Project Number 5-1-3-F

Water Management Strategies against
Water Scarcity in the Alps

Work Package 7
Optimal Ecological Discharge

Dissemination title *Optimal ecological discharge*

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Interreg IV B, Alpine Space Programme

This project is co-funded by the European Regional Development Fund
under Priority 3 - Environment and Risk Prevention

Contents

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1. Optimal ecological discharge

1.1 Main objectives

This WP contributes to the project objectives by defining and applying hydrological and ecological indicators related to optimal ecological discharge under changing regimes imposed by management issues (such as hydropower production) and climate change. It also assesses the resistance and resiliance of ecosystems to water scarcity. Ecological effects on the ecosystem goods and services, as well as mitigation and adaptation strategies such as water re-use are to be suggested. Partners involved in this work package are:

PP1 France: Societé Economique Alpestre (SEA)

PP3 Austria: Amt der Kärntner Landesregierung, Abteilunge 15 und Abteilung 18 (KTN)

PP9 Italy: Provincia Autonoma di Trento – Dipartimento Urbanistica e Ambiente (ProvTn)

PP13 Slovenia: National Institut of Biology (NIB)

PPnonEU1 Switzerland: Swiss Federal Institute of Aquatic Science and Technology (EAWAG)

1.1.1 Definitions

Optimal Ecological Discharge

Optimal ecological discharge describes discharge pattern in terms of minimal base flow requirements and the timing, duration, magnitude and frequency of high flow and flood events most suitable for a creating sustainable habitat conditions for resident biota under different management strategies and climate change

Drought

For the definition of drought and its different types please refer to Hohenwallner et al. (2011). However, an additional type of drought needs to be introduced here: "Drought by hydropower production". Drought by hydropower production can be described as man made low flow conditions induced by exploiting the benefits of hydropower for the production of electricity.

Indicators

An indicator is 'a characteristic of the environment which, when measured, quantifies the magnitude of stress, habitat characteristics, degree of exposure to the stressor, or degree of ecological response to the exposure' and 'provides information on the system's condition'. Indicators serve as tools to assess, in a quantitative way, the condition of a river. When defining indicators according to these objectives, various indicator characteristics need to be considered. They include ecological and social relevance, ease of measurement and interpretation, and cost-effectiveness (Woolsey et al., 2007).

Indices

A numerical scale used to compare variables with one another or with some reference number

Literature cited

Hohenwallner et al. (2011): Water management in a changing environment: Project outcomes and recommendations. Alp Water Scarce Report.

Woolsey, S., T. Gonser, M. Hostmann, B. Junker, A.Paetzold, C. Roulier, S. Schweizer, S.D. Tiegs, K. Tockner, C. Weber and A. Peter. 2007. A strategy to assess river restoration projects. Freshwater Biology**.** 52: 752-769.

2. Overview of the Alp-Water-Scarce Pilot Sites corresponding to their activities

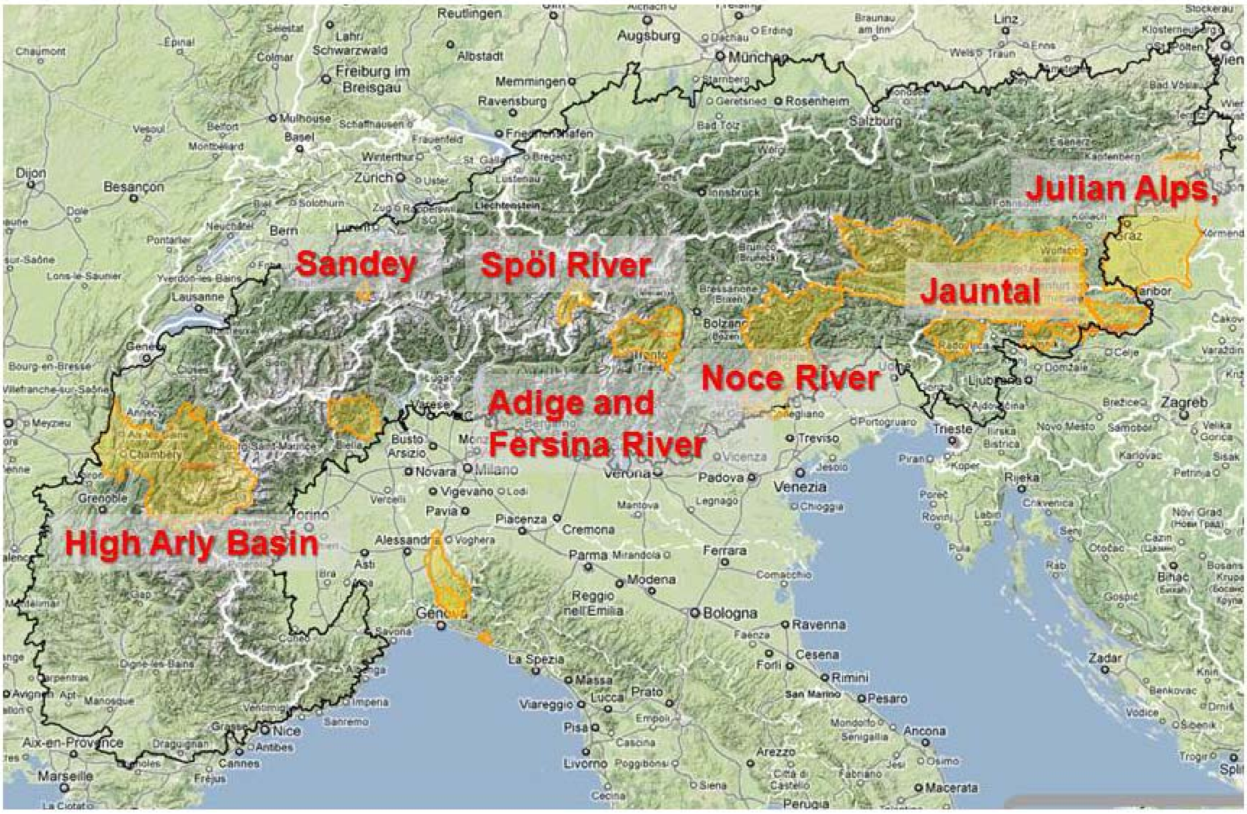

Figure 2: Topographic map of the Alps with the boundaries of the Pilot Sites

2.1 Overview

2.1.1 Spöl (Switzerland)

The Spöl is a canyon-confined river flowing through the Swiss National Park in southeastern Switzerland. The catchment climate is characterized by relatively low precipitation (937 mm/y) and high seasonal and daily variation in temperature (average yearly temperature: 1°C). Vegetation in the river valley is dominated by fir (*Picea abies*) and pine (*Pinus mugo*), whereas alder (*Alnus incana*) is common on river banks. Landuse is restricted in the National Park area. the river originates from the Livigno reservoir (Lago di Livigno) resulting from the Punt dal Gall dam (46°37′ 0′′N, 10°11′30′′E) on the Swiss-Italian border. The Spöl watershed covers 295 km^2. The elevation of Lago di Livigno is 1805 m a.s.l. Below the dam, the Spöl flows 5.5 km before entering the Ova Spin reservoir at an elevation of 1630 m a.s.l. From this reservoir, the Spöl flows another 5.5 km to its confluence with the Inn River, a major tributary to the Danube, in the lower Engadine at Zernez. Construction of the dam was completed in 1970 with a reservoir capacity of 1.64\108 m^3. The dam itself is a double-arch dam, 130 m high and 540 m wide. Most of the water from Livigno reservoir is transferred through a 7.6 km pressure tunnel to the power plant below the Ova Spin reservoir. As a consequence, the Spöl's discharge was reduced from 6–12 m^3/s (peak flows up to 120 m^3/s) before regulation to a constant residual flow of 1.45 m^3/s in summer and 0.55 m^3/s in winter. Since 2000 this constant residual flow is interrupted by 1-3 artificial floods each year to test the potential for water reuse with respect to optimal ecological discharge rather than describing water scarcity in this area.

2.1.2 Sandey (Switzerland)

The Sandey floodplain is the lower floodplain of the Urbach River (3.4 km long, up to 600 m wide, 790-910 m a.s.l.) situated in Canton Bern. The catchment climate is characterized by moderate precipitation (1345 mm/y) and high seasonal and daily variation in temperature (average yearly temperature: 8.8°C). A steep face on the left side of the river confines the valley to the east, whereas the right side of the floodplain opens up into grassland area, bordered by mountain slopes. The floodplain is composed of different habitat patches including the main channel, side channels, islands, open gravel bars, vegetated gravels with successive shrub vegetation, alluvial forest and grassland. In the active floodplain area vegetation is dominated by alder (*alnus incana*) and different willow species (*Salix spec*.). Land use is limited to cattle and sheep growing.
Discharge is driven by glacial melt water and precipitation events. Since 1950, the river has been dammed (5 km upstream of the study area) and about 30% of the average annual discharge is abstracted from the system. Furthermore several levees have been installed throughout the years in the active floodplain area for flood protection.

2.1.3 The Julian Alps, Karavanke and Kamniško-Savinjske Alps (Slovenia)

The Julian Alps pilot site lies in the north-west part of Slovenia on the border with Italy. The catchment area of the pilot site covers approximately 1600 km² shared in the Soča River catchment (772 km²) and the Sava River catchment (818 km²). Approximately half of the area is within Triglav National Park which has been designated as conservation area since 1924. The highest point above sea level is the mountain Triglav (2864 m) and the lowest point is the confluence of rivers Tolminka and Soča (180 m). The area is mainly formed of Triassic and Jurassic carbonate rocks, limestones and dolomites forming a high-mountain karstic plateau. Some Pleistocene sediments and Holocene gravels can also be found. The area has small mostly karstic aquifers with fast vertical percolation. There is a well-developed network of mostly vertical and some horizontal channels.

The Karavanke mountain lies in the north of Slovenia across the border with Austria. With a total length of 120 km, the Karavanke chain is one of the longest ranges in Europe. The highest peak in this mountain range is the mountain Stol (2236 m). It lies on the Periadriatic seam which forms the division between the Adriatic plate and the European plate. Due to the collision of the two tectonic plates the sediment deposits in this area formed fractures and faults in east-west direction. Consequently Karavanke are geologically extremely complex with a mixture of paleozoic rock and mezozoic, oligocen and miocen sediments. Closely connected with complex geology are also very diverse types of aquifers and springs in this area.

Kamniško-Savinjske Alps are located east and south of the Karavanke and are extending over the area of 900 m^2. This mountain range is steep, with deep and narrow, glacier-formed river valleys and the highest peak of 2558 m a.s.l (Grintovec). The geological composition of the area are mainly Triassic and Jurassic carbonate rocks (limestone). The waters from this mountain range drain into two rivers: Kamniška Bistrica and Savinja, which are the part of the Sava River catchment. The aquifers in this area are mostly karstic and porous, with permanent springs emerging at the bottoms of the valleys. The climate over the pilot area is mountainous, with a mean annual temperature of 8 °C and average annual rainfall from 1600 to 3000 mm.

2.1.4 Jauntal (Austria)

Jauntal is the area between the mouthing of the River Vellach into the River Drau and the village Schwabegg respectively the homonymous dam. The catchment climate is characterized by moderate precipitation (1048 mm/y) and high seasonal and daily variation in temperature (average yearly temperature: 7,5°C). The altitude of the pilot site Jauntal is between 360 and 630 m a.s.l.. The average discharge is estimated by 1400 mm and the minimal flow is 900 mm. The area is dominated by subsurface flow in porous aquifer, rivers and some lakes used for bathing and fishing. The main economic sector is agriculture and tourism, therefore the water is mainly used for drinking and irrigation and less for cooling. The water of dam Schwabegg of the River Drau produces electricity. Five gauging station are situated in this area, where drinking water protection zones are exist. The number of inhabitants of the area is about 20.000. Several springs of the Jauntal were investigated by qualitative and quantitative investigation of benthic organism.

2.1.5 Noce (Italy)

Section 1

The Noce River is a fourth order Alpine stream located in the Province of Trento, in north-eastern Italy. The total river length is 105 km and it drains a typical alpine catchment with an area of 1,370 km^2. River basin elevations range from the confluence with the Adige River (198 m asl) up to the peak of Cevedale (3769 m asl). Major tributaries are the Vermigliana, the Rabbies and the Novella creeks. Dominant bed surface grain size ranges from coarse gravel to cobble with coarse sand matrix. In the Noce River basin, hydropower is generated by five plants fed by four artificial reservoirs closed by dams.
The study focuses on two reaches of River Noce, both located in the western part of the basin: the first is upstream from the confluence with the Noce Bianco in the main watercourse (Noce *Pejo-Cogolo*) and the second is about 2 km upstream from the confluence with the Vermigliana stream (Noce *Cusiano*).
For each of these reaches there is one official monitoring point of Trento's A.P.P.A. (Provincial Environmental Agency); at these points, both chemical and biological parameters are determined, at different intervals during the year, in accordance with national legislation.
The Noce *Pejo-Cogolo* reach extends over a length of 200 m with a slope of about 4%. It is the final reach of the branch of the river Noce that rises on Corno dei Tre Signori, chosen for this study because, of all the watercourses with a monitoring point, it seems to be the one where human activities have the least impact. Although this branch is subject to considerable water diversions, it is not affected by hydropeaking and its final reach, on which this analysis is based, is fed by the downflow generated in the interbasin between it and the upstream dam of Pian Palù.
The Noce *Cusiano* reach is part of the main watercourse near the entrance to the Fucine-Ossana-Cusiano floodplain, in which the Vermigliana stream flows into the river Noce. It is located about 5 km downstream from the Noce *Pejo-Cogolo* reach and is affected by intense hydropeaking phenomena caused by the release of turbinated water from the hydroelectric plant of Cogolo. This hydroelectric plant is situated on the river Noce Bianco, about 6 km upstream from the reach selected for this study.

Section 2

The present part of the study is mainly restricted to the area downstream the lowest-elevation plant, near the village of Mezzocorona, where water stored in Santa Giustina reservoir (182,106 m^3 capacity) first passes through the Mollaro reservoir (106 m^3 capacity), and then feeds the Mezzocorona power plant that releases a maximum discharge of 60 m^3/s into the Noce River.

2.1.6 Adige river (Rotaliana plain, Italy)

Mezzocorona power plant is surrounded by the Rotaliana plain, a 30 km^2 agricultural area confined between the Noce and the Adige River. The valley bottom elevation ranges between 198 and 210 m a.s.l., it is intensively cultivated with apple trees and grape vines. This highly valuable agricultural area is crossed by an 130 km ditch network of artificial channels built in the past century to drain the area. Nowadays, the ditch network is used only occasionally for drainage and irrigation, but still requires regular maintenance and suffers from seasonal water scarcity with associated low water quality.

2.1.7 Fersina River and experimental flumes (Italy)

The Fersina stream originates from the Erdemolo Lake in the Mocheni Valley at an altitude of 2005 m, and joins the Adige River at Trento (191 m a.s.l.), it is a snowmelt-fed gravel-bed stream, approximately 14 Km long, with a 171 Km^2 watershed.
The experimental setting (Figure 2.1.7.1) is represented by five 20 m long, 30 cm wide metal flumes, located on the riparian area of the Fersina stream, at the entrance to the Mocheni Valley at 577 m a.s.l.. Flumes A, B, C are 30 cm high; flumes D and E are 50 cm high. The flumes have adjustable longitudinal slope and feeding discharge, they are connected to a loading tank which is directly fed by water diverted from the channel. The flumes were filled with a 10 cm (flumes A, B, C,) or 30 cm (flumes D and E) layer of gravel and sand collected from the riverbanks. The flumes can be set with different discharges and water velocity by 5 sluice-gates.

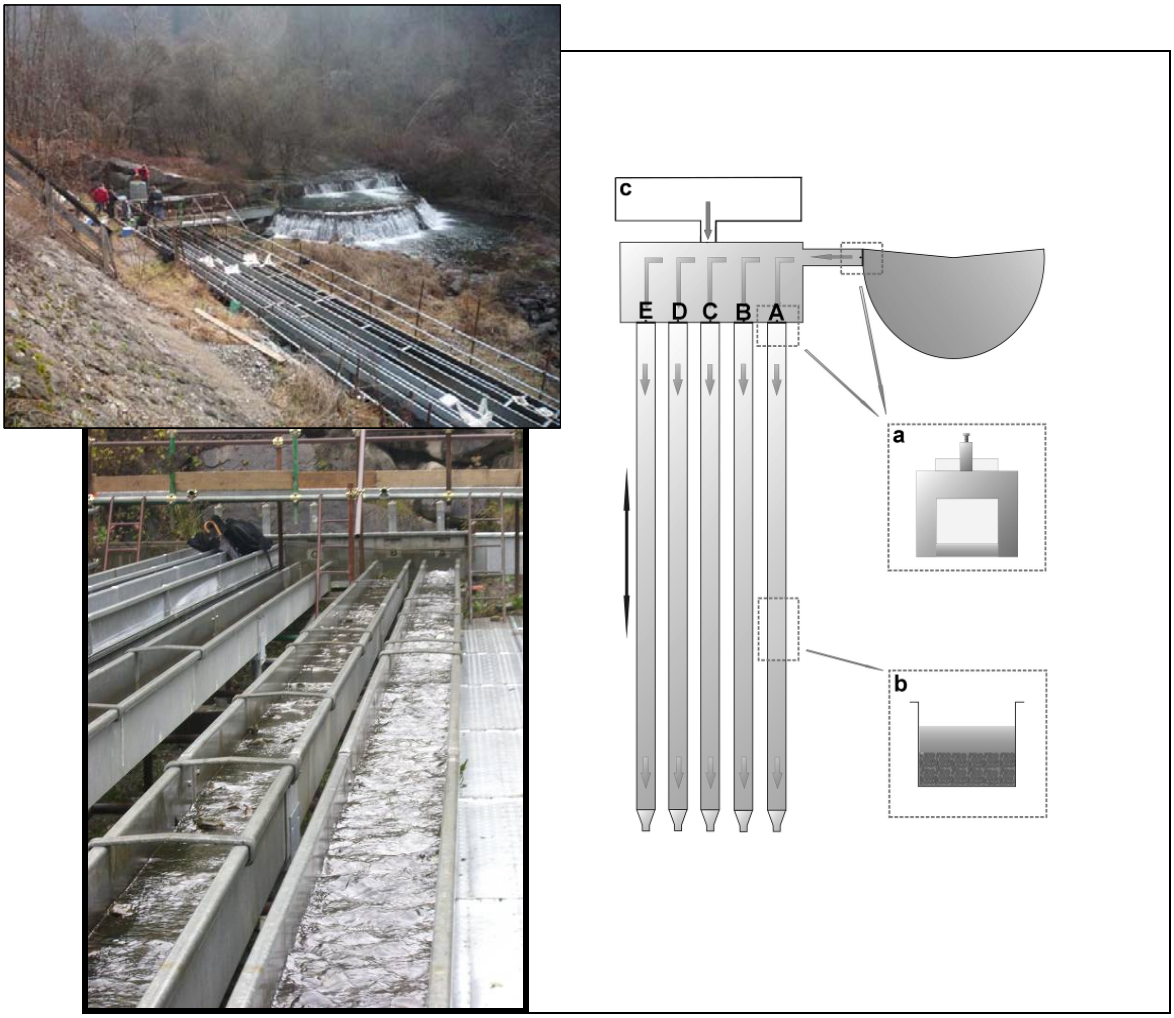

Figure 2.7.1.1: Photos and schematic overview of the Fersina Flumes

2.1.8 High Arly Basin (France)

In the North French Alps (department of Haute-Savoie, next to the town of Megève), this pilot site is situate in high French alpine region, the "High Arly Basin" into the Arly valley.
High Arly Basin pilot site is one of the smallest site of Alp-Water-Scarce project (47 Km²). It is characterized by one watershed well bounded and with a simple hydrological functioning and it benefits of a plentiful resource, approximately 1500-1800mm of water precipitations by year. Influenced by the west winds which bring generally the precipitation, the pilot site knows different characteristics of a mountain climate.

The basin's top is characterized by a torrential functioning. Situated between 2000m at 2525m, this part has many torrents running on rock face. On the other hand, the outside of the basin finishes on two streams running through the town of Megève. Between this two parts, one intermediate area has some little streams and one water table.

Climatology especially hydrologic data is collected by one measurement devices network realized for the Alp-Water-Scarce. Other datas are collected to characterize the High Arly Basin pilot site with local actors, to evaluate takings and pressure on water resource by water's users. To complete the hydrological functioning site knowledge, it's necessary to enlist some data into the computing model (geological, urbanisation, land use, slopes, flows direction, flows network and groundwater, wet zone...).

This data give us the soil permeability's, capacity of infiltration, the slopes impact...

Drinking water, agriculture and tourism activities are the mains uses of water on High Arly Basin. Even if the precipitations are important on the site, some critical times concerning water resource can appear. Indeed, during winter the population increase and the need of water's for artificial snow or water drinking supply are more important. In the same time, the streams are in low water period and aquatic life is weaken.

2.2 Main thematic focus of the different Pilot Sites

2.2.1 Spöl

The primary goal of this study was to test whether implementing a novel disturbance regime through experimental floods would cause a regime shift in ecosystem properties of a flow-regulated river, where the flow regime has been relatively constant for over 30 years. We predicted that ecosystem properties would change in response to the new habitat template of the river that resulted from a more variable flow regime. We evaluated this prediction by testing different population-, community-, and ecosystem-level hypotheses. We expected the floods to have little effect on the physicochemistry of the river because the water source (i.e., hypolimnetic release from the dam) would be the same as before the floods. We hypothesized that the floods would reduce standing stocks of primary producers and eliminate attached moss on bed sediments. The floods should scour bed sediments and dislodge the moss within the first or second year. The pre-flood stream bed was highly armored and the floods should reduce armoring and increase the porosity of bed sediments. Although the study river is nutrient rich, the floods should maintain low periphyton biomass by scouring filamentous algae from bed sediments. We hypothesized that the experimental floods would reduce benthic and transported organic matter in the river. Benthic organic matter should decrease because the bed sediments would be mobilized by the floods and benthic particulates would be flushed from the system. Seston (particulate organic matter) levels should decrease because the floods would reduce standing stocks of benthic organic matter and periphyton that are the primary instream sources of seston.

We predicted that the floods would reduce benthic macroinvertebrate density, biomass, and taxon richness, and result in higher proportions of smaller sized organisms. We hypothesized that disturbance-prone taxa such as large-bodied sessile taxa (e.g., Gammarus fossarum) would decrease in abundance and disturbance-resistant taxa such as small-bodied, highly mobile taxa (e.g., Baetis sp.) would increase in abundance from the floods. The shift in organism size was expected because larger organisms are more associated with taxa that inhabit relatively stable flowing systems, such as before the floods. These taxa should be replaced by more disturbance-resistant taxa that tend to be smaller in size. Lastly, we hypothesized that measured properties would initially become more variable and then less variable as the ecosystem regime shift occurred. The regime shift should be observed by a change in mean values, along with an increase in variation (as coeficients of variation, CV) during the shift. We expected the variation to decrease after the shift, although it would still be higher than before the shift because of the immediate flood effects that cause reductions in organism density and biomass. The regime shift would be related to the changes in the composition of benthic macroinvertebrates, as referred to previously. We also predicted that initial floods would have a greater impact than later floods of similar magnitude because of the regime shift in ecosystem properties and changes in macroinvertebrate composition.

2.2.2 Sandey

In their natural state, floodplains, i.e. the entire channel network and valley bottom area susceptible to flooding, are exceptionally heterogeneous and dynamic ecosystems composed of different aquatic (channels, pools, backwaters, side-arms) and terrestrial (riparian forests, islands, gravel bars) habitats. This complex and heterogeneous array of habitat types undergoes distinct cycles of expansion, contraction, and fragmentation along longitudinal, lateral, and vertical dimensions, primarily as a result of variations in flow. Flow is the key driver regulating ecosystem processes and biodiversity in natural floodplains that transcends multiple temporal and spatial scales. Extremes in flow range from floods exceeding bankfull discharge to low flows interrupted by frequent instream water level fluctuations ("flow pulses") to long-term drought. The magnitude, frequency, duration and timing of flow events structure and maintain habitat heterogeneity, spatial configuration and connectivity, and cause channel migration and turnover in riverine floodplains.

Furthermore these different habitat types differ widely in their habitat properties such as soil texture, temperature and organic matter content, controlling biotic and abiotic processes such as, for example, respiration and nutrient cycling. For example, exposed gravel sites can be described as harsh habitats characterized by extreme temperature variation, high water stress and low productivity. In contrast, riparian forest provides more stable conditions and is rich in resources that sustain high productivity. This habitat diversity and hyporheic exchange results in a high environmental heterogeneity (i.e. the variance in patterns and processes over space and time) Therefore, floodplains may serve as ideal model ecosystems to study the effects of environmental heterogeneity through different habitat properties on ecosystem processes.

The first goal of this study was to quantify the spatio-temporal transformation of floodplain habitats in a Swiss floodplain of national importance from its near-natural state in 1940 until 2007. During this time period, the floodplain was subject to several hydrological and morphological impairments via dam construction and water abstraction in the upper catchment and the installation of several levees for flood protection in the active floodplain. To achieve our goal, we analyzed a 67-year time series of aerial images taken before and after human impairments for changes in the composition and abundance of predominant habitat types (i.e., mature forest, gravel, vegetated gravel, islands, water, pasture and grassland) and in channel complexity within the floodplain. Despite their inherent dynamic nature, the coarse composition (i.e., number of habitat types) and abundance (i.e., the relative proportion of different habitat types to total floodplain area) of habitat elements in natural floodplains seem to remain relatively constant over ecological time periods. This spatio-temporal phenomenon is described as the "shifting mosaic steady state" and is a fundamental process attribute of unregulated river ecosystems. Our working hypothesis was that flow regulation and water abstraction has decoupled the interactive linkage between the natural flow regime and the formation of floodplain habitats inherent to naturally functioning floodplains as posited by the "shifting mosaic steady state" concept. We discuss the application of the concept originally developed for near-natural floodplains as a useful framework to indicate changes in floodplain heterogeneity resulting from human-modified flow regimes or landscape transformation via climate change.

The second goal of this study was to investigate the different terrestrial and aquatic habitats for structural and functional relationships. We measured abiotic parameters, soil and sediment respiration (i. e. the production of CO_2 by autotrophic root respiration, heterotrophic microbial respiration and the respiration of other soil and sediment organisms), and the amount of bacteria and enzyme activities in four terrestrial (gravel, island, alluvial forest and meadow) and one aquatic habitat (channel). The following research goals were addressed: (1) Quantify spatio-temporal variability in different habitat types along the floodplain for soil and sediment respiration, the amount of bacteria and enzyme activities; (2) Determine the main environmental drivers of soil and sediment respiration and enzyme activities; (3) Investigate the relationships between soil and sediment respiration, bacterial abundance and enzyme activities; and (4) Differentiate the large-scale variability of soil and sediment respiration, bacterial abundance and enzyme activities among and within the different habitats of the floodplain. These goals were addressed to improve the holistic understanding of the interaction between habitat heterogeneity and processes in floodplains. Further it was discussed if structural and functional relationships can serve as a framework to assess and monitor changes in floodplain ecosystems in terms of sustainable resource management or increasing environmental pressures such as from climate change or hydropower production.

2.2.3 The Julian Alps, Karavanke and Kamniško-Savinjske Alps

Springs are the areas where groundwater discharge from the aquifer. Its discharge depends on the geology of the aquifer, vegetation cover, precipitation patterns and water use. In order to carry out sufficient water management it is important to know how decrease and changes in precipitation regime will affect groundwater recharge and consequently water chemistry, aquatic biology and drinking water supply.

Aquifers in the area studied are located mostly in carbonates, where the rock is densely fractured; water flows relatively fast through the aquifer and spring discharge respond relatively quickly to high rain or drought. Until this study, little was known about the physical, chemical and biological characteristics of the springs and ground waters in Slovenian Alps. A comprehensive approach, where geology, hydrology, and geochemistry of alpine aquifers were studied at the same time, was needed to satisfyingly interpret the biological data collected. Therefore the objectives of this study were to characterize hydrogeological features of selected aquifers, their typical groundwater and spring invertebrate communities and search for simple biological measures that respond to changes in the precipitation rates and spring discharge over seasons. Our hypothesis was that groundwater and spring invertebrate communities would differ between hydrogeologicaly different aquifers (i.e. retention times of groundwater, rock type, intermittent or permanent spring discharge) and that during periods of drought (low discharge) different assemblages of invertebrates are going to drift from groundwaters as well as inhabiting spring habitat than during high discharge. We predicted that those assemblages would differ in total abundance, species composition, relative composition of higher taxa, proportion between surface and groundwater taxa and selected species traits (i.e. body size, development rates). By testing a variety of biological measures, it would be possible to develop simple biological indicators to detect ecological stress in groundwater and spring ecosystems due to longer periods of decreased water levels.

A group of twelve springs was sampled several times in 2009 and 2010 for the most important physical, chemical and biological parameters. In the field, flow velocity and water levels were measured at each spring and on each sampling occasion. Further, the conductivity, temperature and oxygen concentrations were measured in the spring water. Temperature loggers were installed in each spring to continuously measure T over the two years. In the laboratory, water was analyzed for anions, cations, $\delta^{13}C_{DIC}$, $\delta\ ^{13}C_{POC}$, $\delta\ ^{18}O$, δ D, and tritium in order to investigate the intensity of interaction between groundwater and the surrounding rock (aquifer) and to study the complex pathways of carbon (i.e. the intensity of the organic matter and nutrient input into the aquifer – the vulnerability). Water recharging from the aquifer was filtered through the drift net (mesh size 60 µm). The groundwater invertebrates were collected by putting drift nets into the spring mouths for 24 hours. The invertebrate densities were expressed as number of individuals per 100 m^3 of filtered water. Invertebrates from the spring benthos were sampled by kick sampling and using hand net (mesh size 100 µm).

2.2.4 Jauntal

Beneath the climate induced changes of water household in alpine regions there are man-made aspects that induce drought effects on biota by withdrawal of water e.g. drinking water, irrigation, hydropower, cooling, snowing, industrial use, etc. From the first view the amount, availability and quantity of water in most of the alpine areas may look abundant and not well capitalized. However in some areas the increasing pressure on groundwater resources for diverse technical water supply causing drastic changes, which leads to reduction of ground water level and consequently to a reduced discharge. The spring waters and headwater stream network often get captured and diverted before refuelling groundwater systems. Knowing that streams and rivers affected by water abstraction (e.g. hydropower) sustainable effects the biocenosis of rivers less is known about spring habitat.

Springs and headwater streams are very special habitats that mirror the retention capacity of waters in catchments.

The predicted reduction of water availability in future will influence the discharge variability. This is likely to favour some organisms which are tolerant against less discharge. A typical spring flora and fauna is usually made up by organisms which require a high degree of environmental stability.

The aim of investigations on headwaters in this area is to answer the question if species composition tells about the amount of water discharge. It is an attempt to identify benthic organism as biological indicator which correspond with the amount of discharge, minimal, periodical or permanent.

As it is for the first time that benthic organisms of headwaters get investigated in Carinthia, a careful selection of appropriate headwaters in the pilot site was done. The hypothesis we formulated is:

- Head waters, respectively spring types (porous or fissure) are characterized by a different species composition due to the amount of discharge.
- This should help to identify an indicator (species composition) that gives information about permanent or intermittent discharges.

In the framework of the project Alp-Water-Scarce the Carinthian Institute for Lake Research were commissioned by the Office of the Carinthian government, Dept. 8 – Environment, Water and Nature Protection to conduct appropriate investigations on the headwaters of the pilot site Jauntal. Several springs of the Jauntal were investigated by qualitative and quantitative investigation of benthic organism.

2.2.5 Noce River

Section 1

The aim of this study is to analyse the differences between two reaches of the river Noce, differently impacted by human activities, in order to show both biological changes and morphological changes caused in particular by hydropeaking.
The study is divided into two parts: the first, and main one, focuses on the analysis of the biological communities found at the two monitoring points of Noce *Pejo-Cogolo* (not subject to hydropeaking) and Noce *Cusiano* (subject to hydropeaking); the second concerns the morphological aspects of the two reaches of the watercourse.
The aim was to see if there were any significant differences between the biological communities characterising the two reaches in terms of the variety and diversity of the taxa present. The Rosgen morphological index was subsequently applied to these two reaches in order to ascertain whether they belonged to different morphological types.
In this respect, it must immediately be underlined that the morphological analysis presented in this study does not pretend to be exhaustive, since it is the first attempt to apply the index to the watercourse in question. Furthermore, two reaches of limited extension were purposely selected for analysis in order to capture their characteristics in the immediate vicinity of the monitoring points for which biological data are available.
For the Noce *Pejo-Cogolo* monitoring point it was also possible to conduct a study on the development of the macrobenthic population over the last twelve years (1998 - 2010), with the aim of determining whether this biological evolution could be correlated to climate change. As will be explained in more detail below, however, in this case there were no significant differences over the last decade, an indication that the reach in question has not yet been affected by changes in external forces.

Section 2

The lower course of Noce River is a particularly interesting site for its hydromorphological setting. It is subject to heavy hydro- and thermo-peaking and composed by three reaches clearly displaying different channel morphologies: two single-thread channelized segments at the upstream and downstream end connected by an island-braided multiple-thread segment with dense riparian vegetation on the island surfaces. The present study has focussed on a quantitative statistical field analysis of the hydro- and thermopeaking events and of their propagation dynamics. Water temperature and level have been continuously monitored at 2 minutes intervals for nearly the whole project duration (~ 2.5 years).

2.2.6 Adige river (Rotaliana plain)

In Trentino water resources are mainly used for agriculture and hydropower, with about 85% of non-consumptive uses being for energy production (Autonomous Province of Trento, 2006). Although a pulsating hydropower production is economically very rewarding, it leads to substantial alterations in stream velocity, streambed stability, water temperature and turbidity and reduces riverine habitat diversity. In the lowland zone, streams also suffer from water scarcity, because in mountain areas the valley bottoms have been drained in order to devote land to agriculture, causing the disappearing of lowland freshwater habitats.
In this context, we have tested the quantitative benefits that can be expected from implementation of water management policies promoting different reuses of water. In a global change scenario that foresees a decrease or change of timing of precipitation in the Alps and an increase in water demands, it is paramount to look for water reuse options able to enhance the ecological processes in natural and artificial freshwater ecosystems.
The Rotaliana Plain is a 30 km^2 agricultural area crossed by a 130 km network of agricultural ditches, not used anymore for irrigation purposes, but maintained for flood protection. Water withdrawals for irrigation of the high-quality crops are presently allowed only from groundwater because of minimum flow requirements imposed on the nearby Noce River, from which no withdrawal can be allowed. As a result, the Rotaliana ditches are basically fed by small local springs in the floodplain, which results in reduced summer streamflow. Close to the upstream end of the ditch network, the Mezzocorona power plant intermittently releases 60 m^3/s causing severe hydropeaking in the Noce River. A small part of the hydropeaking water could be diverted into the ditch network without affecting minimum flow, resulting in intermittently increased discharge into the network. Such reuse of water is expected to achieve the following ecosystem benefits which need quantification: 1) aquifer recharge, which can boost vertical connectivity and organic matter cycling (C, N), and increase in water availability for irrigation; 2) restoration and creation of aquatic and riparian habitats which will sustain an adequate biodiversity of aquatic and riparian fauna and flora; 3) improved water quality through increased dilution; 4) easier ditch maintenance (performed by Drainage Boards) thanks to the reduction of aquatic vegetation growth.
During summer 2009 and 2010 the ditch network was investigated in order to characterize the present hydraulic and ecological condition of the channels.
The network was characterized by integrating historical with recently collected topography data, together with information on ditches cover for roughness estimation. The network hydrodynamic response to the upstream input of a hydropeaking wave were simulated through 1D unsteady flow model supported by local hydraulic measurements.
Twenty-seven (27) sampling sites, representative of all the ditch typologies present in the area were identified and chemical-physical parameters such as conductibility, dissolved oxygen and pH, temperature were measured continuously with data loggers. The macrophytes and benthic community were also investigated during the two periods, to and evaluate biodiversity in the area and to use them as bioindicators. Results show a large diversity both for the chemical-physical parameters, with spatio-temporal fluctuations of temperature, flow and conductivity, and for the biological communities. Channels ranged from standing water habitats, with high water temperature and pleustophytic plant communities, to running water ones, with gravel beds and rich benthic populations. The overall results highlight a wide spatio-temporal mosaic of different freshwater habitats, in a relatively restricted area. This reinforces the possible use of these areas to mimic the ecological role of the lost natural Alpine wetlands.
A possible approach to remediate seasonal water scarcity is to connect the extended ditch network to the water discharged by the power plant, thus preserving minimum flow requirements in the river channel, at the same time using part of the released water to dilute pollutants, increase habitat diversity and connectivity, as well as aquatic/terrestrial transitional habitats. Final the aim is to ensure basic ecological processes and sustain an improved biodiversity.
Quantification of the expected benefits was preliminarily achieved through an integrated ecohydraulic approach that combines field investigations and mathematical modelling of ditches – floodplain interactions.

2.2.7 Fersina River and experimental flumes

In most of Alpine streams the alterations of flow regimes are due to an excess (hydropeaking) or lack (Minimum Vital Flow) of variations, with different impacts on diversity and abundance of zoobenthos. The goal of this part of the study was to assess the effects of alterations of the hydrological regime on the same zoobenthic communities, in a pristine Alpine steam, the Fersina River (natural flow), in a set of artificial flumes directly fed by the same stream (constant flow) and in an hydropower impacted reach of the same stream, 200 meters downstream of the flumes (Figure below). The Fersina flumes had been naturally colonized and were kept at constant discharge, simulating the Minimum Vital Flow. Five stations were selected in the natural flow reach of the stream, five in the hydropeaking-impacted reach and five in the flumes (i.e. one for each of the five flumes). Each station was sampled biweekly from mid-February to the end of July 2010 using a Hess bottom sampler and Hester-Dandy artificial substrates. Sampling ended in August, due to damage caused by an exceptional flood.

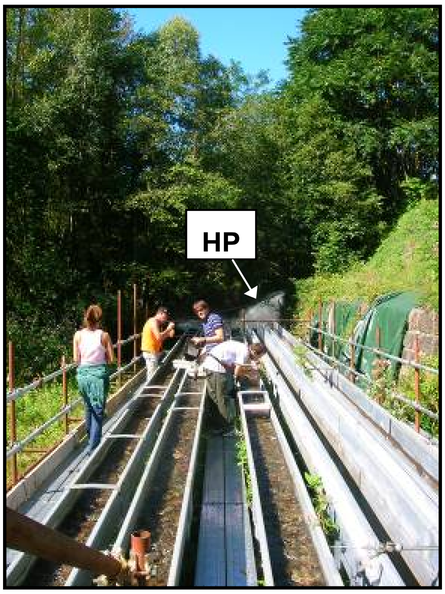

Figure 2.2.7.1. Experimental setting at the Fersina flumes. NFR: Natural Flow Regime site; MVF: Minimum Vital Flow site; HP: hydropeaking site.

2.2.8 High Arly Basin

The main start thematic concerned the biological adaptation at the winter low water’s flow.

Moreover, we wanted observe the aquatic life condition influenced by water taking in streams water, especially concerning water drinking taking and water use for product artificial snow. We try to compare the life in different streams during the winter and summer low water. So, we have realized different sampling in our pilot site stream’s. Physics and chemistry parameters are collected as well as biological data.

3. Indicators and indices considered in pilot sites

For a comprehensive overview of indicators and indices generated in the following studies, please refer to table 3.1 and 3.2

3.1 Spöl

Summary

The Spöl flood project in the Swiss National Park has documented that a flow regime management action can be used to improve the ecological integrity of a flow regulated river. The study further shows that a long-term perspective is necessary as these flow regulated systems often have multiple stressors influencing their ecology. The Spöl river, for example, is fragmented by upstream and downstream reservoirs that constrain the dispersal of aquatic invertebrates as well as influencing the fishery, thus affecting recovery rates in the river. Regardless, the overall ecology of the river has been improved by the adaptive management of the flow regime that integrates ecology with the socio-economic needs of the region.

Detailed investigation

3.1.1 Invertebrates

An NMDS analysis based on taxon densities and taxon relative abundances revealed significant temporal shifts in community assembly (Fig. 3.1.1.1). Based on taxon densities, assemblage composition was similar in 1999 and 2000, then a shift occurred in 2001 with assemblages being grouped from 2002 to 2006. Another shift occurred in 2007 with assemblages grouping together from 2007 to 2010 (stress = 13.6). NMDS from the relative abundance data showed a similar pattern with assemblages grouped in 1999-2000, 2001-2002, 2003 to 2006, and 2007 to 2010 (Fig. stress = 11.5). These data indicate an early period of transition in community assembly between 2001-2003 and a later shift between 2006-2007.

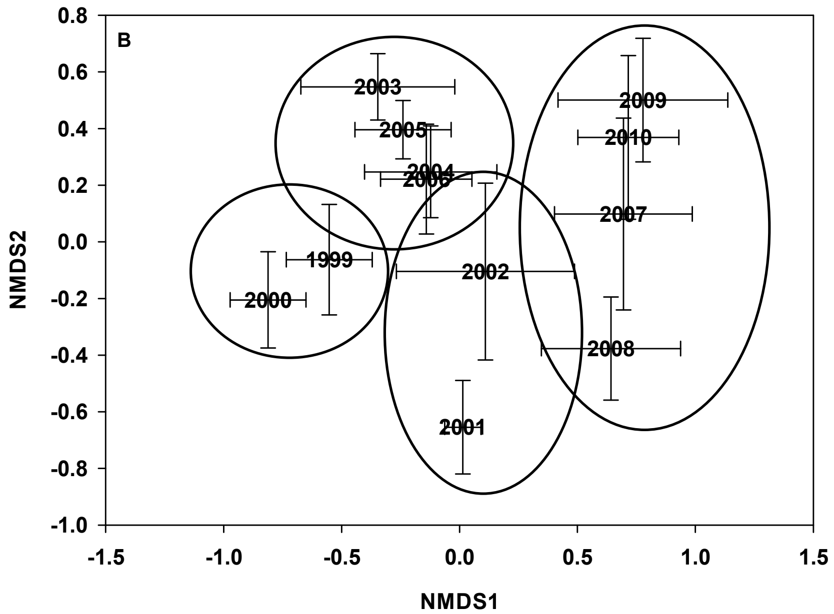

Figure 3.3.1.1: An NMDS analysis based on the relative abundances of benthic macroinvertebrates in the River Spöl. The plot illustrates the temporal changes in assemblage structure over the course of the experimental flood program.

Table 3.1: The AlpWaterScace pilot sites and indicators investigated (see chapter 2 for a detailed description)

Site	Hydraulic conditions	Hydrochemistry	Temperature	Invertebrates	Macrophytes and riparian vegetation	Periphyton	Respiration	Habitat change	Microbial
Springs									
Jauntal		x	x	x		x			
Julian Alps		x	x	x					
Karavanke Alps		x	x	x					
Kamniške Alps		x	x	x					
Rivers									
Arly Basin				x					
Noce	x	x	x	x					
Floodplains									
Sandey							x	x	x
Adige	x		x	x	x				
Experimental Systems									
Fersina Flumes			x	x					
Spöl				x		x			

Table 3.2: The AlpWaterScace pilot sites and indices investigated (see chapter 2 for a detailed description)

Site	Hydraulic conditions	Hydrochemistry	Temperature	Invertebrates	Macrophytes and riparian vegetation	Periphyton	Respiration	Habitat change	Microbial
Springs									
Jauntal									
Julian Alps									
Karavanke Alps									
Kamniške Alps									
Rivers									
Arly Basin		x		x					
Noce				x				x	
Floodplains									
Sandey									
Adige					x				
Experimental Systems									
Fersina Flumes									
Spöl									

3.1.2 Periphyton and benthic organic matter

Moss covered most stones in the Spöl before the flood program with an estimated biomass of 137 $\pm$ 286 g AFDM/m^2. By the third flood in 2000 no moss was observed on stones. The floods reduced the biomass of periphyton in the river (AFDM: $F_{1,67}$ = 19.09, P = 0.0001; chlorophyll-a: $F_{1,68}$ = 35.55, P = 0.0001) (Figure 3.1.2.1). Periphyton biomass reached maximum values between floods in 2001 and 2003 (e.g., 90 g/m^2 AFDM, 400-500 mg/m^2 chlorophyll-a), before decreasing to relatively low values after 2004 (average = 20 g/m^2 AFDM, 27 mg/m^2 chlorophyll-a). Average pre-flood values in 1999 were 29 g/m^2 AFDM and 65 mg/m^2 chlorophyll-a. The floods reduced the amount of benthic organic matter in the river ($F_{1,64}$ = 2.96, P = 0.0001) (Figure 3.1.2.1). BOM reached pre-flood levels (average = 16.8 g/m^2) between floods in 2002 and 2003 before maintaining lower levels after 2004 (average = 4.4 g/m^2).

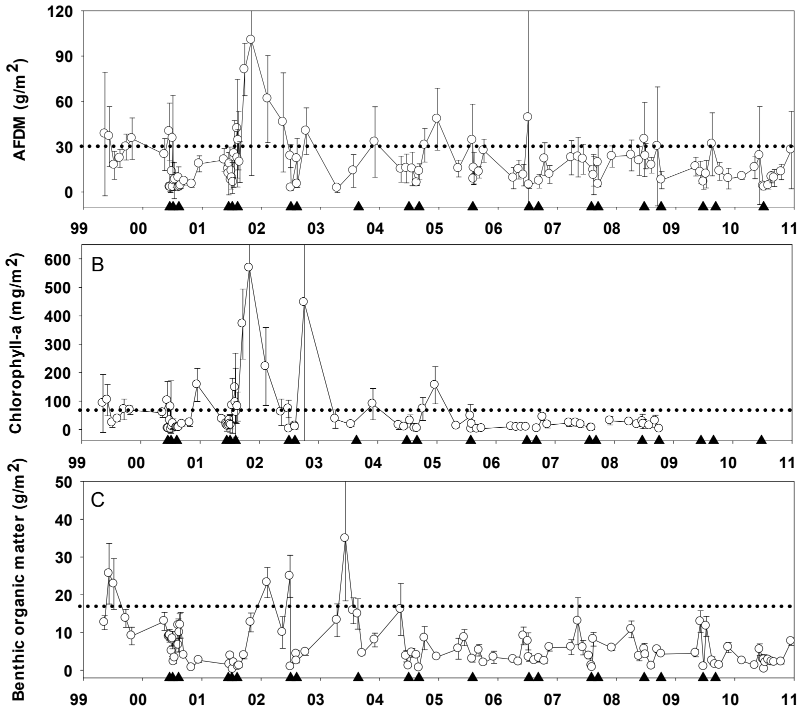

Figure 3.1.2.1: Average (+SD) of periphyton AFDM, chlorphyll a, and benthic organic matter collected in the Spöl River at periodic occasions during the flood program. The triangles represent the times when an experimental flood occurred in the river.

3.1.3 Coefficients of Variation of biotic measures

In all comparisons (density, periphyton biomass, benthic organic matter), the CV (coefficient of variation) increased 2-5 times from the PRE period (pre-2000) to the MID period (2000-2003) (Figure 3.1.3.1). The CV then decreased for 7 of the 9 measures from the MID period to POST period (here, 2004-2006). Sample CVs remained similar between MID and POST for macroinvertebrate density and taxon richness. For five of nine measures, the POST sample CV was intermediate to the PRE and MID CVs. PRE and POST CVs were similar for individual biomass and seston chlorophyll-a.

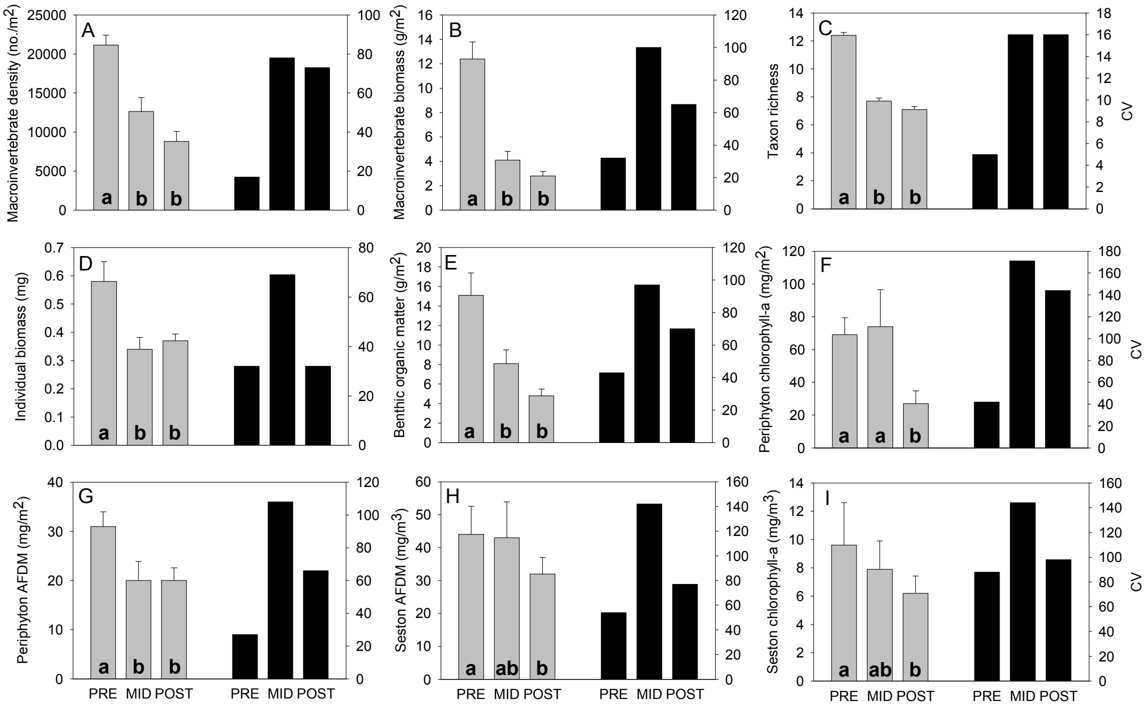

Figure 3.1.3.1: Means (±1 SD, grey bars, left axis label) and coefficients of variation (CV, black bars, right axis label) for (A) macroinvertebrate density, (B) macroinvertebrate biomass, (C) taxon richness, (D) individual biomass, (E) benthic organic matter, (F) periphyton chlorophyll-a, (G) periphyton AFDM, (H) seston AFDM, and (I) seston chlorophyll-a for samples collected before the floods (PRE), samples collected after the first flood in 2000 to those just before the flood in 2003 (MID), and samples collected after the flood of 2003 (POST). Letters within bars indicate significant differences based on Tukey's post-hoc test. Data from Robinson and Uehlinger (2008).

3.2 Sandey

3.2.1 Habitat change

Summary

Natural floodplains are spatially heterogeneous and dynamic ecosystems, but at the same time, a highly endangered landscape feature due to climate change and human impacts such as water storage, flood control and hydropower production. Flow is considered a master variable that shapes channel morphology, and the heterogeneity, distribution, and turnover of floodplain habitats. Despite their highly dynamic nature, the relative abundance of different habitat elements (islands, gravel bars) in natural floodplains seems to remain relatively constant over ecological time periods, and is referred to as the "shifting mosaic steady state" concept. In this conceptual context, we analysed spatio-temporal changes in relative habitat abundance and channel complexity of an alpine floodplain from its near natural state in 1940 before water abstraction and levee construction till 2007 using historical aerial images. Within the first decades of impairment, the relative abundance of floodplain habitats that depend on flood and flow pulses such as parafluvial channels and islands shifted towards a greater abundance of terrestrial forest and grassland habitats. After 1986, the duration and frequencies of high precipitation events (>60 mm 24 h^{-1}) triggering major, channel-reworking floods increased substantially and caused a restructuring of the floodplain and decrease in the abundance of more terrestrial habitat types. These results are contrary to expectations of the "shifting mosaic steady state" concept, yet suggest its potential application as an indicator of landscape transformation, human impacts and climate change on floodplain ecosystems.

Detailed investigations

We used historical aerial images to map the different habitat types of the floodplain over time. All images were taken between July and August when river discharge was similar and below bankfull. Aerial images were scanned at 600 dpi and georectified to a referenced 2007 digital ortho-photo provided by vendors using ArcGIS 9.3 (ESRI Redlands, California, USA). During image rectification, each photograph was re-sampled to a 0.5-m resolution for consistency among all images and resulted in a root mean square error of <1.4 m for all images. Six typical floodplain habitats (i.e., mature forest, gravel, vegetated gravel, islands, water, pasture, and grassland) were quantified for each aerial image. Habitats were delineated by digitizing manually drawn polygons around habitat elements in ArcGIS. From the different vector layers, we calculated the relative proportion of each habitat type as a percentage of total floodplain area to quantify changes in the relative abundance of habitat types between sequential aerial photos. To estimate changes in channel complexity, we derived channel shoreline and thalweg lengths from the ArcGIS vector layers. Shoreline length describes the length of the ecotone between aquatic and terrestrial habitats and was expressed as km per river km. Thalweg length was extracted by converting the channel system from vector to raster format and isolating the center raster cells along the channel continuum. Parameters were used to calculate the sinuosity index (S, eq. 1) and the braiding index (B, eq. 2).

(1) $S = L_T / L_R$
(2) $B = L_T / L_{Tmax}$

Where L_T is the length of the thalweg of the main channel, L_R the length of the whole channel system from the upper to the lower knick-point (straight distance) and L_{Tmax} is the thalweg length of all other channels.

Results showed that the spatial distribution of habitat types changed substantially between 1940 and 2007 (Figure 3.2.1.1). Over 48% of the total floodplain area changed at least once and approximately 5% of the floodplain changed habitats at least four times during the study period.

Table 3.2.1.1: Percentage of the total floodplain area occupied by each of the six habitat types for each measured year of the time series.

Year	Pasture and grasland (%)	Water (%)	Gravel bars (%)	Vegetated gravel (%)	Islands (%)	Forest (%)
1940	52.1	5.7	5.9	12.6	3.6	20.2
1960	67.2	4.0	3.7	3.3	4.1	17.8
1969	69.0	3.9	1.5	3.6	1.7	20.4
1977	69.0	3.4	1.7	3.0	0.9	22.0
1986	67.1	3.7	1.9	3.3	0.2	23.9
1998	69.2	2.8	5.0	2.0	2.9	18.0
2007	69.3	3.8	4.8	2.8	2.1	17.2
Mean	66.1	3.9	3.5	4.4	2.2	19.9
SD	6.2	0.9	1.8	3.6	1.4	2.4
CV (%)	9.4	23.1	51.4	81.8	63.6	12.1

Overall, the habitat composition (number of habitat types) of the floodplain did not change over the study period, but three main time intervals of change in habitat abundance (floodplain percentages) was distinguished. First, a period from 1940 to 1960 when approximately 32% of all habitat types changed spatially (Table 3.2.1.1, Figure 3.2.1.1). Habitat abundance of grassland and pasture increased from 52.1 to 67.2% during this time period, while areas of gravel and vegetated gravel decreased from 5.9 to 3.7% and 12.6 to 3.3 %, respectively. Second, a period from 1960 to 1986 when habitat types overall changed between 15 and 11%. Major changes in habitat abundance were evident for forest habitat, which increased from 17.8 to 23.9%. In contrast, islands decreased from 4.1 to 0.2 % and gravel bars from 3.7 to 1.9 % in habitat abundance (Table 3.2.1.1, Figure 3.2.1.1). All other habitat types remained more or less the same in abundance during this time period. From 1986 to 2007, changes in habitat types were ranged between 18 and 16%, and showed a different developmental pattern compared to before 1986. The abundance of forest decreased by 6.7%, while island and gravel habitats exhibited an increasing trend in area of 1.9 and 2.7 %, respectively. For the entire study period (1940 to 2007), most habitat types decreased in relative abundance, ranging from 9% (forest) to 78% (vegetated gravel), whereas pasture and grassland showed an increase of 28% (Table 3.2.1.1, Figure 3.2.1.1).

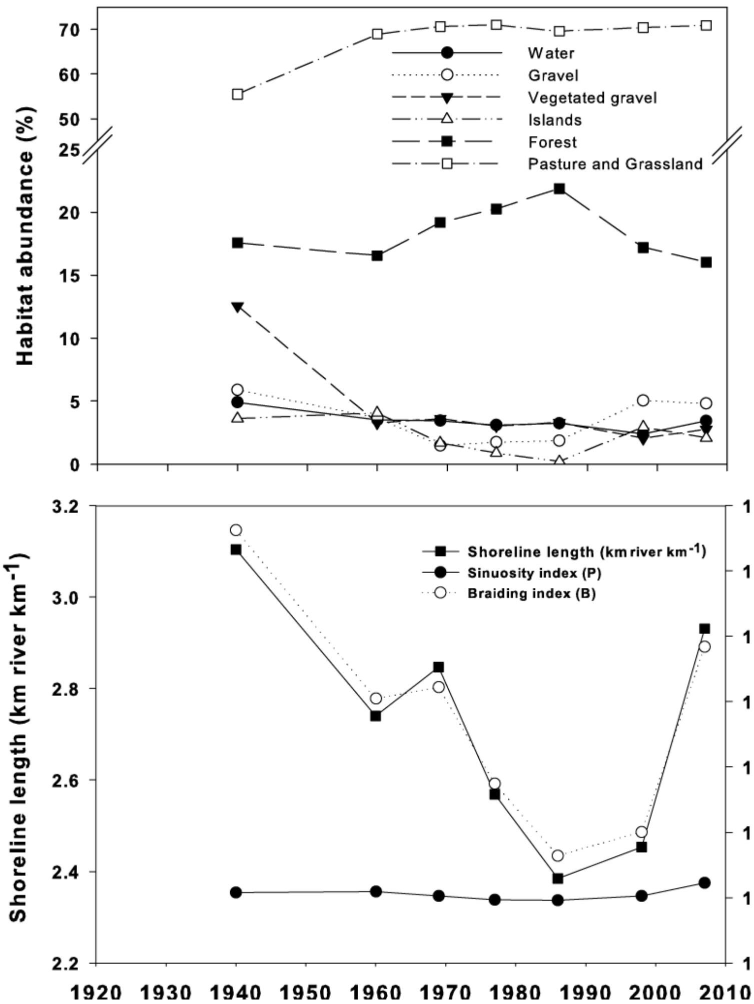

Figure 3.2.1.1: Change in habitat abundance in percent from 1940 to 2007 (upper figure). Change in shoreline length (km km^{-1}), sinuosity index, and braiding index from 1940 to 2007 (lower figure).

Channel complexity paralleled the temporal changes in habitat abundance. From 1940 to 1986, the shoreline length decreased from 3.1 to 2.4 km per river km and the braiding index (B) from 1.75 to 1.25 (Figure 3.2.1.1). Both parameters slightly increased (shoreline length = 0.2 km per river km, B = 0.2) from 1960 to 1969. From 1986 to 2007, both parameters also increased, shoreline length to 1.6 km per river km and B to 2.9, although both values were less than values calculated for 1940. During the whole study period, the sinuosity index remained relatively stable (Figure 3.2.1.1).

The Sandey floodplain is a shifting mosaic of habitat patches that exhibited a distinct change in habitat type abundance and channel complexity over the 67-year study period. These findings are in contrast with results from other near-natural floodplains that show high habitat turnover but a fairly constant habitat type abundance over time, and thus deviates from the "shifting habitat steady state" process for human-impacted floodplains. Importantly, the concept has major applications towards attributing changes in flow due to human impacts (flow regulation, water abstraction) or landscape transformation from environmental change in altering the spatial and temporal dynamics of human-dominated floodplains. For instance, the results of this study indicate that the important linkage between hydrological disturbance (i.e. natural flow regime) and the maintenance of floodplain heterogeneity and process dynamics is decoupled in human-modified floodplains.

3.2.2 Microbial heterogeneity

Summary

Riverine floodplains are among the most biologically diverse and productive ecosystems worldwide. They consist of a mosaic of different habitat types ranging from terrestrial floodplain forests to aquatic channels. They can serve as model ecosystems to investigate the linkage between environmental heterogeneity and ecosystem functioning. In this study, the alpine floodplain of the Urbach in the Bernese Alps in Switzerland was investigated. It is characterized by high volume of hyporheic flow which leads to a down- and up-welling dynamic within the whole system. Up-welling groundwater is frequently enriched by nutrients relative to the surface water, which can influence ecosystem processes in these areas. We measured soil and sediment respiration, bacterial abundance and enzyme activities throughout an annual cycle in four terrestrial (gravel, island, alluvial forest and meadow) and one aquatic (channel) habitats in the floodplain of the Urbach. Additionally, the down- and up-welling dynamic of the system was monitored. The investigated habitat types differed greatly in their substratum composition, water content and temperature. The results display a highly heterogeneous system with low values of respiration, bacterial counts and enzyme activities in the channel and gravel habitats and distinctly higher values in the alluvial forest, island and meadow habitats. Temperature and bacterial abundance were the most important predictors of soil and sediment respiration. The down- and up-welling dynamic could be demonstrated and respiration, bacterial abundance and enzyme activities revealed in several habitats higher values in the up-welling zone. This indicates the importance of up-welling water towards enhancement of ecosystem processes. Overall, we found high functional heterogeneity in measured habitat properties, respiration activities, bacterial abundance and enzyme activities. The results underline a tight and sensitive linkage between habitat heterogeneity and ecosystem functioning in this alpine floodplain. They can serve as a framework to assess and monitor changes in floodplain ecosystems in terms of sustainable resource management or increasing environmental pressures such as from climate change or hydropower production.

Detailed investigations

3.2.2.1 Respiration

A soil respiration chamber (Li6400, LiCor, Lincoln, Nebraska, U.S.A.) attached to a portable Li-6400 infrared gas analyzer (IRGA) was used to measure soil and sediment respiration (i.e. the sum of the autotrophic and heterotrophic respiration; in the following referred to as: respiration) as positive CO_2 flux per m^{-2} s^{-1} in May, August and October 2010. PVC collars of a known volume (8 cm long, 10.5 cm inside diameter) were inserted into the soil and sediments of each predominantly terrestrial habitat type (alluvial forest, island, gravel and meadow), at a depth of 5-7 cm (n=5 per habitat type and date). To avoid effects of prior sampling disturbance, the collars were placed within 1 m^2 but slightly offset from previous sampling sites. According to Norman et al. (1997), insertion of collars results in an initially high CO_2 flux that stabilizes after 10-30 minutes. To minimize this effect of disturbance, collars were placed at least 24 hours before the actual measurements (Buchmann, 2000). To measure the undisturbed CO_2 flux, the soil chamber was set on top of the collars. The measurements were repeated four times per date and site, and values were averaged. Temperature was measured *in situ* by a temperature probe (Testo AG, Mönchaltorf, Schweiz). The soil and sediments enclosed by each collar were collected, enclosed in PVC bags and transported to the laboratory for water, organic matter content and grain sizes as described above.

Respiration of the aerobic aquatic sediments was measured as O_2 consumption over time in plexiglas tubes (32 cm long, 5.2 cm diameter) sealed with rubber stoppers. The autotrophic epilithic algae on the surface were excluded by removing the uppermost centimeters of the sediment. Respiration tubes were half-filled with pre-sieved sediments (≤ 8mm), filled to the top with surface water from the sampling site and sealed. Subsequently, the tubes were incubated *in situ* by burying them into the surface sediment at the sampling site for at least 4 hours. The tubes were buried into the dark in order to avoid artifacts during incubation. Sediment particles >8 mm were excluded because of the random presence or absence of large stones representing a large inactive metabolic volume in a relatively small tube. Temperature and oxygen concentration was measured by an HQ40d Dual-Input Multi-Parameter Digital Meter with a LDO temperature probe (Hach Company, Loveland, Colorado, U.S.A.) before and after incubation. Sediments of each chamber were stored frozen and later analyzed for water, organic matter content and grain sizes.

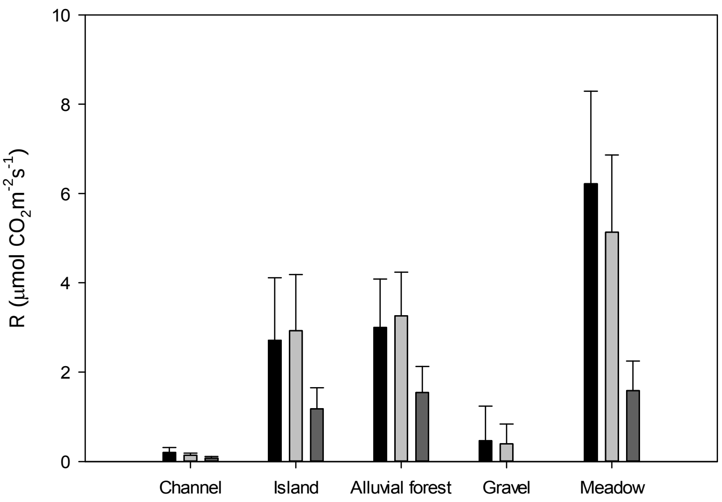

Fig. 3.2.2.1.1: Total uncorrected respiration (average ± SD) of aquatic and terrestrial habitats for three months; n = 20 per habitat type and month in all zones; ▪ = May, ▪ = August, ▪ = October

The system revealed high variability between the habitats concerning respiration (Figure 3.2.2.1.1). The habitat with the lowest respiration values was the channel habitat (mean over three months = 0.140 ± 0.087 µmol CO_2 $m^{-2}s^{-1}$) and the meadow habitat showed the highest values (mean over three months = 4.312 ± 2.544 µmol CO_2 $m^{-2}s^{-1}$). The respiration ranged from 0 µmol CO_2 $m^{-2}s^{-1}$ in the gravel habitat in October to 6.219 µmol CO_2 $m^{-2}s^{-1}$ in the meadow in May. ANOVA showed highly significant differences in respiration between the habitats (df = 4; F = 411.183; $p < 0.05$), although differences between the channel and gravel habitat and the alluvial forest and island habitat were not significant (Scheffé *post hoc*; $P > 0.05$). Respiration differed significantly with season (month) except from August and May (df = 2; F = 119.436; $p < 0.001$). A significant interaction effect between habitat and season could be observed (df = 8; F = 12.001; $p < 0.001$); i.e. the habitats showed a different seasonal trend in their CO_2 efflux.

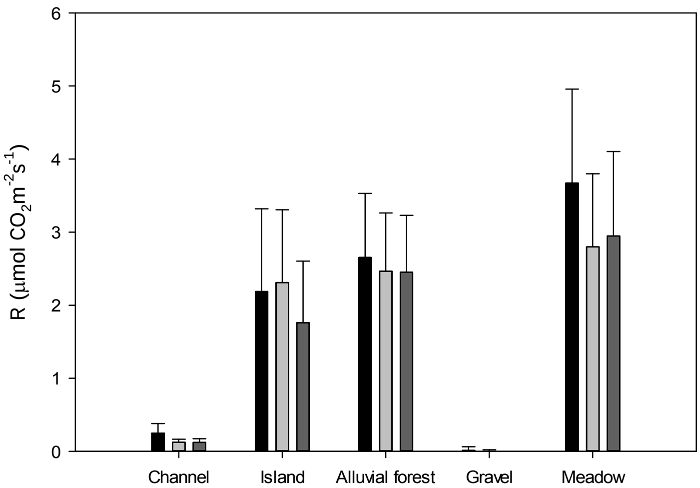

Fig. 3.2.2.1.2: Total respiration (average ± SD) of aquatic and terrestrial habitats standardized at 12 °C; n = 20 per habitat type and month; ■ = May, ■ = August, ■ = October

The respiration data corrected to 12°C displayed a similar pattern as the uncorrected data (Figure 3.2.2.1.2.). The range of values (mean over three months) was lower and reached from 0.006 ± 0.0315 µmol CO_2 $m^{-2}s^{-1}$ in the gravel habitat to 3.139 ± 1.197 µmol CO_2 $m^{-2}s^{-1}$ in the meadow habitat. The highest respiration activity of the island habitat was found in August (2.309 ± 0.997 µmol CO_2 m^2s^{-1}); in all other habitats the highest respirations were measured in May. Respiration differed significantly among the habitats (df = 4; F = 608.405; $p < 0.001$); only the differences between the gravel and channel habitats were not significant. In turn, a stronger variability between the habitats could be shown with the corrected (12°C) respiration data in comparison to the uncorrected. A date effect was apparent (df = 2; F = 4.554; $p < 0.05$) but only between May and August was significant ($p < 0.05$). An interaction effect between the habitats and the months could not be shown (df = 8; F = 1.662; p = 0.107). The corrected data (12°C) had a lower variability concerning the seasonality in comparison to the uncorrected data. The results of this study show the spatio-temporal patterns within and between different habitats in the Urbach floodplain concerning respiration. Harsh environments like the exposed gravel and the channel displayed the lowest values and activities overall. The more stable habitats - the alluvial forest, the island and the meadow - have a lower flooding frequency overall and showed higher activities.Respiration measurements can serve as an indicator to assess and monitor changes in floodplain ecosystems in terms of sustainable resource management or increasing environmental pressures such as from climate change or hydropower production.

3.2.2.2 Microbial composition

Soil and sediment <8 mm were used for the analysis of bacterial abundance and enzyme activity (one small PVC bag with ~ 50 g samples each). The fraction >8 mm was excluded to avoid unevenly distributed large stones representing a large part of metabolic inactive substrate. Plant debris and roots were also removed if possible. Sediment samples of the channel habitats were taken including pore space water to account for dissolved fine material. Samples for the analysis of bacterial abundance were stored at 4°C and samples for analysis of enzyme activity were stored frozen at -20°C.

Eight enzymes that provide insight into the microbial C, N and P acquisition were used for the assays. The enzyme activities were measured by using methylumbelliferon(MUB)-labelled substrates (Table 3.2.2.2.1).

Table 3.2.2.1: Compilation of the investigated enzymes, the analogue substrate, their function, the buffer which was used for the substrate (BI = bicarbonate, MQ = Milli-Q, deionized water) and the corresponding target-nutrient (C = carbon; N = nitrogen; P = phosphorus)

Enzyme	Substrate	Function	Nutrient	Buffer	Reference
α – Glucosidase	*4-MUB-α-D-glucoside*	polysaccharide-degrading; hydrolyzes starch	C	BI	Warren, 1996; Tabatabai and Fu, 1992
β – Glucosidase	*MUB-β-D-glucopyranoside*	polysaccharide-degrading; hydrolyzes celluloses and other β-glucans	C	BI	Sinsabaugh et al., 2008
Xylosidase	4-*MUB-β-D-xylopyranoide*	polysaccharide-degrading ; hydrolyzes hemicelluloses	C	BI	Warren, 1996
Esterase	*4-MUB-acetate*	lipolytic enzyme; catalyzes the hydrolysis and synthesis of acylglycerides and other fatty acid esters	C	MQ	Arpigny and Jaeger, 1999
β – N – acetylglucosa-minidase (NAG)	*4-MUB-N-acetyl-β-glucosaminide*	degradation of chitin but also to several β-1, 4 glucosamine polymers	N	BI	Sinsabaugh et al., 2008
Leucine-aminopeptidase	*L-Leucine 7-amido-4-methyl-coumarin*	hydrolyzes hydrophobic amino acids from the nitrogen terminus of polypeptides	N	MQ	Sinsabaugh et al., 2008
Endopeptidase	*4-MUB-P-guanadinobenzoate*	subgroup of protease, catalyzes the internal cleavage of peptides or proteins (i.e. within the molecule)	N	MQ	National Library of Medicine – Medical Subject Headings, 2011
Phosphatase	*4-MUB-phosphate*	removes phosphate groups from organic compounds	P	BI	Burns, 1978

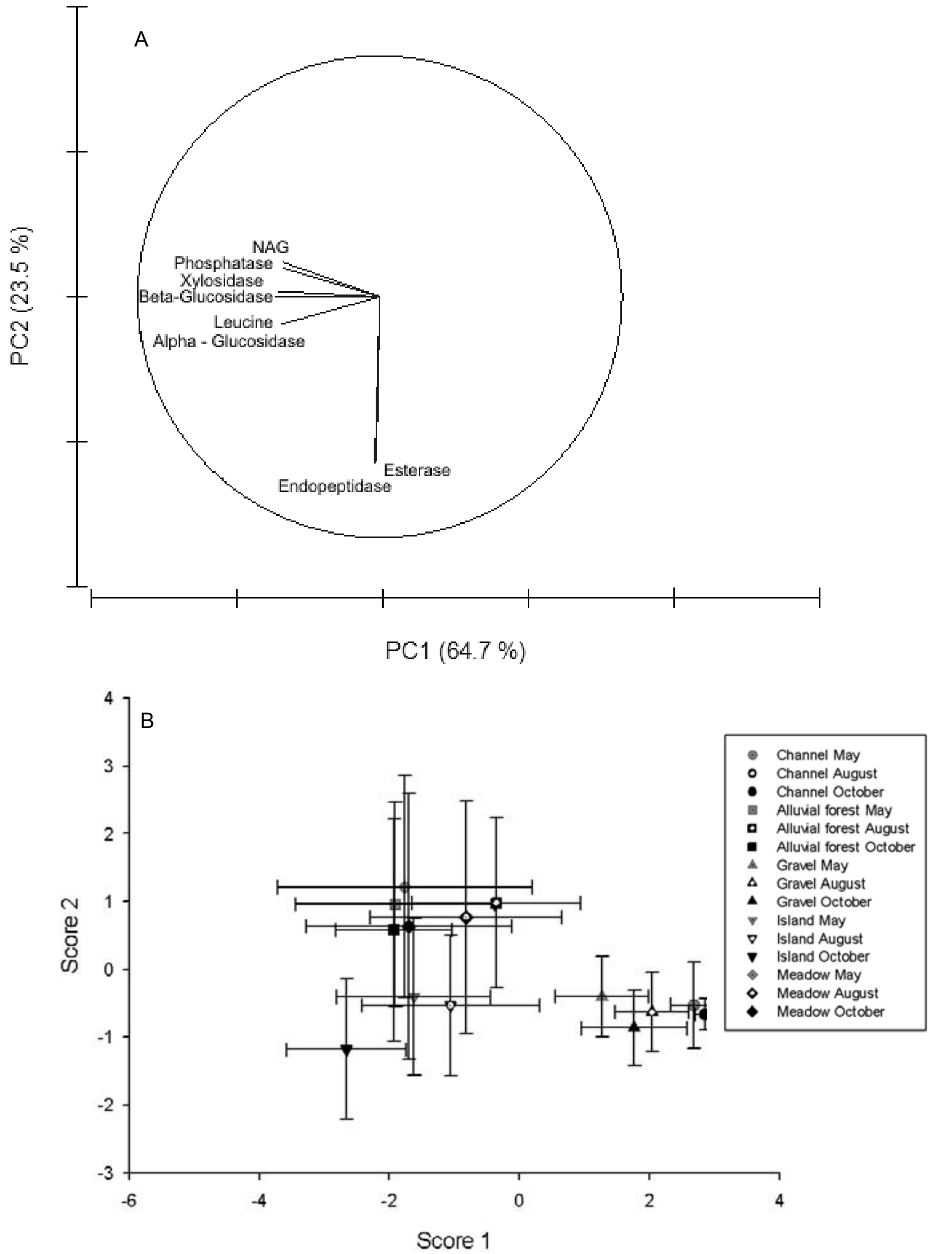

Figure 3.2.2.2.3: Principal component analysis using enzyme activities; (A) Correlation circle (highest possible contribution of a factor to an axis (=1) and the superimposed eigenvectors of the different enzymes; NAG = N-acetylglucosaminidase; the lines of leucine and alpha-glucosidase lie one upon the other; (B) factor map of the different habitats (mean of 12 values per habitat and month); symbols: habitats of three months; lines: standard deviation of score one and two

For the fluorometric enzyme assays five to 10 g of sediment sample (5 g for island, meadow and forest samples; 7 g for gravel samples; 10 g for channel samples) were mixed with 20 ml of autoclaved MQ and vortexed (Vortex-Genie 2, Scientific industries, Inc., New York, U.S.A.) for 1 minute. The reason for the different amounts of sediment is the varying intensity of enzyme activity in the different habitats. Exactly 150 µL of the suspended samples and 100 µL of substrate analogue (final substrate concentration = 400 µM) were pipetted into a 96 well plate. Substrate analogues were buffered in bicarbonate or in MQ, respectively (Table 3.2.2.2.1). Substrate blanks (MUF-substrate plus buffer) were used to assess sample specific quenching due to different suspension colors by adding 50 µl of a 100 µM free MUB solution to the blanks and the corresponding samples. The ratio between blank and sample fluorescence increase was then used as a quenching correction-factor. A standard curve was used to calculate the substrate procession rate. Fluorescence was measured with a microplate reader (Tecan Infinite® 200, Männedorf, Switzerland) at an excitation wavelength of 365 nm and an emission wavelength of 450 nm. Three measurements were performed 10 min to one hour after the addition of the substrates to detect the more active enzymes and two after 15 to 20 hours to detect the less active ones.

The microplates were incubated on a shaker at 15°C in the dark between the measurements.

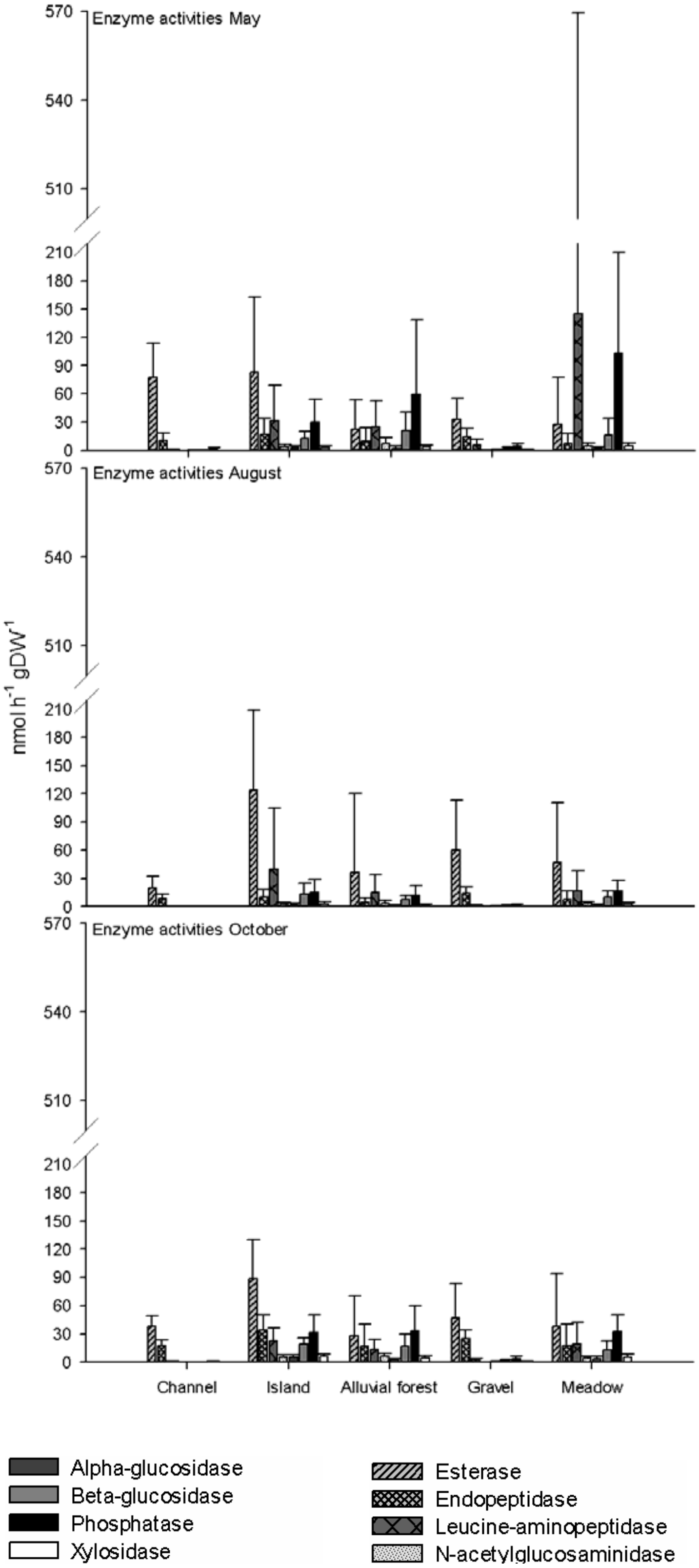

Fig. 3.2.2.2.4: Enzyme activities of eight enzymes (nmol h^{-1} gDW^{-1} (dry weight)) within the different habitats for the months May, August, October; n = 12 (for each habitat and month)

The Principal component analysis (PCA) using the eight different enzymes separated the gravel and channel from the other habitats (Figure 3.2.2.2.3). The habitats of island, alluvial forest and meadow could not be distinctly separated from each other. PC 1 (eigenvalue: 5.18) and PC 2 (eigenvalue: 1.88) explained 64.7 and 23.5 % of the variance, respectively. PC1 was mostly defined by leucine, N-acetylglucosaminidase (NAG), alpha- and beta-glucosidase, phosphatase and xylosidase on the negative axis. PC2 was negatively defined by esterase (eigenvector: -0.688) and endopeptidase (eigenvector: -0.682). The channel and gravel habitats were clustered, all the other habitats were highly scattered regarding their enzyme activities. Enzyme activities were characterized by a high variability within the different habitats (Figure 3.2.2.2.4). In contrast to the other habitats, the channel and gravel habitats enzyme values were low except from esterase and endopeptidase. The highest enzyme activity (values over three months) was measured for esterase, which ranged from 44.899 ± 33.060 (channel) to 98.313 ± 72.332 nmol h^{-1} gDW^{-1} (island), the lowest values were measured for alpha-glucosidase which ranges from 0.057 ± 0.055 (channel) to 3.193 ± 2.240 nmol h^{-1} gDW^{-1} (island). Leucine displayed the largest range of enzyme activity with ranges from 0.403 ± 0.437 (channel) to 60.430 ± 246.192 nmol h^{-1} gDW^{-1} (meadow). Seasonality could be observed with higher values in May and October and lower values in August. All individual enzymes showed a significant site effect (Table 3.2.2.2.2). Concerning the enzyme activities of phosphatase, beta-glucosidase, leucine, alpha-glucosidase and xylosidase, the habitats channel and gravel could be differentiated from each other and from all other habitats whereby the habitats alluvial forest, island and meadow could not be distinguished from each other. The enzyme NAG showed the same pattern but the channel and gravel habitat could not be significantly differentiated from each other. For esterase and endopeptidase, the channel could be differentiated from the alluvial forest and meadow but the forest not from the meadow habitat. Further, it could not be statistically differentiated between the habitats channel, gravel and island (Scheffé *post hoc*; $P > 0.05$). For all enzymes, except for esterase, a significant date effect was observed. An interaction effect could be shown for the enzymes alpha-glucosidase and xylosidase.

Table 3.2.2.2.2: Information about site, date and interaction effect of log-transformed data of different enzymes; *: $p < 0.05$; **: $p < 0.001$, ne: no effect, $p > 0.05$

Enzyme	Site effect	Date effect	Interaction effect
Esterase	**	ne	ne
Endopeptidase	**	**	ne
Leucine-aminopeptidase	**	*	ne
N-acetylglucosaminidase	**	*	ne
Alpha-glucosidase	**	**	*
Beta-glucosidase	**	*	ne
Phosphatase	**	**	ne
Xylosidase	**	**	*

Considering the function of individual enzymes (C, N and P-processing; Table 3.2.2.2.2), habitats could be distinguished roughly into two groups. A first group consisting of the gravel and channel habitats dominated by C and N processing enzymes, mainly esterase and endopeptidase and a second group consisting of the alluvial forest, island and meadow habitat containing the whole range of C, N and P processing enzymes (Figure 3.2.2.2.4). However, all enzyme activities differ seasonally in their intensities. The results show that the spatio-temporal patterns within and between different habitats in the Urbach floodplain concerning enzyme activities. Harsh environments like the exposed gravel and the channel displayed the lowest activities overall. The more stable habitats - the alluvial forest, the island and the meadow - have a lower flooding frequency overall and showed higher activities. Overall enzyme activities can serve as an indicator to assess and monitor changes in floodplain ecosystems in terms of sustainable resource management or increasing environmental pressures such as from climate change or hydropower production.

3.3 The Julian Alps, Karavanke and Kamniško-Savinjske Alps

Summary

In the study from Slovenia an attempt was made to ecologically characterize and asses the vulnerability of the alpine aquifers combining hydrogeochemistry and faunistic surveys using springs as an access points to the groundwater and as macroinvertebrate habitats. Additionally, we tried to develop simple indicators for the rapid detection of significant decreases in groundwater levels. Water from 12 springs located in the alpine region in Slovenia was collected during high and low flows in 2010 in order to carry out geochemical analyses (anions, cations, $d^{13}C_{DIC}$, $d^{13}C_{POC}$, $d^{18}O$, dD, tritium). Concurrently, discharge, temperature, oxygen and pH were measured, and invertebrates drifting from the aquifer and macroinvertebrates inhabiting springs were sampled. The groundwaters studied represent waters strongly influenced by chemical weathering of Mesozoic limestone. The $\delta^{13}C$ of DIC ranged from -15.8‰ to -1.5‰ and indicated less and more vulnerable aquifers. Isotopic composition of oxygen ($\delta^{18}O_{H2O}$), and tritium values range from -12.2 to -9.3‰, and from 6.4 to 9.8 TU, indicate recharge from precipitation. The age of spring waters were estimated to be from 2.6 to 5.1 years. The invertebrates collected differed between the hydrogeological units identified by hydrological and geochemical measurements indicating to be a good predictor of aquifers hydrogeological characteristics. Moreover, the number of invertebrate species and their densities were higher in 2009 than in 2010 which was a "dry" year by means of precipitation in comparison to 2010. It seems that lowering of groundwater water table stimulate more intense drifting of groundwater invertebrates. Hence, the monitoring of invertebrate drift from springs can be a useful tool to assess the level of stress in groundwater ecosystems due to decreasing water levels. Similar observations were noticed regarding spring macroinvertebrates. Higher densities occurred during 2009 in comparison to 2010.

Detailed investigation

3.3.1 Hydrogeochemistry and temperature

The locations and geological settings of studied springs are shown in the Figure 3.3.1.1. The majority of the springs are outlets of the aquifers formed in Mesozoic limestone and dolomite, while one spring is located in Permo-Carbonian shales. They are spreading from 596 to 1236 m a. s. l. All the springs belongs to a group of rheocrene springs. Here the water flows directly out of the ground, usually quite fast and immediately forms a stream without forming the pond. According to radiodecay of tritium, the age of spring waters were estimated to be about 2.6 years for springs located in Julian Alps, about 5 years for springs located in Karavanke and about 5 years for springs located in Kamniško-Savinjske Alps. Fast groundwater flow and short retention times indicated sensitivity of spring discharge to seasonal changes in precipitation. Closer look to precipitation patterns in 2009 and 2010 (Figure 2) shows that precipitation amounts were lower in summer and autumn 2009 in comparison to 2010 in all three mountain ranges. Accordingly, some of the springs dried out during summer 2009 and winter 2010. At the regional scale, 9 springs were permanent springs with variable discharge, while 3 of them completely dried out some of the even up to two months. We were able to observe the drying out by the continuous measurements of water temperature using temperature loggers placed in the spring mouth (Figure 3.3.1.3a,b). This revealed to be a good approach to observe the permanence of spring discharge. In permanent springs, discharge ranged from 0.9 ± 0.2 SD (spring 9) to 215.8 ± 79.7 SD (spring 12) (Table 3.3.1.1). Water temperatures were correlated with the altitude (i.e. high altitude springs had lower temperatures) and were from 4.9 to 7.9°C, with small variation within a spring (between 0.04-0.38 SD per spring), while pH ranged between the springs from 6.8 – 8.1 (Table 3.3.1.1). Mean conductivity, which can be indirect measure of retention times of the water in the subsurface, was from 275 to 83 μScm^{-1} for 12 springs. It fluctuated the most in spring waters from the Julian Alps and was generally higher in 2009 (dryer year) (Figures 3.3.1.4, 3.3.1.5). The conductivity was higher for about 100-150 μScm^{-1} during 2009 in comparison to spring and summer 2010 in all four springs from Julian

Alps. The conductivity therefore indicated lower precipitations and lower water levels in the aquifer as well as slower groundwater flow during 2009.

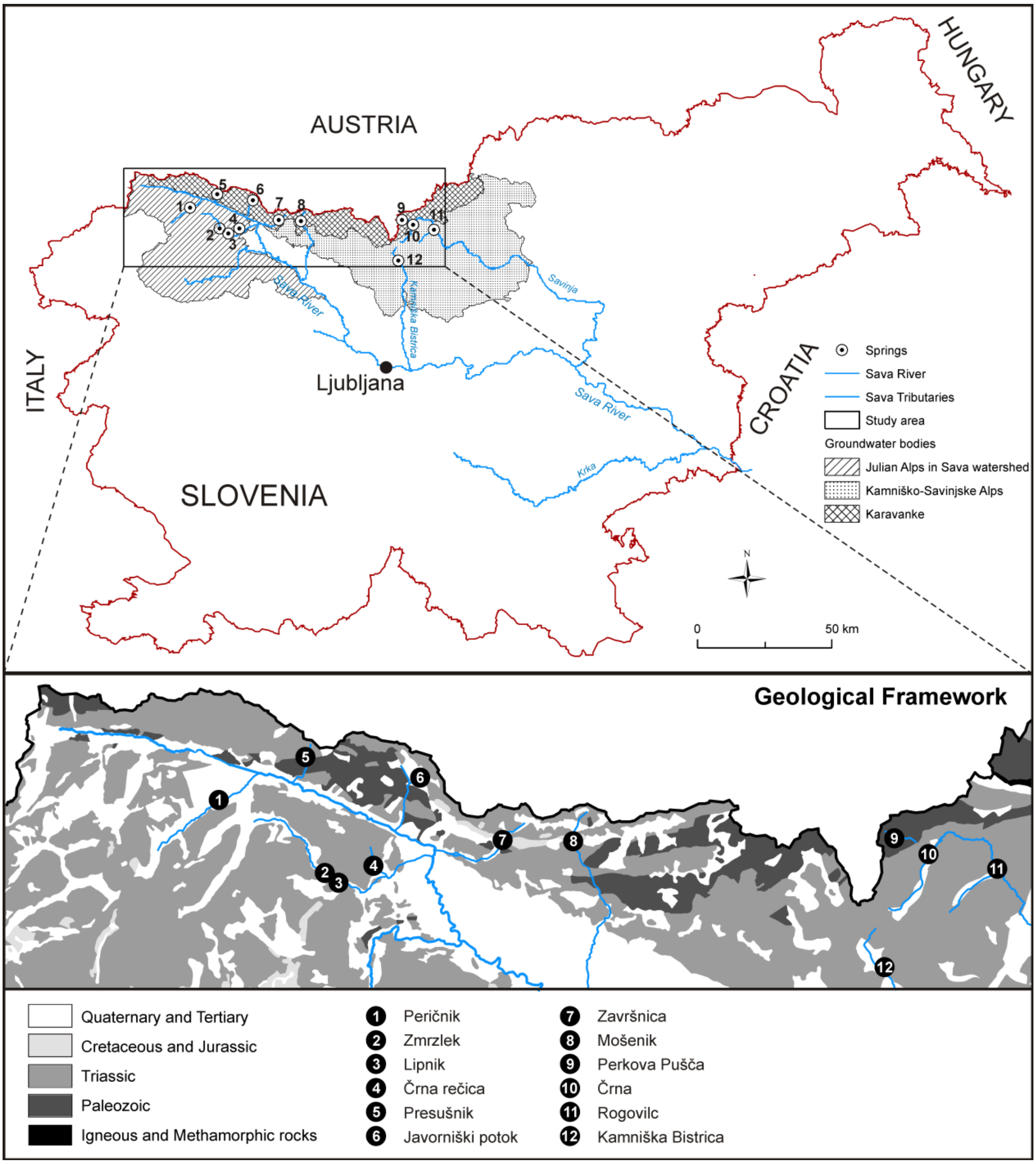

Figure 3.3.1.1: The geology of the area and location of sampling sites. Adapted from Kanduč et al. (Chem Geol, in review).

The nutrients (nitrates, ortho-phosphate) were low in all spring waters, with NO_3 ranging from 1.23 - 3.01 mg l^{-1} and ortho-phosphate being below limit of detection. Similarly, little variation in nitrates was observed during the sampling period. The biogeochemical processes affecting DIC concentrations and $\delta^{13}C_{DIC}$ values were quantified by mass balance calculation, showing that the most important process in all investigated springs is carbonate mineral dissolution with DIC contribution ranging from 48.6% (Črna rečica, location 4) to 86.3% (Kamniška Bistrica, location 12). Only one spring is regarded as an akvitard with no carbonate dissolution (Perkova pušča, location 9). Based on principal component analysis (PCA), where the geochemistry of the spring water was considered, the springs can be separated into 4 groups: 1) springs located in carbonate aquifers, 2) springs located in carbonate aquifers with less soil CO2 contribution, 3) springs located in aquifers with more CO2 contribution, and 4) springs located in non-carbonate aquifers (only 1 sampled as part of this study) (Figure 3.3.1.5). This results of PCA show that spring water as well as the quality of drinking water is controlled primarily by geological composition of the aquifer. The $\delta^{13}C$ of DIC revealed to be a good indicator of the vulnerability of the aquifers (Figure 3.3.1.5). Since vulnerability of groundwaters in Slovenia is highly related with soil profiles (i.e. thicker soils lead to longer infiltration times, which reduces vulnerability to surface contaminants) we can also conclude that $\delta^{13}C$ of DIC values can be used to assess vulnerability of springs. The aquifers with less soil CO2 contribution (higher $\delta^{13}C_{DIC}$ values) and more carbonate dissolution are likely more vulnerable to potential pollution. The investigation of the aquifers from the pilot sites showed that their recharge depends greatly on the precipitation in the area (shallow basin aquifers) and that the residence times of water are short (2-5 years). Short residence times are due to highly permeable rocks with mostly fissured porosity or even karstified rock (wider channels). The measurements of water chemistry and stable isotopes showed to be a good approach to characterize the aquifers geohydrological characteristic. While the data loggers continuously measuring the spring water temperature revealed to be useful to detect prolonged periods of drought and lowering of groundwater levels. The spring waters has more or less stable temperature, but when data sets show extremely higher or lower temperatures this indicates the period when the groundwater level decreased so much that the spring dryed out. The measuring of water conductivity also appeared to be good indicator of low groundwater flow and low spring discharge.

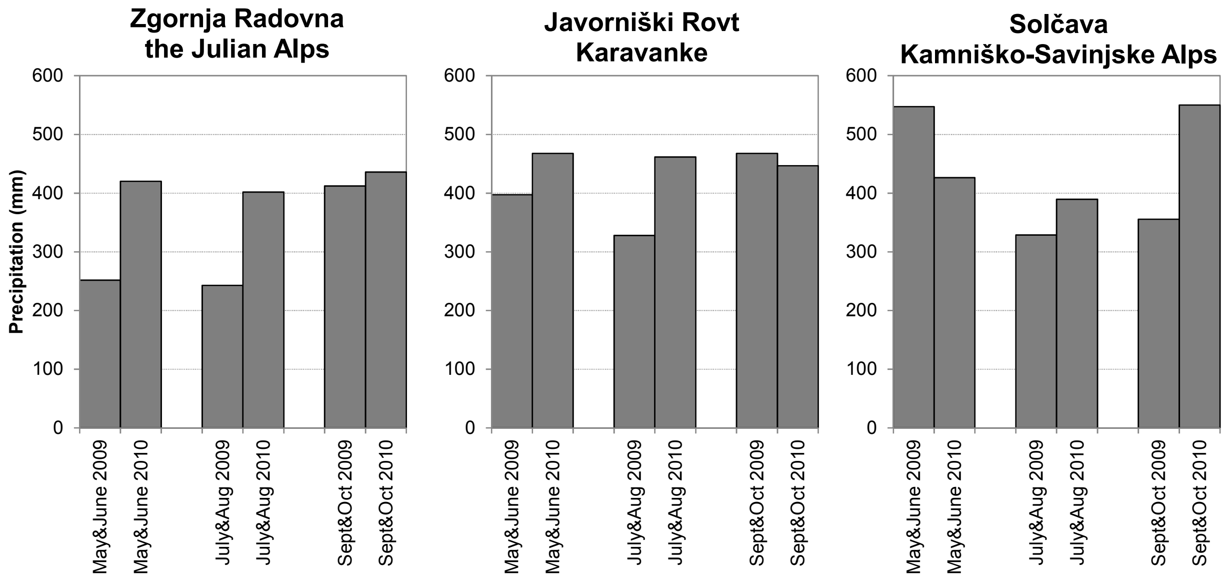

Figure 3.3.1.2: The two-months precipitation (mm) during sampling periods in 2009 and 2010. Data from three meteorological stations are provided by Slovenian Environment Agency.

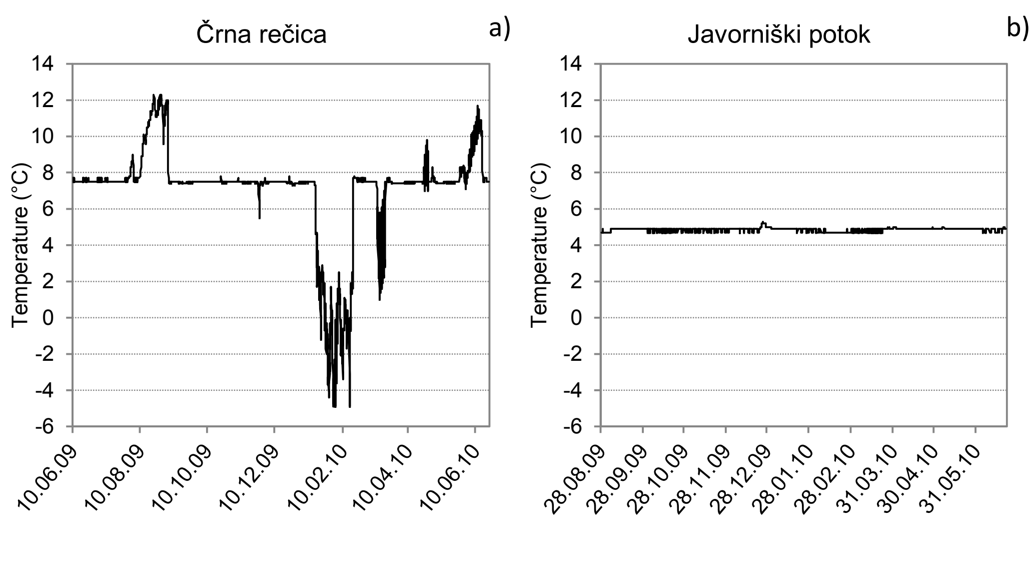

Figure 3.3.1.2: The data from the temperature logger placed in intermittent (a) and permanent (b) spring. The significant increase and decrease in temperature in the left figure are due to drying out of the spring.

Table 3.3.1.1: The main characteristics of the springs from Slovenian Alps. The spring numbers corresponds to those from the Figure X.

	spring name	land use	spring permanence	altitude	mean discharge	SD	temperature	SD	pH	SD
				m a.s.l.	$l\ s^{-1}$		(°C)			
1	Peričnik	Mixed forest	permanent	733	3.5	0.4	6.0	0.2	8.1	0.1
2	Zmrzlek	Mixed forest	*intermittent*	698	10.4	4.3	6.0	0.0	7.9	0.2
3	Lipnik	Mixed forest	permanent	692	14.9	6.6	6.7	0.1	7.7	0.2
4	Črna rečica	Coniferous forest	*intermittent*	653	21.6	7.0	7.5	0.1	7.7	0.2
5	Presušnik	Mixed forest	*intermittent*	1203	1.8	0.5	7.5	1.6	7.9	0.2
6	Javorniški potok	Mixed forest	permanent	1107	32.4	4.7	4.9	0.1	8.0	0.1
7	Završnica	Mixed forest	permanent	941	21.9	5.2	5.8	0.2	7.9	0.1
8	Mošenik	Coniferous forest	permanent	797	27.9	3.8	7.2	0.4	7.9	0.1
9	Perkova Pusca	Coniferous forest	permanent	1236	0.9	0.2	7.4	0.1	6.8	0.4
10	Crna	Mixed forest	permanent	744	9.9	1.8	7.3	0.4	7.8	0.1
11	Rogovilc	Pastures	permanent	645	1.3	0.6	7.8	0.2	8.0	0.1
12	Kamniška Bistrica	Mixed forest	permanent	596	215.8	79.7	5.4	0.2	8.0	0.1

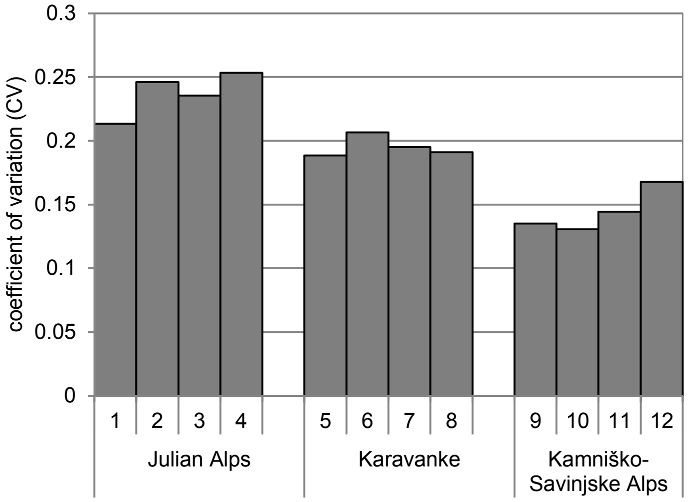

Figure 3.3.1.4: Coefficient of variation for conductivity (µS/cm) measured during spring, summer and autumn samplings in 2009 and 2010. The spring numbers corresponds to those from the map.

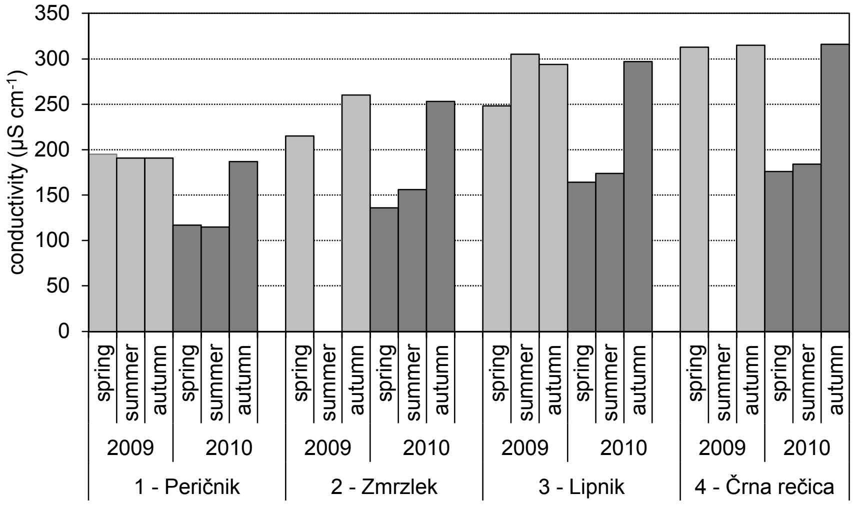

Figure 3.3.1.4: Variation in water conductivity (µS cm^{-1}) over the two years in the springs from Julian Alps.

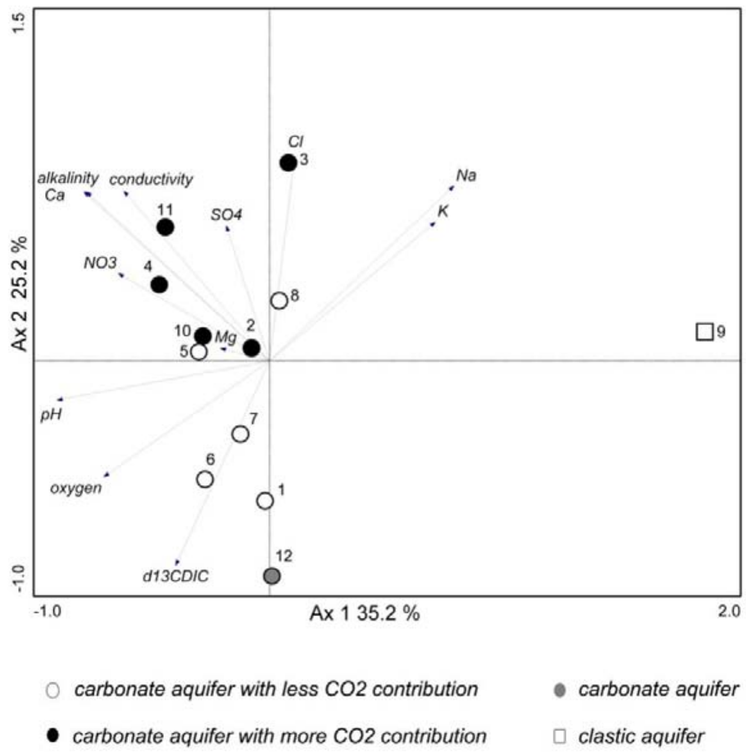

Figure 3.3.1.5: The principal component analysis (PCA) of geochemical parameters of spring water collected in 2010 (Chem Geol, in review).

3.3.2. Invertebrates

Invertebrates were sampled over 2 years in groundwater drift and spring benthos (Figure 3.3.2.1). Since little was known about groundwater and spring fauna in this region until now, a detailed study of the invertebrate communities were carried out. After this a response of communities to variation in discharge and precipitation regimes were investigated. The second part of the study is presented in the next chapter.

Figure 3.3.2.1: The sampling of invertebrate drift (left) and spring benthos (right).

Invertebrate drift

In the drift, the predominant taxa were copepods, followed by ostracods, amphipods and isopods (all Crustacea). Since copepods and ostracods were much more abundant and widespread than amphipods and isopods, we analysed only the data for the former. Altogether, 48 species of Copepoda and Ostracoda were collected from 12 springs (Table 3.3.2.1). Among them *Cavernocypris subterranea* (Ostracoda), *Acanthocyclops sensitivus*, *Speocyclops infernus*, *Elaphoidella phreatica*, *Eucyclops serrulatus*, *Bryocamptus dacicus and Bryocamptus zschokkei* (Copepoda) were the most abundant (300 – 600 individuals per 100 m^{-3}). The most wide spread species occurring in all springs was *Bryocamptus dacicus* (Harpacticoida, Copepoda). The total densities of microcrustacea were the highest in the Julian Alps springs and the lowest in Karavanke springs (Figure 3.3.2.2).

Table 3.3.2.1: List of Copepoda and Ostracoda collected in drift samples in 2009 and 2010. Total densities are given for all springs together. The most widespread species are in bold.

		Julian Alps			Karavanke				Kamniško-Savinjske Alps			
	total density 100 m^{-3} water	Peričnik	Črna rečica	Lipnik	Mošenik	Završnica	Javorniški potok	Presušnik	Kamniška Bistrica	Rogovilc	Crna	Pavličevo sedlo
Cavernocypris subterranea	614	*		*	*	*	*	*		*	*	*
Acanthocyclops sensitivus	410		*	*						*	*	
Speocyclops infernus	**372**	*	*	*	*	*	*	*	*	*		*
Elaphoidella phreatica	354		*	*	*	*			*	*	*	
Eucyclops serrulatus	317				*						*	*
Bryocamptus dacicus	**307**	*	*	*	*	*	*	*	*	*	*	*
Bryocamptus zschokkei	**290**	*	*	*	*	*	*		*	*	*	*
Diacyclops clandestinus	230			*	*	*					*	*
Attheyella wierzejski	110	*		*	*	*	*			*	*	*
Mixtacandona sp. B	105			*								
Diacyclops langiudoides	97		*				*					
Bryocamptus pygmaeus	**77**	*	*	*	*	*	*	*	*	*	*	
Moraria radovnae	77	*		*	*	*	*				*	*
Bryocamptus n.sp.	61	*	*	*		*				*	*	*
Moraria varica	61						*					
Fabaeformiscandona breuili	50							*		*		
Mixtacandona sp. A	45		*	*								
Diacyclops zschokkei	37	*					*	*				
Fabaeformiscandon brisiaca	23											*
Psychodromus fontinalis	22					*				*		*

Moraria poppei	16	*			*	*					*	*
Bryocamptus cuspidatus	15				*	*	*					*
Nitocrella n.sp.	14				*	*	*					
Mixtacandona cf. laisi	13								*		*	
Echinocamptus pilosus	12									*		*
Paracyclops fimbriatus	10		*						*	*		*
Lessinocamptus pivai	9		*	*				*		*		
Bryocamptus rhaeticus	7			*				*				
Ceuthonectes serbicus	6	*	*	*						*		
Potamocypris zschokkei	6									*		*
Austriocyclops n.sp.	6		*						*			
Nitocrella sp.	5		*	*					*	*	*	
Mixtacandona cf. stammeri	5				*						*	
Moraria cf. alpina	5			*			*					
Graeteriella cf. unisetigera	4								*	*		
Paracamptus schmeili	3			*				*				
Potamocypris fulva	3							*				
Potamocypris fallax	3									*		
Mixtacandona sp. C	2			*								
Fabaeformiscandona sp. A	2					*	*					
Attheyella crassa	1		*									
Bryocamptus typhlops	1				*				*			
Eucyclops graeteri	1								*			
Bryocamptus pyrenaicus	1									*		
Nitocrella n.sp. 2	1				*							
Cryptocandona vavrai	1											*
Acanthocyclops vernalis	1									*		
Tropocyclops n.sp.	1								*			

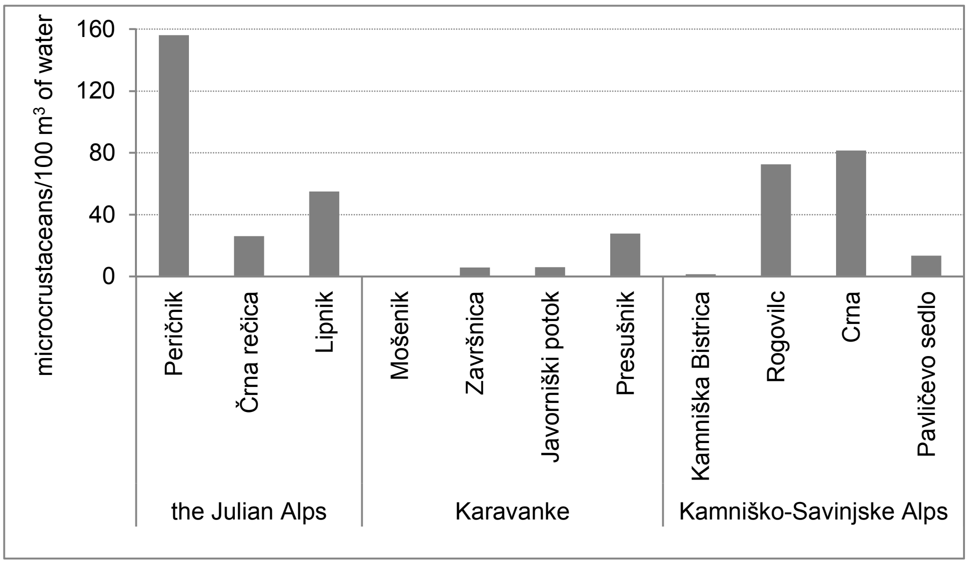

Figure 3.3.2.2. The mean densities (n=6 dates) of microcrustacean drift collected in 2009 and 2010 from 12 springs.

The comparison of groundwater crustacean community using principal component analysis (PCA) revealed high heterogeneity in species composition between the springs (Figure 3.3.2.3). The community in spring Lipnik was different due to abundant presence of *Mixtacandona* species (Ostracoda) and *Elaphoidella phreatica* (Harpacticoida), while Peričnik community differed so much due to high densities of *Cavernocypris subterranea* (Ostracoda). The springs from the Julian Alps had the most abundant and also the most distinct communities indicating that we sampled small localized karstic aquifers with developed vertical and horizontal channels which provide extensive and diversified habitats for the groundwater fauna. In the next chapter, the response of invertebrate drift to variation in spring discharge over two years is discussed.

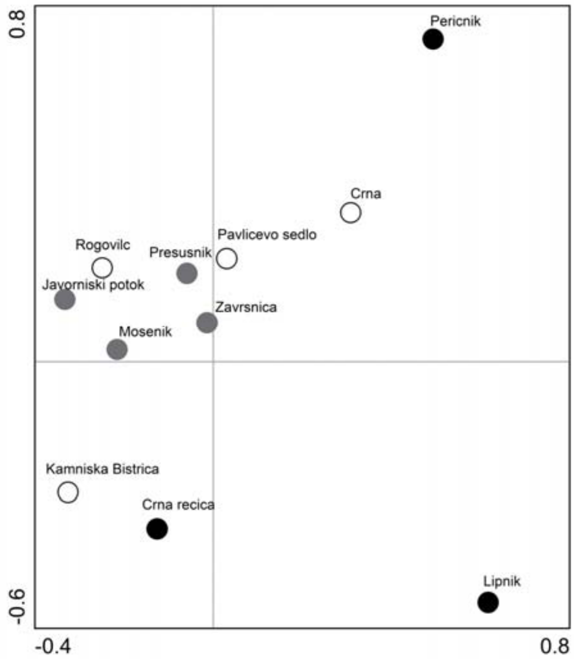

Figure 3.3.2.3: PCA ordination diagram based on species composition of groundwater drift from the Julian Alps springs (black circles), the Karavanke springs (grey circles), and the Kamniško-Savinjske Alps springs (white circles). The first axis explained 16.7% and second 15.8% of variability in community composition.

Spring macroinvertebrates

The EPT taxa are commonly used as indicators of environmental degradation. Some of the studies showed also their sensitivity to flow permanence (Woods et al. 2005). Therefore, we tested if some of the EPT taxa from the springs could be used as indicator of water scarcity in alpine aquifers. Similarly, like in a case of groundwater fauna, little was known about spring macroinvertebrates in this region. Hence, our first step was to carry out the detailed faunistic study and environmental characterization of the springs. The second step was to study the response of EPT taxa to variations in discharge over the two years (2009 and 2010). The most abundant and widespread macroinvertebrate groups in springs were Chironomidae (Diptera), Plecoptera and Gastropoda (Figure 10). In some springs we collected also groundwater amphipods: *Niphargus* sp. Schioedte, 1847. *Gammarus fossarum* Koch, 1836 (Amphipoda), *Protonemura* sp. Kempny, 1898 (Plecoptera), *Dictyogenus alpinum* Pictet, 1841 (Plecoptera) and Chironomidae were the most frequent taxa which were collected from more than a half of the sampled springs. Due to the presence of mostly early larval stages, the identification of EPT taxa was possible only to the taxonomic level of genus or family. Altogether 30 taxa was recorded in 12 springs (Table 3.3.2.3). The taxonomic richness ranged from 7 to 14 taxa per spring. Comparison of EPT communities between the springs by using principal component analysis (PCA) indicated that the most important factors affecting EPT in springs are altitude, water temperature and proximity of springs (Figure 3.3.2.4). The resilience and resistance of EPT to fluctuations in spring discgarge are discussed in the next chapter.

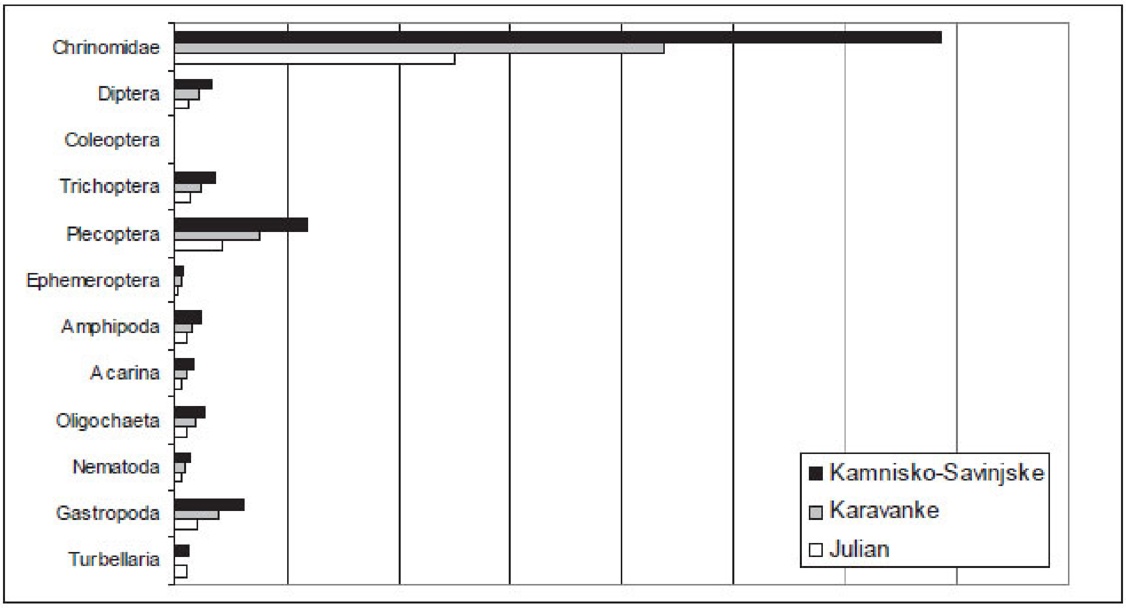

Figure 3.3.2.4: The relative abundances of macroinvertebrates in springs from three mountain ranges.

Table 3.3.2.3. List of EPT taxa collected from 12 springs in the Julian Alps, Karavanke and Kamniško-Savinjske Alps.

		the Julian Alps				Karavanke				Kamniško-Savinjske Alps			
	total density	Peričnik	Črna rečica	Lipnik	Zmrzlek	Mošenik	Završnica	Javorniški potok	Presušnik	Kamniška Bistrica	Rogovilc	Crna	Pavličevo sedlo
EPHEMEROPTERA													
Baetidae	13	8	3	17	4	7	37	1		9	16	1	
Ecdyonurus sp.	16	2							12		2		
Heptageniidae	3	1					2						
Rhitrogena sp.	63	2		4			52			4	1		
PLECOPTERA													
Brachyptera sp.	14			1	4								
Capnia sp.	2			2									
Chloroperla sp.	6	2		2			2						
Dictyogenus alpinum	178	4	5	2	1	1	6	1	3	129	2	11	4
Leuctra sp.	76	1	1	12	3	2	7		15	5	8	15	7
Perloidae	128	1		3						124			
Nemoura sp.	196		1	8	2	5	16	12	5		7	25	16
Nemurella pictetii	31			1	22	3	5						
Protonemura sp.	51	3	37	131	9	23	59	16	21	1	65	3	142
Siphonoperla sp.	1	1											
TRICHOPTERA													
Allogamus sp.	98	9		5		46	2	3	2			7	6
Crynus sp.	8												8
Drusus sp.	281		6	1			35	212		18			9
Glossosoma sp.	39												39
Goeridae	3						1			2			
Hyporhyacaphila sp.	7									6			1
Holocentropus sp.	5					1			1				3
Limniphilinae	15					3						12	
Melampophylax sp.	127					125						2	
Microsema sp.	2					2							
Philopotamus sp.	1			1									
Plectrocnomia sp.	1						1						

Polycentropodidae	11			1		3	3	1			2	1	
Rhyacophiladae	9		2							7			
Tinodes sp.	1	1											
Wormaldia sp.	2								2				
Total tax richness		**12**	**7**	**15**	**7**	**12**	**14**	**7**	**8**	**10**	**8**	**9**	**10**

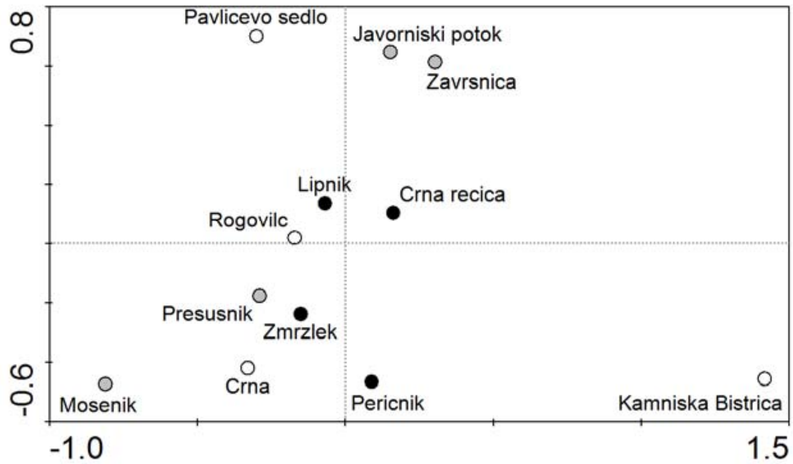

Figure 3.3.2.5: PCA ordination diagram showing similarity between the springs based on EPT taxa. First axis explained 26.3% and second 18.6% of variability in data.

Literature cited

Kanduč, T., Mori, N., Kocman, D., Stibilj, V., Grassa, F., in review: Hydrogeochemistry of Alpine springs from North Slovenia: insights from stable isotopes. Chemical geology, in review.

Wood, P.J., Gunn, J., Smith, H. and Abas-Kutty, A. (2005). Flow permanence and macroinvertebrate community diversity within groundwater-dominated headwater streams and springs. Hydrobiologia. 545: 55-64.

3.4 Jauntal

Summary

The biocenosis of 21 head waters and two rivers known to be intermittent as well as a comparable permanent river were analysed. Chemical and physical parameters such as temperature, pH, conductivity and oxygen content were collected as well as the flow rate and mean flow velocity. Samples of macrozoobenthic organisms and phytobenthos at some selected waters were taken in 2009 and 2010. All organisms' collected, were determined to the maximum possible lowest level. Regarding the macrozoobenthic composition of spring waters a difference between pore and fissure springs were identified. The main factors that are considered to affect the macrozoobenthic biocenosis are the type of spring (pore-/fissure spring), the size of spring and the anthropogenic impact. Nutrient parameters seem to play a less important role. For some species correlations between the abundance and flow rate respectively flow velocity were calculated. A difference between pore and fissure springs were found, which is partly due to the different species inventory. 64 taxa showed correlations over 0.5 (Spearman) with the flow or the average flow velocity. 45 of these were relevant for pore springs, 21 taxa showed a significant correlation with the flow and 22 taxa showed a correlation with the average flow velocity. A high positive correlation (over 0.7) with the flow rate shown in pore springs respectively their runoffs have *Baetis alpinus*, *Baetis rhodani*, *Protonemura auberti*, *Limnius* sp. *Elmis* sp. and *Tinode dives*. *Electrogena*

ujhelyi, *Electrogena* sp., *Leuctra braueri*, *Rhyacophila laevis Wormaldia occipital*, *Parametriocnemus stylatus*, *Stempellinella brevis* and *Ibisia marginata*, however, showed a negative association. High positive and significant correlation with the average flow velocity can be calculated for pore springs respectively their runoffs for *Gammarus fossarum*, *Hydrachnidia* gen. sp., *Baetis rhodani*, *Rhithrogena* sp., *Rhyacophila* s.str., *Tinode* sp., *Simulium costatum.* A negative correlation is shown for *Electrogena ujhelyi*, *Electrogena* sp., *Wormaldia occipital*, *Parametriocnemus stylatus*, *Stempellinella brevis* and *Ibisia marginata*. In fissure springs and their runoffs *Ecdyonurus* sp., *Dictyogenus alpinum*, *Dictyogenus fontium*, *Rhyacophila producta* and *Drusinae* gen. sp. are positively correlated with the amount of flow. A higher positive correlation for the flow velocity was found for *Ecdyonurus* sp. *Dictyogenus alpinum*, *Drusinae* gen. sp., *Drusus monticola* and *Tvetenia calvescens*. Since the majority of species determined are no real spring inhabitants, but euryöke organisms such as *Baetis alpinus*, the correlations refer more to large or small flowing waters. By means of taxa a separation in pore and fissure springs appears and more over, due to their size, these types are sub grouped with regard to the flow rate. Analysing the entire settlement of benthic organism a distinction between smaller and larger pore spring discharges are identified. Hydro morphological similarity of the spring waters and their geographical proximity to each other seem to be responsible for higher correlations. Smaller and larger fissure headwaters on the other hand could not be distinguished. However in strongly anthropogenic influenced spring runoffs (overhead reservoir drainage or heavily modified and dammed spring waters) a biocenotic grouping was found, which in its composition have more in common with small pore springs. It should be noted that for some taxa a significant correlation with the size of waters occurred. These organism can in term of spring water discharge function as indicator and thereby provide evidence for a water withdrawal. In addition, the type of spring, whether pore or fissure and the geographical position and the altitude is crucial for the settlement. It can be shown that a capacity between 10 and 30 liters per second defines a transition zone that due to the benthic population separates a bigger source (> 10 to 30 l/s) from a smaller source (<10 to 30 l/s). An "optimal ecological flow" is given for small pore springs, if the drainage is under this limit and for larger pore springs, when the flow is above this limit. It is more difficult to define a discharge for fissure springs, even if the indicators alone show at a lower flow rate a difference, but considering the total biocenosis no clear shift can be determined. In the study period, the selected reference waters for intermittent streams, River Sucha and River Wackendorf did not dry up. A grouping of these two streams, due to their macrozoobenthic population is rather done by their geographic proximity or by their own headwaters. As an intermittent spring one arm of a periodic headwater was examined. The macrozoobenthic composition showed the highest correlation with the adjacent permanently flowing arm. The phytobenthos showed a small proportion of macro-algae. In pore headwaters with higher nitrate values the eutraphente (nutrient-rich waters favouring) red algae *Hildenbrandia rivularis* could always be found.

Detailed investigations

3.4.1 Invertebrates

The aim of this study was to allow conclusions regarding the optimal and minimal runoff of headwater streams by means of makrozoobenthos as indicator. In addition the benthic biocenoses from intermittent rivers (periodic waters) get compared with permanently flowing ones. The results should help to create a water body specific management. In consultation with the experts of Dept. 8 headwaters with different water conditions and intermittent rivers of the Jauntal were selected to achieve this goal. Pore springs of the Jauntal as well as fissure springs at the foot of the mountain chain Karawanken were chosen. Beneath the species composition and their abundances hydro morphological parameters like flow velocity, slope, water temperature and physical-chemical parameters were collected. Faunistic studies and eco- or hydro morphological reviews of sources and their drains are abundant in the literature and are sometimes an excellent preservation of evidence and representation (e.g. WEIGAND & GRAF, 2007; WAGNER & GERECKE, 2008: GERECKE et al, 2005; CIANFICCONI et al, 1998; CREMA et al, 1996, NATI CANTO, 1998..). FISCHER (1996a) and ROLLOVER (1997), deal with the benthic biocenosis for ecological evaluation of the springs and also describes revitalization projects. Already a zoning of rivers was distinguished by ILLIES (1961), which is until today considered as relevant and refers to the definition of the terms "fish region" or "biocenotic region". The spring area is distinguished in the eukrenal and hypokrenal zone. The eukrenal describes the habitat of the immediate spring area, while the hypokrenal represent the communities of the headwater runoff. Usually a spring is characterized by an relatively constant drainage and a constant cold temperature. The macrozoobenthic organisms living their are cold stenotherm animals (FISCHER, 1996). Springs can be divided into biocenotic types as shown in Table 3.4.1.1 (SCHOENBORN, 1991, "Bavarian spring type catalogue" 2004). In the present study rheocrenes (free flowing sources) as well as the runoffs of tapped springs and overhead reservoirs were recorded.

Table 3.4.1.1: Spring types.

Basic Type	Characteristic Substrate
Flow Spring	organic
(= Rheocrene)	fine material
	large material
	block material
Case Spring	Rock
Seepage Spring	organic
(= Helocrene)	fine material
	large material
Linear Spring	organic
	fine material
	large material
Pool Spring	organic
(= Limnocrene)	fine material
	large material

The term "ecological optimum discharge" was pre-defined by the work group as discharge pattern in terms of minimum base flow requirements and the timing, duration, magnitude and frequency of high flow and flood events most suitable for a creating sustainable habitat conditions for resident biota under different management strategies and climate change."After an initial evaluation of the data some adoptions had to be done. Some headwaters were not sampled a second time, further investigations on the River Sucha were stopped and therefore the river Wackendorf was added to the program.The collected chemical and physical parameters at each location were fairly constant over the season. However, the drainage situations between the springs at the bottom of the Jauntal and in that of the mountain chain of Karavanken seem to be different. While the physical and chemical parameter remains the same, the drainages show quite large fluctuations. For further analysis all data sampled at different dates from the springs and their runoffs were merged and transformed to a semi-quantitative dimension. Thus, for each location a total taxa list with a semi-quantitative frequency list results. The number of total taxa are summarised in Tab. **3.4.1.2** together with the abbreviations of the springs or headwaters.

Tab. 3.4.1.2: Abbreviations of the spring name and number of total taxa.

Spring Name	Abbreviation	Taxa
Dobrowa	DOB	79
Dolintschitschach, undammed eastern branch	DOLU	94
Dolintschitschach, dammed western branch	DOLV	64
Draurain	DRA	113
Einersdorf	EIN	22

Feistritz	FEI	51
Graben	GRA	104
Hainsch	HAI	60
Kanauf	KANN	44
Kuschnig	KUS	33
Kutschej, periodic branch	KUTi	66
Kutschej, permanent branch	KUTM	55
Luscha	LUS	34
Peratschitzen	PER	74
Podrain, below the overhead reservoir	PODHB	54
Podrain, under the tapping	PODFA	46
Pribelsdorf	PRI	54
Spring west from Pribelsdorf	PRIW	84
Wackendorf	WAC	71

All statistical evaluations were performed using software Winstat 3.1. The matrix of the above-mentioned transformation was summarized by means of cluster analysis in terms of semi quantitative taxa composition. It yielded the following results, shown in Figure 3.4.1.1. In total, due to taxa composition, two large clusters of pore and fissure springs (lower branch: pore springs, upper branch: fissure springs) are distinguished. The lower branch of the cluster in Figure 3.4.1.1 consists of two sub clusters, which can be characterized mainly just from the larger and smaller spring discharge at the bottom of the Jauntal just before the river Drau. The springs Dobrowa, Draurain, Graben and Peratschitzen belong in terms of their settlement into one group, while the spring west of Pribelsdorf and the periodically and permanently water-bearing part of spring Kutschej are in the second sub cluster. It is interesting to see that the natural and unaffected part of the spring Dolintschitschach at the foot of the mountain chain Karawanken also belongs in the aggregation of pore springs. According to this Mr. Schlamberger (Dept. 8) confirmed the pore spring nature of the spring Dolintschitschach.

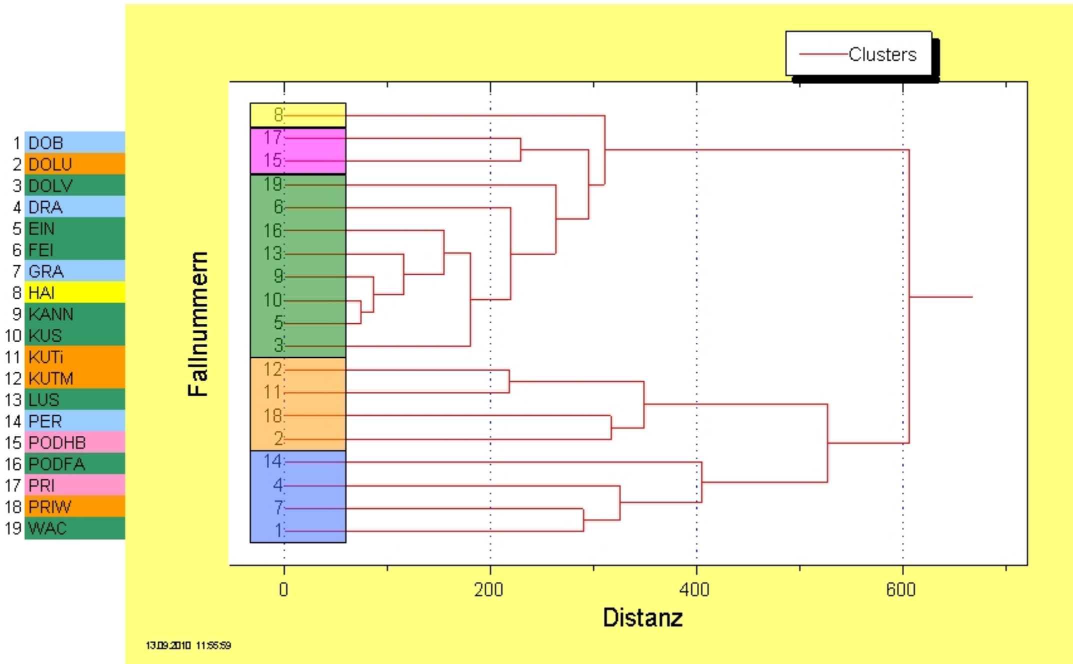

Figure 3.4.1.1: Cluster analysis of the similarity in the macrozoobenthic settlement. Legend: see Tab. 3.4.1.2

The upper part of the cluster (fissure springs) provides a slightly more mixed picture. Again, two main sub clusters can be distinguished. There are, de facto all springs and their runoffs at the foot of the Karavanken, but also the spring Kuschnig and Einersdorf together in one aggregation. A little further away the discharge of the overhead reservoir from the spring of Podrain is associated together with the spring of Pribelsdorf, which is located at the bottom of the Jauntal. The spring Hainsch appears isolated (see Figure 3.4.1.2).

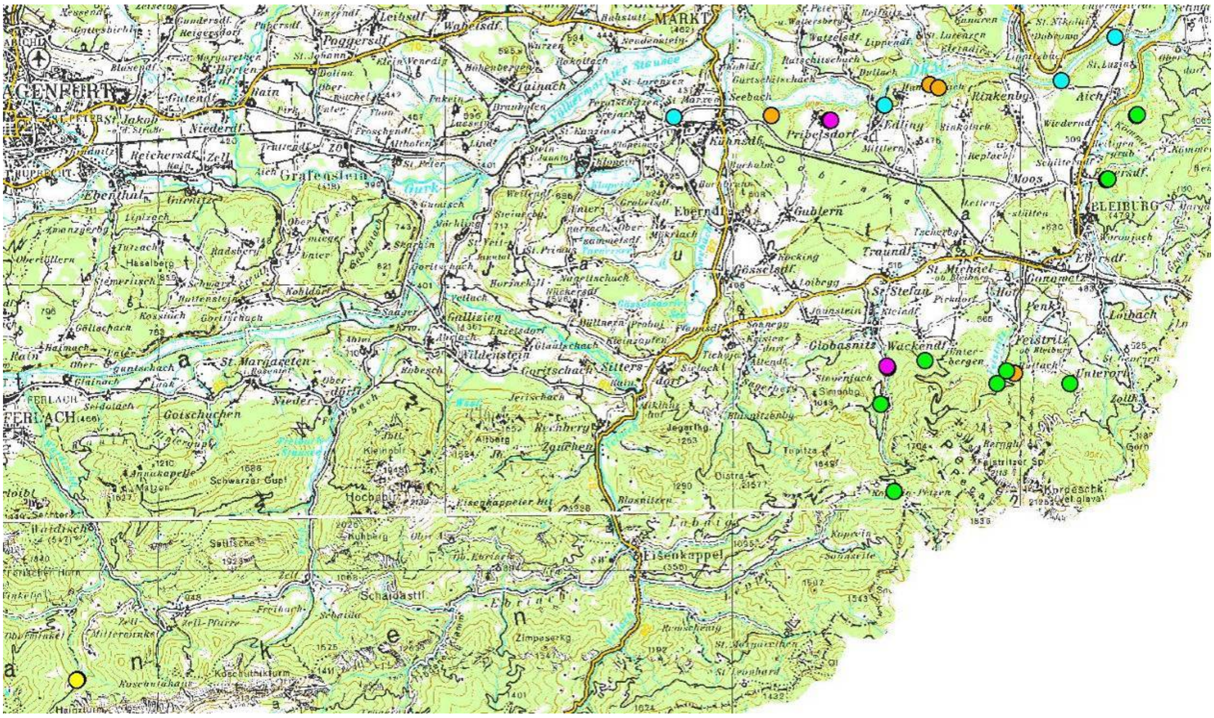

Figure 3.4.1.2: Geographical representation of the result of cluster analysis of the similarity in the macrozoobenthic settlement.

Such a separation of source types was also carried out by GERECKE et al. (1998), there rheocrenes from helocrene and periodical sources on the basis of the meiofauna can be clearly distinguished. In this context WEIGAND (1998) found in the National Park O.ö. Kalkalpen a high faunistic similarity between the main choriotopes. As is done by MAIOLINI & SILVERI (2010), the similarity between the pilot sites of this study was shown by means of the EPT (Ephemeroptera – Plecoptera – Trichoptera) taxa (Figure 3.4.1.33). The group of pore springs Dobrowa-, Draurain-, Graben- and Peratschitzen or the untouched Dolintschitschach, the western part of Pribelsdorf, as well as the two Kutschej springs remains evident under this aspect. The fissure spring cluster is composed differently. The spring of Hainsch is now associated with the spring Feistritz, the Podrain below the spring tapping and the modified Dolintschitschach. The remaining fissure springs now are together with the overhead reservoir of spring Podrain and the spring Pribelsdorf.

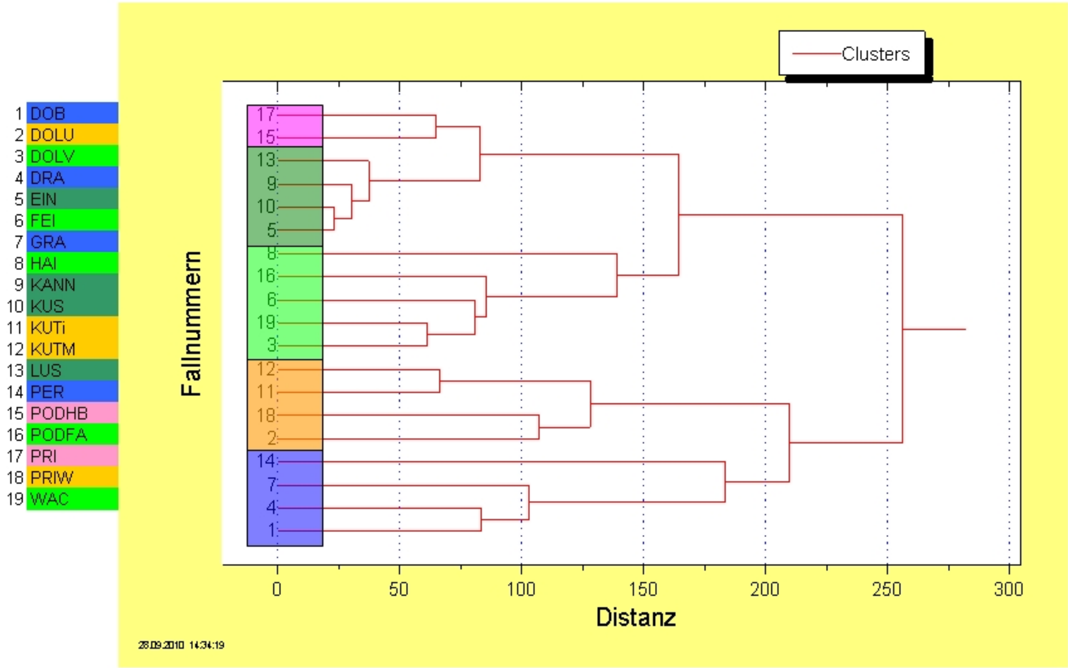

Figure 3.4.1.3: Cluster analysis of the similarity in the settlement macrozoobenthic: EPT. Legend: see Tab. 3.4.1.2

To show the similarities between the sources or headwaters the correlation was calculated by using the Spearman's rank correlation (see annex). The greatest similarities in the composition of taxa is between the pairing of springs Kanauf and the dammed Dolintschitschach (R=0.65), followed by the springs Wackendorf and the dammed Dolintschitschach; Wackendorf and Podrain below the tapping, Graben and Draurain (R=0.58), further the permanent and periodic branch of Kutschej (R=0.56) as well as Pribelsdorf-West and the undammed Dolintschitschach (R=0.53) and the spring Draurain with Dobrova (F=0.52). The above-mentioned greater similarities are in (see annex) marked with blue, similarity higher than 50 % are yellow. It is noticeable, that those sites with only one investigation in spring time show less relation or similarities. Therefore these springs are newly clustered (Figure 3.4.1.4). The larger pore springs Draurain, Dobrowa, Graben and Peratschitzen remain in a cluster, Pribelsdorf-West, the undammed Dolintschitschach as well as the periodic Kutschej show again greater similarities in their benthic colonization. The permanent branch of Kutschej however jumps to the spring Kuschnig respectively Einersdorf. Pribelsdorf and the Podrain below the overhead reservoir remain together. Even the isolated character of Hainsch persists. In any case it is noticeable that except the permanent branch of Kutschej the smaller pore springs of the bottom of the Jauntal now are in opposition to all others. However, in the overall picture of the cluster analysis there is no major interpretive changes found.

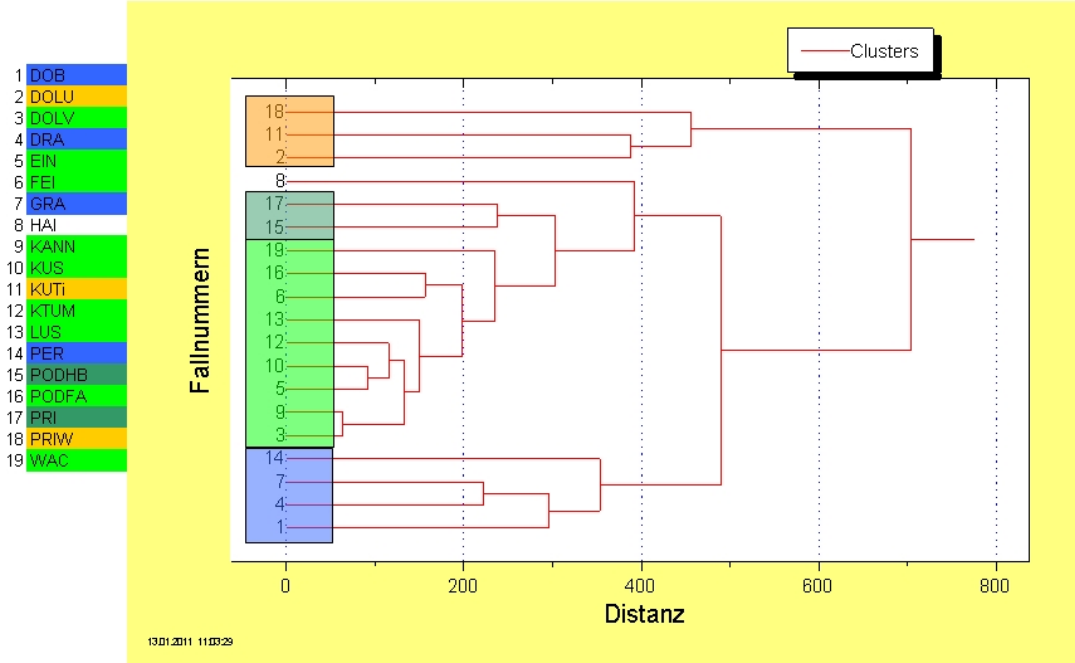

Figure 3.4.1.4: Cluster analysis of the similarity in the macrozoobenthic settlement; Only each spring date. Legend: see Tab. 3.4.1.2

In order to get information about the site and their notional variables a factor analysis was done. In this way four approximately relevant influencing variables are identified. The location of each site in the factor range corresponds more ore less with the composition of the cluster analysis (Figure 3.4.1.5). Dobrowa, Draurain, Graben- and-Peratschitzen springs (marked blue) belong together. Pribelsdorf-West, the two Kutschej springs, Kuschnig and the undammed Dolintschitschach form another group (orange). Two other groups consist on the one hand of the dammed Dolintschitschach, the Wackendorf, Luscha, Feistritz, Kanauf, Hainsch and the Podrain below the tapping (marked green) and on the other hand the somewhat loose group of Einersdorf, Pribelsdorf and the runoff from the overheas reservoir of Podrain (red).

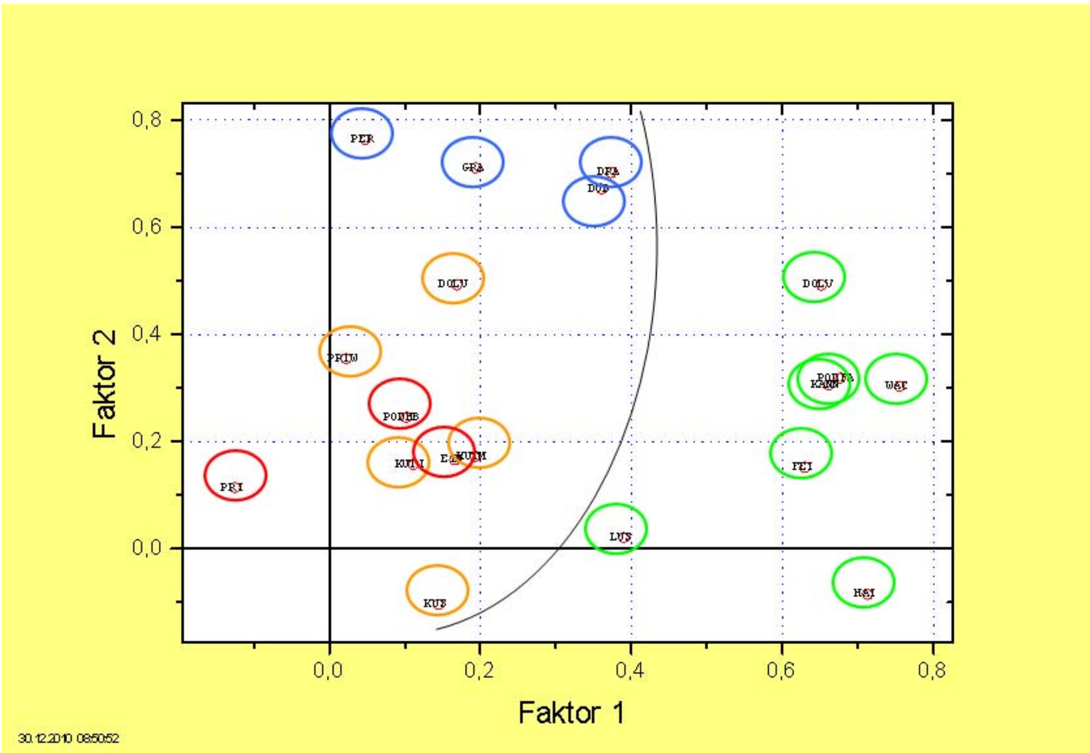

Figure 3.4.1.5: Mapping of the locations in factor range factors 1 and 2.

Along the handmade line in Figure 3.4.1.5 the springs in principle can be separated into large pores springs (left) and fissure springs (right). The red marked sites are morphologically and hydrological heavily modified waters. These are the Einersdorf- and Podrain spring, each below overhead reservoirs and both runoffs are affected and show changing substrate composition (different runoff through the year and the day, flushing and silting of the tank). The third, the Pribelsdorf Spring is anthropogen modified respectively dammed and pools are constructed. It is known from literature that changes of habitat structures are associated with changes of Biocoenosis. Thus, DUMNICKA (2006) was able to demonstrate that anthropogenic changes of the headwaters lead to a large shift of oligochaeta composition. Furthermore, it should be noted, that there is a difference in conductivity between the pore springs at the northern end of the Jauntal and those at the foot of the Karawanken. In general pore springs of course in most cases have a higher conductivity than fissure springs. Further existing chemical data (AKL, Dept. 8) especially regarding nutrient sources show even bigger differences between the pore springs (Table).

Table 3.4.1.3: Mean values of nutrient parameters of five spring waters over the period 2005-2010 (AKL, Dept. 8).

NAME	TYPE	NITRATE [mg/l]	DOC [mg/l]	ORTHO-PHOSPHATE [µg/l]	pH	WATER HARDNESS [°dH] calculat.	WATER-TEMPERATURE [°C]
DOB	Pore	10,16	0,43	3,60	7,51	17,26	10,12
DRA	Pore	4,89	0,52	6,90	7,50	16,11	9,44
GRA	Pore	8,93	0,51	3,81	7,51	16,66	9,71
PER	Pore	3,12	0,47	3,07	7,46	17,10	9,74
PODFA	Fissure	0,56	0,79	3,51	7,72	9,07	8,05

On the basis of site correlation the influence of conductivity is considered as less dominant than the influence of the water body type. Another significant impact might perform the anthropogenic pressure. Nutrient parameters as shown in Table3.4.1.3 play in this case only a minor role.

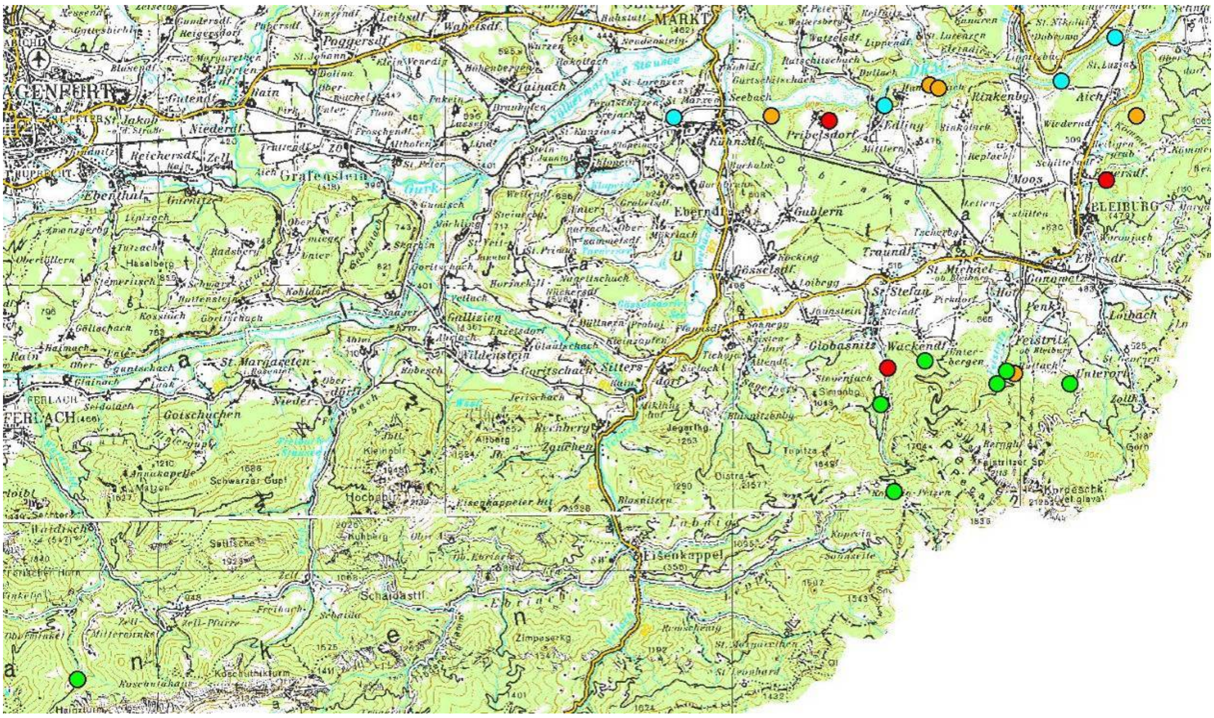

Figure 3.4.1.6: Allocation of sites for factor analysis. Colours explained in the text.

For the illustration in Figure 3.4.1.6 the same colours as in Figure 3.4.1.5 were chosen. The results of the runoff measurements or estimates are classified into size categories from 1 (low flow) to 7 (very large flow) as shown in Table 3.4.1.4. Also the average flow velocity has been categorized. The ranges are between 1 l/s (Kuschnig) to 270 l/s (Hainsch - AKL, Dept. 8).

Table 3.4.1.4: Size categories of runoff (l/s) and velocity (cm/s). Measurements, estimations or data from the Dept. 8 (Draurain and Hainsch).

Spring	l/s	Q-Class	v m	v-Class
DOB	26,3	3	30,9	3
DOLU	4,6	1	17	2
DOLV	16,6	2	23,1	2
DRA	43	3	27	3
EIN	2,2	1	7,7	1
FEI	50	3	35	4
GRA	55	4	39,9	3
HAI	267	7	35	4
KANN	40	3	45	5
KUS	1,2	1	12	2
KUTi	4,8	1	8	1
KUTM	5,4	1	15	2
LUS	10	2	15	2
PER	70	4	21,1	2
PODHB	3,7	1	10	1
PODFA	120	6	50	5
PRI	6	1	20	2
PRIW	1,9	1	9,4	1
WAC	90	4	50	5

Next the individual taxa were set in connection with the flow rate. In this process only taxa were considered, which also occurred with more frequency - which were detected at least four times. Furthermore pores- were separated from fissure springs, because of previous results and of differences in altitude. Finally 34 taxa were estimated that have a correlation of more than 0.5 Spearman Rank Correlation Coefficient (SRCC) with the flow rate or the average flow velocity. 21 taxa showed a significant correlation with the flow rate and 22 taxa showed a correlation with the average flow velocity. Thereby positive and negative correlations appear. *Baetis alpinus*, *Baetis rhodani*, *Protonemura auberti*, *Limnius* sp., *Elmis* sp. and *Tinode dives* show a high correlation (> 0.7) with the flow rate in the pore springs or their runoffs, against this *Electrogena ujhelyi*, *Electrogena* sp. *Leuctra braueri*, *Rhyacophila laevis*, *Wormaldia occipital*, *Parametriocnemus stylatus*, *Stempellinella brevis* and *Ibisia marginata* show a negative relationship. In the fissure springs *Ecdyonurus* sp., *Dictyogenus alpinum*, *Dictyogenus fontium*, *Rhyacophila producta* and *Drusinae* gen.sp. are positively correlated with the flow rate. Since flow velocity and flow rate are factors that depends on river morphology, the SRCC for the average flow rate was calculated too. High positive and significant correlations exist for pore springs and their runoffs for *Gammarus fossarum*, *Hydrachnidia* gen.sp., *Baetis rhodani*, *Rhithrogena* sp. *Rhyacophila* s.str., *Tinode* sp. and Simulium *costatum*, negative correlations for *Electrogena ujhelyi*, *Electrogena* sp. *Wormaldia occipital*, *Parametriocnemus stylatus*, *Stempellinella brevis* and *Ibisia marginata*. For the fissure springs and their runoffs a higher positive relationships is demonstrated for *Ecdyonurus* sp. *Dictyogenus alpinum*, *Drusinae* gen.sp., *Drusus monticola* and *Tvetenia calvescens*. Especially *Leuctra braueri* shows a very high negative correlation in pore springs, with respect to the size of the system respectively the flow rate. This species is high abundant only in very small waters. Conversely, the tendency is also very clear in *Baetis alpinus* and *Baetis rhodani* that are no longer found in smallest water bodies. For the flow velocity, the highest SRCC is achieved by *Gammarus fossarum*. The latter three species mentioned are no classic organism of springs and their runoffs and they additional have a high ecological valence.

Table 3.4.1.5: List of Taxa – biological indicator - with higher and significant correlations with flow rate (Q) and mean flow velocity (v).

Pore Springs - Q		**Fissure Springs - Q**	
Positive	negative	positive	negative
Oligochaeta gen.sp.	Stylodrilus sp.	Crenobia alpina	-
Baetis alpinus	Electrogena sp.	Bythinella sp.	
Baetis rhodani	Electrogena ujhelyi	Hydracarina gen.sp.	
Rhithrogena semicolorata	Leuctra braueri	Baetis sp.	
Protonemura auberti	Rhyacophila laevis	Ecdyonurus sp.	
Elmis sp.	Synagapetus krawanyi	Dictyogenus alpinum	
Limnius sp.	Wormaldia occipitalis	Dictyogenus fontium	
Tinodes dives	Heleniella ornaticollis	Protonemura sp.	
	Parametriocnemus stylatus	Rhyacophila producta	
	Stempellinella brevis	Rhyacophila stigmatica	
	Simulium sp.	Drusinae gen.sp.	
	Ibisia marginata	Drusus monticola	
	Limoniidae gen.sp.	Eukiefferiella fittkaui/minor	
		Rheotanytarsus sp.	
		Tvetenia calvescens	
Pore Springs - v		**Fissure Springs - v**	
positive	negative	positive	negative
Gammarus fossarum	Electrogena sp.	Crenobia alpina	-
Gammarus sp.	Electrogena ujhelyi	Gammarus fossarum	
Hydracarina gen.sp.	Leuctra braueri	Baetis sp.	
Baetis alpinus	Rhyacophila laevis	Ecdyonurus sp.	
Baetis rhodani	Wormaldia occipitalis	Dictyogenus alpinum	
Rhithrogena sp.	Parametriocnemus stylatus	Dictyogenus fontium	
Isoperla goertzi	Stempellinella brevis	Rhyacophila producta	
Nemoura sp.	Simulium sp.	Rhyacophila stigmatica	
Limnius sp.	Ibisia marginata	Drusinae gen.sp.	
Rhyacophila fasciata		Drusus monticola	
Rhyacophila s.str.		Eukiefferiella fittkaui/minor	
Wormaldia copiosa		Rheotanytarsus sp.	
Tinodes sp.		Tvetenia calvescens	
Simulium costatum			

Real spring organisms named in the table above are among the caddisflies *Rhyacophila laevis*, *Rhyacophila producta*, *Synagapetus krawanyi*, *Tinode dives*, *Wormaldia occipital* and to some extent *Rhyacophila stigmatica*, *Wormaldia copiosa* and *Drusus monticola*. The named mayflies are regularly found in springs and their runoffs, but they have their maximum occurrence in rhithral zone. The stonefly *Protonemura auberti* is an inhabitant of krenal and rhithral, as it is *Isoperla goertzi* or *Leuctra braueri* that inhabit the hypokrenal and epirhithral zone like the two types of *Dictyogenus*. The freshwater shrimp *Gammarus fossarum* can be found from krenal up to potamal, *Crenobia alpina* is present in all cold rhithral waters, *Simulium costatum* and *Rhyacophila fasciata radiate* occur from the krenal zone into the top of the rhithral. The above mentioned bloodworms (chironomids) are belonging to the krenal, but also occur in rhithral and to some extent even in the littoral. Using for grouping all sites only the indicators from Table 3.4.1.5, the following picture result as it is shown in Figure 3.4.1.7.

l/s	Q-Klasse	v m	v-Klasse	
3,7	1	10	1	PODHB
6	1	20	2	PRI
1,2	1	12	2	KUS
10	2	15	2	LUS
2,2	1	7,7	1	EIN
90	4	50	5	WAC
120	6	50	5	PODFA
267	7	35	4	HAI
40	3	45	5	KANN
50	3	35	4	FEI
16,6	2	23,1	2	DOLV
5,4	1	15	2	KUTM
1,9	1	9,4	1	PRIW
4,8	1	8	1	KUTi
4,6	1	17	2	DOLU
70	4	21,1	2	PER
55	4	39,9	3	GRA
43	3	27	3	DRA
26,3	3	30,9	3	DOB

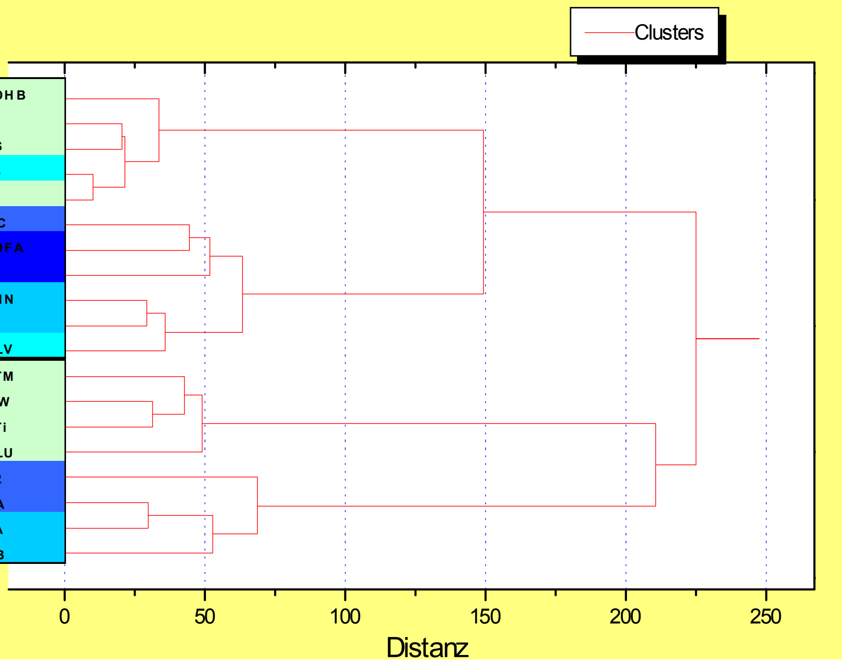

Figure 3.4.1.7: Grouping of spring waters with only the detected indicators.

Resuming the results, first a separation in pore and fissure spring appears and second the two groups are sub grouped according to their size - flow rate. In the upper sub cluster are small separated from larger fissure springs; in the lower sub cluster it is the same for the pore waters. In Figure 3.4.1.7 are beneath the grouping visualisation the flow rates and flow velocities and their classification is given. In summary it can be said that according to the above results, significant correlations are given for some taxa with the size of water and thus they can act as indicator for the tapping of springs. The crossover between larger and smaller headwaters should be carefully formulated in between 10 to 30 l/s. Additional the type of spring (pore- / fissure-), the geographical position and the altitude is to take into consideration too. This means that on the basis of this analysis the term "ecological optimum discharge" have to be applied regarding the above mentioned parameters, especially as fissure springs with a low flow rate have less impact than pore spring. The selected, normally intermittent streams Sucha and Wackendorf unfortunately did not dry out during the project period. Thus, the results obtained are not evaluated in this regard. Nevertheless, an attempt was made to compare the data with other, permanent waters.Among the springs or headwaters two of the periodic sources, however, remained dry in the entire study period. In the area of the Kutschej Spring, a permanent and a periodic branch were visible. The periodic part was frozen throughout the winter and showed at least no surface water. After the melting of snow, the water turned back and at the end of March 2009 two juvenile Nemouridae were found. The results of the investigation on river Sucha were compared with other streams and rivers in the bioregion Southern Alps of the center-alpine basin. Both have water infiltration routes and lose in their flow through the Jauntal water. The river Wackendorf seeps away by several large anthropogenic basins. The benthic species composition of the River Sucha and the River Wackendorf is similar to the composition of the nearby permanently water-bearing streams of the spring Wackendorf and the River Globasnitz. A cluster analyse put these sites together with three study sites of the River Globasnitz: at detention reservoir, at Traundorf (village) and above the village Globasnitz. The abundances of benthic animals are highly variable due to the different seasons. But the number of taxa and the number of families of the comparable sites show similar patterns with the rivers Wackendorf, Sucha and Globasnitz (Figure 3.4.1.8 and Table 3.4.1.6).

Table 3.4.1.6: Number of taxa and families at the compared sites of river Globasnitz at detention reservoir (GLORH), above the village Globasnitz (GLOOHGLO), at the village Traundorf (GLOTRA) and of the river Suche (SUHA), the Wackendorf spring runoff (WACQ), and the river Wackendorf at the beginning of the valley Jauntal (WACB).

	GLORH	GLOOHGLO	GLOTRA	SUHA	WACQ	WACB
Turbellaria	1	1	1	1	1	0
Nematoda	1	1	1	1	1	1
Mollusca	0	1	0	1	3	4
Oligochaeta	11	6	9	9	4	11
Crustacea	2	1	1	3	3	0
Hydrachnidia	1	1	1	1	1	1
Ephemeroptera	7	7	9	10	7	7
Plecoptera	10	14	5	4	7	6
Coleoptera	8	7	7	7	4	3
Trichoptera	11	14	12	17	13	13
Chironomidae	15	21	20	17	14	25
Simuliidae	4	3	4	3	0	0
remain.Diptera	10	9	11	5	7	11
Gesamt	81	86	81	79	65	82

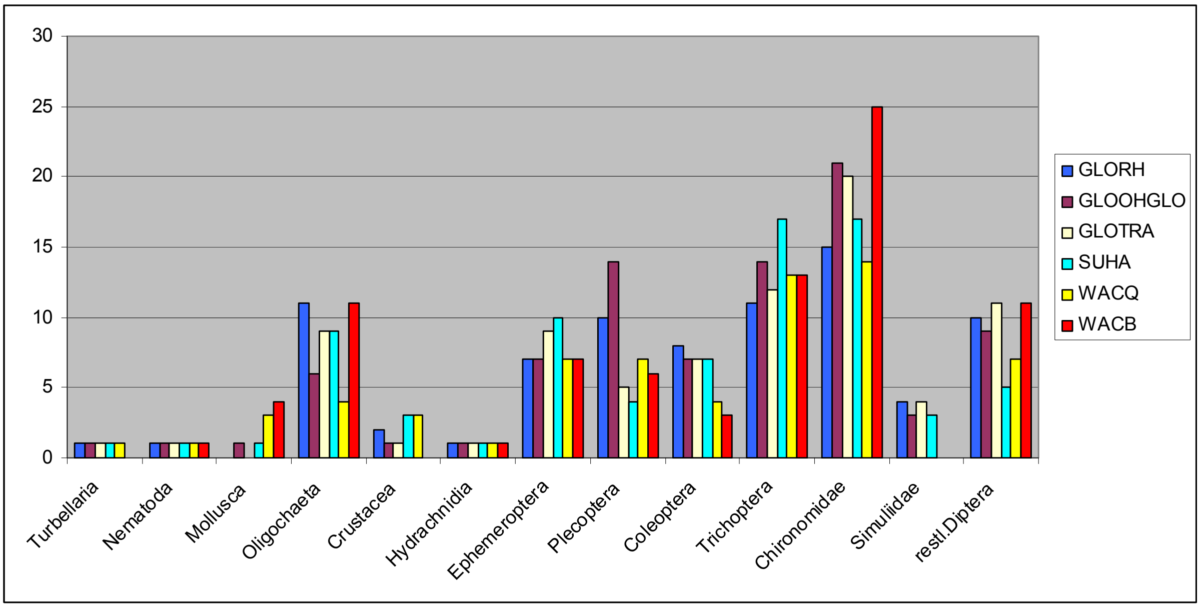

Figure 3.4.1.8: Numbers of taxa and families at the compared points Globasnitzbach at detention reservoir (GLORH9), above the village Globasnitz (GLOOHGLO), at the village Traundorf (GLOTRA); Sucha (SUHA) and the Wackendorf spring runoff (WACQ) as well as the river Wackendorf at the beginning of the Jauntal (WACB).

The geographic proximity for the macrozoobenthic colonization of rivers has obviously a major influence, like it could be demonstrated for the Kutschej springs (Fig. 3.4.1.9). The two branches of the spring Kutschej had in its permanent branch a number of 55 taxa and in its periodic 66 taxa. Even there is no surface runoff in winter the re-colonization is obviously very good, so that more taxa occur than in the permanent water-bearing branch. In particular the total number of Chironomidae, Trichoptera and "remaining" Diptera of the periodic branch are higher than in the permanent one. Of zoological interest is the fact that in both branches of the spring Kutschej the stonefly *Perla marginata* was found only.

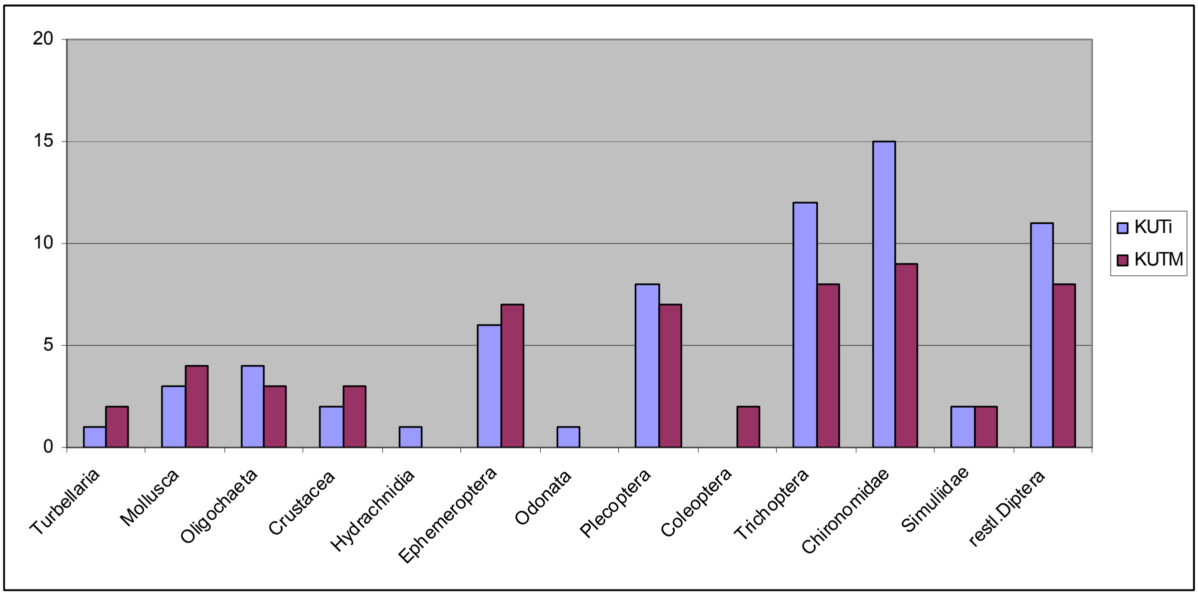

Fig. 3.4.1.9: Number of taxa of each family found in the two branches of spring Kutschej: KUTi: periodic branch, KUTM: permanent branch.

3.4.2 Periphyton

In the autumn of 2009 Phytobenthos surveys were made at 9 sites. For further analysis, the results of those springs and their runoffs were put together. This was done, because in some samples only few algae were found. Probably the methodology (semi-quantitative sampling of areas with periodic fluctuations) is more responsible for the few algae found than the natural variance. As an example the upper branch of the spring system Graben can vary its river bed very much depending on the water delivery. Similar conditions are assumed for the lower branch of this headwater. In this source stream a seasonal changing of discharge and only few species could be found. The spring outlet of the lower branch was found at the foot of a rock formation. The local biodiversity is partly due to the presence of those diatoms (*Navicula contenta*, *Navicula gallica* var. *perpusilla*, *Amphora normanii*, *Achnanthes* .) that well tolerate aerial conditions (like moist rock are). Table 3.4.2.1 records the headwaters included in the cluster analysis.

Table 3.4.2.1: Abbreviation of the spring name and the number of taxa found at one site.

Spring name	Abbreviation	Taxa number
Dobrowa-Spring	DOB	22
Draurain- Spring	DRA	28
Graben- Spring upper branch	OB GRA	24
Graben- Spring lower branch	OB GRA	23
Peratschitzen- Spring upper (western) branch	OB PER	33
Peratschitzen- Spring lower branch (at fish pond)	UN PER	21
Podrain- Spring under overhead reservoir	PODHB	34
Podrain- Spring under the spring tapping	PODFA	31
Wackendorf- Spring	WAC	33

A broad division into three groups results of this analysis (Figure 3.4.2.1). The springs of the Karavanken area with its fast-flowing runoffs (Podrain below the tapping and Wackendorf) are grouped and the spring Podrain below the overhead reservoir builds a separate cluster. This fact can probably be attributed to the morphological characteristics (including the predominantly sandy substrate) of the spring runoff. In the third group, those spring areas are combined, which are further north close to the River Drau.

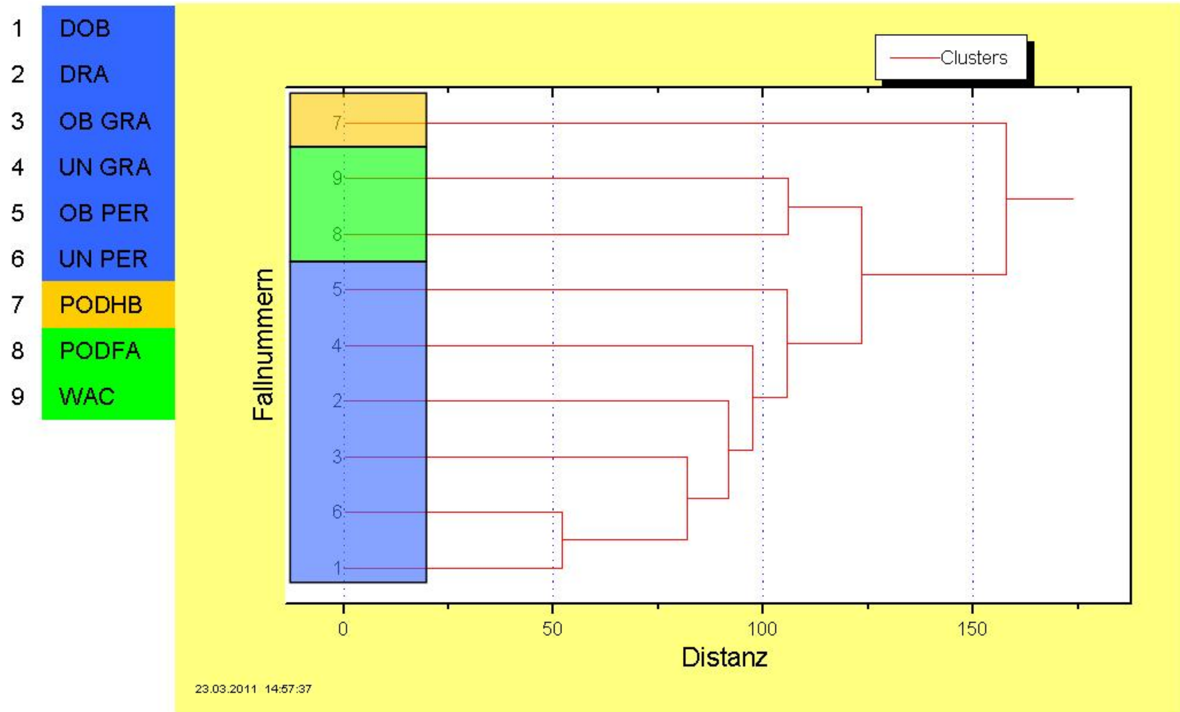

Figure 3.4.2.1: Cluster analysis of the similarity of algae growth

Comparisons of the chemical-physical parameters for these monitoring stations show higher values of nitrate, which significantly seems to affect the composition of algae (see 3.4.2.2).

Table 3.4.2.2: Values (Trophic indices, Saprobie indices) of Algae biocenosis (SI saprobie index, TI Trophic index, N nitrogen).

	Dobrowa	Draurain	Lower Peraschitzen	Upper Peraschitzen
SI Taxa total	15	19	14	19
SI Taxa KA	11	17	14	17
SI KA	1,87	1,77	1,74	1,59
SI total	1,87 II	1,77 II	1,74 I-II	1,59 I-II
TI Taxa total	14	20	14	19
TI Taxa KA	12	19	14	18
TI KA	2,5	2,51	2,33	1,98
TI total	2,5	2,51	2,33	1,98
Classification	eutroph	eutroph	eutroph	meso-eutroph
TI Taxa N	12	13	8	12
TI N	2,54	2,82	2,29	1,90
Classification	eutroph	eutroph	eutroph	mesotroph

In particular, the macro algae *Hildenbrandia rivularis*, which were regularly found in the headwaters of the first four sampling points (Table 3.4.2.2), is considered to be eutraphent, both in terms of the content of total phosphorus and nitrate. The averages of the 5-year period (as well as the 90-percentile) for orthophosphate were all in the status "very good" as it is defined for surface waters (water framework directive). Regarding nitrate content, however, it was a different picture: the spring Podrain below the tapping ranks in the "very good" condition, the remaining spring areas lay above the limits set for a "very good"

(Peraschitzen) or for a "good" condition (springs: Graben, Draurain and Dobrowa). Quantitative analysis of algae biocoenosis can be used to calculate the trophic level and Saprobie index (Table 3.4.2.2 and Table 3.4.2.3).

Table 3.4.2.3: Values (Trophic indices, Saprobie indices) of Algae biocenosis (continuation of Table 3.4.2.2).

	Lower Grabenquelle	**Upper Grabenquelle**	**Lower Podrain**	**Upper Podrain**	**Wackendorfer**
SI Taxa total	15	12	24	25	21
SI Taxa KA	14	11	19	22	19
SI KA	1,37	1,54	1,48	1,49	1,35
SI total	1,37 I-II	1,54 I-II	1,48 I-II	1,49 I-II	1,35 I
TI Taxa total	15	14	25	25	22
TI Taxa KA	14	14	22	23	21
TI KA	1,45	2,29	1,48	1,34	1,58
TI total	1,45	2,29	1,48	1,34	1,58
Classification	Oligo-mesotroph	Meso-eutroph	Oligo-mesotroph	oligotroph	Oligo-mesotroph
TI Taxa N	9	7	16	16	14
TI N	2,37	2,80	1,45	1,04	1,77
Classification	eutroph	eutroph	mesotroph	mesotroph	mesotroph

It is to note that headwaters and their runoffs are methodologically classified as extreme running waters types - or as special types. Algae of such ecosystems usually do not, or only rarely, occur in rivers and therefore are often not classified to the trophic or saprobic level. Therefore, some non-classified species which occurred frequently, were named as "sp. - taxon" so they were not included in the assessment. For some taxon no official classifications exist. With the help of literature a preliminary classification was carried out. As an example, the variations of the taxon *Cocconeis placentala* were assocoiated with the nominal variety as it was done in Switzerland by HÜRLIMANN & NIEDERHAUSER (2006). Without such a provisional classification, varieties that sometimes make up over 70% of the diatom proportion would have dropped out of the evaluation.Due to the limited extent of the classifications the results have, in terms of trophic and saprobic indices, only a limited validity. A significant proportion of biotope-specific types were determined in the samples of the springs and their approximate runoffs. The auto ecological preferences of those species described by KRAMMER & LANGE-BERTALOT (1991a, 1991b, 1997a, 1997b) in the following are briefly characterized in and pictured in the annex.

Table 3.4.2.4: Short description of diatoms found in the samples of this investigation.

Taxa and Description	Occurence
Achnanthes delicatula* ssp. *haukiana	
The environmental focus is oligotrophic, calcareous karst springs	Occur regularly to often in all investigated spring areas (3% - 25%) except of PODHB.
Achnanthes montana	
The environmental focus is in oligotrophic, alternately wet habitats on limestone and primary rocks (moss lawn, trickled over rocks).	Often found at the UN GRA (approx. 18%).
Amphora normanii	
Ecological focusn is in the mountains of wet moss and trickled over rocks, considered to be aerophile cosmopolitan.	Often found at the UN GRA (approx. 10%).
Caloneis bacillum	
Cosmopolitan, common in the mountains, especially in mosses often drizzled over.	Regularly present in all spring areas (except for OB GRA).
Diatoma mesodon	
Sometimes mass occurrences in the mountains, especially in springs are watched.	Exept DRA and PODHB always present, sometimes very frequently (PODFA)
Diploneis oblongella	
Prefers airial locations or well aerated, oligosaprobe waters	Present, but rarely, in three springs (OB PER, UN PER and PODHB);
Navicula gallica* var. *perpusilla	
Prefers moist moss, grass, rocks, often associated with *Navicula contenta*; deposits in reduced-light habitats.	Often present in OB GRA, UN GRA and WAC, otherwise rare to absent.
Navicula contenta	
Cosmopolitan, prefers the air/water interface area (springs that trickle over rocks, moss, grass). Also available in reduced light habitats.	Frequently found in OB GRA and WAC, only rarely in DRA and PODFA, otherwise absent.
Navicula ignota* var. *palustris	
Cosmopolitan, with an ecological focus under conditions of increased osmotic pressure fluctuations. Tolerates temporary drying out of habitat.	sporadic occurrence only in the spring area of WAC.
Caloneis leptosoma	
Is considered to be cosmopolitan in the springs of the mountains and preferred habitat of moss and wet rocks.	Sporadic occurrence in UN GRA,

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3.5 Noce River

Summary

The study is divided into three parts: the first focuses on the statistical analysis of the biological communities found at the two monitoring points on two reaches: the Noce Peio-Cogolo (not subject to hydropeaking) and Noce Cusiano (subject to hydropeaking). The second part of the research investigates the morphological aspects of the two reaches of the watercourse; and the third part refers to the alterations of the hydrological and thermal regimes.

The analysis of the biotic component aimed to assess the significant differences, in terms of the variety and diversity of the taxa present, between the biological communities characterising the two reaches. The Rosgen morphological index was subsequently applied to these two reaches in order to ascertain whether they belonged to different morphological types. The results for the Noce Peio-Cogolo site show that there are no significant changes between the 1998 benthic community and the successive ones. On the contrary, in the Noce Cusiano the values show an increasing level of diversity: in other words the communities in the years following 2002 are increasingly less similar.

With regard to the morphological aspects, the Peio-Cogolo reach still displays its natural characteristics and has not yet been morphologically modified by sudden increases in discharge although these could disturb it irreversibly. The Cusiano reach is constantly subject to hydropeaking phenomena and its morphology now seems to be in equilibrium with the changing rates of discharge.

In general, it seems that these results are consistent with the current state of the two reaches.

The third part of the research assessed the alteration of hydrological and thermal regimes in conditions of potential water scarcity due to the operations of storage hydropower plants with hypolimnetic releases, in the perspective of developing indicators for optimal ecological flow recommendations. The research was conducted in several reaches of the Noce Stream watershed. Firstly, the peaking events were precisely identified and their duration, intensity and time distribution was estimated. Secondly, the spatial analysis of the dynamics of hydro- and thermopeaking waves was studied through the development of the analytical solutions of a coupled hydro-thermodynamic deterministic model for river water temperature.

3.5.1 Macroinvertebrates

Data collection

Two sampling points along the river Noce were identified: the *Pejo-Cogolo* (named "Pejo") site covers a homogeneous stretch not affected by hydropeaking but nevertheless affected by alterations in water flow due to the return of water from the Pian Palù power plant and to natural water inputs from the watershed located downstream from the plant; the *Cusiano* site is, on the other hand, a homogeneous stretch affected by heavy hydropeaking. These sites are also monitored by the APPA for water quality (in different seasons) and for macroinvertebrate presence. For this work, the data analysed was collected during the period of vegetation growth (April to October).
The macroinvertebrate samples were collected by means of a benthos net and were stored in alcohol. Subsequent taxonomical identification was carried out in the laboratory using the Extended Biotic Index (EBI). In this Index, only the presence of the different taxa is recorded, with the quantity being defined only as a category (scarce, abundant etc). For the *Pejo-Cogolo* site, the data used have been collected annually from 1998 to 2010, while for *Cusiano* site data have been collected bi-annually since 2002.

Statistical methodology

In order to identify differences in the benthic community structure at the 2 different sites during the study period, a non-parametric analysis was carried out focused on comparisons and similarity/diversity.
The 3 methods used were the **Q** of Yule, the Similarity Index of Sorensen **S** and the **K** of Cohen, since they are both based on a contingency table, as shown below:

		sample 1	
		present	absent
	present	a	b
sample 2	absent	c	d

Table 3.5.1.1: Contingency table, 2x2

Where - **a** represents the presence in both samples

- **b** indicates only the presence in sample 2 and absence in sample 1
- **c** indicates only the presence in sample 1 and absence in sample 2
- **d** indicates the absence of taxa in both samples compared to the total list of all years

- *Yule's* ***Q*** - This is a methodology designed to measure the concordance between two possibly related dichotomous events. The following formula is used to calculate Yule's **Q**:

$$\mathbf{Q} = (a*d - c*b) / (a*d + c*d)$$

With:
Q = +1 there is perfect concordance,
Q = -1 there is an absence of concordance
Q = 0 there is indifference.

- ***S*** *of Sorensen* – This is the most commonly used index in ecology and supplies a similarity coefficient between two different structures in the biological community. The following formula is used to calculate **S**:

$$\mathbf{S} = 2*a / (2*a + b + c)$$

With:

S = +1, there is an higher similarity

S = 0, there is a lower similarity

- *Cohen's Kappa (**K**)* – This is a statistical coefficient that measures the inter-rater agreement for categorical items. The following formula is used to calculate **K**:

$$\mathbf{K} = (\theta - \theta') / (1 - \theta')$$

Where θ is the relative observed concordance among raters and θ′ is the hypothetical probability of changes in concordance. In this case, Table 3.5.1.1 is used to calculate the probabilities of each observed parameter falling in the present/absent category.

Results

Q, S and K statistical tests were applied to the data on macroinvertebrates with the aim of investigating changes in the structure of the benthonic community at the two study sites, *Pejo* and *Cusiano*, the latter being affected by hydropeaking.
Tables 3.5.1.2 and 3.5.1.3 show the presence of macro-benthonic individuals in each study site:

Loc. Pejo-Cogolo

	taxa	1998	1999	2001	2002	2003	2004	2005	2006	2007	2008	2010
P	*Isoperla*	X	X	X	X		X	X			X	X
P	*Perlodes*			X	X		X	X	X			
P	*Chloroperla*											X
P	Rhyacophilidae	X	X	X	X	X	X	X	X	X	X	X
P	Ceratopogonidae					X						
P	Chironomidae	X	X	X	X	X	X	X	X	X	X	X
P	Empididae	X	X	X					X		X	X
P	Limoniidae	X	X	X	X	X	X	X		X	X	X
P	*Crenobia*		X						X		X	X
F	Simuliidae	X	X	X	X	X	X	X	X	X	X	X
C	*Baetis*	X	X	X	X	X	X	X	X	X	X	X
C	Elmidae		X									
C	Blephariciidae									X		X
C	Lymnaeidae									X		
C	Haplotaxidae	X					X	X				
C	Lumbricidae			X					X			
C	Lumbriculidae			X				X	X			
C	Naididae			X			X	X		X	X	
C	Tubificidae											
C	Enchytraeidae						X					X
Sc	Blephariciidae									X		X
Sc	*Brachyptera*							X				
Sc	*Ecdyonurus*	X		X	X		X	X	X		X	
Sc	*Epeorus*		X	X	X		X					X
Sc	*Rhithrogena*	X	X	X	X	X	X	X	X		X	X
Sc	Brachycentridae			X								
Sc	Glossosomatidae	X										
Sc	Hydraenidae			X	X	X		X			X	
Sc	Psycodidae		X		X		X				X	X
Sr	*Leuctra*	X	X	X	X	X	X	X	X	X	X	X
Sr	*Nemoura*	X	X	X	X	X		X			X	
Sr	*Protonemura*	X	X	X	X		X	X	X	X	X	X
Sr	*Amphinemura*				X							
Sr	Limnephilidae	X	X		X	X	X	X	X	X	X	X
Sr	Sericostomatidae					X						
Sr	Tipulidae					X		X	X			
Sr	Gammaridae						X					

Table 3.5.1.2: List of fauna. Presence of macroinvertebrates at the Pejo site, listed as taxa (taxonomic level required for EBI index - Genera or Family) and grouped according to the most frequent feeding orientation. P = Predators, F = Filterers, C = Collectors, Sc = Scrapers, Sr = Shredders

Loc. Cusiano

	taxa	2002	2004	2006	2008	2009	2010
P	*Chloroperla*				X		
P	*Isoperla*	X	X		X	X	X
P	*Perlodes*	X	X				
P	Rhyacophilidae	X	X	X	X	X	X
P	Chironomidae	X	X	X	X	X	X
P	Empididae			X			
P	Limoniidae	X	X	X	X	X	X
P	*Crenobia*	X	X			X	
F	Simuliidae	X	X	X	X		X
C	Baetis	X	X	X	X	X	X
C	Helodidae	X					
C	Elmidae	X					
C	Lumbricidae						X
C	Lumbriculidae	X	X	X			
C	Naididae	X	X	X			
C	Enchytraeidae	X					
C	Tubificidae						X
Sc	*Ecdyonurus*	X	X	X		X	
Sc	*Epeorus*		X				X
Sc	Rhithrogena	X	X		X		X
Sc	Beraeidae	X					
Sc	Hydraenidae	X		X			
Sc	Psychodidae			X			
Sc	Blephariceridae						X
Sr	*Leuctra*	X	X	X	X	X	X
Sr	*Nemoura*	X	X				X
Sr	*Protonemura*	X	X			X	X
Sr	*Amphinemura*	X					
Sr	Limnephilidae		X	X	X	X	
Sr	Sericostomatidae					X	

Table 3.5.1.3: List of fauna. Presence of macroinvertebrates at the Cusiano site, listed as taxa (taxonomic level required for EBI index - Genera or Family) and grouped according the most frequent feeding orientation. P = Predators, F = Filterers, C = Collectors, Sc = Scrapers, Sr = Shredders

Figures 3.5.1.1 and 3.5.1.2 show different trends in the two study sites: at the Pejo site, benthic organisms with different feeding orientations were distributed homogeneously during the period under analysis (with the exception of 2003 and 2007), while at the *Cusiano* site there was a clear decrease in the number of individuals in all feeding groups, excluding natural predators, up until 2009.

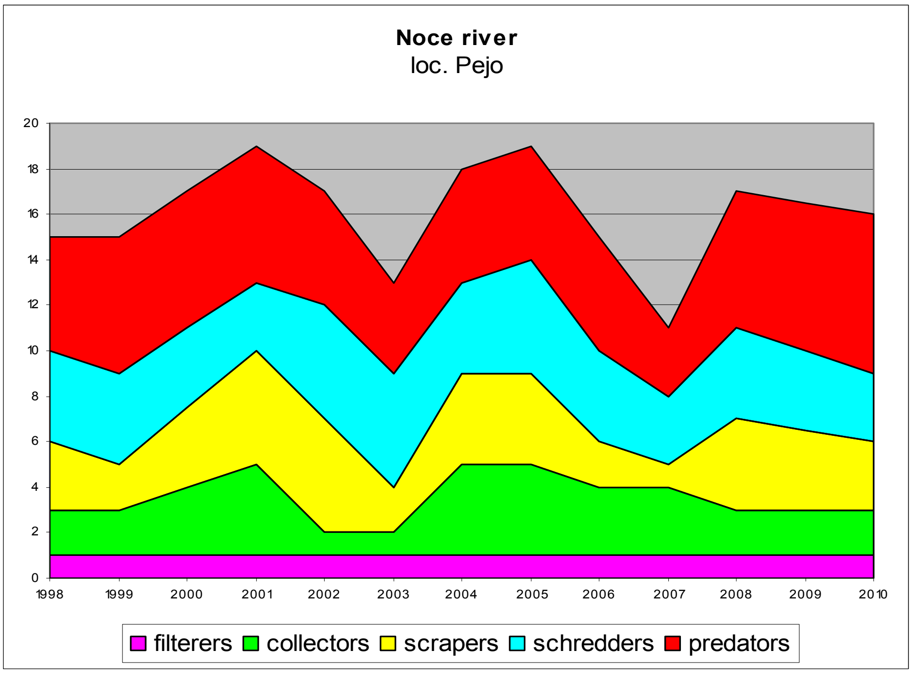


Figure 3.5.1.1: Graph showing macroinvertebrate presence according to feeding orientation over the past 10 years. The X axis represents the years, while the Y axis represents the number of taxa present in each feeding category (as shown in the key).

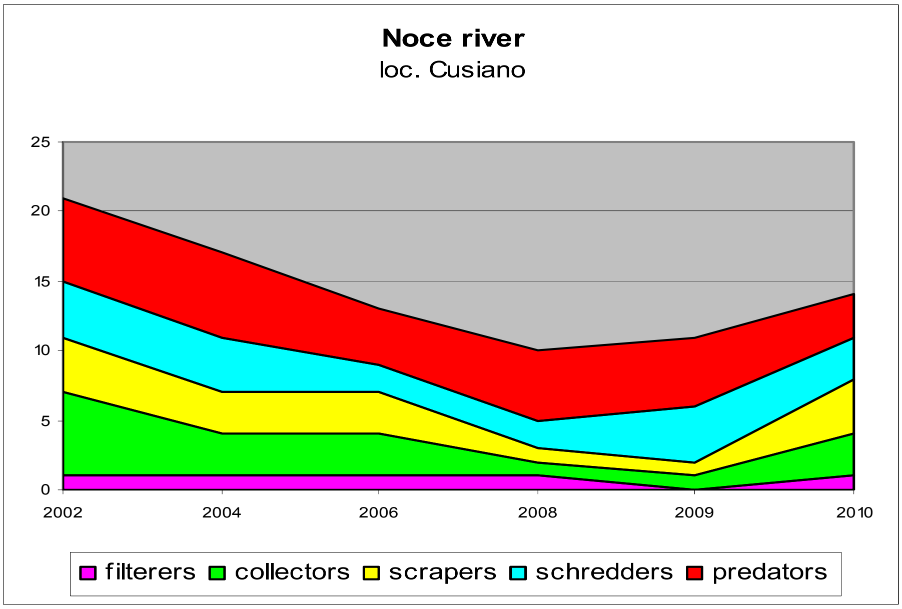


Figure 3.5.1.2: Graph showing macroinvertebrate presence according to feeding pattern over the last 8 years. The X axis represents the years, while the Y axis represents the number of taxa present in each feeding category (as shown in the key).

The older community (from 1998) was compared with the communities in the following years to verify the trend: at the Pejo site, the 1998 list of fauna was compared to the list in the following years, while for the *Cusiano* site, the oldest list dated from 2002.

Q of Yule

	1998
1999	0,89
2000	0,83
2001	0,76
2002	0,85
2003	0,73
2004	0,82
2005	0,88
2006	0,73
2007	0,75
2008	0,92
2009	0,84
2010	0,75
media	0,81
mediana	0,82
st dev	0,07

S of Sorensen

	1998
1999	0,77
2000	0,74
2001	0,71
2002	0,75
2003	0,64
2004	0,73
2005	0,75
2006	0,67
2007	0,62
2008	0,81
2009	0,75
2010	0,69
media	0,72
mediana	0,73
st dev	0,06

K of Cohen

	1998
1999	0,60
2000	0,52
2001	0,44
2002	0,54
2003	0,42
2004	0,50
2005	0,55
2006	0,43
2007	0,41
2008	0,64
2009	0,54
2010	0,44
media	0,50
mediana	0,50
st dev	0,08

Table 3.5.1.4: Q, S, K values for the Pejo site. Missing values, obtained by interpolation, are reported in red.

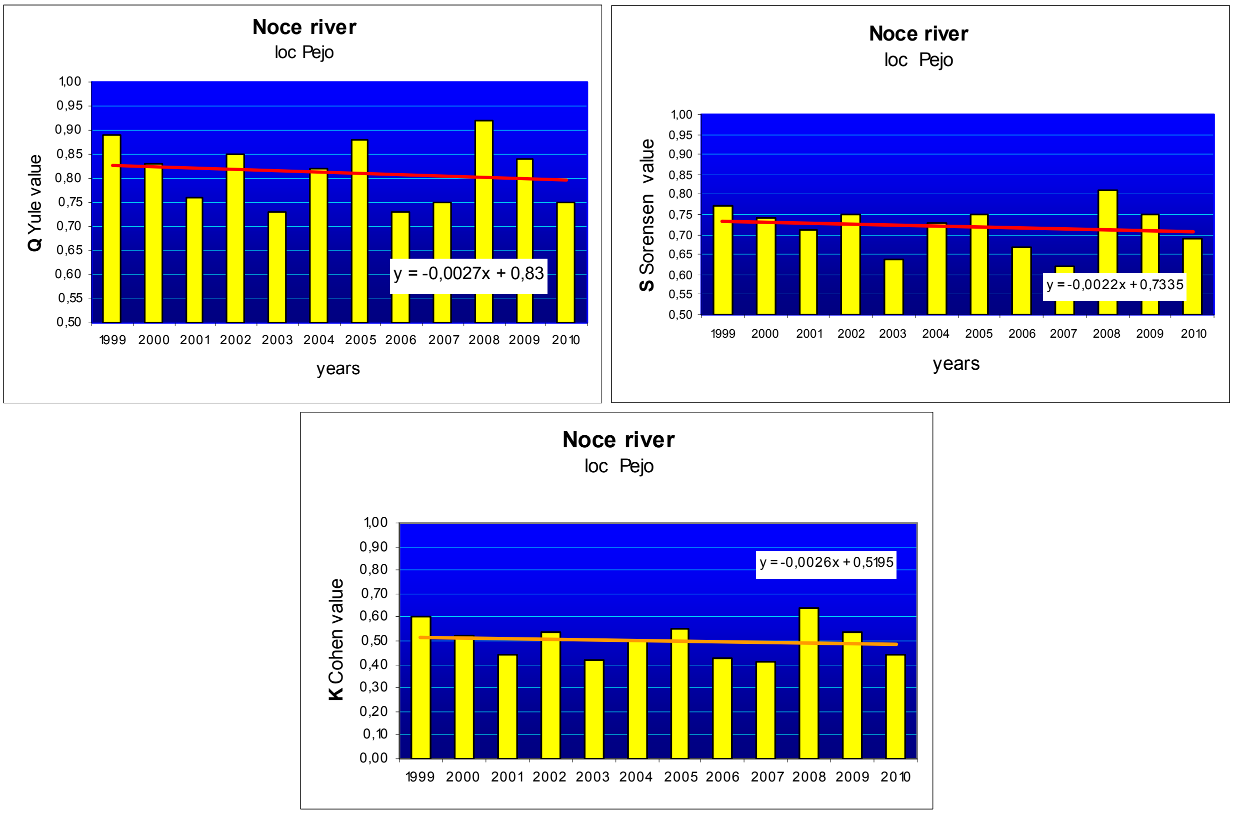


Figure 3.5.1.3: Histograms for Q, S, K tests, with best fit line

There are no significant changes between the 1998 benthic community and the successive ones, as shown in both Table 3.5.1.4 and the histograms. Moreover, the trend lines show a low angular coefficient, indicating stable tendencies.
It is important to note that the Q, S and K tests have a different range of results, so it was necessary to standardise the values of the results in order to be able to carry out a comparison between the tests. The following formula was therefore used to standardise the test results:

$$n = (x - xm) / \sigma$$

where:

n = standardised value

x = value from Table 3.5.1.4

xm = average value (media) from Table 3.5.1.4

σ = standard deviation value from Table 3.5.1.4

Table 3.5.1.4 was therefore updated with the standardised value, as shown in Table 3.5.1.5:

	Q of Yule	**S** of Sorensen	**K** of Cohen
	1998	1998	1998
1999	1,16	0,92	1,30
2000	0,26	0,38	0,23
2001	-0,79	-0,17	-0,83
2002	0,56	0,56	0,50
2003	-1,24	-1,43	-1,10
2004	0,11	0,20	-0,03
2005	1,01	0,56	0,63
2006	-1,24	-0,89	-0,96
2007	-0,94	-1,79	-1,23
2008	1,61	1,64	1,83
2009	0,41	0,56	0,50
2010	-0,94	-0,53	-0,83
media	0,00	0,00	0,00
mediana	0,11	0,20	0,00
st dev	1,00	1,00	1,00

Table 3.5.1.5: Q, S, K standardised values for the Pejo site

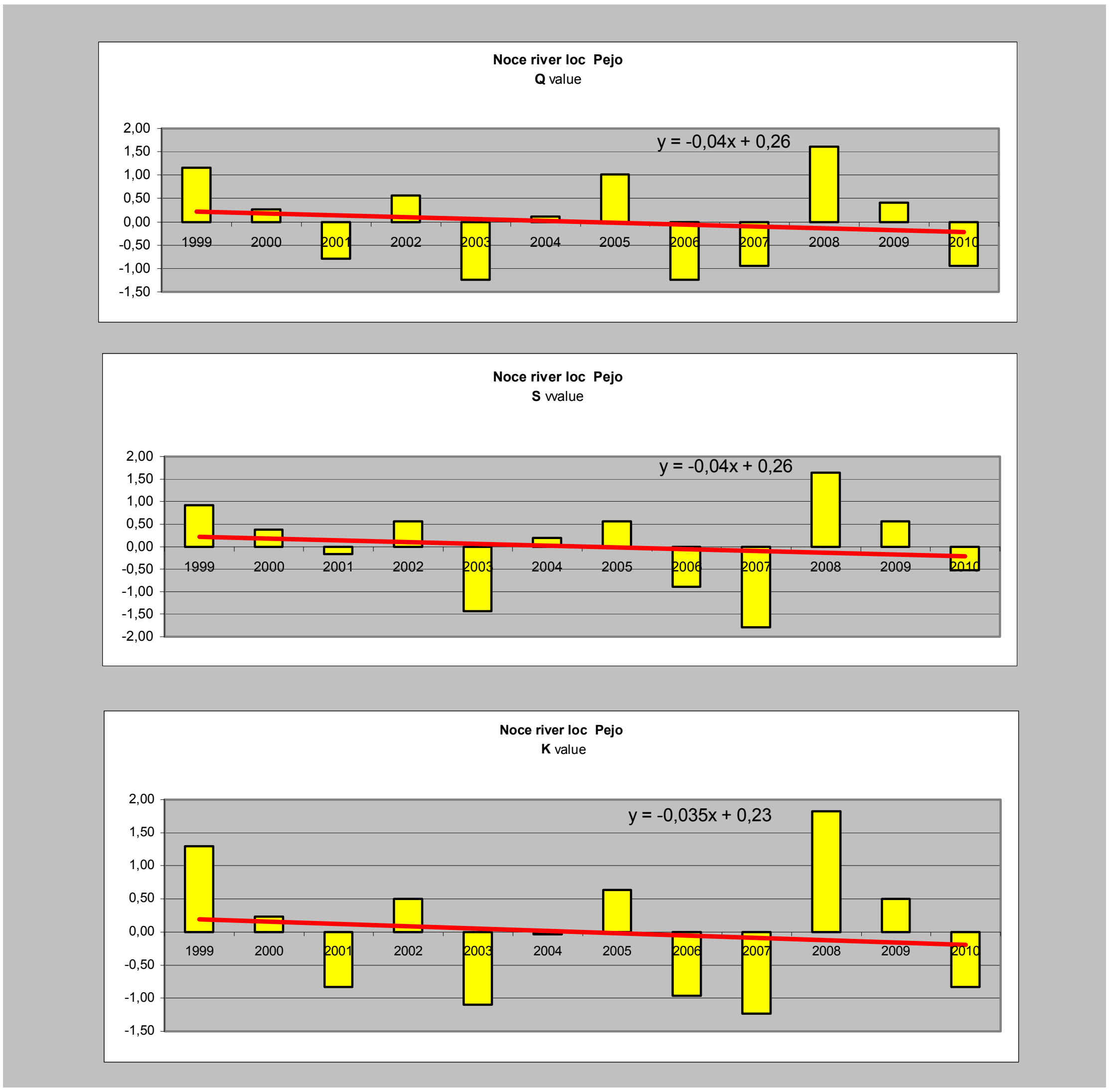


Figure 3.5.1.4: Histogram of the standardised Q, S, K values, with best fit line

The standardised data do not show any major differences, apart from a slight decrease of no statistical significance; it is therefore not possible to determine any differences between the 1998 macroinvertebrate community and the ones in the following years.

The Cusiano site

The same methodology used for the Pejo site was also used for the *Cusiano* site, in order to compare the Q, S, K normal and standardised values.

Q of Yule

	2002
2004	0,76
2005	0,43
2006	0,10
2007	0,34
2008	0,57
2009	0,38
2010	0,06
media	0,38
mediana	0,38
st. dev	0,25

S of Sorensen

	2002
2004	0,79
2005	0,69
2006	0,59
2007	0,56
2008	0,53
2009	0,56
2010	0,57
media	0,61
mediana	0,57
st. dev	0,09

K of Cohen

	2002
2004	0,44
2005	0,36
2006	0,08
2007	0,12
2008	0,17
2009	0,16
2010	0,03
media	0,19
mediana	0,16
st. dev	0,15

Table 3.5.1.6: Q, S, K values for the Cusiano site. Missing values, obtained by interpolation, are reported in red.

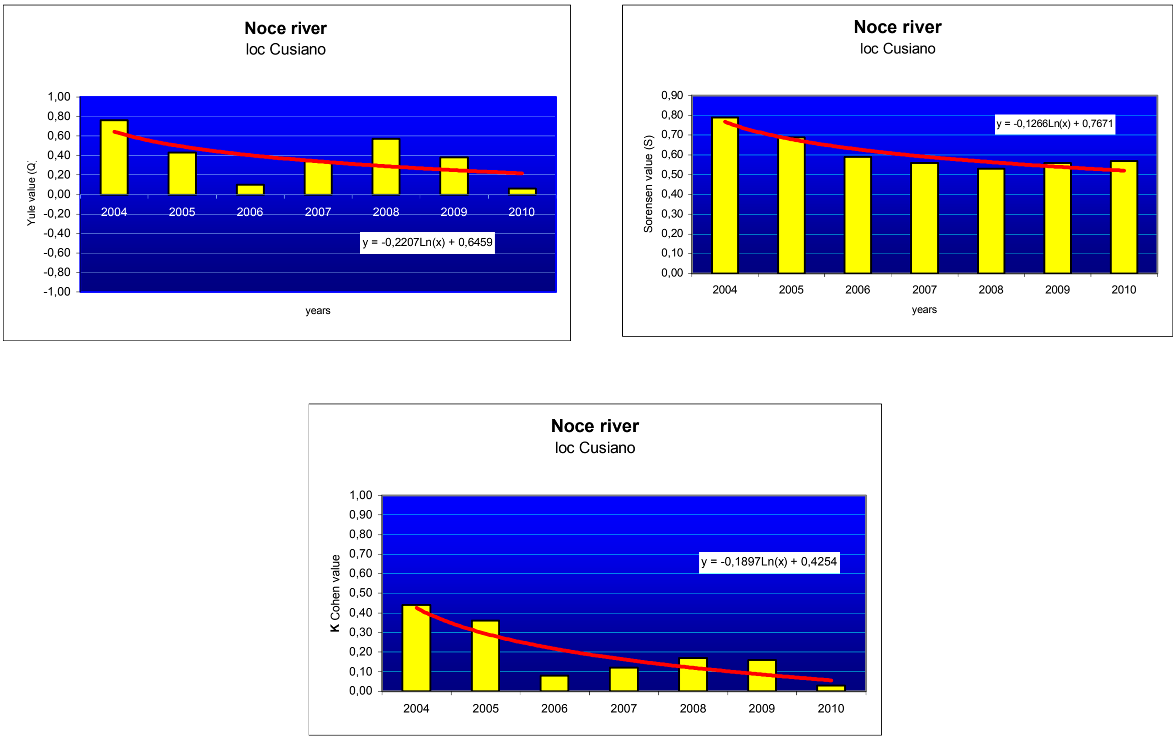


Figure 3.5.1.5: Histograms for Q, K, S tests, with best fit line

Q of Yule

	2002
2004	1,55
2005	0,21
2006	-1,12
2007	-0,15
2008	0,78
2009	0,01
2010	-1,29
media	0,00
mediana	0,01
st. dev	1,00

S of Sorensen

	2002
2004	1,90
2005	0,83
2006	-0,25
2007	-0,57
2008	-0,89
2009	-0,57
2010	-0,46
media	0,00
mediana	-0,46
st. dev	1,00

K of Cohen

	2002
2004	1,64
2005	1,10
2006	-0,76
2007	-0,49
2008	-0,16
2009	-0,23
2010	-1,09
media	0,00
mediana	-0,23
st. dev	1,00

Table 3.5.1.7: Q, S, K standardised values for the Cusiano site

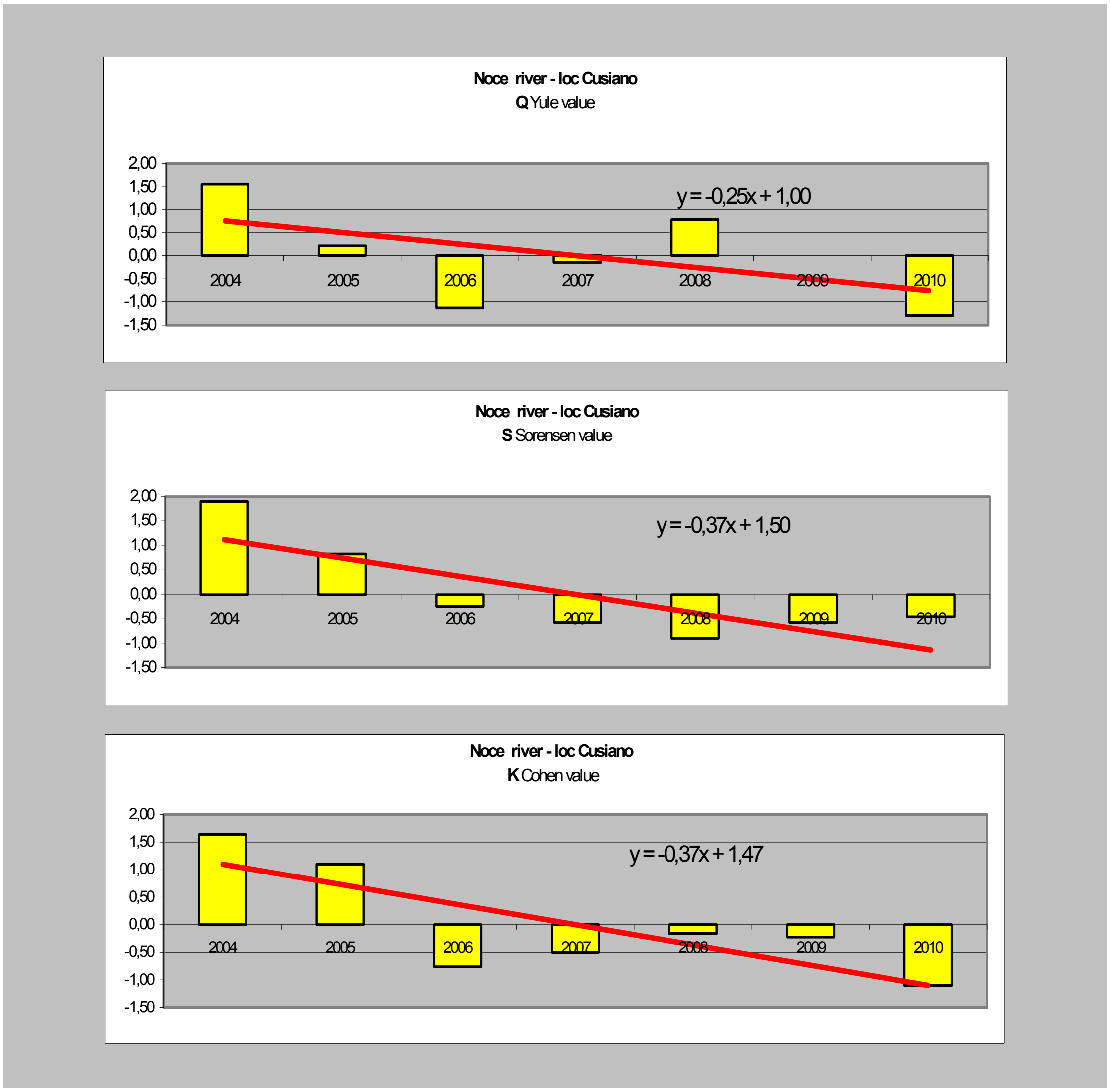


Figure 3.5.1.6: Histograms of the standardised values with best fit line

Figure 3.5.1.6 shows how, unlike the Pejo site, both normal and standardised values show an increasing level of diversity, in other words the communities in the years following 2002 are increasingly less similar.
These results can be correlated to diverse hydraulic conditions and, therefore, to the diverse morphological conditions of the river bed: it in fact changed over the years as a result of frequent hydropeaking phenomena, turning into a simpler, more homogeneous bed and losing its benthic habitat, thus affecting the community.

3.5.2 Morphological characteristic

Extraction of the sections from the DTM with an automated procedure

Once the two reaches had been selected, a GIS (geographical information system) tool was developed for the automatic extrapolation of the geometry of the reaches from the new DTM (digital terrain models) Lidar (1m x1m) of the Province of Trento and from the shapefile of the watercourses.

The tool was used according to the following steps:

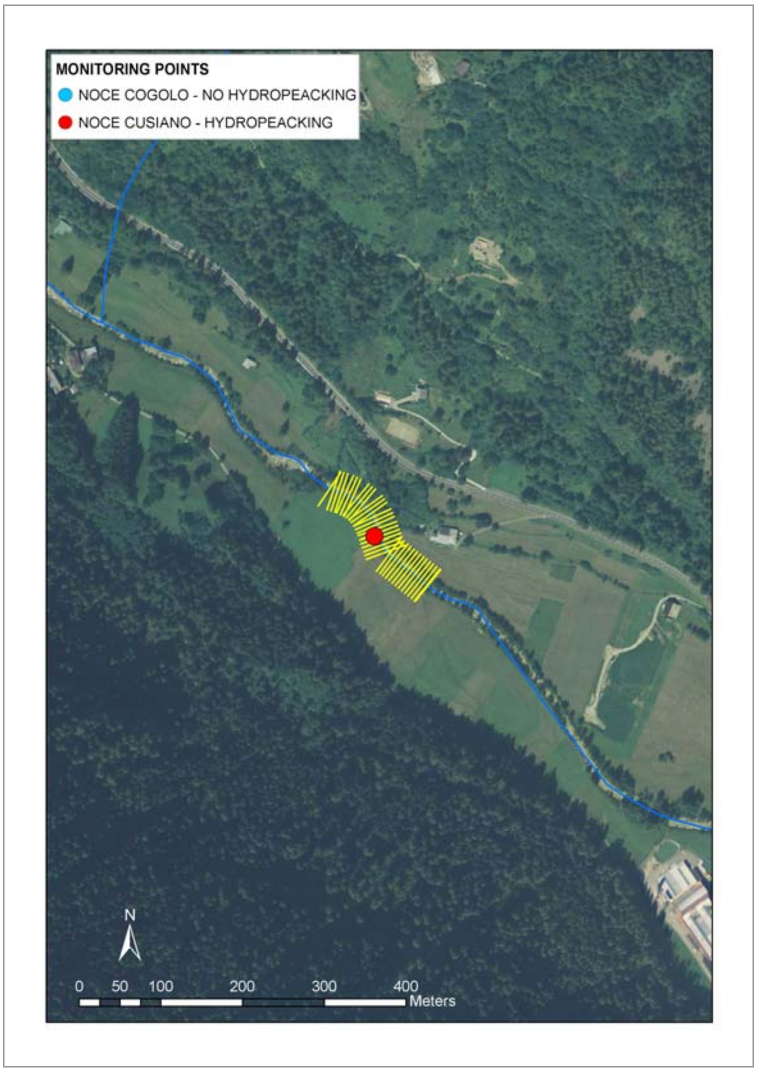
1. loading in ArcGis of the DTM and the shapefile of the rivers;
2. selection of the branch for which the geometry was to be extracted;
3. identification of the specific reach to be analysed either by selecting on screen the start and end points defining it or by introducing their coordinates;
4. establishing the number of sections to be determined within the reach or the distance between the sections;
5. definition of the width of the sections to be determined;
6. determining the number of points from which to extrapolate the single cross-section of the watercourse;
7. based on the options entered so far, the tool analyses the DTM and saves the coordinates for each section of the reach selected in a separate Excel file.

In this case, it was decided to extrapolate the geometry of the two reaches by choosing the following options:

- length of the reaches: approximately 200 m;
- longitudinal distance between the section: 5 m;
- width of the extrapolated sections: 50 m;
- for each section, extraction of the coordinates every transverse metre.

Figures 3.5.2.1, 3.5.2.2 and 3.5.2.3 show the plan and 3D illustrations of the two reaches extrapolated using the GIS tool described above.

Figure 3.5.2.1: Pejo-Cogolo reach: sections extracted



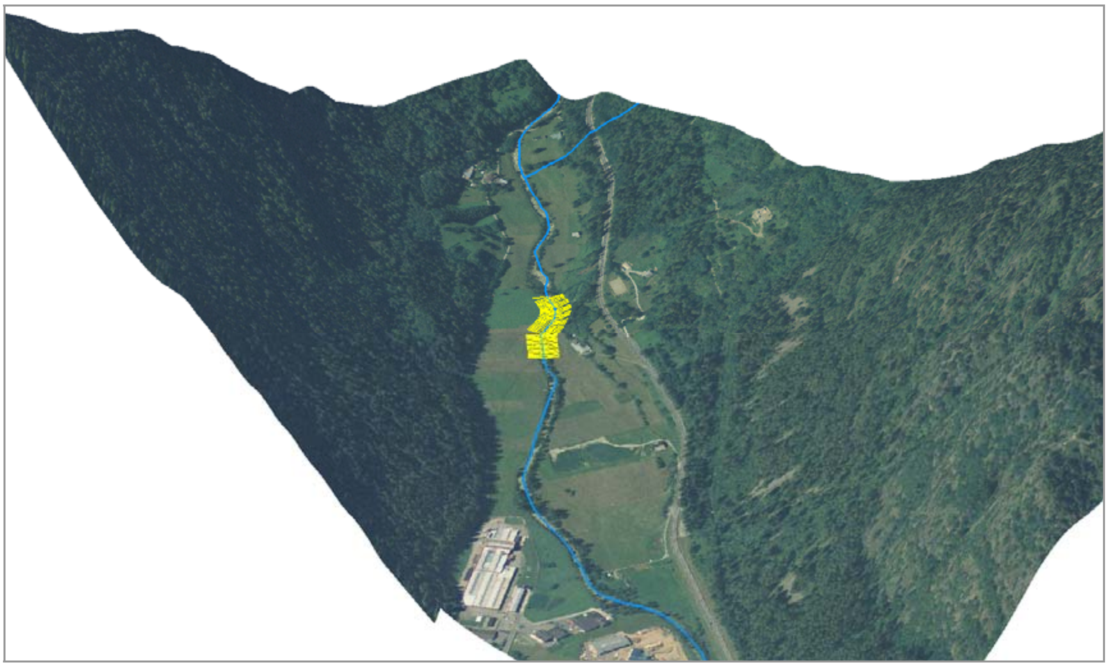


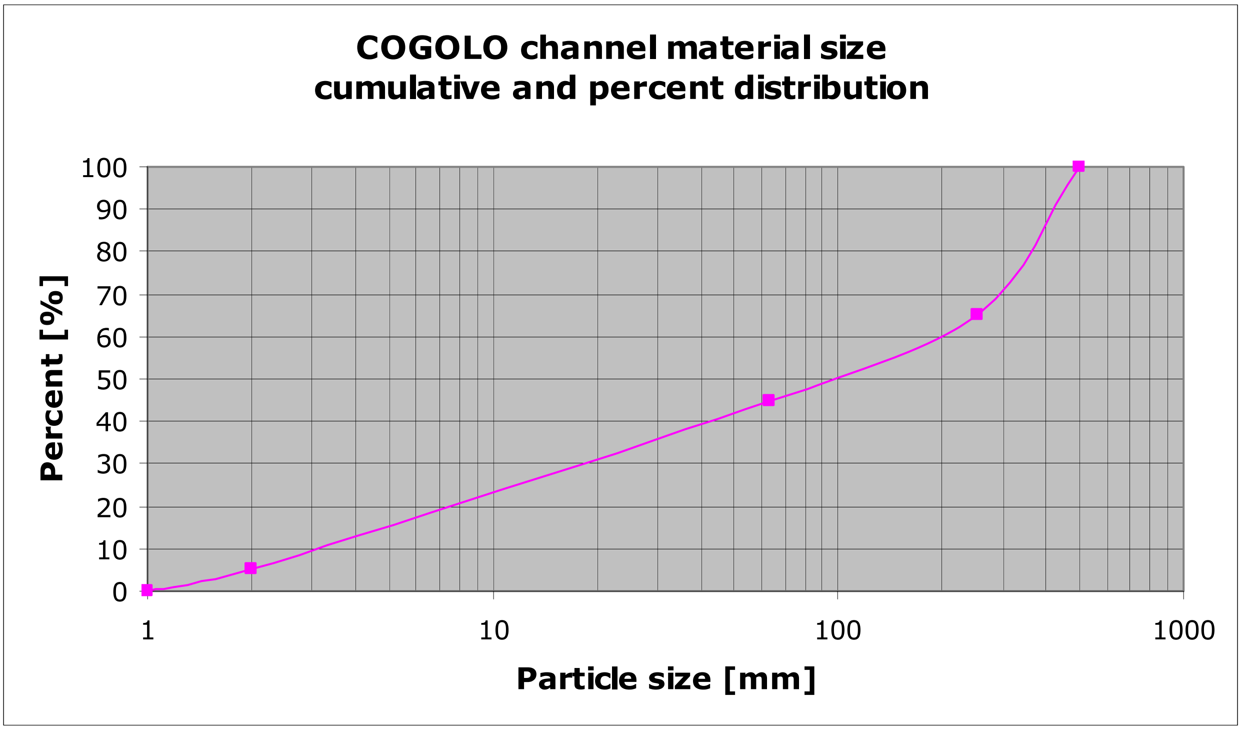
Figure 3.5.2.2: Cusiano reach: sections extracted

Figure 3.5.2.3: Cusiano reach: sections extracted – 3D view

The procedure thus created in effect replaced the traditional survey in the field, which would have been much more costly in terms of time and resources.

Site survey and definition of parameters needed to calculate the index value

It was necessary to carry out an inspection in order to define all the parameters needed for the application of the Rosgen morphological index and in particular the granulometric composition of the riverbed and the position of the Bankfull Stage in the sections analysed. This analysis of the material of the riverbed and of the banks resulted in the two distribution curves shown in Figure 3.5.2.4 and the parameters shown in Table 3.5.2.1.



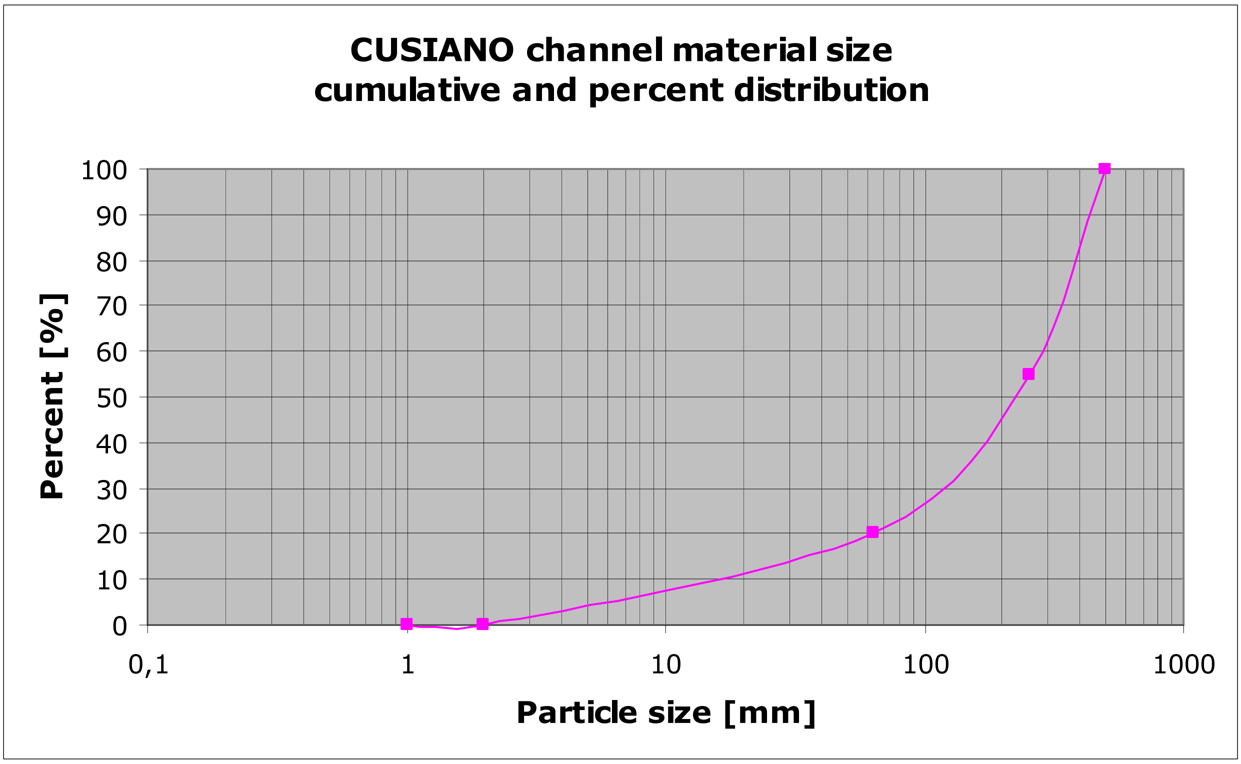


Figure 3.5.2.4: distribution curves of the riverbed and banks material

REACH	D_{50} [cm]	h BANKFULL STAGE [m]
Pejo-Cogolo	10	1 – 1,2
Cusiano	24	0,85

Table 3.5.2.1: parameters of the two reaches

In accordance with the purposes of the stream classification proposed by Rosgen, no fraction was ruled out in the analysis of the bed material. The two granulometric curves, therefore, could suffer possible effects of armouring, most likely in the valley reach of the Noce *Cusiano*. Note that both D50 of the two reaches fall into the fraction of cobbles.
Finally, the measurement of the Bankfull Stage shown in Table 3.5.2.1 refers to the most depressed point of the sections.

Calculation of the index using Excel and cartographic analysis

Once the geometry of the two reaches had been determined in terms of the transversal coordinates of each section and the Bankfull Stage had been calculated for the same reaches, it was possible to calculate the Slope, Entrenchment and Width/Depth parameters (shape and dimension factor).
In this context, Slope is the ratio of the difference in level between the upstream and downstream sections to the length of the reach.
The Entrenchment ratio is the ratio of the width of the flood-prone area to the bankfull surface width of the channel. The flood-prone area is defined as the width measured at an elevation which is determined at twice the maximum bankfull depth.
The Width/Depth parameter is the ratio of bankfull channel width to bankfull mean depth.

By means of cartographic analysis it was finally possible to extract the Sinuosity parameter, the last one necessary for the classification, which is the ratio of stream length to valley length.

Results

Once all the necessary parameters for determining the index had been calculated, it was possible to classify the sections that constituted the two reaches on the basis of the key to classification of natural rivers illustrated in Figure 3.5.2.5.

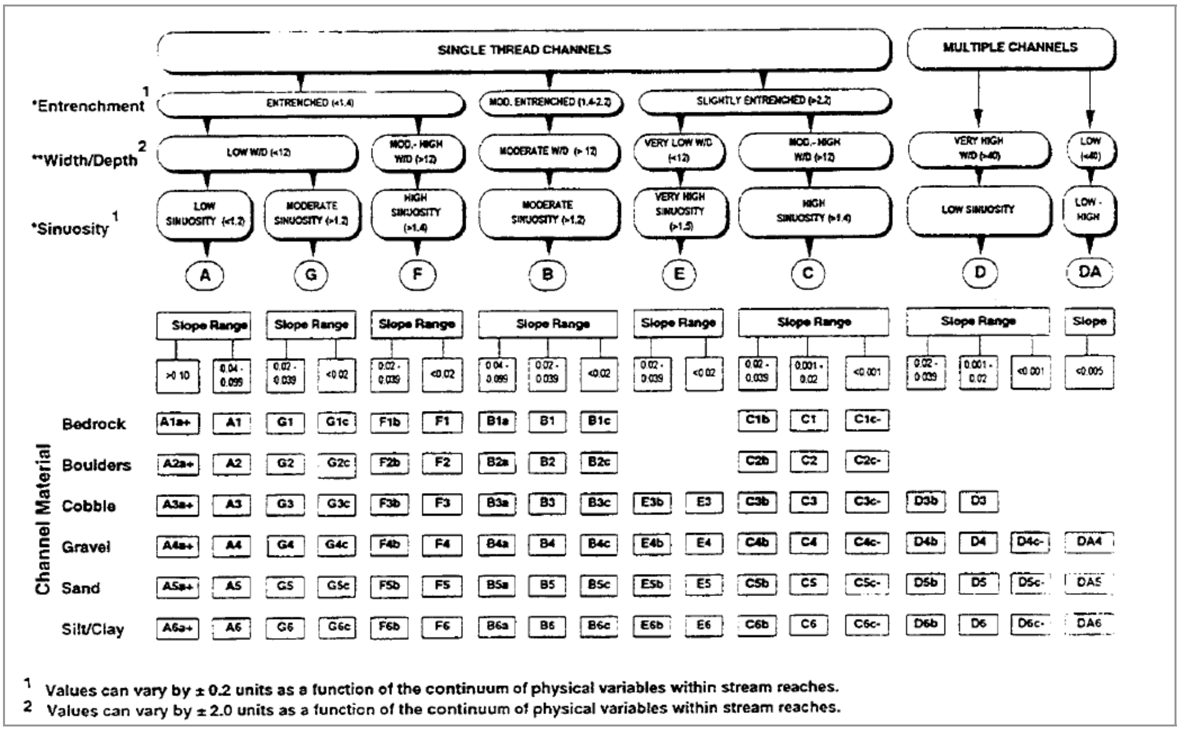
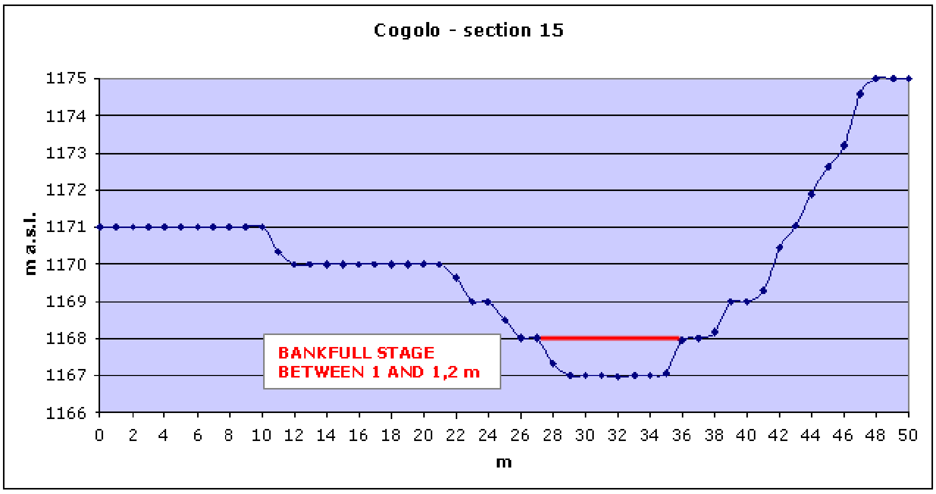


Figure 3.5.2.5: key to classification of natural rivers

By way of an example, the figure shows the results of the classification carried out for two sections belonging to the two reaches examined. Both the reaches were then classified according to the main results of the analysis of their respective sections.

COGOLO –SECTION 15

right			left			
north	east		north	east	X=	70 m
5134981	629722,6		5135017	629756,8		



Entrenchment =	1,50	
Width/Depth =	11,78	
Sinuosity =	1,02	
Slope =	4	%

BANKFULL STAGE = 1m

first 25 m upstream → B3a

then → A3

The main type

GENERAL DESCRIPTION
Steep, entrenched, cascading, step/pool stream. High energy/debris transport associated with depositional soils.

The *Pejo-Cogolo* reach belongs to the A3 type, whose management characteristics are shown in the table below.

Table 3
Management interpretations of various stream types

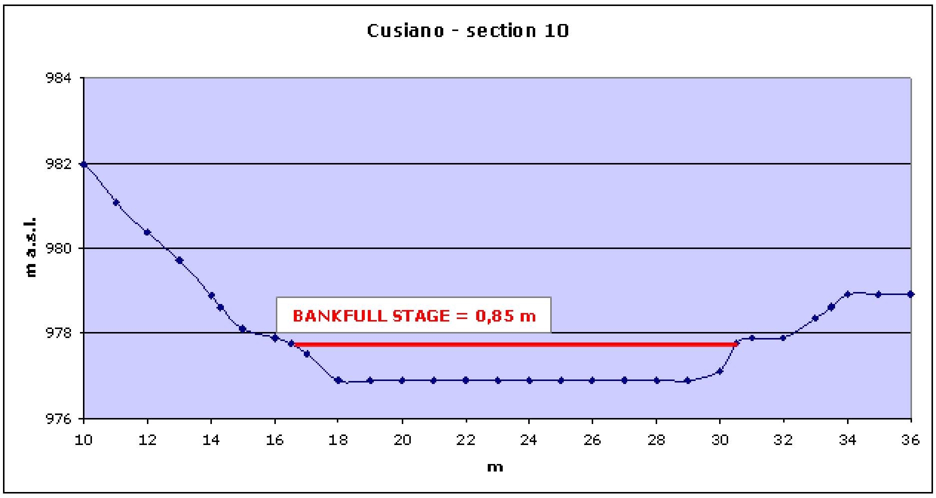
Stream type	Sensitivity to disturbance [a]	Recovery potential [b]	Sediment supply [c]	Streambank erosion potential	Vegetation controlling influence [d]
A1	very low	excellent	very low	very low	negligible
A2	very low	excellent	very low	very low	negligible
A3	very high	very poor	very high	high	negligible
A4	extreme	very poor	very high	very high	negligible
A5	extreme	very poor	very high	very high	negligible
A6	high	poor	high	high	negligible
B1	very low	excellent	very low	very low	negligible
B2	very low	excellent	very low	very low	negligible
B3	low	excellent	low	low	moderate
B4	moderate	excellent	moderate	low	moderate
B5	moderate	excellent	moderate	moderate	moderate
B6	moderate	excellent	moderate	low	moderate

Table 3.5.2.2: management interpretation of the Pejo-Cogolo reach

According to the Rosgen classification, this reach has the characteristics of a morphologically sensitive environment that, if altered, would have a poor recovery potential.

CUSIANO –SECTION 10

right			left									
north	east		north	east						X=	45	m
5130820	632612		5130858	632643,8								



Entrenchment =	1,40	
Width/Depth =	17,69	
Sinuosity =	1,05	
Slope =	3	%

BANKFULL STAGE = 0,85 m

main type → B3

GENERAL DESCRIPTION
Moderately entrenched, moderate gradient, riffle dominated channel, with infrequently spaced pools. Very stable plan profile. Stable banks.

The *Cusiano* reach belongs to the B3 type, whose management characteristics are shown in the table below.

Table 3
Management interpretations of various stream types

Stream type	Sensitivity to disturbance [a]	Recovery potential [b]	Sediment supply [c]	Streambank erosion potential	Vegetation controlling influence [d]
A1	very low	excellent	very low	very low	negligible
A2	very low	excellent	very low	very low	negligible
A3	very high	very poor	very high	high	negligible
A4	extreme	very poor	very high	very high	negligible
A5	extreme	very poor	very high	very high	negligible
A6	high	poor	high	high	negligible
B1	very low	excellent	very low	very low	negligible
B2	very low	excellent	very low	very low	negligible
B3	low	excellent	low	low	moderate
B4	moderate	excellent	moderate	low	moderate
B5	moderate	excellent	moderate	moderate	moderate
B6	moderate	excellent	moderate	low	moderate

Table 3.5.2.3: management interpretation of the Cusiano reach

According to the Rosgen classification, this reach has the characteristics of a morphologically stable environment which has already been modified by hydropeaking, leading to an "armoured" bed and sections resembling a uniform channel.
It is important to underline that the concept of the continuum had to be used in the classification of the two reaches, in particular as regards the Entrenchment parameter in the case of the *Pejo-Cogolo* and the Sinuosity parameter in the case of the *Cusiano*.

Conclusion of the morphological analysis

The implementation of the Rosgen index, according to the "Key to classification" proposed by the same author, made it possible to classify the two selected reaches as belonging to two different types.
According to the table on "Management interpretations of various stream types":
- the Noce *Pejo-Cogolo* reach has a very high sensitivity to disturbance, including increases in streamflow magnitude and timing, and a very poor natural recovery potential (once the cause of instability has been corrected);
- the Noce *Cusiano* reach has a low sensitivity to disturbance, including increases in streamflow magnitude and timing, and an excellent natural recovery potential.

In our opinion, that this result is consistent with the current state of the two reaches. In fact, the *Pejo-Cogolo* reach still displays its natural characteristics and has not yet been morphologically modified by sudden increases in discharge although these could disturb it irreversibly. The *Cusiano* reach is constantly subject to hydropeaking phenomena and its morphology now seems to be in equilibrium with the changing rates of discharge.

Section 2

3.5.3 Physical indicators: Temperature and water level

Variations in time

In the Noce River basin (Figure 3.5.3.1:) hydropower is generated by three plants fed by four artificial reservoirs closed by dams. For this study, we consider two different junctions located, respectively, in the middle and lower parts of the catchment. The first junction is located 5 km downstream the village of Malè where the near-natural Rabbies Stream (yearly mean discharge 3.3 m^3/s, basin area 142 km^2) flows into the Noce River. The second junction is located in the lower basin, near the village of Mezzocorona, where water stored in the Santa Giustina reservoir (capacity of 182 x $106m^3$) first passes through the Mollaro reservoir (capacity of 2 x $106m^3$), and then feeds the Mezzocorona power plant that releases a maximum discharge of 60 m^3/s into the Noce River. The reservoir of Santa Giustina is created by a 152.5 m high dam, with the head at 532.5 m asl, maximum and minimum water levels at 530 and 445 m asl, respectively, and the hypolimnetic intake level at 437 m asl (Edison, 2008).

Data collection

Stream water temperature was monitored with high temporal and spatial resolution at four different locations along the basin through 10 StowAway TidbiT temperature dataloggers which were placed along the main channel at suitable cross-sections up- and downstream of the two junctions. Namely, for the Rabbies junction, gauging sections were chosen near the village of Croviana (immediately upstream the confluence) and near the bridge named 'Ponte Stori' (Figure 3.5.3.1).

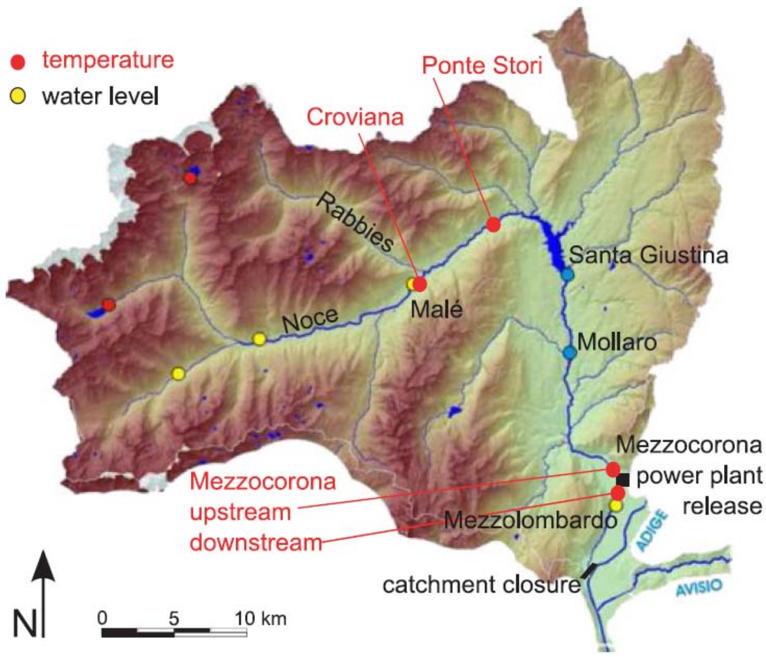


Figure 3.5.3.1: Study site: map of the Noce River catchment with location of the temperature and water level gauging stations (circles).

Although the latter is located a few kilometers downstream the Rabbies- Noce confluence, there are not other significant lateral tributaries. Water level data were recorded at three river sections (Figure 3.5.3.1:): Noce River at Malè (upstream of Croviana), at Ponte Stori and a few hundreds meters downstream the release of the Mezzocorona power plant. For these locations, a discharge-rating curve is also available. A simplified procedure was developed for baseflow separation in order to precisely identify the peaking events and estimate their duration, intensity and time distribution. The procedure is based on establishing a threshold rate of change of the water level time series. The dataset of the water level series on an annual basis is used to obtain the streamflow data throughf a calibrated rating curve. Subsequently, outlayers are removed and a short-scale (e.g. 1 h) moving average is applied in order to smooth high frequency irregularities of the signal, often due to lack of instrumental precision. The second step is the individuation of the base flow (Figure 3.5.3.2:).

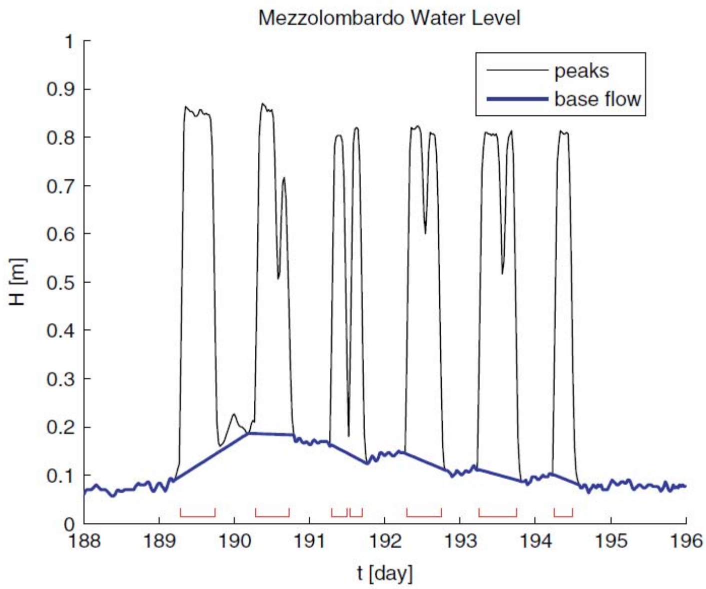


Figure 3.5.3.2: Water levels (black line) in Mezzolombardo from day 188 (Sunday, 8th July 2007) to day 195 (Sunday, 15th July 2007). The base flow (thick blue line) is identified through the procedure

described in the main text. The duration of the peaking events is visualized through the red segments close to the horizontal axis.

Hydropeaking characterization

The characteristics of hydropeaking events change during the year. In some periods, the water is released every day of the week at the maximum rate; in other periods, the requests of the energy market determine a more irregular pattern of hydropower production that causes short-period, irregular water level peaks.

The main result is that the distribution of duration is bimodal, with one peak at about 6–8 h and another one around 8 h for single events (Figure 3.5.3.3, left panel). The distribution is also approximately bimodal considering multiple events, whose number is about 10% larger than single events (Figure 3.5.3.3, right panel). In this case, the peaks are shifted towards shorter durations: the former peak is around 3–6 h, the latter on 15 h. The two peaks can be attributed to two different hydropower generation schemes: half-day production occurring in the morning or in the afternoon only and whole-day production, continuously occurring from morning to evening.

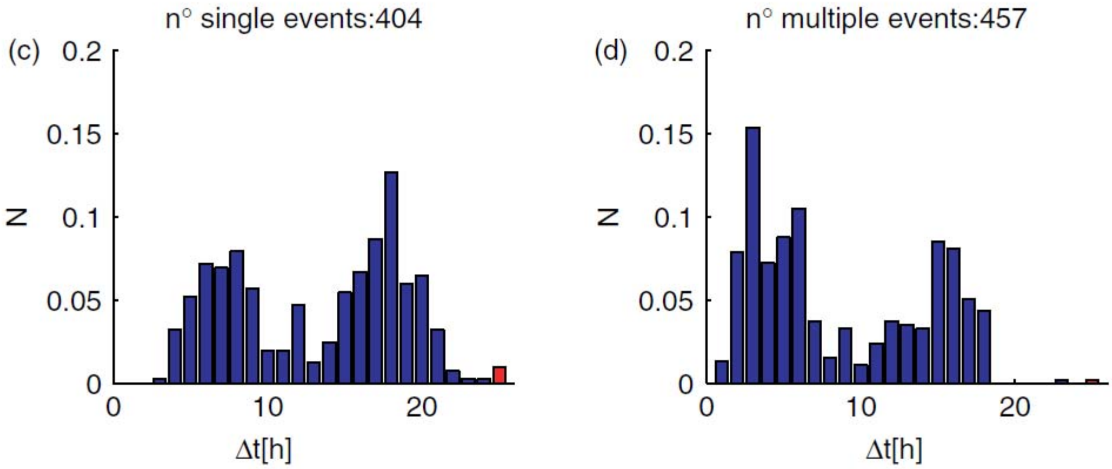


Figure 3.5.3.3: Main characteristics of the hydropeaking events in Mezzolombardo (year 2007). Left panel: frequency distribution of the event duration Δt; right panel: same as left panel but considering separate multiple events. The red bar at 25 h indicates the sum of the events with duration longer than 1 day.

Thermopeaking characterization

Distinguishing the thermal alterations caused by hydropower releases from the diurnal cycle due to the variation in the net external energy input is not an obvious task if only one gauging station is considered. As a rule, a reliable characterization of thermopeaking requires information from two gauging stations, located upstream and downstream of the hydropower release. Applying a suitable time shift to one of the two records allows comparing the upstream and downstream temperature signals, and hence to obtain the net amount of thermal alteration by subtraction. The value of the time shift depends on the mean flow velocity and it can also be determined examining the covariation between the two signals. We refer to the case of the Mezzocorona power plant release in order to investigate in detail the properties of thermopeaking.

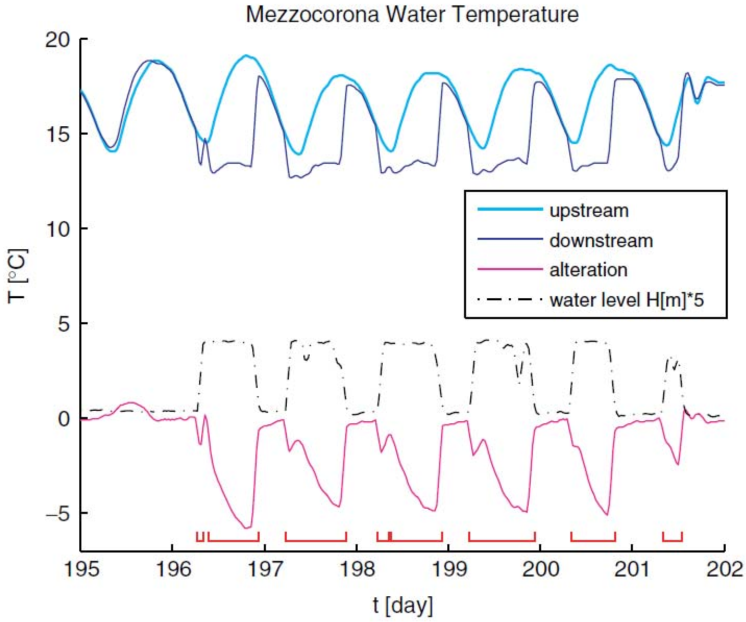


Figure 3.5.3.4: Water temperature upstream (thick cyan line) and downstream (blue line) Mezzocorona hydropower release from day 195 (Sunday, 15th July 007) to day 202 (Sunday, 22nd July 2007). The temperature difference (magenta line) highlights cold thermopeaking events, which are consistent with hydropeaking events (dash-dot black line) as described by the strong water level fluctuations (H is shown on the temperature axis in meters multiplied by 5 °C/m). The duration of the cold thermopeaking events is visualized through the red segments close to the horizontal axis.

Figure 3.5.3.4 shows how the thermally altered streamflow affects the daily temperature cycle. In the period considered (1 week in July) the water temperature in the reservoir is lower than that of the receiving stream, and a cold thermopeaking occurs. The duration of thermopeaking events is compared with the ones estimated by means of water level variations (dash-dot line), showing a noticeable agreement. It is worth noting that the magnitude of the thermal alteration (up to 6 °C in this case) is comparable with the sinusoidal diurnal variation in the undisturbed upstream reach (approximately 5 °C).

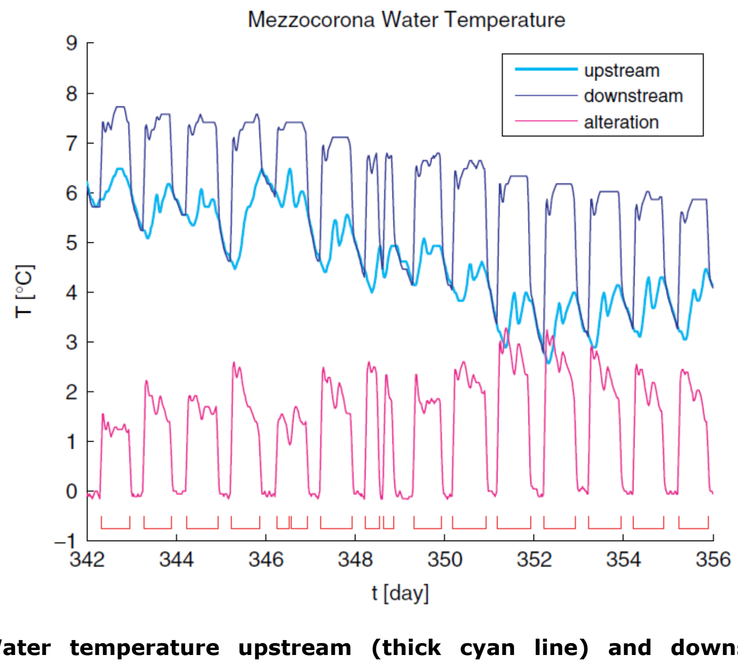


Figure 3.5.3.5: Water temperature upstream (thick cyan line) and downstream (blue line) of Mezzocorona hydropower release from day 342 (Sunday, 9 th December 2007) to day 356 (Sunday, 23rd December 2007). The temperature difference (magenta line) individuates warm thermopeaking events, whose duration is visualized through the red segments close to the horizontal axis.

An opposite behaviour is shown during the winter season, where a warm thermopeaking occurs due to the higher temperature of reservoir water (in particular, for withdrawal of hypolimnetic water if the reservoir may be considered as partially stratified) with respect to the river temperature. This is illustrated in Figure 3.5.3.5, where two weeks of regular hydropeaking are considered. In this case, the thermal alteration (up to 3 °C) is notably larger than the natural daily variation (approximately 1 °C).
A similar analysis has been performed on the lower reach of the Noce, about 8 km downstream of the Mezzocorona power plant release. The gauged time series of water level and temperature are shown in Figure 3.5.3.6.

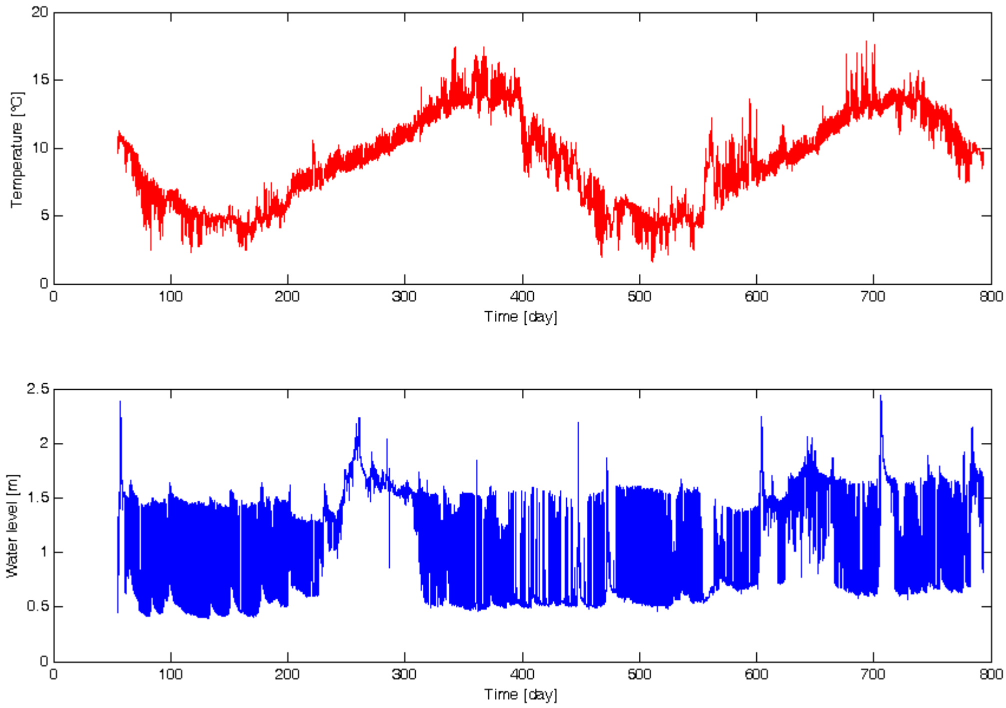


Figure 3.5.3.6: Temperature and water level of the Noce River downstream the Mezzocorona power plant (Dec. 2008 – Nov 2010)

The present study provides a detailed quantification of the short-term alteration of the thermal regime in the Noce River, a typical hydropower-regulated Alpine stream. Besides representing the first study related to the short-term alteration of the river thermal regime in an Italian Alpine basin, it allows a quantitative understanding of the complexity associated with water temperature regimes in regulated streams, particularly at the short time scales affected by hydropower production. The outcomes of the analysis indicate a series of previously unknown thermal effects at multiple scales which might strongly affect biological communities in similar geographical and regulation contexts. Indeed strong biological alterations have been documented (Bruno et al., 2009) in the same Noce River downstream of the Cogolo power station, where strong hydropeaking and thermopeaking occur. It is therefore reasonable to hypothesize that the river biota may also be severely affected by thermopeaking further downstream under analogous regulation effects. In analogy with hydropeaking, we propose the terminology thermopeaking to denote the sharp temperature variations associated with the sudden instream water releases downstream of the power plants. In the Noce River, the rising and falling of hydro- and thermopeaking are comparable with the period of the sampling interval (nearly 30 min), and events might have two typical durations, in the ranges between 5–8 and 15–18 h. The time distribution of thermopeaking reflects the pattern of hydropower production driven by price fluctuations in the energy market and thus often deviates from the fairly regular pattern consisting of energy production during daytime in the working days, which was typical in the Alps until the past decade. Warm thermopeaking occurs from September to January and results in additional (up to 4 °C) heating with respect to that associated with the natural diel fluctuations. On the contrary, cold thermopeaking occurs from March to July and cools down the temperature (up to 6 °C), in contrast with the natural trend that would result in heating during the day. As a consequence, temporal oscillations of water temperature recorded downstream of the release are amplified in the average during winter compared with summer season. Overall, the key differences between natural and man-made temperature variations that can be drawn from the present study are as follows: rates of temperature changes are much faster and contrasting seasonal effects are associated with repeated thermopeaks. The outcomes of this study confirm that the strongest variability applies for the smaller scales, with a strong increase in temperature variability at sub-daily scales. The quantification of thermopeaking events and of their thermal effects at multiple time scales suggest that hydropower regulation have significantly muted the small time scale variability in temperature patterns to which many organisms may have adapted. Conserving or restoring natural temperature patterns in rivers will require attention to these small-scale thermal alterations and to their potential ecological consequences.

Experimental research on the Fersina flumes facilities was necessary to complement the present analysis, by assessing how thermopeaking can affect phases such as larval growth rates, adult emergence or behavioural drift. This appears of specific relevance because many studies have highlighted the importance of river thermal regimes as drivers of ecological processes and of the dynamics of aquatic communities, but based on research conducted at much longer time scales (Ward and Stanford, 1979).

Variations in space

Most studies on hydropeaking and, more recently, thermopeaking, have basically focused on the temporal dimension of this specific alteration of flow regime, through at-a-station measurements of water level and temperature oscillations. Relatively little is known about the spatial extent of river reaches affected by the propagation of these peaking waves, which ultimately requires an in-depth analysis of their propagation dynamics.

A second component of the study related to thermal variations in alpine streams with potential water scarcity due to heavy flow regulation and in the perspective of developing indicators for optimal ecological flow recommendations has therefore been a spatial analysis of the dynamics of hydro- and thermopeaking waves. The present section describes a modeling based approach to investigate the physics of both hydrodynamic and thermal peaking waves propagation in a channelized stream, accounting for the most relevant external energy exchanges. For this purpose, analytical solutions of a coupled hydro-thermodynamic deterministic model for river water temperature have been developed. This approach has been chosen considering that simplified analytical solutions are often powerful tools that facilitate process understanding.

The study has been referred to an open channel receiving a lateral tributary with a different water temperature. This is representative of the typical setting downstream of a reservoir where a channelized stream receives the water discharge released from a hydropower plant. Hydropeaking and thermopeaking are modeled according to the conceptual scheme reported in Figure 3.5.3.7. The undisturbed flow is characterized by constant discharge and temperature values (Q_0, ϑ_0) and receives an abrupt release of water, characterized by values (Q_l, ϑ_l), at the junction where the lateral inflow joins the mainstream (x = xl).

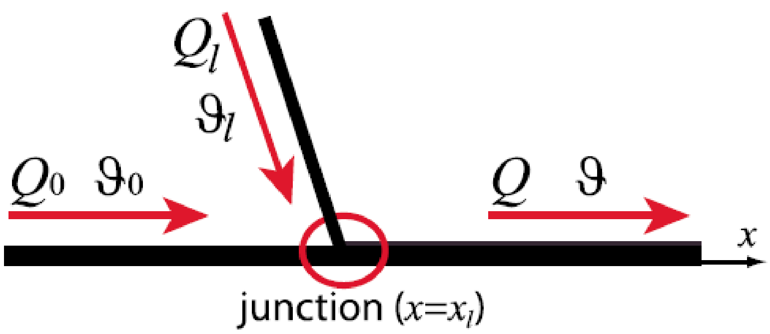


Figure 3.5.3.7: Sketch of the idealized junction causing hydro- and thermopeaking to which the mathematical model is referred.

Here the two peaking waves are approximated by square-shape waves (Q, ϑ) exploiting the typically high rise and fall rates shown by hydro- and thermopeaking. The propagation of the peaking waves in these conditions can be conveniently studied in the framework of a one-dimensional (1D) approach which mainly focuses on the longitudinal thermal and hydraulic variations. Investigating two- and three-dimensional effects associated with vertical and lateral mixing in the nearby region of the lateral inflow is out of the scope of the present study. In this framework the hydrodynamic problem is governed by the Saint Venant equations which read:

$$\frac{\partial A}{\partial t} + \frac{\partial Q}{\partial x} = q_l,$$

$$\frac{\partial U}{\partial t} + U\frac{\partial U}{\partial x} + g\frac{\partial H}{\partial x} + gj = \frac{q_l U_l}{A}\cos\alpha_l, \quad (1)$$

where x denotes the longitudinal axis, t is time, Q = UA is the water discharge, (A, U, H) denote the cross-section area, flow velocity and water surface elevation respectively, q_l is the unit lateral discharge inflowing with an angle al with respect to the x direction, g denotes gravitational acceleration, and j is the friction term

$$j = \frac{U^2}{k_\chi^2 R_h^\chi},$$

with R_h the hydraulic radius (the ratio between the cross sectional area A and the wetted contour B), while $k\chi$ and c are roughness-related coefficients. For example, the Gauckler-Strickler-Manning formulation implies that χ = 4/3 with k_χ = k_s the Gauckler-Strickler coefficient. Assuming that the river is fully mixed both laterally and vertically, temporal and spatial temperature changes along the main flow direction can be described by the 1D advection-dispersion equation (e.g., Rutherford, 1994).

A simplified solution of equation (1) is obtained transposing the square wave from being the boundary to the initial condition. We assume that the wave travels with constant velocity, but we indirectly retain distortions associated with non-linear effects. Wave distortion is accounted for by artificially introducing two different diffusion coefficients for the tail and for the head of the wave, which will therefore be referred to as "artificial diffusion". An analogous procedure has been employed to study the propagation of the thermal wave, by solving the equation of heat transport (convection – diffusion equation) with temperature as passive tracer. Source terms in the equations include all relevant energy exchange between the surface water and the external system.

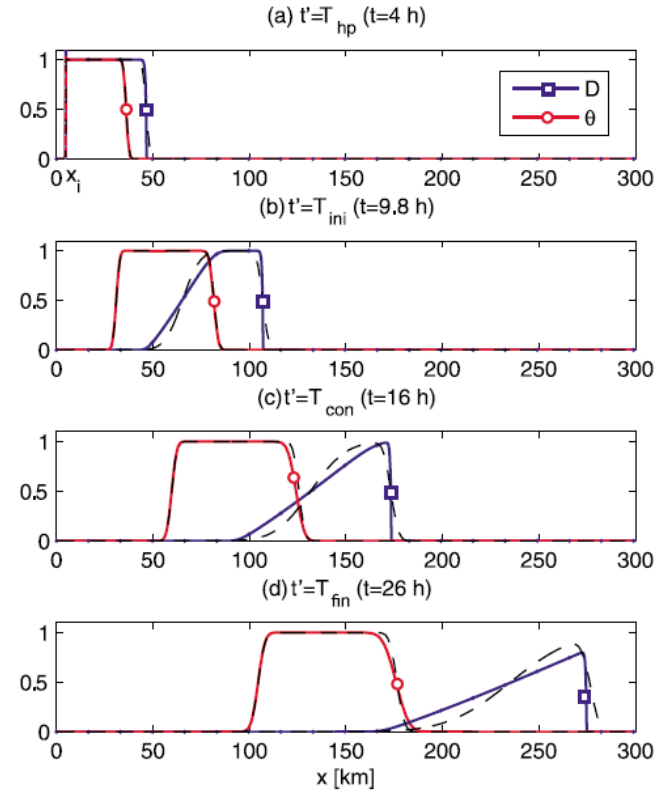


Figure 3.5.3.7: Illustration of the different phases of the hydro- and thermopeaking wave dynamics from numerical results at different times and their analytical approximation (black dashed lines).

The modelling study allows to detect the key properties of hydrodynamic and thermal waves propagation under hydropeaking conditions.
A fundamental physical phenomenon is the downstream propagation of the two waves with different speeds. The maximum possible celerity of the thermal wave is the peak flow velocity, while a good measure of the hydrodynamic wave celerity is the wavefront celerity, which is always higher than the flow velocity. This suggests the possible existence of two stages in the wave dynamics. In a first stage the hydrodynamic wave overlaps with the thermal wave and possibly affects its propagation. In a second stage the hydrodynamic wave is completely separated from the thermal wave. Such behavior is graphically reported in Figure 3.5.3.7, where four stages are distinguished (a to d). (a) just after the end of the release the two waves have different lengths (time $t_i + T_{hp}$); (b) the thermal head front is affected by the decrease of stream velocity (beginning of separation, time $t_i + T_{ini}$); (c) the two waves would be separated if purely convective, in absence of diffusion (time $t_i + T_{con}$); (d) the two waves are almost completely separated (time $t_i + T_{fin}$). Overall, two main characteristic phases of the thermal wave dynamics emerge: the initial overlap and the subsequent separation (Figure 3.5.3.8).
The modelling approach allows to quantify the relevant spatial scales at which the separation phase is likely to occur, in dependence of the geometric and hydraulic condition of a given river reach and on the magnitude and duration of hydro- and thermopeaking waves.

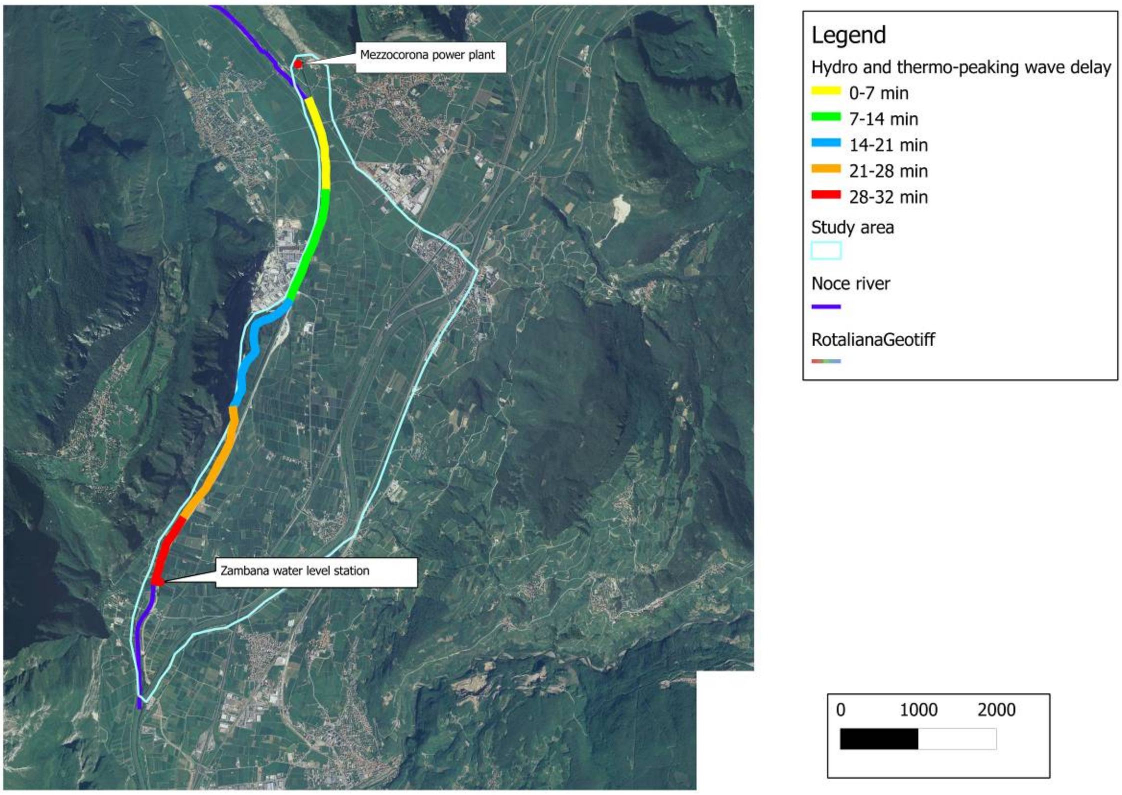


Figure 3.5.3.8: Hydro and thermo-peaking wave delay downstream the Mezzocorona power plant before the junction with the Adige river.

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3.6 Adige river (Rotaliana plain)

Summary

The research conducted in the Adige River (Rotaliana plain) analyzed the contribution of a ditch network to floodplain biodiversity, and assessed its potential to mimic the multifunctional role of extinct natural wetlands through reuse of part of the excess water in the river reach associated with hydropeaking. The hydraulic characterization of the ditches network was performed by integrating historical with newly collected data on topography and ditches cover, water level and temperature measurements. A 1D unsteady flow model for hydropeaking wave propagation in the ditches network was developed. Biological (benthic macroinvertebrates, macrophytes and riparian plants), hydrological and physical – chemical variables were measured at several stations along the network.

The results of the research underline how the ditch network represents a largely variable range of aquatic habitats on a relatively small spatial scale, and their restoration would increase biodiversity in highly anthropic areas, at the same time efficiently storing water resources which otherwise would be quickly withdrawn from the system, at the same time partly damping the hydropeaking waves

3.6.1 Biological indicators: benthic invertebrates and aquatic macrophytes

The idea underlying the experimental activity at this study site is that re-using part of the hydropeaking water into the network of agricultural ditches can improve ecosystem health of the river-floodplain system. An ecohydraulic experimental research has been carried out with the aim of analyzing the contribution of a ditch network to floodplain biodiversity, and to assess its potential to mimic the multifunctional role of extinct natural wetlands through reuse of part of the excess water in the river reach associated with hyropeaking.
A variety of biological and physical – chemical indicators have been measured at the monitoring stations indicated in Figure 3.6.1.1. Conductivity, pH, dissolved oxygen, mean velocity, depth and width of the wet channel and distance from the source as well as water temperature were measured. Species composition and relative abundance of invertebrates and macrophytes were recorded. Phytosociological samples of macrophytes and riparian vegetation were conducted in June 2009 and 2010, zoobenthos was sampled in June 2010, in 25 sampling points along 13 ditches (Figure 3.6.1.1.)

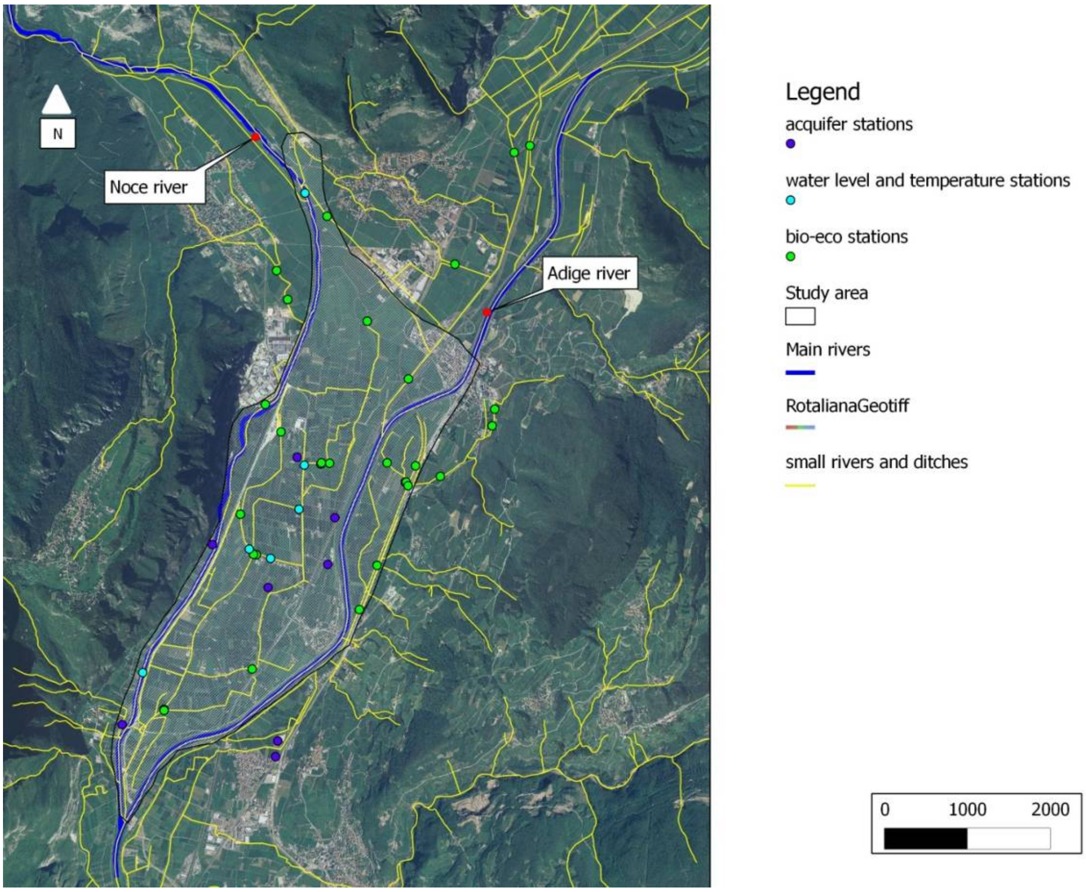

Figure 3.6.1.1: Rotaliana floodplain, macrophytes, zoobenthos, water level and acquifer sampling stations.

We used a standard kick-Surber net with mesh size of 350 ìm to collect benthic samples. We sampled a total area of 0.5 m^2 for each sampling point. All samples were sorted in the laboratory and organisms identified to the lowest possible taxonomical level following Campaioli et al. (1994, 1999). All animals were identified to genus, except for Chironomidae, which were identified to family level. A total of 29 taxa were identified across the 27 sampled sites, for a total of 12920 specimens. We recorded a maximum of 12 taxa with a mean of 5.1 taxa per station, and a maximum of 1668 specimens with a mean of 538 per sampling point. The most widespread taxon was Chironomidae, collected in 22 of 27 samplings, followed by Gammaridae, Dytiscidae and Haliplidae, which were present in 11 samples. Chironomidae and Gammaridae were also the most abundant taxa with a mean of 410 and 110 specimens per sample, respectively. Some other abundant taxa, as Ephemerellidae (*Ephemerella sp.)* were not widespread but locally very abundant (about 90 specimens per sample in 4 samples). Also Simuliidae and Elmidae were locally abundant but not widespread. Some taxa as Dytiscidae and Haliplidae were widespread but not abundant.

Macrophytes are globally considered as good bioindicators for freshwater habitats. Phitosociological samples of macrophytes and riparian vegetation were done both in summer 2009 and 2010 along the ditch network. A total of 36 macrophyte taxa were collected showing a high value of biodiversity for such a restricted area. Vascular plants represented 72% of aquatic macrophytes, aquatic algae 25% and bryophytes 3%. Total macrophyte specific richness per site ranged from 1 to 11 taxa. Three stations presented only one species: two sampling points include only Ranunculus trichophillus and one is a grove of reeds composed only by Phragmites australis. The most widespread species is Typhoides arundinacea, which appears in 16 sampling points. Other frequent species are Callitriche obtusangula with 13 records, Nasturtium officinale (11 records) and Veronica anagallis-aquatica (10 records). 11 species were found only once.
For each sampling site the IBMR Index (AFNOR, 2003) was applied. This index is now used in Italy in order to implement the Water Framework Directive and the use in this study aims also to test its efficiency as bioindicator in alpine lowland areas. The value ranged from 8,

witch corresponds to a very high trophic state, to 12 which indicates low trophic state. The value of the IBMR Index is compared to the value of the reference site for the corresponding geographical region and river macro typology. We obtained the RQE (ecological quality ratio) value which represents the quality of the habitat. In particular (see Figure 3.6.1.2) 3% of the sampling points resulted as poor quality, 62% as moderate quality, 28% as good quality and 7% as very good quality. The overall results highlight a wide spatio-temporal mosaic of different freshwater habitats, with a high value for biodiversity.

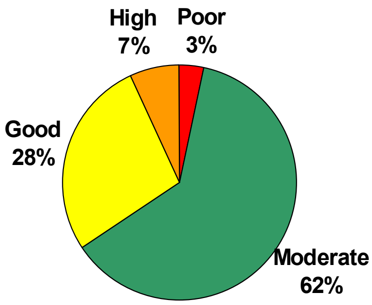

Figure 3.6.1.2: Habitat quality classification of the sampling sites based on the macrophytic RQE (ecological quality ratio).

Quality indices using the zoobenthic community were not used as they are meant for running waters and not suitable for slow flowing waters such as those in the investigated ditch system.

Most impacts resulted to be due to water scarcity and under-exploited management opportunities. The re-use of residual waters from the hydropower station appears to be a good solution to ameliorate the ecological condition of these waterways, and thus to enhance ecosystem benefits as groundwater recharge, nitrogen control and development of an adequate biodiversity.

A Cluster analysis of the stations based on the physico-chemical variables, followed by a discriminant analysis, allowed detecting how the environmental variables contributed to group division in the cluster analysis.

Group A was characterized by low velocity, high mean temperature, and low level of dissolved oxygen: ditch habitats similar to lenthic waters. The vegetation was characterized by the presence of a pleustophytic community, expression of high nutrient load. In fact the vegetation structure in moderately eutrophic ditches is often characterised by a dominance of submerged vegetation, besides emergent species (helophytes), but at very high nutrient loading, the vegetation becomes dominated by a surface layer of pleustophytic plants only, such as duckweed (Lemnaceae) (Janse et Puijenbroek 1998). Several adverse effects are related to this shift to duckweed, principally water becomes anoxic because the oxygen is released into the atmosphere and mineralization occurs mainly anaerobically. Duckweed can be a problem for management because it obstructs water passage and pumping station and must be removed.

The other groups were less clearly characterized and were distributed along a gradient of habitats.

Group B was represented by the smallest and shallowest ditches, with a reduction of habitat and ecological niches. Group B was characterized by concrete substrate, which led to a reduction of width and to an increase of water velocity, and to higher T mean values and to higher velocity. Higher velocity was also probably responsible for the oversaturation of oxygen recorded there. All the releves in group B were characterized by the presence of *Ranunculus trichophillus*. The associations presided by aquatic *Ranunculus* are typically paucispecific, and *R. trichophillus* is the dominant species among aquatic *Ranunculus* in water with high pH (Lumbreras et al., 2009). Due to water velocity this group did not have floating vegetation. Between groups, B is the only which differs from the other ones for macroinvertebrates and in particular, Chironomidae were the exclusive taxon characterizing this group. The rare presence of other taxa in this group probably indicated a low diversity in microhabitat. The concrete substrate led to a reduction in microhabitats and as a consequence a lowered number of taxa.

Group D included the larger ditches, with variable substrate (concrete, silt, sand). It differed also for higher mean temperature and conductivity values compared with the other groups. The vegetation of group D was mainly represented by algae which suggest a high level of organic load; this group of ditches was the only one without helophyte and supra-aquatic vegetation, i.e., no riparian vegetation. The helophyte *T. arundinacea*, the most frequent species in all this study, was present only in this group because larger ditches are more controlled by management with frequent vegetation cutting or they have concrete banks.
Group E was characterized by lowest mean temperature, and intermediate values of conductivity, velocity and pH values. It is the group which included the highest number of sampling point (13). Substrata were mostly represented by sand and silt, this group contained the sampling points with the largest number of plant species (11 taxa of macrophyte in three of them). This group is the only one including all the macrophytic life form typologies and exclusive plants like *Potamogeton crispus*, *Agropyron repens*, *Carex elata* and *C. hirta*.
The results of the research underline how the ditch network represents a largely variable range of aquatic habitats on a relatively small spatial scale. Their restoration would enhance biodiversity in highly anthropic areas, at the same time efficiently storing water resources which otherwise would be quickly withdrawn from the system.
An ecological management of the ditches would restore the hydraulic connectivity with the surrounding territory; the increased inflow would reduce abrupt variations of the physico-chemical parameters, allowing the permanence of more diverse and structured aquatic communities. Such communities would provide ecosystems benefits typical of floodplain wetlands. Moreover, a fraction of the water delivered by each hydropeak would be absorbed from the ditch network, reducing the hydrological and ecological alterations downstream.

3.6.2 Physical indicators: hydraulic and thermal characterization of the ditches network

The hydraulic characterization of the ditches network was performed by integrating historical with recently collected topography data, together with information on ditches cover for roughness estimation. The network hydrodynamic response to different options of the nearby Noce River water re-use into the network was simulated through 1D unsteady flow model supported by local hydraulic measurements. A simplified one-dimensional model for both surface-subsurface flow exchange and conservation of thermal energy was applied to quantify the lateral extent of the riparian region affected by hydropeaking oscillations. The discharge capacity of existing intake structures from the nearby river reach was quantified through direct flow measurements at some indicative sites with personnel of the local Drainage Boards.

Temperature

The temperature regime in the ditch network varied at different time scales. Daily summer patterns exhibited maximum values ranging from 16.1 to 29.1 °C. The daily excursion varied between 2.8 and 11.8 °C. The differences observed depend on the convection and diffusion properties of the surface flow in the ditches system and on the channel shape and structure. A wide variability was also observed for other chemical parameters as conductivity (180-943 μS/cm), pH (8-9,4) and dissolved oxygen, which ranged from super saturation to almost absent.

Network topography and hydraulic properties

In order to quantify the present hydraulic functioning of the ditches network and the dynamics of hydropeaking-withdrawn wave propagation a series of different data types needed to be collected from a variety of sources. (1) Historical data were made available through the archives of the local drainage boards in charge of the network maintenance. These data included maps, cross sectional and longitudinal profiles of the main and secondary ditches (Figure 3.6.2.1). Reliable data in the historical archive were found to start from roughly 1906 onwards. (2) The correspondence of historical data with the present geometry of the ditch network needed to be checked with ground surveys data that have been achieved through high-resolution GNSS-RTK GPS (Leica Geosystems). This finally allowed obtaining the characterization of the present geometry of each individual channel. (3) Hydraulic measurements were performed locally in most of the ditches with the aim to estimate their average roughness properties. Overall, three main typologies of substrate were observed in the network: fine gravel, macrophytes and aquatic vegetation, a mud layer made of very fine particles. With the aim of quantifying a roughness (Manning) coefficient for hydraulic computations, contemporary water level and current meter based discharge measurements were carried out. This also allowed to check the model predictions of discharge partition at some of the main bifurcations occurring at the nodes of the ditch network. (4) Continuous water level and temperature monitoring (Figure 3.6.2.2) has been performed through sensors placed in the main ditches.

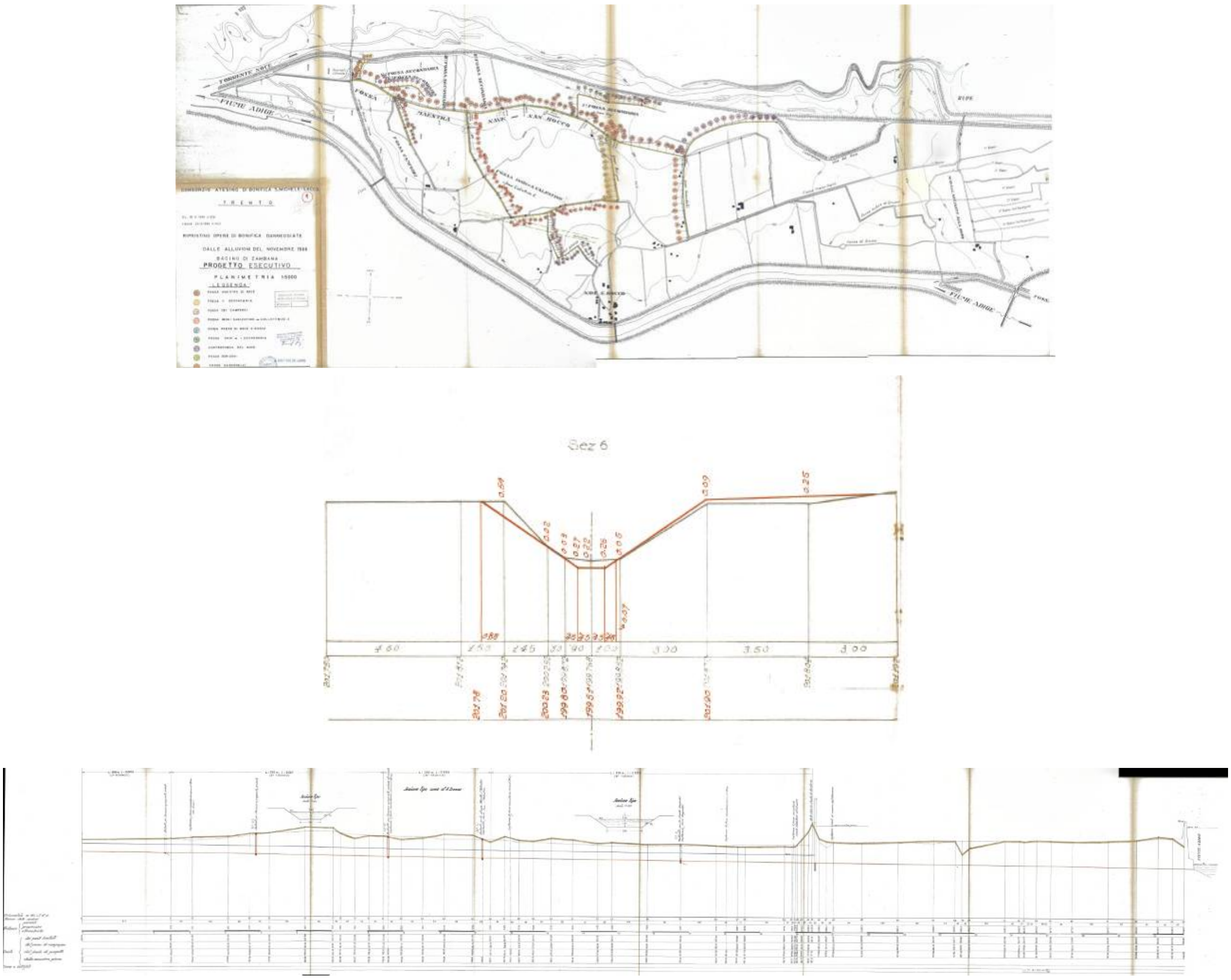

Figure 3.6.2.1: **Example of historical topographic data: map of a portion oft he Rotaliana Plain (upper panel); example of a cross-sectional and longitudinal profile (lower panels).**

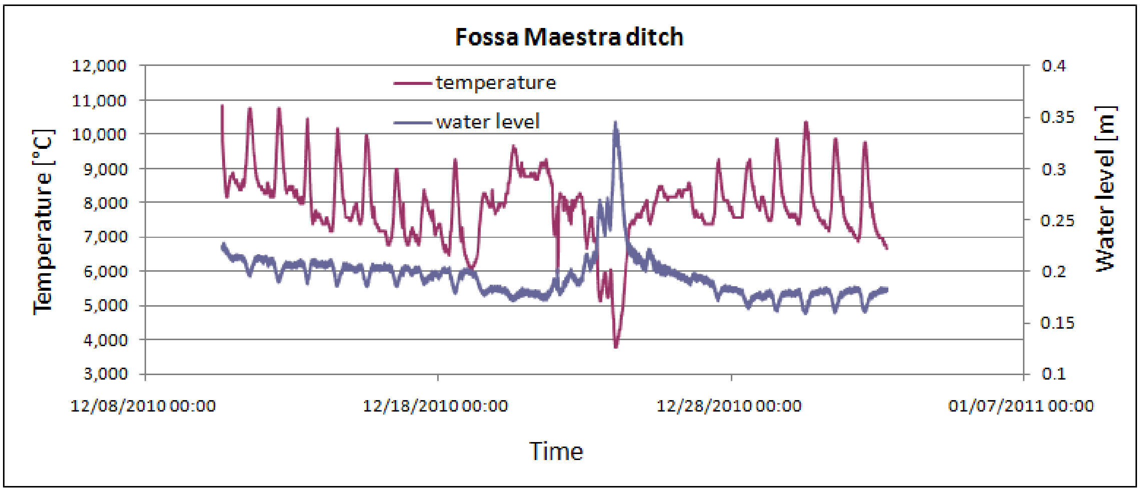

Figure 3.6.2.2: Example of water level and temperature signals at the Fossa Maestra. The data acquisition period in this case has been 1 month at 2′ intervals. The data has been collected during 5 months.

Hydraulic modelling

An unsteady hydraulic model for hydropeaking wave propagation in the ditches network was developed using the freely available and widely used HEC-RAS software on the topographical domain that was built based on the field and hystorical data collection described in the previous section. The hydraulic simulation was restricted to the largest channels, with the overall simulated network planform illustrated in Figure 3.6.2.3.

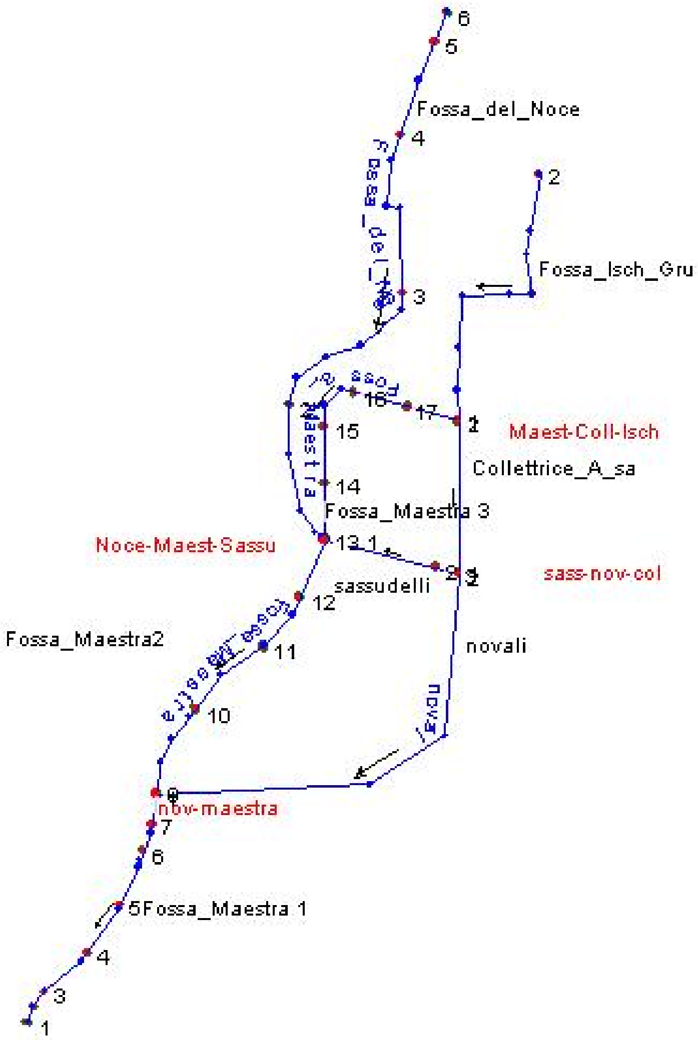

Figure 3.6.2.3: Planform of the ditches network employed for the unsteady hydraulic simulation.

Unsteady hydraulic modeling of the behavior of the main ditches network allowed detecting the key operations that need to be carried out for the withdrawn portion of the hydropeaking wave to affect a large portion of the network, thus increasing the potential for enhancing the expected ecosystem services. Namely, the "Sassudelli" ditch, located in the upper-middle portion of the study network, is characterized by a nearly vanishing slope; for this reason an additional discharge input of only a few hundreds of liters per second would be sufficient to trigger flow reversal, as opposite to the present state with westward flow direction. Achieving eastward flow in the Sassudelli would allow a number of additional nodes and channels to be subject to intermittent hydrodynamic waves and therefore to extend the reuse scenario to the whole floodplain width. Hydraulic modeling also revealed that some structural measures are necessary in the "Collettrice A" and "Novali" ditches in order to remove a 50 cm step that presently determines a much higher water surface elevation in the East portion of the network constituting an additional obstacle to spreading the reuse project across the entire floodplain width. Moreover key indicators to assess the restoration potential, such as hydraulic residence times in surface and subsurface water bodies, extension of terrestrial/aquatic ecotones, dilution rates and variations in the ditches thermal regimes were quantified through the application of the surface-subsurface flow model. Overall, this tool allows reconstructing the inundation dynamics in response to hydraulic waves of different magnitude and duration that correspond to different management options of the intake structure from the Noce River and to the temporal pattern of hydropower production.

Finally, the unsteady hydraulic modeling provided the temporal oscillations of water surface at each location along the ditches network that was needed for the quantification of the additional surface – subsurface water exchanges expected between individual ditches and the riparian aquifer, as consequence of the reused water wave propagation. Such information was used as local boundary condition for the surface-subsurface water exchange simplified model that was developed within the present project and that was applied to estimate the extent of the floodplain that can be affected by the hydraulic exchange and the order of magnitude of the additional aquifer recharge.

A 1D model for the exchanges between the ditches and the floodplain aquifer

With the aim of studying the expected benefits resulting from surface-subsurface water exchanges as a consequence of the re-use of turbinated water into the ditch network, a simple predictive tool was developed to predict the propagation of hydro and thermal waves in the riparian areas. The study system can be represented according to the sketch in Figure 3.6.2.4.

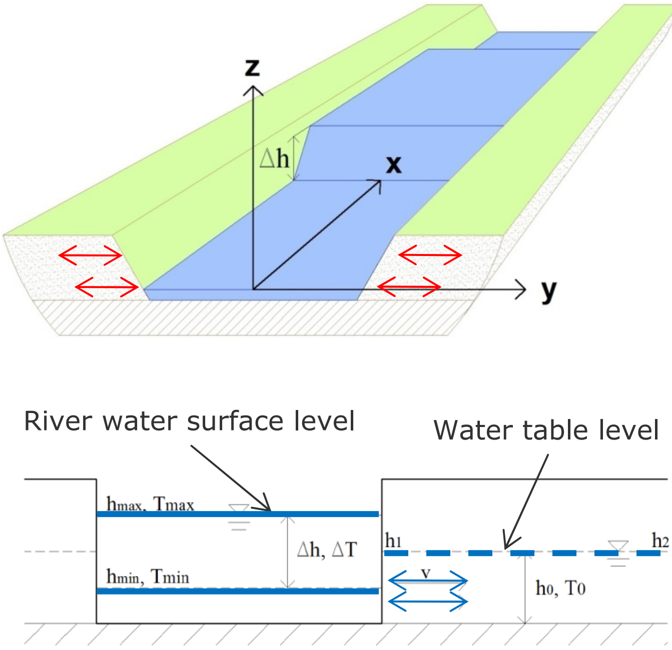

Figure 3.6.2.4: Sketch of the computational domain used by the 1D model for the hydraulic exchange between the ditches and the floodplain aquifer.

The predictive tool is based on the solution of flow (Boussinesq) and heat transmission equation, according to a splitting numerical procedure that solves separately the convection and diffusion problems, acting at different timescales.

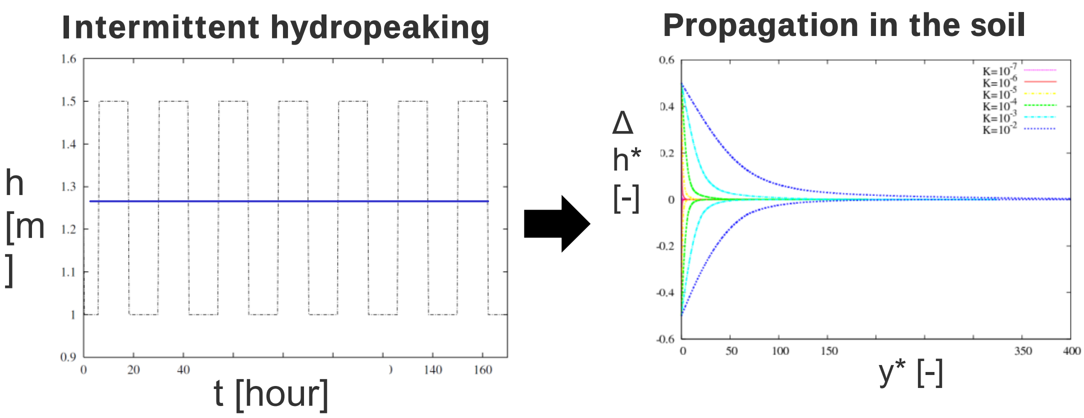

Figure 3.6.2.5: Example of the predicted propagation of the hydraulic wave corresponding to hydropeaking into the floodplain aquifer. Idealized hydropeaking wave in the ditch (left panel); water table at different time steps showing lateral propagation (right panel).

The results quantify the magnitude of hydraulic and heat exchanges between ditches and the surrounding plain as well as provide insight into the dynamics of such interaction. For instance, while hydraulic transport is purely convective in a broad range of soil types, thermal transport is mostly convective in high permeability soils and mostly conductive when the permeability decreases. In particular it is possible to identify the key parameters controlling diel variations in mass and thermal exchanges between the channel and the riparian aquifer. The storage effects of the aquifer and its role in damping hydropeaking waves is also quantified. Finally, the role of longitudinal changes in channel morphology as well as of land cover and seasonal aquifer variations can also be examined through the proposed modelling framework.

Water re-use in the Rotaliana Plain: conclusive remarks

Along rivers crossing Alpine lowland agricultural areas, the possibility of attenuating hydropeaking waves by restoring the flow in nearby agricultural ditches appears to be a very promising way to enhance freshwater biodiversity through the restoration of a mosaic of aquatic habitats on a small spatial scale. Such habitats may also act as proxy for natural wetlands if they are physically restored to a semi-natural condition. The proposed approach also has a good potential for the recharge of the floodplain groundwater, particularly in piedmont alpine areas with relatively high hydraulic conductivity, where surface-subsurface water exchanges are mainly controlled by convection. Groundwater recharge is likely to be more effective for large hydropeaking wave durations and for great length of the total ditches network subjected to water reuse.

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3.7 Fersina River and experimental flumes

Summary

The experimental flumes on the Fersina River were used for a set of simulations to determine the impacts on the benthic community of the hydrological and thermal which occur in the Adige watershed by simulating one hydropeaking wave followed by a thermopeaking one and assessing the effects on benthic invertebrates. Results indicate that the thermal wave following the discharge one may affect the benthos to a higher degree, and these results may be relevant to forecast the summed effect of multiple stressors that occur in heavily modified water bodies.

A second experiment aimed at assessing the effects of an excess (hydropeaking) or lack (Minimum Vital Flow, water scarcity) of variations in the flow regime on the benthic communities of a pristine Alpine steam. The research was conducted in the Fersina River (natural flow), in the set of artificial flumes (constant flow) and in a hydropower impacted reach of the same stream, 200 meters downstream of the flumes. Results results underline the importance of variability in flow rather than velocity in maintaining resilient benthic community.

3.7.1 Invertebrates

Effects of hydropeaking and thermopeaking

Hydropower is a primary and strategic renewable energy source in the Alps, with high economic value and no gas emissions, but with severe local impacts on the ecology of freshwater ecosystems. While the ecological impact of hydropeaking is well known, thermopeaking is still in need of scientific studies (Carolli et al., 2009, 2011; Zolezzi et al., 2010). On a seasonal scale, the daily alterations sum up and in the studied Alpine streams the range is plus 3-4 C° in winter and minus 5-6 C° in summer (Zolezzi et al., 2010).

Because the hydrodynamic and thermal waves propagate downstream with different velocities, with the thermal wave preceding the discharge one (Toffolon et al., 2010), downstream of storage plants with high elevation reservoirs and hypolimnetic releases, the biota experience a first disturbance caused by the hydropeaking wave (catastrophic drift), followed by a second one caused by the sudden temperature change (behavioural drift) Carolli et al., 2011). The time lag between the two events increases with distance from the power station and the magnitude depends on the relative position of the reservoir and the receiving water body.

Aim of this study was to assess the impacts on the benthic community of this sequence of events by simulating one hydropeaking wave followed by a thermopeaking one, in experimental flumes.

Simulations were conducted in the previously described experimental flume system of the Fersina, where we artificially caused a sudden increase of discharge of 2.2 x in flumes B and D (from $5.1*10^{-3}$ to $11.3*10^{-3}$ m^3s^{-1}) and subsequently we decreased abruptly the temperature of 2.4 (flume B) and 3.5 °C (flume D).

Drift samples were collected by filtering the whole volume of water leaving the flumes with drift nets (mesh size 100 micron). For each experiment we collected 4 samples before the experiment for 5 minutes at the beginning of each hour, for the four hours preceding the experiment (four samples x flume, named BF hereafter); during the hydropeaking wave (10 samples x flume, HP hereafter) and during the thermopeaking wave (11 samples x flume TP hereafter) we collected the drift continuously at 2 minutes intervals. Number of collected individuals were transformed in densities and expressed in n. ind. m^{-3}. Benthic density was measured one day before to the experiment by collecting three samples in each flume with a Hess sampler (23.5 cm diameter, 100 µm mesh) and used to calculate mean densities. Propension to drift was calculated as mean drift density/mean benthic density and expressed as n. ind. m^{-1}. Values were calculated separately for the samples collected before the simulations, during HP, and during TP.

The density of drifting invertebrates increased a first time during the hydropeaking simulation, followed by a second increase during the thermopeaking one (Figure 3.7.1.1).

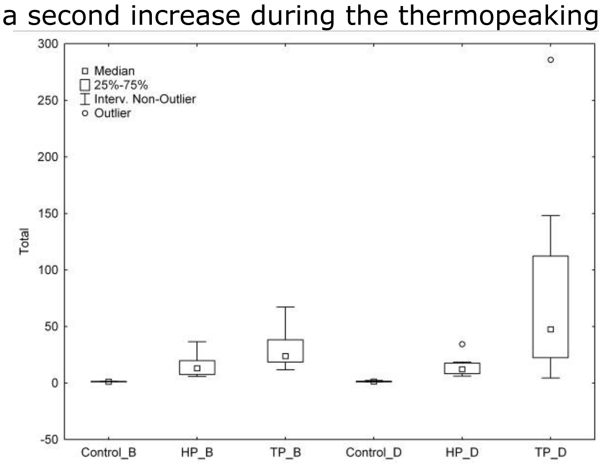

Figure 3.7.1.1: Number of drifting invertebrates collected in the two artificial flumes (B and D) before the simulation (BF), and during the hydropeaking (HP) and thermopeaking (TP) simulations.

The slight but abrupt increase in discharge caused 28 and 24-fold peak increases in drift in flumes B and D respectively (Figure 3.7.1.2), while mean increase before and during the hydropeaking simulation experiment was 12 and 10-fold. During the TP experiment, more invertebrates drifted, particularly in flume B, which experienced a higher thermal shift: peak increase measured 36 in B and 198 in D with a mean increase of 23 in B and 56 in D.

	FlumeB	Flume D
ΔQ ($l\ s^{-1}$)	2.2	2.2
ΔT (°C)	2.4	3.5
A: mean drift before	1.3	1.4
B: mean drift during HP	15.3	14
C: mean drift during TP	29.8	80.9
B/A: drift increase ratio during HP	11.7	9.7
C/A: drift increase ratio during TP	22.7	56
D: max drift before	1. 8	2.2
E: max drift during HP	36.5	34.4
F: max drift during TP	67.1	285.8
E/A	27.9	23.8
F/A	35.8	197.6

Figure 3.7.1.2: Main physical and biological parameters of the hydropeaking (HP) and thermopeaking (TP) simulations. Drift values expressed as ind m^{-3}.

Drift propensity increased in both flumes during both HP and TP simulations, with higher values in the latter. We could identify three kinds of drift responses (Figure 3.7.1.3):
a) taxa which tended naturally to drift (i.e. they were proportionally more abundant in drift before the simulations than in benthos) and responded to the discharge and/or thermal alterations. This is the case of Chironomidae whose drift propensity increased 15- and 22-fold in flume B for HP and TP respectively, and 11- and 51-fold in flume D, and of Simuliidae, whose propensity to drift increased of 14- and 25-fold in flume B, 8- and 41-fold in flume D;
b) taxa which did not tend to drift (i.e. they were proportionally more abundant in benthos than in drift before) but responded to the discharge and/or thermal alterations by increasing their propensity to drift, i.e. Trichoptera, Baetidae, Other Diptera, Acarina, Harpacticoida and Oligochaeta;
c) taxa which were not or only slightly affected by HP and TP, such as Heptagenidae and Plecoptera, whose increase in drift propensity was negligible.

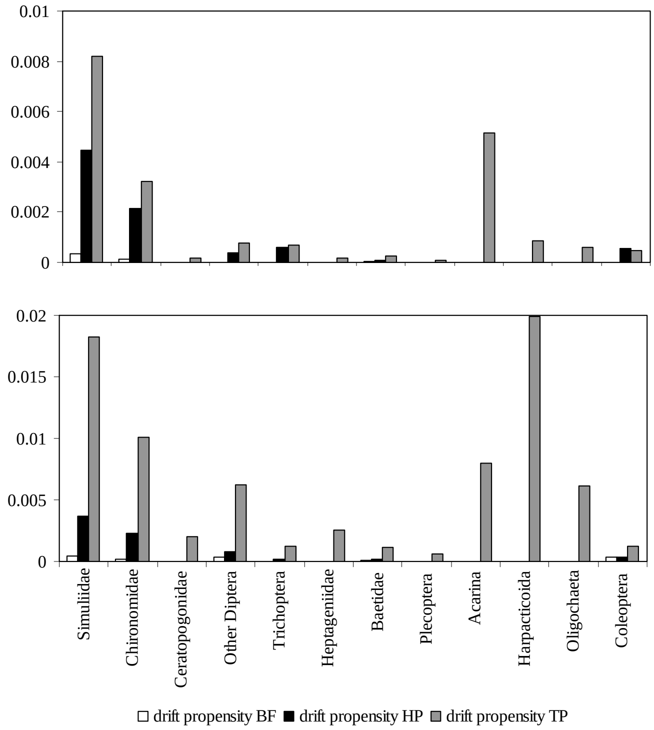

Figure 3.7.1.3: Drift propensity of main taxa in flume B (top) and D (bottom) before simulations (BF), and during hydropeaking (HP) and thermopeaking (TP) simulations.

In a previous study in the same flume system (Carolli et al., 2011) we assessed how abrupt changes of temperature induce drift in benthic invertebrates, and how such drift is probably behavioural, whereas the well-known effects of sudden increases of discharge is the induction of catastrophic drift (e.g. Céréghino et al., 2002; Hay et al., 2008; Bruno et al., 2010). Thus, in the short term, the invertebrate community composition in the hydropeaking-impacted reaches changes because invertebrates undergo behavioral drift in response to thermopeaking and catastrophic drift in response to the discharge wave. Behavioral and catastrophic drifts can occur as distinct events. In this study we attempted to reproduce such effect manipulating discharge and temperature in two artificial flumes. Results indicate that the thermal wave following the discharge one may affect the benthos to a higher degree.

The effects of abrupt increases of discharge and changes of temperature may vary according to the time shift of their arrival. In this case we tested the case of two separate waves. Normally this condition does not occur as in most cases the two waves are concomitant or overlap, as the duration of each hydropeaking event is of several hours and so it over-imposes over the temperature one. More recently though, the operation of power plants is increasingly market-oriented and so short operational shifts (less than one hour) are becoming frequent and so the separation of the two waves may occur at relatively short distances from the power plants. For this reason the results of this study may be relevant to forecast the summed effect of multiple stressors that occur in heavily modified water bodies. Further research is needed to verify the role of the time lag between one hydropeaking event and the associated thermopeaking, and their cumulative effect at different time scales. Finally different taxa behaved differently in reaction to these impacts and thus different communities may be selected by different operational modes of power plants and their relative distance.

Alterations of the Natural Flow Regime

One station were selected in the natural flow reach of the stream (NFR), five in the hydropeaking-impacted reach (HI) and five in flumes (i.e. one for each of the five flumes) (MVF). At each station in the streams and in each flume we set sets of three replicates of Hester-Dandy artificial substrates (Figure 3.7.1.4.), and from February 2010 we started collecting macroinvertebrates every two weeks from one substrates for each replicate/flume. In addition, one quantitative benthic sample per station or flume was collected with a Hess bottom sampler, and Simuliidae were also sampled with specific substrates (floating plastic strips) (Figure below). Each station was sampled biweekly from mid-February; sampling ended in mid-August, due to damage caused by an exceptional flood.

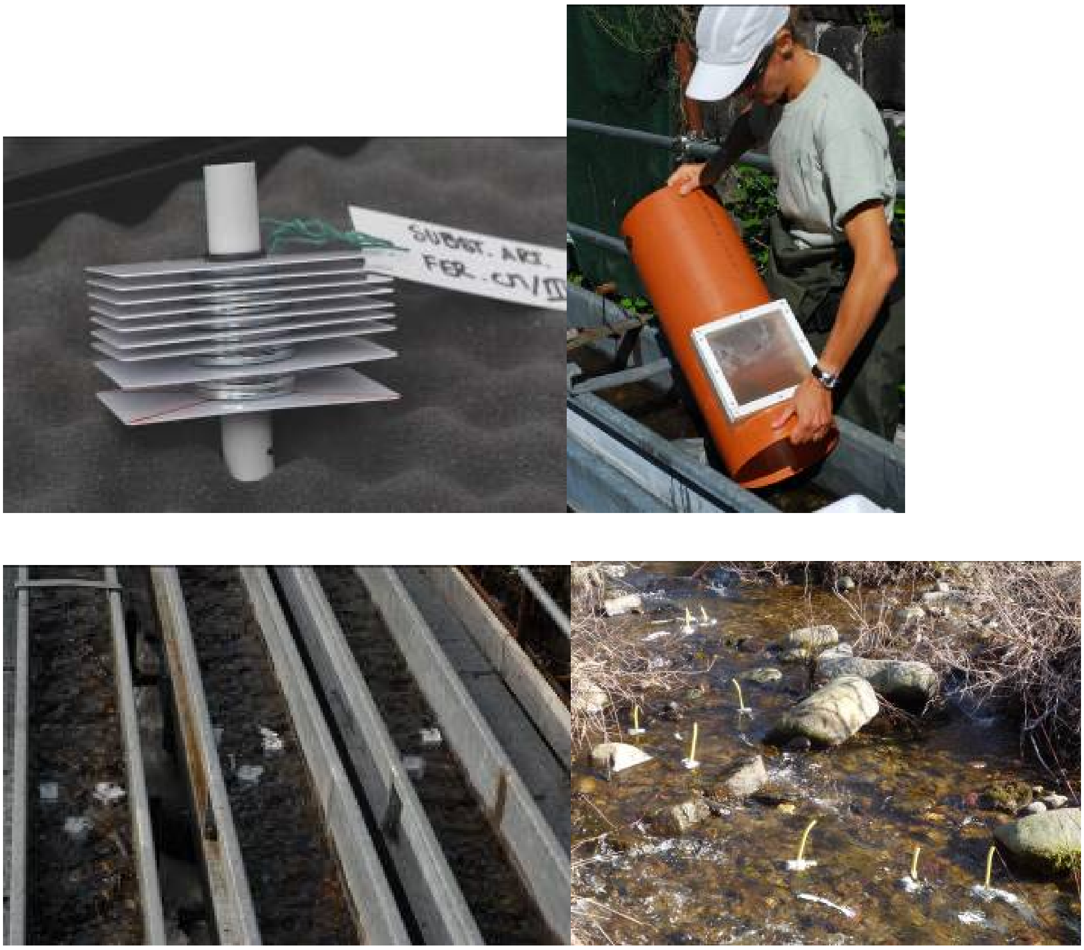

Figure 3.7.1.4: From left to right. Top: Modified Hester-Dandy substrate; Hess bottom sampler. Bottom: set of three substrates in each of the five flumes; five replicates of a set of three substrates in instream station.

A total of 59 taxa were recorded, for the analysis we used Ephemeroptera, Plecoptera and Trichoptera, which were identified to the genus or species level, and selected the most common taxa not belonging to Ephemeroptera, Plecoptera and Trichoptera (i.e. with densities >0.5% of total density for the whole study, and present in >5% of samples). Four taxa were selected: Diptera Chironomidae and Simuliidae, Copepoda Harpacticoida and Coleoptera Elmidae.
We detected a selective effects of the sampling methods, as expected, due to the fact thet substrates had very little FPOM, and were rich in CPOM (leaves, twigs): EPT and Chironomidae were proportionally more abundant in the Hess samples, whereas the substrates were selectively chosen by Simuliidae. Elmidae and Harpacticoida were present only in Hess samples, being probably more strictly associated with FPOM (Figure 3.7.1.5.).

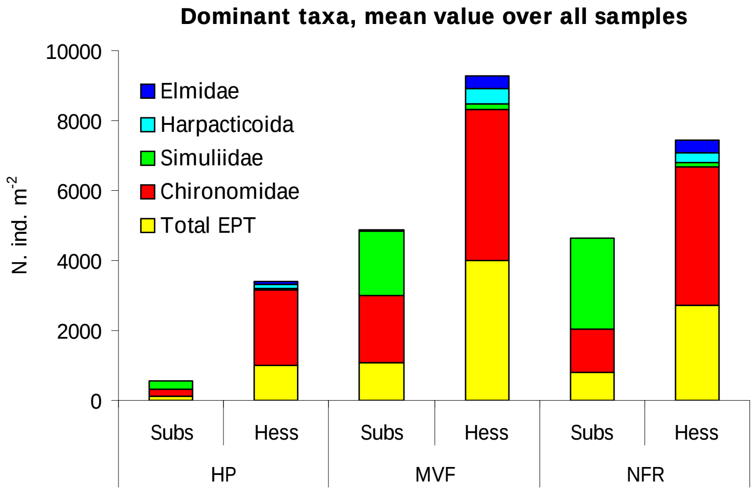

Figure 3.7.1.5: Cumulated mean density (over five replicates of Hesse samples for flumes, over five replicates of substrates for each station) of EPT taxa and dominant taxa.

Because flow determines the amount of organic matter and thus community structrure, we grouped EPT by feeding guilds. In the HP impacted station, substrates had low densities, no predators, few detritivore and filter feeding taxa, the benthic (Hess) samples had low densities, no predators, few detritivore and filter feeding taxa (Figure below). This reduction in abundance and diversity of feeding guilds is probably due to to the removal or organic matter and also of individuals (i.e., high rate of catastrophic drift) due to the sudden increases in discharge.
At the MVF site (i.e., the flumes), (Figure below) substrates were well-colonized and with high abundances from March, a reduced temporal variability, and the community was dominated by generalistic taxa becauseorganic matter accumulates due to the lack of floods. The lack of floods causes accumulation of organic matter that can favor higher abundances in early colonization stages, but in the long term can lead to colmation, habitat homogenization (reduced temporal turnover in feeding guilds).
The NFR station (Figure 3.7.1.6.) had higher seasonality for both substrates and Hess samples, more specialized EPT taxa (large predators such as the stoneflyes *Dinocras* sp., and *Isoperla* sp., passive filter feeders such as the caddisfly *Philopotamus* sp.), indicating a higher resilience for this community.
Overall, results underline the importance of importance of variability in flow rather than velocity in maintaining resilient benthic community, and thus for streams with MVF, the importance of morphology in determining the velocity of the "small" amount of water.

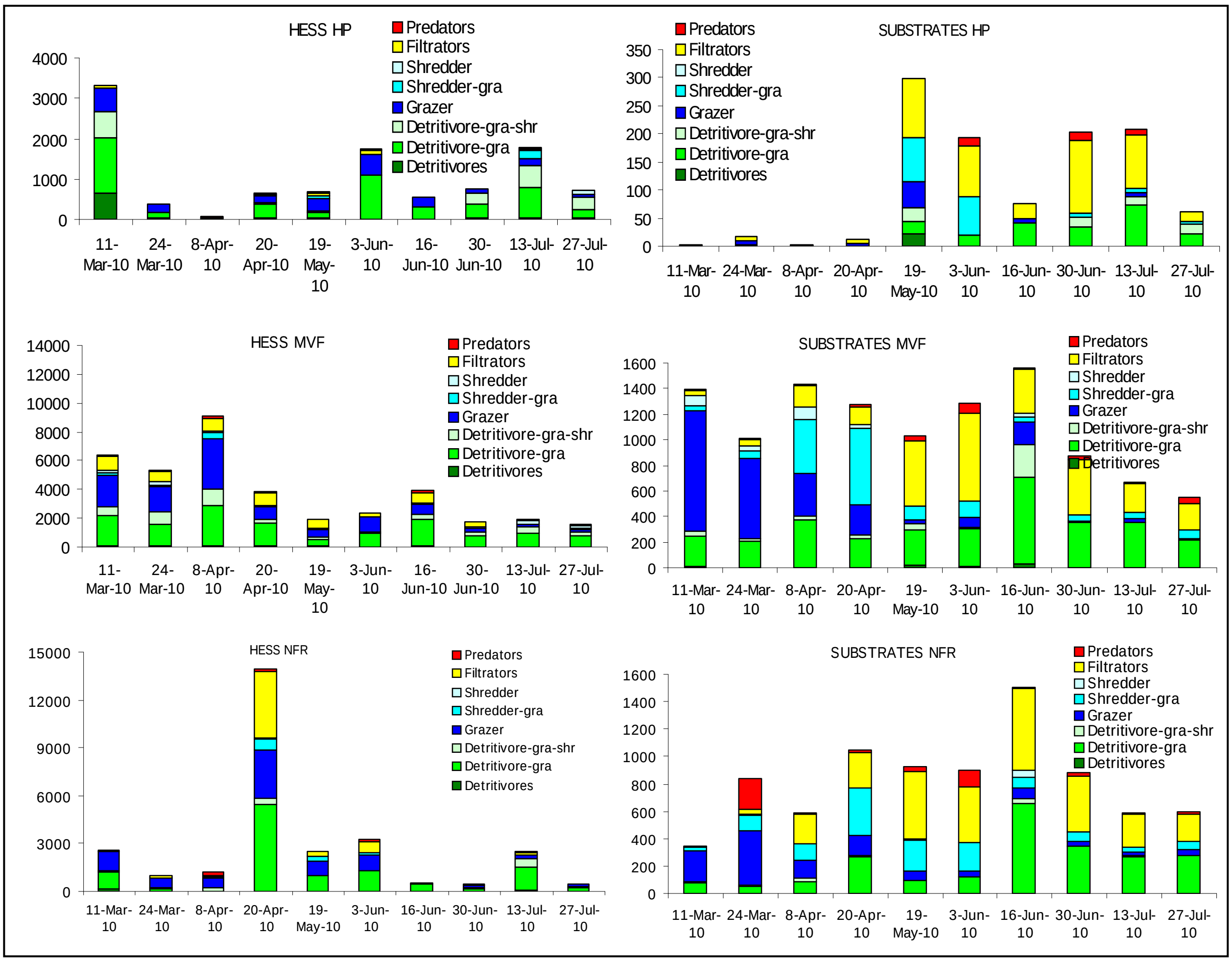

Figure 3.7.1.6. Total abundances of EPT feeding groups (expressed as n. ind. m^{-2}) in substrates and Hess samples for the three flow regimes.

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3.8 High Arly Basin

3.8.1 Indice Biologique Global Normalisé (IBGN)

(Standard global biological index)

NF T90-350 (2004-03-01). Water quality – Determination of the Standard Global Biological Index (IBGN). Not used in Water Framework Directive of European Union.

Source : https://hydrobio-dce.cemagref.fr/normes-et-normalisation

Application field

The IBGN index allows to evaluate general quality of a stream, by the way of an analysis of benthic macro-invertebrates, which are considered as a synthetic expression of this general quality. Applied to a running water site considered separately, this method allows to estimate hydro biological quality in a general typological range except spring zones, some streams downstream from bigger ones and atypical environment as canals and estuaries. Applied comparatively (for example upstream and downstream from a discharge), the method permits to estimate, within the limits of its sensibility, the effect of a perturbation on the environment.

Principle

Sampling of benthic macrofauna (diameter higher than 500micrometers: insects, mollusk, shellfish…) by station, according to a sampling protocol taking in consideration different types of habitat, which are defined by the nature of support and speed flow. After selecting and identifying sampled taxons, we determinate the taxonomical variety of the sample and its indicator group (grouping of taxons). Thus, we can determinate an IBGN index by station, expressed by a note from 0 to 20 (20 which represents the better quality of a stream). Application in mountain, we have to verify if used taxons for the determination of IBGN are present in mountain streams. To complete this sampling, two site of physic-chemistry sampling are created. Different parameters are collected like as nitrates, nitrogen, sulfates, chloride, calcium, phosphate…

4. Using indicators and indices to assess ecological effect on ecosystem goods and services

Ecosystem goods services are the myriad benefits that people derive from nature. These include provisioning services such as food, water, and fiber; regulating services such as climate regulation and pollination; aesthetic services including recreation and spiritual well-being; and supporting services such as bedrock weathering and nutrient cycling. A growing body of work is starting to make the case for how information on ecosystem services can strengthen public and private sector development strategies and improve environmental outcomes. If current trends of ecosystem degradation are to be reversed, it is an urgent priority to integrate ecosystem service considerations into mainstream economic planning and development policy at all scales. Doing so will require tools and approaches that communicate the value and condition of ecosystem services to policy-makers and help them integrate this information with social and economic indicators. Indicators simplify information so that it can be easily communicated and intuitively understood. With indicators, policy-makers can base decisions on evidence, identify and prioritize interventions, track progress toward goals, and inform corrective action in a timely fashion. Up to now most indicators used for ecosystem services have been adopted from narrower environmental fields such as biodiversity, ecology, and climatology, and from economic sectors such as agriculture, forestry, and fisheries. For example, indicators such as crop or livestock production are drawn from economic accounts and agricultural census data. Data for indicators such as tourist visits and spending are drawn from tourism boards. Others, such as carbon storage capacity, deforestation rates, and air quality indexes, are drawn from the environment sector. This reliance on diverse existing indicators provides a necessary starting point for ecosystem service indicators. However, relying on indicators that were developed for other fields should be seen as an interim strategy. The indicators applied in this ecosystems assessments were developed for a variety of purposes. They contribute to communicate communicate the contributions of ecosystem services to human well being and help policy-makers to integrate ecosystem services into broader policy dialogs and decisions and they provide a first framework to quantify impacts from water scarcity and water usage towards the development of an optimal ecological discharge fulfilling the need of ecology and economy.

5. Ecological and management application

In many areas of the world, growing human populations are rapidly depleting available freshwater supplies. During the 20th century, the global human population increased fourfold to more than six billion (6×10^9). Water withdrawn from natural freshwater eco-systems increased eightfold during the same period. Facing an ominous specter of increasingly severe water-supply shortages in many areas of the world, social planners and government leaders are exploring strategies for managing water resources sustainably. This quest for sustainability typically centers on managing human uses of water such that enough water of sufficient quality is available for use by future generations.

In the endeavor to manage water to meet various human needs, however, the water needs of freshwater species and ecosystems have been largely neglected. The ecological consequences have been tragic. The alteration of river flow regimes associated with dam operations has been identified as one of three leading causes, along with nonpoint source pollution and invasive species, of the imperilment of aquatic animals. Freshwater ecosystem services and products valued by society have been severely compromised as well.

The water needs of humans and natural ecosystems are commonly viewed as competing with each other. Certainly, there are limits to the amount of water that can be withdrawn from freshwater systems before their natural functioning and productivity, native species,and the services and products they provide become severely degraded. Water managers and political leaders are becoming increasingly cognizant of these limits as they are being confronted with endangered species or water quality regulations, and changing societal values concerning ecological protection. During the past decade, many examples have emerged from around the world demonstrating ways of meeting human needs for water while sustaining the necessary volume and timing of water flows to support affected freshwater ecosystems (Richter et al., 2003)

In this report we offered a general set of indicators and indices suitable to be integrated into the development of an ecologically sustainable water management program, drawing upon examples from around the Alpine Arc, with a focus on river systems.

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6. Main challenges in ecological quantitative descriptors definition for future integration in water management new strategies

The ultimate challenge of ecologically sustainable water management is to design and implement quantitative descriptors in a water management program that stores and diverts

water for human purposes in a manner that does not cause affected ecosystems to degrade or simplify. This quest for balance necessarily implies that there is a limit to the amount of water that can be withdrawn from a river, and a limit in the degree to which the shape of a river's natural flow patterns can be altered. These limits are defined by the ecosystem's requirements for water. Human extraction or manipulation that exceeds these limits will, in time, compromise the ecological integrity of the affected ecosystems, resulting in the loss of native species and valuable ecosystem products and services for society.
With human uses of water and our understanding of ecosystems continually evolving, the solutions for meeting both ecosystem and human needs will evolve over time as well. Thus, ecologically sustainable water management is an iterative process in which both human water demands and ecosystem requirements are defined, refined, and modified to meet human and ecosystem sustainability now and in the future, rather than a single, one-time solution. This implies an aggressive and continual search for compatibility between ecosystem and human water needs, developing quantitative descriptors which are easy to apply and requires a commitment from all parties to ongoing participation in an active dialogue (Richter et al., 2003).
However present and emerging challenges in water management must take an integrative approach. Most novel developments in ecological quantitative descriptors must be integrated from a holistic catchment scale that incorporates ecological impacts along with societal viewpoints as well as acceptance. Present challenges that require effective quantitative descriptors include hydropower expansion and river restoration, especially in the alpine regions. Hydropower, in particular, must be evaluated in terms of sustaining flow regimes and minimizing the impacts of hydropeaking. New designs are currently being implemented and new descriptors may be needed to measure success or failure. For instance, the 2010 revision of the Swiss Federal Water Protection Law will require incorporation of novel descriptors for restoration measures implemented to reduce the effects of hydropeaking such as through slow release reservoirs or providing more space for some fluvial systems. The increase in small hydropower plants also beckons the need of prioritizing catchments for development while sustaining ecological integrity and ecosystem services.

An emerging issue in most alpine countries is the input of micropullutants into rivers and streams. Micropollutants derive from both point and non-point sources, requiring new techniques for identification and description. New water laws are being implemented currently that require WWTPs to reduce the point source of micropollutants, and other quantitative detection technologies are being developed to better define non-point sources of micropollutants, e.g. from gardens and agricultural lands or urban inputs. Although a variety of options are available to reduce inputs of micropollutants, there is still a strong need for the development of quantitative descriptors and the effects of these pollutants on aquatic environments.

Introduced and invasive species have a long history with humans in the alpine landscape. Although much of the impact from invasives is in more lowland rivers via water transfers across basins, introduced and invasive species are expected and more likely in alpine regions under scenarios of climate change. Novel aquatic taxa are already colonizing alpine streams as these landscapes are being transformed, and adequate quantitative descriptors are needed to measure possible ecosystem impacts and changes in ecosystem properties. Limitations to invasive spread may require some descriptor akin to an early warning system. Restoration may even assist in the spread of some invasive species.

An integrated water management perspective is clearly needed to meet these emerging issues and adapt to unexpected issues that emerge in the future, a difficult task at best, that must include principals of predictive capabilities, uncertainty, and decision processes. Regardless, transparency is imperative in any integrated management program, and should be part of indicator development, especially to reduce conflicts between interest groups while meeting federal regulations. Changes in water laws and implementation of the water framework directive require quantitative indicators for monitoring change, and consistent monitoring is needed to quantitatively test developed indicators. The number and types of indicators will obviously change as new challenges emerge, although most current measures will need to be sustained and integrated to maintain monitoring needs and practices.

We have developed a initial set of quantitative indicators suitable to be integrated in an ecologically sustainable water management program. There are many entry points into this process, but our experience suggests that the development of quantitative descriptors is essential to achieving ecological sustainability.