PERGAMON

Waste Management 23 (2003) 599–607

waste management

www.elsevier.com/locate/wasman

Implications of the spatial variability of landfill emission rates on geospatial analyses

K. Spokas[a,*], C. Graff[a], M. Morcet[b], C. Aran[b]

[a]*University of Minnesota, Department of Soil, Water, and Climate, 1991 Upper Buford Circle, 439 Borlaug Hall, St. Paul, MN 55108, USA*

[b]*Centre de Recherches pour l'Environnement l'Energie et le Dechet (CREED), Veolia Environnement, Zone Portuaire de Limay, 291 Avenue Dreyfous Ducas, 78520 Limay, France*

Accepted 16 June 2003

Abstract

Accurate methods quantifying whole landfill surface flux of methane are important for regulatory and research purposes. This paper presents the results from the analysis of chamber measurements utilizing geospatial techniques [kriging and inverse distance weighting (IDW)] to arrive at an estimation of the whole landfill surface flux from the spatially distributed chamber measurement points. The difficulties in utilizing these methods will be discussed. Methane flux was determined on approximately 20 m grid spacing and variogram analysis was performed in order to model spatial structure, which was used to estimate methane flux at unsampled locations through kriging. Our analysis indicates that while the semi-variogram model showed some spatial structure, IDW was a more accurate interpolation method for this particular site. This was seen in the comparison of the resulting contour maps. IDW, coupled with surface area algorithms to extract the total area of user defined contour intervals, provides a superior estimate of the methane flux as confirmed through the methane balance. It is critical that the results of the emissions estimates be viewed in light of the whole cell methane balance; otherwise, there is no rational check and balance system to validate the results.

1. Introduction

1.1. Methane

There are several contributors to the increasing amount of methane in the atmosphere. These include natural wetland emissions and anthropogenic sources from rice production, enteric fermentation from ruminant livestock, and landfilling of solid wastes (Bingemer and Crutzen, 1987; Bogner et al., 1995; Bogner and Matthews, 2003; Gourlay, 1992; Neue et al., 1994). Methane gas alone contributes approximately 15% to the potential global warming estimates (OTA, 1991). Landfills have been implicated as a major source of atmospheric methane emissions (Kreileman and Bouwman, 1994), comprising about 11% of the total anthropogenic global methane contribution (Blaha et al., 1999; Boeckx et al., 1996). Current estimates based on estimates of solid waste landfilled, the quantity of methane generated, and the net methane emission, suggest that the annual global methane emission from landfills is between 14 and 40 Tg (Boeckx et al., 1996; Bogner and Matthews, 2003). However, these estimates are the results of modeling versus direct measurement. Measured methane emission rates vary over seven orders of magnitudes (0.0004–4000 g m^{-2} d^{-1}) (Bogner and Spokas, 1993; Bogner et al., 1997; Czepiel et al., 1996).

Methane emissions from landfills can be controlled or mitigated. Combinations of installed gas recovery systems as well as the natural attenuation potential of various engineered covers control the rate of gaseous emissions from the landfill surface (Bogner et al., 1995; Christopherson et al., 2000; Mosher et al., 1999). Previous field efforts have shown a significant portion (30–100%) of the methane present in the cover is oxidized to carbon dioxide by indigenous methanotrophic bacteria within the soil cover materials (Bogner et al., 1995; Christopherson, 2000; Jager and Peters, 1985; Mancinelli, 1995; Whalen et al., 1990). In addition, field studies have also shown that if the methane flux is less than the methane oxidizing capacity of the landfill cover, the soil cover bacteria would oxidize atmospheric methane as it diffuses into the soil (Bogner et al., 1995).

* Corresponding author. Tel.: +1-612-625-1798; fax: +1-612-625-2208.

E-mail address: kspokas@soils.umn.edu (K. Spokas).

0956-053X/$ - see front matter
doi:10.1016/S0956-053X(03)00102-8

1.2. Whole landfill emission rates

The estimation of the whole surface emission rate can be problematic when it is measured by limited discontinuous surface chambers. The complication results from the heterogeneity of the resulting flux measurements across the surface of the landfill (Bogner et al., 1995; Cardellini et al., 2003; Jones and Nedwell, 1993; Mosher et al., 1999). The emission of methane from the surface of the landfill is a complex interaction of biological, chemical, and physical processes occurring within the landfill cover soils with all of these processes varying on different spatial and temporal scales. The spatial variation in soil permeability, air-filled porosity, methane concentration in the soil gas, moisture content, and atmospheric pressure all affect methane emission rates. Recently, it has been shown that a barometric pressure decrease of 10 millibars caused a tripling of the methane emissions from a landfill (Cziepel et al., 2003). However, this effect will be site specific. Studies have also shown that the heterogeneity in the surface flux can be related to the distribution of animal burrows in the cover soils (Giani et al., 2002). These burrows create large macropores which act as conduits for increased methane transport.

It can take several days to collect enough chamber flux measurements to describe large landfill surface areas, which could mean changes in the spatial distribution of the flux while measurements are still being collected (Mosher et al., 1999). With a lower sampling density, a regular grid pattern can provide a better estimation than random or cellular stratified sampling schemes (Wang and Qi, 1998). Reliability increased for all sampling techniques with the number of points used in the model (Wang and Qi, 1998).

Commonly the arithmetic mean multiplied by the surface area has been used in past efforts due to the fact that this provides an unbiased estimate regardless of the underlying distribution (Bogner et al., 1995; Cardellini et al., 2003; Livingston and Hutchinson, 1995). However, this technique can bias the estimate for the surface emission depending on the spatial variability and spatial extent of the higher flux regions of the surface. Often there are a few higher flux measurements (often spatial clustered), utilizing the arithmetic mean can over—or underestimate the surface emission rate since all measurements are equally weighed regardless of the ‘‘hot spot’’ area. Heterogeneity spanning four orders of magnitude has been measured on a single site (Pokryszka et al., 1995).

Soil moisture also has a significant controlling role for the methane emissions. Moisture contents of 15–30% (w/w) have been found to be optimal for methane oxidation activity (Boeckx et al., 1996; Giani et al., 2002). In addition, as the soil moisture increases the available pore space for gaseous transport and diffusion is reduced. Both of these factors impact the resulting methane emission. Variability in landfill cover thickness has also been shown to effect the resulting emission of methane to the atmosphere (Nozhevnikov et al., 1993), with thicker covers reducing methane flux.

1.3. Geographical information systems

Geographical information systems (GIS) have been used primarily in the site screening for landfill locations (e.g. Charnpratheep et al., 1997) and assessing the demand for solid waste disposal sites in urban cities (e.g. Leao et al., 2001). GIS has also been used for data presentation and visualization of the spatial relationship between soil gas probe readings (Moore et al., 1995), as well as to visualize the impact of landfills on the methane concentrations in urban areas, indicating that the urban landfills are a major source of elevated methane concentrations (Ito et al., 2001).

The use of geospatial models to estimate the distribution of environmental phenomenon is becoming increasingly popular (e.g. Critto et al., 2003; Gerlach et al., 2002; Leenaera et al., 1990; McBratney and Webster, 1983). Geostatistics is the term used to describe a range of statistical techniques for determining the relationship between spatially distributed values, leading to the estimation of the property at unsampled locations (Chappell, 1998). Geostatistics can interpret the fluctuations in data with respect to spatial and/or temporal variation (Olea, 1991). There are a variety of methods for representing continuous surfaces in digital form using computers (Gumbo et al., 2001). For the application here the digital elevation model (DEM) is the most useful form for the geospatial analysis (Gumbo et al., 2001). A DEM is a collection of geo-referenced elevation points with an interpolated surface. These points can be in a regular or irregular grid arrangement (Gumbo et al., 2001). To have an accurate DEM, it is necessary to acquire high enough data density to capture the features that you want to display. An overview of the geostatistical interpolation methods is given by Carusa and Quarta (1998).

1.4. Semi-variogram

The fundamental tool in the geospatial analysis is the semi-variogram, which determines the amount of spatial dependency (autocorrelation) in the data from the underlying spatial features of the variations (Burgess and Webster, 1980; Chappell, 1998; Oliver and Webster, 1987; Sorey et al., 1998; Webster and Oliver, 1990). The semi-variogram is calculated from the sampling points, and it has been recommended that at least 100 data points are needed for an accurate semi-variogram for a stationary random function (Webster and Oliver, 1992). In the real world, it is impossible to get high enough data density to fully characterize the surface emissions at every point due to practical constraints and timing. Kriging refers to the process of using the spatial dependency to predict the values of a property at unknown locations from the relationship found in the sampled locations. Kriging can be thought of as an optimal predictor (Journel and Huijbregts, 1981). The weights for the kriging analyses are derived from the semi-variogram (Oliver and Webster, 1987). Geospatial models can deal with abnormally large skewness and deviation from normal distributions (Juang et al., 2001). In addition, it has been found that a non-linear kriging model can be applied to predict concentration and volume content averages for highly skewed data sets (Kitanidis and Kuo-Fen, 1996).

There has only been limited use of geospatial techniques for estimating the resulting surface emission from landfills. Cardellini et al. (2002) states that a reliable surface emission estimate can only be accomplished through numerous measurements and a subsequent geostatistical treatment of the data. Borjesson et al. (2000) concluded that the geostatistical analysis provided a qualitative map of the surface methane flux distribution. The goal of this paper was to examine the application of geospatial statistics (kriging and IDW) to improve the quantification of methane emissions versus a simple qualitative representation.

2. Site description

The particular landfill site that was investigated in this study was the Onyx Lapouyade landfill situated near Bordeaux in France. This site has been operating since October 1996, receiving approximately 160,000 metric tons of waste per year. This site consists of two different cover configurations: (1) a final covered zone since 1998 and (2) an operating zone including temporarily covered cells with and without biogas recovery. This is shown graphically in Fig. 1. The area that was investigated for this paper was the final covered Phase I area.

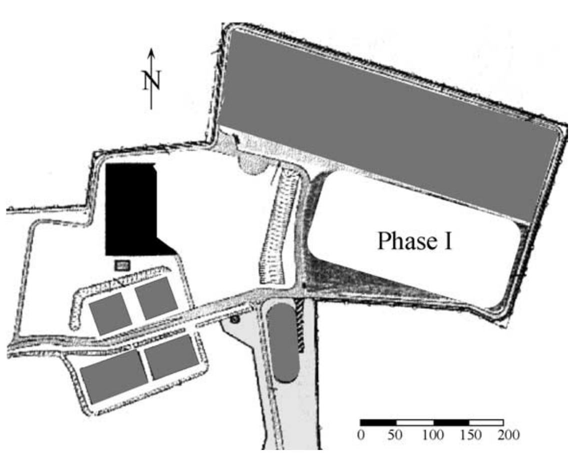

Fig. 1. Overview of Lapouyade field site.

2.1. Landfill methane balance

A mass balance for methane can be applied to the landfill site to better examine the potential pathways of methane from the landfill. This mass balance is given by:

$$\begin{aligned}\text{Methane generated} = {} & \text{Methane emitted}\\ & + \text{Methane oxidized}\\ & + \text{Methane recovered or flared}\\ & + \text{Methane migrated}\\ & + \Delta\text{Methane storage}\end{aligned}$$

Units in each of the above terms are mass per unit time (e.g. kg/day) (Bogner and Spokas, 1993; Bogner et al., 1995). The use of the methane balance equation is critical in examining the scenarios for all the pathways of methane at a landfill site. The various factors of the methane balance were either measured in the field or mathematically modeled. For the field campaign, all of the measurements took place over a period of two weeks with approximately the same climatic conditions (Morcet et al., 2002). There were fluctuations in the barometric pressure between 1010 and 1016 millibars during the 3 days of chamber flux measurements; however, a majority of the measurements took place within a 4 millibar range. Some temporal variability could be attributed to the barometric pressure changes that occurred over the measurement period, as shown by Czepiel et al. (2003). For this analysis, however, the fluctuations in the surface emissions due differences in barometric pressure were ignored.

The methods for determining each term of the methane balance will only be briefly presented as they have already been described (Diot-Morcet et al., 2002). The methane generation model for the Lapouyade site was based on a first-order degradation equation with multiple waste-types as inputs. This type of model has been used successfully in the Netherlands and has been successfully applied to actual landfill sites with excellent agreement (Blaha et al., 1999; Coops et al., 1995; Scheepers and Van Zanten, 1994). The estimate for landfill gas production can be improved by detailed waste characterization that was conducted at the site. This enabled a reliable estimate of the overall production at the various cells. The methane oxidation occurring through the cover soils was measured through a methane isotopic technique (Chanton and Liptay, 2000; Liptay et al., 1998). The methane recovered at the site was quantified through mass flow measurement (concentration×flow rate) from each area examined (Diot-Morcet et al., 2002). The landfill cells at the Lapouyade site each had a geomembrane (HDPE) coupled with other engineered controls to prevent lateral migration of gas through its sides and base. The geomembrane is the rate-limiting barrier, with a permeation rate of 0.58 cm^3 of CH_4 m^{-2} day^{-1} at 101 kPa pressure and 1.5 mm membrane thickness (50% methane test gas) (Lim, 1995; Pauly, 1989). This translates into 400 μg CH_4 m^{-2} day^{-1} per 1.5 mm of thickness. To find the amount of methane that could migrate through the geomembrane liner on each test cell, the area of each cell was multiplied by the permeation rate given above. The change in storage was calculated based on the concentration change seen in the recovered landfill gas multiplied by the estimated gas volume of the landfill (Diot-Morcet et al., 2002). Values for these terms of the methane balance are given in Table 1.

2.2. Measurement of landfill emissions at Lapouyade

There were two methods used to quantify the emission from the Lapouyade site. The first was a tracer method (Trégourès et al., 1999). It relies on concurrent concentration measurements for the methane and an inert tracer gas (here SF_6) released at a known rate upwind from the landfill.

The other technique was an external recirculation flux chamber. The chamber enclosed an area of 0.25 m^2. It uses a pump with a flow rate of approximately 10 l per min to circulate the enclosed chamber atmosphere (15 l) to an outside loop. The rate of methane enrichment in the loop is measured using a laboratory chromatograph transported in a utility vehicle around the site (Trégourès et al., 1999). The rate of concentration increase in the chamber is directly related to the surface emission rate. Negative (uptake) methane emissions could not be measured by this technique. A grid of 20×20 m was followed for the investigation of the Phase I area.

Table 1
Landfill mass balance parameters for Phase I cells at Lapouyade site

	Source (kg day^{-1})	Sink (kg day^{-1})
Methane production	4358	
Methane recovery		3935
Methane oxidation		83.5
Methane migration		20
Change in storage (sink)		50
Totals	4358	4089

3. Results

3.1. Digital representation of landfill surface

A DEM was created for the site using the georeferenced data collected. The DEM with sample locations can be seen in Fig. 2. A spherical variogram was used to determine the kriging weights for the interpolation of the DEM. The variogram for the DEM was modeled in GS+ (Gamma Design Software, 2002) and the kriging interpolation was completed in Surfer (Golden Software, 2001) using the model parameters determined in GS+. Block kriging was chosen on a 2×2 m grid spacing and the interpolation occurred within a rectangular grid that included all of the sample locations. Because of this, some of the areas may be overestimated. This is a result of the boundary of the

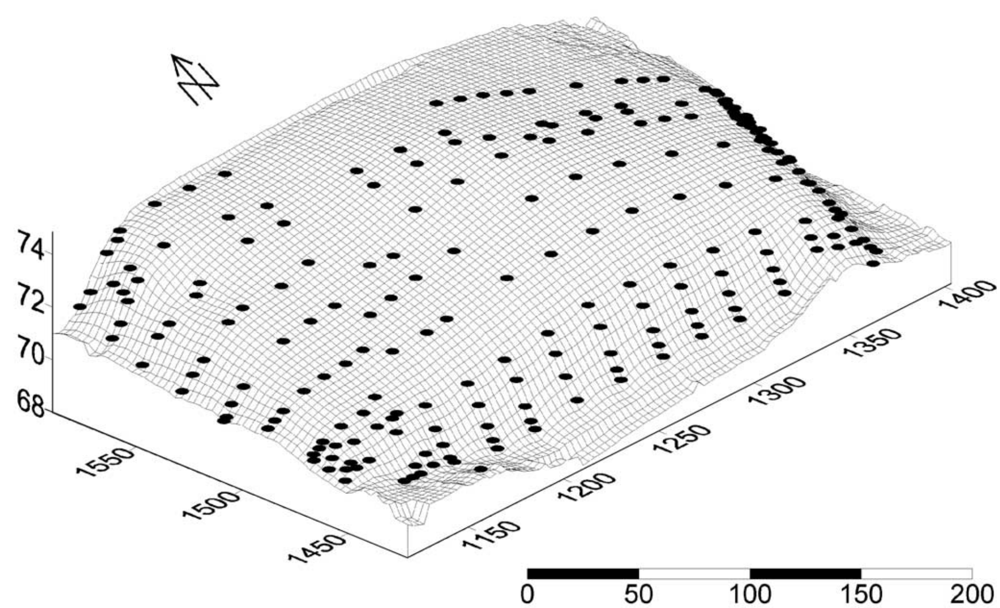

Fig. 2. Illustration of DEM of landfill surface with sample points for Phase I area.

DEM being defined by the edges of the flux measurements and not the true boundary condition.

3.2. Variogram analysis of chamber measurements

The strength of the variogram's spatial correlation can be measured by the nugget-to-total semivariance ratio. The criterion used in this study was similar to that suggested by Cambardella et al. (1994). The variogram is considered to have a strong spatial dependence if the ratio is less than 25% and a moderate spatial dependence if the ratio is between 25 and 75%. If the ratio is larger than 75%, then it has weak spatial dependence. Variogram analysis yielded only moderate spatial structure at this site with a nugget to total semivariance ratio of 0.42 (Fig. 3). The lag distance chosen was the approximate sample distance of 20 m and the non-zero value of the nugget indicates that there is spatial variability at smaller scales than our sampling distance and that measurement error is likely. The deviation from the model variogram can be seen at the small lag distances (< 40 m; Fig. 3). Even though this variogram should not be used for interpolation, the contour map generated by kriging is shown in Fig. 4.

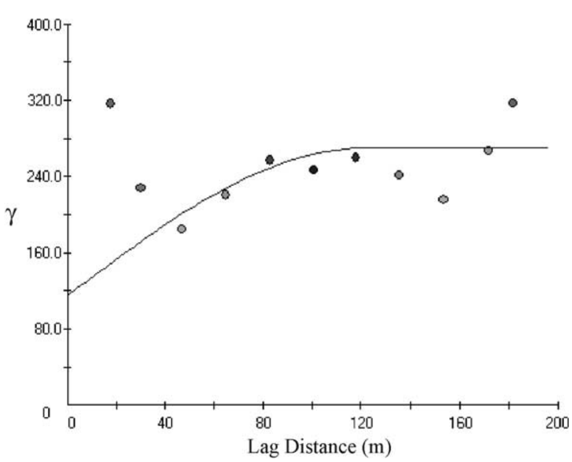

Fig. 3. Semivariogram for the Lapouyade Methane Emission Data.

3.3. Inverse distance weighing (IDW)

Since there was very little spatial structure to the dataset, another technique was applied. The distribution of methane flux emissions was interpolated using inverse distance to a power (squared) in Surfer (Golden Software, 2001) from the methane flux measurements at each sample location. A search neighborhood of 6 m was used so that only the nearest points to the one being interpolated were included in the calculation. This avoided sample locations at significant distances from influencing the interpolated value. IDW is an interpolation technique in which interpolated values are made based on measurement values at nearby locations weighted only by distance from the interpolation location. This technique does not assume any type of spatial relationship, except the basic assumption that near by points ought to be more closely related to one another than more distant points (Davis, 1986; Maguire et al., 1991). The only complication is the resulting contour is a function of sampling density. However, due to the large sampling size at the site, this would not impact the interpretation. Because the data points define the interpolation, the resulting contours are fully representative of the data. This is not always true of kriging and other contour models (Gerlach et al., 2001).

The formula used for IDW is:

$$Z_{\text{est}}^{j} = \frac{\sum\left(\frac{z_i}{(h_{i,j}+s)^2}\right)}{\sum\left(\frac{1}{(h_{i,j}+s)^2}\right)}$$

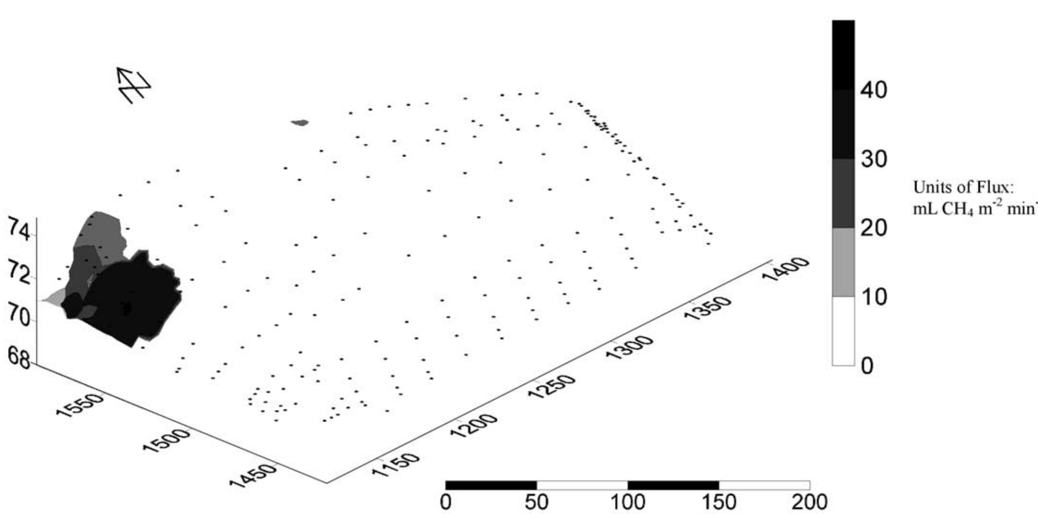

Fig. 4. Spatial distribution of Phase I flux results using kriging.

where: Z^j_{est} = estimated value for location j, z_i = measured sample value at point i, $h_{i,j}$ = distance between Z^j_{est} and z_j; and s = smoothing factor (0 was used).

The contour intervals chosen for the analysis were 0–1, 1–10, 10–100, 100–500, and >500 ml of CH_4 m^{-2} min^{-1}. Once the interpolated grids with the contour intervals were created, the grid volume was used to attain the planar surface area for contour intervals of interest. From these areas, average flux and total flux for the interval were calculated (Table 2). Total fluxes for each interval were added together and an overall flux was calculated for the cell. Fig. 5 displays the contoured results of the IDW analysis of the methane flux across the Phase I cell. There is a large imbalance in the distribution of the flux measurements (Table 2). The results of the geospatial calculations show that 64.6% of the total area contributes less than 1% of the total flux, with the remaining 35.4% of the area of the cells responsible for the remaining 99% of the emissions of methane. This confirms the qualitative use as reported by Borjesson et al. (2000).

The contour map of the landfill emissions produced by IDW and kriging of the variogram show similar trends; however, the smoothing effect of the kriging algorithm (Fig. 4) produces a map with a narrower range of values than the map produced from IDW (Fig. 5). The kriged version of the map does not show the highest actual values of flux in the northwest corner of the landfill, nor does it capture the non-zero flux measurements found on the south facing slope that can be seen on the IDW map. Much of the smoothing effect is a result of the moderate structure of the variogram.

If the emission rates show delineated areas of high and low emission with discontinuities, then a combination of grouping followed by kriging would provide the best estimates (Oliver and Khayrat, 2001; Voltz and Webster, 1990). However, in this dataset there was no clear division, even though the slope areas contributed 90% of the total emission. There were localized discontinuities in the slope faces (Fig. 5), but they were not continuous enough to justify a hard breakline.

Table 2
Results of inverse distance weighting (IDW) analysis of Phase I cells at Lapouyade site

Contour interval (ml CH_4 m^{-2} min^{-1})	Area (m^2)	% Area of total	Number of measurements	Flux (kg CH_4 day^{-1})	% Flux of total
0–0.99	30,921	64.61	189	2.58	0.86
1–9	12,252	25.6	22	59.23	19.83
10–99	4244	8.87	6	101.57	34.01
100–499	429	0.9	3	128.45	43.01
500+	9	0.002	1	6.8	2.28
Totals	47,854	100	221	298.6	100

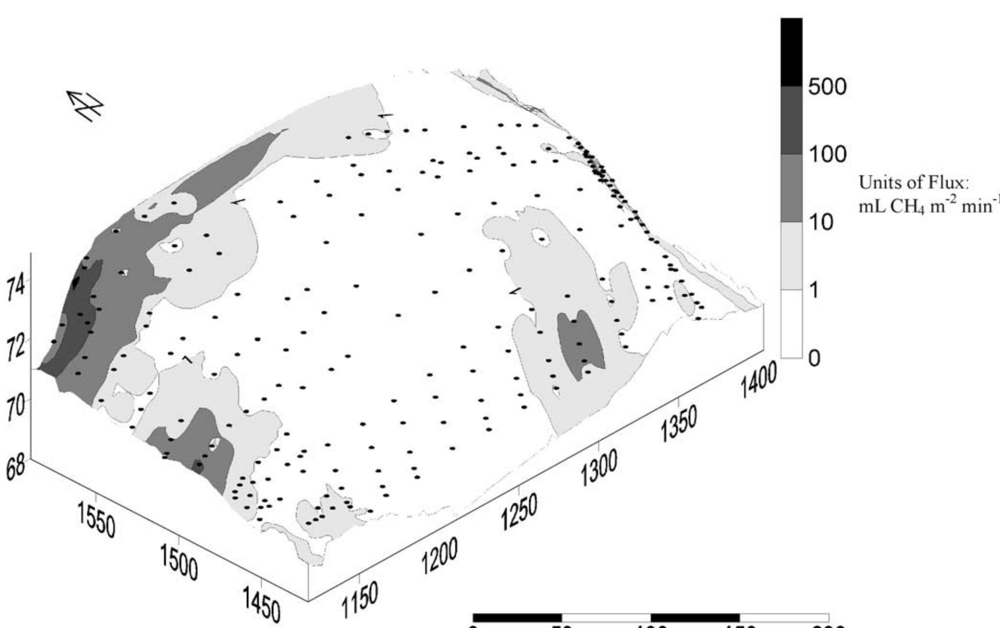

Fig. 5. Spatial distribution of Phase I flux results using inverse distance weighing contouring.

Table 3
Comparison of the various tendency measures, geospatial, and tracer estimates of surface emission rates

	Emission estimate (kg day^{-1})	% Relative balance[a]
Median	0	6.2
Geometric mean[b]	6.6	6.0
Tracer technique	103.3	3.8
Arithmetic mean	140.7	2.9
Geospatial calculation	298.6	−0.7

[a] $\% \text{ Relative balance} = \dfrac{\sum CH_4\text{Sources} - \sum CH_4\text{Sinks}}{\sum CH_4\text{Sources}}(100\%)$.

[b] Geometric mean was calculated substituting 0.001 for a zero flux reading.

Another difficulty of applying ordinary kriging to landfill emission data is that the estimation of errors may vary over orders of magnitude and are highly skewed, so the application of linear methods (ordinary kriging) is not recommended and a better alternative would be disjunctive, indicator, or probability kriging (Cardellini et al., 2003; Kitanidis and Shen, 1996). However, the difficulty is that the models for parameter estimation and validation are still in their infancy and these methods are highly computation intensive which makes these non-linear methods difficult to apply (Kitanidis and Shen, 1996). These methods were not attempted in this analysis.

3.4. Arithmetic and geometric mean calculation

The arithmetic and geometric means as well as the median of the emission measurements were taken and multiplied by the total surface area to arrive at additional estimates for the whole site emission rate. Additional measures of the tendency of the distribution (e.g. mode and harmonic mean) and probability means (e.g. Finney Probit Analysis) were either zero or less than the geometric mean, and therefore were not used in the analysis. The distribution of the resulting flux measurements was highly right-skewed (skewness = 9.7 and kurtosis = 105) with 95% of the flux measurements below 4 ml CH_4 m^{-2} min^{-1} and a maximum of 755 ml CH_4 m^{-2} min^{-1}.

The emission estimate of the geostatistical method was twice the arithmetic mean, 45× higher than the geometric mean, and almost three times the emission estimate of the tracer methodology (Table 3). Tracer and micrometeorological methods can be influenced by difficulties in varying footprint area (due to atmospheric instability) and temporal variability in the fluxes (Laville et al., 1999). This can lead to the tracer method capturing emissions from more than the landfill cell of interest resulting in a positive or negative bias depending on the area's emission values compared to the landfill cell.

The resulting emission estimates were compared to the methane balance (Table 1) and the results are shown in Table 3. The geospatial methodology provided the estimate with the lowest residual in the methane balance. Even though the production rate is modeled, the methane balance does provide a mechanism to compare the potential emission estimates within an order of magnitude. This result of the methane balance coupled with the fact that the IDW is an honest interpolator, establishes the basis for the claim that the IDW is a superior estimator than the mean or tracer methodology at this particular site.

The geospatial (IDW) calculation was assumed to be the most accurate since the data points defined the interpolation and thereby ensured representative contour intervals. This is not always true of kriging and other contour models (Gerlach et al., 2001) as seen in this study comparing the kriged and IDW contour maps (Figs. 4 and 5).

4. Conclusions and implications

The goal of this paper was to examine the potential shortcomings of utilizing geospatial methodologies in determining whole landfill emission rates. These methods offer the potential of calculating whole site emission estimates from limited point measurements, which could lead to improving overall national inventories for global landfill methane emission estimates. The major disadvantage of the chamber measurements having a small footprint also enables detailed spatial distribution studies of the emission from the cover at a site. However, in order for the geospatial analysis to be of value, proper interpolation methodology must be applied. As seen in this study, the slope regions of the cell contributed nearly 90% of the total cell flux. However, there were still spatial discontinuities with the distributions of the emission measurements on the slope, which prohibited the use of grouping the sloped and non-slope areas separately. In order for kriging to be correctly applied, the semi-variogram needs to adequately describe the spatial distribution and the distribution of the emission measurements needs to be continuous. These are very difficult requirements for the landfill emission data to meet.

For the Lapouyade cell, the IDW methodology did provide a more reasonable estimate of the surface emissions as compared to the arithmetic, geometric, and tracer techniques in light of the methane balance at the site. However, it has been shown that ordinary and universal kriging are superior to IDW interpolation (Zimmerman et al., 1999), but due to the lack of spatial structure, kriging could not be used in this study. IDW provides an honest interpolation of the data points, therefore the contour map that is generated through the IDW does not misrepresent any area and all points are considered. This is not the case in kriging where misrepresentations and smoothing can occur since the spatial distribution is modeled for the entire site (Gerlach et al., 2001). This is the fundamental reason why the IDW calculation method is superior to the alternatives.

The major difficulty with soil systems is that the variability cannot be captured by a single variable (e.g. methane emission rate). Therefore, it would be advantageous to determine the spatial variability of other soil properties (e.g. soil moisture, temperature) taken at the same time as emission measurements across the landfill to check for potential covariates that would improve overall emission prediction. This would enable another mechanism to describe the estimated methane emissions with more rapidly collected data that could be collected at higher sampling densities. However, currently these soil processes are inadequately understood for incorporation into a full spatial model (Kitanidis and Shen, 1996).

Acknowledgements

Funding for this project was provided by the Centre de Recherches pour l'Environnement l'Energie et le Dechets (CREED), Limay, France as part of the current METHAN project. The content of this paper does not necessarily represent the views of this agency.

References

Bingemer, H.G., Crutzen, P.G., 1987. The production of methane from solid wastes. Journal of Geophysical Research 92 (D2), 2181–2187.

Boeckx, P., Van Cleemput, O., Villaralvo, I., 1996. Methane emissions from a landfill and the methane oxidizing capacity of its cover soil. Soil Biology and Biochemistry 28, 1397–1405.

Blaha, D., Bartlett, K., Czepiel, P., Harris, R., Crill, P., 1999. Natural and anthropogenic methane sources in New England. Atmospheric Environment 33, 243–255.

Bogner, J., Spokas, K., 1993. Landfill methane: rates, fates, and roles in global carbon cycle. Chemosphere 26 (1–4), 369–386.

Bogner, J., Spokas, K., Burton, E., Sweeney, R., Corona, V., 1995. Landfills as atmospheric methane sources and sinks. Chemosphere 31, 4119–4130.

Bogner, J., Spokas, K., Burton, E., 1997. Kinetics of methane oxidation in a landfill cover soil: temporal variations, a whole-landfill oxidation experiment, and modeling of net-methane emissions. Environment Science and Technology 31, 2504–2514.

Bogner, J., Matthews, E., 2003. Global methane emissions from landfills: new methodology and annual estimates 1980–1996. Global Biogeochemical Cycles 17(2), 34:1–18.

Borjesson, G., Danielsson, A., Svensson, B.H., 2000. Methane fluxes from a Swedish landfill determined by geostatistical treatment of static chamber measurements. Environmental Science and Technology 34 (18), 4044–4050.

Burgess, I.M., Webster, R., 1980. Optimal interpolation and isarithmic mapping of soil properties. I. The semivariogram and punctual kriging. Journal of Soil Science 31, 315–331.

Cambardella, C.A., Moorman, T.B., Novak, J.M., Parkin, T.B., Karlen, D.L., Turco, R.F., Konopka, A.E., 1994. Field-scale variability of soil properties in Central Iowa soils. Soil Science Society of America Journal 58, 1501–1511.

Cardellini, C., Chiodini, G., Frondini, F., Granieri, D., Lewicki, J., Peruzzi, L., 2003. Accumulation chamber measurements of methane fluxes: application to volcanic-geothermal areas and landfills. Applied Geochemistry 18(1), 45–54.

Carusa, C., Quarta, F., 1998. Interpolation methods comparison. Computers Math. Applications 35 (12), 109–126.

Chanton, J., Liptay, K., 2000. Seasonal variation in methane oxidation in landfill cover soil as determined by an in situ stable isotope technique. Global Biogeochemical Cycles 14, 51–60.

Chappel, A., 1998. Using remote sensing and geostatistics to map ^{137}Cs-derived net soil flux in south-west Niger. Journal of Arid Environments 39, 441–455.

Charnpratheep, K., Zhou, Q., Garner, B., 1997. Preliminary landfill site screening using fuzzy geographical information systems. Waste Management and Research 15, 197–215.

Christopherson, M., Linderod, L., Jensen, P.E., Kjeldsen, P., 2000. Methane Oxidation at low temperatures in soil exposed to landfill gas. Journal of Environmental Quality 29 (6), 1989–1997.

Coops, O., Luning, L., Oonk, H., Weenk, A. 1995. Validation of Landfill Gas Formation Models. In: Proceedings from Sardinia '95, Fifth International Landfill Symposium, CISA, S. Margherita di Paula, Calgiari, Italy, 2–6 October 1995.

Critto, A., Carlon, C., Marcomini, A., 2003. Characterization of contaminated soil and groundwater surrounding an illegal landfill (S. Giuliano, Venice, Italy) by principal component analysis and kriging. Environmental Pollution 122 (2), 235–244.

Czepiel, P.M., Shorter, J.H., Mosher, B., Allwine, E., McManus, J.B., Harriss, R.C., Kolb, C.E., Lamb, B.K., 2003. The influence of atmospheric pressure on landfill methane emissions. Waste Management 23. PII: S0956-053X(03)00103-X.

Czepiel, P., Mosher, B., Harriss, R., Shorter, J., McManus, J., Kolb, C., Allwine, E., Lamb, K., 1996. Quantifying the effect of oxidation on landfill methane emissions. Journal of Geophysical Research: Atmosphere 101 (D11), 16711–16719.

Davis, J.C., 1986. Statistics and Data Analysis in Geology. Wiley, New York.

Diot-Morcet, M.D., Aran, C., Chanton, J.P., Bogner, J., Hébé, I., Spokas, K., Graff, C., 2002. Evaluation of the Seasonal Variation of the Methane Mass Balance at a French Landfill. In: Proceedings SWANA 25th annual landfill gas symposium, 25–28 March, Monterey, California, published by SWANA, Silver Spring, MD.

Ito, A., Takahashi, I., Nagata, Y., Chiba, K., Haraguchi, H., 2001. Spatial and temporal characteristics of urban atmospheric methane in Nagoya City, Japan: an assessment of the contribution from regional landfills. Atmospheric Environment 35, 3137–3144.

Gerlach, T.M., Douglas, M.P., McGee, K.A., Kessler, R., 2001. Soil efflux and total emission rates of magmatic CO2 at the Horseshoe Lake tree kill, Mammoth Mountain, California, 1995-1999. Chemical Geology 177, 101–116.

Giani, L., Bredenkamp, J., Eden, I., 2002. Temporal and spatial variability of the methane dynamics of landfill cover soils. Journal of Plant Nutrition and Soil Science (Zeitschrift fur Pflanzenernahrung und Bodenkunde) 165 (2), 205–210.

Gourlay, K.A., 1992. World of Waste: Dilemmas of Industrial Development. Zed Books, London, England.

Gumbo, B., Munyamba, N., Sithole, G., Savenije, H.G., 2001. Coupling of Digital Elevation Model and Rainfall-Runoff Model in Storm Drainage Network Design, 2nd WARFSA/WaterNet Symposium: Integrated Water Resources Management: Theory, Practice, and Cases. Cape Town, 30–31 October 2001.

Jager, J., Peters, J., 1985. Messung der Oberflachenemission von Deponiegas. Stuttgarte Berichte Abfallwirtsch 19, 337–345.

Jones, H.A., Nedwell, D.B., 1993. Methane emission and methane

oxidation in landfill cover soil. FEMS Microbiology Ecology 102, 185–195.
Journel, A.G., Huijbregts, C.H.J., 1981. Mining Geostatistics. Academic Press, New York.
Juang, K.W., Lee, D.Y., Ellsworth, T.R., 2001. Using rank-order geostatistics for spatial interpolation of highly skewed data in a heavy-metal contaminated site. Journal of Environmental Quality 30 (3), 894–903.
Kitanidis, P.K., Kuo-Fen, S., 1996. Geostatistical interpolation of chemical concentration. Advances in Water Resources 19 (6), 369–378.
Kreileman, G.J.J., Bouwann, A.F., 1994. Computing land use emissions of greenhouse gases. Water, Air, and Soil Pollution 76, 231–258.
Laville, P., Jambert, C., Cellier, P., Delmas, R., 1999. Nitrous oxide fluxes from a fertilized maize crop using micrometeorological and chamber methods. Agricultural and Forest Meteorology 96, 19–38.
Leao, S., Bishop, I., Evans, D., 2001. Assessing the demand of solid waste disposal in urban region by urban dynamics modeling in a GIS environment. Resources Conservation, and Recycling 33, 289–313.
Leenaers, H., Okx, T.P., Burrough, P.A., 1990. Comparison of spatial prediction methods for mapping floodplain soil pollution. Catena 17, 535–550.
Lim, K.T., 1995. Mega-molecular Dynamics on Highly Parallel Computers: Methods and Applications. PhD dissertation, California Institute of Technology, Pasadena, California.
Liptay, K., Chanton, J., Czepiel, P., Mosher, B., 1998. Use of stable isotopes to determine methane oxidation in landfill cover soils. Journal of Geophysical Research 103, 8243–8250.
Livingston, G.P., Hutchinson, G.L.. In: Matson, P.A., Harris, R.C. (Eds.), Biogenic Trace Gases: Measuring Emissions from Soil and Water. Blackwell Science, Cambridge, MA.
Maguire, D.J., Goodchild, M.F., Rhind, D.W. (Eds.), 1991. Geographical Information Systems: Principles and Applications. Longman Scientific and Technical, New York.
Mancinelli, R., 1995. The regulation of methane oxidation in soil. Annual Reviews in Microbiology 49, 581–605.
McBratney, A.B., Webster, R., 1983. Optimal interpolation and isarthmic mapping of soil properties. V. Co-regionalization and multiple sampling strategy. Journal of Soil Science 34, 137–162.
Moore, C., Donaldson, C., Bogner, J., 1995. Geographic informationsystems monitor landfills. In: Proceedings from Sardina '95, Fifth International Landfill Symposium, CISA, S. Margherita di Paula, Calgiari, Italy, 2–6 October.
Mosher, B.W., Czepiel, P., Hariss, R.C., Shorter, J.E., Kolb, C.E., McManus, J., Allwine, E., Lamb, B., 1999. Methane emissions at ninelandfill sites in the northeastern United States. Environment Science and Technology 33, 2038–2094.
Neue, H.-U., Lantin, R., Wassmann, R., Adnna, J.B., Alberto, C.R., Andales, M.F., 1994. Methane emission from rice soils of the Philippines. In: CH4 and N2O, National Institute of Agro-Environmental Sciences, Tsukuba, Japan, pp. 55–63.
Nozhevnikova, A.N., Lifskitz, A.B., Lebedev, V.S., Zavarzin, G.A., 1993. Emission of methane into the atmosphere from landfills in the former USSR. Chemosphere 26, 401–417.
Office of Technology Assessment (OTA), 1991. In Changing by Degrees: Steps to Reduce Greenhouse Gases. (US Congress), OTA-0-482, US Government Printing Office, Washington, DC.
Olea, R. (Ed.), 1991. Geostatistical Glossary and Multilingual Dictionary. Oxford University Press, New York, USA.
Oliver, M.A., Khayrat, A.L., 2001. A geostatistical investigation of the spatial variation of radon in soil. Computers & Geosciences 27 (8), 939–957.
Oliver, M.A., Webster, R., 1987. The elucidation of soil pattern in the Wrye Forest of the West Midlands, England. II. Spatial distribution. Journal of Soil Science 38, 293–307.
Pauly, S.. In: Brandrup, J., Immergut, E.H. (Eds.), Permeability and Diffusion Data, Polymer Handbook, third ed. Wiley-Interscience.
Pokryszka, Z., Tauziede, C., Cassini, P., 1995. Development of a method for measuring biogas emissions using a dynamic chamber. In: Proceedings from Sardina '95, Fifth International Landfill Symposium, CISA, S. Margherita di Paula, Calgiari, Italy, 2–6 October.
Scheepers, M.J.J., van Zanten, B., 1994. Handleiding stortgaswinning. Adviescentrum Stortgas, Utrecht.
Sorey, M.L., Evans, W.C., Kennedy, B.M., Farrar, C.D., Hainsworth, L.J., Hausback, B., 1998. Carbon dioxide and helium emissions from a reservoir of magmatic gas beneath Mammoth Mountain, California. Journal of Geophysical Research 103, 15303–15323.
Trégourès, A., Beneito, A., Berne, P., Gonze, M.A., Sabroux, J.C., Pokryszka, Z., Savanne, D., Tauziede, C., Cellier, P., Laville, P., Milward, R., Arnaud, A., Levy, F., Burkhalter, R., 1999. Comparison of seven methods for measuring methane flux at a municipal solid waste landfill site. Waste Management and Research 17, 453–458.
Voltz, M., Webster, R., 1990. A comparison of kriging, cubic splines and classification for predicting soil properties from sample information. Journal of Soil Science 41 (3), 473–490.
Wang, X.J., Qi, F., 1998. The effects of sampling design on spatial structure analysis of contaminated soil. The Science of The Total Environment 224 (1–3), 29–41.
Webster, R., Oliver, M.A., 1992. Sample adequately to estimate variograms of soil properties. J. Soil Sci. 43, 177–192.
Webster, R., Oliver, M.A., 1990. Statistical Methods for Soil and Land Resource Surveys. Oxford University Press, Oxford, England.
Whalen, S., Reeburgh, W., Sandbeck, P., 1990. Rapid methane oxidation in a landfill cover soil. Applied and Environmental Microbiology 56, 3405–3411.
Zimmerman, D., Pavlik, C., Ruggles, A., Armstrong, M.P., 1999. An experimental comparison of ordinary and universal kriging and inverse distance weighting. Mathematical Geology 31 (4), 375–390.