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Management of landfill leachate: The legacy of European Union Directives

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ABSTRACT

Landfill leachate is the product of water that has percolated through waste deposits and contains various pollutants, which necessitate effective treatment before it can be released into the environment. In the last 30 years, there have been significant changes in landfill management practices in response to European Union (EU) Directives, which have led to changes in leachate composition, volumes produced and treatability. In this study, historic landfill data, combined with leachate characterisation data, were used to determine the impacts of EU Directives on landfill leachate management, composition and treatability. Inhibitory compounds including ammonium (NH_4-N), cyanide, chromium, nickel and zinc, were present in young leachate at levels that may inhibit ammonium oxidising bacteria, while arsenic, copper and silver were present in young and intermediate age leachate at concentrations above inhibitory thresholds. In addition, the results of this study show that while young landfills produce less than 50% of total leachate by volume in the Republic of Ireland, they account for 70% of total annual leachate chemical oxygen demand (COD) load and approximately 80% of total 5-day biochemical oxygen demand (BOD_5) and NH_4-N loads. These results show that there has been a decrease in the volume of leachate produced per tonne of waste landfilled since enactment of the Landfill Directive, with a trend towards increased leachate strength (particularly COD and BOD_5) during the initial five years of landfill operation. These changes may be attributed to changes in landfill management practices following the implementation of the Landfill Directive. However, this study did not demonstrate the impact of decreasing inputs of biodegradable municipal waste on leachate composition. Increasingly stringent wastewater treatment plant (WWTP) emission limit values represent a significant threat to the sustainability of co-treatment of leachate with municipal wastewater. In addition, the seasonal variation in leachate production poses a risk to effective co-treatment in municipal WWTPs, as periods of high leachate production coincide with periods of maximum hydraulic loading in WWTPs.

1. Introduction

Landfill leachate is the product of water that has percolated through waste deposits that have undergone aerobic and anaerobic microbial decomposition (Chofqi et al., 2004; Mukherjee et al., 2014). Leachate composition is a function of the type of waste in the landfill (biodegradable or non-biodegradable, soluble or insoluble, organic or inorganic, liquid or solid, and toxic or non-toxic waste material), landfill age, climate conditions and hydrogeology of the landfill site (Chofqi et al., 2004; Slack et al., 2005). A landfill site will produce leachate throughout its working life and also for several hundred years after it is decommissioned (Wang, 2013). As leachate contamination of groundwater, rivers, lakes and soils has the potential to negatively affect the local environment and human population (Ağdağ and Sponza, 2005; Marshall, 2009), the control of a landfill site and appropriate treatment of the leachate it produces is of paramount importance for the current and future protection of surrounding natural resources.

In 2012, 246 million tonnes of total municipal solid waste (MSW) was produced in Europe (equivalent to 487 kg of MSW per person), of which the highest per capita production was Switzerland and the lowest was Romania (Eurostat, 2015). There have been dramatic reductions in the volume of waste being landfilled in many European countries (Ireland, Czech Republic, Slove-

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nia, Norway, United Kingdom, Denmark, Iceland, Austria and Finland (Fig. 1). There has also been a reduction in the number of illegal landfills and an improvement in waste acceptance practices throughout the Member States (EC, 2007). In 2012 across the 28 EU Member States, 34% of all waste treated was sent to landfill, 42% was recycled, 4% was incinerated, and 15% was composted

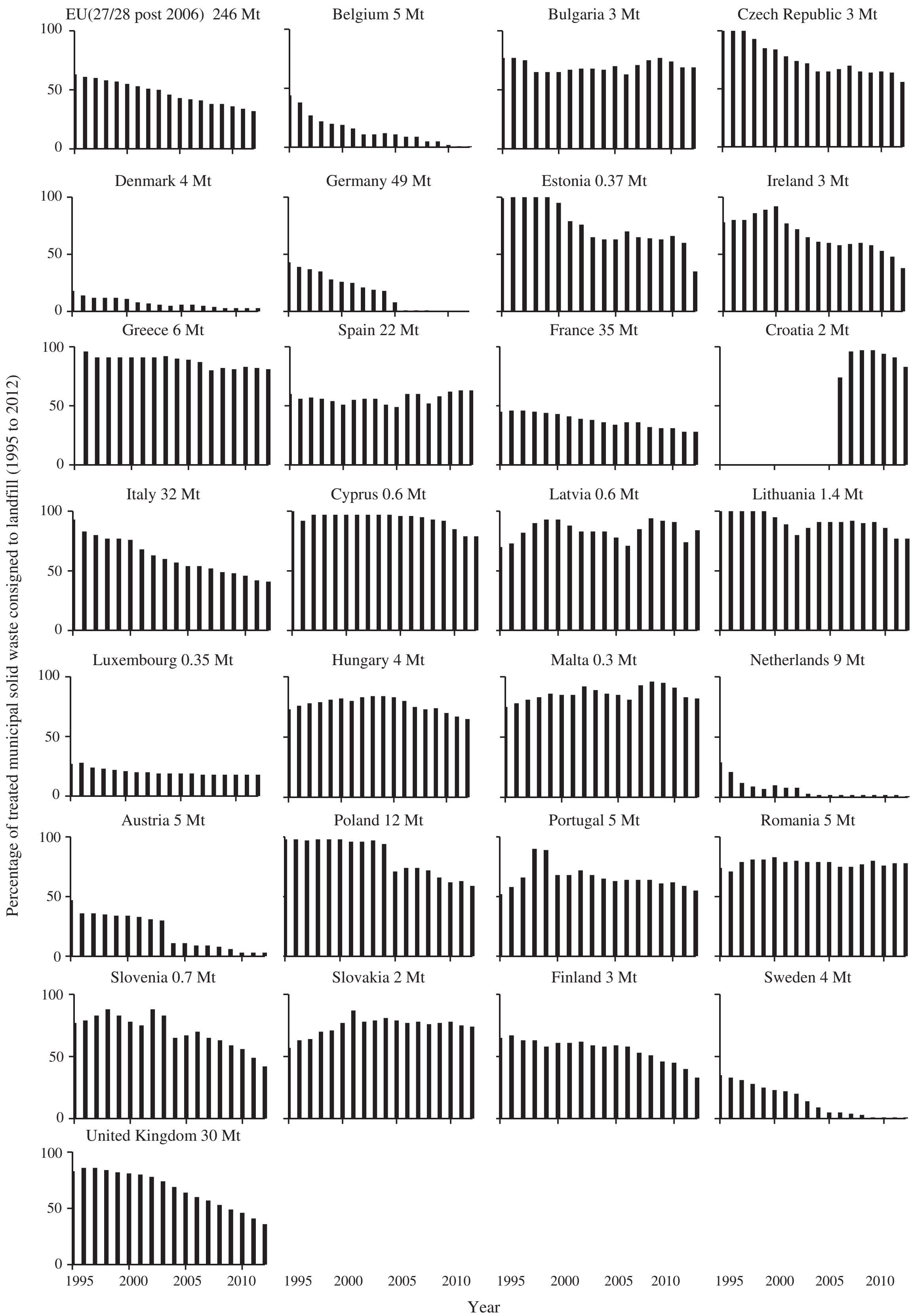

Fig. 1. Percentage treated municipal solid waste landfilled (Mt waste produced) (Eurostat, 2015).

or underwent anaerobic digestion (Eurostat, 2015). The Landfill Directive 1999/31/EC (EC, 2001a) requires that Member States reduce the amount of biodegradable municipal waste (BMW) sent to landfill on July 16, 2016 by 35% of the total amount of BMW generated in 1995. To date Austria, Belgium, Denmark, Finland, France, Germany, Hungary, Ireland, Luxembourg, Netherlands, Spain and Sweden have met these objectives (EPA, 2015; EEA, 2013).

The Landfill Directive, the Waste Framework Directive 2008/98/EC (EC, 2008), the Urban Wastewater Treatment Regulations Council Directive 99/31/EC (EC, 2001b) and the Water Framework Directive 2000/60/EC (OJEC, 2000) are among the main European regulations governing landfilling and leachate management. The Landfill Directive and subsequent Waste Framework Directive directly influenced leachate management practices, specifically leachate collection and disposal routes, which, in turn, influenced leachate chemical oxygen demand (COD), 5-day biochemical oxygen demand (BOD_5) and ammonium-nitrogen (NH_4-N) concentrations and loading. The Directives regulate the nature of wastes that landfills can receive, but also the execution of aftercare (normally 30–60 years). Recent research suggests that aftercare timelines of up to 200 years may be required, which may be reduced to 75 years where effective management practices are in place (Wang, 2013). Current EU policy proposes that all waste is managed as a resource and that landfilling is virtually eliminated by 2020 (EC, 2011; EEA, 2013).

In parallel to the regulations governing landfill management, the Water Framework Directive and Urban Wastewater Treatment Regulations have placed tighter regulations on all discharges to waters, and have resulted in stricter discharge limits being imposed on wastewater treatment plants (WWTPs) (EC, 2001b). Where landfill leachate is treated in WWTPs, plant managers are increasingly concerned over its impact on a WWTP's ability to meet discharge limits – in particular removal of NH_4-N, nitrogen (N) and organic carbon (McCarthy et al., 2010). Throughout the EU, co-treatment of leachate with domestic sewage in municipal WWTPs is common practice. Leachate acceptance by WWTPs is commonly based on the influent hydraulic loading into a WWTP, with a recommendation that leachate volume accepted be less than a certain percentage of annual WWTP load (e.g. 4% in Ireland).

The implementation of these directives has driven significant changes in the landfilling sector by decreasing the volume of waste sent to landfill (Fig. 1) and decreasing the amount of BMW sent to landfill (EPA, 2015). The long-term effect of these changes on leachate composition and treatability is currently unknown. However, it is anticipated that changes in leachate composition, combined with increasingly stringent discharge requirements, will require a multi-actor approach throughout Member States to successfully manage the legacy of landfill activities.

This paper examines leachate management within an EU Member State and explores the legacy of EU directives on landfill leachate. Taking the Republic of Ireland as an example, the objectives of this study were to examine (1) how EU directives have influenced landfill leachate management, and (2) leachate volumes, concentrations and treatability following the implementation of the Landfill Directive.

2. Materials and methods

2.1. Collation of existing landfill data in the Republic of Ireland

Data were collated from the 48 landfills collecting and exporting landfill leachate to WWTPs in the Republic of Ireland (representing 100% of all such landfills). These landfills comprised 22 young (operational/closed less than five years), 10 intermediate (closed more than five year but less than 10) and 16 old landfills (closed more than ten years). The age classification system employed was the same as that used by Renou et al. (2008) to allow for international comparisons. Data pertaining to landfill leachate management, including volumes of leachate collected for treatment, landfill leachate fate (i.e. facility receiving landfill leachate), leachate treatment practices (at the landfill), and available leachate characterisation data, were collated. The volume, strength and fate of all landfall leachate produced in the Republic of Ireland was subsequently determined.

Records detailing hydraulic loading rates and influent load (COD, BOD_5 and total nitrogen (TN)) of leachate to WWTPs were collected from 33 WWTPs in the Republic of Ireland. For each WWTP, the annual influent leachate volume received by the WWTP was expressed as a percentage of annual WWTP effluent volume, and annual influent COD, BOD_5, and NH_4-N leachate loads were expressed as a percentage of annual influent COD, BOD_5 and TN loads. Ammonium was expressed as a percentage of influent wastewater TN at the WWTPs, as the determination of influent NH_4-N concentrations is not a requirement at WWTPs in all cases, while TN analysis is not required at landfills.

Landfill operators of these 48 landfills were requested to provide historic records of waste acceptance, leachate production, characterisation and annual precipitation, and landfill sites were then identified for further study based on data availability.

2.2. Landfill selection and sample collection

Six of the 48 municipal solid waste landfills in Ireland, which were representative of modern engineered landfills (designed and managed post-2001), were selected for an eight-month leachate characterisation study in 2014. These comprised four young landfills and two intermediate aged landfills, which were lined (greater than 90% of landfill area lined) and capped (greater than 91% of area capped). Leachate collection points were chosen to ensure samples were representative of leachate exported off site to WWTPs (i.e. via pipe, manhole, or tanker collection/attachment point). Samples were collected and transported to the laboratory and samples preserved/analysis undertaken within 48 h (APHA, 2007).

2.3. Chemical characterisation of landfill leachate

Leachate TN was measured using a BioTector analyser (BioTector, Cork). Unfiltered COD was tested using Lovibond COD test kits. Unfiltered BOD_5 was analysed using WTW OxiTop meters. Filtered wastewater samples (Whatman GF/C; pore size: 1.2 μm) were tested for NH_4-N, chloride, sulphate and alkalinity using a nutrient analyser (Thermo Clinical Labsystems, Aquachem 150; Finland). Conductivity and pH were determined using a SAC950 sample changer and a Titralab 870. Cyanide was converted to cyanogen chloride by reaction with chloramine-T at pH below 8, without hydrolysing cyanide to cyanate. Following this, cyanogen chloride was reacted with pyridine and barbituric acid to form a red violet complex, and the absorbance of this complex was measured at 578 nm. All analyses were conducted in accordance with standard methods (APHA, 2007).

Total metals (arsenic (As), silver (Ag), copper (Cu), cadmium (Cd), chromium (Cr), iron (Fe), lead (Pb), nickel (Ni) and zinc (Zn)) concentrations were determined by Inductively Coupled Plasma Mass Spectrometry (ICP-MS) Shield Torch System (Agilent 7500a Technologies Inc. USA) following microwave digestion (CEM Discover SPD Microwave Digester) using Trace Metal Grade Nitric Acid (Fisher, UK).

2.4. Statistical analysis

Collated landfill data were analysed using ANOVA and data from the characterisation study were analysed using repeated measures ANOVA in SPSS (IBM SPSS Statistics 20 Core System, Version 20). Logarithmic transformations were required for all variables to satisfy the normality assumption based on checking post-analysis residuals for normality and homogeneity of variance.

3. Results and discussion

3.1. Overview of the impacts of the EU Directives on landfill leachate management practices

The overall number of operational landfills in the Republic of Ireland has decreased from approximately 200 in 1995 to 30 in 2008 (an 85% reduction), and there are less than five landfills currently receiving waste (EPA, 2014). This decrease was primarily due to the closing of many smaller landfills due to the costs associated with meeting licence requirements, which proved prohibitively expensive for smaller operators (McCarthy et al., 2010). In addition, landfill levies have encouraged waste recovery, recycling and the export of waste (Ovens et al., 2013; Fischer et al., 2012; EC, 2008). The shift from a large number of small landfills to fewer large landfills has resulted in the production of lower volumes of stronger leachate internationally (Robinson, 2005). Leachate containment and monitoring are now common practice at all licenced landfills (EPA, 2014) compared to less than one third of landfills in 1995 (Wall et al., 1998). Prior to 1995, landfills were designed based on the 'dilute and attenuate' principle, and leachate was observed to enter the environment either through seepage or overflow (Wall et al., 1998).

There was approximately 1.1 million m^3 of landfill leachate collected from MSW landfills for treatment in the Republic of Ireland during 2013 (EPA, 2014). While young landfills produce less than 50% of total leachate, in 2013 they accounted for 70% of total annual leachate COD load and approximately 80% of total BOD_5 and NH_4-N loads from all landfill types (Fig. 2) (EPA, 2014). Current practice is designed to discharge leachate to sewer (51%) or tankers for removal to WWTPs (48%) for final treatment (including leachate which undergoes treatment at the landfill (EPA, 2014)). Renou et al. (2008) reported that less than 21% of landfill leachate produced in France was co-treated with municipal wastewater in 2002 (similar to Germany where most landfills have separate leachate treatment plants (Rainer Stegmann (*per. com.*)), while in other European countries such as Poland (Kalka, 2012), the majority of leachate is co-treated with municipal wastewater in WWTPs. There is a dearth of publically available data concerning the fate of landfill leachate and leachate treatments used onsite in Europe. The reason for this is twofold. First, landfills (many of which are privately owned and operated) keep such information confidential, as it is commercially sensitive. Secondly, there is not currently a legal requirement for the landfill operators to report data to the EU.

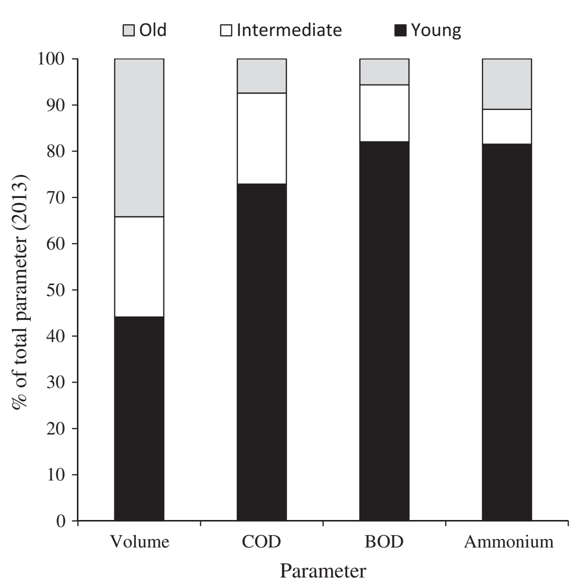

Fig. 2. Leachate volume (m^3), chemical oxygen demand (COD), biochemical oxygen demand (BOD_5) and ammonium load for young, old and intermediate landfills expressed as a percentage of total load from all landfills in 2013 (EPA, 2014).

There a wide range of technologies available for the treatment of landfill leachate, including coagulation (Liu et al., 2012), oxidation (Chemlal et al., 2014), struvite precipitation (Huang et al., 2014), constructed wetlands (Białowiec et al., 2012), membrane bioreactors (Sanguanpak et al., 2015) and biological treatment (Robinson et al., 2008; Syron et al., 2015). The cost of treatment can be divided into operational and capital costs, and can vary significantly depending on leachate type and site-specific conditions (including age of landfill, strength of leachate, volume of leachate produced, standard of construction, rainfall intensity, availability of appropriate receiving waters, and proximity to sewer/WWTP). In practice, landfill leachate collected in the EU is normally held in lagoons located on-site, prior to transfer to WWTPs for treatment (Kalka, 2012; Kurniawan et al., 2010). However, transport and treatment costs may be considerable (Stegmann et al., 2005). Co-treatment has the lowest capital cost but high operation costs, typically in the region of €25 per m^3 of leachate, while alternative treatments have higher capital costs and lower operational costs (i.e. on-site sequencing batch reactors discharging to sewer typically have capital costs in excess of €625,000 per treatment plant and operational costs of €1.90 per m^3 of leachate) (EA, 2007). Leachate treatment technologies currently in use in WWTPs in the Republic of Ireland include basic treatment (such as surface aeration of leachate lagoons and methane stripping (14% of all leachate produced)), or advanced treatments, including dedicated sequencing batch reactors and reverse osmosis systems (10% of all leachate produced). On-site treatment of landfill leachate is uncommon in Ireland compared to some other EU countries such as France, where 79% of leachate is treated on-site (Renou et al., 2008). Further on-site based studies are required to assess the cost effectiveness of co-treatment of landfill leachate compared to on-site treatment.

Analysis of the fate of leachate generated in the Republic of Ireland during 2013 showed that there was no WWTP in which the hydraulic loading of leachate exceeded 4% of the total influent hydraulic load, the threshold recommended by the EPA (Carey et al., 2000). The hydraulic loading of leachate accounted for between 0.01% and 3.8% of WWTP effluent (Fig. 3). Carbon loading ratios were similar, with leachate BOD_5 loading accounting for between 0.01% and 1.8%, and COD loading of between 0.01 and 5.8% (Fig. 3). However, landfill leachate NH_4-N loading accounted for between 0.01% and 33% of influent TN loading to the WWTPs analysed in this study (Fig. 3), which may be a concern for WWTP managers. These results indicate that hydraulic loading limits may not be appropriate when designing leachate acceptance criteria. Where WWTPs are not designed to treat such shock loads of leachate, concerns exist regarding the impact of landfill leachate addition on the biological wastewater treatment processes and the quality of the sludge generated (Çeçen and Aktaş, 2004). In the Republic of Ireland, non-compliance with NH_4-N and TN emission limits values at some WWTPs have been attributed to leachate loading and, in many instances, leachate acceptance has been discontinued in WWTPs. There has been a 30% decrease in the number

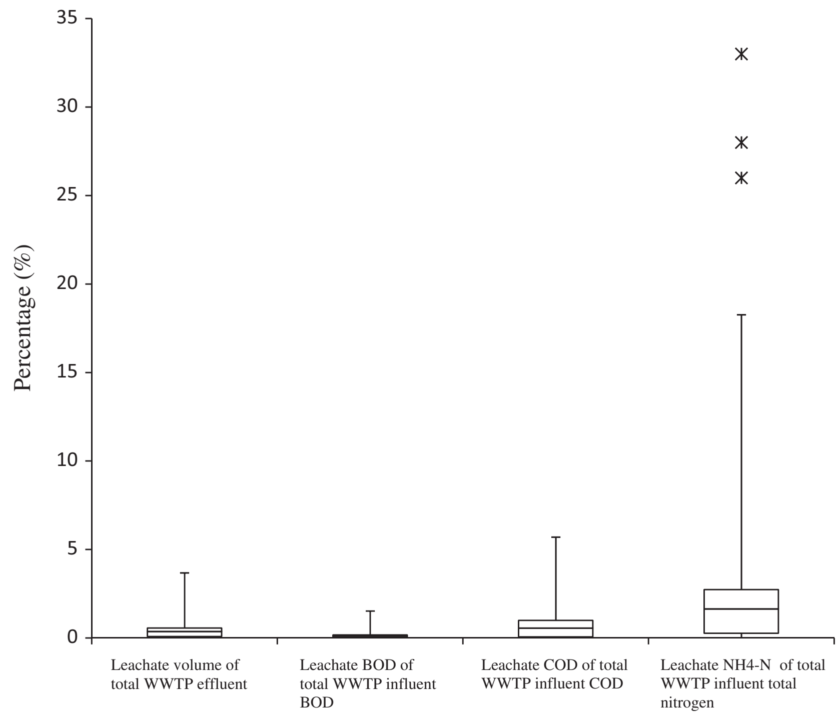

Fig. 3. Boxplot showing leachate volume, chemical oxygen demand (COD), biochemical oxygen demand (BOD_5) and ammonium (NH_4-N) loading expressed as a percentage of wastewater treatment plant (WWTP) effluent volume and influent COD, BOD and NH_4-N load (EPA, 2014). Current practice is to ensure that leachate volume does not exceed 4% of the hydraulic load of the plant (Carey et al., 2000).

of WWTPs co-treating landfill leachate between 2010 and 2015 (EPA, 2014). Increasingly stringent WWTP emission limits will continue to threaten the sustainability of co-treatment of leachate with municipal wastewater. While these data pertain to Ireland, it is likely that these challenges are being faced throughout Europe, particularly in countries such as Hungary, Latvia and Portugal, where over 70% of waste is landfilled (Eurostat, 2015).

3.2. Overview of collated landfill leachate composition and volumes produced in Ireland

As expected, the mean and median concentrations of BOD_5, COD, NH_4-N and chloride were greater in leachate from young landfills than the older landfills (Fig. 4). Median COD was 1100 mg L^{-1}, 693 mg L^{-1} and 221 mg L^{-1} for young, intermediate and old landfills, respectively, showing a decreasing trend with increasing age category (young, intermediate and old) of the landfills. Similarly, BOD_5 concentrations were observed to decrease (from 110 mg L^{-1} to 69 mg L^{-1} and 14 mg L^{-1}) and NH_4-N (from 352 mg L^{-1} to 218 and 98 mg L^{-1}). Young and intermediate leachate BOD_5, COD, NH_4-N and chloride concentrations were not significantly different to each other, but were different to old leachate ($p < 0.05$). These concentrations were in agreement with values reported elsewhere for intermediate (Frascari et al., 2004; Bohdziewicz et al., 2001) and old landfills (Robinson et al., 2008). However, the concentrations for the leachate from young landfills in this study were lower than those reported for young landfill leachate elsewhere (Kjeldsen et al., 2002; Renou et al., 2008; Kheradmand et al., 2010; Ye et al., 2014).

The median BOD_5:COD ratio of young leachate was significantly greater ($p < 0.05$) than intermediate leachate, which in turn was greater than old leachate (median 0.2 > 0.1 > 0.05) (Fig. 5). In a typical young landfill, leachate is characterised by an acidic phase of anaerobic degradation and a BOD_5:COD ratio of approximately 0.85, while older landfill sites have BOD_5:COD ratio of approximately 0.06 which is similar to the median value found in this study (Chofqi et al., 2004). These values were lower than expected for the new leachate, again, likely a result of blending of leachate, and were in agreement with Kjeldsen et al. (2002) for old and intermediate age landfills. The NH_4:BOD_5 ratio of young leachate was less than intermediate leachate, which in turn was less than old leachate (median 0.5 < 0.6 < 0.8). These were not significantly different.

Monthly leachate generation data, collated from 12 landfills, demonstrated seasonal variation in leachate generation in young, intermediate and old landfills. While there was insufficient data to conduct a statistical analysis, leachate volumes were observed to vary by a factor of three between summer (low) and winter (high). This was attributed to precipitation on open areas at young landfills and infiltration, and was consistent with the findings of Robinson (2005). These results may have implications for WWTPs receiving leachate from young landfills during periods of high hydraulic loading. Unless there is adequate storage capacity at the landfill or at the WWTP to buffer high flows during extreme events, there is a risk that WWTPs could potentially be loaded with leachate during periods when they are underperforming due to high hydraulic loading. In the Republic of Ireland, non-compliance with NH_4-N emission limits at some WWTPs have been attributed to accidental shock leachate loading (EPA, 2014).

3.3. Leachate characterisation

The results of the leachate characterisation study are shown in Table 1. There was a strong correlation between TN and NH_4-N ($R^2 = 0.98$, $p < 0.01$). In addition, TN:NH_4-N ratios close to one were observed in young and old leachate, indicating the possible presence of quaternary NH_4-N compounds, which have been found to inhibit biological treatment processes (Tezel, 2009). In this study, the inhibitory compounds NH_4-N, cyanide, sulphate, Cr, Ni and Zn, were present in young landfill leachate at concentrations which

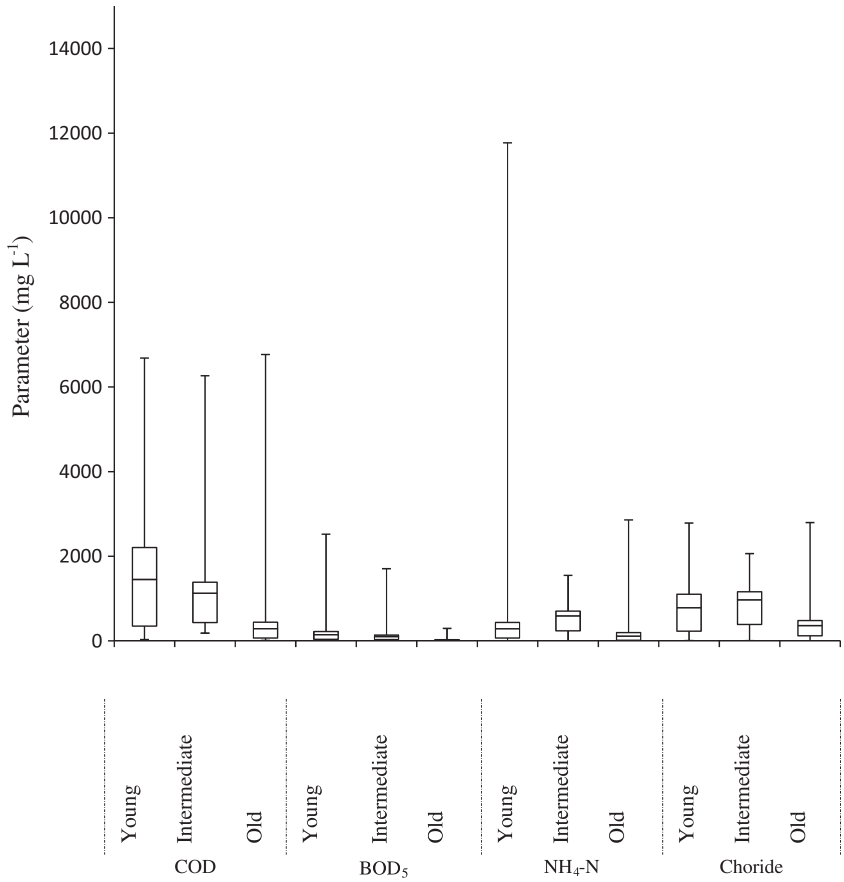

Fig. 4. Boxplot showing leachate chemical oxygen demand (COD), biochemical oxygen demand (BOD_5), ammonium (NH_4-N) and chloride concentrations for young, intermediate and old age landfills.

inhibit ammonium oxidising bacteria (Gerardi, 2002; Henze et al., 2002). Arsenic, Cu and Ag were present in young and intermediate age leachate at concentrations above inhibitory thresholds (Gerardi, 2002; Henze et al., 2002). However, the concentrations were lower than those reported internationally for young landfills (Kjeldsen et al., 2002; Renou et al., 2008; Kheradmand et al., 2010; Ye et al., 2014), indicating that provided leachate undergoes dilution, it is unlikely that concentrations would result in inhibition in co-treatment systems. However, landfill managers considering on-site biological treatment systems must be cognisant of these parameters and ensure that appropriate leachate management procedures are in place at WWTPs receiving leachate.

3.4. Leachate generation rates and trends in engineered landfills

There is a general decrease in the volume of leachate produced per tonne of waste landfilled with increase in opening year (i.e. newer landfills) (Fig. 6a). The exception being Case Study *1997yb* (y: young landfill; b: baled waste) which was unique, as only pre-sorted, mechanically baled material was landfilled at this site. In the case of all of the landfills examined except *2005yd* (y: young, d: diluted with leachate collected from old unlined cells), COD, BOD_5 and NH_4-N concentrations were highest in the first five to ten years after commencement of landfilling and decreased thereafter (Fig. 6b–d). These findings were consistent with Kjeldsen et al. (2002) and Tatsi and Zouboulis (2002), demonstrating the difficulties faced by designers when selecting appropriate treatment technologies and making capital investment decisions. Renou et al. (2008) recommended the use of packaged leachate treatment systems, which can be moved off-site when they are no longer required.

The observed decrease in the volume of leachate produced per unit waste landfilled in the landfills examined as part of this study was attributed to a combination of the effect of increased landfill size and improved leachate management practices – particularly improved containment and diversion of storm water from landfills. These changes are a direct consequence of improvements in landfill management following implementation of the Landfill Directive and WFD. Reducing the volume of leachate has significantly reduced transport costs where leachate is tankered from landfill to WWTP for final treatment; however, stronger leachate may require more advanced technologies to treat it effectively.

It was anticipated that the decrease in BMW fraction landfilled would result in a decrease in the leachate BOD_5 and COD concentrations and a change in leachate biodegradability (as determined based on BOD_5:COD), as it follows that decreasing BMW in waste would decrease biodegradable carbon in leachate (Adhikari et al., 2014). However, it was not possible to demonstrate a change in leachate composition in response to change in the composition of BMW. This this may be attributed to: (1) insufficient available leachate characterisation data to compare pre- and post-2001, which is the year the directive was implemented; (2) major changes in landfill size, management and design during this period, which may also have influenced the strength of landfill leachate (3) potential lag-time between change in composition of waste being deposited and change in leachate composition, which can be up to three years (Robinson, 2005), and (4) the effect of blending young and old landfill leachate.

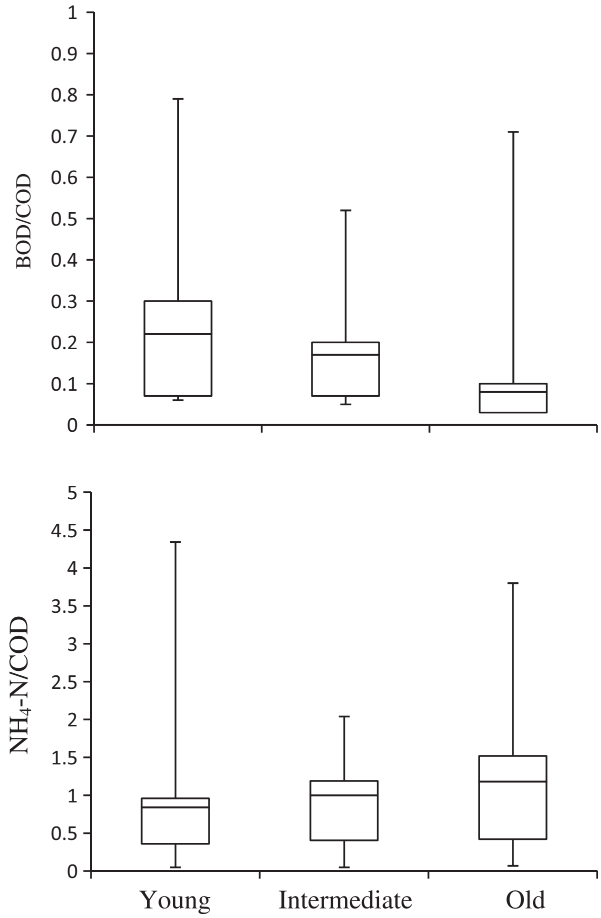

Fig. 5. Boxplot showing leachate chemical oxygen demand (COD) to biochemical oxygen demand (BOD_5) ratio and ammonium (NH_4-N) to COD ratio.

3.5. Environmental policy and management implications for the European Union

Although in decline, landfilling is prevalent in some EU member states (Fig. 1) and many countries throughout the world (Robinson, 2005), and the results of the current study has relevance for these countries. Significant advancements in the past 30 years have reduced the impact of landfills on the environment as a result of the Landfill Directive. The directive is widely considered to be a success and has dramatically improved waste management practices throughout the EU (EEA, 2009). However, the legacy of landfilling activities, particularly landfill leachate production, will potentially be a problem for landfill operators and regulators for many decades to come. The 246 Mt of waste landfilled in EU in 2012 had the potential to produce between 49 and 246 Mt of leachate within the EU (assuming m^3 leachate generated per tonne of waste landfilled of between 0.2 and 1.0) with the cost of leachate treatment estimated to be between €2.10 and €25 per m^3 of leachate (EA, 2007). Leachate treatment will continue to be major concern for landfill and WWTP managers due to the increasingly stringent water quality emission limits placed on WWTPs by Council Directives (EC, 2001b; OJEC, 2000). The problem of leachate management may be compounded by the decrease in landfilling and reduction in revenues being generated, reducing the capacity of landfill operators to invest in treatment facilities. Future research efforts must focus on sustainable options for the treatment of high strength leachate, and data sharing must be encouraged across EU Member States to provide increased information for decision making, especially in accession countries where lessons from Ireland may inform future policy regarding landfilling.

4. Conclusions

The findings of this study are as follows:

1. The implementation of EU Directives has resulted in significant advances in landfill management and protection of the environment from the adverse effect of landfilling.

Table 1
Characterisation of young and intermediate landfills designed and constructed post-Landfill Directive implementation.

Parameter		Young leachate				Intermediate leachate				Inhibitory thresholds[a]	*p*-value
		Range	Mean	Median	St. Dev.	Range	Mean	Median	St. Dev.		
pH	pH	7.6–8.5	8	8	0.28	6.8–8.4	8	7.65	7.52		<0.05
Conductivity	µs cm^{-1}	3089–28430	12664	12615	7316	2606–10440	5307	4502	2416		<0.001
Total nitrogen (TN)	mg/L	120–4027	1352	1000	1120	120–1083	354	279	261		<0.001
Ammonia/TN		0.15–1.00	0.78	0.9	0.25	0.1–0.99	0.74	0.88	0.31		<0.05
BOD_5	mg/L	36–984	342	335	264	6–33	13	11	8		<0.001
COD	mg/L	411–7160	2656	2256	1776	190–748	361	321	141		<0.001
Alkalinity	mg/L	998–9682	3521	3164	2062	10–2100	968	971	571		<0.001
Chloride	mg/L	160–2620	1218	1058	805	130–669	301	290	141		<0.001
Inhibitory compounds											
Ammonia	mg/L	130–4000	1084	772	1005	63–378	203	175	101	480	<0.001
Cyanide	µg/L	6–1164	252	178	278	6–81	30	31	22	100–500	<0.001
Sulphate	mg/L	7.2–1950	394	289	436	21–445	184	117	130	500	<0.05
Arsenic	µg/L	11–412	148	77.5	128	14.6–155	45	30.6	43	50–100	<0.001
Cadmium	µg/L	0.1–7.4	1	0.75	2	0.1–1.6	0.48	0.4	0.51	1000	<0.001
Chromium	µg/L	33–1436	446	279	408	28–284	84	55	74	1000	<0.05
Copper	mg/L	0.003–2.423	0.37	0.1325	1	0.011–0.157	0.04	0.03	0.05	0.1–0.35	<0.01
Lead	µg/L	0.6–1047	56	12	203	0.9–8.2	4	3.45	3	500	<0.001
Mercury	µg/L	0.02–1.07	0.28	0.235	0.24	0.02–2.05	0.32	0.12	1	100–250	<0.05
Nickel	µg/L	10–661	206	186	169	22–151	54	42	36	250–500	<0.001
Zinc	µg/L	10–7639	496	58	1507	10–303	83	60	86	100–300	<0.05
Silver	µg/L	10–2187	252	10	583	10–280	50	10	93	250	<0.05

[a] Threshold of inhibitory effect on heterotrophic organisms in activated sludge wastewater treatment plants (Gerardi, 2002).

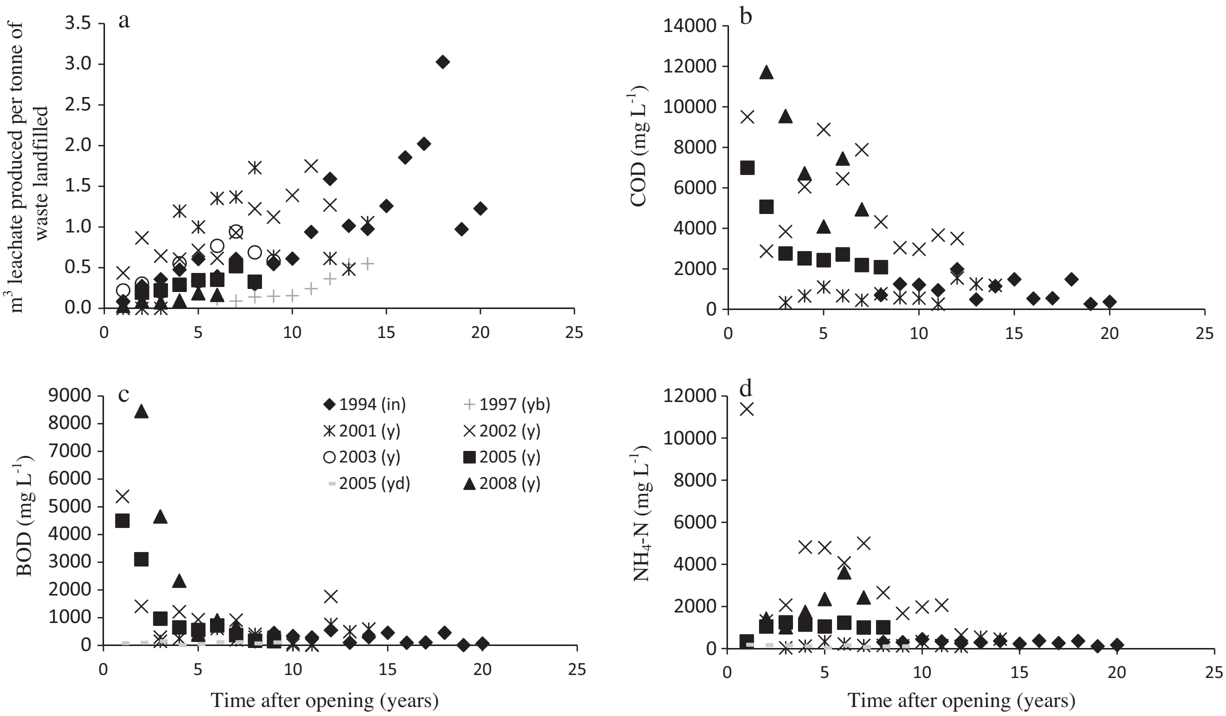

Fig. 6. Relationship between (a) volume of leachate produced per tonne of waste landfilled, (b) chemical oxygen demand (COD), (c) biochemical oxygen demand (BOD_5), and (d) ammonium (NH_4-N) concentrations and time after landfilling started landfilling waste. (YYYY: year landfill opened; y: young leachate, in: intermediate age leachate; b: waste baled before landfilling (EPA, 2014).

2. There is huge temporally and spatially heterogeneity in leachate strength, with young landfill leachate comprising 42% of leachate volume. Young leachate accounts for over 70% of COD and BOD load and 80% of NH_4-N leachate load in the Republic of Ireland. Therefore, treatment of high strength (mainly young) leachate should be a priority.
3. Changes in landfill management, brought about by the EU directives, have resulted in a decrease in the volume of leachate produced per tonne of waste landfilled, and there is a trend towards increased leachate strength (particularly COD and BOD_5 during the initial five years). However, this study did not demonstrate the impact of decreasing BMW on leachate composition.
4. Increasingly stringent WWTP emission limits represent a significant threat to the sustainability of co-treatment of leachate with municipal wastewater.
5. The seasonal variation in leachate production poses a risk to effective co-treatment in municipal WWTPs, as periods of high leachate production coincide with periods of maximum hydraulic loading in WWTPs.

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