Changes in water quality parameters due to in-sewer processes

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Abstract Combined sewer systems contain a large number of organic and inorganic pollutants from both domestic and industrial sources. These pollutants are often retained within the combined sewer system for significant lengths of time before entering sewage treatment works, or being spilt to a watercourse via a combined sewer overflow (CSO) during storm conditions. Currently little knowledge exists concerning the effects of in sewer processes on pollutants. Understanding of in-sewer processes is important for the effective and efficient design of treatment works and CSO chambers and for impact assessments on receiving waters.

A series of studies covering storm and dry weather flow conditions were undertaken with the aim of investigating the nature of in-sewer processes. These studies consisted of marking a body of water with a fluorescent tracer. The tracer was then monitored at a series of downstream sites, and discrete samples collected from the body of water as it progressed through the sewer. The samples were analysed for water quality parameters and these results investigated in tandem with the detailed hydraulic information gained through the tracer studies. The results highlight the hydraulic differences between storm and dry weather conditions such as increased travel times and mixing under storm conditions. The Advection Dispersion Equation (ADE) and Aggregated Dead Zone (ADZ) model parameters have been quantified for the tracer data. The ADE mixing coefficient is shown to increase by an order of magnitude for storm conditions. The ADZ dispersive fraction parameter is shown to be approximately constant with flow. Chemical reactions and decay within the sewer system were found to be consistent with oxygen limitation.

Keywords Combined sewers; mixing; pollution

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Introduction

The majority of chemicals used in domestic applications are disposed of through sewerage systems. Once in the sewers, under normal dry weather flow conditions these chemicals are transported to treatment works where they are either removed or reduced to acceptable levels for the receiving water. Under storm conditions CSOs can discharge part of the flow directly to receiving waters. Historically, design philosophy has been that under storm conditions the majority of the pollutant load will be diluted to acceptable levels. For both conditions it is important to understand the effects of in-sewer processes that may degrade and disperse chemicals and affect other water quality parameters. This allows the design specifications for treatment works to be optimised and the likely impact on receiving water due to CSO discharges assessed. Currently little knowledge exists concerning the degradation and mixing of pollutants in sewer systems. The reactions of primary interest, such as the nitrogen decay cycle, are predominately oxygen driven and are a function of time. Many factors affect the decay rates and interaction of the chemicals present, such as sediment load and bed material age and composition.

Fieldwork methodology

Field studies have been undertaken in an attempt to investigate the temporal change in water quality parameters in a live sewer system. To obtain meaningful measurements of the required water quality parameters it is essential to sample the same body of water as it progressed through the sewer system. This was achieved by marking a body of water with a fluorescent tracer and collecting discrete samples at the centroid of the tracer cloud as it passed a number of downstream sites. A field site was carefully selected to ensure that there were no additional inflows over the reach length, hence changes in water quality could only be due to mixing or chemical reaction.

Site description

A length of combined sewer with no inputs was located to provide quantifiable test conditions. The sewer reach found served a largely domestic catchment and had a 1.5 km connection to a local treatment works through a 375 mm diameter pipe. There was only one small ancillary connection over the reach and during an early site investigation this was found to contain no flow under dry weather conditions. Approximately 1 km of the sewer had recently been rehabilitated providing idealised engineered conditions and minimising infiltration. A further advantage of the sewer was that it had regular manholes that were reasonably shallow and located away from public highways. This made the fieldwork feasible in terms of health and safety and practical for sampling. Figure 1 gives a longitudinal section of the sewer, showing pipe invert levels, water levels during both storm and dry weather flow conditions and the ground or cover level.

Calculations suggested travel times in excesses of 1 hour for the reach. This was confirmed by a preliminary tracer test. Samples were also collected and analysed to ascertain water quality parameter levels, which were found to be typical for a largely domestic catchment (McGhee, 1991). Previous laboratory studies with sewer samples have shown that an hour is sufficient to investigate changes in water quality parameters and chemical degradation for the levels found in the samples (Tanaka and Hvitved-Jacobsen, 1998).

Measurement techniques

Experiments were conducted by injecting a slug of Rhodamine WT fluorescent tracer to mark a body of water. This was monitored at up to seven downstream sites using Turner Designs model 10-005 field fluorometers and/or Turner Designs "SCUFA" submersible fluorometers. The field fluorometers were run in a continuously pumped mode, the SCUFAs were suspended in the sewer flow. Both types of fluorometer output were logged

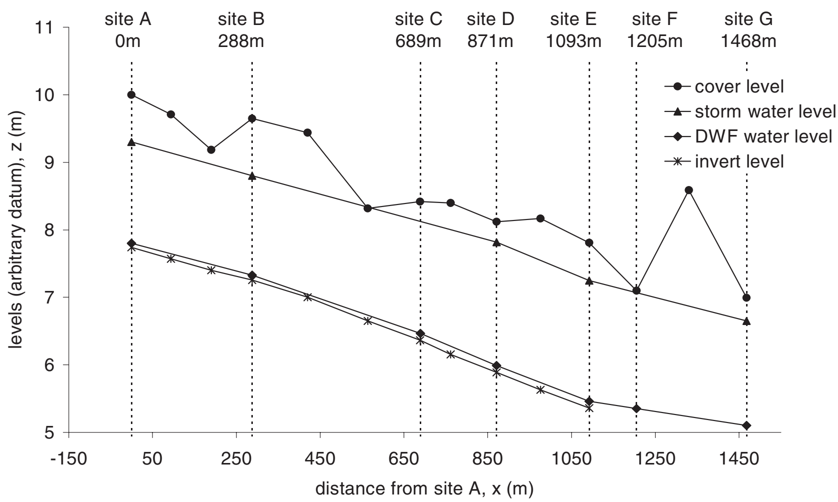

Figure 1 Longitudinal section of sewer system showing sampling points, levels and surcharge levels, injection was at –150 m

at 5 second intervals and monitored visually. Water quality samples were collected as close as possible to the centroid of the tracer cloud as it passed each sample site.

The discrete samples collected were analysed for total suspended sediment, coprostanol, ammonia, total oxidised nitrogen, nitrite, and boron. The water temperature and dissolved oxygen (DO) levels were also measured at selected sites under dry weather flow conditions.

Test conditions

Three tests were undertaken.

- Storm conditions, November 2000, flow approximately 100 l/s, a time of severe flooding in much of the United Kingdom. As a result the sewer conditions on the day were surcharged by approximately 1.5 m throughout the reach, as shown in Figure 1. Sites A, B, D, E, and G, were monitored, the number of sites was limited by equipment availability. Two repeat tracer tests with sample collection were undertaken.
- Dry weather flow conditions, June 2001, flow approximately 30 l/s. All sites shown in Figure 1 were monitored. Three repeats were made.
- Dry weather flow conditions repeated, flow approximately 30 l/s, August 2001. All sites shown in Figure 1 were monitored. Four repeats were made.

Results

Hydraulics

Figure 2 shows complete tracer profiles obtained under storm conditions. Complete traces were only obtained at three sites because of difficulties experienced working with live sewage. At the other sites sufficient data was obtained for timings of sample collection but not for detailed hydraulic analysis. Figure 3 shows traces from dry weather flow conditions, again some difficulties were experienced. However, for the trace shown all 7 sites were successfully monitored for complete traces. The two sets of traces have been normalised by respective injection concentration such that they are directly comparable. Comparison of the two figures immediately reveals the marked difference in travel time for the different conditions. For storm conditions at the furthest down stream site, G, the centroid of the plume arrives around 4,000 seconds after injection, whereas for dry weather flow conditions the centroid of the plume does not arrive until after 6,000 seconds. The shape of the measured plumes is also very different. For dry weather flow the plumes are all approximately Gaussian, whereas the storm condition traces show considerable and increasing skewness. Comparison of measured profiles at site A highlights the initial differences in dilution and retention for the two conditions. Differences in first arrival times for the short distance between injection and first sample site are visually negligible. However, the peak concentration for storm conditions is a fraction of that for dry weather flow conditions, and there is a change in initial spread from some 1,000 seconds for storm conditions to only

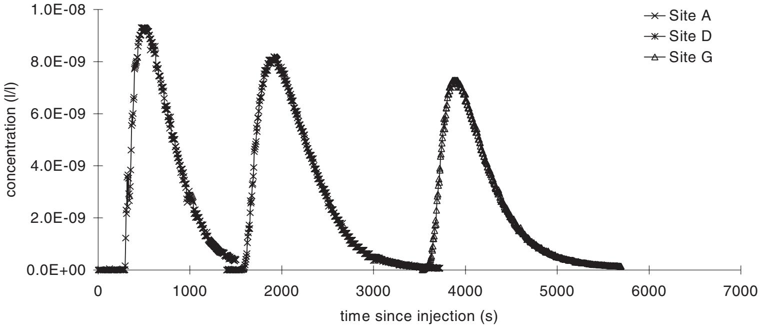

Figure 2 Concentration profiles from storm conditions (complete measurements only obtained at 3 sites)

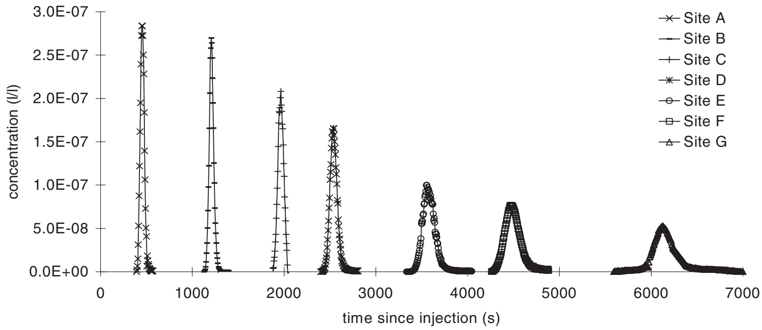

Figure 3 Concentration profiles from dry weather flow conditions (June 2001)

100 seconds for dry weather flow. These features dominate further visual inspection of the data.

The tracer profiles have been analysed to characterise the hydraulic conditions. Two commonly used methods for modelling pollutant transport are the Advection Dispersion Equation (ADE) and the Aggregated Dead Zone model (ADZ), descriptions of these can be found in Rutherford (1994).

ADE based models require two coefficients, a travel or retention time, $\bar{t}$, and a longitudinal mixing coefficient, D. The retention time is the time difference between the centroids of the upstream and downstream plumes. The longitudinal mixing coefficient is a measure of the change in spread between the two plumes and is a function of the temporal variance and travel time. ADE models have been developed from work by Taylor (1954) and are mathematically rigorous, and although technically limited by a number of assumptions made in their derivation, have been shown to be widely applicable (Rutherford, 1994). ADE based models can only predict spatial concentration profiles of Gaussian shape, whereas measured profiles in non-uniform flows are often significantly skewed, as seen in Figure 2.

The second method is the Aggregated Dead Zone (ADZ) model. This technique also utilises two parameters, again the travel or retention time, $\bar{t}$, and a time delay, τ, which is calculated as the difference between the first arrival times at the upstream and downstream sites. The ADZ model uses these coefficients to move the concentration distribution downstream and applies an exponential decay to each element. A simple discrete-time equation for the prediction of temporal concentration distributions can be found in Wallis *et al* (1989). The potential of the ADZ technique for use in urban drainage has been shown by Dennis (2000) in his study on travel time and mixing of solutes in surcharged manholes. The ADZ parameters can be lumped into a single parameter, the dispersive fraction, D_f, as defined by Young and Wallis (1986). The dispersive fraction is a measure of the ratio between the residence time and the total time a tracer spends in a reach and is given by $D_f = (\bar{t} - \tau)/\bar{t}$.

The ADE and ADZ models have been applied to the measured tracer data using optimisation techniques described in Dennis (2000). The results for each set of tests are summarised in Table 1, travel times have been converted into velocities and the dispersive fraction given to facilitate comparison between reaches of different lengths. The velocities for both models are consistent as would be hoped. There is a marked increase in velocities associated with the increased flow under storm conditions. The ADE mixing coefficient is approximately constant for the dry weather flow conditions, however there is an order of magnitude increase in the coefficient due to the increased mixing effects of the storm conditions. It should be noted that the standard deviations of the mixing coefficients are large, particularly for storm conditions. The dispersive fraction is approximately constant with flow rate.

Table 1 Summary results from analysis of tracer data (SD = standard deviation)

		ADE		ADZ	
		D (m²/s)	u (m/s)	D_f (–)	u (m/s)
November 00 Storm	average	0.608	0.54	0.045	0.53
	SD	0.989	0.09	0.045	0.08
June 01 DWF	average	0.064	0.30	0.043	0.29
	SD	0.047	0.09	0.021	0.09
August 01 DWF	average	0.056	0.34	0.035	0.33
	SD	0.050	0.10	0.015	0.10

Prediction of the required parameters limits applicability of models based on either the ADE or the ADZ principles. Both models utilise travel time, this may be estimated from a mean velocity. The ADE mixing coefficient can be calculated for pipe full conditions from Taylor (1954) who gave $D = 10.1du^*$ where d is pipe diameter and u^* is bed shear velocity. This can be estimated from $u^* = \sqrt{(gRS_0)}$ where g is acceleration due to gravity, R hydraulic radius and S_0 hydraulic gradient. This expression considers only mixing due to velocity shear, termed shear dispersion, however the measured mixing coefficient encompasses the delayed storage effects of ancillary structures such as manholes (Dennis, 2000).

The measured water surface profiles as plotted in Figure 1 show relatively little difference between storm and dry weather flow conditions. Hence calculation of D from Taylor (1954) for storm and dry weather flows render approximately the same value, 0.1 m^2/s. Comparison with Table 1 reveals this to be an over prediction for dry weather flow conditions because the pipe was only approximately half full, and an under prediction for storm conditions due to the effects of ancillary structures.

The time delay coefficient required for ADZ modelling can be predicted from the dispersive fraction, D_f. Table 1 shows dispersive fraction to be approximately constant over the flow conditions (given the spread in the data characterised by the standard deviation values). However, there is currently no method for estimating the value of the dispersive fraction without undertaking fieldwork.

Water quality

The temporal variations in suspended solids concentration, boron, coprostanol, ammonia, total oxidised nitrogen and nitrite parameters have been analysed in an attempt to investigate the effects of in-sewer processes on sewer water quality. Values for these parameters are available for all tests and repeats (see "Test conditions" section). The temperature of the sewer flow was on average 14.5°C in June and 16.5°C in August, and showed a general trend to decrease over the reach length by approximately 0.5°C. The maximum dissolved oxygen recorded was 2.4 mg/l, although the average for both dry weather flow tests was 0.8 mg/l. Thus any degradation processes occurring during dry weather flow conditions are expected to be limited by oxygen availability.

Figure 4 shows the trends for suspended solids, coprostanol and boron as a function of sample time. Coprostanol and boron are marker chemicals relating to humans and detergents respectively and are expected to remain constant within the sewer environment. The variation seen in suspended solids concentration reflects the difficulty of reproducible sampling from two phase media. There is also real variation in the type and concentration of solids between repeat runs on the same day. This is illustrated in Plate 1, which shows the dark solids obtained in run 1 and the clearer sample obtained from run 2. The coprostanol and boron concentrations were both greater in the run with the dark solids. Further development of the sampling technique would be required to allow trends in the concentrations of

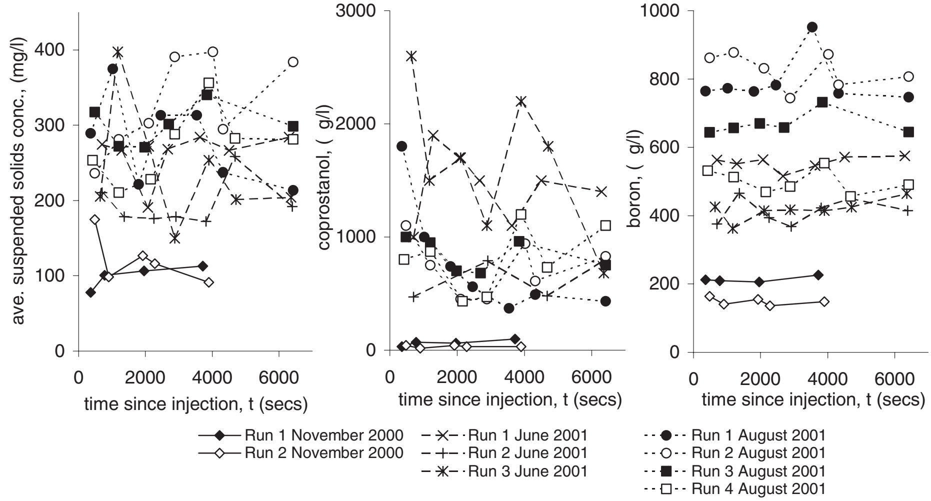

Figure 4 Suspended solids, coprostanol and boron concentrations versus sample time for all tests and repeats

materials adsorbed to solids to be determined. A more uniform input would be expected from a larger population.

The variability in the concentration of coprostanol, present as material associated with the solid phase, is of similar order to the suspended solids variability. Although the suspended solids and coprostanol samples were obtained from separate bottles, filled sequentially, the suspended solids to coprostanol ratios are not consistent between samples. Thus further development of sampling methodology will be required to enable investigation of changes along the sewer length. This may encompass improved synchronisation of sampling at the centroid of the tracer cloud. Less variability is seen in the boron concentrations, as would be expected for a chemical associated entirely with the liquid phase.

The suspended solids concentrations for storm conditions are generally lower than the dry weather flow concentrations. The average dry weather flow suspended sediment concentration is around 280 mg/l, compared with approximately 100 mg/l seen under storm conditions. November 2000 was a period of prolonged heavy rainfall, thus any deposits within the sewer system would have been eroded prior to testing. The observed sediment concentration is probably due to runoff and dry weather flow solids. Hence the sediment load in November is lower than the dry weather flow value due to overall dilution by the storm flow. Coprostanol and boron also show higher concentrations under dry weather flow. Coprostanol concentrations in June average approximately 1,300 µg/l, whereas the August concentrations average about 800 µg/l. Conversely the boron results are higher for August at around 700 µg/l compared to the June average of 500 µg/l. This may reflect the actual human and detergent sewer inputs at the time of sampling.

Figure 5 shows the trends for ammonia, total oxidised nitrogen and nitrite concentrations as a function of sample time. In-sewer decay via the nitrogen cycle may be responsible for the higher concentrations of nitrite and total oxidised nitrogen seen under storm conditions, even though greater dilution will have occurred. Comparison of the dry weather and storm flow concentrations for boron (Figure 4) which may be considered indicative of dilution rates, and ammonia (Figure 5) shows ammonia concentrations lower than expected due to dilution during storm conditions. This is consistent with ammonia removal due to oxidative processes. No decay through the nitrogen cycle is apparent over the test reach (Figure 5), hence any such decay is believed to have occurred upstream.

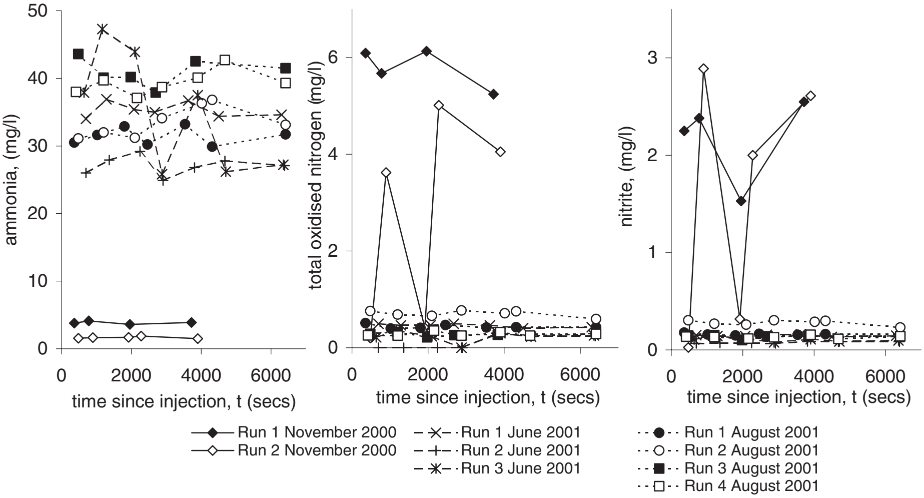

Figure 5 Ammonia, total oxidised nitrogen and nitrite concentrations versus sample time for all tests and repeats

Plate 1 June 2001 samples

Conclusions

- Mean travel times of pollutants under storm and dry weather flow conditions can be predicted from simple velocity area discharge relationships. Velocities have been shown to be increased under storm conditions.
- Traces measured under dry weather conditions exhibit approximately Gaussian forms concurrent with idealised pipe mixing theory. However, under storm conditions the traces exhibit significant skewness. This is believed to be due to the delayed storage effects within surcharged ancillary structures.
- The ADE mixing coefficient has been shown to increase by an order of magnitude from dry weather to storm conditions. Taylor's (1954) expression for pipe flow has been shown to overpredict dry weather flow values and under predict mixing coefficients for storm conditions. This under prediction is believed to be associated with the effects of ancillary structures, such as manholes, which are not considered in the derivation of the expression.

- The ADZ parameter, dispersive fraction, has been shown to remain approximately constant with discharge.
- The maximum dissolved oxygen recorded under dry weather flow conditions was 2.4 mg/l although the average was 0.8 mg/l. Degradation processes occurring under these conditions were limited by oxygen availability. Decay through the nitrogen cycle may be inferred for storm conditions.
- When chemical decay is limited by a lack of dissolved oxygen, prediction of hydraulic mixing may be more important to describe temporal concentration variations. However, such processes are only of significance when the pollutant has a time varying profile.

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