

The effects of reduction of the deposited waste on short-term landfill leachate composition of a landfill: a case study in Norway

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Abstract

Municipal solid waste landfill leachate was sampled over from a landfill receiving varying amounts of municipal solid waste. The investigation aimed to provide information on expected leachate changes in the short term, either after closure of an active landfill, or after a strong decline in the amount of waste deposited at smaller landfills. It was found that during a two year period following a sudden decline in the amount of waste deposited, the levels of various chemical and physical parameters all dropped sharply. The reasons for the decline in discharge levels are thought to be aerobic decomposition taking place in the municipal solid waste just after landfilling, and thus the decline in the impact of this process when there were less fresh waste masses available, and formation of preferential flow paths for the leachate as the municipal solid waste stabilized in the landfill.

Introduction

Long-term changes in municipal solid waste (MSW) landfill leachates have been described and models have been proposed, in order to better predict long-term leachate discharges from landfills after closure (Bozkurt *et al.* 2000; Christensen *et al.* 2001; Barlaz *et al.* 2002; Kjeldsen *et al.* 2002). Degradation processes in landfills change over time, leading to changes in the composition of the discharged leachate (Bozkurt *et al.* 2000; Kjeldsen *et al.* 2002). The chemical conditions in the waste masses in landfills progress through a sequence of chemical phases: an acid phase where fatty acids are oxidized to long-chain organic acids; a methane fermentation process where anaerobic conditions are established in the landfill masses; and finally the maturation phase, which occurs over a time span of hundreds of years, leading to the re-establishment of toxic conditions in the waste masses (Belevi & Baccini 1989; Kjeldsen *et al.* 2002). Based on these phases, changes in landfill leachate composition have been modelled using time frames of 50 to several hundreds of years.

Less work, however, has been carried out on predicting potential leachate development in the shorter (5–10 years) perspective. In cases where landfills receive very variable amounts of waste masses over a time span of 5–10 years, the leachate is likely to change because of the varying waste loads on the landfill. Both in the case of

planned or expected increase in the waste deposited on a landfill, and in the case of planned decrease it is useful to know what changes can be expected in the leachate. This knowledge is of particular importance to the authorities responsible for the environmental regulation of landfills.

The current work is an investigation into variations in the composition of leachate from an active landfill where a large proportion of the total landfilled MSW mass was deposited over a time span of just 4 years, followed by considerably lower deposition rates in the subsequent years. Changes in leachate composition are compared with changes in the mass of waste deposited. The investigation aims to provide information about expected short-term changes in leachate composition occurring after either landfill closure or a strong decline in the amount of landfilled MSW. This information is crucial when predicting the future environmental loads, which that will be inflicted by a specific landfill, and subsequently in the planning of future monitoring and treatment regimes for the leachate.

Materials and methods

Landfill

The landfill studied (Svartasmoget landfill, Stord, Norway) was established in 1993, and is situated on the western coast of Norway. It was established directly on bedrock, without a

bottom liner, and by the end of 2008 had received about 740 000 tonnes of MSW, covering an area of approximately 85 000 m². The area draining to the leachate collection system was held constant from 1993 until 2008. Ground water wells around and under the landfill have been sampled regularly, and no influence from the landfill leachate was detected. The landfill has been described in the works by Øygard *et al.* (2004) and Øygard *et al.* (2005b).

Owing to the closure of a separate large landfill in the region during 1996 (Rådal landfill, Bergen) and the delayed construction of an incinerator plant intended to process the MSW produced in that area, Svartasmoget landfill received relatively large amounts of MSW during a 5-year period from 1995 to 1999 (a total of 440 000 tonnes), returning afterwards to normal running conditions, with only approximately 25 000–30 000 tonnes of MSW deposited annually.

The leachate from the landfill is collected and discharged untreated into the ocean via a single pipeline. Because of the relatively high precipitation (annual rainfall plotted in Fig. 1), and a groundwater flow pattern that introduces groundwater into the leachate stream, the total leachate volume per year is high (170–310 000 m³) relative to the area of the landfill. Based on the average local precipitation and the area draining to the leachate collection system, around 140 000 m³ leachate should be produced each year. Consequently, at least 100 000 m³ of the leachate must originate from the unpolluted groundwater entering the leachate stream under the waste masses. The leachate discharged into the sea is thus rather dilute, with relatively low levels of many leachate components, when compared with levels reported from similar landfills elsewhere (Christensen *et al.* 2001).

Characterization of landfilled waste

There have not been any important changes in the composition of the MSW landfilled during the time the

landfill was monitored, from 1997 up until 2008. Only MSW has been deposited during this period, with organic waste such as sewage sludge and wood being composted at a separate location. Glass and metal were in part source-separated before landfilling of the MSW.

A waste composition analysis of the MSW received in 2002 (2 tonnes of waste analysed) showed that the waste had a composition of 24% paper/cardboard, 19% biowaste (mainly food), 18% plastics, 8% glass, 5% metals, 6% textiles, 0.6% electronic waste and 0.3% paint and other hazardous waste. The remaining 19% comprised wooden products, stone and sand, crockery and nappies. It can be assumed that the waste composition remained relatively constant throughout the years of the investigation, as the landfill only received MSW, and there were few changes in household source-separation or consumption/waste generation behaviour patterns during those years. No incineration residues have ever been deposited at the investigated landfill.

Sampling and analysis of leachate

Between 1997 and 2006, the leachate was sampled monthly, with quarterly sampling from 2007 onwards. On all sampling occasions, the sample was drawn as one single instantaneous sample from the leachate discharge pipe. The leachate was sampled from the central discharge pipeline downstream from the landfill, and thus represents a fresh mixture of all leachate discharged from the landfill. After sampling, the leachate was stored cold (at 4 °C) and dark, taken within 24 h to a laboratory accredited according to ISO 17025, and analysed according to a set programme (nutrients and metals as well as physical parameters). All samples were analysed without prior filtration. pH and conductivity were measured as soon as the samples arrived at the lab. Subsamples for phosphorus and chemical oxygen demand (COD) determination were conserved with sulphuric acid, while samples for metal determination were conserved with nitric acid. The analytical programme (and methods) included anions (EN ISO 10304-1), metals/elements (ISO 11885), nutrients (ISO 6060 for COD and EN ISO 6878 for Tot-P) and physical characteristics of the leachate (ISO 7888 for conductivity and ISO 10523 for pH). Metal levels were quantified after nitric acid digestion of the nonfiltered leachate. Similar analytical methods were used throughout the 10 years of sampling, and all the results are thus comparable.

Results and discussion

The masses of MSW deposited annually onto the landfill are shown in Fig. 2. The sharp increase in landfilled

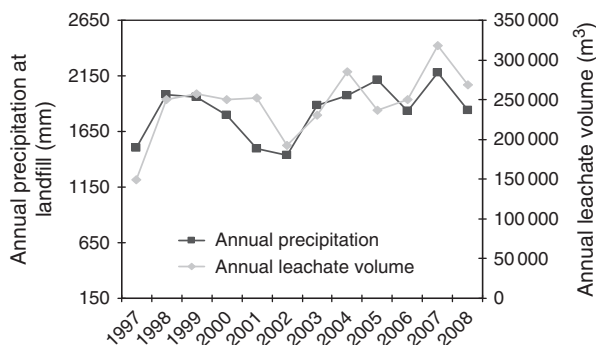


Fig. 1. Annual precipitation (mm) at the location of the landfill, and annual volume of leachate (m³) discharged from the landfill. The mean annual precipitation for the last 50 years is 1660 mm.

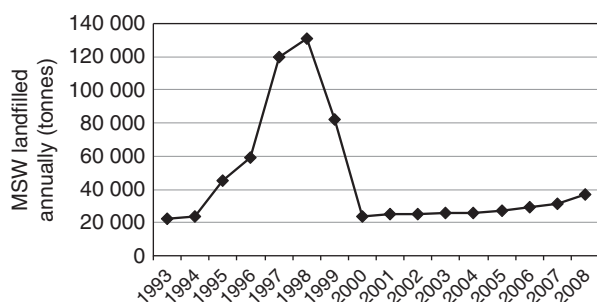


Fig. 2. Masses of municipal solid waste (MSW) landfilled yearly on the landfill from its opening in 1993 until the end of 2008.

masses between 1995 and 1999 included the additional waste from the Bergen area north of the landfill. After the construction of an incinerator plant in Bergen, the landfill returned to normal running conditions, with approximately 25–30 000 tonnes of MSW added each year. The landfill is relatively small on an international scale, and as such the effects of the changes in the yearly deposited MSW are probably more evident on this site than can be expected on larger landfills.

Variations in selected components in the leachate over the period from 1997 to 2008 are given in Fig. 3. The year 1999 was the last of a 4-year period when the landfill received > 60 000 tonnes of MSW annually. In 2000, only 25 000 tonnes of MSW were landfilled. This decrease in landfilled masses seems to have substantially influenced the leachate quality later on.

Annual rainfall at the landfill site and annual leachate volumes are plotted in Fig. 1. The decrease in annual rainfall between 1998 and 2002 should, if anything, result in increased leachate concentrations because of a lowering of the dilution effect. Changes in precipitation can, thus, not be contributing to the observed decrease in leachate concentrations from 1998 to 2001–2002.

The levels of all measured components in the leachate are relatively low, compared with data from other landfills such as those presented by Ehrig (1989) and Robinson & Gronow (1993). This is most likely a result of the dilution effect caused by the relatively high precipitation at the site and by the clean groundwater entering the bottom of the landfill.

Changes in conductivity and easily soluble ions in the leachate

All the six chemical components described in Fig. 3 follow the same trend, with a sharp drop in levels observed in the leachate between 1999 and 2000. Conductivity shows quite large variations throughout the year, leading to large standard deviations. This may be because of periodic events such as strong rainfall leading to dilution of the leachate (Johnson *et al.* 1999). From Fig. 3, no clear

relationships can be seen between annual rainfall (Fig. 1) and either conductivity or chloride concentrations in the leachate. A decrease in conductivity is seen with the increasing rainfall from 2002–2004, but the chloride concentrations show the opposite trend during the same period, thus no significant influence of rainfall on leachate quality can be concluded from the data.

Chloride has previously been predicted to have a quick washout rate from landfills (Øygard *et al.* 2005a), as this anion does not bind to matter in the waste masses (Christensen *et al.* 2001). This assumption is verified here, as the chloride concentrations in the leachate decreased strongly when the amount of MSW deposited at the landfill decreased, between 1999 and 2000. The concentrations of sodium and potassium (Table 1) follow the same pattern as chloride, presumably for the same reason. Like chloride, sodium and potassium are considered to have insignificant binding and retention in landfill masses (Christensen *et al.* 2001), and their concentrations in leachate are therefore largely dependent upon the contact time between the water and waste, and leachate flow. Øygard *et al.* (2005a) calculated the washout rate of chloride from MSW incinerator bottom ash to be around 80% per year. The washout rate is probably lower for MSW, but from this study it is clear that the rate is high, as the chloride concentration in the leachate almost halved as the deposited waste volume halved. At landfill sites in areas with less precipitation, the response in the leachate concentrations of conservative species to changes in the amounts added to the landfill might not be as rapid or as obvious. However, it is also clear that the decline in concentration in the leachate does not represent a washout from the complete waste masses but rather from the masses surrounding preferential flow paths formed in the landfill (Rosqvist *et al.* 2005), as the waste/leachate contact occurs predominantly in and surrounding preferential flow paths, and is limited in other parts of the waste masses.

Changes in levels of COD, nutrients and metals in the leachate over time

The COD and total phosphorus levels in the leachate follow the same trend as the conductivity and the major ions, with a sharp decline in levels between 1999 and 2000. The COD level in leachate is known to be dependent upon the chemical degradation phase the masses in the landfill are undergoing, with the highest COD values found in the leachate produced during the initial acidogenic phase occurring in freshly deposited waste masses, when various fermentation products are discharged (Christensen *et al.* 2001; Vavilin *et al.* 2006). Owing to the large amounts of fresh MSW landfilled throughout 1995–1999, it is likely that, at any time during this period,

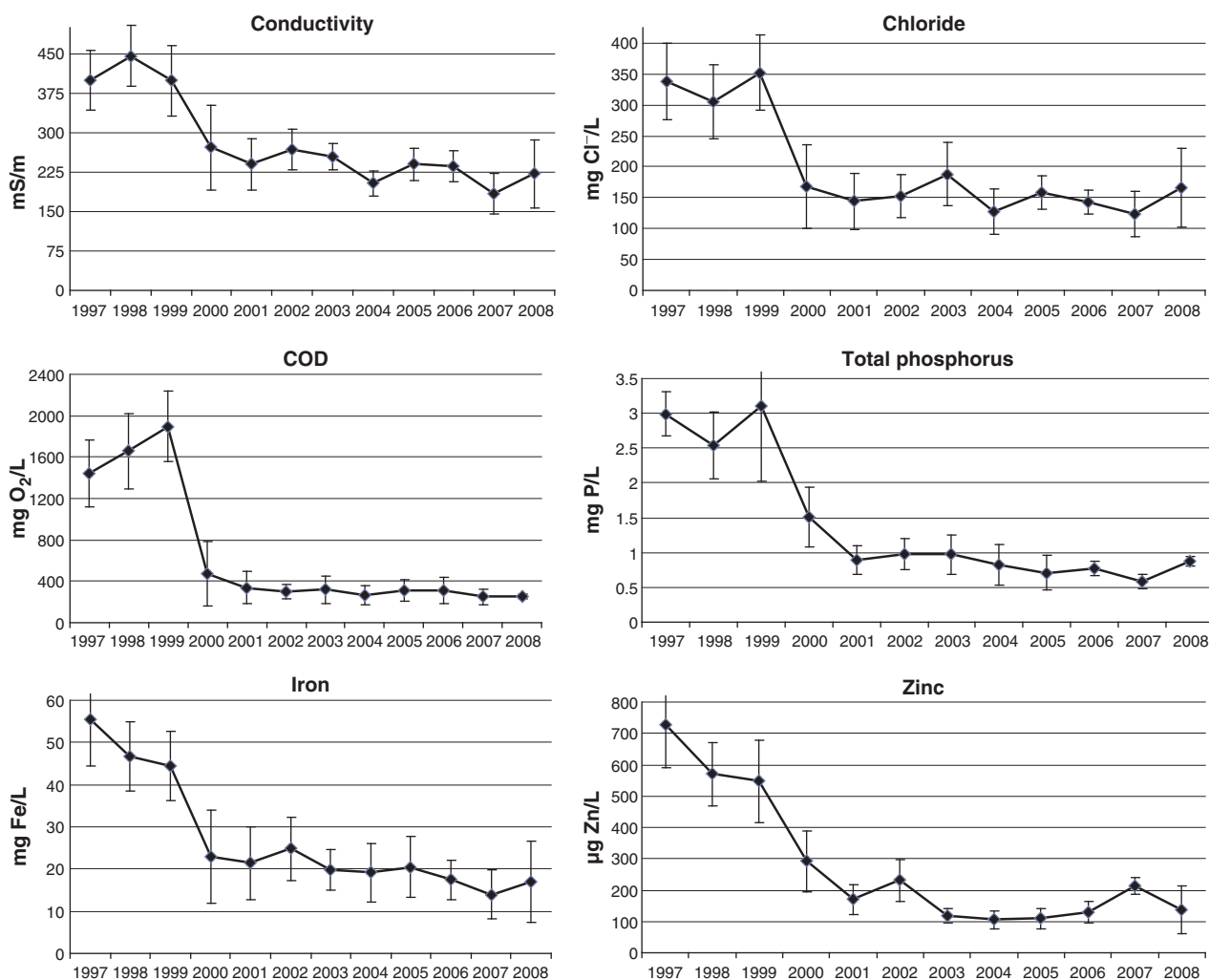


Fig. 3. Changes in levels of selected components in the leachate from the Svartasmoget landfill between 1997 and 2008. $n=12$ for all years except 2007–2008, where $n=4$. Standard deviation of measurements plotted for each year. COD, chemical oxygen demand.

Table 1 pH and concentrations of selected chemical components in the leachate sampled from 1997 to 2008

Sampling year	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008
pH	6.6 ± 0.20	6.7 ± 0.21	6.6 ± 0.20	6.6 ± 0.13	6.6 ± 0.11	6.7 ± 0.09	6.8 ± 0.07	6.7 ± 0.10	6.8 ± 0.13	6.9 ± 0.18	6.9 ± 0.04	6.7 ± 0.21
Total-N	190 ± 44	180 ± 111	240 ± 113	160 ± 70	140 ± 56	150 ± 44	160 ± 54	130 ± 55	140 ± 50	130 ± 64	100 ± 34	140 ± 42
Ammonium-N	190 ± 54	170 ± 108	210 ± 107	150 ± 64	130 ± 55	150 ± 43	150 ± 53	120 ± 53	130 ± 47	130 ± 62	92 ± 30	150 ± 49
Sodium	300 ± 100	250 ± 93	280 ± 140	170 ± 78	170 ± 66	160 ± 50	180 ± 62	130 ± 43	150 ± 54	150 ± 62	110 ± 47	120 ± 46
Potassium	160 ± 48	140 ± 66	210 ± 77	110 ± 43	76 ± 37	88 ± 41	75 ± 36	66 ± 44	68 ± 27	72 ± 31	55 ± 22	66 ± 25
Boron	4.2 ± 1.0	1.7 ± 0.49	2.0 ± 0.69	1.1 ± 0.37	1.0 ± 0.41	1.2 ± 0.30	1.1 ± 0.35	1.2 ± 0.76	1.5 ± 0.72	1.7 ± 0.70	1.2 ± 0.51	1.2 ± 0.61

In 2007/2008, $n=4$ for all results; otherwise $n=12$ per year for all results. Data presented as mg/L ± standard deviation for all the chemical components.

a portion of the waste in the landfill was in the acidogenic degradation phase, rather than the stable methanogenic phase. As time progressed after the deposition of these large MSW volumes, a proportionally larger part of the waste will have entered the methanogenic phase, leading to a corresponding decrease in the COD in the leachate.

However, the pH value of the leachate remained relatively constant throughout the entire sampling period (Table 1), indicating that the acidogenic phase was not a predominant part of the degradation processes in the landfill during this time. The strong buffering capacity of the landfill as a whole may have cancelled out any

acidification of the leachate occurring because of pockets of acidogenic degradation of waste at the top. This theory is supported by the fact that the nitrogen discharged in the leachate is almost entirely present as ammonium (Table 1). In anaerobic landfills, dissolved nitrogen compounds such as nitrate are entirely reduced to ammonium by microbial activity (Christensen *et al.* 2001).

Ehrig (1989) and Robinson & Gronow (1993) found the COD levels in leachate from acidogenic landfills to be 20–50 times higher than in the leachate from methanogenic landfills. Thus, even a small proportion of the landfill being in the acidogenic degradation phase could have significant impacts on leachate composition. As the amount of MSW landfilled decreased from 130 000 tonnes per year to just over 20 000 tonnes, the impact of newly landfilled, potentially acidogenic waste on the combined leachate would have diminished proportionally. Similar effects have been observed for COD in aerobic bioreactor landfills (Jun *et al.* 2007; Giannis *et al.* 2008), but in these cases the stabilization of COD levels in the leachate was achieved within 10–50 days after the establishment of the pilot landfills in these studies. The initial intensive aerobic biodegradation is known to cause high COD discharge, without affecting the pH significantly (Giannis *et al.* 2008). The works by Giannis *et al.* (2008) and Jun *et al.* (2007) support the idea that strong decreases in discharge levels of both COD and total phosphorus can be expected if there is a significant decline in the amount of MSW deposited on smaller landfills.

Leachate nitrogen concentrations (Table 1) did not show the same strong declining pattern as for COD and total phosphorus. This could be because of the relatively strong binding and retention of nitrogen, mainly in the form of proteins, in the waste masses (Burton & Watson-Craik 1998; Jokela *et al.* 2002). Unless this organic matter is actually degraded, the nitrogen will remain largely insoluble, and thus short-term changes in leachate flow will have little effect on the N leaching rate.

For iron and zinc, the declines in concentrations are less pronounced than for COD. However, there is still a significant decline in the discharge of these metals from the landfill after the decrease in the amount of waste deposited. Because the pH of the leachate did not increase much during the 10 years of the investigation, pH change cannot be a significant reason for the decline in the metal discharge.

As mentioned earlier, it is likely that a significant reason for the decline in leachate concentrations is the reduction in the amount of freshly deposited waste, with its lack of channelling. Channelling minimizes the contact between the water flow and the waste (Bendz *et al.* 1997; Rosqvist & Destouni 2000), and therefore a lack of established channels in the waste masses is likely to lead to an increased washout of easily soluble components.

Establishment of preferential flow paths is especially important for leachate quality in young landfills such as this one, as, at this early stage, degradation and settlement of the waste masses has not progressed as far as in older landfills (Bendz *et al.* 1997).

Metals such as zinc are expected to be present mostly bound to particulate ($> 0.45 \mu\text{m}$) matter (Jensen & Christensen 1999; Øygard *et al.* 2007). The rate of discharge of particulate matter via the leachate is likely to change over time. The stabilization of waste masses and formation of preferential flow paths in the landfill would lead to a decrease in the transport of colloidal matter (and thus of Zn and similarly metals bound to colloids) out of the landfill via the leachate.

Iron present in the waste masses will, in an anaerobic landfill, be reduced to Fe(II), which has a relatively high solubility. However, the discharge of Fe(II) from the landfill is still dependent upon contact between the waste and water. Here also, the formation of preferential flow paths results in decreased contact time between the leachate and the Fe(II) present in the waste masses, and thus a decrease in the discharge of iron in the leachate.

Conclusions

(1) Substantially reducing the amount of waste annually deposited on a landfill and subsequently maintaining deposition rates at a relatively low level (or stopping deposition altogether) lead to a sharp decrease in the concentrations of various components in the discharged leachate.

(2) At the landfill studied, the discharge rates of chloride, sodium and potassium, as well as the conductivity of the leachate, decreased sharply after an abrupt decline in the mass of MSW deposited. It is likely that such a decrease is caused by the rapid formation of preferential flow paths in the existing waste masses and the decline in the amount of fresh waste masses in the landfill. Variations in precipitation did not influence the leachate quality significantly compared with the other processes occurring.

(3) Aerobic processes occurring in fresh waste masses lead to a high rate of leaching of COD compounds shortly after the deposition of MSW. As anaerobic conditions are established in the waste masses, their COD level decreases. However, as long as a relatively high volume of MSW is steadily deposited on the landfill, a high COD level will persist in the discharged leachate. A decline in the amount of fresh waste masses available leads to a lowering of leachate COD. Additionally, discharge rates of total phosphorus and COD compounds are strongly affected (lowered) by the generation of preferential flow paths in the settling waste masses. This is also the case for the iron and zinc.

(4) Leachate total nitrogen contents did not change as much as the other components studied, suggesting that a rapid lowering of discharged nitrogen cannot be expected when decreasing or stopping the addition of MSW to a landfill. This is probably because the release of nitrogen is largely dependent on protein decomposition in the waste mass.

(5) The study shows that when predicting the environmental impact of a landfill, it is necessary to take into account possible changes in deposited waste masses. If a decline in the waste masses deposited at a landfill is imminent, and the landfill is of a small size compared with the masses currently deposited, the changes may impact the landfill leachate composition. A possible decline in leachate strength is an important information when assessing the environmental impact of a specific landfill. The decline in the leachate discharge does not imply a generally lower discharge from the landfill, but rather that a higher retention of polluting elements should be expected from such landfills. Over time, similar total discharge levels are likely.

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