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Comparison of the groundwater fauna of two contrasting reaches of the Upper Rhône River

Vergleich der Grundwasserfauna zweier unterschiedlicher Abschnitte der alpinen Rhone

Niphargus (Photo: T. Gonser)

Diplomarbeit

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SUMMARY

In the present study, the fauna of the ground waters associated with two contrasting reaches of the Rhône River (Canton Valais, Switzerland) was investigated. The reach between Sion and Martigny is severely channelized, whereas the reach in Bois de Finges (between Leuk and Sierre) is in a geomorphologically more natural state. The main goal of this study was to find out whether there are differences in the groundwater metazoan populations of the two reaches and whether the ecological integrity of surface geomorphology influences faunal distribution. This knowledge could be put to practical use in the monitoring of river rehabilitation by channel widenings (taking out stream bank enforcments and widening the active channel), which are a common practice to rehabilitate rivers in Switzerland. To date, the reactions of the groundwater fauna to rehabilitation measures have been almost entirely neglected, even though the surface water - groundwater ecotone is recognized to be an integral part of river systems.

In each river reach, 36 groundwater monitoring wells were sampled. The level of the groundwater table, water temperature, pH and electrical conductivity were measured in the field. All organisms larger than 90 µm were collected, determined (on average down to the order) and counted. The overall number of taxa and the overall number of individuals as well as the number of taxa and individuals of different size classes (between 90 µm and 1 mm in size, termed 'small' resp. < 1 mm in size, termed 'large') and functional groups (stygophile, stygobite) were counted. Shannon index and Evenness were calculated. Additionally, water samples were taken and ion concentrations (calcium, chloride, magnesium, potassium, sodium, sulfate and nitrate) as well as organic and inorganic particle contents were determined. The ratios of organic particle content to total fine particle content (POC/TFP) were calculated.

Two- and one-sided Wilcoxon rank sum two-paired tests revealed significant differences between the channelized reach and Bois de Finges in the case of stygophile taxa (particularly *Leuctra major*), 'large' and 'small' taxa (particularly Cyclopoida). In contrast, taxa richness, Shannon index and Evenness did not differ between the two reaches. The variabilty of physio-chemical environmental conditions and of POC/TFP was significantly higher in Bois de Finges. The environmental variables measured did not influence the faunal distribution to an important extent. The following variables that were not tested could serve as possible explanations for the differential occurrence of groundwater organisms in the two reaches: clogging of the riverbed and banks (in the channelized reach) and habitat heterogeneity. The process of clogging ('colmation') reduces interstitial pore space in the superficial layer and disrupts migration pathways, hindering epigean species from reaching the ground water in the channelized reach. The higher habitat heterogeneity in Bois de Finges may promote a higher biodiversity and enables a mixed fauna of stygophile and stygobite as well as organisms of various size classes.

The present study indicates that hydrological exchange and biological connectivity between the river channel and its groundwater aquifer have been impaired in the channelized reach of the Rhône River. In this reach, several river channel widenings are planned for the next 30 years. By using Bois de Finges as a *Leitbild*, the amphibite stonefly *Leuctra major* and other epigean species can be regarded as key indicator species for the monitoring of the revitalization since their presence in the ground water demonstrates intact migration pathways. It can be concluded that the 'vertical dimension of rivers' can bring valuable contributions for the assessment of river ecosystem functions and processes.

KEYWORDS: Switzerland, Rhône River, surface water - groundwater ecotone, interstitial fauna, biodiversity, "Third Correction of the Rhône River", revitalization, channel widening, river geomorphology, river ecosystem monitoring, colmation

ZUSAMMENFASSUNG

In der vorliegen Arbeit wurde die Fauna der Grundwasseraquifere zweier unterschiedlicher Abschnitte der Rhone (Kanton Wallis) untersucht. Der Abschnitt zwischen Sion und Martigny ist kanalisiert, während der Abschnitt im Pfynwald (zwischen Leuk und Sierre) eine naturnahe Geomorphologie aufweist. Die vorliegende Arbeit wurde durchgeführt, um herauszufinden, ob es Unterschiede in den Populationen der mehrzelligen Grundwassertiere der zwei Flussabschnitte gibt, und ob die Natürlichkeit der Geomorphologie des Flusses (d. h. an der Oberfläche) die Verteilung der Organismen beeinflusst. Dieses Wissen kann für das Monitoring von Revitalisierungen mittels Flussaufweitungen angewendet werden. Bis heute sind die Auswirkungen von Revitalisierungen auf die Grundwasserfauna nicht bekannt, obwohl das Oberflächenwasser-Grundwasser-Ökoton anerkanntermassen ein wichtiger Bestandteil von Flussökosystemen ist.

In jedem der beiden Flussabschnitte wurden 36 fest installierte Grundwasserrohre beprobt. Grundwasserstand, Wassertemperatur, pH und elektrische Leitfähigkeit wurden im Feld gemessen. Alle Organismen, die grösser als 90 µm waren, wurden gesammelt, bestimmt (im Durchschnitt bis auf das taxonomische Niveau der Ordnung) und gezählt. Die Gesamtzahl der Taxa und Individuen sowie die Anzahl Taxa und Individuen verschiedener Grössenklassen (zwischen 90 µm und 1 mm, als 'klein' bezeichnet und > 1 mm, als 'gross' bezeichnet) und funktioneller Gruppen (stygophile, stygobite) wurden gezählt. Ausserdem wurden der Shannon-Index und die Evenness berechnet. Zusätzlich wurden Wasserproben genommen und die Ionenkonzentrationen von Calzium, Chlorid, Magnesium, Kalium, Natrium, Sulfat und Nitrat sowie die organischen und anorganischen Partikelgehalte bestimmt. Die Verhältnisse der organischen Partikelgehalte zum gesamtem Feinpartikelgehalt (POC/TFP) wurden berechnet.

Einseitige und zweiseitige Rangsummen-Tests nach Wilcoxon für zwei ungepaarte Stichproben zeigten signifikante Unterschiede zwischen den zwei Flussabschnitten bei den stygophilen Taxa (v.a. *Leuctra major*) und bei den 'grossen' bzw. 'kleinen' Taxa (v.a. Cyclopoida). Hingegen war bei der Gesamtartenzahl, beim Shannon-Index und bei der Evenness kein Unterschied zwischen den beiden Abschnitten feststellbar. Die Variabilität der physikalisch-chemischen Umweltbedingungen und des POC/TFP war im Pfynwald signifikant höher. Die gemessenen Umweltparameter beeinflussten die Verteilung der

Organismen aber nicht wesentlich. Die folgenden, nicht getesteten Parameter könnten als mögliche Erklärungen für das unterschiedliche Vorkommen der Grundwassertiere in den zwei Flussabschnitten dienen: Kolmation der Flusssohle und der Ufer (im kanalisierten Abschnitt) und Habitatsheterogenität. Da Kolmation den Porenraum in der obersten Schicht des Interstitials reduziert und Migrationswege unterbricht, können epigäische Organismen im kanalisierten Abschnitt nicht zum Grundwasser gelangen. Die höhere Habitatsheterogeneität im Pfynwald fördert eine höhere Biodiversität und ermöglicht folglich eine Mischfauna von stygophilen und stygobiten Tieren und auch von Organismen verschiedener Grössenklassen.

Die vorliegende Arbeit deutet darauf hin, dass der hydrologische Austausch und die biologische Verbundenheit zwischen dem Fluss und seinem Aquifer im kanalisierten Abschnitt der Rhone beeinträchtigt sind. Im Rahmen der Dritten Rhonekorrektion sind im kanalisierten Bereich der Rhone für die nächsten 30 Jahre verschiedene Flussaufweitungen geplant. Wenn der Pfynwald als Leitbild verwendet wird, können die amphibite Steinfliege *Leuctra major* und andere epigäische Invertebraten als Schlüsselarten angesehen werden, die für das Monitoring der Revitalisierung verwendet werden können. Diese Arten zeigen eine hydrologische Verbundenheit zwischen dem Fluss und seinem Aquifer an, da ihr Vorkommen im Grundwasser intakte Migrationswege voraussetzt. Diese Schlüsse zeigen, dass die 'vertikale Dimension von Fliessgewässern' wertvolle Beiträge für die Beurteilung von Flussökosystemfunktionen und -prozessen leisten kann.

INTRODUCTION

Most temperate rivers have been heavily regulated during the past few centuries (Buijse *et al.* 2002). Channelization, gravel mining and the building of levees and dams, have altered the face of our rivers and have lead to incised channels, lowered groundwater tables and modified discharge regimes. In particular the alluvial reaches of rivers have been heavily regulated to protect against flooding and to acquire land for housing and agricultural needs. As a consequence, riverine flood plains are among the most endangered landscapes worldwide (Tockner *et al.* 2002). In Canton Zürich, Switzerland, for example, about 67% (2422 kilometers) of the total river length are regulated. Specifically, 27% are placed in submerged culverts, 6% are classified as non-natural reaches, 14% are heavily impaired and the remaining 20% are impaired (Niederhauser 1999).

In recent years, it has been recognized that the need for river restoration is a major challenge for both the current as well as for future generations (Allan & Flecker 1993; Jungwirth *et al.* 2002). River restoration has been defined as the return of the ecosystem to pre-disturbance functions and conditions (Cairns 1991). However, this level of restoration is an ideal that is rarely attainable, much less practiced. In the German *Sprachraum*, 'Renaturierung' ('re-naturalization' resp. rehabilitation), 'Revitalisierung' ('re-vitalization') and 'Aufwertung' ('raising of the ecological value' resp. enhancement) are differentiated (Lachat *et al.* 2001). Renaturalization is the general term that includes both revitalization and 'Aufwertung' and designates the return of an ecosystem to a state that is close to natural ('naturnahe Kulturlandschaft'). Revitalization includes measures that aim to re-establish the dynamic processes of water and bedload balances, whereas 'Aufwertung' involves measures to create alternative habitats that enhance structural and biological diversity. In this paper, the term 'revitalization' will be used since it fits best to the specific case of the Upper Rhône River.

River revitalization offers various advantages to both the environment and the local human population. Not only is river revitalization beneficial for security reasons (protection against downstream flooding) and for securing drinking water supplies (higher surface/ground water interaction), but it also provides recreational opportunities for the residents (Gunkel 1996; Williams *et al.* 1997). In addition, river revitalization schemes can be utilized by scientists for applying and testing ecological concepts (Bayley

1995; Schiemer 1999). A major goal of river revitalization is the preservation and/or enhancement of biodiversity by providing suitable habitats for a wide variety of different species, many of which are endangered (Lachat *et al.* 2001).
There are several distinct ways a river can be revitalized. In Switzerland, one of the more common practices is removing stream bank enforcments and widening the active channel (Peter & Schulz 2002). This practice is called 'channel widening' or 'river enlargment' ('Aufweitung'). In Switzerland, at least 12'500 km (20%) of the total river length require revitalization measures (A. Peter, personal communication). However, since 1990, only about 12 km (< 0.02%) of the total river length have been revitalized using channel widening, and even much less using other methods.

In order to define the ecological deficits of a river system and to assign appropriate revitalization measures, it is indispensable to know the key features, structures, functions and processes of fluvial ecosystems (Ward *et al.* 2001). Only if it is understood how river ecosystems work, a useful revitalization concept can be designed.
One of the most important concepts in river ecology is the four dimensional nature of lotic ecosystems (Ward 1989). This concept stresses the fact that a river system does not only consist of the obvious longitudinal dimension (upstream-downstream connection), but also of the lateral dimension (connection to riparian/floodplain systems) and the vertical dimension (connection to the ground water). Since the extent of these spatial dimensions varies over time, the temporal dimension was introduced as the fourth dimension. The exchange of water, resources and organisms between the three spatial dimensions has been termed 'connectivity' (Amoros & Roux 1988). It is regarded as a crucial factor for the ecological integrity of alluvial floodplain rivers. A 'connected' riparian-groundwater system provides protection from pollutants, clogging of interstices, reductions in habitat heterogeneity and reductions in successional stage diversity, thereby maintaining suitable habitat conditions for diverse biotic assemblages in surface water, riparian and groundwater environments (Ward *et al.* 1998). The transition zones between the channel and the flood plain as well as between the channel and the groundwater aquifer are the basic riverine ecotones (for more details cf. Ward & Wiens 2001). Ecotones have often been observed to harbor an especially rich biodiversity, due to the overlap of two adjacent habitats (edge effect) as well as to the occurrence of communities that are specialized for that ecotone (Naiman *et al.* 1988; Nielsen *et al.* 1992). This applies to aquatic-terrestrial ecotones (e.g. the riparian zone) as well as to surface water-groundwater ecotones (e.g. the

hyporheic zone; ground water associated with riverine aquifers). For discussions concerning the biodiversity of the hyporheic zone, see Marmonier *et al.* 1993, Sket 1999a, Sket 1999b and Danielopol *et al.* 2000.
The role of the hyporheic zone and specifically of the exchanges between surface waters and ground waters for the ecological integrity of the fluvial ecosystem as a whole has been stressed by many authors (e.g. Brunke & Gonser 1997; Boulton *et al.* 1998; Jones 2000). However, connectivity between the surface and the subsurface system is often impaired by anthropogenic activities. Clogging of the interstices of the top layer of the channel sediments, a process also termed 'colmation' (Schälchli 1992; Brunke 1999; Gayraud *et al.* 2002), can be caused by the accumulation of excessive fine sediment originating for example from agricultural activities. Rivers fed by tributaries with upstream reservoir lakes are vulnerable to colmation as well, because the coarse bedload (i.e. gravel, pebbles and boulders) is held back in the reservoir (Sear 1993; Shields *et al.* 2000; Ward & Wiens 2001). This leads to a shift in the bedload composition of the tributaries toward a higher proportion of fine sediments. Additionally, the frequency of flushing floods is lowered. Channel straightening and river regulation worsen the situation in colmated rivers since they cause the possible contact zones between surface and subsurface waters to be drastically reduced (Pringle & Triska 2000). The consequences of colmation are a reduced hydraulic conductivity within the bed, a reduced interstitial space and a decrease of exchange rates across the surface water-groundwater ecotone (Ward & Wiens 2001).
Ecologically sound revitalization concepts should therefore aim to reestablish the connectivity within all three spatial dimensions of the river (Ward & Stanford 1995; Ward *et al.* 1999).

A key component of river revitalization projects is the design of an elaborate monitoring concept. Pre- and post-monitoring is needed in order to assess the success (or failure) of the applied revitalization strategies (Brookes & Shields 1996). Only if the revitalization project is accompanied by an adequate monitoring programme, it will be possible to learn from that project for future revitalizations.
One of the first steps of developing a revitalization project, which is also important for establishing its monitoring concept, is the definition of a *Leitbild*, i.e. an endpoint state (Jungwirth *et al.* 2002). As opposed to the natural reference, which designates the pre-perturbed state, the *Leitbild* describes an ideal near-natural state that includes human

activities in the catchment. The *Leitbild* should be utilized to guide the definition of the specific and realistic target state of the revitalization.

A crucial step in designing a verifiable monitoring concept is defining suitable and quantifiable measures of success over adequate time scales, which is not always easy. Often, the success of revitalization projects is focused on so-called key or target species, i.e. species that have specialized dietary and/or habitat needs (Schiemer 1994). A frequently applied monitoring measure to document the success of a revitalization project from an ecological point of view is biodiversity (Schiemer 1994; Lachat *et al.* 2001). There are different parameters that can be used to measure alpha-diversity, e.g. species richness or numerical indices (e.g. Shannon index). These parameters are often combined with the elicitation of abundance measures, e.g. density of individuals or biomass, to determine dominance resp. evenness of the different species. Biodiversity in alluvial floodplain reaches depends to a large degree on habitat heterogeneity (Ward & Stanford 1995). Habitat heterogeneity is generally higher, the more natural the system is. Up to now, monitoring of biodiversity in rivers has been done mainly by investigating the benthic macroinvertebrates (i.e. organisms that live on or within the streambed) and fishes (Schiemer 1994). However, river fishes can migrate from tributaries and macroinvertebrates are subject to drift (accidental or voluntary downstream transportation). The local community composition thus does not always and not necessarily reflect the actual local environmental conditions. Furthermore, these two classes of organisms can in most cases only be used as indicators of the longitudinal and some of them also the lateral connectivity of river systems. Only some specialized benthic macroinvertebrates and very few fishes occur in the ground water as well and could be used as indicators to monitor vertical connectivity (Ward 1989).

To achieve a holistic approach, a monitoring concept for a river revitalization project should take all integral parts of a fluvial ecosystem into account (i.e. all three spatial dimensions of rivers – longitudinal, lateral and vertical - and the fourth dimension, the temporal dimension). Furthermore, when investigating biodiversity, the highly diverse ecotones (riparian and hyporheic) are of special interest. There have been a few studies that monitored the riverbank flora and fauna (lateral dimension; riparian ecotone) in addition to the benthic macroinvertebrates in the channel itself (Buijse *et al.* 2002). However, monitoring of the vertical dimension, the surface water-groundwater ecotone, has been overlooked in most revitalization projects, even though it is recognized to be an integral part of river systems (Ward 1992). One such study has been conducted on the

Rhône River in France (Brégnier-Cordon alluvial plain) that evaluated the efficiency of management works by investigating the interstitial fauna (Claret *et al.* 1999). Long-term changes in the composition, structure, taxonomic richness and diversity of species traits and habitat-affinities of the groundwater communities were studied in two sites that have been dredged resp. artificially hollowed-out. Dredging was conducted to stop terrestrialization and hollowing-out to lower the surrounding groundwater table. However, these local measures did not involve a whole river revitalization project. Unfortunately, the outcomes of many revitalization projects are not published, and it is thus not known if such projects did monitor the vertical dimension of rivers. Nevertheless, it must be concluded that the response of the groundwater fauna to revitalization measures has been almost entirely ignored.

Different terminologies are used to classifiy the groundwater organisms that live in the interstitial habitat (i.e. the water-saturated spaces between sediment particles in porous environments, e.g. alluvial aquifers). Species inhabiting only the subterranean waters are called **hypogean**, whereas interstitial fauna that can be encountered in surface waters as well is designated **epigean** (Danielopol 1976). The fauna can also be subdivided into the following functional groups (Gibert *et al.* 1994): on the one hand, species that are specialized for and restricted to groundwater habitats, are termed **stygobites**. On the other hand, species with clear affinities to subterranean waters belong to the category of **stygophile** organisms. The stygophile organisms can be subdivided further into the occasional hyporheos (i.e. surface water-species that penetrate short distances into groundwater habitats, but can occur in surface waters lacking groundwater habitats) and the amphibites (i.e. migrating species that require both surface waters and ground waters to complete their life cycle). A third classification differentiates between the evolutionary origin of the organisms (Ward *et al.* 1998). **Primary aquatic** species are either of ancient freshwater lineages or invaded inland waters from the sea. Primary aquatic species lack a terrestrial phase and part of them inhabit subsurface waters throughout their lives. **Secondary aquatic** species reside mainly in surface waters, penetrating only the superficial interstices of the stream-bed. Secondary aquatic species include aquatic insects, most of which have a terrestrial adult phase, water mites and pulmonate snails.

Groundwater organisms actively disperse and colonize new habitats depending on their ecological tolerances and preferences (Danielopol 1991). The occurrence of groundwater organisms is said to be related to their body size. That is, small organisms occur in aquifers that consist of small interstitial pores, and larger organisms can only occur in aquifers with larger interstitial pores (Gibert *et al.* 1994). Additionally, the total available pore space plays an important role for the total faunal density. The following physio-chemical parameters are considered to be of particular importance for the spatial distribution of organisms in the interstices: the organic particle content (with the associated microbial biofilm) as the primary food resource for non-predatory organisms, the total fine particle content (TFP) and constant temperatures. The ratio of organic particle content to total fine particle content (POC/TFP) expresses how much the organic matter is 'diluted' by inorganic fine particles and is therefore a measure of organic particle availability (Brunke & Gonser 1999). Furthermore, there are parameters that constrain the occurrence of groundwater organisms, such as oxygen concentration, toxic substances and pH (Strayer 1994). Also, hydrological factors, such as the local and actual situation concerning downwelling or upwelling, have been shown to be important in controlling the distribution of groundwater organisms (e.g. Dole-Olivier & Marmonier 1992). Only little is known concerning biotic interactions (e.g. competition and predation) and food webs in the hyporheic zone (Stanford & Ward 1988; Strayer 1994). For reviews on factors influencing the spatial distribution patterns of interstitial animals, see Gibert *et al.* 1994, Ward & Palmer 1994 and Ward *et al.* 2000.

Several authors have pointed out that groundwater metazoans can be used as indicators for monitoring purposes (e.g. Danielopol 1989; Danielopol 1991; Malard *et al.* 1998). Many studies that investigated the response of groundwater organisms to chemical pollution suggest using these organisms as biomonitors of water quality (Schmidt *et al.* 1991; Plenet & Gibert 1994). Dole-Olivier *et al.* (1993) inferred that hypogean organisms can be used as tools in studies of landscape change and for environmental management. Already as early as 1990, Pospisil and Danielopol suggested to include the ground water and its fauna in the National Park Declaration of the proposed Danube-wetlands National Park and to conduct comparative studies of the groundwater ecosystem of intact and impaired (i.e. isolated) floodplains. Creuzé des Châtelliers *et al.* (1992) stated it would be worth monitoring the possible recovery of the interstitial assemblages of the Rhine River as a long-term assesment of the efficiency of the restoration measures undertaken on the Rhine.

Monitoring a river to accompany a revitalization project means documenting the state **before and** again **after** the revitalization project **in one area**. Recording the state **in two**

different areas (i.e. areas differing in their ecological integrity) at the **same time** can provide a first clarification to determine if the local **groundwater** organisms are sensitive to varying environmental conditions in the **surface water** habitat at all. Few studies have compared the faunal distributions of rivers or reaches with differing river geomorphologies (Maridet *et al.* 1992; Maridet *et al.* 1996; Olsen *et al.* 2001). Additionally, all of these studies were conducted in natural or close-to-natural rivers. None of them explicitly compared a reach that is anthropogenically impacted with a natural reach and made inferences on the influence of the human alteration of the system on the groundwater fauna. Direct and indirect effects of human activity on ecosystem functions of the hyporheic zone, such as river regulation, mining, agriculture, urban, and industrial activities have been described by several authors (reviews in Gibert *et al.* 1991 and Hancock 2002). However, the specific consequences of those human impacts on the fauna are only touched upon, and no comparative studies have been conducted. Other studies have described faunal assemblages of reaches that are affected by chemical and organic pollutants (e.g. Sinton 1984; Plenet & Gibert 1994), gravel extraction (Creuzé des Châtelliers & Marmonier 1993), channelization (e.g. Creuzé des Châtelliers *et al.* 1992; Marmonier *et al.* 2000) or river incision (Bravard *et al.* 1997). However, these studies do not directly compare the adversely affected reaches with natural reaches of the same region and stream order. There are other studies that do so. A few studies evaluated the influence of water quality on the fauna by comparing a polluted with a pristine reach (Schmidt *et al.* 1991; Plenet & Gibert 1994; Malard *et al.* 1996; Malard *et al.* 1998). In one study, four channelized reaches and one natural reach were analyzed, but not explicitly compared under this viewpoint (Marmonier & Creuzé des Châtelliers 1992). Not a single investigation that compared the groundwater fauna of a degraded reach with an already revitalized reach is known to the author.

The present study was conducted to take a first step into the direction of factoring the vertical dimension into monitoring concepts of river revitalization projects. It can be regarded and used as a pre-study for monitoring projects. The first purpose of this study was to determine whether there are differences between the groundwater biota of a river reach with a degraded geomorphology compared to those of a river reach with a close to natural surface geomorphology. The goal of this study was not to investigate whether revitalization measures have the potential of being successful. This would already be the second step and would be done by comparing a degraded with an already revitalized reach of the same river or by comparing the pre-revitalization state in a certain area with its post-revitalization state.
Additionally, the present study attempted to elucidate why differences in the faunal composition of the two reaches do resp. do not occur. It was investigated if the observed faunal differences could be a consequence of the differing river geomorphologies or rather if they depend on other environmental parameters.

The specific hypotheses behind this study were:

1. Taxa composition, taxa richness (measured by the number of taxa and numerical indices of diversity and evenness) and individual abundances (i.e. density) of groundwater invertebrates differ between the degraded river reach and the more natural river reach. Taxa richness and individual abundances are substantially higher in the more natural river reach than in the degraded river reach.
2. The differences in taxa composition, taxa richness and individual abundances are caused by river geomorphology. Other factors that can influence faunal distribution patterns do not explain the differences in taxa composition, taxa richness and abundance.

In addition to investigating in the suitability of groundwater organisms for the monitoring of river revitalizations, the present study provides a documentation of the present state (i.e. the pre-revitalization state) in the reach that is degraded by channelization.
In the reach with the more natural river geomorphology, in contrast, an inquiry of a near-natural reference state is realized. This reference can be utilized as a *Leitbild* and could thus help to conceptualize the target-state of revitalization projects in this area.

MATERIALS AND METHODS

Study Site

The two reaches of the Upper Rhône River (principal river discharging into Lake Geneva) investigated in the present study are both located in the Canton Valais in southern Switzerland (Fig. 1). There, a large number of groundwater wells, which were installed in the 1990s to record the water level (all wells) and the electrical conductivity (selected wells only), is available for pumping.

The Upper Rhône River is an Alpine gravel-bed stream with a substantial unconfined alluvial aquifer. The longterm (1916 – 2001) annual mean discharge of the river recorded at the gauging station at Sion amounts to 112 m^3/s. The annual discharge regime is characterized by snowmelt in spring/early summer. The last bed-moving flood occured in October 2000.

Table 1 (page 19) gives a short overview of the characteristics of the two river reaches and the differences between them.

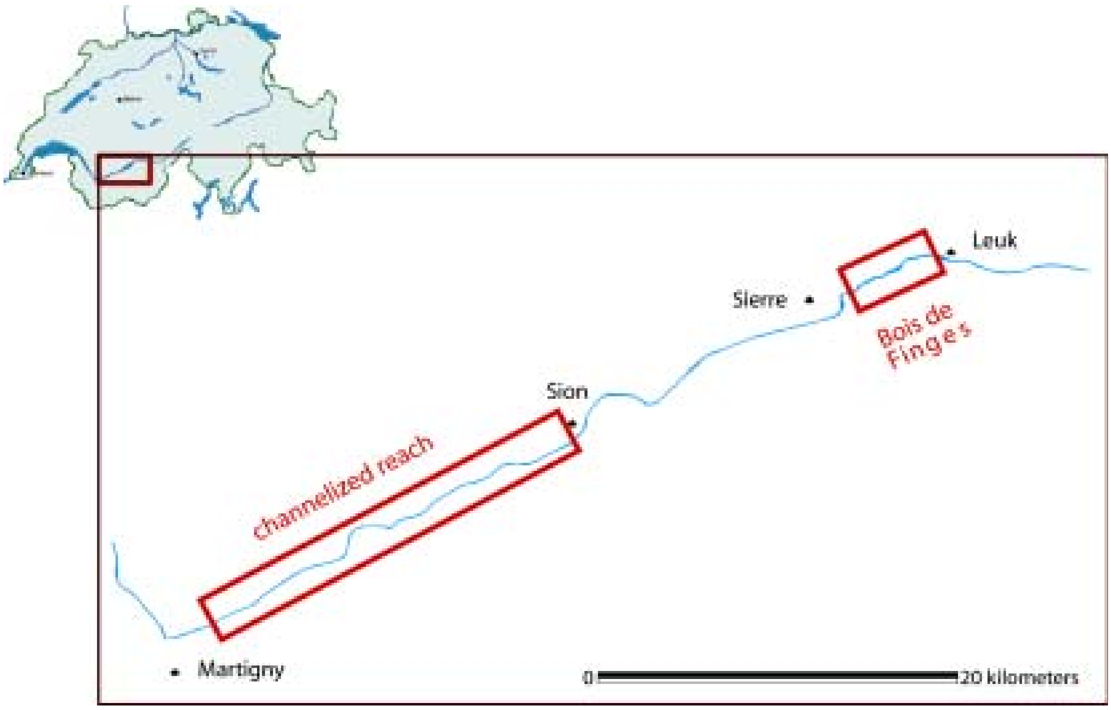

Fig. 1: Location of the study sites along the Upper Rhône River in the Canton Valais, Switzerland (courtesy of Canton Valais)

The geomorphologically more natural river reach is located in the Bois de Finges/Pfynwald ('Pfyn Forest') area, which is situated about 20 km upstream (east) of the town of Sion, the capital of Valais. It encompasses the reach between the town of Susten and the town of Sierre and thus includes an area of about 4 km length and almost 2 km width. Bois de Finges is listed in the Federal Inventory of Alluvial Zones of National Importance ('Auengebiete von nationaler Bedeutung', Object no. 133) and in the Federal Inventory of Landscapes and Natural Monuments of National Importance ('Bundesinventar der Landschaften und Naturdenkmäler', Object no. 1716). It is therefore protected by the Federal Law relating to the Protection of Nature and the National Heritage ('Bundesgesetz über den Natur- und Heimatschutz'). Many endangered species occur in the Bois de Finges area (e.g. Peregrine Falcon (*Falco peregrinus*), Stag Beetle (*Lucanus cervus L.*), Alpine Sea Holly (*Eryngium alpinum L.*)).

In Bois de Finges, the geomorphology of the river is close to natural, apart from some levees (cf. Fig. 2).

Fig. 2: Overview from the terrace of the church of Varen over Bois de Finges (Photo T. Gonser)

However, there are various human impacts on other characteristics of the river, particularly on its discharge regime and bedload input. At the upstream end of Bois de Finges, a run-of-the-river hydropower plant generates energy by diverting most of the river water (up to 62 m^3/s) into a channel that runs parallel to the river and through turbines before leading it back into the river at Sierre. Hence, the Rhône River in Bois de Finges receives only 20 to 40% of the natural discharge. The hydropower plant causes the water level in the low water season (November to April) to be so low, that there is virtually no water left in the river (Schürch 2000). Recharge of the aquifer by infiltration of river water can thus only occur during the high water season (June to August). During the low water season, the Bois de Finges aquifer is recharged exclusively by sulphate-rich subsurface flow and surface runoff from the southern valley side. This causes the establishment of an unsaturated layer beneath the river in the downwelling zone. During the high water season, Rhône River water infiltrates into the aquifer in the northeastern part of Bois de Finges. It contributes about 75% of the groundwater recharge, the remaining 25% originate from the southern hill slope. For more information on the hydrogeology of the Bois de Finges aquifer, please consult Schürch (2000).

There are also several gravel pits that extract large amounts of gravel (about 100'000 m^3/yr) from the riverbed (Jäggi 2001). The Bois de Finges area is composed of various Quaternary geological environments: the Rhône alluvial sediments, the Illgraben fan deposites, the Sierre rockslide deposites and the debris deposites of the southern hill slope (Schürch 2000). A large amount of coarse sediment – up to 300'000 m^3/yr (VAW 1988) – is supplied to Bois de Finges by the Illgraben. For a more detailed description of the geology of the Bois de Finges region, see Burri (1997).

Additional human impacts on Bois de Finges include communities with industrial activities, agriculture, landfills and roads.

The microclimate in Bois de Finges is characterized by cold winters (about –10°C air temperature) and hot summers (about +30°C air temperature). It is one of the driest regions in Switzerland with a longterm (1901 – 1960) annual precipitation mean of only 587 mm/yr (Schürch 2000).

The degraded reach is situated downstream of Bois de Finges. It begins at the downstream end of Sion and ends about 4.5 km upstream of the town of Martigny and is therefore about 20 kilometers long.

The Rhône River has been severely impacted by hydro-engineering in this reach. The river has been placed in a narrow low-flow canal with boulder riprap supporting the banks and artificial levees to contain flood discharges. This study site will therefore be referred to as the 'channelized reach' (cf. Fig. 3).

Fig. 3: Overview from Basse-Nendaz over part of the channelized reach (Photo M. Fette)

Besides the altered riverbed geomorphology, there are also modifications of the discharge regime and the sediment load in this reach. Storage dam hydropower plants located at the tributaries feeding the Rhône River maintain large reservoir lakes that hold back most of the coarse sediment, causing a shift in the bedload composition toward a higher proportion of fines. This type of hydropower plant leads the water through pressure tunnels to generate electricity as needed and thereby causes an altered discharge regime (hydropeaking) and an altered water temperature regime (temperature reduction by cold water from the mountains) in the river (Vivian 1989; Loizeau & Dominik 2000; Poff &

Hart 2002). There are various indications (a calculated discharge balance, temperature profiles of the river and of groundwater wells, data on geochemistry and isotopes), suggesting that the Rhône River in this area is heavily colmated (M. Fette, personal communication). This idea is supported by radon-measurements, showing that the local ground water infiltrated more than 20 days previously (E. Hoehn, personal communication). This means that the water did not infiltrate from the river in the immediate vicinity, but seems to originate either from the hillslopes or from river infiltration further upstream. Also, the contact zone between surface water and ground water has been reduced to a narrow canal in this reach.

Within the framework of the "Third Correction of the Rhône River" ('Dritte Rhonekorrektion') five channel widenings are planned at the Rhône River between Sion and Martigny within the next 30 years (État du Valais 2000). Three of them are located on the left side of the river and cover a total length of 7.3 km and the other two are situated on the right side and encompass 7.15 km. The first channel widening of the Upper Rhône River is projected to be constructed in the year 2004/2005 at Riddes.

Table 1: Geomorphological, hydrological, sedimentological and land use characteristics of the two investigated river reaches. Data on mean bed slope, human impacts on hydrology and reservoir lakes from Landeshydrologie (1992); data on gravel extraction from Jäggi (2001)

Reach characteristics		**Channelized reach**	**Bois de Finges**
morphological	**stream order (Strahler)**	5	4
	channel morphology	constrained (modified)	braided (natural)
	'Ökomorphologie Stufe F' (BUWAL)	non-natural	little impared
	hydroengineering	boulder riprap, levees	levees
	mean bed slope	< 0.5%	1.1 – 12%
hydrological	**human impacts on hydrology**	affected by hydropeaking	effluent discharge reach with 20 to 40% of the natural discharge
sedimento-logical	**water volume of reservoir lakes feeding respective river reach [Mio. m^3]**	731.2	141.6
	catchment area of reservoir lakes feeding respective river reach [km^2]	ca. 387.94	ca. 540.7
	gravel extraction [m^3/yr]	0	100'000
riparian	**anthropogenic land use**	agriculture, industry, transportation	nature reserve (forrest), agriculture, transportation

Sampling Design

Sampling of the sediments in the river channel itself by means of piezometers was not possible because the discharge was too high and it was therefore too dangerous to enter the stream.

All groundwater wells that were known to exist, accessible, no more than a km away from the river and could be pumped, were sampled.

In Bois de Finges, the available groundwater wells are distributed across the entire historical flood plain. In the channelized reach, four transects of groundwater wells were available and sampled: one close to Sion, one close to Chamoson, one close to Riddes and one close to Fully. The transect at Fully consists of wells on both sides of the river, whereas the other transects consist only of wells on one side of the river. Most of the groundwater wells of these transects are within or close to the river embankments. There are also individual groundwater wells close to the river in the channelized reach, three of which could be sampled.

The restriction for pumping was the level of the water table in the well. Since the pumping had to be done with a hand-driven membrane pump in order to protect the organisms from injury, only wells where the groundwater level was not deeper than 7 m below the surface could be sampled. Consequently, 11 wells in Bois de Finges and 22 wells in the channelized reach could not be sampled. Also, another 13 wells in Bois de Finges and 13 wells in the channelized reach could not be sampled because of other reasons (e.g. no water in the well, destruction of the well). Consequently, of all existing wells, 37 wells were sampled in Bois de Finges and 36 wells were sampled in the channelized reach (Fig. 4 and 5 as well as Appendix 1 and 2). For the distribution of the wells (river bank side, hydrological exchange type) and their mean depth and mean distance from the river, see Table 2.

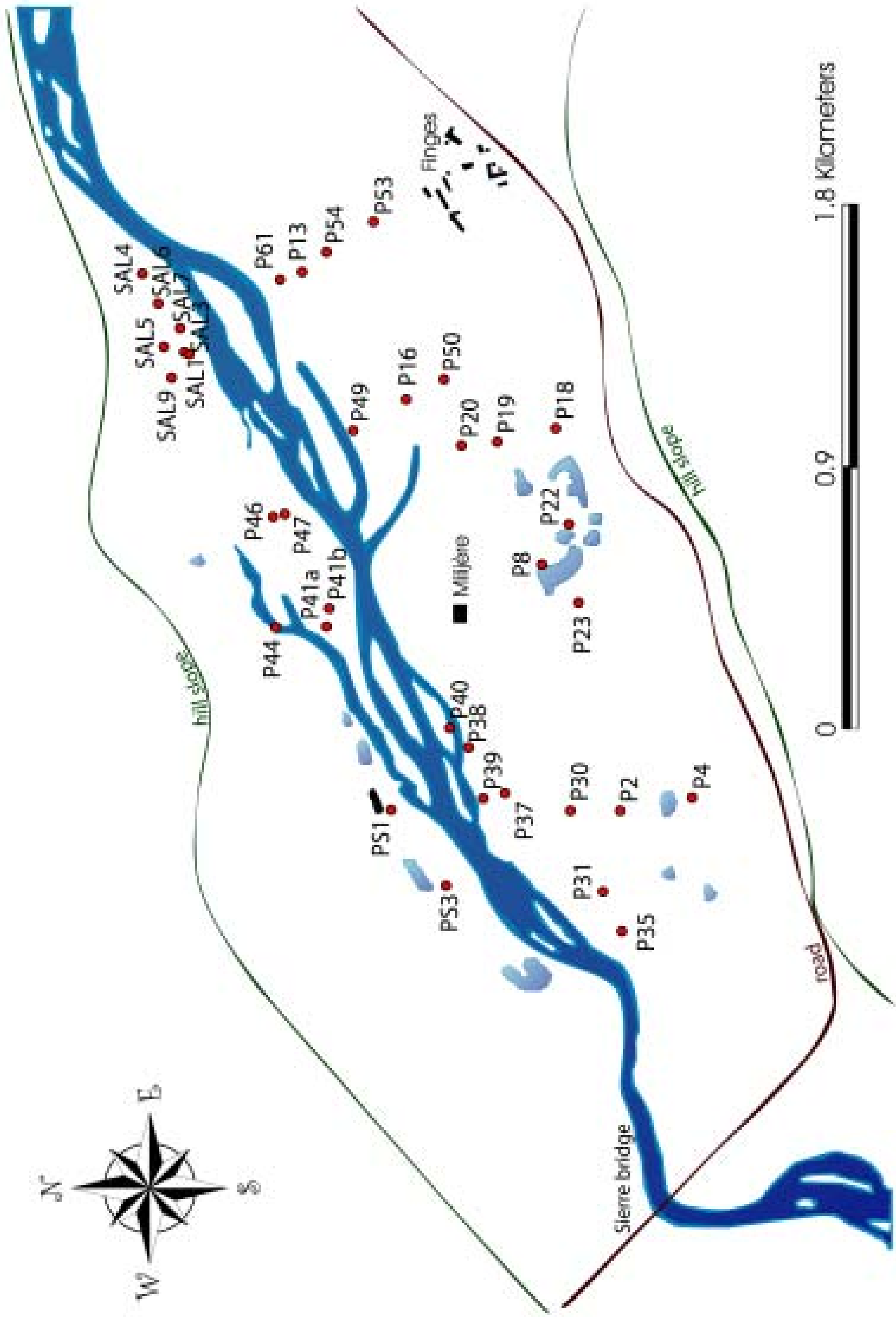

Fig. 4: Map of the sampled wells in the Bois de Finges (courtesy of Canton Valais)

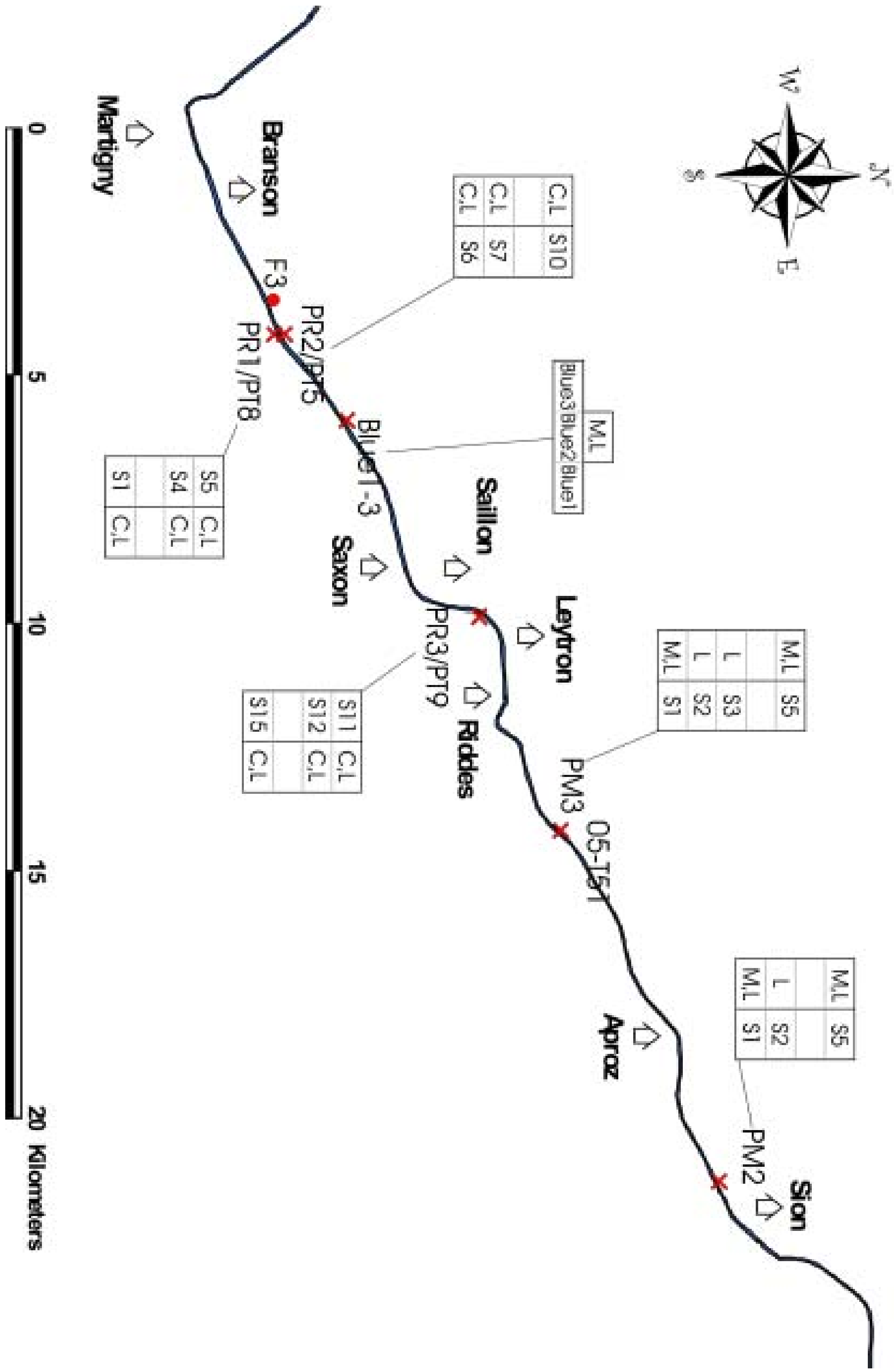

Fig. 5: Map showing the locations of the sampled wells (red dots) and transects (red crosses) in the channelized reach (courtesy of Canton Valais)

Table 2: Distribution of groundwater wells in both reaches and their mean depth and mean distance from the river.

Characteristics of the wells		Channelized reach	Bois de Finges	All wells
total number of wells sampled		36	37	73
river bank side	**left**	50%	62%	56%
	right	50%	38%	44%
hydrological exchange type	**exfiltration zone**	50%	30%	40%
	infiltration zone	33%	38%	35%
	advection zone	17%	32%	25%
depth of the wells [m]	**mean**	8.9	12.07	10.5
	range	4 – 32.2	2.2 – 40	2.2 – 40
distance from the river [m]	**mean**	21.4	292.97	169.2
	range	0 – 58.5	70 - 770	0 - 770

Field Methods

The sampling was done at the beginning of the high water season, on April 18/19 and 25/26, May 9 – 11 and 22 – 24 and June 3 – 5, 2002. Each groundwater well was sampled once.

Fig. 6: Sampling method (Photo T. Gonser)

Before pumping, the depth of the well, the depth of the groundwater table and the water temperature were measured with a depth sensor (A. Ott, Kempten, Germany). The pumping was done with a hand-driven 'Allweiler'-membrane pump and PVC-pressure- and suction-tubes which contain a hard-PVC-spiral (inner diameter 32 mm resp. 25 mm) with 'STORZ'-metal ends (see Fig. 6).

First, the standing water in the well was pumped and discarded. Since groundwater wells can act as traps for groundwater organisms (Bretschko & Leichtfried 1988), the next 2 liters of water with the metazoans that had been collected in the well since the last pumping of the well were filled into a separate 2 l-PE-bottle. This sample was termed 'qualitatitive biota sample'. Subsequently, a defined volume of water (between 15 and 220 l, depending on the content of sand and fine inorganic particles; on average 27 l in the channelized reach and 76 l in Bois de Finges) was pumped and passed through a 90-µm mesh net. The contents retained in the net (organisms and particles larger than 90 µm) were collected in a 500 ml-PE-bottle and termed 'quantitative biota and particle sample'. The water that passed through the net was collected in 12- or. 15-liter plastic buckets, and the electrical conductivity and the pH-value were measured (with a portable WTW-MultiLine P4 F/SET-3; with the specific conductance meter WTW TetraCon® 325-3, standadized at 20°C and the pH meter WTW SenTix 41-3) and turbidity was estimated (Scale: 0 none, 5 highest). Oxygen concentrations were not measured because the oxygen content is altered by the use of the suction pump. Also, a 500 ml-sample was taken from that water in a PE-bottle to determine the ion concentrations and the content of particles smaller than 90 µm in the laboratory. All samples were immediately stored in cooling boxes for transportation.

Laboratory methods

All samples were stored at 4°C in the dark until processing. The majority of the samples was processed within 14 days.

Organisms and Particles > 90 µm

The contents of each of the qualititive and quantitative biota sampling bottles were passed over a 90-µm mesh net and washed thoroughly to remove all suspended matter (i.e. particles < 90 µm). Samples with particle contents of more than 15% of the sample were slurried about 20 times. Then the content of the net was transferred into a counting dish for invertebrates. Under a Leica dissecting-scope (Type CLS 100), all organisms were identified to the lowest possible taxon (i.e. possible without the help of a specialist; bold in Table 4 on page 27), counted, removed from the sample and stored in 70% industrial alcohol. For the qualitative samples, the number of individuals of each taxon was only estimated roughly. Most samples were sorted by students of D-UMNW and controlled by T. Gonser. The only stonefly species that was found was determined by Michel Sartori at the Zoological Museum of Lausanne and found to belong to the species *Leuctra major* Brinck (see Fig. 7 and Table 3).

Fig. 7: Stonefly *Leuctra major* Brinck found in a groundwater well in Bois de Finges (Photo T. Gonser). Lenght: 1.4 cm, excluding antennae and cerci

Table 3: Properties of the groundwater wells in which larvae of the stonefly *Leuctra major* were found and numbers of individuals found (x, y: coordinates (Switzerland), dist. river: distance of the well to the river, 2-liter: No. of *Leuctra major* found in the 2-liter sample, sieved: No. of *Leuctra major* found in the sieved sample, volume: volume of water pumped for the sieved sample)

Properties of the well:						date	2-liter	sieved	volume [l]
ID	x [m]	y [m]	z [m a.s.l.]	depth [m]	dist. river [m]				
P 38	610498	127751	544.76	8.10	135	May 22 2002	20 exuviae	4	220
						June 3 2002		6	50
P 39	610341	127700	544.10	7.80	105	May 23 2002		1	100
PS 1	610297	128008	454.03	8.32	155	June 4 2002		7	100
P 35	609885	127236	539.41	7.92	160	June 5 2002		4	50
P 49	611600	128138	554.96	4.65	145	May 23 2002		1 exuvia	50
P 20	611553	127773	553.34	8.03	475	May 23 2002	1		--
P 46	611306	128410	554.00	4.04	115	June 5 2002		1	170
P 41'	610933	128228	550.14	3.50	110	June 5 2002	1	1	100

Table 4 gives an overview over the systematic classification of all metazoans found. It is probable that most of the differentiated taxa are actually several species since especially the microcrustaceans are a highly diverse group (Rouch & Danielopol 1997; Danielopol *et al.* 1999; Pospisil & Danielopol 2000; Galassi 2001; Danielopol *et al.* 2002).

Table 4: Systematics of the groundwater metazoans found. Bold: keyed to this taxonomical level

Phylum	Class	Sub-class	Order	Sub-order	Family	Genus	Species
Plathelminthes	Turbellaria		Tricladida		Dendrocoelidae	***Dendrocoelum***	
Nemathelminthes	**Nematoda**						
Mollusca	Gastropoda		Basommathphora		**Planorbidae**		
Annelida	Clitellata		**Oligochaeta**				
Arthropoda	Arachnida		**Acari**				
	Crustacea	Phyllopoda	Amphipoda		Gammaridae	***Niphargus***	
			Isopoda		Asellidae	***Proasellus***	
			Cladocera				
		Ostracoda	**Ostracoda**				
		Copepoda	Copepoda	**Cyclopoida**			
				Harpactiocoida			
	Insecta	Ectognata	**Ephemeroptera**				
			Plecoptera		Leuctridae	Leuctra	***Leuctra major***

After having removed all the detectable metazoans from the samples, the quantitative samples were passed through a series of two nested nets, with mesh diameters of 1 mm and 90 µm. All retained particles were washed from the nets into two seperate clean small aluminum dishes with distilled water. Any non-natural contents, such as plastic pieces from the well casing, were removed. The samples were oven dried at 105 °C to constant weight (Memmert GmbH+Co.KG-drying oven, D 06063, model 700). Then they were put in an desiccator (PYREX) for 2 hours before weighing to the nearest 0.1 mg with an analytical balance (Mettler-Toledo AT460 Delta Range®, No. K43343) (→ **dry weight**). Afterwards, they were ashed at 450 °C during 4 hours (Nabertherm®-muffle furnace, Mod L9/11/C6), put in the desiccator again for 2 hours and weighed again (→ **ash weight**). Finally, the contents of the dishes were removed, the dishes were cleaned thoroughly, put in the desiccator for 2 hours and weighed (→ **tare weight**).

Particles < 90 µm

Glass microfibre filters (Whatman GF/F-filters with 47 mm diameter and an average pore size of 7 µm) were furnaced (at 450 °C for 4 hours), left in the desiccator for two hours and weighed (→ **tare weight**). A defined volume of the well-shaken samples was then filtered through them. The filters with the entrapped particles were put back into clean small porcelain bowls and oven dried at 105°C to constant weight. Then they were put in an desiccator for 2 hours before weighing (→ **dry weight**). Afterwards, they were ashed at 450 °C for 4 hours, put in the desiccator again for 2 hours and weighed again (→ **ash weight**).

Ion concentrations

All samples were filtered through glass microfibre filters (Whatman GF/F-filters, see above) before the analysis. The analyses were conducted in the Lehrlingslabor at EAWAG Dübendorf. Calcium-, chloride-, magnesium-, potassium-, sodium- and sulfate-concentrations were determined by ion chromatography. Nitrate was determined with the automated hydrazine reduction method (Downes 1978).

Data Analysis

Preparation of the raw data

The distance of each groundwater well to the river in the channelized reach was derived from detailed plans of the transects (scale 1:250 to 1:375; courtesy GéoVal and BEG). In Bois de Finges, the distance from the river was measured on a 1:5000 map (courtesy Berthod) and converted into the real distance in meters. Inferences concerning the local and actual infiltration or exfiltration conditions were attempted by subtracting the water level in the river (in meters above sea level; courtesy BWG and M. Fette) from the water level in the groundwater well (also in meters above sea level). The term infiltration is used when river water infiltrates into the groundwater body (downwelling), the term exfiltration is used for groundwater seepage into the river (upwelling). However, no data on the actual water levels of the Rhône River in Bois de Finges are available since gauging stations are located only at Brig, Sion and Branson.

The qualitative biota samples were not used for statistical analyses. The counted metazoans of the quantitative samples were converted into number of individuals (of each taxon) per 100 l, using the data on the volume of liters pumped. These values were summed to give the total number of individuals per 100 l (of all taxa together) per sample. The taxa were assigned to two size classes (Table 5): organisms between 90 µm and 1 mm in size (termed 'small') and organisms larger than 1 mm (termed 'large'). For each well, the number of small and large taxa as well as the number of individuals of both size classes were calculated. The taxa were also divided into functional groups (Table 5): stygophile taxa, stygobite taxa and taxa that need to be determined to the species level in order to be sure whether they are stygophile or stygobite and which was termed 'not assignable taxa'. The taxa richness (number of taxa), the number of stygophile taxa , the number of stygobite taxa and the number of non-assignable taxa of each sample were counted. The Shannon-Wiener-Index and Evenness (after Shannon) were calculated for each sample according to the following formulas (Southwood & Henderson 2000):

	n_i: no. of individuals of taxon i
Shannon index = $-\sum_{i=1}^{S} n_i/N * \ln(n_i/N)$	N: total no. of individuals
Evenness = Shannon index/ln S	S: total no. of taxa

Table 5: Classification of the biota into size classes and functional groups. 'Small': between 90 µm and 1 mm in size; 'large': larger than 1 mm. Terminology of the functional groups sensu Gibert *et al.* (1994)

Taxon	Size class	Functional group
Dendrocoelum	'small'	stygophile
Nematoda	'small'	not assignable
Planorbidae	'large'	stygobite
Oligochaeta	'large'	not assignable
Acari	'small'	not assignable
Niphargus	'large'	stygobite
Proasellus	'large'	stygobite
Cladocera	'small'	not assignable
Ostracoda	'small'	stygobite
Cyclopoida	'small'	not assignable
Harpacticoida	'small'	not assignable
Ephemeroptera	'large'	stygophile
Leuctra major	'large'	stygophile (amphibite)

The taxa resp. individuals of the same classes were added in order to compare their relative share to the same taxonomic level. The occurrence frequency of each taxon was calculated by determining the percentage of all samples of one reach in which the taxon occurred. Occurrence frequency indicates the probability of the taxon to occur in a random sample that is taken from a specific site.

For the particles < 90 µm, the measured contents were converted into in milligrams per liter, using the data on the volume of water filtered. For the particles > 1 mm (coarse particles) and those between 90 µm and 1 mm in size, the measured contents were converted into milligrams per liter as well, using the data on the volume of water pumped. For all three particle fractions, the organic content (ash-free dry weight) was determined by subtracting the ash weight from the dry weight. The inorganic content was obtained by subtracting the tare weight from the ashed weight. The total organic particle content and total inorganic particle content were determined by adding all three of the respective size fractions. To obtain the fine particle fraction (particles < 1 mm), the weights of the particles < 90 µm and of the particles between 90 µm and 1 mm in size were summed. The statistical analyses showed that no additional information is gained by using all three size fractions; therefore, only two size fractions (fine and coarse) are reported. The total fine particle content was calculated by adding the fine organic particle content to the fine inorganic particle content. From that figure, the ratio of organic matter particle content to total fine particle content (POC/TFP) was derived for each organic particle size class. For an overview over the particle size fractions measured and calculated see Table 6.

Table 6: Size fractions of the particulate matter: organic particles, inorganic particles, total particles and ratio of organic particles to total fine particles. Bold: parameters included in the results

Organic particles	
< 90 µm	< 1 mm → **fine organic particles (FPOM)**
90 µm – 1 mm	
>1 mm → **coarse organic particles (CPOM)**	
total organic particles (TOM)	

Inorganic particles	
< 90 µm	< 1 mm → **fine inorganic particles**
90 µm – 1 mm	
> 1 mm → **coarse inorganic particles**	
total inorganic particles	

Total particles (org.+inorg.)	
< 90 µm	< 1 mm → total fine particles (TFP)
90 µm – 1 mm	
> 1 mm → **total coarse particles**	
total particles	

Organ. particles to total fine particles
fine organ. part./total fine part. **(FPOM/TFP)**
coarse organ. part./total fine part**. (CPOM/TFP)**
total organ. part./total fine part**.(TOM/TFP)**

Statistical analyses

All of the analytical and part of the descriptive statistics were run with the software S-PLUS 6.0 Professional Release 1 (Lucent Technologies, Inc.). The other descriptive analyses (all graphs except the clustering trees) were performed with the software Microsoft® Excel 97 SR-2.

For an overview of the statistical methods used, see Table 7 (page 33).

Differences between the two reaches: For descriptive statistics, the community composition (in classes) of each reach was depicted in pie-charts, separately for the number of taxa and the number of individuals. Additionally, taxa composition (in the taxonomical level indentified) was visualized in a stacked column-chart (number of individuals) and a 100% stacked column-chart (percentage of individuals). The occurrence frequency was used for a better visualization of the number of taxa that occurred in the two reaches and to show in which reach the individual taxa occured more frequently. Additionally, median, mean and data range of each faunal parameter were calculated. The following faunal similarity indices were calculated: Coefficient of Sörensen, Coefficient of Jaccard, Bray-Curtis Measure, Canberra Metric, Renkonen index and Wainstain index (sensu Krebs 1999). An agglomerative hierarchical cluster analysis grouped the fauna so close together (with the exception of a few wells) that no pattern could be observed and was excluded because it did not add meaningful information.

For analytical statistics, one- and two-sided two-paired Wilcoxon rank sum tests were performed to compare the faunal parameters of the two reaches. This is a robust test of group means of two independent samples by ranking the data. It is sometimes also called 'Mann-Whitney-U-Test'. The two-sided test shows whether the values differ in the two reaches and the one-sided test says if the values of a distinct parameter are larger or smaller in one of the reaches than in the other reach.

Influences on faunal distribution: For descriptive statistics, median, median absolute deviation and data range of each parameter were calculated. To see which groundwater wells were providing similar environmental conditions, agglomerative hierarchical cluster analyses (a method to classify data) were performed and visualized as clustering trees. Also, a normalizing diagram with the ratio of the ion concentrations of calcium to sulfate versus the ratio of chloride to sulfate was made to visualize possible mixing functions. To visualize possible connections between environmental and faunal parameters, scatterplots were graphed.

For analytical statistics, two-paired Wilcoxon rank sum tests (see above) were used to check if it is only the geomorphology that is different in the two river reaches, or if the values of other environmental factors that can have an influence on the biota also differed significantly between the two reaches. Two-paired Wilcoxon rank sum tests (see above) were also used to find out if the environmental parameters of the two reaches differed in their median absolute deviation. To determine which faunal parameters are dependent on which environmental parameters, linear regression and multiple stepwise backward regression (ANOVA) were performed (multiple environmental parameters influencing one faunal parameter). However, they did not contribute any sensible results since their outcomes were very inconsistent. This was probably due to the low correlations between environmental parameters and faunal parameters in general. These results were therefore excluded.

Table 7: Methods used for descriptive and analytical statistics (performed with the software S-PLUS 6.0 Professional, graphs also with Microsoft Excel 97)

Parameter	**Topic**	**Graph/Table to visualize it**
Faunal para-meters	community composition (classes resp. identified taxa)	pie-charts of taxa and individuals
		stacked column-charts of individuals
	simple overview over no. of taxa and their occurrence	100% stacked column-chart of the occurrence frequency
	overview over absolute values of the faunal parameters	Mean, Median, Data Range of the faunal parameters
	percentage of similarity of the fauna in the two reaches	faunal similarity indices
Environ-mental para-meters	overview over absolute values of the environmental parameters	Mean, Median Absolute Deviation, Data Range of the environmental parameters
	similarity of the environmental conditions of the wells	Clustering Tree
	possible mixing functions	Normalizing Diagram
	connections between faunalal and environmental parameters	Scatterplots

Parameter	**Problem/Question**	**Statistical Test to solve/answer it**
Faunal para-meters	faunalal difference between the two reaches present	one-sided Wilcoxon rank sum (two-paired) test
	values of faunalal parameters are higher resp. lower in one of the reaches	two-sided Wilcoxon rank sum (two-paired) test
Environ-mental para-meters	environmental difference between the two reaches present	one-sided Wilcoxon rank sum (two-paired) test
	values of environmental parameters are higher resp. lower in one of the reaches	two-sided Wilcoxon rank sum (two-paired) test
	difference in the variability of the environmental parameters between the two reaches present	one-sided Wilcoxon rank sum (two-paired) test
	variability of the environmental parameters between the two reaches is higher resp. lower in one of the reaches	two-sided Wilcoxon rank sum (two-paired) test
	possible influence of environmental parameters on faunal parameters	Linear Regression
	possible influence of multiple environmental parameters on faunal parameters	Multiple stepwise backward Regression

RESULTS

Differences between the two reaches

Descriptive Statistics: Table 8 gives an overview of the number of taxa found. In the channelized reach, fewer taxa were present than in Bois de Finges (eight resp. twelve). Only one taxon was found in the channelized reach that did not occur in Bois de Finges.

Table 8: Number of groundwater metazoan taxa found in the alluvial aquifer of the Upper Rhône River

Reach	Total no. of taxa found at the site	No. of taxa found exclusively at this site	Total taxa found in both sites together
Channelized reach	8	1	13
Bois de Finges	12	5	

The community composition of each reach (taxomonic level: classes) is depicted in Fig. 8a. Crustaceans dominated in both reaches, but that dominance was less pronounced in Bois de Finges than in the channelized reach. Fig. 8b (page 36) shows the taxa composition in a different way (taxonomic level: as far down as determined). For better visualization, the percentage of the total number of individuals is shown as well. The dominance of the crustaceans was mainly caused by the Cyclopoida and secondarily by the Harpacticoida and the Ostracoda. The Phyllopoda played a minor role. In Bois de Finges, the Oligochaeta were among the more important groups as well. The occurence frequency is depicted in Fig 8c. The Bois de Finges column consists of more different colors, indicating that more different taxa occurred in this reach. As for the channelized reach, the Cyclopoida, which were found in almost 50% of all samples, clearly dominated the occurrence frequency as well.

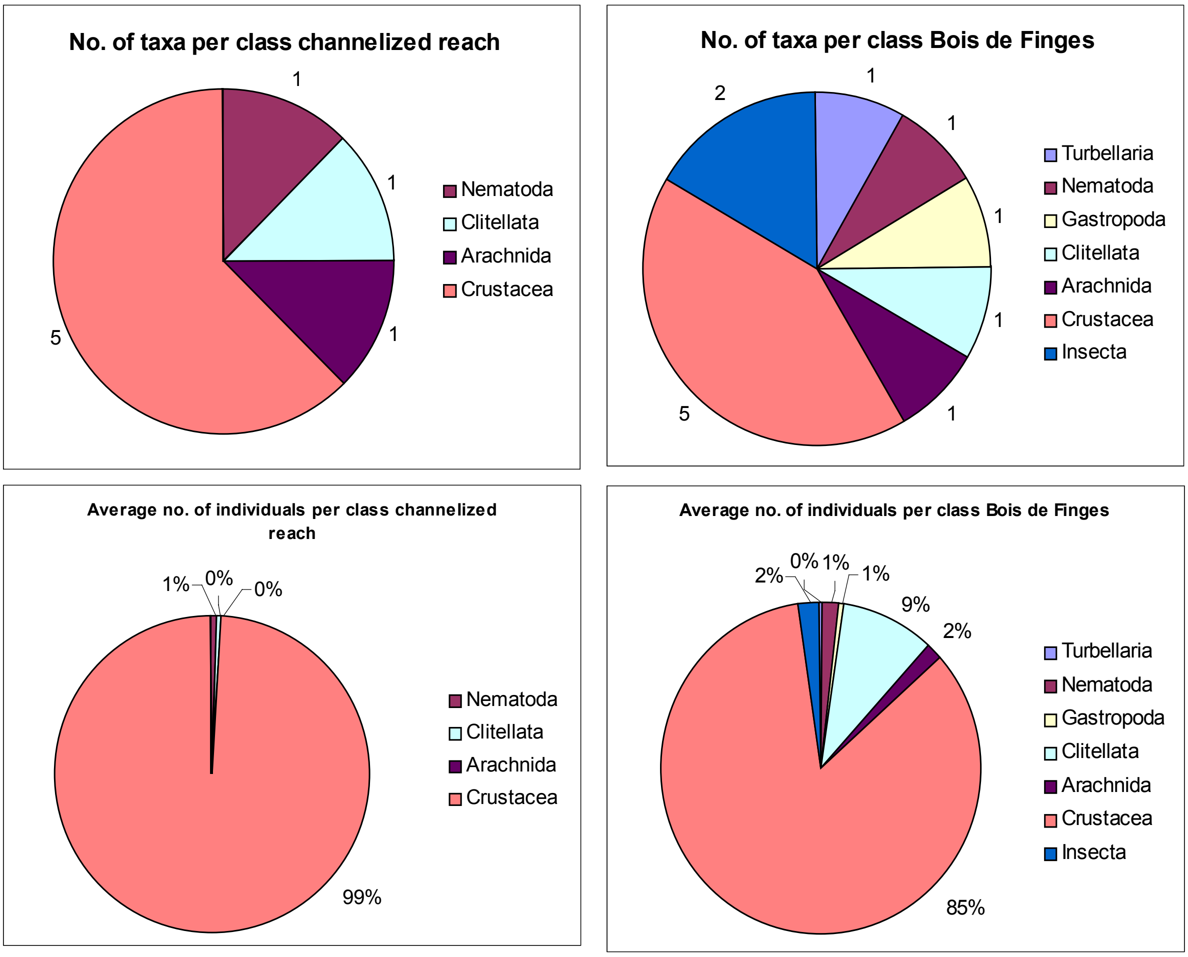

Fig. 8a: Community composition of the groundwater fauna on the class level of the two reaches ‘channelized‘ and ‘Bois de Finges‘. Upper row: number of taxa, lower row: percentage of individuals

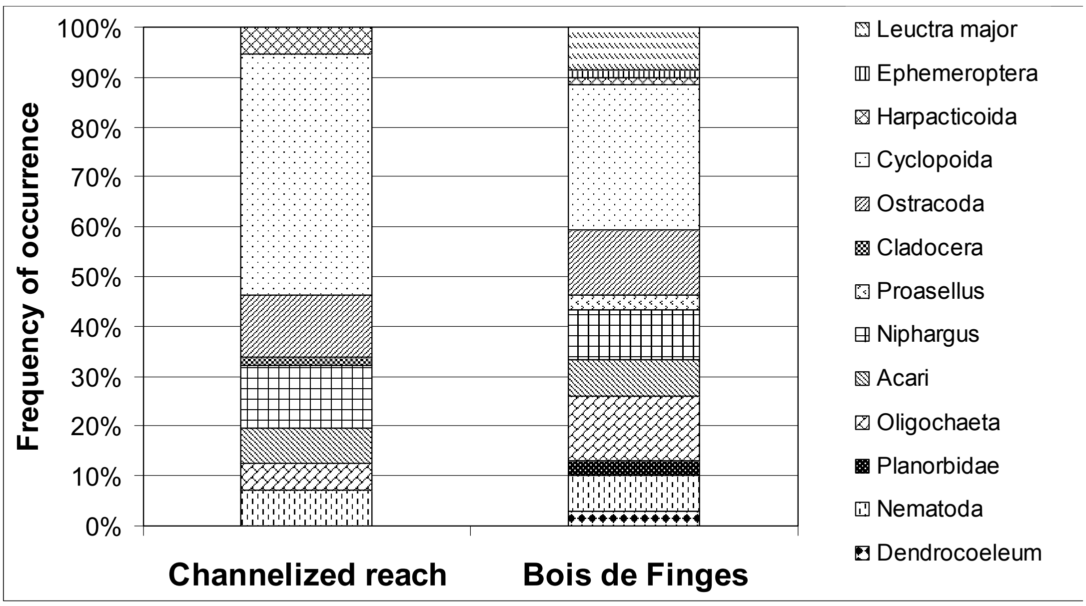

Fig. 8c: Occurence frequency of the taxa in the two groups of samples ‘channelized reach‘ and ‘Bois de Finges‘

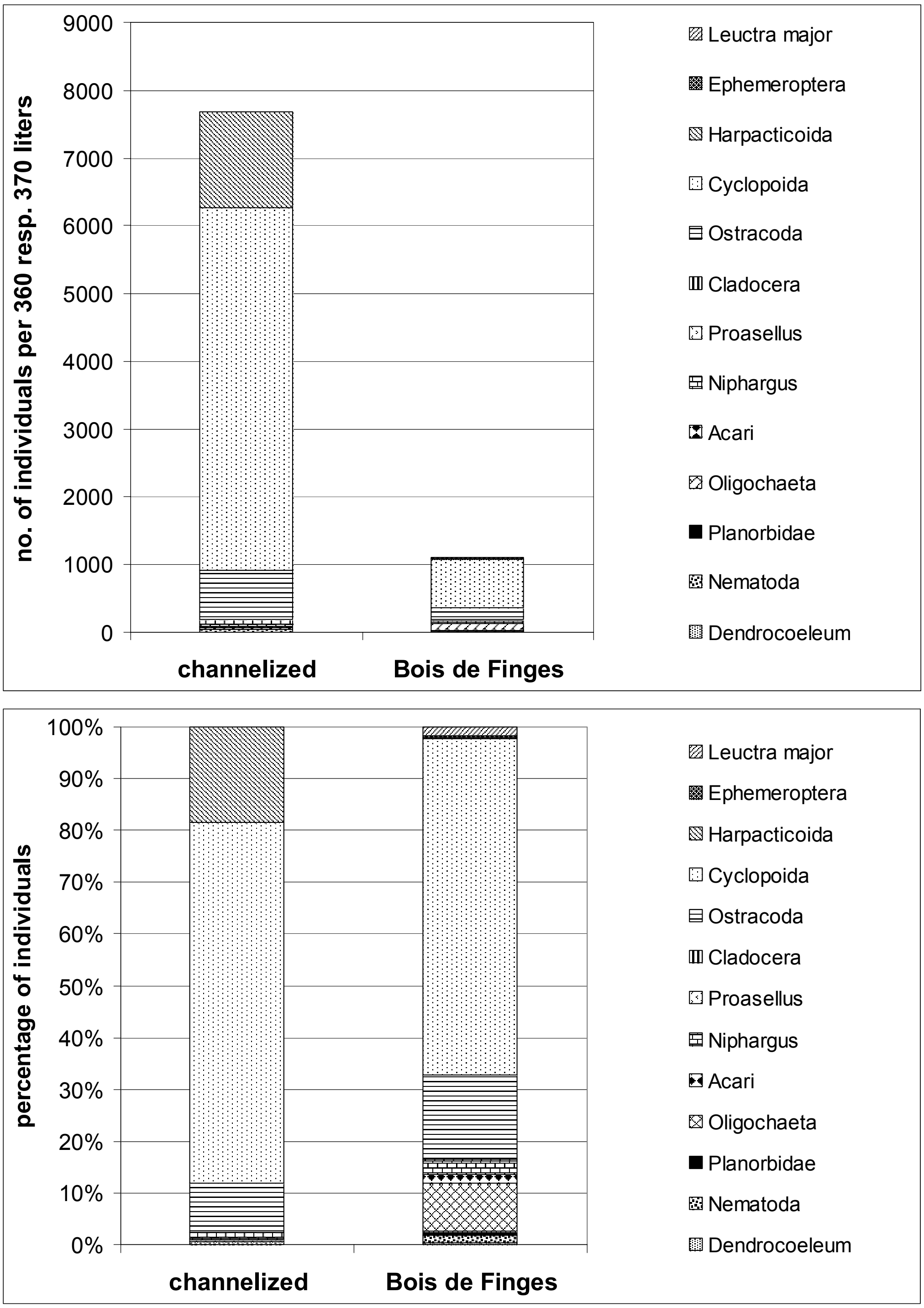

Fig. 8b: Taxa composition of the two reaches 'channelized' and 'Bois de Finges' (pooled supersamples, i.e. sum of individuals per reach for each taxon). Upper chart: absolute number of individuals, lower chart: number of individuals in percentage of total

The values of the calculated faunal similarity indices (Table 9) differ substantially among themselves. The highest percentages of similarity were obtained by indices that take only the number of taxa into account (Sörensen, Jaccard), and the lowest percentages of similarity by indices that take only the number of individuals (i.e. the markedness of dominance) into account (Bray-Curtis, Renkonen). Thus, the two reaches had a relatively high percentage of similar taxa but differed clearly regarding the abundances of the taxa.

Table 9: Faunal similarity between the channelized reach and Bois de Finges as expressed by different faunal similarity indices (calculated according to Krebs 1999)

Name of the coefficient	**Percent similarity**
Coefficient of Sörensen	70.00
Coefficient of Jaccard	53.85
Bray-Curtis Measure	22.57
Canberra Metric	23.70
Renkonen index	0.76
Wainstain index	40.98

Table 10 (medians, means and data range of all faunalal parameters) shows that most of the taxa were found in less than half of the samples (all those whose median equals zero). To get a better idea of the quantities, the mean is shown as well. Average abundances were low, the highest abundances were attained by the Cyclopoida, Harpacticoida and Ostracoda in the channelized reach. The entire group of stygophile taxa was completely absent in the channelized reach.

Table 10: Median, mean and data range of the faunal parameters. 'small': between 90 µm and 1 mm in size, 'large': larger than 1 mm; terminology of the functional groups sensu Gibert *et al.* (1994)

Faunal parameters	Median		Mean		Range	
	channelized reach	Bois de Finges	channelized reach	Bois de Finges	channelized reach	Bois de Finges
no. of Dendrocoelum per 100 l	none	0.00	none	**0.12**	keine	**0.0 - 2.3**
no. of Nematoda per 100 l	0.00	0.00	**1.31**	0.45	**0.0 - 30.0**	0.0 - 8.8
no of *Planorbidae* per 100 l	none	0.00	none	**0.20**	keine	**0.0 - 6.7**
no. of Oligochaeta per 100 l	0.00	0.00	0.80	**2.77**	0.0 - 13.6	**0.0 - 36.0**
no. of Acari per 100 l	0.00	0.00	**0.51**	0.50	0.0 - 6.67	**0.0 - 10.0**
no. of *Niphargus* per 100 l	0.00	0.00	**2.54**	0.77	**0.0 - 26.67**	0.0 - 19.0
no. of *Proasellus* per 100 l	none	0.00	none	**0.12**	keine	**0.0 - 3.33**
no. of Cladocera per 100 l	0.00	none	**0.09**	none	**0.0 - 3.33**	keine
no. of Ostracoda per 100 l	0.00	0.00	**20.33**	4.88	**0.0 - 681.8**	0.0 - 75.0
no. of Cyclopoida per 100 l	**20.00**	2.00	**161.84**	19.40	**0.0 - 3000.0**	0.0 - 400.0
no. of Harpacticoida per 100 l	0.00	0.00	**43.03**	0.03	**0.0 - 1250**	0.0 - 1.0
no. of Ephemeroptera per 100 l	none	0.00	none	**0.12**	keine	**0.0 - 4.0**
no. of *Leuctra major* per 100 l	none	0.00	none	**0.52**	keine	**0.0 - 7.0**
Taxa Richness	1.00	1.00	1.56	**1.87**	0.0 - 4.0	**0.0 - 7.0**
Shannon-Index	**0.15**	0.00	0.30	**0.34**	0.0 - 1.0	**0.0 - 1.4**
Evenness (Shannon)	0.34	**0.41**	0.40	**0.43**	0.0 - 1.0	**0.0 - 1.1**
no.'small' taxa	1.00	1.00	**1.28**	1.14	0.0 - 3.0	**0.0 - 4.0**
no. of 'large' taxa	0.00	1.00	0.28	**0.76**	0.0 - 1.0	**0.0 - 4.0**
no. of stygophile taxa	none	0.00	none	**0.22**	keine	**0.0 - 2.0**
no. of stygobite taxa	0.00	0.00	0.39	**0.57**	0.0 - 1.0	**0.0 - 3.0**
no. of not determinable taxa	1.00	1.00	**1.17**	1.08	0.0 - 3.0	0.0 - 3.0
no. of individuals per 100l	**24.17**	0.00	**213.26**	10.02	**0.0 - 4250.0**	0.0 - 104.0
no.of 'small' individuals per 100 l	**20.00**	4.00	**209.92**	25.38	**0.0 - 4250.0**	0.0 - 475.0
no. of 'large' individuals per 100 l	0.00	**2.00**	3.34	**5.04**	0.0 - 26.67	**0.0 - 43.0**
no. of stygophile individuals	none	0.00	none	**0.74**	keine	**0.0 - 7.0**
no. of stygobite individuals	0.00	0.00	**22.87**	5.98	**0.0 - 681.8**	0.0 - 76.0
no. of not determinable individuals	**15.83**	6.00	**190.39**	23.15	**0.0 - 4250.0**	0.0 - 412.0

Analytical statistics: Over a fourth (7 out of 27) of the variables tested differed significantly between the two river reaches (Table 11). These were the number of large taxa, the number of stygophile taxa, the number of stygophile individuals, the number of *Leuctra major*, the number of total individuals, the number of small individuals, and the number of Cyclopoida.

The one-sided test uses half the P-values of the two-sided test. As a consequence, three more variables became significant with this test: the number of Oligochaeta, the number of large individuals and the number of 'not determined' individuals (Table 11). The other 37% of the variables did not reveal a significant difference ($p > 0.05$).

Table 11: Results of Wilcoxon rank sum tests (= Mann-Whitney-U-Test), two-paired and one-paired, faunal parameters. * p < 0.05, ** p < 0.01, *** p < 0.001; 'small': between 90 µm and 1 mm in size, 'large': larger than 1 mm; terminology of the functional groups sensu Gibert *et al.* (1994)

Faunal parameters	significant difference (two-sided)	Median		sig. higher resp. lower (one-sided)	N	
		channelized reach	Bois de Finges		channelized reach	Bois de Finges
no. of *Dendrocoelum* per 100 l	n.s.			n.s.	36	37
no. of Nematoda per 100 l	n.s.			n.s.	33	37
no of Planorbidae per 100 l	n.s.			n.s.	36	37
no. of Oligochaeta per 100 l	n.s.	lower	**higher**	*	36	37
no. of Acari per 100 l	n.s.			n.s.	35	37
no. of *Niphargus* per 100 l	n.s.			n.s.	36	37
no. of *Proasellus* per 100 l	n.s.			n.s.	36	37
no. of Cladocera per 100 l	n.s.			n.s.	36	37
no. of Ostracoda per 100 l	n.s.			n.s.	36	37
no. of Cyclopoida per 100 l	**	**higher**	lower	***	33	37
no. of Harpacticoida per 100 l	n.s.			n.s.	36	37
no. of Ephemeroptera per 100 l	n.s.			n.s.	36	37
no. of *Leuctra major* per 100 l	*	lower	**higher**	**	36	37
taxa richness	n.s.			n.s.	36	37
Shannon index	n.s.			n.s.	36	37
Evenness (Shannon)	n.s.			n.s.	31	28
no.'small' taxa	n.s.			n.s.	36	37
no. of 'large' taxa	*	lower	**higher**	**	36	37
no. of stygophile taxa	**	lower	**higher**	**	36	37
no. of stygobite taxa	n.s.			n.s.	36	37
no. of not determinable taxa	n.s.			n.s.	36	37
no. of individuals per 100l	***	**higher**	lower	***	36	37
no.of 'small' individuals per 100 l	**	**higher**	lower	**	36	37
no. of 'large' individuals per 100 l	n.s.	lower	**higher**	*	36	37
no. of stygophile individuals	**	lower	**higher**	**	36	37
no. of stygobite individuals	n.s.			n.s.	36	37
no. of not deternimable individuals	n.s.	**higher**	lower	*	36	37

Most of the significantly differing parameters are closely related. The number of Cyclopoida clearly dominated the number of total individuals, the small size class and the functional group 'not determinable'. This explains why, in contrast to expectation, total abundance (total number of individuals) was higher in the channalized reach. On the other hand, *Leuctra major* was the taxon that contributed most to the stygophile group and also to the large size class.

Thus, the initial hypothesis is only partly supported. The presented data indicate that specific taxa and functional or size groups indeed do differ between the two contrasting locations, even though a higher overall biodiversity (measured by the numerical parameters Shannon index and Evenness) in Bois de Finges could not be demonstrated, and the individual abundances of most taxa were not higher in Bois de Finges. In particular, significant differences were observed with organisms that need to be able to migrate between the river and ground water, as well as with organisms of different sizes whose occurence is related to the size of the interstitial pores.

Influences on faunal distribution

Descriptive Statistics: Table 12 gives an overview of the physio-chemical data (median, median absolute deviation, range) of the two reaches.

Table 12: Median, median absolute deviation and range of the environmental parameters. 'fine': < 1 mm,'coarse': > 1 mm, TFP: total fine particles

Environmetal parameters	Median		Median Absolute Deviation		Minimum - Maximum	
	channelized reach	Bois de Finges	channelized reach	Bois de Finges	channelized reach	Bois de Finges
distance to the river	20.00	**175.00**	18.40	**90.00**	0.0 - 58.5	**70.0 - 770.0**
distance class	1.00	**4.00**	0	**2.00**	1.0 - 2.0	**2.0 - 16.0**
depth of the well	7.76	**8.32**	2.19	**2.35**	4.0 - 32.2	**2.2 - 40.0**
depth class	2.00	2.00	0.00	**1.00**	1.0 - 7.0	**1.0 - 9.0**
temperature	8.44	**8.75**	**0.99**	0.65	5.1 - 10.4	**7.4 - 13.2**
pH	**8.05**	7.70	**0.20**	0.10	7.3 - 8.6	**6.2 - 9.76**
electr. conduct.	384.00	**611.00**	58.50	**137.00**	248.0 - 541.0	**170.8 - 2040.0**
turbidity	**3.00**	0.25	**1.00**	0.25	**0.0 - 5.0**	0.0 - 3.5
chloride	**0.21**	0.19	**0.08**	0.06	0.1 - 0.4	**0.1 - 2.1**
nitrate	**0.05**	0.04	0.01	**0.02**	0.0 - 0.1	0.0 - 0.1
sulfate	0.78	**2.58**	0.15	**2.08**	0.3 - 1.1	**0.4 -12.8**
sodium	0.16	**0.24**	**0.06**	0.04	0.1 - 0.3	**0.1 - 1.0**
potassium	0.06	**0.07**	0.01	0.01	0.0 - 0.1	0.0 - 0.1
calcium	1.59	**94.45**	0.42	**20.59**	1.1 - 2.1	**18.5 - 365.7**
magnesium	0.36	**21.20**	0.04	**7.58**	0.2 - 0.6	**6.6 - 93.8**
fine org. part.	**36.59**	3.44	**26.66**	1.79	**3.6 - 1434.1**	0.3 - 193.5
coarse ort. part.	**1.69**	0.58	**1.32**	0.41	**0.0 - 51.0**	0.0 - 13.3
total org. part.	**98.96**	7.77	**87.02**	7.11	**7.5 - 2878.7**	0.6 - 145.6
fine inorg. part.	**2103.54**	73.76	**1754.46**	67.35	**140.3 - 24522.9**	1.3 - 4032.6
coarse inorg. part.	**1.88**	0.79	**1.66**	0.57	0.1 - 64.2	**0.05 - 181.1**
total inorg. part.	**3594.4**	143.15	**3243.56**	140.38	**21.4 - 49078.7**	2.8 - 8065.1
fine org. part./TFP	0.02	**0.05**	0.02	**0.03**	**0.0 - 2.2**	0.0 - 1.3
coarse org part./TFP	0.00	**0.01**	0.00	**0.01**	0.0 - 0.2	**0.0 - 0.4**
total org. part./TFP	0.05	**0.12**	0.04	**0.09**	**0.0 - 4.5**	0.0 - 2.5

In the clustering tree (Fig. 9), the wells of Bois de Finges (green) and those of the channelized reach (red) group into two separate clusters. This indicates that the ground water in the Bois de Finges area and the ground water in the region of the channelized reach are two distinct types of water. Also, when placed into a normalizing diagram with the ratio of the ion concentrations of calcium to sulfate versus the ratio of chloride to sulfate as in Fig. 10, no mixing function can be recognized.

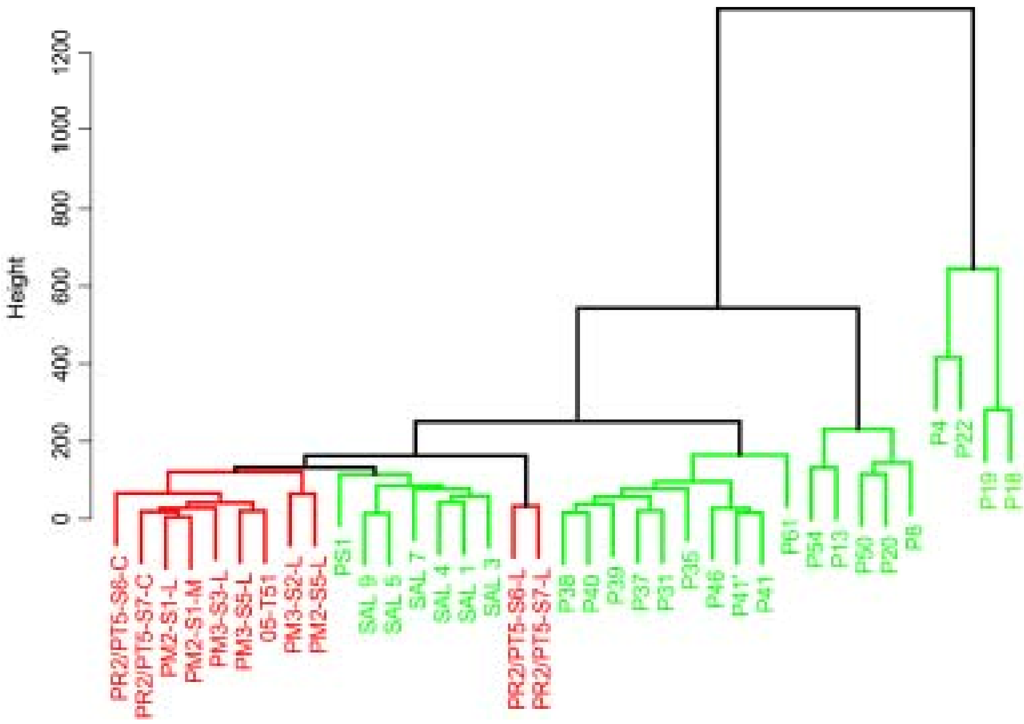

Fig. 9: Agglomerative hierarchical cluster analysis using the following parameters: distance from the river, well depth, temperature, pH, electrical conductivity, calcium, chloride, magnesium, potassium, sodium, sulfate, nitrate, ratio of coarse and fine organic matter particle content to total fine particle content (CPOM/TFP and FPOM/TFP). Red: groundwater wells in the channelized reach, green: groundwater wells in Bois de Finges

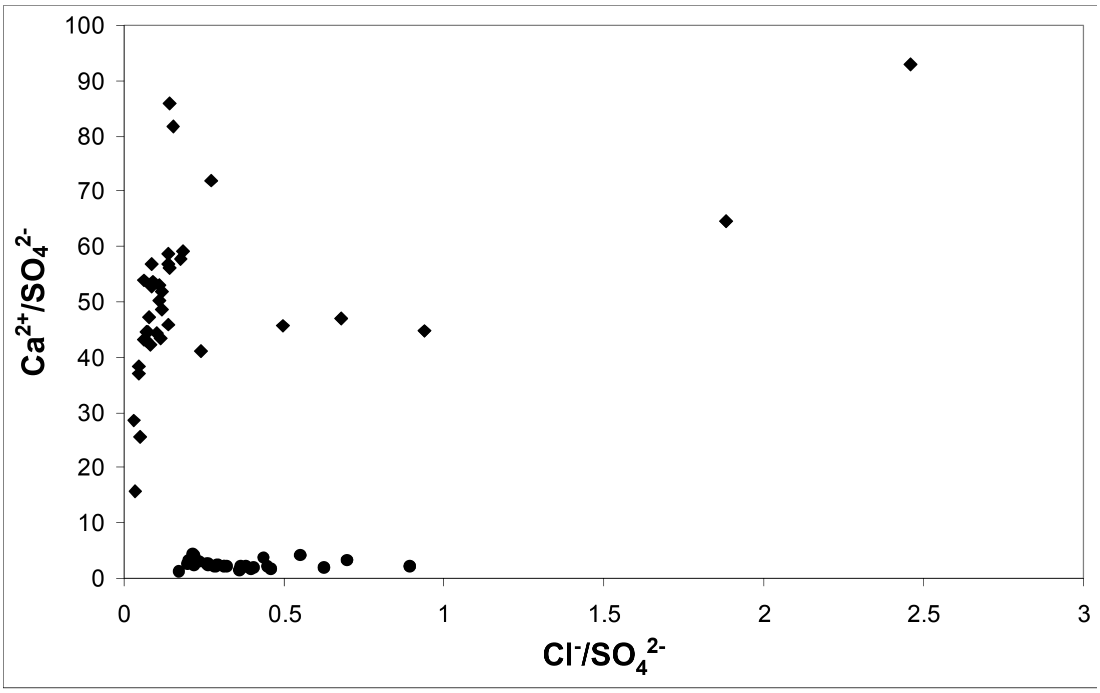

Fig. 10: Normalizing diagram with the ratio of the groundwater ion concentrations of calcium to sulfate versus the ratio of chloride to sulfate. ◊ Bois de Finges, o channelized reach

In Fig. 11, the similarity of the groundwater wells concerning their fine particle content is shown. The wells in which *Leuctra major* was found (orange), cluster together (with one exeption).

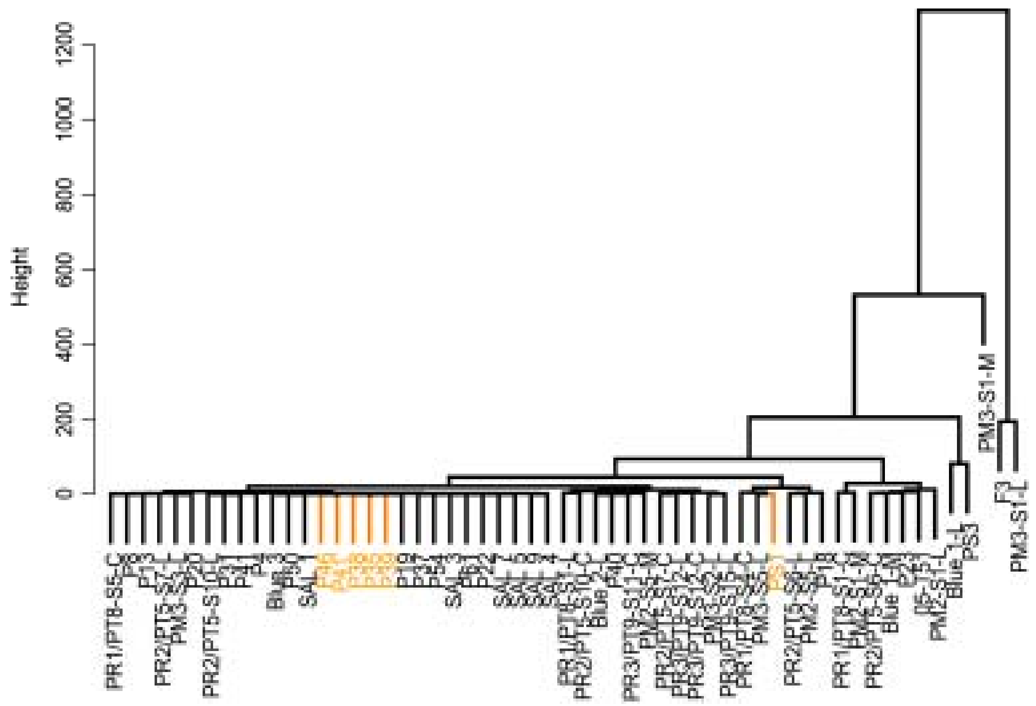

Fig. 11: Agglomerative hierarchical cluster analysis with the parameter fine organic matter. Orange: wells in which the stonefly *Leuctra major* was found

Analytical statistics: The observation that the groundwater bodies of the two reaches seem to be two distinct types of water is supported by the results shown in Table 13. The values of most of the environmental parameters differed between the Bois de Finges wells and the wells in the channelized reach.

Table 13: Results of Wilcoxon rank sum test (= Mann-Whitney-U-Test), two-paired and one-paired, environmental parameters. * $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$; 'fine': < 1 mm,'coarse': > 1 mm, TFP: total fine particles

Environmetal parameters	significant difference (two-sided)	Median		sig. higher resp. lower (one-sided)	N	
		channelized reach	Bois de Finges		channelized reach	Bois de Finges
distance to the river	***	lower	higher	***	31	37
depth of the well	n.s.			n.s.	36	37
temperature	*	lower	higher	*	36	37
pH	***	higher	lower	***	35	37
electr. conduct.	***	lower	higher	***	35	37
turbidity	***	higher	lower	***	32	36
chloride	n.s.			n.s.	15	37
nitrate	n.s.			n.s.	15	37
sulfate	***	lower	higher	***	15	37
sodium	**	lower	higher	**	15	37
potassium	*	lower	higher	**	15	37
calcium	***	lower	higher	***	15	37
magnesium	***	lower	higher	***	15	37
fine org. part.	***	higher	lower	***	29	28
coarse ort. part.	***	higher	lower	***	34	33
total org. part.	***	higher	lower	***	28	26
fine inorg. part.	***	higher	lower	***	29	29
coarse inorg. part.	*	higher	lower	*	34	33
total inorg. part.	***	higher	lower	***	29	27
fine org. part./TFP	***	lower	higher	***	34	28
coarse org. part./TFP	***	lower	higher	***	29	27
total org. part./TFP	***	lower	higher	***	29	26

The median absolute deviation shows that the variabilty of the physical and chemical parameters was significantly higher among the wells in Bois de Finges, indicating that the environmental conditions were more heterogeneous in the more natural river reach (Table 14). Accordingly, the branches of the clustering tree (Fig. 9) of the Bois de Finges wells (green) are further apart from each other than those corresponding to the channelized reach, and there are more subdivisions between them.

The environmental parameters characterizing the organic and inorganic particle contents, show an inconsistent pattern (Table 14). The variability of the absolute contents was much higher in the channelized reach than in Bois de Finges. This is especially true for the fine particles. In contrast, the variability of the ratio of organic matter particle content to total fine particle content (POC/TFP) was significantly higher in Bois de Finges. This means that there were proportionally much more fine particles in the channelized reach.

Table 14: Results of Wilcoxon rank sum test (= Mann-Whitney-U-Test), two-paired and one-paired, median absolute deviation of the environmental parameters. * $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$; ‘fine’: < 1 mm,‘coarse’: > 1 mm, TFP: total fine particles

Environmetal parameters	significant difference (two-sided)	Median Absol. Deviation		sig. higher resp. lower (one-sided)	N	
		channelized reach	Bois de Finges		channelized reach	Bois de Finges
distance to the river	***	lower	**higher**	***	31	37
depth of the well	n.s.			n.s.	36	37
temperature	n.s.			n.s.	36	37
pH	n.s.			n.s.	35	37
electr. conduct.	**	lower	**higher**	**	35	37
turbidity	n.s.	**higher**	lower	*	32	36
chloride	n.s.			n.s.	15	37
nitrate	n.s.			n.s.	15	37
sulfate	***	lower	**higher**	***	15	37
sodium	n.s.			n.s.	15	37
potassium	n.s.			n.s.	15	37
calcium	***	lower	**higher**	***	15	37
magnesium	***	lower	**higher**	***	15	37
fine org. part.	**	**higher**	lower	**	29	28
coarse ort. part.	***	**higher**	lower	***	34	33
total org. part.	***	**higher**	lower	***	28	26
fine inorg. part.	***	**higher**	lower	***	29	29
coarse inorg. part.	**	**higher**	lower	***	34	33
total inorg. part.	***	**higher**	lower	***	29	27
fine org. part./TFP	**	lower	**higher**	**	29	28
coarse org. part./TFP	***	lower	**higher**	***	28	27
total org. part./TFP	***	lower	**higher**	***	28	26

Precisely which factors influence faunal distribution, is difficult to ascertain. Initially, it appears that the data show no connections between the occurrence of the biota and the measured environmental variables whatsoever (Fig. 12). When single linear regressions were made, 47 of 594 calculated regressions were significant (11 or 1.9% with $p < 0.05$, 21 or 3.5% with $p < 0.01$ and 15 or 2.5% with $p < 0.001$; see Appendix 3). If Bonferoni-corrections of these regressions are made, that value is reduced to 7 of 594 (1.2%). The highest R-Square value was 0.37 (Harpacticoida vs coarse organic particles). In general, the faunal parameters that were significantly influenced by coarse organic matter and its ratio to TFP, had the highest R-Square values, but the scatterplots of these relationships show that the high R-Square scores are based on one single, and always the same, value (the PR1/PT8-S1-L-well that had an extremely high coarse organic particle content).

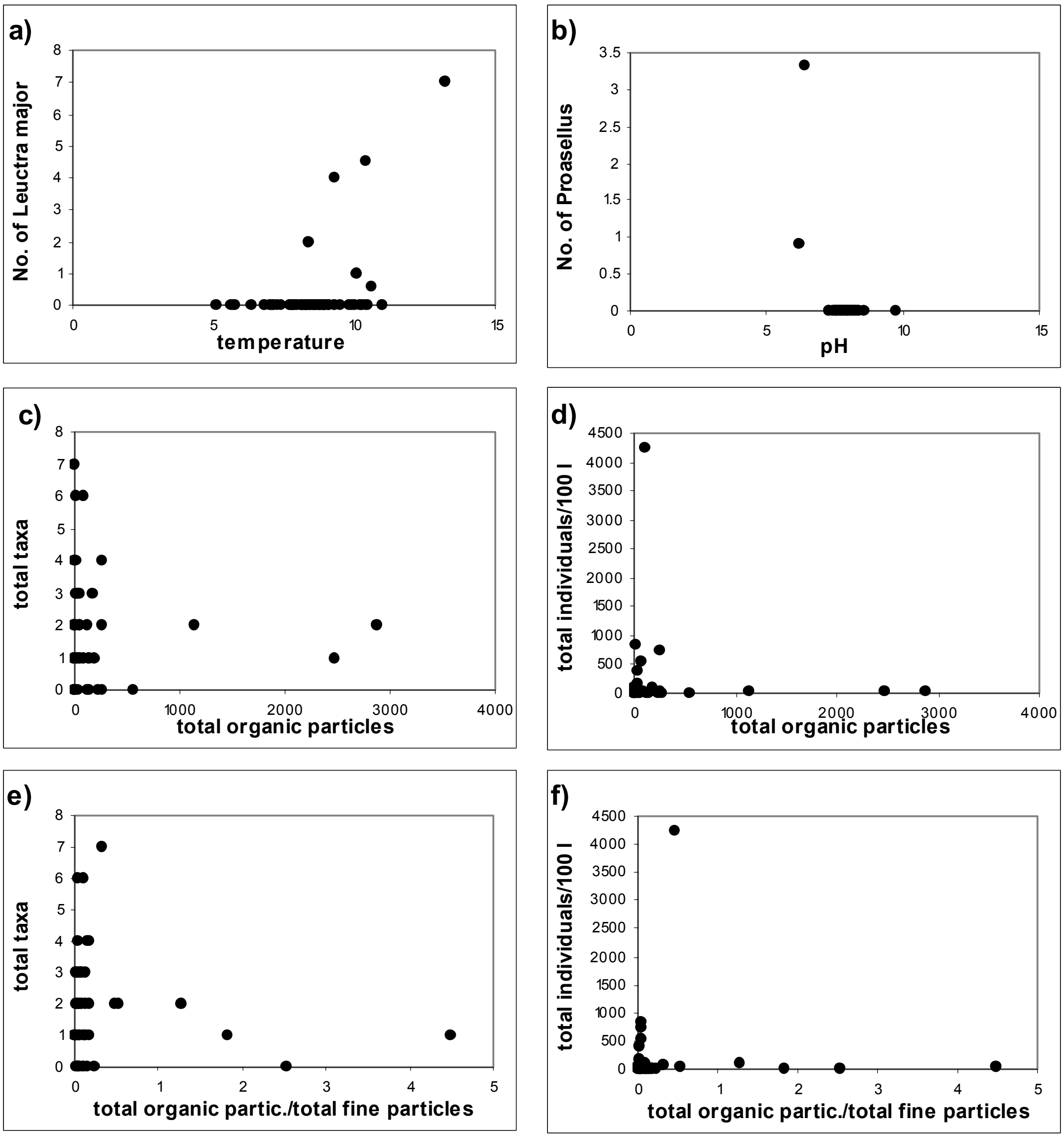

Fig. 12: Selected scatterplots of single regressions with high R-Square values or other interesting features. a) temperature vs no. of *Leuctra major*, b) pH vs no. of *Proasellus*, c) organic particles vs total taxa, d) organic particles vs total individuals, e) POM/TFP vs total taxa, f) POM/TFP vs total individuals. POM: total organic particles, TFP: total fine particles

In summary, it can be stated that the measured environmental variables did not influence the faunal distribution to an important extent, even though the environmental conditions differed between the channelized reach and Bois de Finges. It can be assumed that the environmental parameters did not reach values that would have put any constraints on faunal distribution. Hence, the second hypothesis is supported in part, that is, even though the environmental parameters measured did not influence the faunal distribution, it cannot be concluded that it is the river geomorphology that is responsible for the observed differences in faunal distribution.

DISCUSSION

Methodology

Field Methods: In general, cumulative samples that combine monthly samples over a year should be preferred to sampling just once. However, seasonal variation did not play a major role in several studies (e.g. Rouch 1991, 1992, 1995; Dumas *et al.* 2001), especially not when the fauna is dominated by crustaceans (in contrast to insects because of emergence) as it is the case for the investigated groundwater wells. The environmental conditions of the groundwater habitat are generally more stable than those of surface water habitats.

To be sure that dissolved oxygen is not a limiting factor for the fauna, the use of an oxygen sensor allowing *in situ* measurments up to a depth of 40 meters or pressure pump could be used in further research projects. However, the presented faunal data suggests that oxygen is not a limiting factor in the Upper Rhône River aquifer.

It is generally suggested to gather data of as many environmental parameters as possible that could be of importance. The more parameters which are measured, the higher the chances are that the key variables that influence faunal distribution can be determined. However, a lot of parameters are not easy to measure because of the inherent difficulties in accessing the groundwater habitat and most studies on the groundwater fauna therefore refrain from doing so.

Laboratory Methods: Determination of all taxa to species level by specialists would allow more precise statements in terms of taxa richness and faunal distribution. This procedure, however, is not very efficient for applied research. For monitoring projects, rapid bioassessment is usually preferred since it can be done with a minimal outlay.

Furthermore, it is suggested that the biomass of the collected metazoans is determined, even though it is known that protozoan biomasses are generally higher (Bretschko & Leichtfried 1988). Metazoan biomass would provide a different estimate of overall abundance and could be more valuable than the number of individuals because the organisms differ greatly in size. Here again, the effort is very high and the additional contributions must be evaluated carefully to see if they justify the effort.

Data analysis: It is generally suggested to normalize the raw data. Two things need to be considered (Krebs 1999): firstly, the number of species generally depends on sample size (here: the amount of liters pumped). This is analog to the 'species-area-relationship'. All samples need to be normalized to a standard sample size (usually the smallest). For groundwater sampling, pumping of different volumes of water is unavoidable: on the one hand, animal densities can be so low that large volumes of water need to be pumped. On the other hand, high contents of fine particles can rapidly clog the filtering nets and prevent the sampling of large water volumes. Secondly, the number of species is influenced by the number of collected individuals. For this problem, the method of rarefaction, for example, can be used for standardization. However, the accuracy gained by the application of these methods is not always crucial.
Reasons for the not significantly different numerical faunal parameters in the present study could be the level of taxa determination or that these just are not adequate parameters. For example, Chadwick & Canton (1984) found that so-called "dominance indices" (e.g. Shannon index) respond less to environmental changes than so-called "species richness indices" (e.g. Margalef's index).

Faunal composition and densities: The interstitial fauna was dominated by crustaceans in both study sites, which is not surprising as the mean depth of all wells together is over 10 meters. It has been suggested that in deeper gound water crustaceans are dominant because of the absence of their main competitors, i.e. insects (Sket 1999b).
In general, average metazoan densities are lower in this study compared to other studies. However, groundwater studies in alluvial aquifers have traditionnally focused on the top layer of the riverbed sediments, i.e. the uppermost centimeters down to a depth of maximal two meters. Several studies have shown that faunal abundances are highest close to the surface (e.g. Strayer 1994; Brunke & Gonser 1997). This is the case because in the top sediments, not only the characteristic inhabitants of the hyporheic zone itself are present. Benthic organisms use the river bottom as a refuge from adverse conditions, and epigean species are present because they cannot penetrate much deeper because of their higher oxygen level demands. Additionally, food resources are much scarcer deeper in the sediments because all organic matter that is present in the hyporheic zone ultimately originates from the surface, and they thus do not sustain very dense animal populations. Clearly, the abundances of organisms in the present study with its wells ranging from 2 to

40 meters in depth cannot be compared with the densities of studies that investgated the river bottom to a depth of 2 meters at the most.

Leuctra major was found on average 175 meters away from the river. For Switzerland, this is a novelty, but Stanford & Ward (1988) have found stoneflies up to 2 km away from the Flathead River in Montana, USA. These amphibite species spend one to three years in the groundwater habitat before returning to the surface water habitat for emergence.

Suggestions for further research: Repeating this kind of investigation (recording the state *in two different areas* with differing ecological integrity at the *same time*) in other places would allow the assessment of the importance of these findings for river systems in general. Additionaly, studies that compare the groundwater fauna of a degraded reach with an already revitalized reach should be carried out. This second step would allow to estimate if revitalization measures have the potential of being successful concerning the integrity of the surface water - groundwater ecotone. On the other hand, monitoring (documenting the state *before and* again *after* the revitalization project *in one area*) of the groundwater fauna accompanying river revitalization projects should be intensified. This should not only be done for the monitoring of measures that were specifically undertaken to improve the conditions in the hyporheic zone, like it was the case on the Brégnier-Cordon alluvial plain (Claret *et al.* 1999). Instead, revitalizations should always be accompanied by a monitoring programmee of the groundwater fauna, like Creuzé des Châtelliers *et al.* (1992) suggested it to be done for the Rhine River. In particular, the groundwater fauna of the Rhône River between Sion and Martigny should be monitored during and after the revitalization works that will take place in this area. By doing so, it is possible to take advantage of the present study that has already recorded the present state in the places where channel widenings are planned (cf. Appendix 4). Additionally, suggestions for the *Leitbild*, as well as for the target state and for key species, can be derived from the present study.

Differences between the two reaches

The initial hypothesis of this study, stating that the groundwater fauna differs between the two river reaches investigated, was only confirmed for part of the faunal parameters. The absence of stygophile (resp. epigean) taxa in the channelized reach shows that the groundwater fauna of the degraded reach of the Rhône River is impoverished, since one of the two functional groups which can be distinguished in groundwater metazoans is completely missing. This may indicate that stygophile taxa are indeed sensitive to the environmental 'health' of the surface water habitat (in particular channelization) and can be used as indicators for the ecological assessment of rivers.
The higher number of taxa and individuals of large organisms in Bois de Finges (resp. the higher number of individuals of small organisms and of total individuals in the channelized reach) could indicate that more pore space and more large pores are available in Bois de Finges. This would potentially allow for a broader array of different size classes of organisms to occur in Bois de Finges, which would promote a higher biodiversity. Here again, an impoverishment of the groundwater fauna (in terms of different size classes) of the channelized reach can be detected.

Other authors have reported impoverishments in the groundwater fauna of impaired sites, e.g. in polluted rivers with a bad water quality or along reaches affected by river incision. Danielopol (1976), who investigated the interstitial fauna of the Danube and the Piesting (Austria), found that high pollution in the interstitial habitat eliminated the hypogean fauna and the epigean fauna disappeared in areas with marked chemical reducing conditions. Sinton (1984) investigated macroinvertebrates in a sewage-polluted aquifer (New Zealand) and discovered that incidents of heavy contamination caused high mortality rates. In the ground water along the Rhône River in France, Schmidt *et al.* (1991) detected that interstitial fauna in a chronically polluted site was less diversified than the fauna of two unpolluted sites. In particular, the epigean insect larvae were lacking at the polluted site, whereas the micro-crustaceans were still abundant and diversified. These findings correspond with the results of a study conducted along the Rhine River (Creuzé des Châtelliers *et al.* 1992). In that study, industrial pollution is a possible explanation for the lack of stygobites, for the reduced diversity of stygophiles and for the weakly diversified and weakly abundant insect larvae. An alternative explanation for the dominant microcrustaceans, the rare insects and the reduced or absent hypogeans is the channelization of the river. Plenet & Gibert (1994) revealed that the sites of the Rhône and Ain Rivers (France) that were contamined by zinc or copper had a low number of individuals and low number of taxa. Iron and manganese had an indirect influence on hypogean invertebrate densities in a survey that was carried out by Malard *et al.* (1998) in the Haute-Savoie, France. In the reduced Fe-/Mn-containing zone of the aquifer, very low invertebrate densities were observed. Bravard *et al.* (1997) predicted that the effect of river incision in southeast France on the groundwater fauna is a decrease of biodiversity.

However, most of these studies also observed that if the water quality did not reach an extremely low level that caused toxical concentrations of contaminants or anaerobic conditions, the invertebrate densities were higher than in pristine, oligotrophic water. Other studies detected no influence or even a positive influence of anthropogenic impacts on the fauna. Dumas *et al.* (2001) found that the macrocrustacean assemblages in the Ariège Alluvial Aquifer (France) appeared to be independent of the ranking of the wells along a pollution gradient. Creuzé des Châtelliers & Marmonier (1993) hypothisized that gravel extraction, which induced an increase in groundwater discharge, made it possible for hypogean ostracods to colonize the downstrean end of the Miribel channel (France). The groundwater fauna was thus promoted by the human impact of gravel extraction.

Most of these studies are not of central importance for the present study. In Switzerland, water quality is in general not a problem any more, and river incision cannot be observed at the Rhône River in the investigated reaches. Nevertheless, these studies do show that groundwater metazoans can in principle be sensitive to human impacts in general and faunal diversity and abundance can be influenced adversely or, in some cases, also promoted to a certain degree.

However, what is true for water quality and river incision must not neccessairily be true for impacts on river geomorphology. To date, very few studies have made explicit comparisons of the fauna of reaches differing in their geomorphology. Olsen *et al.* (2001) compared the hyporheic communities of three geomorphologically contrasting reaches of a gravel-bed stream in New Zealand by taking freeze cores to a depth of 50 cm. Taxa richness and total invertebrate desity did not differ significantly between the floodplain site, the ‘forced pool-riffle’-site and the ‘plane bed’-site. Only 4 of the 20 most common taxa showed significant differences in density. Additionally, the depth distributions of the fauna at the different sites were compared. No significant differences were found between the depth distribution of the three sites except for site-depth interactions of three taxa. This study can most readily be compared with the present study, even though the three reaches did not differ primarily in their ecological integrity and the study utilized a different sampling method. In both the present study as well as in the study conducted by Olsen *et al.* (2001), taxa richness did not differ between geomorphologically contrasting river reaches.

In two other studies, Maridet *et al.* (1992 and 1996) compared the vertical distribution of the fauna (and environmental parameters) between three French streams that differed in their geomorphology. In both studies, the freeze-core technique was used and samples were taken to a depth of 60 cm. In the first study, the vertical distribution of the fauna of the Drac River differed significantly from the other two rivers (Galaure and Loire). In the second study, mean densities of invertebrates in the upper layer were significantly higher in the Triouzoune than in the other two rivers (Vianon and Ozange). It was concluded that the vertical distribution of organisms does not follow a uniform pattern in different streams. It is apparent that these two studies can only be compared to a certain extent with the present study since three distinct rivers (and not three reaches of the same river) were investigated and also because the river did not differ primarily in their ecological integrity. Moreover, the present study used different sampling methods and the vertical distribution of the fauna was not a central issue. The significance of these studies for the present work is that they illustrate the occurrence of faunal differences between streams with geomorphologically contrasting reaches.

Other studies did compare contrasting reaches of the same river in the course of the study but they did not explicitly evaluate these differences because they were not the focus of the study. For instance, Ward & Voelz (1990) investigated in the altitudional gradient of the meiofaunal community structure. The sites along the South Platte River that were studied were situated in a valley, several canyons, a meandring and an anastomosed reach and encompassed a total river length of about 420 kilometers. No distinct gradients in interstitial species composition were found. Marmonier & Creuzé des Châtelliers (1992) investigated the longitudinal gradient of the Rhône River in France along 320 kilometers. The five study sites included four by-passed channels and one unregulated reach. Three distinct species groups were found: species only recorded from the upstream sectors, species that were distributed more widely along the length of the river and species recorded exclusively from downstream sectors.

In sum, it can be said that: a) not many studies that are comparable to the present study in terms of the main focus of the research topic and the methodology are available. b) Their findings are contradictory – some studies detect differences, others do not. c) So far there are a few hints that the groundwater fauna of reaches with differing geomorphologies express these differences in their occurrence, depth distribution, richness ot abundance, but more research is needed to substantiate these findings. d) Comparative studies investigating the effects of human alterations of river geomorphology on groundwater assemblages are especially scarce.

Influences on faunal distribution

The second hypothesis which stated that faunal differences are not influenced by physio-chemical parameters, but only by river geomorphology, can neither be confirmed nor rejected with certainity. No strong influences of the measured environmental parameters on the groundwater organisms could be demonstrated, even though the two groundwater bodies differ in their physio-chemical composition. However, the conclusion cannot be drawn directly that it is river geomorphology that controls faunal distribution – river geomorphology is only one among other possible explanations. Furthermore, though clear influences on faunal variables by environmental parameters could not be detected by making linear regressions, this does not necessarily mean that no connection between these parameters exists (Hütte *et al.* 1994). There are several possible reasons why influences cannot be readily detected:

- the relationship is non-linear
- the acquired data cannot show the connection, for example because sample size was too small or because there were too many missing values for some variables.
- the faunal variables are influenced by other variables, like oxygen concentrations, colmation, local and actual situation concerning infiltration or exfiltration, additional hydrological parameters, (micro-)climatic parameters, the season of the year, the life cycle stage of the organisms investigated, the role of protozoans (e.g. biofilms), zoogeography and immigration barriers influencing faunal distribution, patchiness (i.e. metapopulation dynamics) influencing the occurrence of organisms in a certain place, deterministic and stochastic factors, etc.

Other studies have shown significant influences of single physio-chemical environmental parameters on the groundwater fauna. These included for instance the distance from the river (e.g. Marmonier *et al.* 1992), electrical conductivity, alkalinity, ion concentrations of nitrate, sulfate and calcium (Williams 1993), redox conditions, organic matter, temperature (Strayer *et al.* 1997) and POC/TFP (Brunke & Gonser 1999). However, the correlations found in these studies were often low, suggesting that other factors are also important in regulating hyporheic animal communities. Furthermore, there are several studies that did not detect influences of physio-chemical environmental factors on the fauna (e.g. Creuzé des Châtelliers & Marmonier 1993; Rulík 1995; Dumas *et al.* 2001). The non-significant relationships of the present study are thus nothing unusual.

Several studies showed that variables that were not measured in the present study are important in structuring faunal assemblages. In some environments, dissolved oxygen concentrations can be limiting and organisms with a lower tolerance for dissolved oxygen are found in different depths or distances from the river than organisms with a higher tolerance (e.g. Mösslacher *et al.* 1996; Marmonier *et al.* 2000). It has been shown that protozoans (i.e. bacteria and fungi in biofilms) are an important food resource for groundwater metazoans (Bärlocher & Murdoch 1989; Mauclaire *et al.* 2000) and their occurrence and amount is therefore expected to influence the occurrence and abundance of these metazoans. The local and actual situation concerning infiltration or exfiltration (up-and downwelling zones/recharge and discharge/origin of the water) have shown to be an important factor in structuring faunal assemblages (e.g. Dole-Olivier & Marmonier 1992). Additionally, infiltration velocity, groundwater circulation and current velocity play a role in the spatial distribution of the interstitial organisms (e.g. Creuzé des Châtelliers & Reygrobellet 1990; Creuzé des Châtelliers & Poinsart 1991; Vanek 1995). Not only hydraulic parameters, but also subsurface sediment characteristics (i.e. permeability, hydraulic conductivity, porosity, granulometry) influence the distribution of the fauna (e.g. Rouch 1991, 1992; Maridet *et al.* 1996). The role of (micro-)climatic parameters (Mathieu & Amoros 1982), the season of the year (McElravy & Resh 1991) and the life cycle stage of the organisms investigated (Seyed-Reihani *et al.* 1982; Crenshaw *et al.* 2002) has been documented by several studies as well. On a larger scale, zoogeography and immigration barriers can also influence faunal distributions (Strayer 1994). On a small scale, patchiness (i.e. metapopulation dynamics) can influence the occurrence of organisms in a certain place (Dole-Olivier & Marmonier 1992). Thus is becomes clear that many factors are involved in structuring faunal assemblages and therefore, determining which ones are the most important ones for the investigated groundwater system is inherently difficult.

Another factor, which several authors have suggested might exert a strong influence on the inhabitants of the hyporheic zone, is colmation (Schmidt 1994; Ward *et al.* 1998; Ward & Wiens 2001). Colmation is also the parameter that is considered to be the most realistic of the possible explanations for the observed faunal distribution in the Upper Rhône River in Switzerland, even though this could not be directly demonstrated in the present study. Additionally, habitat heterogeneity seems to play a role for the faunal distribution observed by the present study. These assumptions are supported by the following observations:

Firstly, the fact that no epigean taxa could be found in the channelized reach suggests that metazoans cannot migrate between the surface water and ground water in this reach because migration pathways are severed. There is strong evidence that the Rhône River is heavily colmated in this reach (M. Fette, personal communication) and it is assumed that this is the reason for the disconnection of migration pathways as it has been shown by Gayraud *et al.* (2002). This would indicate that an important function of the river ecosystem, i.e. the exchange between rivers and ground waters, has been impaired. The fact that many more individuals of small taxa occur in the channelized reach and more large taxa and individuals occur in Bois de Finges could indicate that there are only small interstitial pores in the channelized reach. This would fit well with the much higher amount of fine sediment particles in that reach. On the one hand, this would mean that the total available pore space would be reduced and faunal densities would decrease, and on the other hand that the larger pores that are needed by large organisms would become clogged and therefore scarcer. The diversity in the size classes of the organisms would be reduced and many organisms that are dependent on larger pore spaces would be excluded from this reach. Hence, higher biodiversity would be prevented by the missing availability of differentiated pore sizes and of overall porespace. However, as the amphibite *Leuctra major* dominates the 'large' size class and since there were no significant differences between the two reaches in the densities of the other, not amphibite taxa of the 'large' size class (Oligochaeta (only significant in the one-sided test), *Niphargus*, and *Proasellus*) it must be assumed that this explanation is not of importance. It seems more likely that the reduced pore size is a side-effect of colmation: the entry of a high proportion of fine sediments that causes colmation, leads to a reduction of the interstitial pore space size in the uppermost layer of the sediment surface, and therefore acts as a barrier for migrating organisms.

Secondly, the higher habitat heterogeneity in Bois de Finges promotes a higher biodiversity and thus enables a mixed fauna of stygophile and stygobite as well as organisms of various sizes. The observed higher dominance of Copepoda in the channelized reach fits well into this picture.

Colmation and habitat heterogeneity are not direct measures of the ecological integrity of river geomorphology itself as a whole, but they are certainly individual aspects of river geomorphology that have a high ecological importance. Both parameters are influenced by anthropogenic activities within the catchment and on floodplain surfaces. Colmation is promoted by the higher proportion of fine sediments released from the reservoir lakes and reinforced by the reduced contact zone between surface and subsurface water which is a consequence of the channelization, i.e. of channel straightening and embankments (Pringle & Triska 2000). Channelization also reduces habitat heterogeneity by reducing the heterogeneity of hydrological exchange because it isolates the river from its flood plain and disconnects the diverse floodplain water bodies hydrologically from the main channel (Hancock 2002). Several studies have found differences in the fauna of the main channel and of other water bodies of the flood plain, such as backwaters or abandoned channels (Dole 1983; Marmonier *et al.* 1992; Ward & Palmer 1994; Danielopol & Baltanás 1996). These studies show that habitat heterogeneity on the flood plain enhances the diversity of the groundwater fauna.

In sum, it seems that channelization does not excert a **direct** influence on the groundwater fauna. Particularly in the case of the Upper Rhône River between Sion and Martigny, the effects of hydropower utilization and channelization overlap and interact. However, channelization influences groundwater assemblages **indirectly** by reinforcing the effects of colmation and by reducing habitat heterogeneity across the flood plain.

Conclusions

Theoretical inferences: The groundwater aquifers associated with two reaches of the Upper Rhône River which differ in their ecological integrity (most obviously concerning river geomorphology) are inhabited by a fauna that evinces specific differences between the two reaches. This shows that groundwater organisms can in principle be sensitive to the ecological state of the geomorphology of the surface water habitat. Therefore, it can be said that even though the direct relationship between river geomorphology and the observed faunal distributions could not be demonstrated conclusively in the present study, river geomorphology could nevertheless be an important factor in structuring subterranean faunal assemblages. River geomorphology seems to be important for the fauna because it is linked the extent of colmation and the degree of habitat heterogeneity. However, the effects of channelization are difficult to separate from the effects of hydropower utilization in the case of the Upper Rhône River. Further research is needed to prove the influence of the ecological integrity of river geomorphology on the groundwater fauna.

Practical applications: The absence of the epigean fauna in the groundwater wells of the channelized reach (between Sion and Martigny) indicates that migration pathways between the surface water habitat and groundwater habitat are disrupted. This is probably caused by the entry of a high proportion of fine sediments in the channelized reach which leads to colmation of the riverbed and banks. In channelized rivers, where the contact zone between surface and subsurface waters is already reduced by river straigthening and embankments, the clogging of the interstices has an even more devastating effect. The hyporheic zone in this reach thus cannot serve as a refugee for epigean organisms. Furthermore, amphibite species that require both surface waters and ground waters to complete their life cycle and that do occur in the Upper Rhône River area, as demonstrated in the Bois de Finges, are inhibited. The impairment of the hydrological exchange and biological connectivity between the river channel and its groundwater aquifer may not only have consequences for the surface and subsurface fauna, but also for groundwater purity and supply in that region.

In contrast, a mixed fauna is present in the near-natural Bois de Finges, with amphibite stoneflies occuring up to several hundred meters away from the river and down to over eight meters depth into the aquifer. An additional reason for the mixed fauna in Bois de

Finges, with epigean as well as hypogean taxa and both large and small organisms, could be the observed greater habitat heterogeneity in that reach.

It is suggested to use the groundwater fauna of Bois de Finges as a *Leitbild* (sensu Jungwirth *et al.* 2002) for defining the target state of the planned revitalization of the Rhône River in the channelized reach. *Leuctra major* and other epigean species can serve as possible key indicator species for the monitoring of that revitalization since their presence in the system documents intact migration pathways. In other words, if these organisms appear in the ground water of the reaches where channel widenings were implemented, the conducted measures must have reestablished hydrological and biological connectivity between surface and subsurface waters. In that case, the revitalization can be viewed as a success from the perspective of the surface water-groundwater ecotone.

It is evident that the mere presence of benthic invertebrates and fishes in a river cannot demonstrate the state of the connectivity between the river and its aquifer. Only if it can be proven that gravel spawning fish can reproduce in the investigated rivers, and/or that epigean metazoans can penetrate the streambed, the ecological integrity of the vertical connectivity can be assessed. The latter can be accomplished by investigating the hyporheic habitat. By doing so, the contribution of the groundwater fauna to overall biodiversity will be taken into account as well.

The findings of the present study are an example that shows that the vertical dimension of rivers can bring valuable contributions for the assessment of river ecosystem functions. Not only may investigations in the surface water-groundwater ecotone indicate where specific problems of a degraded river reach may lie, but they also have the potential of showing whether previously detected problems were solved by revitalization measures. Hence, for a holistic approach, the contributions of the surface water-groundwater ecotone should no longer be excluded from river management concepts.

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APPENDIX

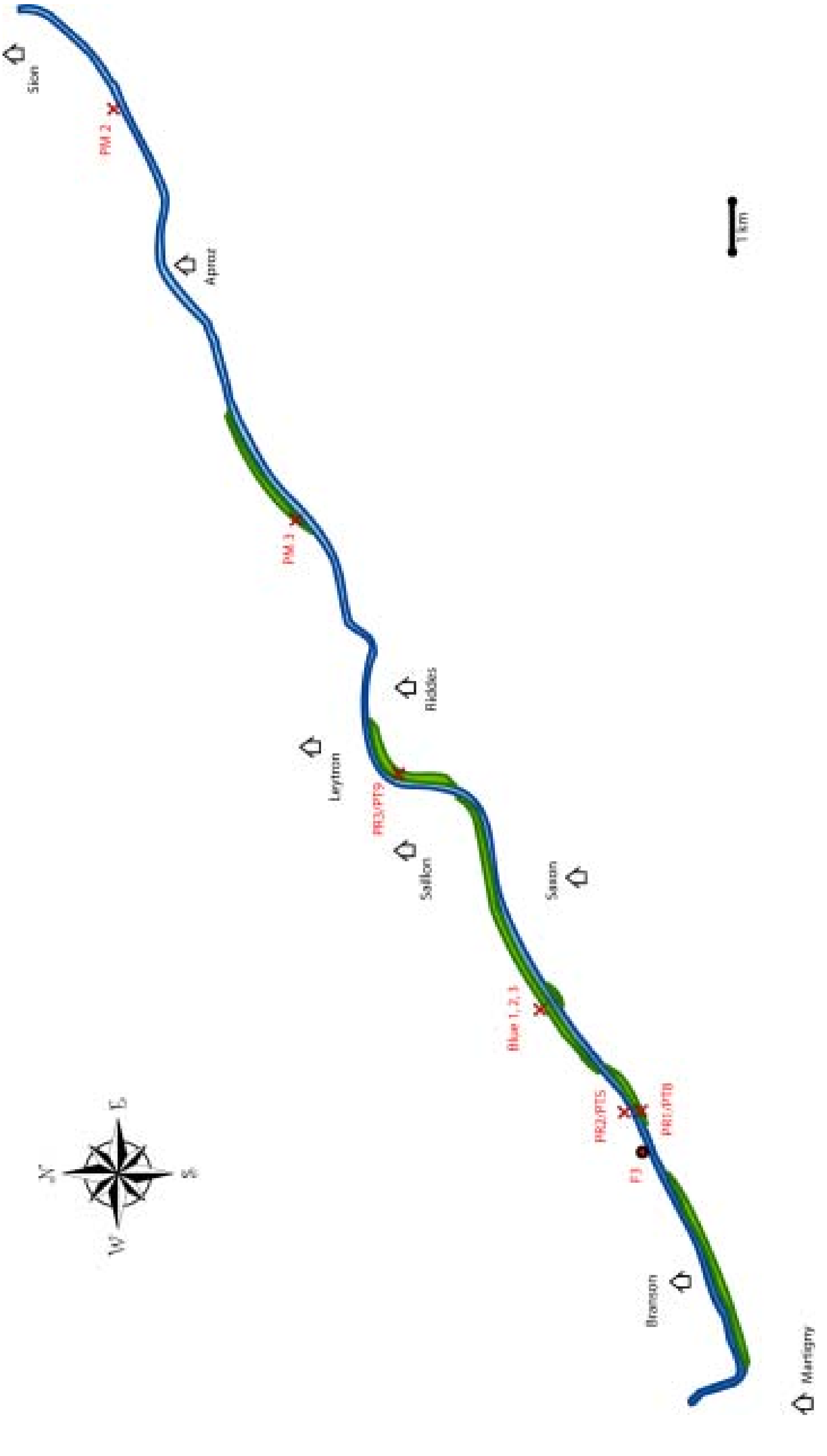
Appendix 1: Technical data of the groundwater wells of Bois de Finges. Coordinates, z soil well and distance to the river: after Berthod (1997), Schürch (2000) and P. Fontana, personal communication

ID well	description of the location in words	stream bank side	river kilo-meter	x coordi-nate well [m]	y coordi-nate well [m]	z soil well [m a.s.l.]	distance to the river [m]	depht well [m]
P 38	a bit upstream of gravel pit 'Pfyn'	left	all bet-ween 83000 and 90000	610498	127751	544.76	135	8.10
P 40	a bit upstream of gravel pit 'Pfyn'	left		610582	127814	546.67	135	7.10
P 39	a bit upstream of gravel pit 'Pfyn'	left		610341	127700	544.10	105	7.80
P 37	a bit upstream of gravel pit 'Pfyn'	left		610358	127629	543.90	175	4.45
P 30	at the height of gravel pit 'Pfyn'	left		610298	127410	541.59	315	8.55
P 2	at the height of gravel pit 'Pfyn'	left		610298	127242	541.86	420	9.05
P 4	at the height of gravel pit 'Pfyn'	left		613044	127001	541.33	625	6.90
PS 3	a bit downstream of gravel pit 'Pfyn', other side of the river	right		610039	127825	542.54	95	8.62
PS 1	across from gravel pit 'Pfyn'	right		610297	128008	545.03	155	8.32
P 31	between gravel pit 'Pfyn' and Sierre Bridge	left		610020	127299	539.93	185	4.71
P 35	between gravel pit 'Pfyn' and Sierre Bridge	left		609885	127236	539.41	160	7.92
P 54	at the height of 'Landgut Pfyn'	left		612219	128227	560.82	425	8.39
P 13	at the height of 'Landgut Pfyn'	left		612150	128310	560.46	315	10.38
P 61	at the height of 'Landgut Pfyn'	left		612120	128380	560.83	245	7.61
P 53	at the height of 'Landgut Pfyn'	left		612319	128067	561.30	610	8.93
P 50	downstream of 'Landgut Pfyn'	left		611780	127833	554.65	495	2.92
P 16	downstream of 'Landgut Pfyn'	left		611708	127960	554.78	355	8.10
P 49	downstream of 'Landgut Pfyn'	left		611600	128138	554.96	145	4.65
P 20	downstream of P49/P16/P50 and further away from the river	left		611553	127773	553.34	475	8.03
P 19	downstream of P49/P16/P50 and further away from the river	left		611564	127651	553.24	585	8.90
P 18	downstream of P49/P16/P50 and further away from the river	left		611611	127457	553.99	770	6.68
P 8	close to 'Rosensee'	left		611145	127505	554.76	595	12.10
P 22	close to 'Rosensee'	left		611278	127413	557.79	705	12.30
P 23	close to 'Rosensee'	left		611011	127381	554.25	670	12.50
P 24	close to 'Rosensee'	left		611012	127382	554.19	670	8.81
SAL 9	across from ‘Landgut Pfyn'	right		611789	128745	559.37	140	32.80
SAL 6	across from ‘Landgut Pfyn'	right		612042	128788	561.82	95	40.00
SAL 7	across from ‘Landgut Pfyn'	right		611956	128718	559.62	85	33.80
SAL 5	across from ‘Landgut Pfyn'	right		611895	128768	560.79	145	30.27
SAL 4	across from ‘Landgut Pfyn'	right		612144	128843	563.67	70	39.72
SAL 3	across from ‘Landgut Pfyn'	right		611871	128700	559.48	70	36.60
SAL1	across from ‘Landgut Pfyn'	right		611874	128716	560.21	95	11.28
P 47	a bit upstream of 'Russubrunnu'	right		611317	128367	554.10	75	4.40
P 46	a bit upstream of 'Russubrunnu'	right		611306	128410	554.00	115	4.04
P 44	close to 'Russubrunnu'	right		610927	128396	550.19	180	2.19
P 41‘	close to 'Russubrunnu'	right		610933	128228	550.14	110	3.50
P 41	close to 'Russubrunnu'	right		610990	128217	549.90	95	5.97

Appendix 2: Technical data of the groundwater wells of the channelized reach. River kilometer, z soil well and distance to the river: courtesy of Bureau d'Etudes Géologiques (BEG) and Vétroz and GéoVal - Ingenieurs Géologiques SA, Sion

ID well	description of the location in words	stream bank side	river kilo-meter	x coordi-nate well [m]	y coordi-nate well [m]	z soil well [m a.s.l.]	distance to the river [m]	depht well [m]
PR1/PT8-S5-C	BEG-transect 'Fully rive gauche' (at Charrat/Fully)	left	43004	576047	109533	461.85	0	4.03
PR1/PT8-S5-L								11.11
PR1/PT8-S4-C				576054	109520	462.85	26.9	5.11
PR1/PT8-S4-L								11.70
PR1/PT8-S1-C				576064	109500	461.48	44.2	5.75
PR1/PT8-S1-L								11.75
PR2/PT5-S6-C	BEG-transect 'Fully rive droite' (at Fully/Charrat)	right	43055	576073	109617	462.00	0.2	4.80
PR2/PT5-S6-L								7.80
PR2/PT5-S7-C				576069	109625	462.72	20	4.33
PR2/PT5-S7-L								10.10
PR2/PT5-S10-C				576056	109650	463.08	38.4	4.22
PR2/PT5-S10-L								8.16
Blue 2-L	3-well longitudinal transect at Mazembroz/ Saxon	right	about 45300	577810	110908	missing	missing	32.20
Blue 2-M								17.90
Blue 3				577785	110888	missing	missing	6.28
Blue 1-L				577834	110925	missing	missing	29.70
Blue 1-M								14.04
F3	Transect at Fully/Charrat	right	42385	575362	109376	462.15	12	6.15
PM3-S1-M	(at) GéoVal-transect 'Riddes' (at Chamoson/ Fey)	right	55429	586056	115203	475.36	1	5.50
PM3-S1-L								5.65
PM3-S2-L				586052	115208	475.49	12.2	5.35
PM3-S3-L				586050	115212	477.83	22.5	9.16
PM3-S5-M				586031	115235	476.46	42	7.98
PM3-S5-L								8.12
05-T51				586026	115241	473.90	58.5	7.01
PM2-S1-L	GéoVal-transect 'Sion' (at the downstream end of Sion/ Salins)	right	63410	593057	118350	485.87	3	7.90
PM2-S1-M								7.89
PM2-S2-L				593043	118376	486.51	33.8	7.71
PM2-S5-L				593034	118397	484.78	54.5	6.79
PM2-S5-M								6.73
PR3/PT9-S12-C	BEG-transect 'Riddes' (at Riddes/ Saillon; point bar)	left	50367	581675	113598	469.41	7	4.50
PR3/PT9-S12-L								8.80
PR3/PT9-S11-L				581670	113600	468.64	0	8.12
PR3/PT9-S11-C								5.12
PR3/PT9-S15-L				581690	113590	468.91	25	7.90
PR3/PT9-S15-C								4.90

Appendix 3: Map showing the planned channel widenings of the "Third Correction of the Rhône River" (green stripes) together with the transects that were sampled in the present study (red dot for single well resp. red crosses for transects). Courtesy of Canton Valais



Appendix 4: R-Square values of single linear regressions with environmental parameters (rows) influencing faunal parameters (columns). Red: $p < 0.05$, yellow: $p < 0.01$, green: $p < 0.001$

R^2	Physio-chemical parameters (including ion concentrations)												
Faunal abundances	distance river	depht well	tempera-ture	pH	conducti-vity	turbidity	chloride	nitrate	sulfate	sodium	po-tassium	calcium	mag-nesium
Dendrocoeleum	.01	.00	.02	.13	.01	.02	.01	.00	.00	.00	.06	.01	.00
Nematoda	.01	.00	.01	.00	.01	.04	.00	.01	.01	.00	.00	.02	.02
Planorbidae	.00	.00	.07	.15	.00	.01	.00	.01	.00	.00	.07	.00	.00
Oligochaeta	.01	.01	.02	.00	.00	.00	.00	.01	.01	.02	.01	.01	.00
Acari	.04	.00	.00	.02	.00	.04	.00	.07	.00	.00	.01	.00	.01
Niphargus	.02	.00	.02	.01	.01	.09	.06	.07	.02	.00	.03	.04	.03
Proasellus	.00	.00	.06	.24	.00	.00	.01	.01	.00	.00	.13	.00	.00
Cladocera	.01	.01	.00	.00	.00	.02	NA	NA	NA	NA	NA	NA	NA
Ostracoda	.01	.01	.06	.00	.01	.01	.01	.02	.01	.02	.00	.02	.02
Cyclopoida	.02	.00	.00	.00	.02	.00	.00	.00	.00	.00	.00	.00	.00
Harpacticoida	.01	.00	.00	.00	.01	.00	.00	.00	.01	.00	.00	.02	.01
Ephemeroptera	.08	.00	.00	.26	.02	NA	.00	.06	.00	.00	.01	.01	.01
Leuctra major	.00	.01	.18	.08	.00	.02	.18	.01	.20	.01	.00	.00	.00
total taxa	.01	.12	.10	.07	.00	.00	.00	.05	.00	.00	.01	.01	.01
‘small‘ taxa	.01	.14	.02	.02	.00	.00	.01	.04	.01	.00	.01	.02	.00
‘large‘ taxa	.00	.04	.17	.11	.00	.02	.03	.03	.04	.01	.00	.00	.00
stygophile taxa	.01	.02	.13	.05	.00	.07	.02	.02	.03	.00	.04	.00	.00
stygobite taxa	.03	.07	.05	.05	.01	.00	.00	.00	.00	.01	.00	.02	.04
not determ. taxa	.00	.10	.03	.02	.00	.00	.00	.07	.00	.00	.00	.00	.00
Shannon-Index	.00	.05	.12	.00	.01	.00	.00	.04	.00	.01	.01	.01	.01
Evenness	.00	.02	.09	.00	.00	.01	.00	.03	.00	.01	.04	.00	.00
total individuals	.02	.00	.00	.00	.02	.00	.00	.02	.00	.02	.00	.04	.02
‘small‘ individuals	.02	.00	.00	.00	.02	.00	.00	.02	.00	.01	.00	.02	.01
‘large‘ individuals	.00	.01	.08	.00	.00	.02	.04	.00	.03	.02	.01	.00	.01
stygophile individ.	.01	.00	.15	.02	.00	.03	.11	.03	.13	.01	.00	.00	.00
stygobite individ.	.01	.01	.05	.00	.01	.02	.01	.01	.01	.02	.00	.03	.02
not determ. indiv.	.01	.00	.00	.00	.02	.00	.00	.01	.00	.00	.00	.00	.00
Total *	1	1	6	2	0	2	1	0	0	0	0	0	0
Total **	0	3	3	2	0	0	1	0	2	0	1	0	0
Total ***	0	0	3	3	0	0	0	0	0	0	0	0	0
Total sig. (all Niv.)	1	4	12	7	0	2	2	0	2	0	1	0	0

Appendix 4 continued:

R^2	organic particle content			inorganic particle content			organic particles/total fine particles		
Faunal abundances	'small' organic particles	'large' organic particles	total organic particles	'small' inorganic particles	'large' inorganic particles	total inorganic particles	'small' organic particles to total fine particles	'large' organic particles to total fine particles	total organic particles to total fine particles
Dendrocoeleum	.00	.00	.00	.01	.00	.01	.00	.00	.00
Nematoda	.00	.01	.00	.00	.00	.00	.00	.01	.00
Planorbidae	.00	.00	.00	.00	.00	.00	.00	.00	.00
Oligochaeta	.01	.00	.01	.01	.01	.01	.01	.01	.01
Acari	.00	.04	.00	.00	.00	.00	.01	.00	.01
Niphargus	.01	.03	.01	.04	.03	.04	.01	.01	.01
Proasellus	.00	.01	.00	.00	.00	.00	.00	.00	.00
Cladocera	NA	.00	NA	NA	.00	NA	NA	NA	NA
Ostracoda	.00	.10	.00	.01	.00	.00	.00	.00	.00
Cyclopoida	.00	.33	.00	.01	.00	.01	.00	.13	.00
Harpacticoida	.00	.37	.00	.01	.00	.01	.00	.14	.00
Ephemeroptera	.00	NA	NA	.00	NA	NA	.00	NA	NA
Leuctra major	.01	.01	.01	.01	.10	.01	.00	.09	.00
total taxa	.00	.00	.00	.03	.00	.03	.01	.00	.01
'small' taxa	.00	.00	.00	.02	.01	.02	.00	.00	.00
'large' taxa	.02	.01	.02	.02	.03	.02	.01	.00	.02
stygophile taxa	.01	.02	.01	.03	.00	.03	.00	.03	.00
stygobite taxa	.01	.00	.00	.04	.05	.04	.02	.03	.03
not determ. taxa	.01	.00	.01	.12	.02	.11	.01	.00	.01
Shannon-Index	.00	.00	.00	.04	.00	.04	.02	.00	.03
Evenness	.01	.01	.01	.03	.01	.03	.03	.00	.06
total individuals	.00	.31	.00	.01	.00	.01	.00	.13	.00
'small' individuals	.00	.30	.00	.01	.00	.01	.00	.13	.00
'large' individuals	.00	.01	.00	.00	.04	.00	.02	.01	.02
stygophile individ.	.01	.01	.01	.02	.07	.02	.00	.06	.00
stygobite individ.	.00	.11	.00	.01	.00	.01	.00	.00	.00
not determ. indiv.	.00	.26	.00	.01	.00	.01	.00	.14	.00
Total *	0	0	0	0	1	1	0	0	0
Total **	0	2	0	1	1	0	0	5	0
Total ***	0	5	0	0	0	0	0	0	0
Total sig. (all Niv.)	0	7	0	1	2	1	0	5	0

List of abbrevations

km	kilometers
m^3	cubic meters
°C	degrees C
ml	milliliters
l	liters
µm	micrometers
mm	millimeters
<	smaller than
>	larger than

List of Tables and Figures

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