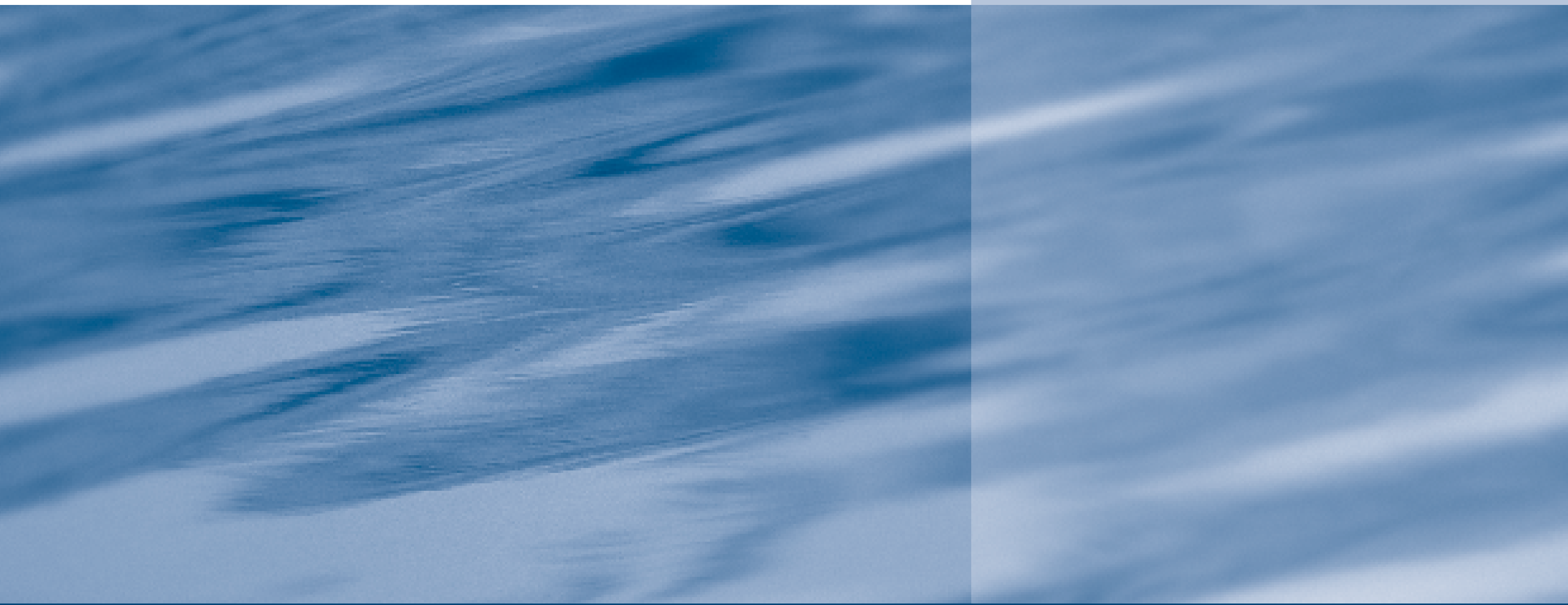

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Groundwater–surface water interactions in the hyporheic zone

Science Report SC030155/SR1

Environment Agency

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This report is the result of research commissioned and funded by the Environment Agency's Science Programme.

Published by:
Environment Agency, Rio House, Waterside Drive, Aztec West, Almondsbury, Bristol, BS32 4UD
Tel: 01454 624400 Fax: 01454 624409
www.environment-agency.gov.uk

ISBN: 1844324257

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Dissemination Status:
Public domain

Keywords:
Hyporheic zone, pollutant, attenuation, groundwater, surface water, interface, interaction, degradation

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Science Project Number:
SC030155/SR1

Product Code:
SCHO0605BJCQ-E-P

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Professor Mike Depledge Head of Science

Executive Summary

The Environment Agency has established a research programme on groundwater–surface water interactions, specifically aimed at pollutant attenuation processes at the interface of groundwater and surface waters, sometimes called the hyporheic zone. This research is needed to provide understanding of the processes that control water flow and pollutant flux between groundwater (aquifers) and surface waters (principally rivers and streams). It is, in part, a response to a requirement for improved understanding of the processes that act at this interface to be able to carry out the Water Framework Directive effectively.

This report reviews the current state of knowledge of the nature of the hyporheic zone, and of the processes that occur at the groundwater–surface water interface. It describes the importance of these processes for pollutant and/or nutrient attenuation and critically evaluates the current definitions and conceptual models of the hyporheic zone.

The biology and ecology of microbial, invertebrate and vertebrate interactions in hyporheic sediments are reviewed, to present a basic understanding of hyporheic ecosystems to river ecosystem managers. Currently, hyporheic and groundwater ecosystems are largely ignored by both hydrogeologists (who focus on water quality and water resources) and freshwater ecologists (who focus on benthic and in-stream ecosystems).

The hyporheic zone is a critical interface between groundwater and surface water environments and is shown to be a dynamic ecotone characterised by steep chemical and biological gradients. Its ecology is an important component on the lotic food web and has a vital role in the cycling and processing of energy, carbon and nutrients. The geochemical and microbial properties of the hyporheic zone are such that it presents significant, but currently little investigated, opportunities for pollutant attenuation that may reduce the impacts of polluted groundwater on a dependent river ecosystem, or vice versa.

The report concludes by identifying knowledge gaps in the existing literature and research priorities that need to be addressed within the context of an integrated catchment management to achieve Water Framework Directive objectives.

Contents

1 Introduction

1.1 Objectives of the Environment Agency's hyporheic zone research programme

The Environment Agency has established a research programme on groundwater–surface water interactions, specifically aimed at the pollutant attenuation processes at the interface of groundwater and surface waters, sometimes called the hyporheic zone. This research is needed to provide understanding of the processes that control water flow and pollutant flux between groundwater (aquifers) and surface waters (principally rivers and streams). It is, in part, a response to new legislation that requires a more integrated approach to the management of the aquatic environment.

The Water Framework Directive (WFD; 2000/60/EU) sets out new and wide-ranging environmental goals (Council of the European Community 2000). One of the primary goals of the WFD is to ensure that water bodies achieve the environmental objectives set out in Article 4 of the directive. These goals include the attainment of good chemical and ecological status for surface water bodies, and of good chemical and quantitative status for groundwater bodies.

The WFD states that if a surface water body fails to achieve good status (i.e., it is poor status) as a result of interactions with, or discharges or abstraction from, an associated groundwater body, then the groundwater body thereby fails to achieve good status. If a water body is of poor status, member states must put in place a Programme of Measures to ensure that its condition is improved so that the water body achieves good status.

Most previous regulatory and management approaches to environmental protection were aimed at particular environmental compartments, or industry and/or work sectors. As a result, past research was largely aimed at understanding the behaviour of water and pollutants in aquifers (e.g., to develop the science needed to carry out the 1980 EC Groundwater Directive) or the behaviour of substances within rivers, which was necessary to carry out the 1974 Dangerous Substances Directive. The policy development, scientific research and management of groundwater and rivers were largely undertaken by separate groups within the relevant organisations. Rivers and aquifers were often considered as separate, essentially unconnected systems. The WFD changes the emphasis on environmental protection and establishes the requirement for a comprehensive approach to integrated catchment management.

As a consequence of the hitherto disaggregated approach, the understanding of flow and pollutant behaviour within aquifers and within river channels is relatively good, but the understanding of processes that occur at the interface of these systems is poor. The Environment Agency's research on the hyporheic zone seeks to establish a fundamental understanding of the controls on flow and pollutant behaviour at the interface of groundwater and surface waters. The term 'hyporheic zone' (*Figure 1.1*) is used to refer to the zone beneath and adjacent to a river or stream in which groundwater and surface water mix. However, existing definitions and conceptual models of this zone vary greatly between scientific disciplines. A series of definitions and conceptual models that incorporate the main hydrological, hydrogeological, microbiological and ecological processes and functions will be developed by the project. This will aid the various workers from different scientific disciplines with an interest in processes that occur at the groundwater–surface water interface to better interpret the resulting impacts on hydrochemistry, sediment chemistry and invertebrate and fish ecology.

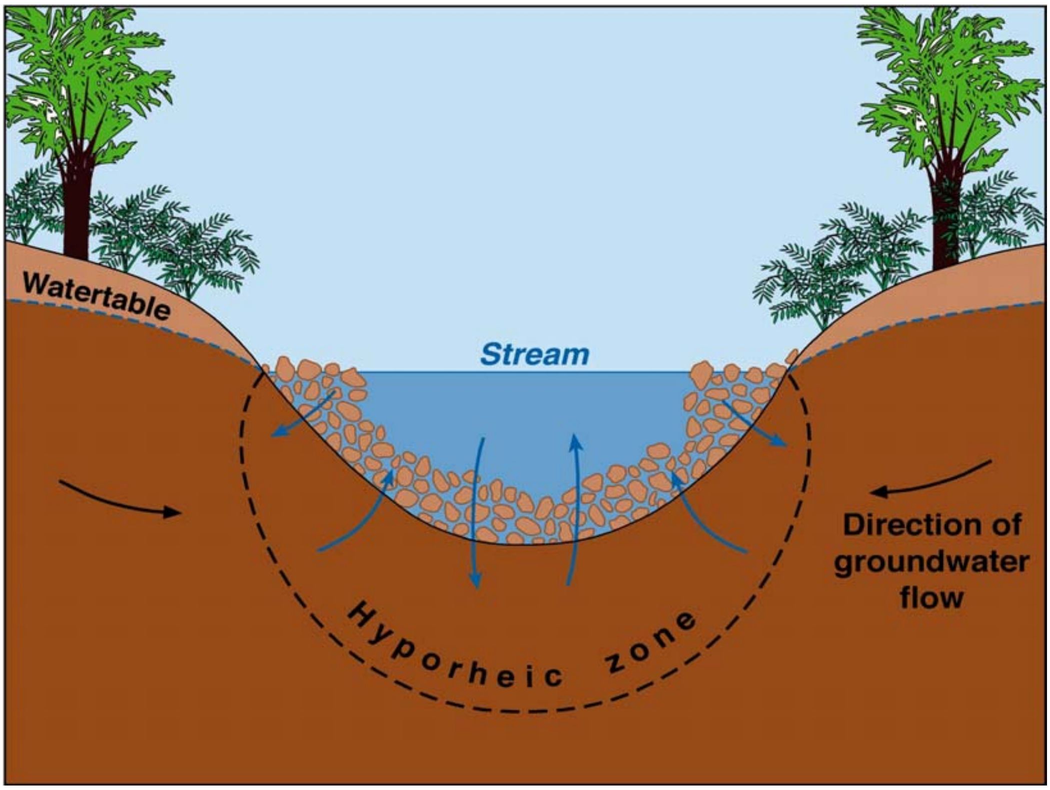


Figure 1.1 The hyporheic zone (after United States Geological Survey 1998).

1.2 Objectives of the report

This report reviews the current state of knowledge of the nature of the hyporheic zone, and of the processes that occur at the groundwater–surface water interface. It describes the importance of these processes for pollutant and/or nutrient attenuation and critically evaluates the current definitions and conceptual models of the hyporheic zone. The report then describes identified knowledge gaps in the existing literature that need further research. The basis for prioritisation of future work by the Environment Agency is a requirement for an ability to predict the effects of hyporheic zone processes on the achievement of WFD objectives, which requires the:

- ability to assess mass flux across the groundwater–surface water interface;
- ability to predict the significance of attenuation processes within the hyporheic zone;
- ability to link hyporheic and/or benthic chemical conditions and ecological health;
- development of reliable and transferable conceptual models of flow and attenuation processes at the groundwater–surface water interface.

2 Regulatory and management context

2.1 New regulatory objectives

Legislation that relates to environmental pollution has existed in England since medieval times, in the form of civil proceeding precedents that concern nuisance. However, the first statutory Acts of Parliament to control water abstraction and pollution were not introduced until the 1960s and 1970s. Anti-pollution legislation in Europe developed throughout the 1970s and 1980s, principally as a consequence of priorities of the European Commission and Parliament. Much of this legislation, such as the 1974 Dangerous Substances Directive, 1980 Groundwater Directive and 1990 Nitrates Directive, dealt with specific parts of the environment, specific industries or specific substances. The different directives were not integrated and, in some circumstances, not consistent with each other. The early environmental directives were typically prescriptive and based on the achievement of defined water quality standards. In many instances the defined water quality standards were not based on scientific rationale, but rather on a political aspiration to achieve zero pollution, which led to the establishment of some standards equal to laboratory detection limits for the relevant compounds.

During the 1990s it became increasingly apparent that a more integrated and comprehensive approach to environmental management was needed. The development of more comprehensive risk-assessment and management approaches occurred during the 1990s (e.g., National Rivers Authority 1992, Department of the Environment, Transport and the Regions *et al*. 2000). These regulatory approaches provided an improved basis for targeted and prioritised action on prevention and cleaning up of environmental pollution. Developing these approaches, the most recent European directives and north American laws are increasingly risk-based. They stress the need to assess the likelihood and magnitude of adverse effects on the wider environment, and to develop more sustainable risk-management approaches that balance costs and benefits.

The WF D (Council of the European Community 2000) is the most significant piece of new legislation on water issues in Europe in recent decades. It seeks to integrate environmental management of the different environmental compartments, such as groundwater; rivers, estuaries and wetlands (*Figure 2.1*), and to set risk-based objectives to protect and improve water quality and water resources (Environment Agency 2002). The WFD contains a timetable for implementation that includes plans to replace existing but superseded directives by 2013.

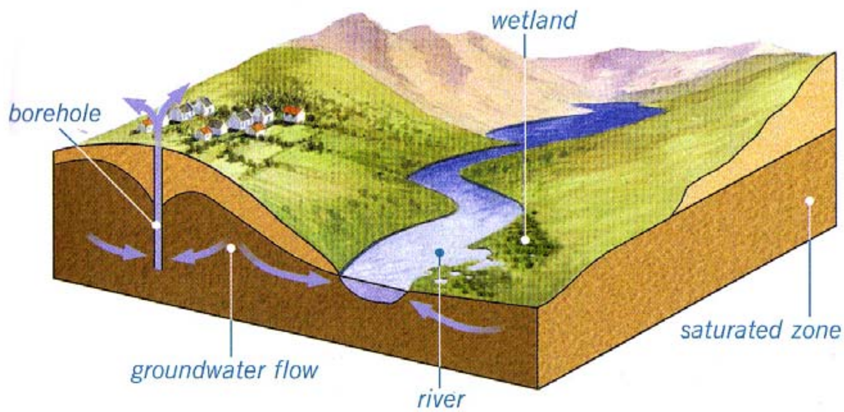


Figure 2.1 WFD requires integrated catchment management (after Environment Agency 2002).

Further initiatives at European Commission and UK central government (Department for Environment, Food and Rural Affairs, Defra) levels are in place to support the implementation and further evolution of the WFD and related policies. Development of policy and research priorities for new directives on soil and sediment systems is currently being considered. The draft Soil Thematic Strategy and the task group reports of the European network on sediments, SedNet, recommend approaches for risk-based management of soil and sediment systems in a manner similar to the WFD for water.

At the UK-level, Defra recently transposed the WFD into UK legislation and similarly carried out other related risk-based environmental legislation, including Part IIA of the Environmental Protection Act 1990 (for contaminated land), the Water Act 2003 (for water management) and the Pollution and Prevention Control Regulations 2000 (for prescribed processes). Environmental regulations are increasingly risk-based and require improved understanding to be carried out effectively, particularly where they relate to processes that span the boundaries of environmental compartments that were previously managed separately.

2.2 Water Framework Directive

The Water Framework Directive came into force on 22 December 2000 and established a new legislative regime for the integrated management, protection and improvement of Europe's rivers, lakes, estuaries and groundwater (Environment Agency 2002). It sets out a series of environmental objectives that must be met within defined timescales. Following transposition into UK law, the initial characterisation of water bodies and economic analysis of water usage was completed in December 2004.

Following a further series of defined stages, a River Basin Management Plan (RBMP) and a programme of measures specific for each water body must be prepared by 2009. Further characterisation, review of the effectiveness of the programmes of measures and updated RBMPs are then to be produced on a six-yearly rolling programme. Environmental objectives, including good status objectives for all water bodies, must be met, where feasible, by 2015.

The status of a surface water body will be determined by the poorer of its chemical or ecological status. Chemical status is determined by compliance against water quality standards prescribed by the European Commission, while ecological status is a measure of the anthropogenic impacts on a water ecosystem. Ecological status is determined by comparison of current ecological conditions against 'reference conditions' that would exist in a pristine surface water body of similar type (altitude, geology, size, etc.).

The development of practical and comprehensive methods to determine ecological status is not trivial. The transferability of approaches between rivers and streams, headwaters and alluvial plains, and naturally oligotrophic and nutrient-rich systems, needs to be considered. Methods applied in different member states are being standardised by a process of intercalibration. The interactions between chemical concentrations and ecological health, as described by its the ecological health's status determination, are unclear. Current approaches to determine river ecosystem health are generally based on investigation of the benthic invertebrate community (Extence *et al.* 1999). Assessment methods that include hyporheic organisms have not been developed (Boulton 2000b) and the fundamental understanding of ecosystem response to specific pollutant concentrations within a complex and dynamic environment is poorly developed.

There are 7816 surface water bodies in England and Wales (Environment Agency 2004). The five most numerous threats to achievement of good surface water status are geomorphological changes (river engineering) 43 percent, nitrate 29 percent, phosphate 28.7 percent, pesticides 17 percent and sedimentation 16.4 percent.

Groundwater status comprises groundwater chemical status and groundwater quantitative status. Methods to assess groundwater chemical status remain unresolved and are the subject of continuing discussion at the European Commission as part of negotiations over a new groundwater directive, which is required by Article 17 of the WFD. Nevertheless, member states made initial determinations of groundwater bodies at risk of failing to achieve good status at the end of 2004. They have used a range of methods, mostly based on compliance with existing directives and national legislation.

The interactions between groundwater and surface water bodies is an important aspect that needs to be understood to properly assess the impacts of pollutants in groundwater on dependent surface waters, and vice versa. The WFD requires that groundwater bodies and surface water bodies be managed in an integrated manner, together with other protected areas, such as designated wetlands. The processes that control water flow, pollutant migration and ecological response at the interface (*Figure 2.2*) are vital to this assessment and are poorly understood.

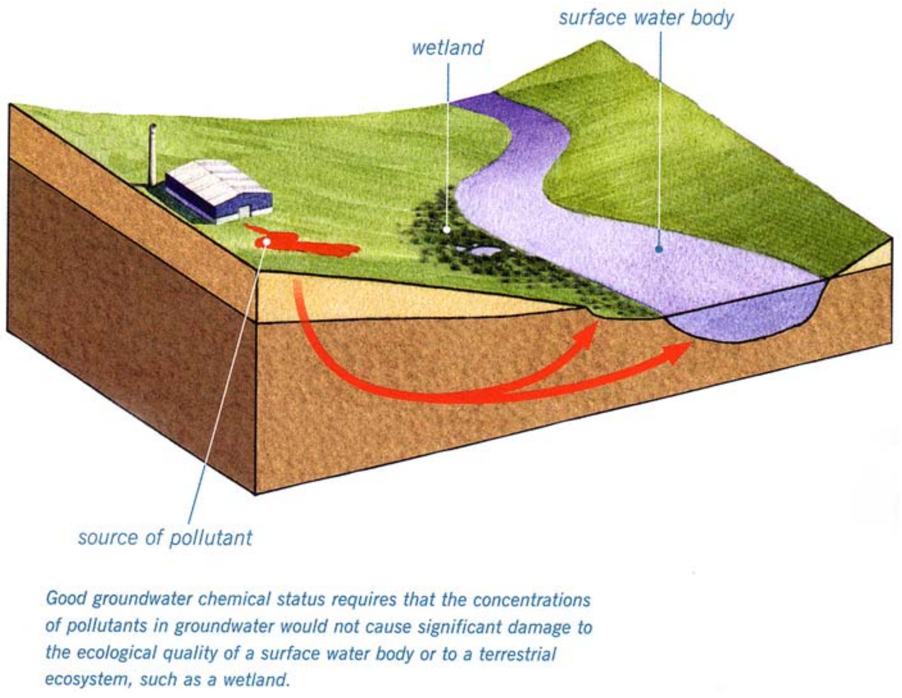


Figure 2.2 WFD requires assessment of groundwater–surface water interactions (after Environment Agency 2002).

Groundwater quantitative status is a measure of the sustainability of water use, in terms of balancing human and ecological needs for water (see *Figure 2.3*). With regard to hyporheic zone issues, the principal areas of interest are the controls on flow across the hyporheic zone (and implications for quantitative status), pollutant attenuation in the hyporheic zone that limits the chemical impacts between the adjacent water bodies, and hyporheic ecology as a component of river ecology.

There are 356 groundwater bodies in England and Wales (Environment Agency 2004). The most frequent reason for groundwater bodies being judged to be at risk of failing their environmental objectives are nitrate 53 percent, over abstraction 49 percent, diffuse urban pollution 22 percent, pesticides 20 percent and phosphorus 17.4 percent.

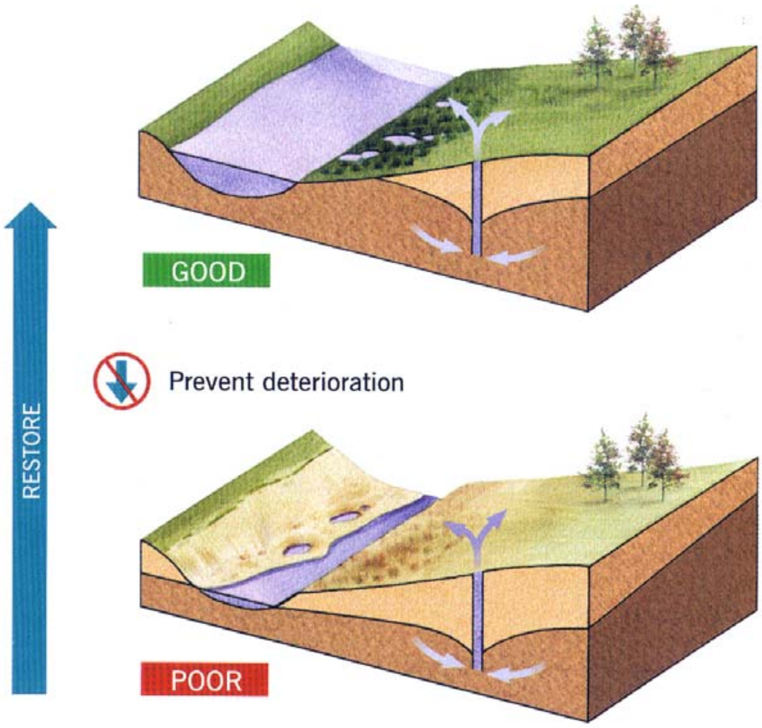


Figure 2.3 Groundwater quantitative status classification and objectives (after Environment Agency 2002).

2.3 Environmental risk assessment

The great majority of contamination risk assessments undertaken by landowners or regulatory bodies are site-specific. Most assessments are undertaken in preparation for the redevelopment of brownfield sites, or to assess sites and/or processes that could pose a future pollution hazard. A tiered approach to risk assessment is recommended in the UK (Department of the Environment, Transport and the Regions *et al.* 2000), and most environmental risk assessments follow a *source–pathway–receptor* analysis method. This approach seeks to identify 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), and to understand the likelihood and consequences of an adverse effect.

Initially as a result of environmental regulation, but increasingly through corporate management because of liabilities associated with both financial and legal risks, the *source–pathway–receptor* approach has been restricted to certain environmental compartments (e.g., pollutant behaviour processes within an aquifer) or site boundary limits (assessing risks associated with contaminants under a piece of land, but ignoring surrounding land owned by others). The Environment Agency's recommended approach for assessing the risks from contaminated soils to controlled waters (Environment Agency 1999) includes the assessment of groundwater pollutant plumes on surface water bodies. However, in reality in-river dilution and attenuation is rarely applied, and compliance is assessed either at a site boundary (landowner compartmentalisation) or at an arbitrary compliance borehole upgradient of the surface water receptor (*Figure 2.4*). The latter generally occurs as a result of a reluctance on the part of problem holders to investigate riparian and/or hyporheic processes on land that they do not own, and a lack of awareness of the attenuation capacity that may exist in the near-river environment. In the USA methods for considering pollutant flux and the groundwater–surface water interface, and the impacts on hyporheic fauna, are being developed in a number of states (Biksey and Gross 2001).

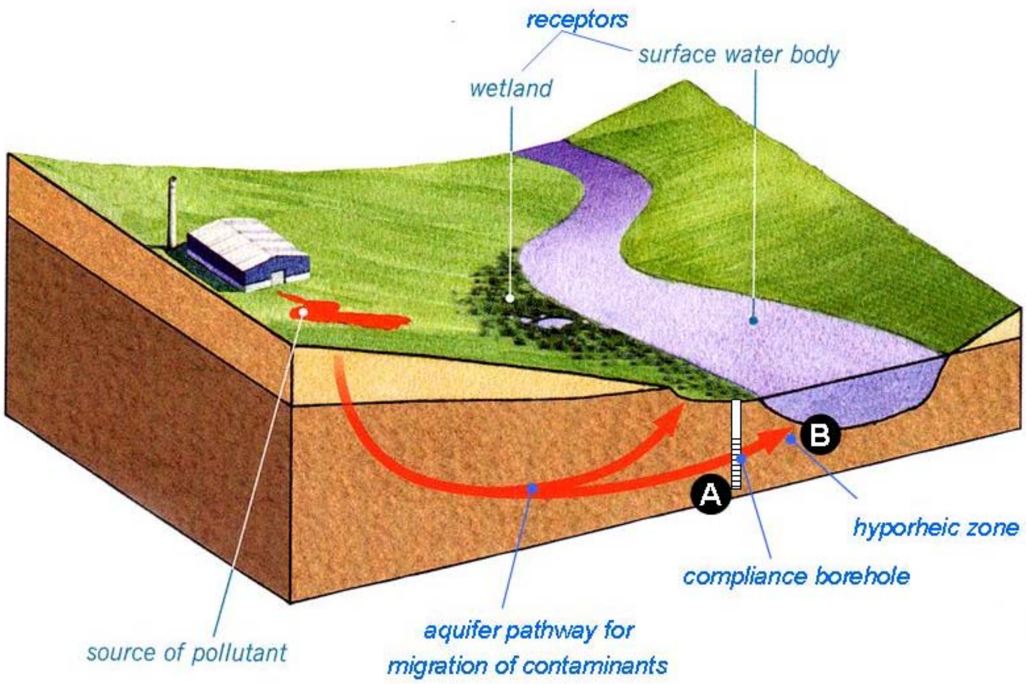


Figure 2.4 Typical contaminated-site assessment using the *source–pathway–receptor* framework. Use of a compliance-monitoring borehole adjacent to the stream (A) excludes any potential attenuation capacity within the hyporheic zone (B). (Based on Environment Agency 2002.)

The next big challenge for environmental risk assessors is to develop risk assessment and management approaches to catchment-scale processes that adopt the rationale inherent in the site-specific assessment methods. Translation of point-source risk assessment techniques to widespread and diffuse pollutants is likely to present many difficulties, but it is needed to tackle the major pollution pressures that affect European waters. At the catchment scale, landowner boundaries and environmental compartmentalisation are meaningless concepts in risk assessment. They will, however, need to be considered at the risk management stage when responsibility for remedial actions is apportioned.

Understanding interfaces, such as those at the boundary of soil and groundwater and at groundwater–surface water boundary, will be critical. This is in part because of the current lack of understanding of processes that act across these interfaces and of the significance of these ecotones. The hyporheic zone has been described as having a pivotal role in the functioning of lotic ecosystems (Palmer 1993), but it is probably unknown to the great majority of experts in the management of agricultural and contaminated land in the UK.

2.4 An aspiration for integrated catchment management

Although the WFD sets out a legal requirement for a more comprehensive approach to environmental (water) management, various organisations have already started moving towards carrying out more integrated regulatory and management approaches. For example, the Environment Agency's Catchment Abstraction Management Strategy (CAMS) seeks to more closely integrate the management of surface and groundwater resources. However, it fails to tackle the integrated management of water quality and water resources. Similarly, the Environment Agency's approach to the management of contaminated land seeks to integrate the assessment of risks to controlled waters with methods for human health (Department of Environment, Food and Rural Affairs and the Environment Agency 2004), but currently fails to fully consider other risks, such as those to terrestrial, benthic or hyporheic ecosystems.

While any assessment framework will necessarily need to concentrate assessors' efforts on the most important processes and mechanisms likely to drive decision-making, the current reliance on compliance against concentration thresholds at arbitrary points in soil or water needs to evolve into an approach that more closely links environmental outcome and effect with exposure, probably in the form of limits on contaminant flux.

2.5 Existing research initiatives

A comprehensive review of existing field-based research infrastructure and facilities for studies of groundwater – surface water interactions and hyporheic processes (Binley 2005) has been undertaken recently on behalf of the Environment Agency. Interested readers should see Binley (2005) for further details.

3 The hyporheic zone

3.1 Definitions and conceptual models of the hyporheic zone

The term hyporheic is derived from Greek roots – *hypo*, meaning under or beneath, and *rheos*, meaning a stream (*rheo* means to flow). A number of definitions for the hyporheic zone exist, the basis for which largely reflect the academic discipline in which they have arisen. The majority of the current literature is presented by ecologists, who have concentrated on the role of the hyporheic zone as a habitat and refuge for freshwater invertebrate fauna and as a location for salmonid egg development. Probably second in scale, the hydrology literature considers processes that control water flow within the hyporheic zone in terms of the exchange of water between the river channel and the adjacent hyporheic sediments. A smaller pool of literature exists within the hydrogeological, geochemical and geomorphological fields.

Some common themes in the definitions of the hyporheic zone in the literature are:

1. It is the zone below and adjacent to a streambed in which water from the open channel exchanges with interstitial water in the bed sediments;
2. It is the zone around a stream in which fauna characteristic of the hyporheic zone (the *hyporheos*) are distributed and live;
3. It is the zone in which groundwater and surface water mix.

White (1993) defined the hyporheic zone as ‘the saturated interstitial beneath the stream bed, and into the stream banks, that contain some proportion of channel water, or that have been altered by channel water infiltration’. Valett *et al.* (1993) describe the hyporheic zone as the ‘subsurface region of streams and rivers that exchanges with the surface’, whereas Triska *et al.* (1989) sought to more precisely define the hyporheic zone as that part of the sub-surface in which both surface and groundwater are present, but surface water exceeds 10 percent of the total volume.

The majority of authors in the freshwater ecology literature emphasise the importance of the hyporheic zone as a dynamic ecotone (Sabater and Vila 1991). For example, Boulton *et al*. (1998) describe the hyporheic zone as ‘an active ecotone between surface and groundwater’, in which water, nutrients and organic matter exchange occurs as a result of hydraulic and chemical gradients, topography and sediment lithology. The ecological significance of the groundwater–surface water interface was first recognised by Orghidan (1959), who coined the term ‘hyporheic biotope’ for the distinct habitat he observed. Stanford and Ward (1993) describe a ‘hyporheic corridor concept’ and consider the hyporheic zone to be a part of the larger groundwater system in which epigean organisms with hyporheobiont life stages are present. Although the hyporheic zone is typically considered by river ecologists to be only a few centimetres to metres thick, Stanford and Ward (1988) and Danielopol (1989) noted that a hyporheic zone delineated by the presence of riverine animals may extend several kilometres from the channel in flood-plain aquifers associated with large unregulated fluvial systems. Brunke and Gonser (1997) conclude that the hyporheic zone can be distinguished from the adjacent environments (the aquifer and stream channel) because it demonstrates characteristics of both, although each characteristic develops its own gradients. Consequently, they consider that exact definitions are too rigid and do not account for the dynamic nature of the ecotone and its functional variability.

To the hydrogeologist the hyporheic zone is a part of the groundwater system. The conventional definition of groundwater is ‘water beneath the surface of the ground in the saturation zone and in direct contact with the ground or subsoil' (Council of the European Community 1980) and 'subsurface water that occurs beneath the water table in soils or geological formations that are fully saturated' (Freeze and Cherry 1979). These definitions encompass water in saturated hyporheic sediments beneath or adjacent to a stream or river, or, indeed, waters below any other form of surface water body, such as a lake. Nevertheless, it is recognised that the concentration and realignment of groundwater flow-lines within highly permeable river-valley sediments result in mixing of the aquifer water with stream water, which normally have different geochemical signatures. Furthermore, the low organic carbon and low microbial biomass environment of most aquifers contrasts with the hyporheic sediments which are relatively rich in organic carbon- and microbial community. As a consequence, it might be assumed that there is an increased potential for biochemical reactions of pollutants transported from an aquifer into the hyporheic zone, or vice versa.

No single scientific discipline has 'owned' the hyporheic zone and it is interesting to observe how each has approached its study. To the freshwater ecologist comparing the hyporheic zone to the stream channel, it is a dark, oxygen- and energy-deficient environment that has low species richness, diversity and density (Gibert *et al.* 1994), and which is difficult to sample and investigate (Palmer 1993, Fraser *et al.* 1996, Fraser and Williams 1997, Findlay and Sobczak 2000). The absence of light prevents photoautotrophic growth and it is often assumed that biological processes in the hyporheic zone are slow and of low productivity in comparison to in-channel processes (Gibert *et al*. 1994). By contrast, hydrogeologists conceptualise the hyporheic sediments as being carbon and microbial community-rich, in comparison to the aquifer sediments with which they are more familiar. The nature of the hyporheic zone is such that it is postulated that the potential for attenuation of pollutants is significantly greater in the hyporheic zone than in a surrounding aquifer.

Consequently, interest has recently started to concentrate on the attenuation capacity of the hyporheic zone for chemical pollutants. Studies have begun to address the hyporheic zone, in isolation or in combination with the riparian zone, as a buffer that may decrease the impact of polluted groundwater on receiving surface waters. Research has principally concentrated on the cycling of nutrients associated with intensive agriculture, such as nitrate (Pinay and Décamps 1988, Triska *et al.* 1993b, Jones 1995, Hill 1996, Cey *et al.* 1999, Pucket 2004) and phosphate (Triska *et al.* 1989, 1993a, Dahm *et al*. 1998, Carlyle and Hill 2001). However, more recent studies on synthetic industrial chemicals have begun to extend the large literature on natural attenuation processes in aquifers to the groundwater–surface water interface below rivers (Atekwana and Krishnamurthy 2004, Conant *et al.* 2004) and within wetlands (Lorah and Olsen 1999, Lorah and Voytek, 2004). Studies that address the processes of industrial pollutant attenuation at the interface of groundwater with lakes, estuaries or coastal environments are less common. However, Reay (2004) describes a study of the behaviour of sewage effluent compounds in a groundwater discharge to the sea, and Dowling *et al.* (2004) describe the processes that control the flux of nitrate from groundwater into the Gulf of Mexico. Westbrook *et al.* (2005) describe the behaviour of a pollutant plume that contains benzene, toluene, ethylbenzene and xzylene (BTEX) and naphthalene at a groundwater–estuary interface.

To the hydrologist, the hyporheic zone is often regarded as an extension of the stream channel, and is the permeable stream-bed deposits that form riffles and palaeochannels through which stream water can migrate. Its characteristics vary, but field experiments, normally comprising tracer tests, show that water can spend a longer duration in the hyporheic zone than in the open channel (Jones and Holmes 1996). Research has concentrated on a number of areas, including the exchange of nutrients and other solutes between the stream and its hyporheic zone (e.g., Dent *et al.* 2001, Jonsson *et al.* 2003) and its implications for ecological systems (e.g., Brunke and Gonser 1997, Findlay 1995, Battin *et al*. 2003), the movement of water through riffle structures and the consequent implications for fish (salmonid) spawning (e.g., Crisp 1990, Malcolm *et al.* 2004a, 2004b) and controls on the evolution of stream water chemistry (e.g., Fuller and Harvey 2000, Jonsson *et al.* 2003).

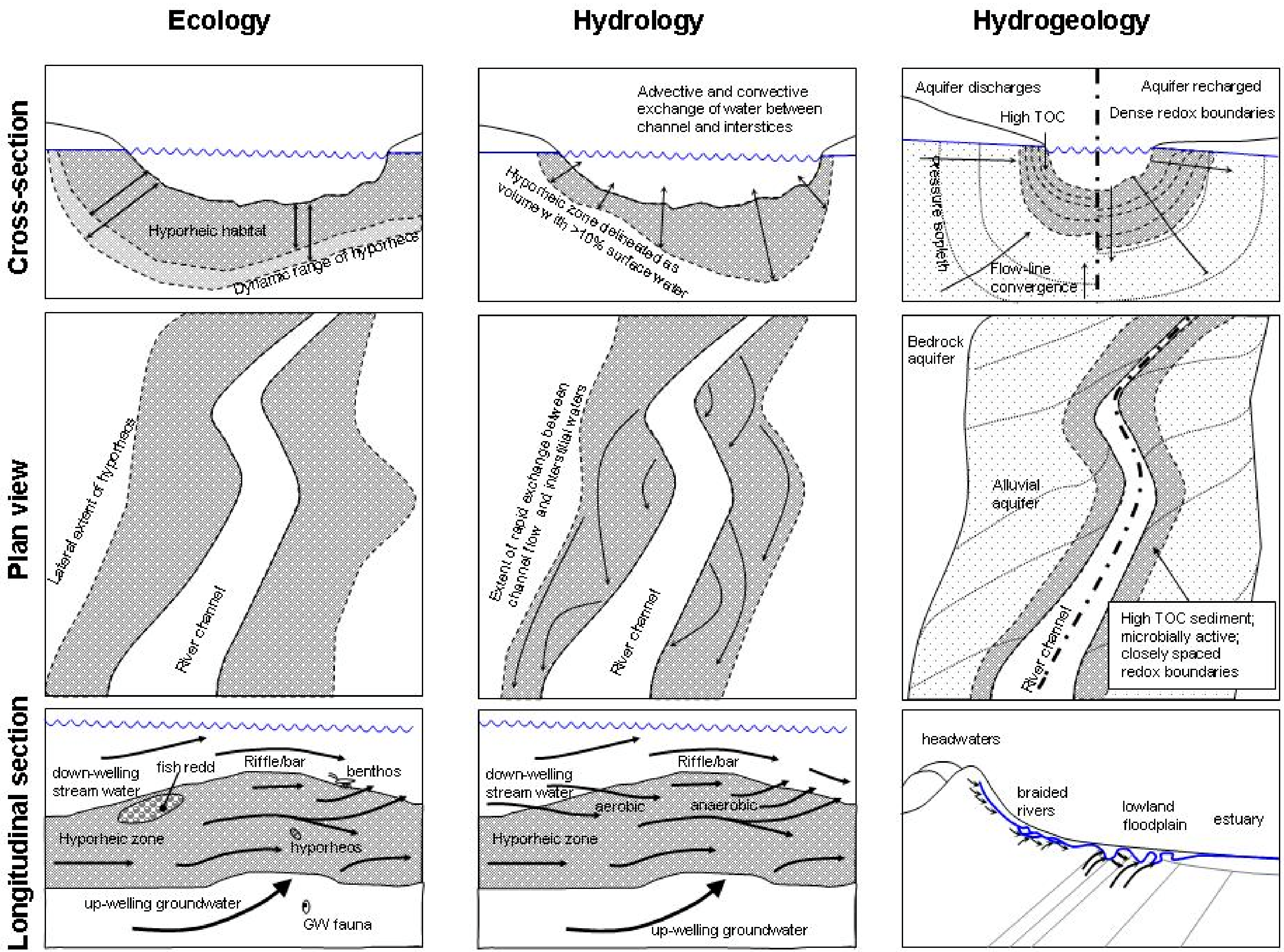


Figure 3.1 Illustrative conceptual models of the hyporheic zone reflect the aims of different workers.

Conceptual models of the hyporheic zone and the processes that occur within it have been prepared by workers in a number of different fields. As a consequence of their research aims, the definitions of the hyporheic zone and metrics used for parameterisations are various. Many descriptions are unclear as to the definition of the hyporheic zone assumed, and there are numerous pleas in the scientific literature for a clearer explanation of the definitions and assumptions made about the hyporheic zone at the outset of research. *Figure 3.1* illustrates the main foci of workers in different scientific disciplines.

Holmes (2000) presents a review of previously described hyporheic conceptual models (*Figure 3.2*) in which the boundary of the hyporheic and groundwater zones varies. In general, the model presented on the left (after Triska *et al.* 1989) is the approach generally adopted by hydrologists concerned with the exchange of water between river channel and permeable strata beneath the riverbed. The model of Gibert *et al.* (1990) is more typical of freshwater ecology descriptions, although many ecologists (e.g., Findlay 1995) may refer to zones E1 and E2 collectively as the benthic and hyporheic zones.

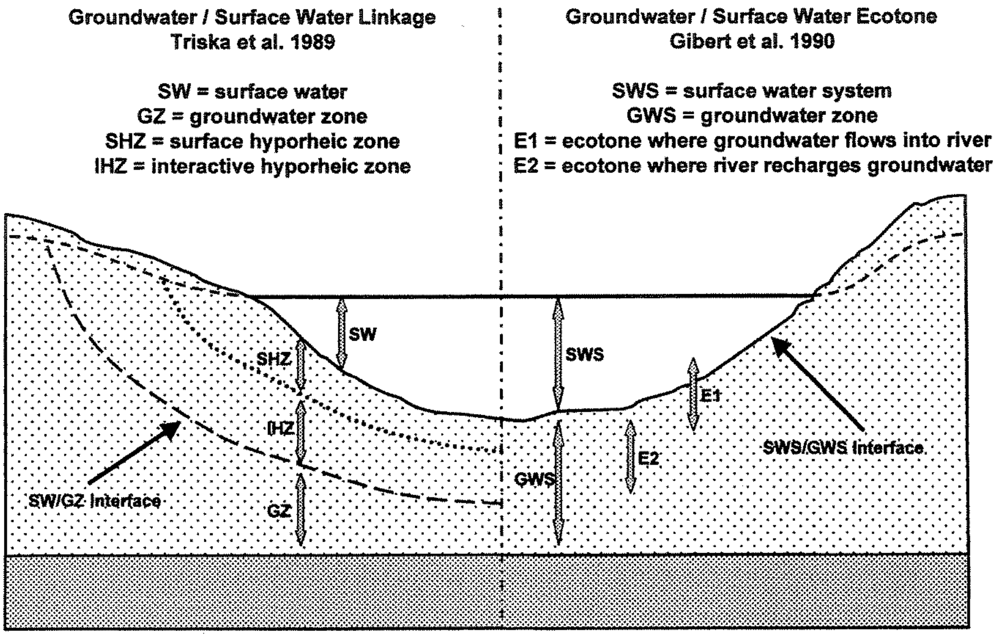


Figure 3.2 Comparison of definitions of groundwater–surface water interface (including hyporheic zone) presented by different authors (reproduced from Holmes 2000, © Academic Press, Inc. 2000).

Figure 3.2 is typical of many conceptual models of the hyporheic zone – presented as a cross section perpendicular to river flow. Other models, principally from hydrologists interested in vertical water-exchange processes along the course of a river, and fisheries scientists interested in the relationship of water flow and fish redd–habitat present conceptual models that have longitudinal section.

Pioneering work by Stanford and Ward (1993) moved hyporheic zone studies forward significantly by proposing the integration of stream channel and hyporheic systems into an integrated river continuum concept – the hyporheic corridor. This concept has helped to dispel the artificial and arbitrary delineation of river channel from sub-surface waters by describing the continuous and dynamic interactions between stream and adjacent sediments within the context of the river's floodplain. It still fails to integrate fully the larger scale hydrogeological flow regime and geochemical processes, or the variability of those interactions, but it represents a major advance in understanding of the hyporheic process.

Very few authors present three-dimensional representations of processes in the hyporheic zone, or attempt to link the large-scale hydrogeological flow regime with processes in the local riverbed sediment. Although the hyporheic zone is commonly described as being a dynamic ecotone (Brunke and Gonser 1997, Boulton *et al.* 1998), temporal variability is rarely included in the conceptual models presented. The development of conceptual models that integrate hydrological, ecological, sedimentary and hydrogeological processes in a four-dimensional framework is needed for a fuller description of the interrelated processes and scales important to hyporheic investigation.

Very few conceptual models link the geological and mineralogical origin of hyporheic sediments, their depositional environment and the consequent geochemical and hydraulic properties of the hyporheic zone. Generic conceptual models of hyporheic zone structure, architecture and geochemistry for a range of typical UK geological terrains would greatly help to clarify the potential for hyporheic attenuation within surrounding geological landscape.

Within the context of the Environment Agency's concentration on the attenuation of pollutants at the groundwater–surface water interface, this report adopts a definition for the hyporheic zone as:

The water-saturated transitional zone between surface water and groundwater.

It is an ecotone with hydraulic and biogeochemical gradients in which there is often mixing of the respective waters, and as a habitat for living organisms it can play an important and protective role for both surface and groundwater. It is often characterised by the presence of dense microbial communities (relative to a typical aquifer population), variably high organic-matter concentrations and a faunal assemblage distinct from both stream-channel and true groundwater. However, its precise extent is best determined in relation to the specific problem being considered and the most important parameters to that problem.

3.2 Delineating the hyporheic zone

Many authors' attempts to delineate the extent of the hyporheic zone bear close association with a desire to define it. Consequently, many of the approaches adopted reflect the scientific disciplines in which they work and the specific research objectives being tackled.

3.2.1 Delineation based on ecology and community structure

The earliest published studies that attempt to delineate the hyporheic zone are on hyporheic ecology. Danielopol (1989) describes how early workers, such as Orghidan (1959) and Schwoerbel (1964), assumed that the hyporheic sediments were solely a refuge for stream-dwelling species, offering stable conditions during periods of environmental stress within the stream channel, such as during flood or drought. The temporal nature of hyporheic refugia resulted in assumed hyporheic zones of only a few centimetres beneath the streambed in porous substrate, such as riffles dominated by gravel and sand deposits.

More recently, it has become evident that hyporheic organisms include a wide variety of permanent and occasional residents (Danielopol 1989) and that the hyporheic zone is populated by complex and dynamic communities of microbial (Barlocher and Murdoch 1989, Storey *et al.* 1999, Findlay and Sobczak 2000), meiofaunal (Ward *et al*. 1998, Hakenkamp and Palmer 2000, Robertson *et al.* 2000a, 2000b) and macroinvertebrate fauna (Coleman and Hynes 1970, Boulton 2000a). Studies of meio- and macroinvertebrate fauna in alluvial aquifers reported the presence of taxa characteristic of the hyporheic zone as deep as 1 m beneath the streambed, and up to a few hundred metres from the channel (Danielopol 1989, White 1993, Dumas *et al.* 2001), which discounts the theory that the hyporheos are simply temporary refugees from the stream channel.

Stanford and Ward (1988) used the distribution of hyporheic biota around the Flathead River (Montana) to identify an even larger hyporheic zone, which had an average depth of 10 m and extended on average about 3 km from the river. Other attempts to define the extent of the hyporheic zone on the basis of faunal assemblages alone have proved difficult because of their complex and temporally variable distribution in the sub-surface (White 1993, Fraser and Williams 1998) and because there is often significant overlap between the domains of hypogean and epigean fauna (Danielopol 1989). However, approaches that combine taxa distributions with water chemistry data have proved more successful (Fraser and Williams 1998).

Studies have also shown that the distribution and composition of hyporheic fauna varies with environmental conditions. Important variables include the direction and magnitude of water flow (Hakenkamp and Palmer 2000), normal seasonal climatic variation (Hendricks and White 1995, Wroblicky *et al.* 1998), climatic extremes including drought and flooding (Hynes 1983, Stanley and Boulton 1993), sedimentary structure (Dumas *et al.* 2001), geological (geochemical) terrain (Cannan and Armitage 1999), pollution (Feris *et al.* 2003a); land-use change (Trayler and Davis 1998) and natural hydrochemical evolution along a stream course (Malard *et al.* 2003). In addition, the system is reported as being dynamic (Gibert *et al*. 1994, Brunke and Gonser 1997; Fraser and Williams 1998), and on a microscale highly heterogeneous or patchy (Dole-Olivier and Marmonier 1992, Ward *et al*. 1998). Williams (1989) describes the system as being one that is more analogous to the oceans with diel tidal fluctuations than to a quasi-static system around which a line can be drawn to delineate the presence of the hyporheos.

Dumas *et al.* (2001) describe how the boundaries of different hydrogeological units can be established by the presence of macrocrustaceans, which are present in an alluvial aquifer, but absent from the underlying and adjacent terrace deposits. Within a single river system in southern England, Cannan and Armitage (1999) describe how the macroinvertebrate assemblage in the groundwater-dominated River Frome (Dorset) changes as the river moves from the Upper Greensands and Gault Clay onto the outcrop of the Chalk, and subsequently onto terrace gravels. Although controls were not present in their work to discount the faunal changes as a consequence solely of moving downstream, they suggest that the mineralogical changes in the groundwater cause the faunal changes, and that changes in the stream macrofauna assemblage will be replicated in the hyporheos.

Malard *et al.* (2003) investigated the relationships between a hyporheic invertebrate community and longitudinal changes in water chemistry along an 11 km stretch of an alpine glacial river. They noted that moving downstream from the glacier the hyporheic taxonomic richness (the total number of different taxa collected) increased and they were able to correlate this with changes in water temperature, increased mixing with groundwater and the import of organic carbon to the stream. Further downstream, in an alluvial floodplain reach, groundwater up-welling increased significantly and was mirrored by a marked increase in the taxonomic richness of the hyporheos. The increased interactions of groundwater and surface water downstream profile and the increase in taxonomic richness suggest that, other factors being equal, large rivers with active alluvial floodplains have a larger and more diverse hyporheic zone than alpine headwater streams.

Stanley and Boulton (1993) contrasted the distribution of hyporheic assemblages with hydrological control in two rivers – a large mesic river in France and a small desert stream in Arizona. They report a strong correlation between the population of hyporheic invertebrates and groundwater flow and/or exchange with the stream. In an up-welling section of the River Rhône, hypogean amphipods increased in abundance from the riverbed downwards (to a maximum study depth of 0.5 m), which was attributed to the increased proportion of groundwater present at depth. Other hypogean fauna displayed similar patterns (Dole-Olivier and Marmonier 1992).

The hyporheic zone has also been delineated in catchments subject to anthropogenic processes. Trayler and Davis (1998) compared the distributions of benthic and hyporheic meio- and macroinvertebrates in the headwaters of four Australian streams with sandy bed sediments. Two catchments were subject to clearfell logging, while the other two were not logged. They observed that the density of benthic communities (in the top 5 cm of sediment) did not appear to be greatly affected, although taxonomic richness was much reduced in the logged catchment. By contrast, the hyporheos (sampled at 30 cm below the streambed) was significantly denser and extended to greater depth in the catchment that was not logged.

3.2.2 Delineation based on tracer tests

The hydrological literature includes reports of several studies that undertook in-stream tracer tests to determine the significance of water exchange between the channel and hydraulically connected sediments (Harvey and Wagner 2000). Most studies sought to identify the relationship between the time solutes and/or water are present within the channel with the time spent in the hyporheic sediments, to determine the hyporheic 'retention'.

Jonsson *et al.* (2003) describe investigations using conservative (^{3}H) and reactive ($^{51}Cr(III)$) tracers injected into the stream channel at a constant rate and measured at points downstream. The objective was to determine the exchange of stream and hyporheic waters and the retention of stream solutes in hyporheic sediments (actually transport into and retardation on hyporheic sediments). They observed that 76 percent of the Cr was removed from the stream over a 30 km reach, and most had partitioned into bed sediments. They noted that exchange of ^{3}H took place over a vertical distance of around 10 cm beneath the riverbed, but the ^{51}Cr only migrated approximately 6 cm into the hyporheic sediments. They determined that the zone of active water exchange extended to about 10 cm below the streambed, but that the ingress of reactive solutes was less (over the timescales of the experiment) because of the effects of sorption processes. They also noted that, although the entry into the hyporheic sediments was rapid, because of a significant concentration gradient the release of the tracer from the hyporheic zone back into the channel was much slower. A combination of sorption and reduced concentration gradient resulted in only 75 percent of the ^{51}Cr being released back into the channel over 45 days.

Fuller and Harvey (2000) applied a similar approach in a creek in Arizona contaminated with mine water to assess the uptake of bromide and metals from the stream into the hyporheic zone. Over a 12 km reach of the creek they observed between 12 and 78 percent removal of metals from the creek into the hyporheic sediments, depending on flow and climatic conditions. Bencala *et al.* (1990) selected lithium for a similar tracer experiment in a stream contaminated with acid metal. Other workers have also used conservative and reactive tracers in combination to determine the exchange between channel and hyporheic zone, including Constantz *et al.* (2003) who released heat and bromide.

Hendricks and White (1991) used chloride and nitrate tracers to delineate the lower boundary of surface water–groundwater mixing at about 50 cm below the streambed, while Triska *et al.* (1989) identified a significantly larger exchange volume that extended in excess of 10 m from the channel. At a distance of 4 m normal to the channel, 88 percent of the water was found to be of stream origin, while at 10 m groundwater dominated, at 53 percent. The retention time in the hyporheic zone was found to exceed the time solute and/or water spent in the channel, and the volume of water in the hyporheic zone (58 percent) exceeded that in the channel (42 percent). In a review of a large number of existing investigations, Runkel (2002) reported that migration in the hyporheic zone accounted for between 0.1 and 68 percent of the total reach travel time.

Although delineation of the hyporheic zone based on hydrological movement does not consider the stream in a landscape context, such as that presented by Stanford (1998), it better defines the stream as an ecosystem and supports the ecotone models of Brunke and Gonser (1997), Boulton *et al.* (1998) and Sabater and Vila (1991).

Boulton (1993) describes a simpler use of tracers emplaced below the sediment surface to identify (quickly and qualitatively) zones of up-welling or down-welling water. Once placed into the sediments, the tracer is only seen in the channel immediately above if the groundwater is up-welling. In zones of down-welling stream water the tracer either goes unobserved or reappears in the stream at some distance downstream where up-welling conditions exist. Yoneda *et al.* (1991) used this principle to identify discharge and infiltration zones based on the presence of ^{222}Rn, which is naturally present at higher concentration in groundwater (through rock–water interactions) than in surface water.

The reliability of in-stream tracer tests have been considered by Harvey *et al.* (1996) and Wagner and Harvey (1997). They consider that, subject to appropriate design and operation, tracer experiments are reliable under most conditions, but it is difficult to obtain meaningful data when there is a high baseflow (groundwater) component of stream flow. Under these conditions, tracer tests that begin with a controlled release within the aquifer and are monitored across the hyporheic zone are likely to yield more useful and interpretable data. There is considerable experience and guidance on groundwater tracer testing in aquifers, but few workers have investigated the movement of tracers across the hyporheic zone, rather merely concentrating on whether a groundwater-released tracer appears within the stream channel.

Further detailed review of the use of tracer tests at the groundwater – surface water interface is presented in Berryman (2005).

3.2.3 Delineation based on geophysical investigations

A limited range of geophysical testing methods has been applied to the groundwater–surface water interface in an attempt to delineate lithological variation, identify the presence and location of pollutants or delineate the hyporheic zone. Different geophysical methods have been applied in various environments, depending on the electrical conductivity, density or other physical property that is expected to vary with the presence of the pollutant or geological layer.

In a study to investigate the migration of a chlorinated solvent plume from an aquifer into a river, Conant *et al.* (2004) used ground-penetrating radar (GPR) to identify stratigraphic layering within sandy clay alluvial sediments and the presence of tetrachloroethene (perchoroethene, PCE). GPR relies on there being a density contrast between the geological horizons, or between a pollutant and the groundwater, to identify the extent of a geological bed or a pollutant plume. Conant *et al.* (2004) used GPR methods to delineate geological strata beneath a river to a depth of 5 m, and to locate a PCE pollutant plume as it migrated from an alluvial aquifer through the hyporheic zone and into a river. Naegli *et al.* (1996) also applied GPR to the hyporheic zone of a clean headwater stream to determine shallow sediment structures, from which they were able to delineate shallow unconsolidated hyporheic sediment from unweathered bedrock.

In a study of the hyporheic zone around a tidal creek in Australia, Acworth and Dasey (2003) used a combination of borehole electrical tomography and cross-creek electrical imaging to locate and delineate the zone in which saline waters from the creek mixed with freshwater in the underlying aquifer. The electrical conductivity of saline water is significant greater than that in freshwater, which enables electrical methods to locate zones of relatively high and low conductivity. Using these geophysical methods, in combination with electromagnetic and gamma-activity logging of control boreholes, Acworth and Dasey (2003) were able to delineate a hyporheic zone, based on the mixing of fresh and saline water, that extended to around 10 m beneath the base of the creek. They also showed that the pattern of mixing was not homogeneous, but that heterogeneities in the hydraulic conductivity associated with a cemented sand layer at depth limited the depth of mixing.

Harvey *et al.* (1997) used electrical conductivity mapping techniques on porewater in lake sediments to successfully locate areas of groundwater inflow. Using a mobile sediment probe lowered from a boat to the lake bed, Harvey *et al*. (1997) delineated inflow zones that could not have been located easily using traditional methods, such as peizometers. Oxtobee and Novakowski (2002) also used electrical resistivity methods, in combination with other techniques, to investigate the processes in a shallow riverbed. They were able to show that water entering the river from the underlying groundwater was principally derived from the uppermost 3–7 m of strata. The technique described, which was applied in a shallow river environment (typically water <1 m deep), could have application in deeper slow-flowing river studies, using the approach described by Harvey *et al.* (1997).

In progress and unpublished studies in the UK funded under the National Environment Research Council (NERC) Lowland Catchment Research (LOCAR) programme include two projects that are using geophysical methods to characterise the groundwater–surface water interface within the riparian and hyporheic zones. Both projects are led by researchers at the University of Lancaster. The projects are using a combination of GPR and electrical resistivity surveys to investigate the sediment structure and identify zones of significant groundwater–surface water interaction. In one, these data are combined with groundwater and surface water chemistry to examine the importance and variability of processes that act at the interface in a Chalk aquifer–stream system in the Pang-Lambourn catchment. The second adopts similar approaches to characterise the interface in a Triassic Sandstone system in the River Tern catchment in Shropshire (Binley 2004, personal communication).

3.2.4 Delineation based on temperature profiling

Tracer tests described in Section 3.2.2 are typically conducted using a suitable chemical or (micro)biological substance. However, it is possible to use heat as a tracer for hyporheic zone studies. Groundwater temperature is relatively constant – long-term records of groundwater temperature in aquifers across England and Wales (invariably less than 100 m deep) indicate typical annual fluctuations between 10 and 13°C, with an average of 11.2°C and a standard deviation of 1.6°C (Smith, unpublished data). By contrast, the temperature of surface waters varies both between day and night, and seasonally. The difference in temperature between groundwater and a river results in a temperature gradient across the hyporheic zone. Dependent on the direction of groundwater flow (whether the aquifer discharges to the river or vice versa), the difference between groundwater and surface water temperature, and the flux of water, a temperature profile develops (*Figure 3.3*).

Temperature profiling in aquifers and rivers has a long history dating back to the early twentieth century and has more recently become an established technique to quantify hyporheic flow (United States Geological Survey 2003). However, a larger literature is related to studies on fisheries management, particularly in relation to studies of commercially important salmonid fisheries (e.g., Acornley 1999).

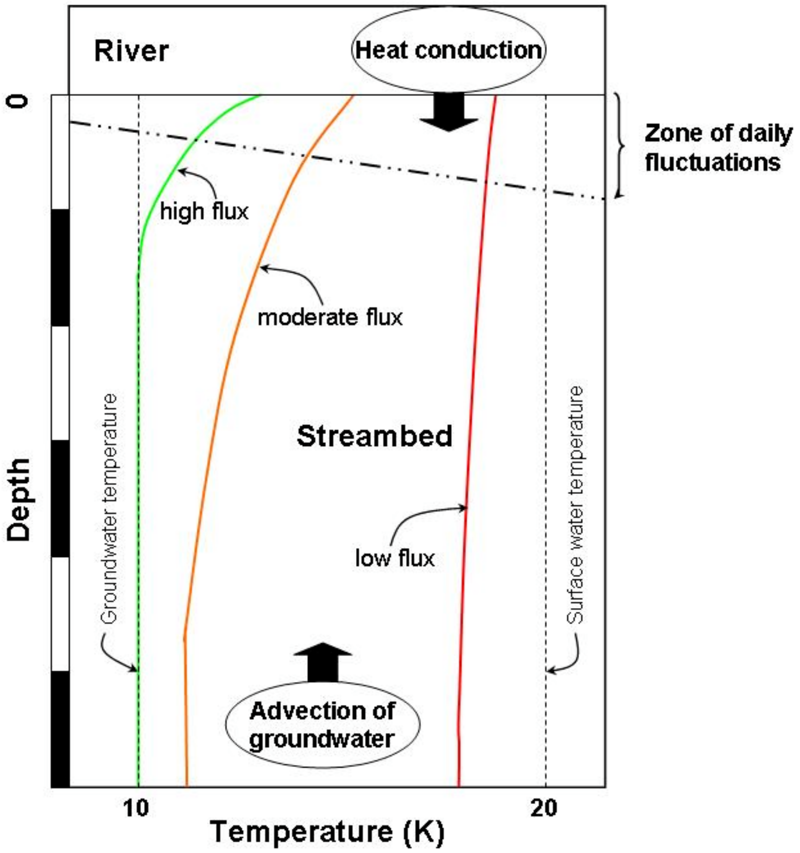


Figure 3.3 Illustrative temperature versus depth profiles through the hyporheic zone for systems with high, moderate and low upwards groundwater flux (based on Conant 2004).

Temperature differences and profiles have been used to identify areas of groundwater discharge and surface water infiltration at a small scale, such as within riffles (White *et al*. 1987, Evans and Petts 1997, Clark *et al.* 1999;,Alexander and Caissie 2003), at river-reach scale (Silliman and Booth 1993, Constantz and Thomas 1997, Conant 2004) and at sub-catchment scale (Constantz *et al*. 2003, Becker *et al*. 2004), and to develop conceptual models of fish habitats and thermal refugia for aquatic animals (Power *et al.* 1999). White *et al*. (1987) and Hendricks and White (1991) describe the thermal profile within riffle structures to interpret the depth of water exchange between a stream and the underlying sediments. Using the combined data, a three-dimensional model is presented (White 1993) of the depth to groundwater, represented as a contoured surface of the depth to groundwater temperature, which was defined as 8°C for the Michigan stream studied. Using this definition, the hyporheic zone thickness was shown to vary between approximately 0.5 and 1.5 m, over a 9 m stretch of the river.

Using miniature temperature loggers installed within a riffle, Evans and Petts (1997) showed that the water temperature at the head of the riffle was close to stream water temperature, while the temperature at the riffle tail was close to deeper groundwater temperature. This was interpreted as showing that surface water down-welled into the riffle at its head, mixed with groundwater and up-welled back into the stream channel at its tail. This is a similar pattern to that described by other authors (e.g., Hendricks and White 1991), but it is not universally observed.

Using data from a field site in Canada, Conant (2004) combined temperature profiling with hydraulic testing and geochemical analysis to quantify groundwater fluxes across the hyporheic zone. He was able to use the data to develop a conceptual model for flow across the groundwater–surface water interface at the site that described five different flow behaviours for baseflow discharges, controlled by hydraulic conductivity and structure of the riverbed sediments. In addition, he was able to show that the hyporheic zone was laterally constrained by the presence of low permeability peaty clay deposits, and that mixing of the groundwater and surface water occurred principally in the zone 2-4 m beneath the riverbed.

The conclusions reached by most workers are that temperature profiling techniques are useful to delineate zones of up-welling and down-welling water (United States Geological Survey 2003). However, further data collection and analysis, such as flux estimates using seepage meters or head measurements, are necessary to quantify water velocity and flux across the hyporheic zone (Becker *et al.* 2004). Importantly, temperature profiling techniques are cheap and easily repeatable. Temperature profiling has also confirmed that the movement of water across the hyporheic zone is commonly heterogeneous both spatially and temporally (White *et al.* 1987, Becker *et al.* 2004, Conant 2004). Combining temperature data with geochemical analyses has shown that misinterpretation of water temperature profile data alone can easily be made. Alexander and Caissie (2003) observed that areas of riverbed with a seepage flux into the channel might simply be assumed to be areas of groundwater baseflow. However by combining water chemistry analyses with temperature profiling, they were able to show that much of the up-welling water was stream channel water that had entered the hyporheic sediments only recently.

3.3 Flow across the hyporheic zone

The hyporheic zone is normally observed and described as being a porous medium, although locally significant heterogeneity exists that gives rise to springs and groundwater seepages more comparable to pipe or fracture flow (e.g., Conant 2004). In many hyporheic systems, Darcy's Law can be assumed to apply, and the flow of water across the interface is a function of the hydraulic conductivity of the sediments and the hydraulic gradient acting across the hyporheic zone (i.e., the head difference between the groundwater and the river). Darcy's Law (Equation 3.1) describes laminar flow in a porous medium.

$$Q = -K.A.\frac{dh}{dl} \qquad (3.1)$$

Where Q = flow (m^3/s), K = hydraulic conductivity (m/s), A = area through which flow occurs (m^2) and dh/dl = change in head (h) over a distance (l) (unitless). The term $-dh/dl$ is commonly written as i, the hydraulic gradient.

The average linear velocity, v, of water can be obtained by dividing by the kinematic porosity, n_e.

$$v = \frac{Q}{n_e} = \frac{K.A.i}{n_e} \qquad (3.2)$$

And the velocity of a linearly partitioning pollutant can be estimated by multiplying by a retardation factor, R_f:

$$u = v.R_f = v.\left(1 + \frac{K_d \cdot \rho}{n}\right) \qquad (3.3)$$

Where u = pollutant velocity (m/s), K_d = aquifer–water partition coefficient (m^3/kg), ρ = bulk density (kg/m^3) and n = porosity (fraction). An estimate of pollutant flow velocity, and subsequently of travel time through an attenuating environment, is a main control on the potential for natural attenuation processes to significantly decrease pollutant mass. Conant *et al.* (2004) observed that the majority of the PCE mass removal within a heterogeneous hyporheic zone occurred in lower permeability silty clay deposits, where slower flow conditions provide sufficient time for reductive dechlorination processes to significantly decrease PCE mass. Less biodegradation was observed in higher permeability areas of the streambed, despite having similar mineralogy and redox conditions.

Field experiments have identified considerable heterogeneity in both the hydraulic conductivity of riverbed sediments (Calver 2001) and the hydraulic gradient across the hyporheic zone within an individual reach. Variations in grain size, particle sorting and packing have a significant effect on hydraulic conductivity at the sediment scale. At the centimetre to tens-of-metres scale, grain size variation, bedding, jointing and other sedimentary structures are important, and at the kilometre scale, changes in geological terrain and the effects of pressure (with depth) significantly affect aquifer permeability. In respect of the hyporheic zone, the effects of sediment parent and bedrock geology largely control the nature of the sediments in the riverbed (e.g., whether they comprise quartz-rich aggregates from weathering of granite batholiths or detrital clays from weathering of marine clays). However, local geomorphological controls on sedimentation and erosion affect the nature of the *in-situ* sediments.

At the local scale, deposition of gravel and sand in point bar and riffle structures results in a more permeable streambed than in areas where clays and silts are located, but such features can also induce non-Darcian flow within the hyporheic sediments (Packman *et al.* 2004). Significant permeability variability is also associated with geological materials that have heterogeneity, which reflects past depositional and diagenetic environments.

Depending on the relative head differences between groundwater and a river, flow may be upwards into the river, downwards from the river into the aquifer, or horizontal, giving rise to flow into and out of the stream (effectively through the stream). Alternatively, there may be no hydraulic continuity between river and aquifer. These basic conceptual models are illustrated in *Figure 3.4* (after Sophocleous 2002).

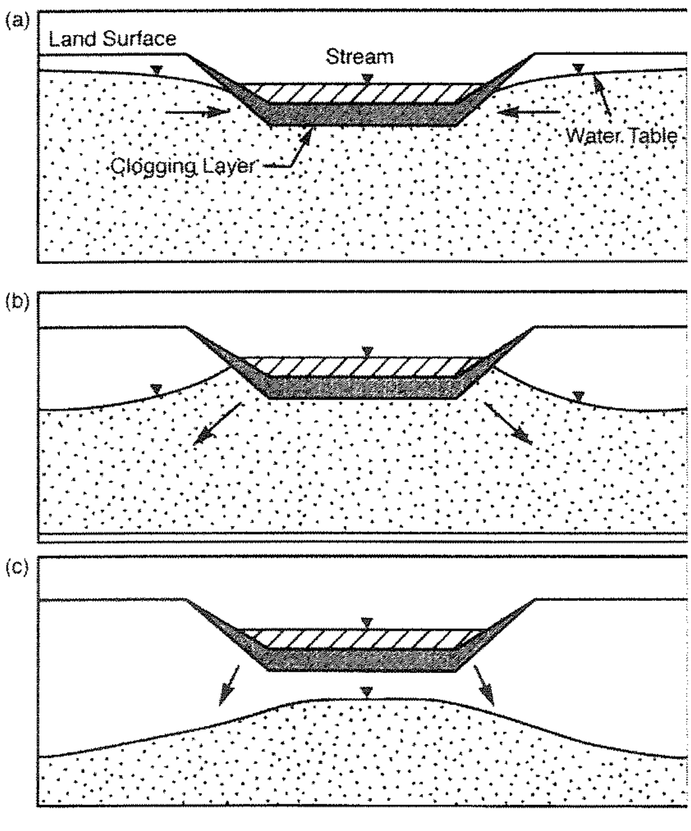


Figure 3.4 River–aquifer relationships for (a) gaining stream, (b) connected losing stream and (c) disconnected losing stream (after Sophocleous 2002, © Springer-Verlag 2002).

Within the hyporheic zone, the hydraulic conductivity may be changed by the action of biological processes. For example, biofilm formation can reduce permeability (Findlay and Sobczak 2000) or sediment reworking by macroinvertebrates, which can increase permeability locally (Boulton 2000a). Chemical processes may cause mineral precipitation or dissolution, which decrease and increase permeability, respectively (e.g., Fuller and Harvey 2000), and physical processes of sedimentation and clogging (colmation), which can reduce riverbed permeabilities (Brunke 1999, Rehg *et al.* 2005).

With the exception of streams affected by severe mine-water pollution, with the potential for significant mineral and/or metal precipitation, colmation appears to have the most significant impact on riverbed permeabilities, and it is certainly the most widespread process, with a regionally significant impact. Colmation is the process of deposition of fine-grained sediments in streambeds, normally as a result of filtering of sediment-containing down-welling stream water by the porous sediments of the streambed and sedimentation (Brunke and Gonser 1997). It can form a layer of low permeability sediment, known as colmatage, which can blind the riverbed (Petts 1988, Schalchli 1992).

The ecological consequences of colmation include loss of habitat and reduced opportunities for colonisation by microscopic fauna, deterioration of refugia for stream invertebrates and diminution of success for fish spawning in gravel beds (Brunke and Gonser 1997). In the most severe cases the colmatage may hydraulically isolate the river completely from the underlying groundwater. Initially this results in locally more heterogeneous interactions between river and aquifer, but in extreme cases can cause hydraulic isolation of long stretches of a river. The causes of colmation formation are numerous and are illustrated in *Figure 3.5*.

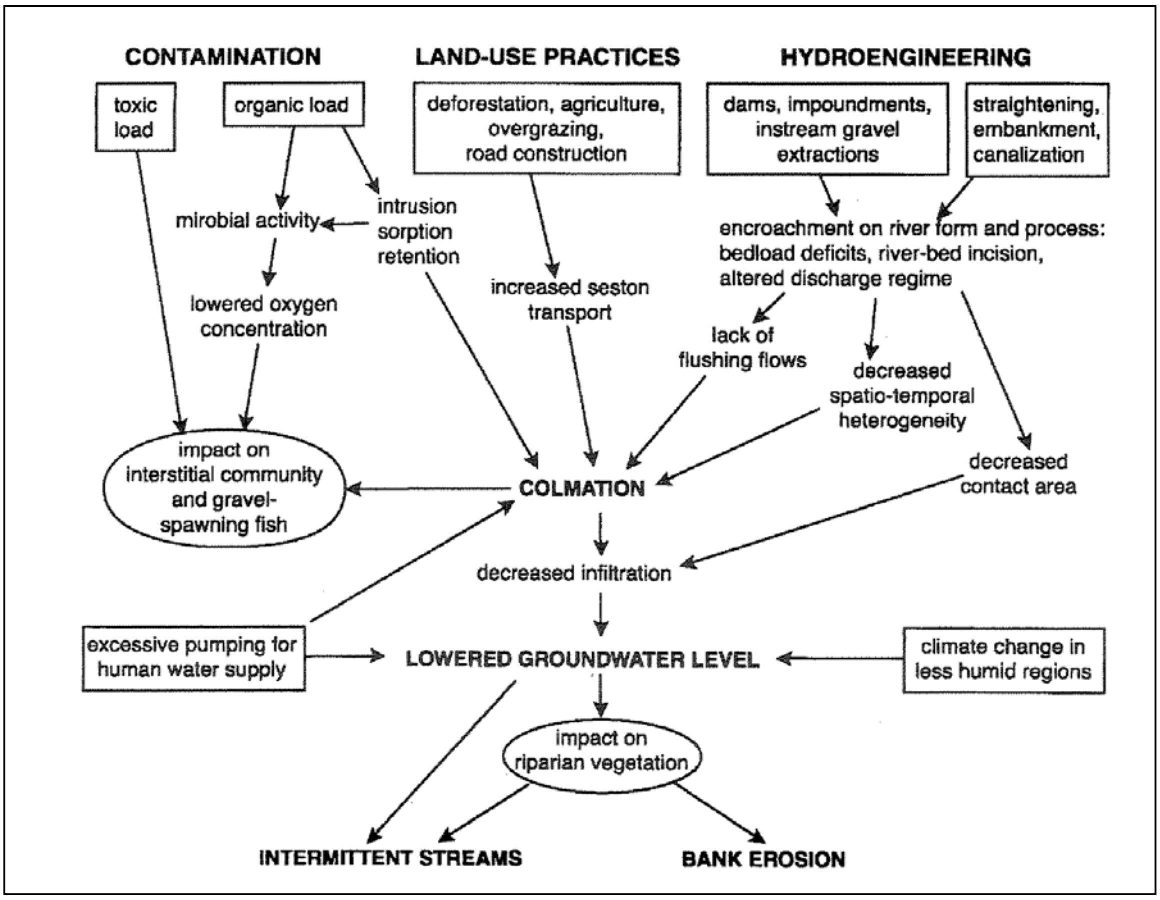


Figure 3.5 Human-induced colmatage-formation processes (after Brunke and Gonser 1997, © Blackwell Science Ltd 1997).

Colmation is most common in heavily regulated rivers subject to excessive soil erosion in the surrounding catchment. Since these catchments are frequently also affected by other human work and activities that lead to pollution, the colmatage can have beneficial effects as a barrier to pollution across the groundwater–surface water interface (Younger *et al.* 1993, Brunke and Gonser 1997).

Direct monitoring of water flow or flux across the hyporheic zone is difficult. Most workers have adopted traditional approaches using standpipes, borehole and/or peizometers to determine the vertical head across the hyporheic zone, and have assumed Darcy's Law is a valid representation to calculate the 'average' flow based on a limited number of hydraulic conductivity measurements. This method works well in homogeneous systems and over large distances, such as when applied to prediction of the flow in a sandy aquifer over a scale or hundreds to thousands of metres. However, the scale of heterogeneities observed in the hyporheic zone coincides with the scale of measurement and results in significant uncertainties in the parameterisation of hyporheic zone properties, such as water flux.

Some workers have tried to overcome this by simply increasing the number of monitoring points (e.g. Conant *et al.* 2004), or by combining a limited number of hydraulic head measurements with other techniques, including temperature profiling (Silliman and Both 1993, United States Geological Survey 2003, Becker *et al.* 2004), seepage meters that collect water moving vertically upwards through the riverbed (Isiorho and Meyer 1999, Harvey and Wagner 2000), tracer tests that provide tracer beak-through times (Bencala *et al.* 1984, Triska *et al.* 1993a), measurement of stream flow at up-stream and downstream ends of a hyporheic study reach to estimate the mean net baseflow input over the reach, *in-situ* flow meters and experiments that monitor the rate of dissolution of plaster of Paris located within the hyporheic zone (Angradi and Hood 1998). Landon *et al.* (2001) reviewed the literature on the effectiveness and accuracy of methods for *in-situ* measurement of hydraulic conductivity in sandy streambeds. They concluded that in shallow sandy sediments, the Hvorslev falling-head permeameter gave more accurate and reliable results than grain size analysis, seepage meters, slug tests or falling head tests. However, they also noted that inter-sample heterogeneity was so great that they considered a large number of tests using any test method that produced an estimate of the heterogeneity to be better than a small number of measurements using the more reliable or accurate methods. A detailed review of techniques for the in-situ measurement of flow across the groundwater – surface water interface is presented in Greswell (2005).

Most field studies identify highly heterogeneous flow conditions that are both spatially and temporally variable. The assumption that hydraulic conductivity values for streambed sediments are homogeneous, or predictable from a small number of measurements, is not consistent with the observed heterogeneity (Calver 2001). Streambed hydraulic conductivities have been shown to vary by seven orders of magnitude, between 10^{-9} to 10^{-2} m/s, although most values lay between 10^{-7} and 10^{-3} m/s (Lee 2000, Calver 2001). Supplementary techniques that have been used to measure *in-situ* hydraulic conductivity of hyporheic sediments include constant and falling head tests (Carenas and Zlotnik 2003), slug tests (Landon *et al.* 2001), solute injection tests (Harvey and Bencala 1993, Harvey *et al.* 1996) and the estimation of hydraulic conductivity from grain size analysis (Wolf *et al.* 1991).

Since one of the controls on natural attenuation is the residence time in an aquifer or hyporheic zone, attenuation potential may vary widely at a site, and between sites (Lee 2000). Any application of monitored natural attenuation to the hyporheic zone therefore needs detailed study of the flow heterogeneity and its impact on residence time and potential for the attenuation processes to occur.

Using fine resolution multi-level samplers or mini drive-point samplers (e.g., Duff *et al.* 1998), water and tracer and/or temperature samples can be obtained at the centimetre scale (Harvey and Wagner 2000). However, in some hyporheic zone studies this may not be a sufficiently fine resolution. Techniques developed in lake-sediment studies have recently been applied to the hyporheic zone, including diffusion sampling using gel probes (Davison *et al.* 2000) for the fine-resolution monitoring of tracer tests. Further development of gel-probe and microelectrode technology, both as a fine-scale measure of tracers and as a measure of redox chemistry, is likely to be a useful development for hyporheic zone studies that will improve resolution down to the millimetre scale.

3.4 Sedimentary structure and facies variability

Fluviatile depositional sequences are, in comparison to other clastic deposits, texturally immature – they contain much fine-grained matrix material in a poorly sorted assemblage commonly containing angular to sub-angular clasts (Tucker 1981). They form mudstones, siltstones and sandstones, the latter dominantly wackes, which can be examined in ancient deposits. The deposits contain clasts and a matrix of variable mineralogy, dependent on the local weathered country rock, and are highly variable between braided and meandering rivers, and alluvial fans, as a result of transport and weathering processes (Tucker 1981).

Sediments deposited in upland braided streams are principally sands, with coarser gravels and cobbles in higher reaches. Channel movement is restricted and the stream forms major elongate sand bodies that contain cross-stratified sediments. Sequential deposition and erosion results in cross-cutting and inter-bedded units (Tucker 1981). By comparison, meandering rivers typically occupy a large alluvial floodplain and the channel migrates laterally as a result of bank erosion and point-bar sedimentation (*Figure 3.6*). The lateral accretion of point bars results in fining-up sequences of sandstone that are commonly cross-bedded and contain increasing proportion of clays and light materials, such as organic matter, in the upper sequences. During floods, silt and clay are deposited on the floodplain, which results in finer grained units overlying coarser sand units (Tucker 1981). These can be observed in the deposits of major UK lowland rivers (Steiger *et al.* 2003a, 2003b).

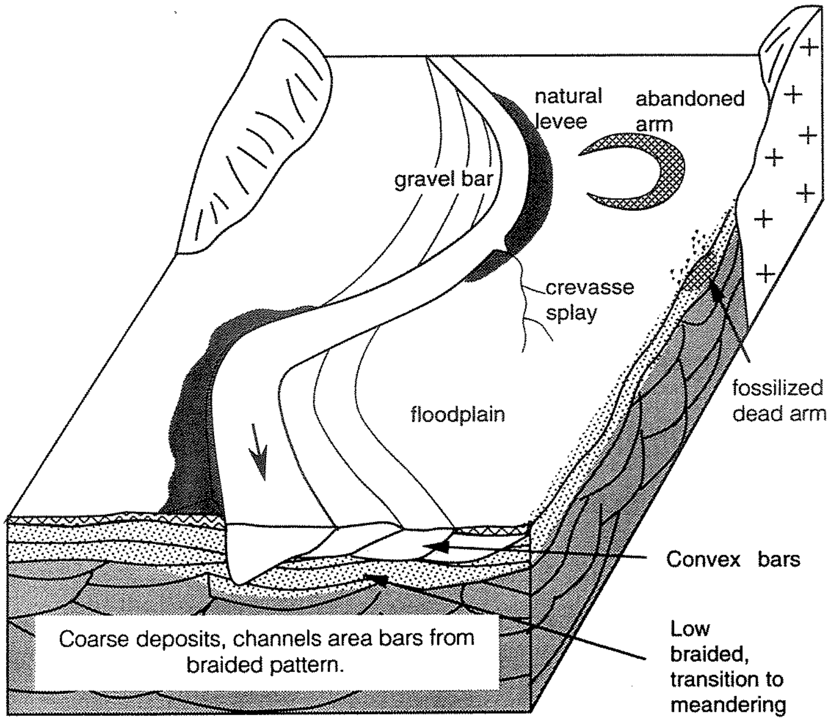


Figure 3.6 Active and buried unit landforms in an alluvial plain (after Creuzé des Châtelliers *et al.* 1994, © Academic Press, Inc 1994).

Within meanders, coarser sands are deposited on the inside of bends where water flow is slower (point-bar accretion), but the outside bank of meanders is often erosional, exposing previous fluvial sedimentary sequences. In lower energy environments, deposition of finer grained sediments occurs on the inside of meanders, while coarser sediments are deposited at the outside of the bends.

A great deal of work has been published on fluviatile sedimentology in relation to the origin, mineralogy, texture and structure of fluvial deposits (Tucker 1981), and subsequently applied in oil-industry interpretation of reservoir structures. Much of the sedimentological literature, however, comprises descriptive and qualitative facies models, whereas hydrogeological use requires quantification of, for example, hydraulic conductivity or mineral composition at a range of scales (Anderson *et al.* 1999).

In the hyporheic zone, sediment geochemistry is likely to reflect sediment origin and depositional conditions. It is expected to be heterogeneous, at each of the sediment, metre and reach scales. As a consequence, the hydraulic conductivity of the sediments, their sorptive capacity (Fontaine *et al.* 2000) and the biodegradation potential (Albrechtsen *et al.* 1997) are expected to vary. Similarly, the lithological structure affects the porosity of the sediments, which has implications for invertebrate distribution and taxonomic richness (Creuzé des Châtelliers *et al.* 1994, Gayraud and Philippe 2003, Olsen and Townsend 2003). Pinay *et al.* (2000) also observed that grain size affected the rates of denitrification in floodplain sediments, and Albrechtsen *et al.* (1997) saw similar effects on the biodegradation of a range of organic compounds in variously graded sediment. Sediment with a clay content below 65 percent did not favour denitrification (Pinay *et al.* 2000). It was suggested that migration of oxygen was limited by the finer grained, low-permeability deposits, which resulted in anaerobic conditions conducive to denitrification.

Coarser sediments, such as gravel beds, are critical for salmonid spawning (Baxter and Hauer 2000). Deposition of finer grained sediments, particularly in regulated, managed rivers, leads to blinding of sand and/or gravel beds, a deterioration of habitat and a decrease in river biodiversity (Hancock 2002).

Further investigation of the extensive sedimentology literature and its potential application within quantitative conceptual models of hydrogeochemical processes in the hyporheic zone may provide useful insights. In particular, the links between depositional environment and sediment hydraulic conductivity (and the potential for groundwater–surface water exchange), and mineralogy and organic matter content (and consequently attenuation capacity) should be investigated and up-scaled.

3.5 Geochemistry of hyporheic zone sediments

The mineralogy of aquifer sediments has been the subject of considerable investigation to determine the intrinsic potential for attenuation of pollutants, principally by sorption processes (British Geological Society and Environment Agency 2002), as well as in terms of the potential of non-aquifer strata as source rocks for fossil-fuel generation (e.g., Tyson 2004). It is known that the mineral composition of aquifer materials, such as clay content and mineralogy, organic matter and Fe/Mn oxide content can affect the degree to which various substances are sorbed onto mineral surfaces. To date few similar studies have been reported for hyporheic materials, which determine the intrinsic attenuation capacity of hyporheic sediments. Most sediment studies have investigated the distribution (and biogeochemical causes) of pollutant concentration, such as heavy metal and hydrophobic organic species, in fluvial and estuarine sediments (Hudson-Edwards and Taylor 2003).

Kleineidam *et al.* (1999a, 1999b) investigated the influence of sediment petrography and organic matter distribution on the sorption of organic substances in a number of fluvial aquifers in Germany and Switzerland. Their work confirmed a relationship between spectrographic composition of the aquifer sediments and the degree of transport, sorting and weathering that is evident in terms of grain size distribution and sediment maturity. Further, the potential for sorption was shown to vary between sediments with differing degrees of sorting and weathering. Kleineidam *et al.* (1999b) did not attempt to interpret the sedimentary architecture of the aquifer sediments to determine whether highly sorbing sediments are associated with particular lithofacies, or whether similar relationships could be observed in hyporheic sediments.

A number of workers have investigated the geochemistry and petrography of sediments in canals and industrialised fluvial basins to identity geochemical controls on pollutant cycling (Owens and Walling 2002, Dodd *et al.* 2003). In particular, Dodd *et al.* (2003) investigated the relationships between P, Fe and organic matter in anoxic canal sediments. They identified that the formation of $Fe_3(PO_4)_2.8H_2O$ – a mineral known as vivianite – may be a significant sink for P in hyporheic sediments. House (2003) notes that vivianite has been detected in anoxic fluvial sediments in rivers that have discharges of treated sewage effluent, but that there is little understanding of the geochemistry of vivianite formation or how widely it is distributed in anoxic and eutrophic fluvial sediments.

Extension of an existing programme of work that is investigating the geochemical composition of British aquifers and aquitards with respect to intrinsic attenuation capacity (British Geological Society and Environment Agency 2002), could concentrate on hyporheic sediments and attempt to relate mineralogy and attenuation capacity to source-rock terrain, depositional environment and sedimentary architecture.

Other studies of hyporheic zone carbon materials have been concerned principally with the import and reaction of leaf litter (Boulton and Foster 1998) or particulate organic matter (POM; Crenshaw *et al.* 2002a, 2002b). These studies examined the effect of organic matter on bacterial and fungal metabolism (Crenshaw *et al.* 2002a, 2002b, Baker and Vervier 2003), water exchange (Boulton and Foster 1998) and faunal distribution (Crenshaw *et al.* 2002a, Boulton and Foster 1998). Little work is reported on the nature and quality of the organic matter in hyporheic sediments and its consequent impact on sorption and microbial communities. However, Richardson (2003) identified studies on organic matter composition as being a priority to better understand the interactions between hydrophobic pollutants and organic mater present in aquifers and sediments.

The few studies on hyporheic zones in large, lowland, regulated rivers in the UK have found that the extensive and active hyporheic zones observed in pristine and unregulated fluvial systems are rare (Petts 2004, personal communication). Rather, the discharge of organic matter in treated sewage effluents and agricultural run-off, reduced bed-load transport and reworking because of the construction of weirs and river navigation works results in deposition of anoxic sediments that are variously carbon- and metal-rich, depending on their origin. The extensive and transitional nature of the hyporheic zone reported for river systems such as the Flathead River in Montana are rarely observed, and a sharp redox boundary between the aerobic benthic sediments and underlying anoxic sediments is often observed. Work on lowland UK hyporheic zones largely ceased at this stage in the early 1990s, when researchers concluded that 'natural', ecologically diverse, zones were not common (Petts 2004, personal communication). The anoxic sediment layer has many properties similar to those described in estuarine and salt-marsh environments (Williams 2003), and there is potential to apply the more extensive research (e.g., on redox-based metal attenuation processes) in those environments to the hyporheic zone of regulated lowland rivers.

The presence of organic carbon rich and anoxic sediments presents potential benefits in terms of the increased potential for the sorption of organic pollutants that discharge from groundwater, denitrification, and anaerobic biodegradation of susceptible substances such as chlorinated solvents. The nature of lowland hyporheic zones and their potential as an anoxic barrier, analogous to an engineered permeable reactive barrier, is worthy of further investigation.

4 Chemical processes in the hyporheic zone

4.1 Attenuation processes in aquifer systems

Chemical and microbial substances in the environment are subject to a range of physical, chemical and biological processes that often act to reduce concentration and mass. Within the sub-surface environment, processes that act to degrade pollutants or retard their movement through soils or aquifers are jointly termed ‘natural attenuation’ processes. The Environment Agency (2000) has defined natural attenuation within groundwater systems as:

‘*The effect of naturally occurring physical, chemical and biological processes, or any combination of those processes to reduce the load, concentration, flux or toxicity of polluting substances in groundwater. For natural attenuation to be effective as a remedial action, the rate at which those processes occur must be sufficient to prevent polluting substances entering identified receptors and to minimise the expansion of pollutant plumes into currently uncontaminated groundwater. Dilution within a receptor, such as in a river or borehole is not natural attenuation.*’

Natural attenuation processes are commonly considered as part of the assessment of risks associated with contaminated groundwater or soils, as illustrated in *Figure 2.4*. Processes that degrade pollutant mass are normally regarded as the most important attenuation processes (Environment Agency 2000), and may take the form of either biotic or abiotically mediated reactions. In most groundwater environments, biotic processes are more significant (in terms of mass reduction) than abiotic processes, particularly for organic pollutants.

Non-destructive processes, such as sorption (to mineral surfaces and organic carbon), hydrodynamic dispersion and volatilisation, act to reduce the concentration of a pollutant, but do not reduce the total mass of the substance present. Retardation processes, including sorption, precipitation and fracture–matrix diffusion, slow the rate at which a substance moves through an aquifer and are collectively termed retardation. Other processes, such as volatilisation and dispersion, do not reduce the velocity of contaminant migration, but result in spreading of the substance through a larger volume of aquifer (dispersion) or partitioning into other non-aqueous phases (volatilisation). Both of these reduce the concentration in groundwater, although the total mass in the environment remains unchanged.

Different attenuation processes are relevant to different substances. Biodegradation processes are typically most important for organic pollutants, though a range of different reactions may be responsible for the degradation, while sorption reactions are often more important for the retardation of metals and other non-degradable substances.

The critical attenuation processes for common organic pollutants such as petroleum hydrocarbons, including the BTEX compounds, are often aerobic and anaerobic biodegradation, where the organic compound acts as an electron donor in a redox processes. Aerobic degradation processes are normally the most rapid for BTEX compounds (Wiedemeier *et al.* 1999). Attenuation of other common pollutants, such as chlorinated ethenes (the ‘chlorinated solvents’) is normally dominated by reductive dehalogenation reactions that only occur in reducing environments (Wiedemeier *et al.* 1999). Sorption of organic pollutants to natural organic matter in sediments may be important, particularly for hydrophobic substances. Sorption is normally described by a partition coefficient for a substance between water and soil carbon, K_d, which describes the equilibrium distribution of the substances between aqueous and solid phases. Substances with high K_d values, such as polynuclear aromatic hydrocarbons (PAHs), are retarded to soil and aquifer materials more strongly than are hydrophilic substances, such as phenol. BTEX and chlorinated ethenes are weakly sorbed and retardation is not normally significant (Wiedemeier *et al.* 1999).

Metals and other non-degradable cations are normally attenuated as a result of sorption onto clay minerals and oxides and/or hydroxides, complexation and precipitation of insoluble metal compounds. The principal attenuation processes for various groups of substances are illustrated in *Table 4.1*.

Table 4.1 Dominant attenuation mechanisms for common contaminant groups (after Environment Agency 2000).

		Non-degradative attenuation mechanisms[1]				Degradative attenuation mechanisms[2]					
Contaminant type	**Examples**	Methylation	Precipitation[3] and cation exchange	Sorption and binding[4]	Volatilisation[5]	Aerobic degradation[6] (contaminant as electron donor)	Anaerobic degradation[7] (contaminant as electron donor)	Reductive dehalogenation (contaminants as electron acceptor)	Fermentation	Co-metabolism	Oxidation/reduction
Chlorinated solvents[8,9] ≥ 3 chlorine atoms	PCE; TCE; TCA; TCM			✓	✓	✓?	✓	✓✓		✓	✓
< 3 chlorine atoms	DCM; VC; DCE			✓	✓	✓✓	✓	✓	✓[10]	✓	
Petroleum hydrocarbons	BTEX; middle distillates			✓	✓	✓✓	✓		✓	✓	
Oxygenates	MTBE, TAME, EBTE				✓	✓✓				✓	
Heavy metals (cationic)	Hg, Cd, Zn, Pb, Ni, Sr, Co	✓[11]	✓	✓✓							
Heavy metals (anionic)	CrO_4, AsO_4, AsO_3, TcO_4			✓✓							
Inorganics	Ca, Mg, Si		✓✓								
Anions	PO_4, BO_3		✓✓	✓✓							
Ammonia	NH_3, NH_4^+		✓✓	✓		✓					
Cyanide	CN			✓		✓					✓
Nitrate	NO_3^-						✓		✓		✓✓
PAHs	Naphthalene			✓✓		✓	✓			✓?	
Creosote	Phenols and phenolics				✓	✓✓	✓✓		✓	✓	
Pesticides	See footnote[12]			✓		✓	✓	✓?		✓?	

✓✓ = primary importance; ✓ = secondary importance; ? = some doubt exists over the process
NB. Dispersion and diffusion are applicable to all contaminants and have, therefore, been excluded.

1. Sorption and redox reactions are the dominant mechanisms that reduce the mobility, toxicity or bioavailability of inorganic contaminants. Precipitation reactions and absorption into the solid matrix of a soil (via occlusion, diffusion into dead-end pores or structural collapse of the mineral around the sorbed species) are generally stable, whereas surface adsorption and organic partitioning (complexation) are more reversible. Contaminant concentrations, pH, redox potential and chemical speciation may release a previously stable contaminant.
2. Unlike metals, which cannot be destroyed, the mass of organic and inorganic compounds (such as ammonia) can be reduced by (bio)chemical reactions that include hydrolysis (reaction with water, acids and bases), photolysis (reaction with

sunlight or with reactive radicals produced by light energy), biodegradation (reaction with enzymes or other biogenic compounds) and oxidation (reaction with oxygen) and reduction–elimination reactions. Generally, the rates of abiotic processes are slow compared with those of biologically-mediated degradative mechanisms, a notable exception being the break down of 1,1,1-TCA. Biological degradation is the dominant process to control the fate and transport of many organic contaminants. Through a series of oxidation-–eduction reactions, dissolved contaminants such as trichloromethane are transformed into innocuous by-products such as carbon dioxide, chloride, methane and water. However, intermediates may be generated that are more toxic and mobile than the original compounds. Transient degradation occurs by the following mechanisms:

- Aerobic degradation – transformation and/or elimination of an organic compound by micro-organisms in the presence of oxygen. This is often the most thermodynamically favoured reaction, providing the greatest energy to the micro-organism.
- Anaerobic degradation – transformation and/or elimination of an organic compound by micro-organisms in the absence of oxygen. Compounds other than oxygen act as electron acceptors, for example nitrate, manganese IV, iron III, sulphate and carbon dioxide.
- Halorespiration – the contaminant acts as an electron acceptor and hydrogen from fermentation of organic compounds acts as the electron donor.
- Fermentation – oxidation and reduction reactions that involve the transfer of electrons between organic compounds.
- Co-metabolism – an enzyme or cofactor produced by a micro-organism fortuitously degrades an organic contaminant. For example, anaerobic co-metabolism ('reductive dechlorination') is a common degradation route for highly chlorinated aliphatic compounds such as PCE and TCE.

3. For example, precipitation as weakly soluble sulphides and carbonates.
4. For example, sorption to iron hydroxides, carbonate minerals and clay matrices.
5. Primarily in the unsaturated zone.
6. Aerobic degradation – oxygen acts as the electron acceptor.
7. Anaerobic degradation – denitrification, Fe(III) reduction, sulphate reduction, methanogenesis (i.e., NO_3, Fe and Mn, SO_4 and CO_2 – in that order, but in part pH dependent) plus chlorinated solvents act as electron acceptors. VC and DCM are the only known examples of chlorinated solvents that undergo anaerobic degradation as electron donors.
8. PCE = tetrachloroethene, TCE = trichloroethene, TCA = 1,1,1-trichloroethane, TCM = trichloromethane, DCM = dichloromethane, VC = vinyl chloride, DCE = dichloroethene.
9. Highly chlorinated solvents do not degrade aerobically.
10. DCM is susceptible to degradation by fermentation.
11. Mercury may undergo methylation and subsequent volatilisation.
12. Chlorinated (lindane), organophosphate (malathion, diazinon), pyrethroid (cypermethrin), triazine (simazine), phenyl urea (isoproturon, diuron), phenoxyacid (mecoprop), cationic (paraquat).

The conditions present in the hyporheic zone, such as steep chemical (e.g., redox) gradients, dynamic exchange of oxygen, carbon and nutrients from the stream channel, a dense microbial and invertebrate community and potential for elevated water temperatures (both relative to aquifer conditions), mean that attenuation processes may be more rapid in hyporheic sediments than in aquifer sediments. However, the residence time in the hyporheic zone is likely to be small relative to that in an aquifer.

To date, the majority of studies of hyporheic attenuation have focussed on agriculturally derived nutrients, such as nitrate and phosphate. A small number of studies on the hyporheic attenuation of BTEX (Westbrook *et al.* 2005) and chlorinated ethenes (Conant *et al.* 2004) have been published in recent years, but there remains significant opportunity to transfer aquifer attenuation science and methods to studies of the significance of these (and other) processes in the hyporheic zone. While some European studies have investigated nutrient cycling in large lowland rivers, such as the Rhône, Danube and Elbe, the great majority of hyporheic research has focussed on small, unregulated headwater streams, often in arctic, alpine or desert environments. Study of the transferability of understanding between headwater streams and large regulated lowland rivers might usefully examine the existing body of diffuse–source pollutant literature.

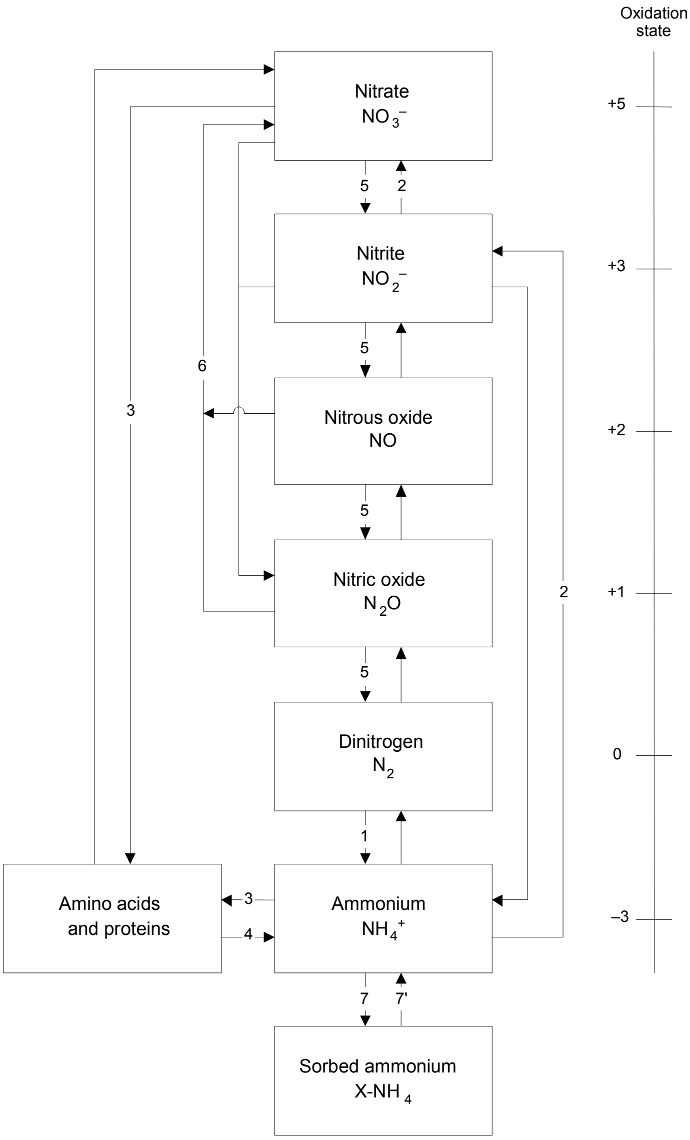
4.2 Cycling of diffuse pollutants

4.2.1 Nitrogen

Nitrogen, along with carbon, oxygen, hydrogen and phosphorus, is one of the most important elements in living matter. Nitrogen is a vital component of proteins and nucleic acids and constitutes around 10 percent of the dry-weight mass of bacteria (Duff and Triska 2000). Bacteria, fungi, algae and plants assimilate nitrogen as nitrate, NO_3^-, or ammonium, NH_4^+.

Dissolved organic nitrogen (DON) is usually scarce in pristine aquatic environments and consequently nitrogen is often the limiting variable in biological productivity. Most nitrogen is bound to organic matter and is not bioavailable. Nitrogen assimilation is generally via mineralisation of NH_4^+, which can be taken up or converted into nitrate by bacteria (Duff and Triska 2000).

Some organisms reduce NO_3^- to nitrogen gas (N_2), a process known as denitrification, while others fix N_2. Virtually all nitrogen transformations in aquatic systems are biologically mediated. Reactions that transform reduced nitrogen (valency state –3) to oxidised nitrogen (valency +5) release energy that is used by bacteria. Reactions that reduce nitrogen require energy from sunlight or organic matter (*Figure 4.1*).



Key:
1. Fixation*
2. Nitrification*
3. Assimilation by plants
4. Mineralisation*
5. Denitrification*
6. Nitrate in precipitation (as dilute HNO_3)
7. Adsorption and desorption

*Microbially-mediated processes

Figure 4.1 Chemical species in the nitrogen cycle (after O'Neill 1985, © Chapman Hall 1995).

In fluvial systems, nitrogen occurs as N_2, NH_4^+, R-NH_2, NO_3^-, NO_2, NO and N_2O. Between NH_4^+ and NO_3^- there is an eight-electron shift in valency.

The major inputs of nitrogen in headwater streams are particulate, associated with leaf litter and horizontal migration from adjacent forest-floor detritus (Duff and Triska 2000), and atmospheric equilibration. DON and dissolved inorganic nitrogen (DIN) inputs are minor in such environments.

Groundwater may provide some nitrogen, though most pristine groundwater has nitrate concentrations below 3 mg NO_3/l. In agricultural catchments, however, application of synthetic fertilisers increases DIN in groundwater, and many aquifers in lowland England exceed the European Union threshold of 50 mg NO_3/l. Point sources, such as treated sewage-effluent discharges and the release of untreated sewage via combined sewer overflows, can increase river and aquifer nitrogen concentrations significantly.

Nitrogen cycling in surface water streams is well documented (Duff and Triska 2000), but investigations of these processes in the hyporheic zone are less well documented. Most studies of nitrogen at the interface of groundwater and surface water environments have focussed on riparian zones and the management of riparian buffers to moderate the nutrient export from agricultural fields to rivers (Cirmo and McDonnell 1997, Burt *et al.* 1999, Cey *et al.* 1999, Hill *et al.* 2000, Angier *et al.* 2002, Puckett 2004). Reviews by Hill (1996) and Puckett (2004) of nitrate removal efficiency in the riparian zone indicate that, at most sites studied, greater then 80 percent of the influent nitrate was removed in the riparian soils and sediments.

Nitrogen cycling in the environment is dominantly a biologically mediated redox-based process. Dissolved oxygen concentrations strongly influence nitrogen transformations, and in hyporheic sediments the concentration of dissolved oxygen is a function of stream–hyporheic zone exchange rates and residence times, consumption of oxygen through respiration and chemolithotrophic processes (Triska *et al.* 1993b). Where consumption of oxygen exceeds its import rate via hydrological exchange processes, anaerobic conditions and a redox gradient develop (Duff and Triska 2000). Reduced and oxidised forms of nitrogen often co-exist under such conditions (Duff and Triska 2000) and steep redox gradients and fine-scale heterogeneity can result in multiple microbially-mediated redox processes in close proximity. This can be further developed where biofilm formation creates chemically diverse environments at a very small scale (Costerton *et al.* 1995).

When oxygen depletion by heterotrophic and chemolithotrophic metabolisms exceeds oxygen input, the system becomes anoxic. At this point facultative anaerobes (bacteria that can survive in the presence or absence of oxygen) switch from oxygen to NO_3^- as the next thermodynamically favourable electron acceptor – a process known as denitrification (see *Figure 4.1* and, for example, Equation 4.1). As oxygen concentration declines further, obligate anaerobes (bacteria that only survive in the absence of oxygen) begin to use other electron acceptors.

$$5CH_2O + 4NO_3^- \rightarrow 2N_2 + 4HCO_3^- + CO_2 + 3H_2O \qquad (4.1)$$

Denitrifying bacteria can use a variety of organic substances as carbon and energy sources. Although denitrification is less thermodynamically favourable than oxygen respiration (Hedin *et al.* 1998), mineralisation of organic matter can be significant in anaerobic hyporheic sediments if NO_3^- is available, which results in a sink for NO_3^-. Although natural NO_3^- concentrations are low, intensive agriculture results in a high NO_3^- concentration in large areas of lowland England. Inputs from industrial and waste-water infrastructure (e.g., leaking sewers and sewer overflows) have similar effects in urban areas.

In low NO_3^- hyporheic zones, the mineralisation of organic nitrogen may be the primary source for denitrifying bacteria (Duff and Triska 2000). Consequently, ammonification and nitrification become closely linked to denitrification in sub-surface flow lines (Jones *et al.* 1995).

Denitrification rates in hyporheic zones have been shown to increase with increasing labile carbon (acetate) concentrations and temperature (Pfenning and McMahon 1996, Hill *et al.* 2000), although increasing NO_3 concentrations did not affect the reaction rate, which indicates that the carbon, rather than NO_3^- was rate limiting in that system. These observations explain why denitrification rates observed in lotic systems are commonly slower in winter than in summer (cooler temperature) and suggest that carbon may be limiting in many unpolluted hyporheic systems (Pfenning and McMahon 1996). Working on the River Garonne in France, Pinay *et al.* (2000) observed that denitrification rates in floodplain sediments were related to sediment grain size. Below a threshold of 65 percent silt and clay they did not observe denitrification, but above 65 percent denitrification increased proportionally to the fines content. It is postulated that the reduction in hydraulic conductivity, and consequent limitation on atmospheric oxygen migration, ensured anaerobic conditions only in clay-rich sediments. Similar relationships between denitrification rate and sedimentology were also seen by Triska *et al.* (1989, 1993b).

Work published on nitrogen fluxes in estuarine sediments has shown that nitrogen cycling in such environments can lead to a release of gaseous N_2 and N_2O to the overlying river channel and the atmosphere (Dong *et al.* 2002). Dong *et al.* (2002) estimate that up to 60 percent of the marine N_2O flux is derived from denitrification in estuarine sediments, which is seen to vary diurnally in response to changing tidal conditions. Denitrification was greatest when estuarine sediments are submerged by tidal water, and in the River Colne at least was greatly influenced by nitrite concentration. Dong *et al.* (2002) concluded that a combination of microbial processes occur in estuarine sediments:

- Anaerobic bacteria transform oxidised nitrogen (as NO_2^- or NO_3^-) $\rightarrow N_2$
- Nitrite denitrifiers transform $NO_2^- \rightarrow N_2O \rightarrow N_2$
- Obligate nitrite denitrifiers transform $NO_2^- \rightarrow N_2$, but no further.

Heterogeneity of flow has also been shown to affect the distribution of denitrification in hyporheic sediments. Down-flowing water was correlated to an increase in the rate of denitrification, which it is suggested results from extensive biofilm development in that location, with a consequent decrease in hydraulic conductivity (Doussan *et al.* 1997) and isolation of the denitrifying bacteria from oxygen in the down-flowing water (Storey *et al.* 2004).

In addition to denitrification during the flow from aquifer to river, exchange of river water from the channel into and out of the hyporheic sediments can further reduce nitrate concentrations (McMahon and Böhlke 1996, Grimaldi and Chaplot 2000) as nitrate-rich water migrates from aerobic surface conditions into anaerobic and denitrifying sub-surface sediments. One possible consequence, as noted by Shibata *et al.* (2004), is that the hyporheic zone can become a net source of ammonium to the river. Since ammonium has a lower environmental quality standard than nitrate (50 µg NH_4/l compared to 50 mg NO_3/l), the potential for secondary pollution by ammonium should not be dismissed.

Studies of riverbank filtration schemes on the River Elbe in Germany (Grischek *et al.* 1998) and the River Seine in France (Doussan *et al.* 1997) have demonstrated rapid microbial denitrification in the riverbed sediments, which has been sustained over many decades. Infiltrating river water with elevated nitrate concentrations is consistently reduced in the first few metres of bed sediments. River concentrations of around 25 mg/l NO_3 in the Seine and 22 mg/l NO_3 in the Elbe are reported as decreasing to <0.5 mg/l NO_3 (Doussan *et al.* 1997) and around 10 mg/l NO_3 (Grischek *et al.* 1998) after the respective river waters have passed through a few metres of riverbed sediment. Doussan *et al.* (1997) report isotopic enrichment of $\delta^{15}N$ of +9.0 per thousand, associated with the riverbed denitrification.

Biofilm development at the river–sediment interface is significant in the River Seine (Doussan *et al.* 1997) and results in a decrease in bed permeabilities. However, modelling suggests that the denitrification efficiency is most sensitive to flow rate through the sediments and the organic matter content of the sediments. Development of biofilm, which takes place across the riverbed reasonably homogeneously increases the residence time within the sediments, and thereby further increases denitrification potential.

Nitrate is the most common groundwater pollutant in the UK and the environmental conditions under which it is converted into other nitrogen species are controlled by redox conditions and carbon supply (assuming that denitrifying bacteria are ubiquitous). Therefore, hyporheic and riparian zone processes are likely to be an important moderating control on the flux of nitrate between groundwater and rivers. Most research has focussed on riparian zone processes in attempting to understand floodplain nutrient processes and their relation to land management practice and river pollution. More recently, studies of artificial river-side barriers to biodegrade nitrate (Schipper and Vojvodić-Vuković 2000, Schipper *et al.* 2004) have been published as part of the remediation (permeable reactive barrier) literature. In the UK, pressure on land is great and the development of floodplains has resulted in an increasing exposure to flood risk and reduced land availability for floodplain management. Current flood-risk management practice in the UK seeks to decrease obstructions to water flow over floodplains, so as to reduce the severity of floods. The planting and management of riparian woodland and forests to improve baseflow water quality would appear to be in conflict with current management practices that are focussed solely on hydraulic issues.

The role of the hyporheic zone in cycling nitrate, and prospects for enhancing these processes to reduce nitrate, are likely to have fewer constraints than riparian buffer-zone development in the UK, because of fewer competing land-use pressures. Investigation of hyporheic nitrate transformation processes in a range of important hydrogeological terrains and the development of generic conceptual models of hyporheic nitrate cycling are likely to be important during the development of programmes of measures under the WFD. Rapid nitrification to increase NO_3^- concentrations has also been observed in hyporheic sediments (e.g., Butturini *et al.* 2000) and attention to the full nitrogen cycle, rather than simply to denitrification processes that may mitigate agricultural nitrate pollution, must be considered.

4.2.2 Phosphorus

There is an extensive literature on phosphorus dynamics in lakes, wetlands, streams and marine environments, but relatively little on the hyporheic zone or aquifer environments (Reddy *et al.* 1999). Phosphorus, P, is often a limiting nutrient in aquatic systems and is subject to complex processes that control its speciation, sorption and fate (Hendricks and White 2000). In aquatic systems, P may be present in a number of forms. It is usual to classify P into dissolved and particulate forms, which can then be further subdivided. Dissolved P is either dissolved inorganic P (DIP) which is generally bioavailable, or dissolved organic P (DOP), which includes colloidal organic P (generally unavailable to biota until it is converted into DIP).

Particulate P may occur within inorganic compounds complexes (e.g., clay minerals, iron hydroxides, carbonate), as particulate inorganic P (PIP), within organic matter complexes or as cellular components, including nucleic acid, phosphoprotein, vitamins and enzymes (particulate organic P, POP; Hendricks and White 2000). In lotic systems, complex interactions of P discharge, erosion and deposition, streambed geomorphology, sediment geochemistry, redox chemistry and P–biota interactions control the distribution of P (Hendricks and White 2000).

P is readily sorbed to metal oxides under oxic and high redox potential (Eh) conditions by a combination of chemical bonding, steric effects and electrostatic forces (Hendricks and White 2000). In particular, P sorbs readily to Fe(III) oxides, although aluminium oxyhydroxides, calcite and clay minerals also have strong affinities for P. As pH decreases, the surface charge on mineral surfaces changes and the sorption capacity for P on Fe and Al oxides and clay minerals increases further (Hendricks and White 2000).

Formation of vivianite, $Fe_3(PO_4)_2.8H_2O$, has also been observed in hyporheic sediments (Dodd *et al.* 2003, House 2003) and may be an important sink for P in certain hyporheic environments. Current understanding of the relative importance of vivianite formation is poor, but it has been reported to be most significant in anoxic and eutrophic environments, such as polluted canals, and may be important in analogous anoxic hyporheic zones of regulated lowland rivers.

By contrast, under anoxic reducing conditions, Fe(III) hydroxide–P complexes break down and PO_4 is released into solution. The redox control on P complexation is likely to be an important control on P retardation in hyporheic zones, where steep and dynamic chemical (including redox) gradients are found. A number of workers have observed that oxic hyporheic sediments provide rapid retention (retardation) of P following exchange from the stream (Mulholland *et al.* 1997, Butturini and Sabater 1999, Bonvallet Garay *et al.* 2001). Further, Carlyle and Hill (2001) identified coupled desorption of P–Fe oxide into aqueous solution as PO_4 with reduction of Fe(III) to Fe(II).

The transfer of P between stream and riverbed sediments is primarily by deposition and resuspension of particulate matter (Reddy *et al.* 1999, Bonvallet Garay *et al.* 2001). Once in the sediments, sorption of P from exchanging water onto sediments has been shown to occur rapidly and to be concentrated in the first few centimetres below the streambed (Butturini and Sabater 1999). Furthermore, the import of oxygen along with P in down-welling stream water results in a deeper aerobic zone in the hyporheic sediments and increases the sorption capacity for P (Reddy *et al.* 1999).

With regard to biotic processes, biofilm production in hyporheic sediments results in highly heterogeneous redox potentials controlled by biofilm structure and processes at the sediment scale. Biotic processes in aerobic areas of the hyporheic zone are significant in terms of mineralisation of complex organic P to soluble bioavailable forms (such as PO_4), biosynthesis of inorganic P and uptake in hyporheic biomass, and cellular P cycling (Hendricks and White 2000). Studies have identified a range of exoenzymes in hyporheic environments, which suggests a microfaunal demand for P, mainly within biofilm communities (Hendricks and White 2000).

P attenuation capacity can be related to hyporheic sedimentary structure and fluid dynamics, since P sorption capacity is directly affected by the exchange of water between the stream and hyporheic sediments (Hendricks and White 2000). Sand and gravel bed sediments with a rapid flow of oxic stream water have an increased capacity for P sorption, while lower permeability sediments and organic-rich sediments, with anaerobic conditions, decrease P adsorption potential (Hendricks and White 2000).

A conceptual model of P dynamics over relatively small-scale riffle structures has been proposed, which relates P sorption and/or mobility to interstitial hydrochemistry within the riffle structure (*Figure 4.2*).

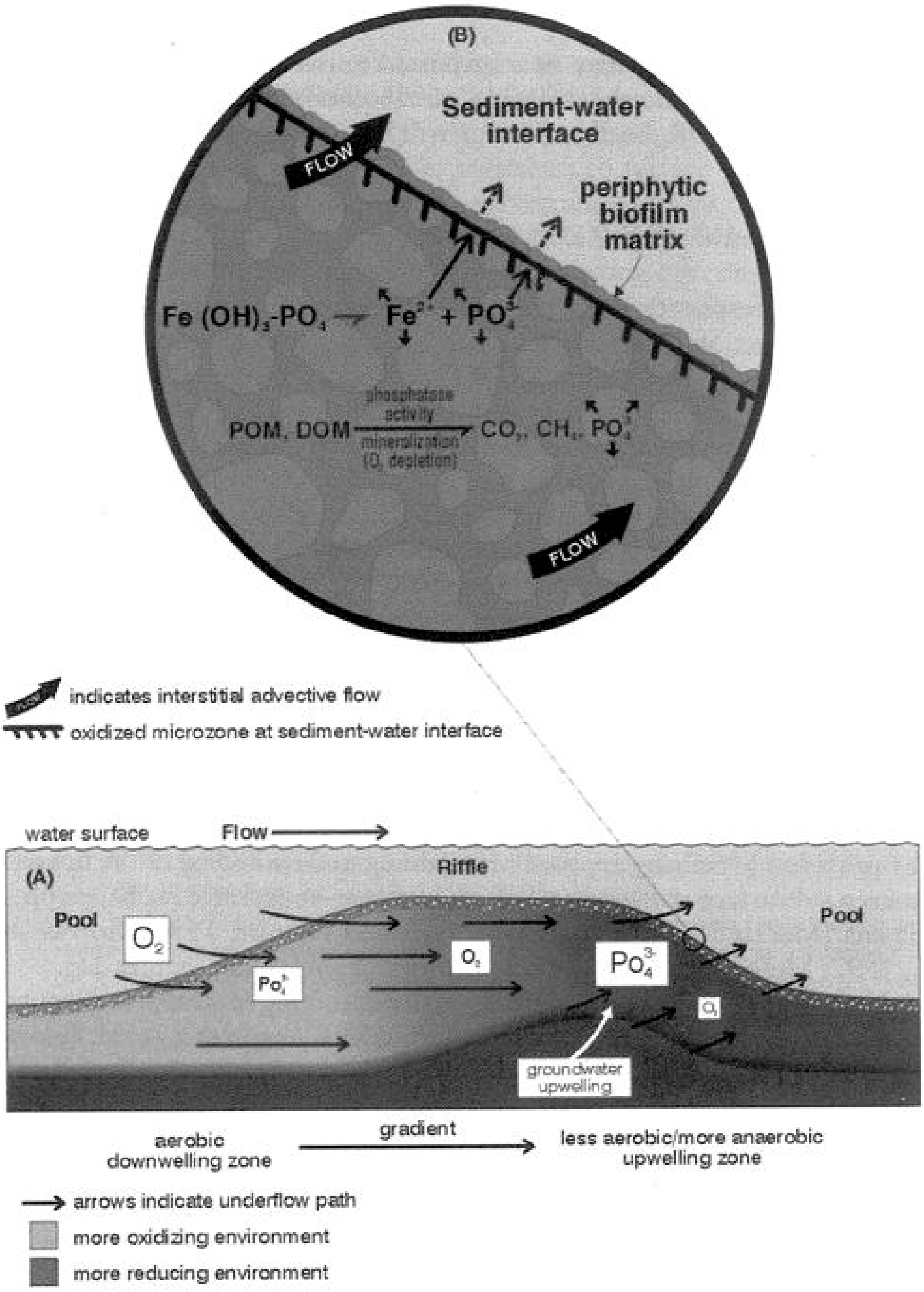


Figure 4.2 Conceptual model of hyporheic flow paths through an idealised riffle, showing changes of oxygen and phosphate concentrations in response to the aerobicity of interstitial waters. Relative concentration is indicated by the size of white box (based on Hendricks and White 2000, © Academic Press, Inc. 2000).

4.3 Attenuation of point-source pollutants

4.3.1 Chlorinated solvents

Chlorinated ethenes – commonly referred to as chlorinated solvents – are one of the most frequently identified groundwater pollutants, as a result of their widespread use in industrial, engineering and dry-cleaning processes, and of their relative recalcitrance in the natural environment. Highly chlorinated ethenes, such as PCE and trichloroethene (TCE), are biodegraded only under reducing conditions. The biodegradation daughter products of TCE, including *cis*-dichloroethene (DCE) and vinyl chloride (VC), are relatively reduced compounds

and have a lower tendency to undergo further reductive dechlorination (Bradley and Chapelle 1998). By contrast, DCE and VC can undergo direct or co-metabolic oxidation in aerobic conditions.
It has been suggested that an environment with sequential anaerobic then aerobic conditions would improve the potential for biodegradation of chlorinated ethenes and their break-down products. Engineered remedial techniques have been developed using this principle, such as the 'Lasagne technique' developed by the US Environment Protection Agency (USEPA) and Remediation Technologies Development Forum (RTDF) programmes, but the required sequential redox conditions also commonly occur naturally at the groundwater–surface water interface (Bradley and Chapelle 1998). Conant *et al.* (2004) observed that a PCE plume did not undergo biodegradation in an aerobic aquifer, but they identified significant anaerobic biodegradation of PCE in the upper 2.5 m of the streambed sediments. The observed concentrations of PCE were heterogeneous, mainly in response to a heterogeneous source and the hydraulic conductivity of the hyporheic sediments, which enabled diffusion and retardation of PCE into low permeability and high organic-carbon areas of the streambed.

Bradley and Chapelle (1998) investigated the biodegradation of the TCE break-down products DCE and VC in hyporheic sediments. Following reductive dechlorination of the TCE in an anaerobic aquifer, they observed that DCE and VC did not degrade rapidly under reducing conditions, but they identified significant oxidation of the DCE and VC to ethene and ethane in the aerobic parts of the near-stream hyporheic sediments of a Florida creek.

Similar investigations into the attenuation of TCE and tetrachloroethane (perchoroethane, PCA) into VC in the hyporheic zone below a wetland at the Aberdeen Proving Ground, Maryland, identified rapid anaerobic biodegradation of the TCE and PCA under reducing conditions, but accumulation of VC (Lorah and Voytek 2004). The study focussed on the deeper reducing environment of the wetland sediments, but it might be expected that the aerobic degradation processes identified by Bradley and Chapelle (1998) may also occur in the near-surface sediments and waters of the wetland.

4.3.2 Mine water and heavy metals

Investigations on the fate of metals in mine-water discharges have included studies on the stream–hyporheic exchanges of metals. Fuller and Harvey (2000) investigated a creek contaminated with mine water in Arizona to assess the uptake of bromide and metals from the stream into the underlying hyporheic zone. They report that the average metal removal from a 7 km reach into the hyporheic zone of the Pinal Creek catchment (Arizona) was cobalt 52 percent, nickel 27 percent, zinc 36 percent and manganese 22 percent. Further investigations indicated that the oxidation of mine-water derived manganese within the hyporheic zone gave rise to manganese oxides that increased the sorption potential for other metals. The continued discharge and oxidation of manganese in hyporheic sediments provided continued sorption capacity for the other metals discharged. Over a 7 km reach of the creek, they observed between 12 and 68 percent removal of metals from the creek into the hyporheic sediments, depending on flow and climatic conditions.

Moser *et al.* (2003) have shown that chromium can be reduced in hyporheic zones. Working on a Cr(VI) groundwater plume, they observed that, as a result of microbial processes and abiotic reactions in the reducing and anaerobic hyporheic sediments, Cr(VI) was converted into Cr(III), the less toxic and generally less mobile form.

Winde and van der Walt (2004) similarly observed the attenuation of uranium in hyporheic sediments as tailings dam-derived uranium in groundwater was precipitated in the hyporheic zone. They were able to show that the rate of attenuation varied during the day in response to temperature, pH and alkalinity, and that sorption of uranium onto manganese and iron oxides was an important retardation process that decreased the flux of uranium into the stream channel.

In the UK, studies on metal fluxes in rivers that drain mining areas and non-mining areas have been compared (Walling *et al*. 2003). Their work highlighted the need to understand the catchment geochemistry and pollutant availability (e.g., changes caused by mining) and the control of the sediment grain size on retardation, storage and transport in the respective catchments.

5 Biology of the hyporheic zone

Early descriptions of sub-surface ecosystems where developed with reference to, and experience of, surface systems (Gibert *et al.* 1994). However, numerous physical and chemical differences between surface and sub-surface environments affect the nature and distribution of hyporheic organisms and their ecological functioning (Gibert *et al.* 1994, Brunke and Gonser 1997, Boulton *et al.* 1998), as summarised in *Table 5.1*.

Table 5.1 Comparison of surface and sub-surface characteristics and the implications for invertebrates (based on Gibert *et al.* 1994).

	Underground system	Surface system
Environment	Constant darkness Habitats have restricted variety (no vegetation) and are small in size Physical inertia, predictable conditions (e.g., temperature)	Light (alternate day/night) Habitats have high diversity Frequent and large environmental fluctuations, low predictability (e.g., temperature)
Organisms	Morphological, physiological and behavioural specialisation to sub-surface environment	No adaptation to underground environment (except ecological one for ubiquitous species)
Biocenosis	Richness, diversity and density: low and variable Biotic constraints: low	Richness, diversity and density: generally high and variable Biotic constraints: generally higher
Functional characteristics	Heterotrophy and allotrophy Low trophic resource polyphagus diet (detritus feeders dominate) Short and simple food webs System with low productivity	Autotrophy Very important trophic resources High diversity of diets and possibility of highly complex food webs System with higher productivity

The most significant physical differences between surface and sub-surface (including hyporheic) environments are light (or its absence), which controls the biotrophic strategies of resident organisms, and the size and shape of habitats within sediments (i.e. the pore-space) which limits the size and mobility of its animals to small creatures with locally distributed community structures (Gibert *et al.* 1994).

Sub-surface animals tend to display characteristics that reflect evolutionary development in a dark interstitial environment, including a lack of pigmentation, blindness or reduced use of eyes, and hypertrophy of sensory organs (Gibert *et al.* 1994). Organisms tend to be relatively longer and thinner than surface forms, as a consequence of limited living space within the sediments, and body appendages are common, often with highly developed chemical and mechanical sensors (Gibert *et al.* 1994, Gayraud and Philippe 2003).

The low energy environment means sub-surface fauna generally have slow metabolic rates and their life-stages may be of longer duration than similar surface forms, with associated lengthened life-cycles and reduced frequency of reproduction.

Hyporheic organisms have been observed to include a wide range of faunal groups, with a variety of life strategies. Some organisms are only found in the hyporheic zone and spend their entire life-cycle in the hyporheic zone – the permanent hyporheos (*Figure 5.1*). Other organisms are surface –dwelling, but spend their larval stages in the hyporheic zone, using it as a nursery and refuge from predators in the river. They return to the surface water for their adult stages and are known as temporary or occasional hyporheos. If organisms are adapted to long (essentially permanent) residence in interstitial habitats, but for a short period during their life-cycle they exit the sediment for the surface water, they are termed amphibites.

Organisms that are adapted to surface dwelling, but accidentally occur in sediment or cave systems, are known as stygoxenes. Although not adapted to sub-surface conditions, stygoxenes can affect ecosystem functioning by acting as prey or predator (Gibert *et al.* 1994). Finally, stygobites are organisms that are adapted to permanent specialised sub-surface conditions. Some ubiquitous forms are widely distributed in caves and alluvial systems, while others, known as phreatobites, are adapted to deep alluvial (interstitial) environments.

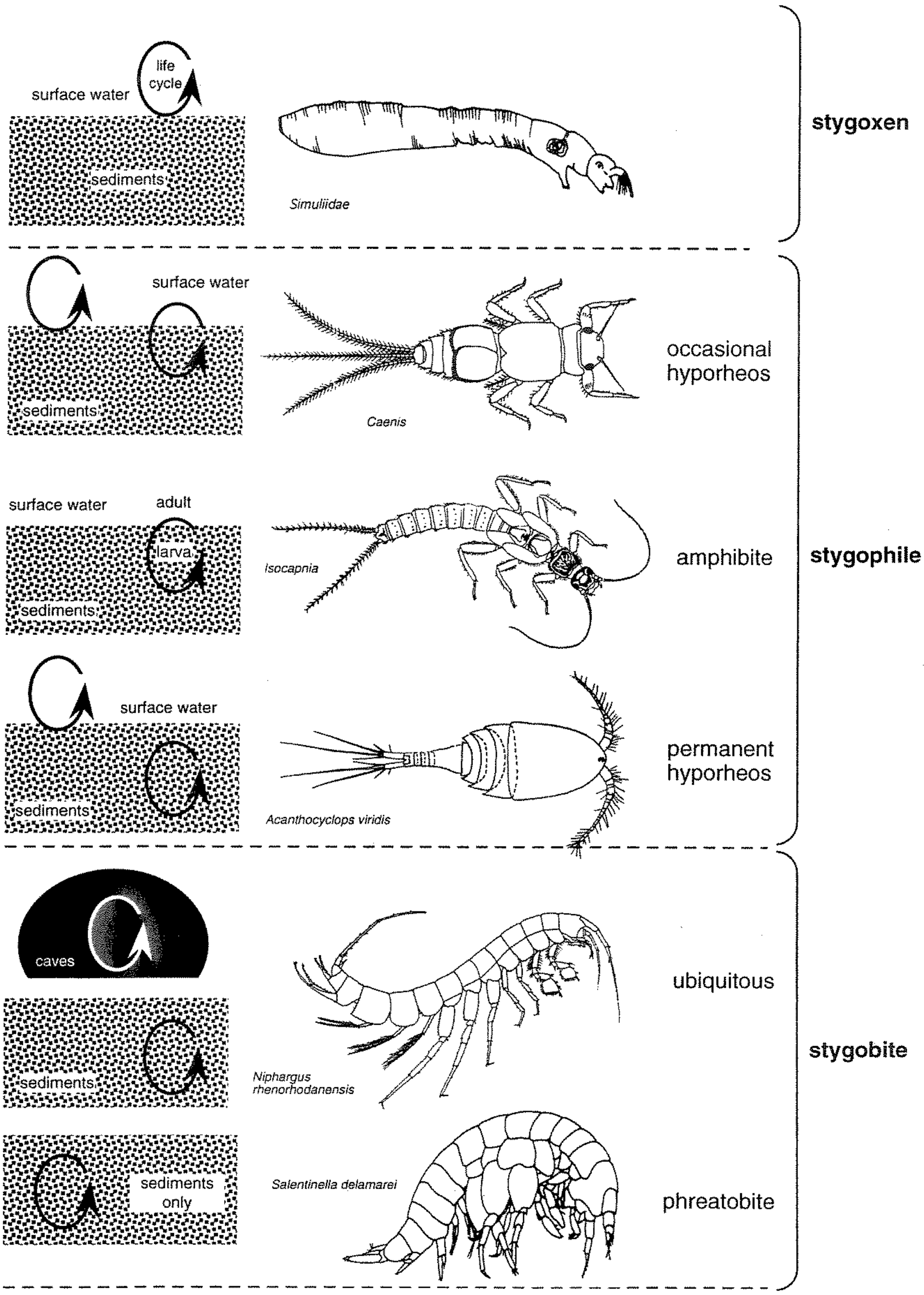


Figure 5.1 A phenologenic classification of fauna associated with the hyporheic zone and groundwater (after Gibert *et al.* 1994, © Academic Press, Inc. 1994).

Lacking light, hyporheic and groundwater food webs are invariably heterotrophic (Gibert *et al.* 1994, Feris *et al.* 2003b). Biological production is largely dependent on the influx of carbon and nutrients from the stream or groundwater. The latter is particularly important in polluted aquifers that contain, for example, elevated concentrations of nitrate, phosphate or carbon from agricultural or industrial pollution sources, because pristine groundwater generally is normally oligotrophic.

Hyporheic fauna include microbes (bacteria, fungi, protozoa), micro-invertebrates, meio-invertebrates and macroinvertebrates. Occasionally, vertebrates, such as fish and amphibians, use the benthic and hyporheic zones for reproduction and refugia (Magoulick and Kobza 2003).

5.1 The hyporheic zone as an ecotone

The hyporheic zone is the interface between groundwater and surface water systems, and as such is subject to numerous physical and chemical gradients, which in turn affect the biological patterns (Sabater and Vila 1991, Brunke and Gonser 1997). The transition zone between groundwater and surface water ecosystems is recognised as a critically important boundary, or ecotone (Vervier *et al.* 1992, Stanford and Ward 1993). While conditions in a groundwater ecosystem are normally relatively stable, conditions in this ecotone change dramatically in both time and space. Magnitude and direction of water, matter and energy transfer vary and have consequent affects on the fauna present (Brunke and Gonser 1997). The hyporheic ecotone may also function as a sink of organic and/or inorganic matter from the catchment and thereby control the metabolic rates in adjacent groundwater systems (Gibert *et al.* 1994).

Although the earliest research on the hyporheic zone presented a picture of a static system, as a result of the interpretation of spot (non-repeated) data collection, it has been recognised more recently that the hyporheic ecotone is a dynamic system that varies daily, seasonally (Wondzell and Swanson 1996a, Wroblicky *et al.* 1998) and in response to climatic extremes, such as floods and droughts (Brunke and Gonser 1997, Boulton 2003, Magoulick and Kobza 2003). Climatic and hydrological heterogeneity give rise to seasonal variations in hyporheic habitats that are also evident as a dynamic microbial community with marked seasonal variation (Feris *et al.* 2003b).

Definitions and hyporheic zone delineation attempts commonly fail to define the parameters and/or processes considered important to the biogeochemical or ecological function being investigated, and different approaches invariably result in different descriptions of the hyporheic zone (Gibert *et al.* 1994). A more profitable approach is likely to be to consider the ecological function of the hyporheic zone, how it varies and the controls on that variability (Brunke and Gonser 1997, Boulton *et al.* 1998).

5.2 Microbial fauna in the hyporheos

5.2.1 Bacteria

Micro-organisms are reported to account for over 90 percent of the community respiration in hyporheic sediments (Brunke and Gonser 1997). The hyporheic microbial consortia are dominated by bacteria that attach to surfaces and are present in biofilms (Gounot 1994, Findlay and Sobczak 2000), where biofilm is defined as a matrix-enclosed bacterial population that adheres to surfaces or interfaces (Costerton *et al*. 1995). Biofilms normally comprise microcolonies of single bacterial taxa in close spatial arrangement with microcolonies of other taxa, within a polysaccharide matrix (Storey *et al.* 1999).

Biofilms play a number of important functional roles (Paerl and Pinckney 1996), including within the hyporheic zone. Firstly, they are known to be critical in the processing and cycling of organic carbon and nutrient (Boulton *et al.* 1998, Chafiq *et al.* 1999). These processes have been shown to provide attenuation of pollutants in channel water as a result of continuous exchange between the channel and hyporheic zone (Moser *et al.* 2003). Additionally, hyporheic biofilms are important in the stream–hyporheic food web (Barlocher and Murdoch 1989, Brunke and Gonser 1997, Pusch *et al.* 1998), which represents the base of the food web on which protozoa (Findlay and Sobczak 2000), invertebrates (Hakenkamp and Morin 2000, Burrell and Ledger 2003) and fish feed (Boulton *et al.* 1998).

Biofilms can be highly organised, with distinct vertical zonation of diverse microbial communities, which reflect, or eventually control, small-scale gradients in chemical or physical parameters (Findlay and Sobczak 2000). Consequently, a hyporheic zone that appears to be aerobic or anaerobic on the basis of water sampling from boreholes or peizometers can, in fact, contain a diverse range of chemical conditions that support a range of biochemical processes at the sediment scale. By associating in a biofilm, rather than moving freely in the water, bacteria benefit from an ability to assimilate complex organic matter and other essential nutrients, such as cations (Gounot 1994), associated with the particles to which they attach (Findlay and Sobczak 2000). However, metabolism within a biofilm is restricted by the rate at which organic matter and nutrients are imported into the hyporheic zone (Findlay and Sobczak 2000). Since groundwater systems are frequently naturally oligotrophic, hyporheic zones with down-welling stream water often support greater bacterial production than areas of up-welling groundwater (Hendricks 1993). Within hyporheic sediments, bacterial production is generally greater in near-surface sediments than in deep sediments (Findlay and Sobczak 2000). Nevertheless, the large surface area of hyporheic sediment available for bacterial colonisation means that it is now recognised as a site of high biological activity, which significantly influences stream ecosystem function (Storey *et al.* 1999).

Bacterial processes in hyporheic sediments have been shown to be important in moderating nitrogen chemistry through nitrification and denitrification processes (Triska *et al.* 1993b) and dissolved organic carbon cycling (Fiebig and Lock 1991, Findlay *et al.* 1993). However, the full role of hyporheic microbial processes has received little study in comparison to studies on stream or soil systems and many questions remain (Storey *et al.* 1999). There are large gaps in our understanding of the structure and function of microbial assemblages in hyporheic environments (Storey *et al.* 1999) and their relation to hydrological, geological and geomorphic controls (Hendricks 1993), and outstanding difficulties in up-scaling small-scale observations to reach or even catchment scale. Studies to assess bacterial processes in hyporheic sediments and its heterogeneity are needed to test the assumption that the hyporheic zone is a microbially active zone that has the potential to attenuate pollutants. Understanding the relation between sediment structure, flow and microbial processes would help to advance the conceptual understanding of the potential for attenuation in different parts of, or types of, hyporheic system.

5.2.2 Fungi

Fungi are known to be important components of microbial communities in soil and on leaf–wood detritus, but little is known about their role in hyporheic sediments (Findlay and Sobczak 2000). Studies have shown fungi to be present and able to colonise within the hyporheic zone (Barlocher and Murdoch 1989, Crenshaw *et al.* 2002a). The deposition of leaf litter, wood and other organic materials in riverbed sediments, and the generally high concentrations of POM in hyporheic sediments, may result in dense fungal colonies and production (Storey *et al.* 1999). In surface environments fungi are more influential than bacteria in the initial decomposition of POM, being able to break down cellulose and lignin, which comprises cell walls. However, bacterial processes dominate the later stages as the particle size is reduced (Storey *et al.* 1999).

It has been suggested that hyporheic fungal abundance, distribution and dynamics may be closely linked to the timing and patterns of coarse, particulate organic-matter entry (Storey *et al.* 1999). At present, particulate organic-matter input is poorly understood, but linking investigations of hydraulic controls on particulate organic-matter input to a study of the role of fungi in hyporheic processing would provide a more complete understanding.

Fungal productivity is likely to reflect the detrital content of the streambed sediments. Some authors have suggested that the majority of the organic matter storage in the total stream system is within the hyporheic zone (Storey *et al.* 1999). In this scenario the importance of hyporheic fungi for organic carbon cycling may be disproportional to their contribution to microbial biomass.

5.2.3 Protozoa

Protozoa are single-celled eukaryotes that lack cell walls. They are intermediate in size between bacteria and larger invertebrates and are generally considered to be an important component of microbial food webs (Findlay and Sobczak 2000). Very little research has been reported on the nature, abundance or productivity of protozoa in hyporheic zones, although Findlay and Sobczak (2000) review two studies that did consider the role of protozoa. In a UK chalk stream, grazing by protozoa was responsible for the removal of 20 percent of the bacterial production, while a similar short-term study in an American creek was extrapolated to produce estimates that protozoan grazing could account for the entire annual bacterial production within the system. Under such conditions, protozoa may be the primary consumer of bacteria and a vital trophic intermediate between bacteria and invertebrates.

5.3 Hyporheic invertebrates

Invertebrate fauna in the hyporheic zone are commonly subdivided, on the basis of organism size, into meiofauna and macrofauna (Gibert *et al.* 1994). Meiofauna are intermediate in size between microbes and macrofauna, and are defined as those creatures between 100 and 1000 μm (0.1–1 mm; Robertson *et al.* 2000a). Macrofauna exceed 1000 μm in length (Hakenkamp and Palmer 2000). Practically, meiofauna are generally separated from other organisms by passing through a 1000 μm filter, while being retained on a 63 μm filter (Robertson *et al.* 2000a).

Invertebrates are an important part of hyporheic ecosystems in terms of their position in the food web between microbial and larger animal, and consequently as a transfer of energy from microbial to higher forms (Hakenkamp and Palmer 2000, Burrell and Ledger 2003). In addition, they are able to manipulate their environment (Danielopol 1989, Mermillod-Blondin *et al.* 2003), which has associated impacts on the physicochemical (Mermillod-Blondin *et al.* 2003) and biogeochemical (Marshall and Hall 2004) environments in which they live.

Meiofauna in particular contribute greatly to the diversity of lotic ecosystems. For example, Robertson *et al.* (2000b) report that ecological investigation of seven streams identified that between 58 and 82 percent of the total species present were meiofaunal. It is commonly assumed, largely on the basis of static studies, that the meiofaunal contribution to stream (including hyporheic) biomass and productivity is small to moderate (Robertson *et al.* 2000b). However, more recent studies, which include a temporal sampling component, suggest that the effects of bioturbation and ‘micro-engineering’ may mean that invertebrate effects on biogeochemical processes are disproportionately large (Marshall and Hall 2004). The consequences of high diversity on the functioning of stream ecosystems is, however, currently unclear (Hakenkamp and Morin 2000, Robertson *et al.* 2000b).

In most hyporheic systems, organisms such as the microcrustaceans, tardigrades, rotifers, small oligochaetes and nematodes are the dominant meiofaunal animals (Boulton 2000a, Hakenkamp and Palmer 2000). However, sediment grain-size (Brunke and Gonser 1997, Hakenkamp and Palmer 2000), water flow direction (Malard *et al.* 2003, Olsen and Townsend 2003), organic matter content (Malard *et al*. 2003), topography (Olsen and Townsend 2003) and physical heterogeneity – often called ‘patchiness’ by ecologists – have all been shown to influence the distribution, composition and taxonomic diversity of hyporheic invertebrates (see Figure 5.2).

Many hyporheic meiofauna have evolved in interstitial habitats and have small size, elongate worm-like shape, adhesive organs (to stick to sediment), armour (as protection from moving and unstable sediment), direct fertilisation and lack of free-swimming larvae (Hakenkamp and Palmer 2000). Meiofauna do not possess a winged dispersal stage common in many lotic macroinvertebrates, but they are nonetheless able to move actively within the interstitial porosity to select a preferred habitat (Danielopol 1989).

Bioturbation during feeding or habitat construction modifies hyporheic zone properties by increasing sediment porosity, permeability and solute transport across the hyporheic zone (Mermillod-Blondin *et al.* 2000). Since invertebrate biomass and productivity have been observed to be proportional to dissolved oxygen concentration, bioturbation may improve productivity locally.

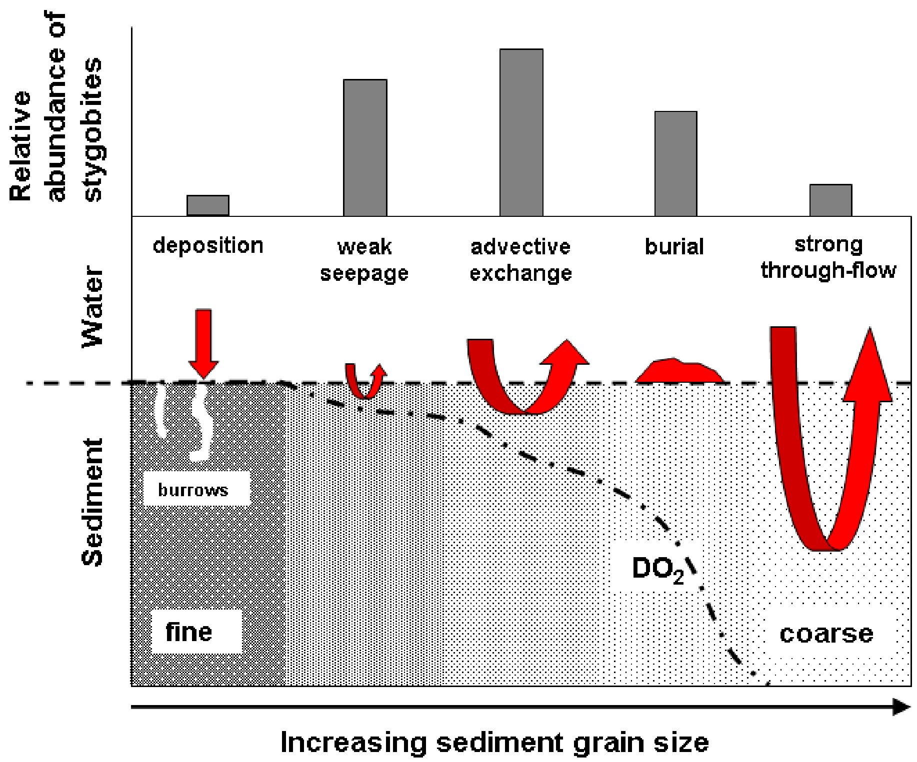


Figure 5.2 Chemical and redox gradients, and water exchanges across the hyporheic zone influence hyporheic zone ecology. (Based on Hancock *et al.* 2005)

Through feeding – normally grazing on biofilm – and subsequent predation by macroinvertebrates and vertebrates, meiofauna provide a trophic link between bacteria and larger fauna (Robertson *et al.* 2000b). Grazing on biofilms can also regulate carbon (Robertson *et al.* 2000b) and nitrogen (Marshall and Hall 2004) cycling in hyporheic sediments. Cophrogy by meiofauna (feeding on macrofaunal faecal pellets) similarly cycles carbon and nutrients (Robertson *et al.* 2000b).

The hyporheic zone may also be used as a refuge by invertebrates. Dole-Olivier *et al.* (1997) observed a heterogeneous ('patchy') distribution of refugia in hyporheic and benthic sediments during flood events. The magnitude of the flood, sediment stability and water flow direction (down-welling areas are preferred) were found to influence the distribution of refugia. Similar studies under drought conditions, however, indicated that the hyporheic zone may be a less important refuge for invertebrates during drying conditions (del Rosario and Resh 2000).

Macroinvertebrates include peracarids, stonefly and mayfly nymphs, and other insects (Boulton 2000a). In most hyporheic zones, macroinvertebrate biomass and production is much lower than that of the meiofauna; however, by virtue of their larger size and mobility, they play an important role in certain processes. They are more constrained in habitat selection than are meiofauna, by virtue of body versus sediment pore size (Boulton 2000a). Macrofaunal respiration may be a trivial part (<5 percent) of the total hyporheic oxygen consumption, but the functional roles of macrofauna (*Table 5.2*) may result in a disproportionate impact on hyporheic ecosystem function (Boulton 2000a).

Table 5.2 Functional roles of hyporheic macrofauna (based on Boulton 2000a).

Process	Effect
Burrowing	Fine-scale alteration of pore size and water velocity; changing redox gradients and supply of matter and/or energy to microbes and meiofauna; aeration of sediments; dispersal of bacteria
Ingestion of coarse faecal pellets	Fine-scale alteration of pore size; compaction of fine sediments; localised production of POM
Excretion	Increase labile dissolved organic matter (DOM) and nutrients, providing favourable conditions for microbes
Grazing on biofilm	Promotion of biofilm processes; role in natural gravel bar filters
Feeding on POM	Disaggregation of coarse POM buried in sediments
Predation on meiofauna	Alteration of population sizes; trophic transfer of energy
Prey for larger fauna	Alteration of population sizes; supply of energy to surface predators
Movement and migration	Transfer of matter and energy among sub-surface zones
Emergence of amphibites	Removal of matter and energy from sub-surface; food for riparian predators

While meiofauna are invariably worm-shaped and able to move through interstitial pore space, some macroinvertebrates, such as peracarid and syncarid crustaceans, are more robust and actively burrow in fine sediment (Boulton 2000a). Macroinvertebrates move to preferential habitat locations (Burrell and Ledger 2003) and are often in dense communities in association with high concentrations of POM and biofilm production (Barlocher and Murdoch 1989).

Various authors, including Notenboom *et al.* (1994) and Boulton (2000a, 2000b), have noted the potential of macroinvertebrate as indicators (biosensors) of anthropogenic impacts, such as pollution. Friberg *et al.* (2003) identified a negative correlation between amphipod populations and pesticide concentration in Danish river sediments, but also observed positive correlations for oligochaeta populations at the same locations. Grumiaux *et al.* (1998) observed deleterious impacts of industrially contaminated river quality and sediments on the macroinvertebrate community structure in a river in northern France. Invertebrate biological monitoring programmes, such as the Environment Agency's General Quality Assessment scheme, are invariably based on measures of macroinvertebrate communities, because of the difficulties of sampling and identifying meiofauna (Extence *et al.* 1999). The classification of rivers in England and Wales, based on benthic macroinvertebrate measures as a surrogate for whole-river biological health, is one of the government's headline indicators of sustainable development. However, the approach takes no account of meiofauna or microbial fauna, or of deeper hyporheic communities, which may be more sensitive to the effects of pollution in associated groundwater environments.

5.4 Fish and other vertebrates

The hyporheic zone has three main roles for vertebrates that inhabit river systems. Firstly, many vertebrates prey on smaller animals, which may either reside in the hyporheic zone or themselves prey on hyporheic fauna. Secondly, some vertebrates may use the hyporheic zone as a temporary refuge in times of environmental (e.g., climatic) stress and, finally, salmonids lay their eggs in hyporheic sediments.

Riverbed sediment food webs are complex and encapsulate fauna in a range of environments across the hyporheic ecotone. River-dwelling vertebrates generally feed either on plants or invertebrates that themselves prey on smaller benthic and hyporheic fauna. For example, Burrell and Ledger (2003) described how insects such as the caddisfly larvae graze on hyporheic biofilm, and in due course become prey for fish during their adult life. Although stream-dwelling vertebrates, such as fish and otters in the UK, are generally the focus for environmental protection and management, the great bulk of the biomass in river systems is in the microbial and invertebrate fauna that underpin the higher predators.

The hyporheic zone is used by a range of animals on an occasional basis as a refuge, particularly during periods of environmental or climatic stress. Certain fish use interstitial hyporheic sediments as a nursery for fry (Power *et al.* 1999), or for refuge for adult fish (e.g., lungfish) during periods of drought (Magoulick and Kobza 2003). Other vertebrates, including salamanders, similarly use the hyporheic sediments in periods of drought.

During cold weather, up-welling groundwater, which is at a relatively constant temperature throughout the year, prevents freezing of river water during the winter months and provides temperature moderation during hot summer periods (Power *et al.* 1999). The thermal stability of up-welling groundwater from the hyporheic zone can provide in-stream thermal refuges for fish and other aquatic animals during winter periods in arctic and alpine environments. Pringle and Triska (2000) state that, as a direct result of up-welling of groundwater over a large length of an Alaskan river, Chum Salmon spawning and the migratory patterns of the world's largest population of Bald Eagles are interlinked.

Salmonids lay their eggs in excavated depressions within riverbed sediments, known as redds (Power *et al.* 1999). The supply of cool oxygenated water is vital for the successful survival and development of salmonid eggs (Crisp 1990, Dent *et al.* 2000). During development, fish eggs require oxygen provided by vertical exchange, which may favour the construction of redds in areas with down-welling water (Jones *et al.* 1995). However, other workers noted that redds constructed in areas of up-welling groundwater benefit from thermal stability (Dent *et al.* 2000).

6 Conclusions: knowledge gaps and research needs

6.1 Priorities for environmental management

Results from WFD Initial Characterisation (Environment Agency 2004) indicate that the main causes for failure of groundwater bodies in England and Wales to achieve the environmental objectives set out in Article 4 of the WFD are, in order of frequency, the following:

1. Groundwater pollution by *nitrate* compromises drinking water protected area criteria and causes nutrient enrichment of dependent surface waters and terrestrial ecosystems;
2. *Over-abstraction* of groundwater causes flow depletion in dependent surface waters;
3. Groundwater pollution by *diffuse urban pollutants* and *industrial pollutants* (particularly chlorinated solvents, petroleum hydrocarbons and gasworks wastes);
4. Groundwater pollution by *pesticides*;
5. Groundwater pollution by *phosphate* causes nutrient enrichment of dependent surface waters and terrestrial ecosystems;
6. Discharge of *mine water* (coal and metalliferous) contaminated with heavy metals and having a low pH into surface waters.

A number of additional pressures pose a risk to the achievement of good status in a relatively small number of groundwater bodies. These pressures include saline intrusion, polluting discharges from landfill and sewage treatment infrastructure, pollution from transport networks, including highways, railways and airports and associated maintenance and storage at depots (e.g., salt storage and de-icing processes), and atmospheric deposition of pollutants to land. Many of these pressures are significant locally, but do not pose a significant risk when considered at a groundwater body scale. With regard to priorities for this research programme, the five pollution pressures listed in the bullet points above are considered to represent the principal groups of pollutants for which improved understanding of hyporheic zone attenuation processes would be most valuable.

More fundamentally, it has become clear from both European Commission (Council of the European Community 2002) and UK (Environment Agency 2004) assessments that current conceptual models of aquifer–river interactions are limited. Consistent conceptual models of flow and attenuation processes within the hyporheic zone that are relevant to the hydrogeological domain are needed. These models need to be developed to assess the impact of pollutant and/or flow modification on the health of ecological communities locally, and of the ecological health (ecological status) of the surface water body at the larger scale.

With regard to WFD objectives, the status of a groundwater body is dependent on the status of the surface water bodies that it discharges into, but the reverse is not the case. That is to say, if a polluted groundwater body causes a dependent surface water to fail to achieve good status, the groundwater body is deemed to be of poor status also. However, if a surface water body is of poor status, the groundwater bodies with which it is hydraulically connected are not automatically condemned as being of poor status. Consequently, for WFD purposes, the Environment Agency's priority is in understanding the influence of attenuation processes as water and pollutants migrate from groundwater into surface waters. Locally, attenuation of pollutants that infiltrate the riverbed or bank is important (Ray *et al.* 2002), but the heavy reliance on river bank filtration in central Europe to provide drinking water from wells in near-river alluvial aquifers is not widely replicated in the UK (Hiscock and Grischek 2002).

Generic conceptual models of groundwater–surface water interactions, and identification of the main parameters needed to characterise the hyporheic zone and quantify the impact of attenuation processes are needed. At the water body and/or catchment scale, precise quantification of the processes in the hyporheic zone is rarely needed, but to make decisions knowledge of the heterogeneity of the system is needed to ascribe confidence limits to predictions made within the risk-assessment process.

The WFD defines a series of typologies for surface water bodies, which includes geomorphological and geological properties. *Development of generic conceptual models of the extent of groundwater–surface water interactions and hyporheic processes for the WFD geological typologies* would be a useful starting point. The geological types are calcareous, siliceous and organic, and it is suggested that the development of generic conceptual models for catchments dominated by chalk–limestone, sandstone, peat and clay-drift could be considered initially. This could usefully be combined with the production of a *compendium and/or catalogue of information on groundwater–surface water interactions and in particular hyporheic zone size and nature in the UK*.

In due course, the process-based models for the hyporheic zone will need to be up-scaled to reach or, ideally, catchment scale: a task that might usefully be undertaken by workers in the Environment Agency and/or University of Sheffield Catchment Science Centre. Lateral heterogeneity within different parts of the river stage, and within regulated and unregulated river systems, will be important.

With regard to the WFD good ecological status objective, it is clear that there remains a significant gap between measuring pollutant concentration and/or flux and water level change, and being able to interpret the nature and magnitude of risks to ecosystem health. Although beyond the scope of the Environment Agency's hyporheic zone programme, it is clear that *further study into the links between environmental pressure and ecosystem response across the hyporheic zone is needed*. The position of the hyporheic zone, together with the riparian and benthic zones, means that they are likely to be critical areas for assessment because of the dynamic conditions and ecological diversity within these ecotones.

6.2 Hyporheic zone architecture

Existing literature on the relation of fluvial sedimentary structure and hyporheic processes concentrates on the interaction of morphological structure and either flow dynamics (e.g., hyporheic flow through palaeochannels) or habitat and/or refugia for hyporheic and benthic fauna. Very little effort has previously been made to link sedimentary structure and facies analysis with pollutant attenuation capacity.

It is recommended that further work should focus on two principal areas:

- On the basis that fine-grained (clay and silt) sediments and sediments rich in organic matter will offer the highest capacity for pollutant attenuation, by sorption and anaerobic degradation processes, the *relationship between sediment geochemistry and depositional environment in pristine or moderately regulated rivers should be investigated*.

  Field studies should investigate whether the deposition of fine-grained sediments in low-energy environments, such as pools and the inside of meander beds, leads to an increased potential to attenuate groundwater pollutants that enter at these locations, and whether gravel and cobble beds (e.g., riffles) offer significantly reduced attenuation capacity. Studies in active river channels might be integrated with the few existing studies on the lithostratigraphy and pollutant attenuation capacity of alluvial aquifers, such as the work by Kleineidam *et al.* (1999b).

- Many of the most important rivers in the UK are regulated lowland systems that are unlikely to have a significant 'classic' hyporheic zone. Instead, they are likely to have a bed of anoxic sediment as a consequence of river regulation (the effects of dams and weirs), deposition of carbon-rich sewage and agricultural effluents, and deposition of solids in run-off from roads and other areas of hard surfacing. Studies of the distribution and geochemistry of anoxic sediments in rivers are suggested, which would bring together studies of hyporheic processes with engineered remediation (permeable reactive barriers). The use of canals as analogues for urban rivers (but with lower flow and longer pollutant and sediment residence times, and ease of sampling) should be considered. The potential for anaerobic biodegradation of chlorinated solvents and precipitation and immobilisation of metals within bed sediments should be investigated. The potential for beneficial attenuation processes needs to be assessed, so that sediment management approaches that rely on removal of sediment can be designed to achieve the optimal environmental and amenity balance.

  Field-based studies of a lowland regulated river should be undertaken to determine the nature, thickness and lateral variability of sediment, and its potential to form an anoxic barrier that might enhance certain attenuation processes.

  Review and assessment of canals and the processes that control sediment–water interactions should be considered in an environment with slower hydraulic processes.

6.3 Flow and hydrology

There are a number of research needs in relation to flow at the groundwater–surface water interface. Estimating *in-situ* flow velocity remains a problem, particularly in deeper lowland rivers that are too deep to easily work with seepage meters. Many of the techniques developed to investigate groundwater–surface water flow in upland headwater catchments with shallow streams will not be applicable in larger rivers, and it is likely that techniques developed to investigate lake sediment processes may be more transferable.

Development, field –testing and/or demonstration of cheap and physically robust in-situ *flow detectors able to detect flow direction, ideally in three dimensions and at low velocities, would be beneficial*. To be of maximum application, *in-situ* sensors should be capable of being connected to data-logging equipment so that long-term records of flow variations within the hyporheic zone can be studied and correlated against seasonal and diel hydrological variations.

The majority of current studies examined the heterogeneity of flow within shallow braided streams or around riffles. This information is of little relevance to lowland rivers, and *studies to establish the degree and variability of flow between larger rivers and groundwater are needed*. In particular, *studies to assess the hydraulic connectivity and water fluxes through regulated riverbeds that may be subject to increased sedimentation* would help to determine whether the anoxic barrier theory described in Section 6.2 has any potential to improve pollutant attenuation, or whether these interfaces are invariably sealed and flow between groundwater and rivers has been prevented.

Revised geological mapping by the British Geological Society and reclassification of geological strata (particularly the chalk) may provide new insights into the geological controls on heterogeneous discharges of groundwater into surface waters. A study should be undertaken to *assess correlations between the location and rate of known baseflow discharges and revised geological (particularly chalk) stratigraphy*.

In both regulated and natural rivers, sedimentation can lead to blinding of the riverbed. The formation of colmation is currently little studied and essentially ignored in the decision-making process relating to river (flood risk) management. *Field and laboratory studies of the processes* *and significance of colmatage on a range of rivers in different geological settings should be a priority*, since this is likely to lead to increased flow heterogeneity and to changes to attenuation potential within hyporheic sediments by affecting flow velocity and hyporheic travel time.

Catchment-scale assessment of groundwater–surface water interactions adopt a simple coefficient to describe the linkage and hydraulic relationship between the two systems – the baseflow index (BFI). *Studies in middle and higher order streams and rivers to determine whether hyporheic flow is homogeneous (very unlikely) or is controlled by heterogeneity in geomorphological, sedimentary, topographic or other factors would help in understanding the influence of sedimentary structure on flow across the hyporheic zone*. This could lead to improved land-use management practices at the locations most likely to provide river baseflow.

Finally, studies that attempt to relate flow to ecosystem health of the hyporheos, benthos and channel fauna would help to better define the required flow conditions (both upper and lower limits) that significantly affect the habitat suitability and ecosystem health of river systems. This is likely to be a more significant issue in low hyporheic flow environments that have been blinded by sedimentation and colmation. *Regardless of exposure to pollutants, does reduced hyporheic flux cause changes in hyporheic, benthic and channel biodiversity, function or processes, and what are the likely effects on overall ecosystem health?*

6.4 Chemical cycling and pollutant attenuation

6.4.1 Diffuse agricultural pollutants

There is a considerable literature on the fate of agricultural pollutants at the interface of aquifers and rivers. However, the great majority deals with nitrogen (and less so phosphorus) in riparian zones.

Nitrate and phosphate are frequent causes for the failure of WFD objectives and it is clear that these will be a major focus for member states in developing and carrying out programmes of measures to address chemical status failures. Regulation of agriculture will inevitably be prioritised and targeted at a regional scale – it is impractical to attempt to regulate at the field scale, although good-practice land management will clearly be adopted at that level. With respect to agricultural pollutant, the Environment Agency needs to understand how aquifer-scale pollution affects the rivers and wetlands in hydraulic continuity. At this scale of management it does not matter whether water enters a river in localised patches, but rather what the rate of mass transfer between the two systems is at a reach scale. *Understanding the significance of nitrogen and phosphorus attenuation processes that occur in hyporheic sediments* will help develop water-quality goals for groundwater (at a groundwater-body scale) that will protect ecological health in the dependant surface water body.

With respect to phosphorus, *elucidation of the geochemical controls on vivianite formation and its potential to act as a hyporheic phosphorus sink requires further study*.

Ultimately, changes to agricultural land use and its management may be required. *Development of process-based conceptual models that can be extended to design artificial improvement of attenuation processes may be desirable*. The conceptual models will necessarily be developed at small scale, but to be useful to land managers they will need to be up-scaled to allow reach or water-body assessment and design of remedial measures.

Conceptual models that include dissolved, colloidal and suspended solid transport through porous and fractured hyporheic zones will need to be incorporated, so that transport processes and nutrient flux can be more easily related to the potential for transport in different geological terrains.

6.4.2 Industrial pollutants

Very little work has been published on the fate of industrial pollutants in hyporheic zones. Although industrial pollutants are the principal group of substances considered in site-specific and urban environment risk assessments, few studies have concentrated on the attenuation processes at the groundwater–surface water interface.

The main industrial pollutants widely observed include the chlorinated solvents, petroleum hydrocarbons, phenols and PAHs. In addition, novel pollutants such as persistent organic pollutants (polychlorinated biphenyls (PCBs) and brominated organics, pharmaceutical compounds and antimicrobial compounds, such as triclosan) are become of increasing concern.

To date a small number of workers have investigated chlorinated solvent and BTEX attenuation processes in the hyporheic zone. There are no reported studies on the attenuation of the novel pollutants listed, although studies are beginning on their attenuation in river channel and aquifer environments.

Since there is so little literature, *efforts are most likely to be of benefit in terms of the most commonly identified organic pollutants (BTEX, chlorinated solvents and phenol), and it is recommended that the research is developed by the application of novel monitoring and investigation techniques to hyporheic studies*. The application of diffusive gel techniques to determine chemical and redox conditions and gradients across the hyporheic zone would provide a novel approach to help elucidate BTEX or solvent attenuation potential in hyporheic zones.

Studies on novel pollutants might initially be at a larger catchment scale to determine the extent of the problem, before concentrating on the local potential for attenuation in suitable environments.

6.4.3 Mine waters

Mine waters are likely to be one of the most frequent causes of water body status failure in the UK, and the number of affected rivers and aquifers is likely to increase in the foreseeable future as watertables in recently abandoned mines continue to rise. Research should focus on two principal areas:

- *The potential for mine water (metals and low pH) derived from mine-waste spoil tips to be attenuated within the hyporheic zone of rivers as pollutants migrate from groundwater into the river via its bed*. The Environment Agency needs to understand how much attenuation capacity can be expected in different environments, and how this affects the requirements for source (spoil heap) treatment;
- Discharge of mine water is often via adits or drains directly into the river, and does not pass through the hyporheic zone. Once in the channel, water and pollutant exchange with hyporheic waters have been shown to remove significant quantities of metals over relatively short stretches of river. The Environment Agency *needs process-based conceptual models of the processes that lead to pollutant attenuation by stream–hyporheic zone exchange* so that this can be factored into catchment management considerations. Equally importantly, processes that lead to remobilisation of retarded contaminants need to be considered – river managers need to be aware of the processes and river management practices that could result in release of retarded metals and inadvertently cause severe river pollution.

6.5 Ecology and microbiology

The priority issues identified in relation to biological studies of the hyporheic zone are listed under the relevant sub-headings:

6.5.1 Microbial issues

6.5.1.1 Bacteria

To date, nitrification is the main process that has been investigated in hyporheic zones, principally as an extension to earlier work in rivers and riparian zones. These studies have demonstrated that it is an important process for converting ammonium into nitrate in aerobic hyporheic systems (Triska *et al.* 1993b). By contrast, up-welling oxygen-deficient groundwater is more commonly associated with denitrification in riparian and hyporheic zones.

Little work has been undertaken to determine the influence on dynamic conditions of coupled nitrification–denitrification reactions in hyporheic zones where geochemical conditions vary, such as in regulated rivers that are subject to managed flow (and flooding) regimes. Similarly, attempts to quantify the links between hydrological and geochemical controls and bacterial productivity are needed.

A number of authors, including Storey *et al.* (1999), recommend work to develop improved methods to determine *in-situ* denitrification rates within stream sediments, and field experiments to compare chemolithotrophic and heterotrophic processes within hyporheic systems.

Although most previous studies considered nitrate, *investigation of other electron acceptors, including sulphate, iron and manganese, is recommended. In particular, bacterial sulphate reduction* is likely to be an important process to consider because of the excess of sulphate relative to oxygen and nitrate in most natural systems.

Further investigation of bacterial community density and processes, and their relation to geomorphological, hydrological and sedimentary environments, would help to establish whether microbial processes are ubiquitous in hyporheic sediments, or whether heterogeneity of bacterial respiration is significant and related to sediment architecture or water flow regime.

6.5.1.2 Fungi

The role of fungi in the hyporheic zone has received little attention to date. However, the relatively high concentrations of coarse POM in hyporheic sediments suggests that fungally mediated processes may be important to the overall ecosystem functioning and respiration rate. Currently, there is very little information on basic attributes, such as fungal abundance, biomass production and species composition in hyporheic sediments. In the context of developing better models of groundwater–surface water interactions for catchment management purposes, the focus should be on *establishing the role and relative importance of fungi on carbon cycling within hyporheic systems*, and its variability with hydrological and geochemical conditions.

6.5.1.3 Protozoa

Protozoa have been little studied in hyporheic systems and present numerous research opportunities. However, with regard to larger-scale catchment processes they are unlikely to be a significant control on ecosystem functioning. It is recommended that further research should initially concentrate on the *role of protozoa in the microbial food web and its influence on wider ecosystem health* (see Section 6.5.4).

6.5.2 Invertebrate issues

Most studies of invertebrate fauna have been made in headwaters of forested streams, or in arctic, alpines or desert environments. There are few studies in temperate lowland rivers of hyporheic meio- and macro-fauna, for which the great proportion of effort has concentrated on benthic ecosystems. Opportunities exist for research in the following areas:

- *Comparison and/or translation of knowledge from headwater and extreme climate studies to lowland UK rivers;*
- *Studies of the invertebrate hyporheic fauna of UK rivers to establish the basic hyporheic ecology;*
- *Studies to investigate the functional relationships between hyporheic and benthic ecosystems, and the food-web linkages at the base of lotic ecosystems;*
- *Development of methods to sample and identify hyporheic fauna to allow this part of the lotic ecosystem to be considered in river-quality surveys;*
- *Studies of the effect of invertebrate burrowing and bioturbation on hyporheic zone permeability, water and nutrient fluxes and chemical (e.g., oxygen) concentration distributions.*

6.5.3 Vertebrate issues

Studies on the relationship and controls of hyporheic conditions to fish spawning have been reported and are considered adequate for the Environment Agency's current needs. However, further work is needed on the interrelationships between hyporheic microbes and invertebrate fauna and higher animals through the river system food web. Current management approaches to river ecosystems frequently focus on providing habitat for indicator species, such as otters, or stocking rivers with fish at artificial levels to meet angling needs. The food-web requirements of higher vertebrates are poorly understood. An assumption is commonly made that the successful reintroduction of, or support for, indicator species by the provision of cleaner river water and suitable artificial habitat implies healthy microbial and invertebrate communities. *Further studies into the inter-relationships between hyporheic and benthic fauna and the higher animals that are commonly selected as marker species of good ecosystem health would provide a more coherent basis for understanding the effects of river management on ecosystem health*.

6.5.4 Ecosystem functioning and health

Although beyond the scope of this programme on hyporheic zone attenuation and processes, it is clear that there remains a *significant gap between knowledge of chemical concentrations in an environment and their impact on ecological diversity and health*. Hyporheic ecology and food webs are poorly understood, so basic research is needed to establish the community structure and functional relationships to allow the subsequent investigation of the effects of environmental stress on those systems.

6.6 Research priorities and suggested funding mechanisms

Priorities for further research are presented in *Table 6.1*.

Table 6.1 Identified research priorities on groundwater–surface water (GW-SW) interactions and attenuation in the hyporheic zone (HZ).

No.	Title and/or objective	
Review and develop initial conceptual models of lowland hyporheic zones		
1	Produce a catalogue and/or compendium of UK hyporheic zones from existing research and river habitat surveys	
2	Review existing understanding of GW-SW interactions in chalk streams in light of revised chalk stratigraphy	
Understanding hyporheic flow systems		
3	Development of methods for rapid 3D *in-situ* monitoring of GW flow	
4	What are the main HZ controls on water exchange and how does exchange of water between rivers and aquifers vary in lowland rivers (*see* headwater studies) temporally and spatially?	
5	Colmation – what are the controlling processes on colmatage formation, and what significance does it have on HZ residence time and flow heterogeneity in regulated rivers?	
6	Do groundwater and surface actually mix in the HZ, or do they just share porosity at different times?	
Hyporheic zone geochemistry		
7	Investigate the relationship between sediment mineralogy and/or geochemistry and depositional environment in a 'pristine' or moderately regulated river. Does attenuation capacity (and hydraulic conductivity) vary significantly with sediment deposition and architecture?	
8	What is the nature of the (immature) organic carbon deposited in hyporheic sediments, and how does its chemistry affect sorption?	
9	Anoxic hyporheic barriers – is there a HZ in regulated lowland rivers, or is there a blinding layer of anoxic, carbon- and clay-rich sediments that can sorb pollutants and improve anaerobic biodegradation processes?	
Natural attenuation and pollutant cycling in hyporheic sediments		
10	What is the potential for attenuation of agricultural nutrients (N and P) in HZs? How do existing conceptual models of N attenuation in the riparian zone extend to hyporheic zones in sandstone and chalk systems?	
11	What are the controls on phosphorus cycling in the hyporheic zone, and how widespread and important is the formation of vivianite as a HZ sink for P?	
12	What is the potential for attenuation of industrial pollutants (e.g., R-Cl or BTEX) in the HZs of urban and other rivers? At what scale can we make predictive assessments reliably?	
13	What is the potential for attenuation of heavy metals from mine water and/or spoil heaps in HZs?	
14	Is the hyporheic zone a sink for certain emerging or novel pollutants, and what effect does channel–sediment exchange have on the fate of novel pollutants, such as triclosan, discharged from sewage works?	
Hyporheic zone ecology		
15	Review and contrast the existing research on headwater hyporheos with the state of the ecology and function of hyporheic assemblages found in lowland rivers.	
16	What is the effect of bioturbation on hydraulic conductivity and water and/or electron acceptor flux in hyporheic sediment?	
17	What are the relationships between HZ flow and/or hydrology and microbial processes and biodiversity in the HZ?	

18	Are the relationships between HZ sedimentary structure, grain-size distribution, etc., and microbial processes and biodiversity in the HZ important?	
19	How significant are fungi on HZ respiration and do high concentrations of POC imported into the HZ increase fungal biomass?	
20	How does the hyporheos respond to pollution pressures, and do biological quality assessment methods that focus solely on benthic fauna adequately describe the health of lotic ecosystems?	
21	How significant are HZ relationships in the lotic food web and what is the role of HZ microbial and meiofauna on river diversity and health?	
Tools and models, including refined conceptual models		
22	Develop a series of generic conceptual models of hyporheic zone geometry, behaviour and attenuation capacity in common UK geological terrains: WFD typologies.	
23	Develop numerical and/or analytical solutions for HZ flow and attenuation.	
24	Up-scale to reach-scale HZ understanding within catchment models.	
25	Develop water body-scale estimates of nutrient attenuation at the GW-SW interface, based on process-based conceptual models, for inclusion in refined Nitrate Vulnerable Zones/Programmes of Measures that incorporate reactive transport processes.	

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Glossary of terms

Aerobic	an environment that contains oxygen; biodegradation or other process that operates in the presence of oxygen
Allotrophy	
Anaerobic	an environment that contains no oxygen; biodegradation or other process that operates in the absence of oxygen
Anoxic	deficient in oxygen
Autotrophy	organisms that use inorganic carbon (e.g., CO_2) as carbon source
Benthos	organisms that live primarily on (not in) stream sediments
Colmatage/colmation	(Formation of) low permeability deposits due to filtration of fine grained material within river-bed sediments
Chemolithotroph	bacteria that utilise reduced inorganic compounds (e.g., Fe^{2+}, H_2) as their energy source
Ecotone	ecosystem at the interface of environmental compartments
Epigean	surface or very shallow sediment environment; inhabited by epigeic organisms
Groundwater body	a distinct volume of groundwater within an aquifer
Heterotroph	bacteria that uses organic compounds as a source of energy and carbon
Hyporheobiont	organisms that live in the hyporheic zone
Hyporheos	assemblage or community of organisms that live in the hyporheic zone
Interstitial	organisms that live among (between) sand grains
Lithotroph	organisms that use inorganic carbon (e.g., CO_2) as carbon source and as an external energy source
Lotic	living or moving in water
Meiofauna	fauna (mostly invertebrates) between 100 and 1000 μm in length
Mineralisation	biodegradation that leads to the transformation of contaminants into inorganic end-products, such as carbon dioxide, water, methane, chloride ions, etc.
Oligotrophic	Nutrient-poor environment
Oxic	In the presence of oxygen
Photoautotroph	bacteria that utilise light as their energy source
Phreatobites	Organisms that are adapted to live in aquifers
Programme of Measures	collection of regulatory, management and fiscal methods to improve environmental quality under the Water Framework Directive
Riffle	deposit of gravel and other coarse sediments within a river; often associated with a deeper pool
Riparian zone	Flood plain and associated areas adjacent to a river
Species richness	see taxonomic richness
Stygobites	organisms that live interstitially and whose bodies are modified for a

	sub-surface lifestyle
Stygophiles	organisms that can occur on or in sediment, but whose bodies are not modified for an interstitial lifestyle
Stygoxenes	organisms that accidentally occur interstitially
Taxonomic richness	a measure of the diversity of taxa or species within an environment