PERGAMON

Water Research 36 (2002) 3054–3062

WATER RESEARCH

www.elsevier.com/locate/watres

The use of salt dilution gauging techniques: ecological considerations and insights

P.J. Wood[a,*], A.P. Dykes[b]

[a] Department of Geography, Loughborough University, Loughborough, Leicestershire, LE11 3TU, UK
[b] Geographical Sciences, University of Huddersfield, Queensgate, Huddersfield, HD1 3DH, UK

Received 1 June 2001; received in revised form 8 November 2001; accepted 16 November 2001

Abstract

The salt dilution (gulp injection) technique is a well established and widely used technique to measure stream discharge, flow velocity and water residence characteristics in small headwater streams. However, the impact of the technique on water quality and instream ecology has been largely ignored in field investigations. A series of experiments were undertaken in a regulated and groundwater-dominated river to examine the effects on aquatic invertebrate drift at high, medium and low discharges. In the groundwater-dominated river, drift significantly increased as a result of the introduction of the saline solution under all flows. Drift increased at the regulated site under low and intermediate flow but not during high flows, probably due to a natural increase in drift associated with spate conditions. Following the application of the saline solution several taxa absent or infrequently occurring in background samples, such as the cased caddisfly, *Agapetus fuscipes*, were recorded. The wider implications of the technique are discussed in relation to short-term pulsed pollution episodes and the management of riverine ecosystems. © 2002 Elsevier Science Ltd. All rights reserved.

Keywords: Macroinvertebrates; Drift; Disturbance; Salt dilution; Gulp injection; Tracer

1. Introduction

A wide range of laboratory flume and artificial stream studies have been undertaken to examine the response of aquatic communities to environmental perturbations [1–3]. To complement controlled experimental lotic ecosystem studies, a wide range of field techniques have been developed by scientists to quantify streamflow, water quality and instream community characteristics [4,5]. These techniques have provided water resources managers and academics with an abundance of information regarding the physico-chemical characteristics of lotic ecosystems and instream ecology. However, the wider implications and potential disturbance associated with the field techniques employed to gather water quality and instream ecology data have frequently been overlooked or ignored [6,7].

Many benthic freshwater organisms have evolved adaptations to withstand predictable natural events associated with hydrological variability [8], with the possible exception of extreme high magnitude and low-frequency events [9]. Anthropogenic impacts, and pollution in particular, can cause severe disturbance and degradation of the environment and aquatic communities [10–12]. However, it is not usually possible to examine the impact of ecosystem stresses associated with pollution on natural systems in real time due to wider environmental, toxicological and ethical concerns. Severe disturbances do not necessarily result in the death or exclusion of all of the organisms within the affected area [13] but may lead to a natural involuntary and/or active response by some members of the invertebrate community when subjected to adverse conditions. One of the most widely studied responses of benthic fauna is the entry of organisms into the water

*Corresponding author. Tel.: +44-1509-223012; fax: +44-1509-223-930.
E-mail address: p.j.wood@lboro.ac.uk (P.J. Wood).

PII: S0043-1354(01)00519-X

column and transport downstream, often referred to as 'drift' [14,15].

Several different categories of drift associated with benthic invertebrates have been identified in the scientific literature: (i) catastrophic drift—usually associated with potentially lethal conditions such as gross pollution [16]; (ii) behavioural drift—entry into the water column in response to a stimulus such as changes in light intensity, foraging activity or the presence of a predator [14]; (iii) distributional drift—usually envisaged as a dispersal mechanism for invertebrate larvae and nymphs soon after emergence from the egg; and (iv) constant drift—low numbers of organisms drifting due to accidental dislodgement from the substratum irrespective of any diurnal activity [17,15,18].

Diurnal measurement of drift has indicated that behavioural drift usually forms the largest component and that most taxa drift during the hours of darkness [19]. Peak drift rates have typically been recorded shortly after sunset and before sunrise [14]. This is probably a behavioural strategy to reduce the risk of predation [20]. However, these clearly identifiable patterns potentially enable those who study lotic ecosystems to clearly distinguish between induced disturbance and natural patterns of aquatic invertebrate drift. This paper examines the impact of the salt dilution gauging (gulp injection) technique on the drift of benthic macroinvertebrates during daylight hours in two contrasting streams under different flow conditions.

2. Study sites

Experiments were undertaken under high winter flow, intermediate and low summer flow conditions in 50 m sections of two contrasting riverine systems: the Little Stour River, Kent, and the River Holme, West Yorkshire (Fig. 1). The Little Stour River is a small chalk stream rising approximately 4 km east of the city of Canterbury. It is typical of most perennial English chalk streams rising from a groundwater spring and with diverse instream substratum patches downstream of the source [21,22]. The study reach had an average width of 7.7 m and a typical background conductivity of around 590 $\mu S\,cm^{-1}$. The river has been extensively managed historically and the study reach was canalised during the

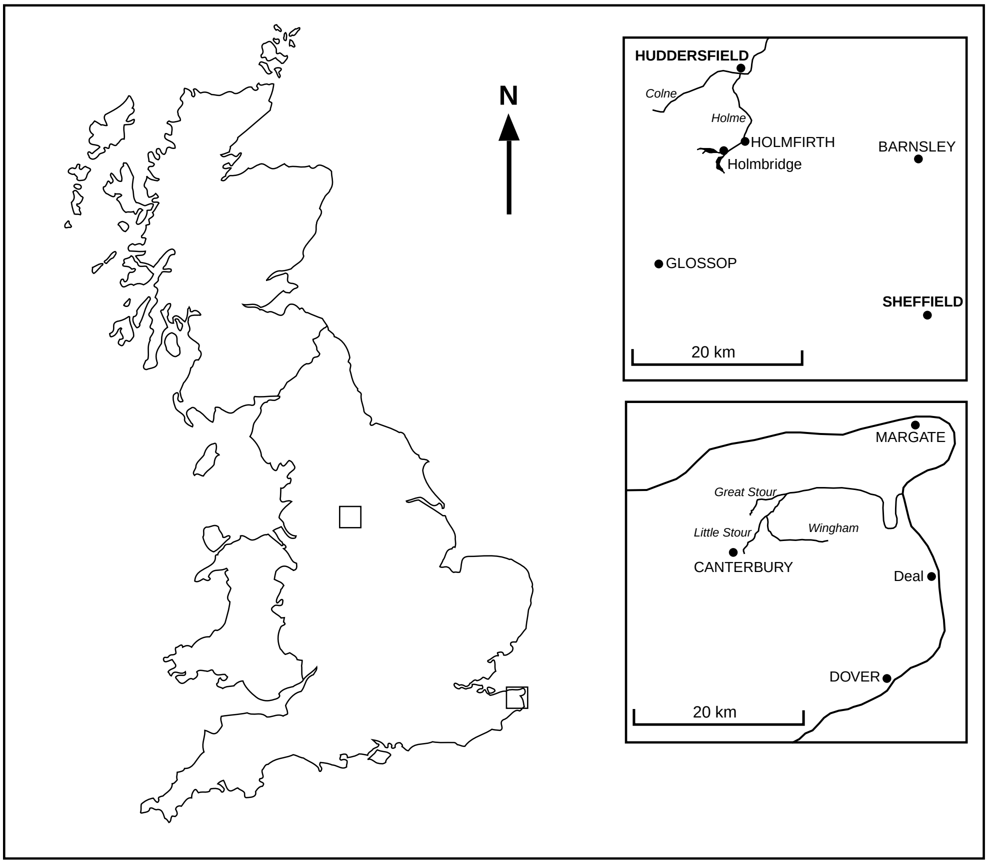

Fig. 1. Location map of the Little Stour River, Kent, and River Holme, Yorkshire.

eighteenth century. The River Holme is a small Pennine stream rising from Carboniferous Millstone Grit sandstones and shales 16.5 km south of the town of Huddersfield. The study reach had an average width of 5 m and a typical background conductivity of around 110 μS cm^{-1}. The study reach had a relatively steep slope with almost continuous riffle/rapids and a substratum typified by gravels and cobbles. River flow has been extensively regulated in the last 150 years by upstream storage reservoirs for drinking water supply, although several are now used to support base-flows in the main river during the summer months.

3. Methods

The dilution gauging (gulp injection) technique is a well-established procedure for measuring the discharge of rocky, turbulent streams where other techniques are unsuitable [4]. The technique has been widely used to determine discharge, but has also been used by freshwater scientists to model the time of travel of pollutants [23] and to characterise backwater/hydraulic dead zones [24]. A range of different tracers have been used, such as sodium dichromate, lithium chloride, rhodamine WT and fluorescein [25,26]. The tracer used should ideally meet all the following criteria: (i) it should be highly soluble in water at stream temperatures; (ii) it should be stable in the presence of light, sediment or other substances in natural waters; (iii) it should be easily detected at low concentrations; (iv) it should be absent or present at such low concentrations in the natural environment that it does not influence measurement; (v) it should be relatively inexpensive; and (iv) it should be non-toxic to stream biota and without long-term impacts on water quality or the environment. In field studies of small turbulent headwater streams, common salt (NaCl) has been the most widely used tracer, not least because it is the cheapest [4,27].

In each of the six experiments undertaken, 1 kg of salt was dissolved in approximately 8 l of stream water and injected into the stream by inverting a bucket containing the tracer solution. The electrical conductivity of the stream water was measured 50 m downstream prior to the injection and at 2 s intervals from the time the tracer solution was introduced into the stream. The 50 m study reach allowed thorough mixing of the tracer and allowed the experiment to be managed easily. Conductivity was measured using a digital field meter, for which a calibration curve had been obtained prior to its use in order to obtain the salt concentrations corresponding with the measured conductivities. The discharge (Q) was derived from the standard equation [28, p. 128]

$$Q = \frac{(c_1 - c_0)v}{\int_{t_1}^{t_2}(c_2 - c_0)\,\mathrm{d}t}$$

where a known volume of the tracer (v) of concentration c_1 is added in bulk to the stream of a known background concentration (c_0) and, at the sampling point, the varying concentrations, c_2, are measured regularly following the introduction of the tracer cloud at time t_1 until all the tracer has passed the monitoring point (t_2).

Invertebrate drift was measured using three standard drift nets 0.4 m wide and 0.25 m high fitted with 250 μm mesh nets, randomly positioned between 40 and 50 m downstream of the point of injection. Each net was positioned so that it could be approached from downstream and changed without causing disturbance to the other nets, which might have biased the results. Nets were changed at regular intervals during the sampling periods due to the potential backwash effect and clogging of the nets with organic detritus and drifting invertebrates [29]. Background drift was measured over 72 h prior to salt injection on each occasion and over a shorter duration (e.g. 0.25, 0.5, 1, 2, 4 and 8 h) up to 24 h following the injection. The saline solution was injected into the river during daylight hours, typically between 10:00 and 11:00 am, so that drift associated with the experimental conditions could be clearly separated from any diurnal patterns associated with variations in diurnal light intensity [19]. The material collected in drift nets was transferred into plastic bags and preserved with 70% industrial methylated spirits in the field to prevent degradation and minimise the emergence of adult insects from nymphs or pupae. All samples were returned to the laboratory for manual sorting, identification and enumeration of invertebrates and exuvia within the sample. Invertebrate community drift data were log-transformed (log e) prior to undertaking analysis of variance (ANOVA).

4. Results

Measurement of discharge using the salt dilution technique produces very accurate results but leads to marked changes in the salt concentration of the stream water over a short period of time (Fig. 2). The traces can be easily plotted and are widely used by hydrologists to determine stream discharge, mean flow velocity [28], and to determine residence characteristics in pollutant travel time or back-water (dead-zones) studies. In each of the experiments, background conductivity levels (± 2 μS cm^{-1}) were regained within 900 s (15 min) following the injection of the saline solution. Discharge in the Little Stour study reach was 0.107 m^3 s^{-1} under low flow conditions (September), 0.265 m^3 s^{-1} at intermediate flow (June) and 0.413 m^3 s^{-1} at high flow (December). Stream discharge in the River Holme was 0.187 m^3 s^{-1} under low flow conditions (October), 0.275 m^3 s^{-1} at intermediate flow (May) and 1.25 m^3 s^{-1} under high flow conditions (February). The

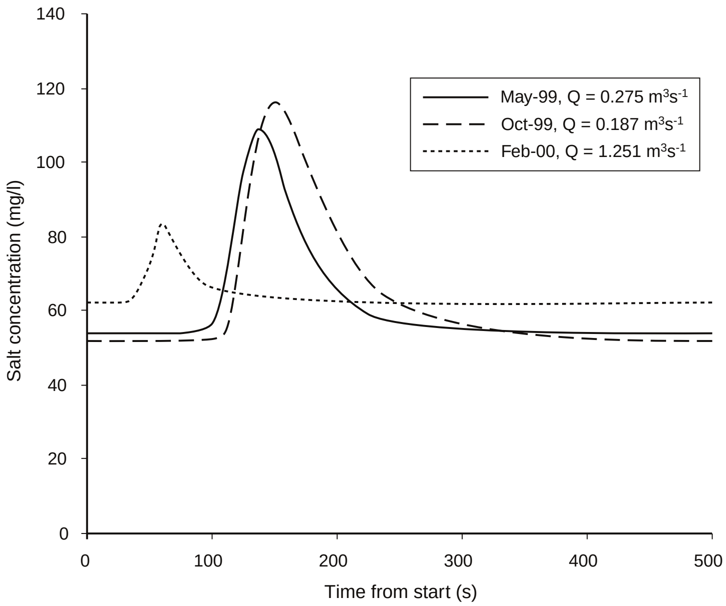

Fig. 2. Variations in concentrations of dissolved salt (mg l^{-1}) with time following injection of the salt solution, 50 m downstream of the injection point in the River Holme.

range of flows recorded in the two rivers reflects the buffered nature of the groundwater-dominated regime in the Little Stour and the highly regulated nature of the River Holme under most flow conditions.

A total of 38 taxa were recorded in the Little Stour River and 20 taxa in the River Holme drift samples. Background community drift rates ranged from 39 invertebrates h^{-1} during low flow in the Little Stour to 345 invertebrates h^{-1} at intermediate flow; in comparison the rates ranged from 20 to 285 h^{-1} in the River Holme. Freshwater shrimp (*Gammarus pulex*) constituted over 70% (71.3–85.2%) of the total drift in background samples during the experiments in the Little Stour. In the River Holme, larval Chironomidae formed the largest component of background drift (51.7–56.7%) followed by Plecoptera larvae (7.2–11.8%) (*Leuctra fusca*, *Leuctra nigra*, *Nemurella picteti*, *Nemoura cinerea* and *Isoperla grammatica*).

The introduction of the salt solution had a marked effect on the number of aquatic invertebrates drifting during the following 24 h in every experiment except at high winter flow conditions in the River Holme (Figs. 3 and 4). Preliminary investigations indicated some variability in drift between individual nets existed, a factor widely acknowledged in previous studies [30,15] and probably reflects the naturally patchy distribution of benthic invertebrates on the riverbed [31,32,22]. However, the introduction of the tracer solution had a greater impact than the between-net variability.

ANOVA of log-transformed drift data demonstrated a significant difference between background and experimental drift rates under all flows in the Little Stour and for low and intermediate flow in the River Holme. However, no significant difference was recorded between the background and experimental drift rates under high flow conditions in the River Holme (Table 1). The experimental conditions led to a significant increase in the drift of individual taxa, notably the two most frequently occurring taxa; freshwater shrimp (*Gammarus pulex*) in the Little Stour and non-biting midge larvae (Chironomidae) in the River Holme.

For some taxa the response was largely confined to one or two experiments, for example the background

Table 1
One-way analysis of variance of background invertebrate drift samples and samples following the introduction of the tracer solution in: (a) Little Stour, and (b) River Holme at different flows

	Sum of squares	Df	F-ratio
(a) Little Stour			
Low flow	5.17	1	58.30*
Intermediate flow	2.78	1	13.86*
High flow	1.11	1	168.87*
(b) River Holme			
Low flow	3.05	1	156.29
Intermediate flow	9.33	1	104.13*
High flow	0.38	1	2.52

* $P < 0.001$.

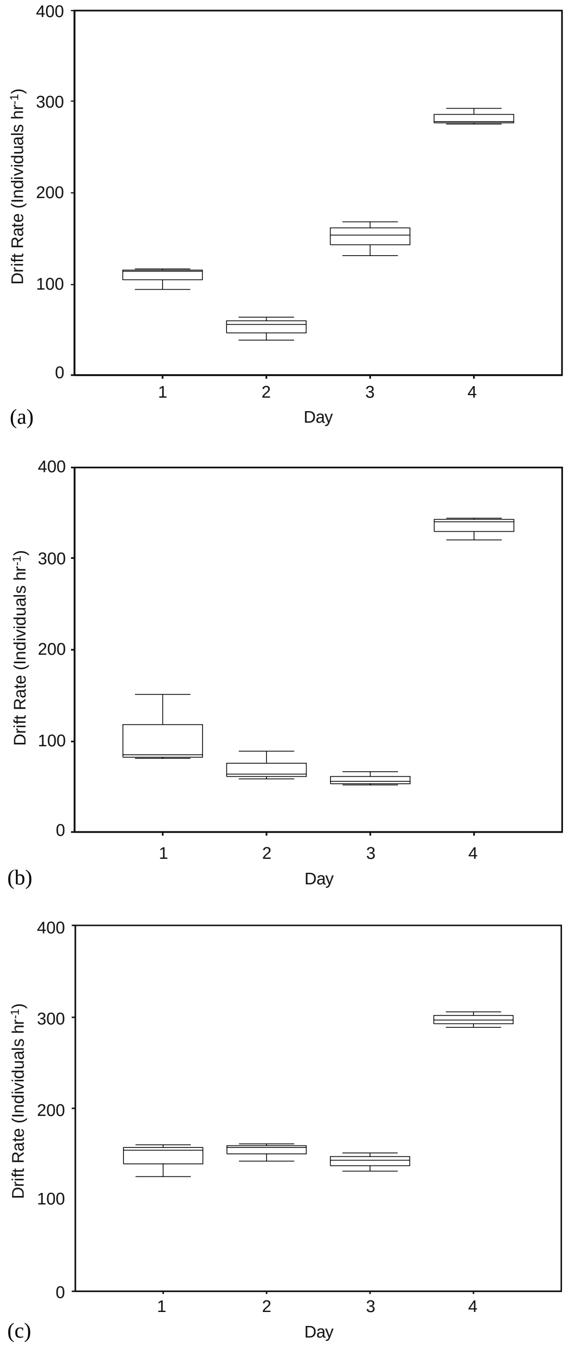

Fig. 3. Box plot of invertebrate drift rates for the 72-h period prior to tracer injection (day 1–3) and for the 24-h period following injection (day 4) at: (a) low, (b) intermediate, and (c) high flow in the Little Stour River.

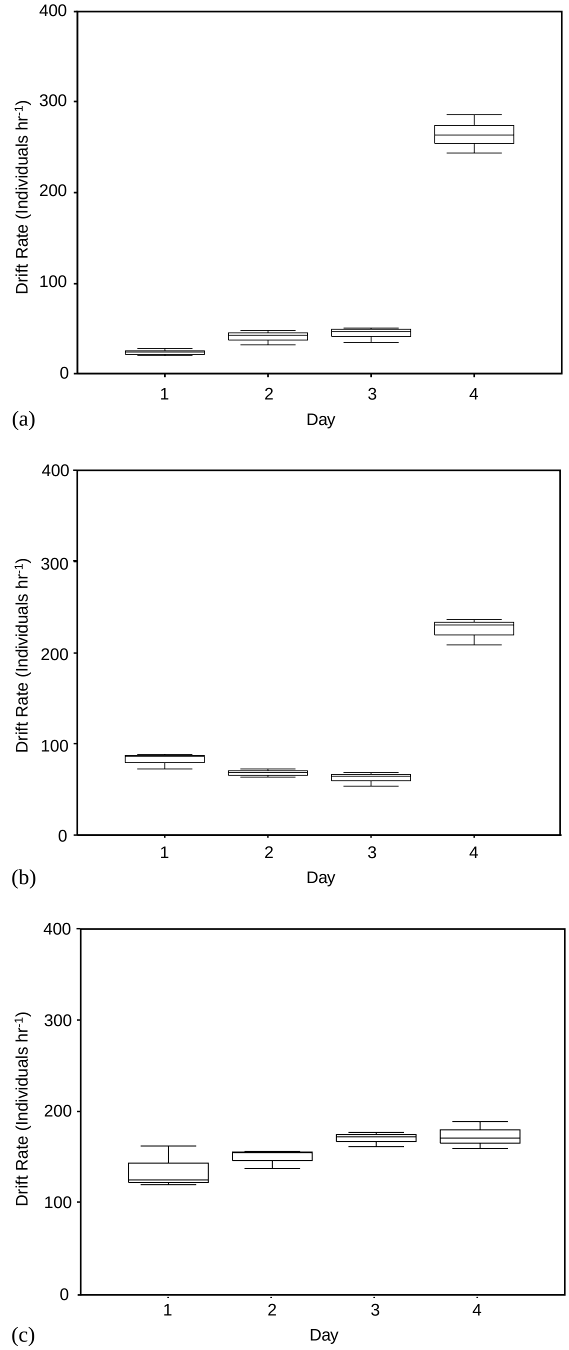

Fig. 4. Box plot of invertebrate drift rates for the 72-h period prior to tracer injection (day 1–3) and for the 24-h period following injection (day 4) at: (a) low, (b) intermediate, and (c) high flow in the River Holme.

drift rate of the cased caddisfly (Trichoptera) *Limnephilus flavicornis* was 8.4 individuals h^{-1} during high flow but rose to $24.3\,h^{-1}$ in the Little Stour following the introduction of the tracer ($t = -10.84$; $P < 0.001$) (Table 2). The most marked response was displayed by another cased caddisfly (Trichoptera) *Agapetus fuscipes*, which was absent from background drift samples at intermediate and high flows in the Little Stour, but had a

Table 2
Independent samples t-test of drift rates (individuals h^{-1}) before and after tracer injection for taxa occurring in more than once season and present in more than three samples

	(a) Little Stour			(b) River Holme		
	Low	Int.	High	Low	Int.	High
Lymnaea peregra	**X**	X	X	—	—	—
Potamopyrgus antipodarum	X	X	X	—	—	—
Oligochaeta	X	X	X	X	X	X
Erpobdella octoculata	—	X	X	—	—	—
Gammarus pulex	$t=-3.63$**	$t=-20.67$***	$t=-5.41$***	$t=-4.21$**	$t=-7.01$***	X
Asellus aquaticus	X	X	X	—	—	—
Leuctra fusca	—	—	—	X	$t=-9.01$***	X
Leuctra nigra	—	—	—	X	X	X
Nemurella picteti	—	—	—	X	$t=-6.39$***	X
Nemoura cinerea	—	—	—	$t=-5.63$***	$t=-9.51$***	X
Isoperla grammatica	—	—	—	X	X	X
Baetis rhodani	$t=-14.43$***	$t=-20.75$***	X	$t=-9.27$***	$t=-12.62$**	X
Baetis fuscatus/scambus	$t=-10.11$***	$t=-10.00$***	X	—	—	—
Ephemerella ignita	X	**$T=-6.88$***	X	—	—	—
Halesus radiatus	X	—	X	—	—	—
Limnephilus flavicornis	—	$t=-10.84$***	$t=-6.45$***	—	—	—
Potamophylax cingulatus	—	—	—	X	—	X
Agapetus fuscipes	X	**$t=-9.04$***	**$t=-11.67$****	—	—	—
Corixidae	X	X	X	—	—	—
Hydrophilidae (larvae)	X	—	X	—	—	—
Elmis aenea (adult)	—	X	X	X	X	X
Elmis aenea (larvae)	X	X	X	X	**$t=-8.03$***	X
Chironomidae	$t=-6.78$***	$t=-11.03$***	X	$t=-8.73$***	$t=-10.74$***	X
Simuliidae	X	X	X	X	$t=-4.14$**	X
Tipulidae	X	X	X	X	X	X

*$P<0.05$, **$P<0.01$, ***$P<0.001$.
Note: X = present; — = absent; for figures in bold equal variances cannot be assumed due to absence of taxa in background samples.

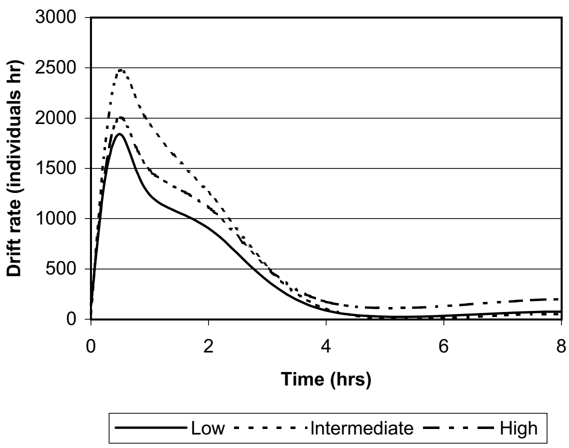

Fig. 5. Invertebrate drift rate (individuals h^{-1}) in the Little Stour during 8-h following the introduction of the tracer solution at low, intermediate and high discharges.

drift rate of 12.3 and 37.6 h^{-1}, respectively following the introduction of the salt solution. The larvae of the riffle beetle (Coleoptera) *Elmis aenea*, also absent in background samples, demonstrated a similar pattern in the River Holme at intermediate discharge with a drift rate of 20.7 h^{-1}.

Examination of common taxa that occurred during more than one flow condition and in at least three samples indicated that seven taxa in the Little Stour and eight taxa in the River Holme had significantly greater drift rates in the 24-h period following the introduction of the tracer solution (Table 2). Detailed examination of drift rates following the introduction of the tracer indicated that the greatest drift rates were recorded in 4 h following the onset of experimental conditions, before a reduction to typical daylight background levels (Figs. 5 and 6).

5. Discussion

Natural and managed riverine systems are subject to considerable physical, chemical and biological variability. At times this variability may be characterised by a frequency, intensity and severity outside a predictable range. This may cause disruption to the

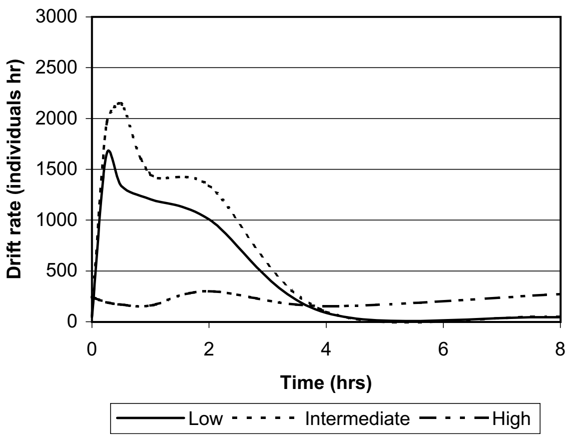

Fig. 6. Invertebrate drift rate (individuals h^{-1}) in the River Holme during the 8-h following the introduction of the tracer solution at low, intermediate and high discharges.

ecosystem, community, or population structure and change resource availability or the physical environment [33]. These events are termed 'disturbances' and may be the result of anthropogenic activity [10–12] or natural events such as droughts or floods [34,35].

In the current investigation, the short-term increase in salinity resulted in a significant increase in drift and clearly indicated the sensitivity of some instream invertebrate taxa to short-duration variations in water quality characteristics. However, the overall impact on the invertebrate community was limited when compared with the density of benthic taxa in both the Little Stour (4628–8730 m^{-2}) and River Holme (1424–4780 m^{-2}) during the study period. The results suggest that the impact of the tracer is confined to a short time period (<24 h) and that a relatively small number of taxa actually entered the drift. The experimental conditions should, therefore, probably be viewed as a low-magnitude disturbance with little spatial and temporal impact. The limited impact of the tracing exercise was highlighted during the high flow experiment in the River Holme, when it resulted in no significant difference in the drift rate. This probably reflects both the dilution of the saline solution associated with the increased discharge combined with behavioural responses of fauna to the high flows experienced during the previous days [36,37].

Some taxa are known to be highly sensitive to short-term changes in salinity [38,39], although others are known to be relatively resilient [40]. A significant increase in background salinity levels may result in physiological stress leading to abnormal invertebrate behaviour, abandonment of larval cases, drift and even death [41,38]. Elevated drift activity was clearly highlighted in the present study, where a number of taxa exhibited a marked drift response following the gulp injection (Table 2). This response was most marked in *Gammarus pulex*, Chironomidae, two-cased caddisfly (*Limnephilus falvicornis* and *Agapetus fuscipes*), and the larvae of the Coleoptera *Elmis aenea* with some taxa completely absent in the background drift samples. This clearly highlights that some taxa are more vulnerable to disturbances associated with saline pollution [39] and that there is a need to understand the response of individual species.

The response of fauna to short-term disturbances associated with the dilution gauging technique provides a valuable insight into pollution impacts, which may be difficult to simulate in artificial streams or flume studies. Disturbances associated with anthropogenic activities are widely known to influence aquatic invertebrate community structure and abundance [42,43]. Natural events, especially flow variability, can significantly increase lotic invertebrate drift [37], although recovery following such events can be quite rapid, depending on the magnitude of the event [44,45]. Disturbances associated with flume and laboratory experimental conditions have indicated increased drift and reductions in the densities of benthic fauna [1–3], although only a limited number of field studies have been undertaken to date [7,46].

Short-term increases in stream water conductivity may occur frequently in the urban environment as a result of runoff from roads following gritting with roadsalt during freezing conditions [47]. In some locations, this may have medium to long-term deleterious effects on instream communities, especially if repeated disturbances occur throughout the year [48,49]. There is clearly a need to develop management strategies to monitor and mitigate potential water quality impacts associated with salt on instream communities in sensitive areas [39].

The wider use of dilution gauging techniques in streams where the use of flow meters and other gauging methods is not possible or appropriate, raises a number of issues. First, there is frequently little or no background information regarding the vulnerability of small streams to pollution or details regarding the structure and composition of the ecological community. This is a particular concern in urban and industrial areas where drainage from roads and/or storm drains may occur [1,49]. Second, small 'headwater' and springhead locations frequently support populations of rare and/or endemic fauna of particular biodiversity and conservation interest [50,51]. An evaluation of the riverine habitats and ecology may, therefore, be required to determine the suitability of the technique in some locations to prevent unnecessary disturbance and degradation of the lotic ecosystem. The methodological approach outlined in this paper could be easily transfered to other locations and should be used to examine the impact of other water tracing compounds (e.g. rhodamine WT and fluorescein) widely used in the field [23].

6. Conclusion

1. The response of the aquatic invertebrate community to salt dilution gauging provides valuable insights into the response of fauna to short-term-pulsed pollution episodes, although further work is required to examine the response under a full range of flows and at different stages of faunal life history.
2. Salt dilution gauging significantly increased the drift of macroinvertebrates in the groundwater-dominated Little Stour River under all flow conditions and in the regulated River Holme under low and intermediate flows.
3. Some taxa which were absent in background samples (*Agapetus fuscipes* and larval *Elmis aenea*) or occurred in relatively low numbers (e.g. *Ephemeralla ignita* and *Limnephilus flavicornis*) were significantly more abundant in drift samples following the dilution gauging exercise using a saline tracer. This demonstrates that some taxa are more vulnerable to saline pollution.
4. The experiments had a relatively short-term effect on drift rates and no impact on benthic macroinvertebrate community abundance could be detected in any of the experiments undertaken. It is, therefore, unlikely that the technique has any long-term deleterious implications for invertebrate communities in most locations.
5. However, a survey of instream ecology is recommended prior to undertaking salt dilution gauging in small streams to identify any fauna of conservation interest and taxa which may be vulnerable to short-term exposure to water tracing compounds.

Acknowledgements

The authors gratefully acknowledge the assistance of Norman Wood, Dr. Maureen Agnew, Kim Wood, Prof. John Gunn, Dr. Richard Hughes, Paul Wanless, Kate Wood, Robin Kenyon, Laura Chapman, Richard Battey, Garry Rushworth and Nina King for field and laboratory assistance. This research was funded by the Division of Geographical Sciences at the University of Huddersfield. Thanks to Prof. I. Reid and M. Greenwood for comments on a draft of this manuscript and Mark Szegner for the production of the figures.

References

[1] Borchardt D, Statzner B. Ecological impact of urban stormwater runoff studied in experimental flumes: population loss by drift and availability of refugial space. Aquat Sci 1990;52:299–314.
[2] Crossland NO, Mitchell GC, Bennett D, Maxted J. An outdoor artificial stream system designed for ecotoxicological studies. Ecotox Environ Safety 1991;22:175–83.
[3] Mitchell GC, Bennett D, Pearson N. Effects of lindane on macroinvertebrates and periphyton in outdoor artificial streams. Ecotox Environ Safety 1993;25:90–102.
[4] Gordon ND, McMahon TA, Finlayson BL. Stream hydrology: an introduction for ecologists. Chichester: Wiley, 1992. 526pp.
[5] Hauer FR, Lamberti GA. Methods in stream ecology. New York: Academic Press, 1996. 674pp.
[6] Furse MT, Wright JF, Armitage PD, Moss D. An appraisal of pond-net samples for biological monitoring of lotic macro-invertebrates. Water Res 1981;15:679–89.
[7] Lake PS, Doeg TJ, Marchant R. Effects of multiple disturbance on macroinvertebrate communities in the Acheron River, Victoria. Aust J Ecol 1989;14:507–14.
[8] Poff NL. Why disturbances can be predictable: a perspective on the definition of disturbance in streams. J N Am Benthol Soc 1992;11:86–92.
[9] Giller PS. Floods and drought: the effects of variation in water flow on stream and rivers. In: Giller PS, Myers AA, editors. Disturbance and recovery of ecological systems. Dublin: Royal Irish Academy, 1996. p. 1–19.
[10] Gillespie WM, Rodgers JH, Dorn PB. Responses of aquatic invertebrates to a linear alcohol ethoxylate surfactant in stream mesocosms. Ecotox Environ Safety 1998;41:215–21.
[11] Hoiland WK, Rabe FW, Biggam RC. Recovery of macroinvertebrate communities from metal pollution in the South Fork and mainstem of the Coer d' Alene River, Idaho. Water Environ Res 1994;66:84–8.
[12] Whiles MR, Wallace JB. Macroinvertebrate production in a headwater stream during recovery from anthropogenic disturbance and hydrological extremes. Can J Fish Aquat Sci 1995;52:2402–22.
[13] Prat N, Toja J, Solà C, Burgos MD, Plans M, Rieradevall. Effect of dumping and cleaning activities on the aquatic ecosystems of the Guadiamar River following a toxic flood. Sci Total Environ 1999;242:231–48.
[14] Brittain JE, Eikeland TJ. Invertebrate drift—a review. Hydrobiologia 1988;166:77–93.
[15] Matthaei CD, Werthmuller D, Frutiger A. An update on the quantification of stream drift. Arch Hydrobiol 1998;143:1–19.
[16] Kreutzweiser DP, Sibley PK. Invertebrate drift in a headwater stream treated with permethrin. Arch Environ Contam Toxicol 1991;20:330–6.
[17] Bogatov VV. Classification of stream benthos drift. Hydrobiol J 1988;24:29–33.
[18] Water TF. The drift of stream insects. Ann Rev Ent 1972;17:253–72.
[19] Huhta A, Muotka T, Tikkanen P. Noctural drift of mayfly nymphs as a post-contact antipredator mechanism. Freshwater Biol 2000;45:33–42.

[20] Young MK, Rader RB, Belish TA. Influence of macroinvertebrate drift and light on the activity and movement of Colorado River cutthroat trout. Trans Am Fish Soc 1997;126:428–37.

[21] Wood PJ, Armitage PD. Sediment deposition in a small lowland stream—management implications. Regul Riv: Res Manage 1999;15:199–210.

[22] Wood PJ, Armitage PD, Cannan CE, Petts GE. Instream mesohabitat biodiversity in three groundwater streams under base flow conditions. Aquat Conserv: Mar Freshwater Ecosyst 1999;9:265–78.

[23] Wilson DA, Labadz JC, Butcher DP. Variation in time of travel in UK river systems—a comparative study. In: Foster IDL, editor. Tracers in geomorphology. Chichester: Wiley, 2000. p. 137–61.

[24] Lancaster J, Hildrew AG. Characterising in-stream flow refugia. Can J Fish Aquat Sci 1993;50:1663–75.

[25] Lyons RG. Identification and separation of water tracing dyes using pH response characteristics. J Hydrol 1993;152:13–29.

[26] Shaw EM. Hydrology in practice, 3rd ed. London: Chapman & Hall, 1994. 569pp.

[27] Gregory KJ, Walling DE. Drainage basin form and process. London: Edward Arnold, 1973. 456pp.

[28] Richards KS. Rivers: form and process in alluvial channels. London: Methuen, 1984. 361pp.

[29] Peckarsky BL. Sampling the stream benthos. In: Downing JA, Rigler FH, editors. A manual on methods for the assessment of secondary production in freshwaters. 2nd ed. Oxford: Blackwell Scientific, 1984. p. 131–60.

[30] Culp JM, Scrimgeour GJ, Beers CE. The effect of sample duration on the quantification of stream drift. Freshwater Biol 1994;31:165–73.

[31] Lancaster J. Scaling the effects of predation and disturbance in a patchy environment. Oecologia 1996;107:321–31.

[32] Pardo I, Armitage PD. Species assemblages as descriptors of mesohabitats. Hydrobiologia 1997;344:111–28.

[33] Resh VH, Brown AV, Covich AP, Gurtz ME, Li HW, Minshall GW, Reice SR, Sheldon AL, Wallace JB, Wissmar RC. The role of disturbance in stream ecology. J N Am Benthol Soc 1988;7:433–55.

[34] Palmer MA, Bely AE, Berg KE. Response of invertebrates to lotic disturbances: a test of the hyporheic refuge hypothesis. Oecologia 1992;89:182–94.

[35] Wood PJ, Petts GE. The influence of drought on chalk stream macroinvertebrates. Hydrol Proc 1999;13:387–99.

[36] Imbert JB, Perry JA. Drift and invertebrate responses to stepwise and abrupt increases in non-scouring flow. Hydrobiologia 2000;436:191–208.

[37] Palmer MA, Arensburger P, Botts PS, Hakenkamp CC, Reid JW. Disturbance and the community structure of stream invertebrates: patch-specific effects and the role of refugia. Freshwater Biol 1995;34:343–56.

[38] Williams DD, Williams NE. Aquatic insects in an estuarine environment: densities, distribution and salinity tolerance. Freshwater Biol 1997;39:411–21.

[39] Williams DD, Williams NE, Cao Y. Road salt contamination of groundwater in a major metropolitan area and development of a biological index to monitor its impact. Water Res 2000;34:127–38.

[40] Metzeling L. Benthic macroinvertebrate community structure in streams of different salinities. Aust J Mar Freshwater Res 1993;44:335–51.

[41] Goetsch P-A, Palmer CG. Salinity tolerances of selected macroinvertebrates of the Sabie River, Kruger National Park, South Africa. Arch Environ Contam Toxicol 1997;32:32–41.

[42] Armitage PD, Blackburn JH, Winder JM, Wright JF. Impact of vegetation management on macroinvertebrates in chalk streams. Aquat Conserv: Mar Freshwater Ecosyst 1994;4:95–104.

[43] Doeg TJ, Koehn JD. Effects of draining and desilting a small weir on downstream fish and macroinvertebrates. Regul Riv: Res Manage 1994;9:263–78.

[44] Matthaei CD, Werthmuller D, Frutiger A. Invertebrate recovery from a bed-moving spate: the role of drift versus movements inside or over the substratum. Arch Hydrobiol 1997;140:221–35.

[45] Milner AM. System recover. In: Calow P, Petts GE, editors. The rivers handbook, vol. 2. Oxford: Blackwell Scientific, 1994. p. 76–97.

[46] Lancaster J, Hildrew AG, Gjerlov C. Invertebrate drift and longitudinal transport processes in streams. Can J Fish Aquat Sci 1996;53:572–82.

[47] Robinson N, Barker J, Burgess W, Fellman L, Licha T, McArthur J, Heppell K, Salgado M, Hasan K, Bradford A, Durham M, Pinto L. The impact of highway drainage on groundwater quality. Jackson Environment Institute, University of East Anglia, Norwich, 1999. 148pp.

[48] Hogg ID, Norris RH. Effects of runoff from land clearing and urban development on the distribution and abundance of macroinvertebrates in pool areas of a river. Aust J Mar Freshwater Res 1991;42:507–18.

[49] Maltby L, Forrow DM, Boxall ABA, Calow P, Betton CI. The effects of motorway runoff on freshwater ecosystems 1: field study. Environ Toxicol Chem 1995;14:1079–92.

[50] Erman NA, Erman DC. Spring permanence, Trichoptera species richness, and the role of drought. J Kansas Entomol Soc 1995;68:50–64.

[51] Williams DD, Williams NE. Invertebrate communities from freshwater springs: what can they contribute to pure and applied ecology? In: Botosaneanu L, editor. Studies in Crenobiology: the biology of springs and springbrooks. Leiden: Backhuys, 1998. p. 251–61.