ECOLOGICAL RESPONSE OF A MULTI-PURPOSE RIVER DEVELOPMENT PROJECT USING MACRO-INVERTEBRATES RICHNESS AND FISH HABITAT VALUE

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Summary

It has been acknowledged that river morphology and hydrology have been intensively altered due to the anthropic demands in floodplain land use and management, flood protection, promotion of navigability or energy production. Rivers were transformed in water *highways*, having lost contact with their surrounding floodplain as well as the plethora of ecological processes and occupants once thriving in these ecotonal zones. The identification of this emerging threat of morphological and hydrological alteration on ecological integrity adds further complexity in the exploitation of hydrosystem resources. These resources are heavily coveted and guarded by different lobbies each having strategic views on future project development. Stakeholders may want to promote hydro-electricity, ecologists a natural reserve, communes may wish to have an increased flood protection and leisure promoters a nautical center. As a result, the proposition of a river development project is certain to face opposition of one party or the other. The motivations of this dissertation are anchored in this context, where various and sometimes conflicting potentials for hydrosystem exploitation remain.

This works aims at contributing scientifically to an innovative approach at the conception phase of a multi-purpose river development project by developing the ecological module to be implemented in the general project's optimizer. The `SYNERGIE` project hypothesis is that it should be possible to identify a synergetic pattern joining the interests of ecological integrity, flood safety, energy production and leisure development. Such a multi-objective river development project would stand more chance of acceptance. This dissertation focuses on the ecological aspects of such a river development project and an application on the regulated Swiss Upper Rhone River. Is expected an ecological answer to a river development project design / management which has to be compatible with Heller's Heller (2007) general `SYNERGIE` project optimizer taking into account all the project poles. The system of interest is composed of a buffering reservoir of *ca.* 1 km^2, a run-of-the-river dam, a hydro power-plant, and an artificial river ensuring longitudinal continuum.

The primary part of the work consisted in an extensive literature review on system understanding, anthropic alterations and quality assessment / prediction tool available. The approach consisted of two levels (1) the general ecological considerations to be followed at the project reservoir scale and (2) the measure of the

downstream ecological response through modeling. General ecological considerations at the reservoir scale were the implementation of an artificial river ensuring longitudinal connectivity, implementation of artificial ecotonal boosters and the allocation of a sanctuary zone with limited public access. The downstream measure of ecological integrity was based on the choice of three taxonomic groups of macroinvertebrates and four ecological guilds (groups) of fish. Mayflies (Ephemeroptera), stoneflies (Plecoptera) and caddisflies (Trichoptera) richness were predicted using simple hydrological and morphological covariates (i.e. substrate, current speed,...) coupled to system specific faunistic surveys. Bank, riffle, pool and midstream fish guilds habitat values were determined using existing methods. By using the simulation results of river development project scenarios as inputs, the ecological response (i.e. the measure of ecological integrity) was computed following the assumptions that high predicted macro-invertebrate richness and high guilds habitat values were linked to a high ecological integrity. An emphasis on the hydropeaking effect in relation with river morphology was performed on macroinvertebrates. They were found to respond well to hydrological and morphological changes induced by river development projects while the approach by fish habitat value encountered limitations in its applicability.

Four multi-objective project scenarios were analyzed, (1) the actual state of the Swiss Upper Rhone River at the Riddes site (VS), (2) a hypothetical hydropeaking mitigation project, (3) a hypothetical bed widening project and (4) a hypothetical bed widening coupled to hydropeakaing mitigation project. The actual state resulted in the worst measure of ecological integrity, with comparable results for hydropeaking mitigation project or the bed widening project. The highest measure of ecological integrity was observed for the coupling of hydropeaking mitigation and bed widening.

These results showed that a multi-purpose project can increase the ecological integrity of the Swiss Upper Rhone River, produce electricity, provide protection from floods and develop local leisure activities. The synergetic effect of the project could be measured by project acceptance. Nevertheless, our knowledge on the hydropeaking effect on hydrosystem should still be completed by more research in order to give more weight to the ecological implication of hydropeaking.

Keywords: *ecological integrity; multi-objective; zoobenthos richness; fish-guilds; model selection; habitat suitability; hydropeaking*

Résumé

Il est reconnu que la morphologie et l'hydrologie des rivières ont subi des altérations profondes à cause des pressions humaines quand à l'utilisation et la gestion du territoire, la protection contre les crues, la promotion de la navigabilité et la production énergétique. Les cours d'eau se sont vu transformés en *autoroutes* fluviales, ayant perdu une bonne partie de leur contact avec leur plaine alluviale ainsi que de nombreux occupants et processus écologiques typique aux zones écotonale. L'identification de la menace causée par l'altération morphologique et hydrologique sur l'intégrité écologique rend d'autant plus complexe les prises de position dans le cadre de l'exploitation des ressources de l'hydrosystème. Ces ressources sont passablement convoitées et gardées par différent lobbies, ayant chacun des vues stratégiques propres quand aux possibilités de développement futur de l'exploitation du système. Des acteurs pourraient vouloir promouvoir la production d'énergie hydro-électrique, les écologistes pourraient revendiquer la création d'une réserve, les communes défendre l'aspect sécuritaire de la protection contre les crues ou le développement de zones de loisir en contact avec l'eau. Il en résulte que chaque proposition de projet de développement riverain à de fortes chances de soulever des oppositions.

La motivation de cette dissertation est ancrée dans se contexte, ou existent des potentiels variés de développement touchant au ressources riveraines. Ce travail vise à la contribution scientifique d'une approche novatrice dès la phase de conception du projet SYNERGIE par le développement du module écologique. L'hypothèse de base du projet étant qu'il est possible d'identifier une configuration synergétique joignant les intérêts de l'écologie, de la protection contre les crues, de la production énergétique et du développement de zones de loisir à l'aide d'un projet multi-objectif. Cette configuration multi-objective devrait contenter chacune des parties et vise à l'acceptation finale du projet. Cette dissertation traite des aspects écologiques d'un tel projet avec une application au Haut Rhône valaisan. Est attendue une réponse écologique au design / à la gestion d'un projet multi-objectif de développement de rivière compatible avec l'optimiseur général de projet. Le système d'intérêt comprend un réservoir tampon d'une surface d'environ 1 km2, d'un barrage au fil de l'eau couplé d'un ouvrage de production hydro-électrique. Une rivière artificielle ferait office de maintien du continuum longitudinal afin de minimiser l'effet obstacle du barrage.

La première partie de ce travail consiste en la revue bibliographique de l'état des connaissances concernant le système d'intérêt, mais aussi l'ensemble des impacts anthropiques ainsi que des outils d'évaluation et de prédiction de la qualité des rivières à disposition. Une approche à deux niveaux a été retenue : (1) des considérations écologiques générales à l'échelle du réservoir et (2) de la mesure de la réponse écologique à l'aval du projet par la modélisation. Les considérations générales à l'échelle du réservoir étant (a) la planification d'une rivière artificielle assurant le continuum longitudinal (b) la mise en place d'une structure artificielle visant à promouvoir la zone écotonale aquatique-terrestre et (c) l'allocation d'une zone sanctuaire d'accès limité. La mesure de l'intégrité écologique à l'aval du projet étant basée sur le choix de trois groupes taxonomiques de macroinvertébrés ainsi que quatre guildes écologiques de poissons. Les richesses des éphémères (Ephemeroptera), plécoptères (Plecoptera) et trichoptères (Trichoptera) ont été prédites à l'aide de variables hydrologiques et morphologiques simples (par ex. granulométrie, vitesse du courant) couplées à des relevés faunistiques du système d'intérêt. Des notes d'habitat ont été prédites selon quatre guildes écologiques de poissons (berges, radiers, mouilles et cours plein) à l'aide de méthodes existantes. En utilisant les résultats hydro-géomorphologiques issus de la modélisation des scénarios par le volet hydrologique comme entrées dans le module, la réponse écologique (la mesure de l'intégrité écologique) est déterminée selon l'hypothèse de base qu'une richesse prédite élevée des macroinvertébrés et des valeurs d'habitat élevées pour les guildes de poissons sont liées à une intégrité écologique élevée. Un effort particulier a été fourni sur l'effet des éclusées avec prise en compte de la morphologie sur les macroinvertébrés. Les macroinvertébrés se sont avérés avoir la meilleure réponse sur les modifications hydrologiques et morphologiques alors que la méthode des guildes de poissons a présentée de nombreuses limitations dans son applicabilité. Quatre scénarios ont été analysés à l'aide de la méthode : (1) l'état actuel du Rhône valaisan au site de Riddes (VS) (2) un projet hypothétique visant à tamponner les éclusées tel que présenté par SYNERGIE (3) un projet hypothétique d'élargissement du lit (4) le couplage de l'élargissement à la mitigation des éclusées. La mesure de l'intégrité écologique étant la moins bonne dans le cas du Rhône actuel (scénario 1), avec des résultats comparables pour les scénarios 2 et 3 et la meilleure pour le scénario 4, couplant l'élargissement à la mitigation des éclusées. Ces résultats ont montré qu'un projet multi-objectif est susceptible d'améliorer l'intégrité écologique du Rhône valaisan, produire de l'électricité, contribuer à la protection des inondations et au développement des activités de loisir. Une mesure de l'effet de la synergie peut être l'acceptabilité du projet, qui devrait se trouver améliorée par l'approche utilisée. Il parait néanmoins que notre connaissance de l'effet des éclusées mériterait d'avantage de recherche de manière à mieux cerner l'implication écologique des éclusées.

Mots-clés: *intégrité écologique; multi-objectif; richesse du zoobenthos; guildes de poisson; sélection de modèle; valeur d'habitat; éclusées*

Contents

Chapter 1

Introduction

1.1 Motivation – the SYNERGIE project

'Methods and strategies for the analysis of possible synergies within a multi-purpose run-of-the-river hydrolectric scheme'

River development projects in Switzerland and other industrialized countries having already the majority of their river regulated are rightfully under great pressures from the social, economical (flood safety and energy production) and environmental lobbies. Our floodplains, rivers and wetlands are heavily impacted by years of anthropic water and land use management (Amoros and Petts; 1993; Poff; 1997). Spatial, financial and ecological resources are scarce and heavily coveted. Hence a single objective project stands virtually no chance of being accepted in today's conflicting conjuncture. Acceptance may only come from a truly multi-objective project addressing social, economical and environmental demands from the very beginning of the conception phase.

Besides the purely technical (and economical) aspects of energy production, flood protection, water storage and others, run-of-the river schemes are under certain conditions susceptible of improving the overall environmental and landscape conditions of our floodplains. Hydrology can be improved by reducing *hydropeaking*[1], increasing low flow discharges and ensuring hydrological variability. Floodplain morphology can be diversified by the creation of non-existing biotopes such as standing bodies of water. Social benefits through the development of a leisure center or landscape perception enhancing can also be addressed.

[1]corresponds to raising or falling discharges caused either by the turning on or off of hydro-turbines (Gore; 1985)

Although broadly studied unilaterally, to my knowledge these aspects were never put together into a single run-of-the-river development scheme at its very conceptual phase. Social, environmental and financial parameters are in strong interaction and yet, difficult for our minds to quantify because of the incompatibility in their qualifying and quantifying units. The conception of a truly multi-objective run-of-the-river scheme becomes a complex system of interacting modules (i.e. social, economical, environmental) in which the optimum becomes hard to distinguish. In order to identify and promote the *synergies*[2] within such a system at its very conceptual phase, a new methodology able to bring together and set the frame for this multiple-objective run-of-the-river scheme optimization is necessary.

The `SYNERGIE` project goes beyond the fundamental aspects of a multidisciplinary approach and is motivated by a strong practical motive. The project is conducted by the *Ecole Polytechnique Fédérale de Lausanne*'s *Laboratory of Hydraulic Construction*. The method aims at identifying the inherent social, economical and environmental parameters, to analyze and quantify their interactions and implement a common model of such a scheme in order to optimize its design and management. The Swiss Upper Rhone River, together with the project of its third correction[3] will serve as a case study for the project.

My responsibility within `SYNERGIE` lies in the environmental pole and hence is referred to the **Ecological Module** (EM). Through the identification of the various ecological players in the context of the scientific principles addressing ecology in a hydrosystem, a scientifically sound *ecological consequence must be predicted from scheme design variables and operation variables*. The resulting *ecological consequence* has to be designed in such a way that it is:

- environmentally relevant and integrative in order to effectively defend the ecological values of such a system
- simple enough so that it can be used practically
- the EM must be integrable in the main optimizer at source code level (i.e. as a *package*)

[2]from the Greek *synergos* meaning working together, *circa* 1660 – refers to the phenomenon in which two or more discrete influences or agents acting together create an effect greater than that predicted by knowing only the separate effects of the individual agents. (`http://en.wikipedia.org/wiki/Synergy`).

[3]`http://www.vs.ch//Navig/navig.asp?MenuID=806`

1.2 General approach

Actors and system understanding

The multidisciplinary context of the SYNERGIE projects calls for a truly holistic inter- and intra-disciplinary perspective. The understanding between actors is primordial, both in terms of vocabulary (linguistic) and substances (technical requirements and limitations, module coupling, etc...). Beyond and of equal importance is the understanding of the ecology at the hydrosystem scale. Continuous meetings between actors filled the inter-disciplinary gaps and a broad range literature review contributed to a holistic perspective on hydrosystem ecology (Luyet; 2005).

The SYNERGIE scheme concept and general considerations

The multi-purpose ambition of the project increases the structural aspect of the scheme, and in order to gain leverage over the river it appeared that a reservoir was an obligate component of the system. Our multi-purpose scheme would therefore enclose one, constrained by specific contextual socio-technical imperatives. Reservoirs are susceptible of having multiple *economical* benefits. It can be seen as a relatively clean and renewable battery, storing energy and releasing it on demand (peak energy), which can be very profitable. Its storage capacity can also be used to mitigate flood peaks, by preemptive drainage and limit damages. Reservoirs are also susceptible to provide *Social* benefits, their aesthetic value may attract people and water-related leisures may be developed.

Ecological water management has to enhance the state of aquatic systems and their immediate surroundings. Such management requires a profound understanding of ecosystem functioning and how communities are associated to their environment. The ecological value brought by the project has to be somehow quantified in order to be inserted in the general optimizer. At a *local scale*, a reservoir is susceptible of creating positive and negative effects on its surrounding elements. The positive effects should be promoted and the negative effects should be minimized. At the *scale of the downstream river*, the reservoir's effect is translated mainly through the management of hydropeaking. In order to take into account these two different spatial scales, it was chosen not to 'normalize' the quantification of both elements but rather to treat them separately: on one hand at the local scale, the reservoir and its surroundings and on the other hand the downstream river.

At the scheme's local scale: the reservoir and its surroundings

Environmental expertise in reservoir development and restoration is quite significant (ETEC; 1992; Lachat; 1986, 1994, 1999, 2000) and others. When designing the reservoir, general ecological recommendations for an ecologically integrated reservoir will be followed. These would encompass the planning of a continuum link between the upstream and downstream of the river through an artificial river, special bank engineering to mitigate reservoir water level fluctuations and enhance lateral connectivity. A positive environmental outcome is assumed providing the ecological recommendations are followed.

At the downstream river scale: hydrological management

At the downstream river scale, the ecological purpose of the reservoir is the improvement of hydrological conditions. The underlying assumption is that by reducing hydropeaking we are able to improve ecological integrity of the downstream river. *Ecological integrity is used to refer to symptoms of an ecosystem's pending loss of carrying capacity, its ability to perform nature's services [...] due to cumulative causes such as pollution* [4]. It is also defined by Angermeier and Karr (Angermeir and Karr; 1994), and is perceived as the *maintenance of all internal and external processes and attributes interacting with the environment in such a way that the biotic community corresponds to the natural state of the type-specific aquatic habitat, according to the principles of self-regulation, resilience and resistance.* In order to justify hydropeaking mitigation we must quantify the consequent ecological improvement. Meaningful ecological indicators must be found in order to develop a scientifically sound prediction tool of environmental conditions through a data-analytical approach (environmental variables used to predict biological communities).

An ecological consequence to scheme design and operation scenarios

The ecological consequences are assessed at the downstream level by a predictive model. The model will be run on a case study for the Swiss Upper Rhone River where the benefits and limits of hydropeaking mitigation will be detailed. The resulting model is developed under the MatLab environment and is `SYNERGIE`-optimizer ready and operational.

[4] http://en.wikipedia.org/wiki/Ecological_integrity

1.3 Thesis structure

The motivation of the work originated from public awareness on ecological degradation linked to river development projects and a need for a sound ecological response to a project was proposed.

The state of the art (Chapter 2) positioned the *river system* (Figure 1.1, light blue) as governed by the concerted action of *hydrology* and *morphology*, which appeared as the two factors potentially affected by a river development *project*, and hence to take into account. Means to assess river quality were also reviewed and the choice of focusing on macro-invertebrates and fish as our *bio-indicators* of the *ecological response* of the *river system* were made based on existing knowledge and ecological relevance.

Materials and methods (Chapter 3) describes the study site of the potential river development *projects* (Figure 1.1, gray) as well as the general ecological considerations to keep in mind and the assessment of the *ecological response* of the downstream river based on bio-indicators.

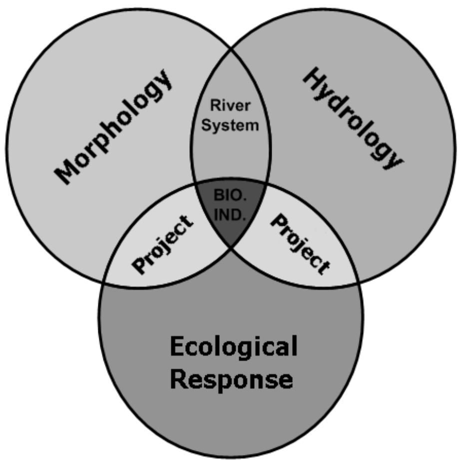

Figure 1.1: Structural synopsis of the work. BIO.IND. stands for bio-indicators (macro-invertebrates and fish)

The effect assessment of *hydrology* in its *morphological context* on the two *bio-indicators* is detailed in Chapter 4 as well as indicators specifics and limitations.

In Chapter 5, four *project* scenario are presented and have their *ecological response* assessed.

In Chapter 5.6 are discussed the project scenario *ecological response* outcome together with their respective ecological implications. Concluding words and perspectives are given in Chapter 6.

1.4 General objectives of the study - a tool for ecology integration in river development

This work was inspired from a multi-purpose river development project, and beside the rather fundamental aspect of model selection for macro-invertebrate richness prediction, there is a very profound concern for the future applicability of this tool. As mentioned in section 1.1, this work is closely bound to one of the project of the Laboratory of Hydraulic Construction directed by Prof. Schleiss at the Swiss Institute of Technology of Lausanne. Public awareness is now such that ecological considerations are granted a status as important as factors such as energy production or flood protection in the decision making processes yielding to project acceptance or refusal. The general objectives concerning this work are:

1. an exhaustive understanding of the ecological principles and processes operating within a river system
2. the state of knowledge of the ecological deficits caused by anthropogenic alterations (e.g. hydrological alterations, bed and banks alterations, etc.) of the system of interest
3. the identification of general ecological considerations applicable at the project scheme scale leading to an improved ecological integrity at the site level
4. the identification and implementation of a set of tools providing a measure of the ecological response to a river project design or management
5. the compatibility of these ecological tools with the other project poles (i.e. energy production, flood protection, landscape integration and leisure development) – these tools have to be in the form of a module that can be considered and operational in an optimizer such as presented by (Heller; 2007)

By fulfilling these general objectives we hope to develop the *ecological* pole of this innovative synergetic approach in multi-purpose river development projects.

Chapter 2

State of The Art

2.1 Hydrosystem habitat, processes and occupants

The purpose of this chapter is to provide an overview of the three major hydrosystem elements (watershed, stream and floodplain) processes that provide aquatic and riparian habitats. This chapter aims at describing these singularly different and yet interconnected systems. Understanding these elements is necessary to evaluate an hydrosystem's integrity and is primordial when attempting to predict ecological consequences following a human intervention on a hydrosystem.

2.1.1 Hydrosystem processes

Climate and *geology* are large scale processes that shape the hydrosystem's physical features (Tockner et al.; 2002; Ward, Malard and Tockner; 2001; Ward et al.; 2002). Directly dependent on the geological features and climate are many finer-scale environmental processes governing the ecological organization of the freshwater habitat throughout the hydrosystem.

Climate

Climate in a narrow sense is usually defined as the average weather, or more rigorously, as the statistical description in terms of the mean and variability of relevant quantities over a period of time ranging from months to thousands or millions of years. The classical period is 30 years, as defined by the World Meteorological Organization (WMO). These quantities are most often surface variables such as temperature, precipitation, and wind. Climate in *a wider sense is the state, including a statistical description, of the climate system*[1]. The climate over a hydrosystem greatly influences the biotic community (hatching, vegetative season length) (Daufresne et al.; 2003), but also the water flow magnitude and timing (Cattanéo et al.; 2002; Poff; 1997) as well as the frequency and magnitude of extreme disturbances (Amoros and Petts; 1993; Cole; 1994). Such disturbances are natural phenomena fully integrated in a hydrosystem that the biotic community must cope with on a daily basis. A region's climate altogether with its disturbances is the major biological filter driving out unsuited species at the profit of more adapted ones. The precipitation pattern over a hydrosystem exerts a direct influence on the surface and subsurface water level variations. The chemistry of the water flowing trough a watershed is also directly affected by the airborne droplets (which may gather pollutants, gases, dust...) that once on the ground charge up with minerals and organic matter from the soil. Key elements (e.g. carbon, nitrogen, phosphorous...) dissolved in the water link the hydrological cycle to other nutrient cycles and contribute to shape the distribution of species. Water percolating further contributes to the hydrosystem's aquifer, which in turn affects greatly ecosystems through influences on patterns of water availability, water flow, water temperature and nutrients availability (Amoros and Petts; 1993).

Geology

The location of the hydrosystem is a tribute to regional geological history. The spatial patterns of surface and groundwater flow are direct consequences of the hydrosystem's geology. The water chemistry of the ecosystems is also strongly influenced by geology. Minerals and organic matter are carried by the surface water and groundwater. Shape, size, and type of the bottom and shore substrates are all function of the hydrosystem's watershed geology. The geology determines where natural barriers may occur in the drainage network and prevent species migration (e.g waterfall). Topography is dictated by geology and climate and is of prime importance in the the shape of the various features of the hydrosystem (e.g. slope, lateral connectivity,...) (Newson et al.; 2002; Richter et al.; 1997; Ward, Tockner, Uehlinger and Malard; 2001).

Hydrosystem vegetation

The hydrosystem vegetation can be seen as the integration of geologic and climatic factors (Grevillor et al.; 1998). Water chemistry is influenced by veg-

[1] http://en.wikipedia.org/wiki/Climate

etation. When decaying, plant matter releases various organic compounds (e.g. humics) that either percolate in the groundwater or are washed out during floods or precipitation. Some of the released compounds are potentially toxic or beneficial depending on the organism, and hence affect community structure. Vegetation can also locally shade a body of water, altering the temperature within an ecosystem. Riparian vegetation provides litter that may serve as food or shelter and adds to the complexity of physical habitats. It plays a crucial role in the behavior of surface water after precipitations by slowing down and holding on runoff. It mitigates the risk of floods following precipitations and provides the water with more time to react chemically with the soil. Soil permeability is also affected by vegetation cover. Finally, vegetation is susceptible to affect the microclimate of hydrosystems by altering its water cycle and modifying the humidity of the surrounding zone (Amoros and Petts; 1993; Cole; 1994).

2.1.2 Freshwater ecosystems

A freshwater ecosystem consists of interrelated freshwater and riparian species and communities linked by a shared environmental regime and a shared physiochemical habitats (Figure 2.1). The spatial extents of these freshwater ecosystems vary and they often have fuzzy boundaries. A hydrosystem can be seen as connected freshwater ecosystems of different scales and natures (Amoros and Petts; 1993). Three major groups are usually isolated: *Flowing water* (lotic systems such as rivers and streams), *Standing water* (lentic systems such as lakes or ponds) and *Transitional systems* (such as wetlands). It is important to stress that although these elements are presented separately, they are naturally in interaction.

Lotic systems - rivers and streams

These ecosystems are easily distinguishable since they are delimited by water flowing along well defined channels. The types of ecosystem as well as the types of organisms living in them are governed by factors such as climate, geology, hydrology, depth, substrate distribution, current speed, distance to spring, temperature and longitudinal, lateral and vertical connectivity.

The ecological integrity of such system is dependent on the nature and relations among key ecological factors. The master variable in a lotic system is *hydrology*, onto which all other ecological factors depend. Terrain, vegetation, watershed climate, plain slope, substrate and others... affect both the large-scale and fine-scale characteristics of the river ecosystem. In addition, lotic systems are dynamic at both the large and fine spatial scale,

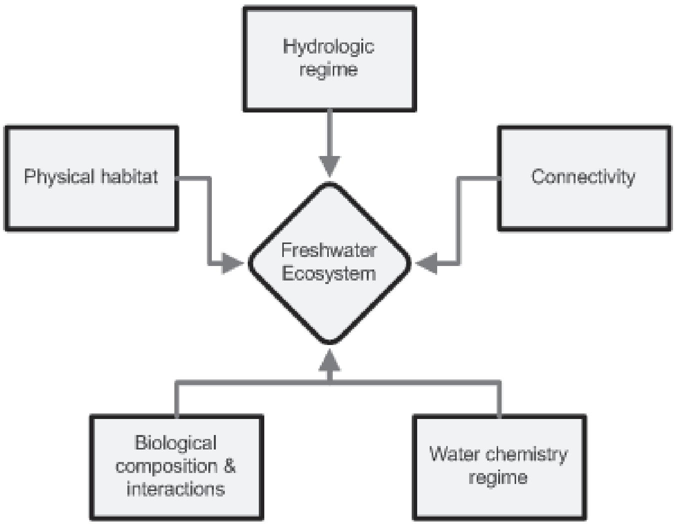

Figure 2.1: Types of major ecological factors for hydrosystem ecosystems, modified from (Silk and Ciruna; 2004).

through the processes of erosion and deposition. A successful conservation of river ecosystems requires consideration of the large and fine scale features of habitat as well as the natural variations over time and the factors that shape them. The *river zone* is historically the first classifier of lotic habitat (Vannote et al.; 1980).

River ecosystems are generally classified in three distinct zones, a *headwater* zone, a *transfer* zone and a lower *mainstream* zone. The headwater zone is usually characterized by a narrow valley and a rather steep slope, which enables the stream to move large debris (e.g. boulders). In this particular zone, erosion is greater than deposition. The headwater zone transits into the transfer zone with a wider stream having a lower gradient. Interactions with the floodplain start to occur at this point, and the river transitions from carrying gravels and finer substrate to depositing the large loads carried from the headwaters. Further downstream is located the mainstream zone, which is characterized by having a deep and wide channel with a low slope, low current velocity and strong interactions with the floodplain. All the river load settles in this zone and processes of *aggradation* or material deposition are frequent (Amoros and Petts; 1993).

These zones and their associated processes transform the river into a flow carrier of energy, materials, nutrients and organisms. Within the various zones of the river system, habitat varies from one *reach* (i.e. length of a river having a specific mixture of geological condition, slope and vegetation) to the other. The variety of habitats amongst reaches in a river adds to the system's overall biodiversity because of the resulting specific ecological conditions occurring. Geologic conditions will affect the behavior of substrate during floods, the nature of the substrate (coarse or fine) and the surface sub-surface water interactions.

River flow contributed by groundwater is termed *baseflow* and is generally a constant and dependable water supply. A key ecological variable such as the water physiochemical pattern is affected by the baseflow. Geology, slope and surrounding vegetation shape the three other ecological key features of habitat in each reach: the *riparian zone* (pool, riffle, bar structures) and the *hyporheic* (sub-river) zone dynamics.

The *riparian zone* can be defined as zone onto which the water overflows when it rises over bank height. Flood waters slow down when overflow occurs, yielding several significant ecological effects such as the creation of new soil and habitat. While flooded, the riparian zone provides slowly flowing water susceptible of providing shelter to river fauna from the turbulent flood flows of the main channel, as well as resource for feeding, spawning and rearing for some species of fish. When the water level falls back, the aquatic biota is able to transit back to the river. Throughout the river, the riparian zone provides bank stability and can be seen as a buffer zone to surface runoff and pollution. The riparian zone also provides habitat for ecotonal species. Reaches in the transfer zone generally have a more developed riparian zone with plant communities varying due to factors such as elevation, soil, flooding frequency, flooding magnitude and interactions with groundwater. Local differences in sediment erosion and deposition further adds complexity and diversity to the floral communities (Glenz; 2005). The riparian zones of reaches located in the lower mainstream zone offer distinct floral communities suited for increased sediment deposition and submersion. These communities are strongly flood dependent doing best in deep alluvial soils and showing high productivity and diversity. Slower current speed allows for rooted or floating macrophytes to develop. *Pools* and *riffles* provide a variety of habitat for aquatic flora and fauna.

Pools are often formed in places where physical obstacles cause the flow in the channel to swing sideways or downward. This causes the erosion of a deeper zone (Chartrand and Whiting; 2000). The most common type of pool is found at river bends, where the flow hits the outer bank and scours out the bank and bed of the channel, and deposits the sediments downstream on the inside bend. Log dams or falls can also create pools.

Riffles are straight and shallow stretches and cause the water flow to agitate up to the water surface, contributing to the water's oxygenation. The substrate of riffles is relatively resistant to erosion and usually coarser than in pools.

Ecologically, pools and riffles differ in more than simply water depth. Pools may be cooler in summer and warmer in winter with relatively little mixing occurring while in riffle, mixing is complete. Riffles are prone to collecting debris (e.g. wood) adding shelter for organisms while pools may collect finer sediment (due to slower current velocity). Pools with the highest ecological quality are deep with slow moving current and overhanging banks, many fish species are very fond of such structures. Young fish sometimes have a better affinity for riffles (Baglinière and Maisse; 1991), where more bite-size food can be found. Filter feeders also tend to prefer riffles since they present a better substrate stability and the improbability of clogging by fines (Logan and Brooker; 1983). Gravel-bottomed riffles are the site of predilection for the spawning of many fish since they guarantee an abundant supply of dissolved oxygen (DO) necessary for the egg development and have a decreased risk of smothering due to the lower quantity of fines. However, riffles may be prone to dessication during months of low flow, while pools may serve as shelters.

Bars are bands of sediments usually deposited following high flow pulses, especially downstream of bends. They are formed by the dramatic current velocity decrease leading to sediment deposition. Bars provide key habitat for a wide range of insect, amphibians and birds (Amoros and Petts; 1993).

The *hyporheic zone* comprises the water-filled interstices under the river, and is generally of similar width than the channel. Occasionally, the hyporheic zone can extend as far as a couple of kilometers beyond the banks. Hyporheic water is often strongly different from surface water (Boulton et al.; 1998). It usually flows at a speed in the order of a couple of $cm{\cdot}h^{-1}$ and usually contains less DO and more carbon dioxide. Hyporheic water temperature is less subject to variations than surface water and usually contains more dissolved minerals. Hyporheic water differs from true groundwater since it usually flows faster, contains more DO, less carbon dioxide and has a higher variation in its thermal pattern. The hyporheic zone may shelter benthic organisms as well as a very specialized hyporheic-obligate community of microorganisms (Brunke and Gonser; 1997; Brunke et al.; 2003; Malard et al.; 2003, 2002). This zone provides a dry season shelter as well as a flood refuge. Since the width, chemistry and temperature regime vary from one hyporheic reach to the other, so does the community composition. Once again, this variation integrates the combinatory effects of the watershed and channel geology and hydrology.

The *hydrological regime* results from both the baseflow and the watershed runoff and is usually characterized by five key ecological attributes (Amoros and Petts; 1993; Junk et al.; 1989; Malmqvist and Englund; 1996; Ward et al.;

2002).

1. the *magnitude* or amount of water going through a fixed point at a time (discharge)

2. the *frequency* or how often a particular condition occurs (e.g. low flow)

3. the *duration* or the length of time that a specific flow condition lasts in time

4. the *timing* at which a particular event occur

5. the *rate of change* or how quickly the flow changes

The *natural flow paradigm* (Poff; 1997) originates from the understanding that freshwater and riparian organisms can bear a range of flow conditions specific to each species (ecological valence). Certain fish will move into the floodplain areas during flood event to spawn, feed or escape predation or unfavorable hydraulic conditions. If this event occurs at the right time of the year and last for the right amount of time, some species will benefit from this event. Similarly, other species may be adversely affected by the same event. The rate at which water level raises or falls is also of considerable importance. Some organisms may be stranded during abrupt level fall (Halleraker et al.; 2003) and others may be flushed away during spates (Céréghino et al.; 2004; Parasiewicz et al.; 1998). Hydrological regime affects other ecosystem conditions as well. Water chemistry, nutrient cycling, oxygen availability, morphogenic processes and thermal regime are tightly coupled to the hydrological regime (Amoros and Petts; 1993; Ward et al.; 2002).

Water chemistry is subject to great changes from reach to reach (Cole; 1994) depending on its location, connectivity and underlying geologic and climatic conditions. The pH determines water toxicity for organisms and also the biodisponibility of various chemicals in the water. A good example is phosphate, which plays a major role in the nutrients dynamics of plants. It tends to dissolve very poorly in waters with high pH and not be available for plants. Nitrogen and phosphorous play a key role in aquatic ecosystem dynamics (Scheffer et al.; 1993), since altogether with carbon, oxygen and hydrogen are the building blocks of life.

Several other dissolved materials play ecologically important roles in shaping the chemistry of lotic system (i.e. silica can be a limiting factor for the development of diatoms). Bicarbonates influence the ability to maintain nutrients in solution as well as the biodisponibility of pollutants; iron and sulfur are also of prime importance for primary producers; and salinity or total dissolved solids equally influence the species communities of a particular stream.

Turbidity affects light penetration and hence primary production, but also predation, and hence plays an important role in structuring the freshwater community.

As previously mentioned, the aquatic communities are often segregated by *water temperature* (Castella et al.; 2001; Tachet et al.; 2000). Cold water communities are typically encountered in the headwaters while warm water communities are usually in the lower mainstream. Temperature affects the metabolism (i.e. the overall reach productivity, egg and larval development, organisms growth…) and temperature changes are often interpreted as a life cycle cues.

There is a 4D connection of the river and its floodplain that is essential to understand in order to approach hydrosystem ecological integrity. There is a *longitudinal*, *lateral*, *vertical* and *temporal* connectivity pattern (Amoros and Petts; 1993; Junk et al.; 1989; Petts; 1984; Vannote et al.; 1980; Ward et al.; 2002).

Longitudinal connectivity is primordial in the movement of energy, material and biota through the entire system.

Lateral connectivity is normally endured by riparian ecosystems, which serve as transitional zones between the aquatic and terrestrial worlds. Riparian ecosystems provide key habitats, structural and trophic material inputs to aquatic lotic ecosystems. Many terrestrial species are also relying on riparian areas as reservoirs of food, water, shelter and migratory networks.

Vertical connectivity is of primordial importance for the exchange of water, dissolved material (e.g. nutrients) and organisms between the surface and the underground.

Temporal connectivity is of prime importance and ensures that the right events happen at the right time for the right duration.

Abiotic factors (hydrology, chemistry, temperature…) shape the biotic composition and interactions in lotic ecosystems and within the different reaches along the stream (Amoros and Petts; 1993; Ward et al.; 2002). A lotic ecosystem is supported by autochtonous and allochtonous production linked by secondary consumers. As one moves downstream a river system, a greater amount of nutrients is autochtonous rather then allochtonous (*The River Continuum Concept* (Vannote et al.; 1980)) which depicts the changes in communities from a river's headwaters to its mouth in relation to habitat characteristics and trophic dynamics.

Lentic systems - lakes and ponds

These ecosystems are distinguishable since they are delimited by inland depressions. The master variable of lentic systems is probably the shape of the basin which has an effect on all the key ecological variables (Cole; 1994). Changes of a lentic system typically occurs on a longer time scale when compared to changes occurring on lotic systems. The most commonly used parameters to describe a lentic water body are surface area, volume, shoreline, maximum depth and maximum length. These characteristics determine the probability of thermal stratification as well as the resultant biological communities (Amoros and Petts; 1993; Cole; 1994; Labadz et al.; 2002; Ward et al.; 2002). Lakes are 3D structures that are sometimes compared to forest ecosystems (Silk and Ciruna; 2004). They exhibit changes in light and temperature from surface to bottom, which structures communities. A zonation between the *littoral*[2] zone, the *pelagic*[3] zone and the *benthic*[4] zone is usually referred to.

The shape of the lentic system, its lateral and vertical connectivity as well as its stratification and wind exposure are factors affecting the hydrologic regime, which in turns governs trophic resources and biotic distribution. A lentic system is subject to *short term* variations, where temporary events (i.e. levels, temperature,. . .), which usually have negligible or little effect on biotic communities. *Seasonal* variations are regular and natural phenomenon for lentic systems and are caused by climate and groundwater flow. *Long term* fluctuations can be due to climate change, may last several years and have profound implications for the biotic communities.

Water chemistry plays a primordial role in shaping the biotic composition and the dynamics of a lentic ecosystem, perhaps to a far greater extent than it does in shaping lotic ecosystems. This is mainly due to the greater physical inertia of still water bodies compared to running water (Cole; 1994). The three major parameters shaping the chemistry of a lentic systems were already defined in section 2.1.2 and are temperature, turbidity, dissolved gases, nutrients composition and concentration.

Productivity and biodiversity shape the biological composition and interactions of a lentic system. Productivity being determined mainly by nitrogen and phosphorous concentrations. The community of a lentic system has learned to cope with the system's productivity regime such that each of the three main lentic zones (i.e. littoral, pelagic, benthic) exhibits a unique set of habitat characteristics each accommodating a particular community.

The littoral naturally provides habitat for the greatest number of lentic-obligates species. Aquatic vegetation provides food, substrate for bacteria, algae and invertebrates as well as shelter for fish (juveniles in particular).

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[2]transition zone between the aquatic and terrestrial media

[3]open water area

[4]lake bottom

The littoral is also usually the warmest, which further stimulates productivity, attracts birds and other terrestrial species (OFEFP; 2004; Traut and Hostetler; 2004).

The pelagic zone can be compared to the 'lung' of the system since phytoplankton activity might produce most of the DO of the system.

The benthic zone can be abundant with crustaceans, mollusks, worms and insect larvae (Cole; 1994).

Transitional systems - wetlands

Transitional ecosystems are characterized by their ability to sustain both aquatic and terrestrial species (Amoros and Petts; 1993; Cole; 1994; Labadz et al.; 2002; Ward et al.; 2002). These transitional ecosystems are zones were water covers the soil or is present either at or near the soil surface for varying durations. These systems are among the most productive and the most threatened ecosystems (Amoros and Petts; 1993). The prolonged presence of water favors the development of specially adapted species. These system's ecological integrity depends on hydrological regime, physical habitat conditions, water chemistry, connectivity and biological interactions operating within.

The *hydrological regime* sets the formation, size, persistence and vegetative composition of the transitional system.

Soil structure and composition are probably the most important physical variables. A good example are hydric soils, which set the frame for a specific microbial fauna yielding specific biogeochemical reactions.

Nutrients are also determinant factors affecting the properties of transitional zones, together with conductivity and pH. DO concentration as well as other chemical variables such as calcium are also important in many physical processes taking place in wetlands.

Transitional zones are connected to the rest of the hydrosystem by both surface water and underground water. Such connectivity must remain intact in order for the wetland to keep its ecological integrity (Grevillor et al.; 1998). Wetland flora has undergone distinct physiological, morphological and reproductive adaptations that enables them to thrive in hydric soils. Some amphibians absorb oxygen through their skin. Many species of crustaceans, insects and mollusks are very fond of transitional zones. Many fish species require wetlands as nursery grounds. Numerous waterfowl birds need transitional zones to nest and rear their young (Wolff; 1994). As conditions vary from a temporary saturation to permanent submersion, the communities will also vary from transitional groupings to species or physiologies adapted to long submersion periods (Westlake; 1975).

These zones have been described as the hydrosystem's *kidneys* since they filter out pollutants, sediments and nutrients from the water. Wetlands are able to mitigate floods during the wet season and guarantee water release in times of drought (Amoros and Petts; 1993; Cole; 1994; Ward et al.; 2002).

2.1.3 Hydrosystem biodiversity

Species distribution in hydrosystems is governed by a combination of factors differing from the ones encountered in a strictly terrestrial or marine system (Amoros and Petts; 1993; Ward et al.; 2002). Catastrophic events (i.e. climatic or hydrologic extremes) cannot be avoided by the freshwater species since migration outside the basin cannot be done readily. The types of ecosystems within a hydrosystem can be very localized and harbor a quite unique community. In most cases, the ecological valence of a species will be restricted by specific requirements (e.g. current speed, temperature, turbidity...) and may vary throughout the species' life cycle. A good example is our river trout *Salmo trutta fario*, which requires very specific condition for its reds in terms of substrate, water quality, current speed... and varying conditions throughout its life cycle (Baglinière and Maisse; 1991; Mills; 1971; Schmetterling; 2000). Crustaceans such as *Asellus aquaticus* can undergo a dessication-resistant life stage, enabling them to occupy temporary water bodies (Tachet et al.; 2000). Most organisms have undergone morphological and behavioral adaptations to maximize the colonization of the habitat offered.
Freshwater fish species have developed kidneys enabling them to excrete the excess water while conserving the minerals necessary for their physiological processes (Cole; 1994). Some aquatic insect larvae will wave their gills in order to create a current of water to extract more oxygen. Hydrodynamic body shapes, means to attach to the substrate or use the water current for egg dispersal are all examples of the evolutions undergone by aquatic species to occupy as many niches as offered by the system (Tachet et al.; 2000).
Submerged vascular plants have also undergone morphological adaptations by developing a root system to hold in place (Westlake; 1975). Organisms living in turbid waters have adapted ways to navigate, predate and avoid predation by sensing chemical or electromagnetic cues (Baglinière and Maisse; 1991).

Usually, nineteen taxonomic phylum are encountered in freshwater ecosystems:

- **Viruses:** are found to be the potential pathogens in many aquatic organisms and humans (e.g. hepatitis). This phylum is not so well documented.

- **Bacteria:** are very adaptable and can be very abundant in the soil, the sea and freshwater systems. Hydrosystem studies were late in becoming concerned with them, with the exception of the blue-green 'algae' and their massive development in eutrophic waters (blooms). In such case, their respiratory demand was greater than their daylight oxygen production, and upon their depth their decay contributed further to oxygen depletion of the water. Some species are known to produce some toxins (Fay and VanBaleen; 1987). Now bacteria are known as a key trophic resource. Most studies focused on their role as mediators in element cycles (i.e. hydrogen, sulfur, carbon, iron, manganese, nitrogen and phosphorus). Bacterias serve as decomposers and recyclers in trophic chains, as agents in biogeochemical cycles and as essential links in detritus communities (Cole; 1994). Some are photosynthetic (i.e. cyanobacteria), and some may be pathogens in aquatic organisms and/or humans. This phylum is well documented but the subject is very broad and diverse. See (Kuznetov; 1976) for classical reference on freshwater bacteria.

- **Fungi:** together with bacteria are the main recyclers of organic substances and decay of dead material. Fungal activity increases the palatability[5] of the substrate to detritus feeders (Nikolcheva et al.; 2003). This phylum is found on most dead material and have the particularity of being able to break down cellulose plant cell walls and chitinous insect exoskeleton. Some fungus are pathogens in humans and aquatic organisms.

- **Algae:** this phylum is represented by a micro- and macroscopic variety (over 8000 known freshwater species) of unicellular and colonial photosynthetic organisms all lacking leaves and vascular tissue. Algae can be the major primary producers in most aquatic ecosystems. Algae are well documented and beside the occurrence of physiologic races as well as taxonomic problems, they were used as indicator organisms as early as 1965 (Brook; 1965).

- **Plants:** photosynthetic organisms, mostly higher plants possessing leaves and vascular tissue. Plants are of primary importance in aquatic systems since they provide spatial, structural and trophic resources for

[5] ability to be eaten

many aquatic organisms. Plants may also act as local thermal buffer sometimes crucial to aquatic organisms. Terrestrial or ecotonal plants are crucial for bank stabilization, flood peak mitigation and surface water - ground water interactions. This phylum is very well documented (Barendregt and Bio; 2003; Westlake; 1975).

- **Protozoans:** are microscopic mobile single-celled organisms found in virtually all freshwater habitats. Most are filter-feeders and affectionate water rich in organic matter, bacteria or algae. The activity of the protozoa – especially ciliates colonies – in the extraction and digestion of bacteria and other suspended particles is the main element of the natural process by which the water supply is cleaned and rendered usable to other organisms. Many protozoans are parasitic on algae, invertebrates or vertebrates.

- **Rotifers:** are near-microscopic organisms widely distributed in freshwater habitats (over 1800 known species). Rotifers are either planktonic or benthic, and graze on plant cells, bacteria and detritus (secondary consumers). Rotifer taxonomy is complicated by many factors including seasonal variation in form and complicated life-cycle. This phylum is very important in the zooplanktonic community of lentic systems and may dominate lotic zooplanktonic communities. Rotifers are documented but their ecology is still obscure (Gilbert; 1966).

- **Myxozoans:** are microscopic organisms with sometimes very complex life cycles and are obligate endoparasites in or on fish. Large documentation on Myxozoans as fish parasite exists (Kent et al.; 2001), but their ecology is still obscure due to their very complicated life cycles.

- **Flatworms:** may be found under free-living or parasitic form. Some use mollusks as intermediate host and are parasites of various vertebrates including humans.

- **Nematodes:** generally microscopic or near microscopic roundworms. This phylum can be parasitic, herbivorous or predatory and are typically found in bottom sediments. Freshwater nematodes are poorly documented.

- **Annelids:** this phylum is comprised mainly of oligochates and leeches. Oligochates are bottom living worms grazing on sediments and are found everywhere in the sediments of ponds, lakes and rivers. A high abundance and low richness (usually restricted to Tubicidae family) is a good indicator of a eutrophic system where they have on occasion been reported to make up over 95% of the benthic community (Brinkhurst; 1966). Leeches are mainly parasitic on vertebrate animals or in some case predatory.

- **Mollusks:** this phylum is comprised of Bivalves (e.g. mussels) and Gastropods (e.g. snails). Mollusk communities can be very rich and tend to form local endemic community assemblages (over 40 endemic prosobranch in Lake Baikal (Cole; 1994)). Gastropods tend to be grazers or predators while bivalves are filter-feeders. Bivalves may contribute to good water quality but are very prone to pollution. Their larva may be parasitic to some fish. Gastropods have received more attention by parasitologists since they serve as intermediate hosts for various flukes often using humans or their animals as definitive host. European mollusks are very well documented and identification keys exist (Tachet et al.; 2000). When mollusks are present in a stream, have a strong potential for a good stream quality indicator with relatively little experience needed.

- **Crustaceans:** this group is mostly a marine group, but those found in freshwater are very important members of the community. They are characterized by a jointed exoskeleton often hardened with calcium carbonate and include larger bottom living species (e.g. crayfish). Most species are either filter-feeders or predators and in some case are parasitic for fish.

- **Insects:** are by far the largest class of organisms known, they comprise over 75% of all the described animal species, but 'only' 3% of species are aquatic (which still represents 25000 to 30000 species, with solely a few hundreds being marine or intertidal (Cheng; 1976). Insects are characterized by a jointed exoskeleton and three pairs of legs. In rivers can be predators, grazers or filter feeders. Insects dominate the intermediate level of the food webs and in some case may be vectors of human diseases (Table 2.1.3)). Aquatic insects are well documented and are commonly used as indicators in river quality assessments. Very good identification keys exist for European aquatic insects (Tachet et al.; 2000).

- **Fish:** are the dominant organisms in terms of biomass and are at the top of the aquatic food web. Fish are well documented, easy to identify and are used as indicators of river quality.

- **Amphibians:** larvae of most species need water for their development. Some species are entirely aquatic. Larvae are grazers and adults

Table 2.1: Order, Common name and Active Aquatic Stages of aquatic and semi-aquatic insects, modified from (Cole; 1994)

Insect Order	**Common Name**	**Aquatic Stages**
Collembola	Springtails	Immature, adult
Ephemeroptera	Mayflies	Naiad
Odonata	Dragonflies, damsflies	Naiad
Orthoptera	Grasshoppers, crickets, katydids	Semiaquatic, none
Plecoptera	Stoneflies	Naiad
Hemiptera	True bugs	Nymph, adult
Megaloptera	Dobsonflies, alderflies	Larva
Neuroptera	Spongeflies	Larva
Trichoptera	Caddisflies	Larva, pupa
Lepidoptera	Moths	Larva
Coleoptera	Beetles	Larva, adult
Hymenoptera	Bees, wasps	Larva (parasitic)
Diptera	True flies	Larva

predators. Very well documented phyla.

- **Reptiles:** most are predators. Some are invasive (e.g. Florida turtle *Trachemys Scripta elegans*). Very well documented phyla in Europe.

- **Birds:** can be closely associated to wetlands and water margins. A few are restricted to rivers and lake systems. Wetlands are often key feeding, staging and resting areas for migratory species. Likely to play a role in the passive dispersal of small aquatic organisms. Very well documented phyla.

- **Mammals:** only few groups are strictly aquatic, several species may be largely aquatic but emerge on water margins. Mammals are usually top predators and grazers, and larger species suffer considerably by habitat alteration and hunting. Very well documented phyla.

To summarize this section (Figure 2.2), the interactions of geology and climate lead to the context of the three elements (lotic, lentic and transitional) susceptible of being encountered in a hydrosystem. The hydrosystem context sets up the natural changes backbone of the major ecological drivers of aquatic ecosystems, namely the hydrological regime, physical habitat configurations, aquatic physical and chemical attributes, the longitudinal, vertical, lateral and temporal connectivity.

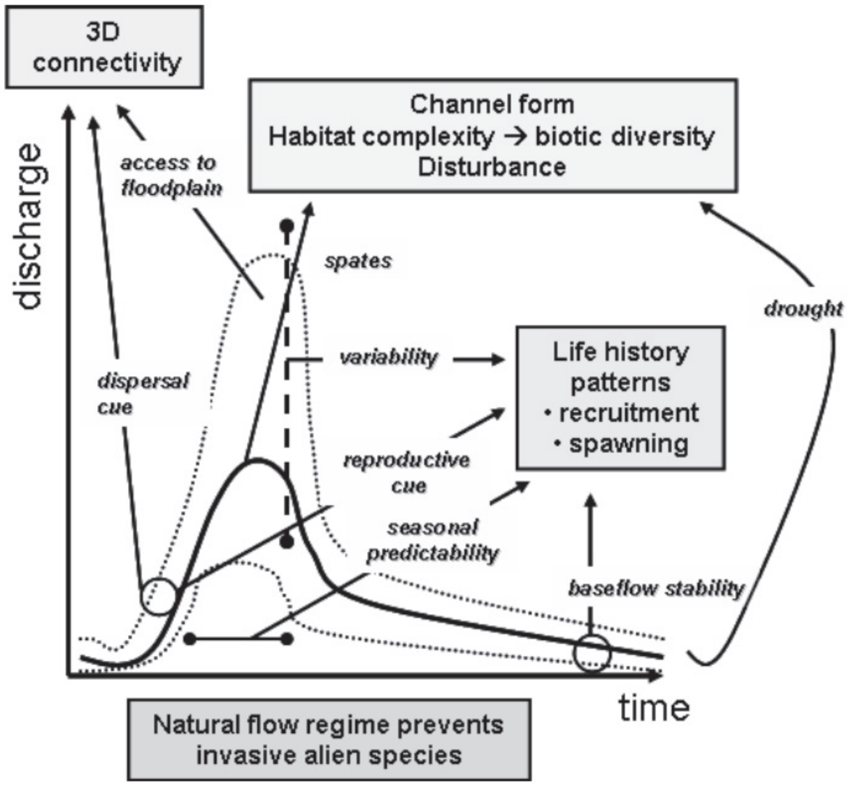

Figure 2.2: Summary figure of the influences of the flow regime on the aquatic biodiversity of a hydrosystem - modified from (Bunn and Arthington; 2002). Geologic and climatic events are the major drivers of the relationships between diversity and aquatic habitat under natural conditions. Life history patterns are influenced by seasonal predictability, reproductive cues as well as spates variability and time

2.2 Anthropogenic alterations of the hydrosystem

2.2.1 Water management

Water demand for various purposes has favored many structural modifications of the hydrosystem (i.e. channelization, dams, diversions, flood control schemes, groundwater extraction…). A tremendous cost to the underlying ecosystems results from these modifications. Existing systems have been fragmented (Dynesius and Nilsson; 1994), ecosystem dynamics altered and unconnected systems have been joined. As a result, fish catch has declined (Jungwirth et al.; 1993; Kerle et al.; 2002), freshwater biodiversity has been lost (Passauer et al.; 2002), floods frequency and severity has increased (Revenga et al.; 1998), floodplain soil nutrients has lowered (Revenga et al.; 1998). In this section will be presented the major threats coming from water use and management on the hydrologic regime, the water chemistry regime, the physical habitat conditions, the 4D connectivity and the biological composition structure and function of the hydrosystems' elements.

Dams may severely disrupt the river's *hydrology* by controlling the flow (Petts; 1984; Poff et al.; 1997). Movements of water, nutrients, sediments and biota is altered longitudinally, laterally, vertically and temporally. Large or infrequent floods, such as 10- or 100-year floods may no longer occur (Bednarek; 2001). This modification has been reported to endanger the ecological integrity of the whole hydrosystem, since regulated flows equally affect downstream lakes and wetlands dependent on a natural hydrology. A major effect of dams has been reported on the four components of *connectivity* (*sensus* Ward (Ward and Stanford; 1995)). By altering the intensity, the extent and the timing of longitudinal connectivity dams will also alter the downstream river's associated lateral connectivity. The passage of sediments, nutriments, debris as well as the upstream/downstream movement of aquatic and riparian species present in 'natural' streams (Lignon et al.; 1995) will be modified. The physical obstruction of both dams and reservoirs impedes and delays the migration of various organisms (Bednarek; 2001).

Lateral connectivity includes the channel dynamics as well as the interactions between the river and its floodplain (Ward and Stanford; 1995) and is equally disturbed by dams. Dams tend to shorten the duration and magnitude of high flows, preventing the river to flow beyond 'bankfull' levels and hydraulically connect with its floodplain (Junk et al.; 1989; Naiman et al.; 1993). As a result, the river fauna can be disconnected from critical spawning, refuge and foraging habitats. Riverine vegetation abundance and diversity has also been reported to react to lateral connectivity alteration (Bornette et al.; 1998). There is often a loss of the naturally occurring periodic flooding, which shows to be particularly harmful for channel rejuvenation and species dependent on allochtonous resources. A reservoir filling tends to simplify upstream river complexity and favor lentic environment (and hence community) at the expense of a lotic one. Spatial and temporal patterns of both charging and discharging of surface water and groundwater are generally dependent on surface flows and it has been observed that reduced surface flows yield a lower groundwater table (Sophocleous; 2002). Groundwater table level is very closely associated with the available instream habitat during dry periods (Brunke and Gonser; 1997). Hydrosystem vegetation is also directly dependent on groundwater levels (Nilsson et al.; 1991, 1997).

Physical habitat is also potentially altered by dams. As mentioned, reservoir tend to modify a lotic habitat into a lentic habitat where lotic organisms will be jeopardized at the benefit of lentic organisms. Stream adapted organisms are replaced by lake adapted (and sometimes exotic invasive) species, potentially leading to cascading trophic effects throughout the hydrosystem (Parker et al.; 1999). Downstream habitat can also be altered due to sediment transit reduction and resulting channel morphology modification. The instream substrate, structures (i.e. pools, riffles, runs) change as the river adjusts to new conditions.

When reaching the reservoir, the settling of sediment particles is encouraged by lower current velocities, altering the *sediment regime* of the downstream river. Many toxic organic contaminants are associated to the sediment (Olsen et al.; 1982). When scouring event move these contaminated sediments in large concentration, they pose a threat to communities. When the sediment transport is interrupted, the downstream channel degrades, the backwater decrease and habitat changes in morphology.

In deep reservoirs stratification can take place in summer, yielding colder and heavier water occupying the bottom layer (*hypolimnion*) and warmer water occupying the top layer (*epilimnion*). When this process of stratification occurs, gas transfers are inhibited between the warmer and oxygen rich epilimnion and colder oxygen-poor hypolimnion. Water release may originate from either of these layers, producing different effects. Water originating from the epilimnion can raise the temperature of the downstream water, altering communities life cycles (e.g. reproductive timing) and growth rates (Frutiger; 2004b). Water release coming from the hypolimnetic layer can lower the temperature of the downstream river, reducing productivity and shifting the community composition toward a colder water one (Petts; 1984).

Reservoirs can also be a significant source of greenhouse gases, especially when vegetation has not been removed prior to flooding. When rotting, vegetation emits carbon dioxide and methane. Greenhouse gases emission from the *Balbina* reservoir in Brazil has a 26 fold greater impact on global warming than the emission from a coal-fired power plant producing the same amount of electricity (McCully; 2001). Lower amounts of greenhouse gases have been reported for deeper and colder reservoirs such as the high altitude dams in Europe (FISRWG; 1998).

Dams bring averaged conditions at the expense of strongly varying conditions, modifying water quality as well as carriage shaping the bed and banks of the river. The biotic integrity, structure and function respond to these changes and are altered. Dams act as a physical barrier disrupting the movement of species, which leads to changes in species composition and a decrease in species' richness. The trends of biotic impoverishment caused by dams include (Silk and Ciruna; 2004):

- disappearance or imperilment of migratory fish (e.g. brown trout - *S. trutta fario*)
- population fragmentation and isolation
- extinction or imperilment of geographically restricted taxa dependent on very specific riverine habitat
- reduction in abundance of flood dependent taxa
- increase in lentic and non-native species

By eliminating or altering flood disturbance there is a modification of the species composition of the streamside vegetation to that of forest types characteristic of unflooded areas (Décamps and Naiman; 1990). When floodplain become deprived of their natural supply of silt, nutrients and moisture they are subject to significant morphological changes resulting in cascading impacts to native species at the benefit of invasive species. The effects of dams are summarized in Figure 2.3.

Surface water diversion (SWD) is the term used in cases where water is redirected from a river, a lake or a wetland for any purpose (i.e. irrigation, flood control, navigation,...). The major effect of SWD on freshwater ecosystems (Figure 2.4) is the reduction of water flow in the main channel. Decreased water levels in the headwaters poses a direct threat to aquatic, riparian and wetland communities. Changes in food supply, in habitat availability, in the nutrient cycles and in water quality were observed (Silk and Ciruna; 2004). Most of the primary ecological impacts are similar to those caused by dams, however SWD has some distinct effects, principally when water is transfered from one geographically distinct watershed to another

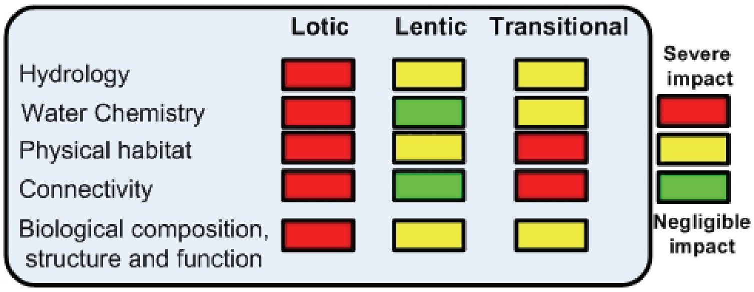

Figure 2.3: Summary of Dam effects on the three elements of the hydrosystem (modified from (Silk and Ciruna; 2004))

(Snaddon et al.; 2000). The risks linked to this kind of water transfer include loss of biogeographic integrity, loss of endemic biota, hydrology alteration, spread of invasive species, spread of diseases and disruption of ecological processes (Arthington and Welcomme; 1995). The hydrologic impact is also comparable to damming. The amount and timing of water in a river is regulated and the resulting impacts on the ecosystem are similar with occasionally an emphasis on the water quality degradation and increased erosion due to loss of root mass (Amoros and Petts; 1993). Longitudinal and lateral *connectivity* is also reduced due to lower and more stable flows. The impact of loss of connectivity has also already been discussed. *Physical habitat conditions* are affected, mainly because of sediment budget alteration. The diversion flows will tend to increase sediment, which can lead to burial of fish eggs and food sources, while dewatered reaches usually erode and move toward a new equilibrium. *The water chemistry regime* is usually altered since reduced flows resulting from SWD increases pollutant concentrations. Besides, pH lowering has been observed altering metal solubility, organic compounds biodisponibility and rates of chemical reactions (Cole; 1994). As a result of decreased flow, water temperature may increase in the summer and decrease in the winter. Both of these changes having consequences for the associated communities, such as reduced DO concentrations in the summer and icing in the winter. Salinity has also been opbserved to increase in cases of decreased flows, stressing the riparian vegetation and aquatic biota.

Changes in the *biological composition and interactions* happen as a result of channel morphology alterations. The establishment of exotic species contribute to the decline of native species (e.g. limit recruitment). Changes in current velocity may interfere with the behavior of flow sensitive taxa such as mayflies (Ephemeroptera) and caddisflies (Trichoptera) (Bernard; 2001; Céréghino et al.; 2004). Suspended sediment load increase can reduce photosynthesis and zooplankton production reducing overall trophic resources.

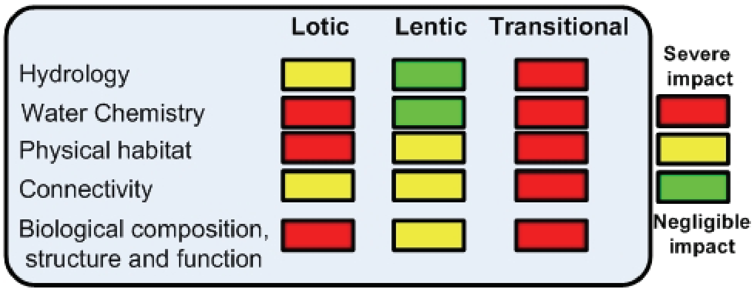

Figure 2.4: Summary of Surface Water Diversions effects on the three elements of the hydrosystem (modified from (Silk and Ciruna; 2004))

The usual modifications leading to an **altered bed and bank structures** include (Dynesius and Nilsson; 1994; Poff; 1997):

- *channelization* – material is physically removed, river shape is altered. Usually done to increase navigability and decrease water residence time
- *armoring* – material such as stones or concrete blocks are added to prevent banks from eroding or collapsing
- *levees* – flood prevention. Have the particularity of sharpening the boundaries between the river and its associated floodplain
- any other structure at or near the shoreline altering flow patterns as well as natural interactions between water and land

There is a great variability in the effects (Figure 2.5) of altered bed and bank structures on the hydrosystem elements. These effects vary spatially and temporally due to climate, soil, channel morphology, hydrology, watershed characteristics, vegetation and land use. Unfortunately, bed and bank structure alterations alter hydrology and water quality, resulting in isolation, fragmentation and ultimately loss of habitat. Erosion is often increased, element structure is strongly simplified, species diversity is lowered and an increase in downstream flooding has been reported (Amoros and Petts; 1993; Bloesch et al.; 1998; Tockner et al.; 2002). Channelization usually reduces overall channel length and width, increases the overall channel gradient and hence alters the *hydrology*. The energy of the flowing water increases and bed scour, bank collapse and erosion are enhanced. While channelization generally reduces natural flood frequency and intensity in the headwaters, it can significantly increase downstream flood (especially when tributaries flooding is in synchrony with the mainstream) (Amoros and Petts; 1993).

Bed and bank alteration generally tends to isolate surface water to it's associated groundwater table, and lowers its level. Water table lowering impacts have been discussed and have a far broader influence than on the channel itself, since they affect the lentic and transitional elements as well.

The *physical habitat conditions* are altered by bedload alteration. Headcutting, lateral bank collapses and overland transport of sediment often lead to channel aggradation and the formation of plugs. Channel blocking may affect seasonal flooding by increase of depth, area and duration of flood. Altered seasonal flooding modify sediment deposition patterns and is susceptible to impact the vegetation composition of the submerged floodplains (i.e. increased submersion time, root hypoxia (Glenz, Iorgulescu, Kuonen, Kienast and R.; 2004; Glenz, Schlaepfer, Iorgulescu and Kienast; 2004)). Besides, channelization converts naturally heterogeneous reaches into a simplified, uniform system reducing habitat diversity (Bundi et al.; 2000; Englund and Malmqvist; 1996; Maddock; 2001; Tockner et al.; 2002). The riffle – pool configuration is altered with channelization, structures such as boulders are removed in order to increase navigability and flow, eliminating important colonization sites. Once the stream is isolated from its floodplain, lateral-connectivity dependent species (for trophic or reproductive needs) become endangered. The altered extent and timing of natural floodplain submersion severely impacts community composition as well as most of the ecological processes. Floodplains, backwaters and transitional zones are important and complex habitats for many resident and migratory aquatic, bird and terrestrial taxa by providing a breeding, resting or feeding grounds (Amoros and Petts; 1993). Armoring has a very negative effect on shoreline and the photic zone of lentic and lotic systems. These zones are naturally highly productive since the shallow water, abundant light and nutrient-rich sediments are ideal for macrophyte growth. Armoring strongly reduces shoreline diversity and habitat, decreasing habitat for insects, fish, amphibians, birds and mammals. Armoring also reduces the filtering capacity of a freshwater system since many of the filtering processes take place in the shoreline (Cole; 1994; Tockner et al.; 2002).

Channelization also alters the *physical and chemical regime* of surface water. Higher instream macrophyte and algae growth has been reported after loss of bank vegetation due to channelization (Silk and Ciruna; 2004). Turbidity, salinity, nutrients concentration, DO, oxygen demand as well as contaminants concentration and pathogens have also been reported to be affected by channelization (Silk and Ciruna; 2004).

The *biological composition and interactions* are also impacted by bed and bank alteration. Organisms linked to bars, backwaters, pools and riffle disappear together with these structures once the bed and banks are sta-

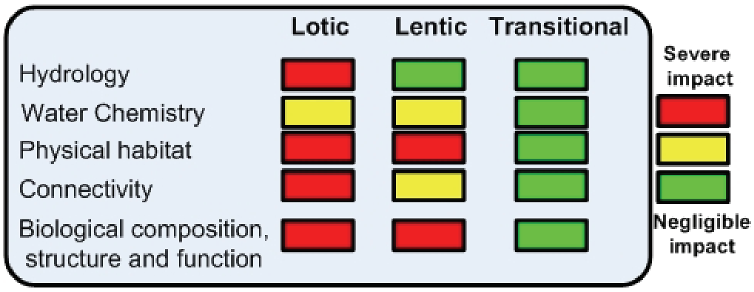

Figure 2.5: Summary of Altered Bed and Bank effects on the three elements of the hydrosystem (modified from (Silk and Ciruna; 2004)).

bilized. Substrate modification usually impedes fish (e.g. *S. trutta fario* to spawn (Baglinière and Maisse; 1991). Mollusks and aquatic insects population and diversity following channelization at the profit of species less sensitive to disturbances such as chironomids (Diptera) which rapidly dominate the invertebrate community. Drifters (i.e. Hydropsychids and Heptageniids), which have a high trophic value for many fish, are typically found in riffle environments and their abundance strongly declines in channelized rivers. Oligochaetes and chironomids are more likely found in channelized river since they thrive in softer and finer substrate (Tachet et al.; 2000).

Hydropeaking corresponds to raising or falling discharges caused either by the turning on or off of hydro-turbines (Gore; 1985). Hydropeaking is usually characterized by:

- its *amplitude* – either by $Q_H - Q$ in $(m^3 \cdot s^{-1})$, Q_H being the discharge during hydropeaking and Q the 'normal' discharge. Similarly, the amplitude can also be described in *(m)* by $H_H - H$ at a given point in a reach or as an average height for a given reach

- a ratio of $\frac{Q_H}{Q}$

- a positive or negative *discharge gradient* $(m^3 \cdot s^{-1} \cdot h^{-1})$ or similarly a height gradient in $(m \cdot s^{-1} \cdot h^{-1})$

- it's *daily frequency* $(event \cdot day^{-1})$

An extensive literature review was performed on hydropeaking , since it is together with morphological alteration gradually becoming the major impacting factor over water quality, which has considerably improved in the past decade (Lek et al.; 2005). The impacts of hydropeaking are probably

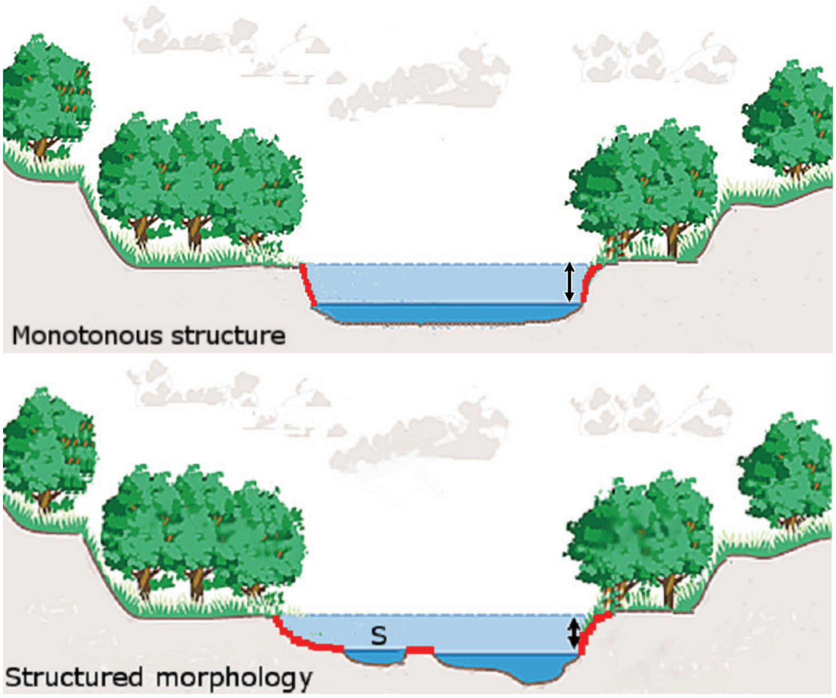

Figure 2.6: Effect of structural complexity on hydropeaking impact on ecosystem – hydropeaking in an unstructured system has little impact (short varial area (in red)) while hydropeaking in a very structured system can lead to stranding zones (S) and has a large varial surface

manifest across all the living organism under river influence and were reported as early as 1939 (Vibert; 1939). They have been formally reported for riverine plants, macro-invertebrates and fish. Despite the growing body of evidence of these relationships, ecologist are still struggling to predict and quantify the biotic responses to hydropeaking. The reason for this struggle may be the difficulty to segregate the direct effects of hydropeaking from the impacts associated with water management and/or land-use management consequent of water resources development. Little is known about the role of hydropeaking event frequency, raising rate, falling rate, magnitude and length of impact on system ecology. River morphology seems to have a considerable weight in the intensity of hydopeaking impacts (Figure 2.6). A *varial* zone along each side of the river is created where aquatic biota cannot live which includes all of the shallow and low-velocity habitats.

During the review of the consequences of a hydropeaking regime on ***benthic macro-invertebrates***, the following major effects were constantly reported in literature, namely:

- an altered emergence patterns (Brittain and Salveit; 1989; Céréghino and Lavandier; 1998a,b; Frutiger; 2004b; Henricson and Muller; 1979; Raddum and Fjellheim; 1993; Ward and Stanford; 1982). Life cycles of macroinvertebrates were disrupted, mainly by average water temperature decrease due to hypolimnetic water releases from reservoirs

- macroinvertebrate stranding due to rapid level fall (Baumann and Klaus; 2003; Brittain and Salveit; 1989; Bunn and Arthington; 2002; Cushman; 1985; Petts; 1984). As previously mentioned, strongly morphology-dependent phenomenon associated with structured bed

- a decrease in the species richness (Baumann and Klaus; 2003; Bonacci and Roje-Bonacci; 2003; Brittain and Salveit; 1989; Cortes et al.; 2002; Mullan et al.; 1976; Munn and Brusven; 1991; Paetzold and Tockner; n.d.; Pozo et al.; 1997; Trotzky and Gregory; 1974)

- an alteration of the communities composition (Baumann and Klaus; 2003; Brittain and Salveit; 1989; Burns and Walker; 2000; Céréghino and Lavandier; 1998a; Céréghino et al.; 2002, 2004; Cortes et al.; 2002; Grubbs and Taylor; 2004; Parasiewicz et al.; 1998; Pozo et al.; 1997; Valentin; 1995). Functional groups proportions as well as individual size distribution were altered

- a decrease of the biomass (Baumann and Klaus; 2003; Bonacci and Roje-Bonacci; 2003; Brittain and Salveit; 1989; Céréghino and Lavandier; 1998a,b; Céréghino et al.; 2002; Henricson and Muller; 1979; Layzer and Gordon; 1993; Morgan et al.; 1991; Moog; 1993; Parasiewicz et al.; 1998; Raddum and Fjellheim; 1993; Trotzky and Gregory; 1974). Has been reported as an overall decrease and as a taxon specific decrease of biomass

- an increase of catastrophic drift (Baumann and Klaus; 2003; Bonacci and Roje-Bonacci; 2003; Céréghino and Lavandier; 1998a,b; Céréghino et al.; 2002, 2004; Gore et al.; 1989; Irvine and Henriques; 1984; Lagarrigue et al.; 2002; Layzer et al.; 1989; Raddum and Fjellheim; 1993). During sudden level rise, many organisms abnormally enter diurnal drift

- genera dominance alteration (Brittain and Salveit; 1989; Céréghino and Lavandier; 1998a; Morgan et al.; 1991; Bunn and Arthington; 2002; Valentin; 1995). Disappearing of sensitive or specialized species (restricted ecological valence)at the profit of generalists (broad ecological valance)

During the review of the consequences of a hydropeaking regime on ***fish***, following major effects were constantly reported in literature, namely:

- a reduced abundance of salmonid species (Bowen and Crance; 1997; Cattanéo et al.; 2002; Freeman et al.; 2001; Moscript and Montgomery D.; 1997)
- an altered community assemblage Baumann and Klaus (2003); Bowen et al. (1998); Kinsolving and Bain (1993); Travnichek and Macenia (1994); Welcomme (1989)
- a reduction in the species richness (Travnichek and Macenia; 1994). This was mainly due to a reduced availability of shallow and slow flowing habitat (Bowen et al.; 1998; Courret et al.; 2006; Freeman et al.; 2001; Valentin; 1995)
- a decrease in the overall biomass (Baumann and Klaus; 2003; Lagarrigue et al.; 2002; Moog; 1993; Parasiewicz et al.; 1998)
- a decrease in the reproductive success (Freeman et al.; 2001; Kinsolving and Bain; 1993; Sparks; 1995). Also due mainly to specific habitat loss.
- stranding of juveniles (Baumann and Klaus; 2003; Courret et al.; 2006; Cushman; 1985; Gehrke et al.; 1995; Perry and Perry; 1986; Petts; 1984; Saltveit et al.; 2001), especially at night and low temperature (Valentin; 1995)
- an overall mitigation effect caused by a behavioral adaptation to hydropeaking (Bernez et al.; 2004; Valentin; 1995; Mesick; 1995)
- a potentially detrimental effect on growth of juvenile brown trout (Flodmark et al.; 2004)

During the review of the consequences of a hydropeaking regime on ***periphyton and mosses***, the following major effects were constantly reported in literature, namely:

- an alteration in the algal biofilm species composition (Burns and Walker; 2000)
- an increased growth of periphyton and mosses below dams (Brittain and Salveit; 1989)
- an increased algal scour in river bed (Petts; 1984)

During the review of the consequences of a hydropeaking regime on ***aquatic macrophytes***, the following effects were reported in litterature, namely:

- reduced growth rate (Blanch et al.; 1999, 2000; Rea and Ganf; 1994)
- reduced seedling survival (Blanch et al.; 1999, 2000; Rea and Ganf; 1994; Rood et al.; 1995)
- alteration of species composition (reduction of exclusive species at the expense of widespread species)(Rorslett and Johansson; 1996; Bernez et al.; 2004)
- reduced species diversity (Rorslett and Johansson; 1996)
- washout of plants (Rood et al.; 1995)

During the review of the consequences of a hydropeaking regime on ***river-margin vegetation***, the following major effects were constantly reported in literature, namely:

- lower plant species richness (Jansson et al.; 2000; Nilsson et al.; 1997)
- lower plant cover (Jansson et al.; 2000; Nilsson et al.; 1997)
- floral community alteration (wind-dispersion favored by hydropeaking) (Jansson et al.; 2000)

During the review of the consequences of a hydropeaking regime on ***river morphology***, the following major effects were constantly reported in literature, namely:

- increase in yearly baseflow stability (Bunn and Arthington; 2002)
- bed incision (Booth; 1990; Montgomery et al.; 1999, 1996; Moscript and Montgomery D.; 1997)
- increased depth and velocities (Bunn and Arthington; 2002; Grubbs and Taylor; 2004; Parasiewicz et al.; 1998)
- reduction of scouring flows / bed armoring (Petts; 1984; Bunn and Arthington; 2002)
- channel widening (Booth; 1990)

During the review of the consequences of a hydropeaking regime on ***river physiochemistry***, the following major effects were constantly reported in literature, namely:

- altered thermal regime (Céréghino and Lavandier; 1998a,b; Cortes et al.; 2002; Flodmark et al.; 2004; Foulger and Petts; 1984; Frutiger; 2004a,b; Grubbs and Taylor; 2004; Kinsolving and Bain; 1993; Lagarrigue et al.; 2002; Pozo et al.; 1997; Raddum and Fjellheim; 1993; Silk and Ciruna; 2004; Ward and Stanford; 1979)

- release of nutrient-rich waters (Cortes et al.; 2002; Foulger and Petts; 1984) Contradictory results found by (Pozo et al.; 1997)

- increased turbidity (Cortes et al.; 2002; Foulger and Petts; 1984; Grubbs and Taylor; 2004; Ward and Stanford; 1984)

- altered conductivity during wave (Foulger and Petts; 1984; Pozo et al.; 1997)

- altered calcium concentration during wave (Foulger and Petts; 1984)

- decrease in pH (Pozo et al.; 1997)

- decrease in silicate (Pozo et al.; 1997)

- increased ammonia (Pozo et al.; 1997)

- decreased amount of stored organic material on the bottom (Raddum and Fjellheim; 1993)

In her PhD thesis, Valentin (Valentin; 1995) states a few observations on hydropeaking effect:

- increase of low flow discharge - an increase sufficient to guarantee that shallow zones and shelters remain submerged throughout the event, main beneficiary are fish, this measure has little effects on invertebrates

- based on the Q_H/Q_L: Q_H being the discharge during the hydropeaking event and Q_L being the discharge at low level – fish biomass was observed to be higher when this ratio is small. However, she states that this relation is strongly dependent on river morphology

- based on current velocities – macro-invertebrate structure alteration and resulting trophic alteration were explained based on very low current speed at low discharge ($< 8cm \cdot s^{-1}$) and speed at high discharge are elevated ($> 30cm \cdot s^{-1}$). This observation was made for relatively slow flowing rivers and is probably not applicable in faster rivers

- event frequency – the higher the frequency, the higher the effects

- on the event discharge – Valentin supposes that lowering the event discharge has a mitigation effect but was not able to state a practical rule. River morphology (shelter availability, maximal current velocity, rising rate, falling rate, etc...) seems to play a predominant role

- on the rising and falling rates - fish seem to be able to react and learn to cope with hydropeaking at a relatively young stage (over 2 months), but effect on the invertebrate is not documented. It is strongly probable that hydropeaking effect can be mitigated by lowering rising and falling rates

Other water use and management threats such as **groundwater exploitation**, are affecting mainly wetlands and lotic systems' connectivity by removing water from systems in close equilibrium.

2.2.2 Anthropogenic land use

Anthropic land use has also been costly for freshwater ecosystems. Agriculture, urbanisation, industrialisation, forestry, mining and recreation all pose potential threats on ecosystems.

The major harm done to freshwater ecosystem by *agriculture* is through water extraction and pollutants emission. Most of the water extracted returns to the atmosphere by evaporation and does not re-enter hydrosystems directly. Water returning after irrigation has usually had a decline in quality since it is usually loaded with anthropic inputs to increase production as well as sediments from eroded soils. The commonly encountered compounds are phosphorous, nitrogen and pesticides.

Land use management for agricultural purposes has also converted most highly-values (nutrient rich) wetlands and riparian areas into cultures, mainly by drainage of the water.

Agriculture alters the soil structure, either by compaction resulting from the use of machinery or trampling by livestock. The soil's percolation capacity as well as water storage capacity is decreased, altering the soil potential for water retention as well as the speed and amount of water rendered to the river. This can have a profound effect on the *hydrology* of associated rivers since water can find itself in the stream a lot faster, further increasing flood pikes. Water extraction has the tendency to prolongate low flow and drought conditions, which can significantly impact the biotic communities as well as the ecological processes. Thornton *et al.* (Thornton et al.; 1999) classified agricultural pollutants as:

- sediments – coming from erosion
- plant nutrients – phosphorous and nitrogen are typically applied in excess concentration leaving a surplus of available nutrients. These compounds accelerate the eutrophication process (algal blooms followed by bacterial explosion fed by algal decay). Water becomes anoxic harming typical aquatic life. Groundwater quality is also threatened by nitrate concentration
- biodegradable organic matter – manure runoff participates to the nitrogen and phosphorous enrichment of water
- heavy metals are found in some fertilizers, pesticides and insecticides. In high concentrations, can affect waterfowl and humans.
- synthetic organic chemicals (e.g. atrazine) usually have a high water solubility and take years to breakdown. A variety of pharmaceuticals ranging from antibiotics to a wide range of animal hormones (e.g. steroids) are the other major source of synthetic organic pollutants. The impacts on the living organisms include altered physiological processes, reproductive impairment and increased toxicity from chemical synergies (Kolpin et al.; 2002; Howe et al.; 1998)
- dissolved solids – salinity increases soil erosion
- microorganisms and their metabolic products, cause an increase in biological oxygen demand as well as diseases
- acidifying compounds. A lower pH increases the biodisponibility of heavy metals to organisms
- macro-pollutants – large debris such as plastic bags. May be harmful to birds and mammals

Urbanisation and industrialization is perhaps the greatest threat to freshwater ecosystem integrity. Alteration of the hydrology, urban sewers and point source pollution are the main consequences of urbanisation on freshwater ecosystems. Roads, parking lots and rooftops of an urban area minimize water infiltration in the soil. Surface water runoff (clear water) enters quickly the river and is susceptible of contributing to the flood peak. Groundwater recharge is generally decreased in impervious areas lowering the water table and altering the baseflow. Wetlands are often removed to make room for urban development and reduce nuisances such as mosquitoes. The removal of riparian vegetation as well as the installation of levees, stream tunnels, channelization, drainage and debris clearing alter directly the *physical habitat conditions*. Processes such as erosion and sedimentation are disturbed by an altered hydrologic and physiochemical regime, rendering life very inhospitable for native communities.

Urban surface runoff is often loaded with a variety of contaminants and has been reported as the prevalent source of pollution in urban areas (Silk and Ciruna; 2004). Urban environment may also have point source contaminants such as pipes discharging from water treatment plants, storm water drains or industrial sites. The contaminants commonly found in urban waters include high temperature water, nitrogen, phosphorous, heavy metals, organic matter, hormones, antibiotics as well as synthetics (Kolpin et al.; 2002; Thornton et al.; 1999). *Connectivity* is equally affected by urbanisation, mainly due to barrier construction and channelization. Water withdrawal adds to connectivity alteration by further reducing water during low flow periods. Hypoxic zones (extremely low oxygen concentration) were reported to create connectivity barriers between reaches. Temperature elevation, DO depletion and increase in sediments are all key factors shaping the aquatic community structure, so it is not surprising to observe that urbanisation alters the *biological composition and interactions* of aquatic communities.

Forestry can contribute to erosion and sedimentation as well as by providing freshwater ecosystems with fertilizers and pesticides. System *hydrology* can also be altered by forestry since runoff and groundwater recharging patterns are impacted. *Physical habitat conditions* are impacted by a decrease in woody debris in the hydrosystem. Trees provide a thermal buffer to surface water, and temperature increases have been reported after trees disappearance. Sedimentation and siltation of bottom substrate have also been reported, altering the physical habitat of lotic, lentic and transitional systems.

Mining can be another major anthropic activity altering freshwater ecosystems but will not be discussed in this case since its effects are very activity-dependent (i.e. coal mining, limestone mining, oil and gas exploitation, sand and gravel,. . .)

Recreation is also susceptible to alter freshwater habitats. Recreation is probably not a long-term impacting factor and can be rather easily mitigated.

2.3 River quality assessment

It is established that hydrosystem's aquatic communities structure and diversity is dependent of many physical, chemical and biotic factors. Physical and chemical variables being themselves strongly dependent on climatic and catchment properties which are in turn influenced by water management and land use ((Amoros and Petts; 1993; Petts; 1984; Tockner et al.; 2002) *and others*).

Hydrobiological studies were able to identify the major driving factors determining freshwater communities, but unfortunately very few (Wright et al.; 1998) were able to establish links between ecological factors and the structure of aquatic communities. Effects such as river morphodynamic, stream order and associated channel properties, watershed area and sources of organic matter.

Of major difficulty is to distinguish the influence of natural characteristics, including naturally occurring disturbances from changes caused by man. Despite these uncertainties, aquatic communities have been used to assess the quality of rivers and many practical methods haver been developed (AFNOR; 1985; OFEFP; 1998b) allowing the detection of the significance of various important environmental variables structuring aquatic communities which have been shown to reveal predictable changes due to natural variability as well as anthropic alterations.

2.3.1 Macro-invertebrate based assessment

Macro-invertebrates are well suited for river quality assessment since a relatively large amount of data exists, their identification is relatively simple in European waters (identification keys (Tachet et al.; 2000)) and they occur in large numbers in most stream types ((AFNOR; 1985; Amoros and Petts; 1993; OFEFP; 1998b; Rosenberg and Resh; 1993) *and others*). Most methods assess site quality in relation to anthropogenic impact, essentially focusing on organic pollution (eutrophication, contamination...). A review of the ecological assessment methods at hand was done by Verdonschot (Verdonschot; 2000). Organic pollution was in the past decades the major impact factor on stream and is now strongly declining at the benefit of other impact factors such as stream regulation and land use. The following macro-invertebrate assessment in ecological hydrosystem management are the most commonly referenced (modified from the literature review by Lek *et al.* (Lek et al.;

2005). Other reviews in (Balestrini et al.; 2004; Mebane; 2001; Statzner, Bis, Dolédec and Usseglio-Polatera; 2001)).

1. *Indexes assessment*

    - *Sparobic indexes* (Liebmann; 1962) – aquatic organisms have different pollution tolerance described in a semi-quantitative way
    - *Diversity indexes* species diversity is assessed under varying disturbances. The most widely used is the Shannon-Weaver index (Shannon and W.; 1949), and is relies on the species richness and abundance. For a review and evaluation of biodiversity indexes, refer to (Boyle et al.; 1990; Hellawell; 1986)
    - *Biotic indexes* combine richness and pollution tolerance. A review of those indexes is given by (De Pauw et al.; 1992)

    These approaches are mainly concerned with organic pollution stressors and are geographically restricted. Single metrics are assumed to increase or decrease along an increase in disturbance.

2. *Rapid assessment techniques* – Environmental degradation is assessed through a number of single metrics based on ecological attributes of biological communities. Six major groups of metrics are usually distinguished (Resh and Jackson; 1993; Resh and McElravy; 1993; Thorne and Williams; 1997)

    - *Richness indexes* – such as the number of overall taxa, – these metrics are considered sensitive to organic pollution
    - *Enumeration indexes* – such as the abundances, % of total EPT taxa and chironomids, % of dominant taxon, number of sensitive taxa, % oligochates – these indices usually consider a disequilibrium between taxa abundances caused by pollution
    - *Similarity index* or *Loss index* – number of taxa in common, community loss index. An overview of commonly used similarity indexes can be found in (Henk; 1981). These metrics usually compare a study site to a reference site
    - *Tolerance* or *Intolerance Biotic indexes* – these metrics include richness and assign tolerance or intolerance values per taxon. Examples in (Alba-Tercedor and Sanchez-Ortega; 1988; Armitage et al.; 1983; Wright et al.; 1984)

- *Functional indexes* – such as % of functional feeding groups (Malard et al.; 2002). These metrics link a disturbance to a food type modification

3. *Community Assessment techniques* – Wright (Wright et al.; 1984) performed a multivariate analysis to classify lotic systems based on their pollution status and used macro-invertebrates types for their assessment and prediction. Derleth (Derleth; 2003) assessed aquatic macro-invertebrates biodiversity as an element of forest sustainability. Verdonschot (Verdonschot; 2000) performed a multivariate analysis of macro-invertebrates and identified macrofaunal site groups described on the basis of organisms abundance linked to environmental variables termed *coenotypes*. These coenotypes interacting to form a functioning web setting the frame for ecological objectives in river management.

4. *Non-taxonomical assessment* – Taxonomical units are grouped into guilds sharing similar characteristics that are subsequently scored. The most commonly found examples are functional groups (e.g. functional feeding group (Cummins and Wilzbach; 1985; Malard et al.; 2002; Tachet et al.; 2000)) and species trait assessment (Gayraud et al.; 2003; Statzner et al.; 1994; Statzner, Hildrew and Resh; 2001; Statzner et al.; 2005)

5. *Macro-invertebrate prediction* – Classification of unpolluted lotic systems as well as prediction of macro-invertebrate community types was performed by multivariate analysis in the well known *River Invertebrate Prediction and Classification System - RIVPACS* (Wright et al.; 1984). Expected macro-invertebrate fauna to be found at a site is predicted giving limited environmental parameters. The observed fauna is then compared to the expected fauna predicted and a measure of the site quality is derived from this comparison. RIVPACS was modified for use in Australia into AusRivAS (Smith et al.; 1999) and ANNA (Linke et al.; 2005). Similar approaches were used in the Benthic Assessment of Sediment (BEAST) (Reynolds et al.; 1995). The 'Instream Flow Incremental Methodology' (IFIM (Bovee; 1982)) use macro-invertebrate preference curves in order to predict habitat availability and surface at varying discharge.

A literature review of the variables most commonly used in macro-invertebrate assessment of river quality is presented in Figure 2.7 and a review of the various stream assessment methods most commonly used is presented in table 4.1.

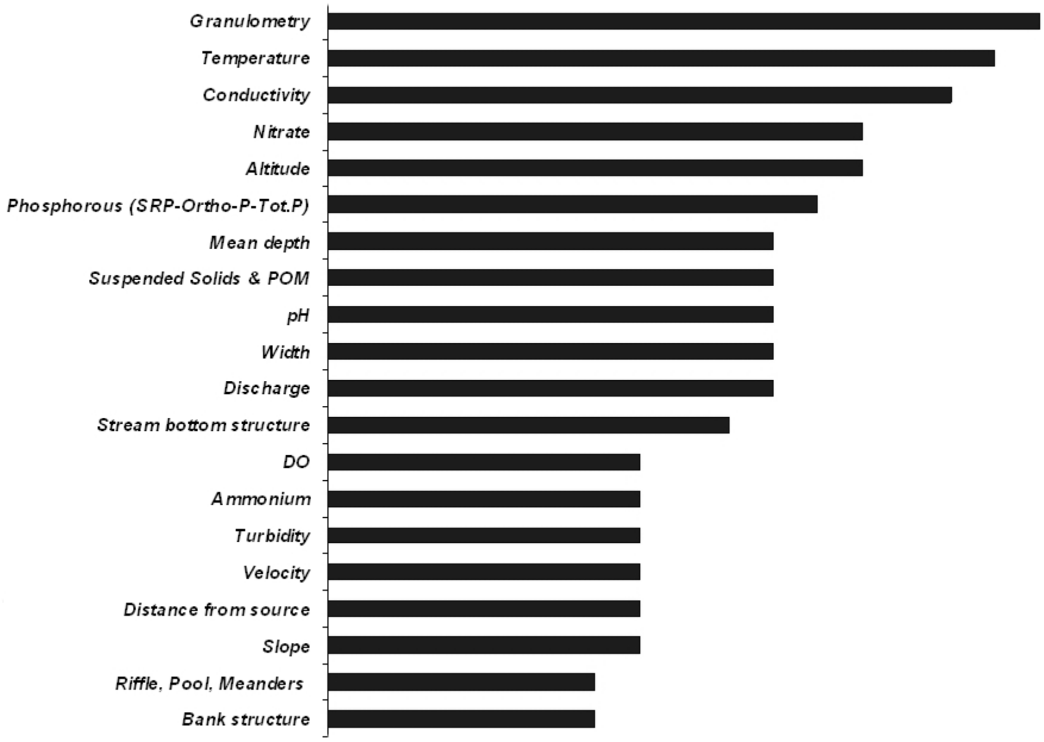

Figure 2.7: Review of the 20 most used variables in macro-invertebrate prediction studies (from (Armitage et al.; 2001; Balestrini et al.; 2004; Casas et al.; 2000; Castella et al.; 2001; De Crespin de Billy et al.; 2002; Dolédec et al.; 2000; Friberg et al.; 2001; Harper and Everard; 1998; Hieber et al.; 2003; Huryn and Wallace; 1987; Knispel and Castella; 2003; Logan and Brooker; 1983; Maioloni and Lencioni; 2001; Malard et al.; 2003; Mebane; 2001; Mérigoux and Dolédec; 2004; Monaghan et al.; 2002; Negishi et al.; 2002; Newson et al.; 1998; Rader and Belish; 1999; Richards et al.; 1997; Robinson et al.; 2002, 2004; Smith et al.; 1999; Statzner et al.; 2004; Turak et al.; 1999; Voelz and Ward; 1989)).

Table 2.2: Macro-invertebrate based river assessment methods most commonly applied in monitoring. Modified from (Hering et al.; 2003).

Assessment system	Country	Reference
Acidification index	S	(Henrikson and Medin; 1986; Johnson; 1998)
AMOEBA	NL	(Ten Brink et al.; 1991)
AQEM	DE, I, S, G, CZ	(Hering et al.; 2003)
AusRivAS and ANNA (Australian version and upgrade of RIVPACS)	AUS	(Linke et al.; 2005; Smith et al.; 1999)
Average Score Per Taxon (BMWP-ASPT)	BG, IR, S	(Armitage et al.; 1983; Chesters; 1980; Wright et al.; 1984)
Belgian Biotic Index	B, P, E, L, GR	(De Pauw et al.; 1992; De Pauw and Vanhooren; 1983)
BMWP Score	GB,S	(Armitage et al.; 1983; Chesters; 1980; Wright et al.; 1984)
Chandlers Biotic Score and Average Chandler Biotic Score	GB	(Balloch et al.; 1976; Chandler; 1970)
Danish Stream Fauna Index (DSFI)	DK, S	(Skriver et al.; 2000)
EKO	NL	(Verdonschot; 1990)
EBEOSWA	NL	(Peters et al.; 1994; STOWA; 1992)
German Faunal Index	D	(Lorenz et al.; 2004)
Instream Flow Incremental Methodology (IFIM)	USA	(Bovee; 1982)
Indice Biologique de Qualité Générale	L,B	(Verneaux et al.; 1982)
Indice Biologique Global Normalisé	F, B	(AFNOR; 1985)
Indice Biotico Esteso (IBE)	I	(Ghetti; 1997)
K-Index (Quality Index)	NL	(Gardeniers and Tolkamp; 1976)
Modified BMWP Score (BMWP-ASPT), Spanish version	E	(Alba-Tercedor and Sanchez-Ortega; 1988)
ONORM M 6232	A	(ONORM; 1997)
Quality Rating System	IR	(De Pauw et al.; 1992; De Pauw and Vanhooren; 1983)
RIVPACS	GB, IR	(Armitage et al.; 1983; Wright et al.; 2000)
ROCI	FIN	(Paasavirta; 1990)
Saprobic Water Quality Assessment Austria	A	(Moog; 1995; Moog et al.; 1999)
Saprobianindex DIN 38 410	D	(DEV; 1992)
SERCON	UK	(Boon et al.; 1996)
Systeme Modulaire Gradué	CH	(OFEFP; 1998a,b)

2.3.2 Fish-based assessment

Fish have an entirely aquatic life cycle and have a relatively long survival time in freshwater compared to aquatic macro-invertebrates. They are easy to catch and recognize, are present in most ecosystems, including impacted ones and are therefore good integrators of the physical, chemical and biological qualities of their habitats. Lotic freshwater fish community ecology is briefly review by Lek (Lek et al.; 2005), and appears historically at three spatial scales:

1. Three most referenced hypotheses address patterns of fish species richness at a global scale. *The species-area hypothesis* (McArthur and Wilson; 1963, 1967; Preston; 1962) which states that species richness increases with surface-area, the *species-energy hypothesis* (Wright; 1983; Wright et al.; 1993) which relates species richness to available energy and the *historical hypothesis* (Whittaker; 1977) which addresses patterns of post-glaciation ecosystem colonization to explain species richness.

2. Biotic zonation (caused by specific geomorphology, temperature,...) or downstream continual species addition explain assemblages at the basin scale. The increase in species following the river is generally attributed to a downstream increase in habitat diversity and stability (Amoros and Petts; 1993; Poff et al.; 1997).

3. Local fish assemblages appear not to be only determined by local processes acting within assemblages but also by processes operating at larger scale (Oberdorff et al.; 1998). In fact many species must undergo migrations sometimes of considerable distance between various habitats in order to complete their life cycle requirements (Amoros and Petts; 1993; Léveque; 1995; Silk and Ciruna; 2004). The driving ecological variables are the integrity of the river continuum, the heterogeneity of available habitats as well as their accessibility.

Fish communities have a long history of indicators of aquatic ecosystem quality (Table 2.3).

Table 2.3: Review of most common methods using fish in water quality assessment. Modified from (Lek et al.; 2005)

Assessment system	Concept	Reference
Bio-typology	Biocenosis, zonation, species richness	(Huet; 1959; Verneaux; 1973, 1976a,b)
CASIMIR	species preference curves yield a habitat value or weighted usable surface	(Jorde; 1996; Jorde et al.; 2001; Schneider et al.; 2001)
ESTIMHAB	statistical model of habitat model outputs	(Lamouroux and Capra; 2002; Lamouroux et al.; 1998; Lamouroux and Souchon; 2002)
EVHA	species preference curves yield a habitat value or weighted usable surface	(Ginot; 1998)
Fish Based Index	species richness and composition	(Oberdorff, Pont, Hugueny and Porcher; 2002; Oberdorff, Pont, Hugueny, Belliard, Berrebi dit Thomas and Porcher; 2002)
Index of Biotic Integrity	species richness and composition, indicator species metrics, trophic function metrics, reproductive function metrics and abundance and condition metrics	(Belpaire et al.; 2000; Karr; 1981, 1999; Oberdorff and Hugues; 1992)
PHABSIM	species preference curves yield a habitat value or weighted usable surface	(Bovee; 1982)
Systeme Modulaire Gradué	species richness and composition	(OFEFP; 1998a,b)

2.4 Research gaps

The lack of an integrated multi-purpose project assessment tool in river development project is the root of the SYNERGIE project's objectives. The state of the art review permitted the identification of various research gaps (RG) of interest at different hierarchical levels.

1. On hydrosystem ecology, and within the context of the SYNERGIE project, the following RG were identified:

    - *the choice of biological indicator(s) (i.e. aquatic macrophytes, diatoms, macro-invertebrate, fish,...) most economically appropriate* *and scientifically pertinent in order to identify and quantify hydrological and geomorphological anthropic alteration in hydrosystem*

- *the use this indicator in a relevant way in river development projects in order to predict an ecological consequence following an engineering action or an operation management*

2. On methodological considerations:

    - *how to minimize prediction error using various modeling methods*

3. On hydrological considerations:

    - *how to quantify and qualify hydropeaking effect in a generalized manner*

2.5 Scientific hypotheses to verify

Based on the research gaps, a set of scientific hypotheses to verify and objectives were formulated when aiming at developing the ecological module of an integrated multi-purpose project assessment tool in river development.

1. simple ecological considerations are sufficient to support and boost ecological integrity at the project's reservoir

2. ecological response can be modeled at the downstream river scale to assess the ecological integrity following a river development project

3. macro-invertebrates richness, and more precisely Ephemeroptera, Plecoptera and Trichoptera (EPT) taxonomic groups as an aggregated index is a good ecological indicators

4. the hour is an adequate time step to predict annual EPT richness dynamics

5. fish guilds (*riffle, bank, pool* and *midstream*) habitat suitability is a good ecological indicator

6. the hour is an adequate time step to predict annual habitat suitability index dynamics

7. the ecological response is function of system hydrology and river morphology

8. it is possible to improve the ecological integrity of the current regulated Swiss Upper Rhone River with a river development project

9. the ecological response of the downstream Swiss Upper Rhone River differs following various river development project scenarios and can be maximized

This work will attempt to provide qualitative answers to these scientific questions.

Chapter 3

Materials and methods

3.1 Study Site – the Swiss Upper Rhone River

The Swiss Upper Rhone River (Rhone) is the major tributary of the Geneva Lake (68% of the total water discharge and particulate matter input) and originates at the Rhone Glacier at an altitude of 1763m and has a catchment area of 5220km^2. The catchment area is comprised of 38% of rocks and glaciers, 62% of pastures, forest and agricultural lands (Loizeau and Dominik; 2000). The Rhone is naturally an alpine braided river that is structurally rich. Its natural dynamic was rich and powerfull, with significant high level waters and important carriage volumes either eroded or deposited. The bed of the main channel could extend over hundreds of meters and was in strong interaction with wetlands.

Figure 3.1: Swiss Upper Rhone location and tributaries. Circles indicate high dams (wall >15 m)

The Rhone lost its natural dynamic around 1850 (Figure 3.2)
and its current profiles result either from its first correction (1860) or its second correction (1920) (Figure 3.3).

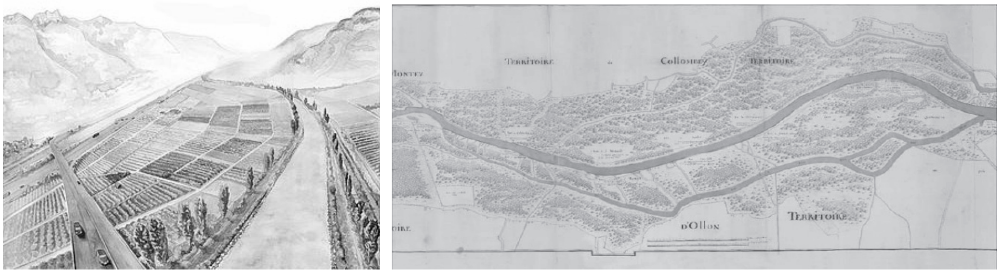

Figure 3.2: Swiss Upper Rhone floodplain in 2006 (WWF Wallis/Lebensraum Rotten Alberto E. Conelli) and as it appeared in 1760 by *F.G. de Rovéra* in (GIDB-R3; 2005)

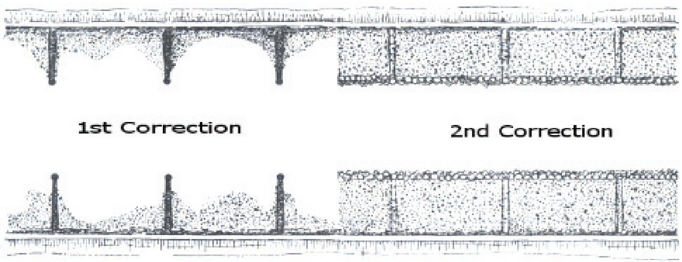

Figure 3.3: Bank design for the first and second Rhone corrections. Modified from Kalbermatten (1964) in (GIDB-R3; 2005)

Less than 1% of the Rhone is currently considered natural. The river bed is either embanked or armored. Its width varies between 30 and 60 m and bank morphology has little influence on bed bottom structure. The bed appears heavily clogged by fines, and even if the Rhone's hyporehic status is not well documented, it is probable that is is significantly impacted by river training, hydrology and carriage alterations. *Morphological* alterations can be summarized by a monotonous linear profile and a lack of structural diversity (i.e. gravel banks, islands, woody debris, riffles or pools). *Hydrological* alterations are caused by water abstraction and high altitude reservoir restitutions. Summer discharge is decreased, morphogenic flood frequencies are lowered, winter discharges are increased (Figure 3.4) and there is significant hydropeaking.

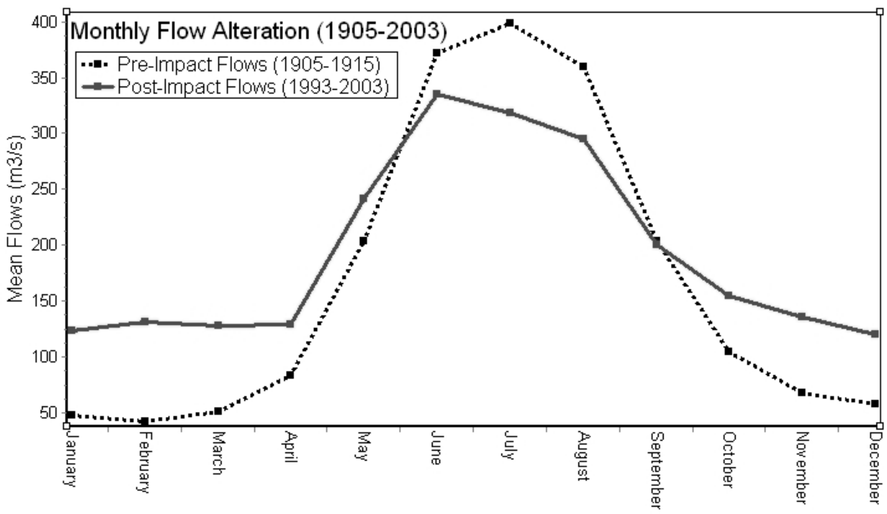

Figure 3.4: Inter-annual monthly discharges for periods 1905-1915 and 1993-2003 at *La Porte du Scex* gouging station. Overall increase in winter discharges and decrease in summer discharges.

Daily water level fluctuations can reach 80 to 90 cm, especially during winter, where water level is usually at its lowest (Figure 3.4).

Daily discharges variations from the order to 1:5 were reported (Baumann; 2004). Hydrological deficits can be summarized by the absence of alluvial dynamic, extremely high current speed and significant hydropeaking and thermal alteration (in average 1^oC colder in summer and 2^{o}C warmed in winter). The overall quality of the Rhone water is good. Yearly average water temperatures are between 6.9 and 7.2^{o}C with winter minimals (January) ranging from 3.8 to 4.1^{o}C and summer maximals (July) ranging from 8.9 to

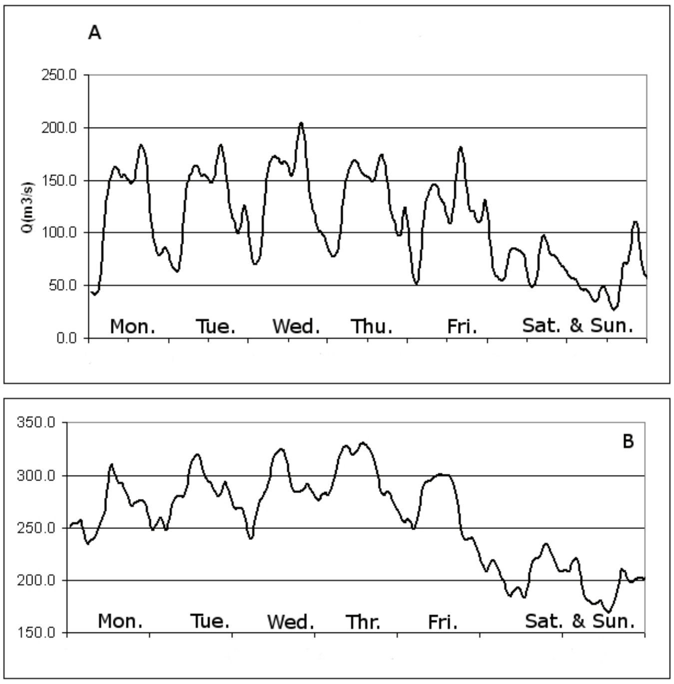

Figure 3.5: Winter (A) and summer (B) weekly discharge at *Branson* gouging station. Febuary and July 1999

9.8^{o}C.

3.2 The reservoir

3.2.1 Scientific context

The reservoir is the instrument allowing hydropeaking mitigation for the downstream river. It is a point landscape element and hence, its own ecological integrity within the hydrosystem is limited. Nevertheless, in the context of a multi-disciplinary project, its ecological integrity as well as its landscape integration should also be addressed, but at a point spatial scale. The use of a reservoir ecology 'model' was not called upon since:

- expertise in restoration and renaturation of lentic water bodies is well established in Europe
- a *model* should always be a system simplifier and called upon when complexity overrides our capacity to deal with the system as a whole
- the ecological success quantification as well as the landscape success quantification is hard if not impossible to achieve objectively since it is very subjective (dependent on a perceived state). This renders system modeling very tricky and in our sense increases the risk for model inappropriateness in this particular case

During the numerous `SYNERGIE` project meetings, it was argued that landscape integration appeared more important at the reservoir level than the ecological value *per se* of the element. A set of various general ecological and landscape considerations was established in order to propose a reservoir with an acceptable ecological and landscape integrity.

3.2.2 General ecological considerations

Heavily regulated hydrosystems are often very poor structurally (see Chapter 2.2). By creating a reservoir one is subject to locally increase hydrosystem structure diversity, which may act positively on the surrounding ecological integrity and landscape perception. The general ecological considerations shall attempt to fill the ecological gaps caused by a monotonous river system, namely:

- *enhance transitional zones* – ecotonal zones are zones of interests for both aquatic and terrestrial organisms as well as specialized organisms encountered solely in these types of zones. The rarity of ecotonal zones in a heavily regulated environment makes it of great biological value and hence should be promoted. It is not an easy tak in a strongly fluctuating environment such as the reservoir but it can be acheived (see Figure 3.6)

- *provide sanctuary zones* – due to the lack of biological zones of interest surrounding heavily regulated systems, sanctuary zones are virtually non-existant. Once the biological potential of the element is set, it is of prime importance to reserve a patch of aquatic, ecotonal and terrestrial terrain to fauna and flora alone, without the disturbances associated with human presence (e.g. dog, trails, litter...). This sanctuary needs not to be large and could be restricted to an island

- *limit invasive species development* – great care has to be taken to spot and eradicate invasive alien species susceptible to develop in an artificially restored environment (i.e. helophytes and macrophytes – macroinvertebrates – fish)

A standing body of water is susceptible of attracting many birds and small mammals typically associated with such environments and often of great biological value due to the rarity of their associated environment (e.g. Punta Funtana pond restoration in VS-Switzerland).

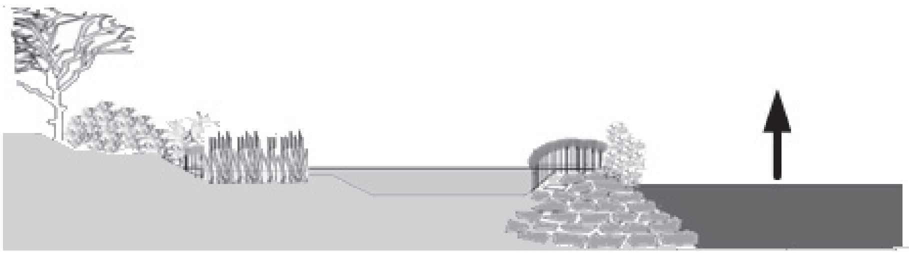

Figure 3.6: Possible method for enhanced transitional zone in a heavily fluctuating reservoir. During low flow the transitional zone remains submersed. Culvert filling takes place during high flows

3.3 Longitudinal continuum

3.3.1 Upstream migration

The principle of upstream fish passage facilities is to attract fish going upward the river to a specific location in the river downstream of the dam (barrier) so as to induce them pass upstream via the opening of a waterway (Larinier and Marmulla; 2002). Trapping and transferring is also observed in some cases. A review of various upstream fish passage facilities is made by (Larinier and Marmulla; 2002) and describes seven different types of facilities:

- *Pool-type fish passes* – the height to be passed is divided into several small drops by forming a series of pools
- *Denil fish passes* – baffles are placed on the floor and/or the walls of a rectangular flume with a relatively steep slope to reduce the mean velocity of the flow. This type of fish ladder is only suited for salmonids
- *Nature-like bypass channels and fish ramps* – channels characterized by a long gradient and the energy is dissipated through riffles and pools positioned along the channel
- *Fish locks* – large holding chamber located at downstream level of the dam and linked to an upstream chamber. Operating principle is similar to a navigation lock. There are doubts in the efficiency of such technique
- *Fish lifts* – fish are trapped and lifted up in a trap or a through with water and emptied at the top of the dam
- *Navigation locks* – similar to fish lock, but even less efficiency
- *Collection and transportation facilities* – usually used as a transitory measure before fish passes are built and operational. Migrants are trapped and transported upstream. An interesting fish trap is described by Pavlov (Pavlov; 1989).

3.3.2 Downstream migration

Downstream fish passage technologies are less advances than those concerning upstream fish passage and may just be the areas most in need of research (EPRI; 1986, 1992). Efforts toward the re-establishment of free movement for migrating fish began with the construction of upstream fish ways and as a consequence, downstream migration problems are only recently addressed. Various means of preventing fish to get in the water intake or to enhance fish survivability exist and range from physical barriers to behavioral barriers and implementation of *fish friendly* turbines. Their results are not equally effective.

Physical barrier

Fish can be prevented from getting into water intakes by using screens with an appropriate mesh size. By placing these screens diagonally to the flow, with the bypass in the downstream part of the screen, fish can be guided toward the bypass. Screen area must be sufficient to create low flow velocities suiting the swimming capacities of species and life stages concerned to avoid fish stranding. Larinier and Travade (Larinier and Travade; 1999) recommend that uniform velocities and eddy-free currents upstream of screens must be provided in order to effectively guide fish toward the bypass.

Behavioral barriers

A wide range of stimuli (hydrodynamical, electrical, visual and auditory) resulted in many experimental barriers such as sound screens, bubble screens, electrical screens and hydrodynamic *louver screens* (Figure 3.7) (Larinier and Marmulla; 2002). Most of the results obtained are not of great use because of their low reliability, their lack of generalization (strong dependence to local conditions of turbidity) and their specificity (as a function of species and size).

A noticeable feature is the louver screen, which has been reported by (Kynard and Horgan; 2001; Odeh and Orvis; 1998). The *louver screen* consists in an array of vertical slats aligned across the canal intake at a specified angle to the flow direction and guide fish toward a bypass (; ASCE).

3.3.3 The Artificial River

The proposed artificial river AR is a hybrid system combining a nature-like bypass channel to a pool fish pass (Figure 3.8). Its channel is characterized by a low gradient and energy is dissipated through a sequence of riffles and pools that mimic sequences in natural streams. This solution is often quite expensive economically and spatially but has the advantages of permitting invertebrate migration and development of associated transitional habitat. Another disadvantage is the need to add a gate due to the strong variation of the upstream level which may cause hydraulic conditions making the fish passage difficult. The pool fish pass is located right before the turbines and ensures that fish having *missed* the nature-like river entrance are not further delayed. The pool pass design criteria should be based on the swimming capacities and behavior of the target species. Recommendations by (Larinier and Marmulla; 2002) state that the drop between pools should be inferior to 0.3 m. The pool volume should be determined by a maximum energy

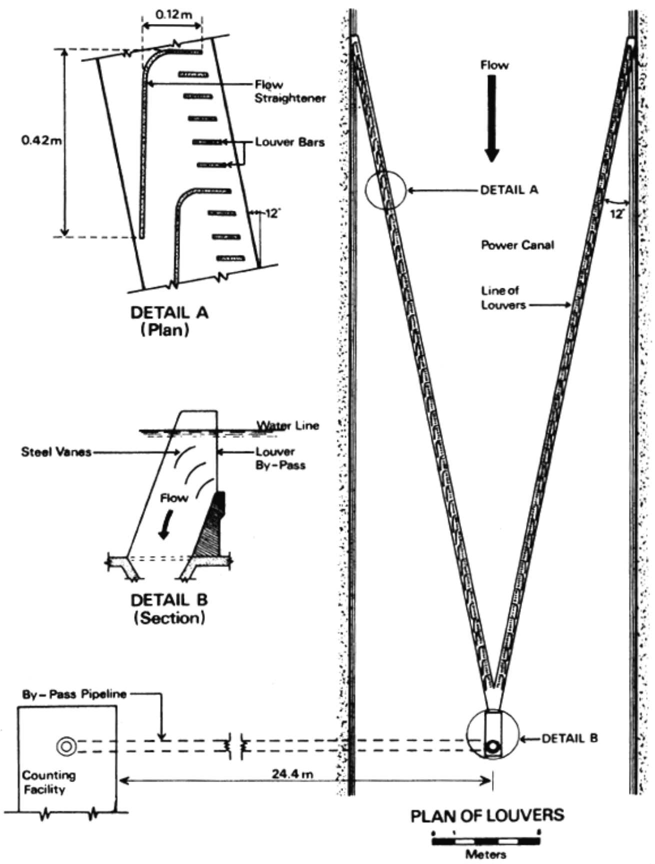

Figure 3.7: Louver deflectors used for guiding Atlantic salmon smolts from a power canal at the East River, Sheet Harbour hydroelectric site in Nova Scotia, Canada. From `http://www.fao.org/DOCREP/003/AA044E/AA044E11.htm`

dissipation in the pool limiting turbulence and aeration (between 200 $watts \cdot m^3$ for salmonids to < 100 $watts \cdot m^3$ for other smaller species and juveniles). The AR discharge has to be sufficient to compete with the river flow during the migration period. It is commonly accepted that the AR discharge must be approx. 5% of the competing flow.

Figure 3.8: Artificial river of the *Ruppoldingen* run-of-the-river power plant – image from `http://www.atel.ch/`

3.4 The Downstream River

3.4.1 Scientific context

In the past decades, water quality was considered the major threat to biodiversity in many aquatic systems. Many countries actively responded by implementing legal instruments resulting in the recent overall improvement of water quality of their freshwater systems. Hydrological and morphological alteration of freshwater systems is now a major emerging threat to biodiversity (Lek et al.; 2005; Pellaud et al.; subm.). The extent of hydrosystem development profoundly affects all of the hydrosystem elements (i.e. lotic, lentic and transitional - see Chapter 2). To my knowledge, there are few indicators taking hydrological and/or morphological alterations into account at the river level (i.e. ecomorphological assessment of the *Système modulaire gradué* (CH) (OFEFP; 1998a,b)) and none at the hydrosystem level. Further river development project raise many issues ranging from energy production, flood safety, ecological awareness, landscape and leisure integration (Reichert et al.; 2004). Many divergent lobbies (ecologists, engineers, economists...) can be expected and hence it becomes primordial to justify as clearly as possible an engineering or management scenario. For ecologists, this means to be able to defend a design or management scenario susceptible to lower energy production or increase structural cost of the scheme (e.g. by proposing a fish ladder, reducing hydropeaking,...). Primarily, ecological objectives must be defined, then appropriate bioindicators (target) must be identified and finally a scenario ecological consequence on the target has to be assessed. It becomes of prime importance for ecologists involved in modern multi-purpose river development projects (such as the `SYNERGIE` project, on Chapter 1.1) to account for the effects of hydrology (i.e. hydropeaking) in its morphological context on their target indicator. At the scale of the downstream river, the ecological objective is hydropeaking mitigation. It is suspected that the ecological integrity of the river will be improved with a reduction in the frequent water level variations and a shift of the hydrology following a more *natural* pattern.

3.4.2 Models to predict the downstream river ecological integration

Macro-invertebrates are very good candidates as bioindicators and their richness can be predicted by models and used to estimate system ecological integrity. The basic assumption being that a high richness represents a high ecological integrity (see Chapter 2). Ephemeroptera (Mayflies), Plecoptera (Stoneflies) and Trichoptera (Caddisflies) richness are assumed to represent the system's macro-invertebrate community, and it is assumed that a high richness of these three groups corresponds to a high ecosystem quality (Amoros and Petts; 1993).

Ephemeroptera have obligate aquatic larvae and are a very diversified group thriving either in lotic or lentic systems, are either swimmers, burrowers or crawlers and can be detritivorous shredders, grazers, filter-feeders or predators (Tachet et al.; 2000).

Plecoptera have obligate aquatic larvae relatively homogeneous morphologically. In their aquatic stage, they are preyed on by many fish or macro-invertebrates. Plecoptera have a higher affinity for lotic systems, with only one European species found in lentic systems *Nemurella picteti* (Tachet et al.;

2000). Plecoptera are diversified longitudinally, and most are encountered in headwater zone (many Nemouroidea). Species number tends to decrease in the transfer and mainstream zone and few species of Chloroperlidae, Perlidae and Perlodidae remain (Tachet et al.; 2000). Their disymetric distribution may induce bias when using biotic indexes such as the IBGN (AFNOR; 1985), where the highest value groups are often based on Plecoptera richness (Tachet et al.; 2000). As a whole, they are reported governed by hydraulic conditions (Céréghino and Lavandier; 1998b).

Trichoptera are insects with aquatic nymphs and pupae, with the exception of the Limnephilidae *Enoicyla*, which has adapted to a terrestrial life. Trichoptera larvae can either be free or occupy a fixed position, in which case they are generally filter feeders, predators or grazers. The majority of Trichoptera larvae are epibenthic, but some may be found a couple of cm in soft substrate. Some free swimming forms also exist (Tachet et al.; 2000). Trichoptera larvae are a trophic source for many invertebrates and fish.

A *macro-invertebrate model* is implemented consisting of (1) the prediction of each group's richness based on hydraulic and geomorphological parameters and (2) the adjustment by the hydropeaking effect.

Fish communities are also excellent indicators of river quality (Baglinière and Maisse; 1991; Karr; 1981) and have been used extensively to monitor rivers throughout the world. As previously mentioned, hydraulic variables are among the most influenced by anthropic river alteration and of particular interest in our case. Literature considers that species show marked preferences for hydro-geomorphic variables such as depth, velocity and substrate distribution (Bovee; 1982; Fragnoud; 1987; Ginot; 1998; Statzner et al.; 1988) and hence that species distribution is under the direct influence of the hydraulic component of the habitat (Lamouroux et al.; 1998). A family of quantitative methods (i.e. Instream Flow Incremental Methodology - IFIM) modeling aquatic habitat suitability were born in the eighties (PHABSIM (Bovee; 1982)). These models are attractive and still widely used with regional modifications (EVHA (Ginot; 1998) and RHYHABSIM (Jowett; 1989)). These models couple a hydraulic habitat model to a biological preference model in order to estimate the impacts of hydrological and morphological changes on fish populations (Lamouroux and Jowett; 2005). They predict habitat values scored between 0 and 1 for a number of fish species or fish guilds. Applying these IFIM models involves the survey of river bed topography coupled to precise measurements of depth and current velocities dependent on the complexity of the hydraulic model used, which also requires prior calibration. Lamouroux and Capra (Lamouroux and Capra; 2002; Lamouroux and Souchon; 2002) proposed a statistical model of the output of conventional IFIM habitat models using a low-effort reach description, mainly depth- and width- discharge relations, substrate size and average flow.

A *statistical fish-guilds habitat value model* will be implemented and complete the macro-invertebrate model in the prediction of the system's ecological integrity.

The resulting outputs had the advantage to predict Habitat Values (HV) for four fish *guilds*[1] based on hydraulic variations and preference curves.

The downstream river ecological response will be based on the macro-invertebrate model output and the fish guild HV model output, following the assumption that high ecological integrity is translated by a high macro-invertebrate richness output and high fish guild habitat values output.

3.4.3 Explicative and response variables

Database used

The data used comes from various sources in literature, namely: Gogniat and Marrer (1984/85); ECOTEC (1996, 1998, 1999, 2004); Baumann (2004); GIDB-R3 (2005). The method used for the benthic fauna analysis is the one of the IBGN (AFNOR; 1985). Each site is sampled 8 times (on a total area of 0.4 m^2) in all substrate types and current conditions. It is highly probable that in some cases the protocol may have been adapted to site specific conditions. In the Rhone, sampling was conducted mainly on the banks, where benthic fauna is most represented (Amoros and Petts; 1993) and where it is technically easier to sample. There may very well be some spatial autocorrelation between neighboring values of a specific variable, but the sampling details available were not sufficient to conduct an autocorrelation analysis. It is also likely that between the different campaigns, there is a variability due to different observers. Sampled organisms were kept in 10% formaldehyde, sorted and identified up to *family* taxonmical level.

Response variables

A total of three response variables are used in the model, all of which are depicted at the *family* taxonomical level.

1. Ephemeroptera richness - y_E
2. Plecoptera richness - y_P
3. Trichoptera richness - y_T

[1]group of species or individuals exploiting similar resources (e.g. food, preference for current velocity, preference for depth,...) within an ecosystem

Explicative variables

A total of 18 explicative variables inputs are used in the model:

1. $x_1 - x_5$: The two dominant substrate classes ($binary[-]$).
   - x_1 – bedrock or blocks, of size $> 200mm$
   - x_2 – cobbles, size ranging from $20 - 200mm$
   - x_3 – gravels, size ranging from $2 - 20mm$
   - x_4 – sands, size ranging from $0.02 - 2mm$
   - x_5 – fines, of size $< 0.02mm$
2. x_6 – Short response hydropeaking integrator. This indicator corresponds to the 90^{th} percentile of ranked 3 hour water level sequences for the preceding two months measurements ($HS[m]$)
3. x_7 – 1-year hourly mean water depth ($Z[m]$)
4. x_8 – 1-year hourly mean current velocity ($S[m \cdot s^{-1}]$)
5. x_9 – Long response hydropeaking integrator. This indicator corresponds to the 7997^{th} ranked 3 hour sequence water level variation ($HL[m]$). Year of interest 3 hours water level variations are ranked, and this integrator corresponds to the 7997^{th} sequence, representing *ca.* the 333^{th} day threshold of non-exceedence. This factor was defined arbitrarily on the ranked 3hr. sequences curve and is a yearly integrator of hydropeaking extent.
6. x_{10} – reach distance from origin ($KmS[km]$)
7. x_{11} – annual hourly discharge - ($Q[m^3 \cdot s^{-1}]$)
8. x_{12} – annual hourly water temperature - ($T[^oC]$)
9. x_{13} – annual hourly bankfull width - ($W[m]$)
10. x_{14} – median substrate size ($d50[m]$)
11. x_{15} – reference yearly (long term) hydropeaking integrator. Reference value for hydropeaking integrator value under undisturbed or objective conditions $HL_{\text{Ref}}[m]$. This value is naturally never null and therefore subsequently withdrawn from the the HL in the fuzzy approximation of the hydropeaking effect order to level it (see Chapter 4).

12. x_{16} – bed-width variability index - fuzzy approximation of ecomorphological characterization of the bed width variability. An input of 1 accounts for a null width variability and is representative of trained channels. An input of 10 accounts for a high bed width variability typically found in natural alluvial reaches

13. x_{17} – depth variability index - fuzzy approximation of reach depth variability. An input of 1 accounts for a null depth variability . An input of 10 accounts for good depth variability such as can be encountered in *natural* conditions

14. x_{18} – bed structure variability - fuzzy approximation of the bed structure variability. An input of 1 accounts for a null variability (uniformly trained structure) while an input of 10 corresponds to a very diverse structure such as the one that can be found in nature (pool, riffle, runs features)

Development of the best EPT richness prediction model

In order to develop the best EPT bioindicator model, the following seven steps are taken (Figure 3.9):

1. obtain a faunistic survey of the E-, P- and T-groups with linked hydrological and morphological information (p. 73)

2. select the best model for the E-, P- and T-groups (p. 78)

3. implement a specific hydropeaking adjustment FIS on each group (p. 97)

4. inject annual hourly hydraulic and morphologic inputs resulting from a scenario's design and management consequences (p. 73)

5. models yield a one year hourly richness indicator for each E-, P- and T-groups (p. 106) which are adjusted by group-specific hydropeaking effect (p. 108)

6. average each group specific annual hourly value to obtain group specific richness indicator

7. sum averaged group indicator values (p.109)

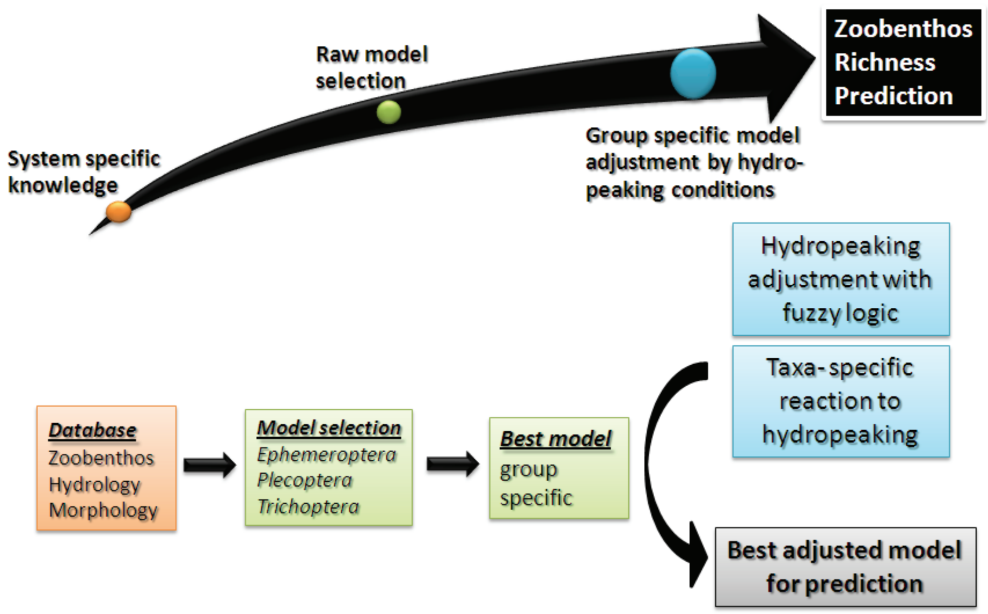

Figure 3.9: Summary of the best EPT richness prediction model development

Chapter 4

Hydropeaking effect assessment on EPT and fish

4.1 Introduction

To my knowledge, the effect hydropeaking has never been generalized on macro-invertebrate richness or on fish habitat. In this chapter we try to test the use of two types of bioindicators on the effect of hydropeaking. When choosing bioindicators, it is important that their ecology is well documented, that they are affected by the effect one tries to assess (i.e. hydropeaking) and that they are typical of the application site. Macro-invertebrate and fish are well documented in European regions, and were both reported sensitive to hydropeaking in literature (see section 2.2.1 on p. 44). This chapter has the following underlying assumptions:

1. Ephemeroptera, Plecoptera and Trichoptera groups (EPT-groups) are assumed to represent the macro-invertebrate benthic community and their richness monitor the ecological integrity of the aquatic environment

2. fish community can be represented by four guilds, namely a *bank* guild, a *pool* guild, a *riffle* guild and a *midstream* guild. The habitat suitability of each guild can be determined and transformed into a weighted usable area (WUA) which can provide insights on the ecological integrity of the aquatic environment

3. hydropeaking effect is closely related to river morphology, and cannot be assessed without taking into account the morphological context into which bioindicators thrive

In order to assess the effect of hydropeaking on the EPT-groups, we (1) select the best richness prediction model for each order based on existing data (presented in the form of a submitted article)(2) quantify the hydropeaking effect taking river morphology into account (presented in the form of an article to be submitted) (3) adjust the resulting models with an order specific hydropeaking effect weight. In order to assess the effect of hydropeaking on fish guilds, we compute each guild's habitat suitability under various hydropeaking conditions.

4.2 Model selection for EPT richness

Modern model selection methods for integrating benthic aquatic richness in river development projects

Pellaud, M.[1], Sardy, S.[2], Schlaepfer, R.[1] and A. Buttler[1]. Submitted to *Ecological Modelling*

Abstract

Conflict of interest in river development projects calls for a multidisciplinary approach. Maybe for the first time in its history, hydrosystem ecology is taken into account at a river development project's conception phase. In literature, ecological integration may appear fuzzy for multiple reasons, one being the inappropriate use of widely available statistical prediction methods of river quality. In this work, we evaluate some parametric and non-parametric models on a dataset of benthic macro-invertebrates from the Swiss Upper Rhone River in order to minimize prediction error. Through such a methodology we aim at developing a scientifically sound and easy to implement model selection protocol for ecologists. Contribution to the clarification of ecological integration in a river development project is generalized through the most appropriate choice of a community structure prediction tool for a specific river-system. We bring forward the potential of the lasso and gradient boosting regression techniques and the limits inherent to the use of non-parametric methods in river-ecology.

Keywords: model selection; macro-invertebrates; lasso; gradient boosting; support vector machines; radial basis functions; river development project; Swiss Upper Rhone River

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4.2.1 Introduction

The consequences of anthropic water-use and land-use management are becoming increasingly important and detectable in the environment (Amoros and Petts; 1993). There is pressure on water demand for public supplies which causes a variety of complex ecological impacts (Poff; 1997; Silk and Ciruna; 2004). Environmental protection agencies legally responded by calling for a more sustainable use of our water resources. River development projects are in the middle of a conflict of interests between public demand for freshwater availability, energy supply demand, flood security, landscape integration, ecological integrity and limited spatial resource. Focusing on an ecological point of view, river development resolutions are often taken by authorities without sufficient clarity on how objectives, outcomes and concerns were considered during the decision making process. This can lead to lack of acceptance by stakeholders and ultimately project failure (Reichert et al.; 2004). River development projects now call for a truly multi-disciplinary approach where economical, social and ecological aspects must be taken into account starting at the project conception phase (Heller et al.; 2006; Pellaud et al.; 2005). Ecological assessment and response (prediction) is complicated by various factors such as: The very *origin of ecological concerns*, or in other words the difficulty to properly distinguish anthropic impact from natural phenomenon in a complex aquatic ecosystem. The *shift in the threats to biodiversity toward a morphological and hydrological alteration* of natural conditions. In the past decades organic pollution was seen as the major impacting factor. The majority of earlier studies assessed river quality based on organic pollution. Efficient legal instruments were implemented (e.g. phosphate ban in washing detergent in Switzerland, sewage connection to a water treatment plant,...) resulting in the overall improvement of water quality. However, few studies focus on such issues in an integrative manner (Lek et al.; 2005). The relative *difficulty to link hydrologic variables to aquatic communities* Hydrobiological studies (Table 4.1) have identified the major factors determining freshwater hydrosystem ecology. Good river quality was quickly linked to species community structure, with poor conditions being reflected by a high abundance of few generalist taxa and healthy conditions reflected by a diversified structure of specialists. Hydrology, connectivity, water physiochemical quality, geomorphology and others were found to have complex impacts on species community structure and were all found to be somehow altered either by land-use or water-use management (Amoros and Petts; 1993; Brittain and Milner; 2001; Brouwer; 1987; Bundi et al.; 1990; Cairns and Heckman; 1996; Poff; 1997; Silk and Ciruna; 2004; Tockner et al.; 2002). Finally, *the inappropriate use of statistical modeling techniques* by some ecologists. From a mathematical point-of-view, the transparency of river studies may also be altered by the choice of modeling and estimating methods used. Thanks to technological development and the broader accessibility of statistical softwares, virtually anyone is able to use complex statistical methods, sometimes too complex for the problem at hand. Using them as a black box is a real danger, leading to adverse effect such as over- or under-fitting, and inappropriate model selection. For instance, many nonparametric methods, if not well tuned, will lead to overfitting the training data set because of the excess of degrees of freedom they offer; the same effect may apply to parametric models when the number of covariates is higher than the number of observations. The so-called *curse of dimensionality* also prevents the use of nonparametric models use when the dimension of the covariates is high. For an excellent view of these issues, see (Hastie et al.; 2001). The goal of this article is to shed some light in the ecohydraulical community on the latter issue, *the use of appropriate statistical regression methods*, so that the researcher can clearly identify whether he is using a (non)linear and (non)parametric model, and a (non)linear (non)Bayesian method of estimation, and whether he is using them correctly. It will also become clear that independent, but identically distributed, training and test sets must be employed to calibrate, test and compare models, which we do for the prediction of aquatic Mayflies (Ephemeroptera), Stoneflies (Plecoptera) and Caddisflies (Trichoptera) Richness of the Swiss Upper Rhone River.

4.2.2 Statistical models and estimators

The regression problem can be stated as follows. Let $\mathcal{C}_N = \{(y_n, \mathbf{x}_n)\}_{n=1,\dots,N}$ be the set of measurements called the calibration or training set of size N, where y_n are the response values (e.g., taxonomic richness) and $\mathbf{x}_n = (x_{n1}, \dots, x_{nP})$ are the vectors of P covariates (e.g., depth, current velocity...). The goal of regression is to predict the response from the covariates by modeling a multivariate function $f(\cdot)$ for which

$$y_n = f(\mathbf{x}_n) + \epsilon_n,$$

where $\epsilon_1, \dots, \epsilon_n$ are independent and identically distributed (i.i.d.) zero-mean measurement errors, often assumed to be Gaussian $\mathrm{N}(0, \sigma^2)$. The form of $f(\cdot)$ can be specified further either in a parametric or a nonparametric way, using a linear or nonlinear expression, as we will see in Section 4.2.2. Once specified, $f(\cdot)$ must be estimated based on the calibration set $\mathcal{C}_N$. We will see in Section 4.2.2 that the estimation can be linear or nonlinear.

Table 4.1: River assessment methods most commonly applied in river and hydrosystem monitoring. Modified from (Hering et al.; 2003).

Assessment system	Country	Reference
Acidification index	S	(Henrikson and Medin; 1986; Johnson; 1998)
AMOEBA	NL	(Ten Brink et al.; 1991)
AQEM	DE,I,S,G,CZ	(Hering et al.; 2003)
Average Score Per Taxon (BMWP-ASPT)	BG,IR,S	(Armitage et al.; 1983; Chesters; 1980; Wright et al.; 1984)
Belgian Biotic Index	B,P,E,L,GR	(De Pauw and Vanhooren; 1983; De Pauw et al.; 1992)
BMWP Score	GB,S	(Armitage et al.; 1983; Chesters; 1980; Wright et al.; 1984)
Chandlers Biotic Score and Average Chandler Biotic Score	GB	(Chandler; 1970; Balloch et al.; 1976)
Danish Stream Fauna Index (DSFI)	DK,S	(Skriver et al.; 2000)
EKO	NL	(Verdonschot; 1990)
EBEOSWA	NL	(STOWA; 1992; Peters et al.; 1994)
Instream Flow Incremental Methodology (IFIM)	USA	(Bovee; 1982)
Indice Biologique de Qualité Générale	L,B	(Verneaux et al.; 1982)
Indice Biologique Global Normalisé	F,B	(AFNOR; 1985)
Indice Biotico Esteso (IBE)	I	(Ghetti; 1997)
K-Index (Quality Index)	NL	(Gardeniers and Tolkamp; 1976)
Modified BMWP Score (BMWP-ASPT), Spanish version	E	(Alba-Tercedor and Sanchez-Ortega; 1988)
ONORM M 6232	A	(ONORM; 1997)
Quality Rating System	IR	(De Pauw and Vanhooren; 1983; De Pauw et al.; 1992)
RIVPACS	GB,IR	(Armitage et al.; 1983; Wright et al.; 2000)
ROCI	FIN	(Paasavirta; 1990)
Saprobic Water Quality Assessment Austria	A	(Moog; 1995; Moog et al.; 1999)
Saprobianindex DIN 38 410	D	(DEV; 1992)
SERCON	UK	(Boon et al.; 1996)
Systeme Modulaire Gradué	CH	(OFEFP; 1998a)

Models

Given P measured covariates, the simplest model is parametric and linear

$$f(\mathbf{x}) = \alpha_0 + \sum_{p=1}^{P} \alpha_p x_p, \tag{4.1}$$

where α_0 is the intercept coefficient and $\alpha_1, \ldots, \alpha_P$ are the coefficients attached to each covariate in $\mathbf{x} = (x_1, \ldots, x_P)$. Note that the *constant model*, $f(\cdot) = \alpha_0$, corresponds to the case where all the other coefficients are set to zero. It is sometimes natural to transform the original covariates, using for instance the logarithm (e.g., $x_{P+1} = \log x_1$), the power (e.g., $x_{P+2} = \sqrt{x_2}$) or the product (e.g., $x_{P+3} = x_1 x_2$). Putting the original and transformed covariates together, the model $f(\mathbf{x}) = \alpha_0 + \sum_{p=1}^{P'} \alpha_p x_p$ is richer and remains linear. Once the model is fitted to the data (see the estimation Section 4.2.2), the validity of the model should be checked, for instance by means of residual plots (Belsley et al.; 1980). If the noise is not Gaussian (for instance Poisson or Binomial), then generalized linear models (Chambers et al.; 1991) can be employed.

While a linear parametric model (4.1) may often be a good model or a sufficiently good approximation to the reality, the residual plots or scientific considerations may reveal that a more complex association exists. Nonparametric additive models (Hastie and Tibshirani; 1999) assume $f(\mathbf{x}) = \alpha_0 + \sum_{p=1}^{P} f_p(x_p)$. They allow more flexibility than (4.1) by letting each univariate function $f_p(\cdot)$ be any smooth function. To fit the univariate functions nonparametrically, two classes of smoothers are local averages (e.g., running mean, running median, kernel smoothers) and expansion-based estimators (e.g., trigonometric functions, smoothing splines, wavelets). The latter assumes each $f_p(\cdot)$ can be expressed as a linear combination of N known basis functions $\{\varphi_{pn}\}_{n=1}^{N}$: $f_p(x_p) = \sum_{n=1}^{N} \alpha_{pn}\varphi_{pn}(x_p)$. The increased flexibility of such linear nonparametric models is reflected by the large number of coefficients that rises from $1 + P$ for a parametric model to $1 + PN$ for a nonparametric model. Like with parametric linear models, generalized additive models have been developed when the noise if not Gaussian. For a review on (generalized) additive models and a recent development using wavelets, see (Sardy and Tseng; 2004).

Even more flexible are projection pursuit models (Friedman and S.; 1981) which assume that $f(\mathbf{x}) = \sum_{q=1}^{Q_{\text{PP}}} f_q(\mathbf{w}_q'\mathbf{x})$, where $f_q(\cdot)$ are again univariate smoothers applied to all the covariates projected into Q_{PP} (unknown) directions $\mathbf{w}_q$. The number of terms Q_{PP} to include is also unknown and part of the model. In a similar spirit, neural networks (see (Ripley; 1996) and references therein) assume, in their simplest form (with a single hidden layer), that $f(\mathbf{x}) = \sum_{q=1}^{Q_{\text{NN}}} \sigma_{\boldsymbol{\theta}_q}(\mathbf{w}_q'\mathbf{x})$, where $\sigma_{\boldsymbol{\theta}_q}(\cdot)$ are simple known functions such as sigmoids or (Gaussian) radial basis functions, which are parametrized by a small number of parameters $\boldsymbol{\theta}$. Both projection pursuit and neural network models are nonlinear and nonparametric. Note that since the neural network's parametric univariate functions $\sigma(\cdot)$ are less flexible than the nonparametric $f_q(\cdot)$ used by projection pursuit, the number of terms Q_{NN} used by neural network is often much larger than Q_{PP}.

Nonparametric kernel-based methods estimate instead $f(\cdot)$ at $\mathbf{x}$ by taking a weighted average of responses y_n around $\mathbf{x}$:

$$\hat{f}(\mathbf{x}) = \sum_{n=1}^{N} K_\lambda(\|\mathbf{x} - \mathbf{x}_n\|) y_n,$$

where $K_\lambda(\cdot)$ is a univariate weighting function which is decreasing with the distance between the point $\mathbf{x}$ to the data points $\mathbf{x}_n$. The regularization parameter λ, called the bandwidth, controls how fast the kernel decreases to zero: when λ is small the weighted average is local, and when λ becomes large the estimate tends to the constant function at all point $\mathbf{x}$, the average $\bar{y} = \sum_{n=1}^{N} y_n/N$ of all the responses. Because the kernel is an isotropic function, the covariates $\mathbf{x} = (x_1, \ldots, x_P)$ should be standardized to unit standard deviation. Borrowing from kernel and expansion-based estimators, radial basis function-based estimators model the underlying association linearly as

$$f(\mathbf{x}) = \sum_{q=1}^{Q} K_\lambda(\|\mathbf{x} - \mathbf{u}_q\|) \alpha_q, \tag{4.2}$$

where $K_\lambda(\cdot - \mathbf{u}_q)$ are now basis functions indexed by a location parameter u_q and scale parameter λ. Kernel- and radial basis function-based models suffer from the curse of dimensionality: when the dimension P increases, a point $\mathbf{x} \in R^P$ has less and less neighbors within the data points $\{\mathbf{x}_n\}_{n=1,\ldots,N}$, unless N increases exponentially with P. In ecology this is rarely the case since N is often bounded by time, financial or experimental constraints.

Estimators

All of the models described in the previous Section require the estimation of coefficients (e.g., $\boldsymbol{\alpha}$, $\mathbf{w}_q$ and $\boldsymbol{\theta}_q$) or hyperparameters (e.g., Q_{NN}, λ), based on the calibration set $\mathcal{C}_N$. A standard method of estimation is to maximize the likelihood function. Assuming Gaussian noise, the maximum likelihood estimate of the coefficients in (4.1) solve the least squares problem

$$\min_{\alpha_0,\boldsymbol{\alpha}} \|\mathbf{y} - \alpha_0\mathbf{1} - X\boldsymbol{\alpha}\|_2^2, \tag{4.3}$$

where $\mathbf{y} = (y_1, \ldots, y_N)$ are the responses of the calibration set, $\mathbf{1}$ is the vector of ones for the intercept, the $N \times P$ matrix X is defined by $X_{np} = x_{np}$ and the ℓ_2-norm is defined by $\|\mathbf{r}\|_2^2 = \sum_{n=1}^{N} r_n^2$. The least squares estimate is linear because a matrix H (called the hat matrix) gives the prediction $\hat{\mathbf{y}} = H\mathbf{y}$, where H does not depend on $\mathbf{y}$. Moreover Gauss-Markov theorem guarantees that it is the linear unbiased estimate with minimum variance. However its large variance can prevent good prediction, for instance when the columns of X are (nearly) collinear.

A remedy is to *regularize* the maximum likelihood estimate. A widely used regularization method is *best subset variable selection*, which amounts to selecting the dimension of the model $p \in \{0, 1, \ldots, P\}$ and taking the p columns of X for which the corresponding least squares value is the smallest among all C_P^p subsets of p columns. Best subset variable selection solves the least squares problem (4.3) with the constraint of having at most p nonzero estimated coefficients. Best subset variable selection is a nonlinear estimator. The crucial ingredient of this method is the regularization parameter p: chosen too small, the bias may be too large; chosen to large, the variance may be too large. Various information criterion help for its selection: Akaike's AIC (Akaike; 1973), Mallow's C_p (Mallows; 1973) or Schwarz's BIC (Schwarz; 1978). From a computational point-of-view, selecting the best subset is a discrete optimization problem, and when the number of covariates P is too large, an approximate solution is often sought by a forward–backward search often referred as *stepwise*.

Instead, the recent *lasso* method (Tibshirani; 1996) constrains the least squares, not by the number of nonzero coefficients, but by their ℓ_1 size, by solving

$$\min_{\alpha_0,\boldsymbol{\alpha}} \|\mathbf{y} - \alpha_0\mathbf{1} - X\boldsymbol{\alpha}\|_2^2 + \lambda\|\boldsymbol{\alpha}\|_1, \tag{4.4}$$

where λ is the regularization parameter. For an optimal λ, the variance-bias trade-off is optimal. The lasso is also a nonlinear estimator. A data-driven selection of λ is $\mathrm{SL_1IC}$ (Sardy; 2006) or the computer intensive V-fold cross validation (Stone; 1974). An older regularization method is ridge regression (Hoerl and Kennard; 1970) which replaces the ℓ_1 penalty by the ℓ_2 penalty: $+\lambda\|\boldsymbol{\alpha}\|_2^2$. This quadratic penalty makes of ridge regression a linear estimator. Note that best subset variable selection can be seen as an ℓ_0 penalized least squares (Frank and Friedman; 1993). It is important to observe that, for the lasso and ridge regression, an important step of the procedure is to standardize the columns of X so that they become unitless before solving the penalized least squares problem. For instance each column of the regression matrix X can be standardized to unit standard deviation. Finally, penalized likelihood functions like (4.4) have a Bayesian interpretation, where the first part corresponds to the log-likelihood function and the second to the log-prior distribution on the coefficients $\boldsymbol{\alpha}$. Along the same vein, *support vector machine* (Cristianini and Shawe-Taylor; 2000) is another regularization method that has been applied for estimating the coefficients of the nonparametric model (4.2). Support vector machine radial basis function does not alleviate the curse of dimensionality however. Gradient Boosting is explained in (Buhlman; 2006; Buhlman and Yu; 2003).

4.2.3 Application

A low cost prediction of river quality is essential, especially in hydrosystem development projects, in order to link a candidate design or operation strategy (i.e. scenario) to a plausible environmental consequence. The Ephemeroptera (E-group), Trichoptera (T-group) and Plecoptera (P-group) faunal assemblages have often been proposed as good integrators, their richness being directly proportional to stream quality (Monaghan et al.; 2002; Tachet et al.; 2000). Our goal is to propose a methodology leading to the most accurate prediction of E-, P- and T-group richness using existing information obtained on the Swiss Upper Rhone River.

Study site

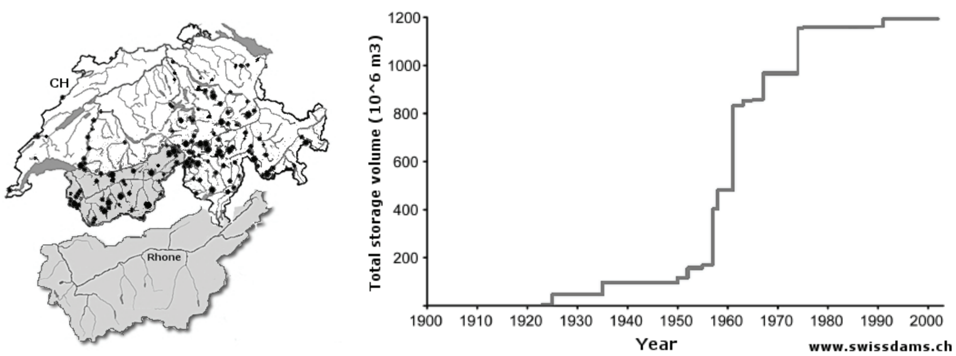

Figure 4.1: Study site: Swiss Upper Rhone River and its watershed. Evolution of the water storage capacity. Dots represent dams and reservoirs in operation. Modified from `http://www.swissdams.ch`

The Swiss Upper Rhone River is the major tributary of the Geneva Lake (68% of the total water discharge and particulate matter input). It originates at the Rhone Glacier at an altitude of 1763m and has a catchment area of 5220km^2 (Figure 4.1).

The catchment area includes 38% of rocks and glaciers, 62% of pastures, forests and agricultural lands (Loizeau and Dominik; 2000). The Rhone discharge was naturally controlled by upstream glaciers but is now subjected to important changes in its yearly flow characteristics due mainly to energy production. The water quality of the Rhone is generally good with little organic pollution (ECOTEC; 1996; DTEE; 2004). By looking at the amount of operating dams within the Swiss Upper Rhone watershed (total storage capacity over 1200e$^6 m^3$) as well as the shape of the main channel, it is evident that the hydrosystems suffers from a high hydrological and morphological anthropic impact (Figure 4.1).

Data used

Data was compiled from five databases (Table 4.2) detailed in: Gogniat et Marrer (Gogniat and Marrer; 1984/85), the ECOTEC Environmental Impact Studies reports on the Cleuson-Dixance project (ECOTEC; 1998, 1999, 2004), LIMNEX (Baumann; 2004) and R3-IBGN (GIDB-R3; 2005). Hourly discharges were obtained through OFEV. See (Pellaud; 2007) for covariate description.

4.2.4 Results

Monte Carlo simulation

To compare some important statistical models and methods of estimation discussed in Sections 4.2.2 and 4.2.2, we design a Monte Carlo simulation using the data described in Section 4.2.3. We consider the following approaches:

- Parametric models: the simplistic constant model (i.e., $f(\mathbf{x}) = \alpha_0$ in (4.1)) used as a benchmark (Method I) and the linear model (4.1) with the coefficients estimated by: least squares (Method II), stepwise (Method III), SL$_1$IC-lasso (Method IV) and Gradient Boosting with Componentwise Linear Models (Method V).

- Nonparametric models: feed forward neural network (Method VI) and Radial basis function model estimated with support vector machine (Method VII).

Table 4.2: Overview of the datasets used. $^\diamond$Five classes of substrate were retained: 1.Block ($> 200mm$) 2.Cobble (20-200mm) 3.Gravel (2-20mm) 4.Sand (0.02-2mm) 5.Fines ($< 0.02mm$).† short response hydropeaking indicator, corresponds to 90^{th} percentile of 3 hr sequence water level variations for 2 months period preceding date of measurement. ‡ yearly hydropeaking integrator, corresponds to 7997^{th} ranked 3 hr. sequence water level variation.

Observations				176
Years				1985→2005
Months				Jan-Feb-Mar-May-Jun-Nov-Dec
	Average	**Std.Dev.**	**Min.**	**Max.**
Explic. Variables				
Binary Covariates				
$x_1 - x_5$: Substrate$^\diamond$ (binary per class)	–	–	–	–
Continuous Covariates				
x_6 :HS†(m)	0.2479	0.0473	0.1524	0.4291
x_7 :Depth(m)	0.52	0.35	0.1	1.05
x_8 :Current speed $m \cdot s^{-1}$	0.52	0.34	0.01	1.39
x_9 :HL‡ (m)	0.2400	0.0165	0.22	0.2705
x_{10} :Distance from origin (km)	80.01	24.30	4	120
Continuous Response Variables				
y_E :Ephemeroptera Richness	1.6	0.7	0	4
y_P :Plecoptera Richness	2.2	1.3	0	6
y_T :Trichoptera Richness	1.6	0.9	0	4

To measure method quality, the Monte Carlo simulation randomly splits the data into a training set and a test set 5000 times. Each time the training set is used to estimate the coefficients of the model and the test set is used to calculate the squared predictive performance (SPP - $\frac{3}{2}$ cross validation (Efron and Tibshirani; 1993)). Figure 4.2 represent the box plots of the 5000 simulation results for the *Plecoptera* taxonomic group and Table 4.3 report their averaged squared predictive performance (SPP) values.

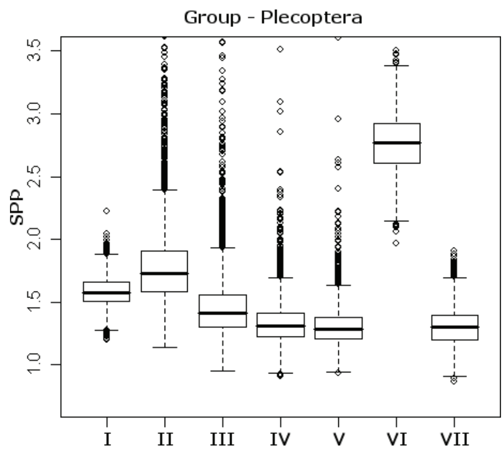

Figure 4.2: Squared Predicted Performance (SPP) of Methods $I - VII$ for P-group.

Parametric Models – Methods I - V

We see that the simplistic *constant model* (Method I) is outperformed by all methods but the Feed Forward Neural Network (Method VI) for the E-group. However, it performs better that least squares coefficient estimation (Method II) and feed forward neural networks (Method VI) for the P-group. It performed best for the T-group (Table 4.3).

Estimation by *least squares* (Method II) was always out-performed by stepwise (Method III), lasso (Method IV), gradient boosting (Method V) estimation as well as radial basis function model estimated with support vector machines (Method VII). Resulting models built on the whole dataset had ten parameters and R^2 values of respectively 0.32, 0.36 and 0.15 for E-, P- and T-groups.

Estimation by *stepwise* (Method III) were out-performed by lasso (Method IV), gradient boosting (Method V) and radial basis functions (Method VII). Resulting models built on the whole dataset had 5 parameters and a $R^2 =$

0.31 for E-group, 6 parameters and a $R^2 = 0.35$ for P-group and 3 parameters and a $R^2 = 0.12$ for the T-group.

Estimation by *lasso* and *gradient boosting* (Methods IV and V) performed best for E-group and P-group. Lasso kept 9 parameters for all groups, with $R^2 = 0.31$ for E-group, $R^2 = 0.35$ for P-group and $R^2 = 0.14$ for T-group when models were built on the whole dataset. *Gradient boosting* (Method V) kept 7 parameters for E-group and P-group with respective $R^2 = 0.30$ and 0.33. Eight parameters were kept for T-group, with a $R^2 = 0.13$ (Table 4.3).

Table 4.3: Estimator quality by averaged SPP for Ephemeroptera (E-group), Plecoptera (P-group) and Trichoptera (T-group).

Group/Method	**I**	**II**	**III**	**IV**	**V**	**VI**	**VII**
E-group	0.503	0.453	0.429	0.409	**0.405**	0.928	0.437
P-group	1.582	1.836	1.461	1.333	**1.302**	2.767	1.307
T-group	**0.759**	0.931	0.859	0.799	0.783	1.228	0.799

Non-parametric Models – Methods VI and VII

Feed forward neural network models (Method VI) always performed poorly (highest SPP). Highest R^2 values were obtained with $R^2 = 0.77$ for E-group, $R^2 = 0.78$ for P-group and $R^2 = 0.48$ for T-group on models built on the whole dataset.

Radial basis function model (Method VII) performed better than constant model (Method I) and least square estimation (Method II) for E-group. It has the second best performance for the P-group and T-group. Resulting models built on the entire dataset had a $R^2 = 0.35$ for E-group, a $R^2 = 0.39$ for P-group and a $R^2 = 0.21$ for T-group.

The best method is highlighted in bold for each group (Table 4.4).

$$\hat{y}_E^{\text{boost}} = 2.16 + 0.43x_2 + 0.02x_3 - 2.96x_6 + 0.10x_7 + 0.68x_8 - 0.01x_{10}, \quad (4.5)$$
$$\hat{y}_P^{\text{boost}} = 9.48 - 0.73x_1 - 0.38x_5 - 1.60x_6 + 0.47x_7 - 22.25x_9 - 0.02x_{10}, \quad (4.6)$$
$$\hat{y}_T^{\text{constant}} = 1.62. \quad (4.7)$$

4.2.5 Discussion

This approach provides a rapid comparison of the tested techniques (Figure 4.2). The ecologist can easily assess which techniques yields the best predictive performance. Method stability is equally displayed, with grouped points indicating a stable prediction and broadly distributed point an unstable prediction. When river development project objectives require predictive

Table 4.4: Coefficient table for Methods $II - VII$ of groups *Ephemeroptera* (E-group), *Plecoptera* (P-group) and *Trichoptera* (T-group), with associated R^2 values. Method I is not included (constant model). The best method (lower averaged SPP) is highlighted in bold.

G	$\hat{\alpha_0}$	$\hat{\alpha_1}$	$\hat{\alpha_2}$	$\hat{\alpha_3}$	$\hat{\alpha_4}$	$\hat{\alpha_5}$	$\hat{\alpha_6}$	$\hat{\alpha_7}$	$\hat{\alpha_8}$	$\hat{\alpha_9}$	$\hat{\alpha_{10}}$	R^2
						Group - Ephemeroptera (E)						
II	2.38	0.13	0.61	0.05	0.14	-0.05	-5.53	0.19	0.81	1.88	-0.01	**0.32**
III	2.87	–	0.53	–	–	–	-4.95	–	0.93	–	-0.01	**0.31**
IV	2.59	–	0.49	0.02	0.05	–	-4.32	0.14	0.79	0.09	-0.01	**0.31**
V	**2.16**	–	**0.43**	**0.02**	–	–	**-2.96**	**0.10**	**0.68**	–	**-0.01**	**0.30**
VI						*—Non Parametric Model—*						**0.77**
VII						*—Non Parametric Model—*						**0.35**
						Group - Plecoptera (P)						
II	10.9	-0.69	0.26	0.10	0.34	-0.57	-4.70	0.85	-0.01	-24.6	-0.03	**0.36**
III	10.86	-1.03	–	–	–	–	-3.76	0.76	-0.52	–	-0.03	**0.35**
IV	10.27	-0.78	0.11	–	0.19	-0.48	-3.41	0.71	–	-23.08	-0.02	**0.35**
V	**9.48**	**-0.73**	–	–	–	**-0.38**	**-1.60**	**0.47**	–	**-22.25**	**-0.02**	**0.33**
VI						*—Non Parametric Model—*						**0.76**
VII						*—Non Parametric Model—*						**0.39**
						Group - Trichoptera (T)						
II	3.98	-0.26	0.81	0.26	-0.04	0.24	1.01	0.47	-0.76	-14.4	0.002	**0.15**
III	4.90	–	0.64	–	–	–	–	–	–	-15.59	–	**0.12**
IV	4.19	–	0.60	0.08	-0.03	0.18	0.57	0.34	-0.42	-13.13	–	**0.14**
V	4.01	–	0.55	0.01	–	0.10	0.12	0.23	-0.21	-11.84	–	**0.13**
VI						*—Non Parametric Model—*						**0.45**
VII						*—Non Parametric Model—*						**0.21**

relevance, non-parametric methods such as radial basis functions estimated by support vector machines (Method VII) may be chosen. However, non-parametric methods are usually hard to interpret and should be used with caution because of the curse of dimensionality. In this particular case, it appears that Method VII performed well, with results comparable to lasso and gradient boosting coefficient estimations (Methods IV, V). Parameter role in shaping the response is not easily found via the use of non-parametric models and when objectives call for a deeper understanding of the system, it is advisable to opt for a parametric method. The lasso or gradient boosting (Methods IV, V), which were shown for groups E-group and P-group to have best SPP present the advantages of parametric models: by having a coefficient value and sign that shape the response variable. The coefficient value can be assimilated to the parameter's effect intensity and the sign the parameters' effect direction. For example, in the P-group, under the gradient boosting method (Method V) the coefficient $\hat{\alpha_9} = -22.25$ implies that parameter x_9 (short response hydropeaking indicator) has a strong negative effect on y_P (Plecoptera richness).

The constant model is our benchmark and is assumed to be the lowest limit for prediction performance acceptability by a model. By running Methods II to VII on our dataset (least square and radial basis functions), we are improving benchmark predictability by over 19.48% for the E-group and over 17.7% for the P-group with Method V (gradient boosting). The constant model (Method I) seems to perform best for the T-group, indicat-

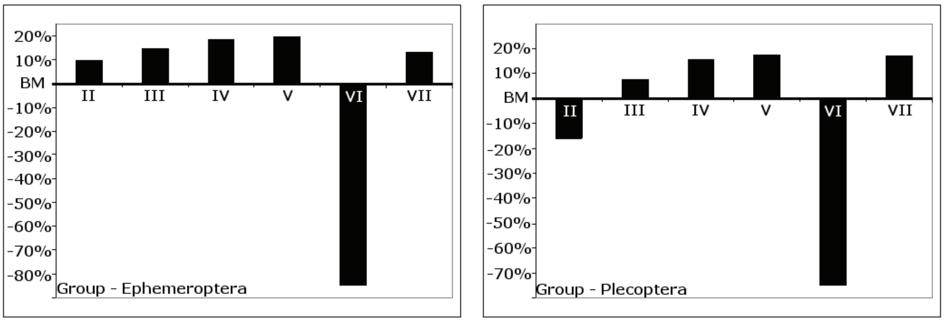

Figure 4.3: Percentages of improvement in reference to the constant model (BM) for Methods II-VII for Ephemeroptera and Plecoptera groups. Trichoptera group results are consistently under benchmark level and hence not represented.

ing that predictability of T-group richness is low and covariates have weak explanatory capability.

In order to contribute to ecological quality assessment or response to a river development project, we propose parametric models with a lasso (Method IV) or gradient boosting (Method V) estimation.

The first model (4.6) explains *ca.* 30% of river E-group richness with an improvement of *ca.* 20% of the benchmark value. The model retains 6 explicative covariates with insights on their effects on E-group richness (Table 4.5).

The second model (4.7) explains *ca.* 33% of river P-group richness with an improvement of *ca.* 18% of the benchmark value. The second model also retains 6 explicative covariates with insights on their effect on P-group richness Table 4.5). The P-group appears heavily affected by hydropeaking (high negative $\hat{\alpha_9}$ value).

The third model (4.7) is the constant function which shows that little information seems to be contained in the covariates to predict Trichoptera richness.

The very high R^2 values of the feed forward neural network ($R^2 = 0.77$ for E-group, $R^2 = 0.76$ for P-group and $R^2 = 0.45$ for the T-group) indicates a strong over-fitting of the data typical non-parametric model suffering from the curse of dimensionality. The model is too complicated and chase all the data points but has virtually no predictive power (high averaged SPP coefficient). When following this approach, it becomes clearer for ecologists to justify the choice and limit of a statistical method or design approach. For instance, in our case it is not advisable to use the Trichoptera group for variable effect understanding or group richness prediction given our dataset and modeling methods. On the other hand we can be confident that the best method were chosen to predict Ephemeroptera and Plecoptera richness and that these two groups are affected positively or negatively by the variables kept.

Ecological integration through pertinent model selection in a river development project is more clearly explicated and can pretend to contribute positively to overall ecological pertinence and ultimately to project acceptance.

Table 4.5: Variable effects on the Ephemeroptera (E) and Plecoptera (P) richness. *N.E.* stands for *No Effect*. Retained from best models (Method V).

Variable	**Effect on group E.**	**Effect on group P.**
x_1 – Blocks	*N.E.*	(–)
x_2 – Cobbles	(+)	*N.E.*
x_3 – Gravels	(+)	*N.E*
x_5 – Fines	*N.E.*	(–)
x_6 – HS	(–)	(–)
x_7 – Depth	(+)	(+)
x_8 – Current Speed	(+)	*N.E.*
x_9 – HL	*N.E.*	(–)
x_{10} – KmS	(–)	(–)

Acknowledgments

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4.3 Habitat Suitability for fish

4.3.1 Physical parameters and field methods

The fish guild habitat value estimates should be interpreted for a stream reach including several pool-riffle sequences and reach length is advised to be at least 15 times the average width (Lamouroux; 2002).

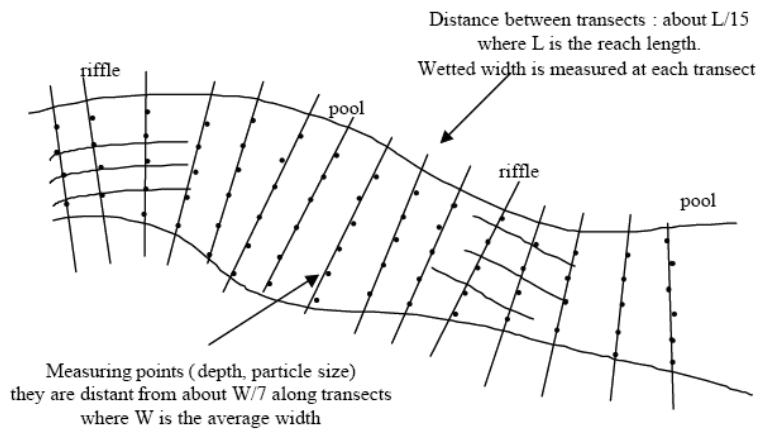


Figure 4.4: Field measurements method (as suggested from (Lamouroux; 2002))

Various measures (Figure 4.4) and/or hydraulic model outputs are used to 'feed' the model and compute the hourly habitat suitability values (HSI), namely:

1. hourly discharge Q_i in $m^3 \cdot s^{-1}$. The hourly discharge can be either a set of recorded values or predicted values coming from a hydraulic model

2. reach of interest transects. A minimum of two transects are needed to define a reach. These cross section should be representative of study's objectives and have a some height per discharge indications

3. reach averaged wetted width (W_i in m). This parameter is determined hourly by a linear or quadratic fit of W_i on Q_i

4. reach averaged water depth (Z_i in m). This parameter is determined hourly by a linear or quadratic fit of Z_i on Q_i

5. average yearly discharge ($\overline{Q_i}$ in $m^3 \cdot s^{-1}$)

6. the average size of bed particles (D in m)

4.3.2 Statistical models for predicting fish guilds habitat values indexes

Guilds were defined by Lamouroux *et al.* (Lamouroux et al.; 1999) from a cluster analysis of preference curves associated with 21 size classes of 11 species generally found in medium to large European streams and having significant microhabitat preferences. Notation is defined on Table 4.6.

- the *pool* guild (Figure 4.5) groups size classes preferring a micro-habitat that is deep, with low current velocities and a fine substrate. Hourly habitat suitability index for Pool guild (HSI_{Pi}) is determined as follow:

$$HSI_{Pi} = \left(0.026 - 0.039 * ln\left(\frac{1}{N}\sum_{i=1}^{N}\frac{Q}{g^{0.5}Z^{1.5}W}\right)_i + 0.013 * ln\left(\frac{1}{N}\sum_{i=1}^{N}\frac{Q}{10 * W}\right)_i\right) * \left[1 + 3.53 * exp\left(-11.09 * \frac{Q_i}{10 * W_i}\right)\right] + e_i$$

The annual habitat value (HV) (HV_P) of the pool guild is computed as follow:

$$HV_P = \int_1^N HSI_{Pi}\, di$$

and represents the yearly integration of pool guild hourly habitat suitability values.

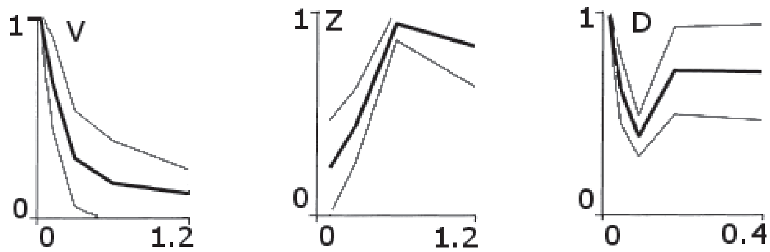


Figure 4.5: Average (+/- standard deviation) preference curves for *Pool* guild. **V** stands for current velocity (m/s), **Z** stands for water depth (m) and **D** for particle size (m). Modified from (Lamouroux and Souchon; 2002)

- the *bank* guild (Figure 4.6) groups size classes preferring a microhabitat that is shallow, with low current velocities and fine sediment. The hourly habitat suitability index for bank guild (HSI_{Bi}) is determined as follow:

$$HSI_{Bi} = \left(0.103 - 0.010 * ln\left(\frac{1}{N}\sum_{i=1}^{N}\frac{Q}{g^{0.5}Z^{1.5}W}\right)_i\right) * \left[1 + 4.17 * exp\left(-23.61 * \frac{Q_i}{10 * W_i}\right)\right] + e_i$$

Similarily, the yearly habitat value index of the bank guild (HV_B) is computed as follow:

$$HV_B = \int_1^N HSI_{Bi}\, di$$

and represents the yearly integration of bank guild hourly habitat suitability values.

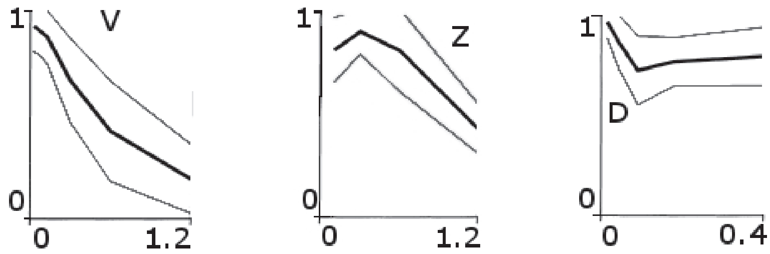


Figure 4.6: Average (+/- standard deviation) preference curves for *Bank* guild. **V** stands for current velocity (m/s), **Z** stands for water depth (m) and **D** for particle size (m). Modified from (Lamouroux and Souchon; 2002)

- the *riffle* (Figure 4.7) guild groups size classes preferring a microhabitat that is shallow, with intermediate to high current velocities and intermediate substrate size. The hourly habitat suitability index for riffle guild (HSI_{Ri}) is determined as follow:

$$HSI_{Ri} = \left(1.074 + 0.281 * ln\left(\frac{1}{N}\sum_{i=1}^{N}\frac{Q}{g^{0.5}Z^{1.5}W}\right)_i + 0.300 * \frac{D}{\overline{Z}}\right) * \left[\left(\frac{Q_i}{10 * W_i}\right)^{0.09} * exp\left(-15.13 * \frac{10}{Q_i * W_i}\right)\right] + e_i$$

  Similarly, the annual riffle guild habitat value index (HV_R) is determined as follow:

$$HV_R = \int_1^N HSI_{Ri}\, di$$

  and represents the yearly integration of the hourly riffle guild habitat suitability index values.

- the *midstream* (Figure 4.8) guild groups size classes preferring microhabitat that is deep and fast flowing, with a coarse substrate. The hourly habitat suitability index for midstream guild (HSI_{Mi}) is determined as follow:

$$HSI_{Mi} = \left(1.352 + 0.713 * \frac{D}{\overline{Z}} + 0.160 * ln\left(\frac{1}{N}\sum_{i=1}^{N}\frac{Q}{10 * W}\right)_i\right) \left[\left(\frac{Q_i}{10 * W_i}\right)^{0.32} * exp\left(-2.87 * \frac{Q_i}{10 * W_i}\right)\right] + e_i$$

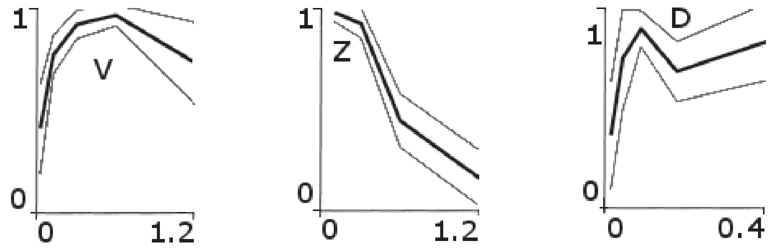


Figure 4.7: Average (+/- standard deviation) preference curves for *Riffle* guild. **V** stands for current velocity (m/s), **Z** stands for water depth (m) and **D** for particle size (m). Modified from (Lamouroux and Souchon; 2002)

Similarly, the yearly midstream guild habitat value index (HV_M) is determined as follow:

$$HV_M = \int_1^N HSI_{Mi}\, di$$

and represents the yearly integration of the hourly midstream guild habitat suitability index values.

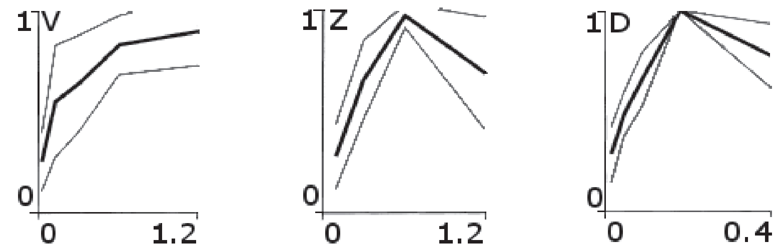


Figure 4.8: Average (+/- standard deviation) preference curves for *Midstream* guild. **V** stands for current velocity (m/s), **Z** stands for water depth (m) and **D** for particle size (m). Modified from (Lamouroux and Souchon; 2002)

The models habitat suitability models were used in New Zealand and were found to be pertinent out of their validation context (Lamouroux and Jowett; 2005).

Table 4.6: Notation used for fish guilds habitat suitability index values and habitat values

Notation	Semantic and unit
g	gravitational acceleration 9.81 ($m \cdot s^{-2}$)
Q_i	water discharge at time i ($m^3 \cdot s^{-1}$)
Z_i	reach-averaged water depth at time i (m)
W_i	reach-averaged wetted width (m)
D	reach-averaged bed particle size (m)
$\overline{H}$	water depth at average yearly water discharge (m)
N	hour number (1 yr. being 8760 hours)

4.4 Hydropeaking and EPT

Fuzzy adjustment of macro-invertebrate groups richness prediction models based on hydropeaking: the importance of river morphology

4.4.1 Fuzzy logic

Fuzzy logic (FL) is the codification of common sense (Zadeh; 1995). It is tolerant of imprecise data, can model highly complex nonlinear functions, be built based on expert knowledge, be blended with conventional techniques and has a shared basis with human language (The MathWorks; 2004; Zimmermann; 1985). The book by Zimmerman is probably the most referred to in FL (Zimmermann; 1985). A classical set can be defined as a container that either includes or excludes an element. In FL, the truth of any statement becomes a matter of degree, through a *membership function* (MF). MF are curves defining how each point in the input space are mapped to a degree of membership ranging from 0 to 1. If X is the input space and its elements are denoted by x, then a fuzzy set A in X can be defined as a set of ordered pairs.

$$A = \{x, \mu_A(x) \,|xX\}$$

$\mu_A(x)$ is the MF of x in A and maps each element of X to a membership value between 0 and 1. FL can be considered a superset of Boolean logic, if fuzzy values are kept at 0 (completely false) or 1 (completely true), standard logical operations will hold. The A *AND* B operation can be solved by the *min(A,B) function*, the A *OR* B operation can be resolved by *max(A,B) function* and the standard operation *NOT A* becomes equivalent to the operation *1-A*. Since there is an underlying function values other than 0 and 1 can be considered for operations. A fuzzy *if-then* rule assumes the form

$$\text{if } x \text{ is } A \text{ then } y \text{ is } B$$

where A (antecedent) and B (output fuzzy set) are linguistic values defined by the fuzzy sets on the ranges X and Y.

We use a combination of Mamdani-type (Mamdani; 1977) Fuzzy Inference Systems (FIS) to address hydropeaking. This particular type of inference system expects the output function to be a fuzzy set and finds the centroid which yields a *defuzzifed* value (crisp output).

4.4.2 Hydropeaking assessment

The hydropeaking assessment is composed of two FIS (Figure 4.9). The first FIS can be seen as the reach-dependent FIS and represents the *weight* of hydropeaking events on the system (Figures 4.9 (A.), 4.10). If the system has little morphological ecological potential, hydropeaking effect will be negligible on it, while on the other hand if the system has a good ecological potential, hydropeaking is very susceptible of affecting it (system sensitive to hydrology).

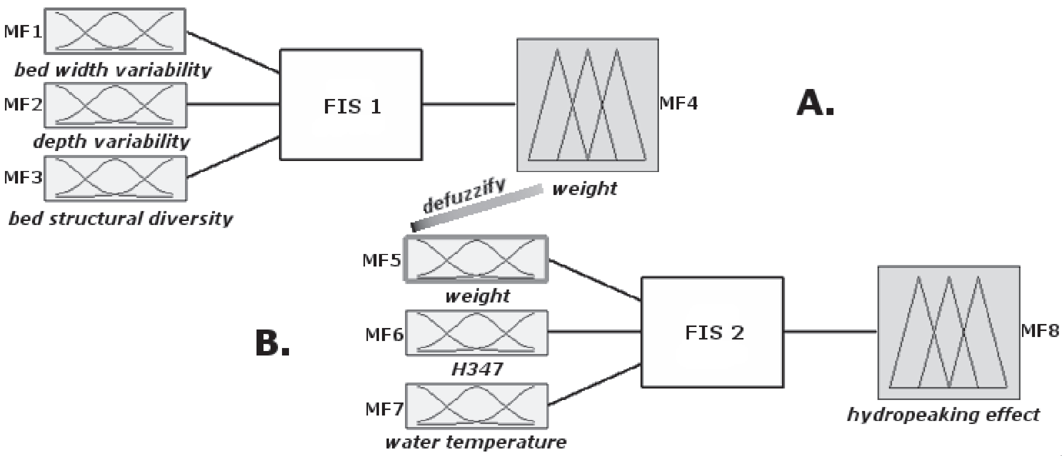


Figure 4.9: Fuzzy Inference Systems (FIS) assessing the hydropeaking effect: MF stands for *Membership Function*. **A.** details the *weigth FIS* and **B.** the *hydropeaking effect FIS*.

The *weight FIS* has all its *low* MF (μ^1_{Low}) characterized by spline-based functions of x, with parameters a and b indicating the lower and upper extremes of the sloped portions of the curve as given by (The MathWorks;

2004):

$$\left\{\begin{array}{rl} 1, & x \leq a \\ 1-2\left(\frac{x-a}{b-a}\right)^2, & a \leq x \leq \frac{a+b}{2} \\ 2\left(b-\frac{x}{b-a}\right)^2, & \frac{a+b}{2} \leq x \leq b \\ 0, & x \geq b \end{array}\right\} \tag{4.8}$$

The *weight FIS* has a *medium* MF (μ^1_{Medium}) characterized by a symmetric Gaussian function depending on the parameters σ and c as given by (The MathWorks; 2004):

$$\mu^1_{\text{Medium}}\left(x;\sigma,c\right) = e^{\frac{-(x-c)^2}{2\sigma^2}} \tag{4.9}$$

The *weight FIS* has all its *high* MF (μ^1_{High}) characterized by sigmoidal functions of x depending on parameters a and c as given below by (The MathWorks; 2004):

$$\mu^1_{High}\left(x;a,c\right) = \frac{1}{1+e^{-a(x-c)}} \tag{4.10}$$

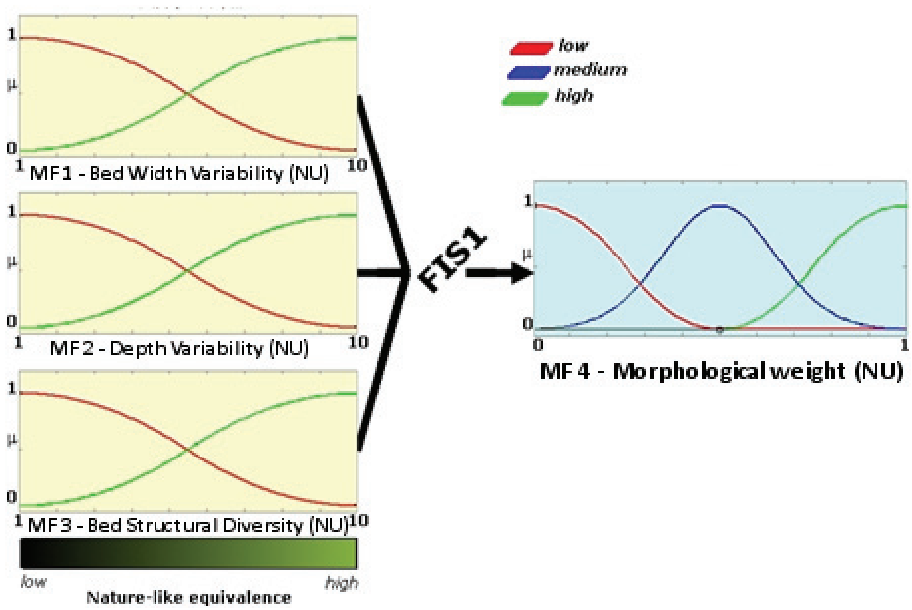


Figure 4.10: **FIS1** *weigth.* Each input (yellow) membership function contribute to the linguistic appreciation of *nature-like* morphological conditions

The *weight FIS* is implemented based on the following set of linguistic variables:

MF1 - bed width variability (*low* and *high*):

- *Semantics*: linguistic characterization of the bed width variations

- *Ecological implication*: Bed width variation provides an insight on the ecological potential of the river. A monotonous bed width is often an indicator of a channelized river, which causes an array of negative effects on the system ecological integrity (see p 41 for effects), limiting its potential for improvement based on a hydrological stand

- *Variable definition*: An input of 1 has a full membership of *low* bed width variability and is representative of trained channels. An input of 10 represent a full membership of the *high* bed width variability function, such as typically found in natural reaches (Figure 4.10). For the *low* MF, parameters $a = 1$ and $b = 10$. For the *high* MF, parameters are $a = 1$ and $c = 5.45$

MF2 - depth variability (*low* and *high*):

- *Semantics*: linguistic characterization of the reach depth variability

- *Ecological implication*: Bed structure provides an insight on the ecological potential of the river. A bed lacking depth variability is typical of an altered system. A monotonous bed is a limit to ecological recovery potential based on hydrological improvement (see p 41 for effects)

- *Variable definition*: An input of 1 has a full membership of *low* depth variability and is representative of altered channels. An input of 10 represents a full membership of the *high* depth variability function, such as typically found in natural reaches (Figure 4.10). For the *low* MF, parameters $a = 1$ and $b = 10$. For the *high* MF, parameters are $a = 1$ and $c = 5.45$

MF3 - bed structural diversity (*low* and *high*):

- *Semantics*: linguistic characterization of the presence of structures (i.e. large woody debris, roots, boulders, underbanks...) in the wetted bed

- *Ecological implication*: a bed lacking structural diversity is typical of an altered system. Without structures, many organisms are unable to find a shelter or a colonizing ground, therefore greatly limiting the potential of hydrology as a factor of ecological improvement in such a system

- *Variable definition*: an input value of 1 accounts for a strictly monotonous bed while an input value of 10 accounts for a highly structured system, with debris, shelters, boulders and all sorts of structural elements that can typically be found in natural systems (Figure 4.10). For the *low* MF, parameters $a = 1$ and $b = 10$. For the *high* MF, parameters are $a = 1$ and $c = 5.45$

MF4 - weight (*low*, *medium* and *high*):

- *Semantics*: overall fuzzy integrator of the morphological component adjustment to hydropeaking (Figure 4.10) ranging from 0 to 1

- *Ecological implication*: this variable represents the overall system sensibility to hydrology. A system showing a variable, structured and diverse morphology has a greater response ability to hydrology than a system limited by its morphology

- *Variable definition*: for the *low* MF, parameters $a = 0$ and $b = 0.5$, for the *medium* MF, parameters $\sigma = 0.15$ and $c = 0.5$ and for the *high* MF, parameters $a = 21$ and $c = 0.75$

The second FIS (Figure 4.11) characterizes the hydropeaking event and yields the *hydropeaking effect* on macro-invertebrate output, taking into consideration both the river ecological sensitivity to hydrology (fuzzy output) and hydrological properties integrating hydropeaking (Figure 4.9 (B.)).

The *hydropeaking FIS* has all its *low* MF (μ^2_{Low}) characterized by spline-based functions of x, with parameters a and b indicating the lower and upper extremes of the sloped portions of the curve as given by equation (4.8)

The *hydropeaking FIS* has a *medium* (μ^2_{Medium}) MF characterized by a generalized bell function depending on parameters a, b and c as given by (The MathWorks; 2004):

$$\mu^2_{\text{Medium}}(x; a, b, c) = \frac{1}{1 + \left|\frac{x-c}{a}\right|^{2b}} \tag{4.11}$$

The *hydropeaking FIS* has all its *high* MF (μ^2_{High}) characterized by sigmoidal functions of x depending on parameters a and c as described in equation (4.10)

MF5 - defuzzified weight output:

- *Semantics*: the extent to which hydrology is susceptible of impacting the system.

- *Ecological implication*: potential for ecological integrity based on hydrology. Depends on system morphological naturalness, if it is low, hydrology has a limited effect on ecological integrity, if it is high, hydrology has a high potential as an impacting factor

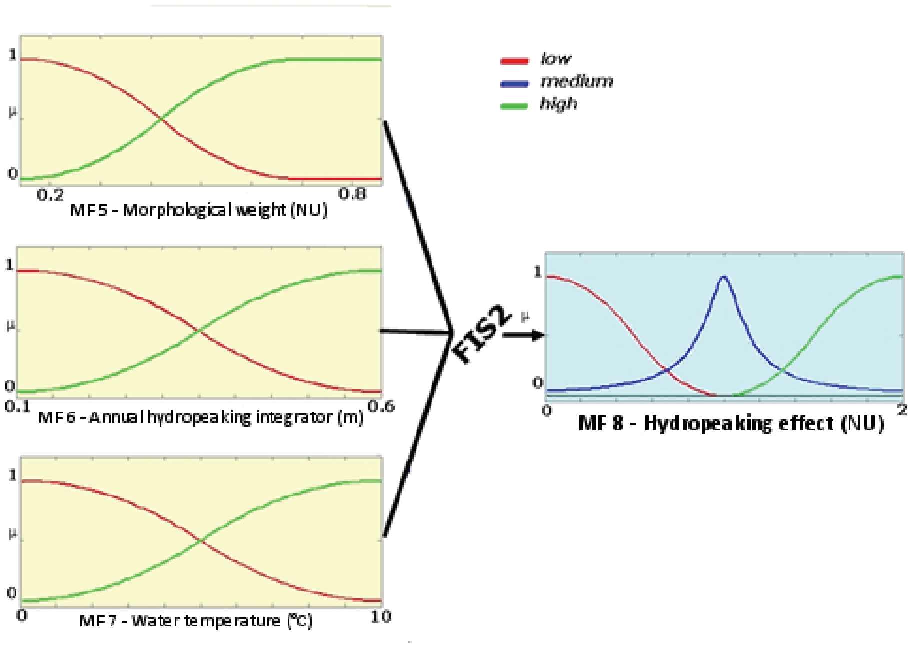


Figure 4.11: **FIS2** *hydropeaking effect*. Input membership functions (MF) are in yellow and output MF in blue. Linguistic assessment of the hydropeaking effect on macro-invertebrates

- *Variable definition*: input value has to undergo defuzzyfication to get 'crisp' estimation (as required by Mamdani-type FIS (Mamdani; 1977)). Defuzzified *weigth* range is determined following lower and upper limit defuzzfied value of MF4). A value of ≤ 0.14 corresponds to a membership of 1 to the *low* MF. A value of ≥ 0.857 corresponds to a membership of 1 to the *high* MF. For the *low* MF, parameters $a = 0.14$ and $b = 0.7$. For the *high* MF, parameters $a = 18$ and $c = 0.42$

MF6 - HL **factor**:

- *Semantics*: annual simulation of 3-hours-sequence water level variations are ranked. This factor corresponds to the 7997^{th} sequence, representing the 333^{th} day threshold of non-exceedence. This factor has to be compared to what can be considered a *natural* or *acceptable* value which is never null. Hourly discharge of 'La Porte du Scex' gouging station were available for year 1907 (Meile et al.; 2005), which was assumed as hydrological reference year. Year 1907 HL ($HL_{\text{Ref}} = 0.0835$) was therefore subsequently withdrawn from HL factor in order not to overestimate it

- *Ecological implication*: this factor serves as one possible yearly integrator of hydropeaking extent. Exceptional high sequences are segregated from the others, which are assumed to shape the biotic communities structures.

- *Variable definition*: indicator ranges from 0.1 m (membership of 1 to *low* water level variations MF) to 0.6 m (membership of 1 to *high* water level variations MF). For the *low* MF, parameters $a = 0.1$ and $b = 0.6$. For the *high* MF, parameters $a = 18$ and $c = 0.35$

MF7 - hourly water temperature:

- *Semantics*: hourly water temperatures

- *Ecological implication*: hydropeaking effect on river organisms is enhanced by low water temperatures (see p. 43)

- *Variable definition*: hourly water temperature of 0^oC has a membership value of 1 to the *low* MF, while a hourly water temperature of 10^oC has a membership value of 1 to the *high* MF. For the *low* MF, parameters $a = 0$ and $b = 10$. For the *high* MF, parameters $a = 0.9$ and $c = 5$

MF8 - hydropeaking effect:

- *Semantics*: fuzzy hourly output of hydropeaking effect ranging from [0 2].

- *Ecological implication*: hydrological adjustment based on the fuzzy appreciation of the role of hydrology in shaping communities richness. In a natural and pristine context, hydrology is a main factor governing system ecological integrity while in a heavily altered system, hydrology has a weaker overall community shaping capability

- Variable description: for the *low* MF, parameters $a = 0$ and $b = 1$. For the *medium* MF, parameters $a = 0.146$, $b = 0.84$ and $c = 1$. For the *high* MF, parameters $a = 9$ and $c = 1.5$

4.4.3 *FIS* rules

Each *FIS* requires its own set of rules. For the *weight* FIS: a total of eight rules were stated following the basic principle that a natural morphology is more sensitive to hydropeaking (high weight) than an altered morphology (low weight) (Table 4.7).

Table 4.7: *weight* fuzzy inference system (FIS) rules: Inputs in forms of membership function (MF): MF1(bed width variability), MF2 (depth variability) and MF3 (bed structure diversity). Resulting output is MF4 (weight)

Rule #	**MF1**	**MF2**	**MF3**	**MF4**
1	low	low	low	**low**
2	low	low	high	**med**
3	low	high	low	**med**
4	high	low	low	**low**
5	high	high	low	**high**
6	high	low	high	**high**
7	low	high	high	**high**
8	high	high	high	**high**

For the *hydropeaking* FIS (Table 4.8), a total of eight rules were stated. Overall hydropeaking effect (MF8) increased with river sensibility to hydrology (MF5 - weight), a high HL (MF6) factor and low water temperatures (MF7).

Table 4.8: *hydropeaking* FIS rules: Inputs are MF5 (weight), MF6 (HL) and MF7 (Temperature). Resulting output is MF8 (hydropeaking effect)

Rule #	**MF5**	**MF6**	**MF7**	**MF8**
1	low	low	low	**med**
2	low	low	high	**med**
3	low	high	low	**low**
4	low	highw	high	**med**
5	high	low	low	**high**
6	high	low	high	**high**
7	high	high	low	**low**
8	high	high	high	**low**

4.4.4 *FIS* implementation

Step 1 of implementation - Fuzzyfication of the input variables

The degree of membership of each input has to be determined generically for the *weight* FIS and hourly for the *hydropeaking effect* FIS. Each input is a crisp numerical value delimited to the input range and the output is a fuzzy degree of membership. For example, to what extent is the bed variable? The figure below shows how bed variation of our hypothetical river (rated on a scale of 1 to 10) qualifies via its MF as the linguistic variable *very low but not quite null* (Figure 4.12). In this case, the bed variability was rated as a 2, which corresponds to $\mu = 1$ for the *low* MF. All the inputs are fuzzified in a similar way.

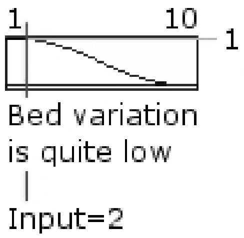


Figure 4.12: Example of fuzzification of the reach bed variation input variable

Step 2 of implementation - Application of the fuzzy operator in the antecedent

We now know the degree to which each part of the antecedent has been reached for each rule. A fuzzy operator is applied to all parts of antecedent in order to obtain one number representing the result of the antecedent for each rule. This number will then be applied to the output function. The inputs to the fuzzy operator are all the membership values from fuzzified input variables and the output will be a single truth value. We follow with our example:

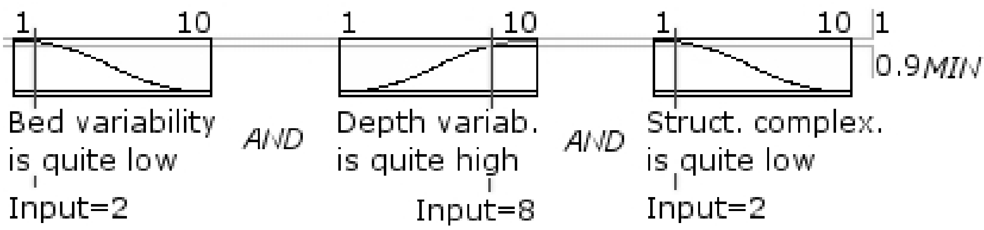


Figure 4.13: Example of fuzzy *AND* operator application, the *MIN* value is kept (0.9)

In this particular case, the fuzzy *AND* operator simply selects the minimum of the three values (*1, 1 and 0.9*), 0.9, and the fuzzy operation for this rule is complete.

Step 3 of implementation - Implication from the antecedent to the consequent

The *consequent* is a fuzzy set represented by a MF, which weights appropriately the linguistic characteristics attributed to it. The input is a single number given by the antecedent and the output is a fuzzy set. Implication has to be implemented for each rule (Figure 4.14).

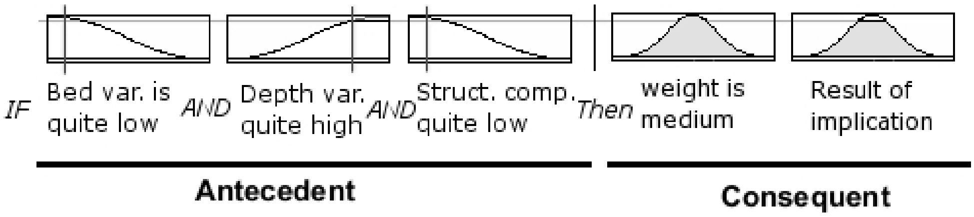


Figure 4.14: Implication method - process automated for each rule

Step 4 of implementation - Aggregation of the consequent across the rules

In a FIS, the final fuzzy set output is based on the testing of all stated rules. The fuzzy sets representing the outputs of each rules are combined into a single fuzzy set through the process of *aggregation*. The input is the list of truncated output fuzzy sets obtained by the implication process for each rule (Figure 4.15)

Step 5 of implementation - Defuzzyfication

The input is the aggregate output fuzzy set and the output is a single crisp number. One of the most commonly used defuzzyfication method is the center of gravity (COG) calculation, which returns the center of area under the curve (Figure 4.16).

The hourly *hydropeaking effect* FIS is implemented in a similar manner

4.4.5 Macro-invertebrate model aggregation

The aggregation of the model consists in the sum of the averaged hydropeaking adjusted group richness. Not all group react similarly to hydropeaking hence the best *Raw* richness prediction model ($\widehat{y}^{*}_{E,P,T}$) undergoes an adjustment with the hydropeaking FIS output ($hydropeakingFIS$) based on the estimation of the ecological response of each group to hydropeaking.

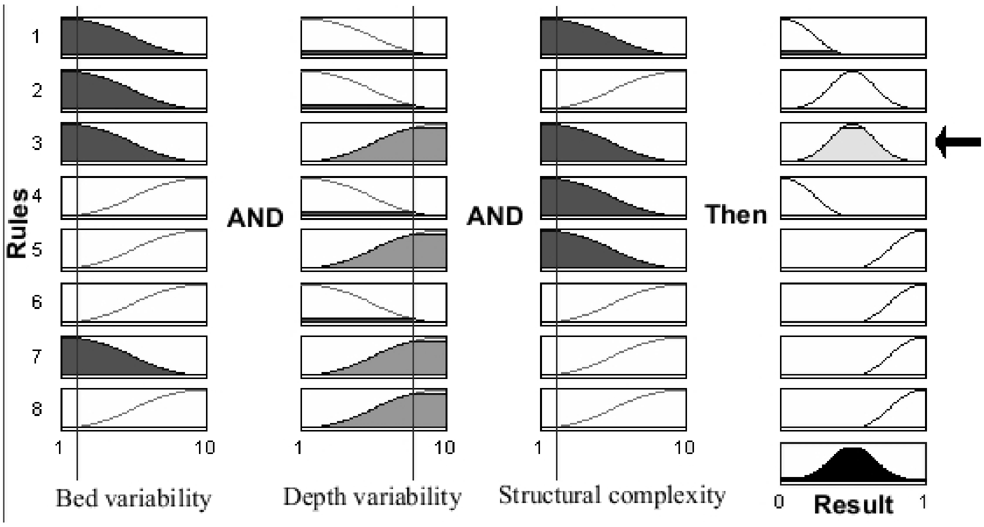


Figure 4.15: Aggregation process - the arrow indicates the rule used in the example

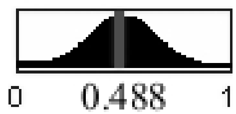


Figure 4.16: Center of gravity (COG) defuzzification yielding the crisp result of the *weight* FIS with inputs of [2 8 2] for *Bed variability*, *Depth variability* and *Structural complexity*

Mayflies (Ephemeroptera) are represented in the Swiss Upper Rhone River mainly by the *Heptageniidae* and *Baetidae* families (Baumann; 2004; Bernard; 2001; GIDB-R3; 2005; R3; 1998).

- *Heptageniidae* - affectionate high gradients. They are very sensitive to anthropic alteration (Bernard; 2001) and particularly hydropeaking. Bad water quality has a negative influence on mayflies richness but should only be a problem near water treatment plants water releases. However, sensitive family, with a score of 10 in the BMWP score Table (Armitage et al.; 1983) and 5 in the IBGN GI value (AFNOR; 1985)

- *Baetidae* - also appear to be very impacted by hydropeaking (Céréghino and Lavandier; 1998a) and to a lesser extent other anthropic alterations. Score of 4 in the BMWP score Table (Armitage et al.; 1983) and 2 in the IBGN GI value (AFNOR; 1985)

Hence hydropeaking is considered to affect deeply this group. Ephemeroptera richness is estimated as follow:

$$\widehat{R}_E = \frac{1}{N}\sum_{1}^{N}\left(\widehat{y}_E^{boost} * hydropeakingFIS\right)$$

Stoneflies (Plecoptera) families reported in the Swiss Upper Rhone River are:

- *Perlidae* - rarely encountered in the Swiss Upper Rhone River, this family is believed to be very sensitive to hydropeaking and organic pollution (score of 10 in BMWP score table (Armitage et al.; 1983), and 9 in IBGN GI value (AFNOR; 1985))

- *Leuctridae* - affected by hydropeaking but quite exigent family in terms of water quality (Bernard; 2001). Family very sensitive to organic pollution (score of 10 in BMWP score table (Armitage et al.; 1983) and 7 in IBGN GI value (AFNOR; 1985))

- *Chloroperlidae* - family appears dependent on slope and vegetation, hence somehow indirectly affected by hydropeaking (score of 9 in IBGN GI value (AFNOR; 1985))

- *Perlodidae* - this family is sensible to bad water quality and to some extent to hydropeaking. Family very sensitive to organic pollution (score of 10 in BMWP score table (Armitage et al.; 1983) and 9 in IBGN GI value (AFNOR; 1985))

- *Taeniopterygidae* - can be found in abundance in autumn. This family is sensitive to hydropeaking and bad water quality. Family very sensitive to organic pollution (score of 10 in BMWP score table (Armitage et al.; 1983) and 9 in IBGN GI value (AFNOR; 1985))

- *Capniidae* - their abundance seems to be in correlation with river width and seem not to be too affected by hydropeaking and bad water quality (Bernard; 2001) (score of 8 in IBGN GI index (AFNOR; 1985))

- *Nemouridae* - affectionate litter, hence are not very developed in the main stream. Very sensitive to hydropeaking and anthropic alteration (score of 6 in IBGN GI value (AFNOR; 1985))

The main interest of Plecoptera to humans is as an indicator taxa, they are usually associated with unpolluted waters. Plecoptera richness is considered heavily affected by hydropeaking (Céréghino and Lavandier; 1998a) and hence is estimated as follow:

$$\widehat{R}_P = \frac{1}{N}\sum_{1}^{N}\left(\frac{2}{7}\widehat{y}_P^{boost} + \frac{5}{7}\widehat{y}_P^{boost} * hydropeakingFIS\right)$$

Caddisflies (Trichoptera) families reported in the Swiss Upper Rhone River are:

- *Rhyacophilidae* - they seem no to be affected by hydropeaking. Only their abundance seems correlated with overall diversity (Bernard; 2001). Score of 7 on the BMWP score Table (Armitage et al.; 1983) and score of 4 in IBGN GI value (AFNOR; 1985)

- *Limnephilidae* - they are not likely to be affected by hydropeaking, bed armoring or anthropic impact. They show a very high abundance (Bernard; 2001). Score of 7 on the BMWP score Table (Armitage et al.; 1983) and 3 in IBGN GI value AFNOR (1985)

- *Hydropsychidae* - not very sensitive to hydropeaking or other anthropic alterations (Nugent et al.; 2002). Score of 5 on the BMWP score Table (Armitage et al.; 1983) and of 3 in IBGN GI value (AFNOR; 1985)

Trichoptera richness is estimated as follow:

$$\widehat{R}_T = \frac{1}{N}\sum_{1}^{N}\widehat{y}_T^{constant}$$

The estimation of the Ephemeroptera, Plecoptera and Trichoptera Richness Index is determined as follow:

$$\widehat{R}_{EPT} = \widehat{R}_E + \widehat{R}_P + \widehat{R}_T$$

4.5 Conclusion

The final R_{EPT} bioindicator can give us an insight on the ecological consequences resulting from a engineering or management scenario and their effect on hydrology and morphology. However, this type of indicator should not:

- be used in a generic manner. The model implementation and selection process has to be specific to each river system and its architecture and variables can differ from the one presented

- be used in systems having contaminated waters since the water quality aspect is not addressed in the model. Water quality is assumed not to be a limiting factor

- under no circumstances be taken as absolute richness values but rather as indicator of the relative ecological integrity based on hydrological and morphological issues

The Baetidae family (Plecoptera) and Limnephilidae (Trichoptera) are very heterogenous families and their ecological interpretation should be linked to species composition of each of these two families (Castella, pers. comm.). The family taxonomic level may hide diversity. The use of such a model is recommended as a tool for decision making in river development projects and can be easily integrated in multi-purpose project optimizer such as the one presented in Heller's work (Heller; 2007) but has to be employed with care.

Literature review of hydropeaking effect on fish revealed that fish are impacted by hydropeaking events. Unfortunately, the extent of the impact seems highly variable and in her PhD thesis, Valentin (1995) noted that young fish were most vulnerable to stranding, especially when dewatering took place at night in cold waters. However, she was also able to observe that fish had a hydropeaking *learning* ability, adding a behavioral component extremely hard to assess in the quantification of a hydropeaking event on fish guilds. Therefore in the scope of this work hydropeaking effect will not be assessed further than by the integration of resulting hourly habitat suitability index that are subject to the *physical* current speed and depth variations. To my sense, this assessment has strong limits to the applicability of the use of fish guilds as hydropeaking indicators in the current state of knowledge and further research on that behavioral aspect should be undertaken.

Chapter 5

Ecological responses of four project scenarios

5.1 Selected models and tested scenarios

In Chapter 4, seven models were tested and for each taxonomical group (E,P and T), the best model were kept as follow:

Ephemeroptera taxonomical group

$$\widehat{R}_E = \frac{1}{N}\sum_{1}^{N}\left(\widehat{y}_E^{boost} * hydropeakingFIS\right)$$

Plecoptera taxonomical group

$$\widehat{R}_P = \frac{1}{N}\sum_{1}^{N}\left(\frac{2}{7}\widehat{y}_P^{boost} + \frac{5}{7}\widehat{y}_P^{boost} * hydropeakingFIS\right)$$

Trichoptera taxonomical group

$$\widehat{R}_T = \frac{1}{N}\sum_{1}^{N}\widehat{y}_T^{constant}$$

The final model aggregation being

$$\widehat{R}_{EPT} = \widehat{R}_E + \widehat{R}_P + \widehat{R}_T$$

The E,P and T groups richness prediction index (R_{EPT}) as well as the four fish guilds habitat values are tested on the downstream river following:

1. an *Actual / no project* variant (scenario 1) – the morphological aspects of the river remain as today's and are coupled to year 1999 hydrogram at the Branson Gouging Station. Using Manning-Strickler equations (FlowMaster[1] software) the link between discharge and current speed, depth and width are established using a weighted Manning coefficient of 0.040 and a channel slope of 0.2659% as given in (LCH; 2005)

2. an hypothetical *bed widening* project (similarities to the project proposed by the third correction of the Rhone River) – the river bed is widened in a hypothetical yet comparable way to that presented by the third correction of the Rhone project (GIDB-R3; 2005). As in scenario 1, year 1999 hydrogram of the Branson Gouging Station is coupled to a widened cross section. Using Manning-Strickler equations (FlowMaster software) with a weighted Manning coefficient of 0.040 and a slope of 0.2659%, the links between discharge and current speed, depth and width are established and hydropeaking integrators are obtained

3. an hydropeaking *buffer basin* such as the one proposed by the `SYNERGIE` project (scenario 3) – the downstream morphology of the Rhone remains as today's and a simulated hydrogram and water temperatures given by Heller (2007) are used to feed the model. Scenario 1's relations of depth vs. discharge, current speed vs. discharge and width vs. discharge are holding

4. the coupling of a hydropeaking mitigation basin to a structural widening of the river bed (scenario 4) – the downstream morphology of scenario 2 holds, together with the simulated hydrogram and water temperature by Heller (2007). Scenario 2 relations of depth vs. discharge, current speed vs. discharge and width vs. discharge are holding

Only the ecological consequences are detailed in this chapter. For insights on energy production, flood safety and leisure potential refer to Heller's work (2007).

5.2 Scenario 1 - Current state

5.2.1 Scenario overview

The Riddes site (Figure 5.2.1) as it looks currently. In December 2000, the Celeuson-Dixance steel-lined shaft (equipped discharge of $75 m^3 \cdot s^{-1}$ and a

[1] http://www.bentley.com/en-US/Products/FlowMaster

maximal production power of 1200 MW in the Bieudron facility[2]) ruptured. The shaft is to be repaired at the end of 2009 and 1999 was the only year of complete functioning of this pipe. Year 1999 is the closest hydrological year to post-reparation conditions (i.e. closest to year 2010 hydrogram). Taken into account any other year would probably under-estimate future hydropeaking conditions and the extra $75m^3 \cdot s^{-1}$ peak releases. This is why year 1999 was chosen as a representative year for no-project / actual state scenario. The river has little interaction with its floodplain, embankment is severe and has not change since its second correction (see page 61). Hydropeaking is significant, especially in winter (see Figures 3.5 p. 64). Detailed description of the scenario 1 situation and covariates used to feed the models is depicted in Figure 5.2.1 and Table 5.1

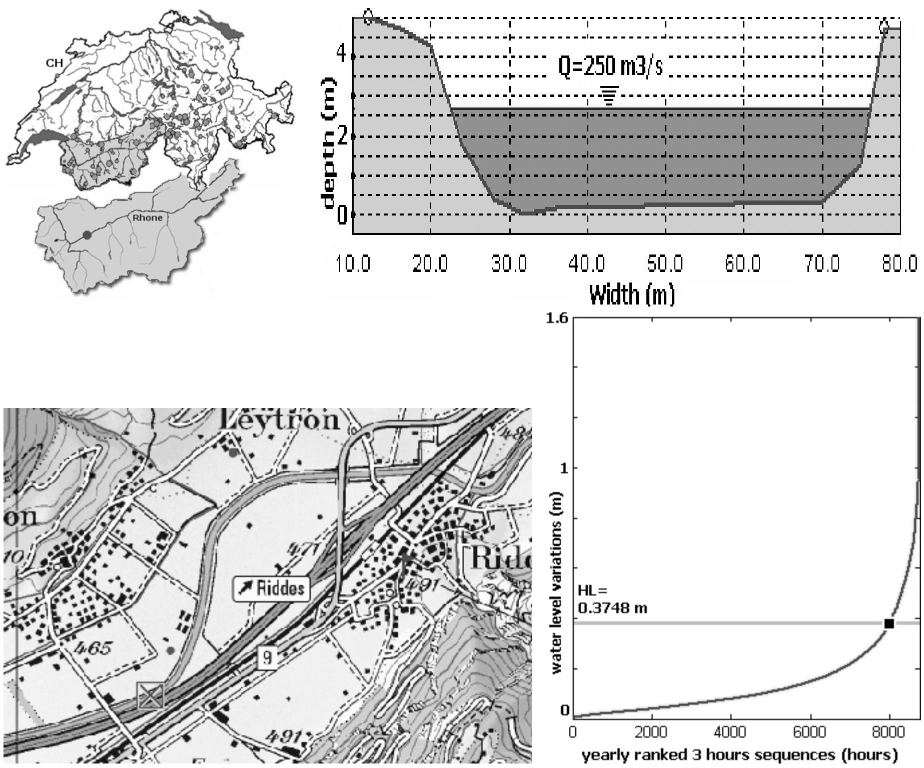
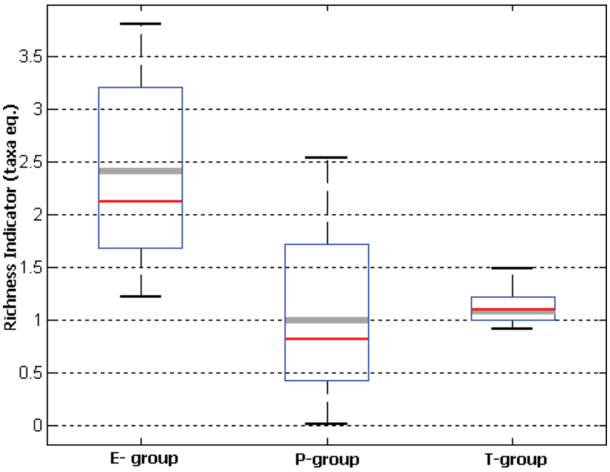


Figure 5.1: Scenario 1 - Current state overview: Site of Riddes (VS-CH) (map from `www.swissgeo.ch`) 1:50'000. Cross section modified from (LCH; 2005), HL factor for year 1999 discharges

[2]http://www.cleuson-dixence.ch/dossier_rupture_pb_avril02-3.pdf

Table 5.1: Scenario 1 – Covariate description:[a]based on annual hourly (i) Depth (Z) vs. discharge (Q) relation of $Z_i = -9e^{-6}Q_i^2 + 0.0111Q_i + 0.4339$ ($R^2 = 0.98$).[b]based on annual hourly speed (V) vs. discharge (Q) relation $V_i = -9e^{-6}Q_i^2 + 0.009Q_i + 0.5$ ($R^2 = 0.95$).[c]based on annual top width (W) vs. discharge (Q) relation $W_i = -4e^{-5}Q^2 + 0.0426Q + 45.794$ ($R^2 = 0.96$)

	Variable	Source	Value	Unit	Mean	Min.	Max.	Stdev.
$x_1 - x_5$	Substrate	Limnex (Baumann; 2004)	Cobbles (x_2) and gravels (x_3)	binary	–	–	–	–
x_6	HS	Branson gouging station (1999)[a]	$f(Q_i)$	$[m]$	0.2757	0.1288	0.4264	0.1182
x_7	Depth	Branson gouging station (1999)[a]	$f(Q_i)$	$[m]$	1.9	0.7	3.8	0.8
x_8	Current Speed	Branson gouging station (1999)[b]	$f(Q_i)$	$[m/s]$	1.6	0.7	2.7	0.6
x_9	HL	Branson gouging station (1999)[a]	0.3748	$[m]$	–	–	–	–
x_{10}	Distance from origin	LCH report (LCH; 2005)	109	$[km]$	–	–	–	–
x_{11}	Water temperature	from OFEV	$f(i)$	$[^oC]$	7.0	0.1	12.1	2.1
x_{12}	Bed width	Branson gouging station (1999)[c]	$f(Q_i)$	$[m]$	51.3	46.9	57.1	2.8
x_{13}	Average substrate size	(Baumann; 2004)	0.028	$[m]$	–	–	–	–
x_{14}	Bed width variability	personal evaluation	1	–	–	–	–	–
x_{15}	HL_{Ref}	*La Porte du Scex* gouging station (yr. 1907) (Meile et al.; 2005)	0.0835	$[m]$	–	–	–	–
x_{16}	Fuzzy bedwidth variability	personal evaluation	1	–	–	–	–	–
x_{17}	Fuzzy depth variability	personal evaluation	1	–	–	–	–	–
x_{18}	Fuzzy structural variability	personal evaluation	1	–	–	–	–	–



Group	Mean	Min.	Max.	Var. Coef.
E	2.4	1.2	3.8	0.3
P	1.0	0.0	2.5	0.7
T	1.1	0.9	1.5	0.1
EPT	4.5			

Figure 5.2: Scenario 1 – Model Results for E-, P- and T-groups. Red bar stands for taxa eq. median, and gray bar taxa eq. mean. Averaged taxa value, minimal, maximal and variation coefficient shown in adjacent table

5.2.2 Scenario 1 – Results

EPT-Group Indicator

The mean EPT Indicator value resulting from the actual state inputs (Table 5.1) is of *ca.* 4.5 taxa (Figure 5.2). The highest group value were found for the E-group, with an averaged result of *ca.* 2.4 taxa, minimal and maximal values of respectively *ca.* 1.2 and 3.8 taxa with a variation coefficient of *ca.* 0.3. This group has the highest annual indicator dispersion.

The P-group comes in second position, with an averaged result of *ca.* 1.0 taxa, minimal and maximal values of respectively *ca.* 0.0 and 2.5 taxa. This group has a variation coefficient of *ca.* 0.7.

The T-group comes last, with an averaged result of *ca.* 1.1 taxa, minimal and maximal values of respectively *ca.* 0.9 and 1.5 taxa. This group has a low annual dispersion with a variation coefficient of *ca.* 0.1.

Habitat value (HV)[3] (Figure 5.3) in the current scenario is best for the *Midstream* guild. This guild has an HV of *ca.* 0.32, with minimal and maximal hourly HSI values of respectively *ca.* 0.09 and 0.42. HSI dispersion is high with a variation coefficient of *ca.* 0.28.

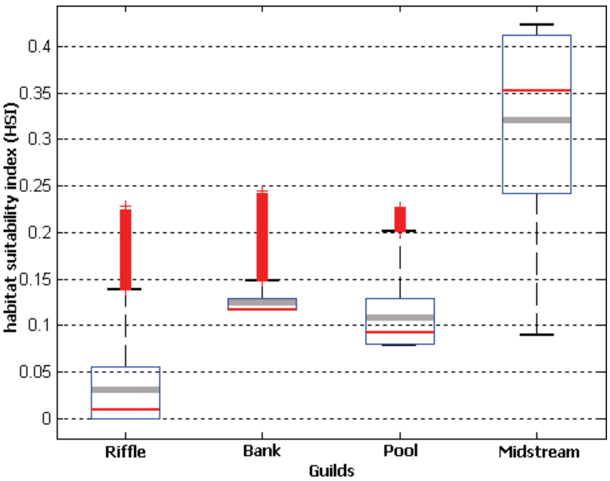
The second highest results are for the *Bank* guild, with an HV of *ca.* 0.13, minimal and maximal hourly HSI values of respectively *ca.* 0.12 and 0.24. HSI dispersion is high with a variation coefficient of *ca.* 0.15.

In third is the *Pool* guild, with an HV of *ca.* 0.11, minimal and maximal

[3] *Habitat Value* (HV)=average of hourly annual *Habitat Suitability Indexes* (HSI)

hourly HSI values of respectively *ca.* 0.08 and 0.23. Variation coefficient for the *Pool* guild is high with *ca.* 0.27.

Last is the *Riffle* guild, with an HV of *ca.* 0.03, minimal and maximal hourly HSI values of respectively 0.00 and *ca.* 0.23. Annual dispersion is high with a variation coefficient of *ca.* 1.67.



Guild	HV	HSI Min.	HSI Max.	Var. Coef.
Riffle	0.03	0.00	0.23	1.67
Bank	0.13	0.12	0.24	0.15
Pool	0.11	0.08	0.23	0.27
Mid.	0.32	0.09	0.42	0.28

Figure 5.3: Scenario 1: Actual state - Fish guilds habitat suitability and habitat value results. Red bar stands for the HSI median while gray bar for the HV value. Minimal, maximal and variation coefficient presented in adjacent table

5.3 Scenario 2 - The bed widening project

5.3.1 Scenario overview

This project would imply a hypothetical bed widening starting at the Riddes section (Figure 5.4). This scenario attempts to assess the ecological consequences of such a development project for the downstream part of the river.

River cross section is assumed to be similar for the whole downstream part of the river (Figure 5.5). Hydraulic inputs of year 1999 were used together with a structurally widen configuration of the river bed. Covariate description and ranges for this scenario are depicted in Table 5.2.

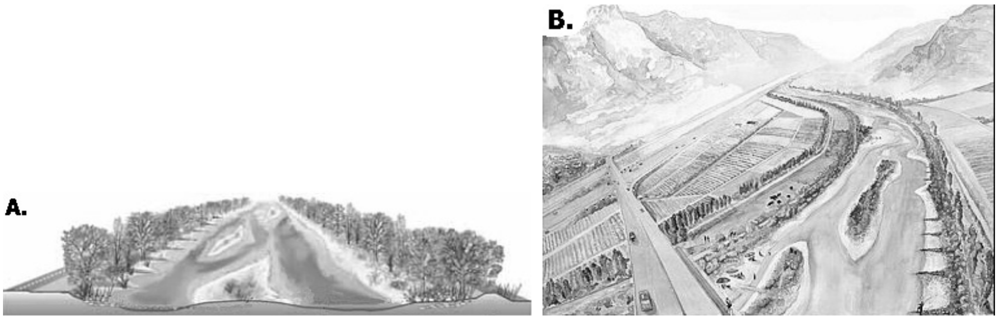


Figure 5.4: Bed widening scenario: (A.) schematic representation of possible bed widening scenario (modified from (DTEE; 2005)) (B.) artistic view from WWF Wallis/Lebensraum Rotten by Alberto E. Conelli

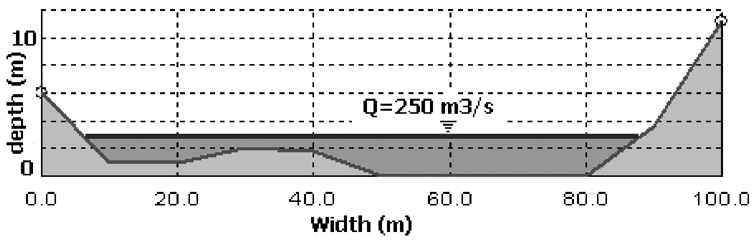


Figure 5.5: Cross section used for bed widening scenario (Scenario 2)

Table 5.2: Scenario 2 – Covariate description:[a]based on annual hourly (i) depth (Z) vs. discharge (Q) relation of $Z_i = -1e^{-5}Q_i^2 + 0.0122Q_i + 0.3764$ ($R^2 = 0.97$).[b]based on annual hourly speed (V) vs. discharge (Q) relation $V_i = 4e^{-8}Q_i^3 - 4e^{-5}Q_i^2 + 0.0131Q_i + 0.3723$ ($R^2 = 0.94$).[c]based on annual hourly top width (W) vs. discharge (Q) relation $W_i = 1e^{-6}Q_i^3 - 0.0013Q_i^2 + 0.4384Q_i + 32.033$ ($R^2 = 0.98$)

	Variable	Source	Value	Unit	Mean	Min.	Max.	Stdev.
$x_1 - x_5$	Substrate	Limnex (Baumann; 2004)	Cobbles (x_2) and gravels (x_3)	binary	–	–	–	–
x_6	HS	Branson gouging station (1999)[a]	$f(Q_i)$	$[m]$	0.3017	0.1296	0.4671	0.1297
x_7	Depth	Branson gouging station (1999)[a]	$f(Q_i)$	$[m]$	2.0	0.7	4.0	0.9
x_8	Current Speed	Branson gouging station (1999)[b]	$f(Q_i)$	$[m/s]$	1.4	0.7	2.0	0.3
x_9	HL	Branson gouging station (1999)[a]	0.3748	$[m]$	–	–	–	–
x_{10}	Distance from origin	LCH report (LCH; 2005)	109	$[km]$	–	–	–	–
x_{11}	Water temperature	from OFEV	$f(i)$	$[^oC]$	7.0	0.1	12.1	2.1
x_{12}	Bed width	Branson gouging station (1999)[c]	$f(Q_i)$	$[m]$	65.7	42.5	76.3	9.4
x_{13}	Average substrate size	(Baumann; 2004)	0.028	$[m]$	–	–	–	–
x_{14}	Bed width variability	personal evaluation	1	–	–	–	–	–
x_{15}	HL_{Ref}	*La Porte du Scex* gouging station (yr. 1907) (Meile et al.; 2005)	0.0835	$[m]$	–	–	–	–
x_{16}	Fuzzy bed-width variability	personal evaluation	1	–	–	–	–	–
x_{17}	Fuzzy depth variability	personal evaluation	1	–	–	–	–	–
x_{18}	Fuzzy structural variability	personal evaluation	1	–	–	–	–	–

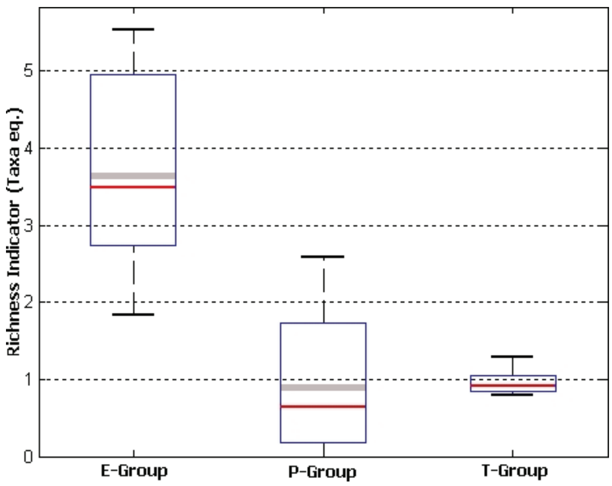
5.3.2 Scenario 2 - Results

EPT-group Indicator

The mean EPT indicator value resulting from scenario 2 inputs (Table 5.2) is of *ca.* 5.5 taxa (Figure 5.6). The highest group value was found for the E-group,with a mean value of *ca.* 3.7 taxa, minimal and maximal values of respectively *ca.* 1.8 and 5.5 taxa. The E-group also has the highest annual dispersion and a variation coefficient of *ca.* 0.3.

The T-group comes in second, with a mean indicator value of *ca.* 0.9 taxa eq., minimal and maximal values of respectively *ca.* 0.8 and 1.3 and a variation coefficient of *ca.* 0.1. The T-group has the most grouped annual values.

The P-group comes in third, with a mean indicator value of *ca.* 0.9 taxa eq., minimal and maximal values of respectively *ca.* 0 and 2.6 and a variation coefficient of *ca.* 0.9.



Group	Mean	Min.	Max.	Coef. Var.
E	3.7	1.8	5.5	0.3
P	0.9	0.0	2.6	0.9
T	0.9	0.8	1.3	0.1
EPT	5.5			

Figure 5.6: Scenario 2 model results for E-, P- and T-groups. Red bar stand for taxa eq. median, gray bar stand for taxa eq. mean. Averaged taxa value, minimal, maximal and variation coefficient shown in the adjacent table

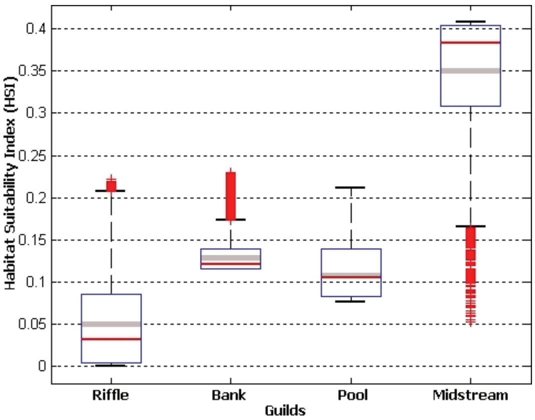
Fish Habitat Suitability Index and Habitat Value

Habitat suitability results and habitat value (Figure 5.7) in scenario 2 are best for the *Midstream* guild, with an HV of *ca.* 0.35, minimal and maximal hourly HSI values of respectively *ca.* 0.05 and 0.41. The *Midstream* guild has the highest annual variability and a variation coefficient of *ca.* 0.2.

The *Bank* guild has the second highest HV value (*ca.* 0.13), with minimal and maximal hourly HSI values of respectively *ca.* 0.12 and 0.23. The variation coefficient for the *Bank* guild HSI values is *ca.* 0.15.

The *Pool* guild has the third highest HV value (*ca.* 0.11), with minimal and maximal hourly HSI values of respectively *ca.* 0.07 and 0.21. The variation coefficient for the *Pool* guild HSI values is *ca.* 0.27.

The *Riffle* guild has the lowest HV value (*ca.* 0.05), with minimal and maximal hourly HSI values of respectively *ca.* 0.00 and 0.22. The variation coefficient for the *Riffle* guild HSI values is *ca.* 1.00.



Guild	HV	HSI Min.	HSI Max.	Var. Coef.
Riffle	0.05	0.00	0.22	1.00
Bank	0.13	0.12	0.23	0.15
Pool	0.11	0.07	0.21	0.27
Mid.	0.35	0.05	0.41	0.2

Figure 5.7: Scenario 2 – bed widening. Fish guilds habitat suitability results and habitat value results. Red bars stand for the HSI median while gray bar stand for the HV value. Minimal, maximal and variation coefficient presented in the adjacent table

5.4 Scenario 3 - The SYNERGIE project: hydropeaking buffer basin

5.4.1 Scenario overview

This scenario is presented in detail in Chapter 1.1, on p. 15. A hydropeaking buffer basin is designed following general ecological considerations (p. 65) together with an artificial river for the maintenance of the river longitudinal continuum (p. 66). The hydraulic and thermal inputs come from the hydraulic simulation by Heller (Heller; 2007). The characteristics of the downstream river (bed width, substrate, height vs. discharge relation, etc...) are considered equal to scenario 1 and covariate ranges are depicted in Table 5.3.

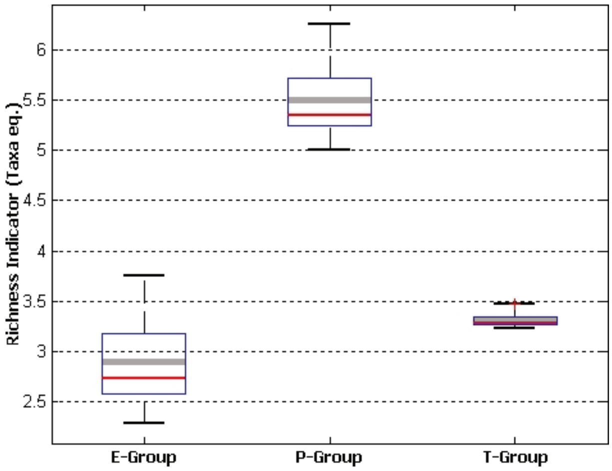
Table 5.3: Scenario 3 – Covariate description:[a],[b] and [c] relations are depicted in Table 5.1 (no river structural changes).† substrate size is expected to increase with lowered hydropeaking

	Variable	Source	Value	Unit	Mean	Min.	Max.	Stdev.
$x_1 - x_5$	Substrate	Limnex (Baumann; 2004)	Cobbles (x_2) and gravels (x_3)	binary	–	–	–	–
x_6	HS	Branson gouging station (1999)[a]	$f(Q_i)$	$[m]$	0.1030	0.0727	0.1377	0.0260
x_7	Depth	Branson gouging station (1999)[a]	$f(Q_i)$	$[m]$	1.8	0.7	3.5	0.72
x_8	Current Speed	Branson gouging station (1999)[b]	$f(Q_i)$	$[m/s]$	1.6	0.7	2.7	0.5
x_9	HL	Branson gouging station (1999)[a]	0.3748	$[m]$	–	–	–	–
x_{10}	Distance from origin	LCH report (LCH; 2005)	109	$[km]$	–	–	–	–
x_{11}	Water temperature	from OFEV	$f(i)$	$[^oC]$	7.1	0.12	13.4	2.2
x_{12}	Bed width	Branson gouging station (1999)[c]	$f(Q_i)$	$[m]$	50.9	46.9	56.6	2.5
x_{13}	Average substrate size	(Baumann; 2004)	0.17†	$[m]$	–	–	–	–
x_{14}	Bed width variability	personal evaluation	1	–	–	–	–	–
x_{15}	HL_{Ref}	*La Porte du Scex* gouging station (yr. 1907) (Meile et al.; 2005)	0.0835	$[m]$	–	–	–	–
x_{16}	Fuzzy bedwidth variability	personal evaluation	1	–	–	–	–	–
x_{17}	Fuzzy depth variability	personal evaluation	1	–	–	–	–	–
x_{18}	Fuzzy structural variability	personal evaluation	1	–	–	–	–	–

5.4.2 Scenario 3 - Results

EPT-group Indicator

The mean EPT indicator value resulting from scenario 3 inputs (Table 5.3) is of *ca.* 11.7 taxa (Figure 5.8). The highest group value was found for the P-group, with a mean value of *ca.* 5.5 taxa, minimal and maximal values of respectively *ca.* 5.0 and 6.3 taxa and a variation coefficient of *ca.* 0.05. The



Group	Mean	Min.	Max.	Coef. Var.
E	2.9	2.3	3.8	0.1
P	5.5	5.0	6.3	0.05
T	3.3	3.2	3.5	0.03
EPT	11.7			

Figure 5.8: Scenario 3 - `SYNERGIE` project basin model results for E-, P- and T-groups. Red bars stand for taxa eq. median while gray bars stand for taxa eq. mean. Averaged, minimal, maximal and variation coefficients shown in adjacent table

T-group comes in second, with a mean value of *ca.* 3.3 taxa, minimal and maximal values of respectively *ca.* 3.2 and 3.5 taxa and a variation coefficient of *ca.* 0.03.

The E-group comes has the lower predicted richness, with a mean value of *ca.* 2.9 taxa, minimal and maximal values of respectively *ca.* 2.3 and 3.8 taxa and a variation coefficient of *ca.* 0.1.

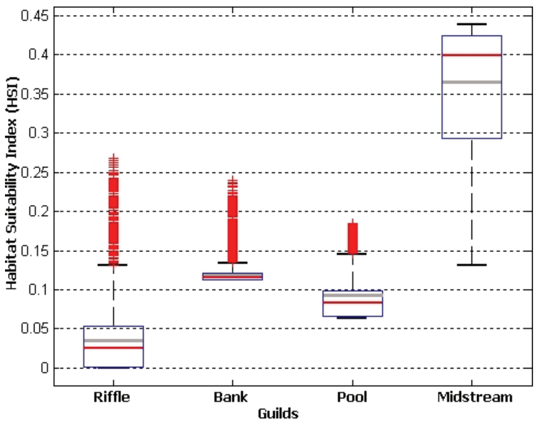
Fish Habitat Suitability Index and Habitat Value

Habitat suitability results and habitat value (Figure 5.9) in scenario 3 are best for the *Midstream* guild, with an HV of *ca.* 0.36, minimal and maximal hourly HSI values of respectively *ca.* 0.13 and 0.44. The *Midstream* guild has the highest annual variability and a variation coefficient of *ca.* 0.2.

The *Bank* guild has the second highest HV with a value of *ca.* 0.12, minimal and maximal hourly HSI values of respectively *ca.* 0.11 and 0.24. The variation coefficient for the *Bank* guild is *ca.* 0.1.

The *Pool* guild has the third HV value of *ca.* 0.09, minimal and maximal hourly HSI values of respectively *ca.* 0.06 and 0.18. The variation coefficient for the *Pool* guild is 0.02.

The *Riffle* guild presents the poorest results, with a HV value of *ca.* 0.03, minimal and maximal hourly HSI values of respectively *ca.* 0.00 and 0.27 and a variation coefficient of *ca.* 0.04.



Guild	HV	HSI Min.	HSI Max.	Var. Coef.
Riffle	0.03	0.00	0.27	1.3
Bank	0.12	0.11	0.24	0.1
Pool	0.09	0.06	0.18	0.2
Mid.	0.36	0.13	0.44	0.2

Figure 5.9: Scenario 3 – `SYNERGIE` project basin - Fish guilds habitat suitability and habitat value results. Red bars stand for the HSI medians while gray bars stand for HV values. HSI minimal, maximal and variation coefficient presented in adjacent table

5.5 Scenario 4 – Coupling of the `SYNERGIE` buffer basin and bed widening

5.5.1 Scenario overview

This scenario would imply a hypothetical bed widening such as depicted in section 5.3 (p. 116) coupled to the implementation of the `SYNERGIE` buffer basin (section 5.4 - p. 120). The input ranges used are detailed on Table 5.4.

5.5.2 Scenario 4 - Results

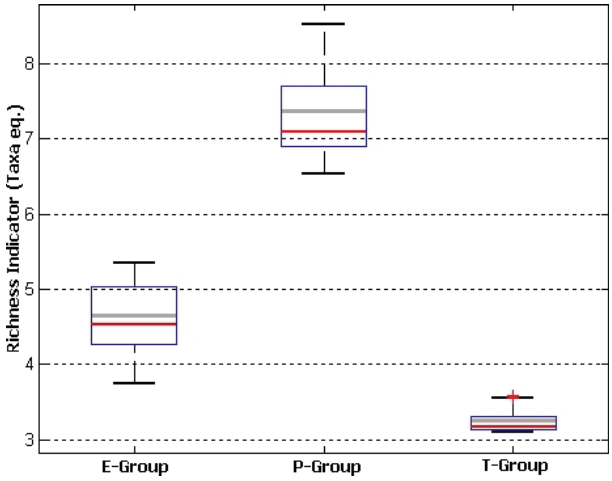
EPT-group Indicator

The mean EPT indicator value resulting from scenario 4 inputs (Table 5.4) is of *ca.* 15.1 taxa (Figure 5.10). The highest group value was found for the

Table 5.4: Scenario 4 – Covariate description:[a],[b] and [c] relations are depicted in Table 5.2 (same hypothetical river structural changes).

	Variable	Source	Value	Unit	Mean	Min.	Max.	Stdev.
$x_1 - x_5$	Substrate	Limnex (Baumann; 2004)	Cobbles (x_2) and gravels (x_3)	binary	–	–	–	–
x_6	HS	Branson gouging station (1999)[a]	$f(Q_i)$	$[m]$0.1126	0.0798	0.1495	0.0281	
x_7	Depth	Branson gouging station (1999)[a]	$f(Q_i)$	$[m]$	1.9	0.7	3.7	0.7
x_8	Current Speed	Branson gouging station (1999)[b]	$f(Q_i)$	$[m/s]$	1.4	0.7	1.8	0.3
x_9	HL	Branson gouging station (1999)[a]	0.1261	$[m]$	–	–	–	–
x_{10}	Distance from origin	LCH report (LCH; 2005)	109	$[km]$	–	–	–	–
x_{11}	Water temperature	from OFEV	$f(i)$	$[^oC]$	7.1	0.1	13.4	2.2
x_{12}	Bed width	Branson gouging station (1999)[c]	$f(Q_i)$	$[m]$	65.5	42.6	76.3	7.3
x_{13}	Average substrate size	(Baumann; 2004)	0.17	$[m]$	–	–	–	–
x_{14}	Bed width variability	personal evaluation	1	–	–	–	–	–
x_{15}	HL_{Ref}	*La Porte du Scex* gouging station (yr. 1907) (Meile et al.; 2005)	0.0835	$[m]$	–	–	–	–
x_{16}	Fuzzy bedwidth variability	personal evaluation	1	–	–	–	–	–
x_{17}	Fuzzy depth variability	personal evaluation	1	–	–	–	–	–
x_{18}	Fuzzy structural variability	personal evaluation	1	–	–	–	–	–

P-group, with a mean richness indicator value of *ca.* 7.3 taxa, minimal and maximal values of respectively *ca.* 6.5 and 8.5 and a variation coefficient of *ca.* 0.1.



Group	Mean	Min.	Max.	Coef. Var.
E	4.6	3.7	5.4	0.1
P	7.3	6.5	8.5	0.1
T	3.2	3.1	3.6	0.05
EPT	15.1			

Figure 5.10: Scenario 4 - bed widening coupled to `SYNERGIE` project basin model results for E-, P- and T-groups. Red bars stand for taxa eq. medians while gray bars stand for taxa eq. mean. Averaged, minimal, maximal and variation coefficient shown in adjacent table

The E-group comes in second, with a mean richness indicator value of *ca.* 4.6 taxa, minimal and maximal values of respectively *ca.* 3.7 and 5.4 and a variation coefficient of *ca.* 0.1.

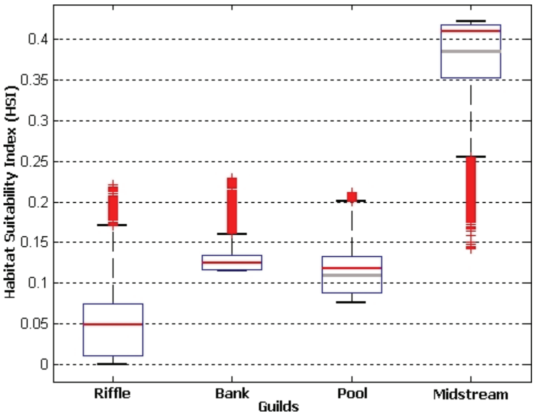
The T-group comes last, with a mean richness indicator value of *ca.* 3.2 taxa, minimal and maximal values of respectively *ca.* 3.1 and 3.6 and a variation coefficient of *ca.* 0.05.

Fish Habitat Suitability Index and Habitat Value

Habitat suitability results and habitat value (Figure 5.11) in scenario 4 are best for the *Midstream* guild, with an HV of *ca.* 0.38, minimal and maximal hourly HSI values of respectively *ca.* 0.14 and 0.42. The *Midstream* guild has the highest annual variability and variation coefficient of *ca.* 0.02.

The *Bank* guild has the second highest position with HV value of *ca.* 0.13, minimal and maximal hourly HSI values of respectively *ca.* 0.12 and 0.23. The variation coefficient for the *Bank* guild is *ca.* 0.8.

The *Pool* guild is in third position with an HV of *ca.* 0.11, minimal and maximal hourly HSI values of respectively *ca.* 0.06 and 0.18. The variation coefficient for the *Pool* guild is *ca.* 0.3.



Guild	HV	HSI Min.	HSI Max.	Var. Coef.
Riffle	0.05	0.00	0.22	0.8
Bank	0.13	0.12	0.23	0.8
Pool	0.11	0.08	0.21	0.3
Mid.	0.38	0.14	0.42	0.2

Figure 5.11: Scenario 4 - bed widening coupled to SYNERGIE project basin - Fish guilds habitat suitability and habitat value results. Red bars stand for the HSI medians while gray bars stand for the HV values. HSI minimals, maximals and variation coefficients in adjacent table

The *Riffle* guild is in last position, with an HV of *ca*. 0.05, minimal and maximal hourly HSI values of respectively *ca*. 0.00 and 0.22. The variation coefficient for the *Riffle* guild is of *ca*. 0.8.

5.6 Limits of annual chronological analysis

5.6.1 Introduction

This chapter focuses on the discussion of the ecological responses of the *EPT group* (R_{EPT}) and *fish habitat value* indicators predicted for each scenario. This chapter consist in:

1. *Intra-scenario analysis* – in depth discussion of each scenario's results for both the R_{EPT} indicator and the fish guild habitat values and their ecological implication

2. *Inter-scenario analysis* – in depth discussion of the various scenarios ecological implications for the system

The chapter will end with some concluding remarks.

5.6.2 Intra-scenario analysis

Scenario 1 - Current state

The Swiss Upper Rhone River in its actual state is heavily impacted hydrologically (i.e. hydropeaking, see Figure 3.5) and morphologically (see Figure 3.3). Hence, we expect its ecological integrity (translated by the R_{EPT} value and fish guild HV) to be low.

Scenario 1 - EPT group richness prediction The aggregated R_{EPT} value for scenario 1 is of *ca.* 4.5 taxa. Ephemeroptera richness index (RE) is the highest, with an averaged model output of *ca.* 2.4 taxa (Figure 5.12, blue band). Graphical interpretation of the model results indicates a seasonal effect and hence an overestimation of RE in winter and beginning of spring, as well as an under-estimation of RE in end of spring, summer and beginning of the autumnal seasons. In accordance with taxonomic surveys (Baumann; 2004; ECOTEC; 1998, 1999, 2004; GIDB-R3; 2005; Gogniat and Marrer; 1984/85), the mayflies susceptible of being encountered are most likely from the Baetidae family, namely *Baetis alpinus* or *Baetis rhodani*. Baetidae are in general not considered especially sensitive to water quality (IBGN GI value of 2 (AFNOR; 1985)). Low mayfly richness represented mainly by *B. rhodani* is typical of heavily regulated rivers (Brittain and Salveit; 1989).

Plecoptera richness index (RP) has the second highest aggregated model output with an averaged value of *ca.* 1.0 taxa. Graphical interpretation of the model output (Figure 5.12, green band) shows a similar trend of seasonality, with HV over-estimations for the cold seasons (end of autumn, winter up to the end of spring) and under-estimation ranging from the end of spring to the first third of autumn. In accordance with taxonomic surveys, stoneflies susceptible of being encountered are most likely from:

- the Leuctridae family (*Leuctra sp.*)
- the Nemouridae family (*Nemoura mortoni*)

Leuctridae and Nemouridae families can be considered as interesting since their presence accounts for a rather good water quality with low organic pollution (IBGN GI value of 7 for Leuctridae and 6 for Nemouridae (AFNOR; 1985)). However, we would expect a much higher richness since Plecoptera (and more specifically the Nemouridae family) are under natural circumstances very diversified in the headwater zones (Tachet et al.; 2000). This is a cue for a low ecological integrity under actual conditions since the water quality of the Rhone is reported as good (R3; 1998; GIDB-R3; 2005). Plecoptera diversity seem to be not only a good indicator for headwater water quality but also a good indicator of hydrological and morphological conditions (as mentioned by (Céréghino and Lavandier; 1998a,b)).

Trichoptera richness index (RT) has the lowest model output, with an averaged richness value of *ca.* 1.1 taxa. The seasonal effect does not appear very marked (Figure 5.12, red band) and caddislies susceptible of being encountered are probably restricted to the Limnephilidae family (*Allogamus sp.*), which are known to thrive under hydropeaking conditions and channelized beds (Bernard; 2001; Frutiger; 2004b).

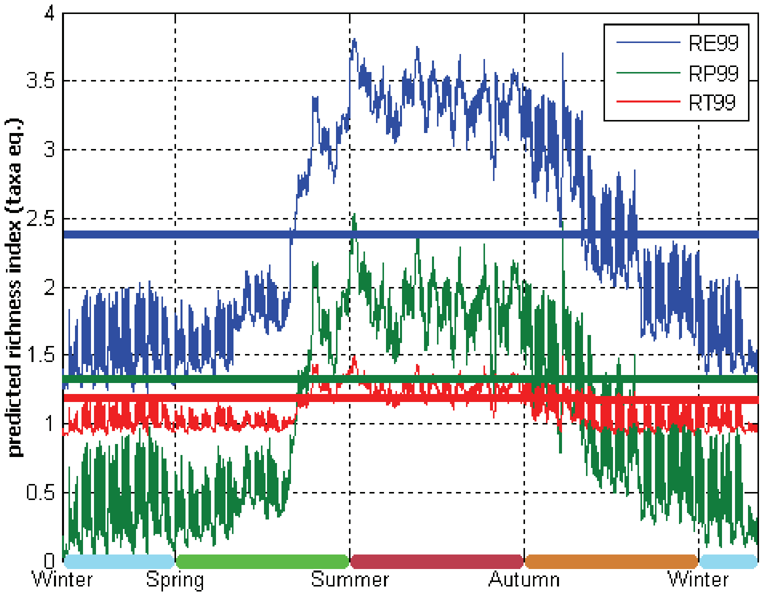


Figure 5.12: Scenario 1 - EPT group richness prediction indexes. Solid lines represent aggregated model output values, RE stands for Ephemeroptera Richness eq., RP for Plecoptera Richness eq. and RT for Trichoptera Richness eq.

We can consider this R_{EPT} value of *ca.* 4.5 taxa to be low and representative of low ecological integrity. The upstream zone of *Finges* is almost not impacted by channelization or hydropeaking and has a similar water quality, with the exception of less suspended solids in the winter season. Observed combined E,P and T richness was up to 11 taxa (Baumann; 2004), clearly indicating a ecological integrity deficit in the Riddes site under its actual conditions.

Scenario 1 - Fish guild habitat values The *riffle* guild has a HV value of *ca.* 0.03 (see Figure 5.13, black line). There is a strong seasonal effect represented by extreme HSI variability in the cold seasons (autumn, winter, and up to the end of spring). The high HSI values are probably due to the low water depths in-between hydropeaking events corresponding to better preferences for the *riffle* guild. In the warm seasons (end of spring, summer and first third of autumn), the high discharges coupled to the trapezoidal structure of the bed are assumed to result in water depths too important to suit the *riffle* guild preferences. A fundamental question is raised by the extreme HSI cold-season (end of autumn, winter up to end of spring) variations on the ecological pertinence of the use of HV in the characterization of the habitat value without the prior knowledge of the response ability of fish guild to sudden temporal change as well as their ability to occupy a very temporary habitat. The issue of the minimal HSI persistence time to be considered ecologically meaningful has to be raised. In other words how long must a HSI value last in order to be available and hence meaningful for a fish guild? It is also very probable that fish response varies with guild (Bunt et al.; 1999). So although we cannot be certain of the under-estimation of

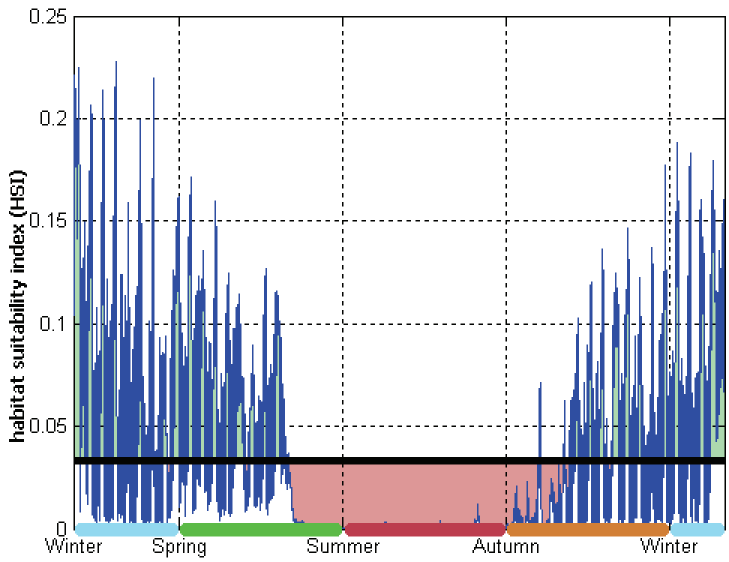


Figure 5.13: Scenario 1 - *Riffle* Guild: Effective habitat value for *riffle* guild under 1999 hydraulic conditions at Riddes site (km 56.31). The black line represents the model aggregated habitat value, the green-fill may possibly account for habitat suitability under-estimation while the red-filled section clearly accounts for habitat suitability over-estimation

the HSI values in the cold season end of autumn, winter up to end of spring, Figure 5.13, green-filled), we clearly see an over-estimation of the HSI values in the warm season (end of summer until autumn, Figure 5.13, red-filled). Ecologically, this implies that the Swiss Upper Rhone River under its actual morphological configuration and hydraulic regime has a rather ineffective hydrology for the *riffle* habitat morphology. It is not really possible to use the absolute HV value as a comparative measure with proportional ecological consequences, however, we assume that a higher HV value calls for a higher guild habitat quality. The high variability in HSI values indicates (max. HSI *ca.* 0.23) that there is a potentiality for higher limit *riffle* guild habitat value under current morphology (maximal HV value may tend toward 0.23).

The *bank* guild has a HV value of *ca.* 0.13 (see Figure 5.14, black line). There is a strong seasonal effect similarly presented by strong HSI variability in the cold season (autumn up to the end of spring) for similar reasons as the *riffle* guild. The high HSI values are produced mainly in winter, in-between hydropeaking events, where water levels are low so that the profile of the river is not trapezoidal anymore and hence has a better fulfillment of the *bank* guild preferences in term of water depth and current speed. The maximal HSI value (*ca.* 0.24) indicates the potential for a higher limit *bank* habitat value tending toward 0.24 under current morphology. The same question about the ecological pertinence of using HV value to characterize the habitat can be raised, where cold-seasons under-estimation (Figure 5.14, green-filled) cannot be ruled out and warm-seasons over-estimation (Figure 5.14, red-filled) is apparent. Ecologically, and taking into account the uncertainty of this guild's response time to sudden water variations, this implies that the Swiss Upper Rhone River under its actual morphological configuration and hydraulic regime is not well suited for fish species categorized in the *bank* guild.

The *pool* guild has a HV value of *ca.* 0.11 (see Figure 5.15, black line). There is a strong seasonal effect similarly presented by strong HSI variability in the cold seasons (second half of autumn up to end of spring) for similar reasons as the *riffle* guild. There appears to be a very slight seasonal effect, with cold-season under-estimation (Figure 5.14, green-filled) and warm-season (end of spring until beginning of autumn) over-estimation (Figure 5.14, red-filled) of the hourly HSI by the annual HV. The high winter hourly HSI values are probably caused by the conditions in between hydropeaking events, with lower discharges and lower current velocities more suitable to *pool* guild preferences. HSI lower values provide an insight on HV at bankfull (*ca.* 0.08) while maximal HSI values provide an insight on actual maximal habitat suitability (*ca.* 0.23). Ecologically, this implies that there is some potential under the actual morphological configuration for higher HV but that the hydraulic regime is not well suited for fish species categorized in the *bank* guild, which is in accordance to observations in literature (DTEE; 2004), where shallow and fast flowing reaches were identified as possible improvement measures for fish.

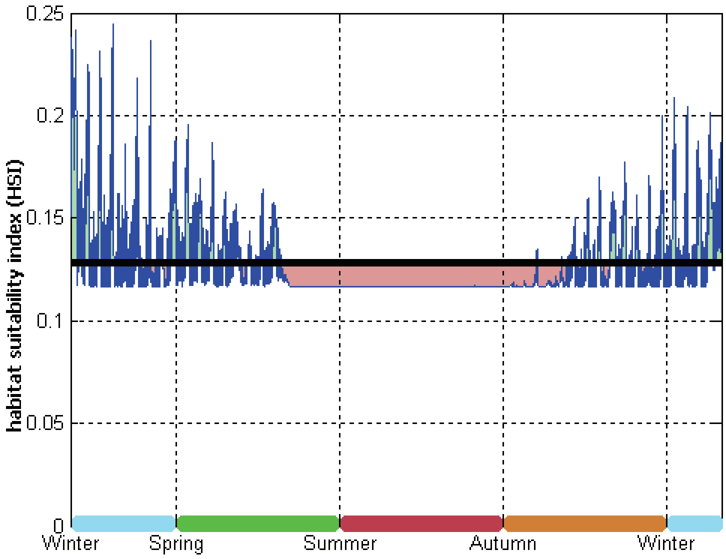


Figure 5.14: Scenario 1 - *Bank* Guild: Effective habitat value for *bank* guild under 1999 hydraulic conditions at Riddes site (km 56.31). The black line represents the model aggregated habitat value, the green-fill may possibly account for habitat suitability under-estimation while the red-filled section clearly accounts for habitat suitability over-estimation

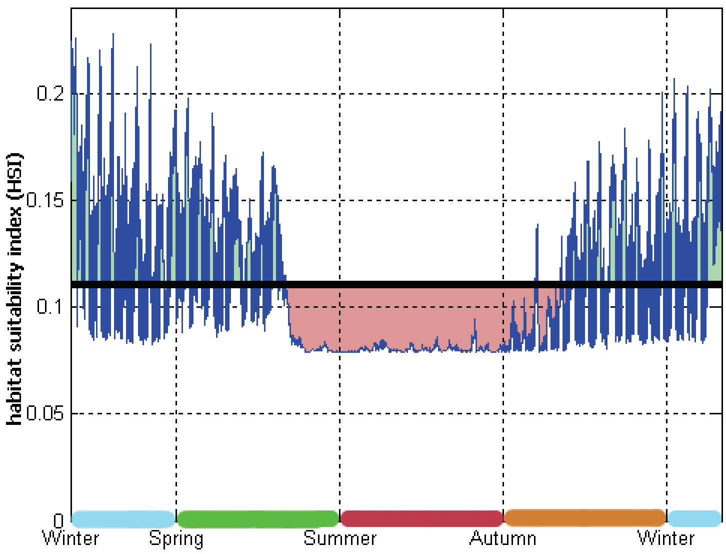


Figure 5.15: Scenario 1 - *Pool* Guild: Effective habitat value for *pool* guild under 1999 hydraulic conditions at Riddes site (km 56.31). The black line represents the model habitat value, the green-filled section the slight under-estimation of habitat suitability and the red-filled section the over-estimation of the habitat suitability

The *midstream* guild has a HV value of *ca.* 0.32 (see Figure 5.16, black line). There is a extreme seasonal effect between the cold (first third of autumn until two-third of spring) and warm (end of spring until beginning of autumn) seasons. In the cold season, the HSI values appear satisfactory for the *midstream* guild, despite their extreme temporal variability (see Figure 5.16, green-filled) and are probably under-estimated by model HV. There is still some extreme variability in the HSI value of the warm season, but HSI appear lower (see Figure 5.16, red-filled) and are probably slightly over-estimated. Cold season HSI variability is probably explained by hydropeaking alone (dial frequency), while warm season variability is probably explained by snow-melt hydraulic response, precipitation (i.e climatic response) and to a lesser extent hydropeaking.

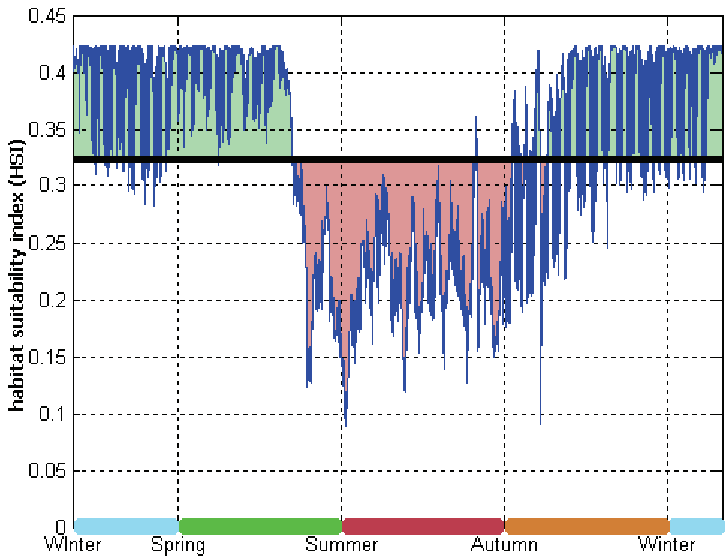


Figure 5.16: Scenario 1 - *Midstream* Guild: Seasonality effect for *midstream* guild under 1999 hydraulic conditions at Riddes site (km 56.31). The black line represents the model habitat value, the green-filled section the under-estimated habitat suitability and the red-filled section the over-estimated habitat suitability

Scenario 1 - Concluding remarks The ecological integrity is generally poor, with a low R_{EPT} richness index to what could be compared in literature (Baumann; 2004; ECOTEC; 1998; GIDB-R3; 2005). From the macro-invertebrate point of view, Limnephilidae would clearly predominate in such a system. Predicted taxonomic diversity is in concert with literature which states that it reaches its maximum in autumn and is at its minimal in febuary (Bernard; 2001). From fish habitat perspective, only the *midstream* guild and to a much lower extent the *bank* guild may find some suitable habitat. Other guilds habitat is probably marginal. This is also consistent with reports in literature stating that the Swiss Upper Rhone is of low interest for fish populations and has a very limited population sustainability potential (DTEE; 2004).

Scenario 2 - Bed widening

The bed widening project does not solve the hydropeaking (Figure 3.5) issue and is actually susceptible of making its effect worse, principally on macroinvertebrates which do not have a good reactive capability (Céréghino and Lavandier; 1998a). However, the structural diversity of the bed has increased, allowing organisms capable of reacting quickly in time to hydrological change (i.e. fish) to find shelter and have less extreme hydraulic conditions in terms of current speed and depth principally.

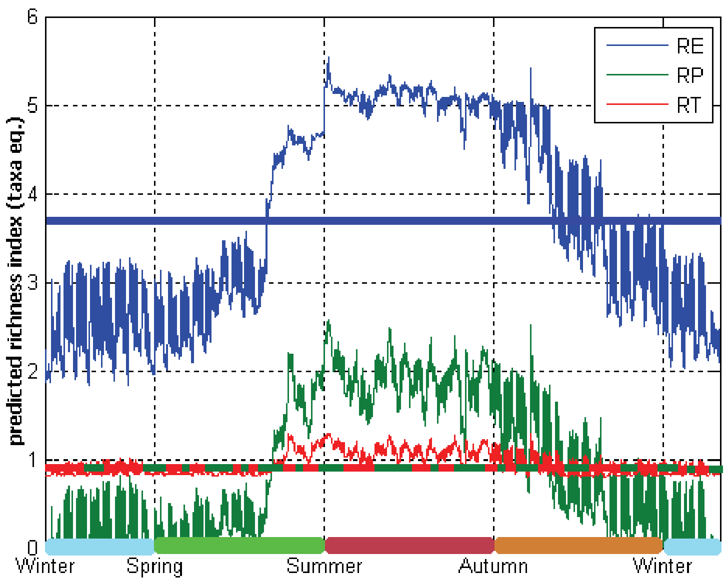


Figure 5.17: Scenario 2 - EPT group richness prediction indexes. Solid lines represent aggregated model output values, RE stands for Ephemeroptera Richness eq., RP for Plecoptera Richness eq. and RT for Trichoptera Richness eq.

Scenario 2 - EPT richness prediction index The aggregated R_{EPT} value for scenario 2 is of *ca.* 5.5 taxa. Ephemeroptera richness index (RE) is the highest, with a model output richness indicator of *ca*. 3.7 taxa (Figure 5.17, blue band). Graphical interpretation of the model results in an important seasonal effect yielding to an overestimation of richness indicator in cold-seasons (end of autumn until end of spring), as well as an underestimation of richness indicator in warm seasons (end of spring to end of autumn). In accordance to taxonomic surveys (Baumann; 2004; ECOTEC; 1998, 1999, 2004; GIDB-R3; 2005; Gogniat and Marrer; 1984/85), the mayflies susceptible of being encountered are most likely:

- from the Baetidae family, namely *Baetis alpinus* or *Baetis rhodani*
- from the Heptageniidae family, namely *Rhitrogena sp.*

Baetidae are not generally considered very sensitive to water quality (IBGN GI value of 2 (AFNOR; 1985)). Their life cycle has been reported to be affected by hydropeaking (Raddum and Fjellheim; 1993). Hepatgeniidae family, however, have a slightly higher IBGN GI value of 5, which somehow supports the river's good water quality. They are reported sensitive to anthropic impacts and hydropeaking in literature (Bernard; 2001; Brittain and Salveit; 1989; Céréghino et al.; 2004).

Plecoptera richness index (RP) has a model output richness index of *ca.* 0.9 taxa (Figure 5.17,green band). Graphical interpretation of the model output shows a trend of seasonality as previously encountered with Ephemeroptera. The richness indicator seems over-estimated in the cold season (end of autumn to end of speing) and under-estimated in the warm season (end of spring until end of autumn). The stoneflies susceptible of being encountered are most likely:

- the Leuctridae family (*Leuctra sp.*)
- the Nemouridae family (*Nemoura mortoni*)

Leuctridae and Nemouridae families have rather high IBGN GI values of respectively 7 and 6 (AFNOR; 1985), which confirm the good water quality. Nemouridae are reported as affected by hydropeaking (Bernard; 2001).

Trichoptera richness index (RT) has the most constant results, with richness indicator model output of *ca.* 0.9 (Figure 5.17, red band). There is a very slight and probably negligible seasonal effect. Ecological interpretation probably restricts the caddisflies taxa suceptible of being encountered to the hydropeaking resistant Limnephilidae family, namely *Allogamus auricollis* (Frutiger; 2004b).

We can consider the value of R_{EPT} of 5.5 to be low and representative of a relatively low ecological integrity when compared to the upstream area of *Finges*, where R_{EPT} of 11 taxa was observed (Baumann; 2004).

Scenario 2 - Fish HV The *riffle* guild has a HV of *ca.* 0.05 (see Figure 5.18, black line). There appears to be a seasonal effect, with extreme variations in the cold season (first third of autumn until two-third of spring) and a probability for a slight under-estimation of the HSI (cf. section 5.6.2 p. 128, Figure 5.18, green-filled area and concerns about HSI persistence effect). A warm-season (last third of spring until first third of autumn) over-estimation of HSI values is clearly apparent (Figure 5.18) and is probably due to high summer discharges resulting in important depths not suitable for *riffle* guild preferences. The high HSI values are produced in the winter, in-between hydropeaking events, where water levels are low so that the river depth permits a better fulfillment of the *riffle* guild preferences (i.e. shallow depths). Ecologically, and taking into account the uncertainty of this guild's

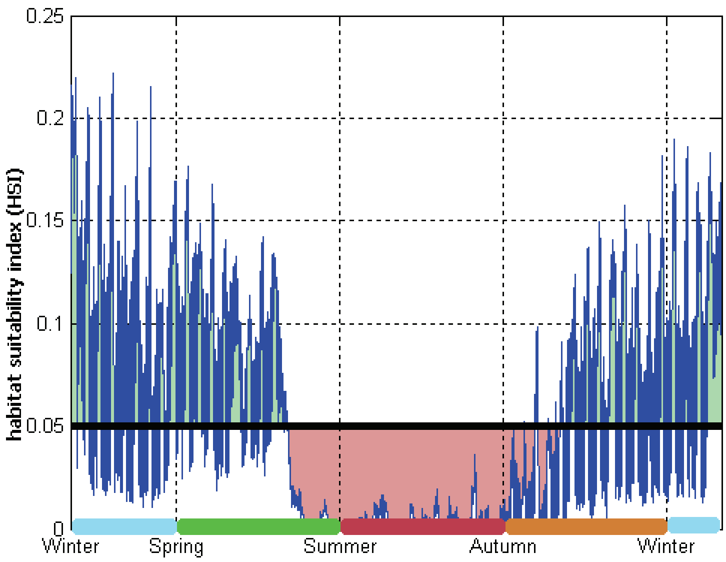


Figure 5.18: Scenario 2 - *Riffle* Guild: Effective habitat value for *riffle* guild under 1999 hydraulic conditions at hypothetical widened Riddes site (km 56.31). The black line represents the model habitat value, the green-filled area the susceptible under-estimated habitat suitability and the red-filled zone the clearly over-estimated habitat suitability

response time to sudden water variations, this implies that a hypothetically widened bed of the Swiss Upper Rhone River under actual hydrological conditions would not be very beneficial for the *riffle* guild. However, this result is debatable since the model scale does not take into account the increase in morphological diversity of the river bed and hence the habitat suitability at a local scale.

The *bank* guild has a HV of *ca.* 0.13 (see Figure 5.19, black line). There seems to be a seasonal effect, at least in terms of HSI variability. Once again, the high HSI values are observed to happen with the low water level in between hydropeaking events that suit better the *bank* guild preferences, mainly in terms of water depth and current speed. Again, cold-season (mid-autumn unitl mid-spring) HV under-estimation cannot be ruled out due to the issue of habitat temporal persistence effect on *bank* guild debated in section 5.6.2 p. 128. However, there is an apparent slight over-estimation of HV in the warm season (end of spring until beginning of autumn, see Figure 5.19, red-filled area). Ecologically, the bed-widening scenario seems

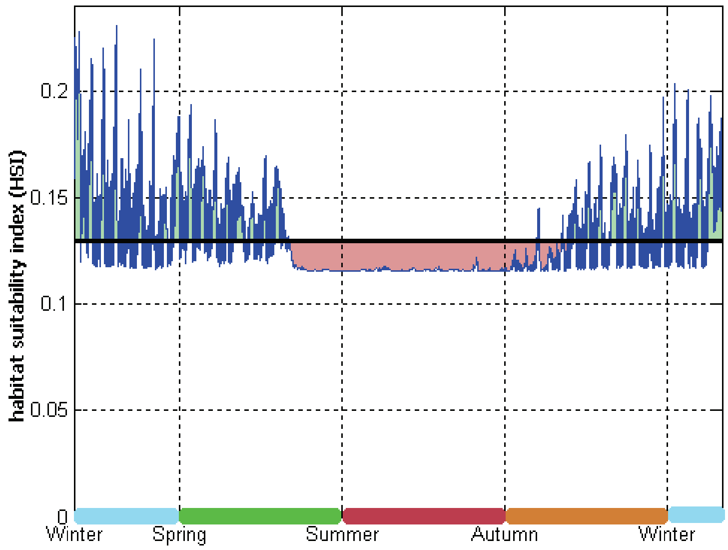


Figure 5.19: Scenario 2 - *Bank* Guild: Effective habitat value for *bank* guild under 1999 hydraulic conditions at hypothetical widened Riddes site (km 56.31). The black line represents the model aggregated habitat value, the green-filled zone the possible under-estimated habitat suitability and the red-filled zone the clearly over-estimated habitat suitability

to fulfill in an acceptable manner the *bank* guild criteria even if intuitively one would be expecting more from the benefits of a wide river, there is still a large amount of water transiting at high current velocities.

The *pool* guild has an HV of *ca.* 0.11 (see Figure 5.20, black line). There is an apparent seasonal effect, at least in term of HSI variation intensity, with a probability for cold-season (beginning of autumn until end of spring) under-estimation of HSI. Warm-season HSI (from end of spring until beginning of autumn) are clearly over-estimated (Figure 5.20, red-filled zone), where high discharge probably result in water velocities too high to fully suit the pool preferences (i.e. low water current velocity). High hourly HSI values are once again situated in between hydropeaking events in the cold season, where low water level still result in depth suiting the *pool* preferences and milder water current speed. Ecologically, a widened bed provides a limited satisfaction on

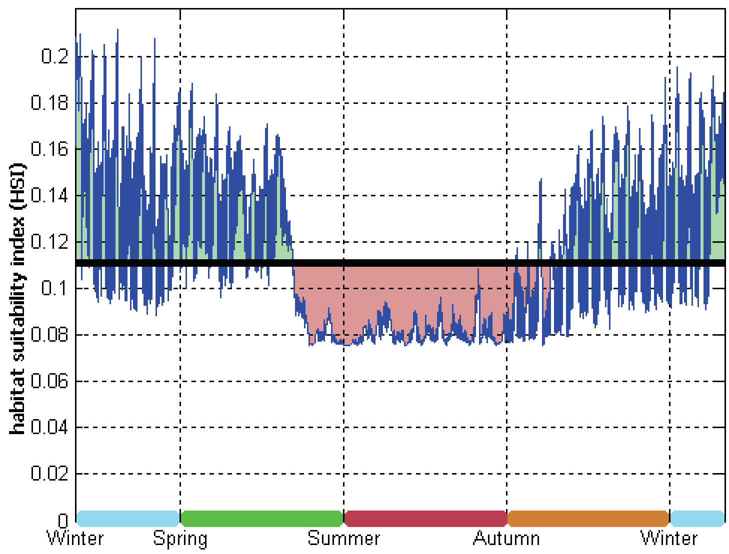


Figure 5.20: Scenario 2 - *Pool* Guild: Effective habitat value for *pool* guild under 1999 hydraulic conditions at hypothetical widened Riddes site (km 56.31). The black line represents the model habitat value, the green-filled zone the possible under-estimation of the habitat suitability and the red-filled zone the clearly over-estimated habitat suitability

the *pool* guild habitat availability.

The *midstream* guild has an HV of *ca.* 0.38 (see Figure 5.21, black line). There is a strong seasonal effect both in terms of HSI variation, with constant important HSI variations in the cold season caused by hydropeaking alone, and extreme warm-season HSI variations caused by snow-melt (i.e. climate), precipitations and to a lesser extent hydropeaking. The *midstream* guild preferences are met (i.e. deep water, high current velocity) in the cold season despite of hydropeaking events. The extreme discharges of the warm season even in the widened configuration result in very important depths coupled to extreme current velocities ($> 1.5 m \cdot s^{-1}$) decreasing HSI values (lowest HSI values observed during flood events). Ecologically, the *midstream* guild preferences are well represented in scenario 2.

Scenario 2 - Concluding remarks Morphologically and in terms of water quality there is a great potential for ecological integrity, which is probably justified by river development projects such as *La 3^{ieme} Correction du*

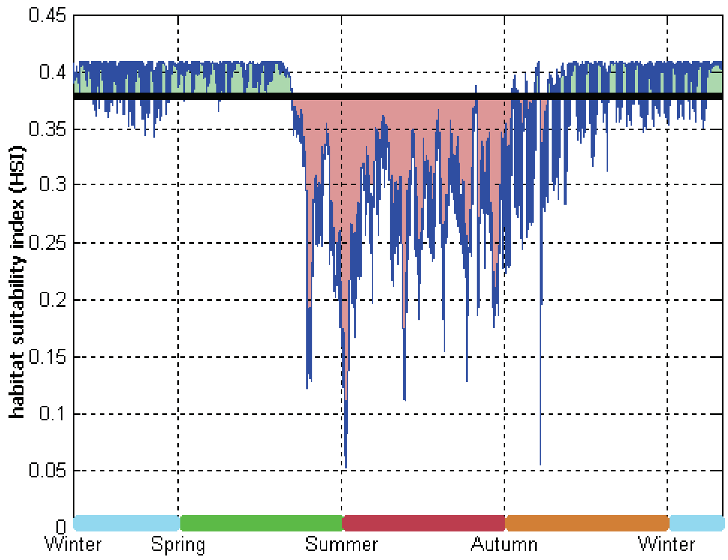


Figure 5.21: Scenario 2 - *Midstream* Guild: Effective habitat value for *midstream* guild under 1999 hydraulic conditions at hypothetical widened Riddes site (km 56.31). The black line represents the model habitat values, green-filled zone to habitat suitability under-estimation and the red-filled zone to habitat suitability over-estimation

Rhone (DTEE; 2004). Unfortunately, ecological integrity is governed and kept down by the extent of the hydropeaking disturbances that stresses immensely the system, with the exception of the *mainstream* fish guild, witch is still able to cope with the environmental conditions. These results truly highlight the necessity to 'deal' with hydropeaking, and especially in a context of bed widening. The model showed that a bed widening project alone would bring little to E-P-T group richness in a environment prone to hydropeaking. Fish communities would probably benefit to a greater extent, since more diverse habitats would be created (DTEE; 2004).

Scenario 3 - The `SYNERGIE` project basin

The `SYNERGIE` project basin (cf. Chapter 1.1 p. 15) solves the hydropeaking issue in the downstream river, and is susceptible of creating a new biotope[4]

[4]coined word proposed firstly in Germany (biotop) based on Greek bios='life or organism' and topos='place'. So biotope is literally an area where life living. More precisely, biotope is an area of uniform environmental conditions providing living place for a specific assemblage of plants and animals. Biotope is almost synonymous with the term habitat but while the subject of habitat is a species or a population, the subject of biotope is a community - `http://en.wikipedia.org/wiki/Biotope`

at the reservoir level. However, one must keep in mind that the longitudinal continuum may be slightly disrupted despite the artificial river and the fish ladder, mainly due to upstream increase in depths and decrease in water current velocities. In this particular scenario, we assumed this effect negligible in model calculation. The structural state of the downstream river remains monotonous with prominent embankment and virtually no bed diversity. It is therefore not surprising to see macro-invertebrates richness prediction indexes do well (index groups sensitive to hydrological conditions, especially Plecoptera and some Trichoptera (Bernard; 2001; Brittain and Salveit; 1989; Céréghino and Lavandier; 1998b)) and fish HV staying rather low except for the *midstream* guild and to a lesser extent the *pool* guild (caused by lack of shallow slow flowing zones (DTEE; 2004)).

Scenario 3 - EPT group prediction index The model averaged R_{EPT} prediction index is of 11.7 taxa. Plecoptera richness index (RE) is the highest, with an averaged model output of *ca.* 5.5 taxa (Figure 5.22, green band). Graphical interpretation of the model results show a slight seasonal effect with an increased and under-estimated predicted richness toward the warm seasons (end of spring and whole of the summer). A slight richness indicator over-estimation is observable in the cold season (autumn up to end of spring). In accordance to faunistic surveys (Baumann; 2004; ECOTEC; 1998, 1999, 2004; GIDB-R3; 2005; Gogniat and Marrer; 1984/85), and ecological affinities, the individuals present are most likely from:

- the Leuctridae family (*Leuctra sp.*)
- the Nemouridae family (*Nemoura mortoni*)
- the Taeniopterygidae family (*Rhabdiopteryx sp.*)
- the Perlodidae family (*Isoperla sp.*).

Individuals from the Capniidae family may also be found. This relatively high presence of Plecoptera indicates on one hand good water quality (IBGN GI values of 9 for Taeniopterygidae family (AFNOR; 1985)) and on the other hand good hydrological conditions since they are a good indicator of hydrological alteration (Bernard; 2001; Céréghino and Lavandier; 1998b).

Trichoptera richness index (RT) has the second highest predicted richness, with an averaged model output value of *ca.* 3.3 taxa (Figure 5.22, red band). Graphical interpretation of the model results show a very constant annual predicted richness above 1 taxa, which indicates that more sensitive taxa from families other than the generalist and hydropeaking resistant

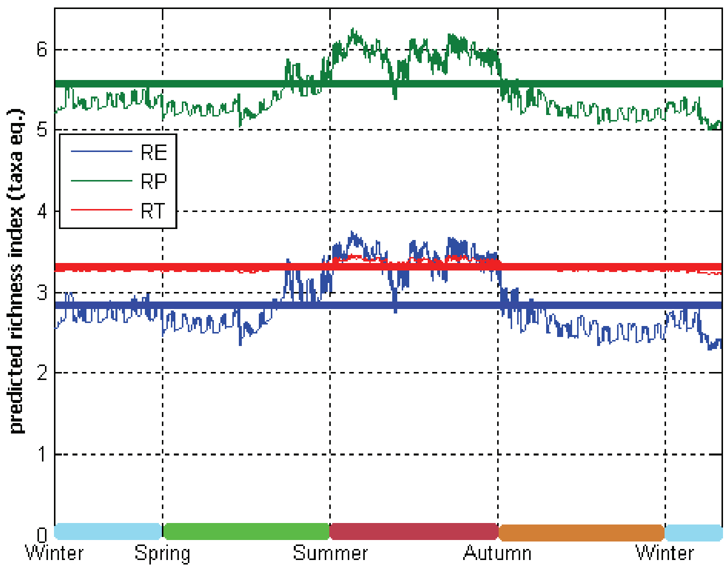


Figure 5.22: Scenario 3 - EPT group richness prediction indexes. Solid lines represent averaged model output values, RE stands for Ephemeroptera Richness eq., RP for Plecoptera Richness eq. and RT for Trichoptera Richness eq.

Limnephilidae (Frutiger; 2004b) such as the exigent predatory taxa from the Rhyacophilidae (Bernard; 2001) family are present. Limnephilidae are sensitive to hydropeaking and bed armoring and would probably be typical of a hydraulically impacted system (Bernard; 2001). Bernard (Bernard; 2001) reported a correlation between Rhyacophilidae abundance and benthic community richness, adding to the interpretation of good ecological integrity under our assumptions (ecological integrity being reflected by diversity). The following taxa can be expected:

- the Hydropsychidae family (*Hydropsyche sp.*)
- the Limnephilidae family (*Allogamus sp.*, *Halesus sp.*)
- the Rhyacophilidae family (*Rhyacophilia sp.*)

Ephemeroptera richness index (RE) has the lowest predicted richness, with a model output value of *ca.* 2.9 taxa (Figure 5.22, blue band). Graphical interpretation of the model results show a slight seasonal variation with a pattern similar to Plecoptera group. The individuals present are most likely from:

- the Baetidae family, namely *Baetis alpinus* or *Baetis rhodani*

- the Heptageniidae family, namely the hydropeaking sensitive *Rhitrogena sp.*

We can consider this R_{EPT} value of 11.7 to be comparable with the *Finges* site diversity, which is only slightly impacted by hydropeaking and has a rather acceptable structural diversity (Baumann; 2004). However, being close to 30 km downstream of the *Finges* site, we could expect a higher richness (Castella et al.; 2001). This predicted richness deficit could be explained by the monotonous structural diversity.

Scenario 3 - Fish guilds habitat values The *riffle* guild has an HV of *ca.* 0.03 (see Figure 5.23, black line). There appears to be a seasonal effect, with extreme variations in the cold-season without a clear probability of HSI under-estimation. The warm season HSI are clearly over-estimated, with much more constant low HSI values probably caused by the high depths of the important summer discharges not very suitable for *riffle* guild ecological preferences. The high HSI values are produced in the cold-season, where water levels are low so that the river depth allows for a better fulfillment of the *riffle* guild preferences. Ecologically, this response is coherent with common sense, since dealing with the effect of hydropeaking alone does not improve the structural variability of the Rhone which remains very monotonous and hence not fulfilling the *riffle* guild ecological preferences.

The *bank* guild has a HV of 0.12 (see Figure 5.24, black line). There seems to be a seasonal effect, not so much in terms of HSI values but rather in terms of HSI variations, which are more important in the cold-season and much more grouped in the warm-season. Once again, coupled to the morphological considerations of the Rhone, it is at much lower waters that HSI values are high for the *bank* guild, probably at levels where banks are not confined to artificial embankment anymore and where depth and current velocity are more suiting for the *bank* ecological preferences. Ecologically, this implies that hydropeaking is not the main driving factor for habitat value in such a morphological context and is not of major concern for the *bank* guild ecological preferences fulfillment.

The *pool* guild has a HV of *ca.* 0.09 (see Figure 5.25, black line). There seems to be a seasonal effect, both in terms of seasonal variability intensity and in terms of HSI values. Cold-season (mid-autumn up to mid-stream) HSI appear generally higher than HV (Figure 5.25, green-filled zone) while warm-season HSI (mid-spring until mid-autumn) appear generally lower than HV (Figure 5.25, red-filled zone). Cold-season HSI variability also appears much more variable, once again, probably due to the lower water corresponding better in terms of current speed to *pool* guild ecological preferences. Ecologically,

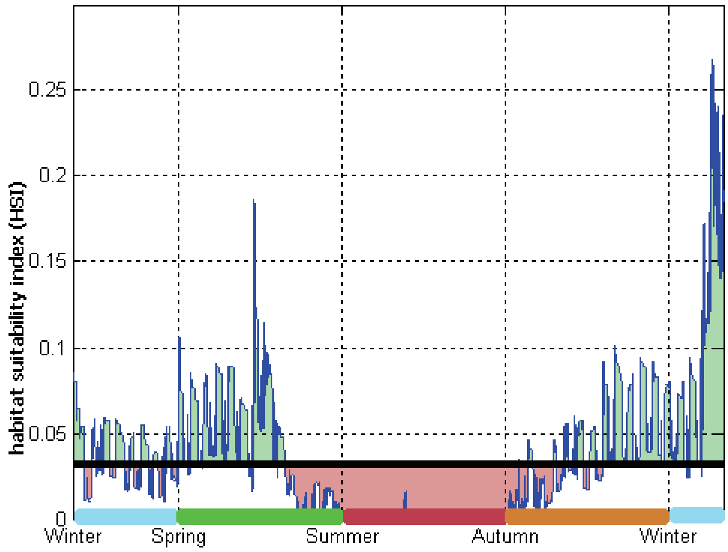


Figure 5.23: Scenario 3 - *Riffle* Guild: Effective habitat value for *riffle* guild under simulated hydraulic conditions (Heller; 2007) at the Riddes site (km 56.31). The black line represents the model HV, the green-filled area the susceptible under-estimated habitat suitability and the red-filled zone the clearly over-estimated habitat suitability

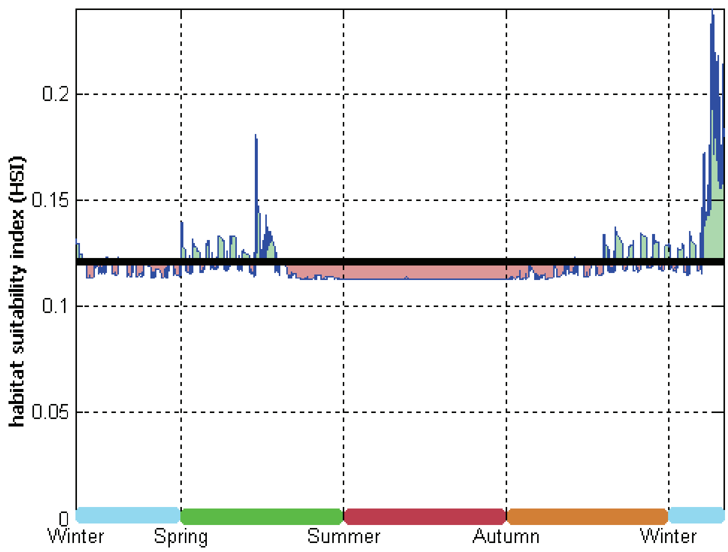


Figure 5.24: Scenario 3 - *Bank* Guild: Effective habitat value for *bank* guild under simulated hydraulic conditions (Heller; 2007) at the Riddes site (km 56.31). The black line stands for the model HV

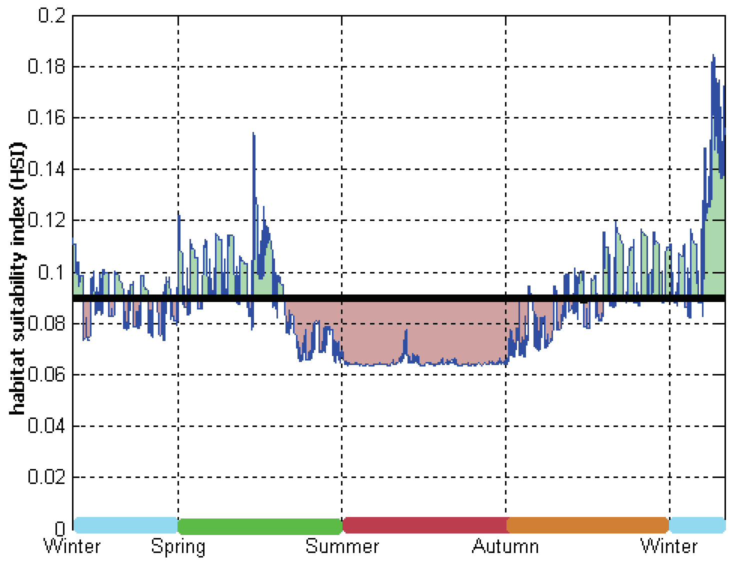


Figure 5.25: Scenario 3 - *Pool* Guild: Effective habitat value for *pool* guild under simulated hydraulic conditions (Heller; 2007) at the Riddes site (km 56.31). The black line the model aggregated HV, the green-filled zone represents the under-estimated habitat suitability while the red-filled zone represents the over-estimated habitat suitability

this implies that the structural component of the river is still the major driving factor of habitat value (DTEE; 2004), and that warm season high water velocities are not very suitable for the *pool* guild.

The *midstream* guild has a HV of *ca.* 0.36 (see Figure 5.26, black line). There is a strong seasonal effect, both in terms of HSI values and in terms of HSI variability. The cold-season (autumn until end of spring) HSI values are the highest, with a clear underestimation compared to the HV (Figure 5.26, green-filled zone) while the summer HSI a clearly over-estimated compared to the HV (Figure 5.26). Besides, warm-season HSI variability is much greater than cold season HSI variability. Ecologically, the *midstream* guild is the most suited for the Swiss Upper Rhone actual configuration and is accommodated by significant depth and water current velocities. However, summer hydraulic conditions are still extreme, probably causing both excessive depth and current velocities to fulfill the *midstream* guild preferences.

Scenario 3 - Concluding remarks Ecological integrity is still insufficient since the R_{EPT} index is similar to what is observed in literature *ca.* 30 km upstream by (Baumann; 2004; ECOTEC; 1998), and would be expected to be higher downstream (Castella et al.; 2001; Vannote et al.; 1980). This trend

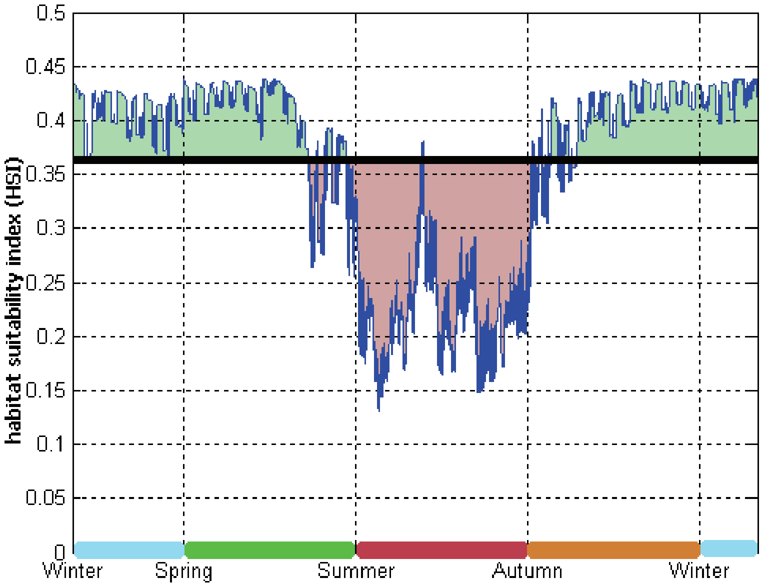


Figure 5.26: Scenario 3 - *Midstream* Guild: Effective habitat value for *midstream* guild under simulated hydraulic conditions (Heller; 2007) at the Riddes site (km 56.31). The black line the model aggregated HV, the green-filled area the underestimated habitat suitability and the red-filled zone the overestimated habitat suitability

is confirmed by the low HV values observed for the *riffle* and *pool* fish guilds caused by the inappropriate river bed morphology in its hydraulic context.

Scenario 4 - hypothetical bed widening coupled to SYNERGIE basin

The downstream river has a corrected hydrology, even if seasonality in terms of water quantities is not respected. The storage volume available does not permit the long term modification of the hydrogram, hence there is still more water in winter and less water in summer than historically due to dam operation and storage (see Figure 3.4). However, the structural diversification allows for the presence of a diverse benthic community and good ecological integration. Fish HV do not appear sensibly higher, but they have a greater potential surface to occupy.

Scenario 4 - The EPT richness indicator The model aggregated R_{EPT} prediction index is of *ca*. 15.1 taxa , which can be considered as good if compared to the upstream, and relatively pristine *Finges* site, where a maximal of 11 EPT taxa were observed (Baumann; 2004). Plecoptera richness index

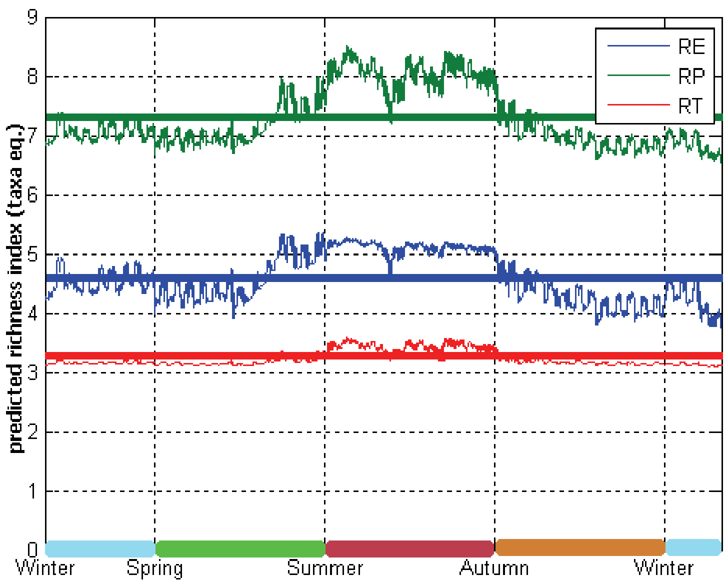


Figure 5.27: Scenario 4 - EPT group richness prediction indexes. Solid lines represent averaged model output values, RE stands for Ephemeroptera Richness eq., RP for Plecoptera Richness eq. and RT for Trichoptera Richness eq.

(RP), has the highest model richness indicator of *ca*. 7.3 taxa (Figure 5.27, green band). Graphical interpretation of the model results indicate a slight seasonal effect yielding a possible over-estimation of the richness indicator result in the cold-seasons (autumn until mid-spring) and a possible under-estimation of the richness indicator result in warm-season (end of spring and whole summer). Highest richness indexes seem to be predicted in the end of summer, witch is close to observations reported in literature for the Swiss Upper Rhone River (Bernard; 2001). In accordance to taxonomic surveys (Baumann; 2004; ECOTEC; 1998, 1999, 2004; GIDB-R3; 2005; Gogniat and Marrer; 1984/85), the stoneflies susceptible to be encountered are most likely:

- the Taeniopterygidae family (*Rhabdiopteryx neglecta, Rhabdiopteryx sp.*)
- the Nemouridae family (*Nemoura mortoni, Protonemoura sp.*)
- the Leuctridae family (*Leuctra handlirschi, Leuctra sp.*)
- the Capniidae family (*Capnia nigra, Capnia sp.*)
- the Perlodidae family (*Isoperla rivulorum, Dictyogenus alpinus*)

Perlodidae are very sensitive to water quality (IBGN GI value of 9 (AFNOR; 1985)) and hydrological impacts (i.e. hydroepaking) (Céréghino and Lavandier; 1998b; Céréghino et al.; 2002). Their diversity is a good cue of good ecological integrity under our assumptions.

Ephemeroptera richness index (RE) has the second highest model output richness indicator of *ca.* 4.6 taxa (Figure 5.27, blue band). Graphical interpretation of the model results indicates a very slight seasonal effect resulting in a possible over-estimation of the richness indicator result in the cold-seasons (beginning of autumn until mid-spring) and a possible under-estimation of the richness indicator results in the warm-season (end of spring, whole of summer). In accordance with reported taxonomic surveys, the mayflies susceptible of being encountered are most likely:

- the Baetidae family, (*Baetis alpinus, Baetis rhodani, Baetis sp.*)
- the Heptageniidae family (*Ecdyonurus sp., Rhitrogena degrangei, Rhitrogena sp.*)

Trichoptera richness index (RT) has the lowest model output HV of *ca.* 3.2 taxa (Figure 5.27, red band). Care must be taken when attempting to interpret its HV since the model for caddisflies richness prediction was weaker (see p. 91). However, the threshold value of 1 taxa represented by the generalist and imopact tolerant Limnephilidae family has been outreached and other more sensitive caddisflies are predicted. In accordance to reported faunistic surveys, taxa susceptible of being encountered are most likely:

- the Hydropsychidae family (*Hydropsyche sp.*)
- the Rhyacophilidae family (*Rhyacophila stricto sensu sp.*)
- the Limnephilidae family (*Allogamus auricollis, Allogamus sp., Halesus sp.*)

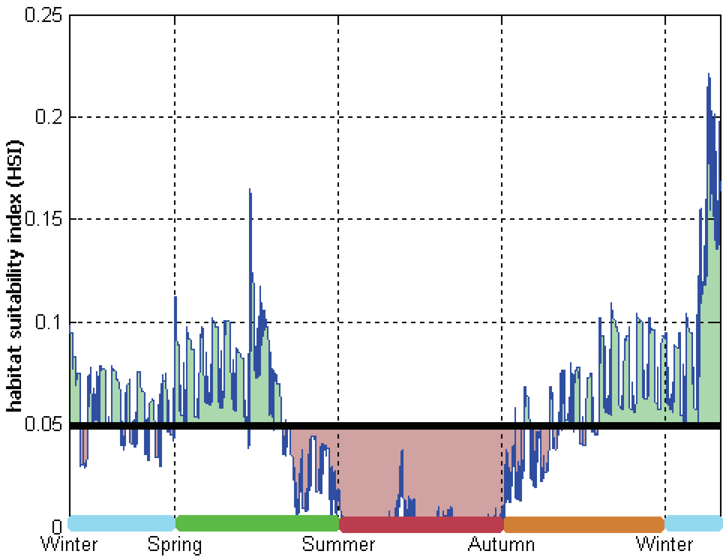


Figure 5.28: Scenario 4 - *Riffle* Guild: Effective habitat value for *riffle* guild under simulated hydraulic conditions (Heller; 2007) and a hypothetically widened bed at the Riddes site (km 56.31). The black line represents the model aggregated HV

Scenario 4 - Fish guild habitat values The *riffle* guild has a HV of *ca.* 0.05 (see Figure 5.28, black line). There appears to be a seasonal effect, both in terms of hourly HSI values and variation intensity. Cold-season variations (mid-autumn up to mid-spring) are more pronounced and a slight under-estimation of HV cannot be ruled-out (Figure 5.28, green-filled zone) while a warm-season (mid-spring up to mid-autumn) HV over-estimation is apparent (Figure 5.28, red-filled zone). Yet again, the high summer flows, regardless of the bed widening and the hydropeaking mitigation do not seem very favorable in terms of the water depth and current speed preferences of the *riffle* guild. Ecologically, hydropeaking mitigation coupled to the described bed widening described on Table 5.4 p. 124 does not seem sufficient to fulfill in acceptable way the ecological preferences of the *riffle* guild.

The *bank* guild has a HV of *ca.* 0.13 (see Figure 5.29, black line). There appears to be a seasonal effect in term of hourly HSI variation extent, that are much more pronounced during the cold seasons (mid-autumn up to mid-spring, Figure 5.29, green-filled zone). Ecologically, the hydropeaking mitigation coupled to bed widening seems to provide some habitat fulfilling the ecological preferences of the *bank* guild.

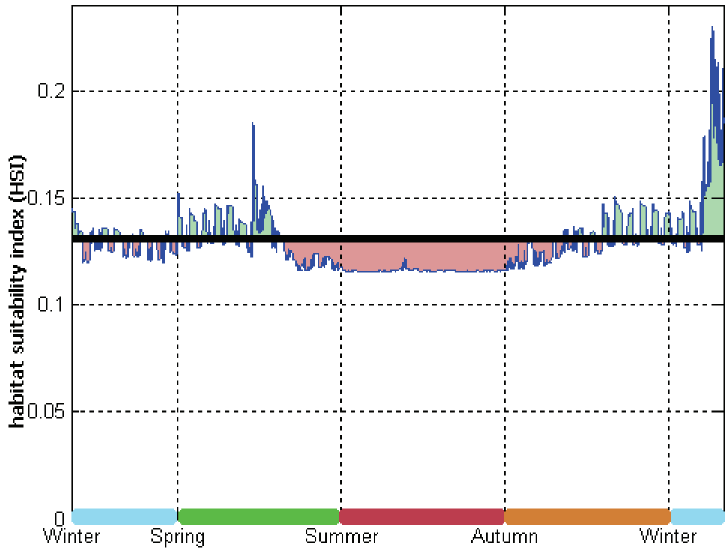


Figure 5.29: Scenario 4 - *Bank* Guild: Effective habitat value for *bank* guild under simulated hydraulic conditions (Heller; 2007) and a hypothetically widened bed at the Riddes site (km 56.31). The black line represents the model averaged HV

The *pool* guild has a HV of *ca.* 0.11 (see Figure 5.30, black line). There is an apparent seasonal effect, both in terms of hourly HSI variability and HV estimations. Cold-seasons (beginning of autumn up to the end of spring) HSI variability is well pronounced and HV under-estimation cannot be ruled out (Figure 5.30, green-filled zone) while warm-seasons HSI variations are grouped and slightly over-estimated by the HV (summer, Figure 5.30, red-filled zone). Once again, this seasonal pattern is due to the high discharges yielding high current velocities – even in a wide configuration such as described on Table 5.4 p. 124 – which are not fully fulfilling the *pool* guild ecological preferences. Ecologically, the hydropeaking mitigation coupled to the bed widening seem to allow limited quantity of the *pool* guild habitat.

The *midstream* guild has a HV of *ca.* 0.38 (see Figure 5.31, black line).

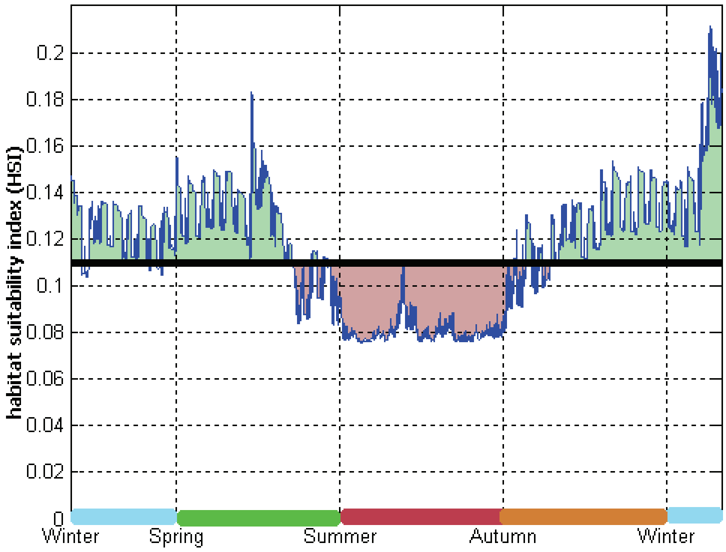


Figure 5.30: Scenario 4 - *Pool* Guild: Effective habitat value for *pool* guild under simulated hydraulic conditions (Heller; 2007) and a hypothetically widened bed at the *Riddes* site (km 56.31). The black line represents the model averaged HV

There is an important seasonal effect, both in terms of hourly HSI variability and possibly in HV estimation. Cold-season (autumn up to the end of spring) variation are grouped and HV is possibly slightly under-estimated (Figure 5.31, green-filled zone). Warm season (end of spring up to summer) variations are more pronounced and HV seems over-estimated (Figure 5.31, red-filled zone).

Scenario 4 - Concluding remarks The ecological integrity of the system is high, mainly for macro-invertebrate taxa compared to what has been reported in literature (Baumann; 2004; GIDB-R3; 2005). The rather low values obtained for the fish HV can be explained by the extent of the bed widening, which given the observed high water velocities and depth during the warm-season are not fulfilling the ecological preferences of the *riffle* guilds, only partially fulfill the preferences of the *bank* and *pool* guild and are satisfactory for the *midstream* guild. Under this type of scenario, specially designed shallow and rapid zone should be planned in order to account for the overall deficiency of such indispensable structures (DTEE; 2004). This scenario yields the highest ecological integrity of the variants.

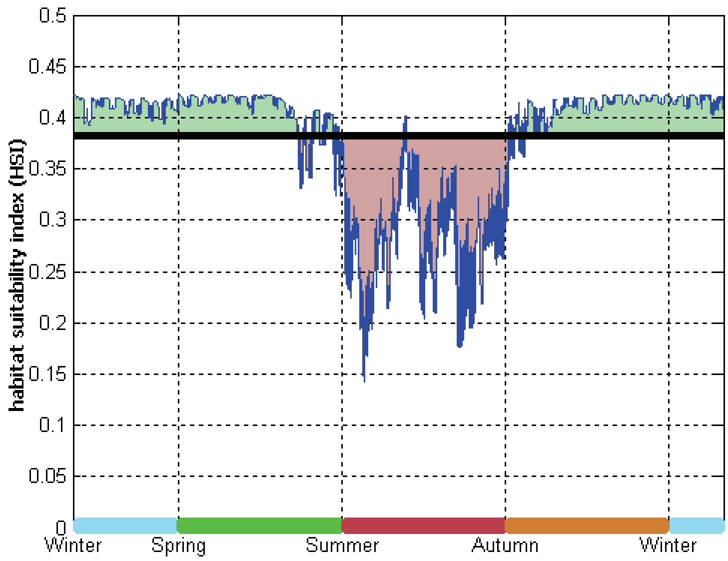


Figure 5.31: Scenario 4 - *Midstream* Guild: Effective habitat value for *midstream* guild under simulated hydraulic conditions (Heller; 2007) and a hypothetically widened bed at the Riddes site (km 56.31). The black line represents the model averaged HV

5.6.3 Inter-scenario analysis

EPT - group richness indicator - general observations

The cold season variation intensities for the RE indicator are greatly smoothed by the `SYNERGIE` basin (see Figure 5.32), which further strengthens the ecological pertinence of the Scenario 4 through the RE index: the hourly variations being much less consequent in both seasons. In Scenario 1, there is a clear $5 + 2$ pattern in the cold-season RE closely linked to the week days subjected to hydropeaking and the week-end days having no hydropeaking. These variations have such a short frequency that by precaution a careful ecologist should be tempted to use a 7-day, or even a 15-day mobile minima to smooth and assess the richness index rather than an average value such as stated in the method. A similar smoothing by minimas on Scenario 4 would have much less impact on the overall results and be much closer to the model predicted richness value output. Exactly the same pattern has been observed on Plecoptera and Trichoptera, so it is probable that *Scenario 1 and 2 (no hydropeaking mitigation, high daily predicted richness indexes variability) richness prediction indexes may be boosted by the very short frequency daily high outputs resulting in limited ecological pertinence.* Smoothing by moving minimas was not performed in the assessment techniques due to lack of knowledge on the response time specific to each E, P and T groups. In other words, it was impossible to decide on the time span used for result smoothing (7-day, 15-day or more).

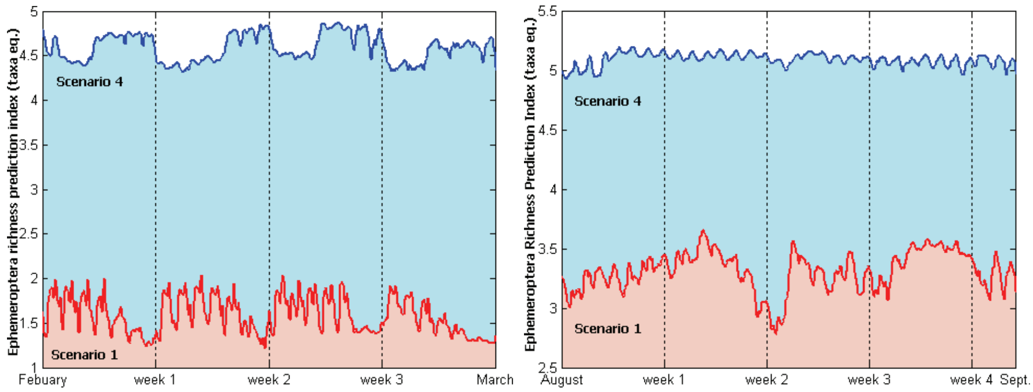


Figure 5.32: Inter-scenario analysis: Group Ephemeroptera, Scenario 1 indexes are represented by the red-filled zone while Scenario 4 indexes are represented by blue-filled zone, Months of Febuary (left) and August (right) are represented

Field calibration of the model would probably enable us to have a clearer idea of the smoothing parameter to use for each macro-invertebrate group. It is probable that water temperature and time of the day may affect the response of organisms to flow variations, which was already reported in literature for fish (Halleraker et al.; 2003; Valentin; 1995).

Ephemeroptera

Little comparison to literature findings is available since the effects of seasonality on hydropeaking have not to my knowledge been reported in literature. The following observations should therefore be considered hypothesis yet remaining to be confirmed and would in my opinion offer great perspectives for further research.

Mayflies predicted richness index is the highest for the bed widening coupled to hydropeaking mitigation variant (Scenario 4, see Figure 5.33, green band). Mayflies appear to be affected both by the bed structure and by hydropeaking: in cold seasons hydropeaking seems to be the limiting factor for mayflies (Figure 5.33, red and black lines), where regardless of the structural aspects, richness indexes are dragged down. A structural limitation is still observed (Figure 5.33, green line vs. blue line comparison). In the warm seasons however, there is clearly a liming effect induced by the structural aspect of the river (see Figure 5.33, blue and red lines comparison and green and black lines comparison). Hydropeaking mitigation seems to have little effect on mayflies in the warm seasons.

It is also of some interest to observe the effect of bed widening without hydropeaking regulation on the short term-amplitude index variations (Figure 5.33, black line), which depending on the response time of mayflies may strongly limit the effect of bed widening without hydropeaking mitigation in the cold seasons.

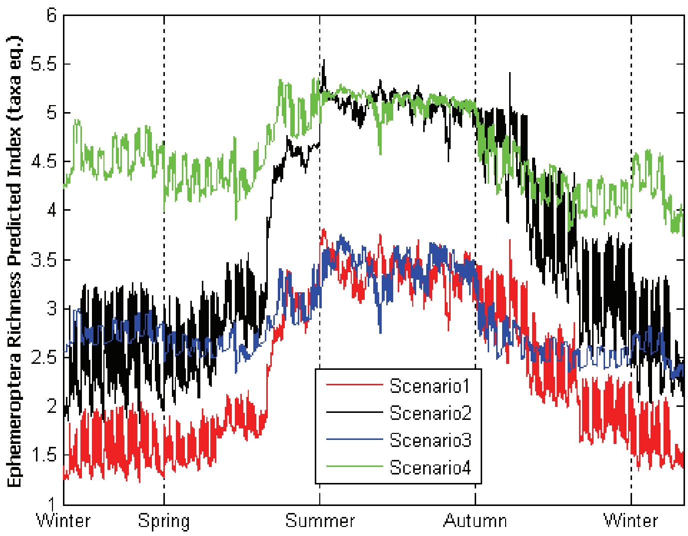


Figure 5.33: Ephemeroptera results for Scenario 1 (red), Scenario 2 (black), Scenario 3 (blue) and Scenario 4 (green)

Plecoptera

There again, to my knowledge, little comparison with literature findings are available on the effects of seasonality and bed structure on stoneflies. The following observations are therefore to be considered as hypotheses remaining yet to be field tested.

Stoneflies predicted richness index is the highest for the bed widening coupled to the hydropeaking mitigation variant (Scenario 4, see Figure 5.34, green band). The overlapping of the scenarios subject to hydropeaking is absolutely striking (see Figure 5.34, red (Scenario 1) and black (Scenario 2) lines). In hydropeaking conditions, it seems that bed widening has a null effect on the Plecoptera richness. This is in concordance with what has been reported in literature (Bernard; 2001; Céréghino and Lavandier; 1998b; Céréghino et al.; 2002) which state that Plecoptera require not only good water quality but good hydrological conditions. This group appears as the most suitable indicator for hydropeaking assessment. A structural effect has a significant influence on stoneflies providing hydropeaking is buffered (see Figure 5.34, blue (Scenario 3) and green (Scenario 4) lines), which really highlights this taxonomic group as the predilection indicator group for headwater river systems. Unfortunately, I would recommend to limit the use of Plecoptera as a hydrological and morphological indicator group in mainstream zones due to its headwater predilection and its decreasing richness in mainstream systems (Tachet et al.; 2000).

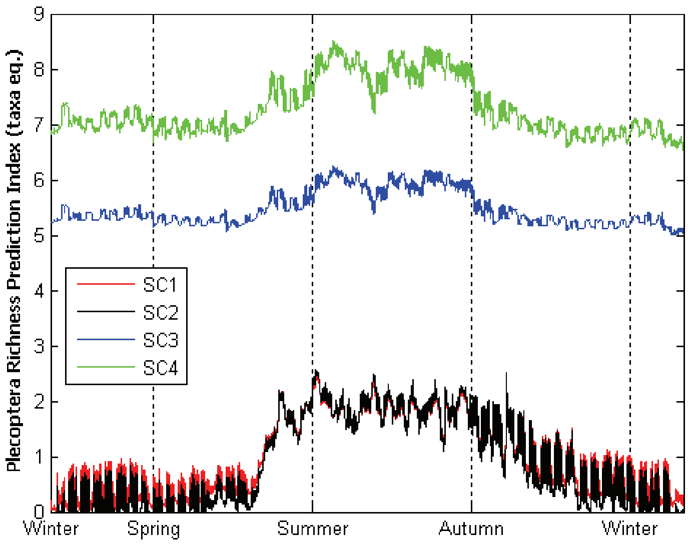


Figure 5.34: Plecoptera results for Scenario 1 (SC1-red), Scenario 2 (SC2-black), Scenario 3 (SC3-blue) and Scenario 4 (SC4-green)

Trichoptera

To my knowledge, little comparison with literature findings are available on the effects of seasonality and bed structure on caddisflies. The following observations are therefore to be considered as hypotheses remaining yet to be field tested. It is also of importance to keep in mind that the prediction model for Trichoptera richness is weaker than the Ephemeroptera richness prediction model or the Plecoptera richness prediction model (see section 4.2 p. 78).

Caddisflies predicted richness index is the highest for the bed widening coupled to the hydropeaking mitigation variant (Scenario 4, see Figure 5.35, green band). The impact of hydropeaking mitigation is still evident though (see Figure 5.35, red (Scenario 1) and black (Scenario 2) lines vs. the blue (Scenario 3) and green Scenario 4 lines). In cases subjected to hydropeaking, it seems that only member of the Limnephilidae family, which are not affected by bed armoring induced by hydropeaking or bothered by sudden water level variations thrive (Bernard; 2001; DTEE; 2004; Frutiger; 2004b). In cases where hydropeaking is buffered, other families seem able to live, which with reserves, points toward a possible use of this taxonomic group's richness as an indicator group for hydrological alteration assessment. In my opinion, more research on the effects of hydropeaking and morphological alteration on this taxa is necessary. Trichoptera may be the group susceptible of being used as an hydrological and morphological indicator in the mainstream zone of rivers, where Plecoptera indicators will not be usable.

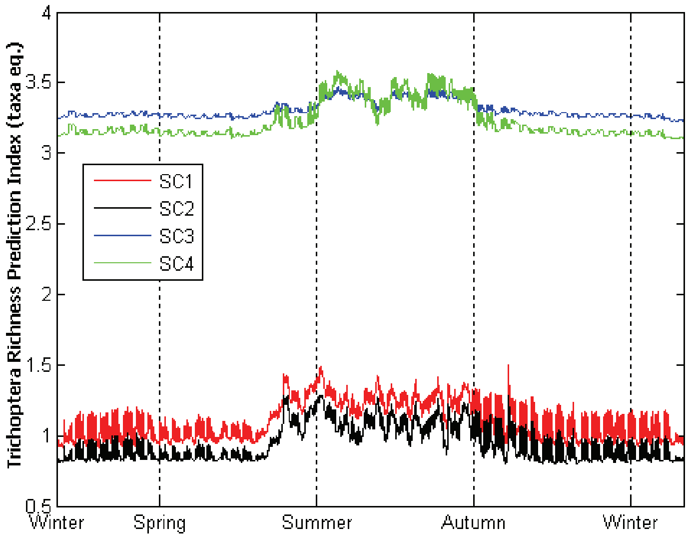


Figure 5.35: Trichoptera results for Scenario 1 (SC1-red), Scenario 2 (SC2-black), Scenario 3 (SC3-blue) and Scenario 4 (SC4-green)

Fish Guild Habitat Values - general observations

A common way of interpreting the habitat values of fish guilds is to convert the HV grade in a Weighted Usable Area (WUA), expressed in ha (Bovee; 1982; Ginot; 1998; Lamouroux and Jowett; 2005; Lamouroux and Souchon; 2002; Pouilly et al.; 1995). For each guild, its WUA is determined as follow:

$$\mathrm{WUA_{Guild}} = \frac{1}{\mathrm{N}} \sum_{\mathrm{i}=1}^{\mathrm{N}} \left(\mathrm{Width_i} \cdot \mathrm{HSI_i^{Guild}} \cdot \mathrm{L}\right)$$

; $\mathrm{WUA}^{\mathrm{Guild}}$ – Weighted Usable Area of each Guild $[m^2]$
; N – number of hours (8760)
; $\mathrm{Width_i}$ – bed width at i^{th} hour
; $\mathrm{HSI_i^{Guild}}$ – habitat suitability index of the guild at the i^{th} hour
; L – reach length $[m]$

In our case, we set the reach length (L) to 56 000 m. Which corresponds roughly to the distance from the Riddes site to the Geneva lake. WUA are computed for each guild in each scenario (see Figure 5.36) This approach

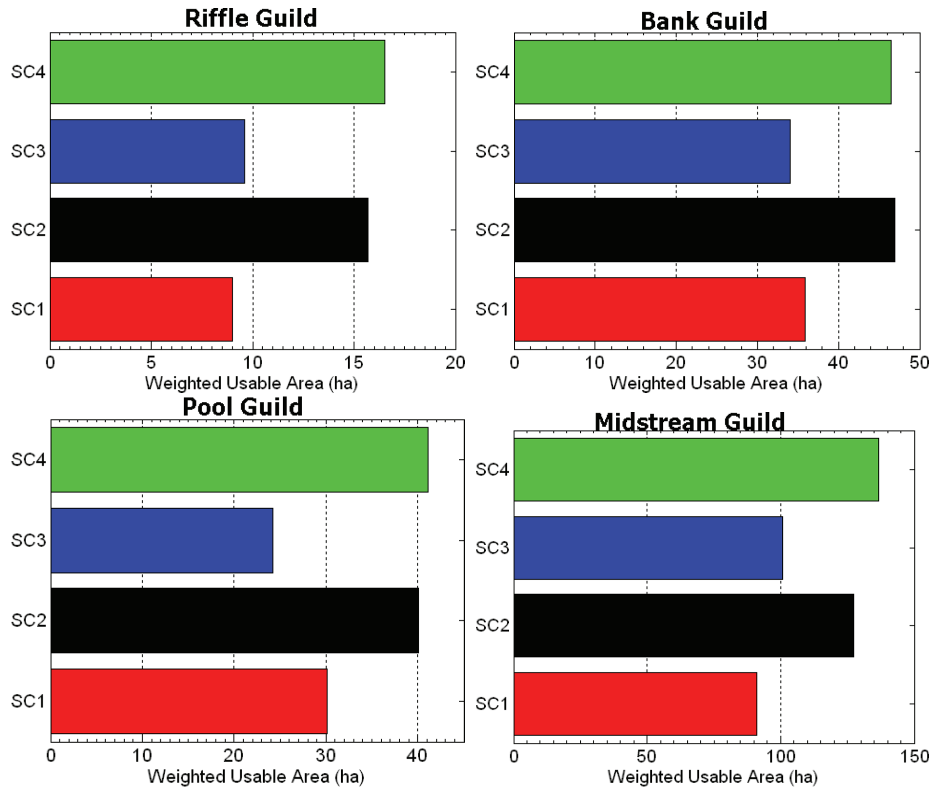


Figure 5.36: Fish guilds Weighted Usable Areas (ha) results for Scenario 1 (SC1-red), Scenario 2 (SC2-black), Scenario 3 (SC3-blue) and Scenario 4 (SC4-green)

clearly highlights the advantages of river bed widening, by providing more guild-specific WUA, even if HV grade remains the same. Hydropeaking mitigation alone (Scenario 3) seems to lower the *bank* and *pool* guilds WUA. This is explained by the more constant flow induced by hydropeaking mitigation and the lack of these in-between peaking event conditions favorable to these two guilds, namely lowered depth and lowered water current velocities.

Chapter 6

Conclusion and Perspectives

6.1 Conclusion

The answers to the scientific questions stated in Chapter 2.5 are:

1. *simple ecological considerations are sufficient to support and boost ecological integrity at the project's reservoir*

   The *obstacle effect* is reported in literature as the major cause of negative effect on the system ecology caused by reservoir. Organisms (i.e. fish and macro-invertebrates in their aquatic phase) are prevented from migrating upstream or downstream. By implementing a *nature-like* artificial river coupled to a fish ladder, the obstacle effect is mitigated for both macro-invertebrates and fish. Besides, the artificial river is susceptible of having a local attractiveness for aquatic macrophytes and associated fauna (i.e. birds and mammals, see Figure 3.8). Another effect consequent of river regulation (training and embankment) is the lack of transitional zones between the aquatic and terrestrial media. By implementing *artificial ecotonal zones* in the reservoir (see Figure 3.6) it should be possible to boost ecological processes specific to transition zones by providing them space and limiting stress caused by daily water fluctuations. These structure also have the advantage of enhancing the landscape integration of the reservoir by maintaining a contact to water in a strongly fluctuating environment. A *sanctuary zone* with limited public access could provide shelter for more sensitive species of birds or mammals. These few simple ecological considerations are expected to contribute positively to the ecological integrity of the reservoir.

2. *ecological response can be modeled at the downstream river scale to assess the ecological integrity following a river development project*

Models assessing the ecological response at the downstream river were used. These models are function of morphological and hydrological parameters of the system and provided insights on the ecological response of the system based on factors directly affected by river development projects (i.e. hourly discharge, bed width, etc.)

3. *macro-invertebrates richness, and more precisely Ephemeroptera, Plecoptera and Trichoptera (EPT) taxonomic groups as an aggregated index is a good ecological indicators*

   Macro-invertebrate richness model selection yielded acceptable predictability for Ephemeroptera and Plecoptera Richness prediction but rather poor predictability in Trichoptera richness prediction. For Trichoptera richness prediction in the Swiss Upper Rhone River our worst prediction threshold (mean) could not be outperformed, indicating a very limited use of this group as an indicator for this particular system. Literature further confirmed that in particular Plecoptera richness could be used as a good ecological integrity measure especially in headwaters. It appears that EPT taxonomic groups richness as an aggregated index is a good ecological indicator as a function of hydrology and morphology. In headwaters however, using Plecoptera taxonomic group alone as an integrator would be more ecologically pertinent for the measure of ecological integrity

4. *the hour is an adequate time step to predict annual EPT richness dynamics*

   Hydrological variations in the Swiss Upper Rhone River are such that it is necessary to have response model based on an hourly time-step. Even if the ecological sense of a predicted EPT richness at an hour i is unreasonable, it serves well as a computational basis in the overall measure of the annual or even seasonal ecological integrity

5. *fish guilds (riffle, bank, pool and midstream) habitat suitability is a good ecological indicator*

   Great care must be observed when attempting to translate the ecological significance of fish guilds habitat values (HV) in terms of a measure of ecological integrity, and hence they have a limited value as ecological indicators for the Swiss Upper Rhone River system. Fish guild HV should be taken as insights or trends but in the actual state of knowledge should not be used directly as ecological indicators. Too many uncertainties remain. Fish were reported to have a behavioral response to hydropeaking effect and a quasi instantaneous response capability to an hydropeaking event. This capability is guessed to be guild specific and not necessarily related to the guilds chosen in our work (i.e. bank, riffle, pool and midstream). Fish guilds habitat value (HV) should be taken as leads for further research perspectives as ecological indicators rather than ecological indicators *per se* of the system's ecological integrity

6. *the hour is an adequate time step to predict annual habitat suitability index dynamics*

   Having stated the limits of the use of fish guilds as ecological indicators, it is hard to say whether the hour is an adequate time step to predict annual habitat suitability index dynamics. It would make sense that in order to integrate rapid hydrological variations (i.e. hydropeaking) in the implementation of fish guilds as potential bio-indicators of ecological integrity, the necessary time step would be in the order of the hour

7. *the ecological response is function of system hydrology and river morphology*

   The expression of hydrological effects is strongly dependent on morphology. A wide and shallow river with smooth banks will be much more affected by a fall in water levels than a trapezoidal deep and narrow river. In the wide and shallow river, large bands may be left exposed and affected by atmospheric conditions, while in a narrow and deep river the effects will be less and little substrate will be left exposed (see Figure 2.6). Hence it appears clearly that these two aspects have to be taken into account jointly when attempting to assess the ecological response of a system following a river development project design or management

8. *it is possible to improve the ecological integrity of the current regulated Swiss Upper Rhone River with a river development project*

   Results for scenarios 2,3 and 4 clearly show that it is possible to improve the ecological integrity of the current regulated Swiss Upper Rhone River with a river development project, both in terms of EPT richness prediction and fish guild habitat values

9. *the ecological response of the downstream Swiss Upper Rhone River differs following various river development project scenarios and can be maximized*

All scenarios have different ecological implications, hydropeaking mitigation (SYNERGIE project) seems to be more beneficiary to macro-invertebrates while river bed widening scenario seems more beneficiary for currently poorly represented fish guild habitats (bank and riffle). The maximization of ecological integrity measure being reached when the reduction of hydropeaking event is linked to a bed widening of the river (scenario 4). This end result is in total coherence with the identified causes of ecological deficits in the Swiss Upper Rhone River, namely channelization and hydropeaking hydrogram, that once mitigated should increase the ecological integrity of the system

6.2 Perspectives

Many perspectives lay ahead of the SYNERGIE innovative optimization method where the importance of ecology is raised at the project conception phase. The SYNERGIE project offers a solution to all hydropeaking hydro-electrical plant releasing water in the Rhone wishing or having to reduce their environmental impact on the Swiss Upper Rhone.

Focusing on ecological considerations, perspectives are lying on further understanding of the hydropeakaing effects on aquatic organisms and means of quantifying the hydropeaking effect taking into account river morphology. Research on hydropeaking effect on fish and further understanding of this effect is required in order to measure the ecological response following a river development project. Fish are easy to catch and identify, are good indicators of water quality, have a strong political value and their ecology is well documented in comparison with macro-invertebrates. Their weak point as an indicator might be their behavioral ability to respond to hydropeaking events which may complicate the assessment of the hydropeaking effect they are subject to. The macro-invertebrate models have partial mathematical validation (general cross validation) but should nevertheless be field validated. The fish habitat value models have been validated by their authors but a validation on the Swiss Upper Rhone River is still to be recommended. Our models should also be field calibrated, and a sensitivity analysis should then be made. In my opinion, the actual knowledge on hydropeaking is not sufficient to clearly state recommendations on hydropeaking tolerance in the Swiss Upper Rhone River and this major issue holds great potential for further research.

Appendix: Determination of HS (covariate x_6) and HL (covariate x_9) hydropeaking integrators

A need for a coherent hydraulic indicator of hydropeaking conditions at various temporal scales was needed in order to attempt assessment. A simple method was applied at two temporal scales. HS, or the *short response hydropeaking integrator* is at a bimonthly scale (set at 60 days – 1440 hours) and determined hourly. The HL, or the *long response hydropeaking integrator* is at an annual scale (set at 8760 hours). The use of such a bimodal hydropeaking integrator was justified by lack of knowledge about the true hydropeaking effect resilience of macro-invertebrate communities as well as the important variations in hydropeaking amplitudes between cold and warm seasons. The HL factor would provide an insight on the overall conditions while the HS factor would target more specifically the seasonal conditions.

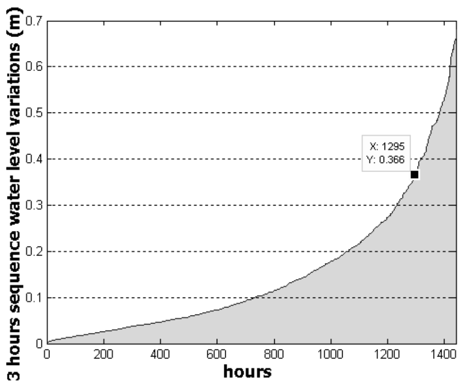


Figure 6.1: HS integrator: 3 hours sequence water level variation for hour 00 of the 01.12.1999

Similarly, historical discharges (year 1907) of the Swiss Upper Rhone River at *La Porte du Scex* gouging station were analyzed in order to attempt looking for what could be considered *natural* daily water level variations and hence provide a daily amplitude threshold recommendation based on natural water level variation.

In order to determine hourly HS, the preceding 1440 hours 3 hours sequence water level variations are ranked, and their 90^{th} percentile (corresponding roughly to the curve inflexion point) was set as the HS hydropeaking integrator value. A time step of 3 hours was chosen because it integrates the consequence of the event at the ecological scale (e.g. an organism stranded by an event has a significant mortality probability after a 3 hour dessication or exposure to cold). The 90^{th} percentile of these 3 hours water sequence variation was chosen because of its representativity of the wide majority of events leaving out really extraordinary occurrences. This was assumed to provide a good representation of overall hydropeaking conditions (see Figure 6.1) as our HL integrator. The same percentile is applied on each annual ranked 3 hours sequences water level variations.

The long response hydropeaking indicator (HL) was determined a similar way (see Figure 5.2.1 on page 113). For the annual 8760 hours, 3 hours sequence water level variations were ranked. Graphical interpretation of the 1907 historic 3 hour sequence water level variation curve at *La Porte du Scex* gouging station resulted in the choice of the 91.324 percentile of ranked 3 hour sequences water level variations.

The choices of the percentile values are highly debatable, and the study of the effects on the results based on these choices has not been conducted and should provide better indications for future guidelines.

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Objectives

Marine and Freshwater Biologist, with a Master in Freshwater Ecology and experience in multi-disciplinary projects, model selection and data analysis. My objective is to put my experience to use in fields of sustainable multi-purpose project development and prospection.

Current Activity

2004–2007 **Environmental scientist - PhD student/teacher assistant** at the Swiss Federal Institute of Technology, Lausanne. Laboratory of Ecological Systems
Thesis directors: Prof. Rodolphe Schlaepfer, Prof. Alexandre Buttler
PhD Subject: Ecological Integration in a Multi-Objective River Development Project. Project led by Prof. Anton Schleiss, Laboratory of Hydraulic Constructions (LCH), Swiss Institute of Technology, Lausanne
Activities: responsible for the ecological pole of the `SYNERGIE` project. My tasks consisted to provide solutions maximizing the ecological added value for a multi-objective project. Assessment of hydrologic and morphologic effect and their mitigation at the river scale and local ecological integrity improvement at the scheme's local scale. Also gave some classes on river quality assessment
Achievements: implementation of a decision making tool for ecological response assessment in a river development project capable of being implemented in a multi-objective optimizer. Publications (3) and Conferences (2)

Academic Cursus

2007	Ph.D. *ès* Sciences – Swiss Institute of Technology, Lausanne
2004	M.S. in Natural Sciences of the Environment – *Diplôme en Sciences Naturelles de l'Environnement* – University of Geneva, Switzerland
2001	Post-implementation environmental impact study of two aquaculture facilities in the *Ria Formosa* lagoon. Under the supervision of Prof. Sofia Gamito, Universidade do Algarve, Faro, Portugal
2000	B.S. in Biology: Marine and Freshwater. University of New Hampshire, U.S.A.
1995	French Baccalauréat (Scientific mention), *Lycée A. D'Orbigny*, La Paz, Bolivia

Relevant professional experience

2006	responsible of the ENAC week river quality assessment module and fieldwork
2004 - present	scientific collaboration in the Ecological Systems Laboratory, ISTE – ENAC – ECOS. Research on alluvial floodplain processes modeling and ecological integration implementation in a multi-objective river development project
2004	TA for the *Ecology* and *Freshwater Wetland Ecology* classes, University of Geneva. Under the supervision of Prof. J.B. Lachavanne and Dr. E. Castella of the Laboratory of Aquatic Ecology and Biology (LEBA). Responsible for field water chemistry laboratory logistics
2001	scientific technician for the environmental impact study of two aquaculture facilities in the *Ria Formosa* lagoon (Algarve, Portugal). Design of experiment, sampling and data analysis. Prof. Sofia Gamito, Universidade do Algarve, Faro, Portugal

Languages

French	Mother tongue
English	Mother tongue equivalent
Spanish	Fluent verbal and written
Portuguese	Basic verbal

Informatics skills

Word processing	$\LaTeX\,2_\varepsilon$, MS Word
Math & statistics	R, MatLab/Simulink, S-Plus, ADE-4, MS Excel
Image & slide shows	The Gimp2, Idrisi, Adobe Illustrator, Adobe PhotoShop, MS PowerPoint, MS Visio
Others	EndNote, IHA

Publications

Pellaud, M., Sardy, S., Schlaepfer, R. and A. Buttler. Modern model selection methods for integrating benthic aquatic richness in river development projects. *Submitted to Ecological Modelling.*

Heller, P., **Pellaud, M.**, Bollaert, E., R. Schlaepfer and A. J. Schleiss. (2006) Multi-purpose shallow reservoir: Synergies between ecology and energy production. IHA River-Flow 2006 Conference Proceedings, Lisbon, Portugal

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francais. Un Gastéropode Planorbidae listé en annexe de la 'Directive Habitats'. *Bull. mens. Soc. linn. Lyon* **74(7-8):**255–269.

Hobbies and Interests

- *hobbies* – camping, squash, coaching (kick boxing & self defense)
- *interests* – promotion of education, travels, mainly in Africa and Asia, cooking, music