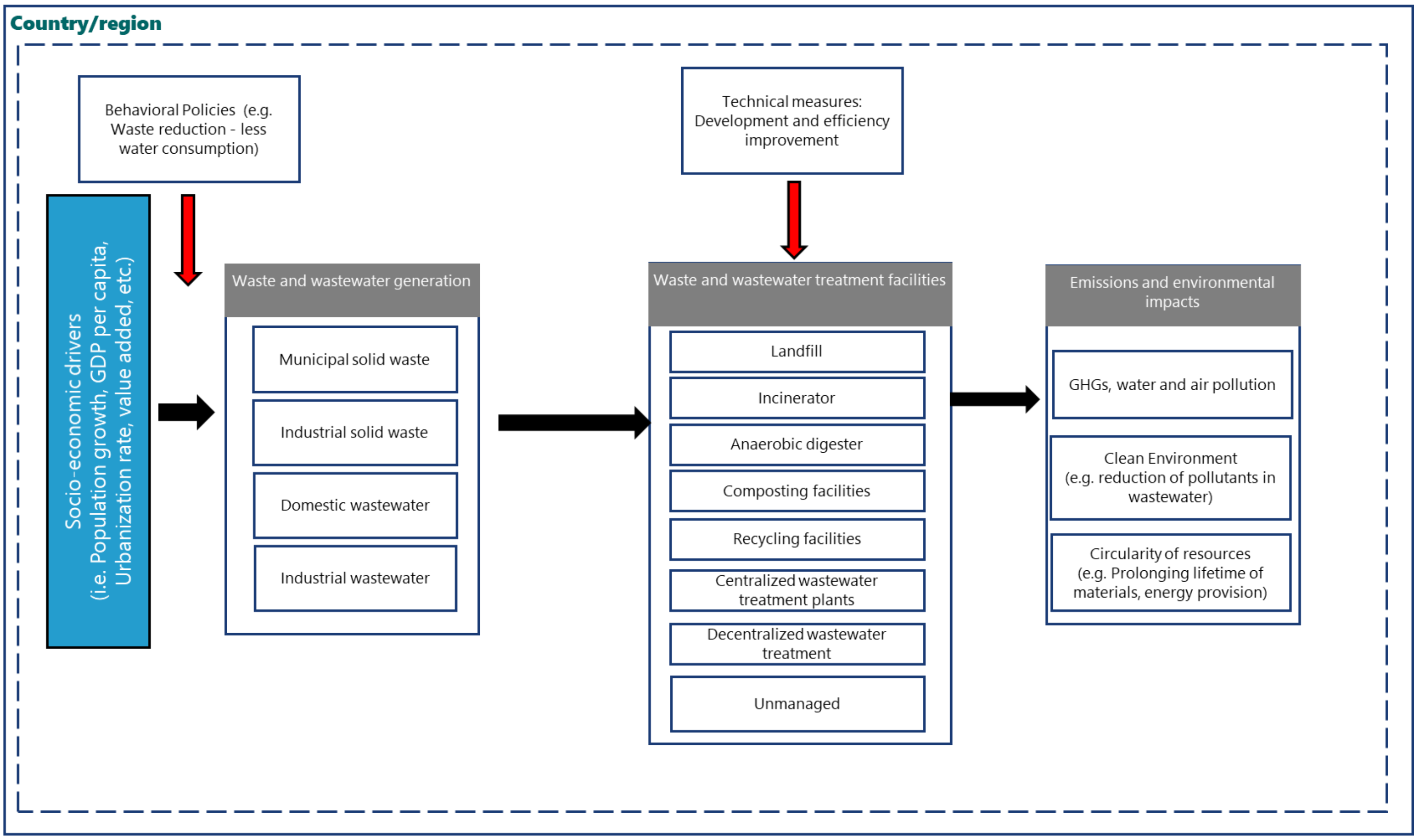

Figure 2. Waste and wastewater framework in GAINS developed by Gómez-Sanabria.

4. Results and contribution of the papers to the objectives of this dissertation

The results presented in this section are in part a summary of the results obtained from the papers plus additional relevant insights achieved during my research that are only partially included in the papers.

4.1 Two level Comparison of Waste Management Systems in Low-, Middle-, and High-income countries (Paper I)

The objective of this review is to investigate the variation of MSW generation rates, composition, and treatment between industrialized and less industrialized cities. Moreover, it also allows comparison of the strategic plans of each city with regards to improvement of the MSW management systems, thus helping me to evaluate the influence of the metabolic stage in waste flows. This review also opens the door to assess the importance of governance on the transformation of socio-metabolic systems. The waste metabolism of the studied cities is carried out using the material flow analysis (MFA) concept. This tool is very often applied to waste management assessments since it shows the need for final sinks and for reuse and recycling measures. It similarly helps to design strategies for recycling and disposal but also highlights the need for the improvement/transformation of product design (Brunner and Rechberger, 2016). The MFAs for current and strategic plans of each city are presented in the supplement of this paper. **Error! Reference source not found.** presents basic characteristics of the selected cities.

Table 2. Profile of selected cities

Country	City	Area (Km^2)	GDP/cap (US$)	Population (million)	Country classification*	Characteristics
India	Ahmedabad	466	2771	6.30	Lower middle income	Major construction/infrastructure boom; Steady population increase; Growing hub of education, information technology and Industry
Mexico	Mexico City	1485	20960	8.90	Upper middle income	Most populated city of the country
Lebanon	Keserwan	336	7750	0.25	Upper middle income	Represents 4.2% of country population and 3.2% of its total area
Colombia	Bogotá	1587	15891	9.70	Upper middle income	Accounts for 25% of the country GDP
Greece	Thessaloniki	19.1	18500	0.32	High income	The second biggest city of the country
Czech Republic	Ostrava	214	17570	0.30	High income	The main industrial centre of the Czech Republic

From this review it can be observed that MSW generation rates tend to increase with income. Ahmedabad has the lowest MSW generation rates but also the lowest GDP per capita while Mexico City has the highest MSW generation rates and the highest GDP per capita. Although Thessaloniki and Ostrava belong to the group of high-income cities, MSW generation rates reach values between those of Bogotá and Lebanon, which are classified as upper middle-income cities. Concerning MSW composition, it can also be noted that the content of organic and inorganic materials changes with income. In general, organics are the main MSW component of the cities with low GDP per capita while its share tends to decrease as income increases and thus inorganic materials become more relevant (Figure 3)

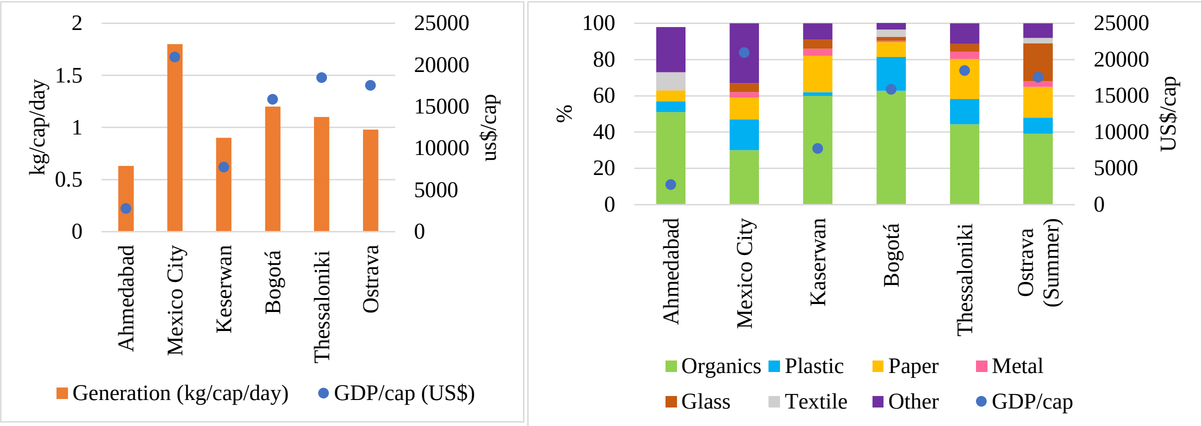

Figure 3.MSW Characteristics. Left panel MSW generation – right panel MSW composition

The generation of MSW can be linked to various factors such as domestic material consumption (DMC), income levels and urbanization levels. Currently, low-income cities, Ahmedabad and Keserwan, dump (controlled) between 85% and 90% of the MSW generated while the other fraction is diverted into composting and material recovery. All the other cities, including Ostrava and Thessaloniki, landfill between (Ostrava) 65% and 90% of the MSW generated. As Ostrava and Thessaloniki belong to EU27 countries, the suggested alternatives included the implementation of the different waste-related EU Directives. For the other cities, it seems that incineration plays a significant role in the MSW management systems, except for Bogotá. Bogotá already had a bad experience with incinerators of MSW and therefore the plan includes the opening of new sanitary landfills. MSW treatment is observed to be coupled with income, hence institutional arrangements, budgets, and infrastructure improve when income rises.

Looking at the institutional and financial arrangements of the metabolism of MSW in the cities one can discern major differences between cities with different levels of income. Higher income cities have a clear structure set in place with well-defined roles and involvement of public and private sectors together with the inclusion of the community. On the contrary, low- and middle-income nations often deal with inefficient institutional structures that lack fundamental policies, adequate budget, and streamlined coordination. Overall, low-income cities (countries) face similar challenges such as lack of funds, poor planning, poor implementation of law and lack of expertise.

For this paper, I developed the strategy for how to pursue this review together with Sophia Ghanimeh. I analysed the general literature and specifically the part dealing with the MSW systems in Bogotá. I was intensively engaged with colleagues in discussing the challenges and alternatives to improve solid waste management systems in all cities included in the study. I drafted the initial version of the paper which was then complemented by the other co-authors. In the revision process, I was responsible for implementing the required corrections and improvements and I responded to the comments of the reviewers.

4.2 Carbon in global waste and wastewater flows – its potential as energy source under alternative future waste management regimes (Paper II)

Based on the insights from Paper I, I proceeded to improve the municipal and industrial waste and wastewater representation in the GAINS model (see Section 3.1.1. and Section 3.1.2). One of the main improvements is the projection of MSW generation and composition according to the income group and urbanization rate. A panel data statistical analysis is used to estimate the elasticities of MSW per capita to GDP per capita and urbanization rate (see Supplement to the paper for the method). The unbalanced panel dataset comprises 684 observations. All variables are specified in logarithmic form in order to provide parameter estimates that can be directly interpreted as elasticity values. In total, elasticities are estimated for five income groups. The results show that the generation and composition of MSW depends on the level of economic development at different stages. While for the low- and middle-income levels elasticity to GDP per capita is found to be between 0.22 and 0.62 without any observed influence of urbanization rate, higher- and upper higher-income levels show elasticities to GDP per capita > 0.80 with a positive influence of urbanization rate. Furthermore, the model is dynamic, meaning that as countries increase GDP per capita, they move to a higher income level and therefore the corresponding elasticities are applied. Another important aspect is the projection of MSW composition based on the elasticity of food waste to GDP per capita. This allows one to represent the impact of economic development on the generation of different waste materials. Furthermore, projections for INW are carried out using the elasticities to value added by manufacturing industry estimated by (Höglund-Isaksson, 2012).

To assess current and expected future carbon flows it is necessary to quantify in the first place waste and wastewater generation and composition. Regarding MSW generation rates, North America is expected to continue generating the highest amount of MSW per capita followed by India. Western Europe is projected to continue increasing MSW generation rates. Africa will increase the generation of MSW per capita but will also be the region with the lowest generation rates by 2050 (Figure 4). In the timeframe studied there is no indication that global MSW generation will reach a peak before 2050.

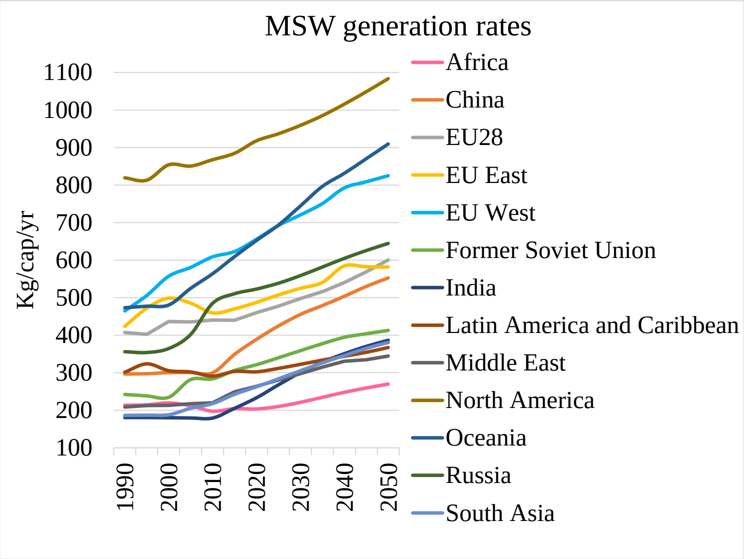

Figure 4. MSW generation rates by region

Figure 5**Error! Reference source not found.** shows the projected composition of MSW in 2050. This is a major aspect as the fractions of MSW determine the type of carbon content (degradable organic carbon or fossil carbon) and therefore the applicable treatment. In regions such as Africa, China, Latin America and The Caribbean and South Asia, food waste accounts for more than 50% of the MSW generated while in most industrialized regions the food fraction accounts for between 20% and 34% of MSW. In the latter regions, paper and plastic are significant fractions.

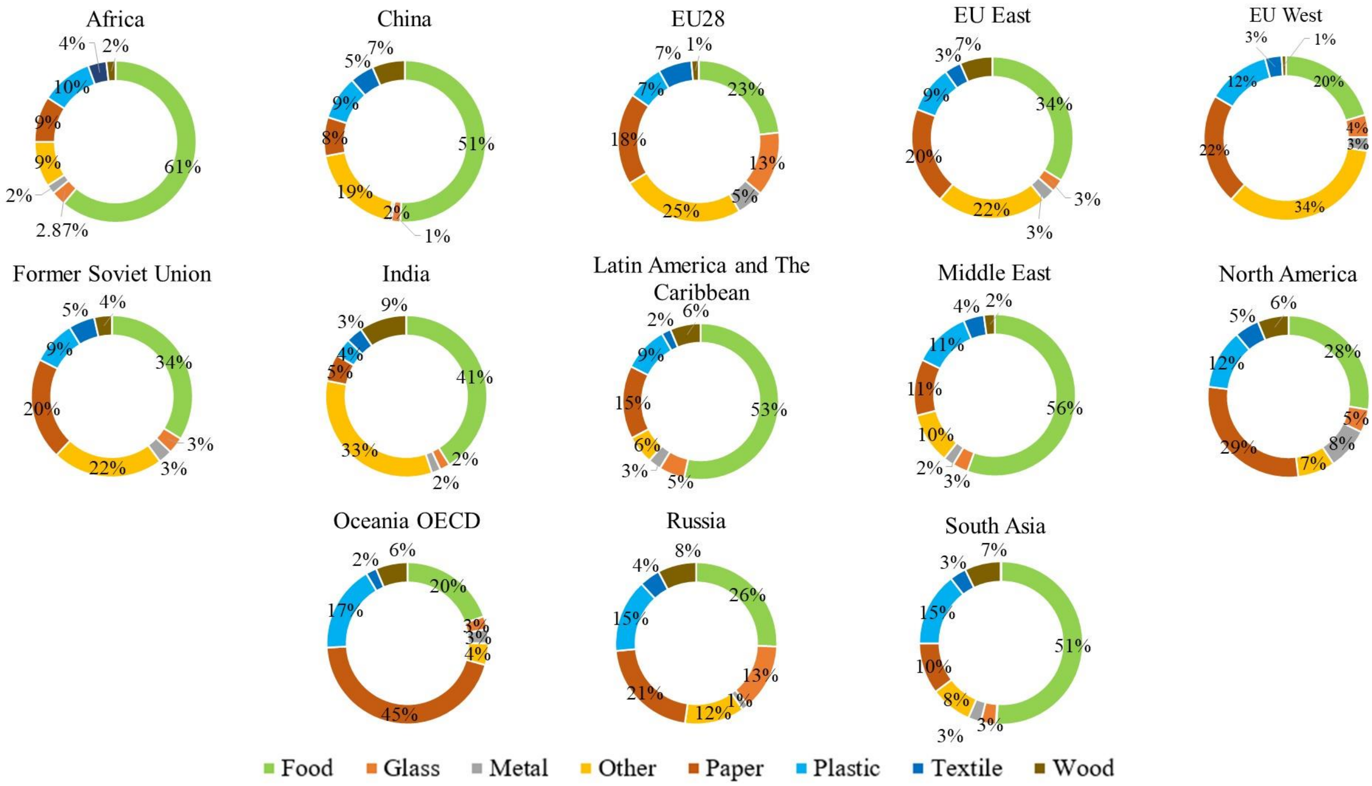

Figure 5. MSW composition by region in 2050

Figure 6 presents INW by type of manufacturing industry. Middle East, Africa and Latin America and The Caribbean are projected to generate the highest INW from food manufacturing industry by 2050. For INW from plastic and rubber manufacturing it is expected that North America, EU28 and Oceania will generate the maximum quantities. North America and Europe account for 18% and 17%, respectively, of the total global plastic production (Plastics Europe, 2019), a situation which somehow coincides with the waste accounting. A similar situation is observed with INW from pulp and paper manufacturing. The projection of INW in manufacturing of textiles clearly reflects the production of textiles in South Asia, as well as the increase of textile production in India and China (Statista, 2019). The same is the case for wood, in which the highest amounts of INW result from regions with a higher share in the market such as North America, Russia, and Latin America (Brazil).

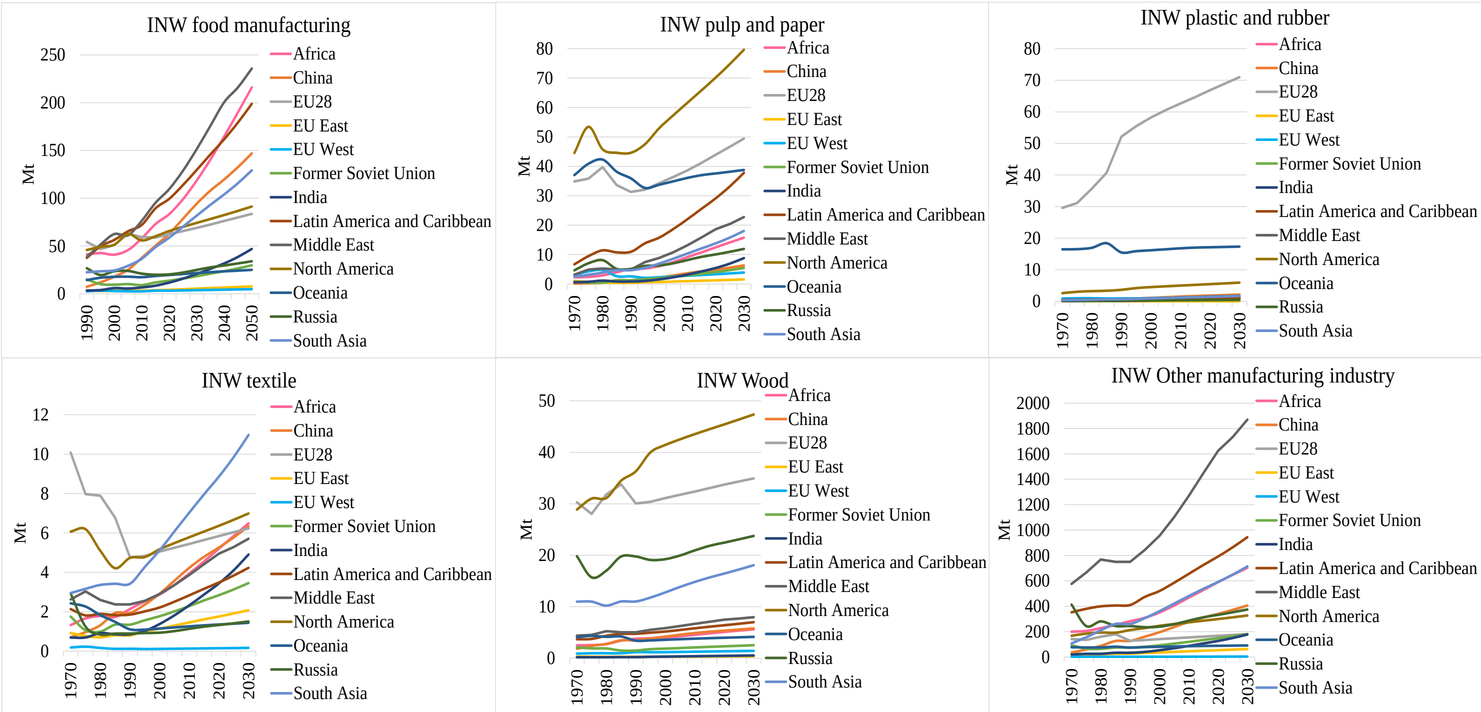

Figure 6. INW generation projections

Once the projections were ready it was possible to determine the carbon in waste and wastewater to quantify the maximum energy that can be generated with an implementation of circular treatment systems and various policy interventions. The different scenarios and technological improvements were then contrasted with the current situation. Modelled scenarios are presented in Figure 7.

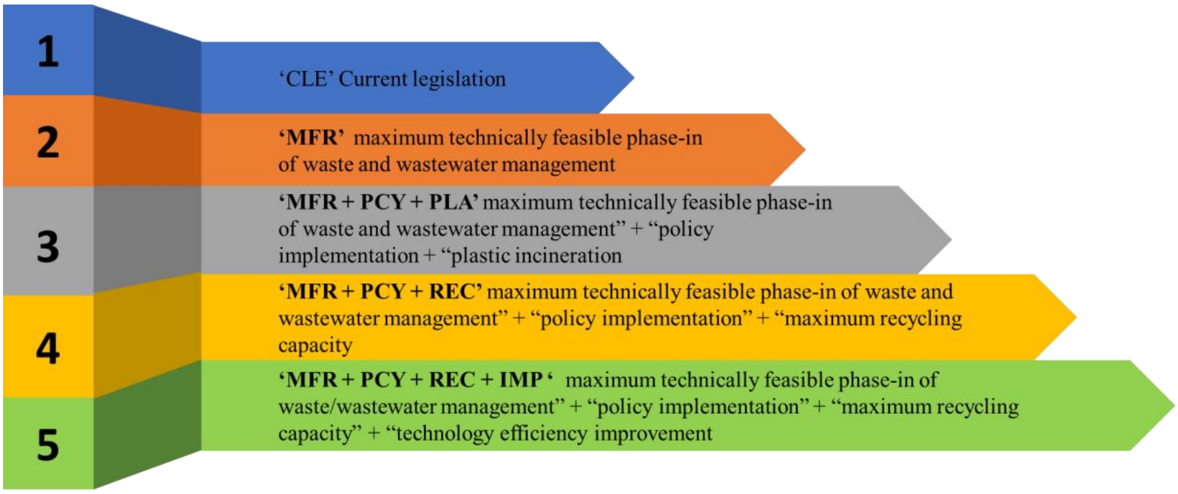

Figure 7. Modelled scenarios

The carbon flows in waste and wastewater systems show that, currently, considerable amounts of carbon are stored in material without being recovered for recycling or made available for energy generation. Currently at a global level, we find that 59% of carbon in waste is lost (dumped, scattered and openly burned without energy recovery), 18% is recycled/composted and 23% is converted to energy. Some 35% of the carbon content in waste comes from industrial waste and 65% from municipal solid waste. In the CLE scenario, an estimated 400 Mt-C is expected to be used annually as an energy source by 2050. With a maximum technically feasible phase-in of waste management (MFR) implementation we estimate that 66% of the carbon in waste could be used to generate energy by 2050, through the use of anaerobic co-digestion or incineration. The global carbon converted annually into energy would then be around 1370 Mt by 2050. With the implementation of food and plastic waste reduction reaching 50% in 2030 on top of maximum technical implementation (MFR + PCY + PLA), the availability of total carbon in waste at a global level is expected to be reduced by 18% in 2050, resulting in around 1300 Mt of global carbon converted into energy by 2050. With an optimum recycling market on top of maximum technical implementation and plastic and food waste reduction policies (MFR + PCY + REC), it is estimated that the total carbon converted into energy would be around 1200 Mt.

Regarding wastewater, the estimates suggest that currently at a global level 57% of domestic and 38% of industrial wastewater is either untreated or discharged after primary treatment. With the current management the amount of BOD – COD going to energy generation by 2050 is expected to be 1 and 10 Mt, respectively. With the improvement of wastewater management focused on energy generation 39% of BOD (refers to urban wastewater) and 91% of COD could be going to anaerobic treatment with energy recovery by 2050, which corresponds to 48 Mt BOD and 205 Mt COD. Extending the treatment capacity (collection rates in urban areas) on top of the technical improvement by 2050 (MFR+PCY+REC+IMP) would increase up to 85% of BOD and 91% of COD going to anaerobic treatment, which corresponds to 78 Mt BOD and 205 Mt COD.

Finally, the estimates suggest that the global implementation of such an ideal system could increase the relative contribution of waste and wastewater sources to global energy demand from 2% to 9% by 2040, corresponding to a maximum energy potential of 64 EJ per year. This would, however, require widespread adoption of policies and infrastructure that stimulate and allow for large-scale waste prevention and separation, as well as highly advanced treatment processes. Giving priority to such efforts would enable circularity of the waste-energy system. The efforts have to be especially taken up by emerging economies as industrialized economies are shifting the low value-added activities to those regions (Krausmann et al., 2017a). Thus, material extraction, manufacturing and therefore waste and emissions are exacerbating the already detrimental structures in place to handle the outputs of the metabolic systems. One of the main problems in reaching sustainability is the lack of agreements regarding international environmental responsibility for countries outsourcing certain economic activities.

For this paper, I evaluated the level of decoupling between economic growth and waste and wastewater generation (organics in wastewater) using the decoupling index (DI) (Fischer-Kowalski and Swilling, 2011) for the period between 1990 and 2015. The results show that at the global level there is no evidence of absolute decoupling of waste and wastewater from GDP, however a certain degree of relative decoupling is observed, a finding which is in line with that from Chen et al., (2020), for example, regarding MSW. MSW, INW and wastewater from domestic and industrial sources have grown more less in unison, with a DI of 0.44 for MSW, 0.33 for INW, 0.30 for COD and 0.37 for BOD. Although in general most of the regions show a type of relative decoupling, the decoupling level, however, differs. Despite the increase of MSW generation in China by 3% between 2005 and 2010 and 19% between 2010 and 2015, GDP has increased at a faster pace, hence a sign of strong decoupling is observed for MSW and BOD. Similar is the case for India. The coupling between INW from manufacturing industry and GDP in regions such as China, South Asia and, to some extent, India, until 2010 reflect the early phases of transition to become more industrialized economies (Krausmann et al., 2017a). As a result of the stagnation and collapse phase between 1990 and 1998 in the Russian Federation and The Former Soviet Union a period of negative decoupling is observed where MSW and BOD generation remain at the same level while GDP decreases at an extremely faster pace. In contrast, INW and COD start decreasing with GDP decrease, although at slower pace. This behaviour somehow recreates the consumption part (MSW and BOD) and the production part (INW and COD). Overall after that period, MSW shows clearly negative decoupling, while, for instance, INW in Russia shows absolute decoupling, a fact that does not directly represent improvement in efficiency in the process but rather could highlight structural changes in economic activity (Krausmann et al., 2017a). Oceania OECD is the only region with strong signs of absolute decoupling between GDP and both MSW and INW and to certain degree BOD and COD, mainly due to the improvement in waste management systems and decrease of waste generation in Japan. Japan is an outstanding example of effective implementation of measures to move to a material-cycle society (Ministry of the Environment, 2012).

Figure 8. Decoupling by region (Index 1990 = 100). Data sources in supplement of (Gómez-Sanabria et al., 2018)

In this paper, I developed the overall scope and strategy of the analysis, carried out the literature review and collected the data and other relevant information. I also carried out the data analysis and developed the model to project MSW. I was intensively engaged with colleagues in discussing the development of the global scenarios. I wrote the first complete draft of the paper, which was then edited by the other co-authors. In the revision process, I carried out the corresponding corrections and improvements and I responded to the comments of the reviewers.

4.3 Technical potentials and cost for reducing global anthropogenic methane emissions in the 2050 timeframe-results from the GAINS model (Paper III)

Following the projections of waste and wastewater and estimations of carbon content, I moved to assess the methane emissions from waste and wastewater sectors and their role on the reduction of global anthropogenic methane emissions. Currently, methane contributes about 18% of global anthropogenic greenhouse gases (based on GWP of 28 times CO_2 over 100 years (IPCC, 2014). Emissions from waste and wastewater account for around 3% of total global anthropogenic GHG emissions which represent around 18% of the global anthropogenic methane emissions (Bogner et al., 2008). Höglund-Isaksson (2012) estimated that about 30% of the global technical methane mitigation potential in 2030 comes from the waste and wastewater sectors. Therefore, even though the contribution from the waste sector is small compared to livestock or oil production (AMAP Assessment, 2015; Höglund-Isaksson, 2012) methane mitigation in this sector is important to control climate change.

Technologies to mitigate methane from waste in landfills include vertical wells and horizontal trenches used as collectors to prevent migration of gas. The gas collected can be flared or recovered to produce energy (Bogner et al., 2008; Yusuf et al., 2012). Alternative treatment options for organic waste are anaerobic digestion and composting. Anaerobic digestion provides a clean fuel from renewable feedstocks which reduces environmental impacts due to the replacement of fossil fuel derived energy (Chynoweth et al., 2001). Composting converts the organic residues into a product rich in humus and plant nutrients, thus, decreasing the use of inorganic fertilizers. Moreover, it is an alternative to divert organic waste from landfills (Gajalakshmi and Abbasi, 2008). For inorganic waste, options such as recycling or incineration involve avoided methane emissions (Bogner et al., 2008). The estimates suggest that under the current legislation, the contribution of CH_4 emissions from waste and wastewater to global anthropogenic CH_4 can increase from 19% in 2015 to 31% in 2050. High technical abatement potentials at about 80 percent below baseline emissions in 2050 are considered feasible for CH4 emissions from solid waste management (Höglund-Isaksson et al., 2020). This assumes it is possible in a twenty-year perspective to extend the infrastructure for source separation, recycling, and energy recovery schemes globally, including a ban on all landfill of organic waste, and allowing for useful utilization of the carbon content of the waste (Gómez-Sanabria et al., 2018).

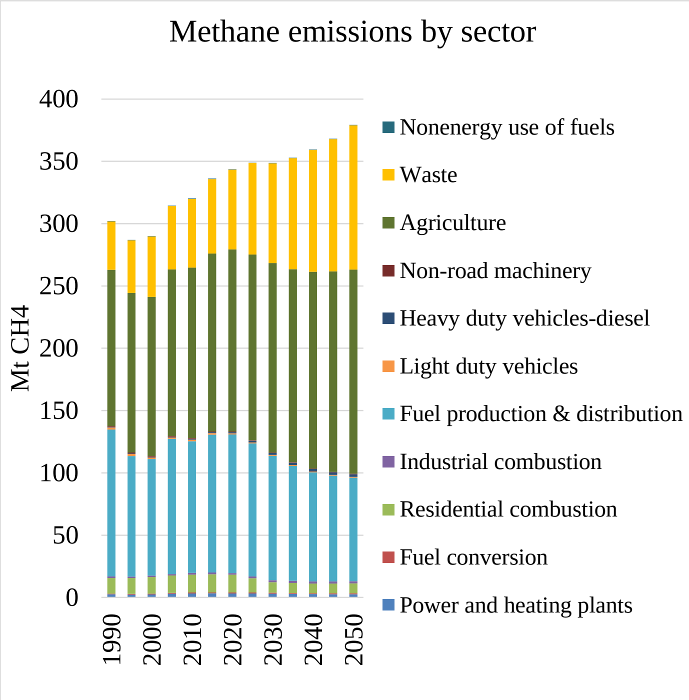

Figure 9. Decoupling by region (Index 1990 = 100). Data sources in supplement of (Gómez-Sanabria et al., 2018)

Source: GAINSv4 Scenario Eclipse_V6b_base. Waste sector by Gómez-Sanabria A.

In this paper, I derived data on the activity levels of the waste and wastewater sectors from literature, official statistics, national reports and expert workshops, developed methods to calculate emissions estimates and projections for both scenarios, baseline, and maximum feasible emissions reduction. I was also engaged in discussions, particularly those related to the waste and wastewater sector.

4.4 Potentials for future reductions of global GHG and air pollutant emissions from circular municipal waste management systems. (Paper IV)

The aim of this paper is to develop long-term scenarios for MSW generation, composition and management and the associated emissions under plausible future socioeconomic development pathways (SSPs). We developed two sets of scenarios, baseline (CLE), and mitigation scenarios (MFR) up to 2050, distinguishing urban-rural scale. It was found that MSW spans a range from 2.01 Gt in 2010 to 2.3 Gt in 2015. Different socioeconomic assumptions in the developed scenarios lead to significant differences in the amounts of generated MSW (Figure 10). The lowest quantities of MSW generation are expected in the SSP3 and SSP4 (3.6 Gt/yr in 2050) because of slow economic growth and inequalities between regions which is reflected in different consumption patterns. On the other hand, in the SSP5 where both income and urbanization rates increase strongly, the MSW generation quantities are estimated at 4.3 Gt/yr in 2050.

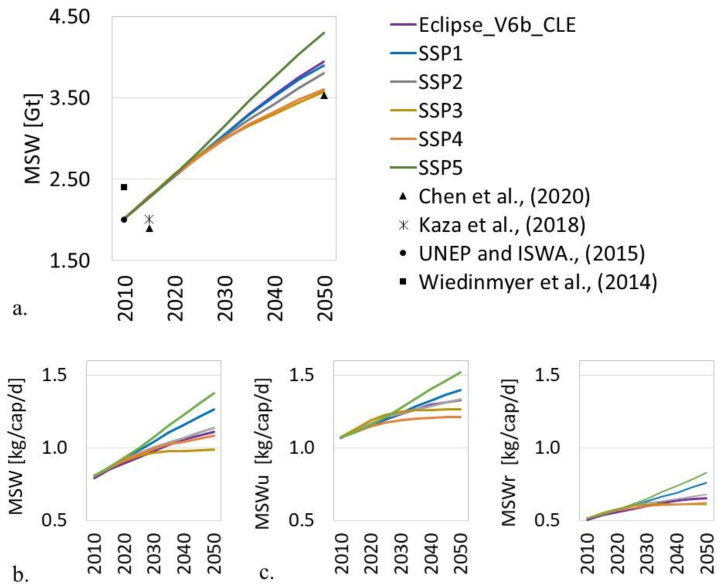

Figure 10 a. Global MSW generation. b. Global MSW generation rates. c. Global urban MSW generation rates. d. Global rural MSW generation rates

Unfortunately, most of the regions having the highest MSW generation quantities have the lowest collection rates. Our estimates suggest that in 2015, 43% of the global MSW collected ended up either in landfills (13%) that are compacted and/or covered but not meeting environmental standards to prevent leakage(Calvo et al., 2005), in unmanaged landfills without any type of management (hereafter referred as dumpsites) (21%), or was openly burned (9%) either directly at the dumpsites (including unintended fires) or in transfer stations. The remaining 29% of the collected waste was either disposed of in sanitary landfills (10%), incinerated (7%), recycled (7%), or composted or anaerobically digested (4%), which is mostly happening in developing countries. From the uncollected fraction, around 20% is estimated to be scattered MSW with a high probability of eventually reaching water courses, and 10% openly burned.

Regarding emission reduction up to 2050 in the different scenarios, the environmental co-benefits will be obtained at different levels depending upon the level of socio-economic development and political and institutional coordination. The different assumptions on policy interventions are then translated into a wide range of future emissions. Figure 11 shows global emission trends for the CLE and MFR scenario families from 2010 to 2050.

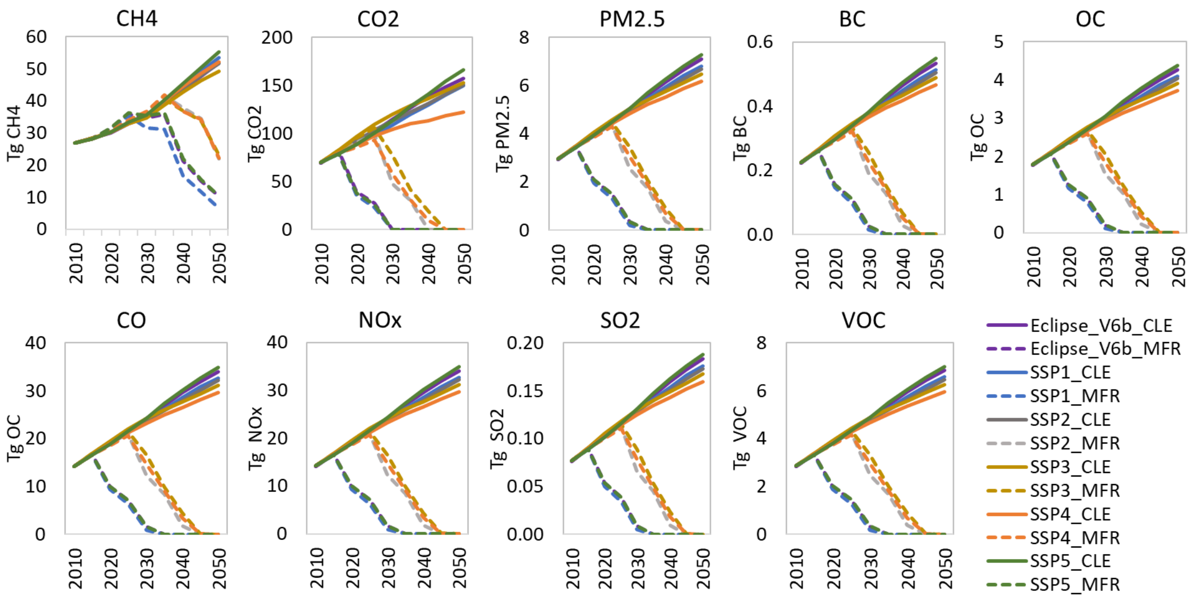

Figure 11. Global emissions from MSW

CH_4 emissions from waste deposited today in landfills will be generated in future years as it depends on the degradability of the organic matter. Therefore, MSW generation quantities and policy adoption at early stages make a significant difference to the trends of CH_4 emissions through the years. For example, towards the year 2035 a slight increase in CH_4 in the MFR families is expected compared to CLE families, except for SSP1_MFR. Towards 2050 CH_4 emissions are expected to be 7 Tg (196 Tg CO_2eq) in the SSP1_MFR, which is 88% lower than in the SSP1_CLE and 30% lower than the expected emissions in the SSP5_MFR and ECLIPSE_V6b_MFR. Our estimates also suggest that CH_4 from landfills in the SSP2_MFR, SSP3_MFR and SSP4_MFR will increase by 3-4% in 2030 and will decline after 2040 reaching a maximum reduction of about 55% in 2050 (22 Tg CH_4 or 616 Tg CO_2eq).

Similar trends are observed with respect to emissions of CO_2, particulate matter, and air pollutants. The percentage reduction of MSW being openly burned translates into the same reduction level of emissions. In the SSP1_MFR, SSP5_MFR and ECLIPSE_V6b_MFR scenarios it will be possible to decrease emissions of CO_2 by about 70% from baseline in 2025 and reach the maximum reduction in 2035. In the SSP2_MFR the maximum reduction will be achieved in 2040 and in the SSP3_MFR and SSP4_MFR in 2045. Our results show a similar situation regarding emissions of $PM_{2.5}$, BC, OC and other air pollutants. Under the development of SSP1_MFR, SSP5_MFR and ECLIPSE_V6b_MFR, the maximum emission reduction potential will be realized in 2035 whereas in the SSP2_MFR it will take 5 years more and for the SSP3_MFR and SSP4_MFR 10 years more than in e.g., SSP1_MFR.

In this paper, I designed the study, performed the projections, emission simulations and analysis, and prepared the manuscript. Other co-authors provided expert guidance during the whole process and contributed to the revision of the manuscript. All authors were involved in the discussions during the process.

4.5 Sustainable wastewater management in Indonesia's fish processing industry: Bringing governance into scenario analysis (Paper V)

This paper presents a case study in which I applied the methodology developed for the global assessment of emissions in the wastewater sector. The case study centres on Indonesia's fish processing industry. Furthermore, it has the novelty that it is the first time that governance is integrated into emissions scenario analysis. The governance variables such as enforcement capacity, institutional coordination and multi-actor networks are incorporated into an analysis of the potential impacts on greenhouse gases and chemical oxygen demand in seven wastewater treatment scenarios.

A description of the scenarios developed is presented in Table 3. Each scenario is designed based on the three main elements: policy, form of governance and technology.

Table 3. Description of the scenarios

Scenario	Policies	Forms of governance	Technology
Business as Usual (BAU)	Current situation - no further enforcement	Current situation	Untreated/anaerobic lagoons
National Wastewater Policy (NWP)	National wastewater policy	No coordination between wastewater and climate agencies	Aeration lagoon plus Activated sludge
Climate Change Policy (CCP)	Climate change policy	No coordination between wastewater and climate agencies	Swimbed
Co-benefits vertical horizontal coordination (CB1vh)	National wastewater policy and climate change policy	Vertical horizontal coordination	Up-flow Anaerobic Sludge Blanket (UASB) plus Activated Sludge (with gas recovery and used).
Co-benefits vertical horizontal coordination (CB2vh)	National wastewater policy and climate change policy	Vertical horizontal coordination	Up-flow Anaerobic Sludge Blanket (UASB) plus Swimbed (with gas recovery and used).
Co-benefits multi-stakeholder network (CB1ms)	National wastewater policy and climate change policy	Multi-actor network	Up-flow Anaerobic Sludge Blanket (UASB) plus Activated Sludge (with gas recovery and used).
Co-benefits multi-stakeholder network (CB2ms)	National wastewater policy and climate change policy	Multi-actor network	Up-flow Anaerobic Sludge Blanket (UASB) plus Swimbed (with gas recovery and used).

Results show that the implementation of NWP and CCP do not deliver co-benefits in terms of simultaneous reduction of COD and GHG emissions due to the absence of multi-level, multi-stakeholder governance. In contrast, the set of CB scenarios, which address environmental concerns by reducing COD concentration in the effluent while reducing GHG emissions from wastewater treatment through multi-level governance, deliver those co-benefits. In that sense, the scenario providing the maximum benefits is the one which combines the highest COD removal efficiencies with the lowest GHG emissions per unit of COD removed. This, in turn, can ensure compliance with the national wastewater standards while reducing GHG emissions from wastewater treatment and therefore supporting the achievement of the Indonesian NDC targets.

Figure 12 shows the relation between GHG emissions and COD removal efficiency. The adoption of any alternative scenarios to the BAU, except for the $CB1_{vh}$ and $CB2_{vh}$, would result in compliance with the national effluent standards by 2030. Interestingly, Figure 4 also shows that the highest GHG emissions are expected from the implementation of NWP. CCP, $CB1_{ms}$ and $CB2_{ms}$ depict similar COD removal efficiencies; nevertheless, $CB2_{ms}$ (UASB plus swimbed) provides the maximum benefits in terms of GHG emissions and is the only option that improves both COD effluent concentrations and mitigates climate impacts. By 2030, the $CB2_{ms}$ removal efficiency would reach a maximum of 98.4% COD removed, while reducing GHG emissions by 60% compared to BAU. GHG emissions in the $CB2_{ms}$ are also 47% lower than in $CB1_{ms}$ and 57% lower than in CCP.

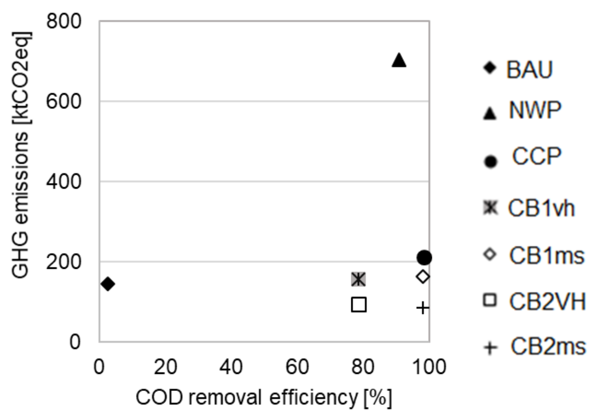

Figure 12**.** Multiple benefits of the analysed scenarios

A summary of the main achievements resulting from implementation of the different scenarios in the year 2030 is presented in Table 4. The scenario providing the maximum co-benefits is highlighted in grey.

Table 4. Achievements by scenario in 2030

Scenario	Policies	Form governance	Technology	Total GHG emissions [kt CO_2eq/year]	COD removal efficiency [%]	Electricity replaced by own biogas [%]
BAU	Current situation	Current situation	Untreated/anaerobic lagoons	143	3	0
NWP	NWP	No coordination	Aeration lagoon+Activated Sludge	703	91	0
CCP	CCP	No coordination	Swimbed	211	99	0
CB1vh	NWP+CCP	Vertical-horizontal coordination	UASB+Activated Sludge+Energy recovery and use	156	79	36
CB1ms	NWW+CCP	Multi-actor network	UASB+Activated Sludge+Energy recovery and use	162	98	36
CB2vh	NWP+CCP	Vertical-horizontal coordination	UASB+Swimbed+Energy recovery and use	94	79	52
CB2ms	NWW+CCP	Multi-actor network	UASB+Swimbed+Energy recovery and use	85	98	52

This article offers a unique perspective on the governance reforms needed to achieve climate and wastewater treatment goals. In many cases legislation exists but the lack of implementation is a major burden for the realization of the objectives. This follows research that argues that managing wastewater is frequently a governance issue (Casiano Flores et al., 2017; Grigg, 2011). Important messages from this article begin with the claim that NDCs and wastewater regulatory standards currently can serve as a starting point for mitigating climate change and improving water quality. However, a critical finding is that the crucial need is not simply strengthening climate and wastewater management policies and measures but aligning policymaking institutions and decision-making processes. A related finding is that strengthening government capacity without coordination can lead to the more stringent enforcement of treatment measures focused solely on COD removal that can surprisingly increase GHG emissions.

In this paper, I developed the concept together with Eric Zusman. I carried out the development of the methodology and the strategy for analysing results. I was directly involved and played a central role in all the discussions concerning the conceptualization and development of the manuscript. I wrote and prepared the paper with contributions from all co-authors. I was responsible for the reviews and answered the comments of the reviewers.

5. Discussion and conclusions

As outlined in the introduction, this dissertation has attempted to quantify the GHG and air pollution emission reduction potentials from implementing circular global waste and wastewater management systems. With the current heated discussion about climate change, circular economy, and sustainability, it is increasingly important to evaluate and recognize the role of this sector as relevant to meet the Paris Agreement to stay at 1.5 degree warming and to achieve the SDGs. This dissertation began with a thorough review of the sustainability concept, the associated Sustainable Development Goals and how Social Ecology can bridge the different agendas by investigating the society-nature interactions at various spatial and temporal scales. In this context, I proceeded to investigate the importance of waste and wastewater in social-metabolism and to identify to what extent this sector can contribute to the reduction of GHGs and air pollutants emissions in the transition to a more sustainable future.

Hence, one of the first steps towards reaching the overall objective of the research was to carry out a comparison of MSW generation, composition, and management in regions with different socio-economic characteristics (Paper I). Here, I showed that MSW highly depends on the economic development of the regions as well as political arrangements in place. MSW management systems in some way reflect the level of development of a region and institutional coordination. This review underlines the common challenges that hinder the development/improvement of MSW management systems in developing countries. These challenges can be summarized as lack of funds, poor planning, poor implementation of law and lack of expertise. Beyond the technical and political arrangements, this review also takes a magnifying glass to existing inequalities on the access to basic human needs including sanitation services.

Paper I opens the space to develop a model to project MSW generation and composition by income level (Paper II). Moreover, the additional estimates of MSW generation in urban and rural areas go beyond the state of the art (Paper IV). This is foremost as MSW represents to a certain extent societies' lifestyle which is directly related to economic development. The method I propose here captures and represents to a higher degree these characteristics includes projections of generation and composition of MSW by income level. This fact that plays a significant role when quantifying the carbon content in MSW and the potential carbon that can be recovered either as embodied in materials or as energy. With this, together with the quantification of carbon content in INW and domestic and industrial wastewater, is possible to assess to what extent MSW management can support the circular economy and the decarbonization of the energy system. We learned that the same disparities observed in the waste management systems between developed and developing countries (and urban and rural areas) also occur in the wastewater management systems. These systems are interconnected and hence the improvement of either of them benefits the other one. For instance, the improvement of waste management systems will reduce the leakage of waste into water courses, thus reducing water pollution and protecting life on water and potentially on land. We also find that, theoretically, the global implementation of circular waste and wastewater management systems can increase the contribution of the waste and wastewater sources to global energy demand from 2% to 9% (64 EJ per year) by 2040. This would require widespread adoption of policies and infrastructure that stimulate for large-scale waste prevention and separation, as well as highly advanced treatment process. Furthermore, the implementation of such systems is key in achieving the SDGs. 12 of the 17 goals include direct targets to improve waste and resource management and sanitation (Velenturf and Purnell, 2017). For instance, the SDG 6, specifically 6.3: “improve water quality, eliminate dumping, minimize the release of hazardous chemicals and materials, halving the proportion of untreated wastewater and substantially increasing recycling and safe reuse globally” (United Nations, 2015). Other SDGs include good-health and well-being (SDG3), affordable and clean energy (SDG7), life below water and land (SDG14 and 15) infrastructure and industrialization (SDG 9). For example, South Africa’s National Solid Waste Management Strategy 2020 includes a specific section stating the contribution of the strategy to the SDGs (Department of Environment, Forestry and Fisheries, 2020).

In addition to the assessment of how circular waste and wastewater systems support the decarbonization of the energy system, Paper III looked at the global technical potentials of reducing anthropogenic CH_4 emissions. This paper included all anthropogenic activities, however, here I only refer to the waste and wastewater sectors. The aim of this paper, together with Paper IV, was to investigate the actual contribution of the CE framework in mitigating CH_4 and air pollution when adopted for the waste and wastewater sectors. This part of the research provides scientific evidence of the potentials and limits of CE to mitigate environmental pressures resulting from human activity. We see that in 2015 the waste and wastewater sector contributed about 18% (61.2 Tg CH_4) of the total global anthropogenic CH_4 emissions. Of this, MSW is responsible for 51% of the emission while INW contributes 18%. Domestic and industrial wastewater make up the remaining 31%. Under the current conditions it is estimated that the contribution of this sector to CH_4 emissions will increase up to 25%. The technical abatement below the 2050 baseline is assessed at 82% for MSW, 74% for INW, 26% for domestic wastewater and 99% for industrial wastewater. In general, high technical abatement potentials at about 80% below the 2050 baseline are feasible from implementing circular management systems in the solid waste sector. In a twenty-year perspective it would be possible to globally extend the source segregation infrastructure, boost recovery and recycling schemes, and ban landfilling of organic waste, thus allowing one to efficiently recover and utilize the carbon in waste.

Paper IV assesses the reduction of particulate matter and air pollution as a result of implementing circular systems. In this paper long-term scenarios for MSW generation, composition and management and the associated emissions are developed considering the SSPs storylines, and distinguishing urban-rural scale. One of the main findings is that the estimates of historical emissions are lower than those used in the Coupled Model Intercomparison Project Phase 6 (CMIP6) (Gidden et al., 2019; van Marle et al., 2017), consequently affecting also future emission trajectories. Our estimations suggest that MSW generation is expected to increase by between 3.7 Gt/yr and 4.3 Gt/yr by 2050 considering the assumptions in the socioeconomic pathways (SSPs). This represents a growth in MSW amounts between 60% and 75% compared to 2015 levels, of which urban areas are responsible for about 80%. The generally high collection rates of MSW in urban areas do not necessarily imply appropriate management. In South Asia, India, China, LCAM and Africa about 80% of the collected MSW is either dumped or openly burned. Furthermore, most of the MSW generated in rural areas is uncollected and thus ends up being illegally dumped, scattered, or openly burned, resulting in several environmental impacts related to air pollution and greenhouse gas emissions. In the baseline (CLE), in which current MSW management practices persist without further policy implementation, emissions to air would increase proportionately to the growth in MSW generation. We then developed a set of mitigation scenarios (MFR) to assess the impacts of abatement measures compared to the corresponding baseline (CLE). The common target of our MFR scenarios is to achieve ~100% collection and management of MSW by 2050 through the implementation of circular MSW management systems to simultaneously tackle emissions of CH_4, CO_2, particulate matter and air pollutants. Co-benefits are obtained at different stages depending upon the level of socio-economic development and political and institutional arrangement. Evidently, all countries would benefit from reduced MSW generation and improved management in the 'Sustainability-oriented' scenario (SSP1_MFR), however, the additional benefit of respective measures is especially relevant for regions generating large MSW quantities and lacking appropriate management systems. We show that the environmental co- benefits of avoided MSW combined with the speedy implementation of anaerobic digestion to treat organic waste and the establishment of source-separated MSW collection to increase the recycling of materials, the 'Sustainability – oriented' scenario, brings major and faster co-benefits in terms of reducing CH_4, CO_2, particulate matter and air pollutants.

Lastly, Paper V integrates governance level within the modelling framework. In this case, a study focusing on the fish processing industry in Indonesia is used to demonstrate that technology could help to achieve the ambitious goals in its Nationally Determined Contribution as well as reduce water pollution. However, the installation and widespread application of these technologies require the understanding of how governance affects the implementation of the existing policies and cooperation across sectors, administrative levels, and stakeholders. Using the GAINS model to assess different technologies and governance options, the study showed that some wastewater treatment approaches would generate considerable methane and carbon dioxide emissions. By exploring different trade-offs, it is possible to identify the option delivering the maximum benefits for both climate and water.

The main scientific contributions from this dissertation are as follow:

- A new model to project MSW generation and composition distinguishing between urban and rural areas.
- Global assessment of carbon flows in domestic and industrial solid waste and wastewater.

- Evaluation of the potential contribution of implementing global circular waste and wastewater systems as a strategy to reduce GHGs, particulate matter and air pollutants.
- Development of scenarios for MSW considering the SSPs storylines which can be used to evaluate goals related to the SDGs and can be included in Integrated Assessment Models (IAMs).
- Demonstration of the relevance of the solid waste and wastewater sectors as an important catalyst in reaching the SDGs and reducing GHGs, particulate matter and air pollutants. Places governance as a key element for the realization and implementation of plans, measures, and policies.

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REVIEW ARTICLE

Two-Level Comparison of Waste Management Systems in Low-, Middle-, and High-Income Cities

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Abstract

Despite the large body of literature comparing prevailing waste management practices at different locations in the world, this article adopts a novel approach of two-level comparison: baseline and strategic plans. It analyses the state of municipal solid waste (MSW) management in six selected cities with different geographic locations and socioeconomic setups. As a first comparison, the current MSW profiles are analyzed to pinpoint the prevailing challenges. From the perspective of governmental institutions, the main obstacles in low- and middle-income cities seem to be the lack of regulations and, most noticeably, the inefficient structure of the waste management sector. Technically, the main challenges are low collection rates, land scarcity, and high transportation costs, as well as the lack of diversified management options. The latter renders the waste sector vulnerable and increases its instability. The second-level comparison addresses the strategic development plans of the studied cities. While cities are planning for various upgrading methods, incineration is perceived by authorities as a practical approach to limit transportation cost and reduce space requirements of landfills. However, special attention should be paid to the environmental impacts of thermal methods in absence of elaborated regulations and strict supervision in low- and middle-income cities. Thus, solutions for high-income cities might not be suitable for developing cities. Methods with lower environmental and socioeconomic impacts, such as anaerobic digestion, were seldom considered in the future plans of the studied cities; yet they can provide economically feasible solutions considering the high organic and high moisture contents of waste in low- and middle-income cities.

Keywords: material flow analysis; waste collection; waste generation; waste management; waste treatment

Introduction

Traditionally, municipal solid waste management (MSWM) has been considered a major municipal concern, posing political, social, and/or financial challenges. The impact aggravates with socioeconomic evolutions, including population growth, income increase, and changes in consumer behavior, leading to continuously increasing waste generation rates (UN-HABITAT, 2010). According to Hoornweg and Bhada-Tata (2012), global municipal solid waste (MSW) generation is expected to rise from 1.3 billion Mg in 2012 to 2.2 billion Mg by 2025. In the lack of proper management, the increasing waste generation rate would exacerbate air pollution problems, land contamination, and water quality degradation. Those constitute major threats to the environment, the economy, and, most importantly, human health (Asase *et al.,* 2009)—with children being the most at risk (Hester and Harrison, 2002).

Most developing and emerging countries are faced with the major challenge of improving their inadequate and unsustainable MSWM. Waste management systems in developing countries are characterized by low collection rates, high rates of open dumping and/or open burning, and low recycling rates (Manaf *et al.,* 2009). Commonly, this has been attributed to financial, technical, and institutional challenges at local and national levels (Pokhrel and Viraraghavan, 2005). On the contrary, waste management in developed countries is generally characterized by efficient and strongly implemented

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policy frameworks, as well as advanced waste management methods and technologies (Sunday *et al.,* 2016).

Therefore, to develop a suitable waste management strategy in developing regions, local authorities and policy makers often tend to rely on the experience of industrialized countries, while taking into consideration social setups (community needs and public conceptions) as well as technical facts (generation rates and composition of the waste). With this in mind, researchers from different countries and backgrounds decided to join efforts and merge local data into an insightful comparative review to provide compelling information to waste managers in developing cities.

Even though several studies investigated different aspects of MSWM at the city level (e.g., Otterpohl *et al.,*1997; Abarca Guerrero *et al.,* 2013; Andreasi Bassi *et al.,* 2017) and compared MSWM in selected cities (UN-Habitat, 2010), this article attempts to provide a two-level comparison of waste management in cities with different income levels. First, the current metabolisms of the different cities are compared and the prevailing challenges are highlighted. Next, the anticipated improvements/changes in MSWM of each city are explained and the ''future'' management systems are compared (once again). Conclusions are made as to the similarities and differences in the approach to improvement adopted by six representative cities from high-, upper-middle, and lower income countries: Ahmedabad in India, Keserwan District in Lebanon, Mexico City in Mexico, Bogotá in Colombia, Thessaloniki in Greece, and Ostrava in Czech Republic.

Methodology

Selection of reference cities

Six cities were selected to represent various population sizes, socioeconomic conditions, political setups, and geographic and climatic conditions, resulting in different waste composition, treatment technologies, and energy and material recovery rates (Hoornweg and Bhada-Tata, 2012).

The sample includes the following: two cities located in Europe (Thessaloniki in Greece and Ostrava in Czech Republic) classified as high-income countries; three cities falling in upper-middle income countries (Mexico City in Mexico, Bogotá in Colombia, and Keserwan District in Lebanon); and one lower-middle income city (Ahmedabad in India) (Table 1)—according to the The World Bank (2018). The selected cities fall in a latitude range of 19.43 (Mexico City) to 23.02 (Ahmedabad) (Fig. 1).

Data collection

Collection of waste generation and management data was carried out differently for the various cities, depending on availability and accessibility of data. In some cities such as Keserwan, Ahmedabad (AMC, 2016a, 2016b; Datta and Kumar, 2016; Kumar, 2016), Mexico City, and to some extent Ostrava, it was possible to conduct interviews with board members of municipalities and/or concerned municipal corporations. The collected data were either of quantitative nature (mainly generation rates and waste flow values) or directly pertaining to the methods and technology adopted in the waste management system. The collected data were found to be consistent with official reports. In other cities, like Thessaloniki, and Bogotá, information was retrieved from official documents, reports, and statistics. Supplementary Table S1 presents a summary of the data collection procedure for each city.

Material flow analysis

The metabolism of the studied cities is visually presented using the material flow analysis (MFA) concept. MFA is defined in Brunner and Rechberger (2004) as systematic assessment of the flows and stocks of material/substances within a system defined in space and time, in which the sources, the pathways, and final sinks are connected. The analysis is based on the law of conservation of matter—the mass balance principle comparing inputs, stocks, and outputs of a process. MFA facilitates data verification and enables the estimation of missing data (Vyzinkarova and Brunner, 2013). It is a widely used tool in waste management, where it shows the need for final sinks and for recycling measures; furthermore, it helps in designing strategies for recycling and disposal (Brunner and Rechberger, 2004).

STAN is a freeware software that supports performing MFA under consideration of data uncertainties (Cencic and

Table 1. Profile of Selected Cities

Country	*City*	*Area (km^2)*	*GDP/cap (US$)*	*Population (million)*	*Country classification*[a]	*Characteristics*
India	Ahmedabad	466	—	6.30	Lower middle income	Major construction/infrastructure boom; steady population increase; growing hub of education, information technology, and Industry
Mexico	Mexico City	1485	20960	8.90	Upper middle income	Most populated city of the country
Lebanon	Keserwan	336	—	0.25	Upper middle income	Represents 4.2% of country population and 3.2% of its total area
Colombia	Bogotá	1587	15891	9.70	Upper middle income	Accounts for 25% of the country GDP
Greece	Thessaloniki	19.1	18500	0.32	High income	The second biggest city of the country
Czech Republic	Ostrava	214	17570	0.30	High income	The main industrial center of the Czech Republic

[a]According to The World Bank (2018).
GDP, gross domestic product.

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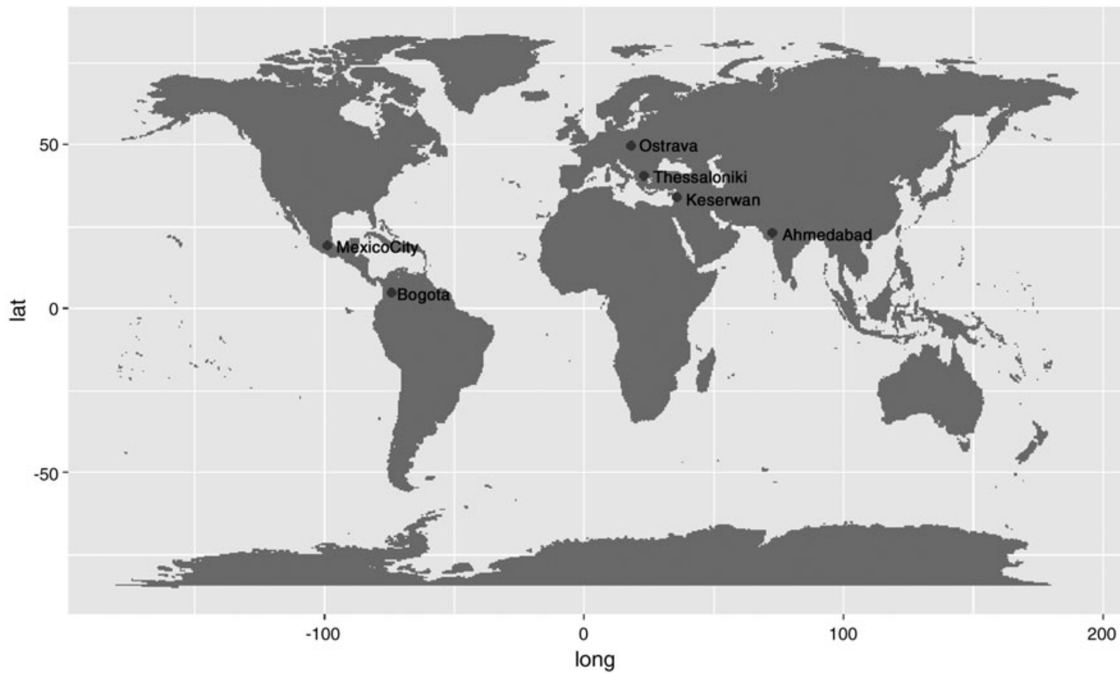

FIG. 1. Map showing location of six studied cities.

Rechberger, 2008). It allows modeling on two layers: the first focuses on goods, including economic goods with a positive or negative market value such as diverse types of waste; and the second layer analyses substances, including chemical compounds or elements (Vyzinkarova and Brunner, 2013). In this study, the analysis is performed at the level of goods, and thus considers the various components of MSW without referring to substance flows.

Solid Waste Characteristics

Waste composition plays a central role when defining a waste management system. It affects the characteristics of the waste such as density, moisture, and calorific value, and controls the selection of the waste technologies (UNEP and ISWA, 2015). The latter determines the type, quantity, and frequency of emission of pollutants and greenhouse gases.

Numerous studies indicate that the composition of MSW depends on socioeconomic characteristics, geographical location, and environmental features. Paper and plastic wastes are the main fractions of MSW in high-income countries, while organic waste prevails in low-income countries (Hoornweg and Bhada-Tata, 2012). Typically, MSW in low- and middle-income countries consists of 40% to 85% from organics. This percentage drops with higher income countries (Edjabou *et al.,* 2015).

Table 2. Solid Waste Composition

Country	*City*	*Generation (kg/[cap·day])*	*Composition (%)*							*Source*
			Organics	*Plastic*	*Paper*	*Metal*	*Glass*	*Textile*	*Other*	
Colombia	Bogotá	1.2	62.7	18.7	8.2	0.8	2.2	4	7.4	Martinez (2016)
Czech Republic	Ostrava (summer)	0.98	39	9	17	3	21	3	8	Žurková *et al.* (2015)
	Ostrava (autumn)		54	4	9	2	20	1	10	
Greece	Thessaloniki	1.1	44.3	13.9	22.2	3.9	4.3		11.4	Municipality of Thessaloniki (2016)
India	Ahmedabad	0.63	51	6	6		2[a]	10	25	Interview with municipal officials
Lebanon	Kaserwan	0.9	60	2	20	4	5		9	Ministry of Environment (LocaLiban, 2015)
Mexico	Mexico City	1.8	30	17	12	3	5		33	In-situ characterization

[a]Includes recyclables (Glass, Fe, and Al).
LSWMP, Local Solid Waste Management Plan.

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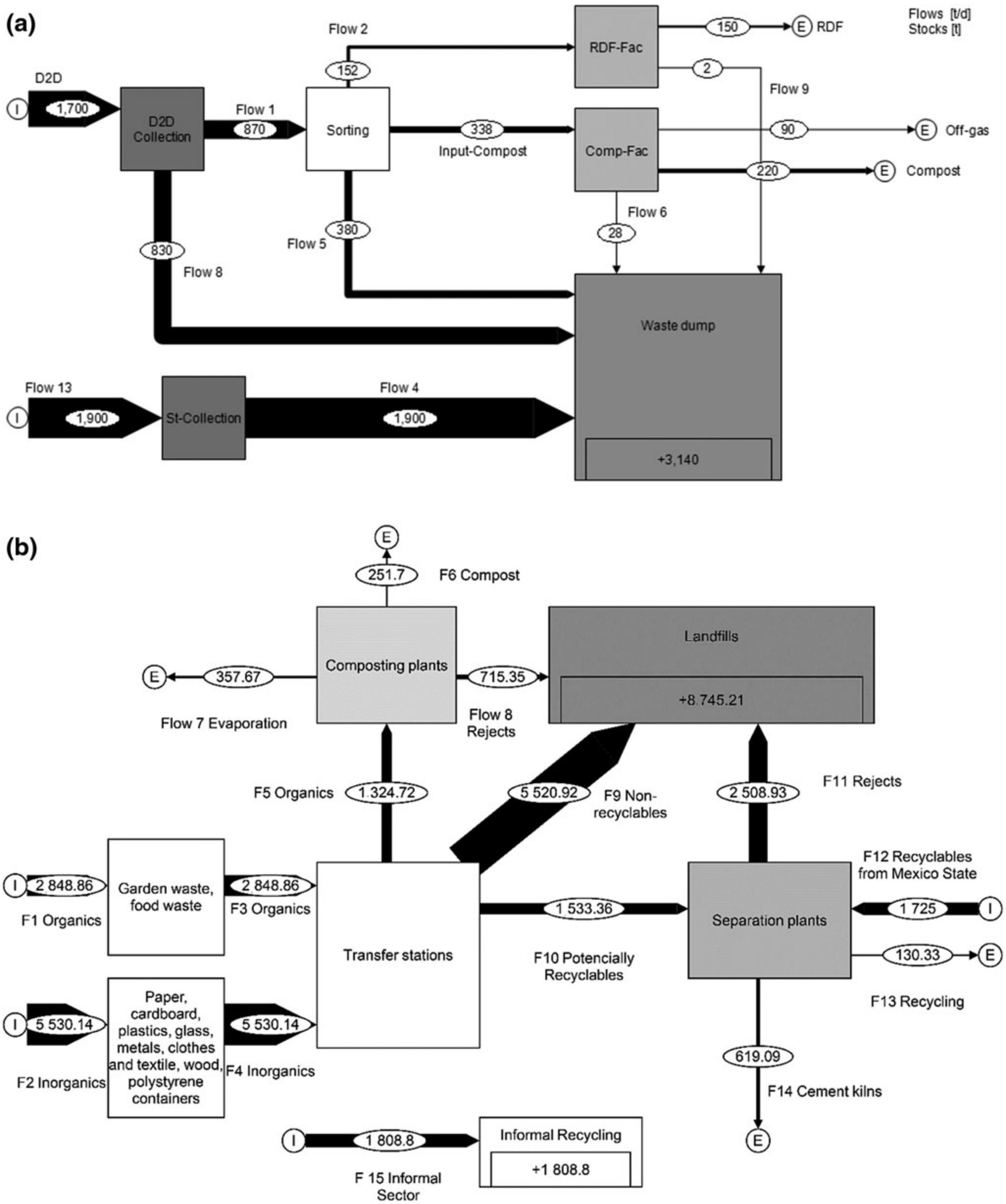

FIG. 2. Material flow analysis of baseline scenarios: **(a)** Ahmedabad, India; **(b)** Mexico City, Mexico; **(c)** Keserwan, Lebanon. MRF, material recovery facility; RDF, refuse-derived fuel.

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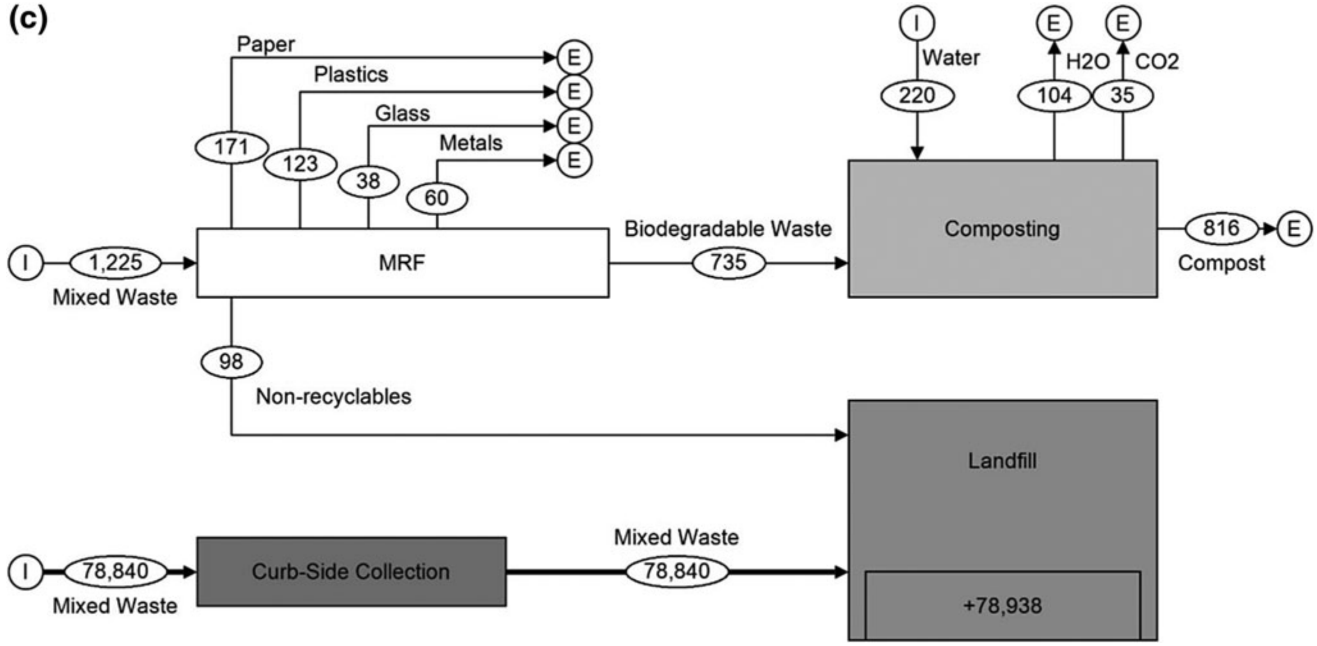

FIG. 2. *(Continued)*

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Like in most nonindustrialized countries, Bogotá and Keserwan have the highest fraction of organics with 63% and 60%, respectively. Ahmedabad comes next with 51% organics and a lower gross domestic product (GDP)/cap than the former two cities. In comparison, Mexico City, Thessaloniki, and Ostrava (summer) have a relative low fraction of organic waste and a higher fraction of paper and plastic materials (Table 2). These characteristics reflect the different lifestyles and state of economic development of the different cities (Hoornweg and Bhada-Tata, 2012). Mexico City has the highest waste generation rate with 1.8 kg/[cap·day], followed by Bogotá with 1.2 kg/[cap·day] and Thessaloniki with 1.1 kg/[cap·day].

Current Metabolism of the Studied Cities

Fate of waste

The current waste metabolism of each of the selected cities is presented in Figs. 2 and 3 in the form of a Material Flow, and detailed flow values and rates are provided in Supplementary Table S2. The MFAs reveal low recycling rates even in high-income countries. The six studied cities do not recycle more than 10%, with about 1% in Keserwan District and almost 0% in Ahmedabad. Yet, the main difference is that in high-income cities, recycling is carried out by an official institution, whereas low-income cities rely on the informal sector. Also, refuse-derived fuel (RDF) application is limited, constituting less than 5% of the total waste stream in Ostrava, Mexico City, and Ahmedabad. Similarly, composting is currently limited (0% to 16%), despite the high organic content of the waste in most of the selected cities. In contrast, landfilling remains by far the most adopted method. Yet, it varies from 65% in Ostrava, with advanced waste sorting programs, and 95% in Bogotá, where sorting is practically absent.

Institutional and financial arrangements

The institutional setup and budget allocation are major determinants of the degree of advancement of a waste management program. One can discern major discrepancies between setups in low- and middle-income versus high-income countries. In the latter case, a clear structure of the institutional arrangement is set in place, with well-defined roles of the involved sectors (public, private, communities, and informal sector) (Schübeler *et al.,* 1996). On the contrary, low- and middle-income nations often deal with inefficient institutional structures that lack fundamental policies, adequate budget, and streamlined coordination. Table 3 presents a summary of the institutional arrangements of the cities in study.

In all considered cities, the ultimate authority for waste management is the Ministry of Environment—the governmental body in charge of issuing environmental laws and regulations. However, the extent of involvement of the different stakeholders varies. In some cities, like Ahmedabad, there is a participation at nearly all levels, including governmental institutions, research centers, and religious bodies. In comparison, other cities rely only on governmental institutions to run the waste management sector. Mexico City and Bogotá are divided into several zones (or delegations) to improve the efficiency of waste collection and transfer plan.

In the case of the industrialized cities of Thessaloniki and Ostrava, the situation is different because Greece and Czech Republic follow the ambitious European Directive 2008/98/EC and Circular Economy Package, which set specific targets to be achieved. One of the targets of the Directive is to reduce the disposal of biodegradable waste by 35% of the amount landfilled in 1995 and to achieve 40% of separate collection of biowaste. Targets on recycling and recovery are as well established by the European Commission. Therefore, there is

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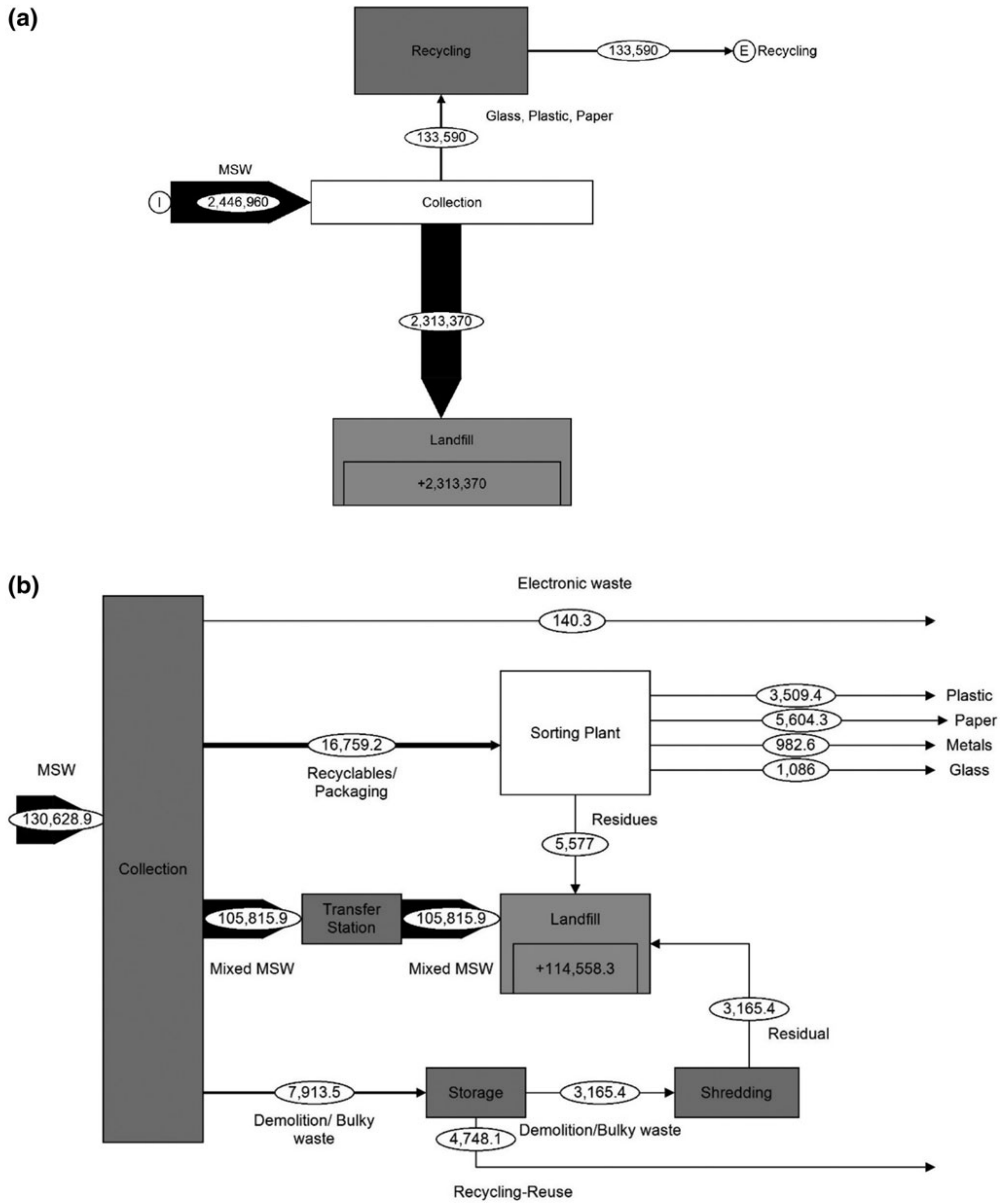

FIG. 3. Material flow analysis of baseline scenarios: **(a)** Bogotá, Colombia; **(b)** Thessaloniki, Greece; **(c)** Ostrava, Czech Republic. MSW, municipal solid waste.

(*continued*)

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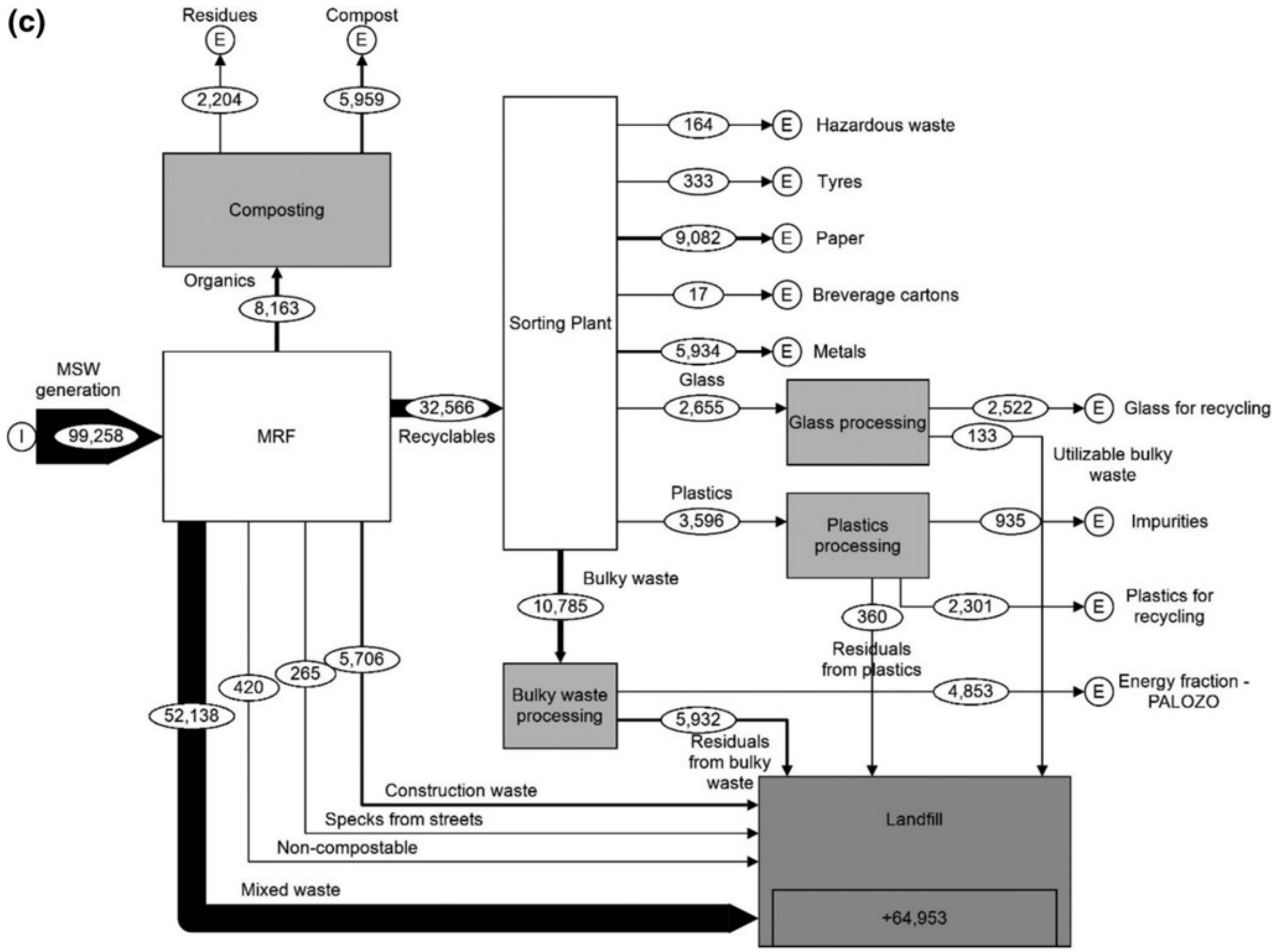

FIG. 3. *(Continued)*

an organized institutional structure that elaborates laws to adopt and implements the corresponding policies.

Prevailing challenges

Even though the developing cities in this study have an institutional structure in place to manage the waste, one of the common challenges is the overlapping functions and lack of communication between the entities, as is the case of Keserwan and Mexico City. In Keserwan, the responsibilities and duties of the different public authorities remain vague and highly interconnected, leading to less-than-optimum planning and poor management of the MSW sector (ELARD, 2004). Similarly, in Mexico City, inefficient communications delayed the implementation of the source separation law (introduced in 2003) to 2011.

Low collection rates are one of the manifold problems faced by developing countries (Hoornweg and Bhada-Tata, 2012). In many cases, low collection rates are associated with mismanagement of the already low budget allocated to the waste sector, in addition to the absence of policies. In Keserwan, the government controls the expenditures of municipalities. It pays directly, from the governmental budget allocated for municipalities, to private waste management companies (without consultation with the concerned municipality). This hinders local initiatives and potential improvements. The impact is aggravated by the fact that the current collection and dumping cost is considerably high (80 to 120 USD/Mg), leading to the depletion of municipality resources (Localiban, 2015). Funds and resources are a major hindrance in implementing a sound solid waste management system in Indian cities as well. Local governments do not have enough funds to carry out even the basic services like collection and safe disposal (Sharholy *et al.,* 2008). Even the very small available funds are often mismanaged. In the case of Bogotá, although collection of (mostly mixed) waste is close to 100% in the metropolitan area, it reaches only 30% in rural areas. As a consequence, uncollected waste is open burned or scattered on the streets and in open drains, causing health and environmental risks (Wiedinmyer *et al.,* 2014).

In addition, vulnerability is a key parameter for the durability and sustainability of the waste management solutions. The lack of diversified waste management methods and the lack of redundancy (dependence on one facility or one main method only) increase the vulnerability of the waste management sector of a city. In the case of Keserwan and Mexico City, governments do not impose any minimum fee or any constrain regarding the type or quantity of waste disposed into landfills. This situation makes

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Table 3. Summary of Institutional Framework of the Cities

City	*Institutional arrangements*
Ahmedabad	The stakeholders of waste management systems include local authorities, concerned ministries, and private contractors. The national and local authorities are the most influential stakeholders in terms of policy development and budget allocation, respectively. From an implementation perspective, the private contractors and the service users (households, civil organizations, and commercial and industrial sectors) play a major role. Other stakeholders include educational and research institutions, political parties, farmers (including poultries and fisheries), health care centers, media, donor organizations, the Chamber of Commerce and Industry, recycling companies, police, and religious leaders.
Mexico City	In Mexico City, the waste is managed by the local government, which sets common procedures and controls the transfer, separation, and landfilling of waste. Mexico City is divided into 16 delegations, each responsible for the collection and transportation of its own waste to the transfer stations. While the Secretary of Environment sets the regulatory framework, every delegation is in charge of the implementation in its area. The delegations have their own vehicles, routing systems, and collection frequencies.
Keserwan	In Lebanon, the distribution of responsibilities among the different authorities is overlapping and confusing (The World Bank, 2004), leading to poor management of the MSW sector. Stakeholders of the waste management sector include, in addition to the Council of Ministers who issue the environmental laws and decrees, the Ministries of Environment, Interior and Municipalities, Energy and Water, Public Works and Transport, Tourism, the Directorate General of Urban Planning, and the Council for Development and Reconstruction. The latter is in charge of signing the contracts with private waste management companies.
Bogotá	The City of Bogota is divided into six different zones to improve the logistics of waste collection. Since 2012, one public company takes in charge the collection of waste in three zones, and three private companies become in charge of the other three zones (JICA and UAESP 2013). The main legislation regulating this new strategy include the CONPES 3530, Decree 312/2006, and the Decree 564/2012. The Resolution 351/2005 regulates the tariffs related to the collection and sanitation services. The involved institutions include the Ministry of Housing, City and Territorial, Ministry of Environment and Development, National Department of Planning, National Council of Social and Economic Politics, Commission for the Regulation of Drinking Water and Basic Sanitation, The Special Administration Unit of Public Services, Mayor's office, and local Mayors.
Thessaloniki	In Greece, the Municipalities are responsible for the management of waste with regard to temporary storage, transshipment, treatment, and recycling. The waste management is realized based on the corresponding Regional Planning for Solid Waste Management (PESDA)* developed by each Prefecture. PESDA assesses the baseline of a region and develops a framework of initiatives and interventions, and sets targets and timetables for the design and operation of Organized Waste Management Facilities. Public limited liability companies (FODSA)* are formed, gathering all municipalities of the same Prefecture. FODSA are competent solid waste management bodies in charge of the implementation of the objectives and actions of the PESDA. They address temporary storage, transportation, processing, reuse, and final disposal of solid waste (Ministry of Public Order and Citizen Protection—General Secretariat for Civil Protection, 2013). The Ministry of Internal Affairs and the Ministry of Environment, Energy and Climate Change specify the waste treatment units, at the national level, needed to meet the requirements of the European Directive 2008/98/EC and the Greek Law 4042/2012 (Stouraiti, 2013).
Ostrava	The structure of the waste management plan of the Czech Republic, driven by the Waste Act § 41 and § 42, applicable EU directives, and methodological instructions of the European Commission (guidance note for waste management plan) issued in June 2012 (Ministry of Environment, 2014). The waste sector is managed by the Ministry of the Environment, Czech Environmental Inspectorate, regional authorities, municipalities (communal environmental offices), Ministry of Health, Ministry of Agriculture, National Hygiene Offices, Czech Trade Inspection, customs administration, and police (Ministry of Environment, 2014). Regional and municipal authorities are in charge of the implementation of waste management legislations. According to Act no. 383/2008 Coll., municipalities have the direct responsibility to manage the waste on their territory. Each community has its own collection system and waste processing and disposal facilities, embedded in a municipal ordinance, and financed by the municipal budget.

*FODSA and PESDA are the acronyms for the original Greek translations of these terms.
FODSA, public limited liability companies; MSW, municipal solid waste; PESDA, Regional Planning for Solid Waste Management.

landfilling an attractive, easy, and inexpensive option, and consequently, the only adopted solution to manage waste. Similarly, in some cases, like in Bogotá, only one landfill is available to serve the whole city. Even though this landfill was forecasted to last until year 2030, the increasing population growth might shorten its lifespan (JICA and UAESP, 2013). In these cases, the whole waste management sector of the city is dependent on this one facility/method. Any interruption or fluctuation in the performance of this only service provider would have catastrophic results—like the case of Keserwan during the Lebanese waste crisis in 2015.

The lack of expertise is another major hindrance in MSWM of developing cities. Specifically, in Ahmedabad, the responsibility of solid waste management, at the level of local governments, is usually assigned to a staff member of low competence (Guerrero *et al.,* 2013). Also, the technical staff are seldom adequately trained to perform their assignments. In addition, awareness campaigns and educational programs are often missing and local communities are unaware of the right behavior required for the success of the waste management plan.

In general, the penetration rate and efficiency of source prevention and separation remain limited in most discussed cities—a common challenge in developing countries (Liu *et al.,* 2017). As an indirect result, the ''informal'' recycling sector plays an important role in management of waste, however, at a lower extent in higher income cities (e.g., Ostrava and Thessaloniki). In general, some improvement in material recovery is observed by involving the informal recycling stakeholders into the ''formal'' MSWM plan. This adds complexity to the already overlapping systems and leads, in many cases, to contradictory opinions.

On the other hand, the current challenges in Ostrava and Thessaloniki are quite comparable because both, Greece and Czech Republic, have to fulfill the European Union (EU) targets. However, the ongoing financial crisis (since 2009) in Greece has affected, among others, the waste management sector in Thessaloniki. The available budget for infrastructure is limited and, in many instances, below the basic requirements of the municipalities for everyday activities.

Finally, a gap is observed between cities in high-income and low-income countries in terms of governance, institutional capacity, and waste-related policies. Institutions are more organized, with properly defined roles, in Thessaloniki and Ostrava, and, to a certain extent, in Bogotá and Mexico City, compared to Keserwan and Ahmedabad. Furthermore, there are clearer and more stringent waste reduction and containment targets in Thessaloniki and Ostrava, compared to other cities—mostly due to the commitments of the EU.

Anticipated Improvements in the Studied Cities

Based on the outcomes and prevailing drivers and challenges of the current waste management practices, the future approaches, planned by the respective governments in each city, are analyzed with the aim of providing a second-level comparison. The MFAs do not reflect exactly the official governmental projects, but the interpretation of the authors (Table 4). The waste flows are calculated in Mg/year.

Ahmedabad, India

Land scarcity seems to be the primary determinant in future planning in Ahmedabad. The city authorities are planning for reduction of waste volume by installing an incinerator with a capacity of 1,000 Mg day-1. For a better exploitation of resources, and to enhance the public image of

Table 4. Challenges and Alternatives to Improve Solid Waste Management Systems

City	*Current methods*	*Challenges*	*Suggested alternatives*
Ahmedabad	RDF (4.2%) Composting (9.4%) Controlled dumping (86.4%)	• Incomplete collection of waste • Challenges in developing countries[a]	• Incineration • landfill for ashes
Mexico City	Material recovery formal sector (3%) Composting (16%) Landfill (81%)	• Inefficient collection of sorted waste • Land scarcity • High transportation cost • Challenges in developing countries[a]	• Improved separation • Incineration before landfilling
Keserwan	Material recovery (10%) Controlled dumping (90%)	• Land scarcity and high land cost • Lack of quality control • Challenges in developing countries[a]	• Governmental plans for incineration • Local authority plans: Material recovery, composting, RDF
Bogotá	Material recovery (10%) Landfilling (90%)	• Inefficient source separation • Land scarcity and high land cost • Lack of coordination between stakeholders (informal sector) • Challenges in developing countries[a]	• Transfer stations • Improved sorting • New landfills
Thessaloniki	Material recovery (10%) Landfill (90%)	• Economic crisis • No organic sorting or treatment	• Source separation • Composting
Ostrava	RDF (8%) Material recovery (19%) Composting (8%) Landfill (65%)	• Low waste management fees • Lack of bans on landfills • Spreading of landfilling • Lack of energy recovery from waste • Separate organic waste collection covers only garden waste	• Improved separation for recycling • Incineration before landfilling • Reduction of biodegradable waste deposited on landfill

[a]General challenges for developing countries: lack of funds, poor planning, poor implementation of the law, and lack of expertise.
RDF, refuse-derived fuel.

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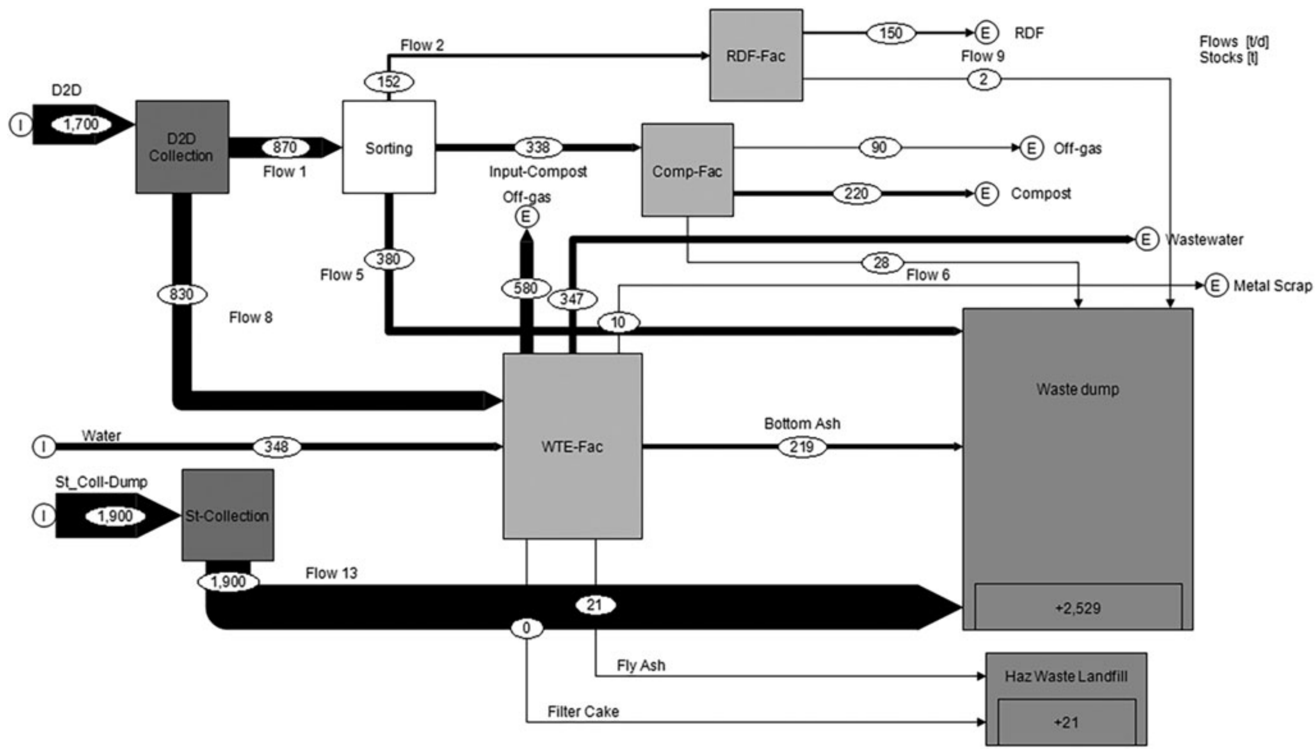

FIG. 4. MFA of alternative MSWM plan in Ahmedabad, India. MFA, material flow analysis; MSWM, municipal solid waste management; WtE, waste to energy.

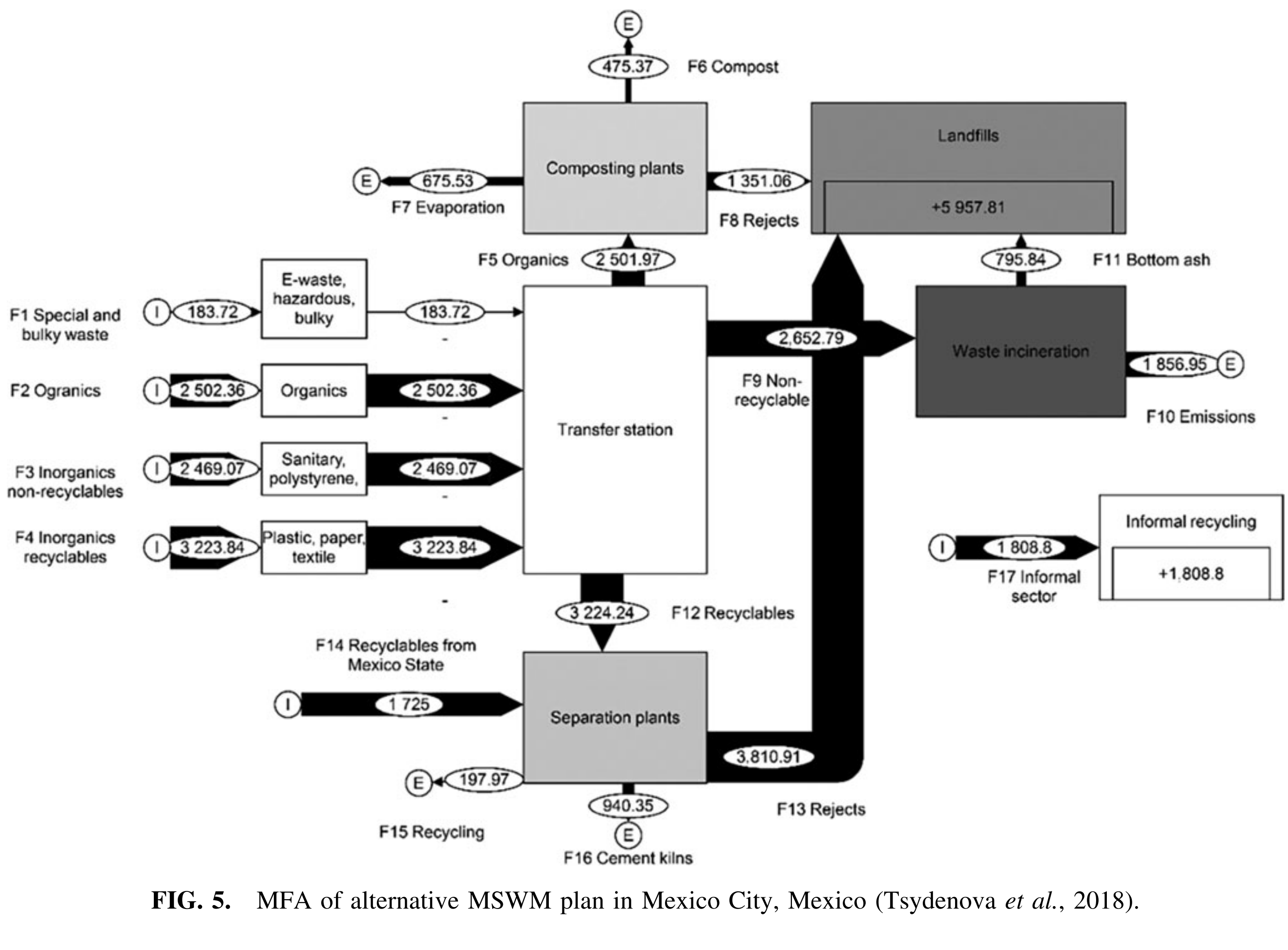

FIG. 5. MFA of alternative MSWM plan in Mexico City, Mexico (Tsydenova *et al.*, 2018).

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the plan, the authorities intend to recover energy from the incineration process. Despite the claimed advantages, Indian nongovernmental organisations (NGOs) and the general public are strongly opposing the waste-to-energy (WtE) plant. This is mainly due to the following: (1) the lack of confidence in the government capacity to implement environmental standards; and (2) the unsuitable composition of waste, with high organic proportion and elevated moisture content—typical of developing countries (Banar and Özkan, 2008). The MFA of the proposed plan is shown in Fig. 4.

Mexico City, Mexico

Similar to Ahmedabad, Mexico City is planning to reduce the volume of generated waste by incineration (Americas, 2017). The main driver behind this decision is the reduction of transportation costs. Since there is no possibility of opening a new landfill in the city itself, the proposed plan reduces the amount of waste to be landfilled outside the city from 8,000 to 2,600 Mg/day. However, the technology is strongly opposed by the local NGOs and general public. The proposed material flow is shown in Fig. 5. After the change of government at the end of 2018, the plan to construct an incineration plant in the city was abandoned (Cuenca, 2019).

Keserwan District, Lebanon

Recently, the Lebanese government ratified an MSW law that allows decentralization and recognizes WtE (incineration) technologies as viable treatment methods, which was strongly opposed by NGOs (Sidahmed, 2015). Similar to the case of Ahmedabad, the reasons for public rejection are, among others, (1) high content of food waste and moisture in the Lebanese waste; (2) convenient weather (moderate temperature and high humidity) for biological decomposition of the organic fraction (50–60%) of the waste; (3) unjustified high cost of incinerators, compared to other viable solutions; and (4) lack of a trustworthy quality control and monitoring of air pollution.

Under the current circumstances, local authorities and municipalities have a focal role in shaping the future of SWM in the country. Recently, the district of Keserwan is determined to initiate an independent SWM system. The currently suggested plan is to have enforced source separation into two categories: ''biodegradable'' waste and ''rest,'' with collection points for bulky and special wastes. The district is launching a private-public partnership to build a facility to

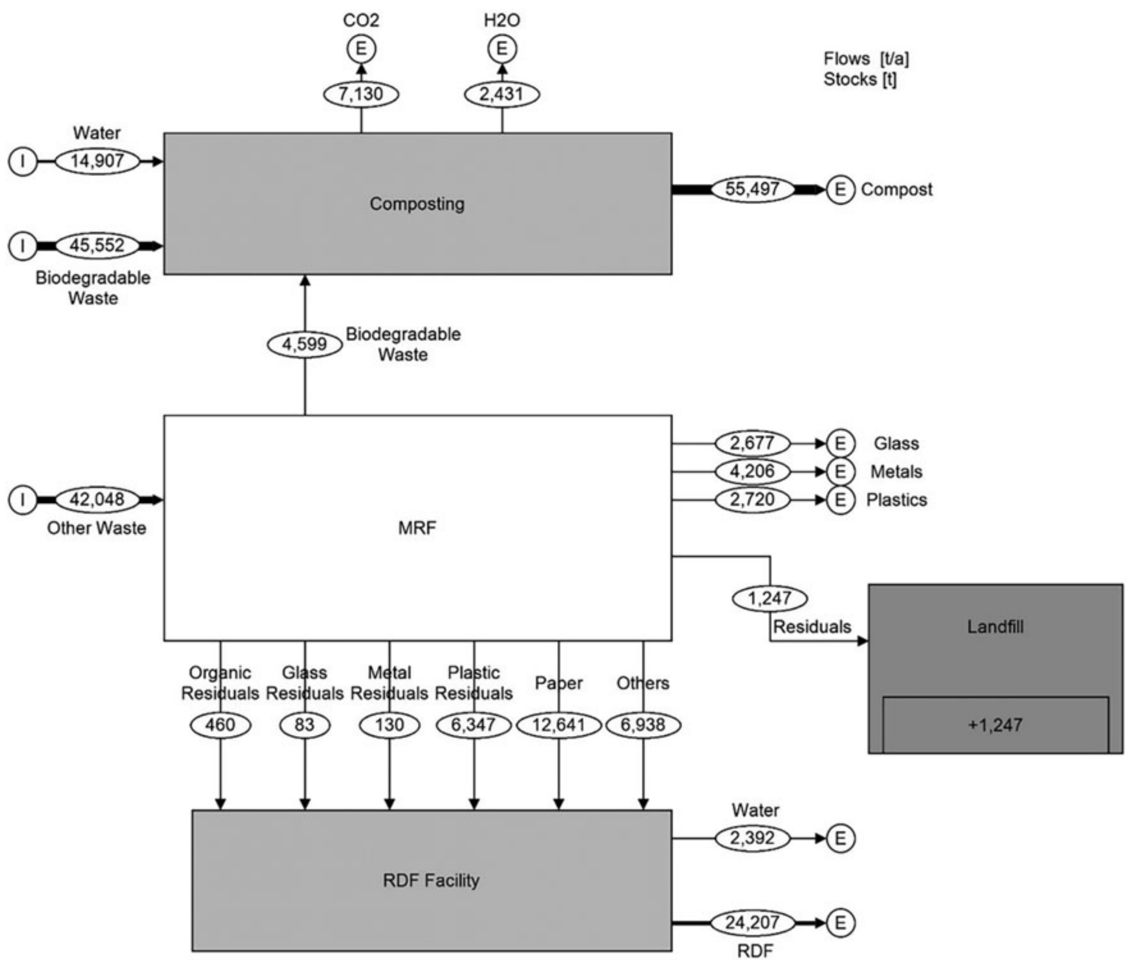

FIG. 6. MFA of alternative MSWM plan in Keserwan, Lebanon.

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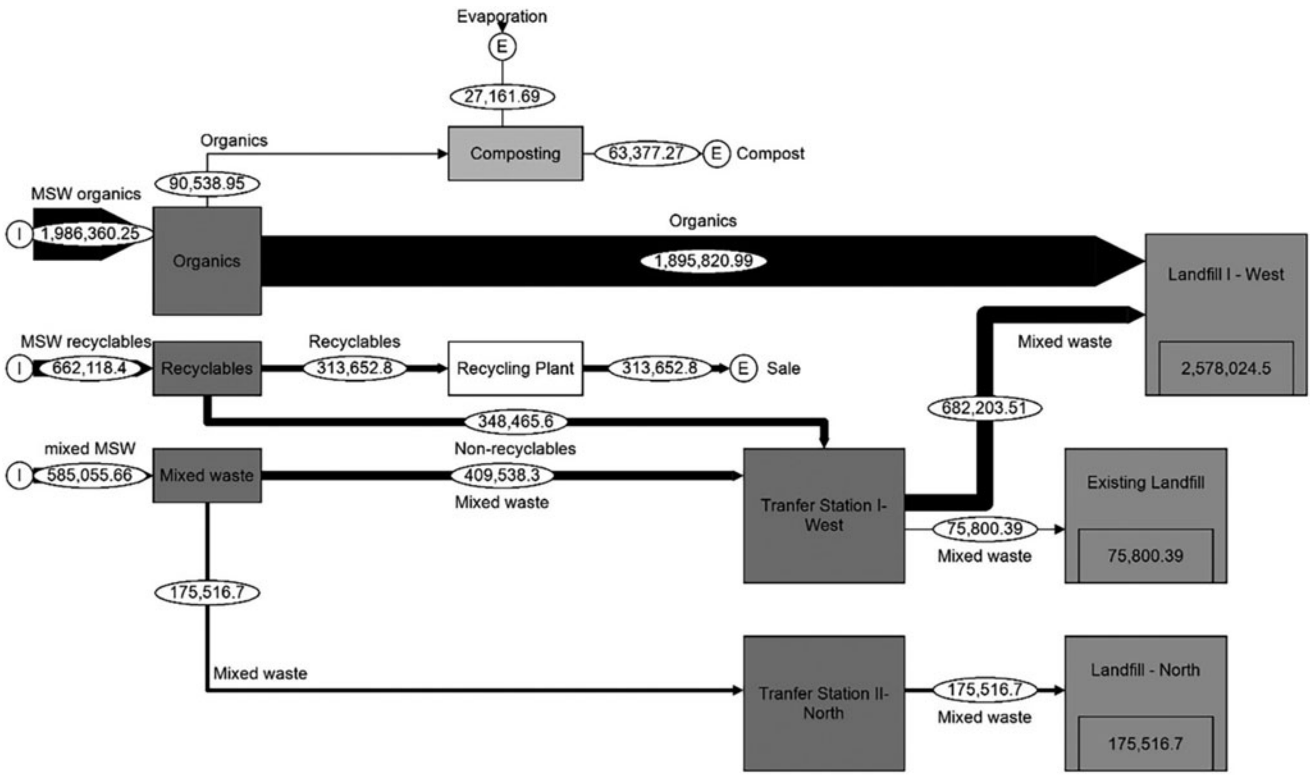

FIG. 7. MFA of alternative MSWM plan in Bogotá, Colombia.

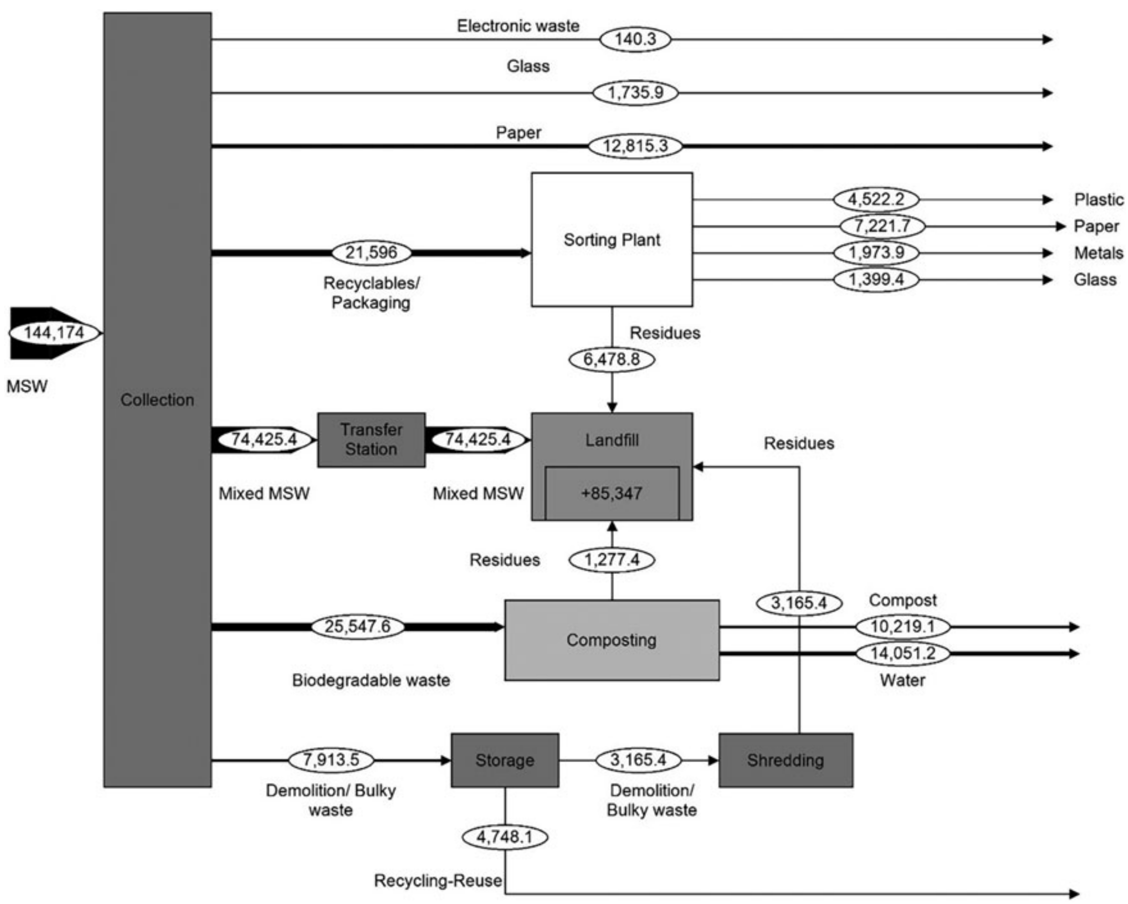

FIG. 8. MFA of alternative MSWM plan in Thessaloniki, Greece.

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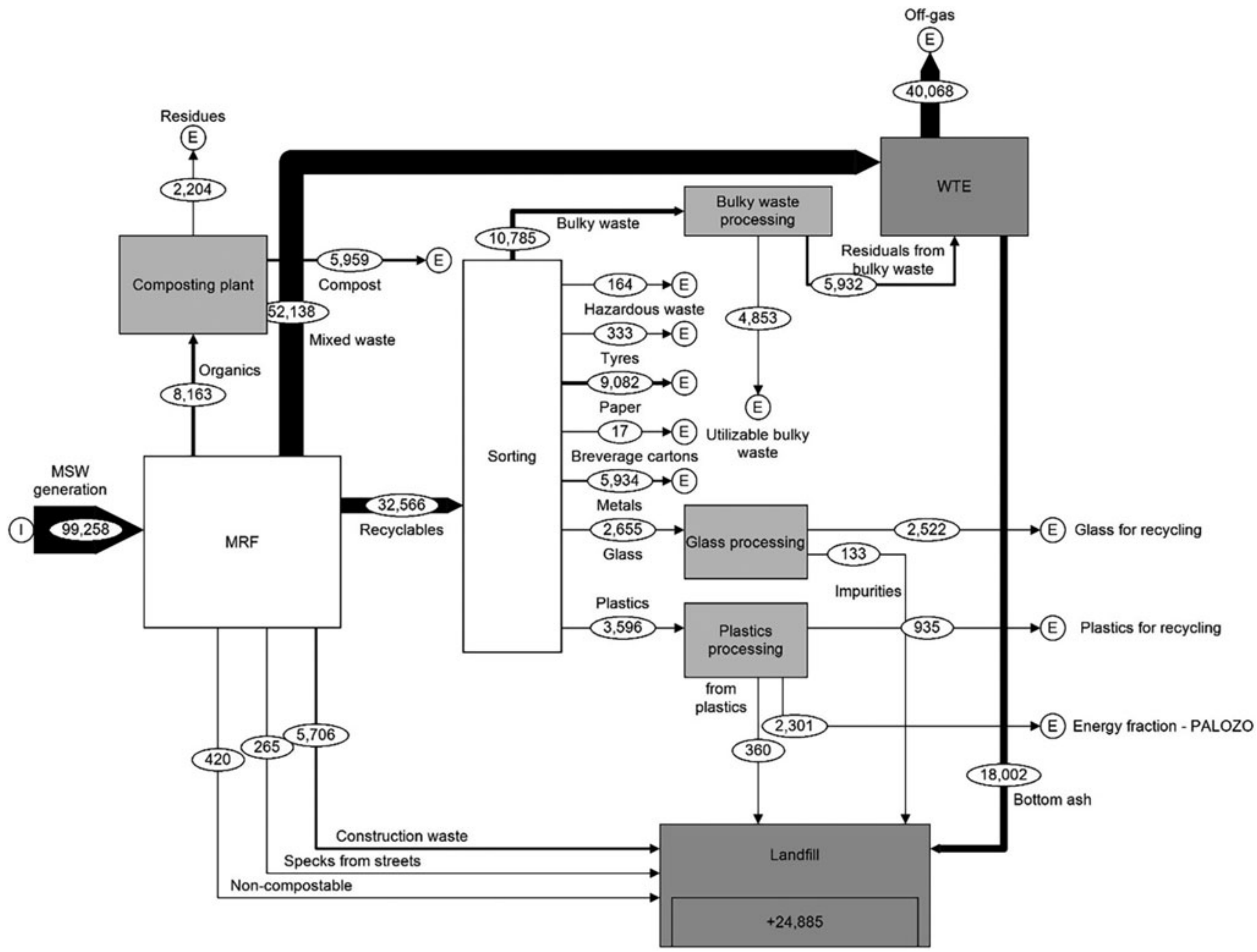

FIG. 9. MFA of alternative MSWM plan in Ostrava, Czech Republic.

sort the recyclables, compost the organics and transform the rest of the waste into RDF (Fig. 6).

Bogotá, Colombia

The Master Plan for the Integral Management of Solid Waste of Bogota provides a comprehensive strategy for the next years, until 2027. Considering the projected increase in population (to 9.1 million) and waste generation (3,230 Mg yearly), the plan focuses on the following: (1) reduction of transportation costs, by establishing two transfer stations; and (2) improving the efficiency of the recycling plant. The MFA in Fig. 7 represents the suggested plan.

Thessaloniki, Greece

To comply with the goals of the EU, the suggested plan encompasses separate collection of biowaste, paper, glass, and electronic waste (Fig. 8). The plan for a waste generation rate of 144,174 Mg/year by 2020 includes a composting facility, with a capacity of 25,000 Mg/year, for the treatment of the separately collected biowaste stream. Based on the targets set in the Local Solid Waste Management Plan (Municipality of Thessaloniki, 2016), 19,900 Mg/year of separately collected paper and 2,500 Mg/year of separately collected glass are taken into account.

Ostrava, Czech Republic

Similar to Thessaloniki, Ostrava needs to fulfill the European Commission directives. Thus, the city is required to ban landfilling of mixed MSW by year 2024. To meet this requirement, the government started to build a WtE plant in the Moravian-Silesian region in 2004 (Fig. 9). The project was planned to be finished in 2010. Despite the efforts of the local authorities and the shareholders of the project, and the approval of the Environmental Impact Assessment Agency, the project has not been implemented yet. For the successful realization of the project, it is necessary to find a suitable strategic partner.

Summaries and Recommendations

The challenges observed in developed countries are addressed from two perspectives: institutional and technical. From an institutional point of view, the absence of policies and, most importantly, the overlapping and inefficient structure of the waste management sector draw a visible line between poor and developed MSWM systems. In high-income cities, like Thessaloniki and Ostrava, every single type of waste is handled by a well-specified entity and follows a clearly defined path. Accordingly, waste solutions should start by defining ''who does what?'' and ''what goes where?'' For each answer to these questions, a set of rules, laws, and code of practices needs to be developed.

From a technical standpoint, two main challenges were identified: incomplete collection of waste and land scarcity in cities. While the solution for the first is obviously budgetary, the solution for the second is not straightforward. Land scarcity is often expressed as ''high cost of transportation'' to remote landfilling areas, which is translated into the ''need to reduce the waste volume.'' The latter is often misconceived by policy makers as the ultimate target, to be reached by any means.

In fact, a closer look at the waste management systems of the discussed cities shows that none of them follows the well-known ''waste management hierarchy,'' in which waste minimization and diversion have priority against disposal (Hoornweg and Bhada-Tata, 2012). Even in Thessaloniki and Ostrava, where the importance of the waste hierarchy is highlighted in the European Directive 2008/98/EC and Circular Economy Package, waste management plans do not seem to tackle waste reduction directly.

Instead, most cities seem to be shifting toward incineration as a mean to reduce the volume of landfilled waste (Supplementary Table S3). While this approach has several advantages, including energy recovery and reduced health hazards of improperly disposed waste, it triggers several alarming burdens. Policy makers see incineration as a ''shortcut'' solution that eliminates/reduces the need for separate collection and/or adequate sorting of waste. However, lack of adequate sorting might result in problematic operation of the WtE plant, especially in developing cities where more than 50% of the waste is organic.

However, the main alarming fact remains the lack of stringent air quality standards and air pollution regulations. This reduces the overall capital and operational costs, because of minimal requirements for air pollution control equipment, and makes it an attractive alternative for investors. However, this leads to inefficient and unreliable supervision and control measures. As a result, higher air pollution impacts are expected in developing cities, which already have high pollution rates (Lu *et al.,* 2017). As a result, a high public opposition is faced in several cities, including Ahmedabad, Keserwan, and Mexico City. In comparison, Bogota, which had a previous unfortunate experience with incineration, is substituting it with simpler solutions. Considering the high cost of the air pollution control system, and the lack of sufficient governmental funds, Bogota decided to establish two transfer stations and construct two more landfills in remote locations (JICA and UAESP, 2013).

Similarly, from a precautionary perspective, the National Waste Management Plan of Thessaloniki (Greek Republic, 2015) avoided thermal treatment of waste and considered it to be a process of high environmental nuisance potential. Instead, the plan opted for ''Energy Recovery'' and ''Energy Utilization of Waste'' concepts that have milder environmental burden. Those include biochemical processes that produce secondary gases or liquid fuels, including biogas recovery from landfills, anaerobic degradation, and biodiesel production from waste oils.

On a final note, the concepts that seem to be mostly undermined in developing cities are as follows: (1) source segregation and separate waste collection, and (2) energy recovery by biochemical technologies. Waste separation, which is often considered impractical, requires long-term planning for behavioral changes. It is efficient only if appropriate regulations are developed and stringently implemented. Also, most cities adopt composting for the treatment of organic waste. Even though composting is economically preferred at small scales, anaerobic digestion can be more feasible at larger scales (Chynoweth *et al.,* 2001).

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Author Disclosure Statement

No competing financial interests exist.

Supplementary Material

Supplementary Table S1
Supplementary Table S2
Supplementary Table S3

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Carbon in global waste and wastewater flows – its potential as energy source under alternative future waste management regimes

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Abstract. This study provides a quantification of the maximum energy that can be generated from global waste and wastewater sectors in the timeframe to 2050, as well as of the potential limitations introduced by different future waste and wastewater management regimes. Results show that considerable amounts of carbon are currently stored in waste materials without being recovered for recycling or made available for energy generation. Future levels of energy recovery when maintaining current states of waste and wastewater management systems are contrasted with those that can be attained under a circular system identified here as a system with successful implementation of food and plastic waste reduction policies, maximum recycling rates of all different types of waste streams, and once the recycling capacity is exhausted, incineration of remaining materials to produce energy. Moreover, biogas is assumed to be produced from anaerobic co-digestion of food and garden wastes, animal manure, and anaerobically treated wastewater. Finally, we explore the limits for energy generation from waste and wastewater sources should the efficiency of energy recovery be pushed further through development of existing technology. We find that global implementation of such an ideal system could increase the relative contribution of waste and wastewater sources to global energy demand from 2 % to 9 % by 2040, corresponding to a maximum energy potential of 64 EJ per year. This would however require widespread adoption of policies and infrastructure that stimulate and allow for large-scale waste prevention and separation, as well as highly advanced treatment processes. Giving priority to such efforts would enable circularity of the waste-energy system.

1 Introduction

The continuous increase of anthropogenic pressure on the environment has brought different disciplines together with the common objective of finding holistic solutions to reduce, mitigate and/or adapt to the negative impacts of human activities. The concept of a circular economy has emerged as a strategy to cope with uncontrollable and unsustainable consumption rates of today's society (Haas et al., 2015). In that context, sustainable waste and wastewater management systems play a significant role in contributing to reduce air and water pollution as well as to decarbonization of the energy system through reducing, reusing, recycling and recovering part of the energy embodied in waste materials and wastewater (Corsten et al., 2013). Various case studies quantifying energy and greenhouse gas emission reductions from waste have been carried out for specific regions, often focusing on a specific management technology, e.g. energy from anaerobic digestion in United Kingdom (Evangelisti et al., 2014), methane generation potential from landfills in India (Mor et al., 2006), determination of fossil carbon in Swedish waste (Jones et al., 2013), energy from waste in the Netherlands (Corsten et al., 2013) or GHG emissions from different waste management technologies in China (Liu et al., 2017). Regarding wastewater, different case studies have shown that anaerobic digestion with biogas utilization can offset the energy consumption in the wastewater treatment process (McCarty et al., 2011; Stillwell et al., 2010). Unique for this study is its wide scope; we estimate global carbon flows from waste and wastewater sources from both domestic and industrial sectors. Previously, the global energy potential from municipal waste has been estimated at 8–18 EJ in 2010 and 13–30 EJ in 2025 (Bogner et al., 2008).

Different waste and wastewater management pathways and policies would have different social, environmental and

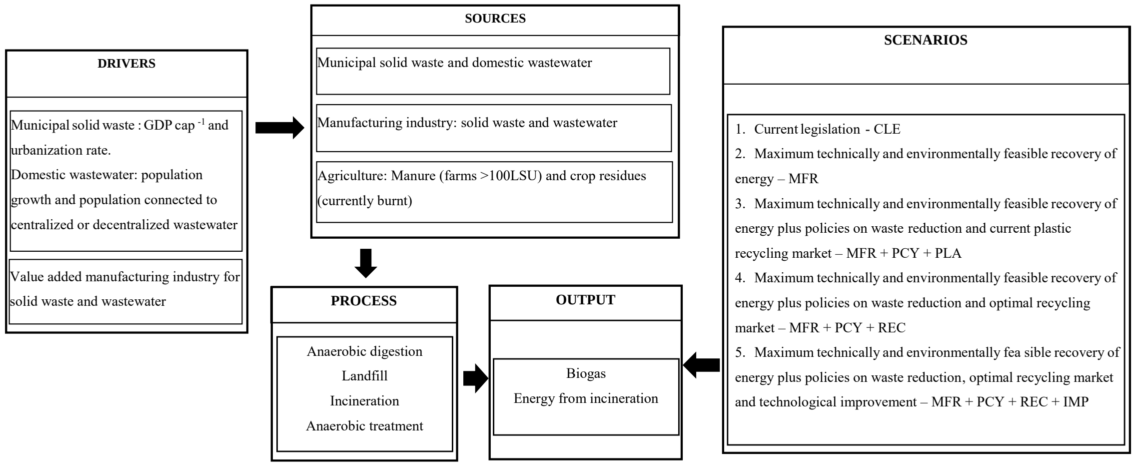

Figure 1. Research framework approach (GDP: Gross Domestic Product, LSU: Livestock Units).

economic impacts. While in the developed world the focus of the management systems have moved towards resource efficiency, developing countries are still facing problems to cope with the large volumes of waste and wastewater generated (Manaf et al., 2009). This has been attributed to financial, technical and institutional problems at the local and national levels (Pokhrel and Viraraghavan, 2005). However, if an economy grows accompanied by enforcement of environmental policies focused on the circularity of the system, climate, health and other environmental impacts caused by poor waste and wastewater management systems could be tackled simultaneously (Ghisellini et al., 2016). Therefore, an examination of the current state and an exploration of future waste and wastewater management alternatives is needed in order to identify an adequate strategy to achieve the maximum benefits for a growing economy. Accordingly, the overarching goal of this study is to investigate the maximum potential contribution of the global waste and wastewater sectors to the decarbonization of the global energy system, as well as to quantify potential limitations on energy recovery from these sources introduced by possible future waste and wastewater policies. The analysis rests on detailed country-/region-specific estimations of the carbon content in current waste flows with simulations of future carbon flows for a range of different waste and wastewater management regimes.

2 Methods

A research framework approach to estimate the current and future carbon content and the maximum energy potential from waste and wastewater at a global level up to 2050, is presented in Fig. 1.

The following section presents a summary of the approach to project industrial/municipal waste and wastewater generation (Sect. 2.1), followed by a short explanation of carbon content determination and maximum energy potential calculation (Sect. 2.2), and then an outline of the assumptions behind the development of the waste and wastewater management scenarios focused on the maximum technically and environmentally feasible recovery of energy (Sect. 2.3). Key assumptions for carbon content, biogas and energy-recovery calculations are presented in Table S1 and equations applied for the different calculations are presented in the Supplement Sect. S2. Furthermore, a section describing the limitations and uncertainty of the waste and wastewater management scenarios is presented in the Supplement Sect. S2.4.

2.1 Wastewater and solid waste projections up to 2050

2.1.1 Industrial wastewater

Industries with a high carbon load in the wastewater can potentially generate biogas when the wastewater is treated under anaerobic conditions (IPCC, 2006). The main industrial sectors considered here as generators of wastewater with a high carbon load are the food industry and the pulp and paper industry. In addition, we consider in a joint category named "other industrial sectors" wastewater with a significant carbon content, i.e. organic chemicals, textile, and leather industries. Activity data used to estimate biogas generation is the content of organics in the wastewater expressed in COD (Chemical Oxygen Demand) terms. Driver for future projections of industrial wastewater (COD content) is the growth in value added in the respective manufacturing industrial sector derived for the period to 2040 from the World Energy Outlook 2017 (International Energy Agency, 2017) and assuming the same annual growth rate between 2040 and 2050 as between 2035 and 2040. Elasticity parameters used for future projections are taken from Höglund-Isaksson (2012). Historical data on industrial production are retrieved from FAOSTAT (2016). Wastewater generation rates by type of industry are taken from different sources (refer to Table 17 in Höglund-Isaksson et al., 2015).

2.1.2 Industrial solid waste

Manufacturing industries considered in this study are food, pulp and paper, rubber, textile, wood and other manufacturing industry. Just like for industrial wastewater, the drivers for industrial solid waste generation projections are the expected growth rates in value added in the respective manufacturing industrial sectors, derived from the World Energy Outlook 2017 (International Energy Agency, 2017). Industrial waste generation elasticity parameters to value added are retrieved from Höglund-Isaksson (2012) and used to project industrial waste generation. Statistics on industrial waste generation quantities are taken from various sources (Supplement Table S2).

2.1.3 Domestic wastewater

Domestic wastewater is defined as wastewater from households (IPCC, 2006), however, may in some cases be mixed with small industry sources. Activity data to estimate biogas generation is the content of organics in wastewater expressed in BOD (Biological Oxygen Demand) terms. Biogas is quantified for domestic wastewater that is centrally collected and treated in a municipal sewage plant. Data on population projections are taken from the GAINS model which are based on data from IEA (International Energy Agency, 2017). Fractions of people connected to centralized/decentralized systems are retrieved from EUROSTAT (2016), OECD (2016) and World Bank Open Data (2016).

2.1.4 Municipal solid waste generation

A new methodology to project municipal solid waste generation and waste composition by income group was developed based on the assumption that average national waste generation rate and composition vary depending on the average national income level (Hoornweg and Bhada-Tata, 2012). Numerous studies (UNEP and ISWA, 2015; Hoornweg and Bhada-Tata, 2012; SWEEPNET, 2012; Wilson et al., 2012) indicate that composition of municipal solid waste depends on socio-economic characteristics, geographical location and environmental features. Paper and plastic wastes are the main fractions of MSW in high-income countries, while food waste dominates in low income countries (Hoornweg and Bhada-Tata, 2012). The drivers used here to project future municipal solid waste generation are GDP per capita and urbanization rate (for extended dataset description and elasticity estimation models, see Supplement Sect. S2). Furthermore, due to the fact that waste composition influences the carbon content and hence the energy recovery potential, projections of waste composition are needed. For future years, the composition of waste is recalculated based on an estimated elasticity of per capita food waste to GDP per capita (for elasticity estimation models see Supplement Sect. S2). After projecting the future generation of food waste per capita, other types of waste are projected to make up the rest of total per capita MSW generated with the relative contribution of non-food waste in 2015 kept constant in future years.

2.2 Carbon content determination and energy-recovery potential

2.2.1 Solid waste

Waste generation quantities and waste composition determine the availability of the carbon content to produce energy. Different waste categories contain different fractions of degradable organic carbon (DOC) and fossil carbon (FC) (IPCC, 2006, vol. 5, chap. 2). At the same time, waste composition determines (to a certain extent) the type of management. In order to quantify the carbon content in industrial and municipal solid waste and the respective flows, the following approach is used (calculations are always carried out globally at the level of 174 countries/regions and with annual results presented for every five years):

- Quantification of DOC and FC in municipal and industrial solid waste using IPCC default values for DOC and FC (IPCC, 2006).

- Identification by country/region of the application rate of current (and future) waste management technologies/systems (EUROSTAT, 2016; OECD, 2016, UNFCCC CRF Tables (2016) and documents referenced in Supplement Table S8). This study distinguishes various management options for each of the solid waste fractions. Description of each of the options can be found in the Supplement Sect. S2.2.1, Table S9. The assessment of the carbon flows is then carried out applying Eqs. (S1) and (S2).

- Estimation of energy recovery from municipal and industrial solid waste: This study identifies anaerobic co-digestion, landfill with gas recovery and use, and waste incineration as the three main treatment technologies to convert waste into a source of energy.

Anaerobic co-digestion

In order to improve the efficiency of the biogas formation processes, different degradable sources are typically co-digested (Berglund and Börjesson, 2006; Singh et al., 2001). Therefore, in addition to food waste (municipal and industrial), manure from dairy cows, non-dairy cattle and pigs that are kept on farms with more than 100 Livestock Units (LSU) and using liquid manure management systems (Höglund-Isaksson, 2015), and agricultural crop residues that would otherwise be openly burned, are included as extra substrates to be co-digested with the waste. Information on manure generation and agricultural crop residues consistent with long-term agricultural projections from FAOSTAT (2012) are taken from the GAINS model. It is assumed that the feedstock for biogas generation contains 80 % manure and 20 % other organic substrate with a water content of 85 %. No trade or exchange of substrates between countries/regions is considered. In cases/countries where one of the substrates is not available or there is a surplus (no needed for co-digestion) in one of them, biogas from single substrate digestion is also considered, albeit adjusting for a lower biogas yield in cases when only manure is digested. Biogas generation is calculated using Eq. (S3) (based on Höglund-Isaksson, 2015) and Eq. (S4).

Landfill

Landfill gas generation is accounted for with a lag of 10 years for fast degrading organic waste and 20 years for slow degrading waste. Landfill gas generation is calculated using Eq. (S5) based on IPCC (2006, vol. 5, chaps. 2 and 3).

Incineration

Energy from incineration is calculated using the Low Heating Value (LHV) of each of the waste fractions. LHV represents the usable heat released from waste and varies according to waste type (Demirbas, 2004). Energy from incineration is calculated using Eq. (S6).

2.2.2 Wastewater

Wastewater generation quantities and composition determine the capacity to generate biogas when treated under anaerobic conditions. Biogas from wastewater treatment is calculated based on the COD content for industrial wastewater and BOD content for domestic wastewater. It is important to notice that in this study the focus of the wastewater treatment is the removal of organic content by using anaerobic treatment as a process to generate biogas. The subsequent wastewater treatment to remove different pollutants depends on the respective legislation at the country level. In order to quantify the organic content in industrial and municipal wastewater and its respective flows, the following approach is used (calculations are carried out by country/region and year):

- Quantification of BOD in untreated domestic wastewater and COD in untreated industrial wastewater using the IPCC method (based on IPCC, 2006, vol. 5, chap. 6, Eqs. 6.4 and 6.6).
- Identification by country/region of the application rate of current (and future) use of wastewater management technologies/systems (EUROSTAT, 2016; OECD, 2016 and documents referenced in Supplement Table S7). This study distinguishes various wastewater management options for each of the two wastewater types. A description of each option can be found on the supplement material Sect. S2.2.2, Table S10. The assessment of the organic material flows is then carried out applying Eqs. (S7) and (S8) based on Höglund-Isaksson et al. (2015).
- Estimation of the energy potential from domestic and industrial anaerobic wastewater with gas recovery. Volumes of biogas from industrial and domestic wastewater treatment are calculated by applying Eq. (S9).

2.3 Waste and wastewater management scenarios

Presented estimates (see Sect. 3) assume a maximum technically feasible phase-in of waste management (in consistency with EU's waste management hierarchy – Directive 2008/98/EC) and wastewater treatment technologies that generate energy while reducing greenhouse gases, air pollution and water contamination on the basis of the circular economy strategy. Five different sets of waste and wastewater management strategies are developed. Implications of costs to implement various strategies are not considered in this analysis. Description of the measures adopted for the different scenarios are presented in the supplement Sect. S2.3.

- *CLE "current legislation"*: The scenario assumes efficient implementation of the existing waste/wastewater legislation. In countries/regions where no waste legislation exists – CLE – represents the current waste management situation.
- *MFR "maximum technically feasible phase-in of waste and wastewater management"*: A scenario that assumes the implementation of the "best available technology" to improve waste and wastewater management systems without regarding costs but considering constrains that could limit the applicability of certain technologies.
- *MFR + PCY + PLA "maximum technically feasible phase-in of waste and wastewater management" + "policy implementation + "plastic incineration"*: The scenario adopts the MFR + policies for reducing the generation of food and plastic municipal solid waste + maintains current municipal plastic waste recycling rates and sends excess plastics to incineration for energy recovery.
- *MFR + PCY + REC "maximum technically feasible phase-in of waste and wastewater management" + "policy implementation" + "maximum recycling capacity"*: This scenario adopts the MFR + PCY + reaches the maximum possible recycling capacity for all waste streams. For wastewater, the scenario includes a capacity to increase treatment of wastewater in urban areas.
- *MFR + PCY + REC + IMP "maximum technically feasible phase-in of waste/wastewater management" + "policy implementation" + "maximum recycling capacity" + "technology efficiency improvement"*: This

scenario adopts the MFR + PCY+ REC + technological development to increase biogas yield formation and to reduce losses during the treatment process for both solid waste and wastewater. Improvements include e.g. adding accelerants (biological or chemical) to improve the metabolic conditions for microorganism growth and therefore biogas formation (Mao et al., 2015), recovery of the dissolved methane in wastewater, and improvement of the biogas recovery rates. For incineration, improvements include an increase of the Low Heating Value, increase in the efficiency of input/air flows and reduction of energy losses during the process.

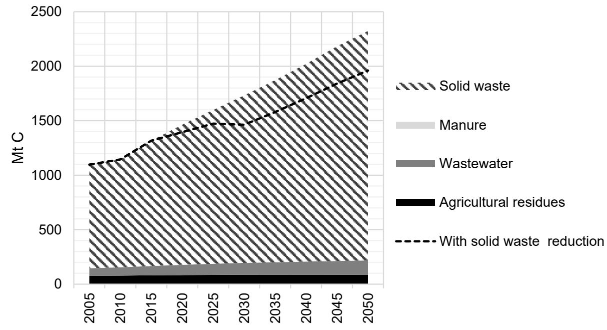

Figure 2. Projected global carbon available.

3 Results

In this section, a summary of the key results at a global and regional level in terms of carbon content and energy recovery from solid waste and wastewater are presented. Regions are aggregated into five groups using the Global Energy Assessment classification (GEA and IIASA, 2012). These groups are: UNFCCC Annex I countries (OECD), Easter Europe and Former Soviet Union (REF), Asia (excluding OECD), Middle East and Africa (MAF) and Latin America and the Caribbean (LAC).

3.1 Total global carbon availability

The availability of carbon in waste and wastewater allows for the quantification of the maximum potential of waste and wastewater as an energy source. Figure 2 shows the projected total carbon available from waste and wastewater sectors at a global level. Currently, the total global carbon in waste and wastewater is around 1400 Mt and is expected to be 2100 Mt in 2050. Municipal and industrial solid waste accounts for 87 % of the total carbon content while wastewater, agricultural residues (currently burned) and manure account for the rest 13 %. Manure accounts for < 1 % of the total carbon available. With future food and plastic waste reduction policies (strategies having an impact on the carbon content in waste), the availability of carbon is expected to be around 1900 Mt in 2050 which is 13 % less carbon compared to the current scenario in 2050.

3.2 Carbon flows in solid waste

Currently at a global level, we find that 59 % of carbon in waste is lost (dumped, scattered and openly burned without energy recovery), 18 % is recycled/composted and 23 % is converted to energy. 35 % of the carbon content in waste comes from industrial waste and 65 % from municipal solid waste. In the CLE scenario, an estimated 400 Mt-C is expected to be used annually as an energy source by 2050 (Fig. 3a). The largest losses of carbon are expected in the ASIA (46 %) and MAF (22 %) regions, where there is currently little or no waste management legislation in place. In low-income countries, collection rates are extremely low and waste disposal is often done in the form of uncontrolled dumpsites and open burning (UNEP and ISWA, 2015). The OECD region accounts for 90 % of the carbon globally converted into energy. OECD countries generate the highest amounts of waste per capita, however, most of the waste is properly managed with just 10 % of the carbon content lost. LAC and REF regions account for the last 21 % of the carbon currently lost at a global level. With a maximum technically feasible phase-in of waste management (MFR) implementation (Fig. 3b), we estimate that 66 % of the carbon in waste could be used to generate energy by 2050, through the use of anaerobic co-digestion or incineration. The global carbon converted annually into energy would then be around 1370 Mt by 2050. Carbon going to landfills with gas recovery until 2030 would serve as a source of energy for some years thereafter. In this scenario (MFR), the share of carbon content used as energy by 2050 would be better distributed between regions having ASIA and OECD countries with around 60 % of the global share (∼ 30 % each). With the implementation of food and plastic waste reduction reaching 50 % in 2030 on top of maximum technical implementation (MFR + PCY + PLA), the availability of total carbon in waste at a global level is expected to be reduced by 18 % in 2050 (Fig. 3c). Nonetheless, the flow of carbon going into the energy sector is reduced by plastic recycling (keeping the current rates of municipal plastic waste recycling) resulting in a global carbon converted into energy of around 1300 Mt by 2050 which is 5 % less than the carbon available in MFR. Although at a global level not significant effect on the carbon into energy flow is observed with the MFR + PCY + PLA scenario, at a country level the situation varies depending on the current level of plastic recycling. If a country has low recycling rates, even with the plastic reduction measure, more plastic would go into incineration. On the contrary, if a country has high plastic recycling rates, the reduction in plastic waste generation would affect the carbon flow to incineration. With an optimum recycling market on top of maximum technical implementation and plastic and food waste reduction policies (MFR + PCY + REC), we es-

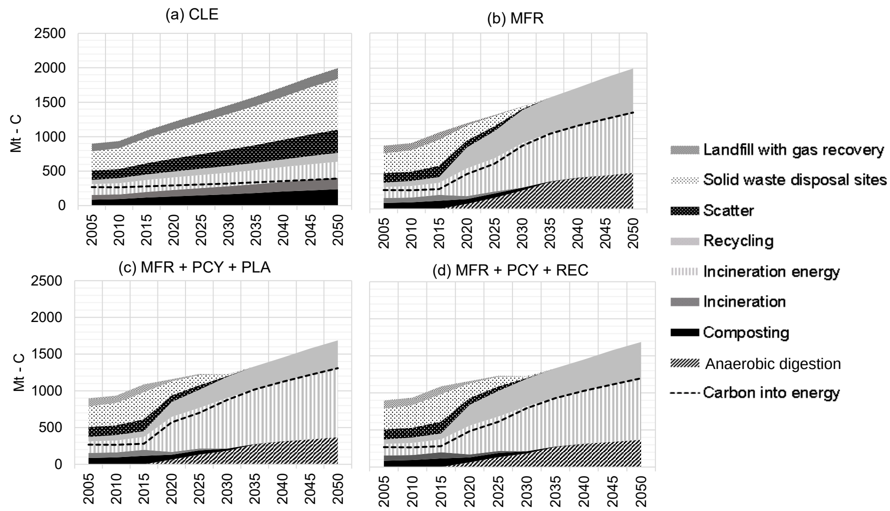

Figure 3. Global carbon content flows in solid waste by scenario.

timate that the total carbon converted into energy would be around 1200 Mt, which is 14 % and 9 % less compared to the MFR and MFR + PCY + PLA, respectively (Fig. 3d). Carbon content flows for the three different scenarios by region are presented in the Supplement in Fig. S2.

3.3 COD and BOD flows in wastewater

Currently at a global level, we find that 57 % of domestic wastewater and 38 % of industrial wastewater is either untreated or discharged after primary treatment. Most of the wastewater is treated under aerobic conditions (42 % domestic and 56 % industrial). The application of anaerobic treatment is rather low for both wastewater types. With the current management the amount of BOD – COD going to energy generation by 2050 is expected to be 1 and 10 Mt, respectively (Fig. 4a). As in the case of solid waste, most of the untreated wastewater is currently discharged in ASIA (50 %) and MAF (21 %) regions (see Supplement Fig. S3). This situation is expected since sanitation and waste management are directly linked (Marshall and Farahbakhsh, 2013). With the improvement of wastewater management focused on energy generation 39 % of BOD (refers to urban wastewater) and 91 % of COD could be going to anaerobic treatment with energy recovery by 2050, which corresponds to 48 Mt-BOD and 205 Mt-COD (Fig. 4b). Extending the treatment capacity (collection rates in urban areas) on top of the technical improvement by 2050, would increase up to 85 % of BOD and 91 % of COD going to anaerobic treatment, which corresponds to 78 Mt BOD and 205 Mt COD (Fig. 4c).

3.4 Maximum energy potential from waste and wastewater

The analysis of the estimation of maximum energy potential from waste and wastewater (before conversion to electricity or heat) shows (Fig. 5) that current energy recovered from waste and wastewater management is around 13 EJ at a global level, which corresponds to 2 % of the total primary energy demand in 2010. 63 % of the total energy recovery originates from waste incineration and 37 % from biogas generation. OECD countries have a share of 81 % of total energy recovered from waste and wastewater at the global level (79 % incineration and 21 % biogas). In general, OECD countries have been improving waste and wastewater treatment systems as a key element of achieving sustainable resource management, of which energy recovery is an essential part.

With the "maximum technically feasible phase-in of waste and wastewater management" (MFR) energy generation would be ∼5 times higher compared to the CLE scenario reaching 66 EJ by 2040, which would correspond to 9 % of the total primary energy demand (∼740 EJ) projected by IEA (International Energy Agency, 2017) in 2040. 81 % of the energy would be recovered from waste incineration and 19 % from biogas. These shares are the result of exhausting the corresponding recycling capacity before sending material to incineration, reducing the waste going to landfills and upgrading/improving wastewater treatment systems with energy recovery. Most of the biogas is generated from solid waste (99 %) while the contribution from wastewater is particularly low (1 %). Wastewater must undergo pre-treatment before entering the anaerobic treatment, which removes organics by 35 %–40 %, reducing the capacity of biogas generation (Cakir and Stenstrom, 2005). Furthermore, a certain fraction (depending on temperature, pressure, salinity) of the

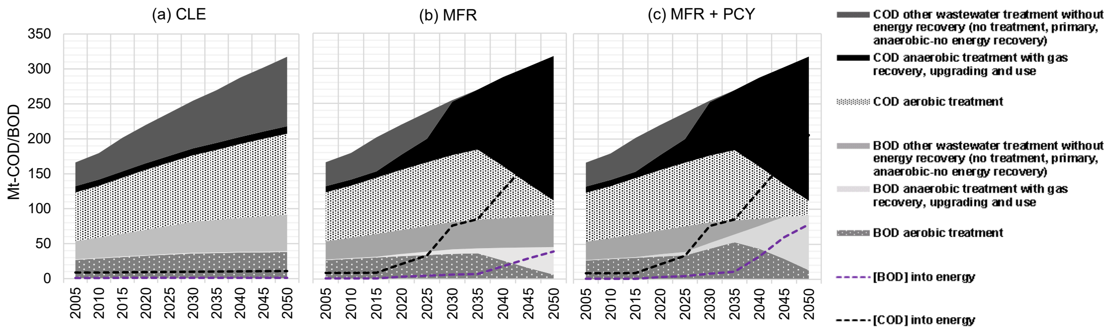

Figure 4. Global BOD (domestic) and COD (industrial) flows in wastewater by scenario.

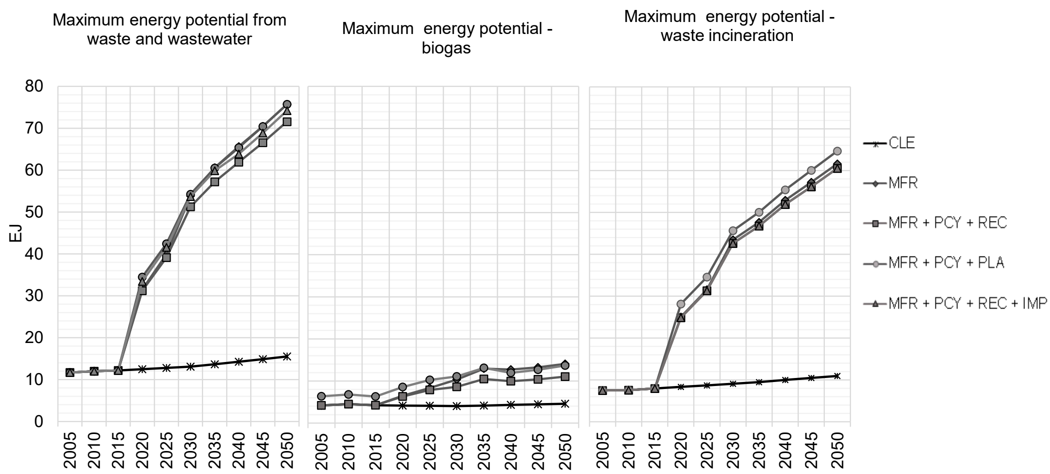

Figure 5. Maximum global energy recovery potential from waste and wastewater treatment by scenario.

methane formed remains in the water as dissolved methane, which diminishes even further the potential for biogas generation (Liu et al., 2014) – a situation which explains the lower share of energy recovered from wastewater.

Moreover, if on top of the technical improvement, policies aimed at reducing food and plastic waste are implemented and plastic recycling rates are maintained at current levels and the remaining plastic material is sent to incineration (MFR + PCY + PLA), energy generation will reach the same level as the MFR strategy alone (66 EJ). Biogas would be reduced by 23 %, falling from 13 to 10 EJ in 2040. Energy available from incineration will increase from 53 to 55 EJ in the same year. Sending the excess of plastic waste into energy recovery compensates for the reduction of plastic generation and increases energy from incineration by ∼ 5 %. Although the concept of waste recovery includes energy recovery, this latter process results in less decarbonization and environmental benefits than material recovery since virgin material is still demanded (Hopewell et al., 2009). However, with the current situation of excess supply in the plastic recycling market (e.g., China's ban on importing recycling plastic after being the leading world's importing country, Velis, 2014) and assuming the success of the plastic waste reduction policy, the "best" way to recover/reuse plastic waste is to convert it to energy through incineration.

However, it is preferable to exhaust the maximum recycling rates before sending material to incineration. Therefore, assuming an ideal market for recyclables on top of food and plastic reduction policies (MFR + PCY + REC), the potential of energy generation is reduced by 6 % in 2040 compared to the MFR and to the MFR + PCY + PLA, resulting in 62 EJ of the energy gains. Hence, the prevention of food and plastic waste generation would not drastically affect the maximum energy recovery potential, but instead have positive impacts towards other sustainability factors. 84 % of the total energy recovered would be from waste incineration and 16 % from biogas.

Finally, the optimal waste and wastewater management scenario for improving the so-called circular economy would be to follow the scenario MFR + PCY + REC + IMP, where the implementation of food and plastic waste reduction policies succeed, the maximum recycling rates of the different waste streams (including plastic) are reached and where waste and wastewater treatment technology improvements increase energy generation and energy recovery efficiency. Once the recycling capacity is exhausted, remaining materials are allowed to enter incineration plants. Organic waste is digested and wastewater is anaerobically treated to produce biogas. The maximum energy potential from waste and wastewater sectors would then be 64 EJ by 2040 which is 9 % of the total primary energy demanded in 2040 as projected by IEA (New Policies scenario 2017). By comparing the CLE to the MFR + PCY + REC + IMP we observe that there exists and estimated additional potential for recovering energy equivalent to 50 EJ per year. In other terms, it means that only 20 % of the maximum capacity to generate energy from solid waste and wastewater would be exploited if current technology and infrastructure are maintained in the future. The success of policies simulated in the improved technology scenario requires waste prevention, reuse, recycling and energy generation, resulting in multiple climate, environmental and social co-benefits.

4 Conclusions

It is recognised that for health and environmental reasons, there is a large potential to improve waste and wastewater management systems at a global level, with immediate action needed in developing countries. We present an estimation of the carbon content in waste and wastewater accompanied by a quantification of the maximum energy that can be generated from global waste and wastewater sectors in the timeframe to 2050 at a global level. Furthermore, we confront different waste and wastewater management scenarios assuming diverse policy measures and treatment pathways and identify an "ideal" system as provider of maximum benefits in terms of energy in support of the decarbonization of the energy system. We find that a scenario that targets at waste reduction, recycling, energy generation and technological improvement would be the policy option that would generate the maximum energy in support of a low-carbon energy system. The management of waste and wastewater focusing on the implementation of this policy option would generate a maximum of 64 EJ of energy in 2040 and 74 EJ in 2050. 82 % of energy would be recovered from waste incineration and the remaining 18 % from anaerobic processes generating biogas. Furthermore, the share of the energy generated from waste and wastewater in the total primary energy demand could increase from 2 % to 9 % at the global scale. Further detailed economic (including recycling markets) and social analyses, taking into account regional and local characteristics would be important to identify potential economic barriers associated with the implementation of the analysed scenarios.

Data availability. Underlying data can be found in the Supplement.

The Supplement related to this article is available online at https://doi.org/10.5194/adgeo-45-105-2018-supplement.

Author contributions. AGS and LHI conceived the idea of the study. AGS developed and performed the calculations, analyzed the results and wrote the manuscript. LHI supervised the findings, supported the development of conclusions and edited the manuscript. PR and WS performed the integration of the global socio-economic drivers needed to carry out the different calculations. All authors discussed the results and contributed to the manuscript.

Competing interests. The authors declare that they have no conflict of interest.

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Supplement of

Carbon in global waste and wastewater flows – its potential as energy source under alternative future waste management regimes

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Table of Contents

S1 Introduction

The information presented here is intended to extend and provide more detailed information on the projections of waste and wastewater (COD and BOD content) generation, underlying assumptions, and data sources used.

S2 Methods

Key assumptions for calculations of waste carbon content and potentials for biogas and energy-recovery are presented in Table S1.

Table S1. Key assumptions to determine carbon content and energy generation

Activity	Variable	Description/assumption	Reference
Solid waste	Maximum carbon conversion	77 % of total organic carbon available is decomposed	IPCC, 2006
Organic waste	Waste conversion rate to biogas	150 m3 biogas kg^{-1} waste	Kigozi et al., 2014
Manure	Water content of manure	85% water content	Höglund-Isaksson, 2015
Manure	Manure conversion rate to biogas	33.53 m3 ton^{-1} manure when manure is co-digested	Based on IEA, 2014
Co-digestion (80% manure + 20% organic waste)	Wet substrate conversion rate to energy	380 KWh/ton wet substrate	Höglund-Isaksson, 2015
Co-digestion (80% manure + 20% organic waste)	Waste conversion rate to biogas co-digestion	65.295 m3 ton-1	Höglund-Isaksson, 2015
Biogas	Biogas from anaerobic digestion composition	60% CH4 + 40% CO2	IPCC, 2006
Landfill gas	Landfill gas composition	50% CH4 + 50% CO2	Spokas et al., 2006
Landfill gas	Gas efficiency collection rate	60%	Spokas et al., 2006
Biogas	Energy from biogas (before conversion)	6.1 kWh m^{-3} biogas	de Mes et al., 2003
Biogas	Biogas thermal value	22 MJ m^{-3} biogas	Spokas et al., 2006
Biogas	Biogas density	1.132 kg m-3	Karellas et al., 2010
	Food waste	5.5 MJ Kg^{-1}	Noukeu et al., 2016
	Plastic waste	27.8 MJ Kg^{-1}	
	Paper waste	16.20 MJ Kg^{-1}	
Incineration solid waste (Low Heating value- LHV)	Wood waste	18.84 MJ Kg^{-1}	Consonni and Viganò, 2011
	Textile waste	19.88 MJ Kg^{-1}	
	Rubber waste	22.5 MJ Kg^{-1}	
	Other waste	5.69 MJ Kg^{-1}	
Industrial wastewater	COD conversion rate to biogas	0.35 m3 biogas kg^{-1} COD	de Mes et al., 2003
Industrial wastewater	Maximum methane production capacity	0..25 kg CH_4 Kg COD	IPCC, 2006
Industrial wastewater	Effluent untreated temperature	30°C	Noukeu et al., 2016
Domestic wastewater	COD conversion rate to biogas	0.84 m3 biogas kg^{-1} COD	de Mes et al., 2003
Domestic wastewater	Country specific per capita BOD taken from IPCC Guidelines 2006	BOD_5	IPCC, 2006. Volume 5. Waste, Table 6.4
Domestic wastewater	BOD conversion rate to biogas	0.84 m3 biogas kg^{-1} BOD	IPCC, 2006Volume 5. Waste, Table 6.2
Methane solubility in wastewater	Methane solubility	45% of CH_4 produced at 30°C	Liu et al., 2014
Primary treatment	COD/BOD removal efficiency	35%-40%	Cakir and Stenstrom, 2005
Anaerobic treatment	COD/BOD removal efficiency	80%	Cakir and Stenstrom, 2005

S2.1 Wastewater and solid waste projections up to 2050

Industrial solid waste: Table S2 presents industrial waste generation by income group classification (see Table S5) and type of manufacturing industry type.

Table S2. Total industrial waste generation in 2010 in Mt

Income group	Food industry	Pulp and paper industry	Rubber industry	Textile industry	Wood industry	Other manufacturing industry	Total	Reference
Low	161	16	3	9	19	958	1167	
Middle low	154	19	12	6	36	1171	1398	Höglund-Isaksson, 2012, Eurostat, 2017, OECD, 2017
Middle	14	3	1	3	3	79	103	
Middle high	23	13	13	2	4	78	133	
High	103	98	47	7	59	338	651	
World	455	149	76	26	121	2624	3452	

Municipal solid waste - Description of data and variables used to estimate waste generation elasticities: The dataset for EU28 countries and some OECD countries covers between 17 and 19 years. For the rest of the countries, the dataset covers between 4 and 10 years. In total, the unbalanced panel data set comprises 684 observations. Data on municipal solid waste generation in kilogram per capita are obtained from different sources (see Table S3). In order to control for the influence of population growth, waste generation per capita is chosen instead of total waste generation as dependent variable in elasticity estimations (Lebersorger and Beigl, 2011). All variables are specified in logarithmic form in order to provide parameter estimates that can be directly interpreted as elasticity values.

Table S3.Dataset description

Country	Years	Waste generation data - Source
EU 28 countries	1995-2012	Eurostat (retrieved 2016)
Japan	1995-2013	OECD (retrieved 2016)
Norway	1995-2014	OECD (retrieved 2016)
Colombia	2003-2011	SSPD 2011
Israel	2001-2013	OECD (retrieved 2016)
Mexico	1995-2012	OECD (retrieved 2016)
Turkey	1995-2013	OECD (retrieved 2016)
Serbia	2006-2013	Eurostat (retrieved 2016)
Macedonia	2008-2014	Eurostat (retrieved 2016)
Malaysia	1996-2000	Department of statistics Malaysia (accessed 2016)
Kenya	1998-2009	
Montenegro	2008-2013	Eurostat (retrieved 2016)
Bosnia and Herzegovina	2008-2013	Eurostat (retrieved 2016)
Australia	2006-2011	OECD (retrieved 2016)
Switzerland	1995-2013	OECD (retrieved 2016)
Peru	2012-2015	Municipalidad Metropolitana de Lima (MML) 2015

In terms of explanatory variables (see Table S4), generation of waste has primarily been linked to economic growth and increases in population and urbanization (Johnstone and Labonne, 2004; Mazzanti and Nicolli, 2011; Mazzanti and Zoboli, 2008, 2009). Income is a major driver of municipal waste generation (Mazzanti and Zoboli, 2008). Gross domestic product has been widely used as the economic parameter to project waste generation (Daskalopoulos et al., 1998).

Table S4. List of variables

Variable	Definition	Source	Mean	Standard Deviation	Minimum	Maximum
Dependent Variable						
MSW	Municipal solid waste generated Kg per person per year	See Table S2	383.71	113.82	101.1	667
Explanatory Variables						
GDP	Gross domestic product USdollar2010 per person per year	World Bank (accessed 2016)	28517.61	20440.94	4945.95	110001.1
UR	Average Annual Rate of Change of the Percentage Urban by Major Area, Region and Country	United Nations -world populations prospects (2014)	71.01	13.17	19	97.73

Elasticity estimation models: Historical data on municipal solid waste generation per capita (dependent variable) are plotted against GDP per capita (independent variable) in order to visualize the relationship between the two variables and to identify possible clusters of municipal waste generation (Fig. S1).

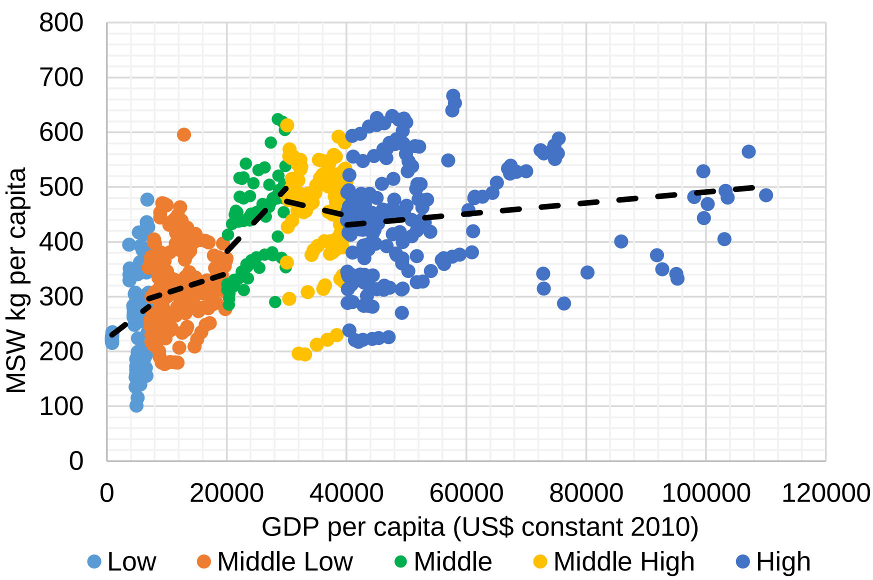

Fig. S1. Municipal solid waste vs GDP per cap.

The definition of the different income groups was carried out based on the distribution of the scatterplot. Table S5 shows the countries belonging to each of the five income groups in 2010 (which is the base year for the projections). Note that in the subsequent projections, countries may over time move out of their initial income group and into a higher income group following an increase in the GDP per capita consistent with the macroeconomic scenario of the IEA World Energy Outlook 2017 (IEA, 2017). Hence, the group distribution of the municipal solid waste generation is dynamic over time.

Table S5. Country by income group in base year 2010

Income group	**Country/region**
High	Austria, Australia, Belgium, Canada, China (Hong Kong and Macau), Denmark, Finland, France, Germany, Iceland, Ireland, Japan (Chugoku Shikoku, Chubu, Hokkaido-Tohoku, Kanto, Kinki), Luxembourg, Netherlands, Norway, Sweden, Switzerland and United States of America.
Middle - High	Brunei, Israel, Italy, Japan (Kyushu Okinawa), South Korea (Busan), New Zealand, Singapore, Spain and United Kingdom.
Middle	Cyprus, Greece, South Korea (Seoul – Inchon, South region), Malta, Portugal, Slovenia and Taiwan.
Middle – Low	Argentina, Caribbean (includes countries in the Caribbean region), Chile, China (Shanghai), Croatia, Czech Republic, Estonia, Hungary, Iran, South Korea (North region) , Latvia, Lithuania, Malaysia (Peninsular Malaysia), Mexico, North Africa (includes Algeria, Morocco, Libya, Tunisia, Sudan), Poland, Romania, Russia (Europe and Asia), Saudi Arabia, Slovak Republic, Turkey and Uruguay.
Low	Afghanistan, Albania, Armenia, Azerbaijan, Bangladesh (Dhaka and rest of Bangladesh), Belarus, Bhutan, Bosnia and Herzegovina, Bolivia, Brazil, Bulgaria, Cambodia, Central America, China (Anhui, Beijing, Chongqing, Fujian, Gansu, Guangdong, Guangxi, Guizhou, Hainan, Hebei, Heilongjiang, Henan, Hubei, Hunan, Jilin, Jiangsu, Jiangxi, Liaoning, Inner Mongolia, Ningxia, Qinghai, Shaanxi, Sichuan, Tianjin, Tibet, Xinjiang, Yunnan and Zhejiang) , Colombia, Ecuador, Egypt, Former Soviet Union States (includes Tajikistan, Turkmenistan and Uzbekistan), Georgia, India (Andhra Pradesh, Assam, West Bengal, Bihar, Chhattisgarh, Delhi, North East (excl Assam), Goa, Gujarat, Haryana, Himachal Pradesh, Jharkhand, Karnataka, Kerala, Maharashtra, Manipur, Orissa, Punjab, Rajasthan, Tamil Nadu, Uttarakhand, Uttar Pradesh, Jammu Kashmir), Indonesia (Jakarta, Java Sumatra and rest of Indonesia), Kazakhstan, North Korea, Kosovo, Kyrgyzstan, Laos, Macedonia, Malaysia (Sarawak Sabah and Kuala Lumpur), Iran, Moldova, Mongolia, Montenegro, Myanmar, Nepal, Other African countries (includes all other African countries), Pakistan (Karachi, NW frontier provinces Baluchistan, Punjab and Sindh), Paraguay, Peru, Philippines (Bicol, Luzol and Manila), South Africa, Serbia, Sri Lanka, Thailand (Bangkok, Central Valley, North Eastern Plateau, Northern Highlands and Southern Peninsula) , Ukraine, Venezuela and Vietnam (North and South).

The panel data analysis is performed to determine the elasticity of the different variables on the generation of municipal solid waste per capita. Pooled OLS, fixed effects and random effects estimator models are run to test the effects of the explanatory variables on municipal waste generation per capita. In the pooled models a single slope is calculated for all countries and the between (cross-sectional) and within (time) variances are bluntly added up. When the cross-sectional variance is eliminated and the slopes are based on time variance only, the model is denoted a within estimator whereas in between models the time variance is eliminated and only cross-sectional variance is considered in the elasticity parameter. In fixed effect models, the within estimator is describing the slope while the country-specific effects are captured as country-specific constants. Finally, random effect model treats the individual effects as random variables and the variance is a weighted average of within and between variance (Hsiao, 1986). Three different tests are applied to select the appropriate model. A Lagrange Multiplier (LM) test is applied to test for the cross-sectional dependence in heterogeneous panels (test random effects vs pooling). An F test is used to test for individual effects based on the comparison between the within and the pooling model and a Hausman test is used to evaluate the difference in vector coefficients between the fixed and random effects models. Here, we explore the possible effects of the explanatory variables on municipal solid waste generation and we test the hypothesis that there are no individual effects, against that there are individual effects. In order to test for a potential presence of homogeneity a Bartlett test is conducted. The Bartlett test is used to test if groups or samples have equal variances, however, the test is sensitive to normality. Therefore, two tests that are less sensitive to normality such as the Chi-square test and Fligner-Killen test are conducted as well (Table S6).

Table S6. Test homogeneity of variances

Test	Hypothesis	Results	Ho
Barlest test	$\mathrm{Ho:}\ \sigma_0^2 = \sigma_1^2 = \cdots \sigma_k^2$ $\mathrm{Ho:}\ \sigma_0^2 \neq \sigma_1^2$	29.407***	Rejected
Chi square test	$\mathrm{Ho:} \sigma^2 = \sigma_0^2$ $\mathrm{Ha:} \sigma^2 \neq \sigma_0^2$	9.48***	Rejected
Fligner-Killeen	$\mathrm{Ho:} \sigma^2 = \sigma_0^2$ $\mathrm{Ha:} \sigma^2 \neq \sigma_0^2$	27.44***	Rejected

The results of the elasticity estimations of municipal solid waste generation to GDP per capita and urbanization rate and the functions for waste generation projections are presented in Table S7. The LM test favoured in all cases the random effect over the OLS model, meaning that there is evidence of significant differences across countries. F test for individual effects favoured always the fixed effect model over the OLS, which means that the fixed effect are non-zero and finally, the Hausman test rejected the random effect model, which assume that the slope coefficients of the two models do not differ and it favoured the fixed effect model. Furthermore, due to the fact that waste composition influences energy generation, projections of waste compositions are relevant. In particular, low income countries tend to have a considerably higher fraction of food waste in the total municipal waste generated than high income countries. Therefore, changes in the future composition of waste are projected based on an estimated elasticity of food waste generation to GDP per capita. Due to limited access to historical data on food waste generation, the elasticity is estimated from a sample of 156 observations of in an unbalanced panel. A fixed effects model was favoured on the basis of Hausman test as the better explanatory model with a resulting elasticity of food waste generation to GDP per capita of 0.42 (Table S7).

Table S7. MSW generation elasticities to GDP and Urbanization rate

Dependent Variable	Unit	Income group (U$ dollars per capita year)	Number of observations	Explanatory variable	OLS	Fixed Effect	Random Effect	LM - test	Hausman -test
Municipal solid waste	Kt per capita	< 7000	98	Constant	4.96 (10.50)***	n.a	2.44 (3.97)***	84.92	3.66
				GDP per capita	0.06 (1.21)	*0.41 (5.38)****	0.36 (5.04)***		
				Urbanization rate					
				R-square	0.01	0.25	n.a		
		>=7000 - <20000	193	Constant	4.15 (8.42)***	n.a	3.68 (6.96)***	38.22	0.002
				GDP per capita	0.16 (3.19)**	*0.22 (3.73)****	0.21 (3.92)***		
				Urbanization rate					
				R-square	0.05	0.07	n.a		
		>=20000 - <30000	75	Constant	-0.92 (-0.56)	n.a	-0.35 (-0.23)	21.99	0.001
				GDP per capita	0.69 (4.30)***	*0.62 (4.23)****	0.62 (4.30)***		
				Urbanization rate					
				R-square	0.20	0.21	n.a		
		>30000 - <40000	108	Constant	5.98 (2.23)*	n.a	2.82 (1.53)	140.56	20.93
				GDP per capita	-0.16 (-0.68)	*0.80 (7.31)****	0.55 (5.40)		
				Urbanization rate	0.43 (2.01)*	*-3.27(-4.43)****	-0.60 (-1.33)***		
				R-square	0.04	0.37	n.a		
		>=40000	210	Constant	3.33 (3.72)***	n.a	-0.10 (-0.07)	52.22	17.43
				GDP per capita	0.17 (2.55)*	*1.07 (8.20)****	0.84 (7.10) ***		
				Urbanization rate	0.18(1.38)	*-1.28 (-3.66)****	-0.67 (-2.28)*		
				R-square	0.043	0.26	n.a		
		All income groups	684	Constant	3.85 (21.10)***	n.a	4.03 (8.61)***		
				GDP per capita	0.24 (17.51)***	0.43 (13.13)***	0.37 (13.72)***	95.64	10.45
				Urbanization rate	-0.08 (-1.52)	-0.45 (-2.27)*	-0.43 (-3.42)***		
				R-square	0.4	0.21	n.a		
Food waste	Kt per capita	All income groups	156	Constant	4.05 (9.32)***	n.a	2.78 (4.29)***		
				GDP per capita	0.05 (1.33)	*0.42 (4.22)****	0.18 (2.85)**	40.54	9.78
				R-square	0.01	0.12	n.a		

Where:, $\varepsilon_{it}=u_i+v_{it}$ is an error term which is separated into an individual effects term and a residual omitted variables term, and $\varepsilon_{it}\sim IID\left(0,\sigma_{\varepsilon}^{2}\right)$ is an error term which are assumed to be normally distributed with mean zero and constant variance.

Although, there are more availability of data for developed countries, it was possible to find a limited set of about ten developing countries for which enough information was available to include in the estimation of elasticities of municipal solid waste generation to GPD per capita and urbanization rates. However, due to a general lack of data from developing countries on food waste generation, the elasticity estimates for food waste generation are based on data from Eurostat (2016) and cover mainly developed countries. In addition, only GDP per capita and changes in the urbanization rate are used as explanatory variables. In reality, many more factors are likely to influence the generation of municipal waste, in particular household-specific factors e.g., household size, type of dwellings, rural or urban, income distribution, etc. It would have been desirable to conduct the elasticity estimations at a more disaggregated level, representing the diverse circumstances within a country, however, this was not possible due to limitations in data availability.

Table S8 presents municipal waste generation rates and composition for the year 2010 (base year for projections). Since yearly information on waste composition is limited (especially for developing countries), the most recent available data is used. References apply to the waste management data as well.

Table S8. Municipal solid waste generation and composition in 2010.

Income group	No. of countries/regions	Municipal solid waste generation			Composition (weighted average across countries)							
		Mt year $^{-1}$	Kg cap^{-1} day^{-1}	Range Kg cap^{-1} day^{-1}	Food	Paper	Plastic	Glass	Metal	Wood	Textile	Other
Low	112	1249	0.67	0.06 - 1.94	0.51	0.09	0.09	0.03	0.02	0.06	0.04	0.17
Middle low	23	246	0.87	0.16-1.51	0.44	0.16	0.09	0.07	0.02	0.06	0.03	0.12
Middle	8	31	1.03	0.85-1.54	0.31	0.24	0.12	0.05	0.05	0.07	0.03	0.14
Middle high	9	107	1.40	0.78-1.90	0.29	0.25	0.13	0.06	0.03	0.02	0.03	0.19
High	22	456	1.77	0.80-2.19	0.25	0.30	0.13	0.06	0.07	0.05	0.05	0.10
World	174	2088	0.83	0.06-2.19	0.43	0.15	0.10	0.04	0.03	0.06	0.04	0.15

Source: Low: Forouhar and Hristovski, 2012, Wiedinmyer et al., 2014, Hoornweg and Bhada-Tata, 2012, Arzumanyan, 2014, Anon, 2009; Bhuiyan, 2010; Zakir Hossain et al., 2014, Penjor, 2007, Viceministerio de agua potable, 2012, Castagnari, 2005, Eurostat 2016, Ministry of Environment PNH, 2010; Mongtoeun, 2015, Bo-Feng et al., 2014; China Statistical Yearbook, 2007; Wang and Nie, 2001, Larochelle et al., 2012; Martínez, 2015, M. Sim et al., 2013, Kumar et al., 2009; Sharholy et al., 2008, Damanhuri et al., 2009; Meidiana and Gamse, 2010; Pasang et al., 2007, Vermenchiva et al., 1999, Sang-Arun and Pasomsouk, 2012, Cvetkovska and Rushiti, 2013, Budhiarta et al., 2012; Manaf et al., 2009, agath P and Hengesbaugh, 2016, Viraraghavan, 2005, Bello et al., 2016; Parrot et al., 2009, Mahar et al., 2007, Organización Panamericana de la Salud, 2001, Department of Environmental Affairs, 2012, ISWA, 2011; Vukmirovic, 2012, Hikkaduwa et al., 2015; Karunarathne, 2015, Tanakwang and Tangtinthai, 2010, International Finance Corporation, 2010, Instituto Nacional de Estadística, 2012, Nguyen, 2005, Thang, 2011.
Middle low: Gonzalez, 2010; Savino, 1999, Hoornweg and Bhada-Tata, 2012, Bräutigam and Gonzalez, n.d.), Bo-Feng et al., 2014; China Statistical Yearbook, 2007; Wang and Nie, 2001, Eurostat 2016, Alavi Moghadam et al., 2009; Damghani et al., 2008, Ryu, 2010, Budhiarta et al., 2012; Manaf et al., 2009, Gomez et al., 2008, Bello et al., 2016; Okot-Okumu, 2012; Parrot et al., 2009; SWEEPNET, 2012,
Middle: Eurostat 2016, Chieueh and Yu, 2006; Tsai and Chou, 2006
Middle High: Wiedinmyer et al., 2014, Ministry of environmental protection, 2012, Eurostat 2016, OECD, 2016 ;Ministry of the Environment, 2012, ISWA, 2011, Bai and Sutanto, 2002, Burnley, 2007; Daskalopoulos et al., 1998
High: Eurostat 2016, Asase et al., 2009, Bo-Feng et al., 2014; China Statistical Yearbook, 2007; Wang and Nie, 2001, OECD, 2016 ;Ministry of the Environment, 2012, EPA, 2012

S2.2 Carbon content determination and energy calculations

S2.2.1 Solid waste

In order to quantify the carbon content of industrial and municipal solid waste and the respective flows, the following approach is used (calculations are always carried out by region for the 174 countries/regions and with annual results presented for every five years):

1. Quantification of DOC and FC in municipal and industrial solid waste using IPCC default values for DOC and FC (IPCC, 2006, Volume 5, Chapter 2).
2. Identification by country/region of the application rate of current (and future) waste management technologies/systems (EUROSTAT 2016, OECD 2016, UNFCCC CRF Tables 2016 and documents referenced in Table S8 supplement material). This study distinguishes various management options for each of the solid waste fractions. Description of each of the options can be found in Table S8. The assessment of the carbon flows is then carried out applying Eq. (S1). and Eq. (S2):

$$DOC_{m,s;j} = W_{s,j} * DMC_{s,j} * DOCd_{s,j} * Appl_{m,s,j} * 0.01 \quad \text{Eq. (S1)} \quad ; \quad FC_{m,s;j} = W_{s,j} * FCC_{s,j} * Appl_{m,s,j} * 0.01 \quad \text{Eq. (S2)}$$

Where: $DOC_{m,s,j} / FC_{m,s;j}$ is the amount of Degradable Organic Carbon (DOC)/ Fossil Carbon (FC) in dry waste type j in sector s (municipal/industrial) going to a specific treatment m ; $W_{s,j}$ is the amount of waste type j generated in sector s (municipal/industrial); $DMC_{s,j}$ is the Dry Matter Content (DMC) in % of wet waste j generated in sector s (municipal/industrial); $DOCd_{s,j}$ is the DOC in % of dry waste j generated in sector s (municipal/industrial); $FCC_{s,j}$ is the fraction of Fossil Carbon in % of Total Carbon in waste j generated in sector s (municipal/industrial) and $Appl_{m,s,j}$ is the application of the waste treatment option m to waste type j generated in sector s (municipal/industrial).

3. Estimation of energy recovery from municipal and industrial solid waste: This study identifies anaerobic digestion, landfill with gas recovery and use and waste incineration as the three main treatment technologies to convert waste into a source of energy.

<u>Anaerobic digestion</u>: Biogas generation is calculated using Eq. (S3) from Höglund-Isaksson, 2015 and Eq. (S4) :

$$BCD = (TS * Y_{cd}) \quad \text{where} \quad TS = MaxM + \left(\frac{MaxM*100}{80}\right) * 0.2 \quad \text{Eq. (S3)} \quad ; \quad BSS = \left(S * Y_{o,m}\right) \text{ Eq. (S4)}$$

Where: BCD is biogas from co-digestion; TS is total substrate; Y_{cd} is the biogas yield of co-digestion when 80% manure - 20% organic waste ; MaxM is the maximum manure available for co-digestion; BSS is the biogas single substrate; S is the substrate and ; $Y_{o,m}$ is the biogas yield when digestion only organic waste or only manure.

<u>Landfill</u>: Landfill gas generation is accounted for with a lag of 10 years for fast degrading organic waste and 20 years for slow degrading waste. Landfill gas generation is calculated using Eq. (S5) based on (IPCC, 2006, Volume 5, Chapter 2 and Chapter 3):

$$LG = ((DOC_{s;j} * 0.77 * F * {}^{16}/_{12}) + (DOC_{m,s;j} * 0.77 * F * {}^{44}/_{12})) * 0.60 * {}^{1}/_{1.132} \quad \text{Eq. (S5)}$$

Where: LG is landfill gas; $DOC_{s;j}$ is the amount of Degradable Organic Carbon (DOC) in dry waste type j in sector s (municipal/industrial) going to landfills with gas recovery; 0.77 is the maximum carbon conversion; F is the fraction of CH4 - CO_2 in generated landfill gas (0.50); $^{16}/_{12}$ is the molecular weight ratio CH_4/C; $^{44}/_{12}$ is the molecular weight ratio CO_2/C; 0.60 is the gas collection efficiency rate and 1.132 kg m^{-3} is the biogas density.

Incineration: Energy from incineration is calculated using the Low Heating Value (LHV) of each of the waste fractions. LHV represents the usable heat released from waste and varies according to waste type (Demirbas, 2004). Energy from incineration is calculated using Eq. (S6).

$$EI = W_{s,j} * LHV_j \qquad \text{Eq. (S6)}$$

Where: EI is energy gained from incineration; $W_{s,j}$ is the amount of waste type j generated in sector s (municipal/industrial) going to incineration with energy recovery (municipal/industrial) and LHV_j is the low heating value of waste typej.

Table S9 presents the different management options implemented for each waste type.

Table S9. Solid waste management technologies

Solid waste management technology	Municipal solid waste								Industrial solid waste				
	Food	Glass	Metal	Other	Paper	Plastic	Textile	Wood	Food	Pulp and paper	Rubber	Textile	Wood
Open burned	X			X	X	X	X	X	X	X	X	X	X
Scattered and/or disposed to water-courses	X	X	X	X	X	X	X	X	X	X	X	X	X
Unmanaged solid waste disposal site - low humidity - < 5m deep	X			X	X		X	X	X	X		X	X
Unmanaged solid waste disposal site - high humidity - > 5m deep	X			X	X		X	X	X	X		X	X
Compacted landfill	X	X	X	X	X	X	X	X	X	X	X	X	X
Covered landfill	X			X	X		X	X	X	X		X	X
Landfill gas recovery and flaring	X			X	X		X	X	X	X		X	X
Landfill gas recovery and used	X			X	X		X	X	X	X		X	X
Low quality burning of waste	X			X	X	X	X	X	X	X	X	X	X
Incineration (poor air quality controls)	X			X	X	X	X	X	X	X	X	X	X
Incineration (high quality air pollution controls - energy recovery)	X			X	X	X	X	X	X	X	X	X	X
Anaerobic digestion	X								X				
Composting	X								X				
Recycling		X	X		X	X	X	X					X

S2.2.2 Wastewater

In order to quantify the organic content in industrial and municipal wastewater and its respective flows, the following approach is used (calculations are carried out by country/region and year):

1. Quantification of BOD in untreated domestic wastewater and COD in untreated industrial wastewater using the IPCC method (based on IPCC, 2006, Volume 5, Chapter 6, Equation 6.4 and Equation 6.6).
2. Identification by country/region of the application rate of current (and future) use of wastewater management technologies/systems (EUROSTAT 2016, OECD 2016, UNFCCC CRF Tables 2016 and some official national documents). This study distinguishes various wastewater management options for each of the two wastewater types. A description of each option can be found in Table S10. The assessment of the organic material flows is then carried out applying Eq. (S7) and Eq. (S8) based on Höglund-Isaksson et al., 2015

$$\mathrm{COD} = \mathrm{WW_i} * \mathrm{P_i} * \mathrm{COD_i} * \mathrm{Appl_{m,i}} * 0.01 \quad \text{Eq. (S7)} \quad ; \quad \mathrm{BOD} = \mathrm{POP_i} * \mathrm{BOD_i} * \mathrm{Appl_{m,i}} * 0.01 \quad \text{Eq. (S8)}$$

Where: COD is Chemical Oxygen Demand (organic degradable material) in industrial wastewater; $\mathrm{WW_i}$ is the amount of wastewater generated per tonne of product in industrial sector i; $\mathrm{P_i}$ is amount of production product in sector i; $\mathrm{COD_i}$ is total organic degradable material content in the wastewater measured as COD in industrial sectors i, BOD is Biochemical Oxygen

Demand (organic degradable material) in domestic wastewater; POP_i is population; BOD_i is per capita BOD (default values used from IPCC, 2006, Volume 5, Chapter 6, Table 6.4) and $Appl_{m,s,j}$ is the application of the wastewater treatment option m to treat domestic/industrial wastewater.

3. Estimation of the energy potential from domestic and industrial anaerobic wastewater with gas recovery. Volumes of biogas from industrial and domestic wastewater treatment are calculated by applying Eq. (S9)

$$\mathrm{BWWI\ (BWWD)} = \mathrm{COD(BOD)} * \mathrm{Appl_{at}} * 0.01 * (1 - \mathrm{Reff_{pt}}) * \mathrm{Reff_{at}} * \mathrm{F_{COD}}\ (\mathrm{F_{BOD}}) * \mathrm{TCF} * (1 - \mathrm{f}) * \mathrm{Y} \quad \text{Eq. (S9)}$$

Where: BWWI is biogas generation from industrial/*BWWD* domestic wastewater treatment; COD is Chemical Oxygen Demand , *BOD* is Biochemical Oxygen Demand in domestic wastewater; $\mathrm{Appl_{at}}$ is the application in % of the anaerobic wastewater treatment to industrial/domestic sector i; $\mathrm{Reff_{pt}}$ is the COD/BOD removal efficiency primary treatment (before anaerobic treatment a primary removal of floating and settleable material is needed (Cakir and Stenstrom, 2005)); $\mathrm{Reff_{at}}$ is the COD/BOD removal efficiency anaerobic treatment; $\mathrm{F_{COD}}$ is the maximum CH_4 production capacity per Kg COD; $\mathrm{F_{BOD}}$ is the maximum CH_4 production capacity per Kg BOD, TCF is temperature correction factor (just for domestic wastewater) (see Höglund-Isaksson et al., 2015. Section 3.4.2) f is the rate of CH_4 solubility (depends on wastewater temperature (Liu et al., 2014) and $Y =$ 0.35 m^3 is the biogas yield per Kg COD removed, 0.84 m^3 is the biogas yield per Kg BOD removed.

One of the challenges of wastewater treatment is the removal of nitrogen and phosphorus to avoid eutrophication of the water bodies. For that purpose, around 35% of the COD in wastewater is needed for biological nitrogen removal (Hu et al., 2011) and hence unavailable for biogas generation. Therefore, an additional estimation of biogas generation representing the balance between COD and nitrogen removal is also carried out. To compensate for the 35% of COD needed for the removal of nitrogen, estimations of biogas generation assuming that the primary sludge is anaerobically digested and partially converted into biogas is also performed for the MFR scenarios. This process is represented in Eq (S10) where $(1 - \mathrm{Reff_{pt}})$ representing the removal efficiency (35%) of primary treatment is removed and a factor representing the 35% COD demanded for nitrogen removal is added $(1 - \mathrm{COD_N})$. However, this process does not add benefits in terms of biogas generation since the effect of adding the COD of primary sludge is cancelled by the COD demanded for nitrogen removal.

$$\mathrm{BWWI\ (BWWD)} = \mathrm{COD(BOD)} * \mathrm{Appl_{at}} * 0.01 * \mathrm{Reff_{at}} * (1 - \mathrm{COD_N}) * \mathrm{F_{COD}}\ (\mathrm{F_{BOD}}) * \mathrm{TCF} * (1 - \mathrm{f}) * \mathrm{Y} \quad \text{Eq. (S10)}$$

Table S10. Wastewater treatment technologies

Wastewater treatment technology	Domestic wastewater			Industrial wastewater		
	Uncollected	Centralized collection	Decentralized collection	Food	Pulp and paper	Other manufacturing industry
Uncollected	X	X	X	X	X	X
Collected but untreated		X	X	X	X	X
Primary treatment		X		X	X	X
Aerobic treatment		X		X	X	X
Anaerobic secondary and/or tertiary treatment without gas recovery		X		X	X	X
Anaerobic secondary and/or tertiary treatment with gas recovery		x		X	X	X
Latrine/ Septic tank			X			

S2.3 Waste and wastewater management scenarios

Description of the measures adopted in the different scenarios are presented below. Each scenario builds on the one before:

- CLE 'current legislation': The scenario assumes efficient implementation of the existing waste/wastewater legislation. In countries/regions where no waste legislation exists -CLE- represents the current waste management situation.
- MFR 'maximum technically feasible phase-in of waste and wastewater management': A scenario that assumes the implementation of the 'best available technology' to improve waste and wastewater management systems without regarding costs but considering constrains that could limit the applicability of certain technologies and assumes a phase-out of waste going to landfills, being dumped or openly burnt. Waste flows are redirected to recycling, treatment with energy recovery, or controlled incineration with energy recovery. The maximum recycling potential of waste streams are applied as follow: 90% of municipal paper and textile waste recycled by 2030 – 80% of municipal plastic and wood waste recycled by 2030. 100% incineration of industrial solid waste by 2030, 100% of food waste treated in anaerobic digesters with biogas recovery by 2050 and 100% of collected industrial and domestic wastewater treated in anaerobic processes by 2050.
- MFR + PCY + PLA 'maximum technically feasible phase-in of waste and wastewater management' + 'policy implementation + 'plastic incineration': The scenario adopts the MFR + policies for reducing the generation of food and plastic municipal solid waste + maintains current municipal plastic waste recycling rates and sends excess plastics to incineration for energy recovery to represent the current recycling market plastic situation. The policies are assumed to

reach a maximum municipal food waste rate reduction of 50% by the year 2030 based on Lipinski et al., 2013 and based on the target adopted by the United Nations Assembly in 2015 of halving per capita food waste at the retail and consumer level as a part of the 2030 Sustainable Development Goals and a maximum municipal plastic waste rate reduction of 50% by the year 2030 as a part of the 2030 Sustainable Development Goals.

- <u>MFR + PCY + REC 'maximum technically feasible phase-in of waste and wastewater management' + 'policy implementation' + 'maximum recycling capacity'</u>: This scenario adopts the MFR + PCY + reaches the maximum possible recycling capacity for all waste streams (including plastic). For wastewater, the scenario includes a capacity to increase the collection (reaching 100%) and treatment of wastewater in urban areas.
- <u>MFR + PCY + REC + IMP 'maximum technically feasible phase-in of waste/wastewater management' + 'policy implementation' + 'maximum recycling capacity' + 'technology efficiency improvement'</u>: This scenario adopts the MFR + PCY+ REC + technological development to increase biogas yield formation and to reduce losses during the treatment processes for both solid waste and wastewater. Improvements include e.g. adding accelerants (biological or chemical) to improve the metabolic conditions for microorganism growth and therefore biogas formation (Mao et al., 2015), recovery of the dissolved methane in wastewater, improvement of the biogas recovery rates. For incineration, improvements include an increase of the Low Heating Value (LHV), increase in the efficiency of input/air flow and reduction of energy losses during the process.

S2.4 Limitations and uncertainty of the waste and wastewater management scenarios

In this study, anaerobic digestion of waste and anaerobic wastewater treatment are analysed independent of the type of anaerobic reactor e.g. Anaerobic Sludge Blanket (UASB), CSTR and Anaerobic filter (AF-Fixed film) (Barber and Stuckey, 1999). Different reactors involve different flow modes, retention times and organic load rates, which are all factors that affect the efficiency of biogas formation (Mao et al., 2015). Furthermore, default IPCC values for biogas rate formation under average normal operating conditions are used to estimate biogas generation. However, it is well known that the microbial community is extremely sensitive and if not properly managed the process would be affected resulting in reduced biogas production (Munk Bernhard et al., 2010).

Regarding incineration and waste heating values a similar situation to the anaerobic treatment is present; incineration is treated as a general technology independent of the type of incinerator. In addition, although a specific Low Heating Value (LHV) is used for each waste fraction, the variability between regions/countries was not taken into account due to a lack of regional data. In general, the scenarios presented do not take into account the losses of substrates during transport and handling, which may result in a lower substrate input actually going into the treatment facilities.

Given the global scope and the wide range of different types of input data going into estimations, it is unavoidable that a certain degree of uncertainty is present in the results. E.g., for developing countries, a lack of country-specific data on quantities of waste and wastewater, implemented treatment modes, and current energy/biogas recovery rates, has been bridged by using default assumptions adapted from neighbouring countries or regions.

3 Results by major world regions

3.1 Carbon content and flows in solid waste

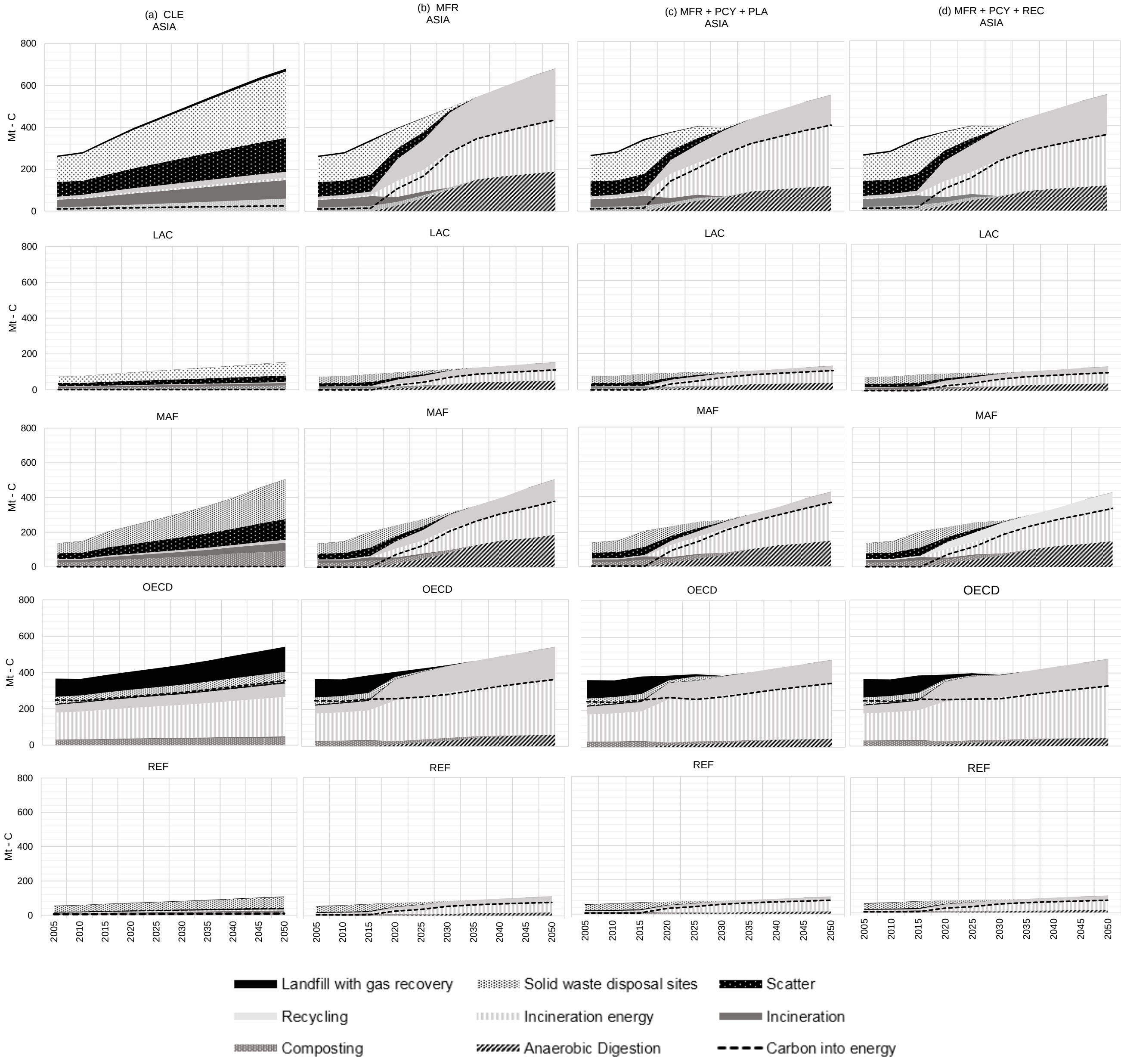

Fig. S2. Carbon flows – solid waste by region

3.2 BOD and COD flows in wastewater

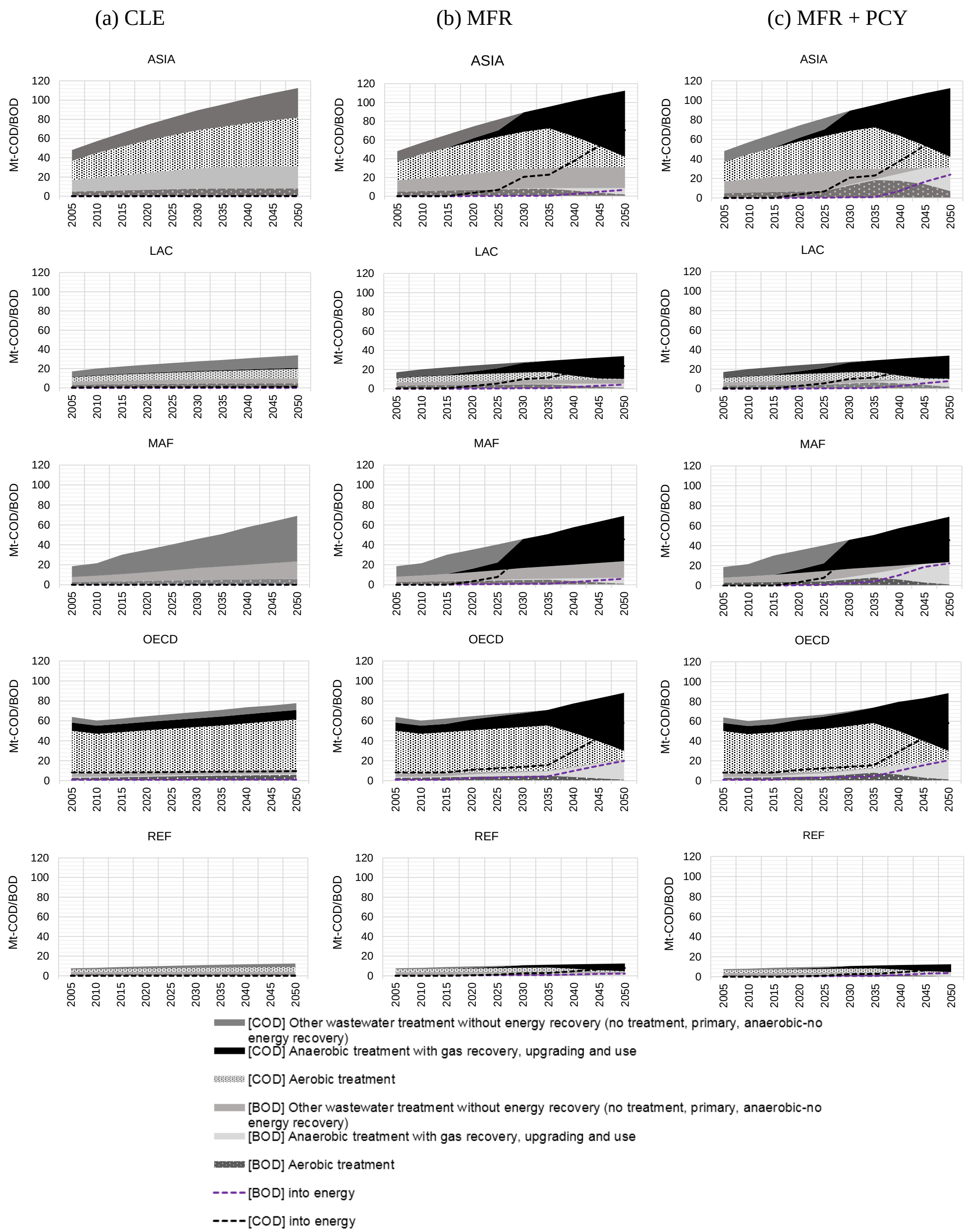

Fig. S3. BOD and COD flows by region

3.3 Maximum energy potential from waste and wastewater

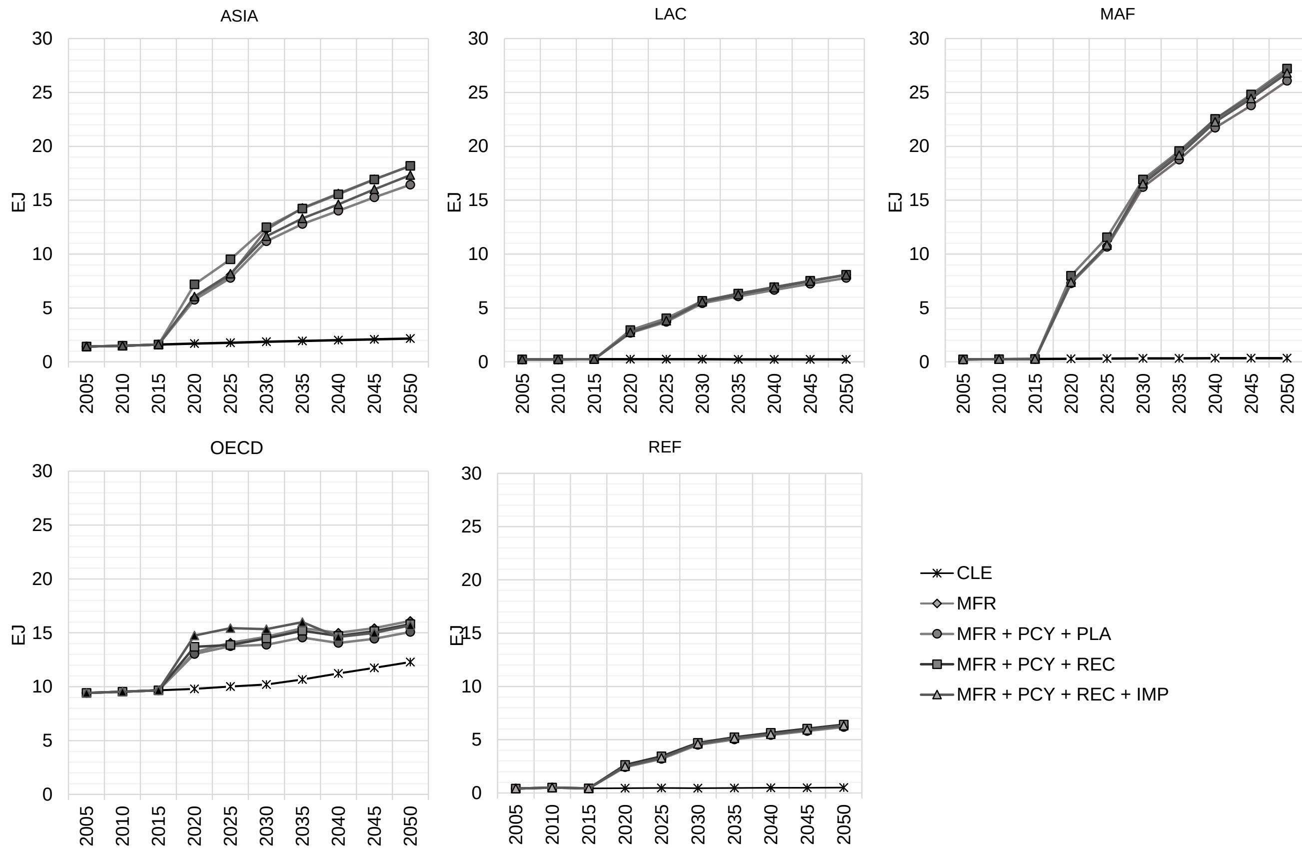

Fig. S4. Maximum energy potential from waste and wastewater by region

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Technical potentials and costs for reducing global anthropogenic methane emissions in the 2050 timeframe –results from the GAINS model

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Abstract. Methane is the second most important greenhouse gas after carbon dioxide contributing to human-made global warming. Keeping to the Paris Agreement of staying well below two degrees warming will require a concerted effort to curb methane emissions in addition to necessary decarbonization of the energy systems. The fastest way to achieve emission reductions in the 2050 timeframe is likely through implementation of various technical options. The focus of this study is to explore the technical abatement and cost pathways for reducing global methane emissions, breaking reductions down to regional and sector levels using the most recent version of IIASA's Greenhouse gas and Air pollution Interactions and Synergies (GAINS) model. The diverse human activities that contribute to methane emissions make detailed information on potential global impacts of actions at the regional and sectoral levels particularly valuable for policy-makers. With a global annual inventory for 1990-2015 as starting point for projections, we produce a baseline emission scenario to 2050 against which future technical abatement potentials and costs are assessed at a country and sector/technology level. We find it technically feasible in year 2050 to remove 54 percent of global methane emissions below baseline, however, due to locked in capital in the short run, the cumulative removal potential over the period 2020-2050 is estimated at 38 percent below baseline. This leaves 7.7 Pg methane released globally between today and 2050 that will likely be difficult to remove through technical solutions. There are extensive technical opportunities at low costs to control emissions from waste and wastewater handling and from fossil fuel production and use. A considerably more limited technical abatement potential is found for agricultural emissions, in particular from extensive livestock rearing in developing countries. This calls for widespread implementation in the 2050 timeframe of institutional and behavioural options in addition to technical solutions.

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1 Introduction

Methane (CH_4) is the second most important greenhouse gas after carbon dioxide (CO_2) contributing to human-made global warming. Keeping to the Paris Agreement of staying well below two degrees warming above the pre-industrial average, will require a concerted effort to curb CH_4 emissions in addition to necessary decarbonization and efficiency enhancements of the energy systems. In the long-term, any remaining anthropogenic CH_4 emissions, e.g., linked to food production, must be offset through negative emission options (IPCC, 2018). Compared to CO_2, CH_4 contributes 28 times more per ton to global warming over 100 years when excluding climate-carbon feedbacks (IPCC, 2013). Because of its shorter lifetime in the atmosphere of 12 years, CH_4's warming potential over twenty years is 84 times that of CO_2 per ton. This means CH_4 accounts for about 40 percent of greenhouse gases' contribution to short-term global warming, which makes it an obvious candidate to target for fast climate change mitigation in the 2050 timeframe (Shindell et al., 2012). Human activities contribute more to CH_4 emissions than natural sources (Saunois et al., 2016) and a swift reduction in anthropogenic CH_4 can even offset climate change impacts of a massive release of natural CH_4 from smelting Arctic permafrost (Christensen et al., 2019).

The fastest way to achieve CH_4 emission reductions in the 2050 timeframe is likely through implementation of various technical options (Pacala and Socolow, 2004). Further abatement potential from institutional changes (Evans and Steven, 2009) and behavioural changes (Abrahamse and Steg, 2013; Camilleri et al., 2019) will be necessary but may take longer to realize. Therefore, the focus of this study is to explore the technical abatement and cost pathways for reducing global CH_4 emissions in the 2020-2050 timeframe, breaking reductions down to regional and sector levels using the most recent version of IIASA's Greenhouse gas and Air pollution Interactions and Synergies (GAINS) model (Amann et al., 2011), denoted GAINSv4 (2019). The diverse human activities that contribute to CH_4 emissions make it particularly valuable with detailed information to inform policy-makers about the potential global impacts of fast actions at the regional and sectoral levels. In addition, we provide insights on sensitivities related to the time and opportunity cost perspectives of the social planner versus private investors.

This study builds on Höglund-Isaksson (2012) by extending the timeframe from 2030 to 2050, updating statistics for historical years to 2015, reflecting recent findings from the literature, and including several methodological improvements of emission estimations, e.g., for the oil and gas sectors (Höglund-Isaksson, 2017; Dalsøren et al., 2018) and waste and wastewater sectors (Gómez-Sanabria et al., 2018). The extended timeframes of this study, to 2015 for historical emissions and to 2050 for future projections, allow for two important insights. First, our bottom-up emission inventory to 2015 attributes a strong increase in atmospheric CH_4 emissions after 2007 (Nisbet et al., 2014; 2019) to a combination of factors; rapid growth in extraction of unconventional gas in North America, extended coal mining in Indonesia, and accentuated growth in waste and wastewater emissions in rapidly developing world regions. Second, the technical mitigation potential of global CH_4 emissions will not be enough for meeting the targets in 2050 of the Paris Agreement. In addition, institutional and behavioural changes will be needed. The GAINSv4 model results add to a limited number of independently developed bottom-up estimates of technical abatement potentials and costs to reduce global CH_4 emissions in the 2050 timeframe (Lucas et al., 2007; Harmsen et al., 2019). Similar efforts have been presented for the 2030 timeframe, e.g., Höglund-Isaksson (2012) estimated marginal abatement cost curves using an earlier version of the GAINS model and USEPA (2006; 2012) presented corresponding cost curves for all non-CO_2 greenhouse gases with Beach et al. (2008; 2015) and Frank et al. (2018) presenting results specifically for the agricultural sector.

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2 Methodology

2.1 Emission estimation

The GAINS model estimates emissions bottom-up, i.e., quantifications of human activities contributing to emissions are multiplied by an emission factor representing the average emissions per unit of activity. Such estimates rely on a wealth of publicly available information to develop internally consistent emission factors across countries, sectors and technologies. The starting point for estimations of anthropogenic CH_4 is the methodology recommended in the IPCC (2006) guidelines, for most source sectors using country-specific information to allow for deriving country- and sector/technology- specific emission factors at a Tier 2 level. For some source sectors consistent methodologies were further developed, e.g., for oil and gas systems (Höglund-Isaksson, 2017) and solid waste sectors (Gómez-Sanabria et al., 2018). The resulting emission estimates are thereby well comparable across geographic and temporal scales and with a possibility to provide plausible explanations for deviations in past emissions. CH_4 emissions are estimated for 174 countries/regions, with the possibility to aggregate to a global emission estimate, and spanning a timeframe from 1990 to 2050 in five-year intervals. For the purpose of better evaluating historical CH_4 emissions, annual estimates for 1990-2015 were produced for this study. Following the general GAINS methodology (Amann et al. 2011), emissions from source *s* in region *i* and year *t* are calculated as the activity data A_{its} times an emission factor ef_{ism}. If emissions are controlled through implementation of technology *m*, the fraction of the activity controlled is specified by $Appl_{itsm}$, i.e.,

$$E_{its} = \sum_m \left[A_{its} * ef_{ism} * Appl_{itsm} \right], \tag{1}$$

where $$\sum_m Appl_{its} = 1, \tag{2}$$

and where	A_{its}	is the activity (e.g., number of animals, tons of waste, PJ gas produced),
	ef_{ism}	is the emission factor for the fraction of the activity subject to control by technology *m*,
	$Appl_{itsm}$	is the application rate of technology *m* to activity *s*.

Hence, for each emission source sector, country- and year- specific sets of application rates for all the possible technologies (including no control) are defined such that application rates always sum to unity.

2.2 Activity data

The GAINSv4 model structure covers all relevant source sectors for anthropogenic CH_4 emissions, for details see Table S1-1 in the Supplement Information (SI). Activity drivers for macroeconomic development, energy supply and demand, and agricultural activities are entered externally in GAINS. For the baseline scenario presented here, the macroeconomic and energy sector activity drivers are consistent with the IEA World Energy Outlook 2018 New Policies Scenario (IEA-WEO, 2018). Growth in global population, Gross Domestic Product (GDP) and GDP per capita are illustrated in Figure 1. This energy scenario assumes that countries comply with the Intended National Determined Contributions (INDCs) to climate change mitigation they pledged in the lead-up to the UNFCCC's COP21 in Paris in 2015, however, it should be noted that these pledges fall short of the Paris Agreement of keeping the earth's warming well below 2°C above the pre-industrial average. How this energy scenario translates into global consumption of different types of fuels is illustrated in Figure 2. Note that for the purpose of this study of improving the understanding of the technical mitigation potentials at the sectoral and regional level, only one baseline has been developed against which future emission reductions are assessed. To provide a full range of possible future developments of global anthropogenic methane emissions, a set of alternative activity scenarios would be

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required. This is however considered out of scope of this paper, as the relative technical mitigation potentials at the sector and regional level will be comparable irrespective of the baseline emission level.

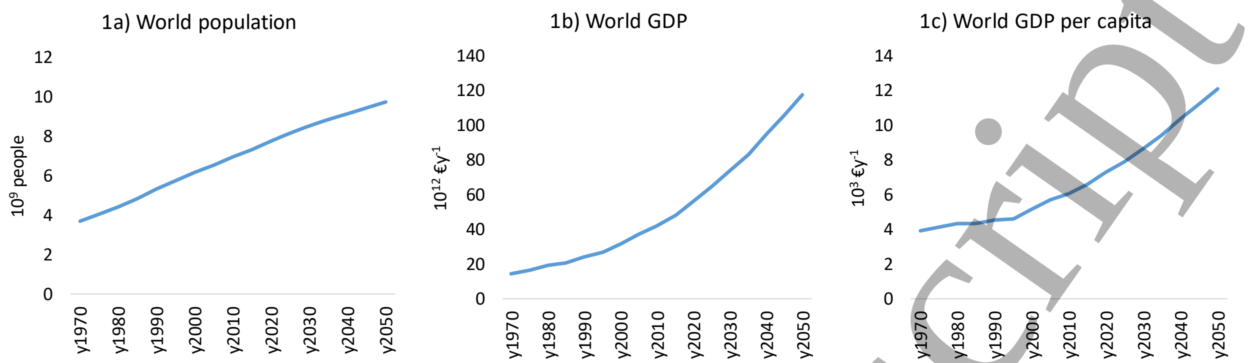

Figure 1: Global projections for population (panel 1a), Gross Domestic Product GDP (panel 1b) and average GDP per capita (panel 1c) 1970-2050. Sources: World Bank (2018) for historical years and projections consistent with IEA-WEO 2018.

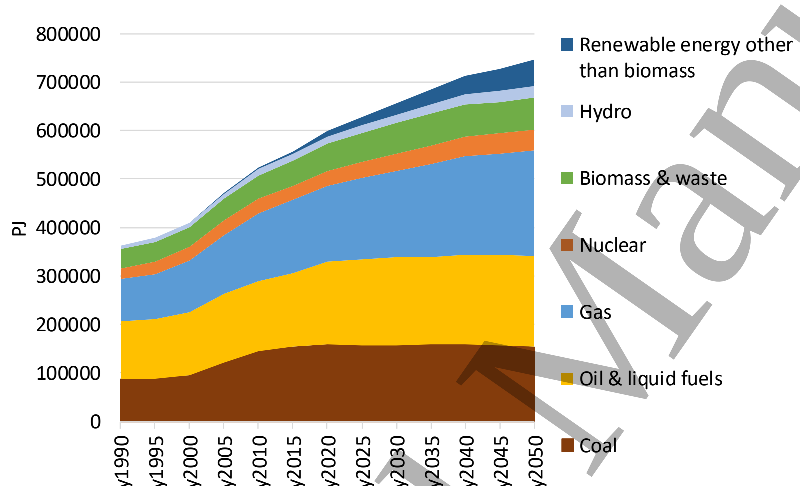

Figure 2: Global energy consumption by fuel in the IEA-WEO2018 New Policies Scenario.

Agricultural activity data are taken from FAOSTAT (2018) with projections aligned to the most recent forecast of FAO (Alexandratos and Bruisma, 2012) and complemented with data from national sources e.g., reporting to UNFCCC (2018) and EUROSTAT (2016) for information about manure management practices, farm sizes etc. The historical and projected changes in global livestock numbers are illustrated in Figure 3.

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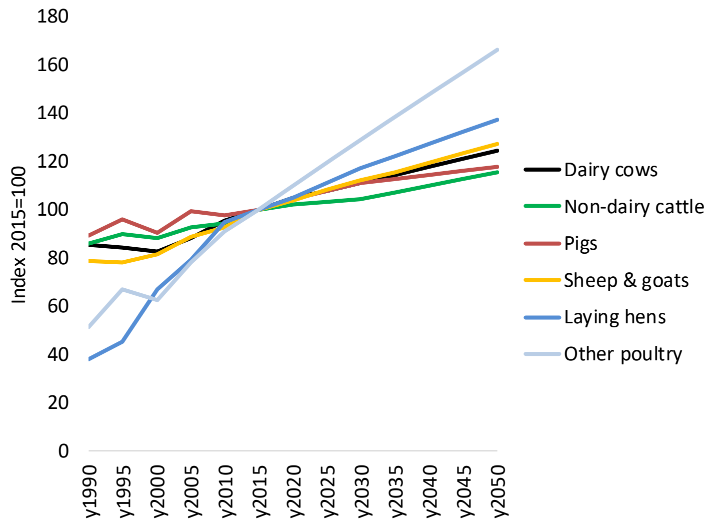

Figure 3: Changes in global livestock numbers relative year 2015 (=100) (FAOSTAT, 2018 and Alexandratos and Bruisma, 2012).

Activity data for the waste and wastewater sectors are derived in GAINSv4 using the methodology described in the Supplement of Gómez-Sanabria et al. (2018). Drivers for the generation of municipal solid waste (MSW) are GDP per capita and urbanization rate, here in consistency with macroeconomic assumptions of the IEA-WEO2018 (see Figure 1). Elasticities for MSW generation by income group are estimated from historical data and reflect the relative increase in average per capita waste generated in response to a relative increase in the average per capita income and urbanization rate. As shown in Gómez-Sanabria et al., higher waste generation elasticity estimates are found for countries with higher incomes. At lower income levels, households primarily generate food waste, while at higher average income levels it is primarily the generation of non-food waste that increase with income. Figure 3 illustrates the global gross generation of waste (i.e., before disposal through scattering, landfill, recycling, incineration or other treatment) for the period 1970-2050 as estimated within the GAINSv4 model. Because of slow decomposition of organic waste in landfills, we account for a time-lag of up to 20 years between disposal of waste to a landfill and the release of CH_4 emissions. To estimate emissions from the year 1990 onwards, it is therefore necessary to estimate waste generation already from the year 1970. As shown in Figure 4, the growth rate for the generation of global municipal solid waste is estimated to increase after 2010, with global amounts growing by 4.5 percent between 2005-2010 and by 14 percent between 2010-2015. Note that for the waste sector the baseyear for projections is 2010 and the 2015 estimate is a model result. The strong increase in global MSW generation between 2010 and 2015 is mainly driven by an expected 20 percent increase in MSW generation in China and India, which follows from the application of a higher MSW generation elasticity as several provinces move into higher average income segments between 2010 and 2015. Although a model result in GAINSv4, the higher growth rate for China after 2010 is confirmed empirically by Chhay et al. (2018) who find that collected and transported MSW in China increased by 1.5 percent between 2005 and 2010 and by 21 percent between 2010 and 2015.

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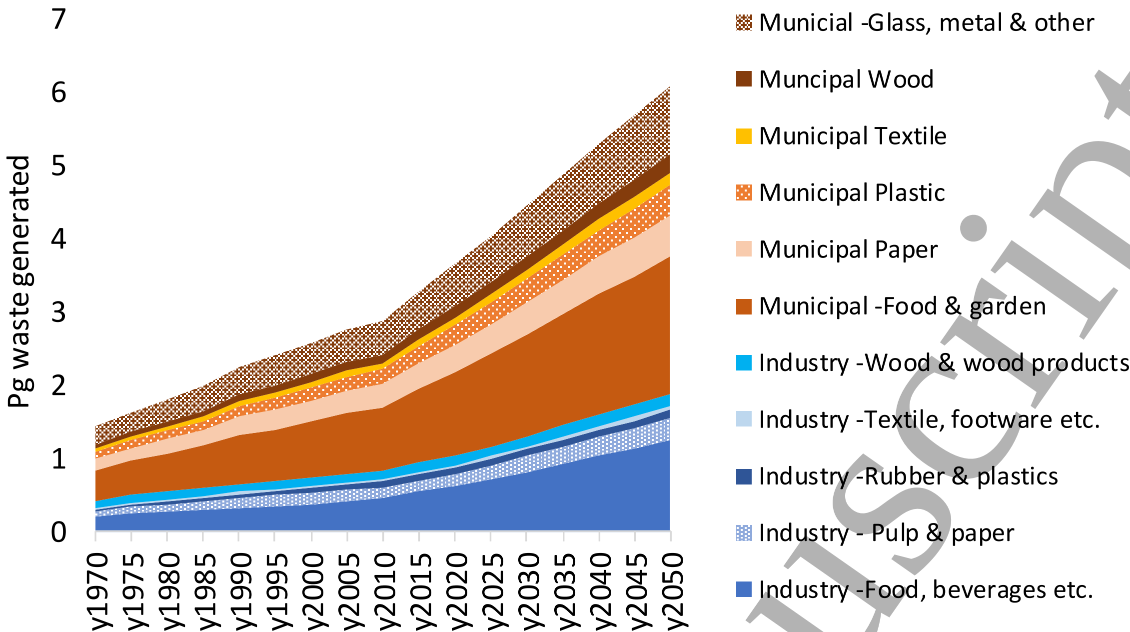

Figure 4: Global generation of solid waste 1970-2050 from municipal and manufacturing industry sources (gross, i.e., before recycling or treatment). Estimated in GAINSv4 in consistency with population and macroeconomic projections of the IEA-WEO2018 following the methodology described in Gómez-Sanabria et al. (2018).

2.3 Emission factors and current control legislation

Sector-specific emission factors are identified both for a no control case and for each control technology applicable to the specific sector in a country. Emission factors are adopted from country-specific information and/or derived in a consistent manner across countries from information on factors determining the country-specific emission factors. Table 2 presents a selection of the most important information sources for CH_4 emission factors in GAINSv4 with a focus on updates made after the publication of Höglund-Isaksson (2012). In addition, a wealth of national information has been fed into individual emission factor estimates, as documented in Höglund-Isaksson (2012, 2017), Höglund-Isaksson et al. (2015, 2018), and Gómez-Sanabria et al. (2018). More sector details are available in Section S6 of the SI.

An implicit assumption in the development of the baseline scenario is that it considers effects on current and future CH_4 emissions from regulations and legislation already adopted as of Dec 2018. Table S4-1 in the SI presents a list of implemented national and regional legislation with direct or indirect impacts on CH_4 emissions that have been considered in the GAINSv4 baseline scenario. Note that future mitigation potentials and associated costs are always assessed as additive to the baseline. Emission reductions and costs incurred by abatement options adopted already in the baseline are not reflected in the estimation of future mitigation potentials and costs.

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Table 1: Principal sources of information for CH_4 emission factors in the GAINSv4 model.

Major source sector	Source sector	Emission factors -prinicpal sources of information
Agriculture	Beef cattle	Livestock emission factors consistent with national reporting to UNFCCC 2016; 2018, complemented with national sources e.g., Xue et al., 2014; Yu et al., 2018; Hansen et al., 2018; FAO 2017ab. For details, see Section S6.4 in SI.
	Dairy cows	
	Sheep Goats etc	
	Pigs	
	Poultry	
	Rice cultivation	IPCC (2006) guidelines (Vol.4, pp 5.45-5.49), complemented with national reporting to UNFCCC (2016,2018) on water regimes and flooding days per year when available.
	Agr waste burning	IPCC (2006) guidelines Section 5.4.2.
Energy	Coal mining	Emission factors aligned with national reporting to UNFCCC (2016) with revisions for China (Peng et al., 2016; China BUR to UNFCCC, 2017; Miller et al., 2019; Sheng et al., 2019), see Section 6.1 in SI and Section 2.6 in SI of Höglund-Isaksson (2012) for details.
	Abandoned coal mines	USEPA (2017) and emissions reported to UNFCCC (2018) for Annex-1 countries, complemented with the assumption of 10% of active hard coal mine emissions, as derived from USEPA (2017), see Section S6.2 in SI for details.
	Domestic energy use firewood	For residential sources, emission factors specified by type of boiler and fuel (Johansson et al., 2004; Kjällstrand and Olsson, 2004; Olsson and Kjällstrand, 2006; Delmas, 1994). For non-residential stationary sources and mobile sources, default emission factors from IPCC (2006; Vol.2, pp.2.16-2.23 and p.3.24).
	Domestic energy use other	
	Industry energy use other	
	Powerplant energy use other	
	Domestic energy use gas	Emission factors for long-distance gas transmission and gas distribution networks (residential and non-residential, respectively) have been aligned with national reporting to UNFCCC (2016) when available, complemented with default factors from IPCC (2006; Vol 2, pp4.48-4.62, Tables 4.2.4 and 4.2.5).
	Industry energy use gas	
	Powerplant energy use gas	
	Gas transmission	
	Gas production	Emission factors from Höglund-Isaksson (2017); US emission factors updated (Alvarez et al., 2018; Zavala-Araiza et al., 2015; Omara et al., 2016), corresponding to average leakage rates of 1% for conventional natural gas, 2.66% for shale gas, 0.58% for coal bed methane (CBM), and 1.65% for tight gas, see Section S6.3 in SI for detials.
	Oil production	Emission factors from Höglund-Isaksson (2017) in consistency with Dalsøren et al. (2018), but with updates for Russian associated gas composition (Huang et al., 2015) and flared gas volumes in 2015 (Elvidge et al., 2016), see Section S6.3 in SI for details.
	Oil refinery	Default emission factors from IPCC (2006; Vol.2, p.4.34, pp.4.52-4.61). For details see Section 2.2. in SI of Höglund-Isaksson (2012)
	Transport Road and Off-Road	COPERT (EMISIA, 2013)
Industry	Industry Brick kilns	AIT (2003)
Waste	Solid waste industry	Emission factors are specified by waste flow for fourteen different waste treatment options, see Gomez-Sanabria et al. (2018) for details on references.
	Solid waste municipal	
Wastewater	Wastewater industry	
	Wastewater domestic	

2.4 Technical mitigation potential and costs

The mitigation potential assessed in the marginal abatement cost curves of the GAINSv4 model refers to feasible reductions in emissions through adoption of technologies defined as installations or applications of physical equipment or material, or modifications in physical parameters affecting emissions. In the short-run, immediate adoption of control technology is assumed constrained by lock-in of investments into existing technology, with successive phase in of new technology modelled

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by sector over the period 2020-2035 and with full effect on emissions from implementation of maximum technically feasible reductions (MFR) only achievable from 2040 onwards. The GAINSv4 baseline scenario assumes no effects on costs and removal efficiencies from technological development as it is assumed that any incentives to adopt (and therefore further develop) emission control technology rely heavily on the existence and stringency of policies directly addressing CH_4 emissions. Hence, without further policy incentives, there are assumed to be no further driver for technological development, which means emission factors for a given technology remain constant over time in the baseline. An exception could be technologies that simultaneously reduce CH_4 emissions and recover/save gas that can be utilized for energy purposes. Adoption of such technologies may arise spontaneously if the future price of gas become high enough to make gas recovery profitable. As the development in future fuel prices is highly uncertain, such technology uptake is not reflected in the baseline scenario, but treated as a future mitigation potential available at a negative cost. In contrast to the baseline scenario, GAINSv4 mitigation scenarios for CH_4 assume additional policy incentives are indeed put in place to stimulate both uptake and further development of CH_4 abatement technology. Assumptions in GAINSv4 about the effects of technological development on removal efficiency and costs for CH_4 mitigation options are presented in Table S5-1 of the SI. Justifications for these assumptions are based on empirical findings of observed developments in control technology following introductions of NOx and SO_2 regulations in the US (Popp, 2003), Japan (Matsuno et al., 2010) and Sweden (Höglund-Isaksson and Sterner, 2010) in the 1990s, as presented in Section 2.5.1 of Höglund-Isaksson et al. (2018).

Unit costs for mitigation of CH_4 per unit of activity are in GAINSv4 calculated as the sum of investment costs, labour costs, non-labour operation and maintenance costs, cost-savings due to recovery or saving of electricity, heat or gas, and non-energy cost savings like avoidance of landfill fees. Unit costs are expressed in constant 2010 Euros per unit of activity. Country and sector specific annual average wages for the agricultural and manufacturing industry sectors are taken from LABORSTA (ILO, 2010) for historical years. Growth in average future wages is proportional to the expected future development in GDP per capita with sector adjustments consistent with growth in sector value added as provided by IEA-WEO (2018). The cost-saving of energy recovery from biogas production or reduced leakage of natural gas during production, transmission and distribution is set equal to the expected future electricity or gas consumer price in industry as taken from the IEA-WEO (2018) New Policies Scenario. Gas recovery refers to the recovery of gas of an upgraded quality of 97 percent CH_4. For some mitigation options, e.g., when biogas is recovered from large-scale anaerobic digestion of food and organic waste, upgrading from 60 to 97 percent CH_4 is necessary for supplying the gas to the grid (Persson, 2003). Costs for upgrading gas have in these cases been included in investment costs.

The total mitigation cost in sector s, country i and year t is defined for sets of application combinations of the possible technologies applicable to the sector. For a given country, year and sector, a technology setting is defined such that the sum of all application rates $Appl_{itsm}$ of possible technologies m (including the no control option) is always unity. The total cost of each technology setting is defined as:

$$TC_{its} = \sum_m \left[A_{its} * C_{itm} * Appl_{itsm} \right], \tag{3}$$

where A_{its} is the activity level, C_{itm} is the cost per unit of activity and $\sum_m Appl_{itsm} = 1$.

The country- and year- specific average cost per unit of reduced emissions is first calculated for each technology available by dividing the unit cost with the difference between the technology emission factor and the no control emission factor, such that:

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$$AC_{itm} = \frac{C_{itm}}{ef_{it}^{No\_control} - ef_{itm}}. \quad (4)$$

Within a sector, the available technologies are first sorted by increasing average cost. The technology with the lowest average cost is ranked the first-best technology and assumed adopted to its maximum applicability in a given sector. The second-best technology has the second lowest average cost and is assumed available for adoption provided it can achieve an emission factor that is lower than the first-best technology. The marginal cost of the second-best technology when implemented in the marginal abatement cost curve (MACC) is the unit cost divided by the additional emission reduction still available for a given sector, i.e.

$$MC_{it2} = \frac{C_{it2} - C_{it1}}{ef_{it1} - ef_{it2}}. \quad (5)$$

In a similar manner, each additional technology available in a sector is added on top of the next best available technology. The result is a MACC built up technology-wise by sector, country and year. Note that if most of the technical abatement potential is exhausted with the first-best technology, the marginal cost of subsequent technologies becomes very high due to the limited additional emission reduction potential. Note also that a technology with both a higher average cost and a higher emission factor than another technology available to a sector will not be adopted at all, since it is both less effective in reducing emissions and comes at a higher cost than other available technologies. Finally, abatement technologies are not always additive, but can also be partly complementary. This is the case e.g., for measures addressing emissions from rice cultivation and enteric fermentation in cattle. For these sectors, we have constructed “combined technologies”, which reflect the overall effect on emissions and costs when more than one measure are implemented simultaneously. For rice cultivation, the first-best technology is improved water management by extending the periods fields are dried out. The second-best technology is improved water management combined with low-CH_4 hybrids and use of soil enhancing amendments. For enteric fermentation in cattle, the first-best technology is breeding for enhanced productivity and animal health and fertility, while the second-best option is to combine breeding with different animal feed changes.

2.5 Uncertainty

Uncertainty is prevalent along many different dimensions both in the estimations of emissions, abatement potentials and costs. When constructing global bottom-up emission inventories at a detailed country and source level, it is inevitable that some information gaps will be bridged using default assumptions. As it is difficult to speculate about how such sources of uncertainty affect resulting historical and future emission estimates, we instead address uncertainty in historical emissions by making comparisons to estimates by other publicly available and independently developed bottom-up inventories, i.e., EDGARv4.3.2 (2018) and CEDS-CMIP6 (2017), and various top-down estimates consistent with atmospheric measurements and inverse model results (e.g., Saunois et al., 2016). Comparisons of global historical CH_4 emission estimates are presented in Section 3.1 and by World region in Section S2 of the SI. The bottom-up inventories adhere to the recommended guidelines of the IPCC (2006), however the flexibility in the recommended methodologies is large as it depends on the availability and quality of the gathered source information. There is accordingly a wide range of possible sources of uncertainty built into estimations in these comprehensive efforts. Having a pool of independently developed inventories, each with its own strengths and weaknesses, can improve the understanding of the scope for uncertainty in these estimates.

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Regarding uncertainty in emission projections and as already discussed in Section 2.2, we only produce one baseline scenario, which is consistent with the economic and energy sector developments of the IEA-WEO (2018) New Policies Scenario. Providing a range of baselines describing different future developments in the activity drivers is out of scope of this study as the intention here is to focus on the relative technical mitigation potentials and costs for reducing emissions at the region, sector and technology level.

Uncertainty in cost estimations is generally high. This is partly a feature of the many dimensions along which uncertainty enters into cost estimates and partly a general lack of detailed information on abatement costs in the literature. There are some uncertainty features that are more systematic than other as they derive from more general assumptions about how investors make decisions about adoption of control technologies. To account for the uncertainty range caused by these particular assumptions, we estimate a range for the marginal abatement cost curves (MACCs). The upper range limit represents the most pessimistic case in the sense that we assume no further technological development and that marginal abatement costs reflect a private investor perspective. Private investors are assumed to operate with a ten percent interest rate on fixed investments, a maximum investment perspective limited to ten years, and no speculation about an expected future increase in energy prices but only considering current (here referring to projected 2020) energy prices when deciding on investments. The lower range limit of the MACC represents the most optimistic case assuming the cost perspective of a social planner and with improving removal efficiencies and declining abatement costs over time due to technological development. A social planner is assumed to take decisions based on a four percent interest rate for fixed investments, considering the entire expected lifetime of the technology, and a future increase in energy prices as expected in the projections of the IEA-WEO (2018) New Energy Policies scenario. Why is it of interest from a climate policy point of view to consider both private investor and social planner perspectives on future abatement costs? The reason is that a social planner, when looking to balance the costs and benefits of climate change mitigation against those of other areas of public spending, e.g., health and education, will need to make such trade-offs on the basis of a low discounting of future values in order to secure opportunities for decent lives also for coming generations. Hence, the social planner's MACCs are suitable for taking decisions about targets for emission reductions that will optimize social welfare. When considering implementation of policies that will actually achieve the socially optimal emission reduction targets, policy maker ought to rely on MACCs estimated from the private investor perspective. These reflect better the higher marginal abatement costs (and higher carbon price levels) needed for private investors to find it profitable to invest in abatement at a level that meets the desired emission reduction targets (Baumol and Oates, 1971).

3 Results

3.1 Historical anthropogenic CH_4 emissions 1990-2015 in GAINSv4

For a good understanding of future emissions, we must first understand the current level and source attribution of emissions. We therefore develop a global inventory of annual CH_4 emissions 1990-2015 and compare it to other global bottom-up inventories as well as to top-down inverse model results. GAINSv4 bottom-up estimates of global anthropogenic CH_4 emissions 1990-2015 are presented in Figure 5. GAINSv4 does not include estimates of emissions from forest fires and savannah burning due to a lack of detailed country-specific information. For the purpose of illustrating total anthropogenic CH_4 emissions in Figure 5, the GAINSv4 estimate of all other CH_4 sources has been complemented with the global estimates of emissions from forest fires and savannah burning from the GFEDv4.0 database (Randerson et al., 2018).

GAINSv4 estimates a decline in global CH_4 emissions in the first half of the 1990s, primarily a consequence of the collapse of the Soviet Union and the associated general decline in production levels in agriculture and fossil fuels (see Regional emission illustrations in Figure S2-1 of the SI). In addition, as described by Evans and Roshanka (2014) and assumed in Höglund-

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Isaksson (2017), venting of associated petroleum gas declined significantly in Russia due to an increase in flaring. It is unclear why this happened, but a possible explanation could be that the privatization of oil production in this period meant that the new private owners were less willing to take the security risks of venting and invested in flaring devices to avoid potential production disruptions. This hypothesis is however yet to be confirmed. Global CH_4 emissions are estimated to remain relatively constant in the second half of the 1990s, but then start to increase in the first few years of the new millennia. This time the primary drivers for growth in emissions are a mix of sources; increased coal mining in China, increased oil and/or gas production in Russia and Africa, rapidly expanding cattle rearing in Latin America, and increased generation of waste and wastewater in China, India and the rest of South-East Asia. The latter driven by population and rapid economic growth. Between 2008 and 2010 there is a brief downturn in emissions following a general decline in economic activity in response to the global financial crisis. After 2010 emissions increase again with principal drivers being; rapidly growing extraction of unconventional gas in North America, increased coal mining in Indonesia, and accentuated growth in waste and wastewater emissions in all rapidly developing regions of the world, including China, India, the rest of South-East Asia, Latin America, and Africa. The latter development would offer a possible explanation to observed increases in atmospheric CH_4 from biogenic sources in tropical regions (Nisbet et al., 2014; 2019). It should however be noted that there is also a small but steady increase in global emissions from livestock, in particular beef and dairy. Emissions from pigs have however seen a slight decline in the last decade due to an expansion in the use of biogas digesters in Europe for treatment of pig manure.

In Figure 5, the GAINSv4 bottom-up estimates are compared with the average top-down estimates of anthropogenic emissions following from inverse model results reconciling bottom-up with top-down measurements of the CH_4 concentration in the atmosphere. Saunois et al. (2016) provide such estimates for three time periods: 2000-2009, 2003-2012, and 2012. As shown, these estimates align quite well with the GAINSv4 bottom-up estimates. Figure 6 illustrates the average and full uncertainty ranges for top-down estimates of emissions by groups of CH_4 isotopic signatures identifiable in the atmosphere and mentioned e.g., in Saunois et al (2016) and Dlugokencky et al., (2011). The isotopic signatures make it possible to distinguish between atmospheric CH_4 from biogenic (agriculture and waste) sources, fossil fuel sources, and burning of biomass sources. GAINSv4 estimates fall within the uncertainty ranges of the atmospheric measurements for all three CH_4 isotopic signature groups. For the biogenic sources presented in Figure 6a, GAINSv4 estimates are close to those by CEDS-CMIP6 (2017) and lower than those by EDGARv4.3.2 (2018). The higher CH_4 emissions from biogenic sources in EDGARv4.3.2 can primarily be attributed to higher annual emissions from wastewater sources than in GAINSv4 (see Table 5.3 in Höglund-Isaksson et al., 2015), in particular for Africa and South-East Asia where GAINSv4 assumes poor conditions for CH_4 formation in areas lacking proper infrastructure for centralized wastewater collection. For fossil fuel sources presented in Figure 6b, the average top-down estimate of CH_4 by Saunois et al. is somewhat lower than the GAINSv4 estimate from year 2000 onwards and considerably lower than the CEDS-CMIP6 estimate for the later years, as discussed in detail below. For emissions from burning of biomass and biofuels presented in Figure 6c, the sum of the GAINSv4 estimate of CH_4 emissions from burning of agricultural waste residuals and the GFEDv4.0 estimate of global CH_4 emissions from forest fires and savannah burning, reveals that the GAINSv4 estimate for these sources falls somewhat short of the average top-down estimate.

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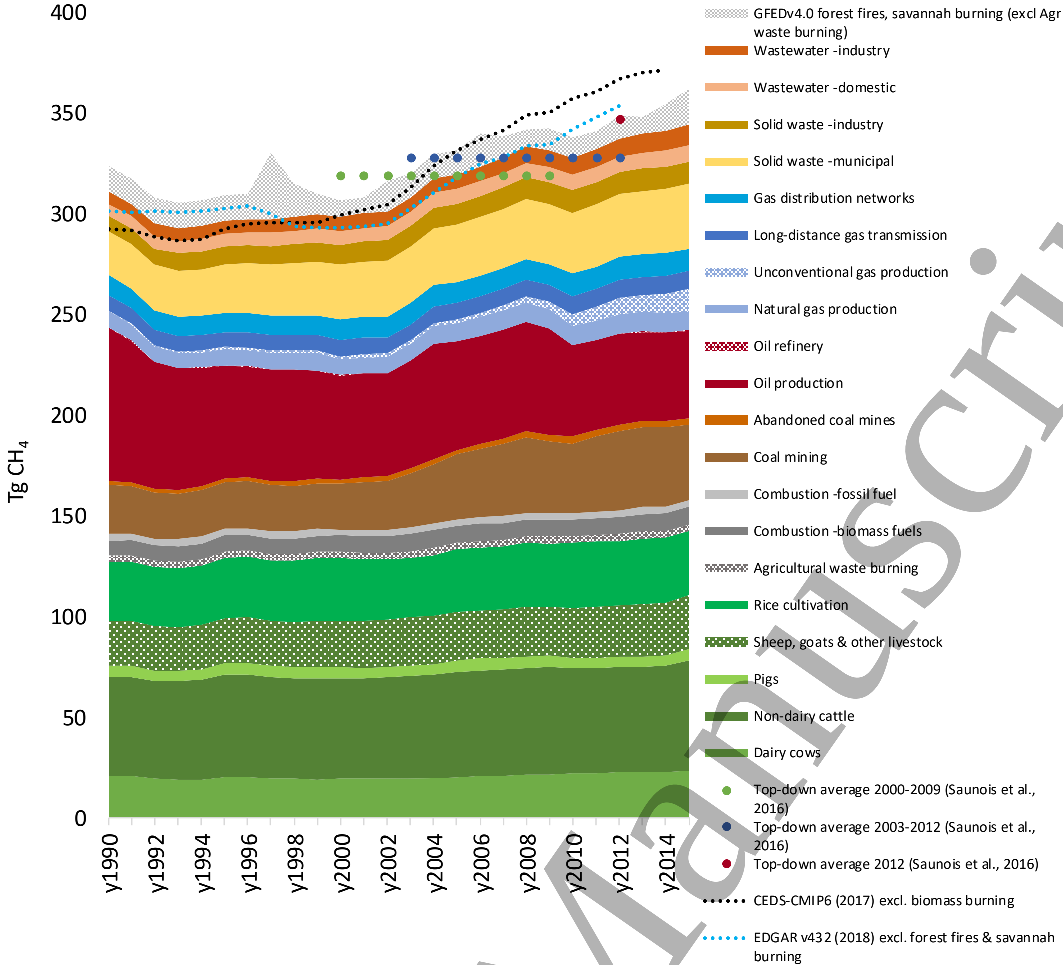

Figure 5: Annual bottom-up estimates of global anthropogenic CH_4 emissions 1990-2015 in GAINSv4 and in comparison to top-down average estimates from Saunois et al., 2016. Note that global CH_4 emissions from forest fires and savannah burning are taken from GFEDv4 (Randerson et al., 2018).

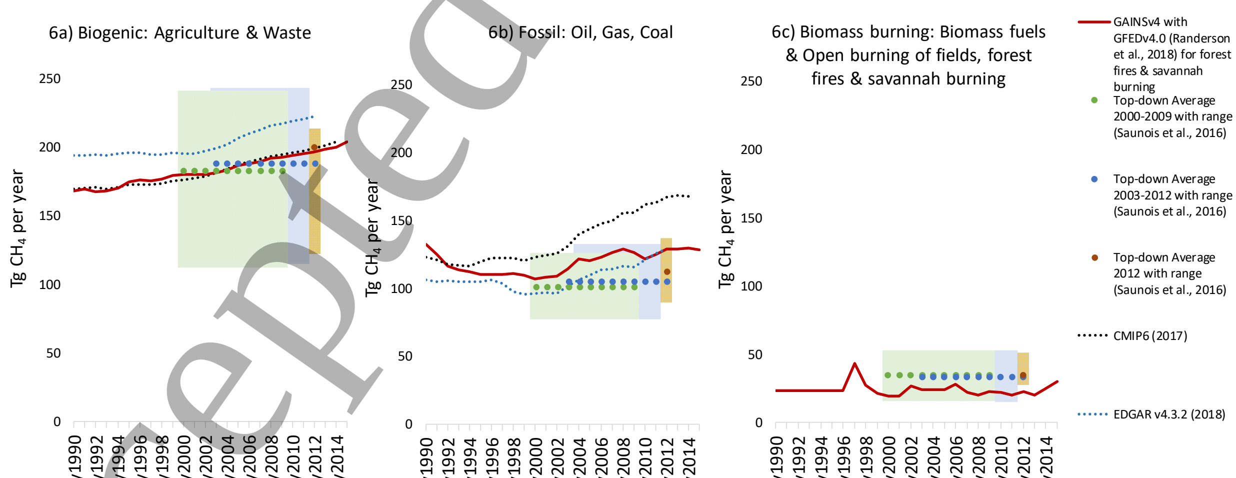

Figure 6: GAINSv4, CMIP6 and EDGARv4.3.2 bottom-up estimates of global anthropogenic CH_4 emissions by CH_4 isotopic signatures and in comparison to the uncertainty ranges (depicted as boxes) and average estimates (depicted as dots) for top-down atmospheric measurements as reported in Saunois et al., 2016.

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Figure 7 displays the estimates of CH_4 emissions from fossil fuel sources by hydrocarbon source and global bottom-up inventory (for further details see Section S3.3 of the SI). In panel 7a, GAINSv4 shows fairly constant estimates of annual emissions of about 80 Tg CH_4 from global oil and gas systems between 1995-2015. Looking closer we see that this seemingly stable emission level is the result of steadily increasing emissions from natural gas extraction, driven by increased gas production in general and shale gas production in particular, and a simultaneous steady decline in emissions from oil extraction. The latter is referred to increased recovery rates for associated petroleum gas, particularly in Russia and parts of Africa (Höglund-Isaksson, 2017). Emissions from oil and gas systems are in the CEDS-CMIP6 and EDGAR v4.3.2 inventories reported as aggregates and it is therefore difficult to know whether the same developments in oil and gas production emissions, respectively, are prevalent also in these inventories. Panel 7b shows how global emissions from coal mining (including from abandoned coalmines) develop over time in the different bottom-up inventories. While GAINSv4 and EDGARv4.3.2 agree quite well, CEDS-CMIP6 estimates considerably higher emissions from this source, in particular for China in the period post-2005. The basis for the higher emissions from coal mining in China in CEDS-CMIP6 is not clear, however, consistent with higher emissions from this source in previous versions of EDGAR (see Table 5.3 in Höglund-Isaksson et al., 2015). Recent results of inverse models (Miller et al, 2019; Sheng et al., 2019) find considerably lower CH_4 emissions from coal mining in China, indicating that also estimates by GAINSv4 and EDGAR 4.3.2 may be on the higher side.

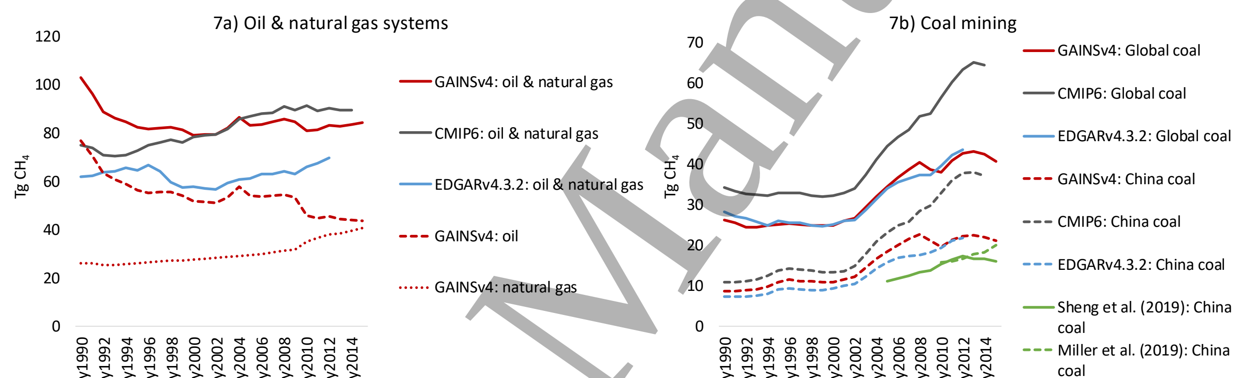

Figure 7: Global fossil fuel CH_4 emissions by source and bottom-up inventory. Global emissions from oil and natural gas systems in Panel 7a and from coal mining activities in Panel 7b.

3.2 Baseline scenario for global anthropogenic CH_4 emissions 1990-2050

A global projection of baseline anthropogenic CH_4 emissions to 2050 consistent with the energy sector developments of the IEA-WEO (2018) New Policies Scenario, is presented in the left panel in Figure 8 in five-year intervals. Baseline emissions are expected to increase close to linearly by about 3 Tg CH_4 per year or 30 percent between 2015 and 2050. Global emission increases are primarily driven by an expected increase in solid waste generation as population grows and countries become richer and by an expected increased extraction of unconventional natural gas. The latter is partly a reflection of a substitution of coal with natural gas and renewables projected in the IEA-WEO (2018) New Policies Scenario and goes together with a decline in emissions from coal mining in the period post-2030 in that particular energy scenario.

Baseline emission developments at a regional level are presented in Figure S3-1 in the SI. For China, baseline CH_4 emissions are expected to continue growing to 2040, but then level off at an annual emission level of about 65 Tg CH_4 due to a decline in coal mining. A strong increase in CH_4 emissions from shale gas production in North America is expected to continue until 2045, when emissions decline due to a projected drop in gas demand in the IEA-WEO2018 New Policies scenario. Due to already adopted climate policy strategies, the European Union is expected to be on track for a decline in CH_4 emissions by

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about 20 percent between 2015 and 2030, however, further reductions will need implementation of additional policy incentives. Continued growth in population and income are expected to drive increases in waste and wastewater CH_4 emissions in Africa, India & South-East Asia. A continued increase in demand for beef is expected to be the prime driver for increased CH_4 emissions in Latin & Central America, while a continued demand for oil drives emission increases in the Middle East. An expected rapid growth in natural gas production in Australia coupled with no phase-out of coal mining, translate into a steady increase in emissions in Oceanian OECD (Australia, New Zealand and Japan) in the period leading up to 2050.

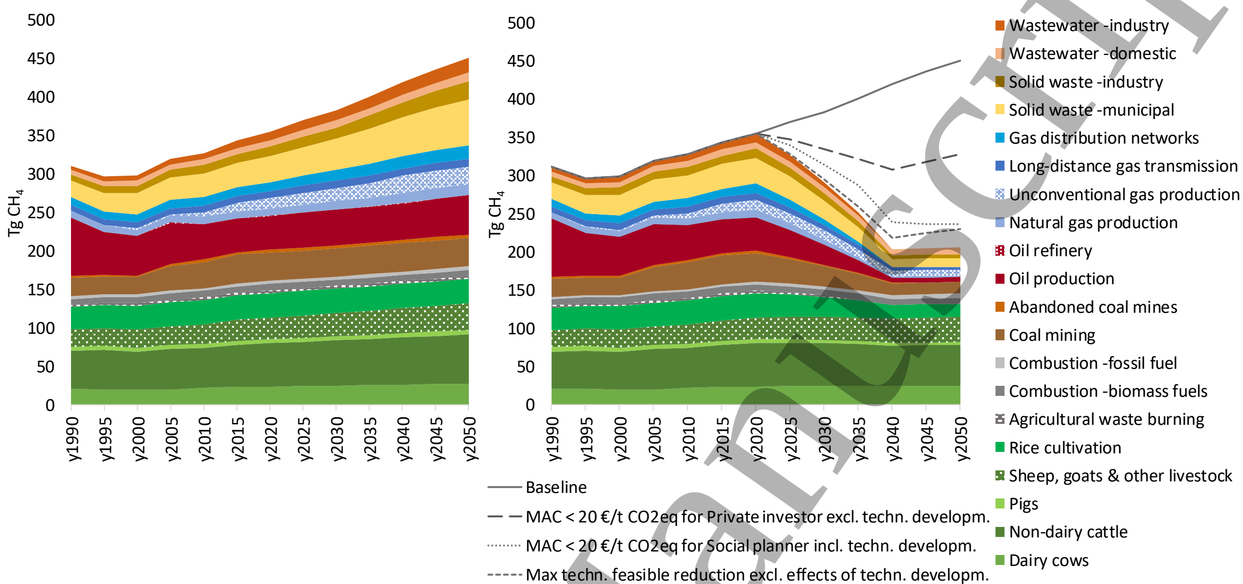

Figure 8: Global anthropogenic CH_4 emissions 1990-2050 in the Baseline scenario (left panel) and with Maximum technically feasible reduction (MFR) including effects of technological development (right panel).

3.3 Technical mitigation potentials in the 2050 timeframe

The maximum technically feasible reduction (MFR) of global anthropogenic CH_4 in year 2050 is estimated at 54 percent below baseline emissions of that year. This corresponds to a global emission level that is 40 percent below the 2015 level and reflects that baseline emissions are expected to grow by 30 percent between 2015 and 2050 (see right panel of Figure 8). The MFR for fossil fuel sources is assessed at 74 percent below baseline in 2050 (see Table 3), assuming full implementation worldwide of at least 98 percent recovery of associated petroleum gas and, in addition, leakage detection and repair (LDAR) programs to reduce unintended leakage during extraction, transmission and distribution of natural gas. Investments into control of fossil fuel emissions would of course become redundant should the World decide on a massive phase-out of fossil fuel use in the next few decades. High technical abatement potentials at about 80 percent below baseline emissions in 2050 are considered feasible for CH_4 emissions from solid waste management. This assumes it possible in a twenty years perspective to extend the infrastructure for source separation, recycling and energy recovery schemes globally, including a ban on all landfill of organic waste and allowing for useful utilization of the carbon content of the waste (Gómez-Sanabria et al., 2018).

The technical abatement potential for agricultural sources is assessed at 21 percent below baseline emissions in year 2050. This includes relatively limited abatement potentials for livestock of 12 percent due to applicability limitations (see Section S3.4. in the SI for details). Large farms with more than 100 LSU contribute about a third of global CH_4 emissions from livestock and for this group we find it technically feasible to reduce emissions by just over 30 percent below baseline emissions in year 2050 (see Figure S6-2 in the SI). The available options include reduction of enteric fermentation emissions through

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animal feed changes (Hristov et al., 2013; Gerber et al., 2013) combined with implementation of breeding schemes that simultaneously target genetic traits for improved productivity and enhanced animal health/longevity and fertility. Increased productivity reduces system emissions by enabling the production of the same amount of milk using fewer animals. The dual objective in breeding schemes is important as a one-eyed focus on increased productivity leads to deteriorating animal health and fertility and a risk that system emissions increase due to a need to keep a larger fraction of unproductive replacement animals in the stock (Lovett et al., 2006; Berglund, 2008; Bell et al., 2011). The enteric fermentation options are considered economically feasible for commercial/industrial farms with more than 100 LSU (Animal change project, 2014) but not for smaller- and medium- sized farms. Breeding schemes are assumed to deliver impacts on emissions only after 20 years and feed changes are assumed applicable only while animals are housed indoor. Emissions from manure management can be reduced through treatment of manure in anaerobic digesters (ADs) with biogas recovery. To be efficient from both an economic and environmental point of view, a certain scale is needed to accommodate both the fixed investment of the AD plant and the time farmers spend carefully attending to and maintaining the process (for details see Section 3.3.1.3 in Höglund-Isaksson et al., 2018). About a third of global livestock CH_4 emissions can be attributed to smallholder farmers particularly prevalent in Africa and South-East Asia. These livestock typically have low productivity and emissions per head and are well adapted genetically to local conditions. We do not consider any technical abatement potential for this group of farmers, because enhanced productivity may not be of primary interest when considering that livestock often fills a dual purpose; beside providing milk and meat it also functions as a mean to store assets and manage risks over time (FAO, 2008; Udo et al., 2011). In absence of access to credit markets and publicly provided health care, the robustness of indigenous breeds may become more important than the increased production that can be achieved by introducing highly productive breeds from abroad. Hence, control of these emissions are closely linked to more general institutional and economic reforms. For CH_4 emissions from rice cultivation, a halving of global emissions is considered possible through improved water management that shorten the period of continuous flooding of fields, combined with a use of low- CH_4 generating hybrids and different soil amendments (see Section S6.5 of the SI for details).

Due to locked in capital of existing technology in the short-run, the cumulative emissions in the MFR scenario is assessed at 38 percent below baseline between 2020 and 2050 (see Table 3). This leaves 7.7 Pg CH_4 or 216 Pg CO_2eq using GWP_{100} from AR5 (IPCC, 2013) released globally between today and 2050 that will likely be difficult to remove through technical solutions. In 2050, MFR leaves 5.7 Pg CO_2eq of CH_4 still released. This is a lot if we consider that to stay at 1.5 degrees warming, IPCC (2018) estimates we must not exceed 10 Pg CO_2eq for all greenhouse gases in 2050 (and be at zero net emissions around 2075). In addition to technical solutions, this calls for widespread implementation in the 2050 timeframe of behavioural options, e.g., human diet changes that reduce meat and milk consumption (e.g, Willett et al., 2019; Springmann et al., 2016; Clune et al., 2017) and general institutional and social reforms indirectly mitigating greenhouse gas emissions in developing countries (Evans and Steven, 2009).

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Table 2: Global baseline and MFR CH_4 emissions in years 2015 and 2050 and cumulative emissions 2020-2050 by source sector.

Emission source sector	Technical abatement options implemented in MFR	Baseline 2015 Tg CH_4	Baseline 2050 Tg CH_4	Emissions in 2050 after Max technically feasible reduction (MFR) Tg CH_4	Emissions in 2050 after MFR: Technical abatement in % below 2050 Baseline	Cumulative emissions 2020-2050: Baseline Tg CH_4	Cumulative emissions 2020-2050: MFR Tg CH_4	Cumulative emissions 2020-2050: Technical abatement in % below cumulative Baseline
Dairy cows	Enteric fermentation: feed changes and breeding to improve productivity and animal health/fertility. Manure management: treatment in biogas digester. Applicable to large farms > 100 LSU.	23.4	27.9	24.8	-11%	804	696	-14%
Non-dairy beef cattle	Enteric fermentation: feed changes and breeding to improve productivity and animal health/fertility. Manure management: treatment in biogas digester. Applicable to large farms > 100 LSU.	55.0	64.0	53.5	-16%	1857	1561	-16%
Pigs	Manure management: treatment in biogas digester.	5.3	5.5	3.2	-42%	165	112	-32%
Sheep & other livestock	Enteric fermentation: feed changes and breeding to improve productivity and animal health/fertility.	26.7	34.3	34.1	-1%	967	881	-9%
Rice cultivation	Improved water management, use of alternative hybrids and soil amendments	32.0	32.1	16.3	-49%	994	659	-34%
Agricultural waste burning	Ban and enforcement of existing bans on agricultural wasre burning.	3.5	3.5	0.0	-100%	110	37	-66%
Combustion of biomass fuels	No technical abatement option identified.	8.5	8.0	8.0	0%	246	220	-10%
Combustion of fossil fuels	No technical abatement option identified.	3.4	5.3	5.3	0%	130	120	-8%
Coal mining	Pre-mining degasification. Ventilation air methane oxidation with improved ventilation.	37.1	36.2	15.3	-58%	1145	666	-42%
Abandoned coal mines	Flooding.	3.5	3.8	0.3	-92%	118	46	-61%
Oil production	Extended recovery of associated gas. Leakage detection and repair programs (LDAR) for unintended leakage.	43.5	51.9	6.1	-88%	1460	612	-58%
Oil refinery & storage	Leakage detection and repair programs (LDAR) for unintended leakage.	0.2	0.2	0.1	-66%	6	3	-46%
Natural gas production	Leakage detection and repair programs (LDAR) for unintended leakage.	9.4	13.8	2.2	-84%	370	162	-56%
Unconventional gas production	Leakage detection and repair programs (LDAR) for unintended leakage.	10.8	22.3	6.6	-70%	592	320	-46%
Gas transmission	Leakage detection and repair programs (LDAR) for unintended leakage.	9.1	10.3	3.8	-63%	305	174	-43%
Gas distribution	Replacement of grey cast iron pipes and doubling of control frequency. Leak Detection and Repair (LDAR) programs.	11.2	17.3	0.4	-98%	461	161	-65%
Municipal solid waste	Source separation with recycling or treatment with energy recovery. No landfill of organic waste.	31.9	60.4	10.9	-82%	1431	653	-54%
Industrial solid waste	Recycling or treatment with energy recovery. No landfill of organic waste.	11.3	23.8	6.2	-74%	533	271	-49%
Domestic wastewater	Upgrade of primary treatment to secondary/tertiary anaerobic treatment with biogas recovery and utilization.	8.0	10.6	7.9	-26%	294	224	-24%
Industrial wastewater	Upgrade of treatment to two-stage treatment, i.e., anaerobic with biogas recovery followed by aerobic treatment.	10.0	18.8	0.2	-99%	464	159	-66%
Total		344	450	205	-54%	12451	7736	-38%
whereof biogenic sources		204	277	157	-43%	7511	5215	-31%
whereof fossil sources		133	164	43	-74%	4700	2364	-50%
whereof biomass burning sources		7	9	5	-40%	240	157	-35%

Figure 9 illustrates the technical CH_4 abatement potentials 2020-2050 by major World region. As expected, the technical abatement potentials are highly region-specific with the largest relative reduction potentials possible in major fossil fuel supplying regions like Russia and the Middle East. Significantly lower reduction potentials are found for regions where agricultural sources dominate CH_4 emissions, i.e., India, Latin America, Oceanian OECD and South-East Asia.

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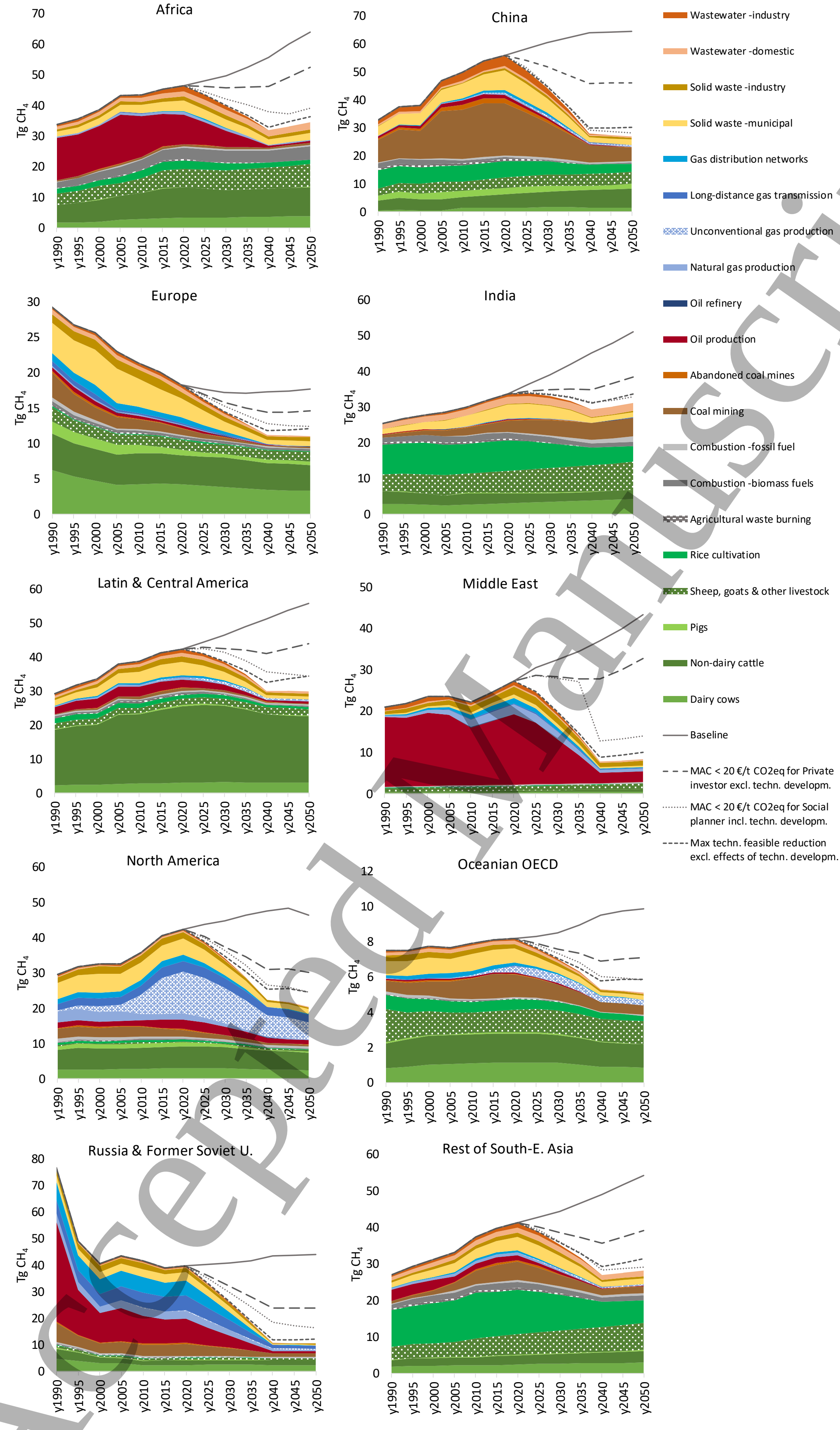

Figure 9: Technical CH_4 abatement pathways to 2050 by major world region and source sector.

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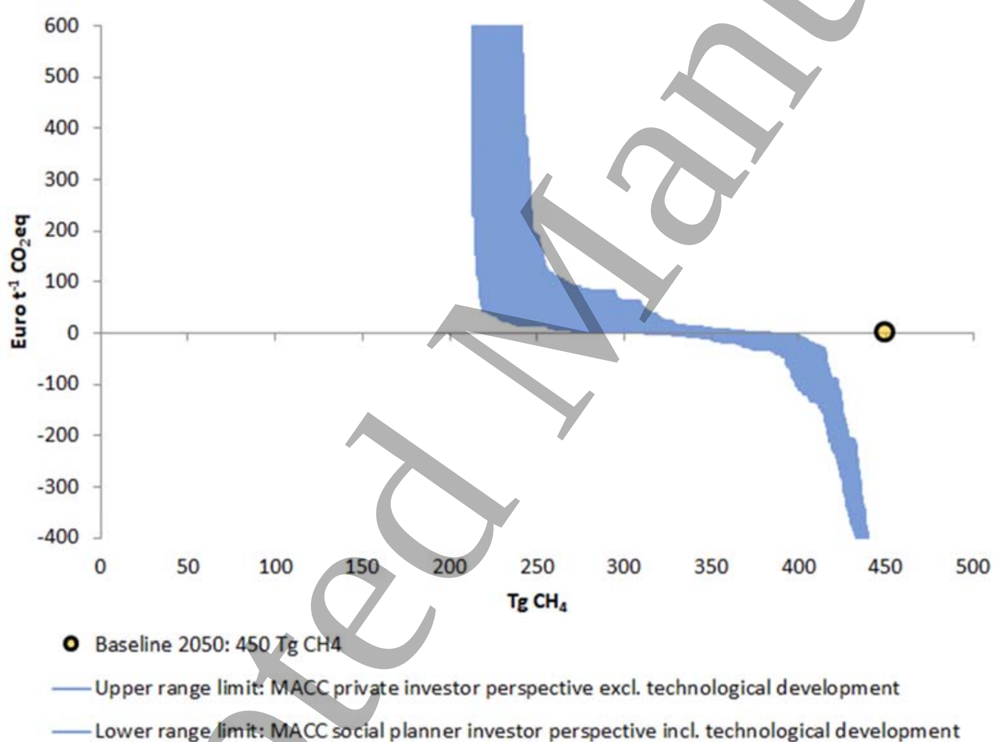
3.4 Marginal abatement cost curves for global CH_4 abatement in the 2050 timeframe

455 The estimated range for the global MACC for CH_4 in year 2050 is presented in Figure 10. The lower range limit of the MACC corresponds to a social planner's perspective and include impacts of technological development, while the upper range limit corresponds to a private investor's perspective and excluding impacts from technological development (see Section 2.5). Starting from a baseline emission level of 450 Tg CH_4 in 2050, a 35 percent reduction is estimated as possible at a zero or negative marginal cost (i.e., at a net profit) at the lower range limit of the MACC, while the same relative reduction would 460 only be possible with the introduction of an additional policy incentive equivalent to 82 €/t CO_2eq at the upper range limit of the MACC. At the lower range limit it is considered possible to almost halve baseline emissions in 2050 at a marginal cost below 20 €/t CO_2eq, while at the upper range limit three quarters of the full baseline emissions are expected to remain at the same marginal cost level. Hence, the marginal abatement costs are highly sensitive to the time and opportunity cost perspective of the investor and to the potential impact from technological development on costs and removal efficiencies. Although policy 465 makers must have a social planner's perspective when determining the optimal allocation of resources to emission abatement in relation to other public goods, they must let a higher MACC guide the setting of carbon price levels to provide enough incentives for private investors to achieve the desired emission reductions in various sectors and regions.



470 **Figure 10: Range of global marginal abatement cost curve (MACC) for CH_4 in year 2050.**

The ranges for the MACCs differ significantly between major source sectors both at a global scale (see Figure 11) and across World regions (see Figure 12). At the lower range limit, more than 85 percent of the global MFR is found attainable at a marginal cost below 20 €/t CO_2eq for all three major source sectors Energy, Agriculture and Waste. At the upper range limit, 475 however, a policy incentive equivalent to the same carbon price level achieves the more modest emission reductions of 57, 71 and 50 percent, respectively. It is evident from the regional analysis that extensive potentials to reduce CH_4 emissions at low costs exist in the fossil fuel production sectors in Russia and the Middle East. Targeting these two sources alone could remove more than 10 percent of global baseline emissions in 2050. An additional almost 10 percent of baseline emissions in 2050 could be removed at a marginal cost below 20 €/t CO_2eq by implementing proper waste and wastewater handling in China,

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India and the rest of South-East Asia. This would likely come with considerable co-benefits in the form of reduced air and water pollution.

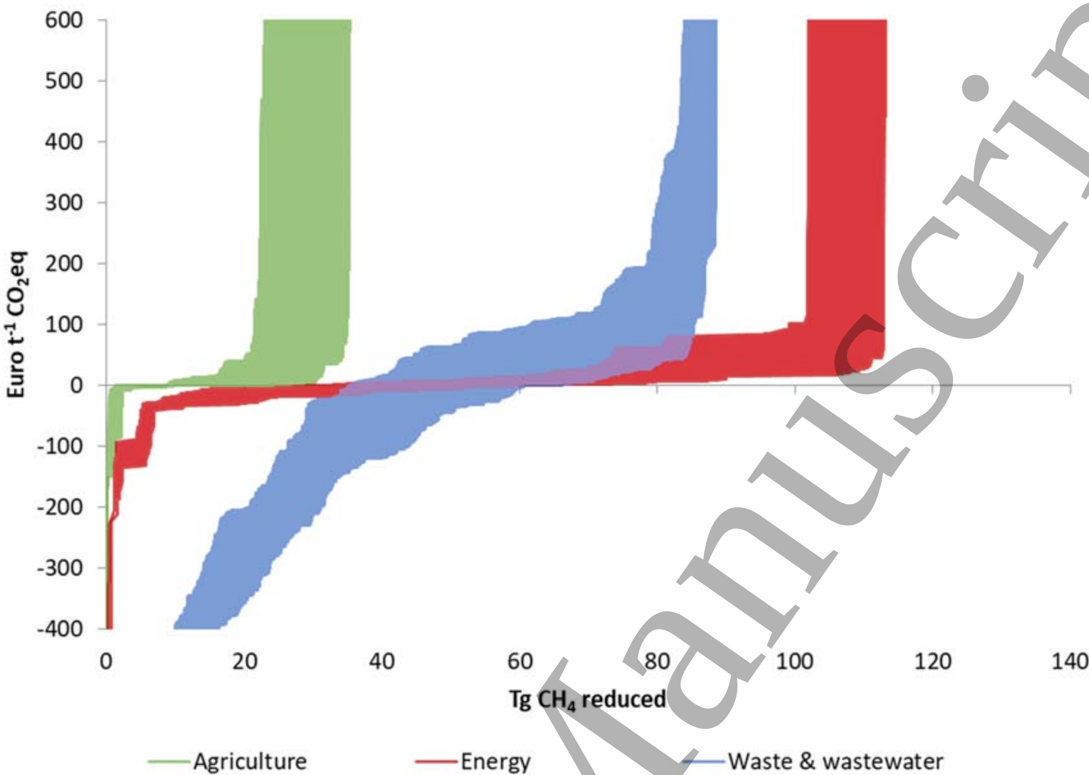

Figure 11: Ranges for global marginal abatement cost curves for reducing CH_4 emissions in 2050 by major source sector.

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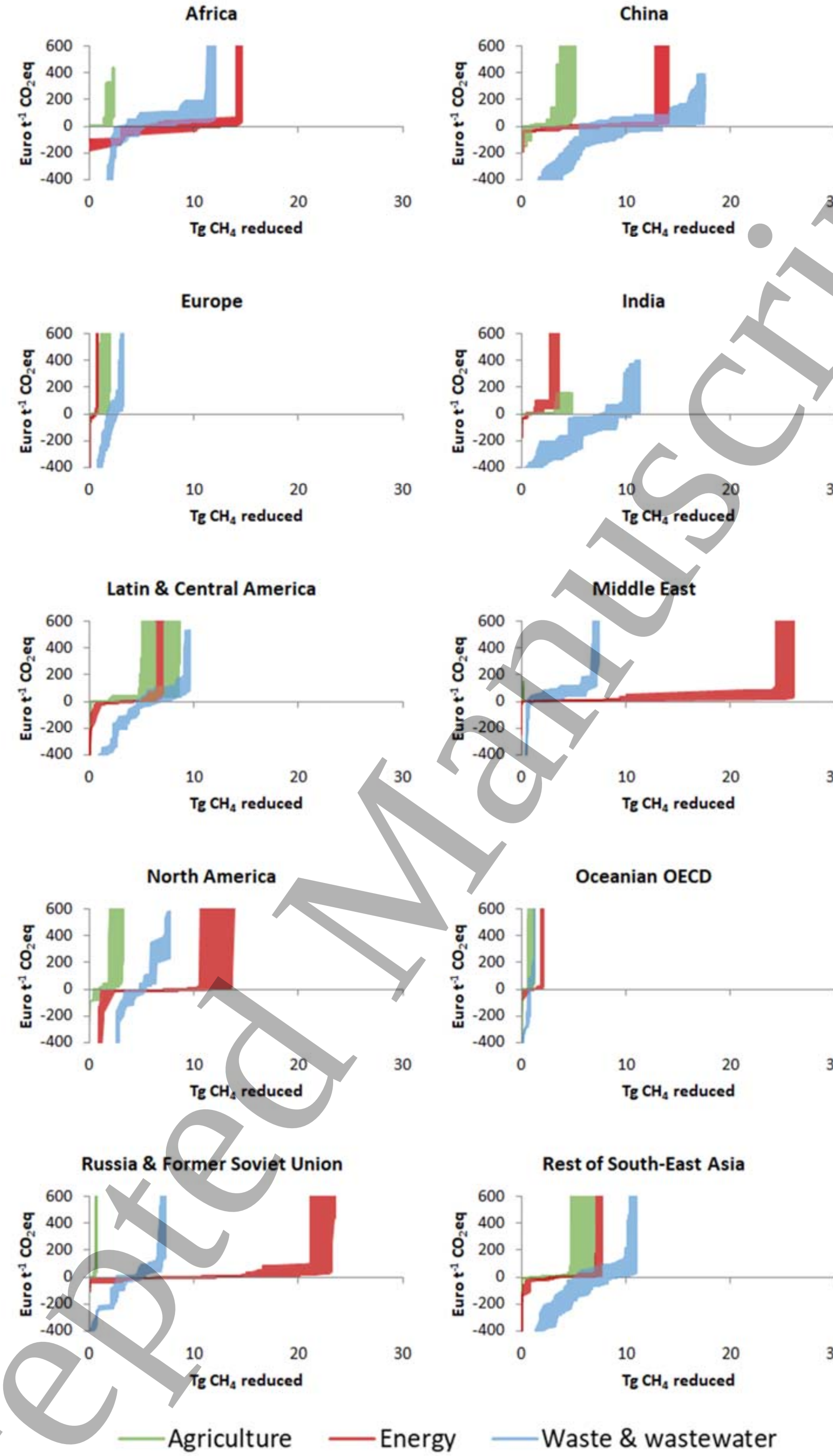

Figure 12: Ranges for marginal abatement cost curves (MACCs) in 2050 by major source sector and world region.

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3.5 Comparison to other studies

The long-run technical abatement potential for global CH_4 emissions in year 2050 has been assessed by Lucas et al. (2007) and Harmsen et al. (2019). Figure 13 illustrates the MFR in total and by sector as estimated in these two studies in comparison to GAINSv4. The different assessments agree fairly well on the long-run technical abatement potential in non-agricultural sectors. Lucas et al. appears generally to be more optimistic than both Harmsen et al. and GAINSv4. The most notable difference is in the assessment of the technical abatement potential for the agricultural sector. Table 3 presents recent estimates from four different studies of global CH_4 mitigation potentials in 2030 and 2050 for this sector. GAINSv4 is slightly more conservative than Beach et al. (2015) in the estimate for 2030, but well within the range estimated in Frank et al. (2018). In the 2050 timeframe, the maximum technically feasible reduction of about 1 Pg CO_2eq in GAINS v4 appears as a middle estimate between the Frank et al. estimate of 0.52 and the Harmsen et al. estimate of 1.7 Pg CO_2eq. The discrepancy can mainly be referred to differences in livestock sector mitigation potentials, where GAINSv4 estimates maximum 12 percent reductions in global manure management and enteric fermentation emissions, respectively. Harmsen et al. estimates 55 and 41 percent reductions for the respective sources and Lucas et al. 50 percent for both sources. This difference can be referred to the applicability limitations introduced in GAINSv4 on the basis of farm size and intensive/extensive systems as discussed in Section 3.3 and Section S6-4 in the SI. Harmsen et al. and Lucas et al. assume almost the same applicability rates for livestock mitigation options across different World regions and no applicability constraints for implementation of enteric fermentation (breeding and animal feed changes) options to the about one third of livestock emissions attributable to smallholder farmers in developing countries. Such applicability constraints apply in GAINSv4 due to the important role livestock herds play in the management of risks for smallholder farmers in Africa and South-East Asia (see Section S6.4 in the SI). GAINSv4 is however considerably more optimistic than Frank et al. about the mitigation potentials of breeding and animal feed changes in year 2050.

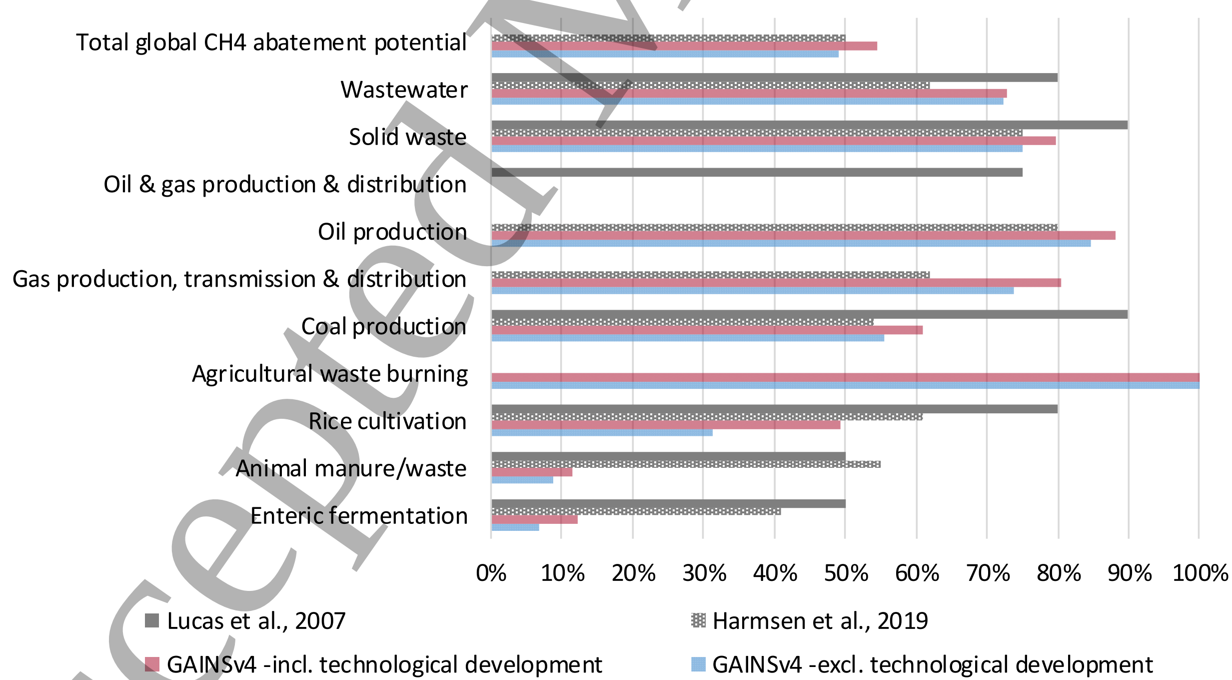

Figure 13: Relative MFR potentials below baseline in year 2050 for global CH_4 emissions in total and by source sector as estimated in GAINSv4 and by Harmsen et al. (2019) and Lucas et al. (2007).

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Table 3: Absolute MFR emission reduction potentials below baseline in 2030 and 2050 for global CH_4 from the agricultural sector, as estimated in GAINSv4 and by Beach et al. (2015), Frank et al. (2018) and Harmsen et al. (2019).

	Maximum technical mitigation potential for CH_4 from global agricultural sources					
	2030			2050		
	Beach et al., 2015	Frank et al., 2018	GAINSv4	Harmsen et al., 2019	Frank et al., 2018	GAINSv4
CH_4 sources	Pg CO_2eq	Pg CO_2eq	Pg CO_2eq	Pg CO_2eq	Pg CO_2eq	Pg CO_2eq
Rice cultivation	0.2	0.2-0.35	0.17	0.37	0.27	0.44
Manure management	0.27	0.04-0.1	0.034	0.13	0.15	0.074
Enteric fermentation		0.03-0.1	0.086	1.2	0.09	0.37
Agric. waste burning	0	0	0.05	0	0	0.10
Total agriculture	**0.47**	**0.27-0.55**	**0.34**	**1.7**	**0.52**	**0.99**

4 Conclusions

Keeping to the Paris Agreement of staying well below two degrees global warming will require a concerted effort to curb methane (CH_4) emissions in the period leading up to 2050. The many diverse sources of CH_4 makes it particularly challenging to design policy instruments that effectively achieve deep emission reductions. A key piece of information for policy-makers is the potential and costs for lowering emissions relatively fast through implementation of technical solutions in various source sectors and world regions. The purpose of this study is to provide such information by exploring future technical abatement pathways for CH_4 using the most recent version of IIASA's Greenhouse gas and Air pollution Interactions and Synergies (GAINS) model.

With a global annual inventory for 1990-2015 as starting point for future projections, a baseline emission scenario to 2050 is developed against which the technical abatement potentials and costs are assessed at a country, sector and technology level. Globally, we find extensive technical opportunities at low costs to control fugitive emissions from fossil fuel production and use. E.g., addressing fossil fuel extraction sources in Russia and the Middle East would remove more than 10 percent of baseline emissions in 2050. An almost as large reduction is expected below 20 €/t CO_2eq from implementing infrastructure for source separation and treatment of solid waste and proper wastewater treatment in China, India and the rest of South-East Asia. The technical abatement potential is considerably more limited for agricultural sources, due in particular to difficulties addressing CH_4 emissions from extensive livestock rearing in developing countries, where the keeping of large herds of robust but relatively unproductive animals often fills a vital function in farmers' risk management.

Overall, we find it technically feasible in year 2050 to remove 54 percent of CH_4 emissions below baseline, thereby leaving 5.7 Pg CO_2eq still released in 2050. This is cause for concern, considering that to stay at 1.5 degrees warming, IPCC estimates we must not exceed 10 Pg CO_2eq for all greenhouse gases in 2050. In addition to technical solutions, this calls for widespread implementation in the 2050 timeframe of institutional reforms e.g., to improve smallholder farmers' access to credit markets and public health services, and behavioural options, e.g., human diet changes that reduce milk and beef consumption.

Finally, we find the marginal abatement costs highly sensitive to the time and opportunity cost perspectives of investors and to the impacts of technological development. Policy makers will need to consider this when setting future reduction targets and carbon price levels to address CH_4 emission reductions. In general, a higher carbon price level than the one found optimal from a social planner's perspective will be needed to stimulate private investors to make market decisions that achieve the desired emission reductions.

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Author contributions: LHI developed the model code, performed emission and cost simulations, and prepared the manuscript with contributions from all co-authors. AGS developed waste and wastewater sector emission estimates and projections. ZK prepared and implemented FAO activity data projections for the agricultural sectors, PR prepared and implemented the IEA-WEO activity data projections for the energy sectors, and WS provided input to methodological discussions and model structure developments throughout the study.

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Supplementary Information to:

Technical potentials and costs for reducing global anthropogenic methane emissions in the 2050 timeframe –results from the GAINS model

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Content:

S1: Activity source sectors of the CH_4 module in the GAINS model

Table S1-1: GAINS model source sectors for anthropogenic CH_4 emissions.

Major source sector	Source sector	Activity unit	Further sub-sectors in GAINS
Agriculture	Beef cattle	M heads	Solid/Liquid manure management; Enteric fermentation/Manure management modelled separately only for animals on liquid manure management; Animals by farmsize (0-15 LSU, 15-50 LSU, 50-100 LSU, 100-500 LSU, > 500 LSU)
	Dairy cows	M heads	
	Sheep Goats etc	M heads	
	Pigs	M heads	
	Poultry	M heads	Laying hens/Other poultry
	Rice cultivation	M Ha	Continuously flooded/intermittently dried out/upland
	Agr waste burning	Mt crop residuals	no further sub-sectors
Energy	Coal mining	Mt coal mined	hard coal/brown coal; pre-mining/during mining/post-mining
	Abandoned coal mines	kt CH4	no further sub-sectors
	Domestic energy use firewood	PJ energy use	By woodstove type
	Domestic energy use other	PJ energy use	By boiler type; by fuel
	Industry energy use other	PJ energy use	By boiler type; by fuel
	Powerplant energy use other	PJ energy use	By boiler type; by fuel
	Domestic energy use gas	PJ energy use	combustion/fugitive emissions; by boiler type
	Industry energy use gas	PJ energy use	combustion/fugitive emissions; by boiler type
	Powerplant energy use gas	PJ energy use	combustion/fugitive emissions; by boiler type
	Gas transmission	PJ gas transported	no further sub-sectors
	Gas production	PJ gas produced	conventional natural gas/shale gas/coal bed methane/tight gas; fugitive emissions from intended venting and unintended equipment leakage estimated separately
	Oil production	PJ crude oil produced	fugitive emissions from intended venting and unintended equipment leakage estimated separately; heavy/conventional and on-shore/off-shore reflected in emission factor assumptions
	Oil refinery	PJ crude oil refined	no further sub-sectors
	Transport Road	PJ energy use	By fuel; by vehicle type (bus/truck/car/light-duty van); by EURO class
Industry	Industry Brick kilns	Mt brick	no further sub-sectors
Waste	Solid waste industry	Mt waste	By manufacturing industry: food, beverages, tobacco/pulp & paper/textile & footwear/wood & wood products/rubber & plastics/other
	Solid waste municipal	Mt waste	By waste category: food & garden/paper/textile/wood/rubber & plastics/other
Wastewater	Wastewater industry	kt COD	By manufacturing industry: food, fat, sugar & beverages/pulp & paper/organic chemical
	Wastewater domestic	M people	centralized collection/decentralized collection of wastewater

S2: GAINSv4 bottom-up CH_4 emission inventory 1990-2015 by sector and major World region

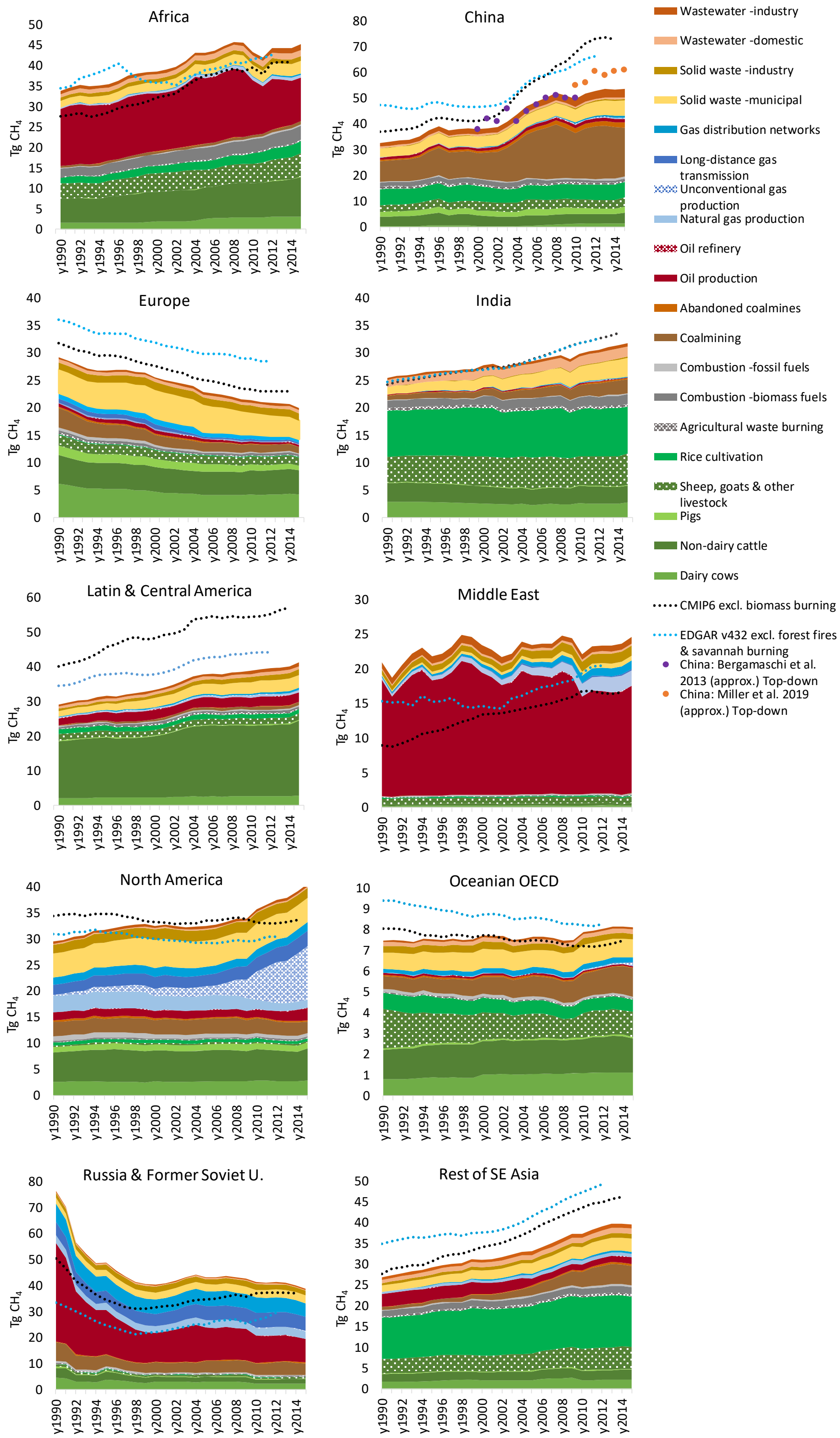

Figure S2-1: GAINSv4 bottom-up emission inventory for CH_4 emissions 1990-2015 by major World region.

S3: GAINSv4 baseline CH_4 emissions 1990-2050 by sector and major World region

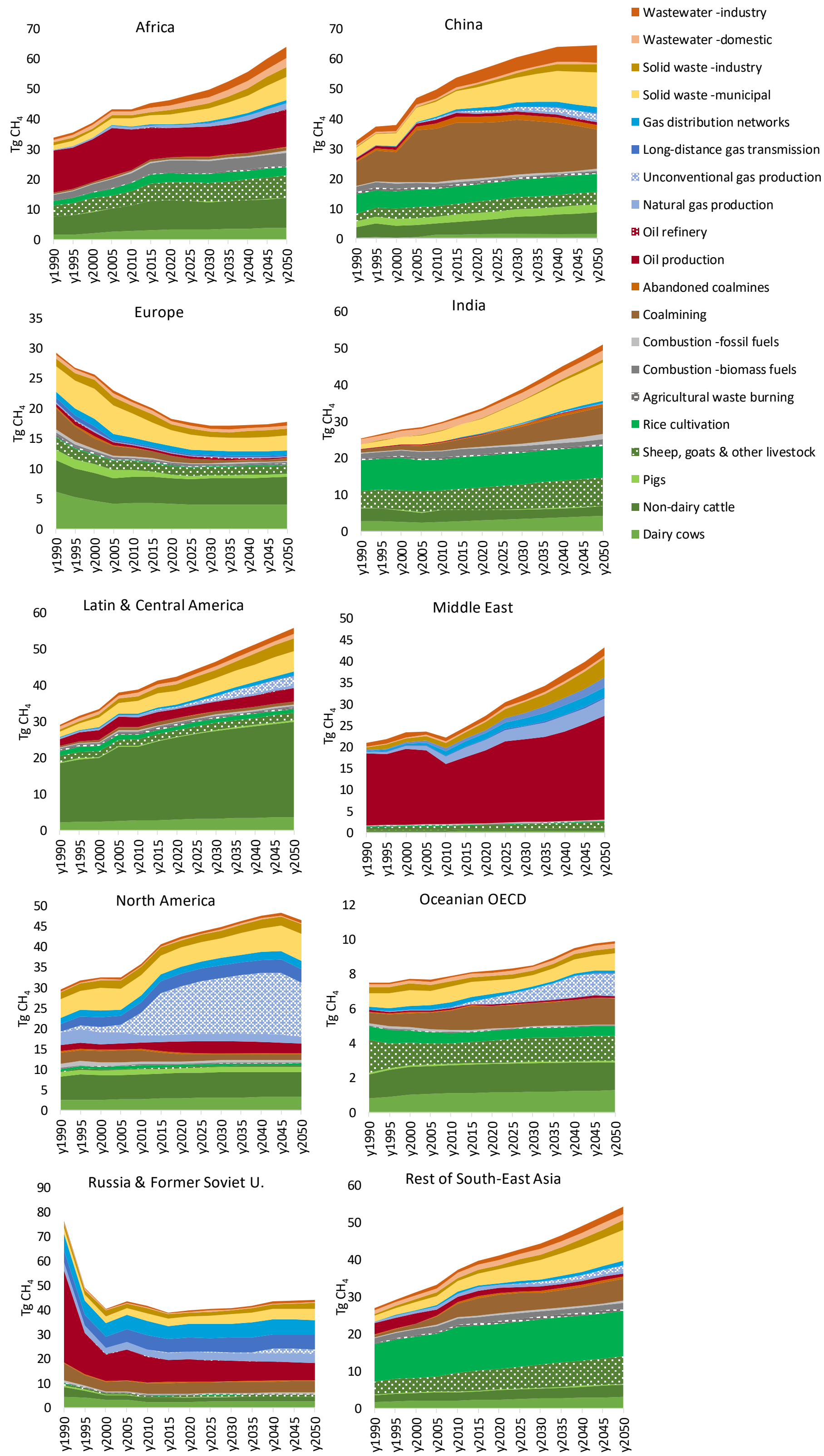

Figure S3-1: Baseline CH_4 emissions 1990-2050 by sector and World region as estimated in GAINSv4.

S4: Current legislation addressing CH_4 emissions implemented in GAINSv4

Table S4-1 provides a list of implemented national and regional legislation with direct or indirect impacts on CH_4 emissions, which have been considered in the GAINSv4 baseline scenario.

Table S4-1: Current legislation implemented in the GAINSv4 Baseline scenario.

Country	Sector	Policy or voluntary initiative	Date of publication/implementation
Algeria	Solid waste	Law relating to the management, control and disposal of waste. In GAINS assumed only partially enforced.	Law No. 01-19 of 12/12/2001
Argentina	Solid waste	Law relating to the management, control and disposal of waste. In GAINS assumed only partially enforced.	Law 25916 of 7/09/04
Australia	Solid waste	Region level legislation. Western Australia: Waste Avoidance and Resource Recovery Act 2007 (WARR Act); Canberra: ACT Waste Management Strategy: Towards a sustainable Canberra 2011-2025; Northern Territory: Waste Management Strategy 2015-2022; Queensland: Waste Avoidance and Resource Productivity Strategy 2014–2024	Regional implementation dates.
Colombia	Solid waste	Integrated waste management plans; Household waste collection, separation and landfill. In GAINS assumed only partially enforced.	Decree 1713/2002. Environment, Housing and development Ministry.
Costa Rica	Solid waste	Law on waste management: collection, separation and final disposal. In GAINS assumed partially enforced.	Law 8839 from 2010
Canada	Oil & gas systems	Requirements for oil and gas producers in the provinces of Alberta, British Columbia, Newfoundland to limit flaring and venting resulting in, e.g., a 40% reduction in venting and a 60% reduction in flaring of solution gas in Alberta. Recently implemented requirements in Saskatchewan and New Brunswick are expected to achieve similar reductions.	Alberta Energy Regulator (2013, 2014); BC Oil and Gas Commission (2013); Canadian Minister of Justice (2009); Saskatchewan Ministry for Energy and Resources (2011); New Brunswick Department of Energy and Mines (2013)
	Solid waste	Provincial regulations in British Columbia, Manitoba, Ontario, Quebec and Prince Edward Island require the collection and utilization and/or flaring of landfill gas (although requirements may depend upon facility size, age, etc.). Under the Provincial regulations in Alberta, facilities can reduce their emissions physically, use offsets or contribute to the Climate Change and Emissions Management Fund.	BC Ministry of Environment (2008); Manitoba Ministry of Conservation and Water Stewardship (2009); Ontario Ministry of Environment (2007); Quebec MDDELCC (2011); PEI Ministry of Environment, Labour and Justice (2009); Alberta Energy Regulator (1998); Ontario Ministry of Energy (2009)
		Province of Ontario has feed-in tariff in support of landfill gas electricity generation.	
	Livestock	Voluntary provincial greenhouse gas offset protocols in Alberta and Quebec address methane emissions from the anaerobic decomposition of agricultural materials (Alberta) and covered manure storage facilities (Quebec).	Alberta Environment (2007); Quebec MDDELCC (2009)
China	Coal mining	Various administrative provisions and programs to increase control and utilization of coal mine gas	Implemented 2005-2007, see Cheng, Wang & Zhang (2010); Miller et al. (2019)
	Solid waste	Law on the Prevention and Control of Environmental Pollution by Solid Waste. In GAINS assumed enforced in Hong-Kong, Shanghai and Beijing, with partial enforcment in other provinces.	Implemented 1995 with Amendment in 2004
Ecuador	Solid waste	Integrated waste management plans; Household waste collection, separation and landfill. In GAINS assumed only partially enforced.	Official registry No 316 -May 2015
Egypt	Solid waste	Law requring solid waste collection, treatment and disposal. In GAINS assumed only partially enforced.	Law 38/1967 on General Public Cleaning and Law 4/1994 for the Protection of the Environment.

Continued Table S4-1: Current legislation implemented in the GAINS Baseline scenario.

Country	Sector	Policy or voluntary initiative	Date of publication/implementation
European Union (EU-28)	EU-wide Climate policies	EU Climate and Energy package 2020: At least 20% cut in GHG emissions from 1990 level. Indirect effect on CH_4 through targets in the energy sector, e.g., 20% renewable energy in 2020 affect CH4 through incentives to extend anaerobic treatment of manure and food waste for recovery of biogas. The Effort-sharing decision provide binding national reduction targets for non-ETS sectors (housing, agriculture, waste, transport).	Adopted May 2009
		EU Climate and Energy framework 2030: At least 40% cut in GHG emissions from 1990 level. Indirect effect on CH4 through targets in the energy sector, e.g., 27% renewable energy, trigger incentives to extend anaerobic treatment of manure and food waste for recovery of biogas. Binding national reduction targets for non-ETS sectors (housing, agriculture, waste, transport) still to be adopted.	Adopted Nov 2018
	Oil & gas systems	EU Fuel Quality Directive: Reduce life-cycle greenhouse gas emissions of fossil fuels by 10% between 2010 and 2020 incl. reductions of flaring and venting at production sites.	EU Directive 2009/30/EC
		Gas flaring is only allowed with specific permission of the government and venting is only permitted in case of emergency.	GMI & EC (2013)
	Solid waste	EU Landfill Directive: Until 2016 reduce landfill disposal of biodegradable waste by 65 percent from the 1995 level and implement compulsory recovery of landfill gas from 2009.	EU Directive 1999/31/EC
		EU Waste Management Framework Directive: The waste hierarchy must be respected, i.e., recycling and composting preferred to incineration/energy recovery, which in turn is preferred to landfill disposal.	EU Directive 2008/98/EC
		Austria, Belgium, Denmark, Germany, Netherlands, Sweden: National bans on landfill of untreated biodegradable waste.	In effect 2005 or earlier.
		Slovenia: Decree on landfill of waste beyond the EU Landfill Directive. Includes a partial ban on landfill of biodegradable waste.	In effect Feb 2014
		Portugal: Target set to reduce landfill of biodegradable waste to 26% of waste landfilled in 1995.	Date of enforcement unclear, but policy in place in 2014.
	Wastewater	EU Urban Wastewater treatment Directive: "Appropriate treatment" of wastewater from urban households and food industry must be in place by 2005 and receiving waters must meet quality objectives.	EU Directive 1991/271/EEC
	Livestock	Denmark: National law on the promotion of renewable energy, which includes subsidy on biogas generated e.g., from manure.	Lov 1392, 2008
Iceland	All sources	No policies specifically addressing methane. Emissions likely small because of small population and cold climate.	Personal info (P. K. Jonsson, 2014)
Indonesia	Solid waste	Current state of waste management implemented in GAINS. Law assumed partially enforced in terms of waste collection and handling.	Waste Management Law of 2008 (No 18/2008)
Japan	Solid waste	High collection rates, appropiate separation systems and adequate waste treatment including recycling, composting and incineration of waste.	Law for Promotion of Utilisation of Recycled Resources (2002)
Kenya	Solid waste	Although Kenya has laws targeted to waste collection and management, implementation and enforcement is weak.	The Environmental Management And Coordination Act (EMCA), 1999
Malaysia	Solid waste	Current waste handling dominated by mostly unmanaged landfills with low collection and recycling rates	Solid Waste and Public Cleansing Management Corporation (SWPCMC) Act, 2007
Mozambique	Solid waste	Current waste treatment is poor with low collection rates	Environment Act (Law 20/97 of October1st)
New Zealand	Solid waste	Waste collection, separation and treatment systems are in place and enforced. Waste minimization assumed partially implemented in GAINS.	Waste Minimisation Act 2008
Norway	Oil & gas systems	Gas flaring is only allowed with specific permission of the government and venting is only permitted in case of emergency.	GMI & EC (2013)
	Solid waste	National ban on deposition of biodegradable waste in covered landfills from 2004.	FOR-2004-06-01-930

Continued Table S4-1: Current legislation implemented in the GAINS Baseline scenario.

Country	Sector	Policy or voluntary initiative	Date of publication/implementation
Peru	Solid waste	Current state of waste treatment systems reflected in GAINS Baseline. Landfills only partially managed, collection rates low in particular in small cities and rural areas.	General Law on Solid Waste Management (Ley General de Residuos Sólidos, 27314)
Phillipines	Solid waste	The GAINS Baseline reflects the current situation. Low collection rates, mainly unmanaged landfills.	Ecological Solid Waste Management Act, known as the Pepublic Act No 9003 (RA 9003)
Russia	Oil & gas systems	In the April 2007 state of the union address, president Putin announced an intent to make better utilization of associated gas a national priority.	Carbon Limits (2013)
		"Estimation of fines for release of polluting compounds from gas flares and venting of associated gas from oil production." (Translation from Russian by A. Kiselev, 2014)	Decree No.1148, Nov 8, 2012 of the Russian Fed. Governm.
		As of 2012, all flared associated gas must be metered or the methane fine increases by a factor of 120.	Evans and Roshchanka (2014)
	Other sources	"About greenhouse gases emission reduction." General policy addressing greenhouse gases, but unclear how methane is specifically addressed.	Decree No.75, Sep 30, 2013 of the Russian Fed. Governm.
Rwanda	Solid waste & wastewater	The GAINS Baseline reflects the current situation. Low collection rates, poor waste & wastewater handling.	National Policy and Strategy for Water Supply and Sanitation Services
Singapore	Solid waste	High collection rates and appropiate waste treatment including recycling, composting, incineration and sanitary landfills.	Environmental Public Health Act, Environmental Public Health (General Waste Collection & Waste Disposal Facilities) Regulations
South Africa	Solid waste	Current waste management shows partial implementation of the law in terms of collection rates, separation of waste and treatment.	National Environmental Management: Waste Act, 2008 (Act 59 of 2008)
Sri Lanka	Solid waste	The GAINS Baseline reflects the current situation. Low collection rates and generally poor management and treatment.	Solid Waste Act 2011
Tanzania	Solid waste	The GAINS Baseline reflects the current situation. Low collection rates and generally poor management and treatment.	Environmental Management Act of 2004
Tunisia	Solid waste	The GAINS Baseline reflects the current situation. Low collection rates and generally poor management and treatment.	Decree no 97-1102 of 2 Juin 1997
United States	Oil & gas systems	EPA's Natural Gas STAR Program: voluntary partnership that encourages oil and natural gas companies to adopt cost-effective technologies and practices that improve operational efficiency and reduce emissions of methane.	USEPA (2014a)
		New Source Performance Standards 2016 for methane from oil and gas systems sources, including Amendment from Sep 2018. Initially requiring oil and gas well owners to schedule monitoring and to repair leakages. The 2018 Amendment significantly relaxed requirements and provided possibilities for exceptions.	USEPA (2018)
	Coal mining	EPA's Coalbed Methane Outreach Program: voluntary program whose goal is to reduce methane emissions from coal mining activities.	USEPA (2014b)
	Solid waste	All landfills fullfill requirements for sanitary landfills. EPA's Landfill Methane Outreach Program: voluntary assistance program that helps to reduce methane emissions from landfills by encouraging the recovery and beneficial use of landfill gas as an energy resource.	USEPA (2014c); Resource Conservation and Recovery Act 1976, 1986
	Livestock	EPA's AgSTAR Program: voluntary outreach and educational program that promotes the recovery and use of methane from animal manure.	USEPA (2014d)
Vietnam	Solid waste	GAINS assumes partially implemented waste separation systems with proper handling and treatment in larger cities, Low collection rates and lack of proper treatment in rural areas.	Law on Environmental Protection 2005

S5: Assumptions on impacts of technological development

Table S5-1 presents GAINSv4 assumptions on impacts of technological development on future emission reduction potentials and costs for CH_4 abatement technologies. For details, see Höglund-Isaksson et al. (2018). Note that the “Technical removal efficiency” refers to the removal potential of emissions in a given country and sector relative a “no control situation”, which is defined as before any abatement technology has been adopted. If a technology has been adopted to some extent already in the baseline, then the remaining removal efficiency will be smaller than the technical removal efficiency. The same applies if there are physical or technical limitations to full applicability in a sector, e.g., animal feed changes are only assumed applicable to animals that are housed indoor. The technical removal efficiency then refers to the removal efficiency for the subset of animals housed indoor.

Table S5-1: Technological development effects 2020-2050 assumed in GAINSv4 for CH_4 mitigation options.

Sector	Methane mitigation options in GAINS	Technical removal efficiency (relative no control when technology is applicable)		Technological development effect on investment and O&M costs
		Current technology	Technology in 2050 (incl. technological development effect)	
Livestock	Anaerobic digestion of manure from cattle and pigs on farms with 100-500 LSU	60% (of manure emissions)	70% (of manure emissions)	-35%
	Anaerobic digestion of manure from cattle and pigs on farms with > 500 LSU	75% (of manure emissions)	82% (of manure emissions)	-35%
	Small-scale biogas digester for farm households in developing countries	50% (of manure emissions)	63% (of manure emissions)	-35%
	Breeding through selection for cows, cattle and sheep > 100 LSU (from 2030)	~ 10% (of enteric fermentation emissions)	~ 26% (of enteric fermentation emissions)	-28%
	Intensive systems: breeding in combination with feed additives > 100 LSU (from 2030)	20-30% (of enteric fermentation emissions)	34-43% (of enteric fermentation emissions)	-28%
	Extensive systems: breeding combined with inter-seeding of natural pastures > 100 LSU (from 2030)	30% (of enteric fermentation emissions)	43% (of enteric fermentation emissions)	-28%
Rice cultivation	Combined option: intermittent aeration of continuously flooded fields, alternative hybrids and sulphate amendments	33%	51%	-35%
Municipal solid waste	Food & garden waste: source separation and anaerobic digestion with biogas recovery and utilization	90%*	93%*	-35%
	Food & garden waste: source separation and treatment in household compost	80%*	85%*	-35%
	Food & garden waste: source separation and treatment in large-scale compost	89.5%*	92%*	-35%
	Paper waste: source separation and recycling	93%*	95%*	-35%
	Textile waste: source separation and reuse/recycling	100%*	100%*	-35%
	Wood: source separation and recycling for chip board production	95%*	96%*	-35%
	All waste categories: well managed incineration of mixed waste with energy recovery	>99%*	>99%*	-35%
Industrial solid waste	Food industry: Anaerobic digestion with biogas recovery and utilization	90%*	93%*	-35%
	Pulp & paper industry: incineration of black liqour for energy utilization	>99%*	>99%*	-35%
	Textile industry: incineration with energy recovery	>99%*	>99%*	-35%
	Wood industry: chipboard production	95%	96%	-35%
	All industries: well managed incineration with energy recovery	>99%*	>99%*	-35%
Domestic wastewater	Upgrade of primary treatment to secondary/tertiary anaerobic treatment with biogas recovery and utilization	93% (of primary treatment emissions)	95% (of primary treatment emissions)	-35%
Industrial wastewater	Upgrade of treatment to two-stage treatment, i.e., anaerobic with biogas recovery followed by aerobic treatment	99% (of primary treatment emissions)	99.3% (of primary treatment emissions)	-35%
Coal mining	Pre-mine degasification on both surface and underground coal mines	90%	93%	-35%
	Oxidation of ventilation air methane (VAM) on underground mines	50%	63%	-35%
	VAM oxidation combined with improved ventilation systems on underground mines	70%	78%	-35%
Oil & gas production	Extended recovery and utilization of vented associated gas	98%	99%	-35%
	Monitoring of temporary flare shutdowns	99%	99%	-35%
	Reducing unintended leakage through Leak Detection and Repair (LDAR) programs	67%	76%	-35%
Gas transmission	Reducing unintended leakage through Leak Detection and Repair (LDAR) programs	75%	82%	-35%
Gas distribution networks	Replacement of grey cast iron pipes and doubling of control frequency	97%	98%	-35%
	Reducing unintended leakage through Leak Detection and Repair (LDAR) programs	50%	63%	-35%
Combustion	Ban on open burning of agricultural waste	100%	100%	-35%

*Reduction relative a no control case defined as disposal to an unmanaged landfill with compacting

S6: Detailed source sector documentation

This section provides additional details on methodologies to estimate CH_4 emissions at the sector level in GAINSv4. The methodology described here builds on the documentation provided in the Supplement of Höglund-Isaksson (2012).

S6.1. Coal mining

The methodology for estimating global CH_4 emissions from coalmines in GAINSv4 has been described in detail in the Supplement of Höglund-Isaksson (2012). In short, emissions are estimated separately for brown coal and hard coal and using separate emission factors for pre-mining degasification, during mining and post-mining activities. In addition, country-specific information about the fractions of coal surface mined and mined underground has been collected and considered in emission estimations. Resulting implied emission factors and estimated emissions in 2010 and 2015 for all coalmining sources are presented in Table S6-1 by country. Emissions from Chinese coal mines make up over half of global CH_4 emissions from this source. Three recent studies (Peng et al., 2016; Miller et al., 2019; Sheng et al., 2019) quantify CH_4 emissions bottom-up from Chinese coalmines with Miller et al. and Sheng et al. also verifying bottom-up estimates with top-down atmospheric measurements and satellite observations. In GAINSv4, we align emissions from coal mining with the findings of these three studies as shown in Table S6-2.

Table S6-1: Implied emission factors for coal mining in GAINSv4 and in comparison to most recent reporting to the UNFCCC (2018).

		Implied emission factors (Gg CH_4/Mt coal)		Emissions in year 2010 (Tg CH_4)		Emissions in year 2015 (Tg CH_4)	
World region	Country	Brown coal	Hard coal	GAINS	UNFCCC (v2018)	GAINS	UNFCCC (v2018)
Africa	South Africa	n.a.	2.36	0.60	n.a.	0.61	n.a.
	Other Africa	0.87	8.38	0.04	n.a.	0.12	n.a.
China		n.a.	5.61	17.7	n.a.	19.1	n.a.
European Union	Bulgaria	0.83	8.56	0.03	0.04	0.02	0.04
	Czech Rep.	0.59	8.26	0.17	0.18	0.12	0.14
	France	n.a.	13.74	0.004	0.00	0.003	0.0004
	Germany	0.07	7.51	0.13	0.13	0.08	0.12
	Greece	1.13	n.a.	0.06	0.05	0.06	0.04
	Italy	n.a.	12.84	0.001	0.001	0.001	0.001
	Poland	0.09	5.94	0.50	0.62	0.50	0.66
	Romania	1.72	13.50	0.06	0.06	0.05	0.04
	Slovak Rep.	2.61	n.a.	0.01	0.02	0.01	0.01
	Spain	0.32	4.44	0.03	0.01	0.02	0.003
	United Kingdom	n.a.	7.66	0.14	0.08	0.08	0.04
	Other EU countries	0.87	8.38	0.01	0.01	0.008	0.006
Eastern Europe	Former Yugoslav republics	0.87	8.38	0.10	n.a.	0.10	n.a.
	Turkey	1.68	8.90	0.15	0.24	0.11	0.09
Western Europe	Norway	n.a.	1.56	0.003	0.002	0.002	0.002
Russia & Former Soviet Union	Russian Fed.	4.53	9.51	2.47	2.23	2.98	2.45
	Kazakhstan	4.01	6.67	0.72	0.97	0.70	0.89
	Ukraine	1.22	22.97	1.26	0.93	0.69	0.56
	Other Former Soviet republics	0.87	8.38	0.01	n.a.	0.02	n.a.
India		0.87	3.84	2.05	n.a.	2.46	n.a.
Latin & Central America		0.87	8.38	0.80	n.a.	0.92	n.a.
Middle East	Iran	1.32	n.a.	0.01	n.a.	0.01	n.a.
North America	Canada	0.54	0.61	0.04	0.05	0.04	0.04
	United States	0.76	2.98	2.75	3.29	2.26	2.45
Oceanian OECD	Australia	1.12	2.89	1.13	0.98	1.37	1.00
	New Zealand	0.81	2.88	0.01	0.02	0.01	0.01
Rest of South-East Asia		0.87	8.38	3.62	n.a.	4.67	n.a.
Global				34.6		37.1	

Table S6-2: GAINSv4 estimate of CH_4 emissions from coalmining in China in comparison to other recent studies.

	China coal mining emissions (Tg CH_4/year)			
Year	GAINS (this study)	Peng et al., 2016	Miller et al., 2019 (approx. adapted from Fig.5)	Sheng et al., 2019
1990	7.9	6.8 (6.0-7.5)		
1995	10.1			
2000	10.1	6.0 (5.3-6.7)		
2005	17.1			11.0
2010	17.7	17.7 (16.7-20.3)	16	15.2
2015	19.1		19	15.9

Emissions from both surface and underground mines can be reduced if CH_4 is recovered through pre-mine drainage up to ten years before the mining starts (USEPA, 2008). Currently in the US, at least 90 percent of degasification emissions from underground coalmines are recovered and utilized (USEPA, 2010). In GAINSv4, this is assumed technically possible in other countries as well. There is, however, only one project known to be recovering and utilizing CH_4 from pre-mine drainage at a surface mine and details about the removal efficiency of this option are uncertain (Sino-US New Energy Sci-Tech Forum, 2009). In GAINSv4, it is considered technically possible to recover 90 percent of the drainage gas also from surface mines. Costs for degasification are taken from Thakur (2006) and include costs for in-mine drilling, underground pipeline costs, and hydraulic fractioning of vertical wells and other gob wells.

Ventilation air methane (VAM) from underground coal mines can be recovered and oxidized through installation of VAM oxidizers (Mattus and Källstrand, 2010). Although the application on coalmines is still in an early phase, the technology is well known from control of odor and VOC emissions worldwide. The technology oxidizes at least 95 percent of VAM when applied to a ventilation shaft. It uses the energy released during the oxidation to keep the process running, which keeps fuel costs limited to the initial start-up phase. For a thermal oxidation process to run without interruptions the CH_4 concentration in the ventilation air needs to be at least 0.3 percent. For some recent installations in China a catalytic oxidation process is in use, which operate with CH_4 concentration rates in the ventilation air as low as 0.2% (Somers and Burklin, 2012). Securing this concentration level without increasing explosion risks (i.e. CH_4 concentrations in the air should never be in the explosive range between 5 and 15 percent), may in some mines require investments in more efficient ventilation systems. A general assumption is made in GAINSv4 that it is technically possible to keep CH_4 concentration levels at a steady rate of at least 0.3 percent, and therefore to install self-sustained VAM oxidizers (Mattus and Källstrand, 2010), on 50 percent of the ventilation air emitted from underground coal mines in all countries. Combining a catalytic oxidation VAM technology with an improved ventilation system is assumed to extend the feasible application of VAM oxidizers to 70 percent of VAM emitted from underground mines in all countries. An improved ventilation system is taken to double the ventilation capacity of the mine compared with a conventional system, thereby doubling the amount of electricity used for ventilation. Costs for VAM oxidation technology and installation are taken from USEPA (2003, p.30) and GMI (2008) and refer to installations in the US and China. Costs for increased electricity use for ventilation in mines are based on information from Unruh (2002) and Papar et al. (1999). No mitigation potential is assumed for post-mining emissions.

S6.2. Abandoned coal mines

Countries reporting CH_4 emissions to the UNFCCC in the Annex-1 category are expected to enter emissions from abandoned coal mines in the Common Reporting Formats (CRFs). The reported emissions make up the activity data for this source sector in GAINSv4. For non-Annex-1 countries, a default assumption is made that emissions from abandoned coal mines corresponds to 10% of active hard coal mining emissions. This assumption is based on US estimates of CH_4 emissions from abandoned coal mines corresponding to 13% of active coal mining CH_4 emissions in 2015 (USEPA, 2017a). Applying this default assumption to China means between 1200 and 1900 kt CH_4 released per year between 2005 and 2015 from this source. In a study funded by USEPA, Collings et al., (2012) analyze CH_4 emissions from 44 abandoned coal mines in the Shanxi province and find that these alone emit an estimated 0.5 bcm or about 350 kt CH_4 per year. Considering that the same report mentions there are likely thousands of abandoned coal mines in China, our estimate for all of China, is likely conservative.

The release of CH_4 emissions from abandoned coal mines typically depends on the status of the abandoned mine, i.e., whether it is left open for venting in order to prevent build-up of explosive CH_4 pockets underground, flooded to prevent CH_4 emissions from escaping, or sealed through cement plugging (USEPA, 2004). For the modelling in GAINSv4, it is assumed that without regulation the no control case is venting. The control option considered is flooding, which is assumed to prevent 90% of emissions compared to the venting case. Sealing is not considered a CH_4 control option in GAINSv4, because to effectively prevent gas leakage, at least 95% of shafts must be sealed (USEPA, 2004), which likely makes it relatively expensive. In contrast, the cost of flooding abandoned coal mines is likely low or even profitable, as abandoned mines can potentially fill an important role in a future transformation to renewable energy. Abandoned coal mines can be used as pumped storage hydroelectric plants (Pujades et al., 2016; Jessop et al., 1995) or flooded and converted to giant floating solar farms as in Huainan, China (China Daily, 2017).

S6.3. Oil and gas production

The methodology for deriving country-specific emission factors for CH_4 from oil and gas systems is described in Höglund-Isaksson (2017). In summary, separate emission factors are derived for emissions from the handling of associated gas, for fugitive emissions from unintended leakages of the equipment, and from downstream leakages from transmission pipelines and consumer distribution networks. Unintended leakages from upstream sources are estimated using IPCC (2006) default emission factors, while emissions from downstream sources use a combination of emission factors from IPCC (2006) and national reporting to the UNFCCC (2016) when available. Emission factors linked to the management of associated gas are derived in a consistent manner across countries using country- and year- specific data on the total generation of associated gas 1990-2012 and the managerial practices for handling of the associated gas. These include the fraction of associated gas recovered, utilized and reinjected, and the volumes of gas not recovered and therefore either flared or vented.

For this study, a few updates were made to take account of additional information provided for Russia, the USA and Canada. For Russia, assumptions on the average composition of the associated gas generated from oil production have been revised based on information provided in Huang et al. (2015). Huang et al. provide information for three different separation stages. Although not completely clear from the source reference, we have interpreted the different stages as stage 1 representing the associated gas flared or vented directly at the wellhead with stages 2 and 3 representing subsequent processing stages. We further assume that the associated gas relevant for our estimations here is to 90% from stage 1 and to 10% from stage 2. The corresponding weighted average composition in vol% is 60.1% CH_4, 8.6% ethane, 17.9% propane, 12.0% other heavier hydrocarbons, and the rest being nitrogen gas and carbon dioxide. This is in contrast to the assumption in Höglund-Isaksson (2017), where the vol% composition of Russian associated gas was taken to be 81% CH_4, 5.5% ethane, 6.6% propane and 5.4% heavier hydrocarbons. Another update concern the recovery rate for Russian associated petroleum gas (APG), which with the recent data from NOAA (Elvidge et al., 2016) suggest that the volume of gas flared from Russian sources is 24.6

bcm in 2016, down from 35.2 bcm in 2010. Using this information to extend Table 5 of the Supplement to Höglund-Isaksson (2017), the resulting recovery rate for Russian APG becomes 68% and is in GAINSv4 applied to all Russian oil production from 2015 onwards.

For the US and Canada, we need to distinguish emission factors for conventional gas production as well as for unconventional shale gas extraction, which has increased rapidly since 2006 due to the development of hydraulic fracturing technology, as illustrated in Figure S6-1. For the US, total gas production increased by 47% between 2006 and 2017.

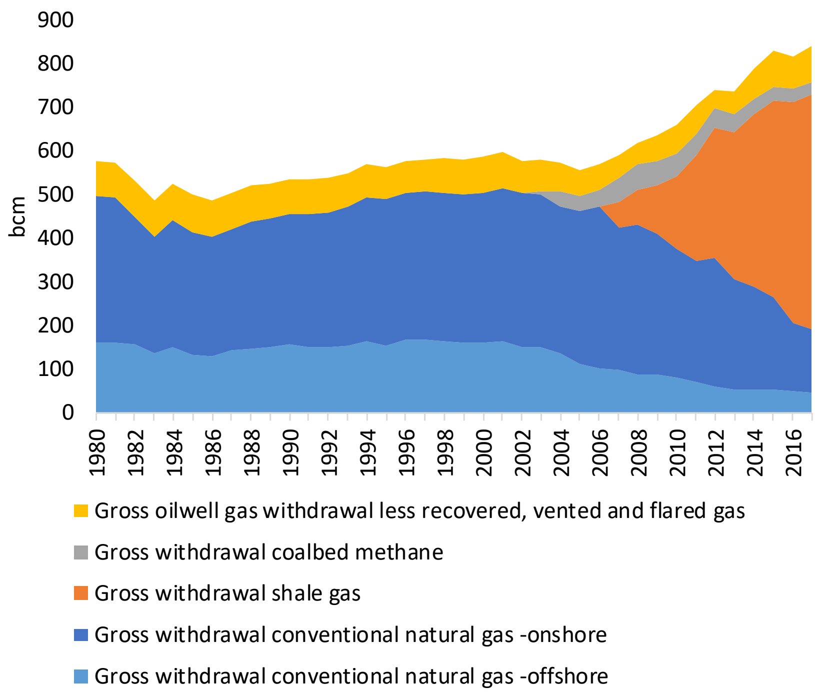

Figure S6-1: US natural gas production by type of gas 1980-2017. Adapted from data retrieved from EIA (July 11, 2019).

There is considerable uncertainty in the literature regarding the average emission factor for fugitive emissions from both conventional and unconventional gas extraction. A general conclusion appears to be that an important reason for the high uncertainty is the highly skewed distribution of emissions with rare super-emitting events contributing to a majority of emissions (Brandt et al. 2013; Zavala-Ariza et al. 2015; Alvarez et al. 2018). Inverse model results show contradicting results concerning whether North American shale gas extraction has contributed to an increase in CH_4 emissions or not. E.g., Turner et al. (2016), Hausmann et al. (2016) and Franco et al. (2016) find strong increases in recent US CH_4 emissions suggesting that unconventional gas extraction could be a likely culprit as much of the increase is measured over regions with such activities. Turner et al. estimate a more than 30% increase in US CH_4 emissions between 2002-2014, with maximum emissions in the South-Central US where unconventional hydrocarbon production is high. However, also livestock production is high in these regions, which adds to the uncertainty in source attribution. Supporting the attribution of recent emission increases to unconventional gas production is a measured simultaneous increase in the atmospheric concentration of ethane (Franco et al., 2016; Vinciguerra et al., 2015), which is consistent with the particularly high vol% of ethane found in US shale gas. In contrast, Bruhwiler et al. (2017) and Lan et al. (2019) find smaller increases in oil and gas emissions than Turner et al., Hausmann et al., and Franco et al., and no firm evidence of a large increase in total US CH_4 emissions 2006-2015. The controversy in the literature also extends to whether conventional and unconventional gas release similar emissions per unit of gas produced or whether considerable differences exist. Few studies (Kirchgessner et al., 1997) are available that measure the average leakage rate from US gas production before 2005 when the boom in shale gas production took off. Comparisons of measured leakage rates before and after the shale gas boom are further complicated by the technological advances in both extraction and emission control technology, as well as the introduction of emission regulations such as 'green completions' (USEPA, 2011). The GAINSv4 upstream emission estimates for US oil and gas sources in 2015 are presented in Tables S6-3 and S6-4. The US upstream emission factors for oil and gas production have been aligned with the average nation-wide estimates of Alvarez et al. (2018, Table 1). Alvarez et al. do not specify emission factors by type of gas produced. This split is in GAINSv4 based on activity data from other references (IEA-WEO, 2018 and EIA, 2019). The leakage rates assumed in GAINSv4 for the US are 0.19% for conventional offshore gas production (Skone et al., 2011), 1% for conventional onshore gas production (Kirchgessner, 1997; Skone et al., 2011; Allen et al., 2013; Cathles, 2012), and 1.65% and 0.58% for tight gas and coalbed methane, respectively (Skone et al., 2011). The leakage rate for shale gas extraction is assumed to 2.66% on average. This assumption was derived by matching the average leakage rate from Alvarez et al. of 1.95% for all upstream oil and gas production in the US in year 2015. An average leakage rate for shale gas of 2.66% is within the relatively large range reported in the literature for shale gas (e.g., Karion et al., 2013; Caulton et al., 2014; Schneising et al., 2014; Peischl et al., 2015; Howarth, 2019). The same average upstream leakage rates by types of gas produced have been assumed for Canadian gas production.

Table S6-3: US emissions (Tg CH_4) from oil and gas systems in year 2015 as estimated by Alvarez et al. (2018), USEPA (2017b) and GAINSv4.

<table>
<tr><th rowspan="2">Emission source</th><th colspan="2">Alvarez et al., 2018 Table 1</th><th rowspan="2">USEPA (2017b)</th><th rowspan="2">GAINSv4</th></tr>
<tr><th>Bottom-up estimate</th><th>Range</th></tr>
<tr><td>Upstream -Production</td><td>7.6</td><td>6-9.5</td><td>3.5</td><td rowspan="2">11.85</td></tr>
<tr><td>Upstream -Gathering</td><td>2.6</td><td>2.42-3.19</td><td>2.3</td></tr>
<tr><td>Downstream -Processing</td><td>0.72</td><td>0.649-0.92</td><td>0.44</td><td rowspan="2">2.58</td></tr>
<tr><td>Downstream -Transmission & storage</td><td>1.8</td><td>1.58-2.15</td><td>1.4</td></tr>
<tr><td>Downstream -Local distribution</td><td>0.44</td><td>0.22-0.91</td><td>0.44</td><td>1.55</td></tr>
<tr><td>Oil refinery & transportation</td><td>0.034</td><td>0.026-0.084</td><td>0.034</td><td>0.014</td></tr>
<tr><td>Total US Oil & Gas supply</td><td>13.2</td><td>10.896-16.794</td><td>8.1 (6.7-10.2)</td><td>16.0</td></tr>
</table>

Table S6-4: GAINSv4 estimate for US upstream oil and gas emissions in year 2015.

<table>
<tr><th>Hydrocarbon produced</th><th>Tg CH_4</th><th>Leakage as % of gas produced</th><th>Principal references for current leakage rates</th><th>MFR leakage rates in 2015</th><th>References for MFR leakage rates</th></tr>
<tr><td>Crude oil</td><td>1.45</td><td>n.a.</td><td>Höglund-Isaksson (2017)</td><td>n.a.</td><td>n.a.</td></tr>
<tr><td>Conventional gas -offshore</td><td>0.05</td><td>0.19%</td><td rowspan="5">Skone et al. (2011) for all gases except shale. Shale leakage rate derived to match Alvarez et al. (2018) for upstream oil & gas CH_4.</td><td>0.18%</td><td rowspan="5">Skone et al. (2011) 'new technology'; USEPA (2016); Saunier et al. (2017)</td></tr>
<tr><td>Conventional gas -onshore</td><td>1.12</td><td>1.00%</td><td>0.50%</td></tr>
<tr><td>Shale gas</td><td>7.90</td><td>2.66%</td><td>1.33%</td></tr>
<tr><td>Coalbed methane</td><td>0.14</td><td>0.58%</td><td>0.29%</td></tr>
<tr><td>Tight gas</td><td>1.19</td><td>1.65%</td><td>0.83%</td></tr>
<tr><td>Sum upstream</td><td>11.85</td><td>1.95%</td><td>Alvarez et al., 2018</td><td>0.98%</td><td></td></tr>
</table>

There are several cost-effective and low cost options available to reduce unintended leakage during extraction and processing of oil and natural gas (USEPA, 2016; ICF International, 2016). Addressing leakages first requires detection. With recent development of Leak Detection and Repair (LDAR) programs, in particular the use of infrared cameras, has lowered the cost of leak detection significantly (ICF International, 2016; USEPA, 2016; McCabe and Fleischmann, 2014). In a survey of LDAR programs in Europe installed to reduce unintended leakages from gas production, transportation and storage facilities, Saunier et al., (2017) find that when used regularly and systematically, LDAR effectively detects leakages. Out of detected leakages, 61 percent are successfully repaired leading to emission reductions of at least 90 percent, while 31 percent are less successfully repaired, reducing emissions by less than 50 percent and sometimes even increasing emissions. In an industry survey of US oil and gas facilities, ICF International (2016) finds that if all facilities are subject to annual LDAR emission surveys, an overall emission reduction of 40 percent is feasible. Drawing on these two studies, we assume in GAINSv4 that it is technically feasible to reduce emissions from unintended leakages by on average 50% when LDAR technology is implemented across all facilities. The cost of LDAR programs is likely to be highly site-specific and to vary with the gas price as reduced gas leakages mean higher profits from gas sales. After detection of leakages, there is a long list of possible repairs that are available at a wide range of costs (see e.g., Table 3-1 in ICF International, 2016). As we do not have access to industry data on the incidence of different types of leakages in global oil and gas systems, it is not possible to make an assessment of the expected number and types of repairs that will be needed and the associated costs. Such assessments exist for US gas and oil systems, based on detailed data reported by industry to the USEPA and complemented by industry surveys (USEPA, 2014e; ICF International, 2016). To estimate costs for gas leakage repairs in GAINSv4, we have sought to align the assumptions on costs with the ranges for the US marginal abatement costs estimated for different industry segments (i.e., production, processing, transmission and distribution).

Maximum technically feasible reduction of CH_4 emissions from the handling of associated gas generated during oil (and to a limited extent gas) production assumes it possible in all countries to recover and utilize at least 98 percent of the associated gas generated. This high level of associated gas recovery is already exceeded in Norway (Husdal et al., 2016a,b; EIA, 2015) and therefore assumed possible to achieve in other countries as well. Costs are taken from OME (2001) and refer to the costs of recovering and processing the gas and transporting it to the nearest EU border either through pipeline or ship, for details see the Supplement of Höglund-Isaksson (2012). In addition to extending associated gas recovery rates to 98 percent, it is assumed technically feasible to further reduce gas venting by making sure as much as possible of the two percent of associated gas not recovered is flared off. Through LDAR programs (USEPA, 2016; McCabe and Fleischmann, 2014), infrared cameras can be installed to continuously monitor flares of associated gas, thereby allowing for the identification and remedy of 'super-emitters', reduce routine venting as well as reduce the number and duration of temporary flare shut-downs caused by unfavorable weather and wind conditions (Husdal et al., 2016b, p.31). To our knowledge, LDAR programs have until now been introduced in Europe to control unintended fugitive leakages from gas processing plants and transmission and distribution networks (Saunier et al., 2017), however, not to control venting of associated gas. The applicability and cost of the technology for this purpose is therefore highly uncertain. As a conservative assumption we assume it possible to reduce venting of unrecovered associated gas by 30 percent if LDAR is implemented across all oil and gas production facilities. The marginal cost is very high (exceeding 500 €/t CO_2eq) as LDAR is assumed applied on top of a 98 percent recovery rate of associated gas and therefore only addressing emissions from the two percent associated gas not being recovered.

S6.4. Livestock

The general methodology used in GAINSv4 to estimate CH_4 emissions from livestock is described in the Supplement of Höglund-Isaksson (2012). Recent revisions concern updates of activity data and reported emission factors to latest statistics (FAOSTAT, 2018; UNFCCC, 2016; 2018) and a review of available technical abatement options for CH_4 described in detail in Höglund-Isaksson et al. (2018). Emissions are estimated by animal types, i.e., dairy cows, non-dairy cattle, pigs, poultry, sheep and goats, buffaloes, and horses, by whether emissions stem from enteric fermentation or manure management, and for dairy cows, non-dairy cattle and pigs, by whether animals are subject to liquid or solid manure management. A recently introduced improvement in the CH_4 module of the GAINS model is a split of the animal categories dairy cows, non-dairy cattle, pigs, sheep and goats by five farm size classes, i.e., less than 15 livestock units (LSU), 15 to 50 LSU, 50 to 100 LSU, 100 to 500 LSU, and above 500 LSU. Information on historical farm-size distributions are taken from EUROSTAT (2015), Ashton et al. (2016), Australian Government (2018), USDA (2011a; 2011b; 2013; 2015; 2016), Arelovich et al. (2011), Beef2Live (2018), Montaldo et al. (2012), Hengyun et al. (2011). Projections of the future development in farm-size classes have been produced for Europe by applying a multi-nominal logistic function weighing in the development observed in historical years from 1990 onwards. To reflect the recent fast-growing development of large dairy and cattle farms in China (Bai et al., 2017), it is assumed in GAINSv4 that the entire future stock increase as projected by FAO (Alexandratos and Bruisma, 2012) is allocated to farms with more than 100 LSU (Bai et al., 2017). For other World regions, farm-size class shares are kept constant in future years due to a lack of historical time-series on which to base a future development in farm size classes. The future development in farm-size classes has implications for future fractions of animals on liquid and solid manure management and on the future applicability of control technology options.

In GAINSv4, country- and animal- specific emission factors have been aligned with the implied emission factors reported to UNFCCC-CRF (2016; 2018) for the year 2010. For dairy cows, both enteric fermentation and manure management emissions per animal are affected by the milk productivity of the cow. This effect is accentuated for highly productive milk cows. To capture this, the no control emission factor for dairy cows is specified as the sum of a fixed emission factor per animal for cows producing up to 3000 kg per head per year and an additional term describing the emission factor per milk yield for milk production exceeding the productivity level of 3000 kg per animal per year. For further details see the Supplement of Höglund-Isaksson (2012).

Technical options to reduce CH_4 emissions from livestock exist for emissions from enteric fermentation and from the handling of manure. The options identified in GAINSv4 are breeding through selection with the dual target of increasing animal productivity while maintaining animal health and fertility, various options to change animal feed, and anaerobic digestion of manure for the production of biogas. A detailed description of these options with references and including expected removal efficiency and costs, is provided in Höglund-Isaksson et al. (2018). Due to limitations posed by economies of scale, the options listed above are considered feasible for large farms (above 100 LSU) with liquid manure management systems and with application limited to the time animals spend indoor. Such intensive systems are typically prevalent in Europe, North America and for a fast growing segment of large industrial farms in parts of Asia, notably China (Bai et al., 2017). In Latin America, parts of the USA, Australia and New Zealand, large-scale extensive dairy and cattle farming dominate, with animals typically grazing outdoor or staying outdoor in feedlots. In GAINSv4, there are no CH_4 mitigation options considered to control manure management emissions from such systems, however, there is assumed to be a potential to reduce enteric fermentation emissions by 10% through breeding and by maximum 30% if breeding is combined with inter-seeding of natural pastures with grass legumes, adding fodder crops and grass legume mixtures. The objective of the latter options is to improve animal productivity by increasing the quantity and quality of the fodder (FAO, 2017). Addressing CH_4 emissions from sheep and goat populations through breeding and changes in animal fodder is only considered feasible for animal on large farms (>100 LSU) in OECD countries. In all other parts of the world, sheep and goat rearing is assumed operated in extensive systems with animals grazing outdoor, genetically well adapted to local conditions, and without feasible technical potential to control emissions.

In GAINSv4, we assume no technical abatement potential for CH_4 from substitution of indigenous low-yielding breeds with highly productive imported breeds for the large number of cows and cattle kept on smallholder farms in Africa and South-East Asia. The reason is that milk and meat production is one out of a number of reasons for keeping livestock, where keeping herds as a mean for storing assets and manage risks over time may exceed productivity in importance (Udo et al., 2011). As smallholder farmers often lack access to formal credit markets and governmental support when faced with incidents of failed crops or illness, keeping large herds of livestock becomes one of few options for managing the risk of life-threatening unforeseen events over time. Substituting robust and to the climate genetically well adapted indigenous breeds with less robust but more productive imported breeds, is under such circumstances unlikely to be attractive to smallholder farmers. Addressing CH_4 emissions from smallholder livestock farmers is likely to require more fundamental economic and institutional reforms aimed at mitigating the risks currently facing this group of farmers.

Figure S6-2 illustrates the limited technical abatement potential for CH_4 emissions from livestock for different animal categories. As shown, technical abatement is almost only limited to large farms with more than 100 LSU. This means that the technical options are only applicable to about one third of global CH_4 emissions from livestock. Another third is estimated from smallholder cattle farms and extensive sheep and goat farms, primarily found in Africa and South-East Asia. No technical options have been found feasible to address these emissions, as explained above. The residual third of global livestock CH_4 is attributed to medium sized farms of 15-100 LSU. With the exception of limited potential from breeding and feeding options applicable to cattle farms with liquid manure management in the 50-100 LSU farm size class, we do not consider the available technical options economically feasible for farms below 100 LSU. Hence, deep future reductions in livestock CH_4 emissions will require additional policy incentives to limit the consumption of meat and milk, e.g., through economic instruments like taxes or by changing consumer preferences by promoting reduced meat and milk consumption for health reasons.

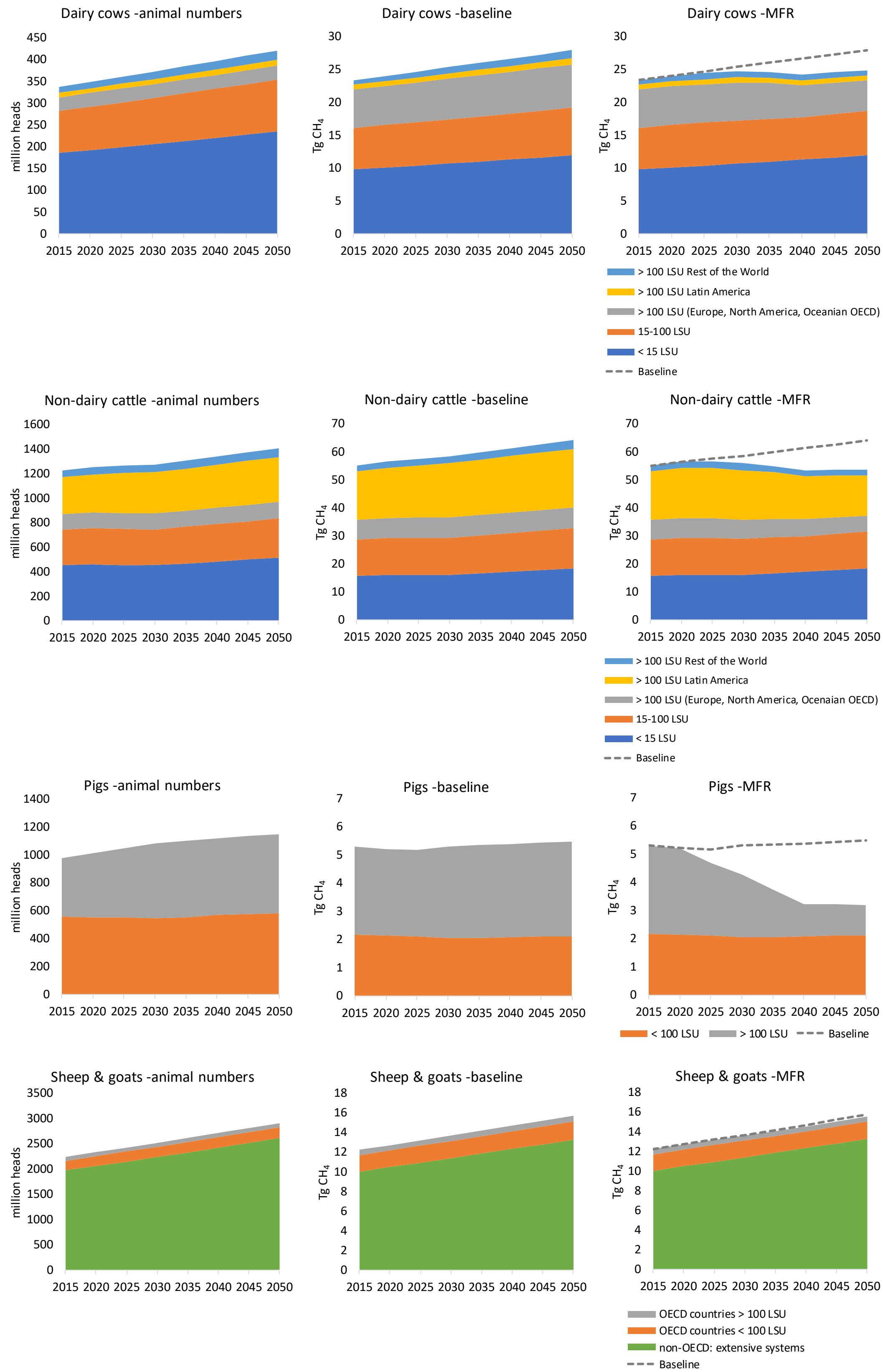

Figure S6-2: Global livestock animal numbers, baseline CH_4 emissions and emissions after Maximum technically Feasible Reduction (MFR), as estimated in GAINSv4.

S6.5. Rice cultivation

CH_4 emissions from rice cultivation result from anaerobic decomposition of organic material in flooded rice fields. Emissions depend on many factors e.g., on the season (wet or dry and season length), soil characteristics, soil texture, use of organic matter and fertilizer, climatic conditions such as temperature and humidity, and agricultural practices (IPCC, 2006, Vol.4, p. 5.45). The emission calculation methodology used in GAINSv4 follows the IPCC guidelines (2006, p. 5.49) and adopts IPCC default emission factors for given water management regimes. The IPCC method is based on the annual harvested area with scaling factors for different water regimes. In GAINSv4, these translate into three cultivation activities:

- *Continuously flooded cultivation area:* fields have standing water throughout the growing season and only drying out for harvest.
- *Intermittently flooded cultivation area:* fields have at least one aeration period of more than three days during the growing season. Compared with continuously flooded rice fields, IPCC suggests that intermittently flooded rice fields emit 27 to 78 percent of continuously flooded fields, where the range depends on if the fields are rainfed or irrigated. GAINSv4 uses the assumption of 50 percent emissions per hectare from intermittently flooded compared with continuously flooded fields.
- *Upland rice cultivation area:* fields are never flooded for a significant period of time and are not assumed to emit CH_4.

Activity data for rice cultivation is measured in million hectares of land cultivated for rice production (FAOSTAT, 2015) and cross-checked with information provided by countries in national reporting to the UNFCCC (2015; 2018). From the same source, we take data on country-specific application of different water regimes, complemented with information from IRRI (2007). For each cultivation activity, country- and technology- specific CH_4 emission factors are identified. CH_4 emissions from rice cultivation in country *i* in year *t* are calculated as follows:

$$E_{it} = \sum_{sm} A_{it} * ef_{i;\,flood}^{IPCC} * h_i * \beta_s * V_{is} * \left(1 - remeff_{sm}\right) * Appl_{itsm} ,$$

where A_{it} is the rice cultivation area in country *i* in year *t*,

$ef_{i;\,flood}^{IPCC}$ is the IPCC default emission factor for CH_4 emissions from flooded rice fields (1.3 kg CH_4 ha^{-1} day^{-1}),

h_i is the duration of the growing season expressed in days per year (=185 days per year),

β_s is an emission scaling factor for water regime *s* (=1 for continuously flooded, =0.5 for intermittently flooded, and =0 for upland rice).

V_{is} is the fraction of rice cultivated land under water regime *s*,

$remeff_{sm}$ is the removal efficiency of technology m when applied to water regime s, and

$Appl_{itsm}$ is the application rate of technology *m* when applied to water regime *s*.

CH_4 mitigation options implemented in GAINSv4 to control emissions from rice cultivation include employment of improved water management regimes, use of alternative rice hybrids increasing yields while suppressing methane generation e.g., through shorter stems, and use of soil amendments e.g., biochar or sulphate-containing amendments.

There are several ways to reduce CH_4 emissions through improved water management; single mid-season drawdown, alternative wetting and drying, aerobic rice production and dry direct seeding (WRI, 2014). A common feature of all water management options is that they reduce CH_4 emissions through decreasing the time that fields are flooded. Differences in local conditions e.g., climatic conditions, traditional farming customs and access to herbicides, water regulation mechanisms or fertilizers, will affect the impact of different water management regimes on yield, labour requirements and methane emissions (WRI, 2014). The choice of preferred water management regime is closely linked to these local conditions. Due to lack of information, we are not able to make a full-fledged assessment of the effectiveness of individual water management regimes in different regions of the world, but will have to resort to making broad assumptions about the effectiveness of water management regimes in general and their associated costs. According to a literature survey by WRI (2014), implementing improved water management regimes on continuously flooded fields have shown to achieve CH_4 emission reductions between 30-90%, with the higher relative reductions found for well-managed fields in the US. As a general assumption in GAINSv4 across all flooded rice fields, an average abatement potential of 20% is assumed achievable in the next ten years, extending to 40% on an annual basis in 2050. If improved water management is combined with other options e.g., low-CH_4 hybrids or different soil amendments (see below for details), the average global abatement potential assumed in GAINSv4 for continuously flooded fields extends to 50%. This estimate takes into account that some areas may be difficult to subject to improved water management due to heavy rainfall during the wet season (e.g., in the Phillippines) or due to unreliable water supply systems or fields that are not well levelled (WRI, 2014). These assumptions are somewhat conservative in comparison to Beach et al. (2015) who estimate an overall abatement potential for global rice cultivation in 2030 at 26.5% below baseline and Harmsen et al. (2019) who estimate 61% below baseline in 2050 for the same source.

A cost estimate of improved water management through drying out of continuously flooded rice fields will have to consider associated operation costs, including cost-savings from reduced water use and higher labour costs due to increased weed growth. In particular in poorer regions where farmers lack access to herbicides, longer periods of dry fields increase weed growth (WRI, 2014; Barrett et al. 2004; Ferrero and Nguyen 2004). According to estimates by Barrett et al. (2004), weed growth increases labour costs by an estimated 20 percent, which is equivalent to about 60 additional work hours annually per hectare in developing countries (Heytens, 1991) and 12 additional work hours annually per hectare in developed countries, where herbicides are used for controlling weed (Shibayama, 2001). Dry direct seeding of rice seedlings have shown to be very effective (45-90% reductions in emissions) for reducing CH4 emissions in the US compared with transplanting seedlings into flooded fields (WRI, 2014; Linquist et al., 2015). The abatement effect is attributed to the one month shorter period of flooding as seedlings grow in dried out fields. The option also contributed to reduced labour input and costs, however, this result appears to be conditional on unrestricted access to herbicides and well managed water tables and may therefore be difficult to replicate in many developing countries. According to IRRI (2007), intermittent aeration of continuously flooded rice fields may reduce water use by 16 to 24 percent. Assuming that continuously flooded rice fields need 1000 mm water input per year (Bouman, 2001) and the global average cost of irrigated water is 0.02 US$ per m^3 (FAO, 2004), then saving 22 percent of water corresponds to a cost-saving of about 30 Euro per ha. In Europe and North America, the cost of irrigated water is higher than the global average, converting into a higher cost-saving of about 70 Euro per ha.

Certain rice hybrids may affect CH_4 emissions. By careful selection of low-CH_4 producing hybrids, emissions can be ten percent lower (ADB 1998). ADB (1998) estimates that Chinese rice yields may increase by as much as 10 to 20 percent from switching to low-CH_4 hybrids. In other parts of the world, where high yield rice hybrids are already in extensive use, potentials for additional yield increases are likely lower. In GAINSv4, the assumption is that the potential reduction in CH_4 emissions from switching to alternative rice hybrids is 10 percent with a 3 percent increase in crop yield, when applied as the sole option. When applied in combination with other options, like improved water management of continuously flooded fields, the removal efficiency of this option is set to 5 percent.

Application of sulphate-containing substrates to rice fields reduces CH_4 emissions because CH_4 producing bacteria compete for the same substrate as the sulphate reducing bacteria (van der Gon et al. 2001). Likewise, application of biochar to soils in rice fields improves soil fertility while contributing to reduced CH_4 emissions because carbon is added in a stabilized form, which inhibits the abundance and activity of methanogens (Han et al., 2016). The costs associated with these options are the costs of acquiring the sulphate-containing substrates or biochar and spreading them on the fields. In GAINSv4, a conservative assumption is that application of these types of CH_4 inhibitors can remove on average 20 percent of CH_4 emissions when applied as a stand-alone option and 5 percent when applied in combination with other options like improved water management.

The country-specific marginal abatement cost estimated for mitigation of CH_4 emissions in rice cultivation in year 2050 ranges from -10 to 40 €/t CO_2eq in GAINSv4.

S6.6. Solid waste

CH_4 from municipal and industrial solid waste is formed and emitted when biodegradable matter is decomposed under anaerobic conditions in landfills or during temporary storage of waste aimed for different types of treatment. CH_4 may also be released during loading or emptying of the reactor when organic waste is treated in anaerobic digesters to produce biogas or during treatment of organic waste in composts. In developing countries, it is common to scatter waste e.g., along riverbeds with the waste eventually ending up in the oceans, or to burn it openly in order to reduce its volume (Wiedinmyer et al., 2014). In both cases anaerobic conditions are unlikely and therefore CH_4 emissions remain very low, however, open burning of waste contribute to high air pollution emissions e.g., PM2.5 and NOx (Andersson et al., 2016; Anenberg et al., 2012; Das et al., 2018). In addition, waste contains a lot of carbon that could be harvested as a source of energy, making scattering and open burning a loss of potentially valuable renewable energy (Gómez-Sanabria et al., 2018). The activity data used in GAINSv4 is the total amount of waste generated before diversion to different types of treatment like recycling, energy recovery or landfill. Amounts of waste generated are first split by municipal or industrial solid waste and then by waste composition for municipal solid waste and by manufacturing industry sub-sector for industrial solid waste. Starting point for emission estimations are historical reported waste generation rates for municipal solid waste and industry waste reported to EUROSTAT (2015) for the EU countries and to the World Bank (Hoornweg and Bhada-Tata, 2012) and various national studies (see Gómez-Sanabria et al. 2018) for other regions. The methodology used to project future generation of waste by estimating waste generation elasticities is described in detail in the Supplement of Gómez-Sanabria et al (2018). The driver for industrial solid waste is growth in value added in the relevant manufacturing industry sectors. It can be expected that municipal solid waste generation per capita is positively related to per capita income (Hoornweg and Bhada-Tata, 2012) and that relative changes in income have a relatively larger effect on waste generation in high-income than in low-income countries. The reason for this being that food waste make up the major part of household waste generated in low-income countries and as countries become richer, it is primarily the generation of non-food waste (paper, plastics etc.) that grows and with per capita food waste generation remaining relatively stable. We used country-level data to estimate waste generation elasticities for different average per capita income intervals using data on income, urbanization rate and historical waste generation for 34 European and 10 non-European countries in the years 1995-2014 (EUROSTAT, 2015; OECD, 2016). Applying the estimated elasticities, future relative growth in the generation of municipal solid waste (MSW) per capita is estimated as a function of the relative growth in GDP per capita and urbanization rate (UNstat, 2014).

CH_4 from waste deposited on landfills is formed and released with a time delay of up to several decades. IPCC (2006, Vol. 5, Ch. 3) recommends the use of a First-order-decay model taking up to fifty years disposal into account. The GAINS model structure does not allow for implementation of a full First-order-decay model. Instead, a simplified structure is used, where the delay between waste disposal on landfills and CH_4 release is accounted for as a lag in the activity data of 10 years for fast degrading organic waste like food and garden waste and 20 years for more slowly degrading waste like paper, wood and textile. The lags correspond to approximate average half-life values for the respective waste types (IPCC, 2006, Vol.5, Tables 3.3 and 3.4).

Table S6-5 presents a summary of the various waste treatment options available in GAINSv4 model structure. The options considered preferable for a given waste category on the basis of overall environmental impacts are indicated with an asterisk. When constructing the marginal abatement cost curves for the solid waste sectors it has been necessary to extend the environmental objectives beyond only minimization of CH_4 emissions, as several of the options available (e.g. scattering and open burning) have dire environmental consequences on air quality and ocean life despite generating minimal CH_4 emissions. Instead the approach has been to identify 'preferred options' and apply them to a maximum technically feasible extent. In the long term, i.e. a timeframe long enough to allow for major infrastructural investments, the reduction potential accounted for in the marginal abatement cost curve for the solid waste sectors reflect the potentials and costs for moving from the current system to a system with an infrastructure supporting maximum source separation for reuse, recycling or treatment in biogas digesters. Any organic waste that cannot be source separated is to be combusted in a well managed (i.e., controlling for dioxins and other air pollutants) incinerator with energy recover and utilization. Hence, in the maximum technically feasible reduction (MFR) scenario, no untreated organic waste is assumed to go to landfills. Information on costs is provided in Höglund-Isaksson et al. (2018).

Table S6-5: GAINSv4 model structure for estimating CH_4 emissions from solid waste sectors.

Options and organic waste source categories		Waste management options included in the GAINS model	Max feasible reduction (MFR) application of preferred option
Options available to all organic waste categories		Incineration with energy recovery (well managed)* Incineration to reduce volume (not well managed) Landfill with gas recovery and flaring Landfill with gas recovery and utilization Landfill with compacting Landfill with cover of earth Unmanaged landfill -predominantly warm/humid conditions Unmanaged landfill -predominantly cold/dry conditions Open burning Scattering (no control option)	In the MFR scenario is assumed that all waste that is not possible to separate, reuse, recycle or treat in an anaerobic digester, is combusted in a well managed incinerator with energy recovered and utilized
Options available to specific organic waste categories	MSW -available to food and garden waste	Source separation & anaerobic digestion with gas recovery & utilization*	100%
		Source separation & household composting Source separation & large-scale composting	Current composting levels maintained to 2030, thereafter move to AD with biogas recovery
	MSW -available to paper waste	Source separation & paper recycling*	90%
	MSW -available to wood waste	Source separation & recycling for chip board production*	90%
	MSW -available to textile waste	Source separation & reuse or recycling*	90%
	Food industry waste	Anaerobic digestion with gas recovery and utilization*	100%
	Pulp and paper industry waste	Black liquor recovered and incinerated for energy purposes*	100%
	Textile industry waste	Incineration with energy recovery*	100%
	Wood industry waste	Incineration with energy recovery*	100%

* Preferred option for given waste category

S6.7. Wastewater

CH_4 emissions are formed when wastewater with a high organic content is handled under anaerobic conditions. Wastewater treatment plants serve to decompose compounds containing nitrogen and phosphor as well as carbon before discharge to a water body. Main gaseous products from wastewater treatment are CO_2 and molecular nitrogen, but also some CH_4. In the GAINS model, wastewater emissions from households and industry are accounted for separately. The activity data used to estimate emissions from domestic wastewater is number of people connected to centralized or decentralized collection of wastewater, respectively. This basically refers to wastewater from

urban and rural populations, except for most industrialized countries where wastewater collection services often include some rural areas as well. Country-specific data on population fractions of wastewater collected centrally are taken from UNFCCC (submission 2014), EUROSTAT (version as of June 26, 2013) and OECD (2015). Country-specific values for the biochemical oxygen demand per person (BOD) are used when available from UNFCCC-CRF (2014). When unavailable, an IPCC (2006, Vol.5, Table 6.4) default factor is used for the maximum CH_4 producing capacity (B_0).
Industry sectors identified by IPCC (2006, Vol.5, p.6.19) as potential sources for CH_4 emissions from wastewater are food, pulp- and paper industry and other manufacturing industries generating wastewater with an organic content, i.e., textile, leather, organic chemicals etc. The activity data for estimating CH_4 emissions from industrial wastewater is the amount of COD present in untreated industrial wastewater. These amounts are derived from production volumes combined with COD generation factors as specified in Table S6-6. Production volumes in ton product are taken from FAOSTAT (2015). Growth in value added by industry is used as driver for future projections. For the pulp- and paper industry, wastewater and COD generation rates reported in literature differ considerably between processes and between developed and developing countries. By comparing reported values from different sources, process specific generation rates are derived as presented in Table S6-6. It should be noted that when using process specific generation rates, for some food industries and pulp- and paper industry the estimated amounts of COD and CH_4 generated from industry come out several times lower than if using the IPCC default factor (2006, Vol.5, Table 6.9). Values for the maximum CH_4 production capacity *(B_0^{COD})* of wastewater from different industrial sectors are based on a literature review presented in Table S6-6. Weighted averages of the values for each process/product for the year 2010 were used to calculate the CH_4 production capacity by sector and country. An IPCC (2006, Vol.5, Table 6.2) default factor of 0.25 kt CH_4/kt COD is applied for the maximum CH_4 producing capacity (B_0^{COD}) when no value was available from literature.

The methanogenic process in the treatment of wastewater is sensitive to daily/seasonal temperature variations as temperature affects the microbiological community and the degradation rate of organic matter (Dhaked, Singh and Singh, 2010). With temperature being a relevant factor for the formation of CH_4 during treatment of domestic wastewater (Luostarinen et al. 2007), the GAINS model includes a country-specific temperature correction factor when deriving emission factors. Data on the rates of methanogenesis at different temperature intervals is adopted from Lettinga, Rebac, and Zeeman (2001), while daily data of the maximum temperature for years 2000, 2005 and 2010 at 25km resolution was taken from the Agri4 Cast Data Portal (JRC, 2015) for Europe and from NOAA (2018) for other parts of the World. No temperature correction factors are applied to emission factors for industrial wastewater, because the temperature is likely to be process-specific rather than determined by the outdoor temperature.

Current applications of different treatment practices for domestic and industrial wastewater are taken from UNFCCC (2014) CRF tables complemented with information from EUROSTAT (version as of June 26, 2013), OECD (data downloaded July 2015) and IPCC (2006, Vol.5, Table 6.5). There are no wastewater options available that primarily target CH_4 emissions. There are, however, several different ways of treating wastewater, which have different implications for CH_4 emissions (Pohkrel and Viraraghavan, 2004 and Thompson et al., 2001). When domestic wastewater is centrally collected and emitted to a water body with only mechanical treatment to remove larger solids, plenty of opportunities for anaerobic conditions and CH_4 formation are created. For this type of treatment, the CH_4 correction factor (MCF) used in GAINS is 1. With well managed aerobic or anaerobic treatment, the CH_4 formation is effectively mitigated and CH_4 emissions can be kept on a negligible level. MCF used in GAINS is 0.01 for aerobic treatment and 0.005 for well managed anaerobic treatment. With less well managed systems the occurrence of anaerobic conditions increase as well as CH_4 formation (IPCC 2006, Vol.5, Tables 6.3 and 6.8). Anaerobic treatment has advantages over aerobic treatment like lower costs, smaller volumes of excess sludge produced, and the possibility of recovering useful biogas, which can be upgraded to gas grid quality (Lettinga, 1995; Thompson et al., 2001). For industrial wastewater, it is assumed that the most effective way to reduce CH_4 emissions is to apply a two-stage process where the water is treated anaerobically with recovery of the biogas in a first stage, which is then followed by an aerobic treatment in a second stage (Latorre et al., 2007). The assumed MCF for this type of treatment is 0.05. In rural areas, domestic wastewater can be collected and treated in latrines, septic tanks or similar anaerobic treatment (USEPA, 1999). Investment costs for sewage treatment are taken from EEA (2005) and operation and maintenance costs from Hernandez-Sancho and Sala-Garrido (2011). Rural wastewater treatment costs are from USEPA (1999).

Table S6-6: GAINSv4 model assumptions for deriving CH_4 emission factors for industrial wastewater sources.

Industry	Product	Wastewater genertion in m3/ton. (range over different studies)	[COD] in kg/m3 Untreated wastewater. (range over different studies)	Maximum CH_4 producing capacity in kg CH_4/kgCOD. (range over different studies)	References
Food	Beer	4.95[a] (1.98 - 7.92)	4[a] (2-6 /1.2 - 125 UK)	0.23 [a-](0.19-0.27)	Debik and Coskun 2009; Kobya, Senturk, and Bayramoglu 2006; Fountoulakis et al. 2008; Şentürk, İnce, and Onkal Engin 2010; Azbar et al. 2004; Azbar et al. 2009; Healy, Rodgers, and Mulqueen 2007; Brito et al. 2007; Rodgers, Zhan, and Dolan 2004; Sharda, Sharma, and Kumar 2013; Shivayogimath and Jahagirdar 2015; Maya-Altamira et al. 2008.
	Vegetables oils[c]	0.8[a] (0.4 - 1.2)	45.5[a] (5 -804)	0.17[a] (0.11 -0.24	
	Wine	2[b] (0.8-14)	30.4[b] (3.1-150)	0.18[d]	
	Sugar Refining	0.69[a] (0.16-1.0)	6.15 [a] (2.3 -10)	NR	
	Meat	13 (IPCC)	5.4 [b] (3 -11)	0.22	
	Dairy Products[e]	3.05[b f](0.19-10)	8.8[b] (0.18 -25.6)	0.22 [b] (0.16 -0.27)	
Pulp	Bleached sulphate pulp	70[a] (30 -110)	1.55[a] (0.10-3.0)	NR	Janssen et al. 2009; Ekstrand et al. 2013; Larsson et al. 2015; Karlsson et al. 2011; Tezel et al. 2001; Chaparro and Pires 2011; Dufresne, Liard, and Blum 2001; Arshad and Hashim, 2012; Thompson et al. 2001.
	Unbleached sulphate pulp	50[a] (20 -80)	1.43[b] (1.35 -2.44)	NR	
	Bleached sulphite pulp	70[a] (40-100)	2.10[b] (0.62 - 8)	0.22[b](0.20-0.24)	
	Unbleached sulphite pulp	70[a] (40-100)	0.80[a] (0.20 - 1.4)	NR	
	Mechanical wood pulp	20[a] (5-50)	6.9[b] (2.71 - 10.37)	0.19[a] (0.12 - 0.27)	
	Semi-Chemical pulp	50[a](20-80)	2.19[a] (0.67 -3.71)	0.19[a] (0.11-0.27)	
	Recovered pulp[g]	20	3	NR	
	Other fibre pulp	20[g]	8.20[a] (7.7 -8.7)	NR	
Paper	Newsprint	9[a] (5-15)	3.5	NR	
	Printing and writing paper	60[h] (60-227)	0.81[a] (0.5-1.11)	NR	
	Recovered paper	12[a] (8 - 16)	0.51[a] (0.43 -0.58) [i]	0.22[a] (0.16-0.27)	
	Household/sanitary/tissue	8.50[a] (5-12)	1.02[a] (0.05-2)	NR	
	Wrapping papers[g]	20	0.08	NR	
	Paper and paperboard other	12[a] (8 - 16)	0.95[b] (0-11)	NR	

a Average
b Median
c Olive oil (Centrifugation and Pressing production processes (most of the data)), sunflower and cotton seed oil
d One study
e Including milk production, cheese, cheese whey, ice cream and butter
f Most of the data (11 total) are below 4.0 (8)
g based on Höglungd - Isaksson .2012
h 60 for UK 227 for Thailand
i Collected after the clarifier

S7: World region aggregations of GAINS model regions

Table S7-1: 174 GAINS model regions used in this study to model global CH_4 emissions.

World regions	174 GAINS model regions used in the modelling of global CH_4 emissions in this study
Africa	Egypt, North Africa (Algeria, Morocco, Tunisia, Libya), South Africa, Other Africa (All other African countries)
China	China (32 provinces)
Europe	EU-28 (28 countries), Norway, Iceland, Switzerland, Albania, Bosnia-H., Kosovo, North Macedonia, Montenegro, Serbia, Turkey
India	India (23 provinces)
Latin & Central America	Argentina, Bolivia, Brazil, Carribean (The Bahamas, Barbados, Cuba, Dominican Rep., Guyana, Haiti, Jamaica, Suriname, Trinidad and Tobago), Central America (Belize, Costa Rica, El Salvador, Guatemala, Honduras, Nicaragua, Panama), Chile, Colombia, Ecuador, Mexico, Paraguay, Peru, Uruguay, Venezuela
Middle East	Iran, Israel, Saudi Arabia, Rest of Middle East (Bahrain, Iraq, Jordan, Kuwait, Lebanon, Oman, Palestine, Qatar, Syria, United Arab Emirates, Yemen)
North America	United States of America, Canada
Oceanian OECD	Australia, New Zealand, Japan (6 provinces)
Russia & Former Soviet Union	Russian Federation (2 regions), Armenia, Azerbaijan, Belarus, Georgia, Kazakhstan, Kyrgizistan, Moldavia, Ukraine, Other Former Soviet Union (Uzbekisthan, Tajikistan, Turkmenistan)
Rest of South-East Asia	Afghanistan, Bangladesh (2 regions), Bhutan, Brunei, Cambodia, Indonesia (4 regions), North Korea, South Korea (4 regions), Laos, Malaysia (3 regions), Mongolia, Myanmar, Nepal, Pakistan (4 regions), Philippines (3 regions), Singapore, Sri Lanka, Taiwan, Thailand (5 regions), Vietnam (2 regions)

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Research article

Sustainable wastewater management in Indonesia's fish processing industry: Bringing governance into scenario analysis

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ABSTRACT

The government of Indonesia has pledged to meet ambitious greenhouse gas mitigation goals in its Nationally Determined Contribution as well as reduce water pollution through its water management policies. A set of technologies could conceivably help achieving these goals simultaneously. However, the installation and widespread application of these technologies will require knowledge on how governance affects the implementation of existing policies as well as cooperation across sectors, administrative levels, and stakeholders. This paper integrates key governance variables–involving enforcement capacity, institutional coordination and multi-actor networks–into an analysis of the potential impacts on greenhouse gases and chemical oxygen demand in seven wastewater treatment scenarios for the fish processing industry in Indonesia. The analysis demonstrates that there is an increase of 24% in both CH_4 and CO_2 emissions between 2015 and 2030 in the business-as-usual scenario due to growth in production volumes. Interestingly, in scenarios focusing only on strengthening capacities to enforce national water policies, expected total greenhouse gas emissions are about five times higher than in the business-as-usual in 2030; this is due to growth in CH_4 emissions during the handling and landfilling of sludge, as well as in CO_2 generated from the electricity required for wastewater treatment. In the scenarios where there is significant cooperation across sectors, administrative levels, and stakeholders to integrate climate and water goals, both estimated chemical oxygen demand and CH_4 emissions are considerably lower than in the business-as-usual and the national water policy scenarios.

1. Introduction

Leading up to the 23rd Conference of the Parties (COP 23) to the United Nations Framework Convention on Climate Change (UNFCCC), the government of Indonesia introduced its Nationally Determined Contribution (NDC). The NDC stated that Indonesia would aim to reduce greenhouse gas (GHG) emissions by at least 29% below business-as-usual (BAU) projections by 2030. A higher pledge of 41% below BAU by 2030 was also submitted contingent upon international financial and other forms of support. Indonesia's NDC breaks down these pledged reductions by sector, outlining possible contributions from key emission sources (Republic of Indonesia, 2016). The NDC emphasizes reductions from preserving Indonesia's forests and shifting to renewable energy; the land use and energy sectors are significant contributors to overall emissions in Indonesia (Wijaya et al., 2017). Yet the NDC also references reducing emissions of GHG from industrial wastewater management.

Indonesia is not only the world's fourth largest populated country, but one of its largest fish and seafood producers. However, due in part to the fast growth of seafood and other industries, more than 70% of Indonesia's rivers are classified as "polluted" (Lorenzo and Kinzig,

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2019). To improve water quality, the government of Indonesia, with support from international organizations, created The Program for Pollution Control, Evaluation and Rating (PROPER). PROPER disseminates color-coded ratings of companies' pollution management performance to generate the kind of public and peer pressure that can induce industrial compliance with national pollution control standards (Torres and Kanungo, 2003). Among the industries covered by PROPER, the fish processing industry performs at the lowest level in terms of environmental performance. Therefore, the Indonesian Ministry of Environment and Forests is making considerable efforts to adopt and implement appropriate wastewater treatment methods (Consultants Co, 2015). Many of these efforts can help achieve water quality and climate change objectives while contributing to the Sustainable Development Goals (SDGs).

The above reasons suggest that Indonesia is an important case to examine the impacts of climate and wastewater management goals in its own right. A few additional factors clarify why Indonesia's experience could offer lessons for other rapidly industrializing countries. Like many other fast-growing countries, although Indonesia has tightened wastewater regulations, the resources to enforce regulations tend to be limited (Asian Development Bank, 2005). As is often the case in rapidly developing countries, industrial capacity to generate effluent outpaces government capacities to regulate pollution. High levels of water pollution are additionally a result of limited access to wastewater management technologies, especially for small and medium sized enterprises (SMEs). Yet another contributing factor behind these challenges is the difficulties of scaling wastewater treatment technologies. Even if a single enterprise installs these technologies, whether many small and frequently dispersed emission sources follow suit is far from guaranteed.

This article suggest that more attention needs to be placed on how governance can help overcome some of the above policy and institutional challenges so as to facilitate the adoption and spread of key technologies. More concretely, governance—the exercise of authority in the pursuit of one or more policy goals—is critical to making links between pollution and climate issues. It also influences whether agencies at multiple levels enforce regulations. It is finally related to whether this sufficient interagency coordination and networks enable the spread of successful solutions (Nanda, 2006; Stoker, 1998; Hewitt de Alcántara, 1998). However, while many of the models developing emission reduction scenarios demonstrate positive effects of sustainable wastewater management on climate change and water pollution, few systematically consider how different levels of governance influence policy enforcement, coordination and networking. The omission of these considerations may lead to modelling results and policy recommendations that diverge sharply from reality (Hourcade and Crassous, 2008).

The main contribution of this article is to better capture that reality by integrating insights on governance into modelling-focused climate policy and sustainable development research. In recent years, some studies have sought to bridge quantitative assessment modelling with qualitative transitions research. For example, a branch of sustainable transitions research has sought to bring in "the types of actors, their goals, strategies, and resources as well as institutional changes" contributing to the spread of sociotechnical innovations (De Cian et al., 2017). However, there remains considerable scope to translate how key actors, agencies and institutions can be incorporated into scenarios that often feature in integrated assessment modelling. A novel way forward is to use research on governance to provide insights into three sets of considerations influencing the spead and scale of technology changes. These insights are critically important because it is often asserted in the water sector that effective resource management is more a governance than technical issue (Casiano Flores et al., 2017; Grigg, 2011). To a significant degree, these three sets of factors also mirror the main modes of governance in recent work on sustainable water governance (Pahl-Wostl, 2019).[1]

The first set of governance insights falls under what is often called government capacity and effectiveness. As that title suggests, these sets of issues involves governments having sufficient financial and human resources to implement a variety of their own regulations (Kaufmann et al., 2010; Rock, 2002). Capacity warrants attention both because responsibilities for implementing climate and wastewater management regulations are increasingly delegated to often underfunded local governments (Asian Development Bank, 2005; Casiano Flores et al., 2019). These issues also merit reflection because the lack of resources cannot only contribute to well-studied implementation gaps (Lester and Goggin, 1998). Finally, capacity is pertinent because increasing resources without sufficient institutional coordination could paradoxically lead to more GHG emissions (see Section Three).

The second set of governance insights concerns whether effective coordination exists between climate and pollution control agencies within and across levels of decision making. Insufficient horizontal coordination can create a disconnect between climate mitigation and development policies, including water management and pollution control policies (Arens et al., 2014; Corfee-Morlot et al., 2009; Peters, 1998). On the other hand, insufficient vertical coordination could lead to challenges acquiring the financial and other resources needed to bring promising technologies to scale. Coordination troubles within and between levels can result bureaucratic turf wars and incoherent policies that are familiar to those working on integrated water resource management and many other contexts (Biswas, 2008; Jones et al., 2019).

The third set of governance considerations involves the spread of successful examples using networks of business and civil society actors in and outside governments or what is often called governance "beyond the government" (Bressers and Kuks, 2013). These networks rely more on informal institutions, trust and voluntary agreements; they also have a higher degree of flexibility that facilitates the sharing of information on innovative solutions and collective learning about which technologies work in which contexts. Networks can complement more the more formal institutions and structures discussed above (Pahl-Wostl, 2019).

Water management experts have cautioned against applying one-size-fits all governance recommendations without an appreciation of context (Ingram, 2013; Suhardiman et al., 2015). However, in this case the three sets of factors outlined above—capacity, coordination, and networks—are related to development that have helped to shape Indonesia's policy and institutions. More concretely, several of relevant changes followed decentralizing reforms in the 1990s that delegated significant enforcement responsibilities for environment regulations to local governments. These reforms did not, however, ensure sufficient numbers of properly trained staff were employed to manage assigned tasks (Rabasa and Chalk, 2001). This meant that regulation No. 82/2001 (water quality), No. 7/2004 (water resources management) and other key sectoral policies (see Table A1 supplement) often suffered from implementation gaps (Arcowa, 2018). These gaps explain why the fish processing industry encountered hurdles ranging from shortages of technical expertise to low levels of government funding (Apip et al., 2015). They also help to understand why simpler aerated treatment ponds became more common than activated sludge technologies or other cleaner technologies (AECOM and Sandec 2010).

Recently, there have been some developments involving governance that may help close these implementation gaps. Some of these involve increases in institutional capacity. For instance, observers have pointed

[1] Pahl-Wostl (2019) argue for meta-governance where governments employ a mix of hierarchies, markets and networks to steer decisions to more sustainable water policies. The third set of considerations in this article combines markets and networks.

out that the growing stringency of wastewater effluent discharge regulations have led to improved water quality in parts of Indonesia (Soedjono, 2018). Others have suggested that Indonesia has embraced the aforementioned PROPER programme to boost compliance with regulations (Consultants Co, 2015; Torres and Kanungo, 2003).

Another notable and relevant set of governance reforms involves Indonesia's response to climate change. Since 2011, Indonesia has placed growing attention to climate policy. This has entailed a core group of experts working with the National Development Planning Agency (BAPPENAS) and relevant line ministries to draft climate change plans. A coordinating unit and several sectoral working groups have been established to better align strategies and plans. To some extent, Indonesia's NDC reflects this cross-sectoral or horizontal integration as it draws from series of sector specific policies and regulations that extend out to 2030. A related set of reforms involves the sharing of plans with provincial and lower level local governments that are then expected to further specify and tailor their content to local contexts (Wijaya et al., 2017).

A final set of reforms has centered on engaging the private sector in wastewater treatment in the fish processing industry. Much of this interest has revolved around developing a regulatory framework for public and private partnerships (PPP). That framework would involve both national and local governments working with overseas donors to attract investments in advanced wastewater treatment infrastructure and technologies. It would also make it a point to engage private sector early and often in the planning (i.e. construction, operation and maintenance) (Asian Development Bank, 2019). A possible consequence of these arrangements is that there would be higher levels of compliance with regulations as a result of the dissemination of cleaner technologies. One of the few locales as Muncar-Banyuwangi that have taken actions in minimizing their wastewater discharges with standards exceeding national regulations suggest just such a possibility (Widodo, 2016).

This study therefore attempts to bring the aforementioned considerations involving governance into an analysis of water pollution and greenhouse gas (GHG) emissions from several wastewater treatment scenarios in Indonesia's fish processing industry. As such, it will not only contribute to wastewater management research specifically, but integrate qualitative insights into quantitative assessment research generally (Meuleman, 2015). In the process, it will help fill an important research gap on whether what is feasible in a model can be achieved in applied settings.

The rest of the paper is structured as follows: Section 2 presents the methods applied to project fish production to estimate Chemical Oxygen Demand (COD) load and GHG emissions up to 2030, along with a description of scenarios disigned to intergrate technological development and governance. Subsequently, in Section 3, the results are presented, including estimations on COD removal efficiencies, GHG emission reductions, together with an analysis of the co-benefits of the different scenarios included and political implications. Section 4 highlights the limitations of the study and lastly Section 5 presents the conclusions of the study with a focus on the way forward.

2. Material and methods

While the previous section suggested the possibility of three sets of governance reforms influencing the adoption and spread of wastewater treatment technologies, it did not offer insights into their implications for water pollution control or climate mitigation. The methods section outlines how those insights will be provided based on the rationale that COD removal efficiencies (%) and GHG emissions ($ktCO_2eq/year$) depend on how different forms of governance affect the type of wastewater treatment technology implemented, i.e., on the adoption of the different scenarios.

The first step is a description of the fish processing industry, followed by a summary of the methods used to estimate COD load and GHGs from the wastewater treatment in the fish processing industry in Indonesia until the year 2030, and then complemented by the description of scenarios. The method described in Gómez Sanabria et al. (2018) is applied to quantify the organic content and biogas generation from anaerobic wastewater treatment. GHG emissions are estimated based on Höglund-Isaksson et al. (2015). For detailed information on technologies implemented, variables and on assumptions and equations applied refer to the Supplement Section A3 and Section A4.

2.1. *Wastewater from fish processing industry*

Fish processing involves three main sets of activities: fish refrigeration, canning and fishmeal processing. The fish refrigeration plant is the where the first step occurs. The process consists of washing and sorting the fish into different groups for sale in packed boxes with ice. The fish to be frozen follows the same steps but requires refrigeration and storage until it is delivered (Björk and Schou Kongstad, 2016). The canning process involves three main sub-processes; reception of raw materials and ingredients, processing (including cooking, washing and canning, and final operations) (Valiño et al., 2007). The fishmeal process involves cooking, pressing, drying and grinding the fish (Green, 2016).

Wastewater from the fish processing contains a mixture of organic substances, nutrients, oil and fats (Purwanti et al., 2018). The characteristics of the effluent varies between the different processes but also depends on the composition of the raw fish (Table 1).

2.2. *Fish production projections, COD load and GHGs estimations*

The paper uses derived COD load in untreated wastewater to assess the organic load removal efficiencies and GHG emissions from different wastewater treatment options. The COD amounts are derived from production volumes combined with wastewater generation rates and COD generation factors (Höglund-Isaksson et al., 2018). National production volumes (from fish catchment and aquaculture combined) are taken from FAO-Fisheries and aquaculture statistics (FAO, 2018). Future production projections to 2030 are based on the Baseline Scenario presented in the "Fish to 2030" (World Bank, 2013) and "Exploring Indonesian Aquaculture" (Phillips et al., 2015) studies. No significant growth in captured fisheries is expected, therefore a 0.4% growth in fish catch is assumed for the whole period until 2030 (Ipsos Business Consulting, 2016; World Bank, 2013). Projections in aquaculture production assume 5.6% annual growth (Phillips et al., 2015). The main growth driver for fisheries in Indonesia is high domestic but also international demand (Ipsos Business Consulting, 2016).

Regional production volumes are based on the regional percentage of production by fish type presented in Phillips et al. (2015) and on the number and size of factories in each region. The regions included are Java, Sumatra, Kalimantan, Sulawesi and Maluku-Papua (Figure A1 in the supplement).

The quantification of COD in untreated industrial wastewater is carried out by applying the IPCC method (IPCC, 2006, Volume 5, Chapter 6, Equation 6.4 and Equation 6.6). The assessment of the GHG emissions and energy generation potentials is based on the removal efficiencies and application rates of the different wastewater technologies adopted. COD removal efficiencies, CH_4 emission factors [$ktCH_4$/kt COD removed], biogas composition and electricity consumption [kWh/kg COD removed] are based on the IPCC Guidelines (2006), Consultants Co

Table 1
Characteristics of effluents of fish processing plants.

Process	Wastewater [m^3/ton]	COD load [kg/ton]	pH
Refrigeration plant	10–30	2–6	6.9
Canning factory	15–30	2.25–4.5	3.8–6.4
Fishmeal factory	12	12	6–7

Source: Based on (Chowdhury et al., 2010)

(2015) and Spokas et al. (2006). Regional CO_2 emission factors [$ktCO_2$/KWh] are adopted from Directorate General of Electricity, Ministry and Mineral Resources, Indonesia; 2016 - Emission factor reference official document (see supplement section A4).

2.3. Wastewater treatment scenarios

Seven different scenarios for wastewater management are developed in the timeframe to 2030. The applied wastewater treatment technologies originate from the study: 'Co-benefits of the wastewater treatment technologies: Indonesia fish processing industry' (Consultants Co.,Ltd, 2015). The phase-in of the technology assumes implementation of 15% by 2020, 50% by 2025 and 100% by 2030 for the 'Business as Usual', 'National wastewater policy', and 'Climate change policy scenarios', respectively. The phase-in of the technology for the further four co-benefit scenarios depends on the areas or forms of governance that are strengthened. For vertical and horizontal integration, a maximum technological phase-in of 80% by 2030 is assumed. The multi-stakeholder network form of governance scenarios allows for 100% of technological phase-in by 2030. Scenarios consider the national effluent standards of the fish processing industry (Decree of Ministry of Environment& Forestry no.5/2014) and the GHG emission reduction targets under the NDC.

Description of the scenarios developed are presented in Table 2. Each scenario is designed based on the three main elements: policy, form of governance and technology. Policies adopted are according to the policies presented in the supplement A1 – Table A1 and are in line with Indonesias's NDC. Assumptions on the form of governance are based on the literature review presented in the introduction. Technological development is based on Consultants Co.,Ltd (2015). A detailed description of scenario narratives is presented in the supplement Section A6 - Table A4.

Table 2
Description of the scenarios.

Scenario	Policies	Forms of governance	Technology
Business-as-usual (BAU)	Current situation - no further enforcement	Current situation	Untreated/ anaerobic lagoons
National Wastewater Policy (NWP)	National wastewater policy	No coordination between wastewater and climate agencies	Aeration lagoon plus Activated sludge
Climate Change Policy (CCP)	Climate change policy	No coordination between wastewater and climate agencies	Swimbed
Co-benefits vertical horizontal coordination (CB1vh)	National wastewater policy and climate change policy	Vertical horizontal coordination	Up-flow Anaerobic Sludge Blanket (UASB) plus Activated Sludge (with gas recovery and used).
Co-benefits vertical horizontal coordination (CB2vh)	National wastewater policy and climate change policy	Vertical horizontal coordination	Up-flow Anaerobic Sludge Blanket (UASB) plus Swimbed (with gas recovery and used).
Co-benefits multi-stakeholder network (CB1ms)	National wastewater policy and climate change policy	Multi-actor network	Up-flow Anaerobic Sludge Blanket (UASB) plus Activated Sludge (with gas recovery and used).
Co-benefits multi-stakeholder network (CB2ms)	National wastewater policy and climate change policy	Multi-actor network	Up-flow Anaerobic Sludge Blanket (UASB) plus Swimbed (with gas recovery and used).

The scenarios are implemented at both the national and sub-national levels. Sub-national regions are selected based on the relative contribution to national production of key aquaculture commodities and farming systems (Phillips et al., 2015) and on information available regarding the distribution of fish factories and number of employees (Consultants Co.,Ltd, 2015). As a result, scenarios have been developed for the following sub-national regions: Java, Sumatra, Kalimantan, Sulawesi and Maluku-Papua. Implementation of costs for the different scenarios were not included in this analysis due to a lack of information.

2.4. Limitations

The study is based on information resulting from a pilot project in which the actual technology tested is the swimbed technology. Some of the parameters used for the development of the analysis are derived from values provided by Consultants Co. Ltd (2015). Retention times, wastewater generation rates, organic load rates, efficiencies on biogas formation and biogas recovery are mostly based on default values which do not differentiate the type of process e.g., refrigeration plant, canning factory and fish meal factory. It is also assumed that different processes operate in optimal conditions. However, it is well known that microbial community is extremely sensitive and, if not properly managed, the process would result in reduced biogas production (Munk et al., 2010).

It is further important to note that the quantification of N_2O is not included in this study. The reason it is not included is that the case study focused solely on CH_4 and CO_2 emissions. However, the authors are aware that N_2O is the third most powerful GHG, having a global warming potential that is 265 higher than CO_2 over a 100 year time horizon (IPCC, 2014) and causes long-term disturbances to the stratospheric ozone layer. N_2O emissions from wastewater treatment plants are the result of the nitrification and denitrification processes (Zheng et al., 2019) occurring mainly in the activated sludge units (Campos et al., 2016).

Regarding electricity generation, it is assumed that the average national fuel mix in electricity production is used in wastewater treatment plants. However, fuel mix might change at a sub-national or regional level. Though average regional emission factors for electricity production are used, the specific regional representation of fuel mix is not taken into account due to lack of relevant data.

The article analyses the implementation of technologies to reduce water pollution and GHGs at the last stage before the effluent enters the environment. Nonetheless, wastewater treatment should be looked at in a holistic manner as wastewater offers a huge potential for recovery of resources. In addition to energy generation, wastewater can also provide 'resources' such as bioactive compounds, antimicrobial agents and natural chemicals (Federici et al., 2009). Furthermore, water recycling and reuse is also important when implementing wastewater treatment systems (Chen et al., 2019).

The development of, *inter alia,* strategies integrating the circular economy framework, health-related aspects and corporate social responsibility could further contribute to the realization of the national sustainable development goals (SDGs). Such aspects—which are also reviewed in the conclusion—would likely further strengthen the case for improved wastewater treatment but an assessment of all of the benefits would require additional data that is not available to the authors.

Beyond the environmental benefits, costs are critical when selecting wastewater management treatment technologies. Factors influencing investment, operation and maintenance costs include flow rate, pollution load, number and type of treatment stages of the facility and removal efficiencies (Hernandez-Sancho et al., 2011). The cost aspect was not part of this study due to the lack of quantitative information in terms of construction, maintenance, and sludge disposal costs for the different technologies. Therefore, it was not possible to carry out the corresponding cost analysis which would be vital in a feasibility study to identify the potential economic constraints that could prevent the adoption of a specific technology system.

3. Results

This section summarizes the key results at the country and sub-national levels in terms of COD removal and GHG emissions from wastewater treatment in the fish processing industry.

3.1. COD removal efficiency

The maximum COD removal efficiencies are expected to be achievable upon the introduction and diffusion of the different treatment strategies and policies. Removal efficiencies show that there is significant potential to improve the removal of organic load and to reduce COD concentration in the wastewater effluent when relevant institutions have adequate human, financial and technological capacities and resources.

Reaching full implementation of NWP and CCP would require overcoming challenges in BAU regarding capacities and coordination in the corresponding institutions. When coming to the implementation of any of the co-benefit scenarios, overcoming these hurdles is even more challenging due to the need to align the agendas of vertical and horizontal (vh) regulatory agencies as well as working with multi-stakeholders and international organizations (ms).

The implementation of the different scenarios will result in an overall increase of COD removal efficiencies of around 12% (for all scenarios) compared to BAU in 2020, and between 43% (NWP) and 46% (for the other scenarios) compared to BAU in 2025. By 2030, the implementation of CCP, $CB1_{ms}$ and $CB2_{ms}$ would result in higher removal efficiencies (98.6%–98.4%) as a result of successful coordination and additional international support in the case of multi-stakeholder (ms) scenarios. Interestingly, $CB1_{vh}$ and $CB2_{vh}$ show the lowest COD removal efficiency (79%) arising from the lack of involvement of multi-actor networks (see Figure A1 in the supplement).

Fig. 1 shows that, although there is an improvement over time in the reduction of COD concentration and load in the effluent resulting from the implementation of the different scenarios, it would not be possible to fully comply with the Indonesian wastewater regulations before 2030.

In the BAU scenario, current technology prevails up to 2030 with a maximum removal efficiency of COD at 2.8%. The low removal efficiency is a consequence of an increase in the national projection of aquaculture commodities production due to insufficient enforcement of the wastewater standards set for the fish processing industry. Full implementation of NWP, CCP $CB1_{ms}$ and $CB2_{ms}$ by 2030 is expected to translate into compliance or even over-compliance with national effluent standards for different fish industry processes in terms of COD concentration (refrigeration plant 200 mg/l, caning 150 mg/l and fish meal factory 300 mg/l) and COD load (refrigeration plant 2.0 kg/ton, caning 2.25 kg/ton and fish meal factory 3.6 kg/ton). Full implementation of $CB1_{vh}$ and $CB2_{vh}$ by 2030 is, however, not expected to be sufficient to meet regulatory standards; rather COD concentration standards are expected to be exceeded by 40% for the refrigeration plant, 53% for the caning and 6% for the fishmeal factory. Concerning COD loads, the standard is expected to be exceeded by 40% for the refrigeration plant and 30% for caning. Full implementation of $CB1_{vh}$ and $CB2_{vh}$ by 2030 would, however, meet the COD load standard for the fish meal factory (Fig. 1). The estimation of the COD discharge load per year after implementation of the different technologies by region and at country level is presented in the Supplement Table A4 and A5.

3.2. Greenhouse gas emissions

Estimates of CH_4 and CO_2 emissions from fish processing wastewater handling have been carried out for the different scenarios (Fig. 2). All GHG are expressed in CO_2eq terms assuming a global warming potential of 100 years (IPCC AR5, 2014). Methane emissions in this article are emissions from the wastewater treatment process at the discharge point and during sludge treatment. Emissions of CO_2 refer to emissions associated with the production and consumption of electricity required for reduction of COD in different processes. The results illustrate that both the choice and scaling of alternative wastewater treatment strategies influence whether the overall impact on GHG emissions will be positive or negative in comparison to BAU.

In the BAU, there is an increase (by 24%) in both CH_4 and CO_2 emissions between 2015 and 2030. This is partly driven by an expected increase in production volumes and partly resulting from the low application of anaerobic lagoons and inappropriate management of the technology. Moreover, considering the current situation, larger quantities of wastewater from fish processing industries are released without previous treatment. Consequently, the circumstances do not fully favor the formation of anaerobic conditions, thus lowering the capacity of CH_4 formation from untreated wastewater. Since anaerobic lagoons require no or little energy (EPA, 2002), emissions of CO_2 related to energy consumption are small.

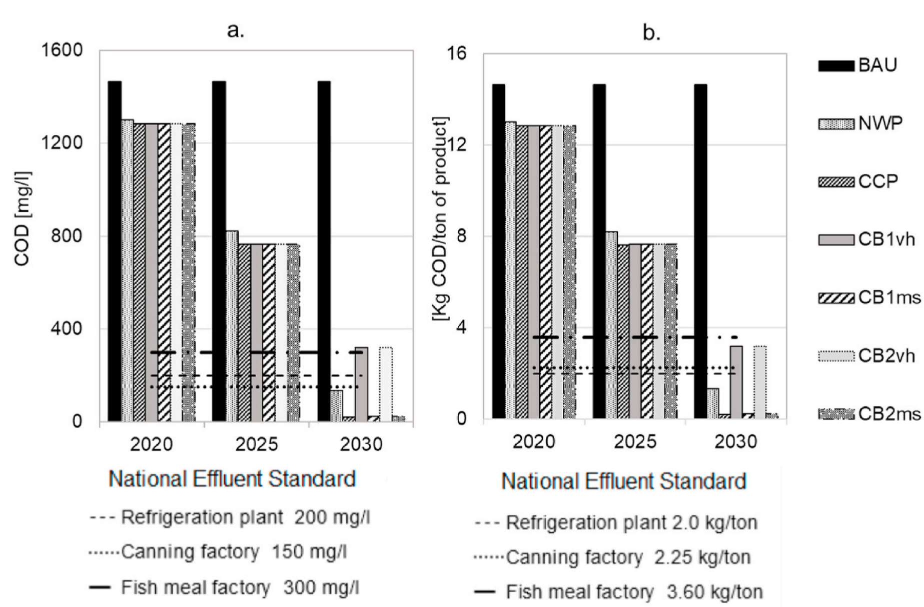

Fig. 1. a. Effluent COD concentration b. Effluent COD load.

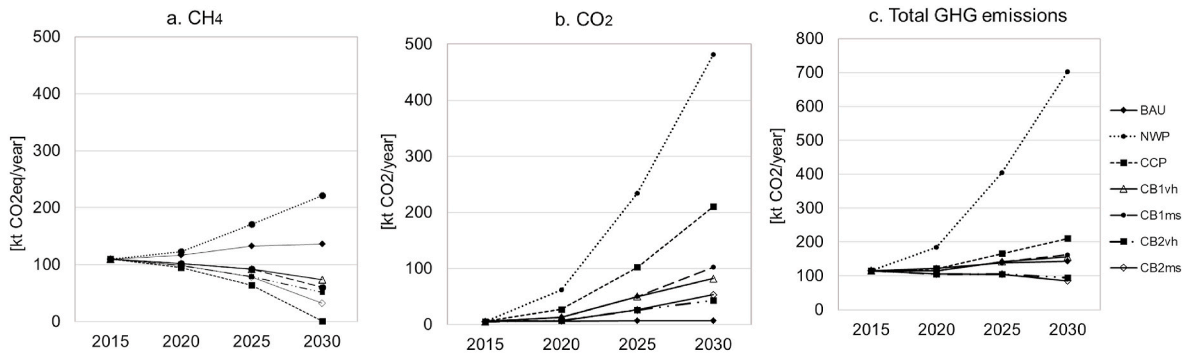

Fig. 2. a. CH_4, b. CO_2 and c. total GHG emissions from wastewater treatment in fish processing industry – Indonesia. Regional figures can be found in the Supplement Figure A2.

In the NWP, total GHG emissions are expected to be about five times higher than BAU in 2030 (Fig. 2c). An expected increase in CH_4 emissions arise from anaerobic conditions during handling and landfilling of the sludge, which more than outweigh the CH_4 emissions from the current lack of treatment (with some use of anaerobic lagoons) in the BAU. CO_2 emissions are expected to be considerably higher in the NWP than in the BAU, due to a higher electricity consumption required by the artificial aeration needed to stimulate biological oxidation in the treatment process.

At the same time, CH_4 emissions from the CCP, $CB1_{vh}$, $CB1_{ms}$, $CB2_{vh}$ and $CB2_{ms}$ are expected to be considerably lower than BAU due to the implementation of improved wastewater technologies. The full implementation of CCP is expected to generate the lowest CH_4 emissions, followed by $CB2_{vh}$ and $CB2_{ms}$ (Fig. 2a). One of the advantages of the CCP is that the technology (swimbed) allows for longer retention times due to sludge recycling (Rouse et al., 2004), which reduces sludge production along with CH_4 emissions from its management. However, this technology has high electricity requirements resulting in high CO_2 emissions, which offset the reduction in CH_4 emissions in terms of global warming impact (Fig. 2b).

Given an assumption that the average national fuel mix in electricity production is used in the energy supply to wastewater treatment, CO_2 emissions in 2030 are expected to be 69 times higher in the NWP than in BAU. This is partly due to a higher electricity consumption in the NWP (2.97 GWh/kt COD removed) than in the BAU (1.39 GWh/kt COD removed) and partly because in the CCP, $CB1_{vh}$, $CB1_{ms}$, $CB2_{vh}$ and $CB2_{ms}$ technologies are expected to increase electricity consumption and associated CO_2 emissions compared with BAU.

It is estimated that the electricity required in NWP will be 562 GWh in 2030, which is 44% higher than CCP, with the latter also using aerobic systems but with a different technology. From the type of technology adopted in $CB1_{vh}$, $CB1_{ms}$, $CB2_{vh}$ and $CB2_{ms,}$ it would be possible to recover and use the biogas generated. The technology adopted in the CB2 set of scenarios requires 30% less electricity per kt COD removed than the technology adopted in the CB1 set. Also, the use of the own biogas as a source of electricity would in 2030 replace 36% and 52% of the required external energy in the CB1 and CB2 sets of scenarios, respectively (Fig. 3).

The expected total GHG emissions in 2030 turn out higher than BAU for all technologies except the CB2 set of scenarios (Fig. 2c). The latter combines an Upflow Anaerobic Sludge Banked technology with Swimbed technology and recovers the biogas to generate electricity for own-plant use. In general, scenarios implementing aerobic treatment alone would result in higher electricity consumption as aeration uses up around 60%–70% of total energy required for the wastewater treatment process (Maktabifard et al., 2018). The advantage of implementing the $CB2_{ms}$ scenario would then be the combination of anaerobic treatment which generates CH_4 that can be used to supply part of the electricity required by the swimbed technology (regional figures displaying GHG emissions can be found in the Supplement Figure A3).

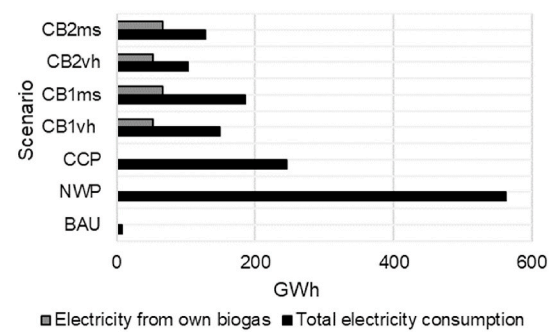

Fig. 3. Electricity consumption wastewater treatment in fish processing industry in 2030 – Indonesia.

Fig. 2c summarizes the total GHG emission trajectories for the different scenarios. The NWP is expected to generate the highest GHG emissions, owing to the largest emissions from electricity consumption. In contrast, the Co-Benefits 2 multi-stakehodler network scenario ($CB2_{ms}$) has the lowest GHG emissions due to a lower energy consumption coupled with recovery and use of the biogas generated from the wastewater treatment to offset part the energy required for the wastewater treatment process.

3.3. Analysing the co-benefits of wastewater treatment

The implementation of NWP and CCP do not deliver co-benefits in terms of simultaneous reduction of COD and GHG emissions due to the absence of multi-level, multi-stakeholder governance. In contrast, the set of CB scenarios, which address environmental concerns by reducing COD concentration in the effluent while reducing GHG emissions from wastewater treatment through multi-level governance, deliver those co-benefits. In that sense, the scenario providing the maximum benefits is the one which combines the highest COD removal efficiencies with the lowest GHG emissions per unit of COD removed. This, in turn, can ensure compliance with the national wastewater standards while reducing GHG emissions from wastewater treatment and therefore supporting the achievement of the Indonesian NDC targets.

Fig. 4 shows the relation between GHG emissions and COD removal efficiency. The adoption of any alternative scenarios to the BAU, except for the $CB1_{vh}$ and $CB2_{vh}$, would result in compliance with the national effluent standards by 2030. Interestingly, Fig. 4 also shows the highest GHG emissions expected from the implementation of NWP. CCP, $CB1_{ms}$ and $CB2_{ms}$ depict similar COD removal efficiencies, nevertheless, $CB2_{ms}$ (UASB plus swimbed) provides the maximum benefits in terms of GHG emissions and is the only option that improves both COD effluent concentrations and mitigates climate impacts. By 2030, the $CB2_{ms}$ removal

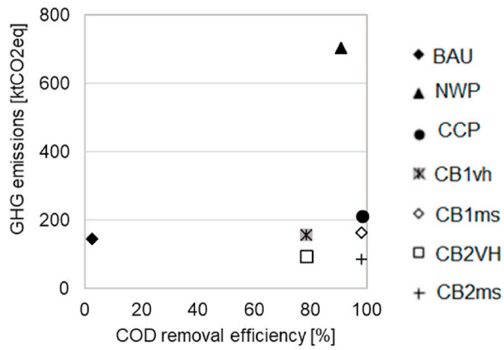

Fig. 4. Multiple benefits of the analysed scenarios.

efficiency would reach a maximum of 98.4% COD removed, while reducing GHG emissions by 60% compared to BAU. GHG emissions in the CB2$_{ms}$ is also 47% lower than in CB1$_{ms}$ and 57% lower than in CCP.

A summary of the main achievements resulting from the implementation of the different scenarios in the year 2030 is presented in Table 3. The scenario providing the maximum co-benefits is highlighted in gray.

3.4. Policy implications

The Indonesian government has established ambitious targets for reducing GHG emissions but also strict wastewater treatment standards. In an attempt to achieve sustainable development, Indonesia is seeking policies that can provide multiple benefits from climate change mitigation and water pollution prevention. Therefore, the identification and dissemination of appropriate wastewater treatment strategies are vital.

Despite strict wastewater legislation, appropriate wastewater treatment facilities (especially in the fish processing industry), are still lacking. This results in effluents with high pollution load which, on the one hand, could potentially create anaerobic conditions facilitating the formation of methane and, on the other hand, contaminate watercourses. If current conditions are maintained, the pressure on water quality will further increase, thereby threatening not only environmental but also health and social conditions. By 2030, without the enforcement of wastewater legislation and taking into account a growth in total production volumes by 24%, COD concentration in the effluent could be as much as 6–7 times higher than national standards for the fish processing industry. Strengthening capacities to implement existing wastewater regulations is therefore essential (Kaufmann et al., 2010; Rock, 2002).

Currently available technology and effective implementation of existing wastewater legislation could decrease water pollution and reduce GHG emissions from wastewater treatment. However, this is only possible with careful choices of treatment technologies and the source of energy. Such choices must consider the COD removal efficiency, the conditions for CH_4 formation and release in the different treatment stages, as well as the energy source and CO_2 emissions from additional electricity requirements. Here again, while enforcement of policies for the outlined co-benefits will be *necessary* for meeting established targets, focusing exclusively on capacities may not be *sufficient* for a sustainable future. To reach both effluent and climate targets in the CB2$_{ms}$—where COD concentration in the effluent would likely be eight times lower than the standard fish processing industry limits and GHG emissions would be 60% lower than in the BAU or around 0.38 ktCO$_{2eq}$/kt COD removed—multiple levels of governments and stakeholders need to work together on shared purposes and common goals (Arens et al., 2014; Corfee-Morlot et al., 2009; Peters, 1998).

In fact, the aforementioned benefits will require a focus on multilevel governance that brings together pollution and climate institutions at national and local levels as well as different networks or stakeholders and international institutions (Pahl-Wostl, 2019; Bressers and Kuks, 2013). Without context-appropriate coordination at different levels (Ingram, 2013), the benefits offered by the CB2$_{ms}$ would not be realized. The misalignment of efforts between agencies and actors might lead to an overly narrow focus on achieving a single objective i.e., water pollution, without realizing it jeopardize progress on another objective i.e., GHG mitigation. Therefore, climate objectives will need to be incorporated into local urban policies (Gouldson et al., 2016). The good news is that some studies have shown that reaching climate goals i.e., NDC targets, requires actions at the sub-national level. An additional piece of good news is that initiatives as ‘United Cities and Local Governments’ are working to reduce GHGs emissions locally and achieve national GHGs reduction targets (Betsill and Bulkeley, 2006). These studies and actions are also supported by work that suggests that national governments can provide the financing to bring to scale promising local innovations (Suhardiman et al., 2015).

Hence, placing more attention on governance for managing wastewater and climate change in Indonesia could help overcome the challenges faced by industries in relation to budget, technology transfer and capacity building (Arcowa, 2018) and support the move towards a reduction in both water pollution and GHGs. The transition to a multi-level, multi-stakeholder forms of governance could support the identification of shortcomings related to the implementation of the current wastewater legislation as well as open several opportunities for policy frameworks targeting multiple objectives.

4. Conclusions

This article offers a unique perspective on the governance reforms needed to achieve climate and wastewater treatment goals. This follows research that argues that managing wastewater is frequently a governance issue (Casiano Flores et al., 2017; Grigg, 2011). It provides that perspective by integrating work on qualitative governance and quantitative modelling research. While there are some limitations to this

Table 3
Achievements by scenario in 2030.

Scenario	Policies	Form governance	Technology	Total GHG emissions [kt CO_2eq/year]	COD removal efficiency [%]	Electricity replaced by own biogas [%]
BAU	Current situation	Current situation	Untreated/anaerobic lagoons	143	3	0
NWP	NWP	No coordination	Aeration lagoon + Activated Sludge	703	91	0
CCP	CCP	No coordination	Swimbed	211	99	0
CB1vh	NWP + CCP	Vertical-horizontal coordination	UASB + Activated Sludge + Energy recovery and use	156	79	36
CB1ms	NWW + CCP	Multi-actor network	UASB + Activated Sludge + Energy recovery and use	162	98	36
CB2vh	NWP + CCP	Vertical-horizontal coordination	UASB + Swimbed + Energy recovery and use	94	79	52
CB2ms	NWW + CCP	Multi-actor network	UASB + Swimbed + Energy recovery and use	85	98	52

approach—i.e. challenges of capturing dimensions of governance or lack of important cost data—there is also clear and compelling messages that should draw the attention of policymakers in and outside of Indonesia.

These messages begin with the claim that NDCs and wastewater regulatory standards currently can serve as a starting point for mitigating climate change and improving water quality. However, a critical finding is that it is not simply strengthening climate and wastewater management policies and measures but aligning policymaking institutions and decision making processes. A related finding is that strengthening government capacity without coordination can lead to the more stringent enforcement of treatment measures focused solely on COD removal that can surpisingly increase GHG emissions. Therefore, decision-makers need to consider governance reforms that consistently deliver policies and measures that exploit the maximum COD removal efficiency as well as the maximum GHG mitigation potential.

Another policy-relevant finding is that it is possible to quantify the benefits that could potentially derive from enhancing governance across levels and actors. The analysis shows that maximum co-benefits would likely result from multi-level, multi-stakeholder forms of governance that is inclined to support the Up-flow Anaerobic Sludge Banked, including gas recovery and use with a Swimbed technology (Scenario $CB2_{ms}$). The substitution of electricity use from external sources due to the recovery and use of own-biogas is one of the main advantages of adopting $CB2_{ms}$. The implementation of CB2 will provide 52% of the electricity required for wastewater treatment, thus, replacing fossil fuels and reducing GHG emissions from electricity consumption. However, the success of $CB2_{ms}$ fully depends on the cooperation and coordination at all levels of governance.

The article also points to a potentially fruitful area of research that could help strengthen that cooperation: that is, more systematically accounting for potentially desirable features of governance. There has been notable headway in assessing some of the key properties and characteristics of good governance across countries that could be useful in this regard. There is also important research that draws on multi-criteria analysis to benchmark the quality of governance in cities (including work on aspects such as performance and efficiency) in cities such as Lisbon. As this work notes, one of the main benefits of this approach is the participatory process of benchmarking performance actually motivates stakeholders to improve performance and efficiency (da Cruz and Marques, 2014; Marques et al., 2015).

A final avenue for future research could focus on exploring strategies that more explicitly integrate the circular economy and co-benefits framework as one of the instruments to reach the national sustainable development goals. Strategies that could be investigated include water and energy use efficiency and material – water recycling. Furthermore, an analysis on the application of these technologies to the whole food industry could shed light on additional water pollution and climate benefits.

CRediT authorship contribution statement

Adriana Gómez-Sanabria: Conceptualization, Investigation, Methodology, Writing - review & editing. **Eric Zusman:** Conceptualization, Writing - review & editing. **Lena Höglund-Isaksson:** Conceptualization, Investigation, Methodology, Writing - review & editing. **Zbigniew Klimont:** Conceptualization, Writing - review & editing. **So-Young Lee:** Investigation, Writing - review & editing. **Kaoru Akahoshi:** Investigation, Writing - review & editing. **Hooman Farzaneh:** Investigation, Writing - review & editing. **Chairunnisa:** Investigation, Writing - review & editing.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

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Potentials for future reductions of global GHG and air pollutant emissions from circular municipal waste management systems

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Article

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Potentials for future reductions of global GHG and air pollutant emissions from circular municipal waste management systems

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Contributions

AGS designed the study, performed the projections, emission simulations and analysis, and prepared the manuscript. GK performed the ambient air pollution concentration calculations. ZK provided expert guidance and contributed to the revision of the manuscript. WS prepared and imported the IAE-WEO activity drivers and provided methodological advice. HH participated in the development of the research and contributed to writing the manuscript. All authors were involved in the discussion during the process.

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Abstract

Recent trajectories of production and consumption patterns have resulted in massively rising quantities of municipal solid waste (MSW). In combination with the large global quantities of mismanaged MSW these increases cause detrimental effects on the environment and climate. Few analyses of the potential environmental co-benefits resulting from the implementation of circular MSW management systems exist. To our knowledge, no global study of possible future scenarios of MSW generation, composition, management, and associated burdens is available that explicitly considers the important differences between urban and rural settings. To help filling this gap, we here develop a systematic approach for evaluating the benefits of implementing circular MSW management systems in terms of their potentials to reduce greenhouse gas emissions (GHG) and air pollution. We also analyse their role in the pursuit of the Sustainable Development Goals (SDGs). Building on the Shared Socioeconomic Pathways (SSPs), we build two sets of global scenarios until 2050, namely baseline and mitigation scenarios. In these scenarios, we assess trajectories of future MSW generation and the impact of MSW management strategies on methane (CH_4), carbon dioxide (CO_2) and air pollutant emissions. We estimate that future MSW generation could increase to at least 3.7 Gt/yr and at most to 4.3 Gt/yr by 2050, depending on the respective SSP storyline. In 2050, we show that the adoption of mitigation strategies in the sustainability-oriented scenario yields earlier, and major, co-benefits compared to scenarios in which inequalities are reduced but that are focused solely on technical solutions. In 2050, the GHG emissions in the sustainability-oriented scenario amount to 182 Gg CO_{2eq}/yr of CH_4, to be released while CO_2, particulate matter, and air pollutants from open burning of MSW can be virtually eliminated, indicating that this source of ambient air pollution can be entirely eradicated before 2050. We conclude that significant potentials exist to reduce GHG, and air pollution if circular MSW management systems are implemented. We also demonstrate that the 6.3 target of the SDG 6 can only be achieved through more ambitious sustainability-oriented scenarios that limit MSW generation and improve management.

Key words: Municipal waste, greenhouse gases, air pollution, methane, SDGs

Introduction

Global quantities of municipal solid waste (MSW) generation have grown massively over the last decades, not only due to population growth but also as a result of economic growth and the consequent changes in production and consumption patterns[1,2]. Estimates suggest that the world population generated 1.9 Gt/yr of MSW in 2015 and is expected to generate about 3.5 Gt/yr of MSW in 2050[3]. High-income countries (in terms of the World Bank income classification) generate more waste per capita per year than low-income countries: they are responsible for 34% of the amount of MSW generated each year, even though they account for just 16% of the global population[4]. These large quantities of MSW generated each year necessitate the implementation of appropriate management systems if the additional associated environmental and health impacts should be avoided that would emerge in the absence of suitable treatment facilities[5]. High-income countries can deploy policies and instruments to cope with the rising MSW flows and hence have cleaner and better-organized waste management systems. Examples include the EU Waste Framework Directive 2008/98/EC[6], the 3R's strategy in Japan [7] and the Resource Conservation and Recovery Act 1976[8], 1986 in the United States. However, high-income countries are still mostly not successful in reducing the amount of MSW generated each year[9]. By contrast, low-income countries often lack suitable management systems, which results from the shortage of funds, poor planning, poor implementation of law and lack of technology and expertise [4,10,11]. Additionally, the outsourcing of resource-intensive production and waste exports from high-income to low-income countries exacerbates the environmental problems resulting from inadequate waste management in many of these countries[12]. Often, open burning, littering and poorly managed landfills are the main ways of waste disposal in low-income countries[4]. Open waste burning results in the release of toxic pollutants, e.g., particulate matter (PM), black carbon (BC), organic carbon (OC), carbon oxide (CO), sulphur dioxide (SO_2), among others, and greenhouse gases (GHG) including carbon dioxide (CO_2) as well as smaller amounts of methane (CH_4)[13–15]. Litter harms wildlife and ecosystems, especially marine life. Global marine litter is currently recognized as one of the biggest sources of ocean's pollution[16,17]. Decomposition of organic matter in landfills can result in the release of CH_4[18], a greenhouse gas that is 28 times more potent per kg emitted than CO_2 in a 100 year timeframe[19], and is also a precursor of tropospheric ozone which alters background ozone concentration and therefore impacts human health[20–22]. In addition to the negative impacts on the environment and climate, these unsustainable practices have well documented adverse effects on human health[23–25]. BC and OC, which are components of $PM_{2.5}$, are associated with pulmonary disease, heart disease and acute lower respiratory infection[26–29]. While reducing air pollution has positive health effects, the impact on the climate system is more difficult to assess. Given the complex interaction between air pollutants and GHGs in the atmosphere, polices that aim at reducing both air pollution and GHG emissions at the same time may succeed to reduce some GHG emission at the expense of reducing cooling effects from specific pollutants such as BC[30].

In the past years, research on waste has gone beyond disposal of wastes to assess the linkages between waste and resource use, climate change, air and water pollution. In that context, various studies have looked at emissions from landfills when assessing sectoral and regional contributions to GHG emissions and abatement potentials[31–34]. Further assessments include the annual National Inventory Submissions of all Parties included in the Annex I of the Convention to the UNFCCC which comprise all reporting on GHG emissions and removals[1]. Current estimates are that landfills contribute about 15% to global anthropogenic CH_4 emissions[31]. Other studies show that open burning of MSW is an important contributor to particulate matter and air pollutant emissions[14,35,36], specifically, it contributes 11% to total global $PM_{2.5}$ emissions and 6-7% to total global BC emissions [35,36]. BC from open burning of waste amounts to 2-10% of global CO_{2eq} emissions[37].

However, very few studies comprehensively assess and model MSW at the global scale. A recent study estimates the global trends and environmental impacts of MSW up to 2100[3] in terms of MSW generation, composition, and treatment, as well as environmental impacts. Other studies look at MSW as a potential source of secondary materials and energy. It is estimated that the relative contribution of energy from waste and wastewater to the global primary energy demand could increase from 2% to 9% by 2040 and deliver 64

[1] https://unfccc.int/ghg-inventories-annex-i-parties/2020

EJ of energy per year (1 EJ = 10^{18} Joules) at the end of this period, if circular management systems are installed[38]. Current estimates are that only around 13% of the global MSW generated is recycled and 5.5% composted[4]. In a trend scenario perpetuating current conditions, this share is expected to increase to 39% in 2050 (includes composting and incineration)[3]. Recycling of waste, including composting and anaerobic digestion, can potentially be boosted in a sustainability-oriented scenario, but so far the extent to which that could be achieved has not been quantified.

Clearly, these assessments provide some insights on the contribution of MSW to GHG and air pollutants emissions as well as a source of energy and secondary materials. However, most of them focus on a single aspect of MSW (i.e., emissions from landfills and open burning) rather than on the MSW management system as such. Studies providing evidence of the potential environmental co-benefits resulting from the implementation of circular MSW management systems are still scare. Furthermore, to our knowledge, no global analysis exists that considers differences between urban and rural settings and assesses how MSW generation, composition, management and associated environmental burdens might change under alternative, plausible future scenarios. We here fill that gap. Our main motivation is to contribute to improved understanding how different societal choices could transform MSW management practices in order to address global climate, pollution, and sustainability issues. To our knowledge, this is the first global study to show how the Shared Socioeconomic Pathways (SSPs) can be translated into emission baselines (CLE) and mitigation scenarios (MFR) for the MSW sector.

We present a new method to globally assess the current and future MSW generation in urban and rural areas and associated emissions as well as their implications for ambient $PM_{2.5}$ concentrations for a range of future population and macroeconomic developments to 2050 using the GAINS model as framework. These are represented by the five SSPs and a scenario consistent with the future macroeconomic and population pathways of the IEA's World Economic Outlook 2018[39]. Two variant scenarios are developed for each of the six future socioeconomic pathways; a 'Baseline - CLE' and a 'Maximum Technically Feasible Reduction – MFR', in which circular municipal waste management systems are implemented globally. This means that landfilling of MSW is restrained, material recycling rates are increased, technological improvements and behavioral measures such as reduction of food and plastic waste generation are assumed to be implemented. Emissions of CH_4, CO_2 (fossil fraction), PM2.5, BC, OC, CO, SO_2, NOx, and NMVOCs are calculated for 184 countries/regions (differentiating urban and rural areas) for the period 2010 – 2050. Results are presented at the level of thirteen world regions and the global aggregate. Based on this comprehensive analysis, we quantify the potential reduction of GHG emissions as well as particulate matter and air pollution through circular MSW systems. We also assess which SDGs can be reached or will be failed under the different scenario assumptions. Our detailed representation of the MSW sector and associated emissions and mitigation potentials can be used as input to Integrated Assessments Models (IAMs) applied to develop emission scenarios for the IPCC, support regional and local scale air pollution studies, and inform local and national governments about the likely developments, environmental consequences, and mitigation opportunities in the MSW sector.

Results

Scenarios of MSW generation until 2050

Different socioeconomic assumptions underlying each of the SSPs lead to significant differences in future MSW flows (Fig. 1). The lowest quantities of MSW generation in 2050 are expected in SSP3 and SSP4 due to slow economic growth and inequalities between regions which is reflected in different consumption patterns. By contrast, in the SSP5 both income and urbanization rates increase strongly, resulting in a growth of the MSW generation quantities estimated at to 4296 Tg/yr. Interestingly, in a sustainability-oriented scenario (SSP1) MSW generation is expected to be just 10% lower than that in the SSP5 by 2050. However, when boosting the SSP1 with the adoption of measures targeted at reducing food and plastic waste (SSP1_MFR), it will be possible to reduce MSW generation by an additional 20% compared to SSP5 quantities by 2050.

The amount of MSW generated, its composition as well as prevalent management systems and policies strongly depend on the dynamics of population and economic activity. We parameterized the drivers of MSW as follows: the most important driver of future MSW generation is GDP. Separate elasticities that relate MSW/cap/yr to GDP/cap/yr are estimated for groups of countries representing four different average income levels under the assumption that MSW generation and its composition are highly dependent on average national income levels. The future composition of MSW is recalculated based on the estimated income elasticity of per-capita food waste generation to GDP/cap/yr. MSW composition fractions estimated separately include food, paper, plastic, glass, metal, wood, textile, and other mixed waste.

Quantities and composition of MSW generated differ between rural and urban populations. Data on rural waste generation are available for a limited number of countries. For countries where data on rural MSW generation are unavailable, rural waste generation is estimated by applying ratios of urban:rural MSW generation per capita for each region that were deriving from the available information for limited number of countries (see Methods). While the uncertainty of the estimate might be high, the split into urban and rural MSW quantities highlights where actions are needed to improve MSW management systems at local levels, allowing for better quantification of impacts and consequently serves better for policy design. Our estimates suggest that urban areas are currently responsible for 70% of the global MSW generated. In 2050 urban areas are expected to generated 80% of the total MSW while rural areas are expected generated the remaining 20%, i.e., MSW per capita in rural areas is expected to be 50% lower than in urban areas. In general, rural per capita MSW generation is much lower than those in urban areas due to their smaller purchasing power. However, in high-income countries these differences between urban and rural areas shrink over time.

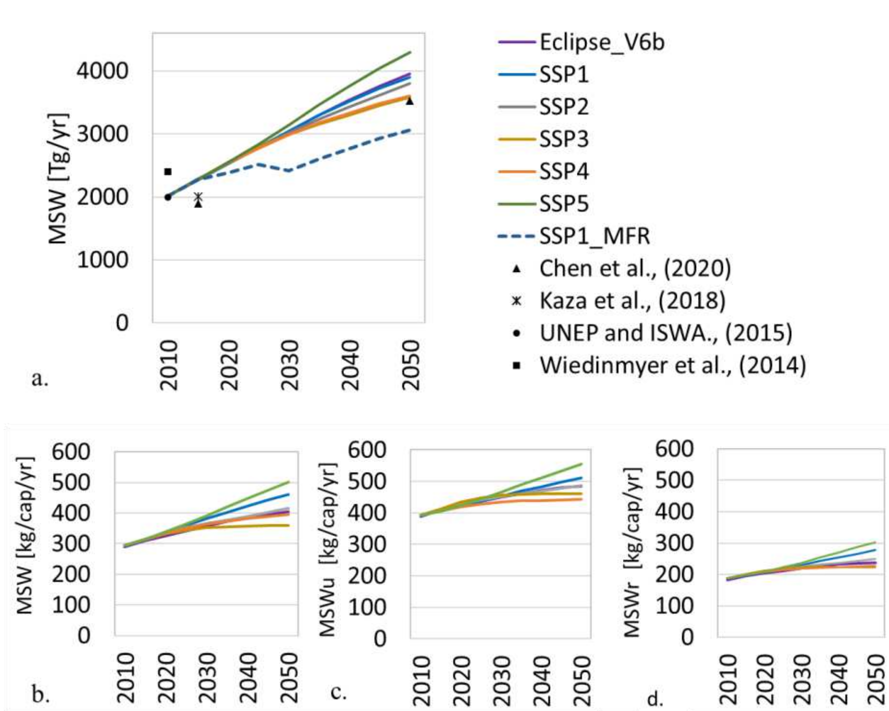

Fig. 1: a. Global total MSW generation. b. Global MSW generation per capita. c. Global urban MSW generation per capita. d. Global rural MSW generation per capita

North America (NAM) is likely to continue having the highest average per capita MSW generation in both urban and rural areas by 2050, followed by Oceania and Europe. China is expected to have the highest growth in MSW generation per capita for urban and rural areas increasing by about 45% compared to 2015. The reason is the stronger economic growth expected in China over the next decade [41]. India is expected to generate about 13% less MSW than China in 2050 across all scenarios. Even though South Asia (SASIA) and Latin America and Caribbean (LCAM) had similar average per capita MSW generation for both urban and rural areas in 2015, per capita MSW generation in Asia is expected to overtake LCAM in 2050 by about 15%. Even though Africa will experience the highest increase on MSW generation compared to 2015, it is likely to continue having the lowest MSW generation per capita in the future (Fig. 2). Supplementary Results. S1 displays total, urban, and rural waste generation by region and scenario.

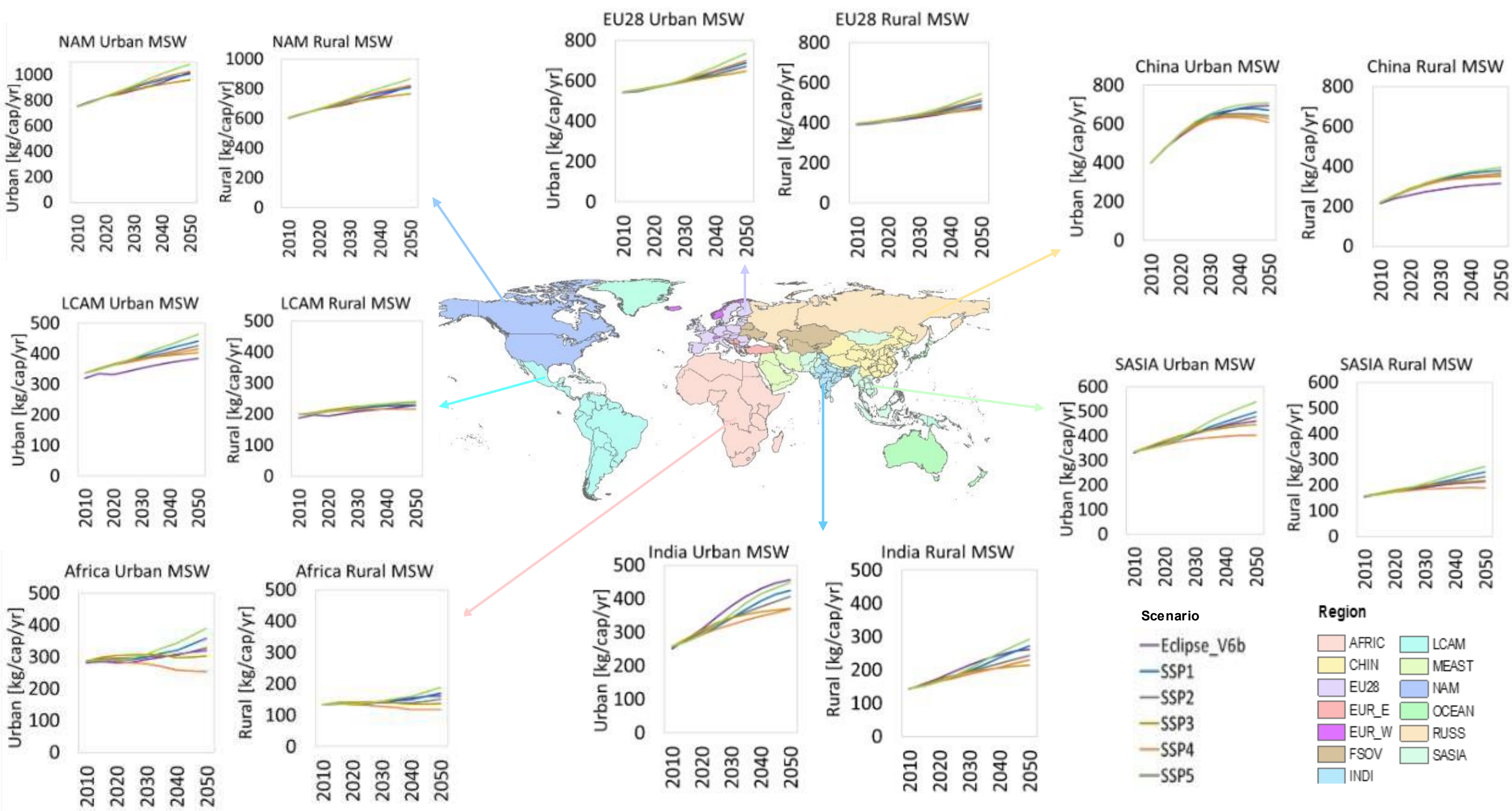

Fig. 2: Municipal solid waste (MSW) generation rates in urban and rural areas by scenario. For high-income regions as NAM and EU28, MSW per capita will remain pretty the same independent of the underlying socio-economic pathway. However, the different pathway trajectories have a strong influence on MSW per capita generation in low, and middle-income regions.

Unfortunately, regions generating the highest amounts of MSW quantities per year have the lowest collection rates and the poorest MSW management systems. Average MSW collection rates in Africa, India, SASIA, and China are estimated to be in average of about 50% - 60%, having urban areas collection rates of ~70% and rural areas ~40%. Moreover, the unsuitable management (i.e., disposed in dumpsites or burned without air pollution controls), of the collected fraction exacerbates the already precarious situation. Based on the detailed MSW activity and management strategies matrix of the GAINS model which comprises eight MSW streams and fourteen treatment technologies [38], our estimates suggest that in 2015, 43% of the global MSW collected ended up either in landfills (13%) that are compacted and/or covered but not meeting environmental standards to prevent leakage[42], in unmanaged landfills without any type of management (hereafter referred as dumpsites) (21%), or was openly burned (9%) either directly at the dumpsites (including unintended fires) or in transfer stations. The remaining 29% of the collected waste was either disposed in sanitary landfills (10%), incinerated (high quality with air pollution controls and energy recovery) (7%), recycled (7%), or composted or anaerobically digested (4%), which is mostly happening in high-income countries. From the uncollected fraction, around 20% is estimated to be scattered MSW with a high probability of eventually reaching water courses and 10% openly burned (Fig. 3). The latter estimates are based on global assessments and detailed country-level studies presented in Table 1 in the methods section.

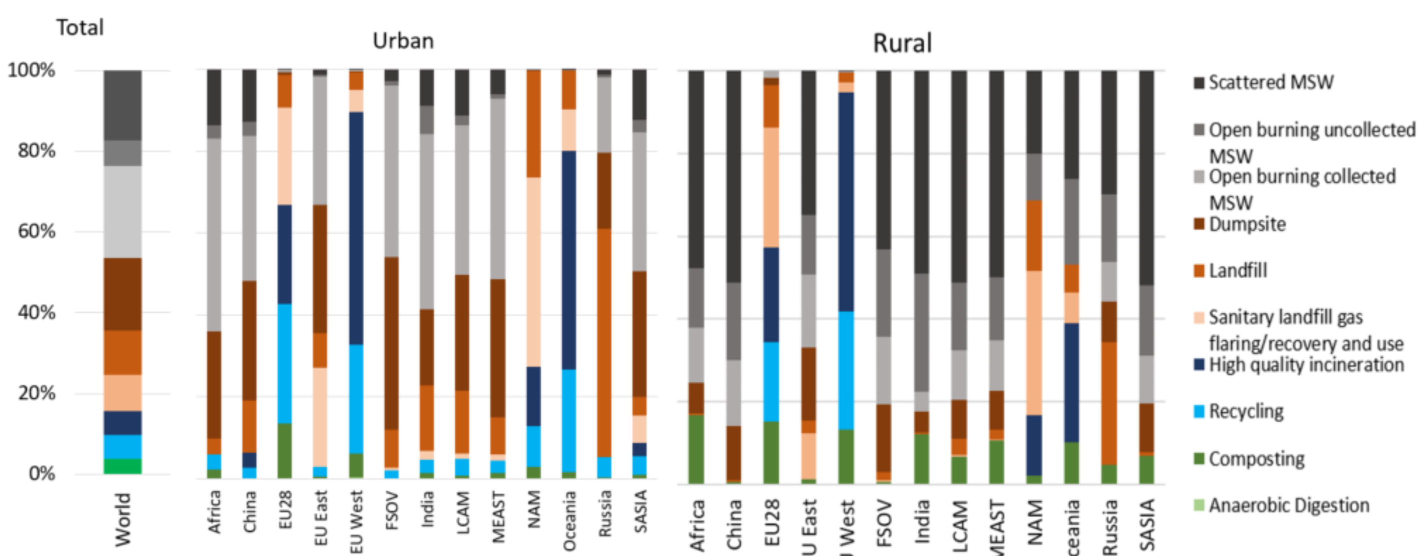

Fig. 3: Municipal solid waste (MSW) management in 2015. Urban areas in low-middle income regions have increased MSW collection rates in last years. However, MSW treatment has not improved at the same pace, hence most of the waste is dumped, scattered or is subject to open burning. Rural areas face an even more challenging situation as in low-middle income regions collection rates are just about 35% - 45%. In general, high-income regions have established suitable MSW treatment systems in both urban and rural areas.

Despite legislation banning open burning of MSW in most of the countries, our calculations indicate that around 16 % of global MSW generated (whereof 55% collected and 45% uncollected), was openly burned, which is equivalent to 380 Tg/yr and 394 Tg/yr in 2010 and 2015, respectively. While in urban areas about 60% occurs either on transfer stations or dumpsites i.e., in the collected fraction, in rural areas is estimated that about 80% of the burning occurs in the uncollected fraction. Rural areas often lack appropriate MSW management systems and therefore the uncollected waste is usually subject to be dumped, scattered or openly burned[43].

If current MSW management strategies are maintained into the future, the expected quantities of MSW disposed of in dumpsites and openly burned would rise proportionally to the increase of MSW quantities. In contrast, in an ideal situation where a circular MSW management system (MFR), is implemented globally, it would be probable to avoid almost all dumping and open burning of MSW in 2050, thereby eliminating the environmental and health burdens associated with current management practices. Circular MSW management systems include restrained landfilling of MSW, increase material recycling rates, technological improvement, and implementation of behavioral measures such as reduction of food and plastic waste generation.

Emissions to air

Our estimates indicate that current CH_4 emissions from MSW handling account for 8 % (28 Tg/yr) of the global CH_4 anthropogenic emissions estimated at 344 Tg/yr in 2015[31]. Under the current management strategies, baseline CH_4 emissions in 2050 are projected to rise by a factor between 1.7 (SSP3_CLE) and 2 (SSP5_CLE) over the amount observed in 2015, increasing the contribution of MSW to 13% of the projected global CH_4 anthropogenic emissions estimated at 450 Tg/yr in 2050[31]. At the regional level, China, NAM, LCAM, and SASIA emitted the higher CH_4 from MSW in 2015. If current conditions are maintained until 2050, then India, Middle East, Africa and SASIA will face the highest growth in CH_4 emissions from MSW, with an increase of about 60% compared to 2015 levels. The expected rise of the CH_4 emissions on those regions is due to the increase of MSW generated, couple with the MSW (mis)management as scattered MSW, dumpsites and precarious landfills (cover or compacted without leakage controls or gas recovery) are the main options to deal with the MSW generated thereby increasing CH_4 emissions.

CH_4 emissions from waste deposited of in landfills today will be generated in future years as it depends on the degradability of the organic matter[18]. MSW generation quantities, composition and policy adoption at early stages makes a significant difference in the trends of CH_4 emissions through the years. In a world implementing circular MSW management systems, the maximum diversion of MSW from dumpsites by 2030 is reached in SSP1_MFR with 91% less compared to the baseline. This is the result of the adoption of MSW reduction measures, speedy implementation of anaerobic digestion to treat organic waste and the establishment of source separated MSW collection systems to increase the recycling of materials. Total elimination of this practice is expected to happen around 2035 in this sustainability-oriented scenario. The adoption of measures is comparatively slower in scenarios depicting high inequalities between and within countries. Therefore, the diversion of MSW from dumpsites takes more time resulting in higher future CH_4 emissions. With the exception of SSP1_MFR in which CH_4 emissions are projected to decrease by 4% in 2030, an increase of about 1%-2% is expected to happen in all other MFR scenarios compared to the corresponding CLE. The maximum CH_4 emission reduction potential by 2050 will be reached in the SSP1_MFR in which CH_4 emissions are expected to decrease by 87% compared to the baseline, thus leaving still 182 CO2eq of CH_4 to be released in 2050. Other scenarios are expected to release more CH_4, namely, SSP3_MFR will leave 646 CO2eq of CH_4 and SSP5_MFR 292 CO2eq of CH_4 to be emitted by 2050 which is 50% and 80% lower compared to the respective CLE counterparts (Fig. 4).

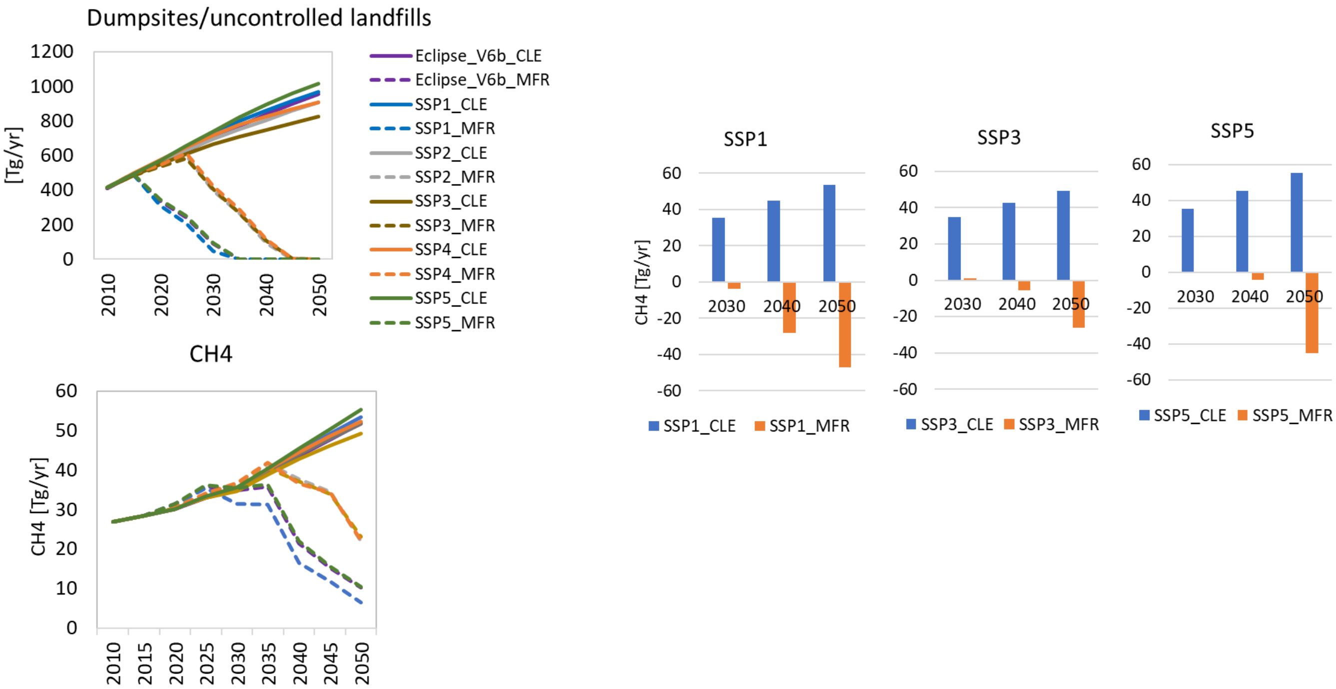

Fig. 4: Global CH_4 emissions under CLE and MFR scenarios. Faster adoption of measures improving MSW systems will result in an early decrease of MSW ending up in dumpsites/uncontrolled landfills and therefore brings quicker reductions of future CH_4 emissions from this source. Supplementary Results S2 presents a detailed analysis of the MFR scenarios.

Emissions of particulate matter and air pollutants depend on the quantities of MSW subject to open burning. Our results suggest that open burning of MSW is responsible for 3.5 Tg/yr of $PM_{2.5}$ in 2015. BC emissions are estimated to be 7% and OC 60% of the $PM_{2.5}$ emissions. Overall, $PM_{2.5}$ emissions from MSW account for 8% of the total global anthropogenic $PM_{2.5}$ emissions. Global anthropogenic BC emissions are estimated at 6.0 Tg/yr (GAINS) of which , following our results, 6% are from MSW burning (see supplement Table S3 for estimates for all pollutants). At the regional level, our calculations indicate that SASIA plus India, China, Africa, and LCAM emitted 89% of the particulate matter and air pollutants from MSW. India and China contributed about 50% and Africa 21% and LCAM the remaining 18% to those aggregate flows in 2015. Although open burning of MSW occurs in the collected and uncollected fraction in both urban and rural areas, most of emissions come from the collected MSW in urban areas. For example, in Indian cities waste handlers burn waste, despite being aware of the ban, mainly due to lack of infrastructure and to prevent accumulation[44]. Furthermore, with the projected growth of MSW generation and if the current conditions prevail into the future then the anticipated global emissions of particulate matter and air pollutants from MSW are expected to nearly double in 2050 for all SSPs. SASIA, India, Africa, China and LCAM are expected to be responsible for 93% of the emissions. Future emissions in the CLE scenarios will increase proportionally to the quantities of MSW open burned. Consequently, the reduction of the fraction of MSW being openly burned translates directly into the same particulate matter and air pollutants emission reduction levels (Fig. 5). In that sense, in the SSP1_MFR, SSP5_MFR and ECLIPSE_V6b_MFR scenarios will be feasible to virtually eliminate open burning and therefore this source of air pollution already in 2030 while in the other scenarios this could potentially happen 10 to 15 years later.

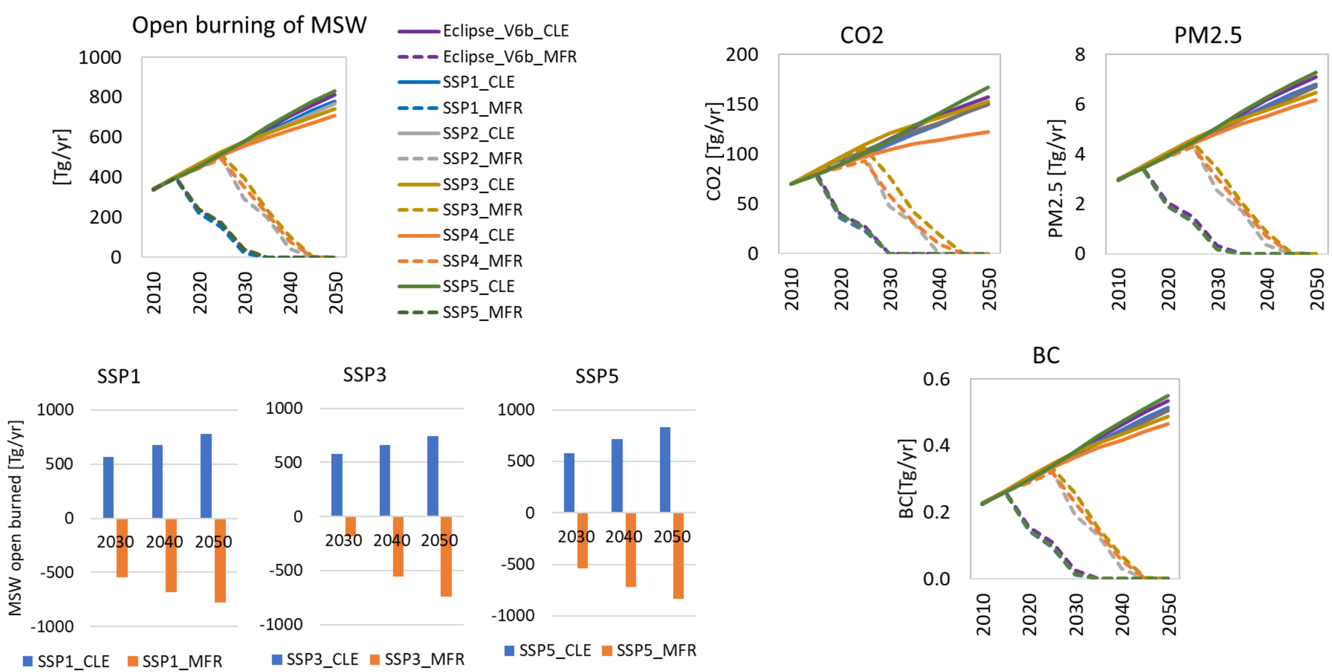

Fig. 5: Global amounts of MSW open burned and related emissions under CLE and MFR scenarios. Reduction fractions of MSW open burned result in the same reduction percentage of particulate matter and air pollutants. Supplementary Results S2 presents a detailed analysis of the MFR scenarios.

At a regional level (Fig. 6), the pre-conditions of the MSW management systems in Europe, Oceania and to certain extent NAM show that the level of effort required to reduce emissions is similar across scenarios. This is the result of the historical evolution on MSW management systems together with the already high-income level and appropriate political arrangements in most of these regions. By contrast, all other regions show high variation across scenarios due to the different dynamics. When comparing the scenarios for regions such as China, India, SASIA, and LCAM, we see that in a sustainability-oriented scenario (SSP1_MFR) a speedier decrease in emissions is observed in urban and rural areas compared to the other scenarios. Moreover, the adoption of circular MSW management systems is slower in scenarios representing a world in which inequalities persist resulting in big differences between urban and rural areas. Consequently, higher emissions are expected across the years.

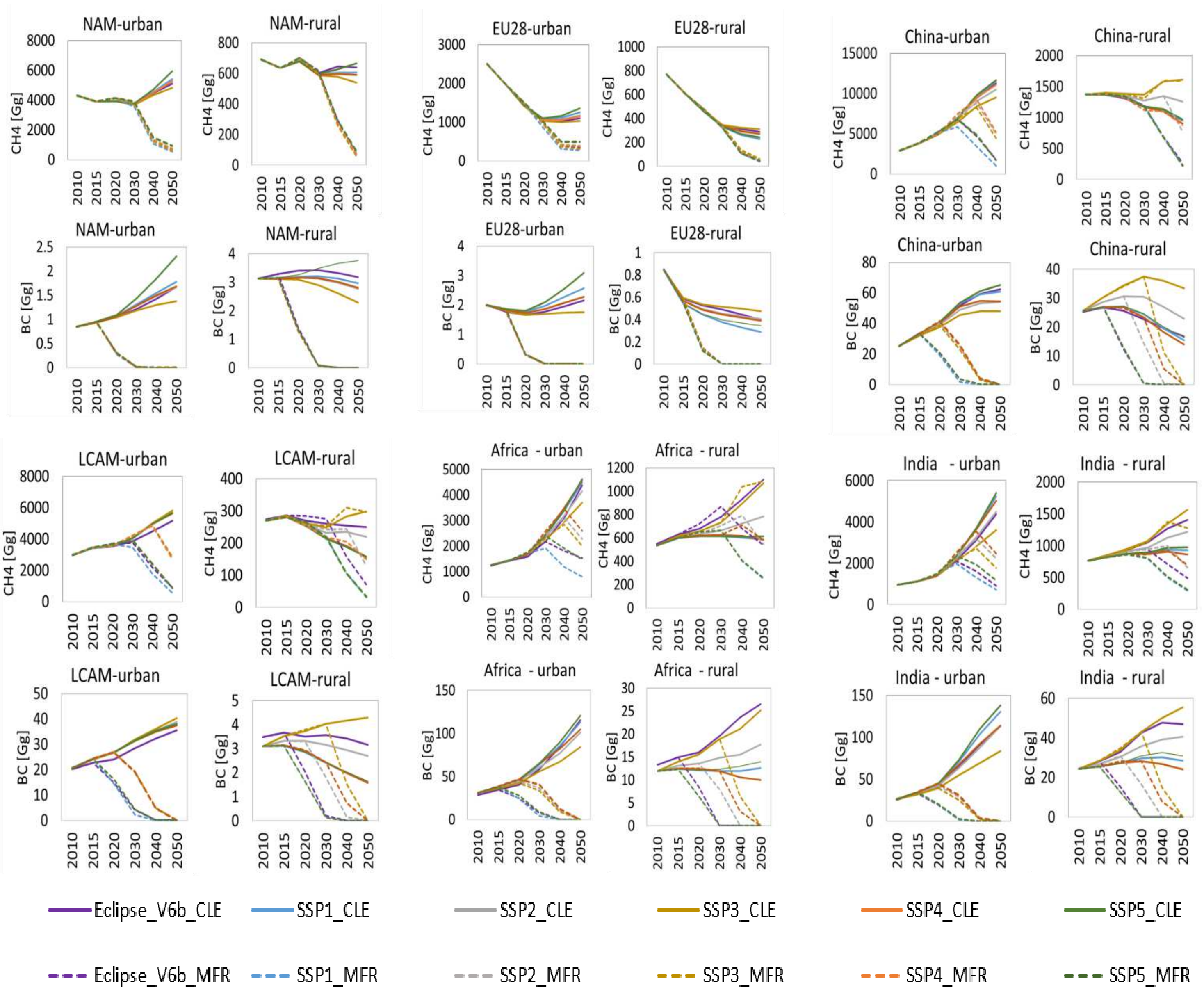

Fig. 6: Regional emissions of CH_4 and BC from MSW. The target of all modelled scenarios is set to reach ~100 % of MSW collection and management by 2050. The environmental co-benefits will be obtained at different levels upon the level of socio-economic development and political and institutional arrangements. The different assumptions on policy interventions are then translated into a wide range of future emissions.

As emissions from MSW burning contribute significantly to ambient $PM_{2.5}$, particularly since the sources are often low-level and spatially located close to population, the improvement of MSW management will also have benefits in ambient $PM_{2.5}$. To illustrate the possible contributions and mitigation potential from this sector, we here quantify the contribution of MSW to $PM_{2.5}$ levels in different world regions. Calculations follow the approach applied in ref[45] and are briefly described in the Methods section below. Differences between the scenarios are driven both by emission changes as well as urbanization trends. Concentrations are highest in India and other South Asia and are expected to grow further under CLE following the emission trends. Other developing regions show similar growth trends but lower absolute concentrations. In China, initial increases level off, peaking around 2035 (SSP1,2,3,4) or 2050 (SSP5). In Europe, North America and Oceania, contributions from MSW burning are much lower since the combustion happens in well-controlled installations and not as open burning. Gradual implementation of better practices and emission controls eventually decreases concentrations to ~zero before 2050 in all MFR cases, although this is achievable at different points in time depending on the SSP storyline.

Discussion

Here we present for the first time a systemic assessment of reduction potentials of GHGs and air pollutants emissions from implementing circular MSW management systems under six future socio-economic development pathways. The assessment includes the development of two scenarios, namely baseline (CLE) and maximum feasible mitigation potential (MFR) for each of the pathways. The explicit representation of urban and rural MSW generation, composition and management allows for a deeper analysis of future plausible management and emission trends. This study can assist national, regional, and local governments in developing strategies to limit the release of emissions into the environment as well as support assessments of feasibility and progress in achieving the UN Sustainable Development Goals (SDGs).

Our results show that future MSW generation quantities are expected to be between 1.7 to 2 times higher in 2050 compared to current levels in all scenarios. Our results also highlight that urban areas are responsible for about 80% and will continue being responsible for the higher share of MSW generated in the future. The generally high collection rates of MSW in urban areas does not necessarily imply appropriate management. In SASIA, India, China, LCAM and Africa about 80% of the collected MSW is either dumped or openly burned. Furthermore, most of the MSW generated in rural areas is uncollected and thus ends up being illegally dumped, scattered, or openly burned resulting in several environmental impacts related to air pollution and greenhouse gas emissions and other health and environmental impacts out of the scope of this study. Our findings also indicate that in urban areas about 60% of the open burning occurs either on transfer stations or dumpsites i.e., in the collected fraction, while in rural areas is estimated that about 80% of the burning occurs in the uncollected fraction.

In the baseline (CLE), in which current MSW management practices persist without further policy implementation, emissions to air would increase proportionately to the growth in MSW generation. We then developed a set of mitigation scenarios (MFR) to assess the impacts of abatement measures compared to the corresponding baseline (CLE). The common target of our MFR scenarios is to achieve ~100% of MSW collection and treatment by 2050 through the implementation of circular MSW management systems to simultaneously tackle emissions of CH_4, CO_2, particulate matter, and air pollutants. Co-benefits are obtained at different stages upon the level of socio-economic development and political and institutional arrangements. Evidently, all countries would benefit from reduced MSW generation and improved management in the sustainability-oriented scenario (SSP1_MFR), however, the additional benefit of respective measures are especially relevant for regions generating large MSW quantities and lacking suitable management systems. We show that the environmental co-benefits of avoided MSW generation combined with the speedy implementation of anaerobic digestion to treat organic waste and the establishment of source separated MSW collection to increase the recycling of materials (SSP1_MFR) yields major and earlier co-benefits in terms of reducing CH_4, particulate matter, and air pollutants. However, more ambitious sustainability-oriented scenarios are crucial to meet the waste related SDGs, specially the 6.3 target which aims at "*By 2030, improve water quality by reducing pollution, eliminating dumping and minimizing release of hazardous chemicals and materials, halving the proportion of untreated wastewater and substantially increasing recycling and safe reuse globally*"[46]. We demonstrate that under the current SSP1_MFR, it will not be possible to totally eliminate scattered and open burning of MSW by 2030. Under this scenario the realization of the objective will be obtained five years later i.e., in the year 2035.

Our analysis also suggest that in 2030, 881 Gg CO_2eq of CH_4 (GWP_{100} of 28 CO_2eq[19]) will still be released in the SSP1_CLE. Nonetheless, this is 13% lower compared to the CH_4 emissions expected in the SSP2_CLE, SSP3_CLE and SSP4_CLE and 11% lower in comparison to the SSP5_CLE and Eclipse_V6b_CLE. Considering that in 2030 high emissions of CO_2 from open burning of MSW would still be released in SSP2_MFR, SSP3_MFR, SSP4_MFR, the total average GHG emissions (CH_4, and CO_2) in these scenarios will sum up to an average of about 1079 CO_2eq, that is 18% higher than the emissions expected in the SSP1_MFR. In 2050, SSP1_MFR leaves 182 Gg CO_2eq of CH_4, to be released. That is 37% lower than the SSP5_MFR and Eclipse_V6b_MFR and 3.5 times lower than the expected emissions in the SSP3_MFR. These variation in emissions can make a substantial difference when considering that the world should stay below 1.5 degrees global warming i.e., the world can emit as maximum as 10 Pg CO_{2eq}/yr of all GHGs in 2050[47].

The reduction of MSW being openly burned translates into the same reduction level of emissions of particulate matter and air pollutants. Under the development of SSP1_MFR, SSP5_MFR and ECLIPSE_V6b_MFR, the maximum emission reduction potential will be realized in 2030 whereas in the SSP2_MFR will take 5 years more i.e., in 2040 and for the SSP3_MFR and SSP4_MFR 10 years more i.e., in 2045. At the same time, MSW combustion contributes to ambient $PM_{2.5}$ – in some world regions, this contribution is substantial. Most low-income countries, and particularly those with already high concentrations, show an increasing trend from this source under all SSPs, highlighting the importance of counteracting. The positive message is that mitigation is possible and the MSW contribution to ambient $PM_{2.5}$ can be virtually eliminated by 2050. However, this will not happen by itself.

Comparison to other studies: Our calculations suggest that the world generated 2289 Tg/yr of MSW in 2015. Estimates from other studies vary from 1999[3] to 2010[4] Tg/yr for the same year. Past assessments estimated global MSW generation between 2000[48] to 2400 Tg/yr [14] in 2010. Looking at MSW generation projections, our estimate for the SSP3 and SSP4 in 2050 are similar to the 3539 Tg/yr projected by Chen et al., 2020 (ref [4]). Our calculations suggest that although the SSP1 represents a sustainability-oriented pathway, MSW quantities in the baseline are foreseen to reach 3901 Tg/yr in 2050, which is only 10% lower than the expected MSW amounts in the SSP5. Our projection for MSW generation in the SSP2 is 3801 Tg/yr while ref[3] estimated a MSW generation of about 3500 Tg/yr in 2050 for the same scenario. However, this estimate is more comparable with our SSP3 and SSP4 projection. The ECLIPSE_V6b_CLE (3948 Tg/yr) is comparable to the SSP1. At the regional level, we find that India is expected to generate about 13% less MSW than China in 2050 across all scenarios. This contrasts findings ref [4], in which projected MSW generation in India was about 40% higher than the projection for China in 2050. However, our finding for India is in line with the projection carried out by ref [49]. Furthermore, the average per capita MSW generation in China is projected to be between 30% - 40% higher than those in India. The fact that estimates for 2010 are lower than those in 2015 and the variability of the results reflect on the one hand, the uncertainty of the data and on the other hand the differences of the methodologies used to derive these numbers. Furthermore, Our estimate of MSW openly burned is 61% lower than the estimate of ref[14], who estimated that 40% or an equivalent of 970 Tg/yr of total MSW generated in 2010 was openly burned (whereof 64% at residential sites and 36% at unmanaged dumpsites) and 57% higher than the estimate of ref[36], who estimated that about 115 Tg/yr– 160 Tg/yr of MSW was openly burned in 2010. Differences in estimated quantities can be attributed to variations in the per capita MSW generation rates adopted referring partly to different data sources, but also to differences in the methodology used to estimate the fraction of waste openly burned. While the assumption in ref[14] refers to a fraction recommended in the IPCC (2006) guidelines, we develop our own method which we believe better represents the complexity of the MSW sector e.g., in terms of the urban-rural split and the country/region-specific MSW composition and MSW management pathways (see Methods). The differences of the estimates puts a magnifying glass on the urgency to develop national standardized MSW reporting systems, which in addition of being key to governments for the implementation and evaluation of MSW treatment, can serve as part of the monitoring system of GHGs, air pollution and SDGs.

Our estimations indicate that current CH_4 emissions from MSW handling account for 8 % (28 Tg) of the global CH_4 anthropogenic emissions estimated at 344 Tg in 2015[31]. Our estimate is 17% lower than the one estimated by ref[35] and which has been adopted within the CMIP6 project [50]. It is difficult to assess the level of agreement between both studies as estimates from ref[35] include MSW and industrial waste while the focus of this study is on MSW and the importance to properly represent the sector for climate and air pollution assessments. However, comparing CH_4 emissions from MSW in the Eclipse_V5a [36] to this study, we can see that the estimate in the latter is 30 Tg /yr or 6% higher.

Recent global CO_2 emissions area assessed at of 39153 Tg/yr in 2015, whereof 130 Tg/yr or 0.33% are generated from waste combustion (including industrial and municipal sources) [35,51]. Ref[14] calculates CO_2 emissions from open burning of MSW of 1413 Tg/yr in 2010, estimate that is around 10 to 15 times higher than that from ref[35,51] and the one from this study.

In 2010, emissions of $PM_{2.5}$, BC, and OC have been assessed at 6.1, 0.6 and 5.1 Tg, respectively[14]. Our estimates are comparatively lower to those results. In contrast, our results for particulate matter are 60% higher than those from ref [36]. In both cases the differences are related to the assumed quantities of MSW openly burned. Other studies[35,51] have estimated BC and OC emissions from waste of 0.7 Tg and 4.2 Tg [35], respectively (Supplementary Results S3 show a comparison of different studies for different pollutants).

Conclusions

Significant potentials exist to reduce GHG, and air pollution provided the implementation of circular MSW management systems. The 6.3 target of the SDG 6 can only be achieved through more ambitious sustainability-oriented scenarios that limit MSW generation and improve management. Similarly, these kinds of scenarios can directly contribute to the achievement of other SDGs, especially SDG 7, 9, 12, 14 and

15. Our results highlight the importance of acting at various fronts, namely, consumers behavior, technological development, technology transfer and institutional coordination. For instance, the benefits from reduction of MSW generation can be jeopardized by social and economic inequalities between and within regions which could restrain the adoption and implementation of measures to improve MSW management systems. Furthermore, for a world focused solely on end-of-pipe solutions will be also beneficial the implementation of policies targeted at reducing MSW generation. The finding is that the development of measures at the consumer side will not bring the expected benefits in terms of emissions reduction if quicker and responsible actions are not taken to bring MSW management systems as an important point in governmental agendas. Finally, we see that the majority of countries have developed some kind of legislation regarding the improvement of municipal solid waste management systems, however, the compliance is highly uncertain. A solid system for the reporting of MSW couple with a transparent systematic follow-up of policy enforcement will help to reduce the uncertainty of the estimates as well as will provide clearer insights into the efforts needed by countries to meet their climate, air pollution and SDGs commitments.

Methods

The methodology for developing MSW generation scenarios and associated greenhouse gas and air pollutant emissions involves the following five elements: 1. Socioeconomic drivers are taken from the Shared Socioeconomic Pathway (SSP) Scenarios for the five SSPs[52] and from the World Energy Outlook and UNDESA[53] for the Eclipse_V6b_CLE (Supplementary Methods S4 presents a short description of the SSPs storylines). 2. The country-specific generation in per capita MSW is driven by expected growth in average per capita income as described in the Supplement of ref [38] and further developed in this study (Supplementary Methods Fig. S2 and Fig.S3 show GDP per capita and urbanization rates) . 3. Estimation of emissions draw on the methodologies presented in ref [33,36,54], but are extended to improve source-sector resolution and accommodate for new, MSW sector-specific, information. 4. Implementation of the current legislation for waste management adopted before 2018. 5. Implementation of circular waste management systems are developed in accordance with the EU's waste management hierarchy - Directive 2008/98/EC[6]. The IIASA-GAINS model is used as a framework to carry out this assessment.

Municipal waste generation (MSW) activity and its characteristics.

Current MSW generation quantities, composition, collection rates, and waste management practices are retrieved from several sources, including national official statistics, peer-reviewed literature, and technical reports (see supplement of Gómez-Sanabria et al., 2018). The driver used to project future per capita MSW generation is GDP per capita. This is linked to MSW generation using elasticities estimated following the methodology first developed in ref[33] and further developed in ref [55]. This methodology is further developed in this study (Supplementary Methods S6). Separate elasticities are estimated for groups of countries representing four different average income levels under the assumption that MSW generation and its composition are highly dependent on average national income levels. Furthermore, MSW composition is recalculated based on the estimated income elasticity to per capita food waste generation. MSW composition fractions estimated separately include food, paper, plastic, glass, metal, wood, textile, and other waste. This last fraction includes ordinary mixed waste and may in some cases also include bulk waste.

Quantities and composition of MSW generated by rural and urban population are different. Data on rural waste generation is available for a limited number of countries, when underlying data on rural MSW generation is unavailable, rural waste generation is estimated by applying different shares related to the specific urban MSW generation rate per capita within specific region and using Eq. (1). This approach is likely to be an improved version of the one-half rural-urban waste generation ratio used by some studies [4,56] because it captures the differences between regions (Supplementary Methods S7 presents the adopted rural urban rates for different regions).

$$MSW_u = MSW_t * \left(\frac{P_u}{P_u + \left(R_{(r/u)} * P_r \right)} \right) \quad (1)$$

$$MSW_r = MSW_t - MSW_u$$

where MSW_t is total MSW generated in a country/region, MSW_u and MSW_r are MSW generated in urban and rural areas, respectively, $R_{(r/u)}$ represents rural per capita MSW generation as a fraction of the per capita urban MSW generation, and P_u and d P_r is rural are urban and rural population, respectively.

Open burning of MSW.

In countries without proper implementation of waste legislation, waste mismanagement is aggravated by poor waste separation at the source, low collection rates and low budget allocated to the waste sector [40]. In the absence of reliable waste management systems, dumping and open burning of MSW, either at residential or dumpsites, become the only alternatives to reduce waste- volumes [13,14]. Total MSW openly burned is estimated here as the sum of the fractions of uncollected MSW openly burned and collected MSW openly burned at dumpsites and transfer stations in urban and rural areas. The starting point to derive the quantities of MSW openly burned is the total MSW generated in urban and rural areas. Waste amounts are then split into collected and uncollected waste for urban and rural areas, respectively. Collected waste includes MSW collected by official authorities but also (recyclable) waste collected by the informal sector. Information on collection rates is gathered from sources presented in [55] and complemented from information available in [4,56]. The fraction of uncollected waste is then split into scattered waste or waste openly burned. The fraction of uncollected waste openly burned is assigned based on the information presented in Table 1, considering the current implementation of waste related legislation, income level, collection rates, and urbanization rate of each region. The fraction of collected MSW openly burned is estimated at 10% - 20% of the waste ending up in dumpsites, partly due to self-ignition resulting from poor management and partly due to deliberate burning to reduce waste volumes. In addition, a fraction of the collected waste is assumed to be burned at the transfer station or before reaching the disposal site, which is the case in several developing countries [57]. Fractions of MSW openly burned, either on the streets or at dumpsites and transfer stations, are dependent on the improvement of the MSW management systems and enforcement of the waste and air pollution legislation. Improvement of waste treatment systems results in reduction of the frequency of MSW openly burned [58]. The quantification of these fractions is however highly uncertain. Literature provides a few different methodologies to estimate the amounts of waste openly burned (Table 1). The IPCC (2006)[18] suggests 0.6 as a representative value for the fraction of total available waste to be burned that is actually openly burned. This assumption is used by Wiedinmyer et al., 2014 to estimate GHGs and air pollutants from open burning of waste. Bond et al., (2004)[59] assumed lower rates of open burning of waste in rural areas in developing countries based on the statement that most of the waste in rural areas is biodegradable. Table 1 also shows that in many cases the default representative value of the IPCC maybe inadequate for several regions.

In general, the quantification of MSW openly burned in region i and year y - $MSW_{(ob)iy}$ is calculated as the sum of MSW openly burned in urban areas $MSW_{(obu)}$ and MSW openly burned in rural areas $MSW_{(obr)}$ applying Eq (2).

$$MSW_{(ob)iy} = MSW_{(obu)iy} + MSW_{(obr)iy} \tag{2}$$

Where,

$$MSW_{(obu)iy} = [(MSW_{(u)iy} * C_{(u)iy} * (ß_{0u} + ß_{1u})) + (MSW_{(u)iy} * (1 - C_{(u)iy}) * ß_{2u})]$$

$$MSW_{(obr)iy} = [(MSW_{(r)iy} * C_{(r)iy} * (ß_{0r} + ß_{1r})) + (MSW_{(r)iy} * (1 - C_{(r)iy}) * ß_{2r})]$$

Where, $MSW_{(u)iy}$ and $MSW_{(r)iy}$ are the total amounts of MSW generated in urban and rural areas, respectively. $C_{(u)iy}$ and $Coll_{(r)iy}$ are the MSW collection rates in urban and rural areas, respectively. $ß_{0u}$ and $ß_{0r}$ represent the fractions of collected MSW openly burned on transfer stations and $ß_{1u}$ and $ß_{1r}$ represent the fractions of collected MSW openly burned at dumpsites in urban and rural areas, respectively. $ß_{2u}$ and $ß_{2r}$ are the fractions of uncollected waste openly burned in urban and rural areas, respectively.

Emission estimations.

Emissions of non-CO_2 greenhouse gases and air pollutants (E) by source (s) and region (i) are calculated in GAINS using Eq (3) [54]:

$$E_{it} = \sum_{sit} A_{is} * ef_{sm} * Appl_{itsm}$$

where A_{is} is the activity data, i.e., the amount of MSW generated before management, ef_{sm} is the emission factor subject to technology m, and $Appl_{itsm}$ is the application rate of the technology m to the activity A_{is}. The GAINS model matrix comprises fourteen different MSW waste management technologies including different types of source separation, recycling and treatment, different types of solid waste disposal sites and different types of incineration technologies and open burning of waste (Supplementary Methods 8). This extensive characterization of alternative treatment flows allows for a detailed representation of the solid waste management system and its emissions at the national/regional level. Emission factors for CH_4 and CO_2 are developed according to the 2006 IPCC Guidelines, Volume 5, Chapter 3 and Chapter 5[18]. PM emission factors are adopted from ref [36]. These are 8.75 for $PM_{2.5}$, 5.27 for OC and 0.65 g/kg for BC. Emission factors for SO_2, NOx and NMVOC are adopted from ref[60] and are consistent with ref [14]. These are 0.5 for SO_2, 3.74 for NOx, and 7.5 g/kg for NMVOC. The $PM_{2.5}$ concentrations are obtained using the annual $PM_{2.5}$ emissions applying a simplified version of the atmospheric calculation in the GAINS model [45]. Those estimates build on a linearized representation of full atmospheric chemistry model simulations. Here, an atmospheric transfer coefficient is developed to related $PM_{2.5}$ emissions to ambient $PM_{2.5}$ concentrations from MSW burning.

Description of the scenarios.

The baseline scenarios associated with the six socio-economic pathways describe the expected developments of municipal solid waste generation and management systems under current legislation 'CLE', hereafter baseline , i.e., assuming no further policies affecting the MSW sector are adopted until 2050. In addition, for each baseline an alternative scenario is constructed, which considers full implementation of circular MSW management systems globally and is referred to as the maximum technically feasible reduction 'MFR' scenario, hereafter mitigation scenario. Note that the technical frontier is explored here without taking account of the cost to implement various waste management strategies.

The MFR scenario is developed according to the SSP narratives and assumes a maximum technically feasible phase-in of a waste management system that is fully consistent with the EU's waste management hierarchy (Directive 2008/98/EC)[6]. This means that a first priority is given to technologies that circulate materials, thereafter to technologies that recover energy, and only as a last resort to well managed landfills. The following maximum recycling potentials of waste streams are applied: 90% of municipal paper and textile waste and 80% of municipal plastic and wood waste can be recycled. It is further assumed that 100% of food waste can be source separated and treated in anaerobic digesters with biogas recovery. These MFR potentials are adopted in consonance with the socioeconomic development for each scenario. Supplementary Methods S9 presents a description of the MFR management narratives specified for each scenario along with the regional aggregation.

Uncertainty

Regarding uncertainty, several data inputs (activity data, emission factors, type of management) go into the estimations and therefore is difficult to do a quantitative uncertainty estimation[3,14]. Historical estimates of MSW generation, collection, management, and related emissions have associated uncertainties resulting from the different definitions of MSW coupled with contradictory reported values for generation and composition. The quality of the data suffers from inconsistencies in the definition of MSW generation across countries [56]. In some cases, amounts reported for MSW generation correspond to the gross quantities of waste collected and in other cases to the MSW quantities left for landfill after quantities separated for treatment have been deducted [61]. In developed countries, in particular in Europe, MSW covers household waste and waste that is similar in nature and composition . In developing countries, data on waste suffers from incomplete characterizations and clear definitions of the fractions and source sectors included in the

MSW are often lacking. These uncertainties are relatively high in developing countries compared to developed countries as in various cases data availability is quite limited in the former case[3]. Additionally, some data reported for generation and collection refers to urban areas rather than national totals [4,40], which makes necessary to adopt assumptions based on dedicate studies for particular regions and expert knowledge to arrive at reasonable national MSW generation rates and attributions to urban and rural waste amounts. These uncertainties become bigger when estimating fractions of MSW openly burned as this information is in most of the cases not attainable. Moving to emission factors, CH_4 emission factors are based on the IPCC Guidelines 2006[18], thereby carry out the uncertainties there described. Emissions factors for air pollutants and particulate matter depend on the composition of waste and burning conditions. Although we adopted the most recognized emission factors in the scientific arena, we acknowledge that large uncertainties are related to the values (uncertainties can be seen in ref[14]). Concerning uncertainty in projections, this is by some means assessed by adopting alternative activity scenarios which allows the comparison of the different estimates and reflect the sensitivities of the proposed measures to input assumptions[63]. In general, there is a global need to improve information on MSW generation rates, treatment and level of policy implementation[3]. Regardless of the uncertainties, we demonstrate the importance of improving global estimates of GHGs and air pollutant emissions from MSW and highlight the considerable role of this sector when assessing the respective mitigation potentials.

Data Availability

The data used for this analysis is available in the Supplementary Information and excel spreadsheet.

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Ethics declarations

The authors declare that they have not conflict of interest.

Supplementary Information

The supplement related to this article is available at

Tables

Table 1. Collection of studies quantifying municipal solid waste (MSW) openly burned.

Source	Scale	Assumption	Results
Sharma et al., 2019	India	Calculation of waste burned at landfills was based on a study in a landfill in Mumbai using average FRP. Fraction open burning of waste 7% - 12%	68 Tg a^{-1} was open burned in India in 2015
Wang et al., 2017	China	In reference to the limited literature, China's averaged proportion of open MSW burning is set to 18.0% at residential and dumpsites and 38.0% at landfills.	The proportion of open burning is estimated from 79.8% in 2000 to 57.0% in 2013
Klimont et al., 2017	Global	IPCC guidelines 2006; CEPMEIP, 2002; EAWAG, 2008; Neurath, 2003. Fraction of open burning of waste is 0.5% - 5% for developed world and 10% -20% for developing world.	Global estimation of MSW openly burned is estimated 115 Tg a^{-1} to 160 Tg a^{-1} in 2010
Wiedinmyer at al., 2014	Global	Follows IPCC guidelines 2006 in which 60% of the total waste available to be burned that is actually burned	970 Tg a^{-1} of waste are globally openly burned. 620 Tg a^{-1} at residential level and 350 Tg a^{-1} at dumpsites.
Hodzic et al., 2012	Mexico City	Assigned percentage of MSW burned according to socioeconomic status. Low and middle-low 60%, mid 30%, mid-high and high 20%. Based on anecdotal evidence with Mexican researchers.	The burned fraction exceeds 4 Gg day^{-1}
Bond et al., 2004	Global	Fraction of burned waste in urban areas base on United Nations Human Settlement Programme, 2000	Worldwide 33 Tg a^{-1}, including 14 Tg a^{-1} in Asia and 5 Tg a^{-1} in Africa

Supplementary Files

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