Pricing landfill externalities: Emissions and disamenity costs in Cape Town, South Africa

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Abstract

The external (environmental and social) costs of landfilling (e.g. emissions to air, soil and water; and 'disamenities' such as odours and pests) are difficult to quantify in monetary terms, and are therefore not generally reflected in waste disposal charges or taken into account in decision making regarding waste management options. This results in a bias against alternatives such as recycling, which may be more expensive than landfilling from a purely financial perspective, but preferable from an environmental and social perspective. There is therefore a need to quantify external costs in monetary terms, so that different disposal options can be compared on the basis of their overall costs to society (financial plus external costs). This study attempts to estimate the external costs of landfilling in the City of Cape Town for different scenarios, using the benefits transfer method (for emissions) and the hedonic pricing method (for disamenities). Both methods (in particular the process of transferring and adjusting estimates from one study site to another) are described in detail, allowing the procedures to be replicated elsewhere. The results show that external costs are currently R111 (in South African Rands, or approximately US$16) per tonne of waste, although these could decline under a scenario in which energy is recovered, or in which the existing urban landfills are replaced with a new regional landfill.

Key words

Externality

External cost

Disamenity

Benefits transfer

Hedonic pricing method

Landfill

1. Introduction

Like all methods of waste disposal, landfilling imposes both financial and external costs on society. Financial costs refer to actual financial outlays associated with establishment, operation and end-of-life management of the landfill site. External costs (or negative externalities), by contrast, refer to costs that are imposed on other parties as a result of the landfill's activities. More generally, externalities can be defined as the positive or negative side effects (external benefits or costs) of a particular economic activity (e.g. landfilling) that are not incurred by those with a direct financial stake in the activity (e.g. the landfill owner or operator); but are instead borne by other groups in society and/or by future generations; or are dispersed throughout society as a whole. Externalities associated with landfilling are not reflected in the financial statements of the landfill owner or operator, but affect social well-being more generally.

Two important types of external costs associated with landfills can be distinguished[1] (see Table 1). Firstly, decomposition of organic wastes produces both landfill gas (LFG) and leachate. Emissions to air (associated with both LFG and transport of waste) impact negatively on both human health and the global climate; while emissions to soil and water (in the form of leachate) impact negatively on both human and ecological health. Secondly, there are 'disamenities' ('nuisances') associated with living in the vicinity of a landfill site (and with waste transport), in the form of noise, odour, litter, vermin, dust and traffic (Eshet et al., 2005, 2006).

[1] Other types of external costs can also be identified; such as opportunity costs associated with the use of land for a landfill site (to the extent that these are not internalised in the purchase price of the land); but these will not be considered here. Emissions and disamenities are generally considered to be the most significant external costs associated with landfills.

Table 1: Externalities associated with landfill disposal

Category	Examples of externalities	Source	Major impacts
Emissions	Methane (CH_4)	LFG	Global warming[a]
	Carbon dioxide (CO_2)	LFG, transport of waste	Global warming
	Local air pollutants e.g. sulphur dioxide (SO_2), nitrogen oxides (NO_x), particulates (TSP, PM_{10}) and volatile organic compounds (VOCs).	LFG, transport of waste	Human health; damage to buildings, crops, forests and water
	Pollution displacement (positive externality/ external benefit)[b]	Energy recovery (LFG is captured and methane content is used to generate energy)	Avoided impacts on global warming/ human health
	Emissions to soil and water	Leachate	Human and ecological health
Disamenities	Odours, dust, wind-blown litter, vermin (e.g. rodents, flies and carrion birds), noise, traffic/congestion, visual intrusion, risk of fires and explosions	Proximity to landfill site, transport of waste. Odours and fire/explosion risks arise mainly from methane emissions.	Nuisance/ unpleasantness

[a] Methane, the concentration of which varies from 40-70% of LFG by dry volume (European Commission, 2000b), and which has a 20-25 times stronger warming effect than CO2 on a molecular basis, contributes about 18% towards total global warming, or 500 million tonnes per year, of which 40 to 75 million tonnes are attributed to emissions from landfills. The methane present in LFG is therefore becoming a significant contributor to global warming (El-Fadel et al., 1997).

[b] All externalities in the table are external costs/negative externalities, with the exception of pollution displacement, which is an external benefit/positive externality

In addition to external costs, there may also be external *benefits* (positive externalities) associated with landfilling. For example, the methane content of LFG can be recovered and used to generate energy (energy recovery). This is an external benefit in that the negative impacts associated with conventional (e.g. fossil-fuel based) energy, including emissions of greenhouse gases and other air pollutants, are displaced (European Commission, 2000a; b; Fiehn and Ball, 2005).

Ideally, different waste management options should be compared on the basis of their overall net costs to society (= financial costs + external costs – external benefits), per tonne of waste. However, unlike financial costs, external costs and benefits are often intangible and difficult to quantify in monetary terms, and are therefore not generally reflected in waste disposal charges or taken into account in decision making regarding waste management options. This implies that decision making will be biased in favour of options with lower financial costs, even though their overall costs to society may be higher (Goldblatt, 2009, Nahman and

Godfrey, 2010). For example, there could be a bias against alternatives such as recycling, which may be more expensive than landfilling from a purely financial perspective, but preferable from an environmental and social perspective.

For example, in the City of Cape Town, South Africa, 2 million tonnes[2] of waste is disposed of at municipal landfill sites each year (average over the last 10 years) (Engledow, 2007, City of Cape Town, 2008a, De Wit and Nahman, 2009). Despite a number of recent initiatives by the City to divert waste from landfills; such as composting facilities and material recovery facilities, education and awareness programmes, and a pilot 'split-bag' collection service, with one bag allocated for recyclables and the other for general waste (City of Cape Town, 2008a); landfilling remains the predominant method of managing solid waste (Engledow, 2007). Only 27% of waste is currently recovered for recycling, composting, chipping etc; with the remainder ending up in the City's landfills (De Wit, 2010).

Moving up the 'waste management hierarchy' (i.e. avoiding, reducing, re-using, recycling, and processing/treating waste as far as possible; and only disposing to landfill as a last resort) requires that the external costs of different waste management options are quantified ('valued') in monetary terms using appropriate valuation methods, in order to ensure that they are properly understood and accounted for in decision making. This will allow all disposal options to be compared on an equal footing, on the basis of their overall costs to society per tonne of waste.

The objective of this paper is to estimate the external costs associated with emissions and disamenities from landfills in the City of Cape Town, per tonne of waste. The focus is on *net*

[2] Tonne = metric ton (1,000kg).

external costs (i.e. external costs less external benefits). In addition, the focus is on *marginal* external costs; i.e., the additional external cost associated with the next tonne of waste landfilled. The analysis is restricted to municipal solid waste accepted for disposal at the three active municipal landfill sites in the City (Bellville South, Coastal Park and Vissershok). This includes household, commercial and some industrial waste, including low-risk hazardous waste, but excludes high-risk hazardous waste, which is managed and disposed of separately at a private facility.

Net costs will be estimated and compared for a number of relevant scenarios, based on the following considerations. Firstly, although leachate is controlled at all three landfills, none of the three sites currently capture LFG, let alone recover methane to generate energy; although there are plans in place for such a system at all three sites (Dentlinger, 2008, Van der Merwe, 2009, Njobeni, 2010). Furthermore, all three existing landfills are expected to reach capacity within the next twelve to fourteen years (City of Cape Town, 2008b, De Wit, 2009a, 2010). A new regional landfill site is therefore being planned to cope with the looming shortage in landfill airspace (City of Cape Town, 2008b, De Wit et al., 2008, Ministry of Local Government Environmental Affairs and Development Planning, 2009), although this will be located some distance away from human populations. This implies that external costs associated with LFG and disamenities are likely to become less significant, while costs associated with transport (e.g. emissions, accidents, traffic and noise) will become more important.

There is therefore a need to compare the costs of the new regional landfill system (in terms of increased transport costs) with those of the existing urban landfill system (in terms of impacts on nearby human populations) (De Wit, 2009b). In addition, the positive externalities associated with planned LFG capture and energy recovery systems need to be considered. Thus, net external costs will be estimated and compared for the following three scenarios:

1. Status quo (existing urban landfills, no energy recovery)
2. Existing urban landfills with energy recovery
3. New regional landfill with energy recovery

The paper is structured as follows. Section 2 outlines the valuation methods used to quantify externalities in monetary terms. Section 3 presents the data used and the results of the valuation exercise, and briefly discusses their implications. Section 4 concludes and provides recommendations for further research.

2. Methods

Quantifying external costs or benefits requires placing a monetary value on the changes in well-being experienced by people as a result of exposure to externalities. Economists have developed a range of valuation methods for quantifying these changes in well-being. Different methods are appropriate for valuing different types of externalities (European Commission, 2000a). A detailed categorisation and description of the different methods can be found elsewhere (for example, in the context of landfill externalities, see European Commission, 2000a, Eshet et al., 2005, 2006). Here the focus is on the two methods that have been applied in this study, namely the benefits transfer method, which is used to estimate the costs associated with LFG and leachate emissions; and the hedonic pricing method, which is used to estimate costs for disamenities associated with proximity to a landfill site. In addition, the method used for estimating transport externalities (including both transport emissions and transport-related disamenities) is described.

2.1. The benefits transfer method

The benefits transfer method involves drawing on existing valuation estimates from other studies, and applying them to the study site in question, making appropriate adjustments for differences between the original study site(s) and the new study site. This requires that the studies from which estimates are drawn were themselves sound and reliable; that, to the extent possible, the original study sites and the study site in question are similar; and that appropriate adjustments are made for any remaining differences between the original sites and the study site in question. Although this method adds an element of uncertainty to the estimates, since there may be important differences between the sites that cannot be fully adjusted for, it is useful in cases where time and budget constraints exist (European Commission, 2000a, Eshet et al., 2006).

In selecting appropriate studies from which estimates can be drawn, and in identifying the specific adjustments that need to be made, it helps to set out the series of causal relationships along the 'impact pathway' leading from waste volumes to monetary costs (or benefits, in the case of pollution displacement) at a given site (European Commission, 2000b). In its simplest form, the relationship between waste volumes and monetary costs can be understood as illustrated in Figure 1.

INSERT FIGURE 1 HERE

Figure 1 shows that the impact chain starts with waste entering the landfill, which creates emissions of various types of pollutants (CO_2, CH_4, leachate etc). These emissions impact upon society and the environment in a number of ways, negatively affecting human well-being (which can be quantified in terms of monetary costs).

The indirect relationship between waste volumes and monetary costs (quantified in terms of monetary cost per tonne of waste) can therefore be broken down into two simpler, more direct causal relationships; i.e. the physical relationship between waste volumes and emissions (quantified in terms of the 'emission factor,' i.e. kg of emissions per tonne of waste); and the economic relationship between emissions and costs (quantified in terms of the 'unit cost,' i.e. the monetary cost per kg of emissions). Together, the emission factor and the unit cost determine the monetary cost per tonne of waste associated with each particular pollutant (European Commission, 2000b).

These relationships are each affected by a number of factors. For pollutants associated with LFG and leachate, *emission factors* are affected by the quantity and composition of LFG and leachate generated; which are in turn primarily determined by:

1. Waste types (general vs hazardous) and composition (in particular, the organic waste content). Since LFG and leachate are both formed through the decomposition of the biodegradable organic waste fraction, the generation and composition of both LFG and leachate are strongly influenced by the proportion of organic waste in the overall waste stream entering the landfill (European Commission, 2000b).
2. Landfill design and operation characteristics; in particular, the presence of LFG and/or leachate management, treatment and/or control features (Eshet et al., 2006). In

modern landfills, emissions of LFG and leachate to the surrounding environment, or at least the associated environmental damages, are mitigated to a large extent through the use of systems for collecting and containing, re-using or treating these emissions. For example, in modern landfills with engineered liners, or where leachate is treated to an acceptable quality, the damages associated with leachate contamination are considered negligible (Eshet et al., 2006). Similarly, LFG can be captured and either flared or used to generate energy (either in the form of electricity, heat, or both); through recovery of the methane content, thereby reducing emissions of methane and other components of LFG (European Commission, 2000b). Older landfills typically don't have these features and are therefore associated with higher emissions of LFG and leachate (European Commission, 2000b).

3. Climate. For example, LFG emissions are affected by general climatic conditions; while leachate generation is affected by water balance in the surrounding environment, which is determined by precipitation and evaporation; as well as soil type (Engledow, 2007).

In addition, pollution displacement benefits associated with energy recovery are affected by:

1. Characteristics of the LFG capture and energy conversion system (LFG collection efficiency; methane concentration of LFG; energy content of methane; form of energy recovered, i.e. heat, electricity, or combined heat and power; energy conversion efficiency; etc) (Powell and Brisson, 1994, Eshet et al., 2006, BDA Group, 2009).
2. The form and efficiency of energy generation that is assumed to be replaced (generally, this will be the predominant form of energy generation in the region/country in question). For example, if energy derived through recovery of

methane is assumed to replace energy that would have been generated at an inefficient coal-fired power station, then the benefits (in terms of displaced pollution) are likely to be far higher than those that would be obtained if a cleaner, more efficient form of energy generation was being replaced (Dijkgraaf and Vollebergh, 2003).

As with emission factors, *unit costs* are also affected by a number of factors. For leachate and local air pollutants, unit costs are affected by

1. Location and physical characteristics of the site (e.g. air pollution impacts are determined by distance to human populations, topography, and prevailing meteorological conditions, e.g. wind directions) (Eshet et al., 2006).
2. The socio-economic characteristics of the exposed populations (population density; and ‘willingness to pay,’ which is affected by both income (which reflects ability to pay), and tastes or preferences for environmental quality).

For global warming impacts associated with CO_2 and CH_4, on the other hand, since the resulting climate impacts are independent of the source of the emissions (BDA Group, 2009); unit costs are the same irrespective of the location and physical characteristics of the site.

Two other factors affecting unit costs are the assumed social discount rate (which reflects the higher value that people place on the present relative to the future) for damages occurring in different time periods; and, for CH_4, the assumed global warming potential (GWP) of CH_4 relative to CO_2 (expressed in CO_2-equivalents) (European Commission, 2000a, Eshet et al., 2006).

2.2. Method for estimating transport externalities

External costs associated with transport are highly context-specific, and should therefore not be estimated using cross-country benefits transfer. Transport emissions depend on the distance of the site from where the waste is generated, as well as vehicle type and weight, travel speed, and waste density and weight (Eshet et al., 2006). Since not all of this information was available for the City of Cape Town, it was necessary to make a number of assumptions regarding the distances involved, and to use estimates of average external costs per tonne-km associated with freight transport from other South African studies. The figures used are discussed in more detail in Section 3.2.

2.3. The hedonic pricing method

Emissions associated with LFG and leachate can be called variable externalities, in that they vary with the amount of waste received. On the other hand, disamenities associated with proximity to a landfill site can be called fixed externalities, in that they arise due to the mere existence of a landfill site, irrespective of the amount of waste received[3]. They are usually experienced by residents living in the vicinity of landfill sites (Fiehn and Ball, 2005), and are inversely related to distance from the site. This conceptual distinction between emissions and disamenities implies an important difference in the way that the associated costs are estimated. In the case of emissions, costs are estimated by deriving a relationship between waste quantities and the resulting monetary impact. On the other hand, valuing disamenities requires that a relationship is established between distance to the site and the resulting monetary impact. Two types of methods can be used to derive this relationship, namely

[3] Disamenities associated with traffic, such as accidents and congestion, are slightly different, in that they increase with the amount of waste received. However, these types of disamenities are dealt with as explained in Section 2.2. Section 2.3 deals only with disamenities associated with living in proximity to a landfill site.

‘stated preference’ methods such as contingent valuation and choice experiments (e.g. Roberts et al., 1991, Douglas, 1992, Garrod and Willis, 1998, Sasao, 2004); and the hedonic pricing method (HPM).

The HPM (Rosen, 1974) is based on the idea that the utility (‘satisfaction’) that individuals obtain from a particular good, and therefore the ‘value’ that they place on that good, is a function of the characteristics of the good. For example, property prices are based on characteristics such as house size, house age, number of rooms, proximity to amenities such as schools, etc. In addition, however, they are affected by aspects of environmental quality; such as air and noise pollution, as well as proximity to ‘disamenities,’ such as landfill sites (Du Preez and Lottering, 2009). To the extent that landfill sites generate disamenities to nearby residents (in the form of noise, odour, litter, vermin, dust and traffic); it is likely that these will be reflected in lower property prices for homes located in the vicinity of landfill sites, relative to homes located further away that are identical in all other respects (Fiehn and Ball, 2005, Du Preez and Lottering, 2009). Provided that all other factors affecting house prices or values can be controlled for, this difference in property prices as a result of proximity to a landfill site can be used to infer a monetary ‘cost’ for the disamenity impacts associated with living near a landfill site.

The HPM is a statistical method through which the effect of environmental quality variables (such as proximity to a landfill site) on property prices can be isolated from all other characteristics affecting property prices, based on data on house prices and characteristics for a large number of properties, using multiple regression. The dependent variable (house price or value) is regressed on a number of independent variables (characteristics affecting house prices or values, including the environmental quality variable in question, such as proximity to a landfill site; as well as other characteristics). In this way, the specific influence of proximity to the landfill site on house prices can be distilled, by holding all other characteristics constant. Once isolated in this way, the influence of the landfill site on property prices can be used to infer (in monetary terms) the impact on human well-being of the disamenities associated with proximity to a landfill site (European Commission, 2000a, Eshet et al., 2005).

A number of studies have used the HPM to estimate disamenity costs associated with landfill sites and other waste management facilities (e.g. Hirshfield et al., 1992, Nelson et al., 1992, Hite et al., 2001, Cambridge Econometrics et al., 2003, Eshet et al., 2007, Du Preez and Lottering, 2009). Brisson and Pearce (1995) review a number of such studies; all of which find a significant negative effect of landfill sites on house prices up to a distance of 4 miles (6.4km) from the site, with house prices increasing by an average of 5-7 percent per mile (3-4% per km) away from the landfill within this radius. This confirms the expectation that proximity to a landfill site will have a negative effect on house prices, as a result of the disamenity associated with living near a landfill site.

3. Data, results and discussion

3.1. Landfill gas and leachate emissions

Studies on landfill externalities tend to report either costs per tonne of waste, or costs per kg of emissions (unit costs); or, in some cases, both. Thus, when transferring estimates from one study site to another, one may use either costs per tonne of waste, and apply these to the waste tonnages at the site in question; or costs per kg of emissions, and apply these to emission quantities at the site in question. Where possible, the latter approach is preferable, because the link between emissions and costs to society is more direct than that between waste volumes and costs to society (see Figure 1); so there is less room for error when transferring estimates from one site to another. However, for sites where there is no monitoring of emissions, or insufficient data to calculate emission quantities, it will not be possible to use costs per kg of emissions. Since this is the case for the landfills in the City of Cape Town; it was necessary to use costs per tonne of waste, and to apply these to the waste tonnages at the three sites.

A number of studies have estimated external costs per tonne of waste associated with landfills. These include Powell and Brisson (1994), which was conducted in the UK context, and which in turn was based on estimates from CSERGE et al. (1993); Eunomia (2002) (Europe), Dijkgraaf and Vollebergh (2003) (the Netherlands), and, most recently, BDA Group (2009). However, Section 2.1 suggested that estimates should only be drawn from study sites which are sufficiently similar to the site in question. Most of the studies listed above were conducted in the UK or Northern Europe, where climatic conditions differ markedly from those found in Cape Town. There are also differences in terms of waste types considered, landfill design and operation standards, soil conditions, population densities, and the form of energy generation that would be displaced through energy recovery, suggesting that benefits transfer from most of these studies would be inappropriate.

The Australia study (BDA Group, 2009), on the other hand, was deemed appropriate, since a range of values are presented based on different LFG and leachate management practices, landfill locations (urban vs rural), and climatic conditions (including conditions similar to those found in Cape Town). Specifically, the average day-time temperature in Cape Town is

22 degrees Celsius, while average annual rainfall is approximately 500mm (World Meteorological Organization, 2011). This corresponds with the ‘wet temperate’ climate category used in the Australian study. All estimates will therefore be derived from BDA Group (2009), based on landfills situated in the ‘wet temperate’ climate zone, and making appropriate adjustments for other factors affecting costs.

Table 2 summarizes the range of monetary costs per tonne of waste for greenhouse gas emissions (CO_2 and CH_4), other air emissions (local air pollutants), and leachate estimated by BDA group (2009) for landfills in a wet temperate climate under a range of different assumptions. Specifically, for both greenhouse gases and local air pollutants, estimates are provided for landfills both with and without LFG capture and energy recovery. Note that the estimates for air emissions refer to LFG emissions only; transport emissions are not included. In addition, the estimates associated with greenhouse gases and local air pollutants under the energy recovery scenario are net of the benefits associated with displaced pollution from coal fired power generation. Estimates for greenhouse gases are based on an assumed unit cost of A$40 per tonne of CO_2 equivalent; a discount rate of 7% over an assumed 30 year landfill operating life and a 50 year post closure period; and an assumed 100-year global warming potential of methane of 21 (BDA Group, 2009)In addition, for local air pollutants, estimates are provided for landfills in both urban and rural locations, as impacts are highly dependent on the size of the exposed population. Finally, for leachate, estimates are provided for both older, unlined landfills; as well as more modern lined landfills. Estimates in Table 2 are provided both in 2008 Australian dollars (A$), as in the original study; and in 2009 South African Rands (R)[4].

[4] 1 US Dollar = approximately 7 South African Rands. 2008 Australian dollars were converted to 2009 Rands based on a 2008 average exchange rate of R6.93 / A$ (http://www.reservebank.co.za), and South African consumer price index inflation figures from http://www.statssa.gov.za/keyindicators/CPI/CPIHistory_rebased.pdf.

Table 2: Range of monetary costs per tonne of waste for different emissions (adjusted from BDA Group, 2009)

Pollutant	Greenhouse gases		Local air pollutants – urban		Local air pollutants – rural		Leachate	
Assumptions	No LFG capture	LFG capture with ER[a]	No LFG capture	LFG capture with ER	No LFG capture	LFG capture with ER	Unlined	Lined
2008 A$[b]	12.00	0.00	0.68	0.97[c]	0.21	0.09	0.02	0.01
2009 Rands	89.13	0.00	5.05	7.20	1.56	0.67	0.15	0.07

[a] ER = Energy recovery. In the "no LFG capture" scenario, it is implied that there is no energy recovery; while in the "LFG capture with ER" scenario, it is assumed that methane is collected and used to generate energy over the operating life of the landfill (but not after closure).
[b] A$ = Australian dollars
[c] Note that the estimate for local air pollution in an urban area is higher with LFG capture and ER than without. In the process of collecting methane and generating energy, although emissions of some components of LFG (including methane) are reduced, emissions of certain other pollutants, some of which have important health impacts (e.g. nitrogen oxides, carbon monoxide, sulphur dioxide and particulate matter), are added or increased (BDA Group, 2009).

Based on the discussion in Section 2.1 on the factors affecting emission factors and unit costs, the following factors need to be taken into account when transferring estimates from one study (in this case BDA Group, 2009) to another (the current study):

3.1.1. Waste types and composition

Both studies focus on the general municipal waste stream. In Australia, however , the organic waste fraction of municipal solid waste is approximately 47% by mass (Ministry for the Environment, 2009); while in Cape Town, the organic waste fraction is approximately 15% by mass (De Wit, 2011). Thus, assuming a one-to-one relationship between the organic waste fraction and the resulting impacts associated with LFG and leachate, all estimates from the Australia study need to be multiplied by 0.32 (= 15/47) to account for the lower organic waste fraction in Cape Town.

3.1.2. LFG and leachate management

At all three Cape Town sites, cells constructed prior to 1990 "were not designed using engineered liners or leachate detection layers, but rather were quite rudimentary in design, therefore the possibility of historical landfill sites polluting groundwater resources and causing soil contamination are real concerns" (Engledow, 2007: 8). However, this study is concerned with marginal costs (costs for the next tonne of waste entering the landfill); with the objective of providing information for assessing waste management options now and into the future. Historical leachate problems associated with waste entering landfills in the past should not enter into the analysis, since these are not affected by current decisions regarding waste management. Newer cells at both Bellville South and Coastal Park are designed with engineered linings, leachate detection and groundwater monitoring systems; while Vissershok has an on-site leachate treatment plant (Engledow, 2007). Leachate associated with waste entering all three landfill sites is therefore being managed according to international standards. Thus, in terms of leachate management, the estimate for 'lined' landfills in Table 2 is appropriate.

On the other hand, none of the three sites currently capture LFG, let alone recover methane to generate energy; although there are plans in place for such a system at all three sites (Dentlinger, 2008, Van der Merwe, 2009, Njobeni, 2010). Thus, for scenario 1 (status quo), we will adopt the estimates associated with no LFG capture or energy recovery for both greenhouse gases and local air emissions. On the other hand, for scenarios 2 and 3, both of which incorporate LFG capture and energy recovery, we will adopt the estimates corresponding to these assumptions.

3.1.3. Climate

Since all estimates used here are derived from the Australian climate category that corresponds with climatic conditions found in Cape Town, there is no need to adjust estimates for differences in climatic conditions.

3.1.4. Factors affecting pollution displacement benefits

BDA Group (2009) assume a 60% LFG collection efficiency, an energy content of methane of 55.52 MJ/kg, a conversion factor of 3.6 MJ/KWh, and a conversion efficiency of methane to electricity of 30%. However, since energy recovery systems are not yet operational in Cape Town, there is currently insufficient information to judge whether these assumptions are applicable in the South African context. As such, the estimates will not be adjusted for possible differences in LFG capture and energy conversion technology.

In addition, BDA Group (2009) assume that energy from coal-fired power stations is being replaced. This assumption is applicable to South Africa, where energy is generated predominately by coal-fired power stations. Thus, no adjustments will be made for factors affecting pollution displacement benefits.

3.1.5. Site location and physical characteristics

Since the impacts of CO_2 and CH_4 on the global climate are independent of their source, there is no need to adjust the estimates for greenhouse gases for site location and physical characteristics (European Commission, 2000b). On the other hand, local air pollution impacts are determined by distance to human populations, topography, and prevailing meteorological conditions, e.g. wind directions (Eshet et al., 2006). Likewise, impacts associated with leachate emissions are relatively site-specific and should therefore be adjusted for site location and physical characteristics (European Commission, 2000b). In Cape Town, the Bellville South and Coastal Park sites are both located in areas with a positive water balance, implying that they are likely to produce significant amounts of leachate (Engledow, 2007). Indeed, Parsons (2002) "presents evidence suggesting that the Bellville South waste site… has in fact impacted on the Cape Flats Aquifer system" (Engledow, 2007: 8). However, adjusting for differences in site location and physical characteristics is somewhat difficult in this case, as the BDA Group (2009) study covers a representative range of different landfill sites in Australia, which are likely to vary significantly in terms of location and physical characteristics. Thus, it was not possible to adjust for site location and physical characteristics in this study.

3.1.6. Socio-economic characteristics

As mentioned in Sections 2.1 and 3.1.5; for greenhouse gases, climate impacts are independent of the source of emissions. As such, the estimates for greenhouse gases in Table 2 do not need to be adjusted for site-specific socio-economic characteristics.

For local air pollutants and leachate, on the other hand, monetary valuations depend on the size of the exposed population, which depends on population density in the area surrounding the landfill; as well as willingness to pay, which in turn depends on both income (ability to pay) and tastes/preferences for environmental quality. These are likely to vary significantly between countries (European Commission, 2000a); particularly between developed and developing countries. Therefore, for local air emissions and leachate; it will be necessary to adjust for population density, income and preferences for environmental quality.

As with site location and physical characteristics, adjusting for population density is somewhat difficult in this case, as the original study covers a range of different landfill sites in Australia. Although separate estimates are provided for local air pollutants in urban as opposed to rural sites, the study is not explicit regarding exactly which sites are included, or their respective population densities. As such, it was not possible to adjust for differences in population density between urban sites in the original study area and in Cape Town; and likewise for rural sites. Instead, the implicit assumption is that population densities around urban sites are similar between the two studies; and likewise for rural sites.

Relative income and preferences for environmental quality can be adjusted for simultaneously using the following formula (European Commission, 2000a):

$$\overline{E}_{SA} = E_{AUS} \times \left(I_{SA} / I_{AUS} \right)^{Q} \tag{1}$$

Where $\overline{E}_{SA}$ = The adjusted estimate (applicable to South Africa)

E_{AUS} = The original estimate (from Australia)

I = Per capita income at purchasing power parity (PPP) rates

Q = Income elasticity of demand for environmental quality

In order to adjust for relative income between countries (I_{SA}/I_{AUS}), it is important to take into account not only differences in income levels, but also differences in purchasing power (the goods and services that can be purchased per unit of currency), which reflect differences in prices (even after adjusting for exchange rates). This can be done by adjusting the estimates using income at purchasing power parity (PPP) rates, which adjust for differences in purchasing power between countries (European Commission, 2000a). Per capita gross national income (GNI) at PPP rates for 2008 (the year to which the Australia data applies) were obtained from the World Bank (2011). SA's per capital GNI (PPP rates, in current international dollars) for 2008 was 10,140, compared to 35,720 for Australia. I_{SA}/I_{AUS} is therefore 0.28 (10,140 / 35,720).

Adjusting for preferences for environmental quality requires that the income elasticity of demand for environmental quality, (Q, that is, the responsiveness of demand for environmental quality to changes in income) must be estimated or assumed. Few studies have attempted to estimate the income elasticity of demand for environmental quality, and results differ widely (European Commission, 2000a). It is often assumed that environmental quality is a 'luxury' good (i.e., that demand for environmental quality increases as income increases); implying that Q is greater than 1 (European Commission, 2000a). However, "very few studies support the view that income elasticities of demand for environmental quality is greater than 1, and a number of studies suggest income elasticities in the order of 0.3" (European Commission, 2000a: 75). Given the lack of agreement in the literature, this study will assume that $Q = 1$. Thus, the Australia estimates for local air emissions and leachate only need to be multiplied by I_{SA}/I_{AUS} (= 0.28) in order to account for socio-economic differences.

3.1.7. Synthesis: Adjustments and scenario analysis

Based on all of the factors discussed in Sections 3.1.1 to 3.1.6, the estimates from BDA Group (2009) (see Table 2) were adjusted as shown in Table 3. All estimates need to be adjusted by a factor of 0.32 to account for differences in waste composition. In addition, the estimates for local air pollutants and leachate need to be adjusted by a factor of 0.28 to account for socio-economic differences. Thus, the net result is that estimates for greenhouse gases must be multiplied by 0.32, while estimates for local air pollutants and leachate must be multiplied by 0.09 (= 0.32 x 0.28).

Table 3: Adjustments to original estimates (in 2009 R per tonne of waste) for transfer to the current study

Pollutant	Greenhouse gases		Local air pollutants – urban		Local air pollutants - rural		Leachate[a]	
Assumptions	No LFG capture	LFG capture with ER	No LFG capture	LFG capture with ER	No LFG capture	LFG capture with ER	Unlined	Lined
Original estimate	89.13	0.00	5.05	7.20	1.56	0.67	0.15	0.07
Adjustment factor	0.32	0.32	0.09	0.09	0.09	0.09	0.09	0.09
Adjusted estimate	28.44	0.00	0.46	0.65	0.14	0.06	0.01	0.01

[a] Due to rounding, the adjusted estimates for leachate in lined and unlined landfills appear equivalent (0.01). In reality, the cost associated with unlined landfills (0.0135) is roughly double that for lined landfills (0067).

In Table 4, the relevant estimates are applied to each of the three scenarios used in the current study, namely:

1. Status quo (existing urban landfills, no energy recovery)
2. Existing urban landfills with energy recovery
3. New regional landfill with energy recovery.

Table 4: Costs (in 2009 Rands) per tonne of waste for each emission under each scenario

	Greenhouse gases	Local air pollutants	Leachate	Total
Scenario 1	28.44	0.46	0.01	28.91
Scenario 2	0.00	0.65	0.01	0.66
Scenario 3	0.00	0.06	0.01	0.07

3.2. Transport externalities

At present, municipal waste in the City of Cape is transported by road to one of the municipal landfill sites; or to one of two transfer stations (Athlone or Swartklip), from where it is compacted and transported by rail to the Vissershok landfill site, which is situated further from the city centre than Bellville South and Coastal Park (Taiwo, 2009). Under the 'regional landfill' scenario, however, all waste will be transported to one of the transfer stations, and then to the new landfill site, which will be located even further from the city centre than the Vissershok site. An increase in transport externalities can therefore be expected under scenario 3 relative to scenarios 1 and 2.

As mentioned in Section 2.2, it was necessary to make a number of assumptions regarding distances travelled (using Google Earth), and regarding external costs per tonne-km (based on a meta-analysis of external costs associated with freight transport in South Africa (Jorgensen, 2009)). Under the status quo, it was assumed that 2 million tonnes of waste is transported annually over an average distance of 15km by road to the nearest landfill site or transfer station. In addition, according to City of Cape Town (2008b) and Taiwo (2009), fifty 20-tonne containers of compacted waste are transported each day by rail from the transfer stations to the Vissershok site (approximately 20km from the Athlone transfer station and 30km from Swartklip). This amounts to 365,000 tonnes of waste transported an average distance of 25km. Thus, based on an average external cost (including emissions, accidents, congestion and noise) of 15.67 cents per tonne-kilometre for road freight transport, and 1.59 c / tonne-km for rail transport[5] (Jorgensen, 2009), the annual external cost associated with transporting waste to landfill is R4.84 million; or R24.22 per tonne of waste. This figure applies to scenarios 1 and 2, which are both based on the current system of urban landfills.

For scenario 3, the new regional site will be located either at Atlantis, which is on average 45km from the transfer stations; or at Kalbaskraal, an average distance of 55km from the

[5] c = South African cents (R1 = 100c). A tonne-km is a unit of measure of freight transport representing one tonne of goods transported over 1 kilometre.

transfer stations. Presumably, compacted waste will be transported from the transfer stations to the landfill by rail. Thus, in addition to being transported 15km by road to one of the transfer stations, the two million tonnes of waste generated annually will also be transported a further 50km (on average) by rail from the transfer station to the new landfill site. Based on the same average external costs referred to above, this equates to R6.28 million per annum, or R31.42 per tonne of waste. However, note that these results are heavily reliant on a number of crude assumptions; notably that in scenario 3 all waste will be transferred to the regional landfill site by rail, for which external costs are significantly lower than for road transport.

3.3. Disamenities

Data for a hedonic pricing study can be obtained based on actual sales within a specific time period (e.g. Nelson et al., 1992), interviews with real estate agents (e.g. Hirshfield et al., 1992), or based on market valuations of a sample of properties within the area (Du Preez and Lottering, 2009). In this study, data was obtained on both actual house sales and market values, based on two sources of information:

1. CMAinfo (2010), an internet-based subscription database service to the real estate industry in the City of Cape Town. It allows for comparative market analysis of any property within the City of Cape Town, by providing information on property characteristics, market value as per the City's property valuation roll, and selling price. In addition, a list of recent sales within a specific radius of a given property can be generated. Among the information provided on each recent sale is sale date, sale price, property size, erf (property) number (which can in turn be used to generate information on additional property characteristics, such as house size), and, importantly, distance

from the reference property. Thus, specifying each landfill site as the 'reference property' in turn, it was possible to obtain information on sale prices and property characteristics (including distance from the landfill site) for a large number of recent sales within a given radius of each site.

2. The City of Cape Town's 2009 property valuation roll (City of Cape Town, 2009), which provides information on the most recent property valuations undertaken by the City. Information is provided on both the market value itself, as well as a number of property characteristics (number of bedrooms, number of stories, house age, house size, etc) underlying the valuation. This data was used to supplement and verify the data found on CMAInfo. In most cases, the data was found to be consistent.

For each of the three landfill sites, the following procedure was followed. First, CMAInfo was used to generate a list of recent sales within a 4km radius of the site. This choice of radius was based on a review of the literature. Although effects have been found up until 4 miles (6.4km) (Roberts et al., 1991, Brisson and Pearce, 1995), some studies suggest that there is no effect beyond 2 – 2.5 miles (3.2 – 4km) (Nelson et al., 1992, Du Preez and Lottering, 2009), and that using too wide a radius may reduce the explanatory power of the model, particularly if the effect is limited to a relatively short distance away from the site.

CMAInfo only displays the 15 most recent sales within a specified radius. As such, it was necessary to begin with a relatively small radius, and then incrementally increase the specified radius so that a sufficiently large sample could be obtained. Unfortunately, a sample generated in this way will have a disproportionate number of properties closer to the landfill site relative to the actual distribution. As the radius increases, the actual number of properties (and recent sales) increases; but only the 15 most recent sales will be shown. Hence, at closer radii, a number of relatively older sales will be captured; whereas at further radii, a number of relatively recent sales will be missed. Nevertheless, in the absence of an alternative means of generating a sample of properties at varying distances from the sites, this had to suffice. In any case, this distribution pattern was not used as the basis for aggregating the sample data to the population; so this was not expected to bias the results.

In this way, data was obtained on 101 recent sales within a 4km radius of the Bellville South site, and 104 for Coastal Park; but only 40 for Vissershok, where housing densities are far lower (properties in this area are typically farms or smallholdings, as opposed to the more densely populated residential areas around the Bellville South and Coastal Park sites). As such, the Vissershok model was dropped from the HPM analysis, and will not be discussed further in this section.

Regressions were run using both actual selling price and market value (as per the 2009 municipal valuation) as the dependent variable (with a slightly different set of independent variables in each case, as appropriate), and compared. It was found that, for both Bellville South and Coastal Park, the model fits the data better (i.e. R^2 is higher) when market value is used as the dependent variable, as opposed to price. Thus, it was decided to use market value as the dependent variable in the analysis.

The purpose of the HPM model is to isolate the influence of proximity to a landfill site on property values, after controlling, to the extent possible, for other property characteristics that could affect value. As such, it is important to include as many characteristics of the property as possible as independent variables in the model. Based on the characteristics for which information was available on CMAInfo and the City's valuation roll, the following model was posited, explaining how variations in property values are determined for properties in the Bellville South and Coastal Park samples:

$$
\begin{aligned}
Value = \ & \beta 0 + \beta 1\ (ErfSize) + \beta 2\ (HouseSize) + \beta 3\ (Formal) + \beta 4\ (HouseAge) \\
& + \beta 5\ (Condition) + \beta 6\ (Storeys) + \beta 7\ (Bedrooms) + \beta 8\ (Roof) \\
& + \beta 9\ (Common) + \beta 10\ (WallType) + \beta 11\ (Garages) + \beta 12\ (Other) \\
& + \beta 13\ (Distance) \qquad (2)
\end{aligned}
$$

Where *Value* is the dependent variable (market value as per the 2009 provisional valuation roll, in Rands) and β_0 is the intercept term; and where $\beta_1 - \beta_{13}$ are the parameters on the independent variables, as defined in Table 5. Note that these independent variables (with the exception of *Distance*) correspond to the range of characteristics taken into account by the City in its property valuations, confirming their appropriateness for inclusion in the model. *Distance*, as a proxy for exposure to disamenities associated with living near a landfill site, is effectively a quality variable.

Table 5: Definitions of independent variables in the HPM model

Variable	Definition	Expected sign
ErfSize	Size of erf (property) in m^2	Positive
HouseSize	Size of house in m^2	Positive
Formal	Dummy variable for whether the dwelling can be characterised as formal (1) or informal (0) housing	Positive
HouseAge	Age of house in years	Negative
Condition	Condition of house (defined as exceptional (7), excellent (6), very good (5), good (4), average (3), to renovate (2) or to remodel (1))	Positive
Storeys	Number of storeys	Positive
Bedrooms	Number of bedrooms	Positive
Roof	Dummy variable for type of roof covering; tile (1), as opposed to sheeting or other (0)	Positive
Common	Number of common walls (walls shared with another property)	Negative
WallType	Dummy variable for type of exterior wall; brick or plastered (1) as opposed to concrete block or other (0)	Positive
Garages	Number of garages (including carports and parking bays, which are assumed to count as 'half' a garage)	Positive
Other	Dummy variable for presence (1) or absence (0) of other features (e.g. swimming pool, security, 2nd dwelling on property, etc.)	Positive
Distance	Distance from landfill site in km	Positive

The above-mentioned model was run for both sites using a range of functional forms, namely linear, lin-log, log-lin, and double-log. For both Bellville South and Coastal Park, goodness of fit in terms of R^2 was above 0.8 across all functional forms, suggesting that the model fits the data very well. In both cases, however, the highest R^2 was obtained with the linear model. Thus, in both cases, the linear functional form was selected. The final form of the hedonic price functions (after omitting variables that are individually statistically insignificant, or that do not follow the expected sign) are as follows. For Bellville:

$$Value = \beta_0 + \beta_1(ErfSize) + \beta_2(HouseSize) + \beta_4(HouseAge) + \beta_8(Roof) + \beta_{10}(WallType) + \beta_{12}(Other) + \beta_{13}(Distance) \tag{3}$$

And for Coastal Park:

$$Value = \beta_0 + \beta_1(ErfSize) + \beta_2(HouseSize) + \beta_{10}(WallType) + \beta_{11}(Garages) + \beta_{13}(Distance) \tag{4}$$

Regression results for the final HPM models are presented in Tables 6 and 7.

Table 6. Regression results for Bellville South (n = 101, R^2 = 0.8911, adjusted R^2 = 0.8829)

	Coefficient	Standard Error	t Stat	P-value
Intercept	166553.6291	27232.7057	-6.1159	0.0000
ErfSize	630.0730	120.9527	5.2093	0.0000
HouseSize	3175.0848	492.1019	6.4521	0.0000
WallType	60880.4860	17661.7955	3.4470	0.0008
Garages	118651.8709	20842.2271	5.6929	0.0000
Distance	22904.7635	17251.5943	1.3277	0.1874

Table 7. Regression results for Coastal Park (n = 104, R^2 = 0.9102, adjusted R^2 = 0.9056)

	Coefficient	Standard Error	t Stat	P-value
Intercept	21377.6620	24502.1355	0.8725	0.3852
ErfSize	259.6496	47.3841	5.4797	0.0000
HouseSize	2414.0623	240.4809	10.0385	0.0000
HouseAge	-2203.3848	799.8161	-2.7549	0.0071
Roof	101822.0954	16175.9088	6.2947	0.0000
WallType	44909.2973	14143.4447	3.1753	0.0020
Other	68644.6893	21508.5006	3.1915	0.0019
Distance	6940.4432	10752.2161	0.6455	0.5202

In the case of both Bellville South and Coastal Park, the overall explanatory power of the model is good (R^2 = 0.89 and 0.91 respectively); implying that 89% of the variation in market values is explained by the independent variables in our model for Bellville South, and 91% for Coastal Park. The coefficients on the explanatory variables all follow the expected signs and, with the exception of the distance variable, are each individually statistically significant. The coefficients on the distance variables imply that, in the case of Bellville, property values increase by R6,940 (about 2%) for each km away from the landfill, *ceteris paribus*. In the case of Coastal Park, as one moves away from the landfill site, property values increase by R22,905 (12%) per km, *ceteris paribus*.

For Bellville, substituting the estimated coefficients and mean values of each explanatory variable (with the exception of *Distance*) into the hedonic price function (Equation 3), the following value-distance function results:

$$Value = 318{,}469 + 6{,}940 \times Distance$$

This implies that the average value of a property adjacent to the landfill site will be R318,469, as compared to R325,409 for a property located 1km from the site, R332,350 for a property located 2km from the site, R339,290 for a property located 3km from the site, and

R346,231 for a property located 4km from the site. In other words, properties adjacent to the Bellville South site are, on average, 8% lower in value as compared to properties located 4km away (i.e., relative to properties located 4km from the site, there is on average a 2% decline in property values per km closer to the site).

Likewise, for Coastal Park (Equation 4), the following value-distance function is obtained:

$$Value = 147{,}517 + 22{,}905 \ \times Distance$$

In this case, the estimated average value of a property adjacent to the landfill boundary is R147,517, as compared to R170,422 at a distance of 1km, R193,327 at 2km, R216,231 at 3km and R239,136 at 4km. Thus, in this case, the effect of the landfill site on property values is far more dramatic. The value of a property adjacent to the Coastal Park site is, on average, 38% lower than that of a property located 4km away (i.e., an approximate decline of 9.5% per km). Thus, while the results for Bellville South are in line with other studies, which suggest a 1 – 6% reduction in house prices per km (see Brisson and Pearce, 1995, Eshet et al., 2006); the results for Coastal Park are well above this range.

These results regarding the influence of the landfill sites on the average value of properties were then aggregated to all properties within the 4km radius of the two sites, as follows. First, based on Geographical Information Systems (GIS) data from the City's Strategic Development Information and GIS Department (Williams, 2010), the actual number of properties within the 4km radius of each site was determined (59,284 for Bellville South, which is situated in a densely populated urban area, and 30,821 for Coastal Park, which is situated in a less densely populated area along the coast).

Then, a uniform spatial distribution of properties away from the landfill site was assumed[6], based on the area of each concentric circle (0-1km, 1-2km, 2-3km, and 3-4km) around the landfill site. For example, for Bellville South, since the area of a circle with 1km radius is 6.25% of the area of a circle with 4km radius (3.14km^2 / 50.27km^2), 6.25% of the properties were assumed to be situated within 1km of the site; etc (see Table 8). The same logic was applied to Coastal Park; although in this case it was assumed that there were no properties within a 1km radius of the site, because there were no properties within this radius in the sample. Thus, in distributing the properties based on the area of each of the three outer concentric rings, the area of the inner 0-1km circle was ignored. Although making the assumption of a uniform distribution is less than ideal, it seems to be standard practice in the absence of more accurate information (see e.g. European Commission, 2000a).

Then, based on the average reduction in value (relative to those located 4km away) for properties within each 1km concentric zone from the landfill site (based on the mid-point within each zone), the total reduction in value due to the landfill was calculated to be approximately R566 million for Bellville South, and R871 million for Coastal Park (see Tables 8 and 9). Although there are fewer properties located within the vicinity of Coastal Park relative to the Bellville South site, the effect of the former on average property values is far more dramatic, resulting in a higher overall disamenity cost associated with Coastal Park as compared to Bellville South.

[6] Although it seems sensible to assume that the distribution of properties in the population corresponds to that in the sample, this was not feasible in the current study because of the way the sample was drawn (see above), which resulted in a disproportionate sampling of properties closer to the landfill site relative to the actual distribution.

Table 8: Aggregation of average disamenity costs to all properties within 4km of Bellville South landfill site

Distance	Area (km^2)	% of total area	No. of properties	Loss of value per property (R) relative to those 4km away	Total disamenity cost (R)
0-1km	3.14	6.25	3 705	24 291.55	90 000 197
1-2km	9.42	18.75	11 116	17 351.11	192 874 915
2-3km	15.71	31.25	18 526	10 410.66	192 867 975
3-4km	21.99	43.75	25 937	3 470.22	90 007 137
Total	50.27	100.00	59,284		565 750 224

Table 9: Aggregation of average disamenity costs to all properties within 4km of Coastal Park landfill site

Distance	Area (km^2)	% of total area	No. of properties	Loss of value per property (R) relative to those 4km away	Total disamenity cost (R)
1-2km	9.42	20.00	6 164	R 57 261.91	352 962 413
2-3km	15.71	33.33	10 274	R 34 357.15	352 985 359
3-4km	21.99	46.67	14 383	R 11 452.38	164 719 582
Total	47.12	100.00	30,821.00		870 667 354

Summing across the two sites, the total disamenity cost imposed on residents living in the vicinity of municipal landfill sites in the City of Cape Town (excluding the Vissershok site) is R1.4 billion. This can be translated to an annual value, based on the "average annual write off of the overall reduction in the real estate price" (European Commission, 2000a: 155). Assuming that 8% of the total reduction is equal to an annual value, the disamenity cost imposed on residents is approximately R115 million per annum.

Assuming 2 million tonnes of waste going to landfill per year (note that this includes Vissershok – we will assume that the disamenity cost associated with this site is zero, and therefore that all disamenity costs associated with waste going to landfill in Cape Town arises at Bellville South and Coastal Park); the disamenity cost per tonne of waste going to landfill in Cape Town is R57.46 per tonne[7] (R115 million / 2 million tonnes). In terms of our

[7] Even though disamenity costs are a fixed externality (i.e. they do not vary with the amount of waste landfilled) and should therefore not strictly be presented as a cost per ton of waste (Powell and Brisson, 1994), doing so is useful for the purposes of comparing the landfill scenarios (so as to be commensurable with emission costs estimated in Section 3.1).

scenarios, this estimate is appropriate for the status quo scenario (existing urban landfills without energy recovery).

For scenario 2 (existing urban landfills with energy recovery); since certain disamenities (such as odours, fire and explosion risks, etc) are strongly correlated with methane emissions (Dijkgraaf and Vollebergh, 2003), we can expect that disamenity costs will be reduced to some extent with energy recovery, as a result of reduced methane emissions. However, it is difficult to assess the exact magnitude of this effect. Assuming a 60% LFG collection efficiency (as per BDA Group, 2009), we could expect that methane emissions would be reduced by approximately 60% in the energy recovery scenario. However, this does not imply that disamenity costs will be reduced by 60% under energy recovery, since methane emissions are not the only source of disamenities. Assuming that 50% of total disamenity costs are associated with odours and other methane-related problems, we might expect that disamenity costs under scenario 2 will be reduced by 30% (50% of 60%) relative to scenario 1. Thus, for scenario 2, disamenity costs are R40.22 (R57.46 – 30%) per tonne of waste.

For scenario 3 (new regional landfill), disamenity costs are likely to be negligible, owing to expected lower population and housing densities in the vicinity of the site, implying that fewer people and properties will be affected (Powell and Brisson, 1994). Indeed, discussions around the location of the new site have emphasised the need to avoid disamenity impacts, to the extent possible (Ministry of Local Government Environmental Affairs and Development Planning, 2009). Thus, for the sake of consistency, we will assume that the disamenity cost associated with this site is zero, as was the case for Vissershok.

However, a number of limitations affecting the HPM study must be borne in mind. Firstly, the analysis is limited to property price effects within 4km of the Bellville South and Coastal Park sites; the effect on properties located further than 4km is excluded, as is the effect on all properties in the vicinity of Vissershok and the new regional site (although both are or will be situated in less densely populated areas). In addition, the effect of other forms of disamenity that may not be captured in property values; is ignored. Finally, it should be noted that landfills in the City of Cape Town have historically been sited in the vicinity of poorer communities (Engledow, 2007, Von Weizsacker et al., 2009). Thus, it could well be that properties in the vicinity of landfill sites are of lower value simply because they are situated in poorer areas; rather than owing to the effect of the landfill site itself. Nevertheless, the results are in line with other studies of disamenity costs, where typical estimates are in the region of US$ 10 per tonne of waste (Eshet et al., 2006).

3.4. Synthesis

The results for emissions and disamenity costs per tonne of waste are summed and compared across the three scenarios in Table 10. Note that these results (both in terms of the overall magnitude of the costs, and in terms of the relative contributions to overall costs of the different components), are in line with other results from the international literature. For example, most studies have found that disamenities contribute half to two thirds of total external costs for urban landfills (e.g. see the meta-analyses presented in European Commission, 2000b, Eshet et al., 2006, BDA Group, 2009); which is also the case here, at least for scenarios 1 and 2.

Table 10: Comparison of net costs per tonne of waste across the three scenarios (2009 Rands per tonne)

	Emissions	Transport	Disamenity	Total
Scenario 1	28.91	24.22	57.46	110.59
Scenario 2	0.66	24.22	40.22	65.10
Scenario 3	0.07	31.42	0.00	31.49

The results in Table 10 suggest that, purely from the perspective of external costs, scenario 3 (new regional landfill with energy recovery) is preferable to scenarios 1 (status quo – existing urban landfills without energy recovery) and 2 (existing landfills with energy recovery). In other words, the expected increase in transport externalities associated with the new regional landfill will be outweighed by the significant reductions in costs associated with LFG emissions and disamenities. Note, however, that this result is highly dependent on two key assumptions regarding scenario 3; namely that (i) waste will be transported from the transfer stations to the regional landfill site by rail, for which transport externalities are significantly lower than for road transport; and (ii) disamenity costs associated with the regional landfill site will be negligible, implying that the site will not be located in the vicinity of human populations.

Finally, in order to properly compare the three scenarios, financial costs also need to be taken into account. In addition, landfilling should be compared with alternative waste management options (such as recycling) on the basis of overall costs to society (financial and external costs). However, estimating financial costs associated with landfilling, as well as financial and external costs associated with alternatives to landfilling, was beyond the scope of this study. This should be the subject of future research.

4. Conclusions and recommendations

This study has estimated the external costs associated with landfilling in the City of Cape Town. This information could be useful to decision makers when comparing waste management options, which should ideally be done on the basis of overall costs to society (including external costs). However, the actual estimates derived in the study must be treated with caution, owing to the various assumptions made in quantifying external costs in the respective scenarios (these will not be repeated here, as all assumptions have been made explicit in the text). The estimates should therefore be seen as providing an indication of the order of magnitude of the externalities, rather than as exact values (Eshet et al., 2005, 2006). At the very least, sensitivity analysis regarding key assumptions would be required in order for the estimates to be used in decision making or for benefits transfer to other study areas (Eshet et al., 2005). Even then, although estimates of the costs associated with externalities can be seen as providing one piece of the puzzle, they do not show the whole picture. Instead, they should be used alongside other sources of information in order to fully assess alternative waste management options (Eshet et al., 2005).

There is therefore an urgent need for further research to help support the estimates provided here, and to underpin future valuation exercises. In particular, there is a lack of quantitative data on the physical quantities underlying the economic estimates, e.g. regarding emission levels, transport distances, etc. There is also a need to assess external costs associated with alternatives to landfilling, so that all alternatives can be compared on the basis of their overall costs to society (as well as on other criteria). Finally, there is a lack of understanding among South African municipalities even regarding the financial costs of waste management, let alone external costs; owing to a failure to implement the principles of full cost accounting[8],

[8] Unlike cash flow accounting, which records cash outlays (expenditures) as they occur, FCA records the actual use or commission of resources, regardless of when money is spent. This is important because significant expenditures are incurred before and after the operating life of specific facilities and services. FCA therefore accounts for the full monetary cost of resources used or committed to MSW activities (direct and indirect

particularly with respect to allowances for closure and post-closure care (Nahman and Godfrey, 2008, Goldblatt, 2009). There is therefore an urgent need to understand how the principles of full cost accounting can be adopted by municipalities in a consistent and coherent way, in order to properly assess the financial costs associated with all waste management options.

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