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BOREHOLE DILUTION TESTS TO MEASURE LEACHATE FLOW AT A RECIRCULATION LANDFILL

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ABSTRACT: The recirculation and infiltration of leachate in landfills may be carried out to facilitate the flushing of contaminants and accelerate the stabilisation of waste. Flushing contaminants through recirculation relies on the movement of fluids through the landfill body to basal drains, which will predominantly be driven by gravity. Leachate recirculation and infiltration measures commenced at de Kragge II landfill (Bergen op Zoom, The Netherlands) in March 2018. Up to 90 m^3/day of treated leachate is recirculated into the top of a 20 m deep, 5 ha landfill cell through 14 horizontal drains installed at the surface. Poor connectivity between the waste and the basal drainage system has resulted in saturated conditions forming in the lower 7-8 m of the landfill. Knowledge about the leachate flow within the waste body is essential for evaluating the success of the stabilisation measures. To investigate the flow regime within the saturated waste, 22 Single Borehole Dilution tests were carried out in 13 piezometers at different depths, between 8.4 and 18.1 m below ground level, and locations across the landfill cell. Tests were repeated in a number of the piezometers to demonstrate repeatability. Flow was measured in all piezometers. Calculated Darcy flow velocities ranged between 0.01 and 1.02 m/day, with the highest velocities measured in the deepest piezometers. Four tests were carried out in one nest of piezometers installed at different depths, with the leachate recirculation system switched off for two days prior to and during the test. Although flows were somewhat higher in two of the piezometers, it was not possible to conclude whether the infiltration of leachate significantly influences flow.

Keywords: recirculation, tracer tests, well tests, hydraulic properties

1. INTRODUCTION

The recirculation and infiltration of leachate in landfills may be carried out to facilitate the flushing of contaminants and accelerate the stabilisation of waste (e.g. Laner et al., 2012; Scharff et al., 2011). Flushing contaminants through recirculation relies on the movement of fluids through the landfill body to basal drains, which will predominantly be driven by gravity. The structure of waste promotes preferential flow and short-circuiting through inter-connected macro-pores that surround zones of lower permeability waste where advective flow is negligible. This is termed dual-porosity flow and has been demonstrated in laboratory and field experiments (Rees-White, 2021, Rosqvist et al., 1997, Woodman, 2015, Woodman, 2017).

Waste is also strongly anisotropic, owing to the horizontal alignment of plate- and sheet-like components under vertical overburden stress. Flow paths may be constrained along and parallel to these horizontal layers, with downward flow restricted to gaps between the low-permeability layers (Woodman, 2014). During flushing, dual-porosity flow and anisotropy may result in areas of waste that are isolated from flow. The efficacy of flushing will, therefore, be highly dependent on the nature and distribution of flow paths within the waste.

Single borehole dilution tests (SBDT) are a single-well tracer test that can be used to estimate the Darcy flow velocity (volume per unit area per unit time) within the saturated zone of porous media (Lloyd et al, 1979, Rollinson et al, 2010). Test data can be interpreted to demonstrate spatial heterogeneity across the landfill, and may also indicate variation in flow with depth, identifying preferential flow-zones and flow paths.

In this research, 22 uniform borehole dilution tests were carried out in piezometers installed at different depths to investigate horizontal flow velocities of the de Kragge II landfill, Netherlands which is being operated under a recirculation flushing regime. The tracing dye rhodamine WT was introduced in a test well and the concentration of the dye monitored over time as the prevailing flow diluted the tracer. The rate of dilution is indicative of the flow velocity and, therefore, hydraulic conductivity in the immediate vicinity of the well.

2. METHODOLOGY

2.1 Test Site

The Dutch Sustainable Landfill Management experiment (iDS) has been operating since 2010 with its primary goal to reduce landfill after-care through enhanced biological stabilisation of the waste body. This is being achieved through aeration at two sites and through the recirculation of leachate at the de Kragge II landfill (hereon termed Kragge), Bergen op Zoom, The Netherlands. The borehole dilution tests were carried out at Kragge.

Kragge is a land raise landfill comprising four cells delimited, though hydraulically connected in the waste, by shallow clay bunds. Cells 1 and 2 are restored with a composite liner, Cells 3 and 4 are overlaid with a shallow, permeable soil cover. The recirculation project only operates in Cell 3.

Cell 3 has an area of 5.6 Ha, an average waste depth of 17 m (20 m maximum depth) and contains ~1 Mt of domestic waste. A 300 mm deep drainage blanket of coarse gravel overlies an HDPE liner at the base of the site. Six parallel, 200 mm diameter perforated drains run approximately north-south through the drainage gravel. The base drains are spaced 25 m apart and link to a common ring main which drains to a single gravity-fed pumping station. Cell 3 infrastructure is shown in Figure 1.

Leachate recirculation and infiltration measures commenced at Kragge in March 2018. An average of 90 m^3/day of treated leachate (periodically supplemented with groundwater) is recirculated into the top of the landfill through 14 horizontal trenches installed ~0.8 m below the surface, perpendicular to the base drains (Figure 1). Each pair of recirculation trenches is operated sequentially for a short period every hour, such that each trenched is dosed two to three times a day. Since it started, 138,000 m^3 has been recirculated into the infiltration trenches. Slow infiltration rates in some trenches resulted in overflow and surface ponding. This was addressed through supplementary shallow drilling along the trenches to encourage infiltration. Poor connectivity between the waste and the basal drainage system and/or significant horizontal to vertical anisotropy in the waste's hydraulic conductivity has resulted in saturated conditions forming in the lower 7-8 m of the landfill.

To support other research projects being carried out at Kragge (not reported here), monitoring piezometers and wells have been installed in Cell 3 (Figure 1). Individual piezometers and 'nests' of closely-spaced piezometer clusters have been installed at different depths within the saturated and unsaturated waste. Piezometers were drilled using sonic drilling techniques which displaces the waste to create a 110mm diameter borehole. Boreholes were installed with 50mm plastic casing with a 2 m screened section at the base. A coarse sand was placed around the screen, bridged with a fine sand and the remaining borehole filled with bentonite to the surface.

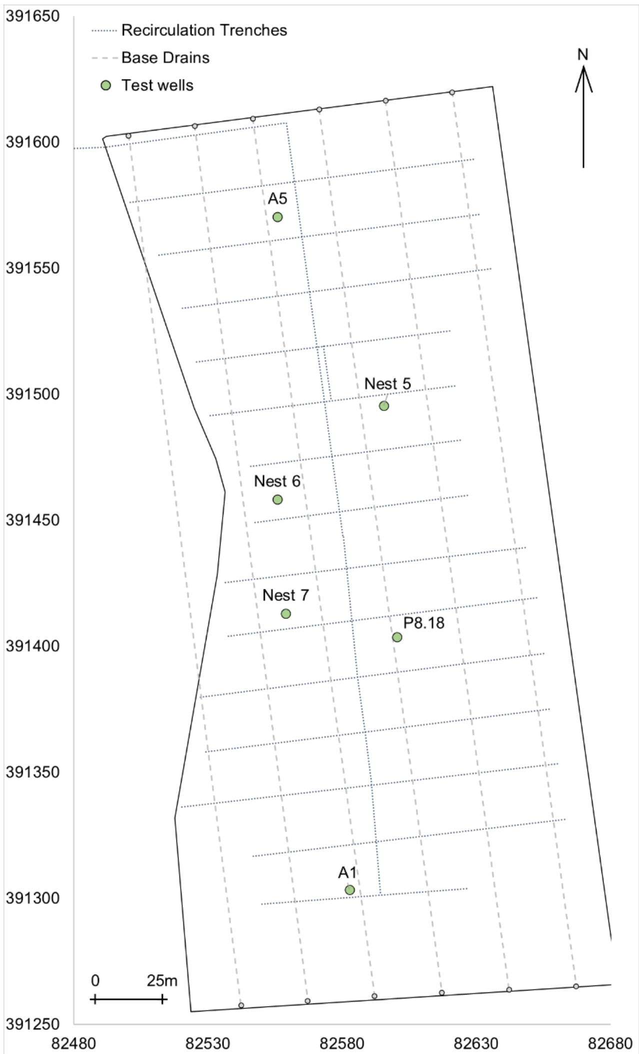

Figure 1. Cell 3 of de Kragge II landfill showing base drains, leachate recirculation system and location of test piezometers.

2.2 Single Borehole Dilution Test (SBDT)

A uniform SBDT is a relatively simple hydrogeological technique requiring a single test well or piezometer. Tracer is introduced throughout the saturated depth in the well and the concentration of the tracer monitored as the prevailing groundwater (or in this instance, leachate) flow dilutes the tracer in the well. The rate of dilution is determined through repeated tracer concentration measurements made at regular horizons throughout the well. The rate of dilution is indicative of the flow velocity and, therefore, the hydraulic conductivity in the immediate vicinity of the well.

Lloyd et al. (1979) report the first use of dilution tests in landfill. They used a fluorescent dye tracer and sampled intrusively at a single location within the borehole giving in a single estimate of flow velocity. Rollinson (2019) reports the results of 28 dilution tests using fluorescent dye tracers, either fluorescein and rhodamine WT, performed at 4 waste disposal sites in the UK.

2.3 Methodology

The fluorescent dye tracer rhodamine WT (RWT) was chosen for the SBDTs. RWT is considered to be conservative and can be accurately measured at low concentrations in leachate using a submersible fluorometer.

At the start of the tracer test, prior to tracer injection, a down-well RWT fluorometer (Cyclops-7F, Turner

Designs) connected to a datalogger (CR10X, Campbell Scientific) with 25m of cable marked at 0.25 m increments was lowered through the leachate column. Background fluorescence measurements were recorded every 0.25 m. This initial background concentration profile was the expected profile to which the tracer test will return to marking the end of the test.

For the SBDT, tracer was only needed in the lowest 2m screened section of the piezometer. Tracer in the blind casing above the screen would not be affected by prevailing flow and might remain in the well for a long duration, dispersing only slowly and potentially influencing the results of other tracer tests. To introduce the tracer only within the screened section, a weighted hose was lowered to the base of the piezometer and a measured volume of tracer was added to the hose. This was chased with leachate from the well to ensure that the tracer was pushed to the end of the hose, which was carefully removed to leave the tracer just within the screened section. A small cone-shaped attachment on the end of the hose was slowly raised and lowered within the screened section to mix the tracer.

Directly after the hose had been removed, the fluorometer was lowered into the well and an initial starting profile of tracer concentration was made at 0.25m depth intervals. The concentration of the tracer was calculated such that the mixed-volume was within the linear range of the fluorometer.

Following tracer introduction, 0.25 depth measurements of tracer concentration were the made regularly (approx. every 30 to 60 minutes) until the tracer concentration had returned to background.

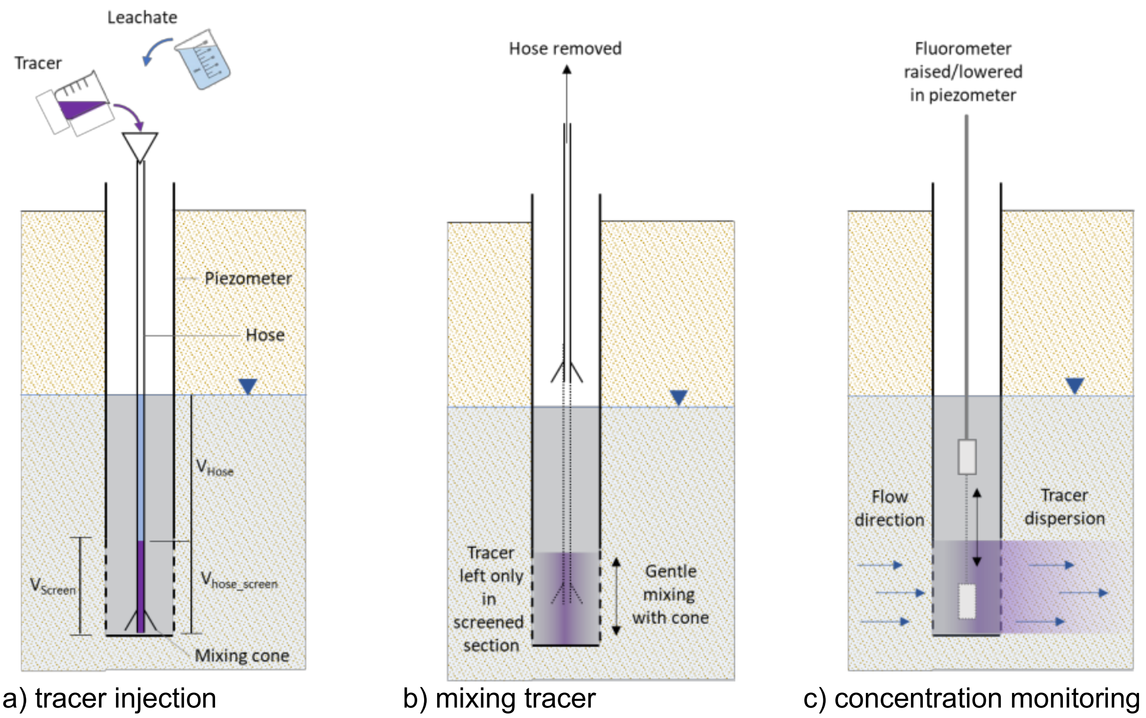

Figure 2. Tracer injection and monitoring technique.

2.4 Data Processing

Several factors can affect the fluorescence intensity measured by the RWT fluorometer. Most significantly, these include temperature, turbidity and the apparent natural background fluorescence of the leachate. Fluorescence is inversely related to temperature, with increasing temperature resulting in a decrease in the measured fluorescence intensity. At Kragge, the temperature of the leachate, measured using down-well sensors, was relatively stable at ~20 °C, although there was some spatial and temporal variation. For the purposes of the SBDT where all monitoring was made in the same well over a relatively short time frame, no temperature compensation was considered necessary.

Turbidity can affect RWT measurements by quenching the signal from the fluorometer and also through light scattering. This will reduce the intensity of the measured output. Conversely, suspended and dissolved organic matter in leachate can result in an increase in the fluorescence output giving a false, albeit low, background signal. Again, for a given test, as all measurements were being made in the same well and it was the relative change in fluorescence that was being measured, no compensation for turbidity or background signal was made.

Calibrating the sensor to actual RWT concentration is, for the reasons described above, difficult owing to the unique chemistry of the leachate in individual wells. However, the output of the fluorometer is linear up to ~1000 ppb and it was the relative change in tracer concentration over time that was required for calculating the flow velocity. Therefore, provided the output was within the linear range, the uncorrected output of the fluorometer (measured in Arbitrary Units, AU) was deemed suitable for use without conversion to tracer concentration.

Dilution test data was analysed using a standard method (see Ward et al., 1998; Pitrak et al., 2007). The method assumes that tracer does not move vertically within the borehole and that dilution is exponential with time. Time-series charts were plotted (natural log of concentration against time) for each 0.25 m monitored interval within the boreholes. The Darcy velocity, q, was calculated for each interval using Equation 1.

$$q = \frac{\pi r m}{2\alpha} \tag{1}$$

Where r is the well radius, m is the gradient of the ln[concentration] versus time graph and α the well borehole correction factor. An example data set is given in Figures 3 and 4.

The value of the borehole correction factor, α, is dependent upon the hydraulic conductivity of the aquifer, the filter pack (if present) and the well screen, and the dimensions of the well screen and borehole. It is generally accepted to use a borehole correction factor of 2 for a well in a homogeneous aquifer that is in good hydraulic connectivity with the aquifer, or if the value cannot be calculated directly (Piccinini et al., 2016; Fahrmeier et al., 2021). Due to a lack of knowledge about the surrounding waste and installation of the filter pack, a borehole correction factor of 2 and an inner casing diameter of 46 mm was used in all calculations.

3. RESULTS

22 SBDT were carried out in 13 piezometers. This included five repeat tests in the same well to demonstrate repeatability, and four repeat tests conducted when the recirculation system was off-line to investigate whether flow was influenced by leachate recirculation. Table 1 gives details of each test including the well depths and whether the recirculation system was operational during the test. The location of the test wells is shown in Figure 1.

Although the tracer injection procedure was followed carefully for each test, in reality it was difficult to accurately get tracer just within the screened section of the well and at the target concentration. In some instance tracer was released in the upper blind section of the piezometer, and in other tests the concentration of the tracer in the screened section was significantly higher or lower than expected. Possible reasons for these discrepancies are discussed below.

The starting tracer concentration, or having an uneven distribution of tracer throughout the well does not effect the analysis. However, as discussed, it was preferable to keep the tracer concentration within the fluorometer's linear calibration range to help simplify the analysis.

Figure 3 shows an example concentration vs depth plot and calculated velocities for test 5.13-3 (see Table 1). Tracer was injected relatively uniformly through the screened section of the piezometer.

Figure 4 shows data from test 7.12-1. In this test, tracer was inadvertently mixed into the entire saturated depth of the piezometer, including within the blind section. The data show that, as the tracer in the screen disperses, the tracer in the blind section remains. Data from just above the screen demonstrates that there is mixing between the two zones, possibly due to turbulence caused by raising and lowering the fluorometer. Only data from within the screened section was used in flow calculations.

Horizontal flow was measured in all the piezometers. Calculated flow velocities ranged between 0.01 and 1.02 m/day. Piezometers 5.13, 7.9, 7.15, and A5, showed considerably lower flow than was measured in other piezometers, even from the same nest or from the same depth. The highest velocities were measured in piezometers A1 and 7.18. Overall velocities were estimated to be highest at the base of the measured section of the landfill.

Full results are given in Table 1 and calculated velocities (averaged if more than one test was carried out) for each monitoring interval, normalised to metres above sea level (mNAP), are shown in Figure 5.

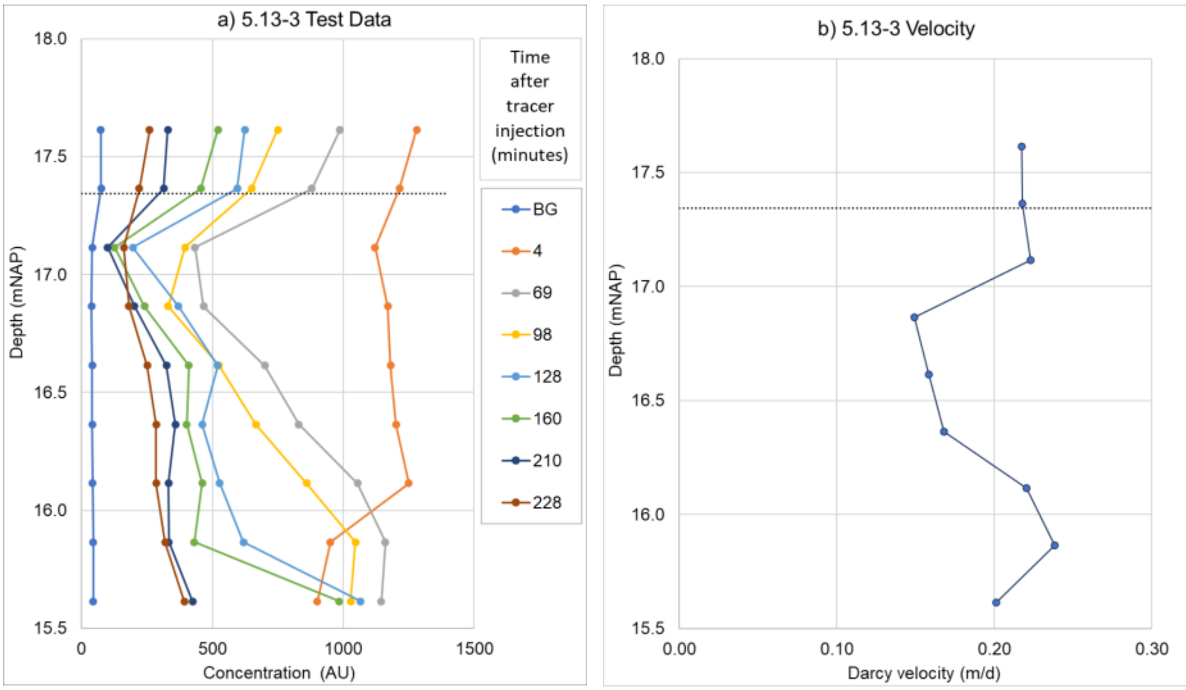

Figure 3 a) fluorometers results for test 5.13-3, b) calculated velocity for each monitoring interval. The dashed line shows the estimated top of the piezometer screen

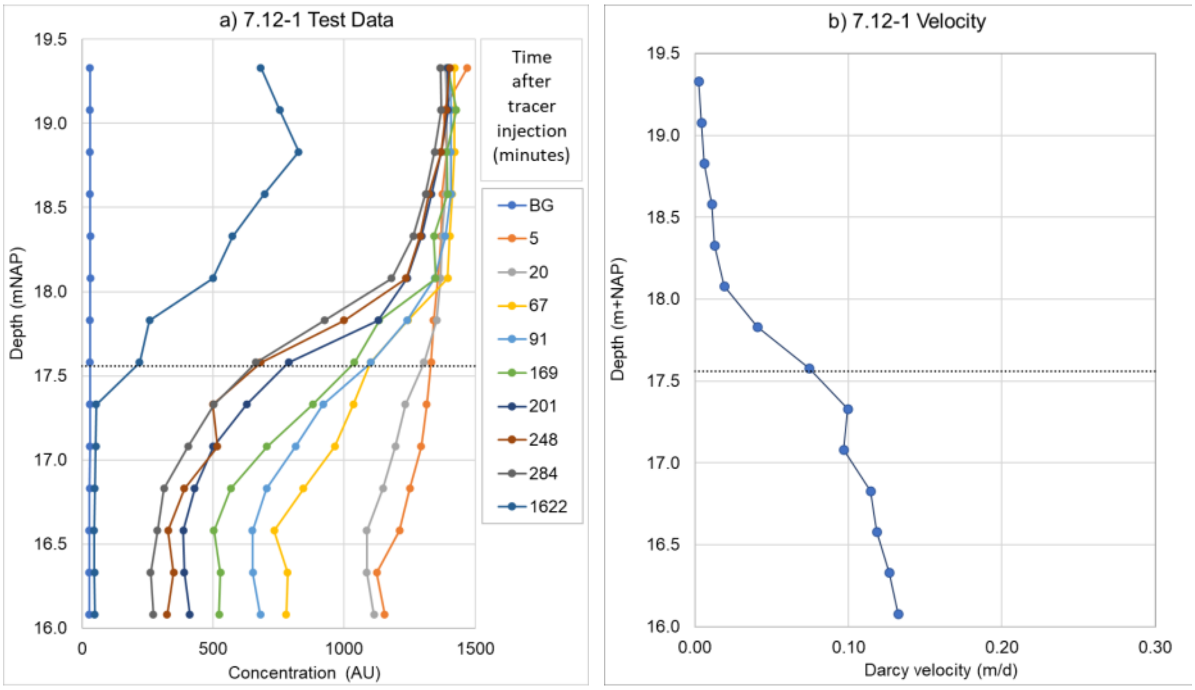

Figure 4 a) fluorometers results for test 7.12-1, b) calculated velocity for each monitoring interval. The dashed line shows the estimated top of the piezometer screen.

Table 1. Test data and calculated flow velocities

Nest No.	Well No.	Test No.	Leachate Recirculation	Depth below ground level m	Screened Depth mNAP	Darcy Flow Velocity m/day	Average Velocity of all tests m/day
5	5.10	-1	Operational	9.62	18.4 to 20.4	0.29	0.54
		-2	Operational			1.02	
		-3	Off			0.30	
	5.13	-1	Operational	12.42	15.6 to 17.6	0.05	0.09
		-2	Operational			0.04	
		-3	Off			0.19	
	5.15	-1	Operational	15.38	12.6 to 14.6	0.07	0.11
		-2	Off			0.15	
	5.18	-1	Operational	17.45	10.5 to 12.5	0.36	0.33
		-2	Operational			0.22	
		-3	Off			0.41	
6	6.11	-1	Operational	10.45	18.3 to 20.3	0.31	0.31
	6.14	-1	Operational	14.41	14.4 to 16.4	0.17	0.17
7	7.9	-1	Operational	9.46	19.3 to 21.3	0.05	0.03
		-2	Operational			0.02	
	7.12	-1	Operational	12.33	16.5 to 18.5	0.11	0.13
		-2	Operational			0.15	
	7.15	-1	Operational	15.26	13.6 to 15.6	0.08	0.08
	7.18	-1	Operational	18.11	10.7 to 12.7	0.91	0.91
	8.18	-1	Operational	16.83	11.6 to 13.6	0.32	0.32
	A1	-1	Operational	8.72	10.7 to 12.7	0.87	0.87
	A5	-1	Operational	8.35	10.1 to 12.1	0.01	0.01

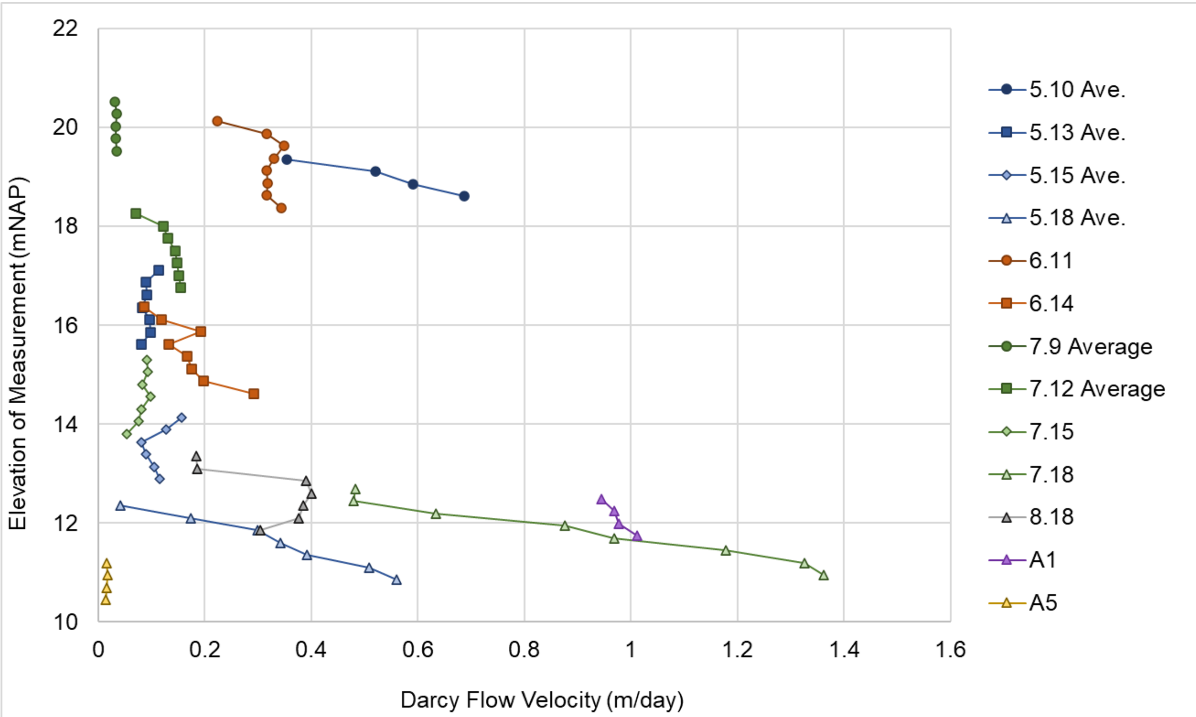

Figure 5. Calculated Darcy flow velocities for each test. Where more than one test was performed in the same piezometer, the average velocity is given. Data is normalised to mNAP

4. DISCUSSION

Accurately injecting the tracer into the screened section of the piezometer was more difficult than expected. In some instances more tracer was injected into the blind section above the screen than within the screen. Reasons for the spread of tracer into the blind section of casing maybe due to over-vigorous mixing after tracer injection or from unintentional mixing due to the fluorometer being raised and lowered in the piezometer. Although the uneven distribution of tracer did not significantly affect the analysis, it did mean that there was often tracer remaining in the piezometer after the test had finished. In some instances, to prevent the risk of interference with later tests, the leachate from the upper piezometer was purged.

In many of the tests, the starting concentration of tracer was considerably lower than had been estimated based on the inside diameter or the screened piezometer and the mass of tracer injected. Although it was assumed that the initial mixing of the tracer from the hose would occur only within the piezometer, it is possible that there was enough free flow of leachate out of the screen to include dilution in the filter pack. Further to this, during the installation of the borehole, a coarse sand was placed around the well screen. For the deeper piezometers in particular, it is possible that bridging occurred in the upper parts of the borehole which resulted in voids between the borehole and the screen. In some instances, assuming that no filter sand reached the base, if the volume of the open borehole is taken into account in the initial mixed tracer concentration, then the measured starting concentrations are much closer to that expected. However, this was not the case for all piezometers.

Although the starting concentration does not affect the analysis, increasing the diameter used in Equation 1 to obtain a better match between the predicted and measured starting concentrations, does result in an increase in the calculated velocity. For example, if the drilled borehole diameter of 110 mm were used (instead of 46mm), the velocities given in Table 1 would increase by ~120 %.

Despite the uncertainty of the influence of a poor borehole installation and possible voids in the filter pack, the results of the SBDT were relatively consistent across all piezometers and across the landfill, although there was more variability with depth. Rollinson (2019) report the results of 28 SBDTs at four different UK landfills, with measured velocities between 0.002 and 0.6 m/day. Rees-White et al (2022) report results of six SBDTs in four UK landfills. Velocities ranged between 0.005 and 0.271 m/day. Both of these reported tests are within the range of those tests reported here (0.01 to 1.02 m/day).

Tests 5.10-3, 5.13-3, 5.15-2 and 5.18-3, were carried out after the surface infiltration system had been turned off for two days prior to testing. The aim was to investigate the influence of infiltration on flow. Figure 6, shows test data from piezometer Nest 5 measured with the surface infiltration system operational and, in each instance, a test completed when the system was off (indicated by a dashed line). Nest 5 is located ~5 m from an infiltration trench to the north, and ~17 m from a trench to the south (Figure 1). In piezometer 5.10 and 5.18, no difference was noted between the tests. In tests 5.13 and 5.15, the velocities at all elevations were higher when the infiltration system was off.

Without additional repeat testing and similar tests in other nests, it is not possible to conclude whether the differences measured in 5.13 and 5.15 are due to the recirculation system or just variation in measured flow. For example, there was greater variation measured in piezometer 5.10 when recirculation was on.

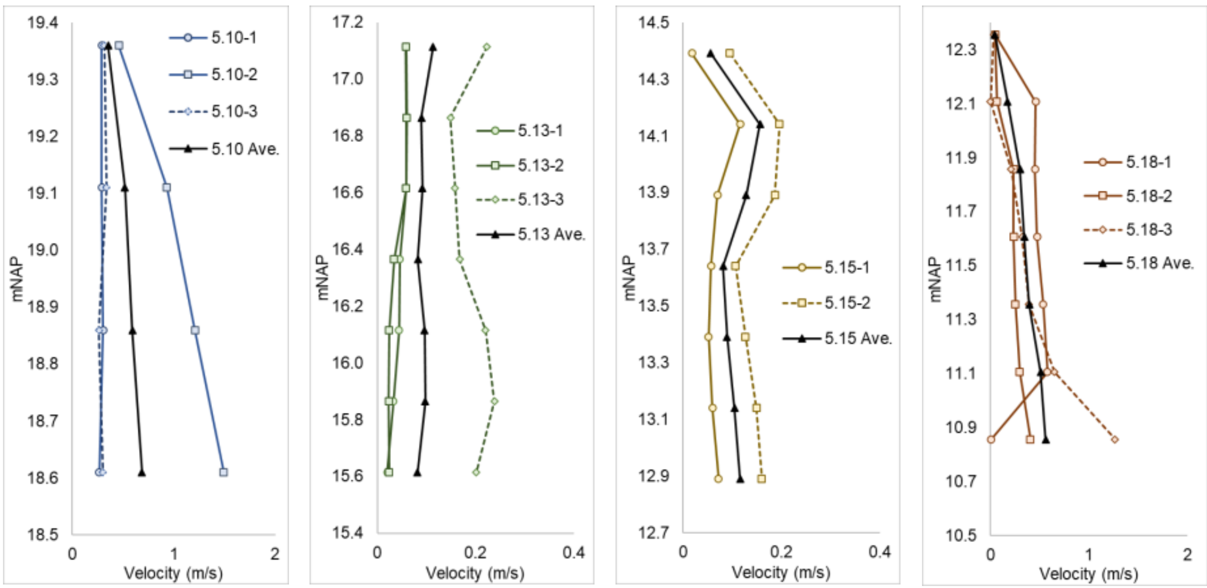

Figure 6. Nest 5 test data showing measured Darcy velocity with recirculation operational (solid lines) and when switched off (dashed lines). Average flows from all tests are also given.

5. CONCLUSIONS

22 SBDT were carried out in 13 piezometers at the de Kragge II landfill, Netherlands which utilises a leachate recirculation system. Tests were carried out at depths between 8.4 and 18.1 m below ground level. Tests were repeated in a number of the piezometers to demonstrate repeatability. Calculated Darcy flow velocities ranged between 0.01 and 1.02 m/day, with the highest velocities measured in the deepest piezometers.

Four tests were carried out in one nest of piezometers with the leachate recirculation system switched off for two days prior to and during the SBDT. Although flows were somewhat higher in two of the piezometers, it was not possible to definitively conclude whether the infiltration of leachate significantly influences flow.

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