CL:AIRE

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technical bulletin

CL:AIRE technical bulletins describe specific techniques, practices and methodologies currently being employed on sites in the UK within the scope of CL:AIRE technology demonstration and research projects. This bulletin describes processes which may affect the fate of groundwater contaminants at the groundwater-surface water interface.

Accounting for the groundwater-surface water interface in contaminated land assessments

1. INTRODUCTION

The groundwater-surface water (GW/SW) interface is the subsurface transition zone between groundwater and surface water bodies. In rivers it is often called the hyporheic zone (Fig. 1) and is characterised by: 1) frequent mixing between stream water and groundwater; 2) often increased biogeochemical activity due to fluxes of dissolved oxygen, nutrients or organic carbon of stream or groundwater origin, and 3) its use as a habitat and potential refuge for stream (or *epigean*) and subsurface (or *hypogean*) invertebrates. In the context of discharge of contaminated groundwater to a stream, this mixing and biogeochemical activity is combined with increased reactivity of near-bed sediments and has the potential to naturally attenuate pollutants. Integrating the GW/SW interface into risk assessments of groundwater contamination could improve predictions of pollutant fate. In addition, the potential impact of contaminant fluxes on aquatic ecosystems in a receiving river should be evaluated as part of any risk assessment.

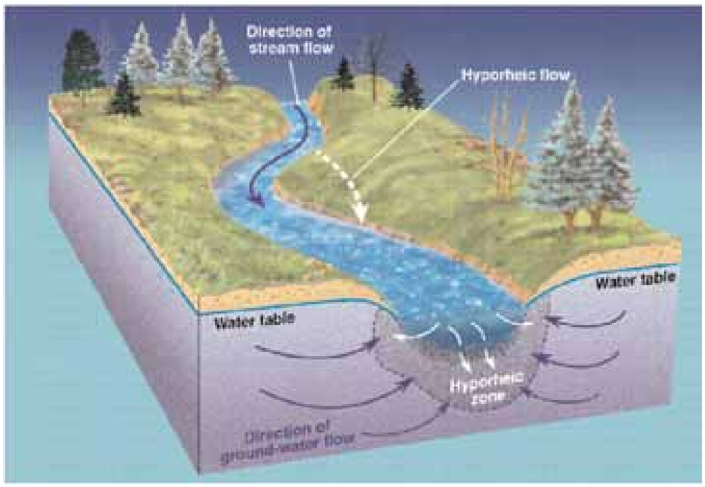

Figure 1. Schematic representation of the groundwater - surface water interface and hyporheic zone. (Reproduced with permission of USGS).

This bulletin aims to raise awareness of processes potentially affecting the fate of groundwater contaminants at the GW/SW interface, and to introduce monitoring and modelling solutions for this specific environment. It outlines legislation, gives an overview of subsurface flow and water exchange patterns at the GW/SW interface and describes processes affecting contaminant fate, emphasising their interdependency with biotic activity in the subsurface and stream environments. The implementation of monitoring strategies through the development of conceptual models is discussed and an introduction to river restoration practices and their potential impact on the management of groundwater contamination is given. Detailed references to the literature are not provided in this bulletin, but can be found in the Hyporheic Handbook (Buss et al., 2009).

2. LEGISLATIVE AND MANAGEMENT DRIVERS

In the UK, as elsewhere in Europe, the EU Water Framework Directive (WFD: CEC 2000) requires improved management, protection and restoration of rivers, lakes, estuaries and groundwater. The first River Basin Management Plans (RBMPs) were published in 2009. They include programmes of measures for each water body, and will lead into a second cycle of RBMPs in 2015. The status of surface water bodies is divided between chemical status (compliance with water quality standards) and ecological status (measure of anthropogenic impacts on ecosystems). For groundwater bodies, quantitative status is defined such that groundwater abstraction does not affect the flow required by groundwater dependent ecosystems to achieve environmental objectives. Threshold values of chemical parameters for groundwater are in part derived from the Environmental Quality Standards of surface water bodies, to ensure that groundwater does not contribute to the failure of the environmental objectives of associated rivers and streams. The WFD therefore requires integrated management of groundwater and surface water, putting the emphasis on stream ecosystem health and encouraging specialists of these two distinct environments to work together. For contaminated sites, the Environment Agency should be consulted to ensure that remediation objectives are consistent with the RBMP of the area and the objectives of the WFD (Environment Agency, 2006).

A tiered approach to environmental risk assessment is recommended in the UK, and most environmental risk assessments use the *source-pathway-receptor* concept. This approach identifies the nature of hazards (the source), the entities that could be harmed or polluted (the receptors) and the routes by which the receptors could be exposed to those hazards (the pathways). However, pollutant attenuation at the GW/SW interface or in the river is rarely considered, with compliance points often situated up-gradient of the surface water receptor (e.g. bank-side monitoring wells). This may be due to the perception that the heterogeneity of fluvial sediments creates great uncertainty in the prediction of contaminant fate. Monitoring techniques in this specific environment are furthermore perceived as costly and technically difficult to implement.

3. GENERAL RELATIONSHIPS BETWEEN STREAMS AND UNDERLYING AQUIFERS

Groundwater contaminants put stream ecosystems at risk in areas of groundwater discharge to the stream (gaining stream). Although this pattern of exchange is likely to be found in many contaminated sites, infiltration of stream water to the aquifer (losing stream) or limited exchange is observed in other areas. In general terms, patterns of exchange between streams and superficial aquifers will be driven by: 1) the distribution and magnitude of hydraulic conductivity within the channel and associated floodplain sediments; 2) the relationship between stream stage and adjacent water table, and 3) the geometry and position of the stream channel within the floodplain (Woessner, 2000).

Losses from a stream occur where the water table is lower than the stream stage. In the UK context, this is more likely to occur in headwaters or in the vicinity of pumped wells. Unsaturated seepage flow occurs beneath the channel in the higher reaches (Fig. 2a). Further downstream, saturated seepage flow occurs where the water table becomes closer to the surface (Fig. 2b and 2c). The location of the transition between unsaturated and saturated seepage flow can vary significantly with time, especially if the gradient of the river and the aquifer storage coefficient are low. Gains from aquifer to stream (Fig. 2d) typically occur in middle reaches, where the water table is higher than the stream stage. Discharge of groundwater can occur on the base and both sides of the stream - or on only one side (Fig. 2e, flow-through case). When stream stage and water table are at the same elevation, no exchange occurs (Fig. 2e, parallel-flow case). In lower reaches, low seepage flow rates towards the river are distributed along large areas, but low permeability infill in alluvial valleys can limit direct discharge of deep groundwater systems to the margins of the alluvial valley floor (Fig. 2f). Furthermore, the proximity of downstream discharge zones such as the sea can divert groundwater flow away from the stream.

The permeability of superficial deposits primarily depends on the type of sediments deposited or eroded in valley fills over long timescales. In the UK, sedimentological heterogeneity of valley fills can lead to strong differences in patterns of groundwater discharge towards rivers. For example, glacial deposits in floodplain or paleo-channels can create preferential flowpaths and discrete groundwater discharges to streams. Similarly, deposits of coarse pebbly alluvial gravel under the valley floor can focus discharge of deep groundwater directly to the stream. Furthermore, when deep formations are in close contact with the stream, the type of aquifer has a strong effect on the style of discharge of deep groundwater to a river. Discrete discharges will be more common in fracture flow systems (e.g. Chalk), with diffuse discharges more common in more intergranular flow dominated systems (e.g. sands and gravels).

4. NUTRIENT AND CONTAMINANT FATE AT THE GW/SW INTERFACE

The hydrological connectivity between streams and aquifers will allow the discharge of groundwater to streams or the infiltration of surface water into aquifers. Hyporheic exchange flow (HEF), or infiltration of stream water into near-stream sediments and return to the stream channel over relatively short distances, is superimposed on these processes. These flow patterns, and the reactivity of near-bed sediments, favour increased dilution, biodegradation, and adsorption or precipitation of mineral phases, which can reduce the concentration of contaminants in the subsurface.

Reactivity of riverbed and riparian sediments

The capacity of riverbed and riparian sediments to attenuate contaminants is mainly related to their grain size and mineralogical and geochemical characteristics such as fraction of organic carbon (f_{OC}), clay content, presence of carbonates and Fe or Mn oxyhydroxides. Smith and Lerner (2008) showed in a UK lowland context that the sorption capacity of the interface zone is greater than the deep Permo-Triassic sandstone aquifer. Biotic activity can locally modify the reactivity of riverbed sediments by modifying their permeability, controlling the availability of particulate organic matter and influencing the precipitation or dissolution of mineral phases. Furthermore, infiltration of fine sediment, composed in part by organic material and favoured by disturbance of the riverbed at high stream flow, can have similar effects. The enhanced reactivity of

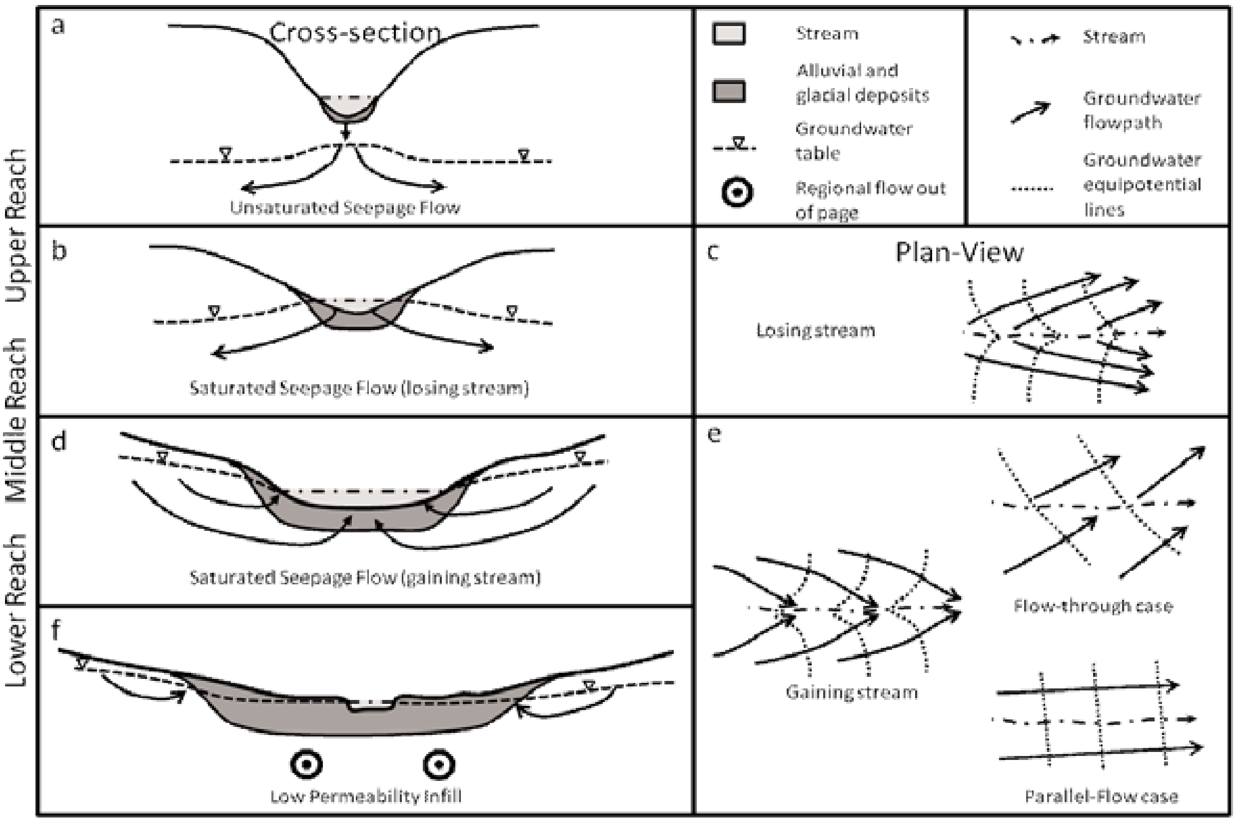

Figure 2. Possible geometrical relationships between rivers and groundwater flow from head water reaches to lower reaches (after Buss et al., 2009 and Woessner, 2000).

riverbed sediments is therefore temporally variable and bed erosion can re-mobilise contaminants sorbed over time.

Development of hyporheic exchange flow

In rivers, HEF is primarily driven by changes in hydraulic head at the riverbed, and by variation in the permeability and in the lateral and vertical development of the superficial deposits where this exchange occurs (Tonina and Buffington, 2009). These parameters control its extent, which can be up to several metres vertically and a few hundred metres horizontally. In the UK, extended HEF will potentially occur in large alluvium deposits, for example in lowland Chalk areas of the South-East; at the opposite, in upland areas of the Pennines, HEF will be more likely restricted to the immediate vicinity of the stream.

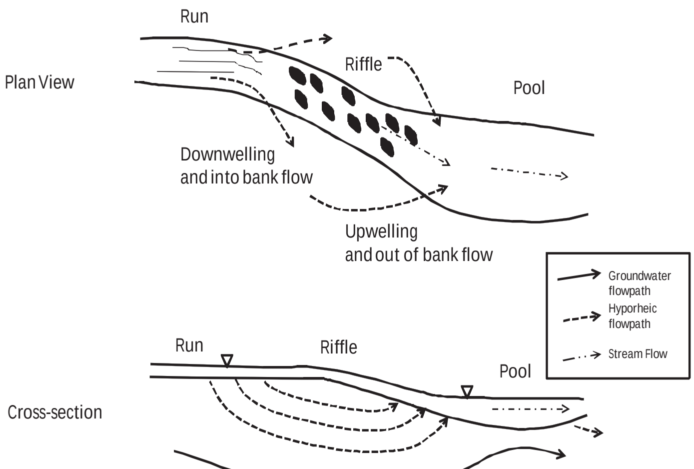

Figure 3. Plan view and cross-section of hyporheic exchange flow due to stream changes in hydraulic head across a run-riffle-pool sequence (after Dent et al., 2001).

Typically, channel geomorphology can induce HEF by creating head gradients in the stream. Pool-riffle-pool or run-riffle-pool sequences (Fig. 3) occur in rivers with relatively shallow water depth, where pronounced changes in riverbed elevation lead to similar changes in the free water surface (Tonina and Buffington, 2009). The downstream fall in head across these sequences drives flow across the riffle. Associated lateral flow in and out of the river banks often occurs and may involve greater fluxes than vertical flows. Based on a modelling study, Storey et al. (2003) found in this setting water that fluxes involved in hyporheic exchange were always less than 0.1% of the stream discharge. This value is nevertheless significant when compared to groundwater fluxes discharging towards streams. Irregularities of river bank and plan-form morphology lead to horizontal head gradients, with similar infiltration in the vertical and lateral direction across bars or meanders (Fig. 4). Numerical flow models show that the range and spatial distribution of hyporheic fluxes and residence times are strongly tied to river plan-form morphology.

Other mechanisms can induce HEF. They include: (1) spatial variation in head at the riverbed due to acceleration of flow over bedforms; (2) deflection of flowing interstitial water away from, or bending towards, the river-sediment interface encountering permeability contrasts (Fig. 5a); (3) changes in the size of the alluvial area and permeability contrasts in near-stream formations (Tonina and Buffington, 2009). As an example at large scale, outcrops of hard bedrock in a riverbed and the associated thinning of floodplain deposits can force flow to discharge into the stream: once flow has passed the constriction, inflow into the floodplain sediments is again likely (Fig. 5b).

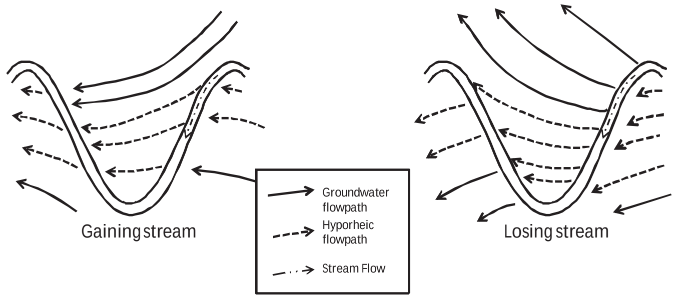

Figure 4. Hyporheic flow due to channel sinuosity in gaining and losing conditions (after Cardenas, 2009).

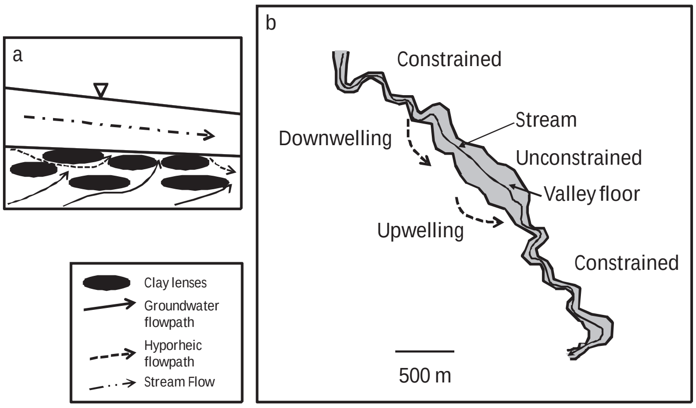

Figure 5. Hyporheic flow induced by a) heterogeneity of permeability distribution in the riverbed at small scale, and b) periodic constriction of the alluvial floodplain by bedrock outcrops (modified from Dent et al., 2001).

These processes are highly variable in time and space. As an example, HEF in winter can be strongly reduced under riffles by higher groundwater discharge and smaller head gradients within the stream (Storey et al., 2003). Variation in the permeability of the shallow sediments is also key, with HEF being strongly reduced in low permeability strata. Apart from spatial variation in sediment types, other processes, often temporally variable, can impact the permeability of the interface. They include: clogging - or *colmation* - of the riverbed by fine materials deposited from the water column and driven into the subsurface by gravity and fluid movement; deposition of fine sediments on the riverbed at the vicinity of submerged vegetation; *bioturbation*, in the form of root holes (but more often active disturbance of sediments by subsurface fauna); and impact of *water temperature* fluctuations in shallow sediments on hydraulic conductivity (e.g. water viscosity decreases with increasing temperature and hydraulic conductivity nearly doubles from 0 to 20°C).

The importance of biological activity at the GW/SW interface

Microbiologically mediated reactions

Streams and groundwater often have distinct redox conditions, as well as contrasting nutrient and organic carbon concentrations. The mixing of these two end-members at the GW/SW interface through development of HEF is one driver of the processes that can attenuate dissolved contaminants (Boulton et al., 2010). The majority of these reactions are destructive and mediated by microbes, which consume the chemicals in respiration and growth. These reactions mostly occur in biofilms, which are a complex matrix of polysaccharide excreted by bacteria, other products and a diversity of microorganisms adhering to surfaces.

Heterotrophic microorganisms, which use organic molecules as a carbon source, tend to dominate this environment. Different biodegradation reactions occur at different redox potentials, which in simple terms indicate which oxidant is dominating the system as an electron acceptor. The biodegradation of organic carbon generally consumes molecular oxygen first and then alternative terminal electron acceptors (TEAs) such as nitrate, oxyhydroxides of Mn and Fe, sulphate and carbon dioxide (Ibrahim et al., 2010). Organic contaminants can be biodegraded as electron donors or acceptors, depending on their character (Conant et al., 2004). As well as destroying organic pollutants, these processes will decrease the concentration of soluble electron acceptors (e.g. nitrate, sulphate) in water, but also release other solutes such as heavy metals or phosphate, associated with the organic matter or Mn and Fe oxyhydroxides (Fig. 6). *Chemolithotrophic microorganisms* can derive their energy from oxidation of inorganic materials like iron, sulphur, ammonium and nitrite. Their activity can lead to the production of nitrate as well as the formation of oxyhydroxides, therefore potentially enhancing the sediment reactivity.

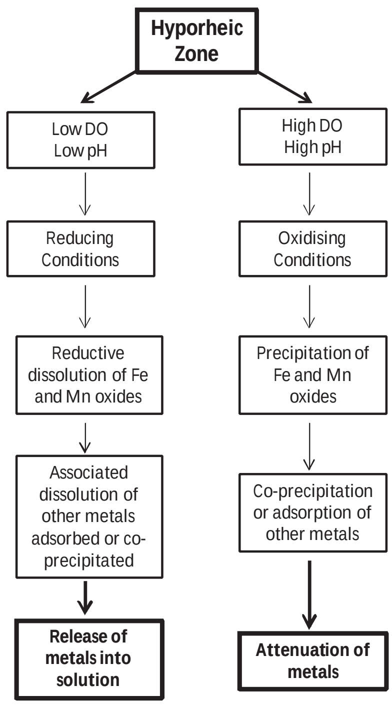

Figure 6. Generic redox and pH conditions in the hyporheic zone which can be highly variable, and lead to the attenuation or release of pollutants such as heavy metals (Gandy et al., 2007).

Biodegradation at the GW/SW interface will be spatially and temporally variable. A primary control of this variability is the solute transit time, due to variability in flowpath lengths, head gradient patterns and the hydraulic conductivity of superficial deposits. As an example, "freshly" infiltrating stream water favours aerobic biodegradation of organic contaminants and precipitation of oxyhydroxides of Fe and Mn, decreasing heavy metal concentrations through co-precipitation or adsorption mechanisms (Gandy et al., 2007). In contrast, more reducing conditions at a greater distance along hyporheic flowpaths (both laterally and vertically) can release heavy metals or reduce concentrations of other organic contaminants such as chlorinated compounds that are used as alternative TEAs in microbiologically mediated redox reactions (Ibrahim et al., 2010). Superimposed on these spatial patterns are temporal variations in solute concentrations, water temperature or groundwater fluxes which will influence contaminant retention and biodegradation rates. Furthermore, episodic burial of particulate organic carbon (POC) following a disturbance of the riverbed during storm events or from sediment aggradation, can provide dissolved organic carbon to drive the above mentioned biodegradation reactions.

The role of fauna and flora

The GW/SW interface provides abiotic conditions (e.g. flow velocities, amplitude of water temperature, substrate stability and hydrochemistry) which are different from those of stream and groundwater environments. This ecotone shelters a community of organisms collectively known as "hyporheos". The hyporheos includes *stygoxenes*, stream organisms only entering the interface through accidental infiltration; *stygophiles*, which have a greater affinity to hyporheic environments and actively exploit resources and the available habitat (e.g. during periods of high stream flow, drought or for protection from predators); and *stygobites* which are obligatory inhabitants of aquatic subsurface habitats. Studies have shown that these communities are strongly influenced by the physical and chemical conditions within the hyporheic zone (Boulton et al., 2010). They potentially play a key role in the context of contaminated groundwater as: 1) their active biotic interactions with benthic communities can impact the stream ecosystem health (and associated objectives of good stream ecological status) even when groundwater contaminants do not directly discharge into the stream; 2) bioturbation effects on sediments can influence transit times and development of microbiological processes; and 3) they play an important role in the cycling of organic matter. Indeed, hyporheic invertebrates redistribute organic matter through direct shredding of POC, including macrofaunal faecal pellets and biofilms, and predation between invertebrate species and by small and juvenile fish.

Aquatic macrophytes (macro-algae, liverworts, mosses and vascular plants) and riparian vegetation have the capacity to remove metals, nutrients or organic chemicals and to change redox conditions through uptake of carbon dioxide and release of oxygen in the subsurface. This capacity, as well as direct modification of sediment structure by the rooting systems, can also influence patterns of microbiological processes and contaminant attenuation at the GW/SW interface (Heppell et al., 2009).

5. DESIGNING AND IMPLEMENTING A MONITORING STRATEGY

A sound understanding of contaminant fate and transport and impacted area is critical in selecting an appropriate monitoring approach, in assessing the required financial and labour resources, as well as identifying issues of access to rivers and private lands. The development of a site specific conceptual model (e.g. Fig. 7) allows one to summarise current understanding and to design monitoring strategies. This model must be regularly refined to incorporate knowledge from new data. The initial aim of a monitoring scheme may not change, but the implementation often follows a trial-and-error process. Indeed, in river environments, it is not unusual to discover that monitoring devices have been damaged by high flow conditions, or that sampling networks are too sparse for the heterogeneity of the site. As with all monitoring strategies, examining existing sources of information through a desk study and conducting a walk-over survey can be invaluable. The following text focuses on the steps that should be followed in the field to characterise the site and ultimately inform better assessment and management of contamination risks. They include: 1) description of aquifer types and surface geomorphic and man-made features controlling the subsurface flow field; 2) assessment of directions of water exchange and quantification of water fluxes at different scales; and 3) establishment of the spatial and temporal variability in hydrochemical and biological conditions.

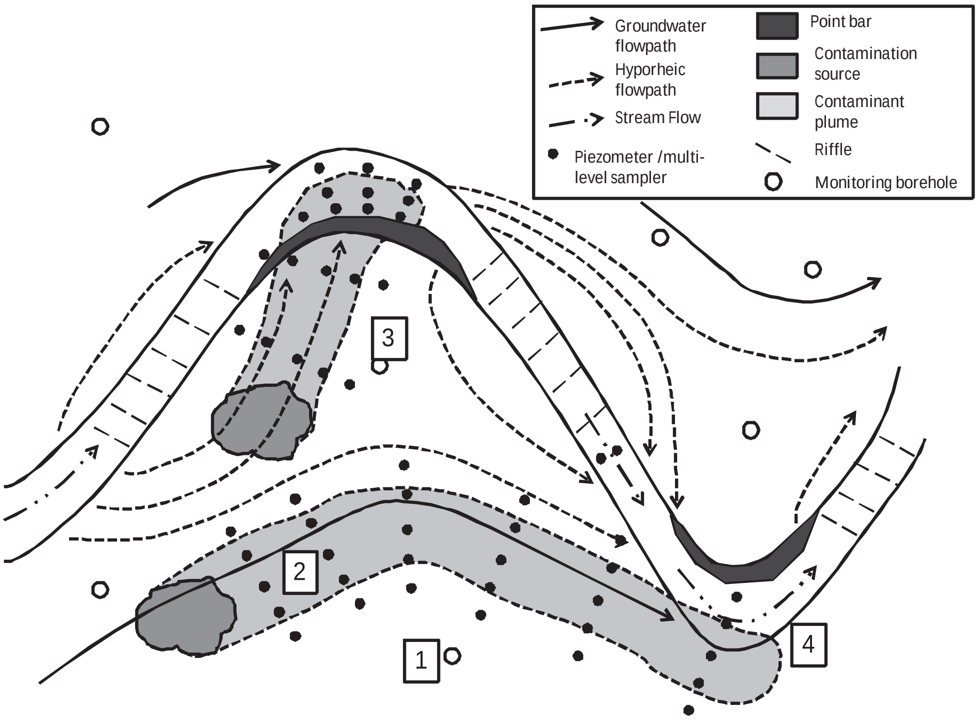

Figure 7. Conceptual model (plan view) of GW/SW interactions at a site affected by the discharge of contaminant plumes towards the stream. In this context, geomorphic features such as meanders and riffles are the main features responsible for the development of hyporheic flowpaths and their interaction with groundwater flowpaths.

Identifying directions of water exchange and quantifying water fluxes

An initial step in the building of the site specific conceptual model is to characterise the geology of the alluvial sediments under and beside the channel. Coring and other subsurface investigations, including surface geophysics, will support three dimensional mapping of the geology. This information, as well as measurements of the hydraulic conductivity (K) of superficial deposits, allow for the characterisation of aquifers, aquitards and the connectivity of units. A mapping of surface geomorphic and man-build features (e.g. meanders, sediment bars, weirs) potentially responsible for the development of HEF will complete this initial conceptual model of flow directions and pathways.

In a second step, directions of water exchange between the stream and aquifer can be assessed by comparing the elevation of the stream and water table in groundwater monitoring wells. Similar measurements can be made using *mini-piezometers* in the riverbed to locate the areas of loss and gain through the stream bed. There is usually a temperature contrast between surface and groundwater, which also provides a powerful way to determine areas of inflow and outflow. Thermal Infrared Imagery (aerial or terrestrial) can be used to detect zones of groundwater discharge, when such inputs modify the stream temperature. Distributed temperature sensing (DTS) detects small variations in temperature along a fibre-optic cable laid on the streambed, which can be related to the exchange of water in both directions at small scales (Henderson et al., 2009).

Finally, water fluxes can be estimated at different scales. In gaining streams, *hydrograph* or *baseflow methods* (Fig. 8) provide a spatially lumped estimate of the magnitude and timing of groundwater contributions. They will not give information about discharge of groundwater at the site (reach) scale, for example due to local variations in sediment permeability or hydraulic gradients between the stream stage and the water table. In well mixed streams, *in-stream tracer tests* can be interpreted using inverse solute transport models (e.g. OTIS-P, Runkel, 1998), to provide reach-averaged estimates of hyporheic exchange fluxes, hyporheic residence times and the dimensions of the hyporheic zone. *Flow gauging* methods using portable flow meters or in-stream tracer tests allow for the quantification of net gains or losses of stream water in specific reaches, provided the difference in flow is sufficiently large relative to the total flow in the stream. Several techniques are available to assess water fluxes at the scale of local geomorphological features (e.g. across a meander or riffle). Direct measurement can be made on the riverbed using *seepage meters*, which isolate a small area of the streambed and measure the flow across the enclosed area. Subsurface techniques include *mini-piezometers* to measure hydraulic gradients and flow direction in the subsurface, or between the stream and the riverbed. Fluxes are calculated following an estimation of K by slug testing or grain size analysis of sediments retrieved by coring (Fig. 9). Finally, water fluxes at the GW/SW interface can be determined using arrays of temperature probes; the approach relies on the fact that heat is transported by flowing water, and that daily fluctuation of stream temperature is detected at a greater depth in a losing reach than a gaining reach.

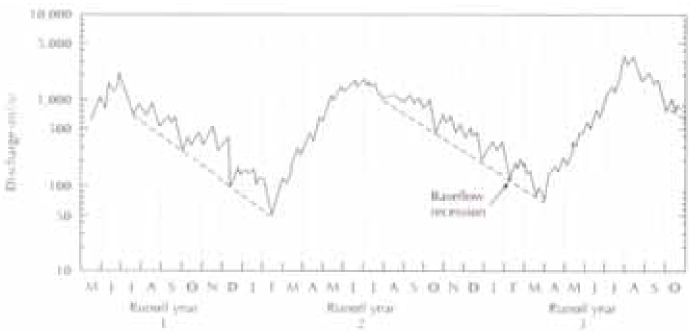

Figure 8. Stream hydrograph with baseflow recessions (Reproduced with permission).

Figure 9. Riverbed cores from the River Don and River Tern (UK). Sampling cores is key to a better assessment of the physical and geochemical features of subsurface sediments which influence water exchange, transit times, microbiological activity and sorption potential for contaminants. (Pictures from T. Ibrahim (R. Don) and N. Riess (R. Tern)).

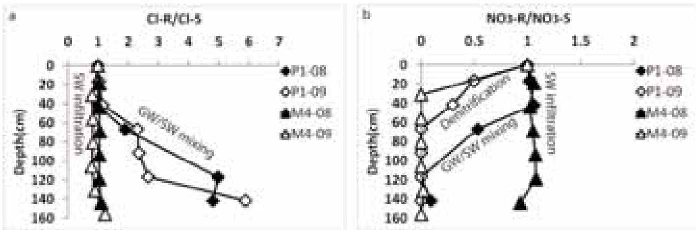

Figure 10. Vertical depth profiles of Cl (a) and NO_3 (b) concentration in riverbed porewater in a riffle, normalised to the stream concentrations (X-R and X-S indicate solute concentration in the riverbed porewater and in the stream, respectively). Cl ratios close to 1.0 at M4 indicate infiltration of stream water into the riverbed, without mixing with groundwater. Increasing ratios with depth at P1 indicate mixing of stream water with a deep groundwater component more concentrated in Cl. Patterns of NO_3 in winter 2008 (P1-08 and M4-08) indicate infiltration of stream water, with limited attenuation at M4, and conservative mixing with a groundwater component depleted in NO_3. In contrast, in Autumn 2009 (P1-09 and M4-09) NO_3 is highly depleted due to *in situ* biodegradation of organic matter (denitrification). (With kind permission from Springer Science+Business Media: Ibrahim et al., 2010, Figures 4 and 6).

Assessing contaminant fate and fluxes

The above techniques provide an understanding of the hydrological connectivity between streams and aquifers, However, they do not quantify the vertical and horizontal extent of dilution, retention and biodegradation associated with the development of hyporheic flowpaths (Fig. 7, zone 3), or the interaction of contaminated groundwater with riverbed and riparian sediments (Fig. 7, zones 2, 3 and 4).

Constant pumping can be undertaken in one or several wells to characterise a contaminant plume, in terms of average concentration and mass flow rates. The pumped water is regularly sampled for chemical analysis, and if additional control wells are sampled downgradient of the contaminant source, the attenuation rate may be estimated. As this approach tends to avoid issues related to the structural heterogeneity of the aquifer, it is called an *integral pumping test* (Kalbus et al., 2006). This type of investigation should nevertheless be undertaken with care, as it could perturb the subsurface flow field at the GW/SW interface.

Environmental tracers are chemical or isotopic compounds that occur naturally or enter the water cycle through human activity. *Conservative environmental tracers* are (relatively) inert substances (e.g. Cl, Br) that can differentiate subsurface flowpaths and sources of water, provided the end-members have different concentrations of the tracer. Vertical depth profiles of environmental tracers allow for characterising the depth of SW infiltration and GW/SW mixing (Fig. 10a). These distributions can be compared with those of reactive solutes, to assess the respective influence of dilution and retention/biodegradation processes on contaminant fate (Fig. 10b). The use of multilevel samplers installed in transects transverse to the direction of contaminant movement (Fig. 7, zones 2 and 3) can allow the evolution of these distributions along subsurface flowpaths to be assessed. Rates of attenuation or release of contaminants can be estimated by comparing fluxes of conservative and non-conservative solutes, possibly using artificial tracers to assess travel times. Finally, sedimentological and geochemical analysis of riverbed cores (Fig. 9) can provide estimates of the sorption capacity of the sediments and potential for contaminant retardation.

Water flow, hydrochemistry and transport of fine sediments in streams often varies at a small temporal scale, in response to the variability of hydrometeorological conditions or to diurnal biotic activity in the stream. This can impact head gradients in the stream and hyporheic exchange, fluxes of oxygen, carbon or nutrients to riverbed and riparian sediments. This small scale variation is often superimposed on seasonal dynamics, impacting water temperature and solute concentrations in the stream. Seasonal variation in stream stage and water table can also affect the general direction of exchange, water and solute fluxes. Sampling at different flow conditions and period through the year can provide a good understanding of the range in flow and hydrochemical conditions, and potential for contaminant attenuation at the GW/SW interface. When a higher temporal resolution is required, measurement probes (e.g. nitrate probes based on UV absorbance) connected to a logger or automatic samplers can be used in the stream and subsurface. They can be combined with passive sampler (e.g. resin accumulating solutes after diffusion through a hydrogel) which integrate concentration profiles over time at high spatial resolution. These devices have the advantage of not modifying the flow field by pumping (Kalbus et al., 2006).

Assessing biological activity and ecosystem response to contaminants

The assessment of solute patterns along flowpaths allows biogeochemical processes to be identified. Further experimentation can aim to better assess the composition and function of microbial communities, or to discuss their response to variations in contaminant concentrations or substrate type. *Experimental chambers* and microcosms (either constructed *in situ* and filled with undisturbed sediment or constructed *ex situ* and filled with disturbed sediment/substrates) can be used to assess biodegradation processes, metabolic parameters and rates in an environment that can be controlled. Similarly, *macroinvertebrate colonisation chambers* can be installed in the sediment to trap subsurface invertebrates and relate vertical variation in community composition to abiotic parameters. More generally, monitoring wells, piezometers or coring techniques can also be used to sample the subsurface biota (e.g. sampling macroinvertebrates, biofilms or free bacteria) or assess its activity (e.g. injection and subsequent pumping of nitrate in wells to assess potential for groundwater denitrification by denitrifier microbes). They can be carried out in parallel with *in-stream* sampling of invertebrates, benthic biofilms, vegetation or fish, with respect to predicted discharge of water to the riverbed. As an example, at zone 4 in Figure 7, the discharge of groundwater contaminants with limited hyporheic attenuation can place the stream ecosystem at risk. In this specific case, it is necessary to assess if groundwater interaction with riverbed sediments is enough to mitigate the contamination, or if some additional measures are needed to manage contaminants before they discharge to the stream.

Modelling approaches

Current risk assessment models for contaminated land and landfills (e.g. the Remedial Targets spreadsheet, ConSim, LandSim) (Environment Agency, 1996, 2003) have no default capability to include contaminant fate at the GW/SW interface. Some models can explicitly account for the hyporheic zone within the transport pathway (e.g. RISC, ESI's RAM) (ESI, 2005). All tend to use simple representations of contaminant attenuation such as linear adsorption and first order biodegradation. In addition, they only represent steady state contaminant transport along a one dimensional pathway, leading to inaccurate predictions where flow and solute patterns are multidimensional and vary strongly temporally. In principle, these analytical solutions or modelling tools can be extended to include different sorption and biodegradation rates in different zones. For example, a one dimensional model which predicts the pollutant concentration emerging into a river (C_{em}) from the concentration at the source (C_s) and an attenuation factor (f) which is a function of travel time in the aquifer (t_{aq}), the retardation factor for sorption (R_{aq}) and a first order biodegradation rate (λ_{aq}) can be written as

$$C_{em} = f(t_{aq}, R_{aq}, \lambda_{aq}) \qquad (1)$$

Extending this to include attenuation in the hyporheic zone gives the form

$$C_{em} = f(t_{aq}, R_{aq}, \lambda_{aq})\, f(t_{hz}, R_{hz}, \lambda_{hz}) \qquad (2)$$

where the subscript hz indicates that different values of the parameters apply in the hyporheic zone. The limitations of using such simple conceptualisations must be remembered when deciding whether to make them more complex.

Distributed groundwater models (e.g. MODFLOW and linked fate and transport codes) have been used to simulate local flow and solute patterns (Storey et al., 2003). Even if they often require significant amount of data, these models are useful to simulate different scenarios representative of the spatial and temporal variability of the GW/SW interface. Geostatistical simulations can account for the sedimentological heterogeneity of the aquifer (Fleckenstein et al., 2006). These approaches remain time and labour intensive.

Towards a re-definition of compliance points

Incorporating the GW/SW interface into groundwater quality management and remediation programmes is needed to comply with legislation that seeks to protect *in-stream* biotic communities. Due to the high spatial and temporal variability of flow and solute patterns at the interface, compliance points must be chosen that identify all areas where stream receptors are at risk with regard to contaminant discharges. Such designs should also identify areas where the potential for contamination attenuation at the interface can be used to reduce costs without putting the ecosystem at risk. Characterising general flow and solute patterns, as well as their expected variation at different seasons or stream flow conditions, using the monitoring and modelling methods described above, is therefore key to guide the installation of compliance points at the GW/SW interface.

6. RIVER RESTORATION

In the UK, contaminated land is often found in areas where rivers and floodplains have been severely degraded over a long period of time. In these areas, the stream ecosystem can suffer from high organic loadings or inputs of contaminants from the subsurface, linked to the development of agriculture, industrialisation and urbanisation. In this case, restoring the river habitat must obviously link with the remediation of groundwater, for example by the attenuation of contaminants in the subsurface before they discharge to the stream. River restoration also involves many other practices, often aimed at restoring the ecological connectivity of river systems, to limit floods or decrease fine sediment loads. As an example, removing *in-stream* obstacles such as weirs will restore the connectivity along the river network (e.g. allowing fish to move from downstream to upstream areas). Repositioning a dyke away from the edge of a river (Fig. 11) will provide room for the river to expand during high flow events, helping to decrease peak flows downstream. This will also favour the river ecosystem by providing new habitats, for example in temporary wetlands in the floodplain. These practices can impact groundwater flow patterns and contaminant fate in adjacent contaminated land considerably. Stream water infiltration into the floodplain can enhance the mobilisation of contaminated sediments and discharge of contaminated groundwater to the stream, enhancing the related risk to the stream ecosystem. It can also favour the development of HEF and therefore enhance the natural attenuation capacity of the floodplain aquifers, potentially reducing costs related to the remediation of contaminated groundwater. These examples show the necessity for contaminated land practitioners to account for the linkage between groundwater remediation strategies and river restoration practices. Due to the long history of alteration of river systems in the UK, single restoration actions often need to be complemented with additional intervention planned in the long-term (Palmer et al., 2005). This long-term river management can be carried out in parallel with a regular monitoring of the subsurface flow field and water quality at the GW/SW interface.

Figure 11. Process of repositioning a concrete lined channel into a new riverbed at Chinbrook Meadows (Greater London, UK), showing original concrete lined channel (left) and new channel floodplain protection (right). Picture courtesy of the River Restoration Centre (RRC).

7. CONCLUSION

The special biotic and abiotic conditions at the GW/SW interface potentially favour the enhanced attenuation of contaminants compared with groundwater systems. This includes enhanced dilution, higher retention on sediments and higher biodegradation rates caused by exchanges of solutes and sediments with the stream, and by interaction with stream and subsurface fauna and aquatic and terrestrial vegetation. The high spatial and temporal variability of these processes can nevertheless put both stream and subsurface ecosystems at risk of failing environmental objectives, such as those of the Water Framework Directive. Any monitoring strategy wishing to consider the GW/SW interface should therefore aim at managing both stream and aquifer environments. It should also consider the effects of measures on flow distributions, water quality patterns and ecological receptors. Due to the complexity of these environmental systems, local or distributed measurements, even when organised in a dense network, often need to be combined with techniques that allow the spatial or temporal integration of flow or solute patterns. Better management of the GW/SW interface often allows the reconnection of aquatic and terrestrial systems through the groundwater media. As illustrated by river restoration actions, this involves integrating the remediation of groundwater contamination into wider objectives, also accounting for ecological and socio-economic benefits.

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