ARTICLE IN PRESS

Environmental Development xxx (xxxx) xxx

Contents lists available at ScienceDirect

Environmental Development

journal homepage: www.elsevier.com/locate/envdev

Carbon footprint reduction potential of waste management strategies in tourism

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ARTICLE INFO

Keywords:
Tourism
Waste prevention
LCA

ABSTRACT

Tourism is one of the major economic factors contributing to growth and jobs worldwide. The number of international travellers has increased more than 50-fold in the past 70 years. However, the contribution of tourism to (municipal) waste generation is also large and is increasing, accompanied by an increase in some environmental and socio-economic impacts. An average value of 1.67 kg waste is now generated per tourist (Obersteiner et al., 2017). Waste prevention and recycling should therefore be major objectives in tourist waste management by municipal authorities.

Within the EU H2020-funded project “URBANWASTE – Urban Strategies for Waste Management in Tourist Cities”, eco-innovative waste prevention and management strategies were implemented in 10 pilot cities with high levels of tourism, in order to reduce urban waste production and improve municipal waste management. This study examined the potential greenhouse gas (GHG) emissions savings for three selected waste prevention and treatment options: food waste prevention, reductions in single use plastic and increased separate collection and recycling of waste. Benefits were expressed per kg waste prevented or diverted higher up the waste hierarchy and per 1000 tourists.

The measures achieved potential GHG emission savings of between 4 and 189 kg CO2-eq. per 1000 tourists, depending on local conditions such as the existing waste management system. Measures tackling food waste reduction and separate collection had low emissions associated with the measure itself, whereas for assessed measures reducing the use of single use plastics by providing reusable alternatives, emissions associated with the measures were relatively high. This was due to the emissions associated with the production of the alternative reusable products. Influencing factors reducing the carbon footprint of waste management in tourism other than the kind of waste focused on were the existing waste management system (especially for biowaste) as well as the practicability and scalability of measures under the divers regional circumstances.

1. Introduction

Tourism is one of the major economic factors contributing to growth and jobs worldwide. According to the World Tourism Organisation (2019), the business volume of tourism at least equals that of oil exports, food products or automobiles. International

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https://doi.org/10.1016/j.envdev.2021.100617
Received 20 June 2019; Received in revised form 11 January 2021; Accepted 16 February 2021
Available online 3 March 2021
2211-4645/

tourist arrivals grew by 7% in 2017, which was the highest increase since 2009 (UNWTO, 2019). A total of 1326 million international tourist arrivals were recorded around the world in 2017, which represented a 50-fold increase compared with 1950.

However, this growth in tourism is accompanied by increases in several environmental and socio-economic impacts. Beside emissions from transport and the impacts of all necessary infrastructure (airports, hotels etc.), there is a high impact on natural resources. An additional major impact of tourism is solid waste generation. Jamieson et al. (2003) estimated that the world's 692.5 million international tourists in 2001 generated 4.8 million tons of solid waste. This amount can be expected to increase by 251% by 2050, according to the United Nations Environment Programme (UNEP, 2019). Service companies within the tourist sector and tourists themselves are major contributors to tourist waste generation. Additionally, littering is a particularly troublesome facet of tourist waste.

In comparison with other cities, tourist cities have to face additional challenges related to waste prevention and management created by the seasonality of tourism and the specificity of the tourism industry and of tourists as waste producers. In particular, in regions with a high variation in tourism throughout the year, appropriate waste management can be a major challenge, as the mass and volume of the solid waste flows generated are season-dependent (Munoz and Navia, 2015). Depending on the proportion of tourists relative to permanent residents, during high season the waste amounts generated by visitors can exceed those generated by the local population. Studies have reported increases in municipal solid waste for tourist areas or regions during high season (e.g. Teh and Cabanban, 2007; Espinosa Lloréns et al., 2008; Shamshiry et al., 2011; Mateu-Sbert et al., 2013). In addition, tourists are not always aware of how waste management in a specific region is intended to function. Ranieri et al. (2014) showed that inefficient behaviour of tourists to separate different fractions of solid waste contributes to the increase in the amount of residual municipal solid waste.

To assess the relevance of tourist activities for waste generation and associated environmental impacts, the actual amount of waste generated by tourists must be determined. The top-down approach attempts to estimate tourists' contribution to the total amount of waste generated by comparing time series of waste generation in a tourist region. In the bottom-up approach, the amount of waste generated by tourists is measured directly in tourist accommodations over a certain period. Both methods have strengths and weaknesses. The top-down approach can only be used for regions with a very high tourist impact and existing time series of data. In addition, European statistics on waste make a distinction between waste generated by economic activities and households, but there are no specific statistics on waste in tourism. In some regions, waste from tourist accommodation is collected together with commercial or with household waste, making data comparison even more difficult. The bottom-up approach focuses more or less only on food and packaging waste and does not include durable goods that are also connected to tourist activities. In addition, Pirani and Arafat (2014a, 2014b) report that there is much variation between hotels as regards how much waste per room is generated on a daily basis, with the rate depending on many variables such as hotel type, guest attributes, guest and employee activities, and occupancy rate.

In an extensive review in which 50 datasets based on the bottom-up approach were analysed, the median amount of "waste generated per tourist and day" was found to be 1.10 kg (the mean amount was 1.67 kg/tourist/day) (Ramusch et al., 2016). Similar results have been obtained using the top-down approach for 10 pilot regions, focussing on residual waste (mixed waste collected from households and other sources) only (Obersteiner and Gruber, 2017).

Although there are still uncertainties, it has been concluded that the contribution of tourism to (municipal) waste generation is both large and increasing (e.g. Cummings, 1997; Dileep, 2007; Pirani and Arafat, 2014a, 2014b; Arbulú et al., 2015; Matai, 2015; Murava and Korobeinykova, 2016). According to Arbulú et al. (2015), tourism can produce more municipal solid waste than other production activities. It is therefore important that the tourism industry continues to improve and adapt its operations towards waste minimisation. If waste is produced, it should be collected, transported, and disposed of in an environmentally sound and cost-effective manner. Improper management of waste can lead to substantial and irreversible environmental impacts, such as increases in greenhouse gas (GHG) emissions, land degradation, resource deprivation, surface and groundwater pollution and loss of biodiversity. As Munoz and Navia (2015) point out, inefficient solid waste management operations can have counterproductive effects in tourist regions, namely higher operating costs and the blight caused by litter and contaminated water, reducing the tourist value of a formerly attractive location. Also, Pirani et al. (2014a,b) conclude that solid waste management is a key aspect of the environmental management of establishments belonging to the hospitality sector.

From a municipal viewpoint, appropriate management of the waste already generated is essential. However, solid waste minimisation in tourist activities should be a major task in future waste management programmes, aimed at reducing the costs of collection, transport and disposal and thus lowering the cost of tourist activities to the local authority (Munoz and Navia, 2015). Often municipalities lack the financial means to enable sustainable solid waste management in tourist areas and therefore need an intervention from all actors to reduce financial and technical pressures and implement sustainable solutions (Chaabane et al., 2019). It is therefore necessary to define focus areas to provide information to policy makers, municipalities and responsible persons in the tourism area which measures have higher or lower impacts and should therefore be focussed on.

To determine the overall impact of tourist activities, environmental assessment covering the whole life cycle has been identified as an appropriate methodology. Environmental assessments of higher priority waste management options, such as waste prevention, are rare (Laurent et al., 2013a). The few that exist often just calculate or discuss the needs and benefits of preventing food waste, rather than evaluating actual prevention measures (e.g. Gentil et al., 2011; Garrone et al., 2014; Giuseppe et al., 2014; Prefier et al., 2016; Mourad, 2016). As pointed out above, the impact of tourist waste generation, and therefore also the potential impact of tourist waste prevention, is very strongly connected to the existing waste management system in a city or region.

The possible options to reach sustainable waste management in tourism are manifold and start with any form of waste prevention. According to the waste framework directive preparation for re-use and separate collection with recycling are the next preferable steps. Nevertheless, attention is drawn to the fact that in each case when applying the waste hierarchy measures shall be taken to encourage the options that deliver the best overall environmental outcome. This may require specific waste streams departing from the hierarchy where this is justified by life cycle thinking on the overall impacts of the generation and management of such waste (European Commission 2008).

Therefore the environmental impact of waste production by tourism must always be viewed in connection with the local waste management system, the assessment has to include actual waste management and different treatment methods, e.g. landfilling, composting, incineration and anaerobic digestion (Bernstad and la Cour Jansen, 2012; Laurent et al., 2013a; b). Qian and Schneider (2016) point out that existing research on waste minimisation within the tourism industry focuses primarily on the hospitality sector, is geographically limited, addresses practices in a cross-sectional manner and generally comprises case studies, mainly about a city, a region and even a single hotel. A limited number of Life Cycle Assessment (LCA) case studies in the tourism sector were found in the literature (de Camillis, 2010) and remain on a general level. Studies on environmental benefits on waste treatment options focussing on tourism are missing and the same applies for more comprehensive studies on the environmental effect of waste prevention and proper treatment focussing on the tourism sector.

Based on existing deficits in research the aim of this study was therefore to identify the environmental impacts of different waste prevention and treatment options connected to tourist activities depending on the existing waste management system. Within the framework of the EU H2020-funded project "URBANWASTE – Urban Strategies for Waste Management in Tourist Cities", eco-innovative waste prevention and management strategies have been tested in 10 pilot cities with high levels of tourism, in order to reduce urban waste production and improve municipal waste management. In each pilot city, strategies aimed at reducing the amount of municipal waste production and supporting the re-use, recycling, collection, and disposal of waste have been developed and tested. To limit the range of possibilities focus was laid on two prevention measures with high topicality and therefore high feasibility that is food waste prevention and reduction of single use plastics. Waste treatment focussed on separate collection and recycling of biowaste.

The cities show huge differences in the state of their waste management strategies. Florence and Copenhagen are among the best performing, with rates of waste recycling of 47% and 32%, respectively, while the other cities and areas perform less well (e.g. the recycling rate is 7% in Tenerife and 23% in Syracuse). Such differences were an advantage in the present study since they allow the waste prevention and management measures to be tested for different starting points, enabling general conclusions on the environmental effect of specific measures.

2. Pilot cities and their current waste management strategies

The pilot cities comprised six coastal cities (Nice (France (FR)), Lisbon (Portugal (PT)), Syracuse (Italy (IT)), Copenhagen (Denmark (DK)), Kavala (Greece (GR)), Santander (Spain (ES))), two insular cities (Nicosia (Cyprus (CY)) and Ponta Delgada (PT)), one peri-urban area (Tenerife (ES)) and one heritage city in the mainland (Florence (IT)).

Since the impact of waste prevention measures depends on the waste management system already in operation, this section presents an overview of the existing waste management system in each city where pilot actions have been implemented. Actual waste generation and treatment in each pilot city are described and the environmental impacts are assessed.

Beside different waste management systems, differences in city area and in number of residents are possible factors influencing the environmental impacts of waste management. The selected cities range in area from 9 km^2 for Puerto de la Cruz in Tenerife to 1400 km^2 for Nice which was the largest region followed by Kavala (Table 1).

According to Fertner and Groβe (2017), the pilot cases can be grouped into three general types based on their spatial characteristics. The first type is large cities or dense urban areas, characterised by a high share of urban area and a low share of nature areas, a high density of population and a dense road network. Among the URBANWASTE pilot cases, Copenhagen, Florence, Lisbon, Nicosia, and Santander are of this type. The second type is large authorities (large municipalities, regions, and metropolitan areas), sometimes with large cities included, but characterised by a considerable rural hinterland. Nice, Kavala, Ponta Delgada and Syracuse are examples of this type. The third type is small cities or municipalities characterised by considerable tourism. The Tenerife pilot case summarises a group of three municipalities (Adeje, Arona and Puerto de la Cruz). Therefore these three municipalities make up the pilot case in Tenerife.

Table 1
Area and population size of the 12 URBANWASTE pilot cities.

Pilot Case	Area, km^2	No. Of inhabitants
Copenhagen	86	601,448
Florence	100	377,207
Kavala	351	70,501
Lisbon	100	504,471
Nice	1400	537,769
Nicosia	21	55,014
Ponta Delgada	233	68,809
Santander	35	172,656
Syracuse	208	123,248
Tenerife/Adeje	106	45,405
Tenerife/Arona	82	79,928
Tenerife/Puerto de la Cruz	9	29,412

2.1. Existing waste management system

Data on waste management and waste amounts were obtained from case study partners (Obersteiner et al., 2017). However, it proved difficult to obtain comparable data, as the exact types of waste behind specific waste streams for which data were reported differed between countries. It became clear that only selected sets of waste generation data (i.e. waste streams) were suitable for use in further analyses. These covered: residual waste, (total) organic waste, selected recyclables (paper & cardboard, glass, metals/metal packaging and plastics/plastic packaging). Co-mingled fractions of recyclables (metals, plastics, and paper & cardboard) had to be used if no separate data were available. Differentiation between data on packaging waste only (metals, plastics) and mixtures of packaging waste with non-packaging waste from the same materials (e.g. metal hangers and frying pans or plastic toys and plastic hangers) was not always possible.

Fig. 1 shows the qualitative composition of municipal solid waste (MSW) generated in the pilot cities, expressed as a percentage. In most URBANWASTE pilot cases the share of separate collected recyclables is still exceptionally low, so the values shown are in line with national data.

After being collected, the waste flows undergo different treatment and disposal processes including mechanical biological treatment (MBT), waste-to-energy (WTE) treatment or direct landfilling with or without gas collection (Table 2). In most of the pilot cases, recyclables are collected as a co-mingled fraction of varying composition. In a final recycling step, the mixture has to be sorted and each fraction is recycled separately. When modelling the environmental impacts, when data were missing the total amount of collected co-mingled recyclables was allocated to the respective fraction based on mean values for those cities where each fraction is collected separately (Table 3).

There is no single solution for appropriate waste management. Thus, each country and each city can have its own solution, based on the existing conditions. However, there are some main issues to be considered when assessing the environmental impact of tourist activities in terms of waste management:

- **Availability of data**: Most of the waste generated by tourists will end up in hotels, restaurants, or other tourist facilities. In some pilot cities, the waste generated in such service institutions is collected separately and ends up as commercial waste, in others it is collected together with household waste and ends up as municipal solid waste, and some cities use a mixture of both systems. Therefore, the data used must be interpreted with care.
- **Treatment of residual waste**: Depending on the disposal pathway used for residual waste (especially if no treatment is applied or the waste ends up in landfill), the waste generated in general and that generated by tourists has a negative environmental impacts.
- **Share of recyclables**: The proportion of waste that is collected separately and recycled also influences the environmental impact of tourists' waste-related behaviour.
- **Separate collection/treatment of organic waste**: Separate collection and recycling of organic waste will result in substantial savings in terms of greenhouse gas emissions.

2.2. Environmental impact of existing waste management

As the goal of this study was not to compare the pilot cities only relative impacts based on waste generation per capita were used. Data on energy use in waste management facilities and the amount of energy replaced by energy production from waste incineration with recovery or biogasification were taken from the latest published in the latest energy datasheets (EU Commission, 2017).

Environmental impacts in terms of global warming potential (GWP) caused by waste generation and waste management activities differed widely between the pilot cities, mainly as a result of two factors: the actual amount of waste generated and the existing waste

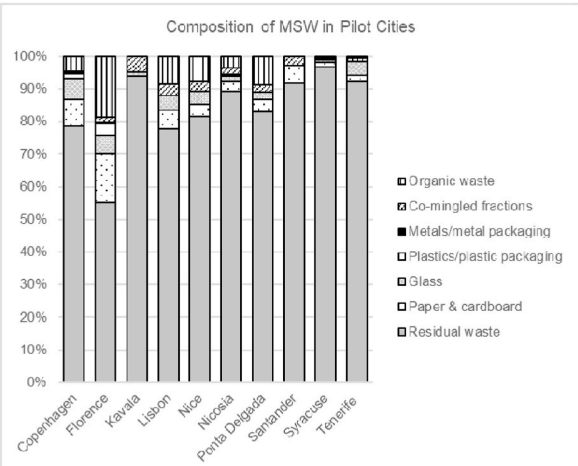

Fig. 1. Composition of municipal solid waste (MSW), i.e. residual waste and separate collected fractions) in 10 pilot cases in 2015.

Table 2
Waste treatment in the 10 URBANWASTE pilot cities. MBT = mechanical biological treatment.

Treatment of residual waste (mixed waste from household and other similar to household sources)	Copenhagen	Florence	Kavala	Lisbon	Nice	Nicosia	Ponta Delgada	Santander	Syracuse	Tenerife
Incineration with energy recovery	x			x	x			x		
Recycling										
MBT		x								
Landfill			x	x		x	x	x	X	x

Table 3
Proportions of different waste types used in Life Cycle Assessment of the 10 URBANWASTE pilot cities. Co-mingled fractions were allocated to different recyclables based on the proportions in other cities.

Waste composition for recycling and treatment in %	Copenhagen	Florence	Kavala	Lisbon	Nice	Nicosia	Ponta Delgada	Santander	Syracuse	Tenerife
Residual waste	81.1	55.3	93.9	77.8	80.1	89.7	83.1	91.8	96.8	92.5
Paper and cardboard	8.7	15.8	3.9	5.9	3.7	3.3	3.7	5.3	1.3	1.7
Glass	4.7	5.9	1.1	4.4	4.0	1.5	2.1	2.2	0.8	4.3
Metals	0.4	0.6	0.2	0.6	2.1	0.3	0.4	0.2	0.0	0.3
Plastics	0.5	3.8	0.8	3.0	2.7	1.7	1.9	0.5	0.6	0.6
Total organic waste	4.6	18.7	0.0	8.4	7.4	3.6	8.8	0.0	0.4	0.6

management system. Different waste treatment measures produce different emissions and therefore have different environmental impacts. It also has to be taken into account that recycling and incineration of waste with recovery often have overall positive impacts, as the production of energy from waste or the production of secondary goods leads to environmental benefits by avoiding the use of primary resources (e.g. fossil fuel). These savings are indicated as negative values in Fig. 2. Only generic data were used for the modelling, so uncertainties were considered, but the results might be slightly worse or even better depending on the technology used.

The results confirmed that landfilling is the worst final waste disposal alternative, while composting and material recovery showed the best performance. Organic waste was shown to make the largest contribution to GWP and was therefore investigated in more detail. This revealed that organic waste in the case cities is treated with four major methods: landfilling, composting, incineration with recovery and anaerobic digestion. Since landfilling of organic waste gives rise to GHG emissions due to methane leakage, this was identified as the most important waste fraction for cities to divert from landfill. The relevant fraction of organic waste resulting from tourist activities is food waste, so efforts to reduce the environmental impact of tourist waste management should focus on separate

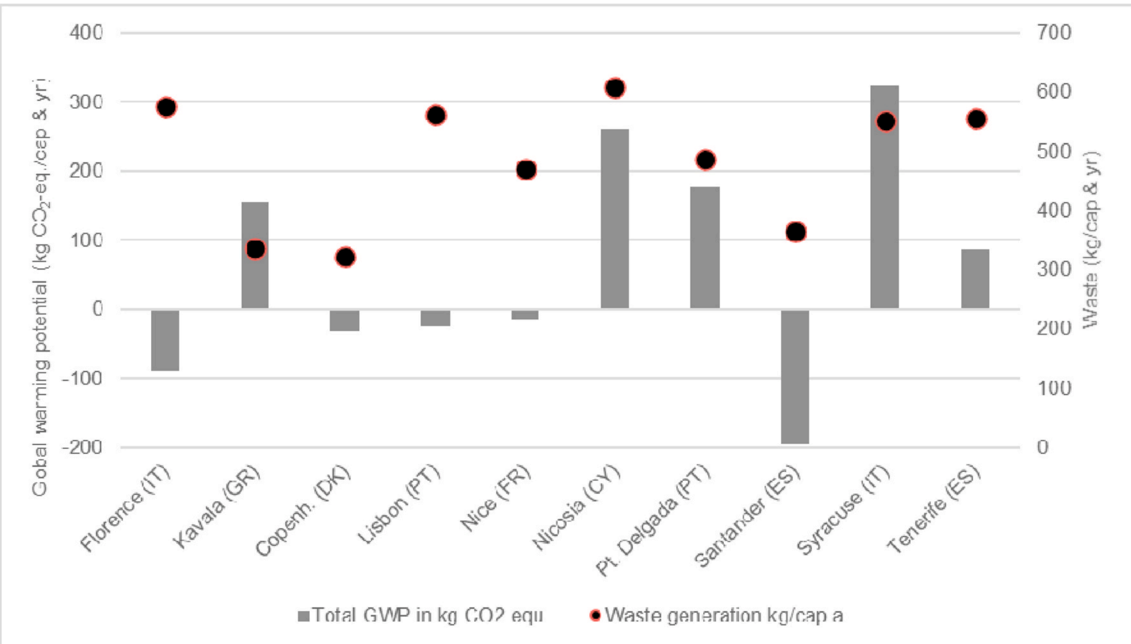

Fig. 2. Waste production and global warming potential (GWP) of waste management per capita and year in the 10 URBANWASTE pilot cases for which data were available.

collection and appropriate treatment of food waste. Significantly lower emissions were achieved in the cities collecting landfill gas or treating the organic waste with any other method. This was mainly due to less methane leakage, but also to the substitution of other products when recycling nutrients and energy from the organic waste.

3. Pilot measures assessed and assessment methods

The impacts of the “baseline” were compared with those of “innovative strategies”, based on primary data collected in the URBANWASTE project. The scope of the waste management strategies tested ranged from waste prevention to increased recycling (Table 4).

The selected strategies were evaluated following a Life Cycle Assessment approach (ISO, 2006). Processes were modelled in GaBi LCA software (GABI, 2019). As the goal of all strategies was to prevent waste or to increase recycling, the functional unit chosen was 1 kg waste prevented or, for measures aiming to increase recycling, 1 kg of waste diverted higher up the waste hierarchy. E.g. diverted from disposal to recovery to recycling to re-use to being prevented to be waste. Within this study the waste hierarchy according to the Waste Framework Directive (European Commission, 2008) was applied. Results were reported against the functional unit.

The system boundary in the study included implementation of the strategy, transports, and treatment of waste. Secondary material produced from waste through recycling was credited for the substitution of primary materials. In cases where materials are sent to waste incineration with recovery, they were linked to an inventory that accounts for waste composition and heating value, as well as for regional efficiencies and heat-to-power output ratios. Credits were assigned for power and heat outputs using the regional grid mix and thermal energy from natural gas. The latter represents the cleanest fossil fuel and therefore results in a conservative estimate of the avoided burden.

Greenhouse gas emissions were characterised as CO_2 equivalents (CO_2 eq.) following the IPCC (2013) characterisation factors over a time horizon of 100 years. Table 5 gives an overview of a selection of most relevant emission factors for assessed activities used within the study. As an example: if 1 tonne of food waste is composted instead of landfilled, 16 kg of CO2eq. are emitted by the composting process while 590 kg CO_2eq. are saved as the waste is no longer landfilled. Hence diversion of 1 tonne of food waste from landfill to composting saved 574 kg CO_2eq.

The strategies were piloted during the period March–November 2018, with the implementation period varying between strategies and between cities.

3.1. *Food waste prevention*

To prevent food waste and redistribute surplus food, three actions were implemented in the pilot cases: donations to charity in Florence; use of a food waste tracker in combination with individually designed actions to prevent food waste in Copenhagen, Kavala, Lisbon, Nicosia, Santander and Tenerife; and use of “doggy bags” to allow customers to bring home their own leftovers in Kavala, Nice and Florence.

A system overview of all food waste prevention options is shown in Fig. 3. In all cases, avoided production of new food achieved by redistribution of surplus food is included as an essential part. The system also includes the current waste management system, as this system is replaced by the action when food is no longer wasted. Depending on the existing waste management options, different food waste recycling and treatment options that are replaced must be considered.

3.1.1. *Donations to charity*

In Florence, four hotels and two charity organisations engaged in a food redistribution activity designed to reduce food waste. During the implementation phase, the hotels donated a total of 795 kg of food and 72 L of beverages, but the donations measure also led to an increase in transports. The implementation phase was evaluated during six months and all surplus food recorded was counted as donated. This value was normalised to a full year in order to give comparable results.

3.1.2. *Food waste tracking*

The theory behind food waste tracking is that waste quantification in itself can be a way to reduce food waste, through increased awareness of the issue in catering units such as hotels and restaurants. Waste can be quantified in many ways but is often neglected due

Table 4
Waste prevention and management strategies in URBANWASTE pilot cities that were assessed in this study.

Measure		Pilot cities
Food waste prevention	Reducing plate waste by offering ‘doggy bags’	Kavala, Nice and Florence
	Improved waste quantification using a food waste tracker.	Copenhagen, Kavala, Lisbon, Nicosia, Santander and Tenerife
	Donations of surplus food to charity organisations	Florence
Reduction of single use plastics	Substitution of single use toiletry containers in hotels.	Lisbon and Ponta Delgada
	Promotion of tapwater use and reusable water bottles.	Florence and Nice
Increase in separate collection and recycling	Waste sorting hotel rooms and common areas. Staff training and provision of waste separation bins.	Lisbon
	Composting of biowaste on site	Tenerife

Table 5
Selection of emission factors applied within this study. *GABI 2019 and own calculations #Saxe et al., (2013)+Williams et al. (2011).

Process/Product	GWP	Unit	Comment
Waste management processes			
Centralised composting of food waste	16	kg CO_2eq./tonne	Open windrow composting (incl. Compost application and crediting)*
Landfill of food waste	590	kg CO_2eq./tonne	Biodegradable waste on landfill with gas recovery*
Incineration of waste	95	kg CO_2eq./tonne	Incineration of municipal waste with energy recovery*
Activities and products required to implement pilot actions			
Production of food container (doggy bag)	0.21	kg CO_2eq./kg of food packed	Container, cradle to grave +
Reusable toiletries container	2	kg CO_2eq./piece	Cradle to grave*
Single use toiletries container	0.01	kg CO_2eq./piece	Cradle to grave*
Water bottle single use	2	kg CO_2eq./piece	Cradle to grave*
Reusable water bottle	48	kg CO_2eq./piece	Cradle to grave*
Products substituted within pilot actions			
Food production	2.1	kg CO_2eq./kg	Production of food composing an average diet #

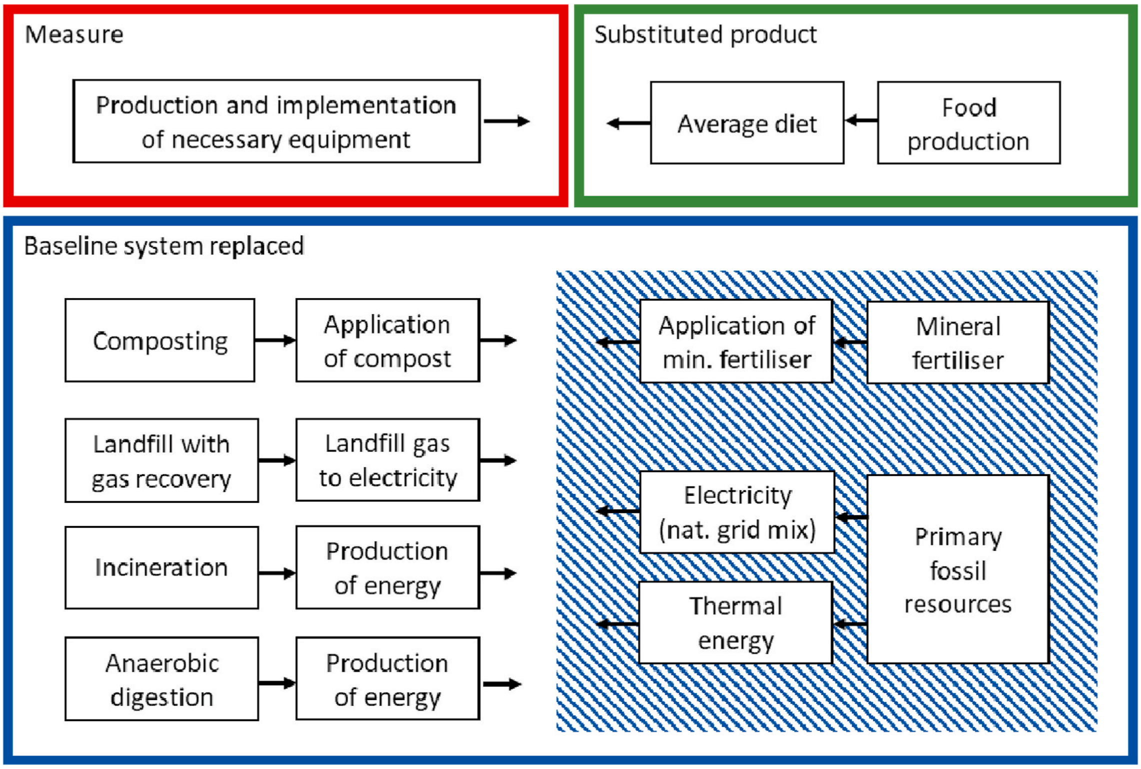

Fig. 3. System overview of pilot measures for food waste prevention. The 'measure' includes all activities directly attributable to the pilot measure (i.e. provision of doggy bags, scales, distribution). The 'Substituted product' includes all activities substituted by the measure (food prevented from wastage replaces other food). As the 'substituted product' is being displaced by the measure, environmental benefits associated with substitution are attributed to the pilot measure. The 'baseline system replaced' includes all waste management activities associated with the disposal of food before the implementation of the pilot measure.

to the extra efforts and long-term commitment required from staff. Therefore, a food waste tracker from the Swedish company Matomatic AB was implemented in the pilot cities. The food waste tracker consists of a heavy-duty scale equipped with a data port connected to a Bluetooth antenna. The weight of the waste placed on the scale is sent to a tablet computer with a quantification application (app) installed. Through this app, the staff can categorise each piece of waste by selecting one of the pre-selected waste categories. Kitchens can choose to quantify between two and nine categories of food waste. In order to provide feedback on the food waste quantifications, the tablet computer uploads the recorded data consisting of a time stamp, kitchen identification number, waste category (including process, meal and food type, where applicable) and mass of waste to an external database. The tracker also provides the possibility to record the mass of food served for each category and number of guests for each meal, in order to provide a value to which food waste can be related. The data collected in the external database are used in weekly reports on the progress of each kitchen that are emailed to the kitchen staff.

The kitchen staff are intended to use the feedback to increase their awareness not only of the amount of food they were wasting, but also what they are wasting and when. This increased awareness is expected to lead to waste reducing actions based on the problems identified. During the implementation phase, the food waste trackers were used for between 1 and 309 days (where food waste was recorded). To quantify the waste reduction, the quantification periods used in this study were divided into two parts for each catering unit included and the first 50% of days were compared with the last 50%. If the quantification period included an uneven number of days, the middle day was allocated to the first period. With this definition of food waste reduction, 21 of the 33 food waste trackers installed succeeded in reducing the waste. The collective reduction from all scales was 18,190 kg. When only considering kitchens, which reduced their waste in the second half of the implementation period, the total reduction was 20,133 kg, or a 959 kg food waste reduction per unit.

The restaurants also implemented additional actions to reduce food waste. In Copenhagen, the measures introduced by restaurants included serving staff surplus food, optimising food purchases and motivating staff through competition with other hotels in the group. In Lisbon, the restaurants communicated their efforts and ambitions through social media, posters and promo-cards. In Tenerife, the restaurants increased use of food scraps and leftovers for the preparation of other dishes, made smaller-sized plates available at the buffet, changed from rectangular to convex trays to keep the appearance of a full tray with less food actually on the buffet and offered reduced portions in the a la carte menu.

3.1.3. Use of doggy bags

A doggy bag enables a restaurant guest to bring the uneaten part of a portion away in order to eat it later. By doing this, the restaurant guest prevents the food from ending up in the waste bin of the restaurant and, if the food is eaten instead of buying new food, it also reduces the amount of new food being produced. Since both production of food and waste management of food waste can generate emissions, avoiding these activities has the potential to reduce emissions.

The implementation phase of this measure was evaluated during two weeks in relation to a baseline week quantification. During the implementation phase, 867 kg food waste were avoided in the restaurants using doggy bags. This value was normalised to a full year in order to give comparable results. The total redistributed food waste was calculated to 23 tonnes per year.

3.2. Strategies to reduce single use plastics

Two strategies with the aim of reducing the use of single use plastics were implemented in four cities. One strategy, implemented in Lisbon and Ponta Delgada by two hotels, involved replacing single use toiletry containers with refillable dispensers. The other strategy promoted the use of public water fountains and reusable water bottles to reduce the use and disposal of single use PET water bottles. The assessment quantified GHG emissions associated with all life cycle stages of the reusable product (production, distribution, re-use, disposal) and included benefits associated with substitution of the former single use product system (production, distribution, use, disposal).

The participating hotels eliminated the use of single use toiletry containers by introducing refillable containers. Two dispensers were installed per hotel room. The hotels provided data on the numbers, weight and material of single use containers replaced and of refillable toiletry dispensers installed. The single use containers in Lisbon and Ponta Delgada weighted 2 g per piece with a capacity of 20 ml and were made of polypropylene (PP). The reusable containers weighted 440 g with a capacity of 300 ml and were made of acrylonitrile butadiene styrene (ABS). In Lisbon 1 kg of single use containers were replaces by 0.3 kg of refillable containers. Ponta Delgada achieved a higher replacement rate of 0.1 kg refillable container per 1 kg of single use containers due to higher hotel occupation rates.

The evaluation of the measure included production, transport, distribution and end of life treatment of the reusable and the disposable products. Disposal routes of ABS refillable dispensers and PP single use bottles were considered based on Eurostat (2018) recycling and recovery rates for packaging. The actual toiletries provided in the different containers were excluded from the assessment.

The European strategy for plastics in a circular economy aims to replace disposable water bottles by encouraging the use of public water fountains in tourist areas and by distributing reusable water bottles to tourists to reduce single use plastics. The measure was implemented in Florence and Nice. Both cities measured the quantity of water distributed from three automatic fountains each and the number of reusable bottles distributed. It was assumed that, prior to the introduction of water fountains, tourists used 0.5 L single use water bottles and that 0.5 L distributed through the fountains displaces one PET bottle. The reusable bottle distributed in the project was an aluminium flask.

The scope of the evaluation included the production, distribution and end of life of the reusable and disposable bottle. The evaluation was based on the conservative assumption that the tourist is on holiday for one week, fills their bottle once per day and disposes of their bottle after one week. The actual number of use cycles will vary but, with no better data available, seven use cycles were assumed to be realistic for this study. Country-specific end of life treatment was assumed according to European Aluminium (2018). End-of-life routes for France and Italy were considered, based on Eurostat (2018) recycling and recovery rates for packaging.

1 kg or 53 single use bottles were displaced by 0.8 kg or 7.5 pieces of reusable water bottles. Bottle weights and production processes were modelled according Botto et al. (2011a, 2011b), Ciafani et al. (2008) and Islam et al. (2018). Single use bottles were made of polyethylene terephthalate (PET) with a polypropylene (PP) lid, a weight of 19 g per piece and a capacity of 500 ml. Reusable bottles were made of aluminium with a PP lid, a weight of 108 g per piece and a capacity of 500 ml.

3.3. Increased separation of collection and recycling

3.3.1. On-site composting in tourist establishments

In April 2018, an electric composter was installed on the premises of the Hotel Tigaiga in Tenerife. The composter is used to recycle food waste on-site into high quality compost, which is then applied to the hotel's gardens. Before implementation of on-site composting, food waste was collected and sent to composting and landfill facilities off-site. The evaluation of this measure aimed to quantify GHG emission savings associated with this action compared to the previous situation, when only a fraction of the food waste collected was composted.

A function associated with on-site composting is that less organic waste requires collection and disposal off-site, so this reduction in emissions from the previous waste disposal route can be credited to the measure. In addition, the on-site composter produces compost that is used as a soil conditioner and fertiliser in the gardens, which reduces fertiliser use on-site.

During the pilot phase of 21 weeks, 3071 kg food waste were produced and composted. This food waste was produced by 20,879 guests staying at the hotel during the pilot phase. Hence each guest produced 147 g food waste that was composted.

The composter required 1.5 kWh electricity per day and about 15 kg sawdust per 100 kg food waste input to operate. During the pilot phase, 1151 kg compost was produced, which is equal to 0.375 kg compost per kg food waste. A macronutrient content of 0.8% N, 0.14% P_2O and 0.22% K_2O was assumed for the composted food waste (Gomez-Barea et al., 2010). This means that 1 tonne of compost potentially displaces 24.8 kg ammonium nitrate, 3.7 kg triple superphosphate and 2.9 kg potassium chloride.

Before the pilot measure was implemented, food waste was collected and, in accordance with the current treatment in Tenerife, 54% was composted and 46% was landfilled with gas collection and recovery. Transport distance to the composting and landfill site was 80 km. For municipal composting, an open windrow composting system according to GABI (2019), including application of compost to agricultural land and fertiliser displacement, was assumed.

3.3.2. Waste separation in hotels

The measure on waste separation aimed to increase separate collection of waste in hotel rooms and common areas of hotels. Data on two hotels in Lisbon were evaluated to see how waste separation could be improved by providing training to staff and providing separate bins in hotel rooms and common areas. Separate collected fractions were plastic, paper, glass and organic waste. These fractions are sent for recycling. Following the avoided burden approach, it was assumed that secondary materials produced through recycling replaced primary materials (Frischknecht 2010). Most of the unsorted waste is send to incineration with energy recovery, while a small fraction is sent to landfill with gas recovery. Energy recovered from incineration was assumed to replace electricity from the Portuguese grid and thermal energy recovered was assumed to replace thermal energy from natural gas.

Collection rates before and after implementation of the pilot measure are shown in Table 6. Before implementation, the recycling rate was 65%, while after implementation it was 74%. As shown in Table 6, the recycling rate did not increase for all waste fractions. In fact, a decrease was observed for paper and organic waste, the reasons for which are unknown. Measured data were provided as volume and converted to mass by using conversion rates according to Umweltberatung (2012).

4. Results

4.1. Food waste prevention

4.1.1. Donations to charities

The food donation measure had an impact from the transportation, but this impact was small in comparison with the reduced impact from substituting virgin food production. This is evident in Fig. 4, where results are disaggregated by measure, food substituted and replaced waste management. Since the current waste management practice in the city is anaerobic digestion, the impact of the replaced waste management system is high when food is donated instead of becoming biogas, as there is no more crediting from the use of the biogas. Hence the credit achieved by the biogas conversion to energy is greater than the burden caused by the anaerobic digestion process. The total amount of redistributed food waste was calculated to 1.7 tonnes per year, corresponding to a 2.0 tonnes CO_2-eq./year net reduction in GHG emissions.

4.1.2. Food waste tracking

In the assessment, it was clear that food waste trackers have a high initial impact from production of materials and transport of a

Table 6
Collection rates before and after implementation of the measure to increase separate collection of waste fractions in hotels.

Waste fraction	Before implementation Mass %	After implementation Mass %
Unsorted waste	35%	26%
Plastic packaging	4%	10%
Paper	20%	16%
Glass	9%	23%
Organic waste	32%	25%
Total waste	100%	100%

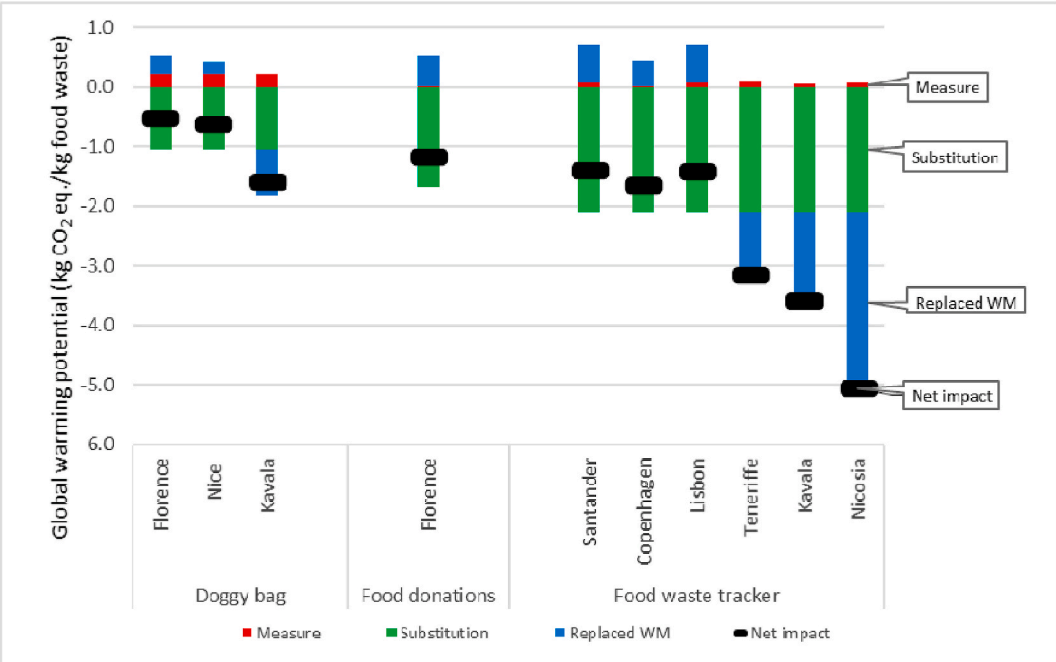

Fig. 4. Global warming potential of food waste prevention measures 'Doggy bag', 'Food donations' and 'Food waste tracker' in the URBANWASTE pilot cases for which data were available, disaggregated by city. WM = waste management.

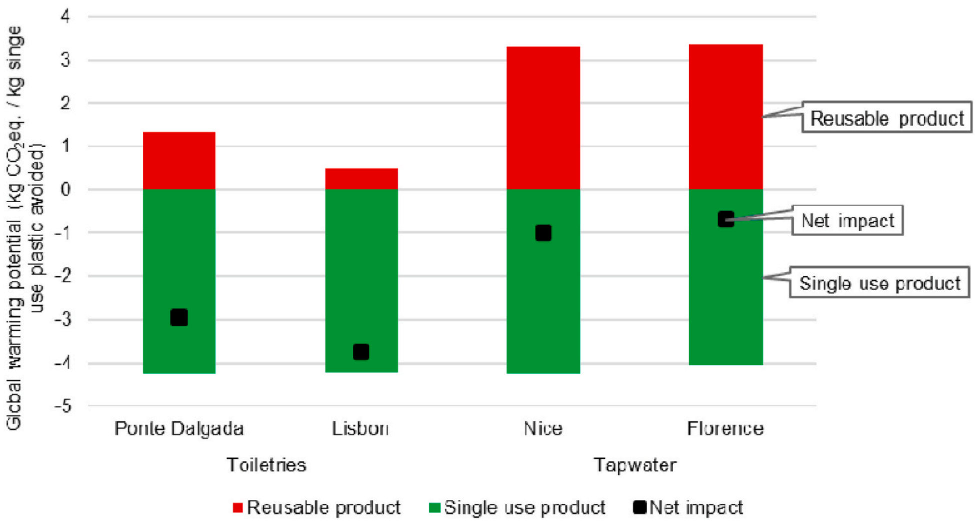

Fig. 5. Global warming potential of measures for reducing single use plastics (toiletries, water bottles) in the four URBANWASTE cities that implemented the measure.

technician for installation. However, since these are fixed impacts related to the waste reduction, they become comparatively low as the trackers are used over time to reduce food waste. Since waste management practices differ between the cities, the waste reduction can have both negative and positive impacts. Cities with waste treatment plants that can recover energy and nutrients from the food waste have less potential to improve than cities where landfilling is the main alternative. Therefore, the food waste trackers proved to have the greatest potential to reduce GHG emissions in the pilot cities of Nicosia, Kavala and Tenerife, since both food production and landfilling of organic waste were avoided in those cases.

The total impact from the waste reducing action of using food waste trackers in restaurants and hotels was 181 tonnes CO_2-eq./year (Fig. 4). The largest impact was achieved in Tenerife, where 69 tonnes CO_2-eq./year were avoided due to this action, while in Lisbon the action also greatly reduced GWP, by 38 tonnes CO_2-eq./year. Since most food waste trackers installed in Kavala were used to a small extent, the recorded waste reduction was small. However, since landfilling of waste was avoided in Kavala, every kg reduction in food waste had a high impact. The opposite trend emerged for Copenhagen, which reduced food waste much more than in Kavala but already had efficient infrastructure to treat food waste, so the potential was not as large and the action did not reduce the emissions as much.

4.1.3. Use of doggy bags

The total redistributed food waste was calculated to 23 tonnes per year, corresponding to a 17 tonnes CO_2-eq./year net reduction in GHG emissions (Fig. 4). Considering the impact in each pilot city from the waste reducing action of introducing doggy bags in restaurants, the measure had the largest impact in Nice, where an estimated 7.7 tonnes CO_2-eq./year were avoided, followed by Kavala and Florence, where 7.5 and 2.9 tonnes CO_2-eq./year, respectively, were avoided.

4.2. Strategies to reduce single use plastics

Fig. 5 shows the GHG emissions associated with implementation of the measures to reduce single use plastics per kg disposable plastic avoided. Activities associated with implementation of the measure, e.g. the life cycle of the refillable dispenser or reusable bottle, is shown as a GHG emission (positive value), while the replaced system is shown as an avoided burden (negative value). Both measures achieved an overall saving in GHG emissions (net impact).

Comparing the two measures, the measure substituting single use toiletry containers achieved higher savings. This is mainly due to the lower impact of the reusable toiletry containers versus the impact of the reusable water bottle.

For the toiletry container measure, difference between cities were mainly associated with different occupation rates of participating hotels. Disposable containers are changed per day and guests, while refillable dispensers are only installed once. Hence the higher the occupation rate of the hotel, the more single use toiletry containers are replaced per refillable container. It was assumed that the refillable container is replaced annually.

4.3. Increased separation of collection and recycling

4.3.1. On-site composting in tourist establishments

Fig. 6 shows the result per kg biowaste composted. The GHG emissions are indicated as positive values, while potential savings are indicated as negative values. An overall saving (net impact) of 0.3 kg CO_2-eq. could potentially be achieved by composting 1 kg of food waste. Most of this saving is achievable by avoiding landfilling.

The GHG emissions associated with the pilot measure were relatively small (0.03 kg CO_2-eq./kg food waste). This includes emissions associated with the closed composting process on-site and application of the compost in the hotel gardens. If the applied compost replaced fertiliser use, this resulted in a benefit of 0.001 kg CO_2-eq. per kg food waste.

Prior to on-site composting, the hotel's food waste was collected and transported to a composting site and landfill. As this treatment is avoided by on-site composting, emissions associated to the previous waste treatment system can be credited to the pilot measure. The replaced waste management system was associated with 48 kg CO_2-eq. per kg food waste. Most (98%) of these emissions were associated with landfilling of food waste.

4.3.2. Waste separation in hotels

Fig. 7 shows the GWP of waste collection and separation before and after the measure was implemented. For systems that provide a

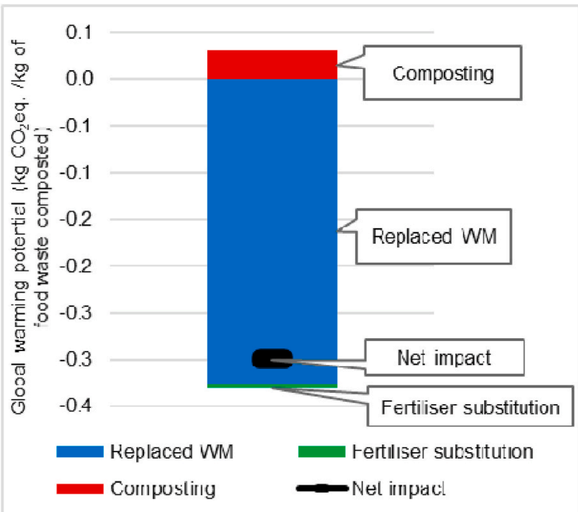

Fig. 6. Global warming potential of on-site composting at a hotel in Tenerife. WM = waste management.

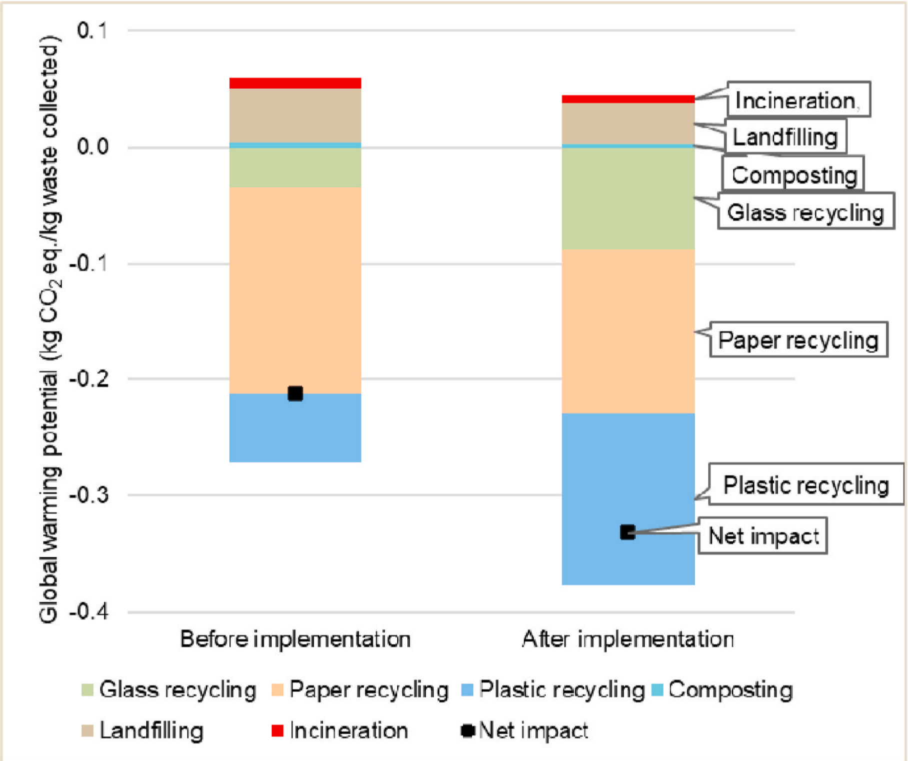

Fig. 7. Global warming potential of separate collection of waste fractions in hotels in Lisbon per kg of waste collected.

net environmental burden, e.g. landfill, incineration and composting, GHG emissions are shown as positive, while for systems creating a net environmental benefit, e.g. recycling, results are shown as negative. The relatively high recycling rate before implementation had already created a net environmental benefit of 0.211 kg CO_2-eq./kg waste. After implementation, the benefit increased to 0.332 kg

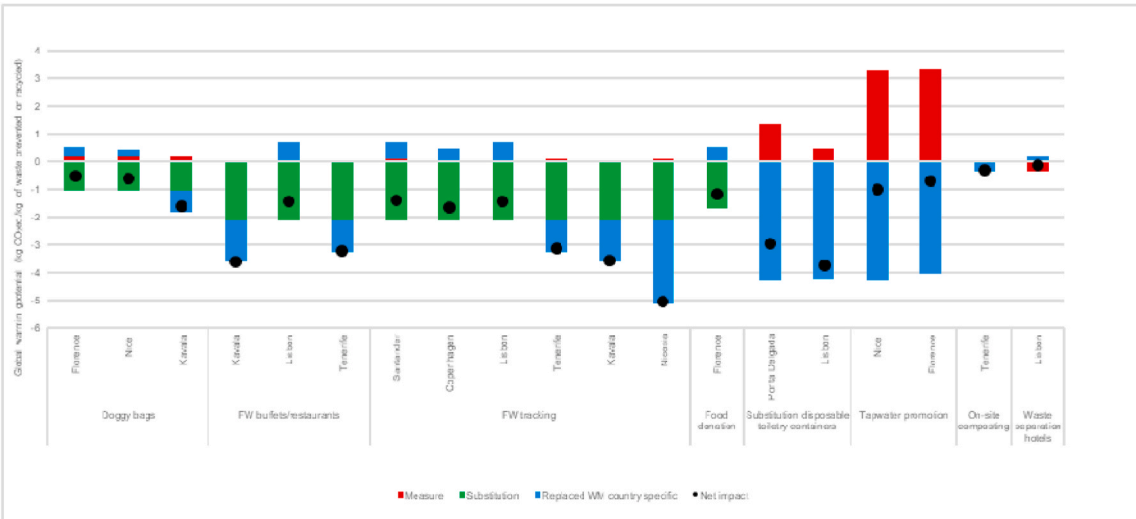

Fig. 8. Global warming potential per kg waste prevented or recycled for each pilot measure tested in the URBANWASTE pilot cities disaggregated by city. Note, however, that these results are based on some critical assumptions which differed between the measures. For food waste reducing measures, assumptions associated with the substitution of food production substantially impacted the results. This is clearly a sensitive part of the assessment, since the substitution also had the largest impact. It was assumed that 100% of the avoided food waste substituted for production of food in general, even though it might have been waste of specific products that was avoided. For the doggy bag measure, it was assumed that only 50% of the leftovers redistributed with the doggy bag were used to substitute for another meal. This was because the primary data collected only included observations up to the point when the leftover food is packed in a bag or donated. What happened afterwards with this food is unknown.

CO_2-eq./kg waste. Hence the measure led to an overall benefit of 0.12 kg CO_2-eq. per kg waste.

The measure achieved an overall increase in separately collected waste in hotel rooms. Compared with incineration or landfill, recycling generates secondary materials that potentially substitute the production of primary materials. Hence less primary material requires production, leading to environmental benefits such as a reduction in biotic and abiotic resource use. The recycling rate increased from 65% to 74% following the implementation of the measure.

5. Discussion

A comparison of all measures, based on 1 kg waste prevented or diverted higher up the waste hierarchy, is shown in Fig. 8. As can be seen, there were substantial differences between cities and between pilot measures.

The GWP of the measures themself were substantially lower for food waste prevention and separate collection than for measures reducing the use of single use plastics. For the two measures aimed at reducing the use of single use plastics, emissions associated with the measure itself were relatively high, in particular for the measure promoting tapwater use. This was due to the impact of production and disposal of the refillable containers and the assumption that one reusable bottle replaces only seven disposable bottles. Further savings could be achieved by prolonging the use phase of the reusable bottles and refillable containers. Measures aiming to increase separate collection and recycling are dependent on available infrastructure to handle an increase in the available waste fraction or implementation of on-site recycling infrastructure such as on-site composting.

The assessment also evaluated GHG emissions associated with the replaced waste management system. Differences were observed between cities disposing of the waste fractions assessed within this study to landfill or incineration without recovery and cities that already recycle or recover these waste fractions.

Food waste prevention measures also achieved potential savings by substituting the production of food. Hence measures reducing food waste in kitchens, buffets and restaurants also reduce the amount of food that is required to prepare one meal, while measures aiming to utilise leftover food potentially replace other meals.

Scherhaufer et al., (2018), Monier et al., (2010) and FAO 2013 report the impact of avoidable food waste in Europe to be between 1.9 and 3.6 kg of CO_2e per kg of food waste. Even if the results are not directly comparable, as Scherhaufer et al., (2018), Monier et al., (2010) and FAO 2013 do not include the burdens of activities associated with the prevention measure itself, they are within a similar range as shown in this study.

The substitution of water provided through bottles and fountains or tabs has been studied based on the function of providing different amounts of water (Botto et al., 2011a, 2011b, Sauer el al. 2009, Dettore 2009, Fantin el al. 2014). These studies results cannot be directly compared with results of this study as results are reported against a different functional unit.

Filimonau el al. (2011) reviewed the carbon footprint of tourist accommodations and concluded that the impacts of single use toiletry products are neglectable. Looking at the impact per kg of waste prevented within this study the carbon footprint reduction potential is comparable to other measures (Fig. 8). When comparing the saving achievable per 1000 tourist the substitution of single use toiletry containers has a substantially lower carbon footprint reduction potential than other measures (Fig. 9). Hence findings are comparable to Filimonau el al. (2011).

Greenhouse gas emissions of different composting systems are studied in literature (Lundie et al., 2005; Boldrin et al., 2009, Martinez-Blanco et al., 2013). Lundie et al. (2005) compared home composting with centralised systems and found, if the home composting system is managed aerobe, greenhouse gas emissions were substantially lower than centralised composting systems (0.3 vs. 52 kg CO_2e per 182 kg waste). Martinez-Blanco et al. (2013) and Boldrin et al. (2009) came to similar results.

The overall effect of the implementation of separate waste collection in hotels was rather low compared to other measures. In general, for measures requiring a switch to separate collection, the existing waste collection system must be considered. Obersteiner and Gruber (2017) were able to show for the analysed pilot cities that the share of total waste generated by tourists is too small to justify a separate collection system only based on activities for tourist location not including households.

Disaggregating the results per kg waste prevented or diverted higher up the waste hierarchy provided an overview of where emissions occur but had some limitations in identifying achievable savings per tourist or city. To identify optimal solutions to assist decision makers and create necessary policy implications the more general view on the possible savings per tourist seems necessary. Therefore, the potential savings were scaled up to 1000 tourists (Fig. 9). The generalisation of results and direct comparison of measures performed in different cities appears admissible based on the assumption that tourism works similar in different regions although one has to take into account possible deviations based on the type of vacation. Savings might be lower for example in regions where already mainly eco-tourism occurs. As for all cities the European energy mix was used. The direct upscaling was not possible for the food waste prevention at buffets and restaurants and the waste separation at hotels measures, due to lack of data. It was also associated with some limitations. For example, for food waste preventing measures it was assumed that one meal equals 500 g and one tourist equals one meal. For measures applied in hotels (substitution of single use toiletry products and on-site composting), it was assumed that one guest equals one tourist. Data on the number of guests were available, each guest was assumed to stay one night. The actual number of nights spend was not available. As Fig. 9 shows, when expressed per 1000 tourists the measures increasing the efficiency of kitchens (food waste tracking) and tapwater promotion achieved a substantial saving, while the replacement of toiletry containers only achieved a relatively small saving.

The food donations measure appeared to be a very promising option in this comparison, but this result was highly dependent on local conditions and assumptions and should therefore be interpreted with caution. For example, the amount of food donated per tourist by the donating hotels was relatively high (160 g/tourist), it was assumed that none of the surplus food was lost at the charity organisation and it was also assumed that the charity organisation would have purchased the same amount of food if they had not

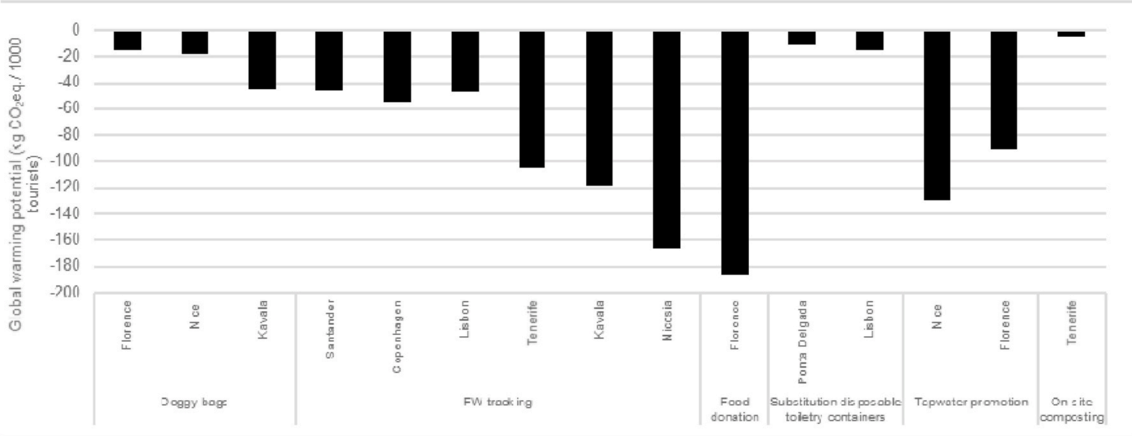

Fig. 9. Global warming potential of each pilot measure disaggregated by city and scaled to 1000 tourists.

received the donations. There is much potential for variations that would reduce the impact from this action, but the results presented are still feasible.

Basing the results on 1000 tourists instead of on kg waste prevented or recycled resulted in significant differences. For the measure involving the use of tapwater, the impact per kg of waste prevented was comparatively low, but the GWP savings per 1000 tourists was high, 130 kg CO_2-eq. in the case of Nice. In this comparison, it emerged as the third promising measure after food donation in Florence (185 kg CO_2-eq./1000 tourists) and food tracking in Nicosia (166 kg CO_2-eq./1000 tourists). However, it must be taken into account that indefinite upscaling, especially of the food donation measure, is not possible.

6. Conclusions

All pilot measures achieved a saving in GHG emissions compared with the situation before implementation of the measure. However, it emerged that the contribution of tourists to overall annual waste generation does not justify general changes in the collection system for recyclables in the interests of tourism alone. The focus in future waste prevention and recycling should therefore be on measures that assist existing systems.

Pilot actions diverting organic waste from landfill by waste prevention and by separate collection and treatment achieved substantial improvements. A promising measure with high potential to reduce the carbon footprint of tourism was the installation of public drinking water fountains (and accompanying information measures), which is relevant at hotel level but can also be implemented by the municipality. Tourists could be encouraged to refill drinking bottles, thus reducing PET bottle waste. The provision of refillable drink bottles together with relevant information on waste prevention could be a possibility. The use of refillable toiletry containers in hotels was identified to give high GHG savings per kg waste, not least because of the low weight of plastic waste. However, the impact per pilot action and per 1000 tourists was low compared with that of other actions dealing with e.g. food waste.

To optimise potential GHG emission savings, several issues must thus be considered by decision makers before being implemented into policies. First, it could be shown that the existing waste management system has a major influence on the results, as found here for the food waste tracking measure implemented in six different cities. Another major influencing factor is the type of waste prevented, as different fractions have different weights and their prevention or recycling has various impacts. The feasibility of measures and their coverage also have an influence, e.g. measures implemented at individual hotels will have lower impacts than measures requiring changes for all hotels, such as separate food waste collection and composting or providing general solutions like tap water.

Since landfilling of organic waste causes greenhouse gas emissions due to methane leakage, this is the waste fraction and route with the highest impact on GHG emissions. Therefore, in regions where organic waste is not collected separately, pilot actions diverting organic waste send to landfill by both waste prevention as well as separate collection and treatment is seen as a substantial area of improvement. Within the different waste management options for organic waste it could be shown that referred to possible carbon footprint reduction potential scaled to 1000 tourist's food waste prevention is preferable compared to composting. Donation turned out to have the largest benefit followed by food waste tracing and the promotion of doggy bags.

For regions with existing treatment of organic waste one of the major improvement areas were the reduction of plastic packaging waste and single use packaging, respectively. Although the impact per kilogram of waste prevented is rather high for substituted disposable toiletry containers, largely due to much higher amount of plastic waste reduced by this option, the promotion and implementation of tap water fountains could result in similar or even greater impacts. In this case additional benefits connected with less littering could be expected.

Compared to all prevention options the separate collection of biowaste showed only minor benefits in terms of GHG emissions and should therefore be the least preferred options in terms of policy implications. If there is no system existing both the shown example of onsite composting but also the introduction of general separate collection seems to be complex and not preferable if only related to the touristic background, based on the comparably low benefits.

Funding

This project received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No. 690452.

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.

Acknowledgements

All persons who have made substantial contributions to the work reported in the manuscript (e.g., technical help, writing and editing assistance, general support), but who do not meet the criteria for authorship, are named in the Acknowledgements and have given us their written permission to be named. If we have not included an Acknowledgements, then that indicates that we have not received substantial contributions from non-authors.

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