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Landfill reactions to society actions: The case of local and global air pollutants of Cerro Patacón in Panama

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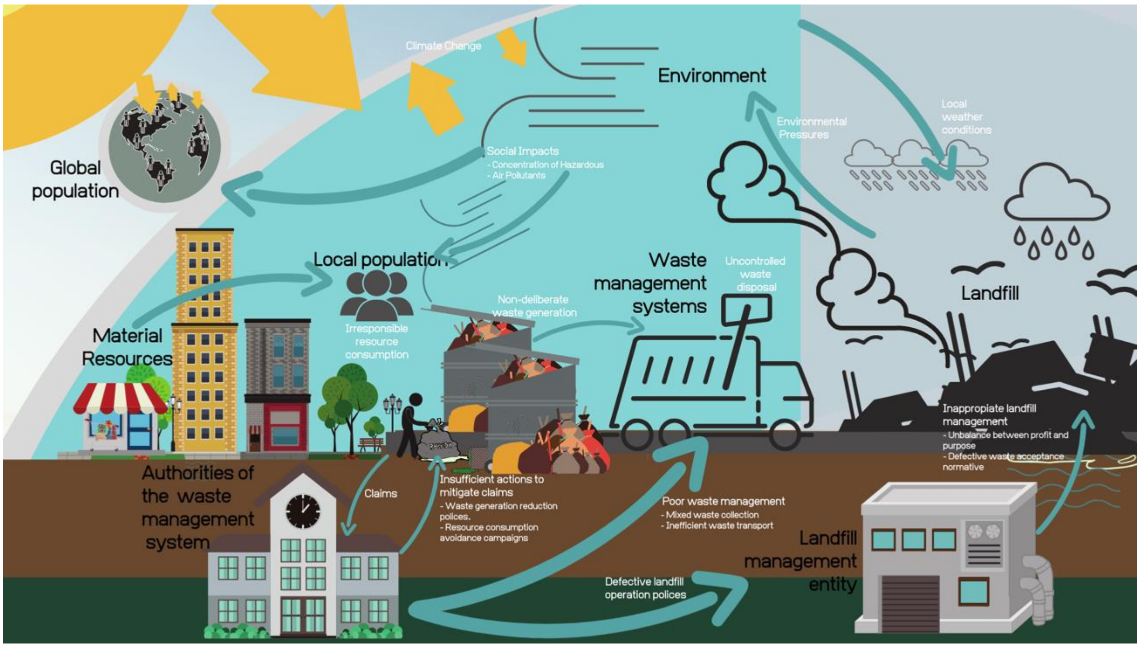
Abstract

This paper studies landfill emissions and the related environmental and health risks in Panama City, with the aim to sensitize the population about the harmful effects of irresponsible resource consumption and non-deliberate solid waste generation that it is disposed of in an uncontrolled manner in landfills. Empirical data on Cerro Patacón, Panama City's landfill was obtained to describe the status of municipal waste disposal. Ten known methane generation models were used to estimate the yearly emission rate of methane from the landfill for a 100-year period starting from its inception in 1986. From the models used, the GasSIM model was chosen to estimate emission rates of six long-term hazardous air pollutants. The AERMOD source dispersion model was used to simulate their atmospheric downwind dispersion by levels of concentration over nearby affected communities; results were mapped in Google Earth. The relative contributions by population of the 32 towns making up Panama City to the forecasted waste generation in 2022 and related hazardous air pollutants emission rates from the landfill were assessed. It was found that Cerro Patacón will generate 45% of the countrywide methane generation by 2022; an average of 47 Gg. The solid waste generated by the 1.5 million inhabitants of Panama City impacts the health of ~73,600 inhabitants in nearby communities through the dispersion of hazardous atmospheric pollutants derived from the landfill. The highest emission rates were from hydrogen sulfide and dichloromethane, which can be largely attributed to the waste generated by the communities of Juan Diaz and Tocúmen. The concentration of Hydrogen Sulfide and Benzene was over the reference concentration (uncertainty factor spanning three orders of magnitude) for all communities and years simulated. The concentration of Vinyl Chloride was over the RfC for all communities and years simulated, except in 2018 for 12 communities.

Keywords: Uncontrolled municipal waste disposal; landfill emissions; air pollution; environmental pressure; Panama City

Graphical abstract



Caption: Interactions between actors of the waste management system and responsibilities for resulting environmental pressures and social impacts.

Highlights

- The anthroposphere non-deliberately generates waste without responsibility allocation
- Uncontrolled waste disposal exerts pressure over the biosphere and atmosphere
- Local and global environmental pressures from Panama City landfill were identified
- Local societal impacts were allocated to 5% of communities affected by the rest 95%
- Data becomes useful when people acknowledge mutual impacts through waste generation

Abbreviations

Phrase	**Abbreviation**
Municipal Solid Waste	*MSW*
Developing country	*DC*
Non-methane Organic Compound	*NMOC*
Hazardous Air Pollutant	*HAP*
Panama District	*PD*
San Miguelito District	*SMD*
Cerro Patacón	*CP*
Reference Concentration	*RfC*
Degradable organic carbon	*DOC*
Zero Order Decay	*ZOD*
First Order Decay	*FOD*

1. Introduction

Landfills require land availability and have negative side effects on the environment and for this reason their placement is often opposed by surrounding residents (Hoornweg and Bhada, 2012). Nonetheless, it remains the most common method for waste disposal worldwide and is considered a reliable and low-cost alternative to final MSW disposal (Caprile and Ripa, 2014; Powrie and White, 2004; Zacharof and Butler, 2004).

In most countries, the costs of using landfills for MSW disposal and household waste collection and transport are covered by public fees that are defined by the (local) authorities based on many different factors, such as household income level, household area, water usage or waste generation. However, the operation and maintenance of landfill sites are often left to private companies who establish a fee per ton of waste, its size depending on the landfill's engineering complexity, that is intended to not only cover the costs but also generate profit (Kinnaman, 2009).

In an effort to promote sustainable waste management, in developed countries more stringent limits on the proportion of organic waste in landfills have driven advanced engineering solutions for landfills sites (Pan et al., 2014). As a result, MSW landfills in developed countries are now mostly used to dispose of inert materials resulting from pre-treated MSW under strict conditions (DEFRA, 2010). This allows the establishment of fixed and transparent landfill operation and maintenance costs due to the relatively homogeneous physicochemical characteristics of the waste (Rigamonti et al., 2016).

DCs, on the other hand, face serious problems of uncontrolled waste disposal due to the absence of separated collection practices and waste treatment before it is disposed of in landfills. In addition to accepting untreated MSW, most landfills also accept sludge, hospital and industrial waste without any pre-treatment and thus retaining bacteriologic activity. This leads to unpredictable landfill operation and maintenance costs, limits the pre-treatment technologies that can be deployed on-site and encourages irresponsible resource consumption (Wilson et al., 2012).

These irregularities affect the local and global environment throughout the landfill's active life, and eventually derive in societal impacts (OECD, 2008). Indeed, landfills are the final stage of the waste management system where uncontrolled waste disposal, combined with local weather conditions—temperature and humidity—and inappropriate management, creates environmental pressures (Sarptaş, 2016). The shifting of landfill costs to the environment is the central concern of the opposition to landfill sites, since such environmental pressures are the chronicle of foretold societal impact.

When the society resents the impacts, claims arise to the waste management authorities for solutions. The authorities need a source of information that will allow them to describe and show society that these impacts are partly caused by the population's own irresponsible resource consumption and non-deliberate waste generation. In this regard, there is an urgent need for reliable estimates and non-steady-state assumptions about the environmental pressures caused by landfill emissions, and there is a general lack of such data in DCs (Bogner and Matthews, 2003).

More in general, there is a need for a holistic perspective of the MSW management system (Chifari et al., 2016; Ziout et al., 2014) in order to understand the extent to which the environmental pressures exerted by landfills is coupled to other aspects of the MSW management system and resource consumption. These insights are also necessary to assess the extent to which claims by the society are caused by uncontrolled waste disposal and inappropriate landfill management, or conversely, are derived from non-deliberate waste generation and irresponsible resource consumption (Chifari et al., 2016; Ziout et al., 2014).

With this information in hand, authorities will be better equipped to create meaningful waste acceptance criteria (DEFRA, 2010) and impose effective operation and maintenance policies on the landfill managers, giving the latter a threshold with which to balance economic profit and socio-ecological interests. Policies to reduce waste generation and campaigns for responsible resource consumption could be fostered in society, and based on the relevant technical, economic and socio-ecological aspects of the entire mechanism of landfill environmental pressure, thus giving the population a reference point for the level of responsibility they have for their own claims.

This paper studies landfill gas emissions in Panama City and the related local and global environmental pressures and human health effects with the aim to raise awareness among the population about the responsibility they bear for the disproportionate impacts mutually exerted through irresponsible resource consumption and non-deliberate solid waste generation that is disposed of in landfills in an uncontrolled manner.

Empirical data on CP, Panama´s main city landfill, was obtained to describe the status of waste disposal. Ten known methane generation models were used to estimate the yearly emission rate of methane from CP for a 100-year period starting in 1986 to generate a picture of the environmental pressure exerted. From the ten models used, the GasSIM model was chosen to estimate also NMOC emission rates, specifically of the six long-term HAPs that presented the higher dispersion rates from CP. The AERMOD source dispersion model, local meteorological conditions data were used to simulate HAP´s atmospheric downwind dispersion by their levels of concentration over nearby affected communities and results were mapped in Google Earth.

To gain insight in the environmental pressure exerted on local communities, the local concentration of HAPs was compared with the inhalation chronic reference concentration RfC [$\mu g\ m^{-3}$], which is an estimate of the level of human exposure through chronic inhalation throughout life that is unlikely to have an appreciable deleterious effect (US EPA, 2011).

Relative contribution to forecasted (2022) MSW generation and HAPs emission rates from the landfill by the 32 towns making up Panama City was assessed to sensitise the population about (potential) environmental and health effects of irresponsible resource consumption and uncontrolled MSW disposal.

2. Case Study

Panama is an upper-middle income DC (The World Bank, 2018) with the most rapid economic growth of all Latin-American cities (Coleman et al., 2014). It is also the world's highest-ranking city in terms of well-being, based on purpose, social, community, physical and financial elements, with 61% of Panamanians thriving in three or more elements (GALLUP, 2014).

The metropolitan area of Panama City consists of two districts, PD and SMD. PD is made up of 23 towns and SMD of 9 towns; a total of 276 communities make up the 32 towns of PD and SMD (INEC, 2000). According to the last census, the population of the metropolitan area was ~1.4 million people, representing 35% of the national population (total land area 75,517 km^2). The national population density is 53 people/km^2, while that of the metropolitan area is 673 people/ km^2; the most crowded in the country (Weitz et al., 2008). Recent studies estimated the MSW generation per capita for PD in 1.55 and for SMD in 1.28 kg inh^{-1}day^{-1} (INECO, 2017).

2.1. Panama City landfill: Cerro Patacón

CP is the landfill for the metropolitan area of Panama City. Its total area is about 130 ha including administration, operation and other non-disposal zones, of which the net disposal area was 53 ha in 2002 (JICA, 2003) and 63 ha in 2016 (AAUD, 2016a). CP receives various waste fractions: MSW (including street sweeping, household waste, and commercial waste assimilable to household waste), sewage and industrial sludge, used tires, clinical waste and construction/demolition (C&D) waste; approximately 60% of this waste is biodegradable (AAUD, 2016b).

Waste disposal has increased by an average of 3.5% per year since CP opened in 1986. From 1986 to 2007, the increment estimates were based on the design stage for achieving its planned lifespan: an average of 1% per year (Weitz et al., 2008). The accelerated immigration and economic growth that began in late 2007/early 2008 (CEPAL, 2009; INEC, 2008) caused a rise in resource consumption that increased the MSW generation rate to an average of 7%. By 2016, waste reception had reached approximately 2,300 tons per day, 40% more than expected for the same year according to its initial design (**Figure 1**).

The initially planned closure date by design was 2037, but actual waste deposited in CP exceeds the design forecast and this rate is expected to continue until at least 2022, the date to which CP closure has been advanced following recent studies of its capacity (AAUD, 2016a). The degree of saturation of CP is evident from a comparison of the maximum elevation of waste slopes reached in 2002 (106 m) to that reached in 2016 (126 m) (JICA, 2003). This was the maximum permitted until a new maximum of 145 m was approved for the same year in order to fulfill the city’s increasing needs for landfill capacity (AAUD, 2016a), despite the augmented risk of hazardous gas dispersion for nearby communities (Paraskaki and Lazaridis, 2005).

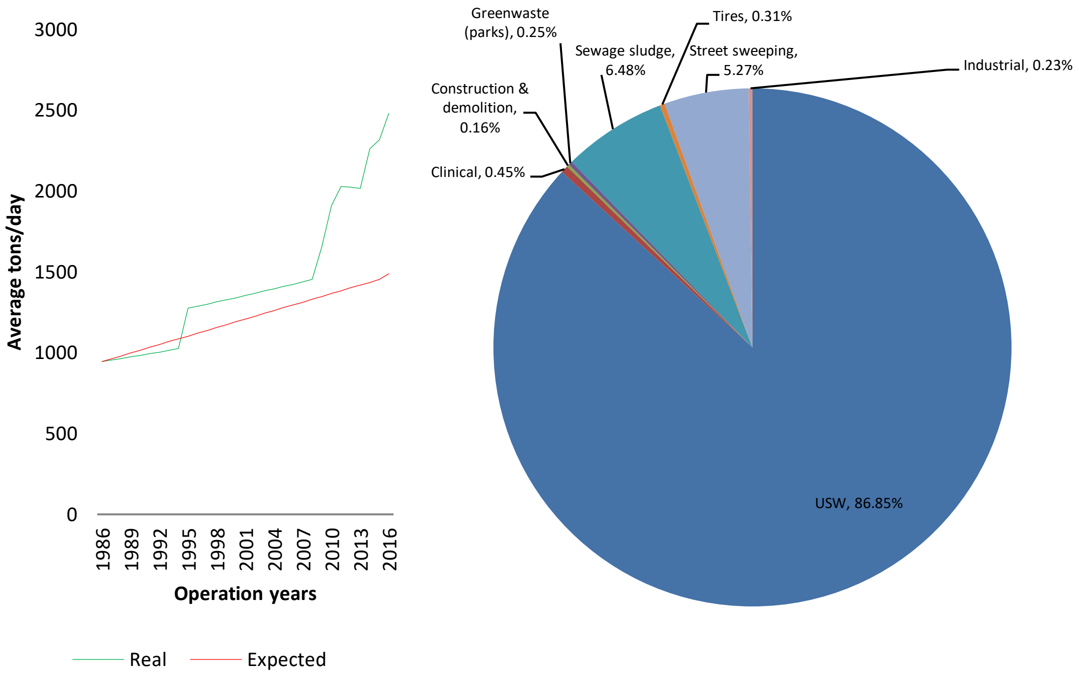


Figure 1. Expected (red line) vs. real data (green line) of the daily waste disposal in tons throughout the lifespan of CP (left), and waste composition in 2016 (right)

2.2. Local and global environmental pressures derived from Cerro Patacón

Biodegradable materials in landfills are decomposed by microbes under anaerobic conditions. This microbial action is highly complex due to unpredictable differences in the degradation rates of the materials that make up solid wastes (Farquhar and Rovers, 1973). The degradation process starts with hydrolysis of solid materials, such as hemicellulose and cellulose, into larger soluble organic molecules, fermentation of which yields organic acids that give rise to methanogenesis. Simple sugars, fats and hemicellulose are easily degraded; cellulose has a moderate degradability, while lignin is resistant to biodegradation under anaerobic conditions. Depending on its availability to bacteria, it can also influence cellulose degradation (Chandler and Jewell, 1980). The intermediate products of landfill waste biodegradation, such as carboxylic acids (R–COOH), carbon dioxide (CO_2) and hydrogen (H_2), generate liquid and gaseous emissions, leachate, and Landfill Gas (LFG) respectively, producing negative environmental and societal impacts (Bogner and Matthews, 2003).

2.2.1.Global Environmental Pressures

2.2.1.1. Methane emissions

The atmosphere is being polluted by LFG acting as a greenhouse gas (GHG). LFG is a combination of methane (CH_4) and carbon dioxide in approximately equal proportions, such that total LFG flow can be assumed to be twice the methane flow. However, traces of other gases have also been found, which constitute no more than 2% of the flow but sum to more than 160 compounds, such as non-methane organic compounds (NMOC), reduced sulfur, and speciated organics (US EPA, 2008). Some NMOCs contain volatile organic compounds (VOC) that can be organic HAPs. Notably, methane and the NMOC nitrous oxide have a Global Warming Potential (GWP) of 21- and 298-times that of CO2 (Frischknecht et al., 2007; Majdinasab et al., 2017).

2.2.2.Local Environmental Pressures

2.2.2.1. Diffusion of toxic gases traces

Local environmental pollution by the waste management sector is now one of the most sensitive social issues in many DCs including Panama (US EPA, 2006a). However, throughout the lifetime of CP, the surrounding community has only been aware of the impacts that their senses can perceive, such as bad smells, noise, vibrations from machines, and surface fires (ATSDR, 2004). The proportion of HAPs present in LFG varies with the local weather conditions and the characteristics of the disposed waste such as the quantity, age, and organic waste content (Paraskaki and Lazaridis, 2005; Sarptaş, 2016). Several NMOCs have been studied to evaluate their HAP properties at the low concentrations at which they are emitted from landfills (Saral et al., 2009). These compounds contribute to air quality deterioration (US EPA, 2006b, 2008), and long–term inhalation throughout life is harmful but imperceptible to people in nearby communities (US EPA, 2011).

2.2.2.2. Landfill surface fires

The lack of proper waste acceptance policies observed in CP is a common problem in DCs (Powell et al., 2016). The combination of uncontrolled waste disposal and inappropriate landfill management (Blais et al., 2010) derives in undesired events like that which occurred in March 2013, when LFG and stockpiles of waste tires combined to produce a heat-generating reaction that caused the worst surface fire seen to date at CP. LFG accumulated in the void spaces in the waste tires (75% volume) and boosted the combustion energy to 28% higher than that of coal, which made the fire difficult to quench both with water and by suffocation (Islam et al., 2009; Pennington, 1996). This fire occupied an area of 30 ha and burned continuously for 10 days, releasing airborne fumes to a radius of 13 km (La Prensa, 2013). No report has yet been published on the environmental damage that resulted from this fire, but studies of similar events have reported increased levels of NMOC, particulate matter, nitrogen oxide, sulfur dioxide, carbon monoxide, polycyclic aromatic hydrocarbon, benzene and dioxin/furan (US EPA, 2008); for the latter, levels reached up to 66 times higher during burning (Weichenthal et al., 2015).

2.2.2.3. Nearby infrastructure explosion risks

As a result of this fire, local authorities concluded that there was a need to assess various issues, including the topography of the landfill's high slopes, where the waste was neither compacted nor covered (AAUD, 2016a). This issue highlighted an environmental pressure resulting from the inappropriate landfill management: risk of explosion by methane migration because the waste was not properly covered with soil.

When LFG is generated, it can be emitted directly into the atmosphere, oxidized to carbon dioxide via an aerobic soil cover, retained within the landfill volume, or migrated laterally to the subsurface. As for the latter case, migration can extend to >300 m in poorly engineered landfills (Kjeldsen, 1996). The most important parameter that controls LFG migration to the surrounding zones is soil permeability to air, which has been found to be strongly influenced by the soil's permeability to water: higher water content results in lower gas migration due to reduced soil porosity (Poulsen et al., 2001).

The soil in CP has a low water permeability (AAUD, 2016a), so despite the high levels of rainfall risk of LGF migration increases during the 3-month dry season (Poulsen et al., 2003). CP is located adjacent to a 4,000-ha protected national park, and the closest residential community is at 50 m distance (Calle 50 squatter settlement) (**Figure 2**). Migration of LFG through soil may result in societal impacts such as explosion hazards in nearby civilian structures, and damage to vegetation due to high concentrations of LFG (Blais et al., 2010; Poulsen et al., 2003).

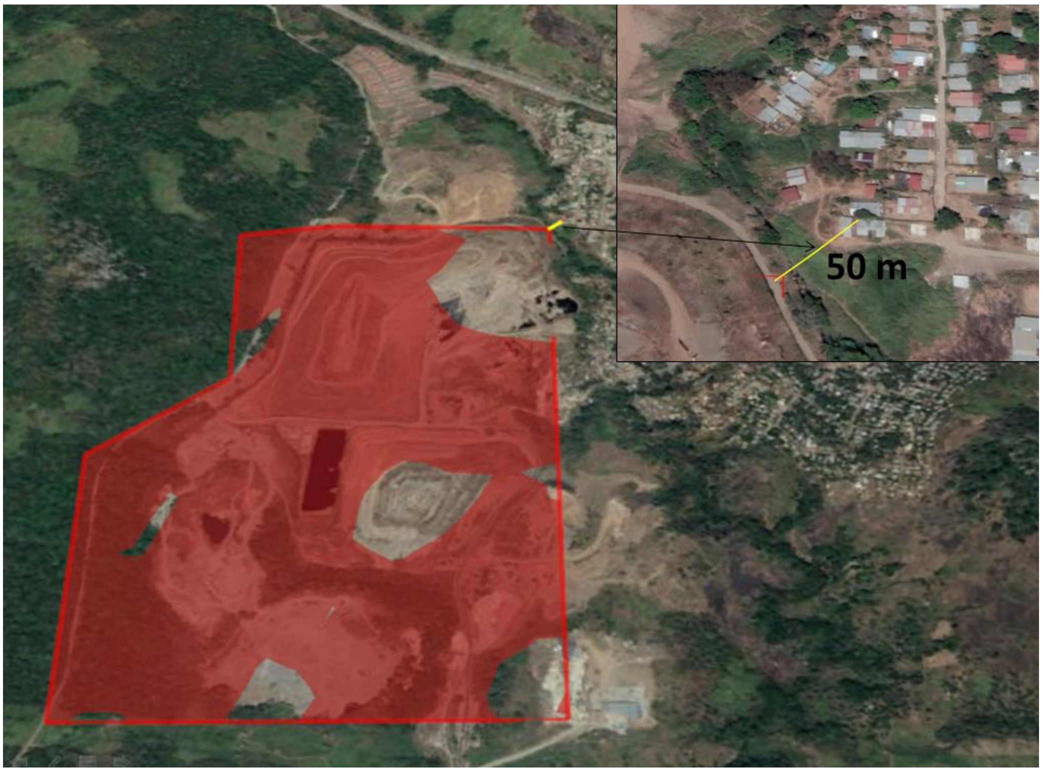


Figure 2. CP landfill area is located adjacent to a protected national park (green zones) and 50 meters from the nearest community.

2.2.2.4. Sludge spills and leachate infiltration in ground and surface water

CP lies in a tropical weather zone, with annual rainfall of 3000 mm and evaporation of 1500 mm. These characteristics are important for CP's waterproofing system, which is not adapted to avoid leachate infiltration to groundwater. Older areas are covered by a 40 to 80 cm layer of clay, while recently constructed areas have a geomembrane and a geotextile, which lacks appropriate systems for gas collection and leachate recirculation (AAUD, 2016a).

Sludge is directly disposed of in a pit, from which a stream flows into a nearby river located no more than 200 m from CP, and from which some communities are supplied with daily-use water. Recent field studies found annual leachate accumulation of >450,000 m^3 in the ponds, with concentrations well above legal limits for various water quality control parameters such as cadmium, lead, nitrate/nitrite, and chlorides in groundwater, and Biological/Chemical Oxygen Demand (BOD5, COD), cadmium, copper, iron, chromium, lead, and sodium in surface water (AAUD, 2016a). The amount and composition of landfill leachate depend on waste type and compaction, landfill hydrology, climate and landfill age, and treatment methods are chosen accordingly (Aziz et al., 2018; Roudi et al., 2018).

2.3. DOC in the waste disposed in CP

As mentioned above, MSW represents approximately 87% of the total waste in CP (**Figure 1**). It is composed of commingled waste, i.e. bulk waste, collected in PD and SMD. The overall composition of MSW when landfilled is 4% metals, 16% paper/cardboard, 18% plastics, 27% organics and 35% others (AAUD, 2016b). It has 80% humidity and 45% carbon content (AAUD, 2016a). The fractions of MSW that contain DOC, which ultimately generates LFG and leachate (IPCC, 2006a), are paper/cardboard, organics (food waste and green-waste) and some materials of the "others" fraction. This is composed of 32 different identifiable fractions, including wood, cellulose, diapers, textiles, glass, etc.

In addition to MSW, DOC containing waste also includes sludge, non-hazardous industrial waste, and street sweeping waste (Sarptaş, 2016). Of the total waste in CP, 73% contains DOC, of which 82% comes from MSW, and the rest from other fractions. Local conditions such as soil cover and weather conditions determine how much of the DOC contained in the waste is available for degradation as methane, as not all biodegradable material can be converted to LFG, and methane accounts for approximately 50% (Oonk, 2010).

3. Materials and methods

Figure 3 shows the methodology applied in this study.

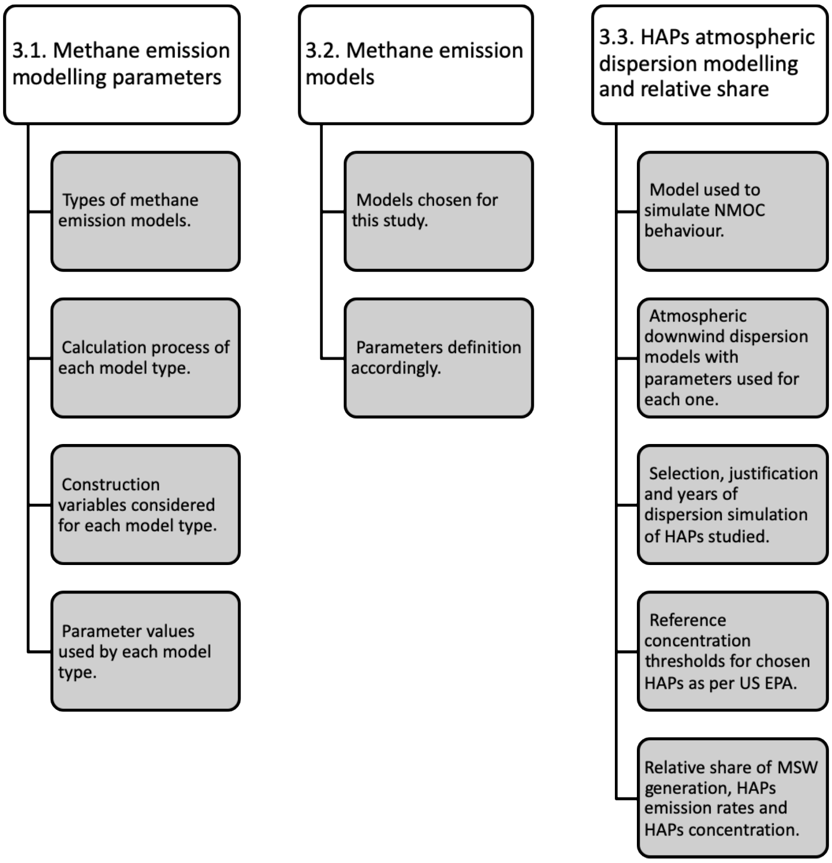


Figure 3. Methodological pathway of this study

3.1. Methane emission modelling parameters

Methane emissions are generally calculated from the methane mass-balance, which is the difference between methane generated and that recovered plus oxidized methane. Oxidation is ~10% of the total methane generation (IPCC, 2006b), and most landfills in DC, including CP, do not have LFG recovery (JICA, 2002; Machado et al., 2009).

Since the extreme complexity of landfill processes makes it impossible to obtain precise data on their outputs, methane emission models are used to assess landfill outputs using field-collected data on a yearly basis, such as the waste disposal volumes and landfill conditions. By modelling the behavior of a landfill site, we can attempt to interpret its environmental pressure (Powrie and White, 2004). Models are constructed using data from different waste categories, which, depending on the country, can differ depending on local definitions of the various waste fractions –e.g. MSW- (Scharff and Jacobs, 2006). There is no perfect model that accurately predicts methane emissions within narrow limits (Oonk, 2010).

Existing models can be empirical, mathematical or numerical. The most well developed and widely used models are the ZOD and FOD empirical models. Second Order Decay empirical models are much less widely used because they are less accurate regarding their complexity (Oonk, 1994), and the reliability of the available mathematical and numerical models (El-Fadel et al., 1989) depends on the availability of input data which is very specific (Majdinasab et al., 2017).

ZOD models are generally used in DC, where there is little or no data on the amount, age and composition of the waste (Oonk, 2010). FOD models are often used in developed countries (Zacharof and Butler, 2004). ZOD models assume that waste produces a fixed amount of LFG for a fixed number of years (Peer et al., 1992), while FOD models assume that LFG forms after waste deposition and that emissions begin to decrease by a given methane generation rate k [yr^{-1}] per half-life $t_{1/2}$ [yr] over a fixed number of years. In this sense, data temporal resolution in this work will be given in a yearly basis. The assumed value of $t_{1/2}$ is 0.693 k^{-1} [yr], in which 50% of the original amount of DOC in waste is biodegraded as methane, depending on the level of moisture, pH, local weather conditions, and the availability of nutrients for methane-generating bacteria, etc. (Ludwig, 2007).

The input for FOD models is usually a “single-phase” or bulk waste amount, -e.g. MSW with homogenous characteristics (Scharff and Jacobs, 2006)-, and thanks to the increasing availability of this type of data, FOD models are increasingly preferred over ZOD models in DCs landfills (Bogner and Matthews, 2003). Moreover, by conducting waste characterization studies, FOD “multi-phase” models can be used to assign parametric values for *k* and methane generation potential L_0 for each waste fraction characterized (Mou et al., 2015). In this case, depending on the model, both parameters can be assigned for different material decay rates, for which the characterized waste proportions must be adapted according to the model’s specific waste input categories -e.g. slow-, medium- and fast-degrading-, and local weather conditions -i.e. wet, dry, tropical, temperate, etc.- (Krause et al., 2016).

The methane generation potential L_0 [m^3 CH_4 Mg^{-1} waste] indicates the total amount of methane emitted from DOC during the methanogenesis phase of waste degradation for bulk waste with a particular composition (i.e. ZOD and FOD models), or for different fractions of waste -i.e. FOD multi-phase models-. Some models express L_0 in terms of mass as Biochemical Methane Potential (BMP) [Kg CH_4 Mg^{-1} waste] (Mou et al., 2014). Others express methane yield as a variable that describes the percentage of carbon actually degraded depending on waste composition, the Biodegradable Carbon (BDC) [kg C Mg^{-1} waste]. In this study, methane generation potential for all models used is expressed as L_0 [m^3 CH_4 Mg^{-1} waste] using a conversion factor of 1.33 kg CH_4 kg C^{-1} to convert carbon mass to methane mass and 0.714 kg CH_4 m^{-3} CH4 to convert methane mass to methane volume (Krause et al., 2016).

In CP, ~73% of DOC contained in disposed waste is biodegradable from which ~10% is aerobically degraded by oxidation occurring in landfill soil cover, and ~90% is anaerobically degraded through the methanogenesis process (IPCC, 2006b). Not all anaerobically converted DOC is bioavailable, it is defined by landfill conditions (Krause et al., 2016). However, suitable landfill conditions in CP allow all anaerobically degraded DOC to be considered bioavailable. Only ~2% of bioavailable anaerobically biodegraded DOC from disposed waste is trace gases, the rest is methane and carbon dioxide in equivalent proportions (**Figure 4**). ZOD and FOD models use the Bioavailable Carbon Factor (BAC_f), dissimilation factor (Ϛ) or the Methane Conversion Factor (MCF), all of which account for the availability of the degradable material due to external conditions (Krause et al., 2016). This value is taken as 100% for anaerobic unmanaged solid waste wet disposal sites (IPCC, 2006a; Scharff and Jacobs, 2006), which is the case of CP.

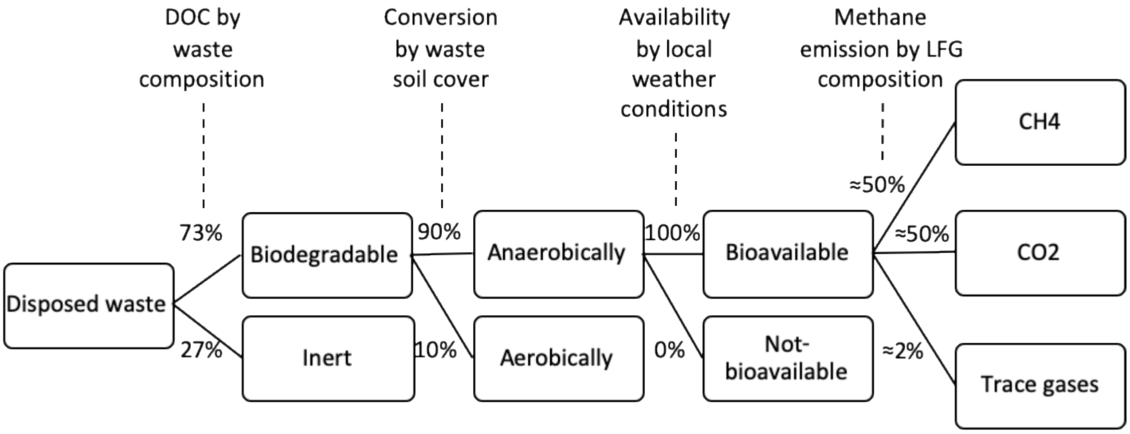


Figure 4. Conditions defining methane emission proportions according to the DOC content of waste disposed in Cerro Patacón.

3.2. Methane emission models

In this study, 2 ZOD, 4 FOD and 4 FOD multi-phase models were used to estimate the average methane generation for CP (**Table 1**). The waste characterization study performed in 2016 was used to benchmark the waste composition for all years modelled since 1986; no other characterization study had previously been carried out in CP (AAUD, 2016b).

For one FOD model (the TNO-model) and all four FOD multi-phase models, waste categories were re-coded to match the input format required by each model and are given as a proportion of the total disposed waste. Parameters k and L_0 were obtained from default values according to each model (Krause et al., 2016; Majdinasab et al., 2017; Mou et al., 2014), for which a weighted-average has been obtained for bulk waste representing 100% of the waste disposed in CP –i.e. commingled waste-; the value obtained serves as a guide for future modeling with no characterization study available.

We computed the yearly average L_0 for the weighted-averages of the bulk waste input values of the five models mentioned above (97 m^3 CH4 Mg^{-1} of waste), and used this as the L_0 value for the remaining models (T&R, LandGem, SWANA FOD, SWANA ZOD, and EPER Germany), as their input is a bulk waste value. Others studies have reported similar L_0 values for bulk waste in wet/tropical landfills (Bentley et al., 2005; Faour et al., 2007; Machado et al., 2009); their k values are the default (Krause et al., 2016; Oonk, 2010; SWANA, 1997).

Table 1. Methane emission models and parameters used for CP

Model		**Waste**		**Parameters**	
Name and reference	*Type*	*Categories*	*Proportions*	*k (y-1)*	*L0 (m3 CH4/Mg Waste)*
		Weighted-average for bulk waste	*100%*	*0.28*	*78*

GasSim (Attenboroμgh et al., 2002)	FOD multi-phase	Paper/Cardboard	19%	0.12	116
		Textiles	6%	0.12	113
		Miscellaneous	6%	0.12	113
		Putrescible	28%	0.69	115
		Fines	8%	0.08	111
		Sludge	6%	0.69	34
		Non-degradable	27%	0.00	0
IPCC (Pipatti et al., 2006)	FOD multi-phase	*Weighted-average for bulk waste*	*100%*	*0.13*	*80*
		Food	18%	0.40	70
		Garden	6%	0.17	93
		Paper	19%	0.07	187
		Wood	5%	0.04	201
		Textiles	6%	0.07	112
		Disposable nappies	4%	0.17	112
		Sludge	12%	0.17	23
		Industrial	1%	0.17	70
		Plastics, other inert	28%	0.00	0
Afvalzorg (Mou et al., 2015)	FOD multi-phase	*Weighted-average for bulk waste*	*100%*	*0.13*	*78*
		C&D waste	1%	0.03	15
		Street cleaning	5%	0.03	27
		Coarse household waste	1%	0.03	112
		Sludge and composting waste	7%	0.21	35
		Refuse	2%	0.03	88
		Household waste	46%	0.21	127
		Vegetable, fruit and garden waste	6%	0.21	66

		Wood	5%	0.10	177
		Inorganic	27%	0.00	0
Central America (Weitz et al., 2008)	FOD multi-phase	*Weighted-average for bulk waste*	*100%*	*0.12*	*84*
		Food Waste	16%	0.23	68
		Fast-decaying green waste	6%	0.23	68
		Other fast-decay organic waste	25%	0.23	68
		Slower-decay green waste	2%	0.03	207
		Paper and Cardboard	14%	0.03	207
		Wood Waste	3%	0.03	207
		Rubber, Leather, Textiles, Bones	7%	0.03	207
		Inorganic waste	28%	0.00	0
TNO (Oonk, 1994)	FOD	*Weighted-average for bulk waste*	*100%*	*0.07*	*164*
		C&D waste	1%	0.10	20
		Street cleaning	5%	0.10	168
		Coarse household waste	1%	0.10	242
		Sludge and composting waste	7%	0.10	168
		Refuse	2%	0.10	168
		Household waste	46%	0.10	242
		Vegetable, fruit and garden waste	6%	0.10	242
		Wood	5%	0.10	242
		Inorganic	27%	0.00	0
T & R (Tabasaran and Rettenberger, 1987)	FOD	Bulk waste	100%	0.03	97

LandGem (US EPA, 2001)	FOD	Bulk waste	100%	0.70	97
SWANA (SWANA, 1997)	FOD	Bulk waste	100%	0.15	97
SWANA (SWANA, 1997)	ZOD	Bulk waste	100%		97
EPER Germany (Scharff and Jacobs, 2006)	ZOD	Bulk waste	100%		97

3.3. HAP atmospheric dispersion modelling and relative share

The Gaussian Plume Model of Atmospheric Dispersion AERMOD (US EPA, 2018) was used to simulate NMOC behavior and atmospheric downwind dispersion based on local meteorological data according to local weather conditions up to 10 km around CP, which is considered as an area-shaped air polluting source. The annual average concentrations of six long-term standard hazardous landfill-derived NMOCs that are considered VOC and HAP are reported in Table 2: Hydrogen Sulphide (H_2S), Vinyl Chloride ($H_2C{=}CHCl$), trichloroethene or trichloroethylene (TCE)(C_2HCl_3), Benzene (C_6H_6), Dichloromethane or methylene chloride (CH_2Cl_2) and Chloroform ($CHCl_3$) (Gioia et al., 1995; Kanabkaew et al., 2014; Paraskaki and Lazaridis, 2005; US EPA, 1991, 1995, 2000).

HAP emission rates from the FOD model GasSim [g s^{-1}] were used as input for AERMOD per m^2 of net waste disposal area at CP for the years 2002, 2018 and 2022. GasSim was preferred over other models because it has been designed with a special inclination toward the potential health effects on the population living near and working on landfills (Golder Associates, 2012). This is of interest given the reported statistical relationship between landfill proximity and adverse effects on human health (Elliott et al., 2001). The resulting concentrations were compared to exposure threshold values of RfC (Dankovic et al., 2015; NITE, 2017; US EPA, 2003) (**Table 2**).

Table 2. Name, formula, RfC [μg m^{-3}] (uncertainty factor spanning three orders of magnitude), relative proportion of the total landfill HAPs emitted and emission rates [g s^{-1}] (at 1 Atm and 25ºC) for the years 2002, 2018 and 2022 of simulated HAPs at Cerro Patacón.

HAP				**Year of simulation**		
				2002	**2018**	**2022**
Name	*Formula*	*RfC*	*Proportion*	*Emission rates*		
Hydrogen Sulphide	H_2S	10	5%	0.1071	0.2	0.262
Vinyl Chloride	$H_2C{=}CHCl$	100	2%	0.0398	0.075	0.097
Trichloroethene	C_2HCl_3	600	2%	0.0321	0.06	0.078
Benzene	C_6H_6	30	4%	0.075	0.14	0.183
Dichloromethane	CH_2Cl_2	600	5%	0.1038	0.194	0.254
Chloroform	$CHCl_3$	100	0.01%	0.0003	6E-04	8E-04

Meteorological data input for AERMOD was obtained from the Albrook station (WMO, 2001); others parameters are described in **Table 3**.

Table 3. Site-specific and meteorological parameters for AERMOD simulation

Parameter	**Value**
CP approximate location (UTM)	17P 656961E 1000248N
Sensible heat flux (W/m2)	40 (Hamza and Muñoz, 1996)
Surface friction velocity (m/s)	0.72 (Cheng and Georgakakos, 2011)

Bowen ratio	0.48 (Lewis, 1995)
Albedo	0.17 (McEvoy et al., 2012)
Average wind speed (m/s)	1.67 (Hidromet, 2017)
Average wind direction (degrees)	315 (Hidromet, 2002)
Average relative humidity (%)	90 (Hidromet, 2017)
Average cloud cover (tenths)	5 (Hidromet, 2017)
Precipitation (mm/hr)	0.34 (Hidromet, 2017)
Monin-Obukhov length (m)	92.39 (Pino et al., 2006)

Relative contributions to MSW generation and related HAPs emission rates from CP by the 32 towns making up Panama City were assessed. HAPs concentrations within affected communities close to CP were allocated by population relative share to assess environmental pressures within the same year.

4. Results and discussion

Results on methane emission behaviour are shown for the 10 models used from the opening of CP in 1986 to 2086; an overall model average behaviour is also presented (**Figure 5**). The behaviour of the total NMOC emissions (**Figure 6**) and 6 major HAPs (**Figure 7**) is shown for a period of 46 years (1986-2032). Simulations of HAP atmospheric dispersion over the area surrounding CP are shown (**Figure 8**) for the years 2002, 2018 and 2022, along with the total population at risk and the HAP concentrations in the affected communities. Radar plots to allocate the MSW generation and HAP emission rates from CP by towns and to allocate EP through HAP concentrations by towns and affected communities were forecasted to 2022 (**Figures 9 and 10).**

4.1. Methane emissions

Oonk (2010) reported differences between model estimations of more than 10-fold. The Central America model reached its highest methane levels by 2023, 92 Gg CH_4, while the TNO model reached its highest value by 1998, 9 Gg CH_4 (**Figure 5**). The LandGEM model behaved in a similar way to that reported by Plocoste et al., (2016) when applied to a tropical area.

The methane emissions of the TNO, GasSIM and LandGEM models had the same order of magnitude to the results reported by Scharff and Jacobs, (2006) for the same landfill; the same is observed for the AFVALZORG, IPCC and LandGEM models in Mou et al., (2015), and for the T&R, TNO, IPCC and LandGEM models in Sarptaş (2016).

Since EPER Germany estimates methane emissions independent of the amount of the methane already generated (Majdinasab et al., 2017), the methane emission curve falls to 0 Gg CH_4 by 2022, the year reported for landfill closure. Similar behavior can be observed in another ZOD model, SWANA, although methane emissions did not fall so dramatically, but rather remained almost constant after the highest peak of 43 Gg CH_4 in 2022, which is the typical behavior of ZOD models (Oonk, 2010).

The SWANA FOD model reaches its peak long before the other models because it considers the effect of waste age on methane emissions by simulating a direct relationship between the methane generation potential (L_0) and the waste decay rate (k). Thus, L_0 becomes over-sensitive to k, and reaches its maximum in the last year in which the interaction between L_0 and k shows a curve-increasing result (Majdinasab et al., 2017).

The behavior of the average curve is similar to that of the LandGEM model, and the peak value by 2022 is very close to the last peak (46.81 Gg CH_4) reported in 2011 by the National Environmental Authority of Panama in their Second National Communication to the United Nations Framework Convention on Climate Change (UNFCCC); however, there is difference of eleven years between the results (ANAM, 2011).

Globally, over 60% of methane emissions comes directly from human activities such as coal mining, the petroleum industry, and MSW landfills, with the remainder being accounted for by other indirect activities and natural sources (US EPA, 2010, 2002). In Panama, cattle raising, manure management, and wetlands account for 40% (80 Gg year^{-1}) of methane emissions (Dennehy et al., 2017; Goopy et al., 2018; INEC, 2014a, 2014b; Philippe and Nicks, 2015). There is no coal mining or petroleum industry, so direct anthropogenic methane emissions, which represent 60% (120 Gg year^{-1}), come from landfills alone; ~45% is emitted by the CP metropolitan landfill (AAUD, 2016b).

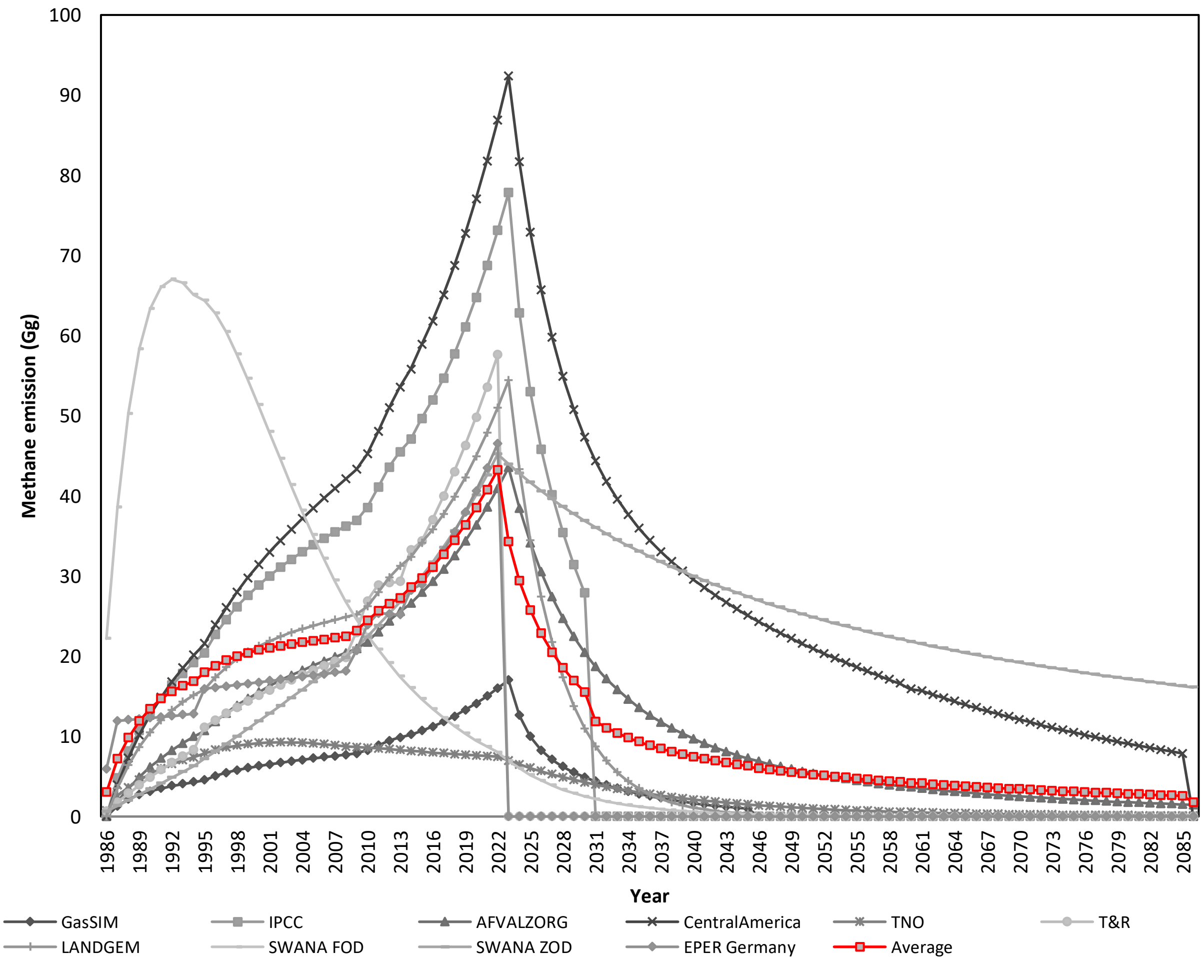


Figure 5. Annual methane emission rates for the ten models used and the average curve

4.2. HAP emissions

According to the US EPA (2005), landfills with NMOC emissions of >50 Mg yr^{-1} require an LFG capturing mechanism. At CP, the total HAP emission per year, which represent 52% of all NMOC generated, reaches a peak of 194 Mg by 2022 (**Figure 6**), so CP should have had an LFG capture system since its opening in 1986. The 6 HAPs simulated in this study account for more than 18% of the total HAP per year, and less than 1% of the total LFG. Hydrogen Sulphide and Dichloromethane had the highest emissions in 2022, both with a peak of ~9 Mg, followed by Benzene, Vinyl Chloride and Trichloroethene, with 13, 4 and 2 Mg, respectively. Chloroform had the lowest emission for 2022 (26 kg, **Figure 7**).

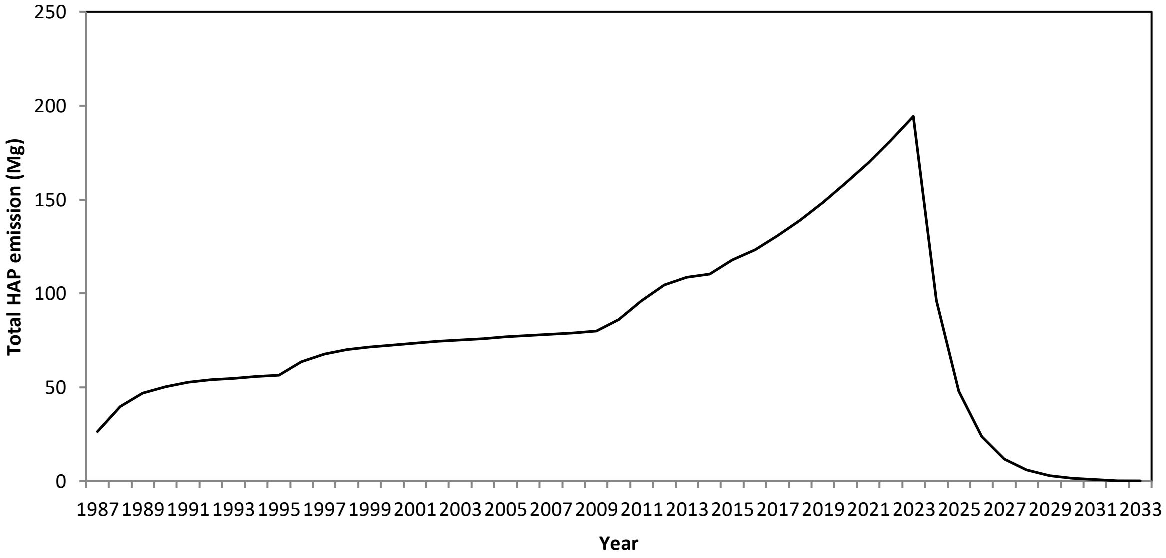


Figure 6. Total HAP emission (Mg) per year in Cerro Patacón.

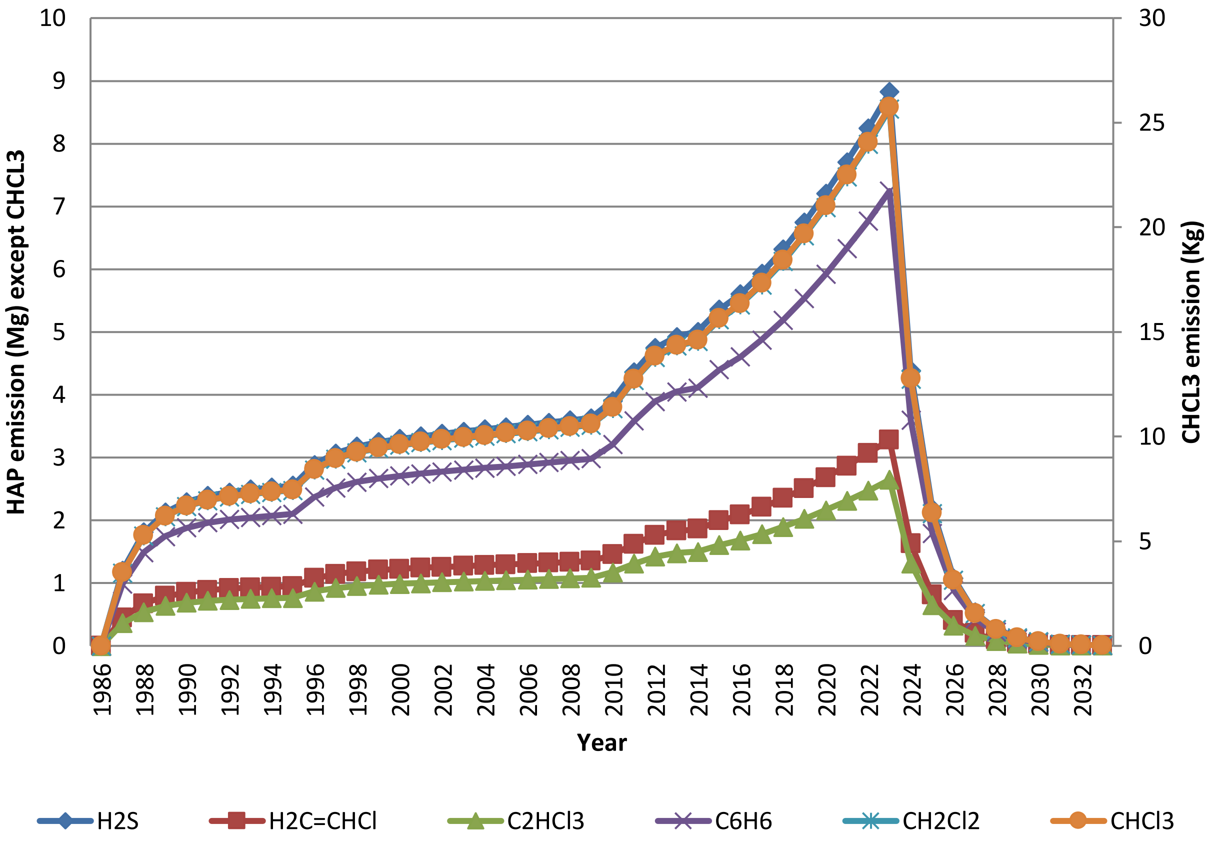


Figure 7. Emissions per year in Cerro Patacón for the six HAPs simulated for the period 1986-2032.

4.2.1.HAP concentrations

A total of 16 communities close to CP, with ~73,600 inhabitants, are affected by low-concentration, long-term HAP emissions (**Table 4**). Communities 1 and 2 are squatter settlements very close to CP, while community 3 is an old low-income rural community that was settled before CP came into existence. Communities 4–8 are recently settled middle-high income condos, while the remainder are older middle-income communities.

Table 4. Approximate population size of nearby communities, and their distance from Cerro Patacón (INEC, 2010)

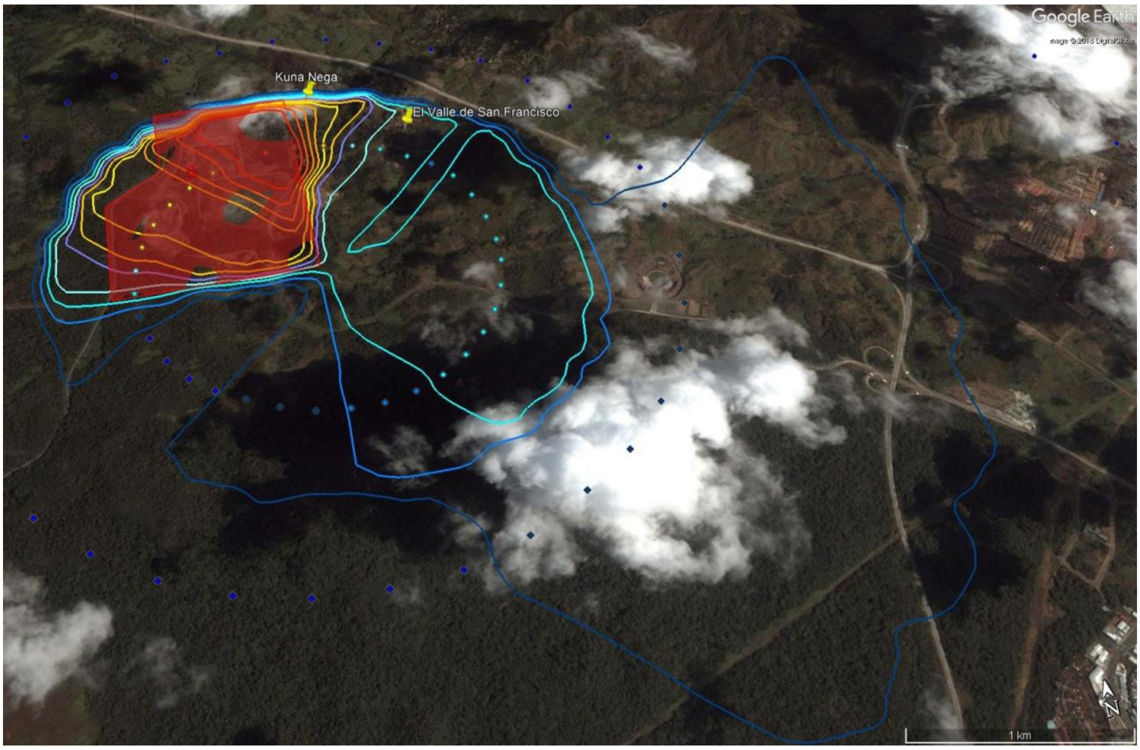
#	**Community**	**Population**	**Distance range (m)**

1	Kuna Nega	2000	<250
2	Calle 50	100	<250
3	El Valle de San Francisco	3000	<250
4	P.H. Altamira Gardens	500	1000-2000
5	P.H. Las Huacas	500	2000-4000
6	P.H. 4 Horizontes	500	5000-7000
7	Paseo Dorado	500	2000-4000
8	Dorado Lakes	500	5000-7000
9	Carrasquilla	5000	5000-7000
10	La Loma	1000	5000-7000
11	El Ingenio	5000	5000-7000
12	Club X	5000	5000-7000
13	Betania	40000	5000-7000
14	Hato Pintado	5000	5000-7000
15	Panacasa	3000	5000-7000
16	Villa de las Fuentes 1	2000	5000-7000
	Total	*73600*	

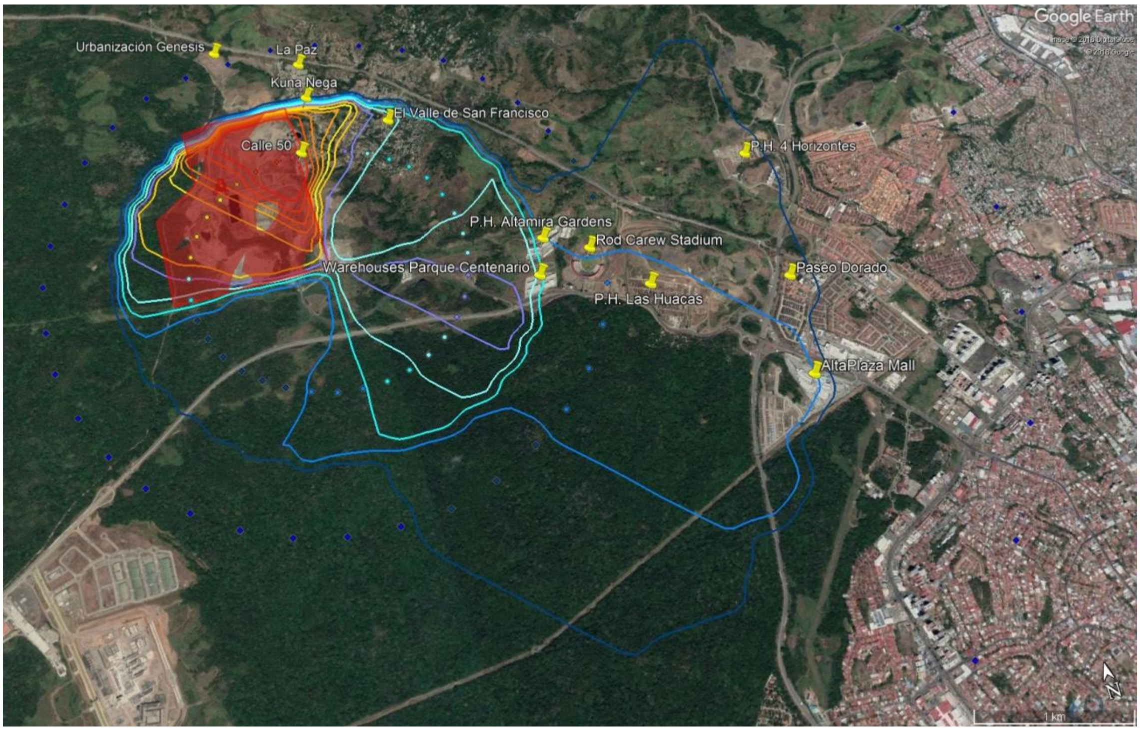
Table 5 show the concentration of the 6 HAPs studied in the 16 communities close to CP. Community numbering refers to **Table 4**.

Figure 8 shows simulations of the HAP atmospheric dispersion derived from CP (red spotted area) up to 10 km around for the years 2002, 2018 and 2022. HAP dispersion is shown from the highest to the lowest concentration according to average air direction patterns, where the orange pattern is the highest concentration and the blue one is the lowest.

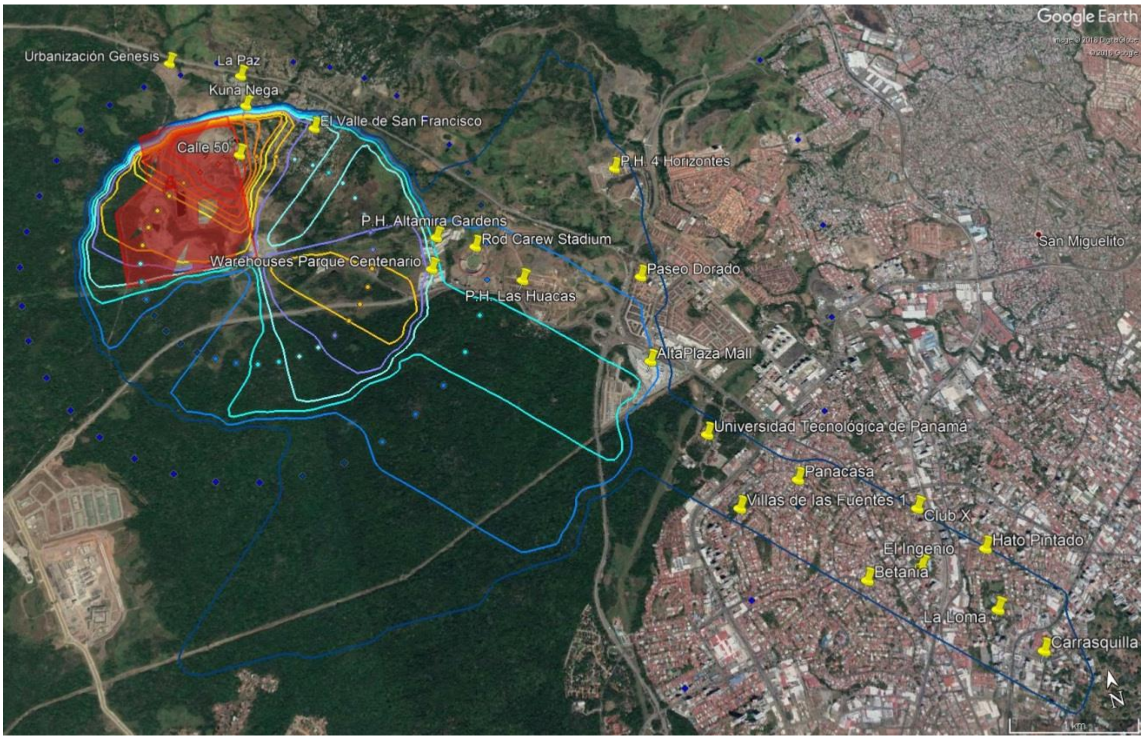
In 2002, the waste disposed of in CP emitted HAPs that affected communities 1 and 3 (these were the only communities existing at that time, **Table 5**), with HAP dispersion extending as far as 4 km from CP to the inhabited areas (**Figure 8a**). In 2018, there was a well-marked difference in population growth around this area (**Figure 8b**), and in this year the landfill produced HAPs emissions that dispersed within the same distance as in 2002; although the concentrations were higher in some of the earlier populated areas (**Table 5**). By 2022, HAP emissions may extend as far as 10 km from CP (**Figure 8c**), thus affecting many more communities (**Table 5**).



a)



b)



c)

Figure 8. AERMOD Dispersion patterns of HAP according to average local meteorological conditions for the years 2002 (a), 2018 (b), and 2022 (c).

The concentration of Hydrogen Sulfide in 2002 is consistent with the field measurement results from JICA (2003b) and is over the RfC (**Table 2**) for all communities and years simulated (**Table 5**). The concentration of Vinyl Chloride is over the RfC for all communities and years simulated, except for communities 5, 6, 7 and 8 to 16 by 2018. The concentration of Benzene is over the RfC for all communities and years simulated. The concentration of Dichloromethane is over the RfC for community 2 in 2018 and 2022. The concentration of Trichloroethene and Chloroform are under RfC for all communities and years simulated. Risk of falling over RfC values beyond 2022 for some HAPs increases with rapid population growth and unpredictable political decision-making that could postpone the closure of CP or continue its operation until the closure year planned under the original design (2037).

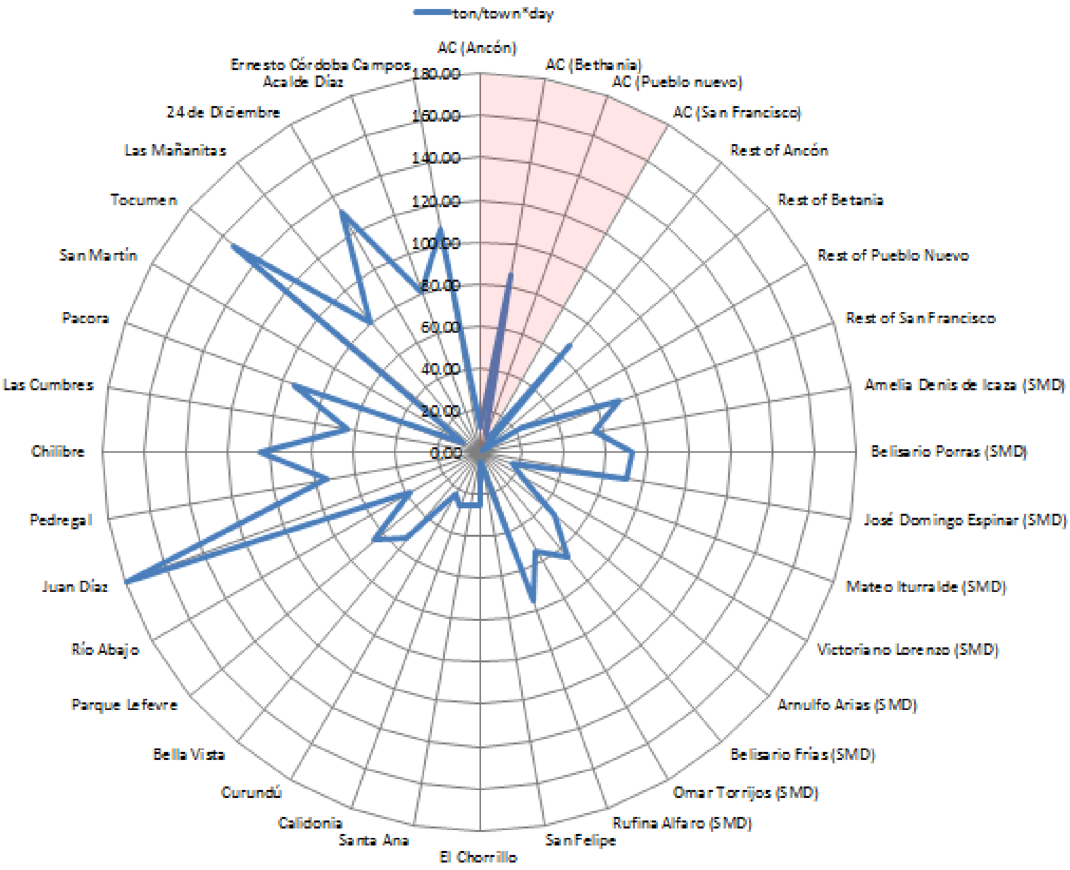
Table 5. HAP concentrations in communities close to CP per year [$\mu g\ m^{-3}$].

HAP	***Hydrogen Sulphide***	***Vinyl Chloride***	***Trichloroethene***	***Benzene***	***Dichloromethane***	***Chloroform***
Community #	*Concentrations*					
			2002			
1/3	**0.4129**	**0.1535**	0.1376	**0.3213**	0.4446	0.0134
			2018			
1/3	**0.5834**	**0.2169**	0.1749	**0.4086**	0.5655	0.0015
2	**0.7001**	**0.2603**	0.2099	**0.4903**	**0.6786**	0.0020
4	**0.2917**	**0.1085**	0.0875	**0.2043**	0.2827	0.0009
5/7	**0.1750**	0.0651	0.0525	**0.1226**	0.1696	0.0005
6/8-16	**0.1167**	0.0434	0.0350	**0.0817**	0.1131	0.0003
			2022			

1/3	**0.6114**	**0.2274**	0.1834	**0.4282**	0.5926	0.0018
2	**0.7337**	**0.2729**	0.2200	**0.5138**	**0.7112**	0.0021
4	**0.3057**	**0.1137**	0.0917	**0.2141**	0.2963	0.0009
5/7	**0.1834**	**0.6822**	0.0550	**0.1285**	0.1778	0.0005
6/8-16	**0.1223**	**0.4548**	0.0367	**0.0856**	0.1185	0.0004

4.2.2. Attribution of landfill emission rates to MSW generation by town and town exposure to dispersed HAPs.

Figure 9 shows the forecasted MSW generation from PD and SMD in 2022 and the corresponding HAPs emission rates from CP by town. Approximately 95% of the MSW expected to be generated by the 32 towns from PD and SMD (1.5 million inhabitants) in 2022 will exert environmental pressure on 16 communities belonging to 4 towns (73,600 inhabitants) of the PD through HAPs atmospheric dispersion. By weighing the HAPs emission rates with the expected MSW generated by town, it is observed that the highest emission rates are for Hydrogen Sulphide and Dichloromethane from the MSW generation of Juan Diaz and Tocúmen followed by Chilibre, Pacora, 24 de Diciembre and Ernesto Córdoba Campos. Within the affected communities, the HAP emission rates from the MSW generated by community 13 exert the highest EP.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AC (Ancón)
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Rest of Ancón
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Rest of Pueblo Nuevo
Rest of San Francisco
Amelia Denis de Icaza (SMD)
Belisario Porras (SMD)
José Domingo Espinar (SMD)
Mateo Iturralde (SMD)
Victoriano Lorenzo (SMD)
Arnulfo Arias (SMD)
Belisario Frías (SMD)
Omar Torrijos (SMD)
Rufina Alfaro (SMD)
San Felipe
El Chorrillo
Santa Ana
Calidonia
Curundú
Bella Vista
Parque Lefevre
Río Abajo
Juan Díaz
Pedregal
Chilibre
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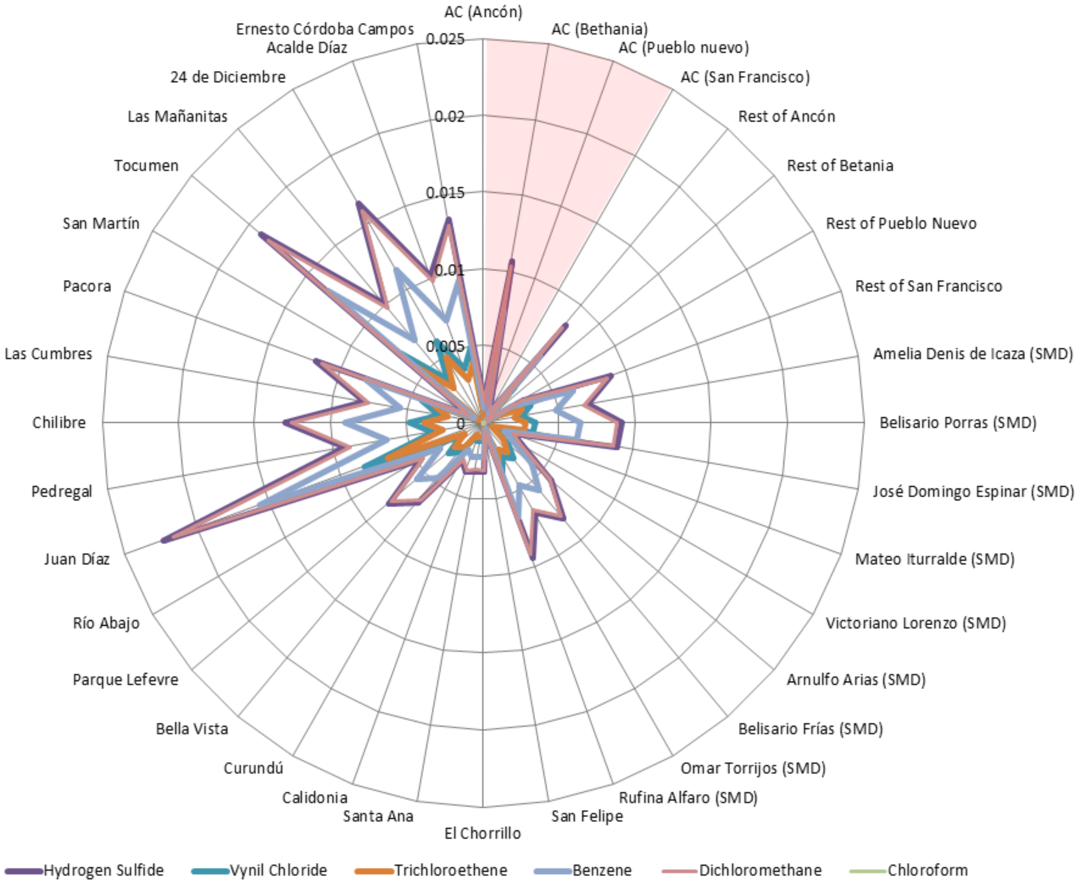
Pacora
San Martín
Tocumen
Las Mañanitas
24 de Diciembre
Ernesto Córdoba Campos
Acalde Díaz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AC (Ancón)
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Rest of Pueblo Nuevo
Rest of San Francisco
Amelia Denis de Icaza (SMD)
Belisario Porras (SMD)
José Domingo Espinar (SMD)
Mateo Iturralde (SMD)
Victoriano Lorenzo (SMD)
Arnulfo Arias (SMD)
Belisario Frías (SMD)
Omar Torrijos (SMD)
Rufina Alfaro (SMD)
San Felipe
El Chorrillo
Santa Ana
Calidonia
Curundú
Bella Vista
Parque Lefevre
Río Abajo
Juan Díaz
Pedregal
Chilibre
Las Cumbres
Pacora
San Martín
Tocumen
Las Mañanitas
24 de Diciembre
Ernesto Córdoba Campos
Acalde Díaz
Hydrogen Sulfide
Vynil Chloride
Trichloroethene
Benzene
Dichloromethane
Chloroform

Figure 9. Expected MSW generation in 2022 (ton town^{-1} day^{-1}) (upper graph) and corresponding HAPs emission rates (g s^{-1}) from CP by Panama City towns. Towns marked with SMD in parenthesis belong to the San Miguelito District, all the others to the Panama District. The reddish zone marks communities (AC) closest to the landfill.

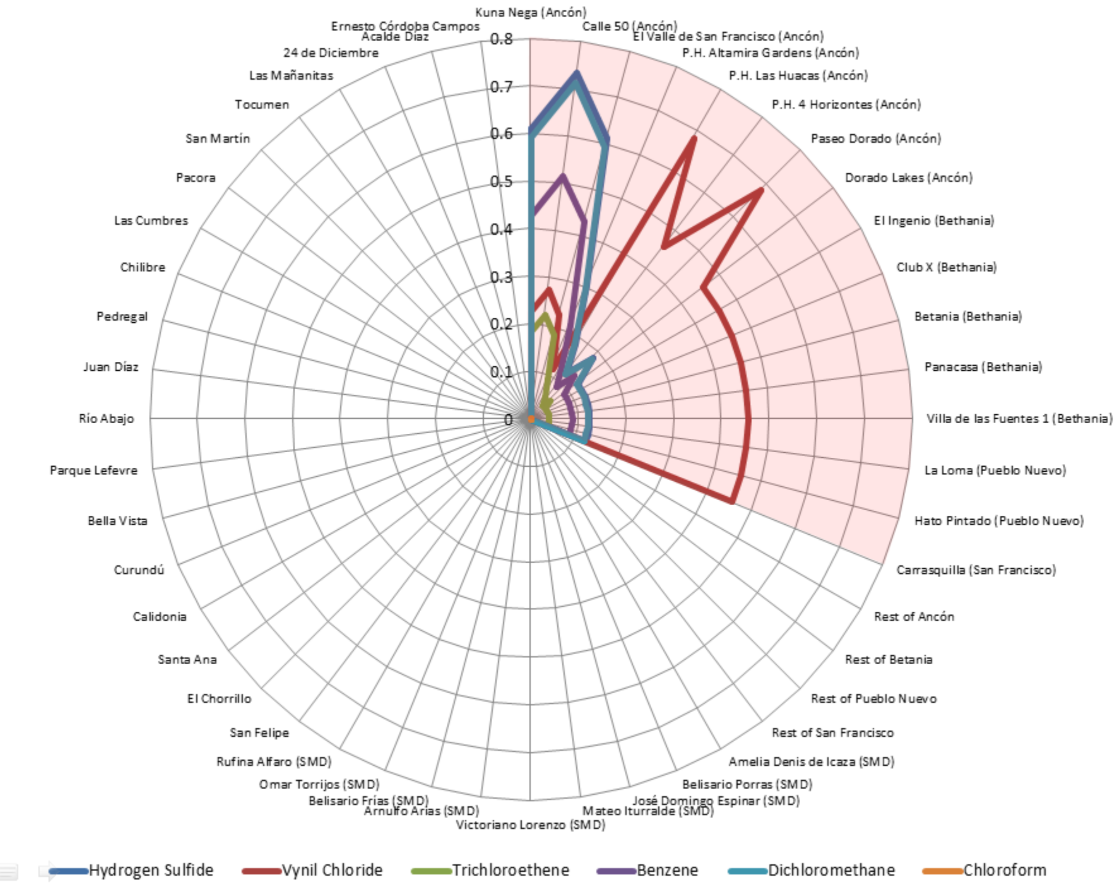

Figure 10. Expected community exposure to dispersed HAPs (in µg m^{-3}) (upper graph) in 2022. Only towns with an exposure exceeding the RF are shown. Towns with SMD in parenthesis belong to the SMD, all the others to the PD.

Figure 10 shows expected exposure to dispersed HAPs emitted from the landfill in 2022 for affected communities of the PD (HAPs concentration exceeding RfC). Community 2, located in the town Ancón, is subject to the highest environmental pressure from exposure to dispersed Hydrogen Sulphide and Dichloromethane, while the communities 5 and 7, located in the same town, are exposed to the highest concentration of Vinyl Chloride. The community 2 is also exposed to the highest concentration of Benzene.

5. Conclusions and recommendations

To reduce global environmental pressures, many DCs including Panama, ratified the Paris Agreement with an action plan based on a carbon pricing mechanism mostly applied to energy generation (Elkahwagy et al., 2017), but waste generation is often left aside because, contrary to energy, measurement mechanisms are rare. Climate change awareness is high but it works in a short life natural attention cycle difficult to engage with for long time because there are not sensible strategies for engagement (Spence et al., 2012). Taking actions on landfills will be necessary to reduce the global environmental pressures exerted.

People engage better with health than with environmental issues, this study presents quantitative results to relate societal impacts with global environmental impacts –e.g. Climate Change- through local environmental pressures –e.g. health hazards- as an alternative to raise awareness in population on the global harms their lifestyles contribute to. With the right engagement tools, awareness on the personal repercussion could contribute to long-term global environmental awareness.

It was found that CP will average a methane emission rate of ~47 Gg by 2022, thus generating ~45% of the total countrywide methane emissions. Waste generated by 1.5 million inhabitants directly impacts 73,600 inhabitants of the Panama district through the dispersion of HAPs derived from landfill. The highest emission rates were from Hydrogen Sulfide and Dichloromethane, allocated to the waste generated by the communities of Juan Diaz and Tocúmen. Calle 50, a squatter settlement nearby CP, bears the highest environmental pressure with the highest concentration of Hydrogen Sulphide and Dichloromethane; it also receives environmental pressures from the highest concentration of Benzene.

Landfill sites will become even more necessary during the coming years due to rapid urbanization in DCs. However, as an alternative to open dumping, restructuring entire waste management systems by introducing waste separated collection, treatment before final disposal, efficient waste transport logistic and better engineered landfills and their appropriate management are required to support healthy socio-economic growth (Bogner and Matthews, 2003).

Despite the fact that local environmental pressures and societal impacts derived are directly proportional to the waste generating society; the path to reliable waste management systems in DCs is a technical and political issue that hardly depends on the society but on the government Will to solve the issue. However, waste generation is it source; the part of the waste management system that society bear responsibility for.

Same way, landfills are the sink of waste management systems, perceived by society as the main cause of environmental pressures, whereas it is merely the source. Societal impacts-mitigating actions are likely to be ineffective if they only target the source of the environmental pressure, rather than its underlying cause.

Environmental justice (Martínez-Alier, 1997) is possible with information mechanisms in place for each inhabitant to acknowledge from their personal perspective the impacts they exert each other with their non-deliberate waste generation. Deep understanding of impacts each inhabitant receive will contribute to opening informed consultation spaces between society and government to foster a solution-oriented decision-making process on the waste management systems.

This study is limited to Panama City, more case studies will be necessary to explore the issue of societal impacts caused by environmental pressures produced in landfills of DCs. It is recommended further research on methods to quantify societal impacts produced by landfill surface fires and nearby infrastructure explosions due to the discrete occurrence of these events. Also, on societal impacts caused by leachate infiltration in groundwater since only communities close to surface water bodies will be affected and different approaches will be necessary. Further research is needed to understand societal impacts from landfills of DCs, not only from an environmental pressure standpoint but multidimensionally, where other criteria to assess societal impacts can be taken into account -e.g. economic, socio-political, cultural-.

This study lacks an uncertainty and sensitivity analysis to verify the quality of the results of the models used, this limitation could be approached in further studies by using only open models allowing variables to be manipulated and understanding their interactions. RfC values uncertainty factors present great variability as per source and reference update. This could affect the interpretation of results, especially when studies are based on models, like in this case. LFG direct field data collection and analysis is recommended in further studies to decrease result uncertainty and increase the reliability degree of the present study.

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References

AAUD, 2016a. 20170731 -E 1.6.4.10 -Reformulacion-ANEJOS [WWW Document]. URL http://gestionderesiduos.com.pa/EstudioAAUD/AAUD - 2016 - 20170731 -E 1.6.4.10 - Reformulacion-ANEJOS.pdf (accessed 9.1.19).

AAUD, 2016b. PLAN NACIONAL DE GESTIÓN INTEGRAL DE RESIDUOS 2017-2027 Análisis y Diagnóstico de la Situación Actual (TOMO I) [WWW Document]. URL http://aaud.gob.pa/plangestion//Docs/ANEXOS/20170731_E 1.3.3.3.5_Propuesta Nuevo Modelo de Gestion_v3.pdf (accessed 9.1.19).

ANAM, 2011. SEGUNDA COMUNICACIÓN NACIONAL: Ante la convención marco de las Naciones Unidas sobre el cambio climático [WWW Document]. URL https://unfccc.int/resource/docs/natc/pannc2.pdf (accessed 11.8.18).

ATSDR, 2004. Health Consultation: A Note of Explanation [WWW Document]. URL http://www.atsdr.cdc.gov (accessed 1.25.19).

Attenborough, G.M., Hall, D.H., Gregory, R.G., McGoochan, L., 2002. Development of a landfill gas risk assessment model: GasSim [WWW Document]. Twenty-Fifth Annu. Landfill Gas Symp. URL https://www.lqm.co.uk/uploads/general/files/GasSim SWANA 2002.pdf (accessed 1.25.19).

Aziz, H.A., Rahim, N.A., Ramli, S.F., Alazaiza, M.Y.D., Omar, F.M., Hung, Y.T., 2018. Potential use of Dimocarpus longan seeds as a flocculant in landfill leachate treatment. Water (Switzerland) 10. https://doi.org/10.3390/w10111672

Bentley, H., Smith, S., Schrauf, T., 2005. Baro-pneumatic estimation of landfill gas generation rates at four landfills in the southeastern United States [WWW Document]. Proc. from SWANA 28th Annu. Landfill Gas Symp. URL https://www.researchgate.net/publication/268398035

Blais, A., Ritch, E., Berlin, R., Clima, R., Glendening, J., Gutierrez, V., Izard, J., Mallow, C., Nelson, R., Swigert, P., 2010. Managing Trash in Colón , Panama : A Case Study of Selected Strategies A Report of the Panama Initiative [WWW Document]. URL https://panama.evsc.virginia.edu/docs/team_panama_report_100621.pdf

Bogner, J., Matthews, E., 2003. Global methane emissions from landfills: New methodology and annual estimates 1980-1996. Global Biogeochem. Cycles 17, n/a-n/a. https://doi.org/10.1029/2002GB001913

Caprile, M.D., Ripa, M., 2014. A Life Cycle Assessment of Landfilled Municipal Solid Waste in Argentina: The Influence of Waste Composition on Greenhouse Gases Emissions and Other Impacts. J. Environ. Account. Manag. 2, 145–162. https://doi.org/10.5890/JEAM.2014.06.005

CEPAL, 2009. PANAMÁ. Evolución Económica durante 2008 y perspectiva para 2009 [WWW Document]. URL https://repositorio.cepal.org/bitstream/handle/11362/25885/1/LCmexL916_es.pdf (accessed 1.24.19).

Chandler, J.A., Jewell, W.J., 1980. Predicting methane fermentation biodegradability. Final report [WWW Document]. Biotechnol. Bioeng. Symp. No. 10. URL https://www.osti.gov/biblio/6810537-predicting-methane-fermentation-biodegradability (accessed 1.25.19).

Cheng, F.Y., Georgakakos, K.P., 2011. Wind speed interpolation in the vicinity of the Panama Canal. Meteorol. Appl. 18, 459–466. https://doi.org/10.1002/met.237

Chifari, R., Lo Piano, S., Bukkens, S.G.F., Giampietro, M., 2016. A holistic framework for the integrated assessment of urban waste management systems. Ecol. Indic. https://doi.org/10.1016/j.ecolind.2016.03.006

Coleman, G., Kalish, I., Konigsburg, D., 2014. Competitiveness: Catching the next wave Panama [WWW Document]. Deloitte. URL https://www2.deloitte.com/content/dam/Deloitte/global/Documents/About-Deloitte/gx-panama-competitiveness-report-08042014.pdf (accessed 1.20.19).

Dankovic, D.A., Naumann, B.D., Maier, A., Dourson, M.L., Levy, L.S., 2015. The Scientific Basis of Uncertainty Factors Used in Setting Occupational Exposure Limits. J. Occup. Environ. Hyg. 12, S55–S68. https://doi.org/10.1080/15459624.2015.1060325

DEFRA, 2010. Waste acceptance at landfills: Guidance on waste acceptance procedures and criteria [WWW Document]. URL www.environment-agency.gov.uk (accessed 9.26.18).

Dennehy, C., Lawlor, P.G., Jiang, Y., Gardiner, G.E., Xie, S., Nghiem, L.D., Zhan, X., 2017. Greenhouse gas emissions from different pig manure management techniques: a critical analysis. Front. Environ. Sci. Eng. https://doi.org/10.1007/s11783-017-0942-6

El-Fadel, M., Findikakis, A.N., Leckie, J.O., 1989. A numerical model for methane production in managed sanitary landfills. Waste Manag. Res. 7, 31–42. https://doi.org/10.1177/0734242X8900700105

Elkahwagy, R., Gyanchandani, V., Piselli, D., 2017. UNFCCC Nationally Determined Contributions: Climate Change and Trade, SSRN. https://doi.org/10.2139/ssrn.2919692

Elliott, P., Morris, S., Briggs, D., Hoogh, C. De, Hurt, C., Jensen, T.K., Maitland, I., Lewin, A., Richardson, S., Wakefield, J., Lars, J., 2001. Birth outcomes and selected cancers in populations living near landfill sites [WWW Document]. URL https://cot.food.gov.uk/sites/default/files/cot/dhlandfillaug01.pdf (accessed 1.28.19).

Faour, A.A., Reinhart, D.R., You, H., 2007. First-order kinetic gas generation model parameters for wet landfills. Waste Manag. 27, 946–953. https://doi.org/10.1016/j.wasman.2006.05.007

Farquhar, G.J., Rovers, F.A., 1973. Gas production during refuse decomposition. Water, Air, Soil Pollut. 2, 483–495. https://doi.org/10.1007/BF00585092

Frischknecht, R., Editors, N.J., Althaus, H.-J., Bauer, C., Doka, G., Dones, R., Hischier, R., Hellweg, S., Köllner, T., Loerincik, Y., Margni, M., 2007. Implementation of Life Cycle Impact Assessment Methods [WWW Document]. Am. Midl. Nat. URL www.ecoinvent.org

GALLUP, 2014. Country Well-Being Varies Greatly Worldwide [WWW Document]. URL https://news.gallup.com/poll/175694/country-varies-greatly-worldwide.aspx (accessed 2.1.19).

Gioia, F., Murena, F., Savino, G., Saha, P., 1995. The release into the atmosphere of hazardous volatiles Part I. Release from porous solids imbued with a liquid mixture in which the volatiles are dissolved. J. Hazard. Mater. 40, 213–235. https://doi.org/10.1016/0304-3894(94)00083-S

Golder Associates, 2012. GasSim 2.5 User Manual [WWW Document]. URL http://www.gassim.co.uk/documents/GasSim User Manual v2.5.8.pdf (accessed 9.26.18).

Goopy, J.P., Onyango, A.A., Dickhoefer, U., Butterbach-Bahl, K., 2018. A new approach for improving emission factors for enteric methane emissions of cattle in smallholder systems of East Africa – Results for Nyando, Western Kenya. Agric. Syst. 161, 72–80. https://doi.org/10.1016/j.agsy.2017.12.004

Hamza, V., Muñoz, M., 1996. Heat flow map of South America [WWW Document]. Geothermics. URL https://www.researchgate.net/profile/Miguel_Munoz19/publication/222894266_Heat_Flow_Map_of_South_America/links/5a07c7ebaca272ed279e6731/Heat-Flow-Map-of-South-America.pdf (accessed 11.21.18).

Hidromet, 2017. Viento - Hidrometeorología de ETESA [WWW Document]. URL http://www.hidromet.com.pa/viento.php (accessed 1.28.19).

Hidromet, 2002. Mapa_Frecuencia_Direccion_Viento_Panama [WWW Document]. URL http://www.hidromet.com.pa/Mapas/Mapa_Frecuencia_Direccion_Viento_Panama.pdf (accessed 1.28.19).

Hoornweg, D., Bhada, P., 2012. What a Waste. A Global Review of Solid Waste Management. Urban Dev. Ser. Knowl. Pap. 281, 44 p. https://doi.org/10.1111/febs.13058

INEC, 2014a. Cuadro 312-04. Total de reses, promedio de reses por hectarea y pastos en la República, según provincia y comarca indígena: Septiembre de 2014 [WWW Document]. URL https://www.contraloria.gob.pa/inec/archivos/P6701312-04.pdf (accessed 1.20.19).

INEC, 2014b. Cuadro 312-07. Existencia de ganao porcino en la República por actividad, según provincia y comarca indígena: año 2014 [WWW Document]. URL https://www.contraloria.gob.pa/inec/archivos/P6701312-07.pdf (accessed 1.20.19).

INEC, 2010. Población en el Distrito de Panamá, por sexo, según corregimiento y grupos de edad: Censo 2010 [WWW Document]. URL https://www.contraloria.gob.pa/inec/Publicaciones/Publicaciones.aspx?ID_SUBCATEGORIA=59&ID_PUBLICACION=362&ID_IDIOMA=1&ID_CATEGORIA=13 (accessed 1.8.18).

INEC, 2008. Comentarios sobre la economía panameña: año 2008 [WWW Document]. URL https://www.contraloria.gob.pa/inec/archivos/P4511COMENTARIO.pdf (accessed 1.24.19).

INEC, 2000. Volumen I: Lugares Poblados: Año 2000 [WWW Document]. 1. URL https://www.contraloria.gob.pa/inec/Publicaciones/Publicaciones.aspx?ID_SUBCATEGORIA=53&ID_PUBLICACION=542&ID_IDIOMA=1&ID_CATEGORIA=9 (accessed 3.6.19).

INECO, 2017. Plan Nacional de gestión integral de residuos 2017 - 2027 [WWW Document]. URL http://aaud.gob.pa/plangestion//Docs/ANEXOS/20170731_E 1.3.3.3.5_Propuesta Nuevo Modelo de Gestion_v3.pdf (accessed 4.29.18).

IPCC, 2006a. 2006 IPCC Guidelines for National Greenhouse Gas Inventories: Solid Waste Disposal [WWW Document]. URL https://www.ipcc-nggip.iges.or.jp/public/2006gl/pdf/5_Volume5/V5_3_Ch3_SWDS.pdf (accessed 11.8.18).

IPCC, 2006b. Good practice guidance and uncertainty management in national greenhouse inventories -waste [WWW Document]. URL https://www.ipcc-nggip.iges.or.jp/public/gp/english/5_Waste.pdf (accessed 1.26.19).

Islam, M.R., Haniu, H., Fardoushi, J., 2009. Pyrolysis kinetics behavior of solid tire wastes available in Bangladesh. Waste Manag. https://doi.org/10.1016/j.wasman.2008.04.009

JICA, 2003. Evaluación de Impacto Ambiental del Sistema de Disposición Final [WWW Document]. URL http://open_jicareport.jica.go.jp/pdf/11712841_01.pdf

JICA, 2002. Characteristics of Solid Waste Problems in Developing Countries [WWW Document]. Support. Capacit. Dev. Solid Waste Manag. Dev. Ctries. URL https://www.jica.go.jp/jica-ri/IFIC_and_JBICI-Studies/english/publications/reports/study/topical/waste/pdf/waste_02.pdf (accessed 6.21.18).

Kanabkaew, T., Rakmak, N., Choosaeng, S., 2014. Assessment of Hydrogen Sulfide Dispersion from Dumpsite Using AERMOD Modeling System. Adv. Mater. Res. 931–932, 650–654.

https://doi.org/10.4028/www.scientific.net/AMR.931-932.650

Kinnaman, T.C., 2009. The economics of municipal solid waste management. Waste Manag. 29, 2615–2617. https://doi.org/10.1016/j.wasman.2009.06.031

Kjeldsen, P., 1996. Landfill gas migration in soil. Chapter 3.1 [WWW Document]. Landfilling of waste: Biogas. URL http://orbit.dtu.dk/en/publications/landfill-gas-migration-in-soil(7f5b6676-c9a8-42c7-9eaf-29236ec35dd2)/export.html (accessed 1.20.19).

Krause, M.J., Chickering, G.W., Townsend, T.G., 2016. Translating landfill methane generation parameters among first-order decay models. J. Air Waste Manag. Assoc. 66, 1084–1097. https://doi.org/10.1080/10962247.2016.1200158

La Prensa, 2013. AAUD: El peor incendio en 10 años en Cerro Patacón [WWW Document]. URL https://www.prensa.com/manuel_vega_loo/AAUD-peor-incendio-Cerro-Patacon_2_3619158050.html (accessed 12.17.18).

Lewis, J.M. (National S.S.L., 1995. The Story behind the Bowen Ratio. Bull. Am. Meteorol. Soc. 76, 2433–2443. https://doi.org/10.1175/1520-0477(1995)076<2433:TSBTBR>2.0.CO;2

Ludwig, V., 2007. User's Manual Central America Landfill Gas Model [WWW Document]. URL https://www.globalmethane.org/documents/models/pdfs/UsersManualCentralAmerica_LFG_model_final_English_REV1.pdf (accessed 9.30.18).

Machado, S.L., Carvalho, M.F., Gourc, J.P., Vilar, O.M., do Nascimento, J.C.F., 2009. Methane generation in tropical landfills: Simplified methods and field results. Waste Manag. 29, 153–161. https://doi.org/10.1016/j.wasman.2008.02.017

Majdinasab, A., Zhang, Z., Yuan, Q., 2017. Modelling of landfill gas generation: a review. Rev. Environ. Sci. Biotechnol. https://doi.org/10.1007/s11157-017-9425-2

Martínez-Alier, J., 1997. Environmental justice (local and global). Capital. Nature, Social. 8, 91–

109. https://doi.org/10.1080/10455759709358725

McEvoy, A.J. (Augustin J., Markvart, T., Castañer, L., 2012. Practical handbook of photovoltaics : fundamentals and applications [WWW Document]. URL https://www.elsevier.com/books/practical-handbook-of-photovoltaics/mcevoy/978-0-12-385934-1 (accessed 1.28.19).

Mou, Z., Kjeldsen, P., Scheutz, C., 2014. Landfill gas generation and emission at Danish waste disposal sites receiving low-organic waste [WWW Document]. DTU Environ. URL http://orbit.dtu.dk/files/100551929/Zishen_Mou_PhD_Thesis_WWW_Version.pdf (accessed 11.16.18).

Mou, Z., Scheutz, C., Kjeldsen, P., 2015. Evaluation and application of site-specific data to revise the first-order decay model for estimating landfill gas generation and emissions at Danish landfills. J. Air Waste Manag. Assoc. 65, 686–698. https://doi.org/10.1080/10962247.2015.1008653

NITE, 2017. Risk Assessment on chemicals-For Better Understanding-5 | Chemical Management | National Institute of Technology and Evaluation (NITE) [WWW Document]. URL https://www.nite.go.jp/en/chem/shiryo/ra/about_ra7.html (accessed 11.23.19).

OECD, 2008. Key environmental indicators, OECD Environment Directorate. https://doi.org/10.2110/pec.72.02.0001

Oonk, H., 2010. Literature review: methane from landfills methods to quantify generation, oxidation and emission. Final report [WWW Document]. URL www.oonkay.nl (accessed 1.25.19).

Oonk, H., 1994. Validation of landfill gas formation models [WWW Document]. URL https://www.rvo.nl/sites/default/files/2013/10/Oonk 1994 %28EN%29 Validation of landfill gas formation models.pdf (accessed 1.25.19).

Pan, S.Y., Du, M.A., Huang, I. Te, Liu, I.H., Chang, E.E., Chiang, P.C., 2014. Strategies on implementation of waste-to-energy (WTE) supply chain for circular economy system: A review. J. Clean. Prod. 108. https://doi.org/10.1016/j.jclepro.2015.06.124

Paraskaki, I., Lazaridis, M., 2005. Quantification of landfill emissions to air: A case study of the Ano Liosia landfill site in the greater Athens area. Waste Manag. Res. 23, 199–208. https://doi.org/10.1177/0734242X05054756

Peer, R.L., Epperson, D.L., Campbell, D.L., Brook, P. Von, 1992. Development of an Empirical Model of Methane Emissions from Landfills [WWW Document]. URL https://cfpub.epa.gov/si/si_public_record_Report.cfm?Lab=NRMRL&dirEntryID=126308 (accessed 1.25.19).

Pennington, D.G., 1996. Effects of Waste Tires, Waste Tire Facilities, and Waste Tire Projects on The Environment [WWW Document]. URL www.ciwmb.cagov

Philippe, F.X., Nicks, B., 2015. Review on greenhouse gas emissions from pig houses: Production of carbon dioxide, methane and nitrous oxide by animals and manure. Agric. Ecosyst. Environ. 199, 10–25. https://doi.org/10.1016/j.agee.2014.08.015

Pino, D., de Arellano, J.V.G., Kim, S.W., 2006. Representing sheared convective boundary layer by zeroth- and first-order-jump mixed-layer models: Large-eddy simulation verification. J. Appl. Meteorol. Climatol. 45, 1224–1243. https://doi.org/10.1175/JAM2396.1

Pipatti, R., Svardal, P., Wagner Silva Alves, J., Gao, Q., López Cabrera, C., Mareckova, K., Oonk, H., Scheehle, E., Sharma, C., Smith, A., Yamada, M., 2006. IPCC Guidelines for National Greenhouse Gas Inventories [WWW Document]. URL https://www.ipcc-nggip.iges.or.jp/SBSTA24/sb24_2006GL_waste_6.ppt

Plocoste, T., Jacoby-Koaly, S., Petit, R.-H., Roussas, A., 2016. Estimation of Methane Emission from a Waste Dome in a Tropical Insular Area. Int. J. Waste Resour. 6. https://doi.org/10.4172/2252-5211.1000211

Poulsen, T., Christophersen, M., Moldrup, P., Kjeldsen, P., 2003. Relating landfill gas emissions to atmospheric pressure using numerical modelling and state-space analysis [WWW Document]. URL http://journals.sagepub.com/doi/pdf/10.1177/0734242X0302100408 (accessed 9.20.18).

Poulsen, T.G., Christophersen, M., Moldrup, P., Kjeldsen, P., 2001. MODELING LATERAL GAS TRANSPORT IN SOIL ADJACENT TO OLD LANDFILL. J. Environ. Eng. 127, 145–153. https://doi.org/doi:10.1061/(ASCE)0733-9372(2001)127:2(145)

Powell, J.T., Townsend, T.G., Zimmerman, J.B., 2016. Estimates of solid waste disposal rates and reduction targets for landfill gas emissions. Nat. Clim. Chang. 6, 162–165. https://doi.org/10.1038/nclimate2804

Powrie, W., White, J., 2004. Landfill process modelling, in: Waste Management. pp. 225–226. https://doi.org/10.1016/j.wasman.2004.02.001

Rigamonti, L., Sterpi, I., Grosso, M., 2016. Integrated municipal waste management systems: An indicator to assess their environmental and economic sustainability. Ecol. Indic. 60, 1–7. https://doi.org/10.1016/j.ecolind.2015.06.022

Roudi, A.M., Chelliapan, S., Mohtar, W.H.M.W., Kamyab, H., 2018. Prediction and optimization of the Fenton process for the treatment of landfill leachate using an artificial neural network. Water (Switzerland) 10. https://doi.org/10.3390/w10050595

Saral, A., Demir, S., Yildiz, Ş., 2009. Assessment of odorous VOCs released from a main MSW landfill site in Istanbul-Turkey via a modelling approach. J. Hazard. Mater. 168, 338–345. https://doi.org/10.1016/j.jhazmat.2009.02.043

Sarptaş, H., 2016. Assessment of Landfill Gas (Lfg) Energy Potential Based on Estimates of Lfg Models. Deu Muhendis. Fak. Fen ve Muhendis. 18, 491–491. https://doi.org/10.21205/deufmd.2016185416

Scharff, H., Jacobs, J., 2006. Applying guidance for methane emission estimation for landfills, in: Waste Management. pp. 417–429. https://doi.org/10.1016/j.wasman.2005.11.015

Spence, A., Poortinga, W., Pidgeon, N., 2012. The Psychological Distance of Climate Change. Risk Anal. 32, 957–972. https://doi.org/10.1111/j.1539-6924.2011.01695.x

SWANA, 1997. Comparison of models for predicting landfill methane recovery [WWW Document]. URL https://www.nrel.gov/docs/legosti/fy97/26041.pdf (accessed 11.12.18).

Tabasaran, O., Rettenberger, G., 1987. Grundlage zur Planung von Entgasungsanlagen [WWW Document]. Müllhandbuch. URL http://wirtschaftsrechtdigital.info/_sid/ZHHL-524258-3Lrx/978-3-503-11666-9 (accessed 9.26.18).

The World Bank, 2018. New country classifications by income level: 2018-2019 | The Data Blog [WWW Document]. URL https://blogs.worldbank.org/opendata/new-country-classifications-income-level-2018-2019 (accessed 7.9.18).

US EPA, 2018. User's Guide for the AMS/EPA Regulatory Model (AERMOD) [WWW Document]. URL https://www3.epa.gov/ttn/scram/models/aermod/aermod_userguide.pdf (accessed 10.7.18).

US EPA, 2011. Integrated Risk Information System (IRIS) Glossary [WWW Document]. URL https://ofmpub.epa.gov/sor_internet/registry/termreg/searchandretrieve/glossariesandkeywordlists/search.do?details=&glossaryName=IRIS Glossary (accessed 1.24.19).

US EPA, 2010. Methane and Nitrous Oxide Emissions From Natural Sources [WWW Document]. URL http://www.epa.gov/methane/sources.html (accessed 9.26.18).

US EPA, 2008. Background Information Document for Updating AP42 Section 2.4 for Estimating Emissions from Municipal Solid Waste Landfills [WWW Document]. URL https://www3.epa.gov/ttn/chief/ap42/ch02/draft/db02s04.pdf (accessed 9.26.18).

US EPA, 2006a. Solid Waste Management and Greenhouse Gases: A Life-Cycle Assessment of Emissions and Sinks [WWW Document]. URL https://nepis.epa.gov (accessed 1.24.19).

US EPA, 2006b. AP 42, Fifth Edition, Volume I - Chapter 2 Solid Waste Disposal (2-4 Municipal Solid Waste Landfills) DRAFT [WWW Document]. URL https://www3.epa.gov/ttn/chief/ap42/ch02/draft/d02s04.pdf (accessed 1.24.19).

US EPA, 2005. US EPA First-Order Kinetic Gas Generation Model Parameters for Wet Landfills [WWW Document]. URL https://nepis.epa.gov/Exe/ZyPURL.cgi?Dockey=P100ADRJ.TXT (accessed 9.26.18).

US EPA, 2003. Toxicological Review of Hydrogen Sulfide [WWW Document]. EPA/635/R-03/005. URL www.epa.gov/iris (accessed 12.11.18).

US EPA, 2002. U.S. Greenhouse Gas Emissioms and Sinks, 1990-2016 [WWW Document]. EPA 430-R-18-003. URL https://www.epa.gov/ghgemissions/inventory-us-greenhouse-gas-emissions-and-sinks (accessed 1.20.19).

US EPA, 2001. Volume III: Chapter 15. LANDFILLS Revised Final [WWW Document]. URL https://www.epa.gov/sites/production/files/2015-08/documents/iii15_apr2001.pdf (accessed 1.25.19).

US EPA, 2000. Chloroform [WWW Document]. URL https://www.epa.gov/sites/production/files/2016-09/documents/chloroform.pdf (accessed 12.11.18).

US EPA, 1995. 2.4 Municipal Solid Waste Landfills [WWW Document]. Compil. Air Pollut. Emiss. Factors AP-42. URL http://www.epa.gov/ttn/chief (accessed 1.24.19).

US EPA, 1991. Dichloromethane; CASRN 75-09-2 [WWW Document]. URL https://cfpub.epa.gov/ncea/iris/iris_documents/documents/subst/0070_summary.pdf#nameddest=rfc (accessed 11.23.19).

Weichenthal, S., Van Rijswijk, D., Kulka, R., You, H., Van Ryswyk, K., Willey, J., Dugandzic, R., Sutcliffe, R., Moulton, J., Baike, M., White, L., Charland, J.P., Jessiman, B., 2015. The impact of a landfill fire on ambient air quality in the north: A case study in Iqaluit, Canada. Environ. Res. 142, 46–50. https://doi.org/10.1016/j.envres.2015.06.018

Weitz, M., Coburn, J.B., Salinas, E., 2008. Estimating national landfill methane emissions: An application of the 2006 Intergovernmental Panel on Climate Change Waste Model in Panama. J. Air Waste Manag. Assoc. 58, 636–640. https://doi.org/10.3155/1047-3289.58.5.636

Wilson, D.C., Rodic, L., Scheinberg, A., Velis, C.A., Alabaster, G., 2012. Comparative analysis of solid waste management in 20 cities. Waste Manag. Res. 30, 237–254. https://doi.org/10.1177/0734242X12437569

WMO, 2001. International current condition weather stations [WWW Document]. URL https://www.wunderground.com/about/faq/international_cities.asp (accessed 1.28.19).

Zacharof, A.I., Butler, A.P., 2004. Stochastic modelling of landfill processes incorporating waste heterogeneity and data uncertainty, in: Waste Management. pp. 241–250. https://doi.org/10.1016/j.wasman.2003.12.001

Ziout, A., Azab, A., Atwan, M., 2014. A holistic approach for decision on selection of end-of-life products recovery options. J. Clean. Prod. 65, 497–516. https://doi.org/10.1016/j.jclepro.2013.10.001