Quantifying sediment deposition and the spatial variability of sediment-associated metals in ponds treating urban diffuse pollution

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Abstract Studies examining the accumulation of sediments within retention ponds have tended to provide spatially averaged estimates of deposition rates and have not accounted for the spatial variability of deposition and the potential role that developing morphological and sedimentary structures play in the routing of flow and potential contaminants within retention ponds. Additionally, studies examining the potential metal contamination of retention pond sediments have tended to examine individual cores at metre intervals. In this paper, the utility of ground-penetrating radar to provide a rapid means of delineating present pond-morphology and to also provide an estimate of sediment deposit thickness within retention ponds is explored. The spatial variability of metals in pond sediments at the centimetre-scale is also examined. Results show that metal variability around the inlet is high at the centimetre-scale, and that a degree of dependence exists between direction of flow and metal contamination.

Key words morphology; ponds; retention basins; Scotland; sediment; sediment deposition; spatial variability; SUDS; urban stormwater

INTRODUCTION

The prediction of anthropogenically-enhanced global warming invoking more extreme rainfall conditions and increasing flood risk in Scotland (Black & Burns, 2002), in combination with high rates of urbanisation increasing point-source and diffuse pollution over the next century, dictates a need for long-term sustainable intervention and management of watercourses at a wide range of scales. Retention ponds (or basins), as part of SUDS (Sustainable Urban Drainage Systems) techniques, are promoted for controlling flood risk and diffuse water pollution within urban catchments. However, being a relatively new approach little is known about their long-term sustainability if potentially contaminated sediment is permitted to accumulate. The need for a greater understanding of the long-term operation and maintenance requirements of retention ponds was identified in early studies (Rowney *et al.*, 1986), particularly as sediment accumulation over time leads to a reduction in the capacity of the pond (Färm, 2002) to accommodate and attenuate large runoff volumes. The physical processes controlling deposition rates within individual ponds need to be understood since it has been identified that spatially localised deposition can actively force a readjustment of the hydraulics within the pond, leading to possible short-circuiting of flow and a commensurate reduction in sediment residence time (Fennessy *et al.*, 1994).

Early studies of sedimentation processes within retention ponds concentrated on gaining a broad understanding by examining spatially averaged rates of deposition, which were shown to be highly site specific. Yousef *et al.* (1994) reported a very slow rate of deposition throughout the basin studied (0.00783 m $year^{-1}$), in contrast to Striegl (1987), who reported a maximum deposition rate of 0.02 m $year^{-1}$ in a small urban Chicago lake. Marsalek *et al.*'s (1997) study of the Kingston stormwater pond, Ontario, Canada, showed a similar accumulation rate of 0.02 m $year^{-1}$. This study also confirmed that the nature of deposition was spatially variable throughout the pond. Larger fractions were preferentially deposited near the inlet, with finer fractions being distributed more evenly throughout the pond. Färm (2002) also reported spatial variation in sediment accumulation within a Swedish retention pond 18 months after its construction, with 0.05–0.08 m near the inlet and 0.015 m at the outlet, giving deposition rates of 0.01–0.053 m $year^{-1}$. The gradient demonstrates preferential deposition nearer the inlet—a morphological structure corroborated by other similar studies (e.g. Striegl, 1987; Mesuere & Fish, 1989). These structures should, according to the observations of Marsalek *et al.* (1997), contain coarser sediments around the inlet, with finer sediments nearer the outlet, giving a laterally graded distribution of sediments. It is important to understand the development of these morphologies and their associated sediment distribution as they will also inform the spatial distribution of potentially toxic metals, which tend to be found in greater concentrations around finer sediment grains deposited within the basin (Horowitz, 1991).

Studies examining metal concentrations within retention pond sediment show that concentrations also tend to be highly spatially variable. At the pond scale, metal concentrations in the literature do not appear to follow consistent spatial patterns, especially between the inlets and outlets. Färm (2002) reported a decrease in concentration for all metals between the inlet and outlet. In contrast, Marsalek *et al.*'s (1997) study showed a marked increase in metal concentration around the outlet for Cd, Cu, Pb, Ni and Zn. In a pond studied by Mallin *et al.* (2002), an increase in metal concentration was observed around the outlet when compared with the inlet. Conversely, Cu concentrations in the <63 µm and <125 µm fractions decreased between the inlet and outlet of a retention pond in Oregon, USA (Mesuere & Fish, 1989).

These examples show that distance from inlet, at the pond scale, exerts no consistent control over the spatial deposition of metals within retention ponds, though it is widely assumed that grain-size, particularly the finer fractions, controls metal deposition since finer fractions adsorb more contaminants than larger ones, due to their greater specific surface area. Furthermore, sampling strategies have generally aimed to provide a good areal coverage of the pond, but methodological and cost constraints in obtaining samples have confounded this, and samples have typically been taken only at metre intervals. Additionally, the spatial accuracy and precision with which individual sediment samples are obtained has generally been overlooked. It is therefore intended that the relationships that exist between physical sedimentation, developing morphological structures, and areas of localised metal contamination within retention pond sediments are examined. These will inform pond design and maintenance, in particular the potential development of a method for rapidly assessing the location of sediments of greater metal contamination within retention ponds, through geomorphological observation. As part of the broader research, this paper reports on results from recent work that examines: (1) the utility of ground-penetrating radar as a method for rapidly delineating pond-bottom morphology and present sediment depths within retention ponds; and (2) the processes controlling metal deposition within pond-bottom sediments by quantifying their spatial variability.

STUDY LOCATION

The retention pond investigated is located in central Scotland (OS map reference: NS972662). The pond is large compared to other SUDS ponds in Scotland, with a surface area of approximately 1.65 ha and a water volume of 23 020 m^3. It functions as a regional control in the local SUDS treatment train, draining (by way of linear wetlands and swales) two large commercial distribution centres. The pond was constructed in 2000 and does not have an artificial liner since the substrate of boulder clay has a naturally low permeability. It has a maximum water depth of 2.1 m in dry conditions.

GROUND-PENETRATING RADAR: A NOVEL APPROACH

Feasibility study

Ground-penetrating radar (GPR) is a geophysical technique that identifies non-invasively subsurface electrical discontinuities by the emission, propagation and reflection of high-frequency electromagnetic impulses (Neal, 2004). Despite widespread use in many fields, few have examined the applicability of GPR to water bodies. It is likely that this is due to the increased scattering and attenuation of the electromagnetic signal that occurs on passage through water, leading many to assume that it is not a viable technique, particularly in waters that have high electrical conductivity. Nevertheless, one study examined the use of GPR for delineating riverbed morphology during a flood (Spicer *et al.*, 1997). Using the signal from a 100 MHz GPR antenna suspended above the river channel, a continuous transect of bed-surface features was constructed during a flood event but the low frequency only provided a theoretical depth resolution of 11.3 cm. Although no subsurface detail was extracted, this study proved that GPR was conducive to the delineation of bed-form remotely.

A successful trial of using GPR to delineate pond-bottom morphology and sediment depth in the retention pond was conducted in August 2006. In two separate runs, a 450 MHz antenna and a 200 MHz antenna were secured in contact with the bottom of an inflatable boat in a co-polarised, common offset configuration beside a differential GPS (global positioning system) rover unit. The base-station for the differential GPS was set-up on a tripod beside the pond. The differential GPS recorded location every second, while the GPR was left to run continuously. The boat was then moved in a series of approximately linear transects along the long-axis of the pond dependent upon the planform geometry of the pond and avoiding highly vegetated sections of the pond. To validate the GPR results, pond water depth was also measured on the same day using a Plastimo Echo-sounder and also a ranging pole.

Results and discussion

The data for both GPR-runs were post-processed using ReflexW 2D. Corrected GPR data for the 450 MHz antenna demonstrated an ability to acquire a water-sediment boundary with 0.02 m depth resolution, increasing to approximately 0.055 m for sub-surface sediment features. The resolution achieved by the 200 MHz antenna was lower: 0.06 m for the water–sediment boundary and approximately 0.15 m for the sub-surface sediment features. Water depths could be identified in 81.5% of all traces for the 450 MHz antenna, and in 99.2% of all traces for the 200 MHz antenna. The loss of data for the 450 MHz antenna compared with the 200 MHz antenna was anticipated due to the higher degree of signal attenuation experienced at higher frequencies. High electrical conductivity in the water body can be a confounding factor in the use of GPR, increasing signal attenuation and scattering. However, average electrical conductivity on the day of study was 387 $\mu S\ cm^{-1}$, low in comparison with GPR trials that were undertaken previously to test its applicability. The conduct of further field research on ponds using GPR needs to avoid times of high pond water electrical conductivity, for example as often occurs in temperate and cold climates in winter due to the wash-off of salt applied to roads.

Relationships were examined between post-processed data for both antennae and also between the individual antennae and the echo-sounded depths measured within the pond. Since each location where a GPR trace was obtained for the 450 MHz antenna was rarely in the same location as the GPR trace locations given by the 200 MHz antenna, a program was developed to extract trace points based on a search area (kernel) centred around each point. This allows the efficacy with which depths at the same location are measured by both antennae to be correlated (based on this 2 m × 2 m kernel). The program was also used to extract the locations of the echo-sounded depth points in order to compare them with the depths recorded for each antenna (Fig. 1). A similar degree of statistical explanation of the echo-sounded depth measurements was achieved by both the 450 and 200 MHz antennae. However, the 450 MHz antenna provided more realistic results as the gradient of the regression line was closer to 1 and the *y*-intercept closer to 0. This was expected as the depth resolution was better for the 450 MHz antenna than for the 200 MHz antenna. The depths measured by both antennae (Fig. 2) were highly significantly correlated ($r = 0.887$; $p = 0.000$).

The subsurface resolution of the data from the 200 MHz antenna is too low to detect any sediment deposits accurately. Post-processing of subsurface (sediment) depth data from the 450 MHz antenna has been challenging as no clear horizon was continuously represented throughout each GPR trace to delineate the initially constructed pond bottom. Therefore, there is a risk of misinterpretation if the depths estimated from the GPR traces are not validated by manually measured sediment

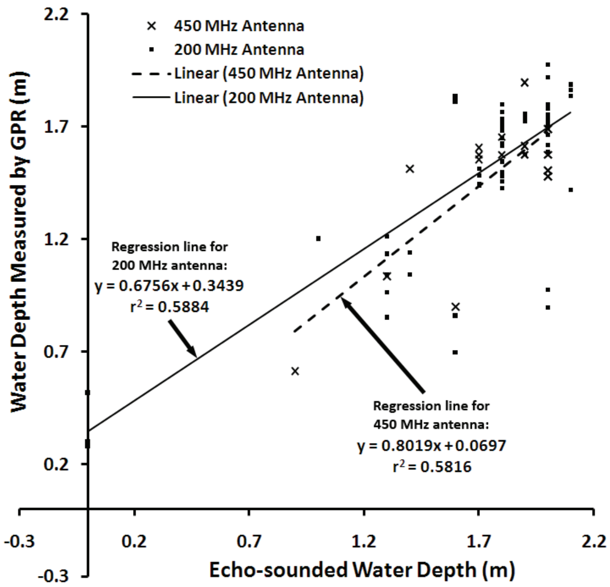

Fig. 1 The relationships between water depth measured by echo-sounder and water depth estimated from the 450 and 200 MHz antennae. For measurement error (range of GPR antenna resolution), see Fig. 2. Measurement error in echo-sounded water depths was ±0.05 m.

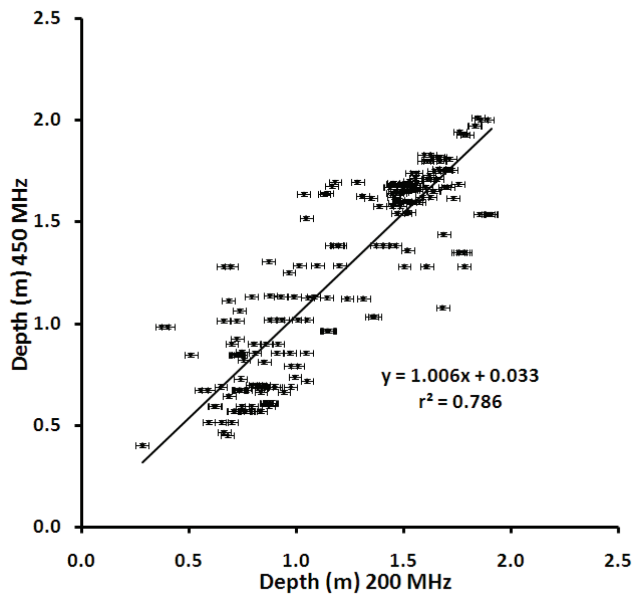

Fig. 2 The relationship between water depths estimated by the 450 and 200 MHz antennae (x-error bars for 200 MHz antenna = ±0.03 m and y-error bars for 450 MHz antenna = ±0.01 m).

depths throughout the pond. Sediment cores extracted from the pond in December 2006 were between 0 and 0.15 m thick, but it was difficult to distinguish the pond-bottom sediment from the deposited material in some cores. Sediment depths estimated from selected traces from the 450 MHz antenna (August 2006) were 0–0.22 m (±0.0275 m error). Typically, the greater sediment depths were recorded by the GPR at greater water depths and were of relatively uniform thickness throughout the main body of the pond. However, the boundary between water and sediment in the traces was "fuzzy" rather than abrupt, probably due to a boundary layer of suspended sediment that graded into more stable deposits. Sediment depths found at the periphery of the pond tended to be thinner (~0–0.1 m). It can be inferred from the 450 MHz GPR traces that time-averaged deposition rates at the pond's periphery can vary between 0 and 0.017 m $year^{-1}$, and 0–0.037 m $year^{-1}$ for the deeper areas of the pond.

SPATIAL VARIABILITY OF METALS IN RETENTION POND SEDIMENTS

Methodology

Pond sediments were sampled in March 2006 with minimal disturbance in a 1 m^2 "open-box" quadrat positioned approximately 3 m south of the main inlet to the retention pond. The positions and elevations of 30 randomly selected points within the quadrat were recorded using a Trimble 5600 series total station and cores of between 0.04 and 0.13 m depth were collected from these locations. The sediment samples were returned to the laboratory, air-dried, disaggregated and ground. The oven-dried moisture content and organic matter content (through loss on ignition) were determined for each sample. Total metal concentrations (Cr, Cu, Fe, Ni, Zn) were measured by flame atomic absorption spectrometry (Solaar M Series, Unicam) in digests of the ashed sediment samples prepared using hydrochloric and nitric acids. All analyses were performed in duplicate, with certified reference materials and blanks for laboratory quality control.

Results and discussion

Mean metal concentrations for Cr, Cu, Fe, Ni, and Zn within the quadrat were compared with results from SUDS ponds in Dunfermline, Scotland (Heal *et al.*, 2006) and standards (Table 1). The measured metal concentrations were generally similar or lower to the other Scottish SUDS ponds. This is perhaps due to the relative maturity of, and the greater traffic densities around, the other ponds.

Table 1 Sediment concentrations for total potentially toxic metals, compared with other Scottish retention ponds and standards for aquatic sediments. Units are mg kg^{-1} dry weight, except for Fe (%). Values are mean ± 1 standard deviation. Values in **bold** exceed aquatic sediment and/or contaminated land standards.

Determinand	This study (n = 30)	Halbeath[a] (n = 49)	Linburn[a] (n = 77)	Pond 7[a] (n = 62)	Wetland[a] (n = 123–126)	Standards for aquatic sediments[b]
Cr	22.6 ± 2.6	70.7 ± 65.8	78.2 ± 87.0	**118 ± 110**	76.7 ± 102	110
Cu	21.9 ± 3.1	18.8 ± 9.22	20.9 ± 15.3	16.3 ± 6.42	17.4 ± 7.44	110
Fe	**4.32 ± 1.18**	**4.41 ± 1.10**	**4.74 ± 1.68**	3.87 ± 0.873	**7.16 ± 3.04**	4
Ni	22.4 ± 6.4	63.3 ± 48.4	68.4 ± 39.8	**83.9 ± 61.4**	63.6 ± 57.5	75
Zn	69.7 ± 9.5	78.4 ± 72.9	110 ± 89.4	77.0 ± 24.8	93.1 ± 43.1	820

[a] Ponds located in Dunfermline, Fife, Scotland (aggregated results for all samples taken between 1999 and 2003.) See Heal *et al.* (2006).

[b] Severe effect level, Ontario Ministry of the Environment (1993).

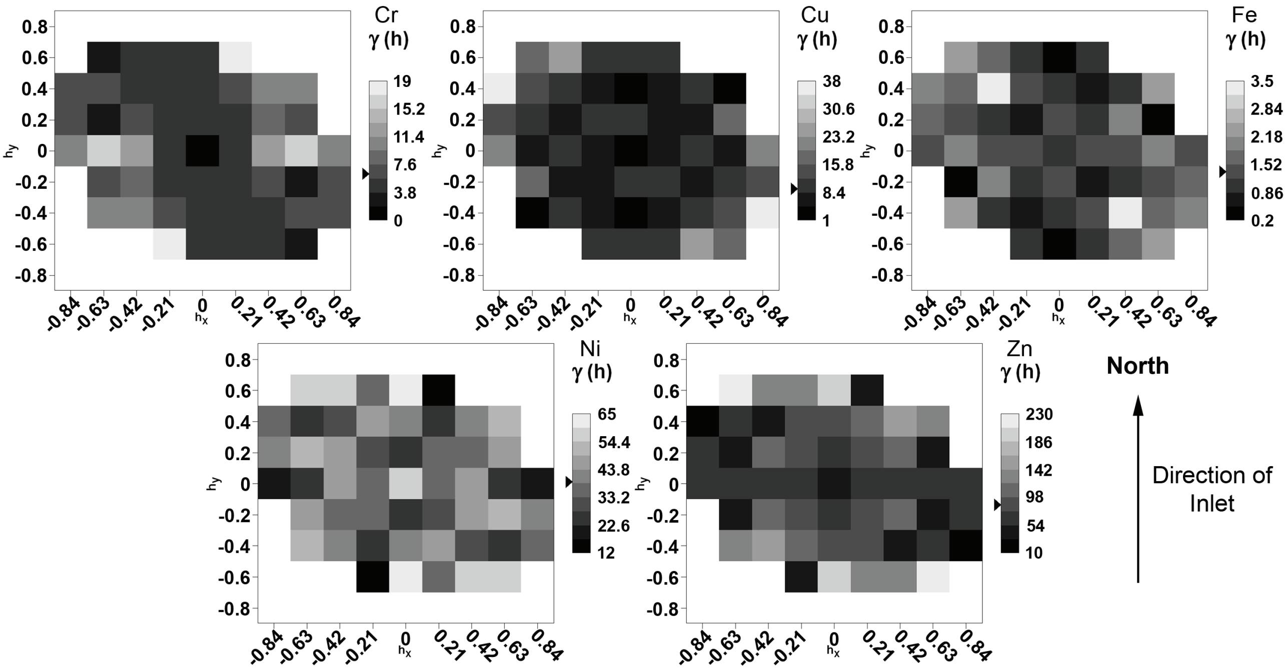

Fig. 3 Semivariogram surfaces for Cr, Cu, Fe, Ni and Zn sediment concentrations for the 1 m^2 quadrat study.

Semivariogram surfaces (Fig. 3) were created showing the variation in the data with respect to both distance (lag) and direction. Generally, spatial data sampled from a continuously varying population will show greater similarity for points closer together than those further apart. Overall, the patterns do not reflect this, demonstrating a highly variable surface at the centimetre scale. However, closer inspection shows that a degree of anisotropy exists in a north–south direction (between 0°N and –15°N) for Cr, Cu and, to some extent, Fe, in which the overall semivariance (γ (h)) is low along these axes compared to semivariances in other directions. This means the variability in metal concentration between samples is not as great along this axis, which also corresponds with the direction of the inlet and the main flow direction in this area. It is possible that higher velocities of flow along this axis do not fully permit the settling of sediments and associated metals, which are conveyed further into the pond.

With $n = 30$, the smallest lag that could be achieved was 0.21 m while keeping an optimum number of pairs of points in each lag. It appears that spatial variability is occurring at a finer level so increased resolution and more points are required in the semivariogram surfaces to quantify this detail. Relationships were also examined between metal concentration and sediment deposit morphology. The 30th percentile in cumulative elevation was used to delimit areas of "low" and "high" relief in the 1 m^2 quadrat. Two-sample t-tests were conducted to determine if there were significant differences ($p < 0.05$) between metal concentrations in these two areas. The null hypothesis was accepted in all cases showing that the sediment surface elevation around the inlet exerted no obvious control on metal concentration.

CONCLUSION

Ground-penetrating radar, coupled with differential GPS, provide a rapid means with which to acquire a continuous representation of present-bedform morphology in retention ponds, providing that the water electrical conductivity and the volume of submerged vegetation in the pond are low. Subsurface detail tends to be more difficult to acquire, though this is improved if higher-frequency GPR is used, if sediment depths are thick, or if the water depth is shallow. At the centimetre scale, metal concentrations in sediment at the pond inlet were highly variable. Micromorphological sedimentary structures that develop around the inlet do not exert a strong control on the variability observed, but inlet geometry and the angle at which water flows into the pond appeared to reduce the variability along its axis for Cr, Cu and Fe concentrations in sediment. Further analyses of these data, particularly with grain size information, are needed to provide a fuller explanation of the controls upon metal concentration in pond-sediment deposits. Nevertheless, the high degree of spatial variability in metal concentrations in sediment measured at the inlet alone suggests that care should be taken in devising sampling strategies for assessment of the extent of sediment contamination in retention ponds, particularly if few samples are used to estimate pond-scale metal concentrations in sediments.

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