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Hydraulic performance assessment of passive coal mine water treatment systems in the UK

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ABSTRACT

Hydraulic performance assessment of passive treatment systems has been conducted for UK's Coal Authority mine water treatment systems. The study aims to improve the understanding of the hydraulic factors that govern contaminant behaviour, such that future design of treatment systems is able to optimise treatment efficiency and make performance more predictable, and improve performance over the long-term. Assessment of the hydraulic behaviour (i.e. residence time and flow pattern) of the treatment systems was accomplished by means of tracer tests. The tracer tests were undertaken at eight UK Coal Authority mine water treatment systems (lagoons and wetlands) within Northern England (main study areas) and part of southern Scotland. A modelling approach using a tanks-in-series (TIS) model was adopted to precisely analyse and characterise the residence time distributions (RTDs), in an effort to account for the different flow patterns across the treatment systems. Generally, lagoon RTDs are characterised by a greater flow dispersion compared to wetlands (i.e. higher dispersion number, D and lower number of TIS, n). Consequently, the hydraulic efficiency, e_λ for lagoons is much lower than wetlands (mean of 0.20 for lagoons compared to 0.66 for wetlands). Implications for design and maintenance of mine water treatment systems are discussed.

1. Introduction

Hydraulic performance in passive treatment systems is often associated with the hydraulic residence time within the system (e.g. Martinez and Wise, 2003; Lin et al., 2003; Kjellin et al., 2007). The time a fraction of water spends in a system may reflect the patterns of water movement across the system and the degree of treatment received of polluted waters (Thackston et al., 1987). The relative importance of residence time as a measure of hydraulic performance of passive treatment systems, in particular within wetland-type treatment systems, has been discussed in many studies (e.g. Thackston et al., 1987; Kadlec, 1994; Werner and Kadlec, 1996; Martinez and Wise, 2003; Persson et al., 1999; Goulet et al., 2001). However, detailed investigations on the actual residence time to reflect the hydraulic performance of the treatment systems have not been widely explored in the application of passive treatment of coal mine water in the UK.

Current design practice for aerobic wetlands treating net-alkaline mine waters in UK applications are based on the zero-order kinetics for pollutant removal i.e. the commonly used area-adjusted removal as recommended by Hedin et al. (1994). Lagoons are designed to allow nominal 48 h of retention time. A knowledge that iron removal under aerobic conditions follows first-order kinetics model for pollutant removal has been the basis for a recommended alternative method for the design of such systems (e.g. first-order removal model by Tarutis et al., 1999). Both of these approaches are based on the plug-flow assumption, which is not the case in real systems. An increasing knowledge of the hydraulic behaviour (e.g. flow pattern across a treatment system) requires a better understanding of the hydraulic factors relating to treatment system performance. Such an assessment has not been widely explored in UK mine water treatment systems. The extent to which actual systems deviate from an ideal flow pattern is the subject of study here. Such an investigation is particularly of interest to better understand the impacts of the residence time (and hence the flow pattern) have on the hydraulic performance of the system, which potentially has an effect on pollutant removal. This is compounded by first-order removal kinetics of some contaminants e.g. iron; removal is not only dependent on the chemical factors i.e. concentration, pH, dissolved oxygen but also the time it takes to attenuate the pollutant (e.g. Jarvis and Younger, 2001; Goulet et al., 2001). Understanding the actual flow pattern rather than assuming plug-flow, which is rarely the case in actual systems is a key

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objective of this investigation. This is somewhat related to recent work by Sapsford and Watson (2011), in which a pH-dependent first-order removal rate constant have been developed for typical mine water conditions in the UK, and are found to be in the range of $0.04\,d^{-1}$ (pH 6)–$372\,d^{-1}$ (pH 8) for first-order oxidation rate constant, and $12.5\,d^{-1}$ for first-order sedimentation rate constant. This retention time-based approach seems to be appropriate for it takes account of the chemical characteristics of mine water and is based on the first-order kinetics for iron removal.

Certainly, the primary pollutant of concern in UK coal mine water discharges is iron. Settlement lagoons and aerobic wetlands have been regarded as 'proven technology' for passive treatment of net-alkaline, ferruginous mine waters within the UK applications of passive treatment (PIRAMID Consortium, 2003). However, one of the limitations of the current design practice for these passive mine water treatment systems are that hydraulic factors are not being accounted for in the design of such systems. This has significantly led to limited understanding of the hydraulic characteristics of the treatment systems which, together with the knowledge of the geochemical processes governing pollutant removal, are central in the assessment of the overall treatment system performance. With regard to the variation in current treatment systems performance i.e. iron removal (e.g. Jarvis and Younger, 1999, 2001; Younger, 2000; Younger et al., 2004; Kruse et al., 2007; Johnston et al., 2007) and hydraulic residence time (e.g. Kruse et al., 2009), this study aims to investigate the hydraulic factors influencing performance of passive mine water treatment systems, specifically in settlement lagoons and aerobic wetlands.

2. Materials and methods

2.1. Site description

The study was undertaken at eight UK Coal Authority mine water treatment systems within Northern England (main study areas) and part of southern Scotland, which consist of mine water treatment wetlands and settlement lagoons (Fig. 1). The systems were designed to treat net-alkaline (i.e. alkalinity > acidity), ferruginous mine water with design flow capacity ranging between 10 and 88 L/s and influent iron concentration from 6 mg/L to 60 mg/L. Irrespective of lagoon or wetland, these include a range of relatively small to very large systems, of between 600 and 11,400 m^2 treatment area. In typical applications of passive treatment within the UK coal mine water treatment systems, settlement lagoon serve as a pre-treatment unit preceded by an aeration cascade, aimed at removing about 50% iron by means of hydrolysis and settlement of ferric hydroxides, prior to final polishing in wetland system (or a series of wetlands) (Younger et al., 2002).

2.2. Field sampling and measurements

2.2.1. Tracer test implementation

The hydraulic factors influencing treatment performance were assessed by means of conducting tracer tests to experimentally determine the actual residence time within the treatment systems. Background tracer concentrations were pre-determined from samples taken prior to the tracer experiments (Lin et al., 2003; Wolkersdorfer et al., 2005) so that any changes due to tracer addition could be detected and that the actual mass recovered could be precisely determined. The method employed for the tracer test was the slug tracer injection, where a known amount of tracer was injected into the inlet of the treatment system and complete mixing with the flow was assumed (Kilpatrick and Cobb, 1985). To ensure maximum mixing, the tracers were dissolved with the mine water and were poured directly into the turbulent zone at the inlet discharge point. Throughout the full series of tracer experiments three types of tracer have been employed; sodium bromide, Na-fluorescein and sodium chloride. Dual-tracer tests were conducted whenever possible for verifying the results (Kusin et al., 2009).

Samples of mine water were automatically collected by Aquamatic Auto Cell P2 Autosamplers equipped with 24 × 1 L polyethylene bottlers at specific time intervals (estimated based on the system nominal residence time). The tracer test typically takes between 24 and 72 h, depending on the system nominal residence time. On-site measurement for bromide was not available, but other tracers i.e. Na-fluorescein and NaCl were measured in the field. Outlet autosamplers were installed for capturing the recovered bromide tracer. The bromide was analysed in the laboratory using a calibrated Dionex 100 Ion Chromatograph. Na-fluorescein was continuously measured in the field using a calibrated Seapoint fluorimeter (detection limit 0.2 μg/L) which was set up to take the concentration readings at 5 min intervals. Alternatively, the Na-fluorescein was also analysed in the laboratory using a Varian Cary Eclipse Fluorescence Spectrometer whenever on-site measurement was not possible and/or for verifying the results. The sodium chloride tracer was measured as electrical conductivity which was recorded using an Eijelkamp CTD Diver at 5 min intervals.

2.3. Flow measurements

The flow rates during the tracer tests were continuously monitored at the outlet of the systems using an Eijelkamp CTD Diver and a BaroDiver for atmospheric pressure correction. The water heads were ascertained using water levels data recorded by the Eijelkamp CTD Diver. The BaroDiver semi-continuously measured the water and atmospheric pressure, temperature and conductivity at a point behind the 90° V-notch weir where the water was not disturbed by the sharp notch. Velocity–area method was used whenever other methods were not possible e.g. in an open channel at the inlet of treatment system. The water depth was measured using a graduated pole and a flow impeller was set at 0.6 of the depth measured downstream from the surface (Brassington, 2007; Shaw et al., 2011). The velocity was measured using a Valeport Model 801 Electromagnetic Flow Meter suspended in the water pointing in an upstream direction. Measured flow rates ware taken within 5–10% precision (PIRAMID Consortium, 2003; Brassington, 2007). For precision, timely data of the measured flow rates were ascertained (i.e. 5, 10 and 15 min intervals) and are used to compare between dynamic (timely data) and average flow using moment analysis. This enables precise measurement and analysis to be made as a means for demonstrating that the flow rates were consistent throughout the duration of the tracer tests.

2.3.1. Tracer flow-pattern modelling

A modelling approach was adopted as a means for comparing the actual tracer residence time distribution (RTD) responses during the tracer tests with the expected theoretical response, to assess system performance under current design. Modelling of the tracer RTD curves was applied to determine the main RTD characteristics (i.e. mean residence time, variance (spread of tracer) and mode (peak of tracer)) which will then lead to determination of treatment system performance metrics i.e. system volumetric efficiency (e_v), RTD efficiency (e_{RTD}) and the overall hydraulic efficiency (e_λ).

The tracer RTDs were modelled based on a tanks-in-series (TIS) model, which is believed to represent a good approximation of most wetland conditions and/or systems similar to it (Kadlec and Wallace, 2009). The model is simple and can be used with any kinetics (Levenspiel, 1972), and is a widely applied tool for treatment

Fig. 1. Locations of the coal mine water treatment systems studied (not to scale).

wetlands and ponds (Kadlec and Knight, 1996). The TIS model lies between the two commonly known ideal systems (i.e. plug-flow and completely-mixed), neither of which appear to be the case in real wetland conditions (Kadlec and Wallace, 2009). Based on the TIS model, the number of tanks-in-series, n, for tracer RTDs was determined. This represents the ideal n TIS for RTD as in non-ideal systems (Levenspiel, 1972). The n TIS essentially indicates the spread of tracer residence time from the mean, thus showing how the flow would deviate from ideal plug-flow. Analysis of system hydraulic characteristics using the TIS model was obtained from three different approaches, whichever fits the data well (i) TIS from moment, (ii) TIS from least squares errors and (iii) delayed TIS from least squares (Kadlec and Wallace, 2009).

The TIS fit for residence time distribution is represented by the gamma probability density function (Levenspiel, 1972; Kadlec and Wallace, 2009):

$$E(t) = g(t) = \frac{1}{\tau_i^n \Gamma(n)} t^{n-1} \ \exp\left(-\frac{t}{\tau_i}\right) \quad (1)$$

where $g(t)$ = gamma distribution for residence time (d^{-1}); $\Gamma(n)$ = gamma function of n, $= \int_0^\infty t^{N-1} \exp(-t)dt$, to allow n being accounted as a non-integer variable, or $(n-1)!$, if n is an integer (d^{-1}); n = number of tanks-in-series (unitless); t = time (d); τ_i = mean residence time in one tank (d). Accordingly, the resulting parameters from the gamma distribution function for tracer RTD were computed from the following:

(i) mean residence time for the whole system, mean, $\tau_m = n\tau_i$
(ii) spread of tracer from the mean, variance, $\sigma_\theta^2 = n\tau_i^2$
(iii) time for the peak tracer, mode, $\tau_p = (n-1)\tau_i$

Following Eq. (1), if the n and τ_i are straightforwardly taken from the values calculated from moment analysis (see Kusin et al. (2010) for details), the resulting parameters are termed as TIS model from moment (Kadlec and Wallace, 2009). Conversely, a more robust approach was used to minimise the summation of squared errors between the TIS model and the observed data and is termed TIS from least squares (LSQ) method. Whenever there is delay between tracer injection and first tracer detection, delayed TIS model from LSQ was used (Kadlec et al., 1993; Chazarenc et al., 2003; Kadlec and Wallace, 2009).

$$g(t)=\frac{1}{\tau_i(n-1)!}\left(\frac{t-t_D}{\tau_i}\right)^{n-1}\exp\left(-\frac{t-t_D}{\tau_i}\right) \quad (2)$$

where t_D = delay time (d); τ_i = tracer residence time in one tank (d).

Results from a TIS model can in fact be compared to those of a plug-flow with dispersion (PFD) model (another commonly applied model for tracer flow studies). According to the PFD model, system dispersion number, D, for systems with relatively large extent of dispersion ($D > 0.01$), was calculated according to the appropriate boundary conditions as follows (i.e. for closed-closed system (Levenspiel, 1999)):

$$\sigma_\theta^2 = 2D - 2D^2(1-e^{-1/D}) \quad (3)$$

where σ_θ^2 = system dimensionless variance $=\sigma^2/t_m^2$ (unitless).

Hydraulic efficiency (e_λ) for the treatment systems was determined using Eq. (4) (after Persson et al., 1999), reflecting both the fractions of water involved in the flow-through and the mixing characteristics of water movement characterised by the behaviour of system dispersion.

$$e_\lambda = e_v e_{\mathrm{RTD}} \quad (4)$$

where $e_v=(t_m/t_{an})=(V_{\mathrm{eff}}/V)$; t_m = actual mean residence time (d); t_{an} = actual nominal residence time (d); V_{eff} = system effective volume; (m^3) V = system nominal volume (m^3); Q = flow rate (m^3/d), and e_{RTD} = residence time distribution efficiency = $(1-\sigma_\theta^2)$; σ_θ^2 = system dimensionless variance (unitless). Note that the term $(1-\sigma_\theta^2)$ is an equivalent form of $(1-1/n)$ (Kadlec and Wallace, 2009). In the case of a TIS system, the τ_{an} was determined as the following (Kadlec and Wallace, 2009):

$$\tau_{an}=\tau_{in}\left(\frac{1}{N}\sum_{j=1}^{N}\left(\frac{1}{1-(\alpha j/N)}\right)\right) \quad (5)$$

where α = water loss fraction, $\alpha = 1 - R = 1 - Q_o/Q_i$ (dimensionless); N = total number of tanks; j = tank number counter i.e. $j = 1, 2, \ldots, N$ (unitless). Computation of this actual nominal residence time (Eq. (5)) gives the advantages of accurately determine the fraction of water involved in the treatment as it takes account of the flow pattern during tracer test, in addition to effect of flow changes between inlet and outlet.

3. Results and discussion

3.1. Wetland systems

3.1.1. RTD characteristics for wetlands

The results of the TIS modelling applied to the actual field data from the studied wetland systems are presented in Fig. 2 (illustrated on the same scale for ease of comparison). Here, only the best TIS fit to actual RTDs are shown, which were used for evaluation of the hydraulic performance characteristics for each of the investigated wetland system. Throughout the discussion, it is worth noting that residence time may differ between systems due to factors such as design configuration (length-to-width ratio), season, age of operation and the role of vegetation (which seems to have an influence on the residence time, albeit the effect of seasons complicates interpretation). The wetlands are planted with *Typha latifolia* and *Phragmites australis* and are varying in their stage of maturity during the tracer tests i.e. from very sparsely-vegetated reeds during first year operation until maturely-developed reeds. The hydraulic inefficiencies of a treatment wetland may be attributed to the non-uniform distribution of vegetation causing non-uniform resistance to flow (Martinez and Wise, 2003). For instance, the reeds' growth can be largely influenced by the seasonal variation (growing or non-growing season) and/or the age of the wetland system itself e.g. year-to-year reed development. However, the qualitative link between vegetation effect and hydraulic performance was only through the observations of the treatment systems monitored, though a quantitative assessment of such a variation would appear to be more useful. Furthermore, there appear to be multiple influences that affect the distribution of residence time in the treatment systems because generally more than one influence may be present, and therefore it is difficult to know exactly the effect of one compared to the other. A summary of the hydraulic characteristics of the wetlands is given in Table 1.

The RTD characteristics and performance metrics from the TIS model for wetland systems are shown in Table 2. An example of tracer TIS modelling to show year-to-year changes in system hydraulic performance of a wetland system at Lambley, Northumberland has been presented in Kusin et al. (2010), which forms part of the discussion here. The tracer RTDs are presented in the form of normalised RTDs (Fig. 2) to illustrate the effects of different experimental conditions, i.e. tracer concentrations, flow rates and duration of test between the treatment systems on tracer RTDs. Clearly there are distinct TIS shapes between the wetland systems, characterised by the number of TIS, n, which ranges from 2 to 5 TIS (Table 2). These correspond (as an inverse relationship) with system dispersion number, D, which ranges from 0.104 to 0.577. In theory, a greater n TIS would indicate a greater

Table 1
Summary of hydraulic characteristics of wetland systems.

Variable	Whittle[a]	Whittle	Allerdean Mill	Strafford	Mousewater	Cuthill
Season of monitoring	Winter	Winter	Spring	Summer	Summer	Summer
Age of system (years)	5.0	8.0	1.5	0.8	5.8	7.0
System area (m^2)	2400	2400	1066	1690	8400	2744
System volume (m^3)	721	458	224	423	2142	571
Length- to-width ratio (unitless)	5.5	5.5	2.1	5.7	3.5	3.5/5.3[e]
Water depth (m)	0.3	0.19	0.21	0.25	0.26	0.21
Flow (L/s)	25	23.07	7.91	14.04	31.9	10.61
Tracer recovery (%)[b]	60.3[c]	82.56[c]	67.40[c]	103.57[d]	77.98[c]	67.98[c]
		113.64[d]	76.58[d]		94.33[d]	71.99[d]

[a] Data from Kruse et al. (2007).
[b] Calculated as the percentage of total mass recovered from the amount of tracer added.
[c] Denotes bromide.
[d] Denotes Na-fluorescein.
[e] L-shaped wetland, L:W ratio of first cell/L:W ratio for second and third cell.

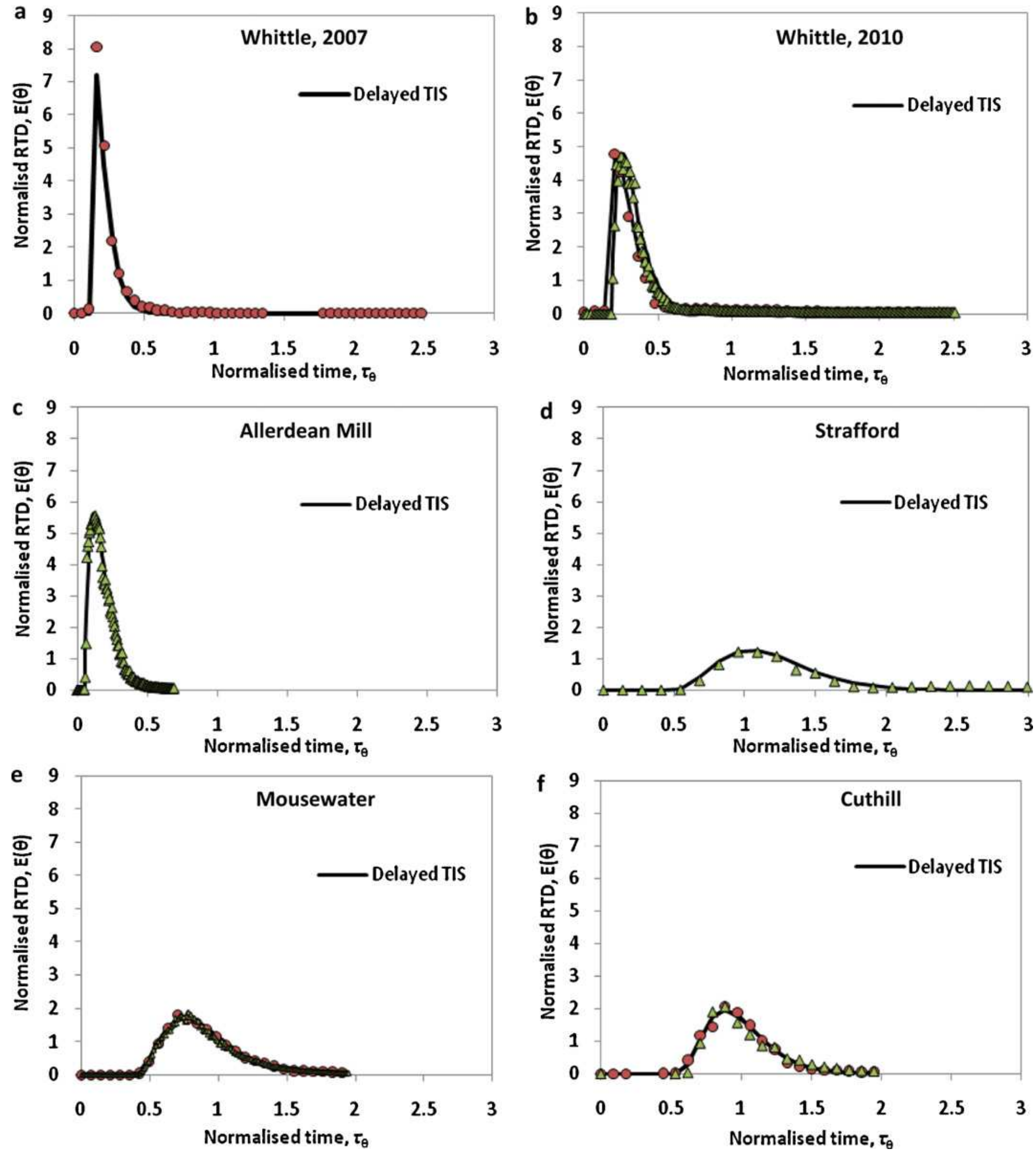

Fig. 2. Normalised TIS model fitted to tracer RTD data for wetland systems plotted on the same scale for comparison. Red circles denote bromide tracer, green triangle denote Na-fluorescein tracer. TIS are the delayed TIS model from least squares, which represent the best fit to actual RTD data.

Table 2
RTD characteristics and performance metrics of wetland systems from the TIS model.

Treatment site	Nominal residence time, τ_{an} (day)	Actual mean residence time, τ_m (day)	Mode, τ_p (day)	Dimensionless variance, σ_θ^2	Dispersion number, D	No. of TIS, n	RTD efficiency, e_{RTD}	Volumetric efficiency, e_v	Hydraulic efficiency, e_λ
Whittle	[a]0.258	0.053	0.095	0.606	0.577	1.65	0.394	0.204	0.080
Whittle	[a]0.237	0.088	0.064	0.511	0.407	1.96	0.489	0.370	0.181
	[b]0.240	0.102	0.078	0.506	0.400	1.98	0.494	0.424	0.209
Allerdean Mill	[b]0.317	0.107	0.069	0.507	0.401	1.97	0.493	0.338	0.167
Strafford	[b]0.225	0.362	0.318	0.187	0.104	5.4	0.813	1.605	1.305
Mousewater	[a]0.742	0.698	0.320	0.238	0.138	4.2	0.762	0.941	0.718
	[b]0.742	0.699	0.596	0.244	0.142	4.1	0.756	0.942	0.712
Cuthill	[a]0.771	0.468	0.290	0.240	0.139	4.2	0.760	0.607	0.461
	[b]0.771	0.468	0.289	0.247	0.144	4.1	0.753	0.607	0.457

All parameters are unitless unless otherwise stated (units in brackets).
[a] Tracer test using sodium bromide (NaBr).
[b] Tracer test using sodium-fluorescein (Na-fluorescein).

Table 3
Summary of hydraulic characteristics of lagoon systems.

Variable	Acomb[a] East	Acomb[a] West	Acomb East	Acomb West	Whittle[a]	Whittle	Bates right L1	Bates right L2	Bates left L1	Bates left L2	Strafford	Allerdean Mill	Mousewater	Cuthill
Season of monitoring	Winter	Winter	Winter	Summer	Winter	Spring	Summer	Summer	Summer	Summer	Summer	Spring	Summer	Summer
Age of system (years)	5	5	7	7	5	8	5.8	5.8	5.8	5.8	0.8	1.5	5.8	7.1
System area (m^2)	375	375	375	375	900	900	2850	2850	2850	2850	850	883	3036	726
System volume (m^3)	1050	1050	1050	1050	1305	1305	6242	6242	6242	6242	2040	1325	6527	1089
Length-to-width ratio (unitless)	1.5	1.5	1.5	1.5	3.0	3.0	2.0	2.0	2.0	2.0	4.5	4.7	1.2	3.2
Water depth (m)	2.8	2.8	2.8	2.8	1.65	1.65	2.2	2.2	2.2	2.2	2.4	1.6	2.2	2.0
Flow (L/s)	6.25	5.8	6.25	6.5	25	25	78.69	78.69	58.55	58.55	14.04	9.67	36.72	10.72
Tracer recovery (%)[b]	82.14[c]	82.14[c]	61.75[c]	77.74[c]	82.05[c]	95.97[c] 119.29[d]	130.62[d]	155.24[d]	69.19[d]	65.78[d]	69.69[d]	76.58[d]	84.72[c] 76.00[d]	67.66[c] 77.68[d]

[a] Data from Kruse et al. (2007).
[b] Calculated as the percentage of total mass recovered from the amount of tracer added.
[c] Denotes bromide.
[d] Denotes Na-fluorescein.

amount of complete-mixing, and the more it approximates an ideal plug-flow system (Persson et al., 1999). Therefore there should ideally be a small degree of dispersion resulted from this flow pattern.

Individual assessment of the wetland systems using n and D showed that the Whittle wetland (during 2007 tracer test) had the highest dispersion number, D of 0.577 (corresponding to $n = 1.7$), while the Strafford wetland indicated the lowest system dispersion number of 0.104 (corresponding to $n = 5.4$). The poor flow mixing effect and variation from the ideal flow pattern at Whittle wetland may be attributable to the apparent short-circuiting across the system following its fifth year operation. Field observation showed that the wetland is a mature system with well-established reeds and a substantial build up of dead plant material (Kruse et al., 2007). This can lead to the development of channels through the wetland, and thus a rapid transit of flow across the system, but also there is portion of water that remained in the system longer (i.e. the long RTD tail). In contrast, the Strafford system is only in its first year operation, with notably very sparse reed growth, but the flow was well-distributed within the system (i.e. a low dispersion number). Note that this is in contrast to Lambley wetland during its first year operation, during which the flow was very dispersed (Kusin et al., 2010). Thus, it appears that whilst reeds may result in short-circuiting, neither are they necessarily essential to ensuring good hydraulic performance.

The greater flow mixing effect seen at Strafford wetland was presumably due to the presence of deep zones and islands created as planting blocks near the inlet and in the middle of the wetland system. This results in a greater mixing and redistribution of flow, and hence a more well-distributed flow across the system. Persson (2000), in a simulation of surface flow wetland shapes of a Scandinavian stormwater treatment wetland, found that an island placed in front of the inlet improves the hydraulic performance with respect to effective volume and degree of mixing. Thus, the flow in the Strafford wetland was found to be much less dispersed than that of Whittle wetland, despite having a fairly similar system length to width ratio.

3.1.2. Hydraulic performance metrics for wetland systems

The different TIS shapes resulted in the mean residence time for the wetlands ranging from 0.05 to 0.7 days. However, direct comparison of these mean residence times will not necessarily imply efficiency of the systems e.g. longer residence time does not necessarily mean an efficient system, because such a system would probably have a long nominal residence time (i.e. due to large system volume or a relatively low flow per treatment volume). Instead, an appropriate measure to compare between these system residence times is by assessing the metric of system volumetric efficiency, e_v which is a measure of mean relative to nominal residence time. Generally, the wetland systems have a range of e_v of 0.204–1.605. Again, the lowest was found in the Whittle wetland during the 2007 tracer test, whilst the longest was found in the Strafford wetland in the 2009 tracer test. These, apparently correspond with the systems dispersion characteristics i.e. the n TIS and D, as discussed earlier. The high e_v in the Strafford wetland (mean residence time 60% higher than nominal residence time) shows that the wetland system was capable of retaining a large proportion of water volume during flow passage through the system, whilst enhancing a more distributed flow across the system and thus encourage a long water travel time. Sherman et al. (2009) reported a mean residence time of 50% greater than the nominal residence time in a free water surface wetland treating effluent from a mine's wastewater treatment plant in Australia. However, the reason of this larger mean relative to nominal residence time was not reported.

The wetland systems' e_{RTD} (which is a hydraulic performance metric of the dispersive flow behaviour, i.e. the extent of a system deviation from an ideal flow pattern) ranges from as low as 0.394 to as high as 0.813. The low e_{RTD}, e.g. 0.394 in Whittle wetland, would possibly mean that the flow within the system appears to be dominated by greatly dispersed fractions of flow, given by the large system dimensionless variance of 0.606 (60% of water was attributed to a deviation from its mean). Similarly, high e_{RTD} (e.g. 0.813 in the Strafford wetland) may be attributable to a lower extent of flow dispersion as given by the relatively low dimensionless variance of 0.187 (only about 18% of the flow was attributed to a deviation from its mean).

As noted in Table 2, the overall hydraulic efficiency, e_λ ranges between 0.08 and 1.305. Again the lowest and highest values were found from the Whittle and Strafford wetland respectively. This shows that the TIS model results in consistent hydraulic performance characteristics for these treatment wetlands, e.g. greater e_λ at Strafford wetland resulted from its greater e_v and e_{RTD}, strengthening the conclusion that these inter-related hydraulic parameters (i.e. n TIS, D, σ_θ^2, e_v, and e_{RTD}) are very important for determining the overall system hydraulic efficiency. Of the two parameters e_v and e_{RTD}, the first has a greater impact on the system hydraulic efficiency, e_λ (e_v; $R^2 = 0.998$, e_{RTD}; $R^2 = 0.861$ both $p < 0.05$). This would mean that a hydraulically more efficient system would be achieved if a longer mean relative to nominal residence time prevails, thus providing greater potential for pollutant attenuation and degree of treatment received. Discussion on a site-to-site basis of the other wetland systems is not included here because the reason for the differences in system dispersion characteristics is generally due to the presence of flow short-circuting effects, creating a significant deviation from ideal flow patterns, as characterised by the n TIS and D.

3.2. Lagoon systems

3.2.1. RTD characteristics for lagoon systems

As with the wetland systems, the TIS model fitted to actual tracer RTD data for lagoon systems are shown in the normalised forms and are illustrated on the same scale for comparison (Fig. 3). As before, only the best TIS fit are displayed for each RTD from the treatment lagoons. A summary of lagoon hydraulic characteristics is given in Table 3. The lagoons show a relatively greater extent of system dispersion compared to wetland systems as characterised by the greater dispersion number, D, and the lower number of TIS,

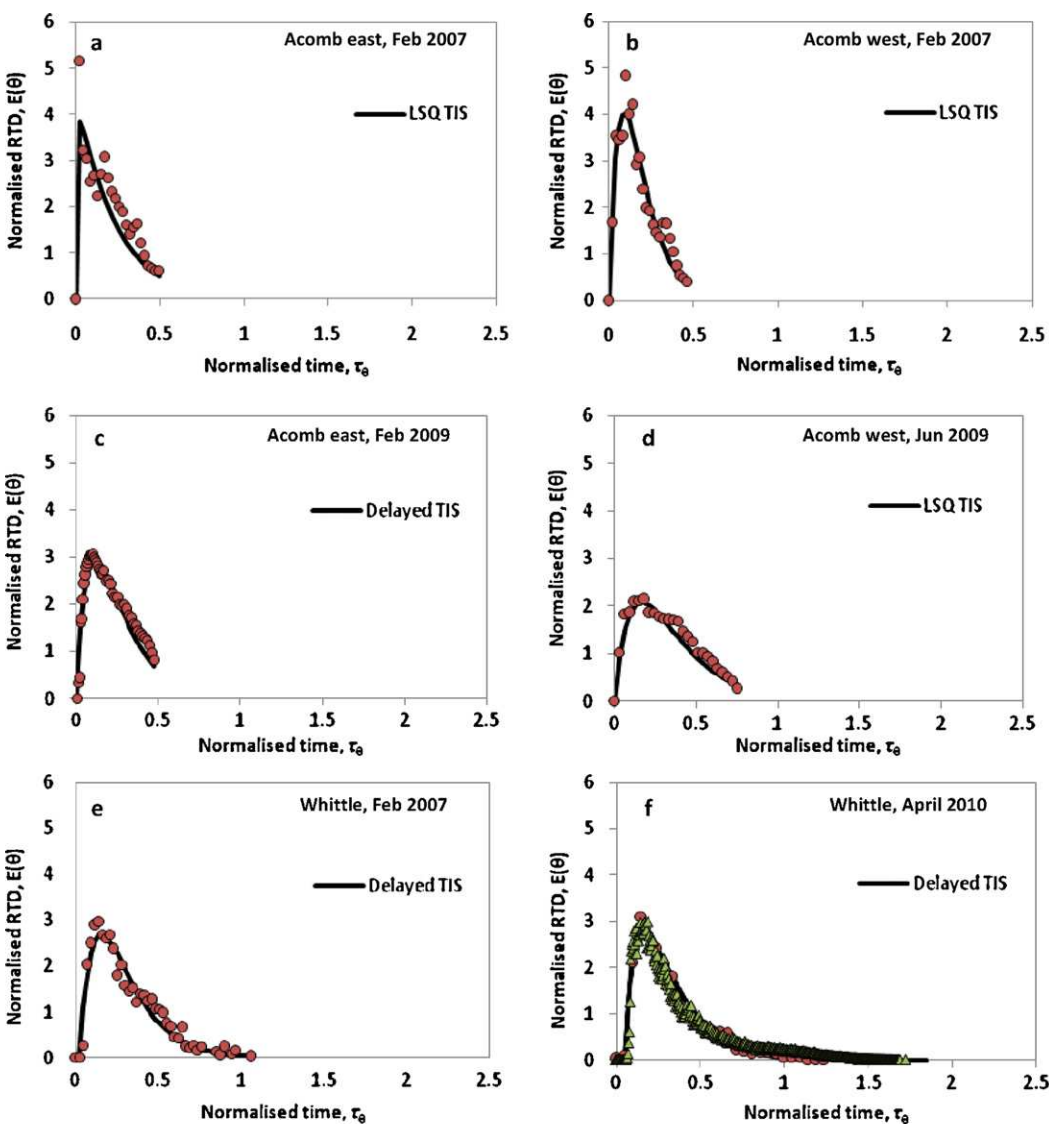

Fig. 3. Normalised TIS model fitted to tracer RTD data for comparison in wetland systems. Red circles denote bromide tracer, green triangle denote Na-fluorescein tracer.

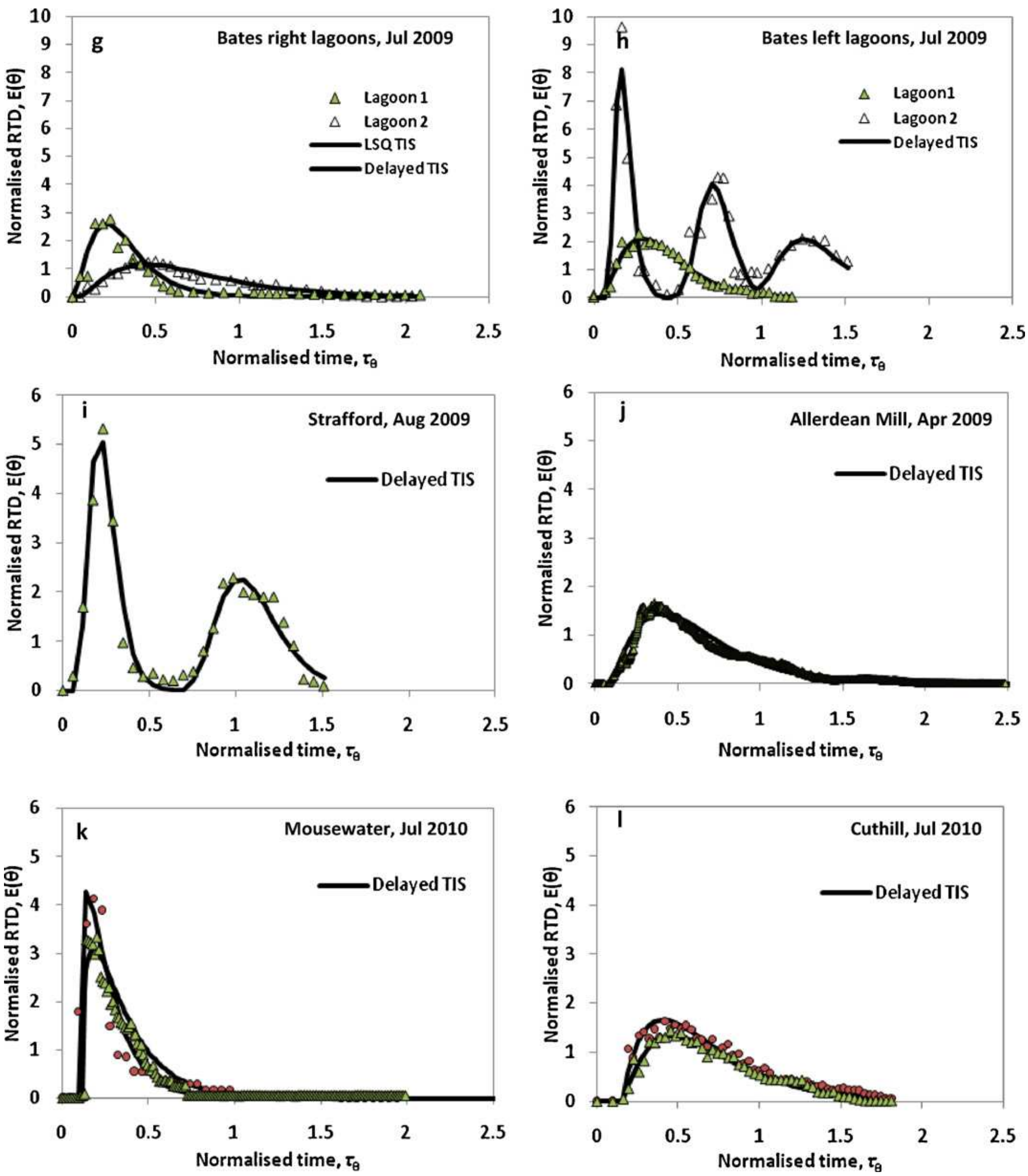

Fig. 3. (*Continued*)

n. Overall, the lagoon systems have D in the range 0.173–0.976, which corresponds (as an inverse relation) with n TIS of between 2 and 3 TIS (Table 4). Note that the resulting parameters from the parallel TIS model were not accounted for in this comparison. Two-path flow pattern is apparently seen from Bates (left) lagoon 2 and Strafford lagoon; when a parallel path model is warranted, this suggests that there is something seriously wrong with the system that needs improvement (Kadlec and Wallace, 2009). Therefore, for consistency these systems were excluded for comparison of system hydraulic characteristics resulting from the TIS model. The highest D of 0.976 (the system with greatest dispersion from ideal plug-flow) was observed in the Acomb east lagoon (2007), which corresponds with $n = 1.4$, while the lowest D of 0.173 was found in Bates (first lagoon on the left side) which corresponds with $n = 2.95$. These reflect the apparent short-circuiting of flow as seen from the very sharp and quick peak of RTD from the Acomb east lagoon.

3.2.2. Hydraulic performance metrics for lagoon systems

The resulting mean residence times for the lagoons were found to be in the range of 0.159–0.952 days. The system volumetric efficiency, e_v, which is used to compare between the mean relative to nominal residence times ranged from 0.07 to 0.745. The lowest e_v was found in the Mousewater lagoon, while the highest was found in the Bates lagoon (the second lagoon to the right side). The comparatively low e_v found in the Mousewater lagoon was presumably due to a relatively low mean residence time compared to the nominal residence time coupled with the large volume of the system. Such a skewed RTD, with $n = 1.4$, suggests that the effect of flow short-circuiting was also pronounced within this system and hence a large fraction of water was rapidly transmitted across the surface of the lagoon.

The greater dispersion characteristics of the lagoons have resulted in a relatively lower system RTD efficiency, e_{RTD} (0.265–0.714), compared to wetland systems. These correspond (as an inverse relationship) to system dimensionless variance, σ_θ^2, of 0.286–0.735. The greatest estimated dispersion in a lagoon system, $\sigma_\theta^2 = 0.735$ and $e_{\mathrm{RTD}} = 0.265$, was found in the Acomb east lagoon during the 2007 tracer test. The reason for this was likely due to the fact that this lagoon system received a very high influent iron concentration (approximately 40 mg/L). This was coupled

Table 4
RTD characteristics and performance metrics of lagoon systems from the TIS model.

Treatment site	Nominal residence time, τ_{an} (day)	Actual mean residence time, τ_m (day)	Mode, τ_p (day)	Dimensionless variance σ_θ^2	Dispersion number, D	No. of TIS, n	RTD efficiency, e_{RTD}	Volumetric efficiency, e_v	Hydraulic efficiency, e_λ
Acomb (E) 2007	[b]1.944	0.433	0.027	0.735	0.976	1.36	0.265	0.285	0.075
Acomb (W) 2007	[b]2.082	0.382	0.192	0.498	0.388	2.00	0.502	0.183	0.092
Acomb(E) 2009	[b]1.981	0.505	0.235	0.549	0.467	1.82	0.451	0.255	0.115
Acomb (W) 2009	[b]2.082	0.734	0.347	0.528	0.433	1.90	0.472	0.353	0.167
Whittle 2007	[b]0.605	0.179	0.095	0.510	0.405	1.96	0.490	0.295	0.145
Whittle 2009	[a]0.642	0.269	0.138	0.575	0.514	1.74	0.425	0.419	0.178
	[b]0.642	0.278	0.145	0.570	0.504	1.75	0.430	0.433	0.186
Bates (R) 1	[a]0.918	0.281	0.184	0.346	0.222	2.89	0.654	0.306	0.200
Bates (R) 2	[a]0.881	0.656	0.404	0.411	0.284	2.43	0.589	0.745	0.439
Bates (L) 1	[a]1.237	0.516	0.355	0.286	0.173	2.95	0.714	0.417	0.297
Bates (L) 2	[ac]*0.108*	*0.227*	*0.208*	*0.208*	*0.118*	*4.8*	*0.792*	*2.098*	*1.661*
	[c]*0.409*	*0.448*	*0.496*	*0.079*	*0.041*	*12.6*	*0.921*	*1.095*	*1.008*
	[c]*0.547*	*0.545*	*0.504*	*0.228*	*0.131*	*4.4*	*0.772*	*0.997*	*0.769*
	[d]*1.234*	*1.144*	*0.953*	*0.168*	*0.093*	*5.9*	*0.832*	*0.928*	*0.772*
Strafford	[ac]*0.891*	*0.352*	*0.216*	*0.195*	*0.109*	*5.1*	*0.805*	*0.395*	*0.318*
	[c]*0.672*	*0.663*	*0.551*	*0.170*	*0.094*	*5.9*	*0.830*	*0.987*	*0.820*
	[d]*1.564*	*0.795*	*0.322*	*0.594*	*0.552*	*1.7*	*0.406*	*0.508*	*0.206*
Allerdean Mill	[a]1.554	0.952	0.610	0.385	0.257	2.6	0.615	0.612	0.377
Mousewater	[a]2.285	0.159	0.083	0.730	0.973	1.4	0.270	0.070	0.019
	[b]2.285	0.211	0.108	0.727	0.959	1.4	0.273	0.093	0.025
Cuthill	[a]1.185	0.552	0.296	0.519	0.419	1.9	0.481	0.466	0.224
	[b]1.185	0.622	0.259	0.465	0.345	2.2	0.535	0.525	0.281

Italicise values are excluded from the comparison of system hydraulic performance (not valid for TIS model comparison). All parameters are unitless unless otherwise stated (units in brackets).

[a] Tracer test using sodium bromide (NaBr).
[b] Tracer test using sodium-fluorescein (Na-fluorescein).
[c] Resulting parameters from parallel TIS model (two-paths modelling approch (Chazarenc et al., 2003)).
[d] Resulting parameters from moment analysis.

with hydrogen peroxide dosing into the lagoon system to provide greater oxidation and hence precipitation of the iron, which ultimately settles as iron hydroxides, and in turn largely accumulate at the bottom of the lagoon. This appeared to favour development of significant preferential flow paths across the surface of the lagoon, which resulted in a very quick, skewed RTD peak of the tracer. This effect was reduced during the 2009 test, following sludge removal in 2008, when a lower dimensionless variance, σ_θ^2 (0.549), and a greater mixing efficiency, e_{RTD} (0.451), was evidenced. The Acomb east lagoon volumetric efficiency was improved during 2009, with more volume being involved during flow-through, and an overall improvement in system hydraulic efficiency, i.e. e_λ from 0.075 to 0.115. Therefore regular ochre sludge removal/dredging is important as to maintain or improve hydraulic performance, and the discussion above quantifies the level of this improvement.

Overall, the lagoon systems have values of e_λ from as low as 0.019 to as high as only 0.439. The lowest value was found at the Mousewater lagoon, while the highest was found in Bates (the second lagoon to the right side). Clearly, this was reflected by the metric of e_v where system volumetric efficiency is of substantial importance to the overall lagoon hydraulic efficiency (e_v; $R^2 = 0.953$, e_{RTD}; $R^2 = 0.849$ both $p < 0.05$).

3.3. Implications for the design and maintenance of mine water treatment systems

General trends (i.e. mean data for all wetlands compared to mean data for all lagoons) showed that efficient treatment performance for iron removal corresponds with greater hydraulic efficiency in wetlands compared to lagoon systems. The greater hydraulic efficiency in wetlands was mainly attributed to a greater volumetric efficiency in the wetland systems, which thus results in a longer relative mean residence time for retention and attenuation of iron. On average, 83.86% of iron load is removed from the treatment wetlands compared to 45.36% in lagoons (Table 5).

For lagoons, residence time is an important variable for the design of such systems. Current observations showed that low actual residence time in the lagoons seems to be an impediment to further improvements in treatment performance. Therefore, performance can be optimised by ensuring a greater volumetric efficiency (hence residence time). Based on the experiences of the tracer tests, this appears to be achievable by providing a large length-to-width ratio system (up to a ratio of 4.7), but also a greater depth (i.e. maximum of 3.0 m), though only if systems are regularly maintained i.e. dredged. Greater L/W was found to significantly correlate to greater depth, although this may imply a coincidence of engineering design rather than any sort of cause and effect. For wetlands, the use of area-adjusted removal rate formula appears to work well for the design of aerobic wetlands, despite the observed concentration-dependence of iron removal processes. This was shown by the hydraulic performance and treatment efficiency of the wetland systems which is far better than lagoon systems.

Nevertheless, first-order removal formula (TIS basis) would appear to be a more appropriate approach to the design of mine water treatment systems since it takes account of the flow pattern effect on pollutant removal processes, in addition to the first-order kinetics (concentration-dependence) of iron removal. For the systems studied, it has been showed that it is appropriate to take account of both the flow pattern effect and first-order kinetics in designing lagoons and wetlands. However, there is no evidence from actual systems to show whether such criteria (residence time and treatment area) estimated from a TIS first-order approach would enhance removal efficiency and hydraulic performance.

Regular sludge removal (yearly), particularly from lagoons, is very important because of rapid depth reduction due to build up of ochre and debris (7–49% effective depth reduction per year). Such ochre accumulation significantly reduces the effective volume of the systems (which is especially pronounced in lagoons), whereby apparent streaming effects result in very short residence times in the systems, and hence reduce the time available for

Table 5
Summary iron removal for coal mine water treatment systems.

	Lagoon			Wetland		
	Mean (S.E.)	Median	Min/Max	Mean (S.E.)	Median	Min/Max
Fe in (mg/L)	18.27 (2.79)	18.12	4.36/34.1	7.59 (1.56)	6.70	3.01/20.8
Fe out (mg/L)	8.85 (1.46)	6.45	1.30/24.88	0.88 (0.12)	1.01	0.11/1.7
Flow in (L/s)	33.30 (9.01)	17.95	5.85/83.32	34.40 (7.96)	24.04	7.91/84.83
[a]Fe loading rate (kg/d)	35.39 (7.66)	18.41	3.77/128.95	20.58 (4.71)	14.82	2.06/47.92
[b]Fe removal efficiency (%)	44.50 (5.42)	49.36	13.88/85.19	85.42 (2.71)	85.11	57.66/98.60
[c]Fe load removal (%)	45.36 (5.11)	46.38	13.88/85.19	83.86 (3.55)	85.45	47.69/99.01

Data present mean data and standard error of mean (S.E.) in parenthesis, median, and minimum (min) and maximum (max) values of performance data for lagoons and wetlands (n = 14 for lagoon and 10 for wetland).

[a] Calculated as $Q \times$ Inf Fe; where Q = flow rate (L/s).

[b] Calculated as (Inf Fe − Eff Fe)/Inf Fe × 100.

[c] Calculated as Q(Inf Fe − Eff Fe)/Q*Inf Fe × 100.

pollutant attenuation. In wetlands, thinning of reeds is also important to maintain effective flow movement so as to prevent short-circuiting effects, although at the same time the presence of reeds appears to provide significant potential for physical filtering of precipitated iron hydroxide and adsorption onto living and dead plant material. Thus, thinning of reeds is recommended whenever apparent channelisation would otherwise dominate the flow pattern, and therefore limits the capacity for adsorption and settlement of precipitated iron hydroxide.

4. Conclusions

The TIS modelling approach has been successfully applied to charaterise the flow patterns in UK passive mine water treatment systems. Delayed TIS from least squares method yielded the best fit to actual RTDs, since it took account of the transport delay often encountered in tracer detection. This modelling approach served as the basis for computing the system hydraulic efficiency.

Variations in flow pattern have a significant impact on the residence time distribution across the systems. The tracer flow modelling showed that both lagoons and wetlands were greatly dispersed from an ideal plug-flow pattern ($D > 0.01$). This effect was more pronounced in lagoons. The lagoon systems had dispersion in the range of 0.17–0.97, whereas wetlands had dispersion in the range 0.10–0.58. These correspond (as an inverse relationship) with n in the range 1.4–2.95 in lagoons and between 1.65 and 5.4 in wetlands. Accordingly, lagoons had hydraulic efficiency in the range 0.02–0.44, whilst in wetlands hydraulic efficiency ranged between 0.08 and 1.31 (mean of 0.20 for lagoons compared to 0.66 for wetlands). Clearly, hydraulic efficiency is much lower in lagoon systems. This reflects the more dispersed flow patterns within lagoon systems (i.e. portions of water that exit the system fast and also portions that leave the system more slowly, i.e. the long RTD tails). Additionally, the presence of dead zones with large amount of ochre sludge has significantly reduced much of the lagoon's effective volume (and hence residence time). Regular sludge removal (yearly) is therefore recommended if efficient hydraulic performance is to be maintained, because lagoon depth and volume appeared to rapidly decrease over time due to accumulation of ochre and debris. Providing a large L/W ratio lagoon system with also a greater depth may help improve volumetric efficiency (and hence longer residence time for greater iron attenuation) though only if systems are regularly maintained i.e. desludged.

Regardless of wetlands or lagoons, the flow patterns across the studied systems are characterised by the n TIS of 1.4–5.4, which corresponds to dispersion number, D, of 0.10–0.97. These can be compared to a number of instances from the United States, Australia, Spain and France for which n is between 0.3 and 10.7 for free water surface (FWS) wetlands (Kadlec and Wallace, 2009). Interestingly, none of these FWS systems as summarised by Kadlec and Wallace (2009) receive polluted mine water. The results from this study have successfully situated the coal mine water treatment systems in the UK within the range of typical wetlands (for wastewater, stormwater and agricultural treatment) based on the hydraulic performance characterisation from the TIS model.

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References

Brassington, R., 2007. Field Hydrogeology, 3rd ed. John Wiley & Sons Ltd., England, p. 264.
Chazarenc, F., Merlin, G., Gonthier, Y., 2003. Hydrodynamics of horizontal subsurface flow constructed wetlands. Ecol. Eng. 21, 165–173.
Goulet, R.R., Pick, F.R., Droste, R.L., 2001. Test of first-order removal model for metal retention in a young constructed wetland. Ecol. Eng. 17, 357–371.
Hedin, R.S., Nairn, R.W., Kleinmann, R.L.P., 1994. Passive Treatment of Coal Mine Drainage. United States Department of the Interior, Washington, DC, p. 35 (Bureau of Mines Information Circular 9389).
Jarvis, A.P., Younger, P.L., 1999. Design, construction and performance of a full-scale compost wetland for mine-spoil drainage treatment at Quaking Houses. J. Chart. Inst. Water Environ. Manage. 13, 313–318.
Jarvis, A.P., Younger, P.L., 2001. Passive treatment of ferruginous mine waters using high surface area media. Water Res. 35, 3643–3648.
Johnston, D., Parker, K., Pritchard, J., 2007. Management of abandoned minewater pollution in the United Kingdom. In: Proceedings of the International Mine Water Association Symposium, 27th–31st May, Cagliari, Italy, pp. 209–213.
Kadlec, R.H., 1994. Detention and mixing in free water wetlands. Ecol. Eng. 3, 345–380.
Kadlec, R.H., Bastiaens, W., Urban, D.T., 1993. Hydrological design of free water surface treatment wetlands. In: Moshiri, G.A. (Ed.), Constructed Wetlands for Water Quality Improvement. CRC Press, Boca Raton, FL, pp. 77–86.
Kadlec, R.H., Knight, R.L., 1996. Treatment Wetlands. Lewis Publishers, Boca Raton, FL, p. 893.
Kadlec, R.H., Wallace, S.D., 2009. Treatment Wetlands, 2nd ed. CRC, Press, Boca Raton, FL, p. 1016.
Kilpatrick, F.A., Cobb, E.D., 1985. Measurement of discharge using tracers. Water-Resources Investigation Report. US Geological Survey.
Kjellin, J., Worman, A., Johansson, H., Lindahl, A., 2007. Controlling factors for water residence time and flow patterns in Ekeby Treatment Wetland, Sweden. Adv. Water Res. 30, 838–850.
Kruse, N., Gozzard, E., Jarvis, A., 2007. Determination of hydraulic residence time at coal authority mine water treatment system and their relationship to iron removal. Report to the Coal Authority. University of Newcastle Upon Tyne, UK, p. 21.
Kruse, N., Gozzard, E., Jarvis, A., 2009. Determination of hydraulic residence time in several UK mine water treatment systems and their relationship to iron removal. Mine Water Environ., http://dx.doi.org/10.1007/s10230-009-0068-6.

Kusin, F.M., Kruse, N., Mayes, W.M., Jarvis, A.P., 2009. A comparative study of hydraulic residence time using a multi-tracer approach at a coal mine water treatment wetland, Lambley, Northumberland UK. In: Proceedings of the 8th ICARD, 22–26 June 2009, Skellefteå, Sweden.
Kusin, F.M., Jarvis, A.P., Gandy, C.J., 2010. Hydraulic residence time and iron removal in a wetland receiving ferruginous mine water over a 4 year period from commissioning. Water Sci. Technol. 62 (8), 1937–1946.
Levenspiel, O., 1972. Chemical Reaction Engineering, 1st ed. John Wiley & Sons, New York.
Levenspiel, O., 1999. Chemical Reaction Engineering, 3rd ed. John Wiley & Sons, New York, p. 664.
Martinez, C.J., Wise, W.R., 2003. Hydraulic analysis of Orlando easterly wetland. J. Environ. Eng. 129, 553–559.
Lin, A.Y.C., Debroux, J.F., Cunningham, J.A., Reinhard, M., 2003. Comparison of rhodamine WT and bromide in the determination of hydraulic characteristics of constructed wetlands. Ecol. Eng. 20, 75–88.
Persson, J., 2000. The hydraulic performance of ponds of various layouts. Urban Water 2, 243–250.
Persson, J., Somes, N.L.G., Wong, Y.H.F., 1999. Hydraulics efficiency of constructed wetlands and ponds. Water Sci. Technol. 40 (3), 291–300.
Consortium, PIRAMID., 2003. Engineering guidelines for the passive remediation of acidic and/or metalliferous mine drainage and similar wastewaters. In: European Commission 5th Framework Programme Passive In Situ Remediation of Acid Mine/Industrial Drainage (PIRAMID). University of Newcastle Upon Tyne, UK, p. 166.
Tarutis, W.J., Stark, L.R., Williams, F.M., 1999. Sizing and performance estimation of coal mine drainage wetlands. Ecol. Eng. 12 (3–4), 353–372.
Thackston, E.L., Shields F.D.Jr., Schroeder, P.R., 1987. Residence time distributions of shallow basins. J. Environ. Eng. (ASCE) 113 (6), 1319–1332.
Sapsford, D.J., Watson, I., 2011. A process-orientated design and performance assessment methodology for passive mine water treatment systems. Ecol. Eng. 37 (6), 970–975.
Shaw, E.M., Beven, K.J., Chappell, N.A., Lamb, R., 2011. Hydrology in Practice, 4th ed. Taylor & Francis, London, p. 560.
Sherman, B.S., Trefry, M.G., Davey, P., 2009. Hydraulic characterisation of a constructed wetland used for nitrogen removal via a dual-tracer test. In: Proceedings of the International Mine Water Association Conference, 19th–23rd October, Pretoria, South Africa, pp. 548–556.
Werner, T.M., Kadlec, R.H., 1996. Application of residence time distributions to stormwater treatment systems. Ecol. Eng. 7, 213–234.
Wolkersdorfer, C., Hasche, A., Gobel, J., Younger, P.L., 2005. Tracer test in the Bowden Close passive treatment system (UK)-preliminary results. Wissenschaftliche Mitteilungen 28, 87–92.
Younger, P.L., 2000. The adoption and adaptation of passive treatment technologies for mine waters in the United Kingdom. Mine Water Environ. 19, 84–97.
Younger, P.L., Banwart, S.A., Hedin, R.S., 2002. Mine Water: Hydrology, Pollution Remediation. Kluwer Academic Publishers, Dordrecht, Netherlands, p. 442.
Younger, P.L., Jayaweera, A., Wood, R., 2004. A full-scale reducing and alkalinity-producing system (RAPS) for the passive treatment of acidic, aluminium-rich mine site drainage at Bowden Close, County Durham. In: Proceedings of the CL:AIRE Annual Project Conference, April 2004, London.