COST 626

European Aquatic
Modelling Network

Proceedings

from the final meeting
in Silkeborg, Denmark
19-20 May 2005

Editors:

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Martin Baptist
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COST: European Cooperation in the field of Scientific and Technical Research

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Contents

Preface

This *proceedings* is the final result of the COST Action 626, European Aquatic Modelling Network. The report summarise 5 years of information exchange, scientific discussions, crew exchange and collaborative research in the format of scientific papers. More than 150 people from a total of 16 participating countries have contributed to COST Action 626. Through nine joint Working Group meetings a wide range of subjects have been discussed. All participants are most grateful to the organisers of COST 626 meetings in Brussels, Stuttgart, Trondheim, Vigo, Oulu, Aix-en-Provence, Salzburg, Madrid and Silkeborg. Several smaller Working Group or Small Group meetings have also provided useful collaboration and we are grateful to the organisers in St Pée sur Nivelle, Ghent, Wallingford, Lyon, Silkeborg, Oulu and Klagenfurt. In total, 30 scientists have been able to visit another COST 626 partner through Short Term Scientific Missions, providing a possibility of profound collaboration. We also want to thank the 6 invited speakers from countries outside Europe for their valuable input to COST 626.

At the start of COST 626, we focused on describing the state-of-the-art in data sampling, modelling, analysis and applications of river habitat modelling. We were organised in three Working Groups (Raw data; Modelling and Application) and made a public availably report: "State-of-the-art in data sampling, modelling, analysis and applications of river habitat modelling". The report can be downloaded from our web-site, www.eamn.org. Information about participants, meetings, reports and other documents can also be found on, www.eamn.org. Even though COST 626 will finish in June 2005, the web-site will be maintained further.

The last years of COST 626 we have focused on collaborative research within seven Topic Groups:

Abiotic and biotic data management
Fuzzy logic and other statistical techniques
Flow variations (interactions between flow regime and ecology)
Winter conditions for fish
Scaling
Modelling for The Water Framework Directive
Floodplain vegetation modelling

A wide range of universities, research institutes, companies and nationally-funded programs and projects have provided time and resources to fulfil this report, while COST has provided printing and of course the important networking facilities that made it all possible. We are all thankful to the large numbers of national funders and to COST.

The final COST 626 meeting is hosted by the Danish Environmental Institute in Silkeborg, Denmark, 19-20 May 2005. The final meeting will summarize collaborative research within the Action. The Scientific committee for the Final COST 626 meeting is:

Martin Baptist, Harm Duel, Michael Dunbar, Peter Goethals, Atle Harby, Ari Huusko, Anton Ibbotson, Helmut Mader, Morten Lauge Pedersen, Stefan Schmutz and Matthias Schneider.

Papers in the proceedings are printed in an alphabetical order. All papers have been through a review of their abstracts, but there has been no final review except from layout corrections. The content of these proceedings can be freely copied or printed as long as the authors' permission is obtained and the citation refers to this volume, i.e:

Author, A., Co-author, B., and Co-author, C. 2005. Title of paper. *Proceedings*, Final COST 626 meeting in Silkeborg, Denmark. (In Harby, A. et al (editors) 2005).

We encourage any reader of this paper to contact the authors to obtain more information or discuss the content of the paper.

We certainly hope that the end of COST 626 is just the start of a wider international collaboration on methods and models to assess river habitats in the best possible ways.

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Atle Harby	Ari Huusko	Anton Ibbotson	Helmut Mader
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Cost-Effectiveness-Analysis of Heavily Modifyed River Sections. Case study Drau (Austria)

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ABSTRACT: Various sections of the river Drau, Austria, are affected by hydropower production. Within a pilot study, one of these sections was investigated using a newly developed decision support system (DSS) called "RiverSmart". This is a system, in which measures are evaluated from an ecological point of view and total costs are estimated. The model-like character allows the combination of measures in the form of scenarios, which are compared with the actual condition. At present, the ecology in the considered river section is impacted in several ways (minimum flow, weir, reservoir flushing and changes of the natural discharge). The ecological status of the investigated river section was assessed for different scenarios using the DSS RiverSmart model. The comparison of the measure scenarios shows that an increased dynamic of discharge combined with an optimal discharge profile has nearly the same ecological effect as a steady dotation raise. However, the increase of mean flow is much more expensive.

1 INTRODUCTION

Hydro power features a very good CO_2- and energy balance in comparison with other power generation systems. Thus, from the global point of view, hydro power is an ecologically sustainable way of producing energy. On the other hand, the use of hydropower seriously interferes with river ecosystems. Various studies on the ecological effects of hydropower stations (Moog, 1993; Parasiewicz et al., 1998) show that ecological enhancements can be achieved by using improved technologies in hydropower production. The demand for more ecologically oriented hydro power already caused first approaches for a "green electricity certification" (Bratrich & Tuffer, 2001). This certification implicates a differentiation of the product "power" and offers producers of "green energy" a chance to resist the international price erosion even having higher financial efforts. Beside the pricing pressure due to the electricity market liberalisation, govermental regulations of the EC-Water Framework Directive (WFD, EUROPEAN COMMISION, 2000) force power companies to act. Within the EU, concepts to implement measures for rivers have to be worked out in order to achieve a good ecological status - or good ecological potential for the heavily modified waterbodies - until the year 2017. The choice, which river sections should be considered as heavily modified waterbodies is subject of many current case studies (CIS Working Group, 2002). To fulfill the accounts of the WFD in the future the power companies, but also river engineers have to focus on ecological enhancements of running waters. In view of limited "ecological budgets" it is advisable to examine possible alternatives of measures in terms of their cost-benefit relation. Considering the diverted river section Rosegg of the river Drau as an example, two options for both improving river ecology and reducing current operational expenses are discussed:

1) Reduction of sedimentation by modification of the river bathymetry in combination with enhanced flood management.
2) Installation of a supplementary turbine at the weir and raising the minimum flow.
The investigation was performed by using the Decision Support System RiverSmart (Egger et al., 2005).

2 AREA UNDER INVESTIGATION

The diverted reach under investigation is situated on the river Drau in Carinthia (Austria). The power station Rosegg is part of a conjointly operated chain of ten hydro power stations. They have a regular working ability of 2,623.7 GWh/year and a height of 175.7 m within a length of aprox. 147 km.

Figure 1. Location of the area under investigation on the river Drau.

Table 1. Power station Rosegg – hydraulic and hydrological facts.

Parameter	Quantity
Catchment area	est. 7.000 km²
Average water flow/year	205 m³/s
Flood water discharge volume	HQ1: 970 m³/s
Height	24 m
Length of the diverted river section	6,5 km
Engine output power station Rosegg	593,7 MW

3 RESEARCH METHOD

The estimation is based on a cost-effective-analysis of the actual state of 1998 (scenario 1) and on two scenarios of measures (scenario 2 "changed bathymetry and increased flow dynamics"; scenario 3 "increased dotation"). The ecological effectiveness and the additional costs of scenarios 2 and 3 are calculated and compared .

3.1 Ecological evaluation

The ecological evaluation is output of the Desicion Support System RiverSmart (Egger et al., 2005). Impacts on the running water ecosystem are investigated for the different scenarios and are added up in impact categories. These are classified into 5 categories (very low, low, average, high, and very high).

The reference situation (Leitbild) for the evaluation is established by 29 evaluation criteria. It is assumed that every impact category causes for each evaluation criteria a deviation of the reference situation (Mader et al., 2005). This deviation is specified as the degree of achievement (DOA; 0 to 100%). To merge the separate degrees of achievement, only that impact, which influences a certain evaluation criteria the most, is taken into account. The resulting minimum DOAs are integrated to a total degree of achievement by the arithmetic average value of all evaluation criteria. The total degree of achievement determines the "predicted ecological status". This status has 5 evaluation categories according to the EU-Water Framework Directive (WFD). The first category, "high status", corresponds to the reference situation, whereas categories 2 to 5 document the gradual deviation from the reference situation ("good status", "moderate status", "poor status", "bad status"). The WFD provides to attain at least a "good status" for all running water systems by the year 2017.

3.2 Acquisition of costs

The cost-analysis considers the costs of measures which are necessary for the realization of the evaluated scenario. An estimation of the costs without detailed planning is afflicted with great uncertainties. The stated values can only show the dimension. The cost-estimation is split into two specifications: output-change (in %) and cost change (in Euro), both in comparison with scenario 1.

4 RESULTS

4.1 Defining a reference situation

The reference situation of the reach under investigation can be described as an obligation maeander with alternating gravel bars, silver pasture forest seam and adjacent grey alder alluvial forrest (see fig. 2).

System entities	Abiotic system components	Abiotic parameters	*Specification of the ecologic Leitbild*	Type of impact (intensity of impact - 1: light, 5: very strong): Regulation (1)	Bank protection (1)	Weir above the section (5)	Weir below the section (2)	Flood-Management in the impoundment-space (5)	Management of sediment - excavation (4)	Management of sediment - reservoir flushing (5)	Water extraction (4)	Management of the natural cover to keep free the riverbed (4)	Stabilization of the bottom (1)	Minimum of degree of achievement	Mean value components of the system	Mean value layer of consideration
				Degree of achievement (%)												
System elements	Hydrology	Magnitudes of average discharges	*high*	100	100	100	100	75	100	100	13	100	100	13		
		Duration of average floodplain inundation	*moderate*	93	100	100	100	67	93	100	97	100	100	67		
		Relative distances to groundwater surface	*moderate*	90	100	100	100	100	90	100	67	100	100	67	49	
	Water body	Spatial extension of the aquatic area	*moderate*	90	100	100	100	100	97	100	33	100	100	33		
		Magnitude of average water depths	*high*	93	100	100	100	100	90	100	50	100	100	50		
		Magnitude of average flow velocities	*moderate*	90	100	100	100	100	97	100	33	93	100	33	39	
	Stream course development	Degree of stream course development	*low*	93	100	100	100	75	93	100	100	95	100	75	75	
	Stream channel morphology	Richness of bed forms and bed patterns	*high*	95	100	25	100	80	75	0	80	85	100	0		
		Channel dimension	*moderate*	90	93	67	100	100	90	100	17	83	100	17	9	
	Bank\Riparian morphology	Spatial extension of the riparian zone	*moderate*	90	97	100	100	100	100	100	33	100	100	33		
		Structural quality of the riparian zone	*moderate*	97	90	83	100	100	90	83	87	100	100	83	58	
	Floodplain morphology	Spatial extension of the floodplain zone	*moderate*	97	100	100	100	100	100	100	100	100	100	97		
		Structural quality of the floodplain zone	*high*	95	100	100	100	100	100	100	80	100	100	80	88	
	Physico-chemical characteristic	Water temperature	*low*	100	100	100	100	100	100	100	50	100	100	50		
		Water turbidity	*very low*	100	100	100	100	100	90	0	100	100	100	0		
		Amount of organic matter	*very low*	100	100	100	100	100	100	0	40	100	100	0		
		Amount of harmful inorganic matter	*very low*	100	100	100	100	100	100	0	40	100	100	0	13	47
System dynamics	Morphodynamics	Degree of instream morphodynamics	*moderate*	90	100	0	100	87	67	83	67	90	100	0		
		Bank erosion and riparian dynamics	*moderate*	90	97	33	100	67	77	87	100	77	100	33		
		Degree of floodplain morphodynamics	*moderate*	97	97	100	100	67	100	100	100	100	100	67	33	
	Hydrodynamics	Discharge dynamics - short term	*very low*	100	100	100	100	90	100	0	80	100	100	0		
		Discharge dynamics - medium term	*moderate*	100	100	100	100	93	100	83	83	100	100	83		
		Discharge dynamics - long term	*moderate*	100	100	100	100	100	100	100	100	100	100	100		
		Dynamics of flooding	*high*	95	100	100	100	50	75	100	95	90	100	50		
		Groundwater dynamics	*high*	100	100	100	100	100	100	100	50	100	100	50	57	45
System connectivity	Lateral connectiviy	Connectivity of the river to floodplain water bodies	*very high*	90	100	100	100	50	50	100	0	100	100	0	0	
	Vertical connectivity	Connectivity of the river to groundwater	*very high*	100	100	100	100	100	100	100	100	100	100	100	100	
	Longitudinal continuum	Local longitudinal continuum	*very high*	100	100	0	70	100	70	100	30	100	100	0		
		Regional longitudinal continuum	*very high*	100	100	0	70	100	80	100	30	100	100	0	0	33
Total degree of achievement (in %)																42
Predicted ecological status class																3,0

Figure 2. Detailed evaluation conclusion: Drau, section diverted stretch, scenario 1 (actual state 1998). The minimal degree of achievement of each evaluation criteria is marked grey.

4.2 *Scenario 1: actual state of 1998*

In scenario 1 the state of the river in the year 1998 is considered, which is before an integrated ecological study (Petutschnig et al., 2002) was inaugurated. The constant year-round minimum flow in the diverted river section is 5 m^3/s and is delivered through a turbine installed in the weir. For flow rates higher than 400 m^3/s, additional water is flowing into the diverted river section. To act along with operational safety as well as flood-security an intensive management of sedimentation accumulation in the diverted river stretch is required.

The most effective impacts are the categories "water extraction" and "sedimentation management by means of flushing". The weir body is a barrier for fishes and sediment. The total DOA is 41%. This gives a forecasted ecological status of 3,0 (moderate status).

4.3 *Scenario 2: modified bathymetry and increase of flow dynamics*

In Scenario 2 following measures were implemented: a) The modification of river bathymetry for minimizing the sedimentation rate, b) ecological measures, e.g. the enhancement of morphological variety in the river bed, c) the increase of flow dynamics, d) an optimized flood management scheme and e) the re-establishment of the river continuum. The advanced sediment and flood-management in combination with meliorated morphology provides significant ecological improvements concerning the system components hydrology, river morphology and morphodynamics. Hence, an ecological status class of 2,0 (good status) can be achieved, which is an improvement of one level.

The additional costs of scenario 2, compared to scenario 1, result from an estimated production decrease of 25-30%. Additional river bottom stabilization (0.1 Mio Euro) and a facility for fish migration (1 Mio Euro) have to be provided.

Cost savings can be derived by reduced efforts for sediment removal (0.15 Mio Euro) and natural coverage management (0.05 Mio Euro).

4.4 *Scenario 3: dotation raising*

In scenario 3 - additionally to scenario 2 - the minimum flow was increased from 5 to 150 m^3 and delivered to the diverted stretch by an additional turbine. However, the reservoir flushing is still part of the operation plan. Due to the changed operational mode sediment excavation within the diverted river stretch can be reduced. With the proposed set of measures, including the increase of minimum flow, it is possible to achieve a predicted ecological status of 1,5 (very good – good status).

The additional costs of scenario 3 amount, analog to scenario 2, to 1.0 Mio Euro for the fish migration facility and 0.1 Mio Euro for measures on the bottom stabilization. In scenario 3 the construction of a new power house (and with it expenses of 1.2 Mio Euro) is an essential expense factor. Increased minimum flow results in a production decrease of 20-25% in comparison with scenario 1.
Cost reductions of 0.25 Mio Euros result from the sediment management.

5 COST-EFFECTIVENESS-ANALYSIS

In table 2 the identified predicted ecological status is compared with the estimated costs (changes in energy production and expenses in comparison to scenario 1).

Table 2. Cost-effectiveness-analysis of the scenarios at the river Drau/Rosseg.

	forecasted ecological status	changes in energy production	Expenses (in comparison with scenario 1)
Scenario 1	3.0	0 %	0
Scenario 2	2.0	- 25 % to – 30 %	est. 0.9 Mio.
Scenario 3	1.5	- 20 % to – 25 %	est. 12.9 Mio.

Regarding scenario 2 "changed bathymetry and increased flow dynamics", a good ecological status as claimed in the WFD could be reached in the diverted river section. Ecological measures must preferably include sediment management by means of flushing. Essential in this section is an ecologically optimised manner of sediment management: River bathymetry has to be designed in a way that it allows a maximum wetting of the river bed with minimum water quantity (to inhibit the appearance of willows) but also avoids the deposition of sediments at reservoir flushing. Simultaneously the safety of the surroundings against flooding needs to be warranted. Beside the positive ecological effects, the optimized shaping of the discharge profile allows a minimisation of the running operational costs (costs of excavation and natural caverage management).

In scenario 3 a significant increase of mean flow is planned. This measure leads to higher investment costs (12 Mio Euro for the establishment of a power house at the weir) for the operating company, but also, compared to scenario 2, to a lower production decrease.

6 CONCLUSION

The comparison of the measure scenarios 2 and 3 shows that an increased dynamic of discharge combined with an optimal discharge profile (scenario 2) has nearly the same ecological effect as a steady dotation raise (scenario 3). However, the increase of mean flow is much more expensive.

7 LITERATURE

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Modelling the influence of vegetation on the morphology of the Allier, France

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ABSTRACT: Understanding the interactions between vegetation and the morphology of rivers is becoming increasingly important in view of modern river management and climate change. There is a need for predictive models for the natural response of rivers to river rehabilitation. One way to study the effects of river rehabilitation is to study natural reference rivers. The Allier in France is considered as a landscape reference for the to-be-restored Border Meuse in the Netherlands. The Allier is highly dynamic, large amounts of sand and gravel are transported during floods and its morphology changes considerably from year to year. The riparian vegetation is characterised by pioneer species on the low-lying dynamic point-bars, herbaceous vegetation and grass on the higher parts and extensive softwood floodplain forests, mainly consisting of poplars, on the older and higher floodplains. Due to the river dynamics, this river shows natural rejuvenation of vegetation such that older forests are removed by erosion and young pioneer vegetation can start growing on the point-bars. This model study investigates the role of vegetation on the morphological changes of a single flood event that took place in December 2003. A state-of-the-art 2-DH morphodynamic model was applied in a 6 km^2 study area. This model accounts for the effects of vegetation on the hydraulic resistance and on the reduction of bed shear stress and subsequent bed load sediment transport. The model results show that vegetation has a pronounced effect on the hydro- and morphodynamics. The results also reveal that this model has only limited success in simulating the observed morphological changes. Recommendations for further model development will be made. It can be concluded that vegetation is an important factor for the habitat template in gravel bed rivers, but our knowledge is at present insufficiently advanced to accurately predict the morphodynamic changes in this section of the Allier.

1 INTRODUCTION

There is an increasing need for understanding and predicting the interactions between vegetation and the morphology of rivers. Vegetation affects the morphology through slowing down and diversion of flow (Baptist et al., in press). A changing morphology leads to changes in the habitats of fish and other fauna.

The objective of this study is to simulate the influence of vegetation on morphology with a state-of-the-art numerical hydrodynamic and morphodynamic model. The morphological changes of a flood event in a meandering gravel bed river were carefully mapped, together with the vegetation characteristics of the study area. Results and shortcomings of different model techniques will be discussed.

2 STUDY AREA

The study area is part of the Allier, France. The Allier is a gravel bed, rain-fed river that originates in the Massif Central and joins the Loire River in Nevers, about 400 km downstream of its origin. The study area of about 6 km^2 in size lies 5 km upstream of the town of Moulins, France. It is located in the meandering section of the Allier and it is part of a nature reserve in which most of the river banks are unprotected. The Allier is considered as a landscape reference for the to-be-restored Border Meuse in the Netherlands. The Allier is highly dynamic, large amounts of sand and gravel are transported during floods and its morphology changes considerably from year to year. The riparian vegetation is characterised by pioneer species on the low-lying dynamic point-bars, herbaceous vegetation and grass on the higher parts and extensive softwood floodplain forests, mainly consisting of poplars, on the older and higher floodplains. Due to the river dynamics, this river shows natural rejuvenation of vegetation such that older forests are removed by erosion and young pioneer vegetation can start growing on the point-bars (Baptist et al., 2004). The rejuvenation also leads to the presence of large woody debris, mainly trees, into the river.

3 MATERIAL AND METHODS

A state-of-the-art two-dimensional depth-averaged (2-DH) numerical model was applied in which the drag forces of the vegetation are decoupled from those of the sediment, leading to a more accurate description of the bed shear stress (Baptist, 2005). For the latter approach, one needs estimates of the stem diameter and densities of different vegetation types. Model details on boundary conditions, grid size, initial conditions, etc. will be reported in De Jong (in prep.).

Two field campaigns were carried out in July 2003 and July 2004 to collect data on the terrain topography and the vegetation distribution. These campaigns were organised jointly by Delft University of Technology, Faculty of Civil Engineering and Geosciences, WL | Delft Hydraulics, Utrecht University, Department of Physical Geography, Radboud University, Department of Environmental Studies and Meander Consultancy and Research, under the heading of the Netherlands Centre for River Studies.

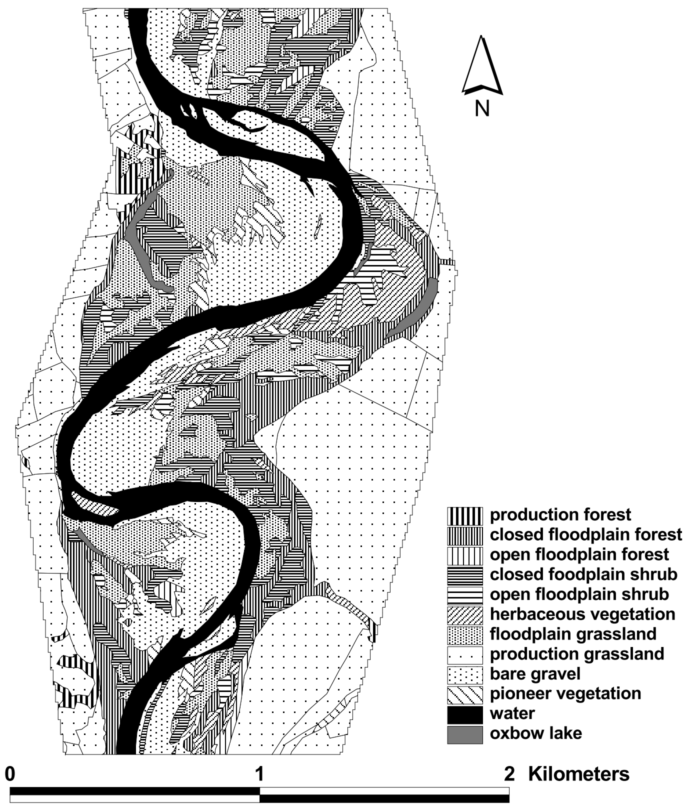

Figure 1. Vegetation types in the Allier study area.

A Real-Time Kinematic Differential Global Positioning System (RTK-DGPS) was applied to obtain terrain coordinates in x, y and z direction with an accuracy of about 5 cm in each direction. Approximately 3000 elevation points have been collected in each field campaign in order to map the floodplain heights. The morphology of the river bed was obtained by levelling river cross-sections. Interpolation of the elevation data on a 20 x 20 m rectangular grid resulted in a Digital Elevation Model of the study area.

Vegetation structures were identified and mapped in the field to obtain a ground truth for the analysis of stereoscopic aerial photos taken in the year 2000. The vegetation in the area was classified based on the main vegetation types present, see Figure 1. For forests and shrubs an additional qualification was made with respect to their horizontal distribution (open or closed cover). A closed cover is defined as more than 60% cover, and an open cover is defined as between 20% and 60% cover (Breedveld & Liefhebber, 2003). At less than 20% cover of shrubs or trees, the vegetation type is based on the dominant vegetation, usually grassland. Vegetation characteristics height, diameter and density were obtained for floodplain forest and shrub (Wijma, 2005). Estimates of vegetation properties of grassland, herbaceous vegetation and pioneer vegetation have been obtained from measurements in Dutch floodplains (Van Velzen et al., 2003a, b).

Table 1 presents the vegetation types that were distinguished in the study area, and their properties height (*k*), diameter (*D*), and density (*m*). The drag coefficient (C_D), needed to compute the vegetation resistance, is assumed equal to 1.

Table 1. Vegetation types and their properties.

Type	*k* (m)	*D* (m)	*m* (m^{-2})
Production forest	10	0.042	2
Closed floodplain forest	10	0.042	1.2
Open floodplain forest	10	0.042	0.4
Closed floodplain shrub	5	0.01	10.2
Open floodplain shrub	5	0.01	3.4
Herbaceous vegetation	0.5	0.005	400
Floodplain grassland	0.2	0.003	3000
Production grassland	0.1	0.003	4000
Pioneer vegetation	0.1	0.003	50

4 RESULTS

A flood event that took place in December 2003 led to major changes in the morphology of the study area.
Figure 2 presents the measured sedimentation and erosion between July 2003 and July 2004. It shows the vertical changes in topography, so when a 3 m high bank is washed away, it results in 3 m erosion. In the middle western part of the study area (x,y = 675600, 2167000), a 10 m high cliff is present, leading to 10 m of erosion. Horizontal rates of erosion amounted up to 50 m. West of the northern point bar (x,y = 675900, 2167900), an oxbow lake is completely filled up with 2-4 m of sediment. At several locations on the point-bars, large scour channels can be found, due to short-cut flows.

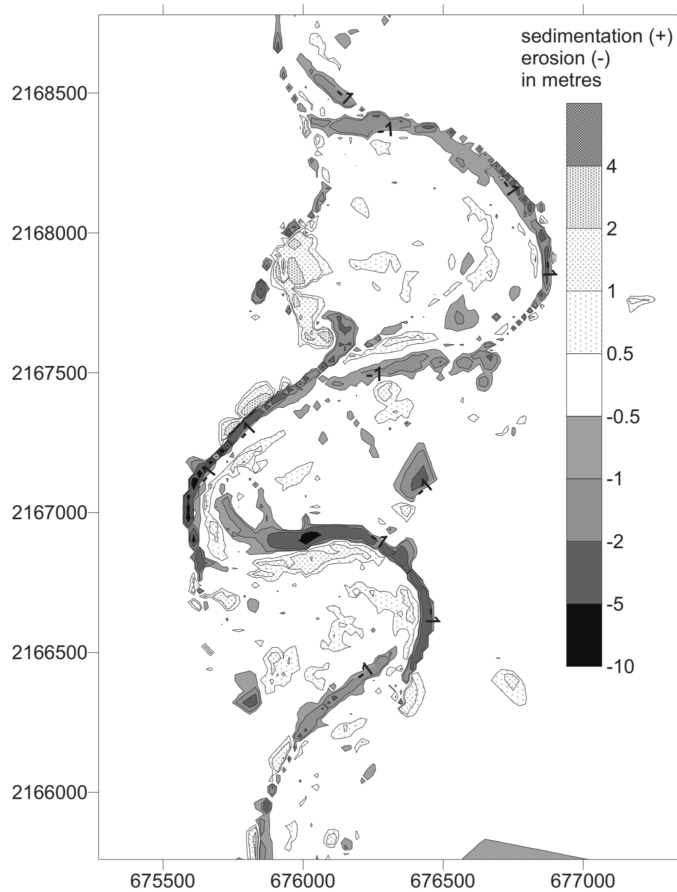

Figure 2. Measured sedimentation (+) and erosion (-) between 2003 and 2004, in metres.

The numerical model results are presented in Figure 3 and Figure 4. Figure 3 shows the results of the model including the effects of vegetation on the flow, the bed shear stress, the sediment transport capacity and the morphology. Figure 4 shows the model results when the vegetation is completely left out of the model, i.e., there is no effect on slowing down flow velocities or redirecting flows. A comparison of these results with the measurements shows that, generally speaking, patterns of erosion and sedimentation are simulated much better with the model including vegetation influence, but a critical evaluation can only conclude that the simulation results are still not very good. The filling of the oxbow lake has not been simulated, and erosion rates are generally underpredicted. Furthermore, sedimentation patterns on the higher parts are missing.

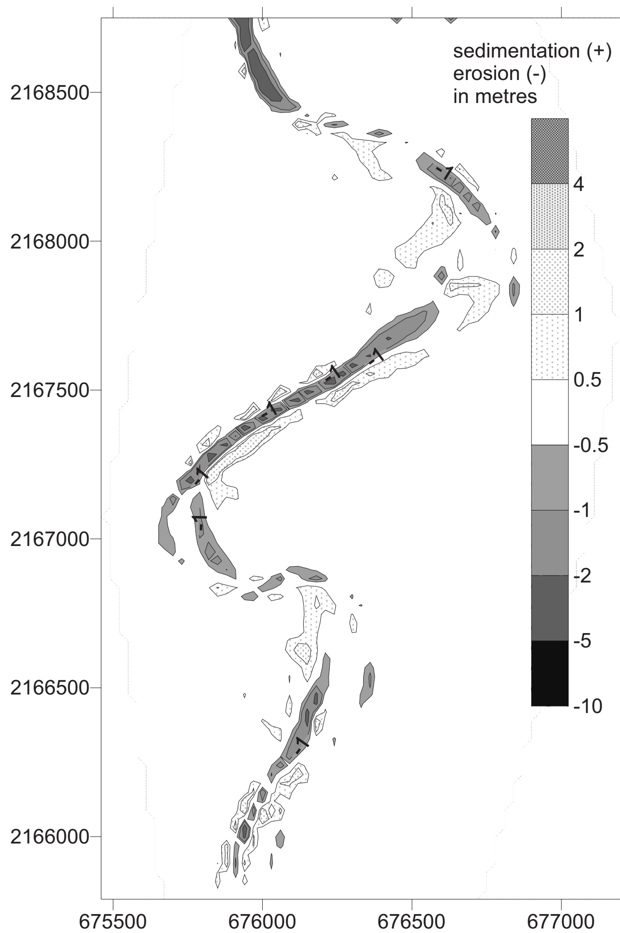

Figure 3. Numerical model results for the simulation of morphological changes, including vegetation influence.

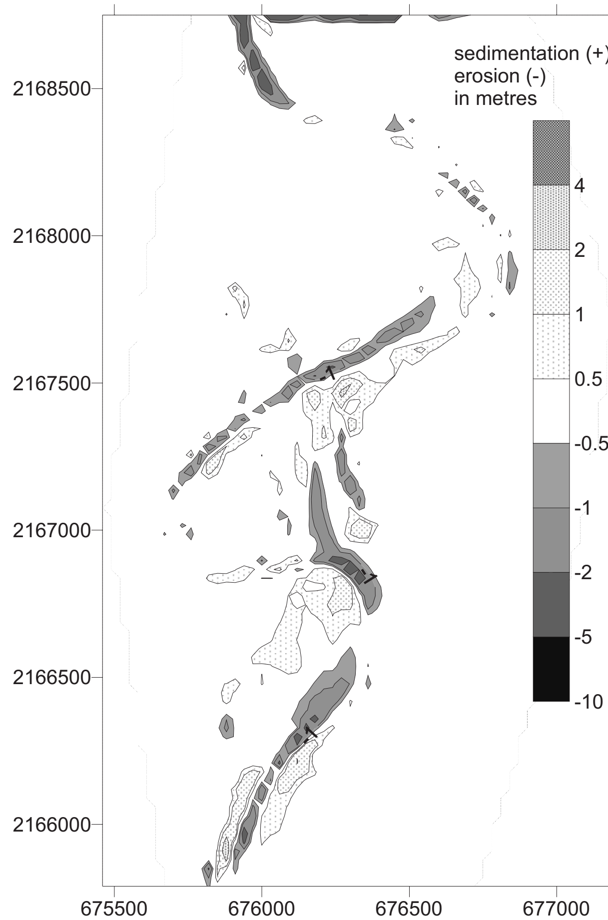

Figure 4. Numerical model results for the simulation of morphological changes, without vegetation influence.

5 DISCUSSION AND CONCLUSION

This study has shown that it is important to include the influence of vegetation in hydrodynamic and morphodynamic models. This will greatly improve model simulations, yet the results are still not very satisfactorily. The main causes for this are that a (semi)-natural river such as the Allier has many features that are still difficult to model. The Allier has armoured layers, a wide range of sediment sizes ranging from course sand to course gravel, steep, cohesive banks and plenty vegetation. In the current state-of-the-art version, only the effect of vegetation has been accounted for, which is already a major step forward. However, in future model applications, more model improvements are needed, i.e. the computations must be made with graded sediment, and bank failure mechanisms must be included.

On the other hand, a process-based model such as applied here is not the only way to go. One can also start from a top-down approach and carefully describe and quantify (!) spatial patterns in rivers, associated with vegetation patterns.

Ultimately, it is our desire to have a morphodynamic model that is capable of predicting changes in sand and gravel bed rivers either due to natural processes, or as a result of human measures. Such a model can for example be applied in the Border Meuse, where large-scale flood protection measures, gravel mining and river rehabilitation is planned. The morphodynamic computations can then be coupled to Habitat Evaluation Procedures (HEPs) to predict changes in habitat suitability for flora and fauna.

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The hydromorphological picture of meso-scale units of rivers

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ABSTRACT: The presented work serves two purposes. One contributes to the development of a scaling and classification system of fish habitat in small-medium sized stream based on hydromorphological units (HMUs). The other purpose is to compare salmon productivity in upland and lowland rivers, for the intra-institutional project of Centre for Environment and Hydrology (CEH).

Mesohabitat mapping and typing suffers from over-categorizing and inaccurate definition of class elements. We employed two existing systems used for partly similar purposes, the Norwegian Mesohabitat Classification Method and the River Habitat Survey of the UK in order to compare their results and difference in their application.

Data were collected during three field campaigns in the UK and Norway. We tested the hypothesis that there are more similarities within the HMUs in both methods in terms of physical parameters than between them. These physical parameters include or are based on river depth, surface and mean flow velocities, surface flow type and dominant substrate size. We found that the first principal component describes the HMUs best as a single parameter, however we could not cover the whole range of available HMU types in our analysis due to limited hydromorphological variation in the selected rivers.

The first two field campaigns was carried out in combination with the internal CEH project, under the umbrella of a COST Action 626 Short Term Scientific Mission (STSM) during summer 2002.

1 INTRODUCTION

Meso-scale classification of rivers has been used for decades in hydrology and ecology. Recent research has demonstrated a large potential for using this in eco-hydraulics. Habitat modellers have to look at complex systems (e.g. catchments), where problems inherent in applying models developed for small scales for larger scales need to be overcome. The use of hydromorphological units (HMUs or meso-scale classes) extends information and helps to overcome the problems arising from scale alteration. The procedure is called upscaling, and is done by means of a system based on meso-scale sized classes.

Besides the numerous issues related to the practical application and technical details related to such methods, there is also a need to describe the HMUs by means of measurable physical parameters. These parameters should allow explicit description of each HMU type.

2 BACKGROUND

In our study we used the Norwegian Mesohabitat Classification Method (NMCM), which classifies HMUs by estimating and observing four parameters, surface pattern, mean depth, mean surface velocity and relative gradient. See Borsányi *et al.* (in press) for the details on this method. The four parameters used in the NMCM vary within each HMU to some accepted extent, and these variations differ in magnitude from each other in the different HMU types. We decided to take point samples of the four selected parameters from HMUs and analyse them by statistical means to get a better picture in what ranges and how the parameters vary.

We collected data during field campaigns in three rivers. Two small rivers were selected in the UK by CEH and one in Norway. The two rivers in Scotland were Cruick Water and Water of Tarf. The study section on Cruick Water lays by Newtonmill, and on Water of Tarf by Tarfside Farm, both in SE Scotland. The river in Norway was actually a side channel of Nidelva in Trondheim, middle Norway.

The sampling strategy was meant to cover whole river reaches, and the samples were collected independently from the actual HMU layout. The sampling followed a regular random structure, meaning that the samples were taken from nodes of an imaginary grid stretched on the surface of water, which had no relation to HMU layout or other hydromorphological features. We sampled 5-8 points in cross sections following each other in regular distances of about 2-10 meters. Altogether in 405 of such nodes surface flow type, mean and surface velocity, depth and substrate composition data were collected. Depth was measured to the closest centimetre by a measuring pole, and velocities to the closest millimetre per second by propeller instruments. After sampling at points, all reaches were surveyed according to the NMCM. All samples of data were collected independently from the HMU survey, the point positions were assigned to the HMUs later in a GIS.

Guidelines presented by and tools provided with the works of Gordon *et al.* (1992), Townend (2002) and Johnson and Kuby (2004) were used for assistance in statistical analyses. The calculations were carried out in Microsoft Excel 2003 and Minitab 14.

The purpose of our statistical test was to show that all different HMU types (as in the NMCM, altogether 8 types) do differ from each other.

3 RESULTS

The combined maps of HMU surveys and positions of sampling data points on the three sites are shown on Figure 3.1. HMUs in the NMCM are noted by letters and numbers, such as A, B1, B2, C, D, E, F, G1, G2 and H. These represent mesohabitat features, like glides, pools etc.

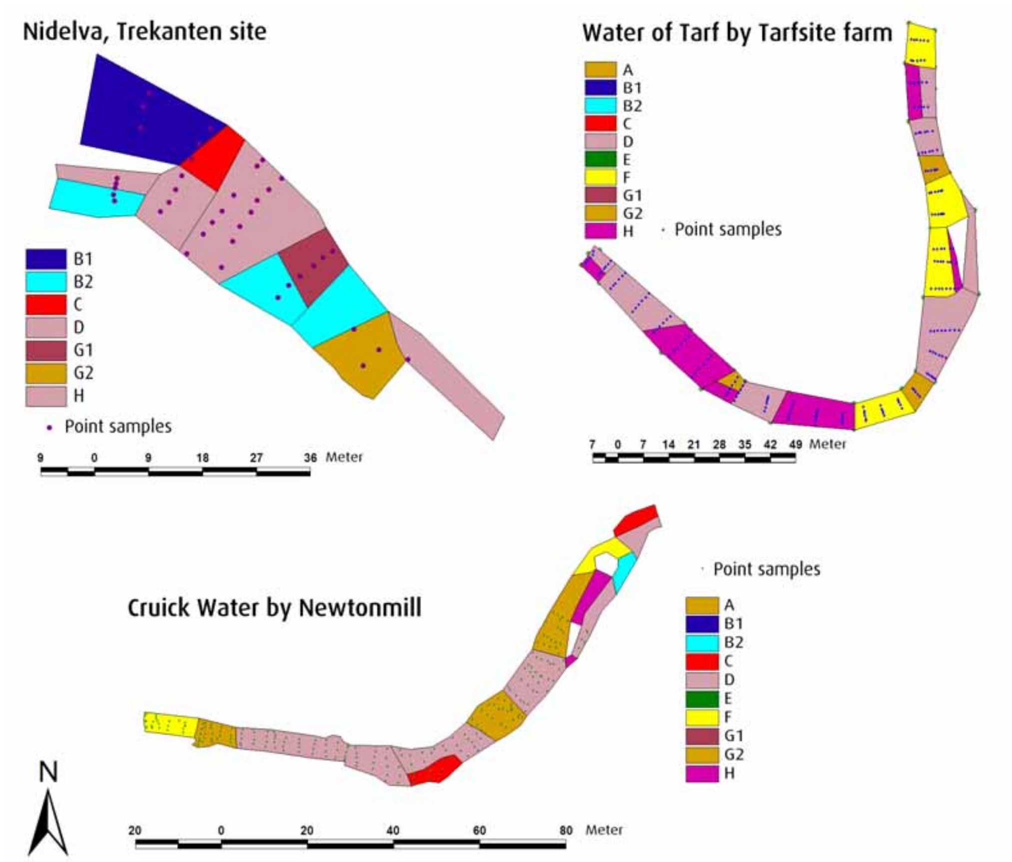

Figure 3.1 HMU survey and point samples at Nidelva, Norway (top left), Water of Tarf (top right) and Cruick Water in Scotland.

Table 3.1. Example of collected data on Cruick, Tarf and Nidelva

River	ID	X	Y	Z	Dep (m)	V_{mean} (m/s)	V_{surf} (m/s)	Subs	Surf	HMU
Nid	21	114.9	104.01	98.84	0.82	0.425	0.443	GR	RP	B1
Nid	22	112.99	101.43	98.85	0.82	0.789	0.789	GR	SM	B1
Nid	33	104.61	109.89	98.64	1.05	0.434	0.529	GR	RP	B1
Nid	34	103.69	107.65	98.63	1	0.542	0.568	GR	SM	B1
Nid	35	103.2	104.2	98.89	0.83	0.638	0.655	PB	RP	B1
Nid	11	126.19	76.07	99.02	0.6	0.703	1.132	GR	RP	B2
Nid	13	127.96	78.1	99.01	0.6	0.651	0.798	GR	RP	B2
Nid	23	111.78	99	98.95	0.7	0.807	0.763	GR	SM	C
Nid	25	110.17	96.22	99.3	0.32	0.282	0.382	GR	RP	D
Nid	26	109.14	93.89	98.98	0.68	0.339	0.408	GR	SM	D

Table 3.2. Codes and sizes for categorical variables used in the sampling and in the analysis. Extended from Raven *et al.* (1998)

Substrate diameter (mm)				**Surface Flow Type**		
CL	code 1	Clay	<0.002	**BW**	code 5	Broken Standing Waves
SI	code 2	Silt	<0.02	**UW**	code 4	Unbroken Standing Waves
SA	code 3	Sand	<2	**RP**	code 3	Rippled
G	code 4	Gravel	<16	**SM**	code 2	Smooth
P	code 5	Pebble	<64	**NP**	code 1	No Perceptible Flow
CO	code 6	Cobble	<256			
BO	code 7	Boulder	>=256			

Table 3.1 shows part of the data collected at Nidelva to explain the data structure. The header "ID" shows a unique identification for each sampling point, "Dep" stands for depth, "V_{mean}" and "V_{surf}" for mean and surface velocities respectively, "Subs" for substrate code and "Surf" stands for surface flow type. For this latter two the categories from Raven *et al.* (1998) were used (this work commonly known as the RHS report in the UK). The categories are presented here for convenience in Table 3.2.

Table 3.3 summarizes the number of sampling points per river and per HMU where depths, surface and mean velocities, surface flow types and substrate composition were measured. Sampling points where any of these was missing are not included. The table shows that HMU types "A", "B1", "B2", "C", etc. are under- or not at all represented. For example in the case of type C, we see 10 sampling points from Cruick, one point from Nidelva and none from Tarf. There also seems to be little overlapping between HMU types on the three sites, as for example types B1 and B2 are only present in Nidelva, or type practically only in Tarf. Such distortions were expected to limit our possibilities to follow our original plans for our tests.

Table 3.3. Number of sampling points collected with complete set of data in Cruick, Nidelva and Tarf

River	HMU	Number of points with complete data
Cruick	C	10
	D	119
	F	22
	G2	67
	H	1
Cruick Total		219
Nidelva	B1	5
	B2	4
	C	1
	D	22
	G1	4
	G2	3
Nidelva Total		39
Tarf	D	67
	F	56
	G2	16
	H	49
Tarf Total		188
Grand Total		446

4 DISCUSSION

First we tested how often the HMU survey succeeded or failed when classifying the positions of the point samples. This is checked by looking at the number of correctly and incorrectly surveyed HMU types. The verification is based on our measurements, while the survey is based on estimation of parameters. The measurements give a limited possibility to tell expected HMUs for that very point.

Table 4.1 shows the results. The main diagonal (from top left to bottom right, greyed) shows the percent and number of correctly classified points in during the surveys. The table also shows into which other HMUs the wrongly predicted points should have been classified.

For example in the first row we see that altogether 5 sampling points were surveyed as points in HMU type B1, however, based on the depth, surface velocity, surface flow type and gradient, only 4 (80%) confirmed the features of HMU type B1, one point (20%) actually had the characteristics of HMU type C.

We note that no HMU type A was either surveyed or expected in either of the three rivers (a row for A is missing), and that though no HMU type E was surveyed (row for type E is also missing), it was found in two sampling points, wrongly classified as type F-s. HMU types B1, B2, C and D were mostly correctly (or equally correctly and wrongly in case of B2) classified. This is shown because percentage values in the diagonal at these types are higher than any other values in the same rows respectively. HMU types F, G1, G2 and H were mostly incorrectly classified. Note that all wrongly surveyed HMU types have broken surface. Looking at the row of type F for example, we see that out of 78 sampling points surveyed to fall in F type areas, only 32 (41%) had the characteristics of type F. 44 points (56%) had actually characteristics of type H. In case of G2 it is important to note, that though points of this HMU type was mostly wrongly surveyed to be G2 types (39=45% of 86=100% correct), still among all the actual types the sample points fall in, the correct type dominated. 24 points had actually type B2, 18 points type D, 1 point type G1 and 4 points type H characteristics. Interestingly, the sampling points classified as points in HMU type H mostly showed characteristics of type D (29=58% of 50=100% cases).

Summarizing, we see that the HMU survey of the NMCM classifies areas in rivers that do not have similar characteristics. The physical features of the different HMU types vary to some extent, and therefore a simple comparison of their mean values seems to be insufficient to provide basis to distinguish between HMU types. Since we are interested in similarities/differences between HMU types, those with few samples should be excluded from the further analysis. These are HMU types A, B1, B2, C, E and G1, and so this subchapter focuses further on to compare HMU types D, F, G2 and H.

Table 4.1. Comparison of surveyed and expected point-HMU relations. Numbers show the number of sampling points and percentages the actual correct and wrong proportions for each surveyed HMU type.

	Expected HMUs									
Surveyed HMUs	**B1**	**B2**	**C**	**D**	**E**	**F**	**G1**	**G2**	**H**	*Total*
B1	80% 4		20% 1							*100%* *5*
B2	0%	50% 2						25% 1	25% 1	*100%* *4*
C	9% 1	0%	73% 8	18% 2						*100%* *11*
D	1% 3	11% 23	3% 7	72% 149				1% 3	11% 23	*100%* *208*
F					3% 2	41% 32			56% 44	*100%* *78*
G1	50% 2	50% 2								*100%* *4*
G2		28% 24		21% 18			1% 1	45% 39	5% 4	*100%* *86*
H		24% 12		58% 29				6% 3	12% 6	*100%* *50*

We continue width exploratory data analysis. We see that we collected three numerical variables (depth and 2 velocities) and two categorical variables (substrate class and surface flow type class). This latter would limit the possibilities of applicable tests, and therefore are converted to numerical categories. This is possible, because classes of these variables actually represent gradually increasing substrate sizes and (sort of) wave heights or relative energy loss reflection on the water surface.

We tested the normality of the data by means of Probablity plots (not shown here) and performing the Anderson-Darling test for the three numerical variables registered in all three rivers per HMU types. The plots display the 95% confidence interval, and the numerical values displayed are means, standard deviations, number of samples, Anderson-Darling values and probability values. The results show that in case of depth samples normality is probably not achieved in the sample in HMU types D, F and G2, as p-values here are below 0.05. All velocity samples are probably normally distributed except surface velocities in HMU type D.

Since we aim at utilizing all five variables in our analysis, and these are mixed in distribution or excluded so far from the analysis, we look for principal components, which may both reduce the number of variables to be used in the tests and thereby simplifying our methods and at the same time keeping the features of the original dataset. Principal component analysis (PCA) is performed on the complete set of the five variables, testing for correlation between them. The scree plot is shown on Figure 4.1. The plot displays the component number with the related eigenvalues (which are the variances of the principal components) of the correlation matrix.

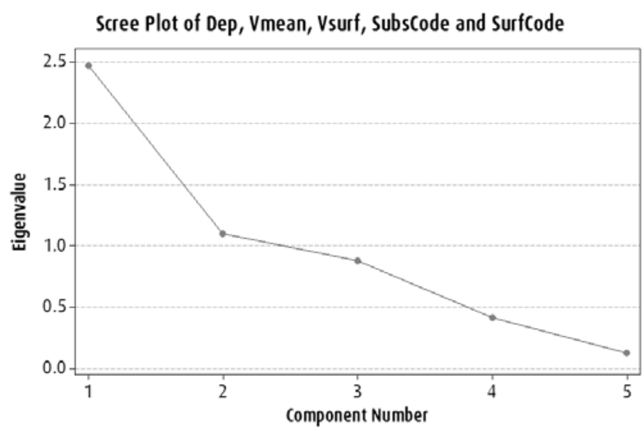

Figure 4.1. Scree plot of depth, mean and surface velocities, substrate code and surface code to determine the number of principal components for the PCA analysis

The plot does not show a clear flattening shape as principal component values increase. Components 2 and 3 have almost similar effect on the data, and 4 and 5 follows the descending trend of the graph. We could either select only one component (because only component 1 provides clearly higher eigenvalue than 1, or select three components, as they might divide the rapidly increasing and flattening parts of the graph. Normality tests (not shown here) carried out on the first three principal components tell us that component 1 is likely to provide normal distribution in all four HMU types, whereas components 2 and 3 are probably not normally distributed in HMU types D, F, G2 and D, H respectively. In order to maintain data integrity, we must use the same amount of principal components for all HMU types, and therefore we must use only one, which is noted PCA1 further on. The coefficients for the first three components are shown in Table 4.2.

Table 4.2. Coefficients of the PCA for calculation of the first three components

Variable	PC1	PC2	PC3
Dep	0.092	-0.853	-0.383
Vmean	-0.577	-0.214	0.095
Vsurf	-0.577	-0.255	0.099
SubsCode	-0.245	0.345	-0.902
SurfCode	-0.516	0.208	0.143

We continue with testing the analysis of means between the four HMU types based on PCA1, which is a linear combination of our five original variables. The results are shown in plot-form on Figure 4.2. The decision limits above and below the centerline are shown for each HMU. The error rate or alpha level was chosen as 0.05, which is reflected in the value of the decision limit lines. If a point falls outside the decision limits, then there is significant evidence that the mean represented by that point is different from the total mean of all samples. This is the case in HMU types D, F and G2. In case of H, this test does not provide significant evidence that its means differ from the total mean.

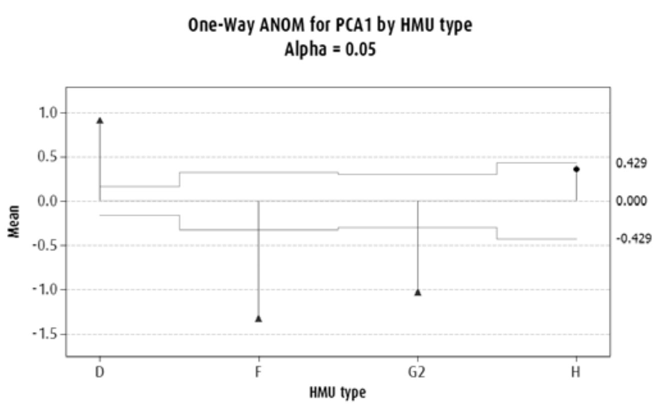

Figure 4.2. Analysis of means of principal component 1 in HMU types D, F, G2 and H.

5 CONCLUSIONS

We conclude that the data we collected in the three rivers does not allow an objective separation of HMU types of the NMCM, due to lack of sampling points in types A, B1, B2, C, E and G1. Analysis of the remaining data shows significant differences between types D; F and G2 in relation to the total mean, but no evidence was found showing differences between points in type H and the total mean. The comparison was not possible in case of either of the separate HMU types based on the five parameters, however the first component in PCA provided basis for the further analysis.

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Coupling habitat suitability models and economic valuation methods for integrated cost-benefit analyses in river restoration management: ecosystem oriented flood control study on the Zwalm river basin (Belgium)

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ABSTRACT: Scientific tools which can bridge the gap between the economic market and the natural market of (aquatic) ecosystems are of paramount importance to attain sustainability characterized by a growth of the economic development allowing ecosystem repair and regeneration. For this purpose, data collection and preparation should be based on insights in the river processes at different spatial and temporal scales, but also include the needs of the managers, allowing discussions among all involved participants and thus to make decisions transparent. Cost-benefit analyses are good instruments for this goal and predictive models embedded in a decision support system can be valuable tools to deliver the necessary data for the in-depth analysis of the different restoration options.

In Flanders, flood control is mainly based on the use of weirs. These weirs are often insufficient to avoid flooding during intensive rain events, and many sites in Flanders are yearly confronted with damaged houses. Therefore, in the Zwalm river basin, natural flooding areas will be reused and the effects of weirs will be minimized, as they strongly affect the ecosystems (migration and habitat conditions by the altered water depth and flow velocity). To get a better understanding of the involved costs and potential benefits of these interventions, habitat suitability models are used in combination with economic valuation methods. Based on this case study in the Zwalm river basin, major difficulties and pitfalls of these methods will be illustrated in combination with the advantages for integrated river management.

Ecological effect assessment through the combination of mechanistic and data driven models

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ABSTRACT: In view of recent legislations like the Water Framework Directive and REACH, there is a growing need for a proper assessment of the ecological effects of anthropogenic disturbances. The former legislation focuses on the good ecological status, while the latter regulates environmental and human health risk of chemical substances. A way to combine these two goals, i.e. to evaluate ecological impact, is to simulate ecological effects using current ecological modelling capacities. In the field of modelling, two approaches can be used to perform the latter simulations: mechanistic (i.e. food-webs) and data-driven modelling (e.g., artificial neural networks for habitat suitability modelling). In this research an attempt is made to combine the strengths of both approaches by applying each of them in the proper context, i.e. the first approach is expected to serve for WFD purposes, while the second approach seems more suitable in the REACH context. Both approaches are evaluated with a SWOT analysis and their use will be demonstrated with a case study.

INTRODUCTION

Recent EU-legislation, like the Water Framework Directive (WFD) and the Registration, Evaluation and Authorisation of CHemicals (REACH) respectively aim at protecting European surface waters and assessing the environmental effects of chemicals in surface waters. Conducting field experiments to fulfil these tasks may be a too expensive approach, given the extent of this legislation. The potential of ecological models for use in ecological effects assessment has been shown in Barnthouse *et al.* (1986), Barnthouse *et al.* (1992) and Suter II (1993). In fact, ecological modelling may be the only option for assessing chemical effects under circumstances where field experiments cannot be conducted (Naito *et al.*, 2002). In general, two approaches can be followed when performing (ecological) modelling: the mechanistic and the probabilistic approach. The first approach mathematically synthesizes available knowledge into a predictive framework (e.g., Park and Clough, 2004), while the latter consists more of a data-driven process (e.g., Borsuk *et al.*, 2004). Which approach will be preferable in a specific case depends on the required properties of the model. Therefore, the characterization of the strengths, weaknesses, opportunities and threats (SWOT) of both approaches is required. The goal of the presented work is to make this analysis for both approaches and compare the findings in order to obtain an optimal framework in which both approaches are combined. A hypothetical case study will be performed to demonstrate the strength of this combined approach in ecological effect assessment.

1 SWOT ANALYSIS OF MECHANISTIC ECOSYSTEM MODELS

1.1 Strengths

Mechanistic ecosystem models can be thought of as simplifications of current knowledge about ecological processes, which enable to simulate at least general trends of ecosystem behaviour (Bartell *et al.*, 1992). This knowledge based approach in ecosystem modelling can act as a filter for erroneous data-driven model simulations in two ways. First, the possible range of simulations when different parameter values are set will be limited. This can be demonstrated by the following example: if a batch culture of algae is supplied with a limited amount of nutrients, the size of the algal population will be limited to a maximum value. As such, the model will never simulate the population to be larger than the latter defined maximum, no matter what values the parameters of the model are set to. When further developing the model for its use in a specific case, the knowledge-based skeleton is provided with appropriate parameter values which permits the simulation of the phenomena of interest. Since these parameters in mechanistic models are used to describe knowledge-driven processes, the range of possible parameter values is limited to what is biologically possible and as such simulations are filtered in a second way. Another advantage is the structure of these models, i.e. an aggregation of different known processes, facilitating communication of modelled processes (i.e. model components) between scientists. Different implementations of the same process can be compared without the need for a thorough investigation of the complete models.

1.2 Weaknesses

Ecological modelling in general and ecosystem modelling in particular is associated with high costs and a high demand of expert knowledge input (Pastorok *et al.*, 2002). High costs are mainly due to the time necessary to obtain simulations which give a fair approximation of observed system behaviour and to the monitoring effort of these observations (Alewell, 1998). Input of expert knowledge is associated with the difficulty of finding appropriate parameter values for the implemented processes and as such with the risk of yielding an overparameterized model (Omlin, 2001). Most mechanistic ecosystem models are characterized by an extensive list of parameters: reported parameter numbers from published ecosystem models range from 25 (Hanratty and Liber, 1996) to 120 (Zhang *et al.*, 2004). If these parameters are not known or can not be estimated with enough precision, they rather contribute to output uncertainty (Jørgensen, 1995) instead of assisting in the elucidation of observed ecosystem behaviour.

1.3 Opportunities

In view of recent EU-legislation as the Water Framework Directive (WFD) and the Registration, Evaluation and Authorisation of CHemicals (REACH), there is an opportunity for mathematical models in general in assessing anthropogenic effects on ecosystems (Campbell and Bartell, 1998). The WFD primarily focuses on the maintenance of a good chemical (i.e. exposure related) and ecological status (i.e. biodiversity related), while REACH requires the assessment of effects of chemicals on the environment. Especially REACH seems to create the niche where mechanistic ecosystem models can operate, since these models provide insights in ecosystem functions and hence in the way how these are affected by chemicals.

1.4 Threats

The timing of the cited legislation may be the most important threat for the use of mechanistic ecosystem models in a regulatory context. Given the extensive list of parameters, as discussed above, there is a need for standardized calibration procedures in mechanistic ecosystem modelling, although some efforts have been done by Jorgensen *et al.* (2002), Loehle (1997) and others. Only in a minority of papers the used calibration procedure is reported and, when mentioned, solely consists of a simple visual comparison of observations and simulations. This approach, resulting from a lack of available general calibration tools, gives rise to excessive calibration efforts. The need for calibration on a case to case basis therefore threatens the feasibility of the use of mechanistic ecosystem models.

2 SWOT ANALYSIS OF DATA-DRIVEN ECOSYSTEM MODELS

2.1 Strengths

Because of the variety of techniques in data-driven ecological modelling, strengths mainly depend on the used technique. Fuzzy logic based methods (Zadeh, 1965) introduce the advantage of linguistic properties. Other techniques (e.g., Bayesian Belief Networks, as in Trigg *et al.*, 2000) produce models which have so called learning capability if large data sets are available.

2.2 Weaknesses

Data-driven models mainly rely on the availability of good quality monitoring data. Unlike mechanistic models, predictions mainly depend on monitoring data sets and the use of rather subjective expert knowledge (Adriaenssens, 2004).

2.3 Opportunities

As cited above, recent EU-legislation creates opportunities for ecological modelling. Mechanistic models, however, are merely capable to calculate water quality variables and some biomass information about the biota present. Details on the composition of these communities are mostly not modelled by the latter type of models. Since the WFD primarily focuses on the maintenance of a good ecological status, models describing habitat suitability may assist in this.

2.4 Threats

Apart from the threats which are listed in Goethals (2004), the difficulty of choosing the appropriate technique when performing data-driven ecological modelling is an extra difficulty. An overview of the latter is given in Verdonschot and Nijboer (2002)

3 COMBINED APPROACH

In this section, we will propose a new way forward and elaborate one example to demonstrate an efficient combination of both modelling approaches. A hypothetical example is developed.

3.1 Methodology

Long-term dynamics of populations under anthropogenic stress in a river ecosystem, may be of interest to regulators to support river management. Assuming biodiversity studies are available, preferably at or near locations of interest, a mechanistic ecosystem model then allows to simulate the dynamics of the populations which are known to be present from the biodiversity data. However, the latter model would not account for changing habitat suitability as a result of anthropogenic disturbances. Therefore, an artificial neural network (ANN) was used to predict habitat suitability of the upper part of the Zwalm river given some measured structural and physical variables. These results serve as an indication of the ecological status of the system, as defined in the WFD and can subsequently be used as input for a mechanistic ecosystem model, as developed by De Laender *et al.* (in preparation). The latter model is implemented as an object oriented structure in WEST® (Hemmis, NV). Only those populations whose presence is predicted by the ANN are implemented in the mechanistic model. This allows for the simulation of the dynamics of these populations, given the environmental boundary conditions and chemical contamination due to anthropogenic disturbances. Results obtained in this phase of the combined framework can serve as an effect estimate on the ecosystem given a defined exposure concentration of the chemical of concern. The latter can be used as the basis of ecological risk assessment. In Fig 2, the flow chart of this methodology is shown.

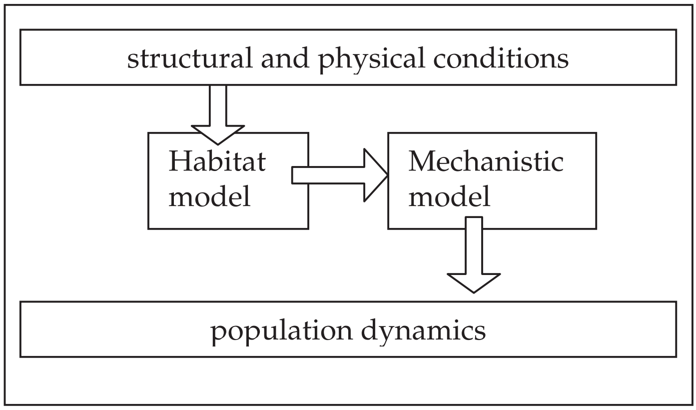

Figure 2. Flow chart of proposed methodology

4 DISCUSSION AND CONCLUSIONS

The presented methodology can be seen as an integrated way of ecological modelling, needed for the proper assessment of anthropogenic disturbances in aquatic ecosystems. Next to the discussed in this paper, both models can also be combined in some other ways. An example is the use of a mechanistic model as an explanatory tool for important factors in the used ANN. Herein, the predictive power of ANN is combined with the explanatory power of mechanistic modelling. The general strength of this combination is the incorporation of habitat defragmentation into ecological effect assessment. As stated by Bartell *et al*. (2003), environmental effects also consist of effects due to habitat degradation and as such, only models which also incorporate the latter can be thought of as highly realistic. Care has to be taken, however, that underlying assumptions of both models are not conflicting. Some work has been done by Mackay and Robinson (2000) to examine the latter. Opportunities for these models are created by the cited legislation. The most limiting factor may be time and data restrictions, although this also holds true when both models would be used separately.

ACKNOWLEDGEMENTS

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Ecological models to support the implementation of the European Water Framework Directive: case study for the zebra mussel as indicator for the ecological quality for macrobenthos

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ABSTRACT: As a consequence of the European Water Framework Directive there is a growing interest for ecological modelling tools to support the implementation of this directive. There is an urgent need for modelling tools to analyse and assess the relationships between the biological quality elements and the hydro-morphological and physical-chemical properties of the water bodies. This study is an assessment of applicability of habitat model for the Zebra Mussel (*Dreissena polymorpha*) on the basis of numerical performance indicators, sensitivity analyses and practical simulation exercises. The model was applied in Lake IJssel to test their usefulness for assessment of the ecological quality of the water bodies and to design measures to improve the ecological status.

1. INTRODUCTION

The main objective of the European Water Framework Directive (WFD) is to achieve good chemical and ecological status by the year 2015. The ecological status is determined by four biological quality elements: phytoplankton, macrophytes, macroinvertebrates and fish. In addition, hydrodynamic and physico-chemical quality elements have to taken into account as well. One of the key scientific premises of the WFD is that relationships between the biological quality indicators and both hydromorphodynamic and physico-chemical properties of surface waters are sufficiently well understood to enable cost-effective management of rivers, lakes and coastal waters to achieve the ecological objectives. Although some research has been carried out to establish this kind of relationships, our present understanding of the links between the biological quality elements and both water quality and hydromorphology, is generally not adequate to support measure programming to target the ecological objectives (Heiskanen & Solomini, 2005; Duel et al., 2005). There is a requirement for a better understanding of the relationships between anthropogenic pressure, water quality and biological response, so that effective programmes of measures can be introduced to enhance and protect aquatic ecosystems. In addition, there is an urgent need for ecological modelling tools to assess the effectiveness of measures to improve the ecological quality of rivers and lakes at risk. In this paper, different habitat modelling techninques are presented that can be used to analyse and assess the ecological quality of rivers and lakes. Habitat modelling techniques are widely accepted approaches for ecological assessments of rivers and lakes in both the Netherlands and Flanders (Duel et al., 2003; Goethals, 2005). However, habitat models are moastly developed for fish species. However, the number of models for macroinvertebrates as well as macrophytes is increasing (Jorde et al., 2001; Duel et al., 2003; Goethals, 2005). In this paper, the applicability of habitat models for macroinvertebrates is demonstrated using the Zebra Mussel (Dreissena polymorpha) as an indicator for ecological quality of water systems. There are several techniques to develop habitat models. This paper will focus on expert knowlegde based habitat models.

2. MATERIAL AND METHODS

2.1 Approaches

Although habitat models are generally accepted tools to assess the ecological impact of restoration or rehabilitation measures, the realibility of habitat models is often unknown. For acceptance of the habitat modelling results by the water managers, it is essential to assess this reliability. There are two approaches to analyse the reliability of habitat models: (1) to carry out a sensitivity and uncertainty analysis to address the uncertainty in the preference curves in the habitat models (2) to validate habitat models by comparing the models results with field data.

For large fresh water lakes, a habitat model for Zebra Mussel was developed by Duel and Specken (1993) based on habitat preference data of Central European lakes and expert knowlegde. The performace of this model was tested using Lake IJssel in the Netherlands as a case study. A panel of experts was asked to estimate the uncertainty range of the parameters of the preference curves used in the habitat model. Uncertainty analysis was carried out using Monte Carlo simulation tests to quantify the uncertainty in predicted habitat suitability. In addition the model results were compered with data from a monitoring programme.

2.2 Study site: Lake IJssel

Lake IJssel is with its 1200 km^2 one of the largest fresh water lakes in Western Europe. It is supplied with water by the river IJssel, the northern branch of the river Rhine in the Netherlands (Figure 1). With respect to biodiversity, Lake IJssel is an area of international importance (Wolff, 1989). Due to a large fish biomass and a high density of fresh water mollusks large numbers of piscivorous birds and molluscivorous birds are present in the area, especially during the winter period (Van Eerden, 1998). The macroinvertebrate community is dominated by Zebra Mussel (*Dreissena polymorpha*). This species has colonised Lake IJssel shortly after damming the Zuiderzee, a coastal water in connection with the Wadden Sea. According to the requirements of the WFD, Lake IJssel is designated as a heavily modified water body. In this study, the Zebra Mussel is selected as a biological indicator for ecosystem quality of Lake IJssel.

2.3 Modelling tool

The uncertainty analysis of the habitat model for Zebra Mussel in Lake IJssel was carried out using HABITAT modelling software. HABITAT is a GIS-based framework application that allows for the analysis of ecological functioning of study areas in an integrated and flexible way. HABITAT can be applied to analyse the availability and quality of habitats for individual species. Moreover, it can be used to map spatial ecological units (e.g. habitats, ecotopes) and predict spatial changes in habitat suitability for example due to human interventions. Users can use predefined habitat evaluation modules for individual species, or can define new modules to suit their needs for specific applications.

Figure 1. Location of Lake IJssel in the Netherlands.

2.4 Habitat model evaluation

The confusion matrix (Fielding & Bell, 1997) was used as a starting point to evaluate the model performance. Evaluation of the results was based on two criteria, the percentage of Correctly Classified Instances (CCI) and the weighted Kappa ( ) (Cohen, 1960; Fleiss & Cohen, 1973). The CCI is defined as the number sites where the modelled habitat suitability class was the same as the monitored one, divided by the total number of sites. The weighted Kappa is a simply derived statistic that measures the proportion of all possible habitat suitability classes that are predicted correctly by a model after accounting for chance.

3. RESULTS

The habitat suitability for Zebra Mussel in Lake is sensitive (limiting) for water depth, substrate and mud layer. Surprisingly, the habitat suitability for Zebra Mussel is not determined by water quality parameters. Although the uncertainty ranges in the preference curves estimated by the panel of experts were sometimes up to an index value of 0.4, the average habitat suitablity (figure 2) is comparable with the modelling results without uncertainty. As the standard deviation is used as a measure for uncertainty, the standard deviation is less than an index value of 0.1 in this study.

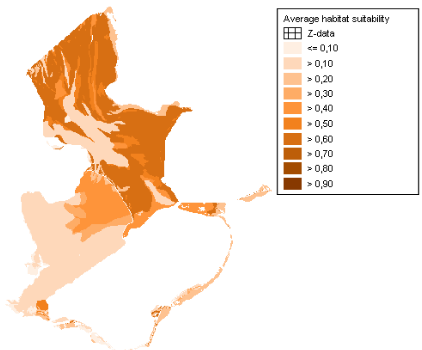

Figure 2. The average habitat suitability (0=not suitable; 1=optimal habitat quality) for Zebra Mussel in Lake IJssel based on 200 Monte Carlo simulation runs (from Duel et al., 2003).

Since an important criterion for the acceptance or rejection of the model results is the comparison of the model results with field observations, the results of the model simulations for zebra mussel were compared with the densities of zebra mussel observed in field surveys. The results are summerised in table 1. Based on the confusion matrix, CCI is 47% and Cohen's Kappa value of 0.21. When the habitat model is predicting high suitability index, high mussel density is found in 34% of the monitoring sites as well. When low suitability is predicted, the misclassification rate is low: less then 10%.

Table 1. The validation of the expert based habitat model for Zebra Mussel by compering model results and field monitoring data.

Predicted suitability	High	low
Monitoring results		
High density	26%	2%
Low density	51%	21%

4. DISCUSSION AND CONCLUSIONS

Sensitivity and uncertainty are important criteria for credibility of models (Van der Molen 1999). During sensitivity analyses the appropriateness of the model structure can be examined and during uncertainty analysis the uncertainties in model predictions can be addressed and quantified to a certain extent. A measure for the acceptability of the uncertainties in habitat modelling results for applications in water management is the probability that the uncertainties result in a range of an index value of less than 0.1 (Duel et al, 2000; Van der Lee et al., 2002). When more than 90% of the calculations result in an uncertainty of less than 0.1, the habitat model is considered to be suitable for applications. Considering the application for Lake IJssel, the habitat model for Zebra Mussel is acceptable. On the other hand, there is a striking misclassification for high suitability predictions. This is probably caused by wintering diving ducks as they feed on Zebra Mussels during the wintering period. Predation by diving ducks was not included in the habitat model for Zebra Mussel.

Since uncertainty is an important criteria for credibility of models and model results, it is re-commended to include uncertainty analysis in the procedures for the application of habitat evaluation measures and to make the uncertainties in habitat evaluation for every application explicit.

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Scaling: patterns of with-site and between site variation from hydraulic and macroinvertebrate datasets

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ABSTRACT: In trying to understand how water abstractions and dam operations impact river ecology, we first need to understand the effects of natural flow variability. Habitat models have often been applied to assess impacts, but they do not really give us adequate information about a desirable regime and the significance of seasonal flow events, particularly if the emphasis is on communities rather than single species. Many countries have historical biomonitoring datasets, particularly for macroinvertebrates, which can be used to help elucidate such relationships. However, there are several difficulties that have to be overcome when one uses such datasets. Firstly we need to define flow-sensitive ecological metrics which should respond to hydrological variation. In the UK this has already been done with the LIFE methodology, this has also been taken up by the EU STAR project. Secondly, it is the within-a-site variation that we are interested in, however often the between-site variation masks the interesting relationships. In this paper I discuss how we can make these linkages using a technique known as mixed effects modelling. We recognise that there are multiple levels of variation within these large datasets, and partition this variation accordingly. Similar techniques can be applied to when physical habitat data are aggregated across sites, as we instinctively know that habitat varies strongly both between and within sites, but overall which is more important? Understanding these patterns for physical habitat data is the first step to developing physical survey methods which allow us to work at larger scales, but which also remain quantitative.

1 INTRODUCTION

In trying to understand how water abstractions and dam operations impact river ecology, we first need to understand the effects of natural flow variability. Habitat models have often been applied to assess impacts, but they do not really give us adequate information about a desirable regime and the significance of seasonal flow events, particularly if the emphasis is on communities rather than single species. Also, with the coming of the Water Framework Directive, there is increasing interest in broad-scale screening assessments of pressures and impacts, and also in typologies. Many countries have historical biomonitoring datasets, particularly for macroinvertebrates, which can be used to help elucidate such relationships.

However, there are several difficulties that have to be overcome when one uses such datasets.

- We need to define flow-sensitive ecological metrics which should respond to hydrological variation.
- We need to “unmask” the interesting within-site links between flow and the community, which can easily be masked by between-site differences.

In this paper I outline the basis of the LIFE score, and the need to generalise its response. I then go on to describe the usefulness of mixed effects models for answering these kinds of questions, with examples from three case studies, all of which are still in progress.

1.1 The LIFE Index

The LIFE index (Lotic Invertebrate index for Flow Evaluation, Extence et al., 1999) was formulated to test whether it is possible to link changes in benthic invertebrate community structure with indices of historical river flow at a gauge close to the sample site. The LIFE index can be calculated from species or family-level bio-monitoring data. Every taxon is assigned a velocity preference (termed a flow group by Extence et al.) from I to VI (based on literature data), and five abundance categories are used. Implicit in the 'velocity' preference is preference or avoidance of silty substrates. A matrix is then used to give a combined score for each taxon in the sample of between 1 and 12. The scores for all taxa are added together, and the average score is the LIFE index. It is important to note that the index *is expected to* be sensitive to natural and artificial flow changes; it thus allows an extrinsic hypothesis to be tested. This approach is different to the one commonly adopted in multivariate statistics, which allows more freedom for the data to describe any patterns it likes (ter Braak and Šmilauer, 1998; Clarke and Warwick, 2001). There are arguments for and against both approaches, however it cannot be denied that management agencies need simple indices of community response, and that modelling the response of such simple indices to natural and anthropogenic variation is simpler than modelling multivariate responses.

Extence *et al.*, 1999, demonstrated that correlations exist between LIFE score and moving averages of historical flows (e.g., Figure 1). LIFE is currently being used in England and Wales as part of the implementation of Catchment Abstraction Management Strategies and the Water Framework Directive.

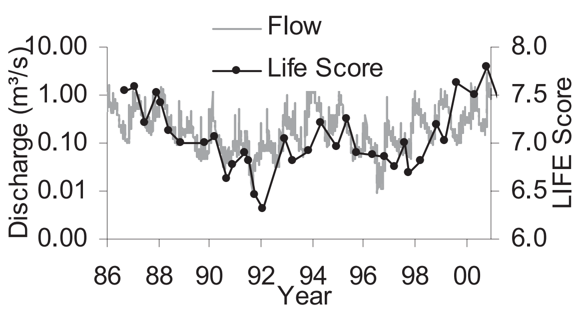

Figure 1. Example LIFE score and river discharge time series. Note the log scale on the left.

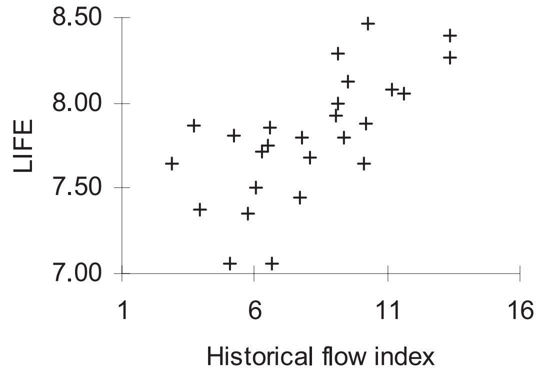

Figure 2. Example relationship between LIFE score and historical flow at one site.

Using simple regression or correlation analysis, the initial work on the LIFE score demonstrated that it responded to flow, whether calculated at species or family level. However, these relationships were defined on a site by site basis, where there were good runs of data, with samples taken for 15 years or more. To prove more widely applicable, the method needed to be demonstrated across a wider range of sites, which are likely to have fewer samples. Such monitoring sites also differ naturally in their LIFE score, and in their habitat quality.

1.2 The need to generalise LIFE across sites

What was needed was a single model, or a few models, which integrated data across many sites. There are formidable challenges to achieve this, particularly to separate out any signal from noise. Ideally, rating curves would be produced for each site, to convert discharge into velocity as the driving variable. As these are not available, a more basic standardisation of flows was used, standardising each site's flow record by dividing by long term mean flow. For each sample date, flow indices were summarised by chosing six month periods before the sample was taken, and then calculating a flow duration curve for that six months, and extracting Q95, Q50 and Q10. Obviously there are many other options for generating flow indices, but that is not the subject of this paper, which is rather about the general techniques.

Mean LIFE score also varies between sites. This could be because of natural factors, in which case it is possible to use the RIVPACS model (Wright et al., 2000) to derived an observed / expected LIFE score. However, LIFE should also be affected by artificial flow alterations, or habitat quality, and in some cases, the overall mean score is likely to be of less interest than the slope of its response to flow, this is certainly the situation in Case 1, described above and below.

1.3 Hydraulic diversity in rivers

Habitat models have been widely applied to determine environmental flows on a case by case basis. The hydraulic component of habitat models grew out of one-dimensional approaches designed for engineering studies. Yet even in the early days, it was recognised that for habitat purposes, it is important to consider both longitudinal and lateral hydraulic diversity. Aside from a few exceptions (e.g. Stewardson and McMahon, 2002, Lamouroux and Jowett, 2005), little progress has been made in synthesising the underlying data from such studies to deduce general patterns in hydraulic diversity, although some progress has been made with modelling longitudinal hydraulics (e.g. Jowett, 1998). Analysing hydraulic diversity patterns between sites presents multiple levels of variation: lateral and longitudinal, as well as variation with flow. This would be difficult to model in a traditional linear modelling framework, but the benefits from such as model are considerable, not least because generalised hydraulic models would be a useful additional component to models of how the biotic community respond to flow, for example the LIFE score models mentioned here.

1.4 Mixed effects models

Mixed effects models are a class of statistical model which allow efficient parameterisation when there are nested (or hierarchical) data structures and sources of variability where we do not explicitly select their levels, but want to model a broader population. So in the first two cases considered in this paper, at the lowest level in the hierarchy, we have a basic regression between historical flow in the months before a sample, and the LIFE score of the sample. There are several samples from a site, somewhere between five and fifteen, and we restrict ourselves to autumn samples only. Stepping up one level in the hierarchy, we look across sites. Our set of sites are just a fortuitous sample, they were selected because they had enough data. If we are interested in how LIFE score varies with flow, site is to some extent a nuisance factor, there are many differences between sites that we can at best partially control for.

There are at least three options to analyse a dataset such as this. We could undertake a separate regression for each site. However, because of the relatively few data points per site, the slopes of these regressions will often be highly uncertain, and we are swamped by noise. Alternatively, we could undertake one regression, with site as a factor. This has the advantage of using a single overall error variance, adding power to the analysis. The simplest model would be to allow a coefficient for each site to adjust the site's intercept (ie mean LIFE score). Going further, one might allow both intercept and slope to vary by site.

However, we are beginning to make ourselves a problem, to model site to site variation we have one or two parameters per site in the dataset. If we have 100 sites, then we have 100 parameters describing the intercepts, and possibly another 100 describing slopes. Such a model has two main disadvantages, firstly it is inefficient, in the sense that we are using up degrees of freedom in this site factor which we would prefer to keep for the error term in the model.

Secondly we are estimating parameters for the sites in the dataset, often we are more interested in the site to site variation in general. If we only have a few samples per site, it becomes impossible to set up such a model and we are left with the situation of having lots of data points but we cannot interpret what they mean.

Mixed effects models offer a way forward in this situation. They are mixed in that they allow us to include both "fixed" and "random" terms in the analysis. Continuous variables are fixed terms, as would be categorical variables, when the levels of the categories mean something. So a variable like habitat type would also be fixed. But variables such as site, or replicate within site, whose levels don't mean anything (ie Site1, Site2 etc), can be considered "random", and modelled with an normal distribution with two parameters, an overall mean and a standard deviation. So in the case of describing how mean LIFE score varies across sites, we can model this as a normal distribution. Similarly, between-site slope of the LIFE response to flow can also be modelled as another normal distribution with two more parameters. There needs to be one further parameter, the covariance between intercept and slope, but this can be minimised by centering the explanatory variables to have mean zero, this is implicit in the flow standardisation mentioned above anyway.

To put this more formally, we have:

Y_{ij} is raw or O/E LIFE score for site j, observation (time point) i,
$i = 1...n_j$ where n_j is number of observations for site j
$j = 1...N$ where N = number of sites
X_{ij} is our within-site explanatory variable such as standardised flow.
N(x,y) signifies a normal distribution with mean and standard deviation parameters x and y to be estimated by the model.

Jumping straight to the more complex random slope and intercept model due to lack of space, we can formulate this as:

$$Y_{ij} = \beta_{0j} + \beta_1 X_{ij} + e_{ij}$$

Where:
$\beta_{0j} \sim N(\beta_0, \sigma^2_{b0})$
$\beta_{1j} \sim N(\beta_1, \sigma^2_{b1})$
$cov(\beta_{0j}, \beta_{1j}) = \sigma^2_{b01}$
$e_{ij} \sim N(0, \sigma^2_e)$

Mixed effects models also provide a more general framework to handle nested data structures, which are common in environmental data. These can be handled with least squares linear modelling only so long as the data are balanced (this is rarely the case in observational studies): least squares also does not handle random effects themselves, instead giving "fixed" parameter estimates.

On the downside, parameter estimation is more complex, and there is no analytical solution corresponding to the least squares regression cases above. Instead, models are fitted by an iterative algorithm (generalised least squares), which is able to partition variation between and within groups. This does mean that there is no direct equivalent of the R^2 measure so beloved of many people, however it should be noted that R^2 is NOT a good measure of model performance or fit. Simple assessment of mixed-effects models can be under taken by comparing parameters with their standard errors, with a parameter value of more than 2x the SE being a useful guide to significance (for comparison, this can also be undertaken for traditional linear regressions). Models can be compared using a likelihood ratio test, this can be used (with care) to compare traditional linear regressions with their equivalents using mixed/random effects.

2 CASE STUDIES

2.1 Does LIFE respond to modelled flow?

Two issues that we frequently encounter are:

- we often have many biological monitoring sites which are not close to gauging stations
- these sites often have a few samples, often fewer than say 10.

We would dearly love to use such data in deducing flow-ecology relationships. Although to some extent, each sample site is unique, we hope that there is some commonality in the relationships, indeed this generality is the cornerstone of high-level policies such as the WFD.

Gauged flow data can be transposed to ungauged sites, if the artificial influences between the gauge and site can be quantified. Mixed effects models can then provide a useful way of pooling the data across sites, whilst still allowing each site to retain some of its own character.

Here the utility of the approach is illustrated with data from six sites on the River Enbourne, a tributary of the Thames in the UK. Each site only has samples in a few years. We would like to allow the intercept (ie the mean LIFE score) to vary across sites, and to test whether there is a significant slope to the LIFE-flow relationship, and whether it differs between sites. Undertaking separate regressions for each sites gives predictable results, some slopes are significant at p=0.05, others are not. Undertaking one regression, including a site x flow interaction, tells us that overall slope is not significant (p=0.22), although one site has a particularly steep slope. These two approaches represent opposite ends of a continuum. Separate regressions have little power as there are few data points per site. A single regression pays no attention to the nesting of the data and only has one residual term when in fact there are two: within sites (associated with flow), and between sites. Both of these approaches use up 12 degrees of freedom for parameters, as there are 31 data points this leaves 19 dfs for the residual term.

The results of a mixed effects analysis are interesting. There are fewer parameters (5 instead of 12), and at this level, the p-value for the LIFE-flow relationship is much less, 0.056 this is because the slope parameter is greater (Table 1).

Table 1. Comparison of LIFE – flow models incorporating only fixed, and both fixed and random effects.

Model	Slope	SE slope	Pr(t)
a. Fixed, flow*site	0.29	0.23	0.22
b. Fixed (flow), random (site)	0.47	0.23	0.05

Using likelihood ratio tests, we can compare models with and without random effects, and models with varying numbers of random effects (intercept only, or intercept and slope) (Table 2).

Table 2. Comparison of fixed vs random and two random effects models

Model 1	Model 2	P value
Random intercept and slope	Fixed, flow*site	<0.0001
Random intercept	Random intercept and slope	0.08

Figure 3 gives a graphical representation of some of these results. The mixed-effects approach allows the regression line for a site to "borrow" from the relationships across all the sites, giving less extreme lines for the more extreme cases, and those sites with more scatter in the data.

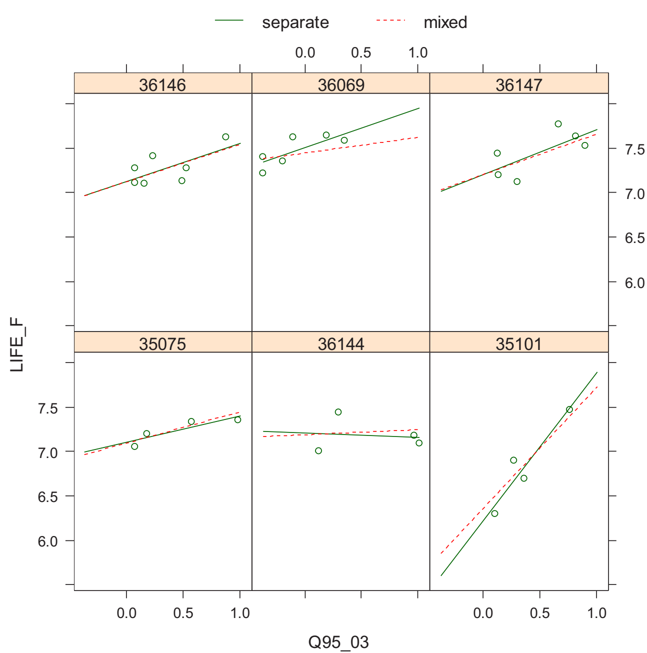

Figure 3. Comparison of regression lines from a mixed effects model and separate regression lines for each site. Each panel represents a site. The x-axis is historical flow in the three months before the sample, the y-axis is LIFE score.

2.2 *Are some rivers more or less sensitive to flow?*

The requirement of the European Water Framework Directive for the identification of sites at risk of failing to achieve Good Status means that simple screening criteria are required to assess, amongst other pressures, flow regulation. It is logical to suggest that screening criteria for waterbodies at risk should be based on some proportion of a flow statistic, for example a proportion of mean flow or Q95. However, should the same criteria be used for all water bodies, or are some more sensitive than others? Although sensitivity to natural flow variation and sensitivity to flow regulation are not necessarily the same thing, understanding the former should tell us something about the latter, especially given our imperfect knowledge about the latter. One measure of sensitivity of a site to flow could be the slope of the flow-LIFE score relationship: a steeper slope implies a site where the ecology is more sensitive to flow. As usual, our estimates of the slopes for any individual site can be highly uncertain.

I used a dataset of 190 macroinvertebrate monitoring sites to see whether differences in slope could be related to environmental variables, specifically those commonly used in predicting reference conditions and also those catchment characteristics controlling hydrology. Here the model intercept is a nuisance parameter, we know that sites vary in their mean LIFE score, for example because of (unmeasured) habitat diversity, but what is of real interest is the slope of its response to flow. A fixed effects model with 190 (uncertain) parameters for intercept and another 190 for (uncertain) slope is rather unmanageable compared to a mixed effects model with 5 parameters. This is particularly the case as we are interested in the relationships for sites in general, not just the sites in our dataset. This analysis showed that when flows were expressed as a proportion of the mean flow, the slopes of response of LIFE to flow were greater in more “upland” catchments.

2.3 *Hydraulic variability at various scales*

This final example concerns variation in river depths and velocities. The WFD requires us to manage rivers at the catchment scale, and to monitor key hydromorphological variables. Yet most knowledge about how these variables influence river ecology is only applicable at small scales of say 5-20 channel widths. If we are to upscale or to work at larger scales, we need to know something about the larger scales over which water depths and velocities vary. Many institutes hold data from reach-scale habitat modelling surveys, it would be nice if we could use our existing data from habitat models to tell us something about larger scale variability. Two key issues are:

1. Depths and velocities also vary with flow
2. Depths vary in two dimensions and and velocities vary in three dimensions

Following the principle of building up models one step at a time, I will combat issue 1 by considering hydraulic data at Q50 only, and issue 2 by only considering mean cross-sectional depths and velocities. Both of these assumptions can be relaxed at a later date.

A very simple model can be built which considers for each of depth and velocity, their variance components within and between sites. This is virtually the simplest random effects model, having a single random effect: site. Variation within sites goes into the "error" term, and there are no fixed effects. The model is then

$$Y_{ij} = \beta_{0j} + e_{ij}$$

Where:
$i = 1..n_j$ cross sections for each site
$j = 1..N$ sites
$\beta_{0j} \sim N(\beta_0, \sigma^2_{b0})$
$e_{ij} \sim N(0, \sigma^2_e)$
Y is either depth or velocity.

The two σ^2 terms are the variance components we are interested in, and as they are variances, the total variance in the data is the sum of the individual variances. If $\sigma^2_{\beta 0}$ is high compared to σ^2_e, then between-site variance is dominant and we should perhaps be collecting our data points more widely-spaced. If the converse is true, then our strategy of habitat modelling relatively short representative sites is perhaps valid. An analysis of 64 PHABSIM sites from the UK, collected by CEH, University College Worcester, and WS Atkins Consultants, gave the results in Table 1.

Table 1. Variation in mean cross-section depth and velocity within and between 64 UK PHABSIM sites

Variable	Depth	Velocity
$\sigma^2_{\beta 0}$: between site variance	0.019	0.009
σ^2_e : within site variance	0.21	0.018
$\sigma^2_e / (\sigma^2_{\beta 0} + \sigma^2_e)$ (%)	53	67

The results show that the proportions are not the same for depth and velocity, for depth it is very nearly 50:50, whereas for velocity, 2/3 of the variability in the dataset is within sites. These results suggest that there is no easy shortcut to sampling at larger scales and that larger scale sampling programmes could run the risk of confusing smaller and larger scale variation.

3 CONCLUSIONS

These are all preliminary results, compiled in haste to give readers a flavour of the generality and power of mixed effects models. These models are efficient, and allow one to make best use of data where there are nested levels of variation and where samples are drawn from a larger population of interest. They are particularly flexible with uneven replication which is all too common in environmental data. Not recognising nesting when it is present can make for erroneous conclusions. Mixed effects models can also be extended to model cross-correlations and non-constant variance structures.

A few years ago, the SAS package was the only route into mixed-effects models, and its arcane syntax and idiosyncrasies could be daunting and off-putting. Now, other packages are available, for example add-ons for the R and S-Plus computing environments, and stand-alone, for example MLWin. These are still not necessarily easy, some of the concepts are challenging, they are however extremely flexible. Nevertheless, it is high time that such models received more attention in environmental science, and in hydro-ecology in particular.

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RiverSmart: A DSS for River Restoration Planning

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ABSTRACT: DSS RiverSmart is a computer based decision support system for hydromorphological, ecological, and cost-effectiveness evaluation of river engineering measures. A new approach based on site-specific abiotic and biotic reference and expert knowledge is used to evaluate the status quo as well as scenarios of engineering measures. To apply the model, a limited amount of field data is needed. Application scale varies from individual river stretches of approximately one kilometer to river segments of tens of kilometers. Small scale field data of high resolution, e.g. results of abiotic/biotic monitoring, numerical hydraulic, morphodynamic and microhabitat modeling are used for calibration. The tool has the potential to explore and develop River Basin Management Plans and programs of measures as well as hydropower schemes in coherence with the ambitious goals of the European Water Framework Directive. Yet, DSS RiverSmart was applied at the river Drau and Traisen, Austria.

1 INTRODUCTION

To manage rivers in a watershed context is an old postulation of many river ecologists. With the release of the European Water Framework Directive (WFD) in December 2000, it is nowadays real European water policy. This new era in Europe's approach to rivers is actually challenging water administration, river managers and hydropower companies throughout the EU.

Researchers and engineers have to support the implementation of the WFD with sound solutions for several scientific and technical problems, and especially with respect to the strict timetable of the WFD. In the first phase of implementation, research groups, mostly teams of specialized aquatic biologists only, have focused on the development of IBI's (Indices of Biotic Integrity, Karr 1981) to be used for the assessment of waterbodies. Assessment methods on biotic indicators (fish, benthic invertebrates, macrophytes and phytobenthos, phytoplankton) will certainly play a key role in monitoring programs to find, out whether a river is at critical state or not. However, River Basin Management Plans (RBMP), including detailed programs of measures, have to be developed and implemented until 2009. We postulate that biotic monitoring and IBI's alone cannot deliver the complete information necessary for the set-up of those RBMP's. In addition, models and tools for ecological predictions based on abiotic drivers, which are responding to engineering measures, are needed. Only such tools will allow the comparison of potential programs of measures - a relevant strategic planning aspect in the phase prior to the final release of a fixed RBMP.

Physical habitat simulation models like classical PHABSIM (Bovee 1982) or it's European relatives - e.g. HABITAT (Alfredsen 1996), CASIMIR (Jorde 1996, Schneider 2001) or 5M7 (Le Coarer & Carell, 2000) - already support the abiotic issue. However, these models have been developed for applications on the local scale (tens and hundreds of meters, "project-scale"). Consequently, in real planning-practise those models have been applied on the river segment scale or in the above mentioned context just sporadically (EAMN, 2004). Upscaling of those models and tools is consequently a major task, the ecohydraulics research community faces today (EAMN, 2004). With regard to this, our inter-disciplinary project team started to develop the decision support system DSS RiverSmart (Strategic model for the analysis of rivers based on typologies). Goal of this project is to develop a tool which

- allows a quick and cheap pre-evaluation of ecological benefits and costs of potential river engineering measures,
- which is coherent to the approaches and needs of the WFD,
- which complements biotic monitoring assessment methods and already existing model capabilities for detailed project planning (e.g. microhabitat models),
- which is applicable to individual river reaches and large river segments.

This paper explains and discusses the basic concepts and evaluation methods so far developed within the DSS RiverSmart project. A case study application at the river Drau, Austria, is given by Angermann et al. 2005.

2 CONCEPTUAL MODEL FOR THE RIVERINE LANDSCAPE

In general, river and floodplain ecosystems or "riverine landscapes" can be described as a complex of three partial systems: (i) the aquatic part which is represented by the permanent water bodies (main channel, side channels, oxbow lakes etc., depending on the size of the river), (ii) the amphibian part represented by the riparian zone, river banks, gravel bars, shallow islands, etc., and (iii) a semi-terrestric/terrestric part represented by a more or less wide floodplain zone. Within DSS RiverSmart we followed another functional hierarchical approach to describe the global system. The ecosystem "riverine landscape" is built up in this representation by three system entities: (i) Level I - “system elements“, which are the real physical elements; (ii) Level II - “system dynamics“, which represents temporal variability of water, substrate, soil, nutrient and energy flux in the ecosystem and (iii) Level III - “system connectivity“, the spatial-temporal network between the system elements.

The abiotic parts of the system entities are hierarchically further defined by specific "abiotic components". Those again are further determined by 29 selected "abiotic parameters" (Figure 1).

Level I	Level II	Level III	Level IV
Ecosystem	**System entities**	**Abiotic system components**	**Abiotic parameters**
River-ine land-scape	System elements	Hydrology	Magnitudes of average discharges Duration of average floodplain inundation Relative distances to groundwater surface
		Water body	Spatial extension of the aquatic area Magnitude of average water depths Magnitude of average flow velocities
		Stream course development	Degree of stream course development
		Stream channel morphology	Richness of bed forms and bed patterns Channel dimension
		Bank\Riparian morphology	Spatial extension of the riparian zone Richness of bank forms and structural quality of the riparian zone
		Floodplain morphology	Spatial extension of the floodplain zone Richness of floodplain features and structural quality of the floodplain zone
		Physico-chemical characteristic	Water temperature Water turbidity Amount of organic matter Amount of harmful inorganic matter
	System dyna-mics	Morphodynamics	Degree of instream morphodynamics Bank erosion and riparian dynamics Degree of floodplain morphodynamics
		Hydrodyna-mics	Discharge dynamics - short term Discharge dynamics - medium term Discharge dynamics - long term Dynamics of flooding Groundwater dynamics
	System connec-tivity	Lateral connectiviy	Connectivity of the river to floodplain water bodies
		Vertical connectivity	Connectivity of the river to groundwater
		Longitudinal continuum	Local longitudinal continuum Regional longitudinal continuum

Figure 1. Hierarchical classification framework actually used within DSS RiverSmart to describe the abiotic part of the ecosystem "riverine landscape".

Each of the 29 parameters can be addressed by a set of five linguistic described value classes ("very low", "low", "medium", "high", "very high"). For a more precise determination of the parameters a list of indicators has been developed, which are used as background information for the experts and for the calibration and validation of the model.

The abiotic condition of each river section under investigation is evaluated by adapting or affecting this set of fuzzy parameters. The response of the biotic system is not directly linked to the abiotic parameters, but is addressed in a second step inherently.

3 METHOD OF ECOLOGICAL EVALUATION

The evaluation method for measures is strictly impact oriented and based on expert knowledge. The whole method of evaluation and the procedure necessary to run DSS RiverSmart can be described in 6 steps:

Step 1: Select river sections

In a first step the river sections for separate investigations have to be determined, e.g. a minimum flow affected section will be a separate section, the impoundment upstream of a dam will be another section and so on.

Step 2: Define reference situation

The reference situation is derived from historic data or regional available river typology information. It describes the riverine landscape without basic and irreversible changes of the habitat conditions and the biocoenosis (Muhar et al. 2000). Based on this, the former river section is "reconstructed" by adapting the values of each of the 29 valuation parameters to one of five defined value-classes ("very high" to "very low"). By this, each river section under investigation gets its own regional reference.

Step 3.: Transformation curves

Impacts of potential measures are expressed in form of transformation curves (Figure 2). In these curves a referenced effect of a specific type of impact (e.g. "water withdrawal") with a specific intensity is transfered to a degree of achievement (DoA). The DoA shows how far the value of the parameter corresponds to its occurrence in the reference state. Its extremes are 0 % DoA (no achievement at all) and 100 % DoA (equal to the parameter value in the reference situation, which means there is no impact on this parameter at all).

Transformation curves are developed mainly by expert knowledge. Calibration and validation with the help of numerical hydraulic and/or morphological models is not mandatory, but helpful to improve the quality of those functions (see Mader et al., 2005).

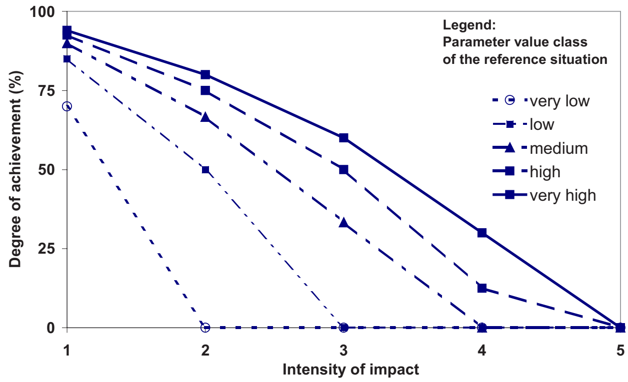

Figure 2. Example for a set of transformation curves for the impact type “water withdrawel“ and the abiotic parameter “discharge“ which can occur in the reference situation in 5 value-classes (very low, low, medium, high, very high).

In this case the impact type "water withdrawal" with an "intensity" of 3 (e.g. water withdrawal with acceptable minimum flow) will result in a river with a "high" natural discharge in a "degree of achievement" of 50 %.

For each impact type, valuation parameter, and parameter value-class a transformation curve has to be defined by experts. This results in a knowledge base, which is the core of the tool. Documentation of the matter behind these curves is very important to improve the transparency of the results and to improve the transferability of the curves to other situations or projects. So far, DSS RiverSmart offers a set of transformation curves developed in the first model applications in two Austrian rivers. These curves are now used for a sensitivity study and model validation.

Step 4: Aggregation
A scenario of several measures within one river section is actually treated as follows:

- determine DoA for each impact type and parameter separately on the lowest hierarchical level (abiotic parameters) and select most significant impact for each abiotic parameter (minDoA's).
- aggregate those minDoA's on the component level for each component member.
- aggregate component DoA on the system entities level
- calculate total DoA out of the system entities.

Currently, a minimum rule and the arithmetic average is used for parameter aggregation.

Cumulation and compensation effects which certainly occur are neglected so far. In the models end version these aggregation steps will therefore be realized by a composite programming technique (Bardossy et al. 1984).

Step 5: Derive ecological status
From the total DoA the predicted ecological status of the river section ("predicted status") is derived, separately for the hydro-morphological and the different biotic states. The predicted biotic status is calculated based on the abiotic conditions and specific "biotic sensitivities". These biotic sensitivities depend on the biotic group under investigation (e.g. "fish ecological sensitivity") as well as the biotic reference condition (e.g. historic fish species composition etc.). In accordance with the definition in the WFD the predicted status of the river is expressed in a numeric schema, which ranges from 1.0 (high status) to 5.0 (bad status). In the end we get results for the predicted hydromorphological status as well as different specific biotic states (fish, macroinvertebrates, macrophytes, algae, floodplain vegetation) of the river.

Step 6: Results
Evaluation results are presented in form of tables which point out the underlying system hierarchy. Thus it's easy to identify those parameters which respond very sensitively to existing impacts. At the same time it is obvious which impact has a special negative or positive effect and which restoration measures should be prioritized. Positive and negative ecological effects resulting from a set of measures can be compared with the status-quo as well as the reference situation.

4 COST-EFFECTIVENESS-ANALYSIS

In addition to the ecological evaluation, the approx. costs and, in case of hydropower, the potential consequences for renewable energy production, are calculated for the set of measures under investigation. In a cost-benefit/cost-effectiveness-analysis different scenarios can be compared in their economic, energetic and ecological consequences.

Based on this, recommendations are made towards a holistic and economically realistic river basin management plan. End-users can use the results to decide which scenario should be analyzed in more detail in the phase of detailed project planning.

5 CONCLUSION

DSS RiverSmart is conceptualized for the evaluation of engineering measures (waterpower, river regulation, river restoration etc.) in river sections and segments of variable length. The tool can be used to compare different scenarios by weighting costs of a set of measures against ecological benefits and, in case of hydropower, energy outputs. Results can serve as a basis for river managers and hydropower companies, e.g. to decide which planning scenario is worth being studied in detail. Hence, DSS RiverSmart can be used for the elaboration of RBMP's, the key element of the WFD.

Up until now, the methodology was applied to two Austrian rivers. Simultaneously, more research is carried out to explore the models sensitivity and to improve its ecological evaluation method (composite programming). The biotic approach of using "biotic sensitivities" has only been tested for fish and is not validated yet.

Future case studies on other Austrian rivers, as well as the river Neckar, Germany, and the Durance, France, will be used to further improve the methodology developed in this project.

6 ACKNOWLEDGEMENT

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MesoCASiMiR – new mapping method and comparison with other current approaches

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ABSTRACT: Within the field of ecohydraulics, habitat modelling is a useful tool for designing and assessing river restoration programs, fishery enhancement, and environmental flow assessment. At present, most fluvial fish habitat modelling occurs on a micro scale, with a maximum extent of several hundred metres. The cost of such methods becomes unfeasible at the management scale, which may require considering several kilometres of river. Research is being conducted internationally on effective methods for upscaling present modelling techniques to an intermediate "meso" scale. MesoCASiMiR provides a new mapping method with the aims to use the established biological input data from the micro scale model CASiMiR, which are based on fuzzy logic, and to simulate the suitability, the fragmentation and the connectivity of meso habitats. Case studies were performed in which the MesoCASiMiR method was compared with other current mesohabitat mapping approaches based on ease of application, time required, amount of detail provided, output, and subjectivity. The results indicated that all methods had strong and weak areas based on the mentioned criteria, depending upon their original design purpose.

1 INTRODUCTION

In order to proceed with river remediation schemes, habitat simulation models are advantageous for the comparison of different proposals and to ensure that a proposed renaturation project truly is better for the ecological situation of the river. They are also beneficial for assessing the environmental impact of newly planned measures to ensure that the river ecology is not compromised by the new plan. In this way, simulation models can also be used as a tool within an Environmental Impact Assessment and an Integrated River Management.

1.1 Macro- vs. Microhabitat

Part of the problem facing decision-makers is the matter of differing spatial scales. For management applications, the river must be viewed at a scale of catchments or river sectors. However, habitat modelling has tended to occur at the level of actual living space for a fish. Where a planner may need to look at a stretch of river many kilometres long, modellers may be looking at sections of river which are only several hundred metres long. The cost in time and finances is prohibitively high to analyze a management-sized sector using these methods.

1.2 Upscaling

This problem of differing scales has led to a pressing present-day research concern: that of upscaling. The goal of this area of research is to scale up from already established microhabitat river assessment methods to the intermediate mesoscale. The goal of mesohabitat models is to provide data at a scale that is meaningful to planners, yet is still valid for fish habitat modelling. At the mesoscale, a river consists of a series of smaller habitat reaches and types. These individual reaches are what microscale models examine. They may be a pool or a set of rapids, for example. The challenge is in developing a modelling method that is detailed enough to be valuable and accurate, and provides consistent and reliable results, yet is reasonable enough in time and labour costs that it may be applied to very long stretches of river.

1.3 Objectives

The purpose of this study (Young, 2004) was to compare the habitat mapping portions of four mesohabitat approaches under development at different universities. The comparison is based on the method's: ease of application (whether special equipment or extensive experience and expert knowledge is required), time needed, and subjectivity.

Two case studies were performed: one on the River Eyach in the German Black Forest, and the second on the Upper Danube. A stretch of river which was unwadeable was included to compare how the methods responded when measurements were not possible and estimations were necessary. The four methods were:

- Meso-Scale Habitat Classification Method Norway (MSC), developed by Norwegian University of Science and Technology and Sintef Energy Research Ltd, Norway (Borsányi et al, 2003)
- MesoHABSIM, under development at the University of Massachusetts (Parasiewicz, 2003)
- Rapid Habitat Mapping (RHM), developed at University College Worcester (Maddock et al, 2001)
- MesoCASiMiR, under development at Universität Stuttgart

Within this paper it is not possible to describe all four methods. Therefore only the new approach of MesoCASiMiR is briefly characterized in the following chapter and for the other methods it is referenced to the mentioned publications.

2 MESOCASIMIR

The general idea in the development of MesoCASiMiR is to use the established biological input data from the micro scale model CASiMiR, which are based on fuzzy logic. In opposition to the micro scale in MesoCASiMiR the hydromorphological parameters are not calculated by hydraulic modelling but mapped by visualisation and single measurements, where a clearly classification is not possible. Figure 1 shows an overview of the key portions of the data sheet.

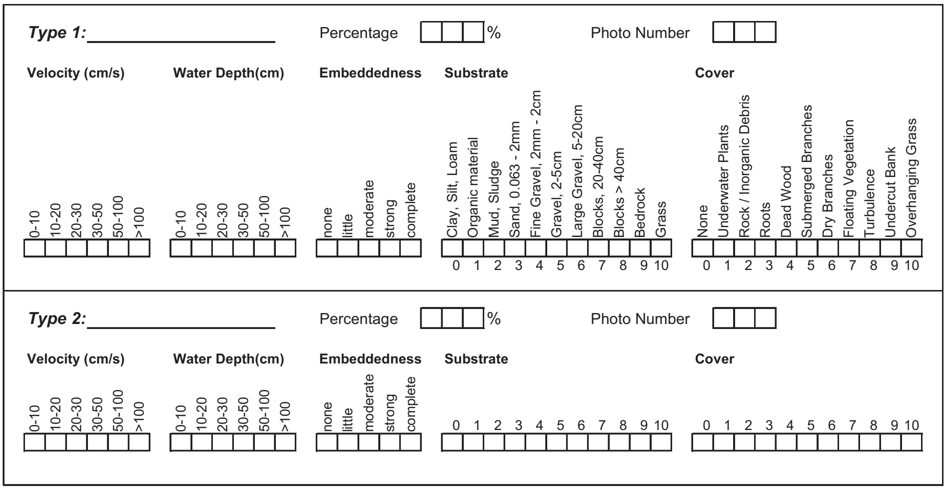

Figure 1. Key portions of MesoCASiMiR data sheet.

MesoCASiMiR data requirements are primarily composed of visual observations supplemented by physical measurements to "calibrate" the estimations of the surveyor. River reaches are not separated by a set distance, but rather by observed river variation and are determined by the mapper. Extraction weirs and confluences represent significant flow changes and always mark a reach boundary. Other reaches may be delimited to account for heterogeneity, and allow for new classifications to account for the new habitat types. GPS measurements are recorded at reach boundaries and the width of the river is estimated. Once the mapper has decided upon the reach boundaries, the preliminary data from the datasheet is recorded, including, for example, the height and GPS co-ordinates of any weirs or man-made obstructions. The habitat types within the selected reach are then identified. Habitat types are areas with distinctly different characteristics than other parts of the channel. For example, one type may be a deeper main channel, with a second type as the slower margins, and a third type a shallow gravel bar. A maximum of 5 habitat types may be selected for each reach. Each type is classified in terms of representative velocity, depth, embeddedness, substrate, and cover values. Photographs are taken of each habitat type. The collected data is then used as input for the MesoCASiMiR. Using fuzzy rules, the program calculates habitat suitability indexes from 0 to 1 for each of the inputted habitat types for a selected fish species and lifestage. At the stage of program development during this study, only the parameters velocity, depth, substrate, and cover are used in the suitability index calculation, though studies to include embeddedness, connectivity and habitat fragmentation continue. Studies also continue on visualization techniques. As a first step, the numerical suitability indexes were the only provided output.

3 COMPARISON: TIME

3.1 Overview

Two segments were mapped with each of the four methods and timed. The first was a 2.6 km wadeable section of the river Eyach, the second a 1.2 km unwadeable section of the Upper Danube. The time required for RHM was estimated for the wadeable portion, as it was performed alongside the MesoHABSIM method.

3.2 Wadeable

For the wadeable stretch, a significant difference can be seen between MSC and the others. This is because the parameters can, for the MSC method, usually be estimated from the shore, with only occasional measurements required for confirmation.

MesoHABSIM was significantly slower than the other methods. This was due to the difficulty of getting into the random number positions. It often required some awkward manoeuvring, multiple river entrance and exits, and walking against the flow, all of which required extra time.

The RHM-PHABSIM time was estimated relatively high because of its maximum depth and velocity measurements. Once the mapper is in the river, the measurements themselves do not add a significant time cost. The main time consumer at the observed river reach is safely entering and exiting the river. However, these two values are not truly a part of analyzing the habitats.

MesoCASiMiR also had a measurement component in which entering and exiting the river increased the time. Time was also consumed with MesoCASiMiR by estimations. The mapper had to make decisions about where to end the reach, what percentages of each type, and which values were representative for each type. These decisions took longer than simply measuring a value. A minor factor that prolonged MesoCASiMiR was the quantity of awkward paperwork for each reach.

Figure 2 gives the comparison of time consuming at the wadeable river Eyach.

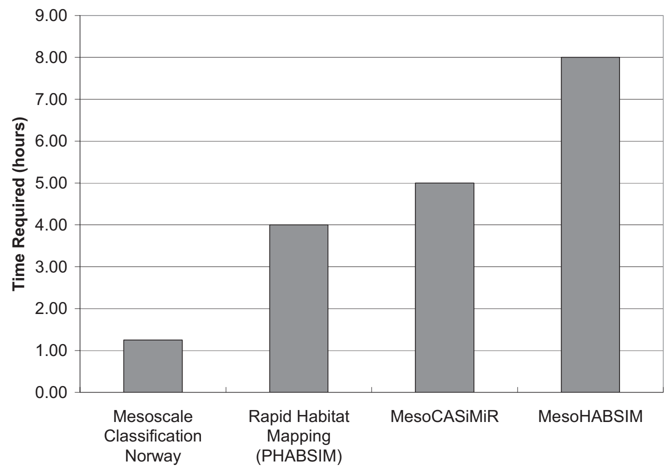

Figure 2. Time Requirements at river Eyach (wadable)

3.3 Unwadeable

With the removal of the possibility of measurements, thereby eliminating the time required for entering and exiting of the river, all methods became faster, with the most significant changes being with RHM (3.7 times faster) and MesoHABSIM (3 times faster), see Figure 3.

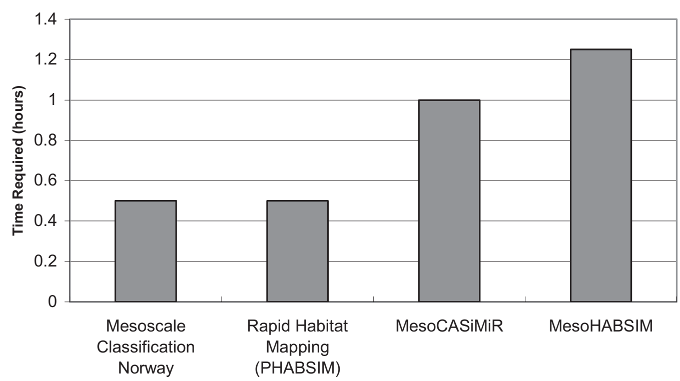

Figure 3. Time Requirements at river Danube (unwadable)

The MSC-Norway method was only slightly faster, as there were not many measurements taken even in the wadeable stretch.

MesoHABSIM continued to be the slowest because it required 7 estimations at random number locations, though the margin between it and MesoCASiMiR was much smaller. The boundaries of the habitat unit had to first be determined before the mappers could walk back to determine how to delineate the imaginary grid.

The speed increase for RHM illustrates how great the entry/exit time requirements were.

MesoCASiMiR also decreased its time requirement significantly, but slightly less dramatically than RHM and MesoHABSIM. It still had a high paperwork requirement and a large number of decisions to be made on the part of the mapper. Comparison: Detail

4 COMPARISON: DETAIL

4.1 Hydraulic and Morphological Parameters

MSC provided the least amount of detail, followed by RHM, then MesoCASiMiR and MesoHABSIM provided the most detail (see Figure 4).

	Velocity	**Depth**	**Substrate**	**Cover**	**Gradient**	**Surface Pattern**	**Embeddedness**
MesoScale Classification Norway	Decision Tree, greater or less than 0.5 m/s	Decision Tree, greater or less than 0.7 m	Considered as a possible second level of detail, not accounted for on the map itself	n/a	Decision tree, greater or less than 0.4%	Decision tree, smooth surface or standing waves greater than 0.05 m	n/a
Rapid Habitat Mapping PHABSIM	Maximum recorded	Maximum recorded	Recorded for the unit in general	n/a	Considered as part of the qualitative definition	Considered as part of the qualitative definition	n/a
MesoCASiMiR	Maximum of 5 types per reach, 6 classes	Maximum of 5 types per reach, 6 classes	1 observation for each type	1 observation for each type	n/a	n/a	1 observation for each type
MesoHABSIM	7 measurements	7 measurements	7 observations, representative chosen for the entire reach	Attributes selected for the unit	Low gradient checkbox	Considered as part of the qualitative definition	n/a

Figure 4. Comparison of Detail of hydraulic and morphological parameters

However, the nature of the detail must also be commented upon. The MSC decision tree has only 4 levels with 2 options each. The flow velocity classes, i.e., are divided in >0.5 m/s and <0.5 m/s.

RHM uses a characterization-list of channel geomorphic units (CGU), where informations about velocity and depth are included in a descriptive way. RHM can also be adapted to be less detailed, by removing the maximum velocity and depth measurements.

MesoHABSIM provides a histogram for velocity, depth, and substrate values along with the qualitative habitat unit classification. This provides an overview of the heterogeneity of the reach. When using the data entry form on the hand-held computer, which the developers have programmed, there is also an option for stating whether cover features are present or abundant. Multiple cover types may be selected. It is possible, and occurred in several reaches, that none of the 7 values fell within the representative type.

MesoCASiMiR is quite detailed in that it allows for up to 5 types with 5 to 11 alternatives per parameter. It is missing the gradient and surface pattern observations of the other methods, but these variables are not necessarily as key for fish habitat modelling as the other parameters.

4.2 Spatial Level of Detail

MSC allows for the smallest types, with up to 3 types laterally requiring only that they be a river width long. This is especially important for larger rivers where there may be more lateral diversity than longitudinal. This restriction was initially intended in order to prevent the map from becoming overly fragmented and, as with the requirement that each type be a minimum of one river width long before being mapped, from becoming overly detailed. It may be, however, that this will need to be adjusted for the method's application on larger rivers.

RHM has been adapted during other applications so as to allow for the added detail of lateral divisions. In this study the habitat units were mapped cross-section equal, in the same way as for the MesoHABSIM method. Thus, information i.e. of shorebanks are getting lost.

MesoCASiMiR uses percentage divisions of habitats within a section, so smaller parts can be considered.

5 COMPARISON: SUBJECTIVITY

5.1 Overview

The issue which all of the methods faced to some extent was subjectivity. Subjectivity was observed by having two observers with approximately equal training mapping the same section of river with each of the methods without consultation with the other mapper.

The first set of subjectivity observations was taken along a mostly wadeable stretch of the river Eyach that included a large weir with its associated backwater. The stretch was relatively homogeneous laterally, but included several different habitat types longitudinally. The second set of subjectivity observations was along a partially wadeable stretch on a branch of the Upper Danube within the re-naturated Blochinger Sandwinkel. There was more lateral heterogeneity than within the Eyach.

5.2 Meso-Scale Classification Noway

The subjectivity test along the Eyach had some surprising results. Considering how straightforward the decision tree was, there was little variability expected. However, there was some variability observed. This is likely because, as measurements from the other methods confirmed, much of the upstream segment was on the border of the decision tree classes of 0.7 m deep and 0.5 m/s velocity.

The second subjectivity observation showed little variation between the two observers. The main points of difference surround the question of surface gradient, with the second observer selecting more areas of high gradient, such as the run and rapid, and the first observer classifying those same areas as having lower gradients, as a glide and deep splash respectively.

Overall, as the Upper Danube results illustrated, this method was found to be reasonably objective and repeatable.

5.3 Rapid Habitat Mapping

The subjectivity results along this section were not surprising. The problem of subjectivity and user variation for qualitative habitat classifications is well documented. The problem may have been reduced by the selection of a better classification system, but the fundamental problems would have remained. Descriptive words such as “moderate“, “deeper“, and “shallow“ are, by their very nature, dependent upon the experience and perceptions of the user as opposed to the objective conditions of the river itself. Every mapper will have a different definition of “deep”. A mapper coming from a mountainous region will have a different concept of gradient and surface turbulence than a mapper whose experience has been confined to lowland streams.

The method could be improved by having a set, yet flexible, quantitative classification system rather than leaving the selection up to the user, and by diagrams of the surface pattern and channel bottom of the different types in addition to the written descriptions.

5.4 MesoHABSIM

The observations of habitat types have the same subjectivity concerns as expressed under RHM.

The substrate values were found to be less subjective than for MesoCASiMiR. This is because the choriotop definitions include the possibility of smaller grain sizes within the larger class. For example, mesolithal (6-20cm) is defined as „fist- to hand-sized cobbles with a mixture of gravel and sand“, thus eliminating the problem of debating whether the dominant substrate is larger cobbles which happens to have smaller gravel surrounding them, or smaller gravel which happens to have larger cobbles embedded within it.

The seven velocity, depth, and substrate observations have two issues regarding subjectivity and repeatability. The first is that, by selecting a random number grid, subjectivity is decreased, as observers are not required to select representative locations but needed simply to take the observations from a given point. Repeatability suffers under the random number system, however.

5.5 MesoCASiMiR

The level of subjectivity in MesoCASiMiR was surprising. It had been hypothesised that this method would be highly repeatable, but the results showed otherwise. Between the two subjectivity reaches, several problems were identified:

Mappers selected different start and end points for reaches. It had been expected that using percentages would minimize the effect of this, as parts of a type which had been cut off by ending the reach at a certain point would be accounted for as a percentage of the next reach, but this was not found to be true in practice. With different sized reaches, the types are assigned different percentages and different qualities are selected as being representative.

Mappers selected different categories in cases where the velocity, depth, or substrate was close to a class boundary.

Mappers selected different locations within the river as representative of the type. When the two mappers observed the same point of the river, the same values for velocity, depth, substrate, and cover would be selected. However, when the mappers were required to select a point as representative of the type, the mappers would select different locations, leading to different results.

Embeddedness was not well defined and requires a clearer procedure. Within the subjectivity study, the two observers rarely selected the same embeddedness values. However, this parameter is not yet attached to the model.

The cover values were unclear. Where small amounts of cover were present, one observer would record the cover while the other would record “no cover“.

6 CONCLUSIONS

The four methods observed in this study all had advantages and disadvantages related to the way in which they developed and the specific purpose for which they were designed. They cover the spectrum of mesohabitat mapping from its quickest to its most detailed.

MSC provides a quick method to gain an overview of the river with high repeatability and no expert knowledge required. The decision tree solves the problems of subjectivity presented by qualitative methods, and the final product, which is immediately available, allows the users to develop a clear image of the hydromorphology of the river section. With a few adaptations, it can be applied to all rivers and provides a good tool for comparison. On the other hand, this method provides little detail and becomes more complicated when additional parameters, such as substrate, are desired. At the moment there is no multi-variate model for the interpretation of the data as it relates to the actual aquatic populations and the users are left to find their own methods of interpreting the data in regards to fish habitat for decision making.

MesoHABSIM is quite detailed and the model itself should provide accurate information. However, the mapping portion provides data for only part of the model, specific biological data being required as well. The proper application of the method requires skill and experience on the part of the mappers, and quite a lot of time. This method is the opposite extreme within mesohabitat mapping as compared to the MSC method, whose key goals were to be quick and applicable by non-experts.

Rapid Habitat Mapping is also a quick way to get an overview of a river segment, and serves the purpose of providing a rough idea of habitat proportions for further microscale modelling. However, this method is rough, and needs the mappers to return for in-depth microscale mapping for the results to be of use. It is less a mesohabitat method, than an upscaling tool. This is what it was designed to be, but in being so it is not comparable to a purely mesoscale methodology. Many of the subjectivity issues that arose regarding habitat definitions have been dealt with since the study was performed and a new set of definitions have been produced. A handbook including illustrations to help with classification would improve the method further.

MesoCASiMiR is detailed and quantitative. It truly is a mesohabitat method, collecting data for use of a model developed for that scale. It deals with real numbers as opposed to general descriptive terms. However, mappers are often unable to distinguish between classes at this scale, making the value of such real numbers questionable.

7 FOLLOWING ADAPTATIONS AS A RESULT

Aside from comparing the different approaches one of the main aims of this study was to improve the MesoCASiMiR mapping method. In the meantime the mentioned problems are minimized due to essential improvements on mapping methodology:

To exclude the subjectivity in estimating the percentage of each habitat division, now the habitats are drawn directly into maps, even on paper prints or on digital maps on a Pocket PC with the aid of GPS.

To minimize the subjectivity in selecting different categories in cases where the velocity, depth, or substrate was close to a class boundary, now MesoCASiMiR uses overlapping class boundaries. This method was proved yet in several ensuing studies.

To include embeddedness – essential for modelling of spawning habitats – an extensive study has been arranged at University of Stuttgart resulting in a detailed field guide, which assures a minimization of subjectivity.

To decrease the subjectivity and difficulties in assessing the substrate and cover conditions, for both parameters a dominant, a subdominant as well as a present value is mapped.

Furthermore the whole visualization of the mapped data and the habitat simulation is done in ARC GIS. MesoCASiMiR can be started from ARC GIS as an extension.

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Habitat assessment in flood risk management and the Environment Agency

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ABSTRACT: The Environment Agency is the competent authority responsible for the implementation of the Water Framework Directive in England and Wales. One of the issues facing the Agency, as both the environmental regulator and in the work that it carries out to manage flood risk, is the assessment of the status of habitats and how this may change. The paper will review some of the emerging tools and techniques being developed for use in assessing instream habitat in England and Wales and some if the issues faced in their implementation.

1 THE ENVIRONMENT AGENCY AND FLOOD RISK MANAGEMENT IN ENGLAND AND WALES

The Environment Agency was established by the 1995 Environment Act and is a non-departmental public body covering England and Wales. The Agency is sponsored largely by the Department for Environment, Food and Rural Affairs (Defra) and the Welsh Assembly Government. It employs over 10,000 people covering a diverse range of activities including flood risk management, pollution control, water resource management, conservation, fisheries management and waste licencing. Our activities range from influencing Government policy and regulating major industries at a national level, through to day-to-day monitoring and managing incidents at a local level.

The Agency has a key role in Flood Risk Management (FRM) in England and Wales, with a budget of £564m allocated to this task in 2005-06. Within FRM, policy, strategic guidance and administration of legislation is the responsibility of the Defra in England and the Welsh Assembly Government in Wales. The Agency is the principle operating authority in England and Wales, with a duty to supervise all matters relating to the delivery of flood defence.

The Agency is also an operating authority with permissive powers to carry out flood risk management works on main rivers and sea defences. There are over 600 other operating authorities generally providing more local services on ordinary watercourses (i.e. those that are not classified as main-rivers) and coastal protection.

In addition to the above, our role in managing flood risk may develop in the future as a result of the recent *Making space for water* consultation carried out in the autumn of 2004 by Defra. The consultation was designed to help develop a new strategy for flood risk management in England and Wales and the first Government response (Defra, 2005) indicates that in the future our approach to FRM may develop to include:

A more holistic approach that includes environmental, social and economic benefits
A 'whole catchment' and 'whole shoreline' approach
Working towards giving overarching, strategic responsibility for flood and coastal erosion risk management to the Agency
Working towards integrated management of urban drainage
More climate change research
Improved planning guidance

Rural land use solutions, such as wetlands and washlands
The increased use of managed realignment of coasts and rivers
The addition of coastal erosion on our flood risk maps and looking into including other sources of flood risk (e.g. groundwater, urban drainage).

2 FLOOD RISK MANAGEMENT AND THE WATER FRAMEWORK DIRECTIVE

The Environment Agency is the competent authority responsible for the implementation of the Water Framework Directive in England and Wales. As set out in Section 1, it may also carry out flood risk management works and has a supervisory duty for other operating authorities. For example engineering works may:

Damage habitats as a result of construction activities (eg. through the release of fine sediments)
Lead to the loss of habitat variability as a result of channel straightening and regrading
Result in the loss of flood plain habitats through the construction of flood walls/embankments
Result in the loss of coastal and estuarine habitats through the construction of sea walls/embankments and "coastal squeeze" (Leggett et al. 2004).

This presents major challenges in ensuring that the Agency, and others, deliver effective FRM whilst also meeting the requirements of the Water Framework Directive by improving the future quality and status of aquatic habitats.

The potential developments in the approach to FRM in England and Wales as identified in Section 1.l will greatly assist us in meeting this challenge. However, we also need to ensure that we have the knowledge and tools required to:

Identify and monitor the quality of aquatic habitats, and
Where justified, enable us to design and carry out flood risk management works sustainably.

To achieve this a number of research studies have been carried out within the Agency's Science Programme and the Joint Defra/Agency Flood and Coastal Erosion Risk Management R&D Programme. Some of these are described in more detail below.

3 SETTING RIVER HABITAT OBJECTIVES

The Agency is developing a series of tools designed to assist in the assessment of river habitat quality and identify and prioritise objectives to improve habitats where required. Setting River Habitat Objectives (RHOs) helps to support management approaches to improve the physical structure of rivers and the main tool used to achieve this objective is the River Habitat Survey (RHS). RHOs aim to:

Characterise river reaches, and the impacts affecting them
Describe the nature and quality of river habitats
Offer a diagnostic tool for identifying river and land management problems
Improve management decisions to protect and enhance river habtiats
Establish a framework for identifying and prioritising river habitat improvements
Provide a means for detecting change and measuring the impacts of river management.

RHOs are used because it is increasingly recognised that good environmental quality of rivers depends as much on physical structure (habitat) as it does on water quality. In addition, the Water Framework Directive also requires an assessment of hydromorphology as background in classifying ecological quality. As a result, the development and piloting of RHOs is a priority for the Agency and forms a Biodiversity Strategy Target to help promote water and wetland conservation.

The RHO process is broken down into six stages as shown in Figure 1 below:

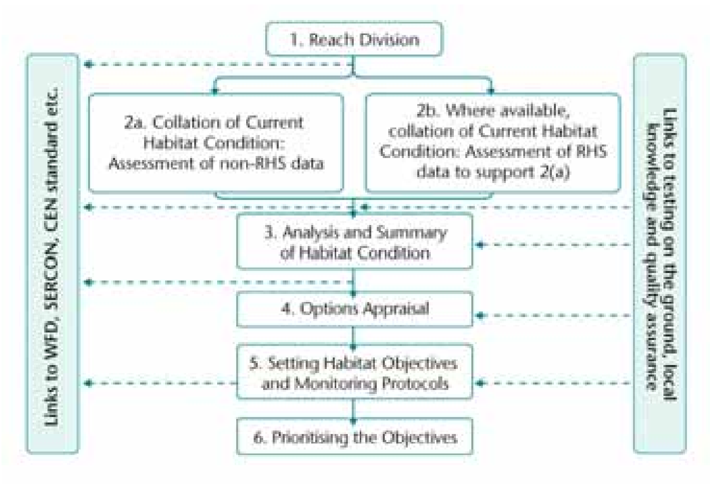

Figure 1. The six stages of the RHO methodology.

3.1 River Habitat Survey

A key tool for setting RHOs is the River Habitat Survey which provides the core site-based morphological data. The main uses of RHS are:

Catchment evaluations
Habitat suitability for species
WFD
HD
Rehabilitation/restoration
Flood defence
EIA

The RHS system includes a database that contains over 15,000 sites from across the UK. For reaches where RHS has been carried out, the data can be used to verify the existing morphological conditions and assumptions made from maps and photographs. The RHS database can also be used to establish what the overall character of the reach may be given certin geopmorphological and land-use scenarios. Slope, distance from source, height of source and site altitude are used to cluster similar RHS sample sites in a so-called "context analysis". Nearest neighbour site clusters can be used to generate the likely habitat character of a reach based on the representative group of sites at a regional or national scale.

The RHS process is used to set provisional General Habitat Quality (GHQ) classes for the site in question, which can then be used in identifying options for future site management as set out in Figure 2.

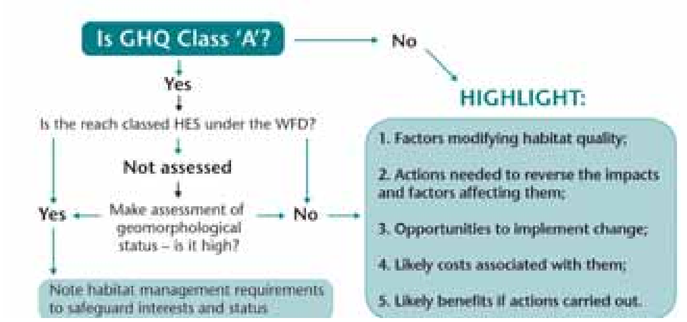

Figure 2. Options appraisal for River Habitat Objectives

3.2 Geomorphological components to RHS

The Defra/Environment Agency Joint Flood and Coastal Erosion Risk Management R&D Programme has been working to develop geomorphological and floodplain components to the River Habitat Survey technique. This incorporates data collected on river and floodplain morphology in the field, and uses desk-based information collated in a GIS. The objective is to provide a strong tool to study river-floodplain interactions and will be of particular use in FRM. It enables the characterisation and comparison of floodplain processes and habitats and the sensitivity of channel and floodplain to change. It also provides input to strategic floodplain hydraulic modelling and the design and evaluation of river management schemes, both key factors in the design of sustainable FRM works.

3.3 Linking geomorphology to ecology

We are currently working to improve our ability to directly link habitat assessment techniques with ecology to better identify the status of habitats. One emerging approach is designed to allow the comparison of observed vs expected 0+ salmon populations. This is driven by a relatively good understanding of the relationship between young salmon and their habitat. The technique uses Bayesian statistics to make the best use of existing information on physical habitat. The physical information required by the assessment is obtained from map-based sources (eg DTM) and existing field surveys eg River Habitat Survey. Missing data is emulated using Bayesian stats. This allows the development of a prediction of 0+ (young of the year) salmonid density that can be compared with actual from field surveys. Maps of uncertainty associated with the analysis are produced.

3.4 Sustainable management of river conveyance

In addition to the emerging tools that allow the improved assessment of river habitats in FRM, and their improvement, it is also important to ensure that new and ongoing river management activities are carried out as sustainably as possible. A new tool (the "Conveyance Estimation System" (CES)) is designed to facilitate the design of sustainable FRM works in rivers and also the improved management of aquatic vegetation. The CES system provides an easy and logical approach to assessing/allocating roughness across channel and floodplain, and calculation of water level. It also indicates uncertainty bands and uses UK-relevant roughness data, including vegetation types and River Habitat Survey data. The system is supported by a Conveyance Manual and includes software which is being made available in commercial user packages:

in 1-D modelling software (iSIS – available now, Infoworks – coming)

as "CES Standalone" specifically for channel maintenance (forthcoming).

4 CONCLUSIONS

Balancing Flood Risk Management activities and the need to maintain and improve habitats to meet the requirements of the Water Framework Directive is clear challenge faced by the Environment Agency, Defra and other FRM operating authorities. To meet this challenge a range of useful tools are being developed that help to identify impacts to the wet environment and to enable the better design of FRM works to minimise risk. These are being taken forward, through further research and development of tools and through pilot studies to test and demonstrate their effectiveness. In achieving this we are adopting a collaborative approach to research and the implementation of tools and techniques and hope to develop this, both in the UK and internationally, in the future.

Assessing Luxembourg river health from macroinvertebrate communities: methodological approach and application

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ABSTRACT: River health can be assessed thanks to biological monitoring, in particular of the benthic macroinvertebrate communities. In Luxembourg, a river typology was first determined on the basis of selected abiotic variables. Then, specific macroinvertebrate assemblages were identified for each type. For this purpose, a set of sites were chosen according to habitat quality in order to fit a predictive model. The latter, first determines relative probabilities for a test site to belong to a given type on the basis of environmental variables. In a next step, probabilities of occurrence of taxa are set as the weighted average probability among each type. Finally, an index of water quality is derived from the deviation of the observed fauna from the expected one. This model is implemented on the basis of an environmental typology. Thus, the present approach is closely dependent on the adequacy between the environmental typology and the macroinvertabrate communities. Results are consistent when compared with another independent biological index (I.B.G.N.). Although the present approach for stream bioassessment works on individual stream reaches, predictions are provided from meso-habitat to macro-habitat scale environmental variables. This can be explained mostly by the fact that the sampling effort encompasses comparable microhabitats at each sampling site, decreasing their potential influence upon the model. This qualitative predictive model is thus an adequate tool to assess river health even without taking into account abundances of macroinvertebrates.

1 INTRODUCTION

The European Union Water Framework Directive (EU WFD 2000) recommends river health assessment on the basis of biological monitoring. The organisms most commonly used in stream water quality monitoring are periphytic algae, macrophytes, fish, and benthic macroinvertebrates (Lenat & Barbour 1994). Benthic macroinvertebrates are likely to play an important role in the Water Framework Directive implementation since most European countries have a tradition in monitoring benthic invertebrates. Furthermore, besides organic pollution, macroinvertebrates are well suited to detect acid stress, habitat loss and overall stream degradation (Hering *et al.* 2004). Like in most European countries, benthic invertebrates are also commonly used in Luxembourg (*e.g.* Dohet *et al.* 2002).

A large variety of methods for sampling and assessing the benthic macroinvertebrates are available. Some of them developed recently in order to fulfill the requirements of the EU Water Framework Directive must be mentioned here. The AQEM approach of assessing the ecological status of running waters is mainly based upon metrics (*e.g.* Hering *et al.* 2003). Artificial neural networks are also applied as an approach for the analysis of ecological data (*e.g.* Park *et al.* 2003, Walley & Fontama 1998). Decision trees also play an important role in predictive modelling and attempts are made in order to enhance their predictive power, *e.g.* with genetic algorithms (D'heygere *et al.* 2003). Finally, besides multimetric approaches that use the values of several indices to measure biotic conditions, many studies use multivariate methods and more particularly prediction systems, that measure the distance of the observed fauna to an expected reference community (*e.g.* Clarke *et al*. 2003, Logan 2001, Moss *et al.* 1987, Wright *et al.* 1993). Because of their similarities, these predictive models are often named as "RIVPACS" models according to the acronym Wright (1995) and coworkers derived from the name of the original assessment scheme (River Invertebrate and Classification System). The model developed in this paper to assess river health in Luxembourg is mostly founded on the latter concept.

Nevertheless our objective was not only to build an effective river bioassessment tool but was also to develop a model that fulfils the requirements of the EU WFD. Thus, a river typology was first determined on the basis of selected abiotic variables (Ferréol *et al.* 2005). This typology then served as the basis for the development of a predictive model of the macroinvertebrate fauna. The different steps of the model development are presented as well as selected results concerning its statistical validation.

2 MODEL CHARACTERISATION

2.1 Environmental typology in Luxembourg

The first step of the elaboration of the model was to define a functional stream typology. Only mesological data were considered so as to preserve the independence between biotic and abiotic data. Thus, further biological predictions are provided from environmental data. Six stream types are defined in Luxembourg (Ferréol *et al.*, 2005). Typological keys are principally global size of the stream, altitude and water mineralization. The latter is well correlated to the geological difference between the calcareous and non-calcareous parts of the country.

2.2 Site selection

Site-specific predictions of the macroinvertebrate fauna to be expected in the absence of environmental stress need a selection of reference sites or of sites close enough to reference conditions for each stream type (Wright *et al.* 1993, Reynoldson *et al.* 1997). A preliminary set of sites with low organic pollution and low land use intensity was selected. Land use intensity is supposed to increase from forests to pastures and agriculture to urban areas. This first selection of least impacted sites was then validated thanks to the SEQ-physique method, which is an easy and fast method for characterizing the physical quality of river beds and banks (Charrier *et al.* 2002, Raven *et al.* 2002). Finally, 62 sites were selected as reference sites in order to build the model. As sites of some stream types do not have a sufficient high quality, they must be considered as sites with the highest ecological potential available for their respective types.

2.3 Prediction of the macroinvertebrate fauna

The different steps of the modeling are displayed in Figure 1. The first step (mentioned as (1) in Fig. 1) is the computation of the relative probabilities for a test site to belong to each stream type on the basis of selected environmental variables.

These variables characterize the meso-habitat and the macro-habitat (for the geological aspect) because it is also the scale of the typology. Moreover, they are selected for their stability in time in order to be independent on seasons and easily sampled in the field. Hence, the assumption of habitat temporal stability is met. The statistical method involved here is a linear discriminant analysis. Environmental data are located in the plane of the analysis and the relative distance from that point to the average position of a different type is transformed into relative probabilities. So, the notion of gradient is introduced because limits between types are neither fixed nor absolute.

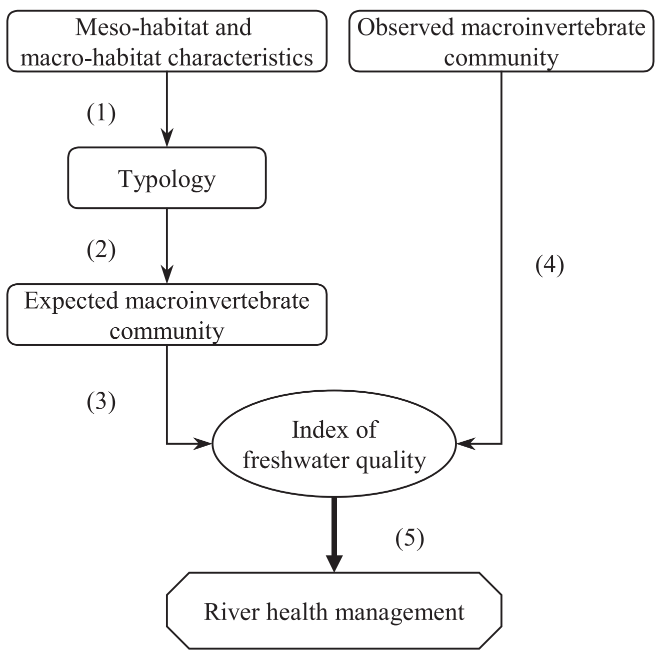

Figure 1. Graphical representation of the predictive model for benthic macroinvertebrate communities and its application

The step (2) deals with the biological modeling. Reference sites set theoretical occurrence probabilities for each taxon among each type. Under the assumption of unstressed conditions, the expected probability P of finding a particular taxon at a test site is estimated from the proportion of reference sites in each environmental type where the given taxon is present, weighted by the test site's probabilities of belonging to each environmental type. Only highly expected taxonomic units are considered (*i.e.* with P > 75%) in order to focus only on taxa expected under reference conditions. Three taxonomic levels are considered (*i.e.* species, genera and families) as well as three seasonal levels (spring, fall and both).

Steps (3) and (4) define respectively the Expected score (E) and the Observed score (O). E is the sum of the respective probabilities of occurrence within the listed taxa (see above). O is the number of listed taxa occurring in the test sites. The O/E ratio is the final index of assessment and represents the proportion of observed taxa among those mostly expected under unstressed conditions in given habitat conditions. Table 1 displays an example of calculation for an impaired test site.

Table 1. O/E index computation example (family taxonomic level, both seasons level)

Taxa	Presence	Probabilities of occurrence (P>0.75)
Baetidae	v	1.000
Simuliidae	v	0.985
Chironomidae	v	0.985
Limnephilidae	v	0.981
Elmidae		0.980
Rhyacophilidae		0.977
Hydraenidae		0.955
Lebertiidae		0.897
Hygrobatidae		0.897
Hydropsychidae		0.892
Gammaridae	v	0.889
Lumbricidae		0.854
Sphaeridae	v	0.850
Ancylidae		0.827
Lumbriculidae	v	0.818
Leptophlebiidae		0.809
Sperchontidae		0.803
Dytiscidae	v	0.798
Lymnaeidae	v	0.796
Ceratopogonidae		0.796
Psychomyiidae		0.776
Tubificidae	v	0.772
O/E index	Number of observed taxa	Sum of probabilities
0.52	10	19.34

2.4 *Quality class boundaries establishment*

The O/E ratio is not informative in itself. Values need to be classified in order to provide a pertinent assessment of the river quality. This last step (5) is important because it determines the decision to start or not a remediation of the river.

Test sites are ranked in five quality classes depending on the value of the O/E index. The limit between the higher (High) class status and the second one (Good) is set as the unit. Thus, this threshold corresponds to a situation where at least, all taxa expected are present at the test site. Therefore, unsressed sites belonging to this higher class status may be characterized by more observed taxa than expected. The limit between Good status and the third one (Average) is very important because it also determines the limit at which a remediation of the site will be necessary. That limit is defined as the 5th percentile of the O/E ratios computed from reference sites. All the other class limits are then set as thirds of the previous limit (Average/Bad limit as 2/3 of the value of the Good/Average limit and Bad/Very bad limit as 1/3 of the value of the Good/Average limit).

3 TESTING OF THE MODEL

Outputs of the model fit very well with what is expected in the field. To assess the performance of the model, the results of the predictions are compared with another independent bioassessment method. Thus, test sites not used to build the model and encompassing all the range of water quality classes are assessed with both the present model and the I.B.G.N. method (AFNOR, 2004). In the example illustrated in figure 2, the identification level for the model is the species level and samplings were performed in autumn. The I.B.G.N. index involves macroinvertebrates to be identified up to the family level. The confrontation of results (Figure 2) displays a linear trend between O/E and I.B.G.N. indices. The Pearson's product-moment correlation is 0.84 (p-value $< 10^{-16}$).

Results seem to be less rigorous for the O/E index because the value 1 corresponds with an I.B.G.N. close to 15 although the High status is defined for values equal or higher than 17. However, the I.B.G.N. index is not stream type specific. On one hand, some types do not have sites that could be evaluated as High and on the other hand, some types have a natural reference fauna that is not considered in the I.B.G.N. index (e.g. streams with a high water quality but with a low productivity)

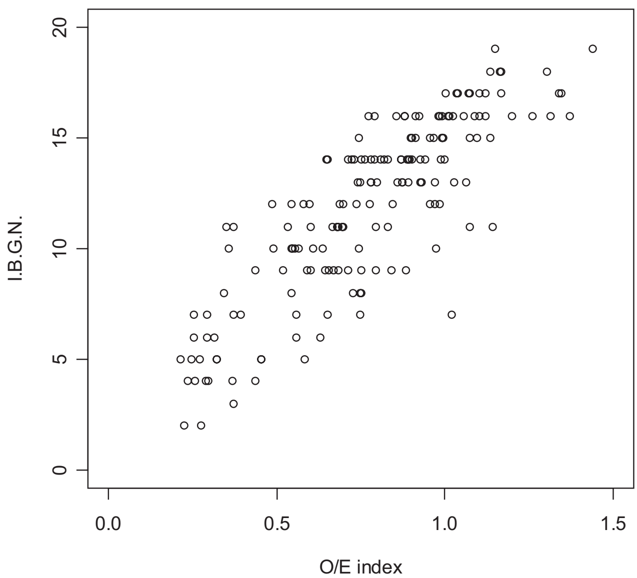

Figure 2. Results of O/E index at species level against I.B.G.N. results for test sites covering all the range of water quality classes.

4 APPLICATION

4.1 Relationship between habitat features and predictions of the model

Adequacy between the environmental typology and the macroinvertabrate communities has been checked. The results of a correspondence analysis (Jongman *et al.* 1995) performed on the macroinvertebrates data in absence/presence for all reference sites plotted whit their respective types (ellipses and stars) are presented in Figure 3. All types are well separated one from another except types I and II.

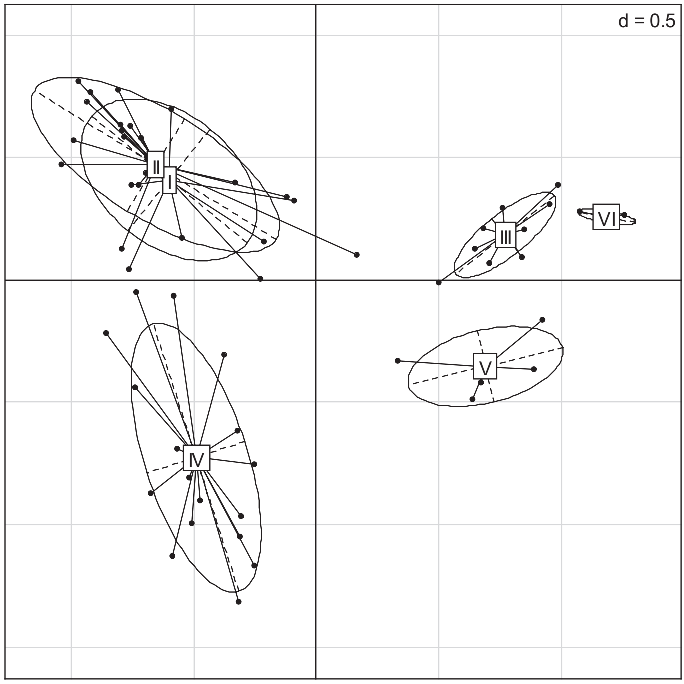

Figure 3. Site scores of a correspondence analysis performed on absence/presence data from reference sites. Sites are grouped according to their respective environmental type (marked from I to VI).

Therefore, the abiotic typology is strongly linked with biotic assemblages. Results are less obvious when using the complete biological dataset because a pollution gradient appears in the ordination and polluted sites aggregate (not displayed here). Concerning types I and II, the first one is characterized by sites closer to the stream sources and in relative higher altitude than the second one. However, dimensions and geology are quite similar. If macroinvertebrate communities, as analyzed by correspondence analysis, are also similar, a bias can be expected for the banding of the index. Anyhow, as the predictions for a test site are weighted by relative typological probabilities and as average positions of those two types are also close in the linear discriminant analysis plane, final results are not strongly affected by that bias. Further studies are then needed in order to determine whether types I and II can be merged in only one type.

This approach to stream bioassessment works on individual stream reaches. Actually, samples collected in different microhabitats at a stream site are assumed to reflect the condition of the whole stream. However, this hypothesis has not been verified and it may be hazardous to generalize predictions of species assemblages from individual stream reaches to the whole stream ecosystem. In particular, the influence of different microhabitat distributions on the predictions of invertebrate fauna generated by the model, should be tested. Furthermore, this model was constructed from data obtained by a relative extensive sampling strategy. The variability associated with smaller sample size on the calculation of O/E ratios, should be assessed if the model has to be used by water managers with standardized sampling protocols.

4.2 *River health management*

As referred by Boulton (1999), an appropriate choice of indicators, rigorous sampling and analysis, and careful data interpretation must be matched with effective communication to policy-makers and the public. The present model allows an assessment of biological condition by comparing the taxa observed at sites of unknown condition with the biota expected to occur in the absence of stress. By predicting the actual taxonomic composition of a site, the model also provides information about the presence or absence of specific taxa. If ecological information (sensitivities to different stressors, biological or ecological traits of taxa) is available, this information can lead to derived indices and diagnoses of the stressors most likely affecting a site (Hawkins *et al.* 2000)

Finally, the main advantage of the present model is the complete independence between biotic variables and the process of stream classification. River typology being determined on the basis of selected environmental variables only, this model potentially enables the use of different biological indicators (e.g. diatoms, macrophytes, fish). Actually, different taxonomic groups respond to different environmental factors and it is very unlikely that site groupings based on macroinvertebrates be used as a surrogate for diatoms, macrophytes or fish assemblage classification (Paavola *et al.* 2003). By using a single classification of sites and multiple taxonomic groups as biological indicators, the assessment by water authorities of the ecological effects of pollution and landscape alteration might be greatly facilitated.

5 CONCLUSION

Although the present approach to stream bioassessment performs well on individual stream reaches, predictions are provided for environmental variables from a meso-habitat to a macro-habitat scale. This can be mostly explained by the fact that the sampling effort encompasses comparable microhabitats at each sampling site, thus decreasing their potential influence upon the model. The expected macroinvertebrate fauna is computed from environmental variables and from a typology implemented without biological data. Therefore, the results show the importance of habitat characteristics for the prediction of benthic macroinvertebrate community composition. With an appropriate sampling effort, information is relevant at a meso-habitat scale.
Although the RIVPACS methods are based on a biotic classification instead of the environmental typology used by the present model, this important difference does not affect the relevance of the results.

ACKNOWLEDGEMENTS

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Critical Approach to Reference Conditions Current Evaluation Methods in Rivers and an Alternative Proposal

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ABSTRACT: Most currently used methods in Spain for evaluating and simulating reference conditions are reviewed. Present reductionism tendency of abusing biotic indexes in the implementation of the Water Framework Directive is discussed, because of loosing basic biological information and precision.

Ideas for an alternative proposal are presented. This proposal should integrate the understanding of the physical and geomorphic processes that sustain biological communities. The importance of the finest level of taxonomic resolution of the data used in the models that simulate and evaluate reference conditions is remarked in order to manage most natural stream reaches properly.

INTRODUCTION

The WFD requires a strong shift in the traditional River Management, from a vision of the rivers as a source of hydric renewable resources with different uses, according to its physico-chemical characteristics, to an integrated view of the water body as an ecosystem, where the ecological status has to be evaluated, according to its "naturality".

This change of management concepts has to be transferred to the way of assessing and monitoring water resources or water bodies. The idea of water quality as a main objective to be maintained in rivers has been replaced by the issue of improving their ecological status, in terms of biology, hydromorphology and physico-chemical characteristics.

In Spain, after some decades of river water quality monitoring based on physico-chemical conditions only, the bio-assessment methodologies were first applied by González del Tánago *et al*. (1979) and García de Jalón & González del Tánago (1986). New indexes adapted to the Iberian conditions were later on defined (Alba-Tercedor & Sáchez-Ortega, 1978; Munné *et al*., 1998), and more recently proposed for evaluating reference conditions and ecological status of rivers (Alba-Tercedor *et al*., 2002; Munné *et al*., 2003).

This paper tries to reinforce the strength and limitations of these indexes for establishing reference conditions, advocating for a more taxonomically precise ways of defining the biological communities, and promoting the use of "similarity indexes" to evaluate the ecological status or rivers, in terms of the "distance" from the species composition and structure of the original communities.

The interest of defining reference conditions in geomorphological and hydrological terms and for riparian conditions is also discussed, having a large scientific information for biological and physico-chemical ecological status criteria and variables, but a significant much less documentation and experience on hydromorphological quantitative characteristics and evaluation.

BIOLOGICAL INDICATORS BASED INDEXES: LIMITATIONS OF THEIR USE FOR THE WFDA

Although the concept of water quality is evaluated in relative terms due to the use which is dedicated, it was generally admitted that water that potentially could be used in greater number of uses had a greater 'water quality'. In this sense, high mountain streams with crystalline clear waters have absolute greater water quality than the naturally turbid, mineralized and enriched waters of lowland rivers.

The selection of indicator organisms of water quality has been based on these principles, having been markedly biased towards species that lived mainly in mountain streams (Rhithron), where the water is normally clear and less mineralized, against those living in lowland rivers (Potamon), where the water is more turbid and mineralized due to natural processes. Rhithron species have been considered "intolerant species", with a higher weighting ratio in the indexes, whereas Potamon species are generally considered as 'tolerant' species, having lower weighting ratios.

The biological-indicator concept included in the biotic indexes has been very useful for summarizing general conditions of running waters, and easily understood by civil engineers and technical staff in charge of river management without any ecological or biological knowledge. These indices oversimplify the survey results, as they are obtained by adding the 'weight factor' associated to each species of the list. The weight factor of the species was given on subjective assessments for the species tolerance limits or sensitivity to organic pollution.

Also, we must be conscious that under different pollution types or different pressures the same species to may be tolerant to ones and sensible to others. Most Plecoptera species are well known to be sensible to organic pollution, whereas they are tolerant to heavy metals and acidification.

However, for the implementation of the WFD it is required to go further, and evaluate not only the integrated value of the biotic index reflecting water quality, but to assess if all the main species which should be in the river are in fact there, independently of their weight factor, and to estimate how much the present community differs from the original or defined as a reference.

In this situation, a more precise taxonomic identification of river fauna and flora than that required by the biotic indexes (genus or family level) is necessary, and a comparative study of the present situation in relation to the reference condition seems to be demanded, without any previous consideration of the species as indicators of water quality, neither of the scoring system of prescribed values of good and bad conditions, like those reflected in the traditional biological indicator based indexes.

IMPORTANCE OF SPECIES IDENTIFICATION FOR ASSESSING THE ECOLOGICAL STATUS

The importance of species identification was pointed out some time ago by Resh & Unzicker (1975) and García de Jalón *et al.* (1981), preventing from the mistakes and lack of accuracy in the use of water bio-indicators without the adequate precision of taxonomic determination.

These ideas are nowadays relevant in the context of the WFDA defining reference conditions for biological communities, taking into account that the sensibility of different organisms to different human pressures may greatly vary from one species to another, even among the species in the same genus, and with greater emphasis among species belonging to the same family.

This is especially true for those genus or families with a broad ecological spectrum that include many species, which often are community dominants, like *Baetis* and *Hydropsyche*. As an example *H. exocellata* and *B. rhodani* are well known tolerant species to eutrophication and organic pollution, while on the contrary *H. tibialis* and *B muticus* are sensible species to that pollution.

INTEREST OF SIMILARITY INDEXES IN THE CONTEXT OF THE WFDA

Similarity indexes were proposed by Hellawell (1986) to be used on stream bio-monitoring, especially for aquatic organisms, and more recently Winward (2000) has suggested their use for monitoring vegetation resources in riparian areas. Similarity indexes can be very useful for quantitative comparison of present vs. reference conditions, by means of identifying key species and comparing their abundance and space and time distribution in present conditions with those considered as “natural”.

Similarity indexes that mathematically fluctuate between zero and one are the most interesting for the WFDA. They can use qualitative data (presence/absence of species like those used by Jaccard, Sorensen, etc.), or quantitative data (abundance of species as proposed by Raabe, Czenowski, etc.).

Also, these similarity indexes may be used directly as the EQR ‘ecological quality ratio’ for each metric, as the index value for the reference conditions is always one (identity), and the value of the index would be directly the EQR.

Furthermore, the issue of determining thresholds between “very good” and “good” ecological status may be undertaken in a simple manner, when we have different reference sites for a river type. In this case, the minimum value of similarity between two communities from these reference sites can be considered as a threshold between these ecological status.

GEOMORPHOLOGICAL AND HYDROLOGICAL INFORMATION OF REFERENCE CONDITIONS

Much of the effort that has been done until now in the implementation of the WFD has corresponded to biologists and ecologists working in different flora and fauna groups, but very few contributions have appeared from geomorphologists or hydrologists trying to define more precisely the hydromorphological conditions mentioned in the WFD.

Many scientific works have been carried out in the context of the WFD trying to verify river classifications by their biological meaning, but in many cases, biological conditions are deeply altered by water pollution or flow regulation, and the geomporphological attributes like those proposed by Rosgen (1996) or Montgomery & Buffington (1997) can be the only natural criteria for river characterization and classification, being very useful in defining the reference conditions of the fluvial habitat.

Within each geomorphological group, several hydrological flow regime types can exist, and their reference conditions and ecological status can be defined following the hydrological alteration indicators proposed by Richter *et al*. (1997). Magnitude and duration of average flows, magnitude, duration, frequency and timing of extreme flows, predictibility and flow change ratio are selected by these authors as hydrological parameters with significant biological meaning, which can be relatively easy evaluated in most of the cases and compared with natural flow regimes.

Finally, the riparian conditions should be evaluated not only in terms of the composition and structure of present vegetation, but also including temporal dynamic variable of this vegetation (ex. woody species regeneration) and hydrological functionality of the fluvial space, in terms of dimensions, connectivity with the active channel (access for flooding), permeability, etc.

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Validation of ecological models for the European Water Framework Directive

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ABSTRACT: Water system models are since several decennia proposed as promising tools to improve river management. Numerous modelling and data mining techniques have been developed, but the particular strengths and weaknesses of these techniques remain unclear. This is partly because there is lack of sound methodologies and criteria (indicators) to assess the models' qualities for practical use in decision support.

This study compares the evaluation of predictive models of macroinvertebrates on the basis of numerical performance indicators, sensitivity analyses and practical simulation exercises. The models are developed for the Zwalm river basin, on the basis of artificial neural networks. Four macroinvertebrate taxa are used as key species to illustrate both model development and evaluation within the context of the implementation of the European Water Framework Directive.

Life in the ice lane: A review of the ecology of salmonids during winter

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ABSTRACT: The main objective of this presentation is describe the results of a review article that we have been writing. In this review, we have summarized the latest information about the survival, habitat use, movement and biotic interactions of salmonids as it relates to the prevailing physical conditions in rivers and streams during the winter. Such information should be of use to both ecologists and resource managers who have interests in identifying where bottlenecks in fish production lie and in effective management of boreal streams. We mostly focus on behavioral and ecological aspects of overwintering fish, but because of the close linkage between physical habitat and fish ecology, both physical and biological elements are discussed.

1 INTRODUCTION

Winter conditions in boreal streams are generally associated with low water temperatures, various ice phenomena, low discharge rates as well as decreased sunlight and heat radiation (Prowse & Gridley 1993). Ice processes undoubtedly have a major influence on the ecology of animals living in boreal streams, through their effect on the timing, duration, and magnitude of flow and water levels. Despite the general belief that conditions in winter strongly influence survival and population size of fish, the ecology of fish has not been as extensively studied in winter as in other seasons (Hubbs & Trautman 1935, Cunjak 1996). This is presumably a consequence of the difficulty associated with sampling in winter. Much previous work conducted in winter has been carried out at water temperatures above freezing without ice cover. Thus we know little about the behaviour of fish under ice (Robertson et al. 2003, Roussel et al. 2004), and little experimental work has been conducted on the impact of different ice conditions on fish (but see Finstad et al. 2004a). However, technological developments are improving our ability to study fish in icy conditions (e.g. Greenberg & Giller 2000, Alfredsen & Tesaker 2002, Robertson et al. 2004), and as a consequence, new insights will likely follow (e.g. Roussel et al. 2004,

Johnston et al. 2004, 2005). Information on how ice affects the ecology of fish should have consequences for how we manage fishes in boreal rivers.
The main objective of the this article is to summarize the latest information about survival, habitat use, movement and biotic interactions in salmonids as it relates to the prevailing physical conditions in rivers and streams during the winter. Here we consider winter as a period with ice formation and low water temperature, reaching freezing or near-freezing water temperatures by mid-winter.

2 PHYSICAL CONDITIONS OF THE LOTIC ENVIRONMENT

Physical river habitat conditions depend on various variables, the most important of which are related to stream flow distribution, morphology, cover, temperature and water quality (Cunjak 1996, Heggenes et al. 1993, Tesaker 1998, Alfredsen & Tesaker 2002). During winter, different types of ice form in sub-arctic streams (e.g. anchor ice, frazil ice, surface ice), which in turn affect these physical habitat conditions. As proposed by Prowse and Gridley (1993) and Cunjak et al. (1998), winter time can be divided into three main periods: early winter (freeze-up), mid winter (stable conditions) and late winter (ice break-up). In addition, ice regimes can be characterized based on river type (Table 1). Ice formation in small, steep rivers is dynamic, in some cases changing throughout the entire river, whereas ice conditions in larger rivers are generally more stable. Anthropogenic impacts may also alter the ice regime. Hydropower production, particularly in high-head systems, can result in altered discharge regimes and the release of warm water into rivers. In regulated rivers, the ice regime is characterized by repeated ice break-ups and increased ice production. In some cases rivers have been dredged to prevent ice jamming, and this will alter both substrate composition and flow conditions in the reach.

Table 1. Ice processes categorized by seasonal and river characteristics.

Ice regimes	Small, steep rivers	Large rivers	Regulated rivers
Freeze up	Border and skim ice formation	Border ice	Border ice
			Dynamic ice formation
	Dynamic Freeze-up	Ice cover formation	Reduced ice growth
Main winter	Extended dynamic freeze-up	Stable ice cover	Repeated ice break-ups
	Anchor ice dams	Open riffles	Local ice runs
	Local ice runs		Increased dynamic ice formation
Ice break up	Thermal ice break-up	Thermal ice break-up	Repeated mechanical ice break-ups

3 FISH SURVIVAL

Winter is often recognized as a bottleneck for survival for lotic fish populations. The data-at-hand show that survival rates of fish and their eggs during winter vary considerably, not only between rivers and over the winter season, but also between years. For example, in studies spanning up to 17 consecutive years, the annual variation in survival rates has been shown to be substantial for brown trout *Salmo trutta* (15-84%), brook trout *Salvelinus fontinalis* (35-73%), Atlantic salmon *Salmo salar* (43-75%) and coho salmon, *Oncorhynchus kisutch* (16-84%) (Needham et al. 1945, Hunt 1969, Holtby 1988¸ Cunjak & Randall 1993). Surprisingly, few studies have actually compared survival rates in winter with other seasons, which is necessary if one is going to make any general conclusions about winter functioning as a bottleneck for survival. Recent studies indicate that there may not be any general seasonal bottleneck. Instead, survival rates are low in different seasons in different studies, sometimes in connection with episodic events such as floods and droughts (Elliott 1993, Smith & Griffith 1994, Elliott et al. 1997, Cunjak & Therrien 1998, Olsen & Voellestad 2001, Letcher et al. 2002, Lund et al. 2003, Carlson & Letcher 2003). Survival has been shown to be lowest in spring (Elliott 1993), in autumn and early summer (Carlson & Letcher 2003), in winter (Letcher et al. 2002) or to not differ appreciably between seasons (Olsen & Vollestad 2001, Lund et al. 2003). These results indicate that there may be a complexity of environmental and biological factors affecting the survival of fish. In some rivers the set of prevailing conditions in winter, such as severity and duration of the winter, together with quality and suitability of habitats, may act as the bottleneck to survival, whereas in other rivers prevailing conditions during other seasons may be more limiting.

Numerous studies have reported positive relations between size and winter survival, with individuals under some minimum size in autumn being most prone to mortality (e.g. Hunt 1969, Quinn & Peterson 1996, Meyer & Griffith 1996, 1997, Johnston et al. 2005). However, some studies have reported the opposite, i.e. a negative relation between size and winter survival (e.g. Needham et al. 1945, Carlson & Letcher 2003) or no relationship whatsoever (e.g. Lund et al. 2003, Johnston et al. 2005). A recent study by Finstad et al. (2004b) presented evidence that YOY mortality may be linked to levels of energy stores at the onset of winter rather than body size per se. In summary, the main causes of mortality in winter are believed to be (1) depletion of energy reserves in combination with harsh physical conditions, (2) predation and (3) accidents. Thus, over-winter survival should be relatively high if fish have high energy stores in autumn, dwell in habitats with substantial cover, and perform low-risk activities to avoid predation (Cunjak et al. 1998, Finstad et al. 2004b, Johnston et al. 2005).

4 ACCLIMATION OF FISH TO WINTER

Ambient water temperature regulates the basal metabolism of poikilothermic animals such as salmonids. Thus, with dropping temperatures at the onset of winter, metabolic needs “gear down” and salmonids shift to an energetic “save” mode (e.g. Heggenes et al. 1993). Somatic growth ceases (Cunjak & Power 1987b, Cunjak 1988, Bradford & Higgins 2001) or decreases (Cunjak et al. 1987, Metcalfe & Thorpe 1992), even though nocturnal feeding continues throughout the winter (Cunjak & Power 1987, Cunjak et al. 1987, Cunjak 1988, Heggenes et al. 1993, Simpkins & Hubert 2000, Finstad et al. 2004c). Assimilation efficiency is low, however, and energetic deficiencies are common especially during the acclimatization period in early winter (Cunjak & Power 1987b, Cunjak et al. 1987). Deficiencies are not easily overcome during the course of winter and usually continue until water temperatures warm up again in spring (Nicieza & Metcalfe 1997).

5 MOVEMENTS AND HABITAT USE OF FISH – SEEKING FOR SHELTER

As water temperature decreases in autumn and it becomes unprofitable for fish to remain in energetically costly, fast-velocity sites, they shift to areas where they can conserve energy (Fausch &Young 1995, Cunjak 1996). This means that fish of all sizes occupy nearly the same kinds of local habitats in terms of velocity and substrate (e.g. Heggenes et al. 1993, Mäki-Petäys et al. 1997, Armstrong et al. 2003). In preferred winter sites the velocity is low, usually less than 10 cms^{-1}, and the substrate structure is diverse. Velocity and substrate seem to be the most important selection criteria in winter (Cunjak & Power 1986). Winter habitat use, especially for young salmonids, may resemble summer habitat use in terms of average global velocities for the habitats that they use, but fish in winter search for local microhabitats where nose velocities are close to zero. Depending on fish species and life stage, the shift from summer to winter habitats may thus involve movements ranging in length from some metres to over 100 kilometres (e.g. Rimmer et al. 1983, West et al. 1992, Gowan et al. 1994, Young 1998). The shift may occur within or between differing stream sites, sections or even between macrohabitats such as the main river and tributaries or a river and an estuary (e.g. Brown & Mackay 1995, Erkinaro 1995, Young 1998, Bramblet et al. 2002, Lenormand et al. 2004). Typically the shift occurs at water temperatures between 3 and 6°C (Jakober et al. 1998, Nykänen et al. 2001, Bramblet et al. 2002), but temperatures of 10°C for Atlantic salmon (Fraser et al. 1993) and 10-14.5°C for adult European grayling (Nykänen et al. 2004) have also been reported.

During winter, most fish seem to be more or less active, although they rarely move far (Cunjak 1996, Jakober et al. 1998, Bradford et al. 2001, Muhlfeld et al. 2001). If long distance movements are made, they are usually related to unstable ice conditions, such as accumulation of frazil and anchor ice in preferred habitats (Brown 1999, Brown et al. 2000, Simpkins et al. 2000), or to high discharge events (Brown et al. 2001).

Availability of cover has been shown to influence the number of fish that overwinter in an area (Tschaplinski & Harman 1983, Meyer & Griffith 1997, Harvey et al. 1999). Fish found in habitats with little structure in autumn have been observed to move into sites with more complex structure in winter (Mitro & Zale 2002). Small fish are probably better able to use crevices within the substratum, whereas large-bodied individuals may have to move into pools or other deep, slow velocity areas to find suitable shelters from ice and predators (McMahon & Hartman 1989, Cunjak 1996, Robertson et al. 2003). In addition to pools, other slow-velocity habitats, such as off-channel ponds (or alcoves), logjams, undercut banks, swamps, side channels, beaver ponds and tributaries have been identified as suitable overwintering areas for fishes (Bustard & Narver 1975, Tschaplinski & Hartman 1983, Swales et al. 1986, Chisholm et al 1987, Swales & Levings 1989, Nickelson et al. 1992, Cunjak 1996, Harper & Farag 2004). In some systems, areas influenced by groundwater provide the only refugia for overwintering (Power et al. 1999).

6 BIOTIC INTERACTIONS STILL IN OPERATION

Intra- or interspecific competition occurs either indirectly through competition for resources or directly through agonistic behaviour (Wootton 1990). Salmonids have been shown to be less aggressive in winter than in summer (Hartman 1963, McMahon & Hartman 1989, Heggenes et al. 1993, Whalen & Parrish 1999) and territorial behaviour may be less important at this time (Bustard & Narver 1975, Cunjak & Power 1987a, Griffith & Smith 1993).

Several authors have reported overlap in habitat use by juvenile salmonids during winter, suggesting the potential for interspecific competition (Glova 1986, Mäki-Petäys et al. 2000, 2004, Heggenes & Dokk 2001). Heggenes & Dokk (2001) found that niche overlap between juvenile salmon and brown trout is higher during the winter than in summer. Harwood et al. (2002) suggested that competition for food and resources may affect overwintering survival in salmonids as they found that Atlantic salmon became less nocturnal or occupied shallower water when in sympatry with brown trout than when alone.

There is also potential for intraspecific competition as different age classes of the same species have been reported to use similar habitats or food resources during winter (Mäki-Petäys et al. 1997, Whalen & Parrish 1999, Amundsen et al. 2001). The spatial niche of brown trout has been shown to be narrow during winter (Mäki-Petäys et al. 1997) and as a consequence different cohorts may use similar microhabitats. Large individuals have also been observed to dominate over their smaller conspecifics during winter (Gregory & Griffith 1996, Vehanen et al. 1999), even if levels of aggression are lower in winter than in summer (McMahon & Hartman 1989, Harwood et al. 2002).

7 MORE RESEARCH IS NEEDED

There is still much we do not know about the effects of winter conditions on the ecology of salmonids. Ice conditions, for example, are in many cases believed to have negative impacts on fish populations, particularly in regulated and dredged rivers, and yet no studies have quantified winter survival or other effects on fish populations in regulated rivers (Saltveit et al. 2001). Attempts at modeling freeze-up and ice dynamics at local scales, such as individual rapids, have not been satisfying (Alfredsen & Tesaker 2002), although models devised for larger scales, such as river sections, have been successful (Reiter & Huokuna 1995). Predicting the formation and dynamics of frazil and anchor ice and their effects on the behavior and microhabitat selection of salmonids is largely unknown. Surprisingly, even basic habitat preference curves used in habitat-hydraulic modelling are almost totally lacking for young salmonids such as

Atlantic salmon and brown trout (but see Mäki-Petäys et al. 1997, 2004, Armstrong et al. 2003). In summary, we believe that future research should be directed towards (1) being able to predict the dynamics of freezing and ice processes at different scales, especially at the local scale, (2) studying fish behavior, habitat use and preference under partial and full ice cover, (3) evaluating the impacts of man-induced modifications on the ecology of salmonids in winter (e. g. flow regulation, land-use activities) and (4) identifying methods to model and assess winter habitat conditions for salmonids.

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Application of multi scale habitat modelling techniques and ecohydrological analysis for optimized management of a regulated national salmon water course in Norway

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ABSTRACT: To enable salmon friendly hydropower operation we made an attempt to rank and give priority to indicators of hydrological alteration of relevance for maintenance of preferable habitat conditions for Atlantic salmon. Mesohabitat modelling has been tested out for ca 30 km of the river. The regulated part of the river has been visually mesohabitat classified on various discharges (5 – 41 m3/s). Data on juvenile densities at various seasons and locations of spawning by Atlantic salmon and brown trout are linked to physical variables within the major mesohabitats in GIS, to analyse the relationship between fish data, mesohabitat classes and flow. In the entire part of the river affected by regulation, altered erosion and sedimentation processes are possibly degrading the habitats over time. Downstream of the hydropower station, rapid rise and falls in flow do occur, possibly harmful for juvenile fish inhabiting the shallow parts of the river. In the bypass section, the mesohabitat maps show that deep habitats are mainly missing. Alternative ways of spending environmental flows are suggested to possibly improve the habitat conditions for salmon. River channel adjustments optimised for several life stages of salmon in parts of the river are also being tested out in river Surna.

1 INTRODUCTION

1.1 Background

Norway have among the best salmon rivers in the world. However, decline in A. salmon stocks are evident in catch statistics in Norway during the last centuries. The reasons for this are complex, and a mixture of human interventions and natural variations in stocks (NOU 1999). A third of the Norwegian salmon watercourses are affected by hydropower regulations altering the volume and timing of flow and temperature conditions to various extend. There has been an increasing awareness for taking environmental constrains to help the fish stocks. The authorities nowadays require mitigations like minimum flows, restrictions in run of the hydropower plants and stocking of salmon parr and/or smolts to compensate for possible losses of acceptable river habitats for salmon.

2.1 Study area

In 2003, the Norwegian parliament selected river Surna as one of several national salmon watercourses, and mitigations will be given priority in these rivers to protect the Atlantic salmon. River Surna has an altered flow regime due to hydropower regulation since in the late 1960's. The river has a mean annual flow of 56 m^3/s, and a total catchment area of 1200 km2. Atlantic salmon can migrate up to 56 km from the sea in the main stem. Of the lowermost anadromous reach ca 18 km are downstream the outlet of the power plant, with significantly altered seasonal flow and water temperature, like receiving cool water during summer from two bottom intakes of a mountainous reservoir (Follsjøen). Based on an expert judgment (prior to this study) a minimum flow requirement at 15 m3/s is established all year downstream the power plant. Further upstream, ca 13 km of the river is affected by by-pass regulation (Figure 1). The Trollheim power plant produces normally 369 GWh during summer and 436 GWh during winter. The power plant may in periods create rapid and frequent flow variation with stranding of juvenile fish as a consequence.

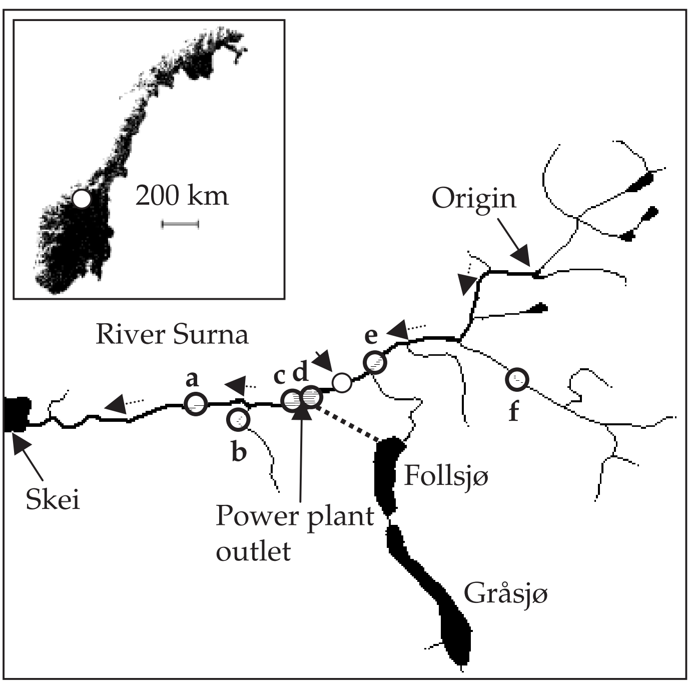

Figure 1. The regulated river Surna, located in Mid-Norway (on the top of Europe!) from origin (the rivers Rinna and Sunna) to the river outlet at Skei and the Follsjø reservoir. Black circles with letters refers to the following locations refeered to in the text; a – Honstad, b – Vindøla, c – Skjermo, d – Harang, f – Løsetli (Rinna).

There is an ongoing debate about the development and fate of the salmonid fish populations in river Surna. Many of the sport fishermen and locals are concerned about the development, blaming the regulation to have the main responsibility for the negative development. Still, the river catches of A. salmon, have been among the top 20 in Norway during the last years.

Previous biological assessments of river Surna have suggested low discharge just upstream of the outlet of the power plant as a migration bottleneck for adult salmon. Significantly many spawning redds were also found here by Lund et al. (2003). Further, stranding of fish and dewatering of redds due to variable flow through the power station may lead to increased mortalities. Factors affecting timing of and survival during smolt migration is also not well understood, but Hvidsten and Møkkelgjerd (1987) found high predation rates of smolt by cod in the estuary of Surna. Hvidsten and Hansen (1988) found that stocking of smolt at high discharge is an effective mitigation effort to increase recapture rates of migrated smolts. Saltveit (1990) found significantly smaller yearlings of both A. salmon and brown trout downstream compared to upstream of the power plant in river Surna. He stated that those species smoltify one year later, and this decrease the production of salmonids due to the cold summer water.

In this study we focus on;

1) the importance of the physical habitat conditions like various part of the hydrology and temperature for the Atlantic salmon habitats by linking physical characteristics driven by discharge to fish data.
2) suggesting some mitigations regarding environmental flow and habitat improving efforts to possibly optimise both the A. salmon habitats and hydropower production.

2 MATERIAL AND METHODS

2.1 Indicators of Hydrologic Alteration

The Indicators of Hydrologic Alteration (IHA) – Methodology is a way of assessing the degree of hydrologic alteration arising from human water- and land uses by defining 32 biologically relevant hydrologic parameters. These parameters are organized in 5 statistics groups that are derived from the flow components *magnitude and frequency, timing, duration* and *rate of change* as changing those due to e.g. regulation are likely to affects the ecosystems and geomorphology (Richter 1997). A Norwegian modification of the IHA have been carried out at 5 gauged or simulated Q-stations in Surna by Dangelmaier (2004).

2.2 Large scale hydraulic modelling

More than 62 profiles have been measured in for various purposes covering more than 31 km of Surna. We have used different land survey techniques including differential GLONASS, Topcon type for measurement of topography points together with water lines at various flows.

The profiles and water line have been incorporated into HEC-RAS (US Army Corps of Engineers 1991). Hec-Ras is a hydraulic river system simulator based on geometric transect data as foundation and hydraulic data for calibration based on the Manning's formula. The model is one-dimensional and can perform static and dynamic calculations of different parameters and variables like water level, water velocity, water surface area, energy gradient and Froude's number among other things. The accuracy of the model depends on the amount of transect per distance and the calibration data available.

The Hec-Ras model of the river Surna downstream the power plant consists of 49 measured transects, on the 18 km reach. This gives an average of 390 m per transect. The transect localizations were chosen to represent the major topographic change in the river and edges of classified mesohabitat types. The calibration was based on consequent water level measurements done on three known discharge distributions in the river. Water level sensor data from four different locations, was also found very useful for quality control of the calibration of the model.

Upstream the power plant, another setup of HEC-RAS was done based on 60 profiles (both measured and interpolated), covering the 13 km of river.

2.3 Mesohabitat mapping and analysis

Mesohabitats were surveyed at several relevant flows both up and downstream of the power plant, covering from 5 to 40 m3/s. We used a method for classification based on observations or estimations of four quantified physical variables: surface pattern (wave height < or > than 0.05 m), surface gradient (slope < or > than 4%), surface velocity (< or > than 0.5 m/s) and water depth (< or > than 0.7 m). This system builds on the widely applied riffle-run-pool methods, but aims to be more detailed, flexible and objective than the previous approaches. The 10 classes are found relevant for classification of A. salmon habitats (Borsanyi, 2003).

The mesohabitat mapping was done from rubber boat, wading or on foot, and sketches were drawn on maps in 1:5000 scale. Handheld GPS and distance measure instrument was partly used during survey. In general, only units longer than they were wide were delimited, due to the scale of this classification. All data was reprocessed into ArcView GIS system of Surna. More than 900 random point measurements velocities, depth and partly substrate have been done in various mesohabitats and flows, to verify the classification.

24 Small scale habitat modelling

At selected reaches, classical small scale habitat-hydraulic modeling has been conducted, on parts of the river covering several of the major mesohabitat types of River Surna. Details about this are given by Kitzler et al (2005) and Sundt et al (2005). The latter have even compared various habitat modeling techniques to test the impact of various hydraulic vs. fish preference data.

In a habitat improving context, Stickler et al (2004) also used River 2D as a tool for optimizing an planned ice channel to create more suitable habitat conditions for all life stages of Atlantic salmon on parts of the by-pass section. All small scale habitat modeling have used general fish preference from snorkeling observations in many Norwegian rivers (Harby et al, 1999)

2.5 *Fish data sampling and fish preference*

Density estimates of juvenile Atlantic salmon and brown trout were done by standard electro-fishing procedures of 26 evenly spread monitoring stations. Fishing was done here in 21.-28. August 2002, and September/October 2003. Partly the same sites have been fished as well by Saltveit (1990). In addition, we established 10 more untypical locations in the by-pass section to cover the main mesohabitats represented in Surna, and get data on spatial and temporal variation. Hence, these locations were fished both in September and November 2003. All fish was length measured, and some were taken for further age determination.

Mapping of spawning redds and areas were done by wading and scuba diving, sketches were drawn on maps in 1:10 000 scale. This was done by biologists from NINA, 18-20 November 2002 from outlet of the hydro power station to the sea, and 11–15 November 2003 from Rindal to Skei (Figure 1) (Lund et al, 2004).

General preferences for juvenile salmon have been used in several of the sub activities in the project (see Sundt et al., 2005), thus no juvenile data except for the density estimates from traditional electro fishing have been collected so far.

3 RESULTS

3.1 *Indicators of Hydrological Alteration (IHA)*

Dangelmaier (2004) has analysed the hydrological alteration at several location, both up- and downstream of the power plant in the main stem of Surna (Figure 1). The analysis shows of Honstad several striking changes prior (1951-1969) versus after regulation (1970 – 1988). The annual 1 – 90 days max discharge has on average decreased by 21 – 35 % after regulation (Table 1). Further, the duration of extreme events like high/low pulses have decreased on average with 55 – 79 %, and the same with the max 24 hour fall and rise in discharge. However, the numbers of high/low pulses have increased after regulation downstream the power plant. A further analysis of the discharge data based on hourly sampled data, showed more than 120 dewatering episodes that might lead to stranding of fish during the last years (Table 2).

Table 1. Summarized IHA analysis giving ratio regulated/unregulated discharge from two stations downstream power plant (Honstad and Skjermo) and three upstream in the by-pass section, based on combination of gauged and simulated discharge. The 1-7 day min or max is mean annual extreme value. E.g. mean winter Q after regulation was 35.2 m3/s at Skjermo while before was 17.2 m3/s giving a ratio of 2.0.

	Honstad	**Skjermo**	**Harang**	**Dønnem**	**Løsetli**
mean winter	2.1	2	0.7	0.88	0.79
mean spring	1.1	0.9	0.55	0.87	0.78
mean summer	0.6	0.8	0.37	0.59	0.49
mean autumn	1.3	1	0.54	0.8	1.38
average runoff	1	1	0.5	0.77	0.8
1 day max	0.8	0.6	0.51	0.78	1.21
1 day min	1.3	4.8	0.56	0.9	0.33
3 day max	0.8	0.6	0.52	0.8	0.98
3 day min	1.5	3.8	0.62	0.96	0.35
7 day max	0.7	0.6	0.51	0.8	0.98
7 day min	1.7	3.4	0.65	0.93	0.38

Table 2. An analysis of number of dewatering episodes downstream Trollheim power station in Surna in the period 2001-03-01 to 2003-11-23, based on hour discharge data from Skjermo gauging station.

		Reduction in discharge pr hour ($m^3/s/h$)		
		10 - 20	**20 - 30**	**> 30**
Discharge (m3/s)	**15 - 60**	84	5	4
	60 - 120	10	0	2
	> 120	17	1	3
	Sum	**111**	**6**	**9**

A decrease in mean discharge occurred after regulation in the summer months for all locations affected by the regulation (May – August). On the other hand, the mean discharge have on average been higher after regulation for all the other months downstream the power plant, and most during mid winter conditions (e.g. more than 2x mean discharge for December and January). The annual lowest 1 – 90 day minimum discharge has increased on average by 26 to 104 %

after regulation. The Julian date for beginning of snowmelt and winter/spring 1 day max has been earlier after regulation.

In the by-pass section upstream from the outlet of Trollheim power station, the hydrological alterations have turned out differently, and no minimum flow is established here. The analysis of IHA at Løsetli (1958- 1968) versus (1969 – 1980) showed that mean monthly discharge is decreased on average by more than 50 % in February, March, June, July after regulation. By 6 to 30 % in January, April, May, and actually increased in the other months. The annual 1 – 90 days minimum and maximum discharge is lowered and beginning of snowmelt is delayed after regulation.

The analysis of hydrological alteration showed that low flow (1 – 90 days minimum flow) events are increased by 1.5 to 4 times after regulation and the mean seasonal flow are more than doubled during winter and reduced the rest of the year downstream the outlet of the power plant. The degree of flow alteration varies for various in the by-pass section. The mean seasonal flows have been reduced up to 64 % while the climatic variation before and after regulation have actually led to increased mean flow and max floods in some months after regulation in the uppermost part.

3.2 Temperature alteration

In parts of the year the main contribution of flow in river Surna, is supplied from the power plant. This water has its origin from the two bottom intakes of the Follsjø reservoir at 375 masl. The reservoir may be up to more than 50 meter deep, leading to a severe alteration of the water temperature in river Surna. Typically the water temperature downstream the power plant is therefore 2-5 C colder than upstream the power plant during summer (Tjomsland, 2004). Reduced water temperature decrease the growth of juvenile salmonids, and possibly decreased production and survival of salmon and trout smolts as a factor of increased smolt age (Saltveit, 1990). In addition, the hydropeaking may be even more harmful in rivers with slow growing yearlings, thus the smallest juveniles of salmonids is most affected by stranding (Halleraker et al, 2003). Several studies have found an almost linear relationship between activity and water temperatures for juvenile salmonids, and no threshold switch temperature (e.g. Bremset, 1999).

In contrast, in the cold part of the year the intake arrangements make the water through the power plant to be typically more than 2 C warmer than upstream the power plant. The altered temperature is affecting large parts of the river on its way to the sea. On average, the 9.5 km of river between Skjermo and Honstad is warming up the water less than 1 C in July (Table 3). During winter a cooling of ca 2 C occurs on average in the coldest months on the same reach. Increased water temperature during winter leads to lack of ice cover and unfavorable energy consumption of fish. Both factors may lead to sublethal effects and possibly increase mortality of juvenile salmonids (Finstad et al., 2004).

Based on simulated and measured water temperatures, the impact of mitigations of surface intake in the reservoir and/or modified power productions on the mean water temperatures at Skjermo and Honstad are shown in Table 3. The results show that only a new intake arrangement that bring winter cold surface water of Follsjø to Surna, ensure river water to become below 0 C. On average, river Surna may return to surface ice condition in the lower 8-10 km with surface water in December – February with such a mitigation. Reduced run of water through Trollheim power plant is the most efficient mitigations to increase summer temperatures in river Surna. Still, intake of surface water in July, will on average rise river temperatures more then 1 C, giving a significant contribution to higher growth rates of juvenile salmonids. Various temperature regimes have been tested on a growth model from a neighbor catchment. The results from the tested mitigations shows that several action, like temperature depended power production and/or arrangement of a new surface intake may give a significant contribution to increased production of smolts in river Surna.

Table 3. Impact on mean water temperatures (C) pr month (M - no) in the period 1998 – 2001 from various intake (bottom and surface) in Follsjø reservoir and production of water through Trollheim power plant (pp), based on a combination of measured and simulated data, modified from Tjomsland (2004). Norm. = present situation, -50 % is half production through pp, -90% is 90 % production . *In italic is mean discharge through power plant and in the by-pass section in the same period.*

	Skjermo						**Honstad**			***Discharge (m3/s)***	
	Bottom intake			Surface intake			Bottom	Surf		*pp*	*By-pass*
M	Norm	-50%	- 90%	Norm	-50%	- 90%	Normal			*m3/s*	*m3/s*
1	2.4	2.2	1.5	1.6	1.5	1.0	0.9	**-0.6**		*24.1*	*5.7*
2	1.4	1.2	0.8	1.4	1.2	0.8	0.5	**-0.2**		*22.1*	*15.4*
3	1.2	1.1	0.7	1.5	1.3	0.9	0.6	0.2		*26.2*	*6.5*
4	1.5	1.4	1.1	1.8	1.7	1.3	1.2	1.4		*28.5*	*24.5*
5	4.4	4.4	4.3	4.5	4.4	4.3	4.1	4.3		*21.4*	*48.3*
6	7.5	8.1	9.5	8.3	8.7	9.7	8.0	9.4		*30.7*	*21.2*
7	10.1	10.7	12.3	11.5	11.9	12.8	10.8	13.0		*22.6*	*17.6*
8	10.9	11.2	12.0	11.3	11.6	12.3	11.0	11.6		*29.1*	*17.2*
9	10.8	10.7	10.6	10.3	10.3	10.3	10.8	10.4		*27.7*	*5.4*
10	7.8	7.5	6.5	7.3	7.0	6.2	7.8	7.3		*22.6*	*8.6*
11	4.1	3.9	3.3	3.3	3.2	2.8	3.1	2.6		*22.0*	*15.8*
12	3.0	2.8	2.1	1.3	1.3	1.1	1.0	**-0.6**		*23.5*	*6.8*

3.3 Hydraulic diversity and water covered areas

A further evaluation and quantification of discharge steps that are most likely to give stranding is given in Figure 2. The figure shows only minor loss of water covered areas between 37 and 30 m3/s, and an increasing relative impact of dewatered areas especially at discharge less than 13 m3/s (below the current minimum flow requirement). Water level sensors in various mesohabitats in Surna, documented average water level drops of 0,5 -1,5 cm pr m3/s. Typically the lowest water drops occur in mesohabitats with most stranding areas. We also classified the river margins subjectively based on factors known to have impact on stranding of juvenile salmonids; like slope, substrate composition, cover and mesohabitat. By linking this to the hydraulic modeling results, we have quantified recommended threshold values for environmental justified dewatering speed of the power plant in Surna, to reduce stranding significantly based on experience from previous experimental studies (Halleraker et al., 2003; Saltveit et al, 2001). The recommended dewatering scheme is given as a table to the power company. E.g. from 22 to 15 m3/s, more than 22 min pr m3/s is suggested, while only more than 5 min pr m3/s is needed from 50 to 35 m3/s. At particularly critical parts of the year, like the first weeks after swim up of yearlings, rapid dewatering should be avoided. In addition, dewatering of the riverbanks should be done in darkness if possible. Despite all this mitigations, ramping will not exclude stranding totally, and e.g. stranding in pocket water may still occur.

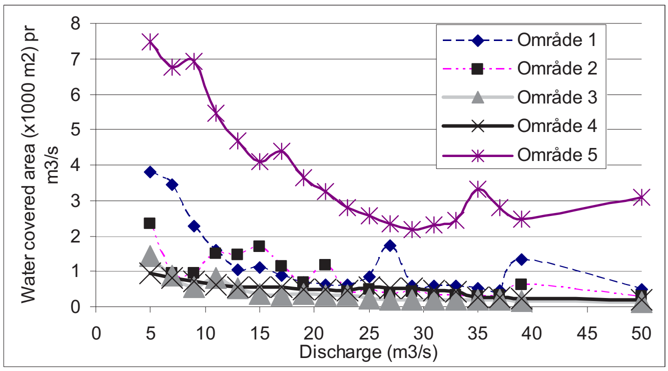

Figure 2. Dewatered areas (x 1000 m2) pr m3/s reduced flow from 50 – 4 m3/s split on sub areas (område 1-5) in river Surna. Data from Hec-Ras simulations calibrated at 6, 21 and 40 m3/s.

3.4 *Mesohabitat versus flow*

Mesohabitat was mapped at three flows both upstream and downstream of the power plant (Table 4). Mesohabitat classification is a cost-efficient method to cover large areas on a catchment scale. Still, better incorporation of fish data linked to the mesohabitat characteristics needs to be achieved. In

Figure 3 spawning registrations is linked to the mesohabitat maps at the dominating flow during the spawning period.

Table 4. Percentage distribution of mesohabitats in Surna at selected discharges. Letters refers to meso-habitat classes according to Borsanyi et al. (2003), depending on surface chraracteristics, slope and surface velocities.

Discharge	Sum A,E,F H	Deep areas			Shallow areas		
		B1	C	G1	B2	D	G2
40 m^3/s* downstream	4.0	47.7	19.9	9.2	9.8	5.3	4.0
35 m^3/s** upstream	4.4	34.1	2.7	24.3	17.2	1.2	16.1
10 m^3/s,**upstream	0.6	0.8	6.5	0.1	38.8	33.8	19.4

Deep areas >0.7 m. C/D < 0.5 m/s, B and G >0.5 m/s. B - smooth surface, G – turbulent habitat

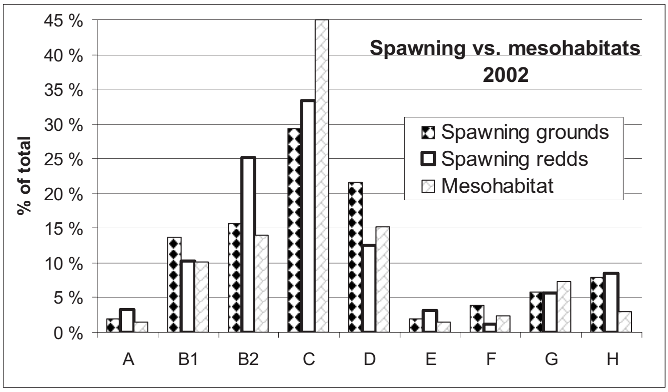

Figure 3. Proportion of total registrated spawning grounds (n= 51) and spawning redds (n= 564) in Surna from outlet of power plant to Skei in November 2002 by NINA (Lund et al, 2003) versus proportion of mesohabitats mapped at 16 m3/s.

4 DISCUSSION AND RECOMMENDATIONS

In the first phase of this project, we have based our recommendations on updated knowledge and general links between physical conditions and preferences of salmonids (mainly juveniles). Better knowledge on the seasonal habitat use and the habitat value of the large calm pool areas in the lower parts of Surna is of major importance for the overall conclusions and final environmental flow requirements. Bremset (1999) have found extensive use of pools by juvenile brown trout and Atlantic salmon in several rivers in Norway including river Vindøla – one of the tributaries to Surna. He even found 2,5 times more parr in pools than in riffles. Based on his diving observations, plenty of good deeper habitat may still be present in river Surna at flows below the present minimum flow requirement of 15 m3/s. Monitoring of salmon stocks, based on electrofishing is widely used in Norway. Electrofishing by wading may give biased results due to the shortcomings of the method of covering all major mesohabitats in salmon rivers. So other fish data sampling techniques need to be applied.

The analysis of hydrological alteration developed by Richter et al. (1997) is a strong tool for systematic hydrological characterization with relevance for biology and morphological processes in rivers. The management of rivers in Norway is advised to standardise this or similar analysis as a tool for better comparison of environmental flow and for evaluation of mitigation to make better transferability between rivers.

The IHA analysis of several parts in Surna, have documented that the level of low flow situations have been drastically increased downstream the power plant after regulation. It is very likely that this 1,5 – 4 times increased level of the yearly 1 – 30 days minimum flow have increased the production of salmon in Surna. Climatic variations also play a significant role here, and have even created higher maximum floods after versus before regulation in the upper part of the by-pass section. Still, severe floods may lead to flushing of the smallest fry, so a short term effect the dominating reduced magnitude and duration of floods due to regulations is probably reduced loss of the youngest year classes of salmonids. According to the law the power company is stocking Surna with juvenile salmon as a mitigation effort. Despite low recaptures of the stocked parr or smolt, this is considered to improve the salmon production (Lund et al., 2004).

The following prioritised impacts of the regulation in Surna have been found in the first phase of the project, with suggestions to mitigations:

1. Rapid and frequent dewatering of the river banks, particularly early in the growing season for salmonids, and during daytime midwinter. This leads to stranding of juvenile fish. However, seasonal habitat use and the importance of the large deep pools in river Surna is not enough understood. The following mitigations may be applied:

a. Run of the Trollheim power plant should be adjusted to know strategies that reduce stranding significantly. This includes to follow our recommended guidelines for ramping and stabilize the flow as much as possible early in the growing season for juvenile salmonids and do dewatering in darkness. Procedures and technical solutions for doing this should be evaluated further.

b. To establish more optimal substrate and cover condition for juvenile salmonids under the dewatering zone may give a significant contribution to the production of salmonids downstream the power plant and reduce stranding.

c. Drop out of the power plant may cause severe stranding. A bypass is expensive and must be evaluated versus the cost for other mitigations.

2. Altered temperature regime compared to natural situation, due to bottom intake from the high-head reservoir. Decreased water temperature is evident during summer. Most likely this leads to reduced growth rates, increased smolt age and increased stranding of juvenile salmonids. During winter the increased water temperatures leads to lack of ice cover and possibly altered energy consumption of the salmonids and consequently increased mortality.

Suggested mitigation:

a. Establishment of a surface intake in the Follsjø reservoir.

b. Temperature depended run of the power plant.

Both mitigations are found to give significant contribution in parts of the year to diminish the temperature alteration and hence increase the smolt production. Only surface intake in the reservoir ensure ice cover for larger areas of river Surna downstream the power plant during mid-winter. Reduced flow through the power plant especially in July will give a boost to the fish growth, but must eventually be considered versus the loss of water covered areas. Our large scale modeling and mapping of mesohabitat at various flows are useful tools in this respect.

3. The IHA analysis and large scale hydraulic modeling of river Surna upstream the power plant, showed critical periods with low flow and unfavorable habitat conditions. In longer periods of the year this part is shallow and deeper areas are rare. Unfavorable habitat for all life stages of salmonids is then dominating.

a. Habitat improving effort by rehabilitating optimal salmon habitat is being tested out on a ca 800 m reach of Surna, based on a habitat optimalisation by use of River 2D (Stickler et al., 2003). This may serve as sufficient mitigations for ensuring good enough ecological potential in this modified water body within the water framework directive, but must be evaluated versus the establishment of a environmental flow requirement also in this part of the river. A first step in micro scale habitat modeling for quantification of optimum minimum flow requirements is given by Kitzler et al (2005) and Sundt et al. (2005), showing optimum flow at 8 – 10 m3/s at a modelled reach of the by-pass section.

The impact of altered sediment transport and erosion patterns affecting the substrate composition and embededness in particular, but is not really quantified in the first phase of the project. Alfredsen et al. (2004) showed clear indications for increase sedimentation of deeper areas. This may be an interesting following up, and hence optimum substrate and cover are for sure crucial for optimal raring habitats for salmonids, particularly during winter in rivers without ice-cover like Surna.

We see the main potential for more optimal power production in river Surna by a more flexible minimum flow requirement below the power plant. Our investigation have shown that no major change in mesohabitat composition occur between 15 and 6 m3/s below the power plant. The minimum flow requirement may be linked to a % of a gauging station in a natural flowing river in the region.

We believe multi-scientific approaches by linking the physical conditions to fish data are powerful tools for more optimized management of large regulated rivers like river Surna.

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Norwegian mesohabitat method used to assess minimum flow changes in the Rhône River, Chautagne, France. Case study, lessons learned and future developments - methods and application

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ABSTRACT: Eight bypass sections of the Rhône River are being rehabilitated in order to improve ecological status. At Chautagne, most of the flow is diverted to a canal leading to a hydropower plant. From July of 2004, the environmental flow to the approximately 6 km old branch of the Rhône River was increased from 10 (winter) – 20 (summer) m^3/s to 50 (winter) – 70 (summer) m^3/s. A scientific study program will investigate changes in the fish and invertebrate population. The Norwegian method of classifying the river section into maximum ten different physical meso scale morphological classes (mesohabitat classes) was applied at 10 m^3/s and 70 m^3/s. The results show a change in mesohabitats after the flow increase. The dominating class is still deep and low velocity pools (type C), but a higher physical diversity occurs at 70 m^3/s. Diversity indices from Simpson (1949) and Shannon & Weaver (1962) show increased diversity which is more pronounced if we leave all pool habitats outside the calculation. The analysis also show less dominance (O'Neill 1988) and more dissected (Li and Reynolds 1993) composition of mesohabitats at 70 m^3/s compared to 10 m^3/s. The results show that mesohabitats of low velocity and low depth without surface waves (type D) almost disappears at 70 m^3/s. Mesohabitats of high velocity and both high and low depths without surface waves (types B1 and B2) cover less than 5 per cent of the total area at 10 m^3/s, but cover almost 20 per cent of the total area at 70 m^3/s. This will give impacts on the composition and abundance of fish and invertebrates. Rheophilic taxa like *Hydropsyche* sp. (Trichoptera), *Heptagenia sulphurea* (Ephemeroptera) and *Potamopyrgus antipodarum* (Gasteropod) should be favourised by the increased flow while more limnophilic species like the Ephemeropteran taxa of the *Caenis* genus or the Trichopteran taxa *Hydroptila* sp. *Polycentropus flavomaculatus* or *Psychomyia pusilla* will find less amount of suitable habitat at 70 m^3/s. Impacts on the fish population will be shown and discussed in twin paper by Olivier et al (2005). Verification of manual estimates of depth and water velocity were done in 10 (low flow) and 24 (high flow) mesohabitats. In total, 6 % of the depth measurements and 0,3 % of the velocity measurements were outside the range of the mesohabitat at low flow. At high flow, 16,2 % of the depth measurements and 19,6 % of the velocity measurements were outside the range of the mesohabitat. Even though there are still some challenges to look at dynamics and heterogeneity among mesohabitat classes, we believe our method may be a useful tool to compare different flow situations and their impact on the invertebrate and fish population structure even in large rivers.

1 INTRODUCTION

Eight bypass sections of the Rhône River are being rehabilitated in order to improve ecological status. Since 1980 most of the flow is diverted to a canal leading to a hydropower plant at Chautagne. From July of 2004, the environmental flow to the approximately 6 km old branch of the Rhône River was increased from 10 (winter) – 20 (summer) m^3/s to 50 (winter) – 70 (summer) m^3/s. A scientific study program will investigate changes in the fish and invertebrate population. This paper describes methods, models and some of the first results for physical conditions and impacts on invertebrates. Olivier et al (2005) will give more details on impacts for fish.

2 STUDY SITE

The dam and hydropower plant in the Rhône river at Chautagne was built in 1980. The mean annual natural daily discharge of the Rhône River at the site of Chautagne is estimated to 450 m^3/s. In the period 1997-2004, the flow in the bypass channel was at minimum flow 78 % of the time. 12 % of the time the flow was still less than 100 m^3/s, 9 % of the time it was more than 100 m^3/s and only 1 % of the time more than 500 m^3/s. Figure 1 shows a graph of the observed mean daily flow in the hydropower canal and the downstream total of the flow for 2001-2002. Even at 50-70 m^3/s, most of the river meandering bed is not wetted, leaving large gravel bars dry. However, floods passing over the dam reshape the river bed frequently regardless the environmental flow. Figure 2 shows a picture of the study site. Figure 3 shows a map of the upper Rhône river basin where Chautagne is located.

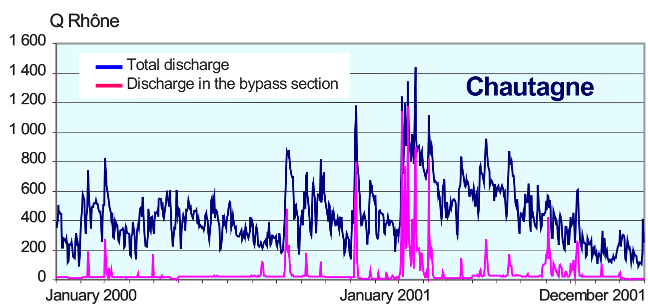

Figure 1. Hydrograph for the Rhône River at Chautagne 2001-2002.

Figure 2. A photo of a part of the bypass section of the Rhône river at Chautagne at low flow in summer, i.e. 20 m^3/s.

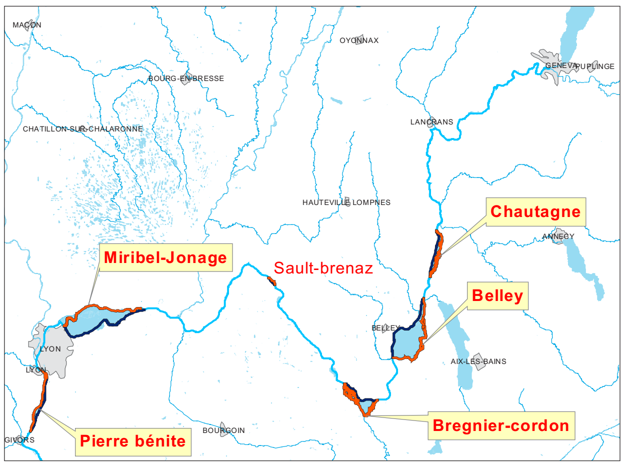

Figure 3. A map of the upper Rhône River showing the location of Chautagne.

3 METHODS AND MODELS

3.1 Mesohabitat mapping

The Norwegian method (Borsányi et al 2003) of classifying the river section into maximum ten different physical meso scale morphological classes (mesohabitat classes) was applied at 10 and 70 m^3/s. The classification is carried out by manual observations and estimations of four key factors: surface pattern (wave height), surface longitudinal gradient, water velocity and water depth. The factors are generalised for a wet area of at least on river width in the length direction. It is possible to make maximum 3 classes across the river. Figure 4 shows the composition of the mesohabitat classes (geomorphologic units). Each class is defined by following the rules given in table 1 and 2.

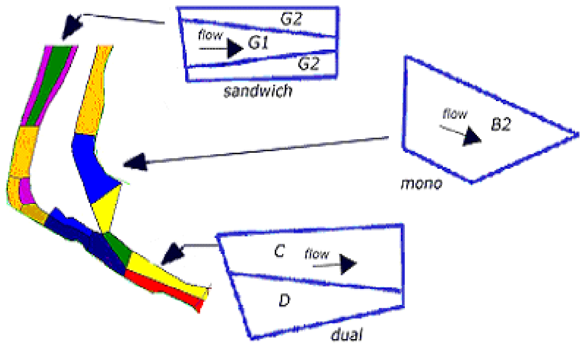

Figure 4. Class layout and configuration from Borsányi et al (2003).

Table 1. Criterion for classification of stream reaches from Borsányi et al (2003). Empty cells in "Class" are due to the fact that some combinations are practically impossible in natural rivers.

Surface pattern	**Surface gradient**	**Surface velocity**	**Water depth**	**Class**
Smooth	Steep	Fast	Deep	**A**
Smooth	Steep	Fast	Shallow	
Smooth	Steep	Slow	Deep	
Smooth	Steep	Slow	Shallow	
Smooth	Moderate	Fast	Deep	**B1**
Smooth	Moderate	Fast	Shallow	**B2**
Smooth	Moderate	Slow	Deep	**C**
Smooth	Moderate	Slow	Shallow	**D**
Broken	Steep	Fast	Deep	**E**
Broken	Steep	Fast	Shallow	**F**
Broken	Steep	Slow	Deep	
Broken	Steep	Slow	Shallow	
Broken	Moderate	Fast	Deep	**G1**
Broken	Moderate	Fast	Shallow	**G2**
Broken	Moderate	Slow	Deep	
Broken	Moderate	Slow	Shallow	**H**

Table 2. Criterion and limits between categories.

Variable	Criterion	Limits
surface pattern	smooth	wave height < 5cm
	broken	wave height > 5cm
surface gradient	steep	slope > 0,4 %
	moderate	slope < 0,4 %
surface velocity	fast	> 0,5 m/s
	slow	< 0,5 m/s
water depth	deep	> 0,7 m
	shallow	< 0,7 m

Mesohabitats were classified at low flow (approximately 10 m^3/s) 23-24 April 2004 and at high flow (approximately 70 m^3/s) 15 December 2004 and 9 February 2005. During all campaigns, mesohabitats were drawn on copies of air photos of the river using a boat and by wading. The same crew was used for all campaigns. Points marked on a handheld Meridian GPS were used to obtain correct positions. Data were post-processed in ArcGIS superposed to the digital photos and existing digital maps.

3.2 Landscape metrics

Through the use of spatial metrics derived from areas like landscape ecology (Li and Reynolds 1994, Malcolm 1994, O'Neill et al. 1988, Turner et al. 1989) and signal processing (Shannon and Weaver 1962), the spatial habitat arrangement in the river can be described. The metrics classifies either composition or configuration in the landscape. Through definition of the habitat patches, we can combine the physical variables into compound habitats and use it in the analysis of the habitat variability. We can also look at interfaces between adjacent patch types and identify the important edges e.g. between resting and feeding areas (Bovee 1996). The spatial metrics can be combined further with time series of flow to show the changes in spatial composition and configuration over time, or to compare different flow scenarios.

In this study, landscape metrics are used to compare diversity, contagion, patchiness and dominance at low and high environmental flows. We used diversity indices from Shannon and Weaver (1962) and Simpson (1949). To investigate the dominance of mesohabitat classes, we used the index from O'Neill et al (1988). Li and Reynolds (1993) contagion RCI index was used to analyse the contagion. A landscape is clumped if this index is greater than 0.66, and dissected if it is less than 0.33. Calculations of landscape metrics were done in ArcGIS and MS Excel.

3.3 Physical conditions within mesohabitats

Within each mesohabitat class the local velocity and depth may show a high variability, even though the mesohabitat class is supposed to be defined as a homogenous object. To investigate the local variability across and within mesohabitat classes, at least 30 random measures of velocity and depth were taken in 10 different mesohabitats at low flow and 24 different mesohabitats at high flow. In general, we tried to measure variables in at least two sites of each important mesohabitat. The measurements were conducted by wading and the use of Ott C2 current meters for velocity measurements and a ruler for measuring depth, substrate size, subdominant substrate size and roughness. The embededness was also estimated. At sites not accessible by wading, a RDI ADCP (Acoustic Current Doppler Profiler) of 2400 or 1200 MHz was used to measure local velocity and depth in non-random transects.

We used FST-hemispheres (Statzner and Müller, 1989) to measure near-bed hydraulic forces at the point where the Hess sampler had been used for invertebrates (see text in 2.4). This simple method involves the use of 24 standard hemispheres of identical size (diameter 7.8 cm) and surface texture, but different densities. Hemispheres are exposed sequentially on a small weighted horizontal plexiglass plate on the stream bottom and the heaviest hemisphere just moved by the flow defines the instantaneous flow condition near the stream bed.

3.4 Invertebrate sampling

A total of 60 invertebrate sample-units were taken in spring 2002 (n=20), in summer 2002 (n= 20) and in winter 2003 (n= 20) using a Hess type sampler (area 0.05 m2, mesh size 200 m). The sample-units were taken regularly over the total reach length and across the width of the river.

3.5 Habitat models for invertebrates and fish

The mesohabitat method may be used in combination with habitat-hydraulic models like PHABSIM (Bovee 1982), EVHA (Ginot et al 1998) or HABITAT (Harby and Heggenes 1995) to show how habitat conditions vary within a mesohabitat unit and then within the river classified into mesohabitats as shown in Harby et al (2004). Statistical habitat models like ESTIMHAB (Lamouroux and Capra 2002) may also be used.

In this part of the study, we have chosen to assign a preference to each mesohabitat class for each taxa or group of invertebrates and fish. This is a generalized approach hopefully suitable to study intra- and interspecific relationship and competition between the species. Invertebrate preferences are made from a limited *in situ* data sampling to demonstrate the method and will be supplied with more data later. Fish preferences are taken from the literature (Lamouroux et al 1999) with some modifications based on *in situ* data sampling (see Olivier et al 2005).

4 RESULTS

4.1 Mesohabitat classification

The classifications at low and high flow are shown in figure 5 and 6. The Mesohabitat class C dominates at both flows. The area of mesohabitats at each flow is shown graphical in figure 7 while the numbers of each mesohabitat at both flows are shown in figure 8. Figure 9 compares the mesohabitats at low and high flow when the area of mesohabitat class C is removed. The total wetted area increases with 173 903 m^2 from 742 747 m^2 at 10 m^3/s to 916 649 m^2 at 70 m^3/s.

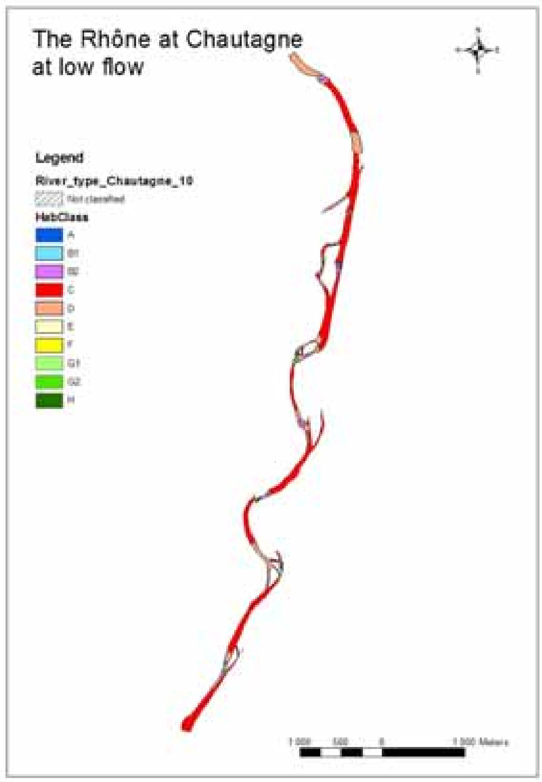

Figure 5. Mesohabitat at low flow.

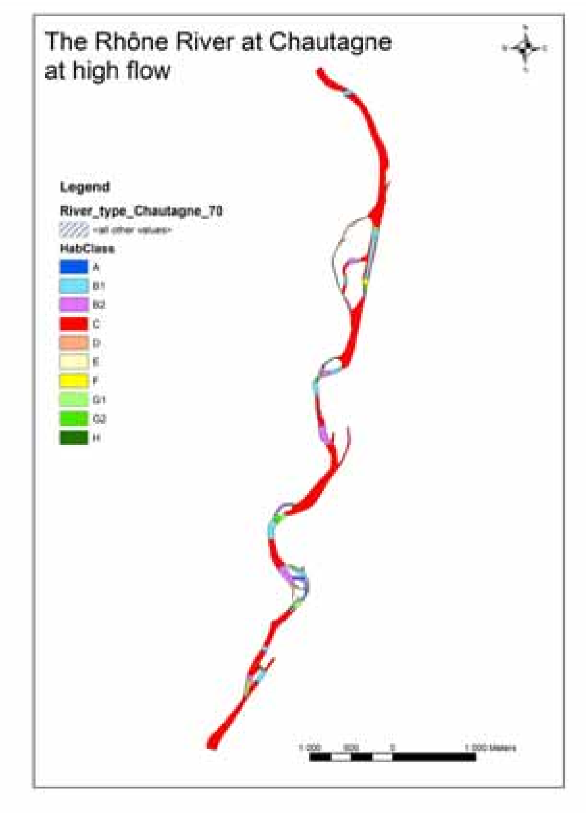

Figure 6. Mesohabitat map at high flow.

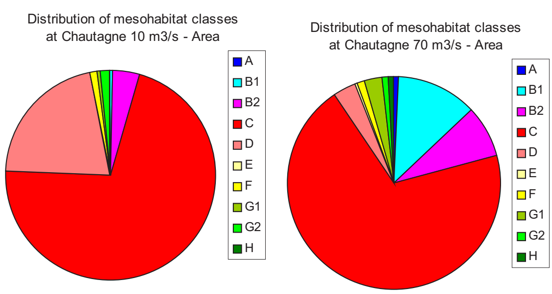

Figure 7. Distribution of mesohabitat areas at low (left) and high (right) flow.

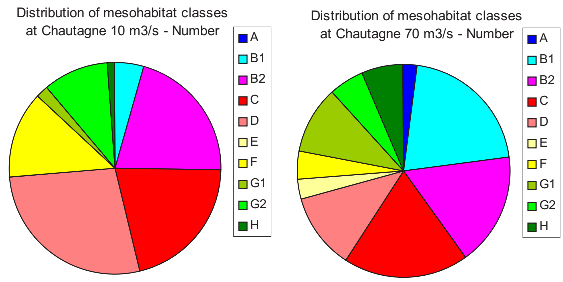

Figure 8. Distribution of mesohabitat numbers at low (left) and high (right) flow.

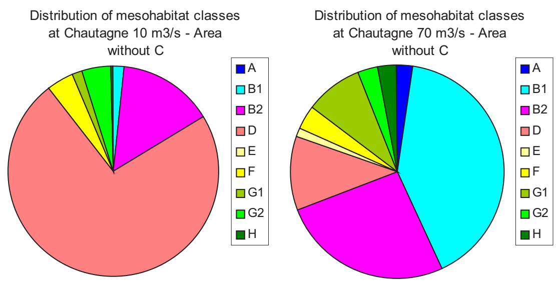

Figure 9. Distribution of mesohabitat areas without class C at low (left) and high (right) flow

Change in mesohabitat from low to high is shown in figure 10. Table 3 gives the total area at each flow, the change in m^2 and percentage and the proportion of change.

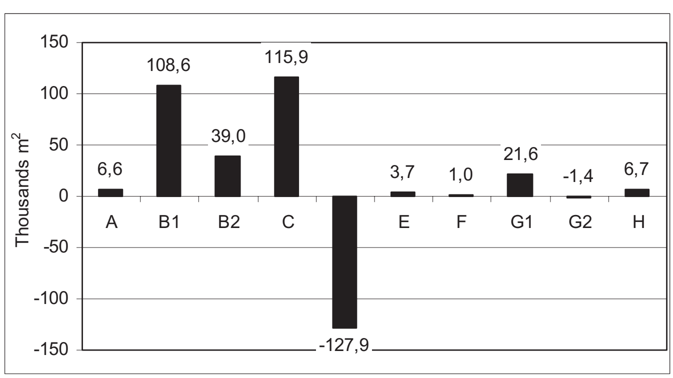

Figure 10. Mesohabitat change by increased environmental flow in area [thousands m2].

Table 3. Area and change in mesohabitats at low and high flow. "Prop. Change" is calculated by comparing the class proportion at low flow (class area divided by total area) with the class proportion (class area divided by total area) at high flow.

Class	Area $10 m^3/s$ [1000 m^2]	Area $70 m^3/s$ [1000 m^2]	Difference [1000 m^2]	Change [%]	Prop. change [%]
A	0	6,6	6,6		0,72
B1	3,4	112,0	108,6	3150	11,76
B2	32,0	71,1	39,0	122	3,44
C	525,5	641,4	115,9	22	-0,78
D	158,8	31,0	-127,9	-81	-18,0
E	0	3,7	3,7		0,40
F	9,2	10,2	1,0	11	-0,13
G1	3,2	24,8	21,6	683	2,28
G2	9,7	8,3	-1,4	-14	-0,40
H	0,8	7,5	6,7	810	0,71

4.2 Landscape metrics

Table 4 shows the results for some selected landscape metrics at low and high flow.

Table 4. Some landscap**e metrics at low and high flow.**

Index (see chapter 2.2)	*Low flow*	*High flow*
Shannon-Weaver index	0,88	1,11
Simpson diversity index	1,82	1,95
Dominance index	1,42	1,20
Contagion RC1	0,60	0,40

Mesohabitat C dominates at both flows (figure 6). To investigate the remaining landscape (figure 8), we removed mesohabitat C from the data and analysed the data with results given in table 5. The index of contagion is not calculated while neighbours cannot be identified without all mesohabitats present in the analysis.

Table 5. Some landscape metrics without mesohabitat class C at low and high flow.

Index (see chapter 2.2)	*Low flow*	*High flow*
Shannon-Weaver diversity	0,93	1,50
Simpson diversity	1,78	3,89
O'Neill et al dominance	1,37	0,80

4.3 Physical conditions in mesohabitats

Figure 11 shows measurements of depth and velocity at low flow conditions in the 6 most frequent mesohabitats. Figure 12 shows measurements of depth and velocity at high flow conditions.

In total 60 measurements of FST were taken randomly within three different mesohabitats. The frequency of FST numbers is shown in figure 13. The data show a clear difference in FST numbers between the mesohabitats B2, D and F. FST numbers between 3 and 7 (4,5 average) are found in mesohabitat D. In mesohabitat F, only FST numbers 7,8 and 11-15 are found, with an average of 12,3. The widest range of FST numbers (3-15) is found in mesohabitat B2, but the average of 9,1 is well in-between the average of D and F.

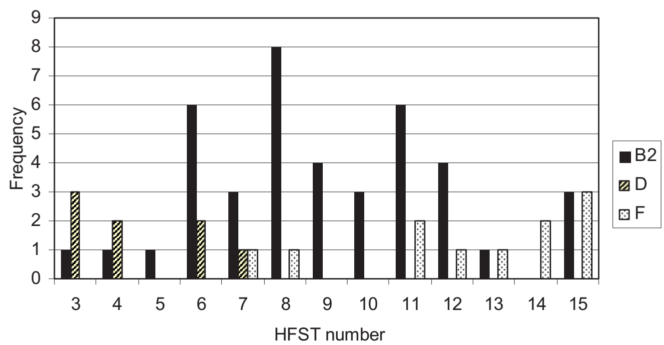

Figure 11. Point measurements of depth and velocity in different mesohabitats at low flow.

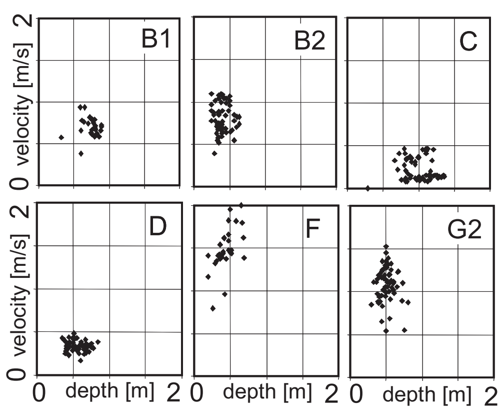

Figure 12. Point measurements of depth and velocity in different mesohabitats at high flow. Notice changed scale in upper right graph.

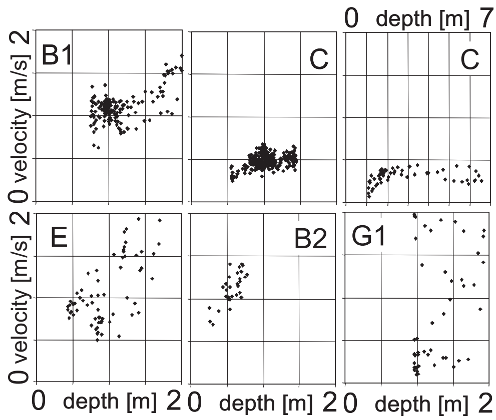

Figure 13. Distribution of measurements of FST in three mesohabitats.

4.4 Invertebrates

Field measurements of standard hemispheres at each invertebrate sample-unit made it possible to make preference functions for many invertebrate taxa. We used Schmedtje's method (1995) based on non linear regression analysis to define preference curves (see details of the method in Mérigoux and Schneider 2005). Depending on the model parameters, and thus the shape of the curves we could define 4 preference categories (limnobiontic, limnophilic, rheophilic and rheobiontic) and therefore 4 groups of taxa depending to their affinity for FST-hemispheres numbers. Table 6 shows the preference for some species while figure 14 shows how preference is modelled for two example species, one rheophilic and one limnophilic.

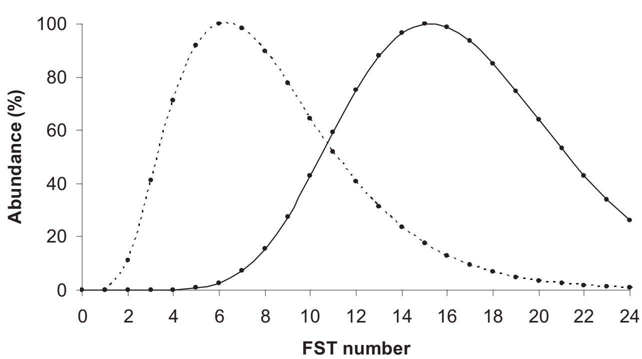

Figure 14. Examples of two FST-hemisphere preference curves modelled from invertebrate and local FST measurements. *Ancylus fluviatilis* (bold line) with a maximum of individuals in FST number >10 is a rheophilic species in spring at Chautagne whereas *Psychomyia pusilla* (dotted line) with a maximum of individuals in FST number < 10 is a limnophilic species in summer at Chautagne.

Table 6. Near-bed hydraulic conditions preferences for some of the invertebrate taxa encountered in spring and summer. ("S": Summer and "Sp": spring). Preferences might vary during ontogenesis and therefore between seasons. "FST pref." is a rough description of the range of FST numbers where a taxon was most abundant (data from the modelled preference curves, see text). The four preference categories refer to taxa with high affinities to low hydraulic conditions (limnobiontic), affinities to low hydraulic conditions (limnophilic), affinities to high hydraulic conditions (rheophilic) and strong affinities to high hydraulic conditions (rheobiontic).

Taxon	FST Pref.	Category
Caenis sp.(S)	3-7	Limnobiontic
Polycentropus flavomaculatus (S)	3-7	Limnobiontic
Hydroptila sp. (S)	3-7	Limnophilic
Psychomyia pusilla (S)	3-7	Limnophilic
Hydropsyche sp. (S+Sp)	6-12	Rheophilic
Heptagenia sulphurea (S+Sp)	6-12	Rheophilic
Gammarus sp. (juveniles) (S+Sp)	6-12	Rheophilic
Potamopyrgus antipodarum (Sp)	6-12	Rheophilic
Hydropsyche contubernalis (Sp)	8-15	Rheophilic
Ancylus fluviatilis (Sp)	8-15	Rheophilic
Baetis luther (S+Sp)	8-15	Rheophilic
Rhyacophila s. stricto sp (S+Sp)	8-15	Rheobiontic
Hydropsyche exocellata (Sp)	8-15	Rheobiontic

5 DISCUSSION

5.1 Mesohabitats and landscape metrics

Figure 5-7 show that mesohabitat C dominates the distribution. This is quite normal in large rivers at low flows, remembering that even our "high flow" is a small flow compared to natural mean flow or floods. There are also quite large variations among C classes; depths of 7m have been measured at some points in the river. Variations in velocity between 0 and 0,5 m/s may also be found within class C, naturally giving quite different conditions for invertebrates and fish. However, physical conditions in a typical class C habitat do not vary much with flow compared to variations in other classes. In addition, class C is probably not the most ecological important zones (Olivier et al 2005), and hence we did not focus on studies of class C. However, the presence of class C may play an important role for other habitat types and the function of the aquatic ecosystem.
At low flow, class A and E are missing and B1, G1 and H cover a very small area.
At high flow, all classes are present, but A, E, G2 and H cover small areas.
Classes B1, B2, C and G1 increase their area a lot when flow is increased, while class D is reduced with 81 per cent (figure 10 and table 3). The change in class C may not be very important because it is only a relative change of 22 per cent, but the other changes may have an important ecological impact.

The diversity of mesohabitats is higher at high flow, calculated either with the index of Shannon-Weaver or Simpson (table 4). At low flow, there is also a higher dominance by certain habitat classes (C and D) than at high flow. While the Li and Reynolds's contagion RC1 index is closer to 0.66 at low flow, the landscape is considered more clumped at low flow than high flow (table 4). The analysis of diversity in mesohabitats show what is natural to expect in a regulated river with strongly reduced flow; a degradation in diversity.

When class C is removed from the analysis (figure 9), it is a lot more clear that high flow conditions provide more diverse habitat conditions (table 5).

5.2 *Physical conditions in mesohabitats*

Measurements of depths and velocities were made to verify the mesohabitat classification. We can expect most measurements points to fulfil the required definitions of each mesohabitat class (table 2), but we can also expect some measurements not to fulfil these requirements due to the scale and size of the geomorphologic units forming the mesohabitat classes. Figure 10 and 11 show some examples of point measurements in the 34 mesohabitat units used for verification. In total, 6 % of the depth measurements and 0,3 % of the velocity measurements were outside the range of the mesohabitat at low flow. At high flow, 16,2 % of the depth measurements and 19,6 % of the velocity measurements were outside the range of the mesohabitat. 20,1 % of the depth measurements and 7,0 % of the velocity measurements were outside the range of the mesohabitat when measurements in mesohabitat class C was left out of the analysis at high flow. The results indicate that the classification of mesohabitats at low flow is more consistent and reliable than at high flow. However, the sampling method may also play a role. At low flow, only wadeable sections of the river were sampled. At high flow both wadeable and non-wadeable sections were sampled. At low flow, the sampling within a mesohabitat was always random using two randomly chosen numbers to give direction and distance to the next measurement point. However, at high flow the sampling was done in non-random transects due to navigation requirements. These factors may play an important role when comparing measured values with optic estimates (classification). At high flow, the diversity of habitats is higher and hence also maybe the diversity within mesohabitats, giving more measurements of depth and velocity outside the range of the mesohabitat.

In large rivers like the Rhône River, there will naturally be a large variation among mesohabitats of class C as long as the depth may vary from 0 to 10 m. The Norwegian mesohabitat method is not developed to assess large rivers. If the analyses of deep and slow habitats are important, we recommend using another method. In our study, we may run a separate investigation of large pool habitats. Since mesohabitat class C does not change much with the flow, this is not a field of great importance when studying the change in habitat.

5.3 *Shear stress and invertebrates*

At low flow the proportion of mesohabitat D is considerable, i.e. with FST values in the range of 3-7 with an average of 4,5. This kind of mesohabitat is favourable to species like the Ephemeropteran taxa of the *Caenis* genus or the Trichopteran taxa *Hydroptila sp. Polycentropus flavomaculatus, Psychomyia pusilla*. All these taxa were found in those hydraulic conditions in summer with a discharge of 20 m^3/s.

At high flow, this kind of mesohabitat is almost not present. We assume that similar FST numbers can be found in mesohabitat C, which is the mesohabitat covering the largest area at both flows (72 % at 10-20 m^3/s and 74 % at 50-70 m^3/s). However, even if mesohabitat C has the same range of FST numbers as mesohabitat D, the water depth is much larger. Benthic invertebrates usually do not colonise these foodless habitats and we can not expect that these habitats will compensate the decreased area of mesohabitat D at high flow.

A wide range of FST numbers (3-15) can be found in mesohabitat B2, but the most frequent FST numbers are between 6 and 12 with an average of 9. These values of FST correspond well to the requirements of taxa like *Hydropsyche sp., Heptagenia sulphurea* and species of the *Gammarus* genus both in spring and summer and *Potamopyrgus antipodarum* (Gasteropod) in spring only. At low flow, B2 represent only 4 % of the area, but is increased to 10 % of the total area at high flow, favourising these taxa.

FST numbers in mesohabitat E and F are generally higher than in other mesohabitats, and our measurements in F confirm this with a range of 8-15 and an average of 12. We have no measurement in mesohabitat E, but we assume that the FST numbers are comparable. Species like *Hydropsyche contubernalis, Hydropsyche exocellata* and *Ancylus fluviatilis* in spring and *Baetis lutheri Rhyacophila s. stricto sp*. both in spring and in summer are found in large numbers at FST around 12. As the total area of mesohabitats E and F are going to double from low to high flow, we assume that this will favourise these taxa.

In general, there is a clear shift in mesohabitats from slow and shallow habitats with FST numbers below 7 to faster habitats with FST numbers around 9 and 12. We realise that the number of FST sampling sites is low and also not representative for the whole range of mesohabitats and flows. This should be investigated more in details in the future. Doing FST measurements is quite time-consuming, so we hope to find other and less time-consuming ways of linking invertebrate abundance to mesohabitat classes.

5.4 General comments

The diversity of mesohabitats will increase when flow is increased from 10 (20) m^3/s to 70 (50) m^3/s and generally lead to a more diverse ecosystem. The increased environmental flow will lead to an increase in areas with mesohabitat B1 and G1, and this will favour species preferring rapid and deep lotic environments. Areas with mesohabitat D will almost disappear, leaving lentic species or taxa preferring slow and shallow environments with little preferred habitat. However, due to the special scale of the mesohabitat units, we may find large amount of D type habitats along the shore and edges of many other habitats, probably leaving a significant area of shallow and slow habitats also at high flow. These changes will be reflected in the invertebrate (see chapter 5.3) and fish (see Olivier et al 2005) composition. The future bypass channel at Chautagne will then probably obtain an ecology a little bit closer to the natural conditions of the Rhône River, with increased populations of rheophilic species (see Olivier et al 2005 for details on fish population).

5.5 *Further use and future developments*

The study will be continued during the next years in order to measure the availability of different mesohabitats, especially in correspondence with the evolution of invertebrate and fish population structure. At Chautagne, we have a unique possibility to compare predictions based on model results with the actual development of the invertebrate and fish population.

Based on the promising results and possibilities to conclude in this paper and in Olivier et al (2005), we believe that the Norwegian mesohabitat method may be a useful tool to compare different flow situations and their impact on the invertebrate and fish population structure even in large rivers with not only salmonid species. In fact, we propose the method being more applicable to river systems with a varied ecology of species requiring very different habitat conditions. We also think the method may be used more precisely if sampling of fish and invertebrates can be closely related to mesohabitats. However, we do not have solid evidence of the ecological significance of mesohabitat classes, while it is not proved that fish or invertebrates will distinguish between the classes.

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Not just the structure: leaf (CPOM) retention as a simple, stream-function-oriented method for assessing headwater stream mesohabitats and their restoration success

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ABSTRACT: Traditionally, when assessing stream habitats the focus has been placed on structural endpoints, such as flow and depth fields, or species diversity and other community attributes. The evaluation of ecosystem processes, i.e. functional endpoints, has got less attention. Here we forwarded a hypothesis whether the leaf retention could be used as a simple, stream-function-oriented method for assessing mesohabitat quality and restoration success. The biotic communities of streams in temperate zone forested areas are highly dependent on organic material, such as the leaf fall from the riparian trees in autumn. Consequently, a high retentive capacity of a stream could indicate beneficial conditions for benthic organisms, which in turn would propagate up in stream food webs. Our experimental results verified the view that complex stream bed structure indicates high leaf retention, and vice versa. Considering the elemental importance of leaf litter to forest streams the retention rate seems to be a good candidate for a simple stream-function-oriented tool for assessing the success of rehabilitation of streams with reduced bed heterogeneity.

1 INTRODUCTION

Traditionally, assessments of habitat quality in streams have focused mostly on fish, particularly habitat quality for and abundance of salmonids or some other indicator species (e.g. Bovee 1982, Huusko & Yrjänä 1997, Pretty et al. 2003). However, considering the extreme variation between stream sites, e.g. the monitoring programs of stream restoration success are unlikely to detect changes in these kinds of single structural elements unless the differences between reference and treatment sites are huge (Brooks et al. 2002, Muotka & Laasonen 2002, Pretty et al. 2003). Instead of monitoring structural elements, Brooks et al. (2002) advocated the use of ecosystem-level measures as indicators of stream quality at local scales. Of course, not all ecological processes can be studied efficiently, thus suitable instruments should be selected on the basis of the characteristics of the system to be assessed, and the practicability of quantification.

In temperate zone forest streams the leaf fall from riparian trees is often the primary energy source. Many species of detritivorous benthic animals in streams are highly reliant on inputs of leaf litter for their nutritional requirements. The large amount of alloctonous input and its retention have been shown to play an important role in structuring stream communities (Vannote et al. 1980, Cummins et al. 1989, Richardson 1991), often dominated by organisms adapted to shred leaf litter. However, the concentration of this resource varies considerably seasonally, the major input of alloctonous detritus to small boreal streams occurring during the autumnal leaf fall. Several recent studies have indicated that invertebrate populations, especially shredders, are frequently resource limited under seasonal conditions (e.g. Richardson 1991). On the other hand, coarse particulate organic matter (CPOM) decomposition often correlate positively with shedder densities (e.g. Webster & Benfield 1986). Dodson & Hildrew (1992) and Dodson et al. (1995) showed by their litter retention manipulations in certain English streams that animals were responding primarily to food supply rather than by using extra organic debris as habitat or by reacting to any hydraulic changes associated with enhanced retention. It is proposed therefore that increasing litter retention in streams may serve to increase their invertebrate productivity, resulting also in increases in productivity of fish populations (Elliot 1986). The enhancement of retention is thus very important to the question of stream enhancement. In all, because of its elemental role in the stream food webs, CPOM retention could be an obvious candidate for a measure of a functional end-point of stream management (Brooks et al. 2002).

Here we put forward a hypothesis whether the litter (CPOM) retention could be used as a simple method for evaluation of the effects of habitat enhancement measures. Good litter retention would indicate beneficial effects on the resource potential in different trophic levels, resulting in increased productivity and community diversity. In this paper we present results from artificial leaf-litter retention experiments conducted in laboratory flumes with different physical nature. We also give results from the artificial leaf-litter retention experiments carried out in a forest stream in order to standardize a practice for field studies.

2 MATERIAL AND METHODS

2.1 Laboratory experiments

The experiments were carried out in four indoor artificial flumes located in the Finnish Game and Fisheries Research Institute, Kainuu Fisheries Research and Aquaculture Station, Paltamo, Finland. The flumes used were made of fiberglass, and they were 6 m long, 37 cm wide at the bottom and 40 cm wide at the top of the walls. The water was lead to flumes from two large tanks (2000 l), one tank delivering into two flumes. During the experiment the tanks were continuously filled by two submersible pumps from the main water storage of the aquaculture station. The bed structure set out at the flumes was mimicking a simple channelized (two flumes) and a complex enhanced (two flumes) stream bed differing from each other mainly by the amount of stones present. A wire mesh (5mm mesh size) closed the flume at the down-end.

Retention was estimated as a percentage of artificial leaves retained out of a known number of leaves released, i.e. the percentage of leaves that did not reach the wire mesh, was used as the index of retentiveness (Speaker et al. 1984, Lamberti & Gregory 1996). We used 5*6 cm sized plastic leaves in the experiment. These leaves were approximately the same size as leaves of many common riparian trees and bushes (e.g. *Betula* sp., *Populus* sp., *Alnus* sp., *Salix* sp.) in northern Finland. By a pilot study we considered that also the buoyancy of the artificial leaves used was approximately the same as that of natural leaves of *Alnus* sp and *Betula* sp. The experiment trials were carried out at three different flows (2.7, 12.4, 18.3 ls^{-1}) mimicking a continuum from a low to a high flow, and three slopes (0.0, 1.8, 3.7 %) in both stream bed structures. Each trial was repeated six times. In each trial, 25 artificial leaves were released in each flume. The leaves were spotted under the water surface and were spread evenly across the flume width. The time interval between the release and the retrieval of the drifted leaves from the down-end wire mesh was five minutes.

After the experiment runs depth and velocity in the flumes at all slopes and flows were measured by a measurement rod and Schiltknecht MiniAir 2 Flowtherm with a propeller size of 20 mm. The flume was divided into 5x10 cm cells, and water depth and velocity was measured following a stratified random sampling design with each 10 cm longitudinal strata having one randomly chosen measurement cell across the flume width. Velocity was measured from the centre of each cell at 2 cm above the bottom. A bed profiler (Ziser 1985) was used to obtain a trace of the bed profile in the flow direction. Three parallel profiles were measured along the whole length of both flume bed types. The principal variables to describe the flow in the flumes were calculated from the measured variables by using the formulas presented in Davis & Barmuta (1989). Mean roughness height, which was taken as two times the mean of the standard deviations of the height of the roughness elements in three rows measured separately, was in channelized flumes 5.36 cm and in the rehabilitated flumes 7.31 cm. The roughness density, which relates the plan area of roughness elements to the total area of the flume bed, was 7.9 % in channelized and 36.7 % in restored flumes.

2.2 *Field experiments*

To develop and standardize a methodology for to examine the retention capacity of a stream, and to apply it to estimation of rehabilitation success of dredged streams in the field, experiments were carried out in two wadable reaches of a typical forest stream (River Alajoki, mean annual flow 1.0 m^3s^{-1}, during the study 0.4 m^3s^{-1}) located near the above mentioned research station. The study reaches consisted of riffles and runs with some small pool-spots. The same 5*6 cm sized plastic leaves were used in these experiments as in the laboratory tests. A 15 mm mesh size wire net was secured across each study site to block off the leaf transport. To find out a reasonable number of artificial leaves to be released in a trial, 100, 200, 300, and 600 leaves were released at a time into a stream reach, and the leaf-catches at the downstream block net were compared. The leaf-catches at the net were controlled by every fifteen minutes up to 210 minutes to figure out the point when the leaf drifting was stabilized. These tests (with four replicates) were carried out in stream reaches of different lengths (30 m and 50 m) to find out a good compromise for the length of the study arena.

3 RESULTS

3.1 Laboratory experiments

Flow, slope and bed structure (i.e. roughness height and roughness density) all affected leaf retention significantly (Table 1). Litter retention was regulated by the arrangement and abundance of bed roughness elements, interacting with flow and slope. The effect of flow and slope on retention was closely linked to water depth, because the probability of a leaf coming into contact with the bed elements decreased with increasing depth. Simplified flumes were about half as retentive as enhanced ones. There was a clear decline in the retention rate in the channelized flumes as the flow and slope increased, in contrast to cobble-dominated enhanced flumes, which tended to remain retentive with increased flow and slope (Figure 1). Local abundance or absence of particular retentive structures and their roughness height was estimated to be more important in determining retention than flow or slope alone. The local fine-scale distributions of leaves retained were determined by the location of specific features of bed roughness elements.

Table 1. Three-way ANOVA table for the effects of channel slope (S), flow (F) and bed structure (B) on the retention of artificial leaves in the experimental flumes.

Source of variation	SS	df	MS	F	p
S	1344.3	2	672.2	116.6	<0.001
F	2089.5	2	1044.8	181.3	<0.001
B	1379.6	1	1379.6	239.4	<0.001
S + F	278.8	4	69.9	12.1	<0.001
S + B	36.4	2	18.2	3.2	<0.05
F + B	66.1	2	33.1	5.7	<0.01
S + F + B	92.3	4	23.1	4.0	<0.01
Error	518.7	90	5.8		

3.2 Field experiments

The field experiments revealed that the best practicality of quantification the stream retentiveness was to conduct the test in a 30 m long study arena (in streams not wider than 15 m) with 300 leaves allowed the drift two hours in a trial (Figure 2). Smaller number of leaves released gave often 100 % retentiveness estimates although experiments with large number of leaves showed clear variation for the same stream reach. Accumulation of leaves to block net ceased almost totally after two hours, yielding < 5% of the total in all the test runs. As shown by laboratory experiment, obviously the greatest differences between low and high retentiveness of streams can be found at low or mean discharge. Thus, it is recommended to carry out tests below the mean flow of a stream. However, to get the full picture of the retention capacity of a stream its better to run the experiments e.g. at three discharges spanning from low to high ones, with replicate test runs at each case.

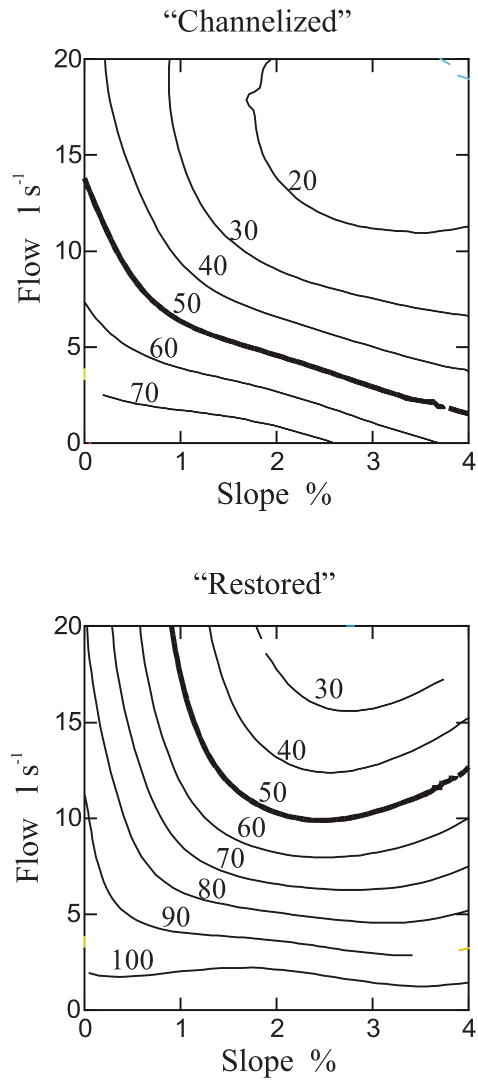

Figure 1. Retentiveness contours (in % terms) of artificial leaves by slope and flow in the two channel bed structures.

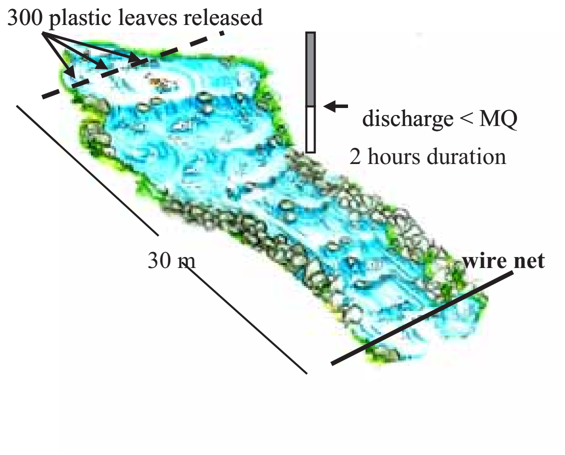

Figure 2. An example of the basic measures of a simple field-test for to quantify the retention rate of a stream section.

4 DISCUSSION

In this article we have given mainly methodological aspects for the examination of leaf retention in headwater streams. The experimental results verified the view that complex substrate structure in the stream bed indicates high leaf retention, and vice versa. Preliminary results from our ongoing field studies have given supportive evidence for the results of these experimental studies, in addition to those reported in literature (e.g. Speaker et al. 1984, Petersen & Petersen 1991, Muotka & Laasonen 2002, Lepori et al. 2005). As the effects of CPOM manipulation are known to propagate up in stream food webs (Richardson 1991, Dodson & Hildrew 1992, Dodson et al. 1995), management measures that enhance the availability of organic material to benthic animals may also benefit e.g. salmonid fish populations.

The results support the use of retention rate as a simple stream-function-oriented tool for assessing the success of stream rehabilitation and management. A marked advantage of the leaf release approach is that the results are achievable at relatively short time and with low costs. In Finland we have applied this methodology to assess the impact of restoration of boreal forest streams channelized for timber floating (Huusko et al. (unpublished), Muotka & Laasonen 2002), but, considering the importance of leaf retention on stream functioning, the methodology should be applicable for example to assess enhancements in a wide range of streams with reduced bed roughness heterogeneity. The standardized practice for field studies presented above is well adapted for the northern forest streams, but most probably it should be modified to commensurate with the scale of the habitat to be investigated as well with the number of leaves released when applied to elsewhere.

Assessing the ecological effects of stream management efforts by any methodology entails two basic comparisons (Barmuta 2002). Firstly, was the monitored variable positively affected by management efforts, i.e. did it produce any ecological change? To answer this managed sites should be compared with their impacted conditions. On the other hand, managed sites should also be gauged against a specified guiding image of a healthy stream within the same watershed used as a target for management actions. CPOM retentiveness tests make no exception to these.

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Links between hydrological regime and ecology: The example of upstream migrating salmonids

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ABSTRACT: There is now a pressing need for models, methods and systems for setting instream flow requirements that protect the integrity of the biota and ecosystem functions that inhabit running waters. Most of the techniques currently used are simplistic and reductionist in their approach. Frequently they concentrate on one or a few key species and ignore other parts of the ecosystem that those key species frequently also depend on. Current methods have tended to concentrate on physical habitat modelling which is only one variable in the myriad of interactions that flow has with other important factors such as chemistry and temperature. This review makes use of literature on migration flows for upstream migrating salmonids where there is a good deal of information to demonstrate how different elements of the flow regime, timing, duration, frequency and magnitude impact on migration rates and highlights some of the problems of linking hydrological regime to ecological impact

1 INTRODUCTION

There is now a pressing need for models, methods and systems for setting instream flow requirements that protect the integrity of the biota and ecosystem functions that inhabit running waters. Most of the techniques currently used are simplistic and reductionist in their approach. Frequently they concentrate on one or a few key species and ignore other parts of the ecosystem that those key species frequently also depend on. Current methods have tended to concentrate on physical habitat modelling which is only one variable in the myriad of interactions that flow has with other important factors such as chemistry, temperature and substrate movement etc.

Elements of flow regime (timing, frequency, magnitude, duration and rate of change) are frequently ignored in these models even though such hydrological variability is important for ecosystem integrity (Poff & Ward, 1989) and some have suggested that where conservation are objectives of river management, then targets must reflect natural flow variability (Richter et al 1996). If this is a practical way of managing flow regimes the question then remains as to how close to the natural flow regime do flow targets need to be set. Richter (1997) has proposed a 'Range of Variability Approach' (RVA) that uses 32 hydrological parameters to describe the natural flow regime and that initial flow management targets should not vary outside 1 standard deviation from the mean or the 25th to 75th percentile of each of these parameters.

Whilst the RVA provides a mechanism for approximating the natural flow regime statistically, the biological implications of such variations are less clear and are likely to vary amongst the hydrological parameters. So, for example, a 1 SE deviation from mean date of a flood event (timing) may have more impact on facilitating upstream migration of adult salmon than a 1 SE deviation in the number of high pulses every year (frequency and duration).

This paper describes the scientific literature and data from a number of UK rivers on the relationship between upstream salmon migration and flow regime to determine what type of information is available on the relationship between flow and organism or ecosystem response.

2 ADULT SALMON MIGRATION.

Early papers emphasise the importance of freshets for moving salmon from the estuary into the river (Huntsman,1948). Baxter (1961) says to induce migration of fish into a stream with only residual flow there is a necessity to provide flows in the form of freshets and many authors (Calderwood, 1908; Hayes, 1953; Harriman 1961) report the use of these in initiating migration from the estuary. However, freshets require consideration of:

their timing,
the amounts of water to be released,
the frequency of their release and
their duration.

For example Harriman (1961) concluded that freshets before 15 May did not start an early run of salmon but freshets after that would. And Baxter (1961) says that the amounts of water required for a freshet will vary with the size of the stream because of the relationship between a.d.f. and stream width.

Hayes (1953) working on the Le Have River, Nova Scotia found that under natural conditions the river fell to 50 $ft^3 sec^{-1}$ but found that 200 $ft^3 sec^{-1}$ was needed to maintain a good run of fish. His general aim was to keep the river running at 400 $ft^3 sec^{-1}$ for as long as possible and add freshets up to 1600 $ft^3 sec^{-1}$ at intervals. After three years the author concluded that:
large or small freshets are capable of moving fish,

major runs can occur without the need for natural or artificial freshets and can be maintained by a steady flow of water during the run season,
artificial freshets moved fish into the river from the head of the tide but were not sufficient to bring fish into the estuary,
the reverse of a freshet which was to reduce the water and then increase it again could act like a freshet in bringing fish into the river and
some freshets had no effect and it was concluded that there had to be a supply of salmon in the estuary if freshets were to work.

Hayes (1953) proposed a plan for water control based on his observations which included:

maintaining flow at 400 $ft^3 sec^{-1}$ for as long as possible, allowing natural freshets to take their course,
not wasting water on large numbers of freshets,

timing positive freshets to reach the head of tide at dusk during periods of spring tides and onshore winds and

using inverse freshets when water resources are very low.

Baxter (1961) has addressed the issues of how much, how often and for how long. He says that the amounts of water required for a freshet will vary with the size of the stream because of the relationship between a.d.f. and stream width. However, his estimation was that salmon would ascend most rivers in flows varying from 30-50% of a.d.f. in the lower and middle reaches of rivers and at 70% of a.d.f. in the upper reaches. This was for rivers with gradients of 1:90 to 1:300 in upper reaches and 1:400 to 1:700 in middle to lower reaches. Rivers outside this range may require differing amounts of flow which can be estimated from observation. However, spring fish (February - April) or the larger salmon that run in the early part of the year require more water than summer run fish and it is estimated they require 50 - 70% of a.d.f. (Baxter, 1961). This was attributed to the effect of the lower temperatures on fish activity.

There are potentially some important indirect considerations of the flow requirements for fish as changes in flow rate will undoubtedly impact on the concentration of oxygen and toxic pollutants as well as having an indirect effect on temperature. This is an important consideration in the setting of flows and other authors (Alabaster, 1990; Milner, 1989) have suggested that temperature plays an important part in determining the tendency of salmon to migrate. On the River Dee the numbers of salmon migrating reduced considerably as mean weekly maximum temperatures reached 21.5°C. Similarly the numbers of grilse migrating in the Miramichi River declined at high temperatures (Alabaster *et al*., 1971). Inhibition of migration is thought to occur at temperatures lower then 5°C (Milner, 1989). Jensen, Heggberget & Johnsen (1986) tried to correlate a variety of physical and environmental variables with the numbers of salmon passing through a fish pass between June and September. They found that in addition to flow, temperature showed a significant relationship with the number of migrants.

The behaviour of salmon in relation to flow and its interactions with other environmental factors will be dependent on the date they enter the river and the position in the river system in which they eventually spawn (Webb, 1992). At times of low flow fish accumulate in estuaries. It is not known how acclimatisation of river flows and the length of time spent in the estuary effects the migration into fresh water (Milner, 1989). And the frequency of the freshets will depend on the positioning of tributaries, the distance between the source of the freshet and the estuary and the location of the spawning grounds (Baxter, 1961). Where the residual flow is not supplemented by natural flow from tributaries it may be necessary to have frequent freshets to bring salmon into the river at regular intervals, or where the residual flow only affects spawning grounds they may only be needed later in the year to bring spawning salmon into the tributary.

The duration of freshets may be important. Some authors report that salmon migrate on the falling portion of a freshet (Trepanier et al., 1996; Huntsman, 1948; Huntsman, 1939, Hayes, 1953, Brayshaw, 1967, Stewart 1968, Dunkley & Shearer, 1982) and the duration of this stage will determine the length of time and therefore the distance that the salmon migrate. Baxter (1961) suggests they need not be more than 18 hours, 12 of which should be at the full rate.

A number of authors have tried to measure the flows that salmon use for migration. For example on the River Avon Brayshaw (1967) observed peaks of migrating salmon appeared to occur at 75 - 100% of a.d.f. Allan (1966) described the salmon counts at a counting fence on the River Axe and the frequency at various flows. Most salmon ascended through the trap at flows between 28 and 243 $ft^3 sec^{-1}$. Banks (1969) used Allan's data to calculate a figure for the number of salmon per flow day and found that the peak of this value occurred at between 147 and 267 $ft^3 sec^{-1}$. This data is difficult to relate to a.d.f. for this river but presumably it could be done.

Stewart (1966) describes results for the River's Lune and Leven where fish moving upstream are automatically recorded. He found on the river Lune that 80% of all salmon move upstream on flows equivalent to 190% of a.d.f. or 460 $ft^3 sec^{-1}$. A peak of movement occurs at 400 $ft^3 sec^{-1}$ (equivalent to 167% of a.d.f.) and a lesser peak occurs at 175 $ft^3 sec^{-1}$ (equivalent to 73% of a.d.f.). These peaks correspond to movements of spring and summer fish. On the River Leven the peak of salmon migration occurred at a flow equivalent to 86% of a.d.f. However comparisons between the two rivers were difficult because the counter was 26 miles up river on the Lune and at the head of the tide on the Leven which was also partly regulated by Lake Windermere. Swain & Champion (1968) found on the River Axe that appreciable numbers of salmon utilised flows less than 65% of a.d.f. for migration but this occurred at night. Cragg-Hine (1985) observed that over several years during the summer months fish tended to utilise the higher flows but the migration flow range varied from year to year depending on prevailing flow conditions. Studies of fish counts over a counter on the River Lune between 1974 and 1979 showed very great variation in the flow utilised for migration from year to year depending on the flows that were available. In general fish utilised flows that were slightly higher than those available but in 1979 when the water level was consistently higher the mean migration flow was less than the mean available flow. The summer migration flow range was approximately 10 - 82% of a.d.f. for that river.

This observation of fluctuating relationships between flow and salmon movement between years was noted by Smith (1991) who stated that it would alter depending on prevailing flow conditions. On the Aberdeenshire Dee during a low flow year in 1989 the flows used for entry to the river were much lower than in previous years. A similar observation was made by Sambrook & Broad (1989) on the River Tamar where salmon were observed to enter the river at low flows.

3 CONCLUSIONS

Elements of timing, magnitude, frequency and duration of flows and interactions with other factors are important when considering ecologically sensitive flows for upstream migrating salmonids.

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Conceptual framework for assessment of ecosystem losses due to reservoir operations

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ABSTRACT: Worldwide, dams and reservoirs impart change in ecosystem function through simple presence in the landscape and subsequently through operational modification of river and floodplain processes. Operational modification is primarily manifested through discontinuity in downstream gradients and departure from the natural flow regime. A physical process-based conceptual framework is described that enables assessment of operational losses to ecosystem function in river floodplain systems. The framework utilizes combined empirical analyses and numerical simulation to compare pre-reservoir and post-reservoir conditions through linkage of available habitats to the corresponding flow regimes and to the fluvial processes present at those locations. The framework results in a suite of parameters that facilitate linkage of the assessment to relevant investigations of ecological and biological processes. The framework may be utilized to quantify functional losses at existing or proposed facilities, to identify and evaluate restoration potential, and to optimize facility operation. Examples from the Kootenai River, USA and Canada are given.

1 INTRODUCTION

1.1 Background

Human civilization has utilized the world's rivers for thousands of years, reaping the benefits of irrigation, transportation, flood control, power generation and recreation. As a result of our sustained development of these resources, river systems have been altered significantly worldwide, with nearly 60% of major river basins fragmented by large dams (Revenga et al, 2000). In the continental United States, 75,000 dams contain storage volume nearly equalling one year's mean runoff (Graf, 1999).

Dams and reservoirs impart change in the environment simply through their presence in the landscape, with the magnitude of change unique to each facility. Placement of a large dam can resemble an intervention approaching geologic significance, while small facilities may have little influence on the surrounding environment.

Dams and reservoirs may also impart change in the environment through facility operations over a sustained period. Operational impacts develop through discontinuity in downstream gradients (eg. sediment supply, water quality) and modification of the natural flow regime. These impacts lead to secondary changes in fluvial and floodplain processes, affecting the high spatial and temporal variability of available habitats characteristic of river floodplain systems (Richter et al, 1996; Poff et al, 1997).

1.2 *Need for Operational Losses Assessment Methodology*

A significant volume of research concerning the ecological function of regulated rivers has been conducted over the past two decades (eg. Stanford et al, 1988; Ward & Stanford, 1995; Ligon et al, 1995; Richter et al, 1996, Poff et al, 1997). A robust, systematic methodology aimed at quantifying operational impacts to river floodplain physical processes will provide a tool to aid in understanding the relationship between facility operations and ecosystem health in an effort to lessen impacts. This tool will also contribute to efforts to assess operations-based losses that have occurred to date. The systematic approach is applicable to river systems in general that have altered flow regimes and floodplains and therefore for a majority of rivers affected by the European Water Framework Directive. Restoring the ecological integrity of a disturbed system requires at least a partial restoration of the physical processes that drive the ecological functions. In the pilot study presented here we try to establish the framework to analyze and quantify the losses caused by dam operations and differentiate them from losses due to other human activities. Once the framework is established, its individual modelling tools can be selected fitting to a specific situation.

2 CONCEPTUAL FRAMEWORK

2.1 *Framework Requirements*

Several elements are necessary to ensure framework utility and enable application to a range of reservoirs with a variety of operational objectives. These include facility to isolate operational impacts from other basin perturbations, means to assess the manifestation of operational impacts on downstream physical function, and subsequent linkage to biological and ecological processes. Morphological adaptability enhances framework transferability between river basins. Lastly, predictive capacity allows evaluation of proposed restoration actions.

2.1 *General Framework Algorithm*

The general framework algorithm is represented schematically in Figure 1. The algorithm flows through five modules to develop function analyses for various reference scenarios. Paired comparison of these function analyses allows segregation of particular reservoir operational impacts. The five modules are described here.

2.3 *Contemporary Basin History Module*

The basin history input module involves description of recent basin factors, modifications and management actions not attributable to reservoir operation that may have impacted ecosystem function in the study area. Primary components include climate change, and land and resource management actions. These history elements are arranged to develop reference basin scenarios with particular characteristics. Reservoir operations combine with these scenarios to produce the physical river floodplain environment.

2.4 Reservoir Operations Module

Reservoir operations may impact river floodplain function though hydrologic alteration and through discontinuity in downstream gradients. The degree of impact varies greatly between reservoirs. The reservoir operations module involves description of the hydrologic regime and discontinuity environment characteristic of distinct periods of interest.

2.5 Modeling Scenario Definition Module

Basin history and reservoir operation scenarios are combined into reference modeling scenarios. Separate scenarios might be developed for pristine, immediate pre-dam, immediate post-dam, contemporary and other conditions. These reference modeling scenarios become the building blocks for the paired comparisons that allow isolation of the impacts of particular operational strategies, or allow evaluation of the potential benefit provided by a proposed restoration action.

2.6 Function Analysis Module

The physical function of each reference modeling scenario is analyzed through empirical methods and numerical simulation. The results of the function analysis are distilled into a suite of parameters that quantify and describe the physical river floodplain function under a particular reference scenario.

2.7 Scenario Comparison Module

The last step in the process of isolating and quantifying the physical impact of a particular reservoir operation strategy is comparison of the function analysis results for paired reference scenarios. The paired reference scenarios are typically arranged to examine how an operational strategy may have in turn impacted biological processes of particular significance for the river floodplain system under study. This step produces a results matrix, summarizing the operational effects and providing input for subsequent ecological or biological modeling.

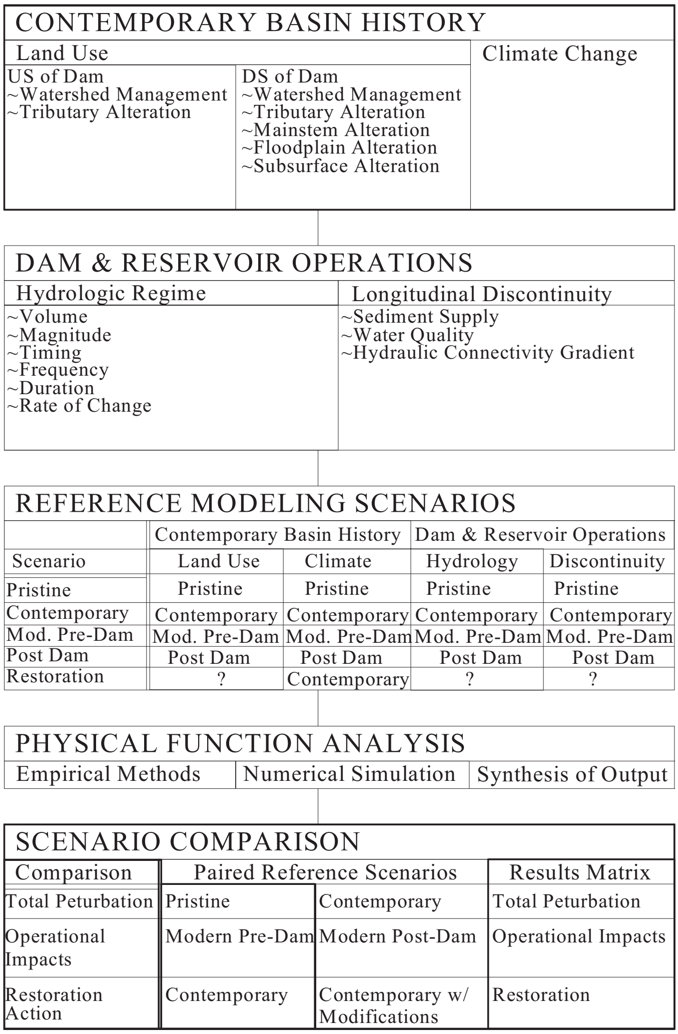

Figure 1. General conceptual framework for assessing reservoir operational impacts on downstream river floodplain physical processes.

3 FRAMEWORK EXAMPLE APPLICATION

3.1 Kootenai Basin Description

The Kootenai Basin (Figure 2) is an international watershed (spelled Kootenay in Canada, Kootenai in the U.S., the latter is used here with the exception of place names) originating in the northern Rocky Mountains of eastern British Columbia, Canada. The Kootenai River flows south into Libby Reservoir whose 145-km length straddles the Canada-USA border (Hoffman, et al. 2002). From the southern terminus of the reservoir at Libby Dam near Libby, Montana, the river changes course to the west and then to the northwest near the border between the U.S. states of Montana and Idaho. From the town of Bonners Ferry, Idaho, the Kootenai flows generally north again crossing the international boundary before emptying into Kootenay Lake. Located 357.1 river kilometers above the mouth, Libby Dam and Reservoir is a 130-m high flood control and hydropower production facility that impounds flows originating in the upper 23,300 sq. km. of the 50,000 sq. km. watershed. Kootenay Lake is a naturally formed lake whose levels have been regulated since the construction of Corra Linn Dam near Nelson, British Columbia in the 1930s (USACE, 1984).

Libby Dam and reservoir was completed in 1973. Several adverse impacts have been attributed to the facility, including negligible recruitment of native fisheries (Hoffman et al, 2002) and limited regeneration of riparian forest (Polzin & Rood, 2000). An assessment of ecosystem losses due to Libby Dam and Reservoir operations is currently being led by the Kootenai Tribe of Idaho with the cooperation of several government and non-government entities.

3.2 Contemporary Basin History

The Kootenai Basin has been intensively managed starting in the late 1800s. Perturbations (w/ timeframe) considered most pertinent to the Kootenai River / Libby Dam assessment include: floodplain diking, drainage and conversion to agriculture over approximately 21,000 hectares between Bonners Ferry and Kootenay Lake (1900-1940s), channel dredging coincident with floodplain diking (1900-1940s), completion of Corra Linn Dam (1932), dredging of the natural Kootenay Lake outlet upstream of Corra Linn Dam to reduce outlet hydraulic control (1930s), treaty signed to increase winter levels and decrease spring levels in Kootenay Lake, impacting natural backwater profiles to Bonners Ferry, ID (1938), and completion of Libby Dam (1973) (dates of events from Tetratech, 2004).

Comparison of time series from unregulated gaging stations (Environment Canada #08NG6005 Kootenay River at Wardner, BC & #08NG6065 Kootenay River at Fort Steele, BC; USGS #12302055 Fisher River near Libby, MT) for the periods pre-dating and following the construction of Libby Dam show stable trends in climatic conditions.

3.3 Reservoir Operations

The natural flow regime for the Kootenai River was characterized as ‘Rocky Mountain snowmelt dominated’ (Hoffman et al, 2002), resulting in a high, sustained peak in late spring, followed by a recession to base flow by September, and low winter flows. The hydrologic regime of the Kootenai River has shifted significantly since Libby Dam was completed (Figure 3). As the facility is operated for flood control and hydropower generation, spring peak flows are approximately half of historic levels while winter flows have more than doubled. Other hydrologic characteristics have also been affected (rate of change, frequency, duration). Note that a slightly modified experimental flow plan has been in place since 1993, largely focused towards recovery of the endangered white sturgeon *(Acipenser transmontanus).*

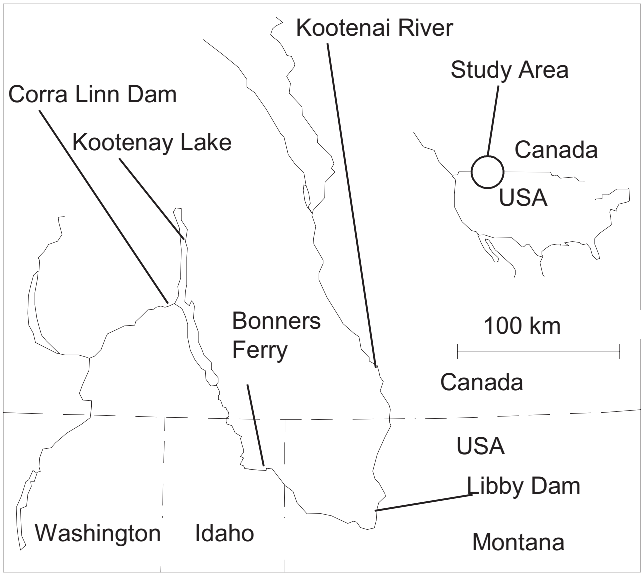

Figure 2. Location map showing Kootenai River Basin.

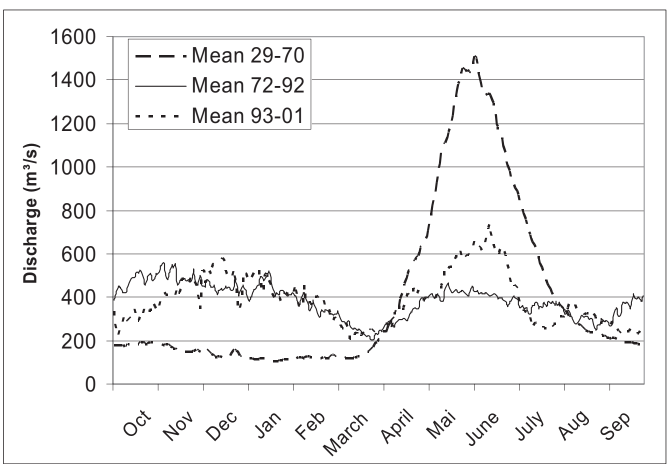

Figure 3. Timing and magnitude for Kootenai River flows at USGS gage # 12305000, Kootenai River at Leonia, adjacent to Idaho-Montana state boundary.

Libby Reservoir has also affected downstream gradients, trapping virtually all incoming sediment (Tetratech, 2004) and a majority of nutrients, stopping fish passage and resulting in slightly cooler (1-2 deg. C) summer and slightly warmer (3 deg. C) winter downstream water temperatures (Marotz et al, 2001).

3.4 Modeling Scenario Definitions

As can be understood from the previous paragraphs, several conditions each for the Kootenai River mainstem channel, floodplain, downstream backwater profile, and operational environment must be considered and segregated in order to isolate the downstream operational impacts of Libby Dam. These various basin history and operational elements have been arranged into the modeling scenarios summarized in Table 1.

Table 1. Kootenai Basin Reference Modeling Scenarios

	Contemporary Basin History		Dam & Reservoir Operations	
Scenario	Land Use	Climate	Hydrology	Discontinuity
Pristine	Pristine	Historic	Natural	Historic
Pre- Dam	floodplain: diked, drained & leveled main channel: dredged DS backwater: Per IJC 1938	Historic	Natural	Historic
Post-Dam (1975-1992)	floodplain: diked, drained & leveled main channel: dredged DS backwater: Per IJC 1938	Historic	Regulated (1975-92 regime)	sediment, nutrients: interrupted temperature: sel. withdrawal
Post-Dam (1993-present) /current	floodplain: diked, drained & leveled main channel: dredged DS backwater: Per IJC 1938	Historic	Regulated (1993-present regime)	sediment, nutrients: interrupted temperature: sel. withdrawal
Restoration Action	?	?	?	?

Table 2. Kootenai Basin Scenario Comparison

Comparison	Paired Reference Scenarios		Results Matrix
Total Peturbation	Pristine	Post-Dam (1993-?)	Total Perturbation
Facility Operation Impact (1975-92)	Pre-Dam	Post-Dam (1975-92)	Facility Operation . Impact (1975-92)
Facility Operation Impact (1993-?)	Pre-Dam	Post-Dam (1993-?)	Facility Operation . Impact (1993-?)
Restoration Impact	Post-Dam (1993-?)	Restoration Action	Restoration Impact

3.5 Function Analysis

Additional details and brief examples of the function analysis for the Kootenai Basin will be included in the full paper and presentation.

3.6 Comparison Evaluations

The reference modeling scenarios are then paired as shown in Table 2 to isolate various aspects of Kootenai Basin perturbation. Examples of the comparison evaluations will be included in the full paper and presentation.

4 CONCLUSIONS & OUTLOOK

Dams and reservoirs may impart change in the environment through their presence in the landscape and through facility operations over a sustained period. A systematic methodology to quantify operational impacts to river floodplain physical processes has been developed and is described briefly in this manuscript. This tool will aid in understanding the relationship between facility operations and ecosystem health and contribute to efforts to assess operations-based losses that have occurred to date.

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A Hierarchical Approach for Riparian and Floodplain Vegetation Modelling - Case Study "Johannesbrücke Lech"

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EXTENDED ABSTRACT: Up to recently, the ecohydraulic modelling community has focused mainly on aspects of fish habitat and fish population modelling. Only few attempts have focused on floodplain vegetation modelling. One of the reasons may be, that floodplain and riparian vegetation cannot be treated in the same way of quasi-static conditions as within the traditional IFIM approach, but has to be treated fully dynamically. However, as floodplain and riparian vegetation is a key element in each river restoration and nature-oriented flood protection measure, there is an increasing need throughout Europe to cope with this issue. Within COST Action 626, some members of the floodplain vegetation group therefore started to develop a new floodplain succession model.

This new model represents space in a hierarchical framework scaling up plant individuals to plant cohorts, to ecotops, to cross sections and to river segments. Necessary downscaling acts vice versa. The modelling concept is based on expert and literature knowledge, which helps to simulate the complex ecology and growth of selected key species (annual herbs, grasses, shrubs, trees) in a simplified way. To cope with the history of weather extremes, hydrology and disturbances, floodplain succession is routed through time based on a daily time steps (see Figure 1).

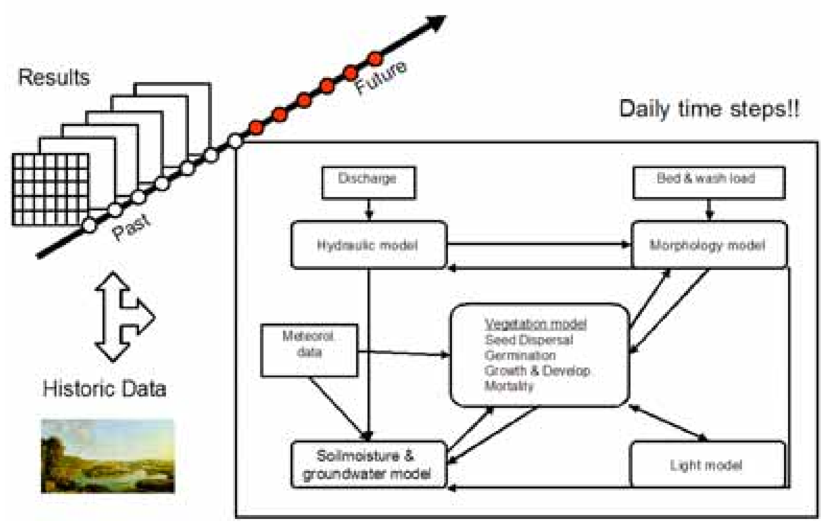

Figure 1: The modelling framework used for floodplain succession modelling is linking abiotic and biotic submodel

Species-related sprouting, growing and leaf canopy information as well as expert knowledge on abiotic tolerances are linked and interact with abiotic conditions (evapotranspiration, soil-moisture, hydraulics, morphodynamics, light extinction by canopy). Within each time step new abiotic and biotic conditions are simulated. In a self-organizing way new plants can be created each time step, compete for light, space and water, can grow, are partly damaged or have to die within a model run.

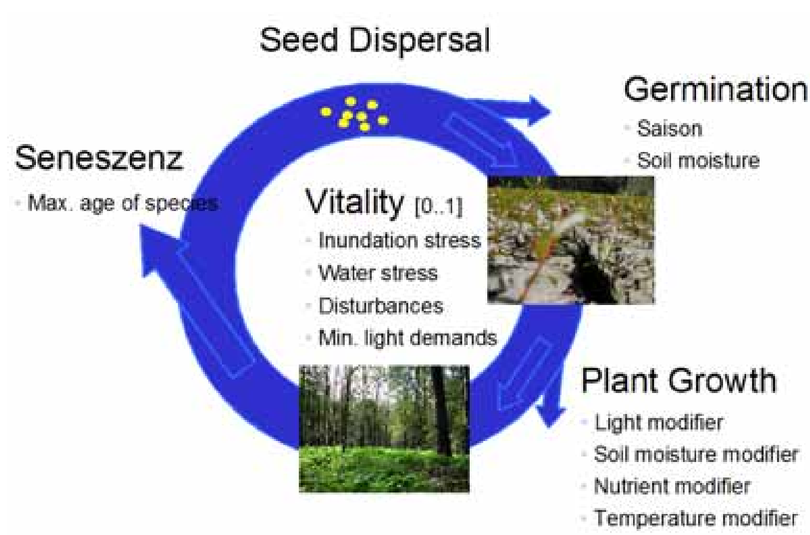

Figure 2: The simulated life-cycle of plant individuals. Expert and literature knowledge is used to parameterise the ecology of different key species.

Development and testing of the first model prototype was performed for the river Lech, Austria. A historic situation around 1850 and the actual situation is simulated and compared on different points along a cross section. Within this presentation we will focus on the overall concept and especially on the hierarchical modelling schema, we used for up- and downscaling. First modelling results for our study area "Johannesbrücke Lech" will be presented and discussed.

Flow variability and habitat selection of young Atlantic salmon. A case study from river Surna, Mid-Norway

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ABSTRACT: Habitat selection of Atlantic salmon in rivers partly depends on physical parameters like water velocity, water depth, substrate size and cover. Therefore the purpose of the study was to display how varying discharge will change habitat conditions and preference of habitat for young Atlantic salmon. The study site is situated in Mid-Norway at river Surna, where topography data and information on hydraulics for three different flows (2.1 m^3/s, 4.4 m^3/s and 6.5 m^3/s) were collected. The investigated part of the river, called Sandehølen, is about 350 m long and consists of different meso-scale habitat classes, including two pools. The collected data provided the input for the 3D hydraulic model SSIIM. In addition to the validated models, further discharges have been simulated. The results from SSIIM have been used in the HABITAT model in combination with general summer preferences for juvenile salmon in order to study the habitat conditions, focusing on pool conditions for different flows. The obtained information from HABITAT on preferred areas for Atlantic salmon was compared with collected fish data in different meso-scale habitats. The final results can be transferred from the micro-scale through meso-scale habitat classification to larger river reaches. All meso-scale habitat classes showed more avoidable area with higher discharge, due to rise of velocity.

1 INTRODUCTION

1.1 General information

In general, habitat models combine the information on living space so called "Habitat" with the preferences in habitat for selected species. The result is the Habitat suitability for the modeled part of the river. Above all with the implementation of the water framework directive in the European Union and in Norway this aspect plays an important role in the future, considering that until the year 2015 all freshwater resources shall be of an ecologically sustainable level of quality and quantity - effectively restored to so-called "good chemical status" and "good ecological status". In regulated rivers, good or maximum ecological potential are the main goal. Habitat models will therefore be an important contribution concerning the evaluation of the ecological status of freshwater and finding environmental flow targets to ensure healthy rivers.

1.2 *Objective of the study*

In this study, habitat modeling was used to show, how the variation of discharges during the year is likely to affect the aquatic ecosystem for young Atlantic salmon, for a river reach, influenced by flow regulation due to hydropower production.

In recent years river management has been based mainly on summer studies, which have been carried out in shallow fast flowing river reaches. The lack of winter and pool habitat studies shows, that the influence on variations in discharge and their effect on the aquatic ecosystem are still not totally understood. Therefore this study will include the investigation on two pools and the result from the habitat model will be compared with electro – fishing data. Atlantic salmon and especially young fish, have often been used as target species for habitat models, due to their economic value, high sensibility concerning environmental changes and flow variability. In this study we focus on condition for young Atlantic salmon, although during sampling of biotic data, brown trout has been considered too.

1.3 *Methods*

The study site was divided in different mesohabitat classes, based on the classification tree from Borsányi et al. (2002). With the help of this method, impacts on the varying habitat suitability due to change in flow can be transferred to a larger river scale. The study site Sandehølen at river Surna, has been modeled with the help of simulation tools. In this case the three dimensional hydraulic model SSIIM and the model Habitat, both developed at the NTNU, Trondheim, Norway, have been used.

Fish habitat modeling is detailed modeling of a large system. While the size of one fish is very small compared to the size of the river, it is necessary to study parts of the river system in detail (Harby, 1994). In general, only parts of the river system are studied in detail due to the fact that modeling of the whole river is too expensive and time consuming. The selected study sites must give a representative range of the river system in hydraulic and biologic way. Therefore the chosen study site included different meso-scale classes.

2 STUDY SITE

The study was carried out at river Surna, Mid-Norway. The total catchment area has the size of about 1200 km^2 and has mountain character. The yearly mean discharge constitutes 56 m^3/s, at the estuary. The river is influenced by hydropower production, which influences water discharge and temperature at 2/3 of the anadromous part of the river. Downstream the power plant outlet, the flow is altered from natural one and is characterized by higher discharge during winter, due to power production. In the by-pass section upstream the power plant outlet, where the study site Sandehølen is situated, the discharge is reduced due to by-passes from tributary rivers from Surna, to the reservoirs

(Dangelmaier, 2004). The investigated part is about 350 m long and contains different meso-scale habitats. Besides shallow areas, the study reach consists as well of two pools.

3 MODELING OF PHYSICAL HABITAT CONDITIONS

To obtain information on physical habitat conditions a 3D hydraulic model has been used. SSIIM, a three-dimensional numerical model developed for simulation of sediment movements in water intakes with multi-block option was developed at the Department of Hydraulic Engineering at the Norwegian Institute of Technology. SSIIM is an abbreviation for Sediment Simulation In Intakes with multi-block option. The program itself was made for use in River/Environmental/Hydraulic/Sedimentation Engineering. 3D hydraulic modeling calls for geodetic point information spread over the whole study site. In comparison to 1D modeling where transects are measured, multi dimensional modeling requires data collection over the whole reach. Therefore, data sampling was carried out in summer 2004 at river Surna for 3 different discharges, which was used in the following step to develop a hydraulic model. Following three discharge rates have been modeled and verified: Q = 2.1m^3/s, 4.4m^3/s and 6.5m^3/s. The topography survey was done with the total station Leica TC 307 and a Topcon DGPS, while velocity and discharge measurement was done with the help of a Sontek Flowtracker, based on the Doppler effect. In addition to sampling of information on topography, discharge, velocities in order to enable validation of the models and substrate have been measured. Besides topography data and flow rate, waterlevel in the downstream transect serves as input for the three dimensional hydraulic model SSIIM.

The grid generation is probably the most important part in the modeling process, for achieving convergence, with the use of SSIIM. Thus a high degree of non-orthogonality in the grid leads to slower convergence. The waterlines, respective the flows, have been used as important tools for the grid generation. The model calibration was done with the help of the roughness. In many models just a general roughness parameter is set. The Sande part can be described as inhomogeneous concerning the substrate size. While in the most upstream part, bed substrate is rough with large blocks up to 0.4 m in diameter, in the most downstream part and in between, the roughness is much lower with small parts of sand banks. As well, in the two pools the substrate has been defined as much finer than in the upstream part. Therefore, a “bedrough file”, containing detailed information on distribution of substrate size within the study site, has been implemented in addition. Another important fact was to verify the models. This was done with the help of the measured velocities in different random points with known coordinates. For obtaining a better understanding from functionality of SSIIM and influence from roughness, the model was carried out with the “bedrough file” and without. Concerning the verification of the model, the case with included “bedrough file” gave best correlation of measured velocities in field with calculated velocity values in SSIIM for Q=4.4m^3/s and Q=6.5m^3/s. For these models good correlation between measured and simulated velocities was shown. Above all, for low water, the influence of the roughness from the riverbed is very high and leads to problems in the hydraulic calculation. Up to now, this problem for low water depths, has not yet been solved satisfactorily. However, for eco-hydraulic investigations, above all these low discharges are of main interest. The low flows have an influence on the habitat available and are often influenced by anthropogenic use of water as it is the case at river Surna. As well at river Surna the low flow caused problems so that the grid had to be changed to avoid dry cells. In general, the influence of roughness changes with the water depth. The properties of the longitudinal profile change with fast variation in depth, through different roughness elements. This makes determination of hydraulic parameters like water depth (hydraulic radius R = A/U), discharge and water velocity difficult. Local differences in velocities, which are not considered in SSIIM due to high roughness and difficult determination of hydraulic parameters, explain the difference of measured velocities with the modeled ones. The interaction between structure elements and the fluid happens through eddies. By formed eddies the resistance grows so that convergence in SSIIM is complicated. Heavy grid modification was the only way to achieve convergence. In addition to the modeled discharges, other flows up to Q= 35m^3/s have been simulated for gaining information on varying habitat suitability in the study site. The discharge rate at 35m^3/s, describes a small spring flood event in the investigated site. Meso-scale habitat was defined as glide in the upstream area, followed by a deeper, slower area classified as walk and a large pool in the downstream part at 4.4 m3/s. The purpose of this study was to obtain information about velocity and depth distribution for different flows and meso-scale habitats and assignment to a habitat model. For achieving information on the final habitat suitability of areas within the study reach, work has been continued, using the habitat suitability model HABITAT, based on information on physical habitat conditions for varying flow, which was achieved with the help of the hydraulic model described above and comparison of these conditions with general summer preferences for young Atlantic salmon gained from snorkeling observation in many Norwegian Rivers (Harby et al, 1999).

4 RESULTS

In this chapter the calculated areas of preference, avoidance and indifference with the Habitat suitability model for Young Atlantic salmon are demonstrated. The results, available as “Habitat plots” are shown for different discharges so that change in availability of living space due to flow variability can be interpreted. Based on univariate preference curves for salmon parr for summer conditions, comparison with the physical habitat results from the 3D hydraulic model SSIIM has been done and led to the final Habitat plots. These plots, allow detailed information on spatial habitat distribution and give an insight into change in habitat with flow variability. Suitable habitat is described as potential or possible suitability considering parameters depth, velocities and substrate. Suitable area for salmon parr can be found for the modeled low flows, varying around 5m^3/s above all in the upper part, where velocities vary between 18 cm/s up to about 46 cm/s. The depth varies between 20 and 70 cm. However shallow river banks and border areas are avoided. Compared with information given by Hendry et al. (2003) it can be confirmed that young Atlantic salmon finds suitable area between pools and fast flowing shallow areas, where the water velocity is accelerating and the water depth decreasing. The substrate has been defined in these areas, with sizes of about 10-50 cm, which salmon parr prefers. Preferred habitat is given as well in areas of the large pool for discharges around $6m^3/s$. The grid for $Q=2.1m^3/s$ had to be modified heavily in order to make the SSIIM model run. This model is highly affected by the influence of roughness parameters. Above all the shallow part at the downstream end on the orographic right side had to be eliminated in the grid. In this area we find one large and shallow gravel bank. Figures 1,2,,3 give an overview about varying habitat suitability for three different discharges.

Figure 1: habplot $Q=4.4m^3/s$

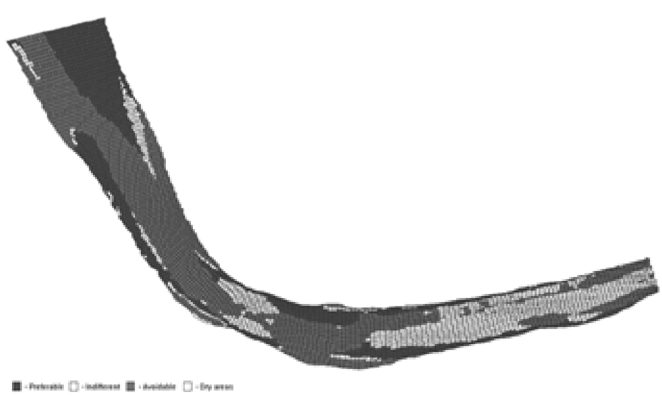

Figure 2: habplot $Q=22m^3/s$

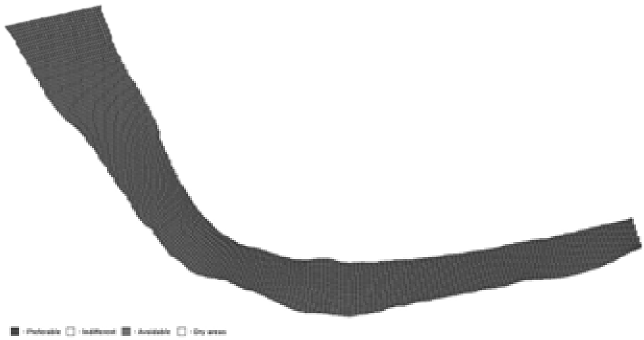

Figure 3: habplot Q=26m³/s

Avoidance of area is given in the shallowest parts of the river surrounding the large pool and close to river banks for low discharge. In these areas flow velocities are low. With rising discharge shallow and avoided areas are disappearing, while on the other hand already deep parts grow to avoided areas. This can be seen in figure number 2 where areas of avoidance around the large pool are even larger. What can be seen is that the area in the uppermost part turns from preferred to indifferent area, due to higher flow velocities when discharge is higher. On the other hand the shallow river bank areas turn to preferred area, because of higher water depths. The higher discharges are mainly characterized by avoided area. This is due to higher velocities and high water depth in case of high flows. In total the results given in the Habitat plots have been combined in following tables, which show change of habitat due to flow variability. Figure number 4 and 5 show habitat suitability for the whole study site, while figures number 6 until 11 describe habitat conditions for the separated meso-scale classes. The habitat classes have been defined for Q=4.4m³/s.

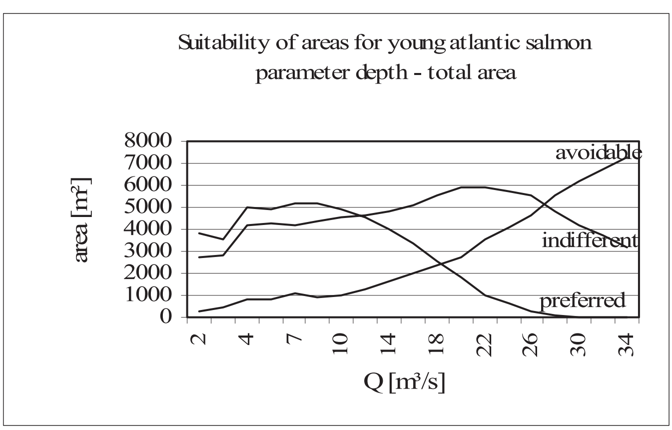

Figure 4: Depth suitability curves, whole reach

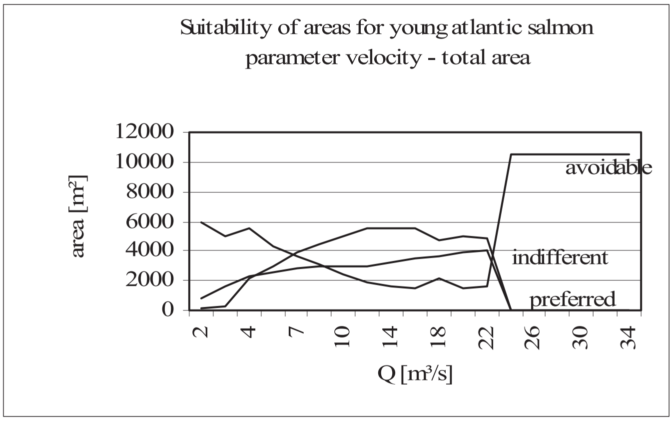

Figure 5: Velocity suitability curves, whole reach

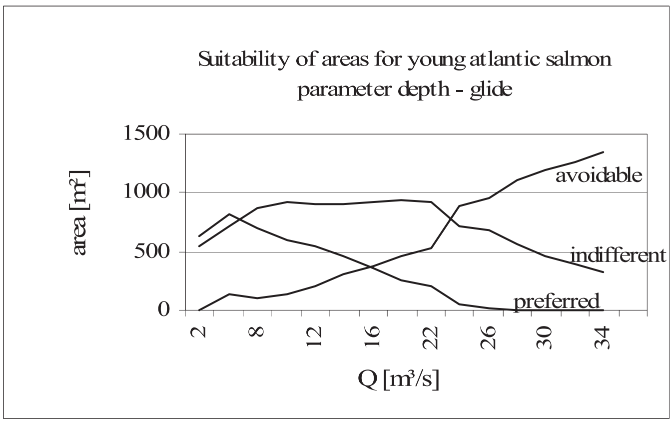

Figure 6: Depth suitability curves, glide

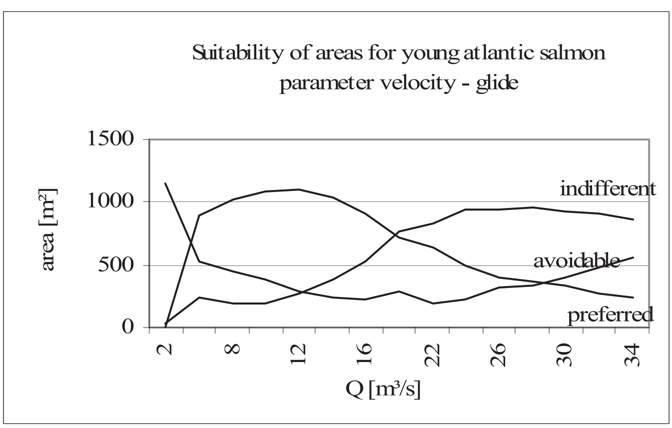

Figure 7: Velocity suitability curves, glide

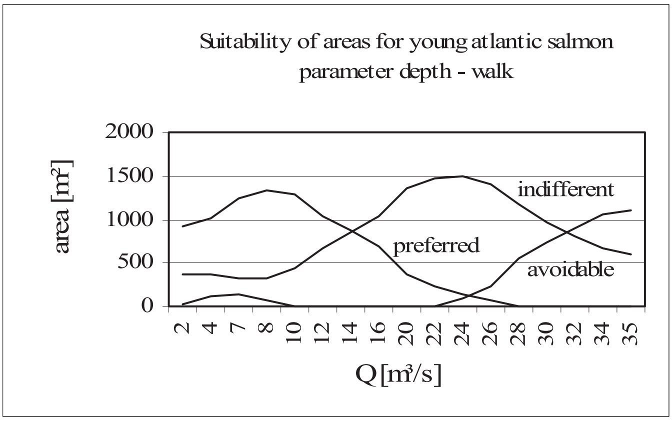

Figure 8: Depth suitability curves, walk

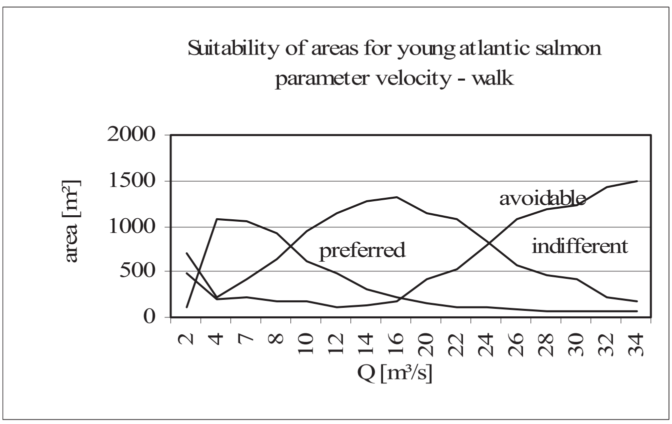

Figure 9: Velocity suitability curves, walk

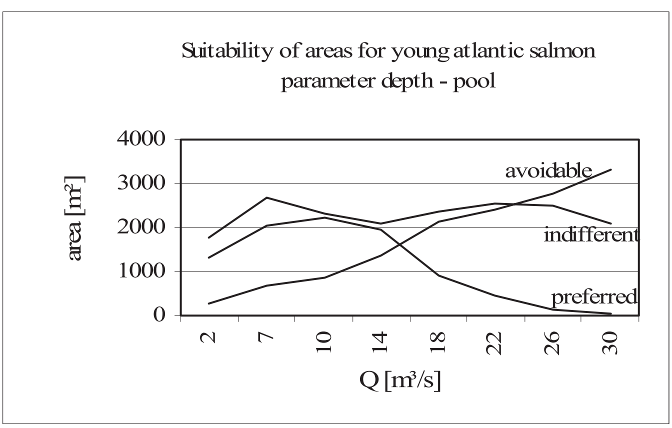

Figure 10: Depth suitability curves, pool

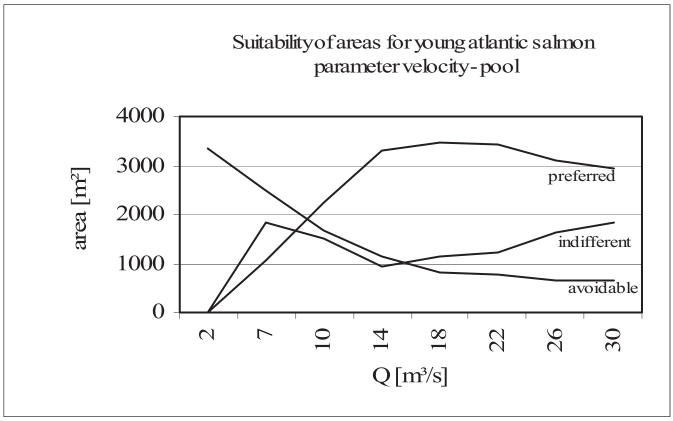

Figure 11: Velocity suitability curves, pool

Thus the model for low discharge did not correlate very well in velocities, the sharp bend is caused in all figures above.

During field work respective meso-scale habitat classes have been electro-fished in three runs each.

The following figure shows distribution of classified meso-habitat classes and fishing stations

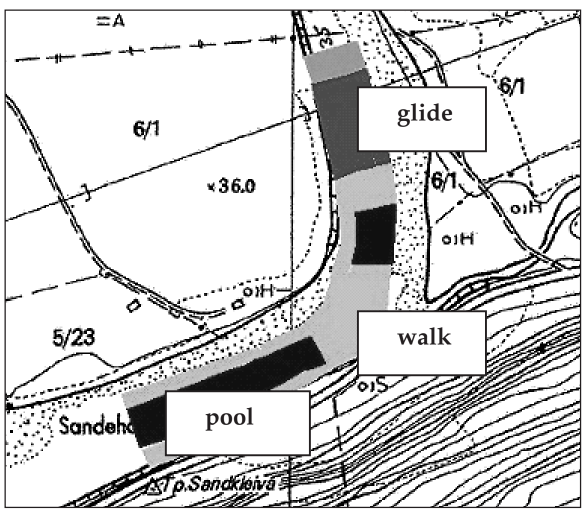

Figure 12: Meso-habitat distribution at 4.4 m3/s and Fishing- stations

Table 1 gives the number of caught fish in the different meso-scale classes by electro-fishing in August 2004 and estimated numbers, based on different runs, received from Norwegian Institute of Nature Research.

Table 1. Number of young Atlantic salmon and estimated numbers [n/100m²] in the different Meso-scale habitat classes at 2.1 m3/s for summer conditions (Ugedal et al., unpublished).

Density	salmon 0+		salmon 1 y. & older	
[n/100m^2]	collected	estimated	collected	estimated
Meso-scale habitat class				
glide	62	77.1	23	25.8
walk	87	49.7	13	6.0
pool	110	159.1	39	48.9

In general, fish densities in the study site are high at summer low flow conditions. Above all in the meso-scale habitat class pool, many young Atlantic salmon have been caught during field work when flow was 2.1m^3/s. As well in the meso-scale habitat class walk, the number was high.

Comparing the electro-fishing data with results from habitat modeling, it has to be mentioned that above all in the uppermost part of the reach the model are showing preferred area at low flow. The fish data confirms relatively high densities in this part. For the large pool, mesohabitat pool results from modeling and fish data show different results. While model tools show avoidable area for low flow, the fish data gave the highest fish densities. This can be explained by the fact that substrate may play an important role in this area. Looking at the separated plots for parameters depth, velocities and substrate we can see that the single plots for depth and velocities show different results. While the depth preference is high, velocities are not preferred due to too low velocities at low flow.

5 DISCUSSION

Based on Habitat modeling of one varied reach in the by-pass section of river Surna, the impacts of varying discharge on habitat suitability for young Atlantic salmon have been quantified. Parameters considered in the Habitat model have been depth and velocity distribution, substrate information and preference curves for Young Atlantic salmon. Through the band width of modeled discharges from low to medium high flow, much of the annual discharge range except flood has been simulated. The availability of suitable habitat for young Atlantic salmon is important because of the high importance of salmon in Norway and to fulfill the guidelines set by law, evaluating the effect of a possible environmental flow requirement and to fulfill the requirements for the water framework directive of ensuring acceptable ecological potential. It still does not exist an environmental flow requirement for the by-pass section, upstream the power plant outlet. Furthermore river Surna is protected as a national water course for Atlantic salmon. For the habitat modeling process no local preference curves could be used, because up to now, only electro-fishing data is available.

The modeled flows from the study site led to explicit results. Above all the separation of different meso-scale habitat classes makes extensive use of results possible. The results can be transferred through up scaling to a longer river reach. Therefore separation of meso-scale classes was important and shows results in habitat suitability in every class.

In general, high flows show small preferred area, due to high velocities and depths. However, like results of electro-fishing show, that although avoided area is high in the river for low and high discharges, high densities of fish have been observed. This is most probably due to occurrence of preferred substrate. If we look at separated habitat plots for single parameters it can be seen that mostly just one fact is decreasing suitable area. For discharge 2.1m^3/s depth suitability would be given while velocities are too low. For the three modeled discharges enough suitable habitat area is available for flow 4.4 and 6.5m^3/s. Fish data shows, that in general fish densities are high for the whole reach. Above all, as well the numbers for the pool is high, but modeling with SSIIM and Habitat showed avoidable area for low flow. For higher discharges the trend shows for the whole river, but as well for the separated meso-scale habitat classes pool, glide and walk, that avoidance of areas is increasing. The low flow shows mainly avoidable area, but as mentioned the model could not be optimized due to high roughness influence. However suitable area does not guarantee high densities. Results from habitat modeling always show the potential suitable habitat due to chosen parameters. If fish is using the suitable area in reality, depends on many other factors than velocities, substrate and depth. Habitat availability depends as well on predators, temperature or amount of oxygen.

Therefore these factors should be included in future studies. The advantage of upscaling can be used in the future for obtaining substantial information on river health. Final conclusion and information of optimum Q for each mesoclass is given in the following part. Comparison of different Habplots for varying discharge gives following maximum of preferred area. Comparison of plots for combined parameters velocity and depth has been carried out. The uppermost part, classified as glide shows most preferred area with Q= 8m^3/s. For this flow rate velocities are not too high and shallow areas for lower flows at the river banks turned into deeper areas. The same could be noticed for the classified walk in the bend. Here optimum discharge varies between 6.5 and 8 m^3/s and finally the pool shows highest amount of preferred area for Q=10m^3/s. Noticed in this case, was that above all for the velocity plot the preferred area increased with rising discharge up to 10m^3/s.

It is also interesting to obtain information about long term habitat suitability due to climate change.

Furthermore modeling difficulties due to high roughness should be investigated more, in order to avoid convergence problems and result interpretation.

For more information on low flows in the Sande part, comparison of results could be done as well with results of Stickler et al. (2004), using the two dimensional hydraulic model River2D for creating an optimized ice channel in the adjacent areas of Sande. River2D enabled better modeling of low flows, useful for comparison and result completion from SSIIM.

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Computational approach for near bottom forces evaluation in benthos habitat studies

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ABSTRACT: Characterization of near bottom hydraulics is still a challenging issue in habitat studies. One measurement device which can be used is the FST hemispheres method introduced by Statzner & Müller (1989). Over the past 15 years, FST hemispheres became widely accepted in Germany as a tool for describing benthos habitats. A lot of preference curves were found for different benthos species. Although very appreciated by biologists developing preference curves, measurements with FST hemispheres can be very difficult and time consuming if the complete characterization of a river reach by different discharges is needed. They are also restricted to small water depths (up to 60 cm) and special kinds of substrate. There is an urgent need to replace these laborious measurements with a computational method. A new approach which allows calculation of FST hemispheres' distributions over the river reach is presented. The main idea is to combine information from 2D hydraulic modeling and a special kind of substrate mapping to calculate forces acting on a hemisphere. The model takes into account recent developments in characterization of a vertical velocity profile in gravel-bed rivers. Some results of field measurements showing the capabilities of the new model are also presented.

1 INTRODUCTION

FST hemispheres have been proposed as a method for rapid characterization of near bottom hydraulics and, consequentially, for the assessment of the forces acting on benthos organisms (Statzner & Müller, 1989). The measurement equipment consists of 24 hemispheres of identical radius (3.9 cm) but different densities and a ground plate. The plate is placed horizontally on a river bottom and hemispheres one by one are exposed to the current. The number of the heaviest hemisphere just moved by the given flow is used as the result of the measurement.

From the very beginning, the method is a controversial subject in benthos research, see for example Frutiger & Schib (1993), Statzner (1993), Heilmair & Strobl (1994). Some important questions, like "can FST hemispheres be an indicator for the forces acting on benthos animals, which are an order smaller in size than FSTs" or "is there any advantage of the FST hemispheres in comparison to conventional methods (for example, velocity measurements with micro-propeller)", are still not fully answered. Anyhow, FST hemispheres became a widely accepted tool for the characterization of benthos habitats, especially in Germany. A lot of preference curves were found for different benthos species. This data is commonly used in ecological studies, for example to define minimum flow requirements in diverted streams (see among others Statzner et al. 1990, Jorde 1996).

Although conceptually very simple in use, measurements with FST hemispheres become very laborious and time consuming if the complete characterization of a river reach by different discharges is needed. As a rule of thumb, 100 measurements randomly distributed over the reach are needed for one single discharge. The method is also inapplicable at water depths over 70 cm and some substrate types (for example bedrock). Many authors tried to overcome these problems by establishing calibration curves. Popular parameters against which FST numbers were calibrated are bottom shear stress (Statzner et al. 1991, Dittrich & Schmedtje 1995) and flow velocities (Heilmair & Strobl 1994, Mader, H., pers. comm.). Although for specific rivers these curves can be valid, they are generally not transferable to other river reaches. Dittrich (1995) showed, that calibration curves against bottom shear stress are strongly dependent on substrate characteristics. The same is true for calibrations against mean column velocity.

There are also approaches that allow to obtain complete FST hemisphere distributions for the specific river reach by a given discharge. Here, the "statistic" models of Lamouroux et al. (1992) and Scherer (1999) should be mentioned. As a basis, they use information on FST distributions obtained in the past for different river reaches. Deficiencies of such models are, that they are using too few specific reach parameters and should be possibly calibrated for the every application case.

In this paper we present a method for calculating the FST numbers (densities), which is based on force balance of a hemisphere in a velocity column. It integrates local river information on substrate, mean column velocity, and water depth, which is an advantage in comparison to the common calibration curves. As a result, maps of spatial FST hemispheres distributions can be produced, which are an alternative to the results that can be obtained with "statistical" approaches. These maps can be used in minimum flow studies as well as in more complex restoration/rehabilitation river projects where there is a need to combine near bottom flow information with other parameters relevant for benthos habitats.

2 FORCE BALANCE FOR A HEMISPHERE IN VELOCITY COLUMN

Considering an FST hemisphere placed on a ground plate in velocity column (see Figure 1), the following force balance equation can be written:

$$F_{drag} - F_{frict} = 0 \tag{1}$$

with :

$$F_{frict} = \mu \cdot \left(F_{w'} - F_{lift}\right) \tag{2}$$

where F_{drag}, F_{lift} = drag and lift forces, F_{frict} = friction force, $F_{w'}$ = weight of the hemisphere in water, μ = friction coefficient between the hemisphere and the plate.

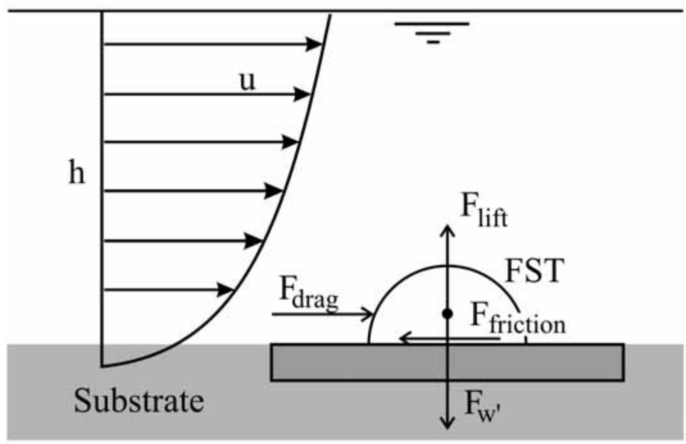

Figure 1. Forces acting on a hemisphere in velocity column.

From the similarity considerations drag and lift forces are defined in a following way:

$$F_{drag} = \frac{1}{2} c_{drag} \cdot \rho_w \cdot u^2 \cdot A_h \tag{3.1}$$

$$F_{lift} = \frac{1}{2} c_{lift} \cdot \rho_w \cdot u^2 \cdot A_h \tag{3.2}$$

where c_{drag}, c_{lift} = drag and lift coefficients, ρ_w = density of water, u = reference velocity, A_h = hemisphere area, perpendicular to flow direction.
Weight of a hemisphere in water is given by:

$$F_{w'} = V_h \cdot g \cdot (\rho_h - \rho_w) \tag{4}$$

where V_h = volume of a hemisphere, g = acceleration due to gravity, ρ_h = density of a hemisphere.
Therefore, the density of the hemisphere just moved by a given flow is:

$$\rho_h = \rho_w \cdot \left[1 + \frac{3}{8} \frac{u^2}{g \cdot r_h} \left(c_{lift} + \frac{c_{drag}}{\mu} \right) \right] \tag{5}$$

From the experiments with bluff bodies (not streamlined objects) it is known, that in a uniform velocity profile drag and lift coefficients are only depended on the Reynolds number $Re = u \; r_h \; \nu^{-1}$. Hemispheres placed on the plate in the river represent a far more complicated case. Flammer & Tullis (1970) investigated free surface flow past hemispheres with smooth bottom conditions and found out that at relative submergences $h/r_h > 4$ the drag coefficient is independent of the Froude number. This value represents a restriction on the minimum water depth for FST measurements. Authors also postulate that drag is independent from the form of the velocity profile (uniform in contrast to fully developed logarithmic), but this can be an artefact, because it is difficult to evaluate the influence of depth in laboratory flume without running into problems with side walls influence. Generally, additional research of drag and lift coefficients is needed, which, in this work, is done by means of numerical flow simulation.

Scherer (1999) modified the original FST plate to reduce suction between the plate and the hemisphere and to eliminate the backflow effect resulting from the hand during positioning of the hemisphere. By tilting experiments he found that the mean value of the friction coefficient μ for the new plate is equal 0.46 and varies significantly from 0.32 to 0.6.

3 TURBULENT VELOCITY PROFILE OVER HYDRAULIC ROUGH SURFACE

There is no doubt, that essentially the flow velocity distribution near the river bottom determines the flow forces acting on the FST hemispheres. As in most lotic environments hydraulically rough flow predominates, we want to present the state-of-the-art in the theory of turbulent flows over rough surfaces and we base this upon the recent work of Bezzola (2002).

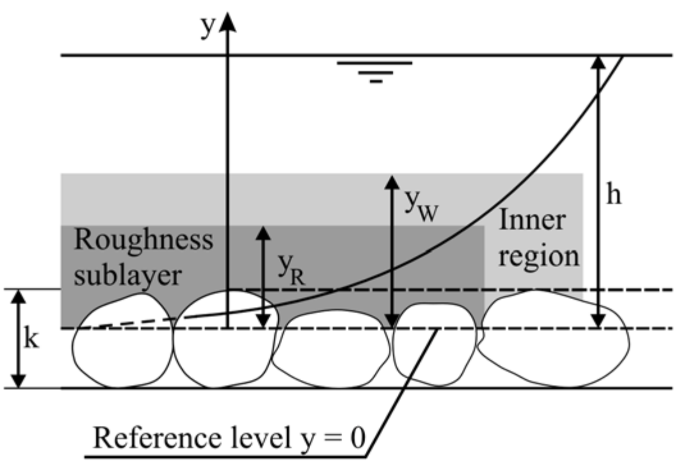

Figure 2. Definition sketch of the velocity profile over rough bed.

In his work, he summarizes all published data on measurements of turbulent velocity profiles, concentrating especially on cases with small relative submergence. On the basis of the mixing length approach of Prandtl, he develops a new model for the velocity distribution. The main element of Bezzola's model is the assumption that not the total shear stress, but only the portion of turbulent shear stress controls the mean motion. Additionally, the thickness of the roughness sublayer is introduced as a roughness parameter. The latter is an analog to Nikuradse's equivalent sand roughness k_s, but in contrast to the already existing approaches, it is not dependent on relative submergence. This means that for a given substrate configuration, this parameter has to be measured only once and needs no further adjustment. The new velocity profile (see definition sketch in Figure 2) is given by equations:

$$\frac{u}{u_*} = c_R \cdot \left(\frac{1}{\kappa} \cdot \ln\left(\frac{y}{y_R} \right) + 8.48 \right) \qquad \text{for } y \leq y_w \tag{6.1}$$

$$\frac{u}{u_*} = c_R \left(\frac{1}{\kappa} \ln\left(\frac{y_W}{y_R} \right) + 8.48 \right) +$$

$$+ \frac{2}{3} \frac{h}{\kappa \cdot y_W} \left[\left(1 - \frac{y_W}{h} \right)^{\frac{3}{2}} - \left(1 - \frac{y}{h} \right)^{\frac{3}{2}} \right] \quad \text{for } y > y_w \tag{6.2}$$

with u = mean velocity on level y, u_* = shear velocity, c_R = turbulence damping factor, κ = von Kármán constant, y_R = thickness of roughness sublayer, y_W = thickness of near-wall (inner region), h = water depth.
The turbulence damping factor is given by:

$$c_R^2 = 1 - \frac{y_R}{h} \quad \text{for } \frac{h}{y_R} > 2 \tag{7.1}$$

$$c_R^2 = 0.25 \frac{h}{y_R} \quad \text{for } 0 \leq \frac{h}{y_R} \leq 2 \tag{7.2}$$

The thickness of inner region is also dependent on relative submergence and defined as:

$$y_W = h \quad \text{for } \frac{h}{y_R} \leq 1 \tag{8.1}$$

$$y_W = y_R \quad \text{for } 1 \leq \frac{h}{y_R} \leq 3.2 \tag{8.2}$$

$$y_W = 0.31h \quad \text{for } \frac{h}{y_R} > 3.2 \tag{8.3}$$

The thickness of the roughness sublayer can be estimated by measuring the turbulence intensity profiles. These values have their maximum on the top of the roughness sublayer and stay more or less constant or get smaller if moving downwards from it.

4 NUMERICAL FLOW SIMULATIONS OF HEMISPHERES IN VELOCITY COLUMN

To obtain values of drag and lift coefficients for FST hemispheres the computer program FLUENT (Fluent 2003) was used. It is a state-of-the-art software for modeling 2D or 3D fluid flow and heat transfer in complex geometries. As work is still in progress, only some aspects of numerical flow simulations will be highlighted in this paper.

4.1 Objectives

The influence of following parameters has to be evaluated by numerical flow simulations:

substrate,
velocity, and
relative submergence.

The log profile of Bezzola was chosen as a theoretical basis for the calculations. Therefore, substrate characteristics are represented by the thickness of the roughness sublayer y_R. As a reference velocity, the mean column velocity at 40% of depth (but not the near bottom velocity) was chosen for the following reasons: mean column velocity is a quite common parameter in environmental studies and can be easily measured or modeled with 2D hydraulic programs; as a velocity gradient at 40% of depth is much smaller than near the bottom, the error in mean velocity determination will be much smaller in case of a wrong assumption on the reference level of the profile.

Also the influence of the hemisphere's vertical position in the velocity profile has to be investigated. According to the standard field procedure (Statzner & Müller 1989), the ground plate has to be placed on the river bottom in a horizontal pit and leveled with substrate tops, which is no problem for bottom materials of small size. On the contrary, in rivers with coarse substrate this is difficult, if possible at all, to perform.

4.2 Numerical model

A computational domain was chosen upon guidelines of Flammer & Tullis (1970). The size of the domain is 20X50X25 cm. The width of 50 cm, which represents half of the domain due to the symmetry of the problem, was necessary to insure that side walls do not have an effect on the flow field. A hemisphere is placed 50 cm downstream from the inflow boundary.

The problem is modelled as a channel with symmetry boundary condition at the top (without free surface). As there is no tool in FLUENT that allows to parameterize large roughness and it also makes no sense to reproduce every stone, the following was done: the bottom of the channel is modelled as a moving boundary and the hemisphere is positioned at the level $y = 0.25y_R$. On the inflow boundary the velocity profile, given by Equations 6.1 and 6.2, is applied.

Steady state calculations are performed with the standard *k-* turbulence model. To check the model performance, experiments of Flammer & Tullis (1970) with smooth bottom conditions were reproduced. In case of the uniform velocity profile, a very good agreement for the drag coefficient in the entire range of the tested Reynold's numbers was obtained. Using the fully developed log profile, we got values that were about 25% smaller. Apparently, there is no detailed data on the reference values of depths and velocities for every experiment, so the different values for the mean velocity could be the reason for this discrepancy.

5 FIELD MEASUREMENTS

Two objectives were set up for the field measurements. The first one was to verify the applicability of the new velocity distribution law. The second was to check if the FST hemispheres numbers can be estimated from detailed velocity measurements. The pilot field measurements were done at the river Alz in Bayern (Germany).

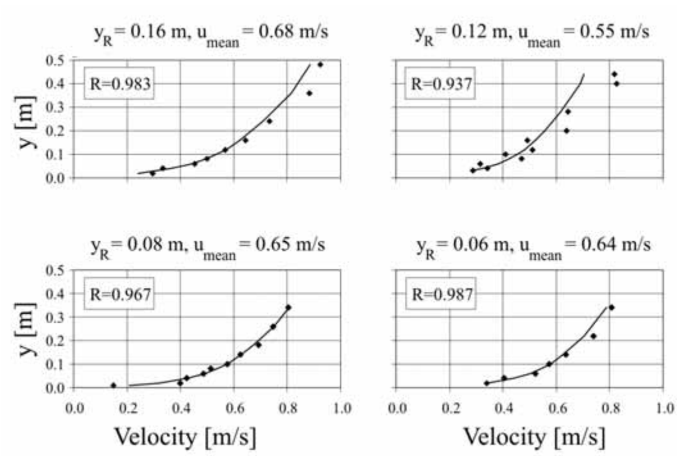

Figure 3. Examples of measured and fitted velocity profiles.

5.1 Methodology

Measurements were performed as follows: at first, the FST measurements were done, and water depth, visual substrate characteristics, special point features and wind conditions were noted. After the ground plate was removed, a flow velocity profile was measured with 3D-ADV. In total, 21 profiles and parallel FST measurements were done. Another 30 points were taken, but only with the velocity measured at 40% of depth.

The applicability of Bezzola's log-law was verified the following way:

1 From the vertical profiles of the turbulent intensities, the thickness of the roughness sublayer was estimated.
2 According to Equations 8.1 – 8.3, the thickness of the inner region was calculated.
3 By varying the reference level position $y = 0$ of the velocity profile, the best log fit was found for the inner region.
4 Water depth and the thicknesses of a roughness sublayer and of an inner region were adjusted.
5 Fit over the entire water depth was recalculated.

After the profile was approximated by the log fit, the vertical position of the FST ground plate was recalculated. For cases in which the latter was above the reference level position of the log-law profile, detailed 3D calculations of the drag and lift coefficients were done.

5.2 Results

Points for detailed velocity measurements were selected within two river cross-sections with widths of 34 and 28 m respectively. Substrate was mostly represented by stones with diameters ranging from 5 to 20 cm.

The theoretical velocity profile of Bezzola has a surprisingly good correspondence to the measured velocity profiles. From 21 profiles only two did not have a log form, the first one was situated 2 m behind a big boulder, the second was measured in a local depression. In the Figure 3, the four adjusted profiles are shown. Fourteen fits had a coefficient of determination higher than 0.94, two larger than 0.81, and the two remaining 0.77 and 0.52. The coefficient of determination was generally higher for profiles with high water depths and for profiles that were measured with the longer time interval (60 seconds versus 30 seconds). It should be also mentioned, that it was difficult to estimate the thickness of a roughness sublayer for shallow depths, thus its value was taken in analogy to the profiles with larger depths.

For the nine profiles (in which the adjusted position of the ground plate was above the zero reference level) drag and lift coefficients were calculated using above mentioned numerical model and the corresponding FST densities were obtained with the Equation 5 (see Table 1). For the remaining nine profiles, the adjusted reference position was higher than the position of the ground plate and no numerical simulations with log profile could be performed.

Table 1. Computed and measured FST numbers.

Depth	y_R	$u_{40\%}$	c_{drag}	c_{lift}	FST N computed	FST N measured
m	m	m/s	-	-	-	-
0.31	0.12	0.45	0.25	0.33	6	8
0.39	0.08	0.65	0.22	0.26	8	8
0.52	0.08	0.72	0.23	0.27	8	8
0.48	0.12	0.55	0.24	0.28	7	8
0.40	0.09	0.57	0.29	0.20	7	6
0.15	0.07	0.11	0.31	0.37	1	1
0.16	0.11	0.22	0.40	0.40	4	5
0.27	0.06	0.36	0.25	0.36	5	5
0.40	0.13	0.39	0.25	0.3	5	6

6 CONCLUSIONS

The pilot measurements gave a positive result in two respects. First, the theoretical velocity distribution of Bezzola could be applied for a majority of the measured profiles. Second, FST numbers, estimated on the basis of the detailed velocity measurements, are close to those measured in the field. Further measurements should be performed for fine substrates and smaller river widths. Also, a method should be developed for the robust estimation of the roughness sublayer thickness, as the measurement of velocity and turbulent intensity profiles is very time consuming.

After the numerical investigation on how substrate, mean velocity, and relative submergence influence the drag and lift coefficients will be completed, predictions on FST distribution based on 2D flow simulation will be possible. Mean column velocities and water depths from 2D hydraulic model, combined with substrate mapping will form a basis for the calculation of the local FST numbers.

We also recognize that the distributions obtained using the values from 2D modeling will probably not match the distributions obtained by 100 field FST measurements. FST hemispheres work generally on a much finer scale than a 2D hydraulic model. This behaviour should be more apparent in rivers with coarse substrates, which should be checked by further measurements.

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“HydroSignature” software for hydraulic quantification

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ABSTRACT: A quantitative description of physical habitats is essential for ecohydraulic modeling.

For a given discharge and for any part of the aquatic space, a hydrosignature represents the calculation of the surface or volume percentages in a depth and mean velocity cross classification.

Free software has been developed to provide the automatic calculation of hydrosignatures for several types of vertical measurements described by their depth and current velocity:

- Non spatialized data sets for a given surface,
- Cross sections,
- Meshing based on finite elements represented by triangles or four angles in the horizontal plane.

As hydrosignatures are visualizations of the hydraulic diversity, they can be used for instance to help verify the validity of fauna habitat models.

1 INTRODUCTION

In an ecosystem of running waters, the distribution of aquatic communities such as fish and demersal invertebrate is dependant on the local hydraulic conditions (i.e. Aadland 1993; Jowet 2003).

Therefore, the two hydraulic parameters depth and velocity are often measured or modeled for lotic ecology studies.

However, for ecohydraulic modeling purposes, certain scientists such as Statzner et al (1988) and Inoue & Nunokawa (2002) insist on the value of hydraulic complexity descriptive measurements on a micro-habitat scale, which should not only be summarized in terms of mean values.

Within the limited scope of quantifying the hydraulic diversity based on mean depth and velocity couples, a representation in terms of hydrosignatures is therefore proposed.

When it comes to calculating the hydrosignature of a part of the aquatic space referred to as a “hydraulic unit”, it is necessary to first define the velocity and depth classes. The velocity/ depth plane is therefore divided into crossed classes. A surface hydrosignature consists in calculating the surface percentage of each crossed class of the unit in the horizontal plane.

Figure 1 illustrates the construction of a surface hydrosignature, not to mention that the same type of calculation can be carried out in terms of volume. A hydraulic unit is three-dimensional and can be broken down into verticals. As each vertical has its own depth and mean velocity, it only belongs to one crossed class. A volume hydrosignature involves calculating the volume percentage of each crossed class.

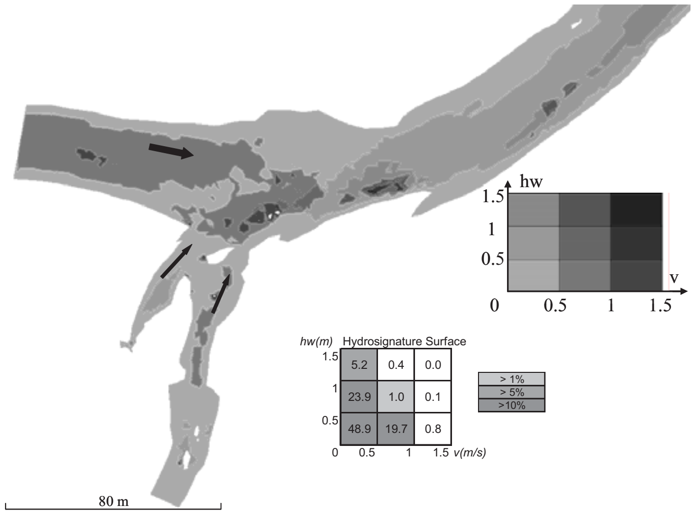

Figure 1. Definition of the crossed classes in terms of velocity and depth; representation in the horizontal plane of a stream reach illustrated using a colour per crossed class. Surface hydrosignature of the stream reach obtained by calculating the surface percentage of each crossed class.

The data required for hydrosignature calculations can be based on different types of verticals to which depth and mean velocity hydraulic values are associated.

- Non-spatialized data sets on a given surface,
- Cross sections associated with a representative length in the direction of the flow,
- Meshing based on finite elements represented by triangles or four angles in the horizontal plane.

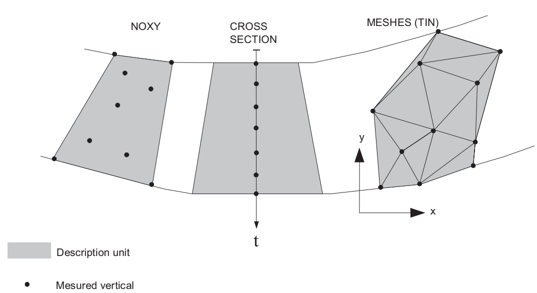

Figure 2. Illustration of the different types of hydraulic descriptions in the horizontal plane based on mean velocity/ depth verticals.

The free HydroSignature software program operating under Windows XP was used to facilitate these calculations, which can be downloaded from the address: http://hydrosignature.aix.cemagref.fr/.

2 METHODS

2.1 Définitions, nomenclature

u: hydraulic description unit number.
hw [m]: depth.
v [m /s]: mean velocity for the vertical.
t [m]: abscissa of a vertical in relation to a relative reference specific to the cross section.
l [m]: length representative of a cross section in the current/ flow direction.
w [m]: width of a unit.
J: total number of crossed classes belonging to a hydrosignature grid.
j: number of a crossed class.
A [m²]: water surface area
V [m3]: volume
ej,je: hydroelement – a hydroelement can be defined as an element of volume in a description unit (u) that belongs to only one crossed class j (Fig. 3).
je: number in the unit (u) of a hydroelement of the crossed class j.
JE: total number of hydroelements belonging to the crossed class j of the unit (u).
Aj,JE [m²]: horizontal surface of all the hydroelements belonging to the crossed class j.

$$A_{j,\,JE}(u) = \sum_{je=1}^{je=JE} A_{j,\,je} \quad (1)$$

Vj,JE (u) [m3]: volume of all the hydroelements of the crossed class j.

$$V_{j,\,JE\,(u)} = \sum_{je=1}^{je=JE} V_{j,\,je} \quad (2)$$

ccaj: percentage in terms of the horizontal surface of the crossed class j.

$$cca_j\,(u) = \frac{A_{j,\,JE}}{A(u)} \cdot 100 \quad (3)$$

ccvj: volume percentage of the crossed class j..

$$ccv_j\,(u) = \frac{V_{j,\,JE}}{V(u)} \cdot 100 \quad (4)$$

For a hydrosignature:

$$\sum ccv_j = 100 \qquad (5)$$

$$\sum ccv_j = 100 \qquad (6)$$

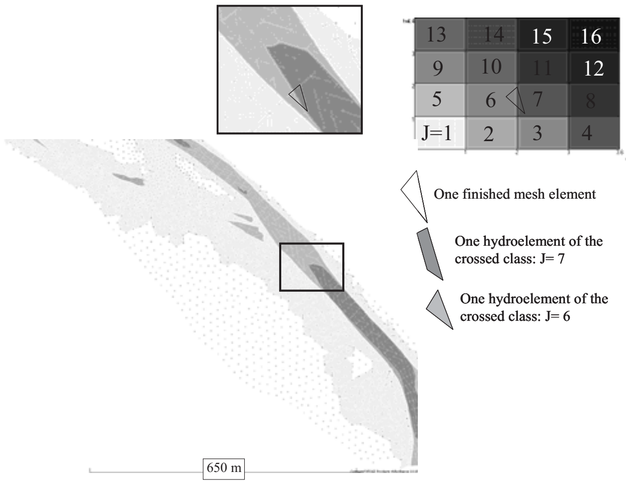

Figure 3. Visualization of a hydroelement in a mesh.

2.2 *Meshing*

In this method, all verticals are spatially identified by the co-ordinates x and y, with z representing the altitude at the bottom of the stream. A triangular irregular network (TIN) can be constructed to interconnect the verticals within the horizontal plane.

Figure 4. Visualization of a mesh in the horizontal plane.

Each triangle of the xy plane defines a finite element of the polyhedral volume including 9 edges that are composed of three verticals, which are themselves interconnected by three segments on the surface and three segments at the bottom.

The basic principle behind the hydrosignature calculation of a meshed unit consists in breaking down – when necessary – each finite element of the mesh into hydroelements.

This breaking down is performed by linear interpolation in space (x,y,z,d,v) based on the vertical co-ordinates in this space. The percentages of crossed classes are therefore calculated based on hydroelement horizontal surfaces or volumes according to equations 1 & 3 or 2 & 4 (Figure 5).

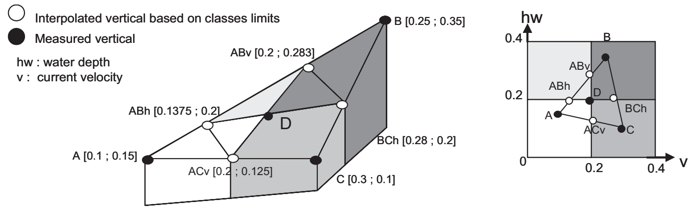

Figure 5. Breaking down a finite element of a hydroelement mesh.

In the case where a unit is described in the horizontal plane by a quadrangle mesh, each quadrangle is broken down into four triangles all sharing the same centre of gravity whose co-ordinates in space (x,y,z,d,v) represent the mean of the co-ordinates of the four verticals having generated the quadrangle. At this stage, all calculation techniques developed for a TIN type mesh are therefore applied.

2.3 *Example using the software for TIN meshing.*

In order to calculate the hydrosignature of units produced by a TIN mesh, the software requires 3 "text" format files (ASCII), with one file defining the depth and velocity classes and the remaining two input files defining the mesh characteristics.

The following notations were used:

Text in italics: alphanumerical data provided by the user.

Crossed class file : *2cc.txt*

hw(m) *hw hw hw hw.hw*

v(m/s) *v v v v*

Input file 1 defining the unit meshes :*1.txt*

[3ANGLES]

[HEAD]

description provided by user

[/HEAD]

{x y z hw v}

[UNIT *unit_name*]

n1 n2 n3

n1 n2 n3

…….. ;

n1 n2 n3
[/UNIT]
[UNIT *unit_name*]
n1 n2 n3
n1 n2 n3
…….. ;
n1 n2 n3
[/UNIT]
……….
[/3ANGLES]

Input file 2 associated with file 1 defining the verticals :*1*_XYHV.*txt*
x y z hw v
x y z hw v
x y z hw v
……….
x y z hw v

The references *n1 n2 n3* mentioned in the hydraulic unit descriptions of the input file 1 refer to 3 line numbers in file 2, in other terms 3 verticals defining a finite element of the mesh. File 1 can contain as many units as the user requires, with each unit being described by one or several meshes/ finite elements defined by series of 3 verticals known by their position (line No.) in the associated input file 2.

The above-mentioned files need only be flagged in order to calculate the hydrosignatures by units and the overall hydrosignature (sum of the units) contained in the object file. It is also possible to automatically create another object file in Microsoft Excel in order to directly visualize the results.

Other than the volumes and horizontal surfaces, the HydroSignature software is also capable of automatically calculating the mean and extremal hydraulic characteristics of units.

2.4 Cross sections

In this method, the units are composed of cross sections and the verticals are spatially identified according to an abscissa t belonging to each cross section, with this axis being located at the intersection of the water surface and the vertical plane perpendicular to the flow containing the cross section. The length l that is representative of the cross section in the flow direction must have already been measured.

The basic principle behind calculating hydrosignatures in this case is similar to that of meshing. Linear interpolations are performed in the vertical plane of the cross section in order to break down the latter into trapeziums. These trapeziums define the hydroelements for which the third dimension is represented by the length l in the flow direction.

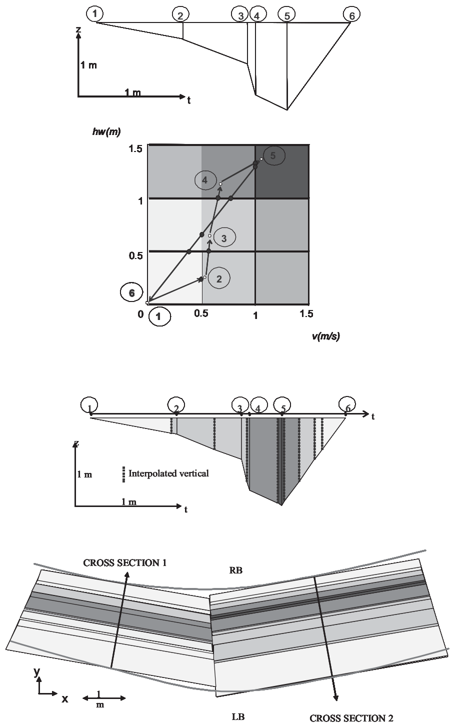

Figure 6. Breaking down cross sections into hydroelements.

Other than a class file, the HydroSignature software must also have an input file of the type:

[TRANSECTS]
[HEAD]
description provided by user
[/HEAD]
[tape(m) hw v]
[TRANSECT *unit_name*]
[LENGTH(m) *l*]
t hw v
t hw v
…….
t hw v
[/TRANSECT]
[TRANSECT *unit_name*]
[LENGTH(m) *l*]
t hw v
t hw v
…….
t hw v
[/TRANSECT]
……………..
[/TRANSECTS]

Another method of calculation is offered by the software in the case where the user can provide len-gths representative of different left bank and right bank cross sections. However, the current version does not take into account completely spatialized cross sections for which the co-ordinates of the banks and the limits of representativeness are known.

2.5 NOXY Non-spatialized data.

With this method, the depths (hw) and mean velocities (v) of verticals are recorded without reference to their position; the verticals do not have co-ordinates (no x no y = NOXY). The dimensions of a unit are determined by a length l and one or several widths.

As the relative positions of the verticals are unknown, the exact representativeness of each vertical in terms of the horizontal surface is also unknown. Each vertical is therefore considered to represent an equal proportion of surface. Each vertical produces a hydroelement with constant depth and velocity values.

Other than a class file, the HydroSignature software must have an input file of the type:

[NOXY]
[HEAD]
description provided by user
[/HEAD]
[hw v]
[UNIT *unit_name*]
[LENGTH(m) *l*]
[WIDTHS(m) *w w w*]
x y z hw v
x y z hw v
……….
x y z hw v
[/UNIT]
[UNIT *unit_name*]
[LENGTH(m) *l*]
[WIDTHS(m) *w w w*]
x y z hw v
x y z hw v
……….
x y z hw v
[/UNIT]
……………
[/NOXY]

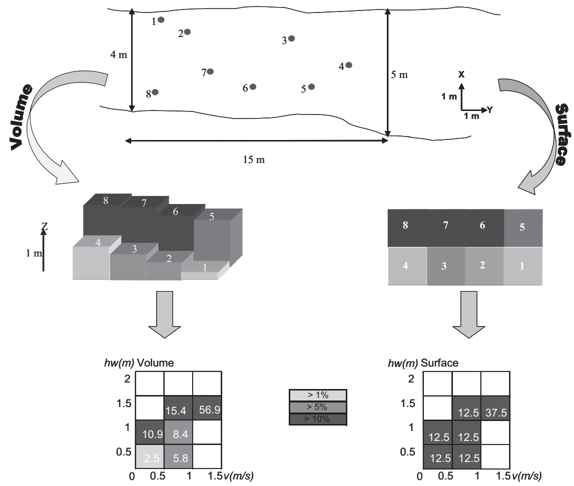

Figure 7. Non-spatialized NOXY data generating hydroelements.

The NOXY method does not enable interpolations between verticals. For the same vertical density, it is not as easy to reconstruct the hydraulic continuity as it was with the previous methods, which means that the calculated hydrosignatures are less accurate. Other methods were therefore developed by Scharl & Le Coarer in 2005, who recommend the semi-spatialization of verticals when carrying out on-field measurements. The NOXY3 method recommended in the article called "Morphohydraulic quantification of non spatialized data sets using Hydrosignature software" requires very little time and considerably improves descriptive accuracy.

3 RESULTS

Between the cartographic representation in the horizontal plane of the crossed classes and a surface hydrosignature, the spatial distribution of the hydroelements is lost. However, Figure 8 illustrates that the presence of a managed section can be identified within the general structure of a hydrosignature. In this case, two peak values can be observed in the surface hydrosignature of a stream reach in the Loire River, which represent respectively a channel and a sill.

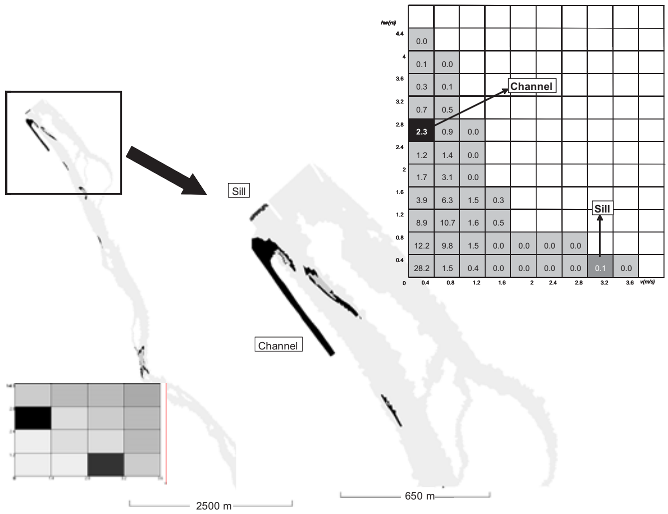

Figure 8. Visualization of the velocities/ depths in the horizontal plane and the surface hydrosignature: focusing on the managed section (7 km of the Loire River at 80 m3/s, EDF data).

4 DISCUSSION

Hydrosignatures make it possible to both quantify the hydraulic diversity and visualize the distribution of the latter in the velocity/ depth plane. The most accurate mathematical solution used to calculate hydrosignatures consists in using a mesh based on spatialized depth/ velocity couples. These calculations can also be performed using cross sections or non spatialized data. The databases that represent these verticals can come from measurements or hydraulic & statistical modeling.

Considering that the spatial distribution of aquatic organisms is directly related to the hydraulic characteristics, hydrosignatures can be used as a tool in ecohydraulic modeling. As the depth and velocity variables are not independent, it seems wiser to calculate their distribution using bivariate data.

Within the scope of habitat/ fish modeling, the representation of hydroelements in the horizontal plane was used by Baras (1992), whereas Grift (2001) used bubble charts to represent quantitative hydraulics in the velocity/ depth plane.

The in-stream flow incremental methodology (IFIM, Bovee 1982) is widely used in ecohydraulics for fish/ habitat modeling (Parasiewicz & Dunbar 2002). This modeling implements curves that are taken into consideration on an independent basis. We recommend:

- Associating the hydrosignatures corresponding to the sampling that contributed to their construction with preference curves of fish taxa for the variables hw and v,
- Systematically checking the hydrosignatures of the reaches simulated during hydraulic modeling at different flow rates, in order to make sure that the validity range of the fish habitat's biological model is respected.

We hope the HydroSignature software (http://hydrosignature.aix.cemagref.fr/) will enable hydrobiologists to test this method in their own research and studies.

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WEB SITE

http://hydrosignature.aix.cemagref.fr/

The effect of flow regulation on channel geomorphic unit (CGU) composition in the So a River, Slovenia

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ABSTRACT: This paper examines the effects of flow regulation on the size, spatial distribution and connectivity of channel geomorphic units (CGU) of the So a River in Slovenia. A river channel survey was completed along three reaches, i.e. an unregulated stretch (reach 1), and two regulated reaches with lower discharges, (reach 2 and 3).

Results demonstrated significant differences in the CGU composition between the unregulated and regulated reaches. The unregulated stretch was dominated by the glides and relatively fast-flowing and turbulent features whilst regulated reaches were dominated by slow flowing pool CGU's. River regulation also reduced the size of the CGU's. CGU's tended to be shorter, and hence there was greater habitat division or fragmentation evident in the two regulated reaches. Therefore flow regulation in the So a River alters the dominant types of CGU's present, significantly reduces the size of CGU's, and affects the longitudinal distribution of types by reducing habitat connectivity and creating greater habitat fragmentation.

1 INTRODUCTION

Physical habitat plays an important role in determining 'river health' and influencing the structure and function of aquatic communities (Stalnaker 1979, Aadland 1993, Pusey et al. 1993, Maddock 1999, Gehrke and Harris 2000, Maddock et al. 2004). Traditional assessment of both physical habitat and biotic communities (e.g. fish and macroinvertebrate populations) has tended to focus on sampling at discrete points, or along small (i.e. <200m) stretches of river channel ('small' scale). Results from sampling at disparate points are then extrapolated to the sections of river inbetween ('upscaling') to provide catchment wide assessments (at the 'large' scale), or make river management recommendations (e.g. for environmental flows). However, extrapolation without an understanding of the nature of the river between sampling points and hence a knowledge of whether they are truly representative of the river inbetween is questionable.

Furthermore, it has been argued that the an understanding of river systems at the 'intermediate' scale (i.e. 1-100 km's of stream length) may be more appropriate for studies examining physical habitat impacts on fish. Fausch et al. (2002) have argued that river habitat assessment should concentrate on assessing reaches at the 'intermediate' spatial scale rather than at disparate points or representative reaches in order to recognise the river landscape as a spatially continuous longitudinal and lateral mosaic of habitats.

To facilitate this approach, a range of river habitat mapping methods and classification systems have been developed. Surveys are normally completed as part of aquatic habitat modelling studies, either to model physical habitat availability directly from mapping results, or to identify representative reaches for further and more detailed data collection. River habitat mapping aims to identify the types and spatial configuration of geomorphic and hydraulic units. Physical habitat units have been defined and classified by many authors, leading to an array of terms in use to describe the physical environment utilised by the instream biota. The terms used to describe these units differ between authors and include 'channel geomorphic units' (CGU's) (e.g. Hawkins et al. 1993), 'mesohabitats' (e.g. Tickner et al. 2000), 'physical biotopes' (e.g. Padmore 1997) and 'hydraulic biotopes' (e.g. Wadeson 1994). Newson and Newson (2000) provide a review of the use of some of these terms and the differences between them.

Identification and mapping of channel geomorphic units can be accomplished in a variety of ways including in-channel measurements (Jowett 1993) or with the use of air photo interpretation and/or airborne multispectral digital imagery (Hardy and Addley 2001, Whited et al. 2002). The most common approach however is to walk the relevant sector of river and use subjective visual assessment (Hawkins et al. 1993, Maddock et al. 1995, Parasiewicz 2001).

In addition to the need to assess rivers at the most appropriate scale and along continuous reaches, others have called for the translation of key concepts that are well established in landscape ecology to be translated to riverine environments (Wiens 2002). These key concepts include patch dynamics, habitat connectivity, complexity and fragmentation, and the importance of understanding river ecosystems at a range of spatial scales. This requires a shift in traditional ways of conceptualising and sampling river habitats. A recent study examining macroinvertebrate assemblages has demonstrated the importance of this new approach (Heino et al. 2004). River habitat mapping is likely to underpin an understanding of the links between physical habitat dynamics and instream biota in general, and particularly for fish species.

The aim of this paper is to highlight that in addition to the routine use of habitat mapping results (to describe the types, locations and proportions of physical habitats present along a reach), these field data can also be used to evaluate habitat size, connectivity and fragmentation. This is highlighted with the use of a case study to examine the influence of flow regulation on these factors.

2 SITE DETAILS

The So a River rises in the Slovenian Alps, flowing for 95km through Slovenia before crossing into Italy and discharging into the Adriatic Sea. It has a catchment area of 1576 km^2 and is predominantly underlain by limestone, but the lower parts the river run over flysch and quaternary gravels. The So a has a flashy flow regime, with high flows occurring at any time of year. The lowest flows are experienced both in summer and winter months with generally higher snow-fed flows in spring and rain fed flows in autumn. The So a River is well known for the presence of Marble Trout and recreational (white-water rafting) opportunities.

The river is regulated for hydro-power production at the Podsela Dam and Ajba Dam. Water is abstracted from the impoundment upstream from each dam. It then flows along a bypass channel to the hydropower plant and is subsequently augmented back to the river channel further downstream. Therefore, bypassed sections with reduced flows exist below each dam.

No long term flow records are available to describe the pre- and post river regulation flow regimes exactly, but it is clear that the hydro-power scheme abstracts the vast majority of water for long periods of time, leaving by-passed sections of river with greatly reduced flows. Prior to 2001, the highest possible abstraction rate at Podsela Dam was 96 m^3s^{-1} and the measured flow below the Podsela Dam for most of the year is 0.2 m^3s^{-1}. The highest possible abstraction rate at the Ajba Dam is 75 m^3s^{-1} whilst flow releases until 2003 were normally 0.5 m^3s^{-1}.

In order to assess the impact of these reduced flows on physical habitat type, size and fragmentation, three reaches of river were assessed. **Reach 1:** an unregulated 5.14km stretch of the river between Volarje and Tolmin flowing through a broad open floodplain; **Reach 2:** on a 4.20km by-passed section of the river affected by abstraction below the Podsela Dam that flows through a confined river valley bordered by bedrock walls; and **Reach 3:** another regulated part of the river below the Ajba Dam (4.95km long) with a relatively intermediate-sized valley floor. The three reaches are illustrated in Figure 1 below.

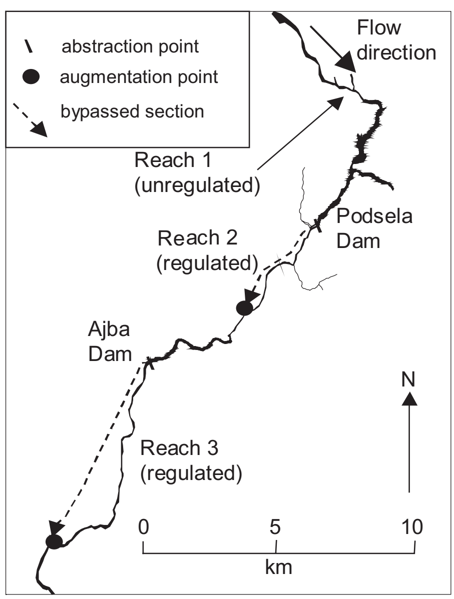

Figure 1. Site Location.

3 METHODS

Each reach was mapped to examine CGU composition and distribution. Mapping was undertaken between 5th–8th July 2004 inclusive, following established procedures (Maddock and Bird, 1996). Each reach was navigated primarily on foot; a small boat was used to traverse the non–wadeable reaches. Field assessment involved a combination of visual assessment and physical measurement. CGU's were identified using a modified version of the Hawkins et al. (1993) classification system. Descriptions of CGU's are highlighted in Table 1.

Table 1. Description of Channel Geomorphic Units (after Hawkins et al. 1993).

CGU (Mesohabitat)	Turbulence	Brief Description
Fall (Fa)	Turbulent & Very Fast	Vertical drops of water over a full span of the channel, commonly found in bedrock and step-pool stream reaches.
Cascade (Ca)	Turbulent & Very Fast	Highly turbulent series of short falls and small scour basins, frequently characterised by very large substrate sizes and a stepped profile; prominent features of bedrock and upland streams.
Chute (Ch)	Turbulent & Very Fast	Narrow steep slots or slides in bedrock.
Rapid (Ra)	Turbulent & Fast	Moderately steep channel units with coarse substrate, but unlike cascades posses a planar rather than stepped profile.
Riffle (Ri)	Turbulent & Moderately Fast	The most common type of turbulent fast water CGU's in low gradient alluvial channels. Substrate is finer (usually gravel) than other fast water turbulent CGU's, and there is less white water, with some substrate breaking the surface.
Run (Ru)	Less Turbulent & Moderately Fast	Moderately fast and shallow gradient with ripples on the surface of the water. Deeper than riffles with little if any substrate breaking the surface.
Glide (Gl)	Non-Turbulent Moderately Slow	Smooth 'glass-like' surface with visible flow movement along the surface; relatively shallow (compared to pools).
Pool (Pl)	Non-Turbulent & Slow	Relatively deep and normally slow flowing, with finer substrate. Usually little surface water movement visible. Can be bounded by shallows (riffles, runs) at the upstream and downstream ends.
Ponded (Pd)	Non-Turbulent & Slow	Water is ponded back upstream by an obstruction, e.g. weir, dam, sluice gate etc.
Other (O)		Used in unusual circumstances where feature does not fit any of recognised types.

Boundaries between each CGU were visually identified from the bankside or boat, and their locations mapped using a Trimble GeoXT 12 channel GPS receiver with sub-metre accuracy. Channel width and water width were recorded to the nearest metre using a Bushnell Yardage Pro distance measurer at a representative point within each CGU.

Substrate sizes present (based on the Wentworth classification) were identified and assigned to 'dominant', 'subdominant' and 'present' categories. Maximum depth for each CGU was estimated to the nearest cm using a measuring staff and the average water column velocity was measured at 0.6 of the water depth from the surface, using a SEBA Mini Current Meter in order to confirm hydraulic characteristics within and between CGU's. The proportion of the surface area of each CGU taken up with instream cover (e.g. instream macrophytes, large woody debris) and overhanging cover (e.g. from overhanging trees and boughs) were visually estimated to the nearest 10 percent. The presence of lateral-, point- and mid-channel bars, their location (e.g. left or right bank), and whether they were vegetated (>50% of surface covered) or unvegetated (<50%) were also noted to provide additional descriptive information. Photographs were taken of each CGU and their numbers recorded.

Using MapInfo 7.5 software, GPS data were combined with digitised maps at 1:50,000 scale and data recorded during the survey to create maps showing the CGU locations. The measured width and length data were used to calculate total water area in each reach and for individual CGU types in each reach.

4 RESULTS

Results demonstrated significant differences in the CGU composition between the unregulated and regulated reaches (Fig. 2).

A reach dominated by fast and turbulent CGU's will have bars focused on the left-hand side of the diagram. As bars become increasingly skewed towards the right-hand side, then this indicates the channel is dominated by slower flowing and non-turbulent CGU's.

The unregulated stretch (reach 1) was dominated by the glides (55%) with the rest of the reach consisting of relatively fast-flowing and turbulent features (runs, riffles and rapids). The dominant feature of both of the regulated reaches were the slow flowing pool CGU's occupying 44% of reach 2, and 76% of reach 3, with glides, runs, riffles and rapids forming the remainder of the CGU's.

Physical measurements of CGU length and water width enabled the calculation of the extent that the reduced discharge in the regulated reaches was dewatering the channel and reducing the size of the CGU's (Table 2).

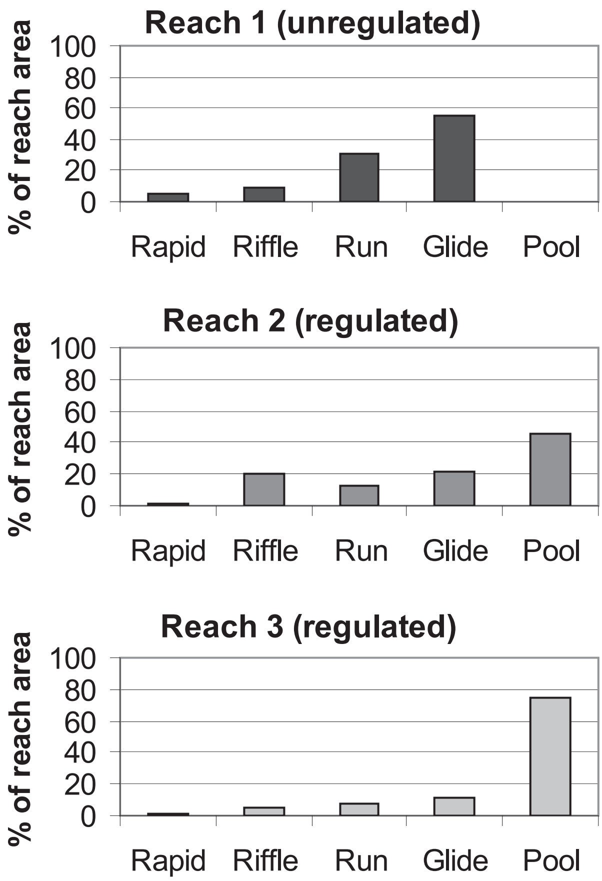

Figure 2. CGU proportions in each reach.

Table 2. Length and average water width of each reach

Reach No.	Length (km)	Average CGU water width (m)
Reach 1 (unregulated)	5.142	58.0
Reach 2 (regulated)	4.195	18.4
Reach 3 (regulated)	4.949	29.2

The average CGU size in the unregulated stretch (reach 1) was 58m wide, compared to 18.4m in reach 2, and 29.2m in reach 3. A direct comparison of CGU size (width and length) is illustrated in Figure 3 below. This highlights the impact of flow regulation in reducing average CGU size in reach 2 and reach 3.

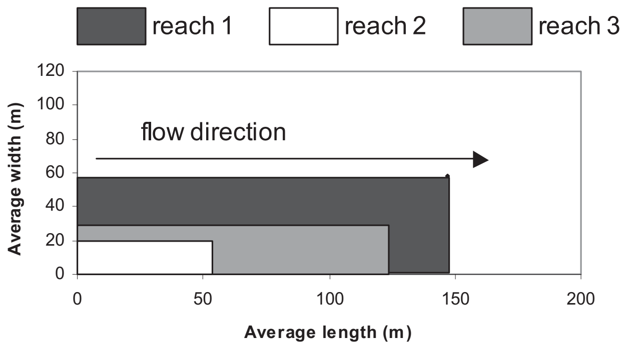

Figure 3. Average length and width characteristics of CGU's in each reach.

In order to examine the effect of regulation on the degree of CGU fragmentation, the average number of units per km was calculated. A relatively large number indicates the channel is dominated by more CGU's and hence they are shorter and more fragmented, whereas a smaller number indicates the reach has fewer units occupying greater longitudinal distances. Results are illustrated for each reach in Table 3 below.

CGU's tended to be shorter, and hence there was greater habitat division or fragmentation evident in the two regulated reaches, particularly reach 2 (18.12 CGU's per km) compared to the unregulated reach (6.81 CGU's per km).

Table 3. Number and fragmentation of CGU's along each reach.

Reach No.	Length (km)	Total number of CGU's along reach	Number of CGU's per km
Reach 1 (unregulated)	5.142	35	6.81
Reach 2 (regulated)	4.195	76	18.12
Reach 3 (regulated)	4.949	40	8.08

5 DISCUSSION AND CONCLUSION

This study demonstrates that when utilising river habitat mapping results in the routine sense, i.e. to examine the types and proportions of CGU's present in discrete reaches, the impacts of river regulation are evident. Using the case study of the So a River, the unregulated reach was dominated by glides and relatively fast-flowing features, whereas the effects of abstraction in the regulated sections created reaches dominated by slow flowing pool type CGU's. The effects of local geomorphology, such as valley gradient and width are also likely to influence CGU presence and when conducting a field-based study such as this, these factors cannot be controlled between reaches. However, reach 1 occupies a broad, wide open floodplain, and reach 2 a narrow confined valley. The confinement in reach 2 may be expected to constrain channel and water width and lead to increased water velocities and a greater proportion of fast flowing turbulent units here. Despite this, the opposite is true; reach 2 has a greater proportion of slow flowing (pool) units than reach 1, demonstrating that the impact of river regulation is evident from habitat mapping results despite influences of channel morphology rather than because of them.

Reduced discharges from abstraction in the downstream reaches (2 and 3) has significantly reduced average water width when compared to the unregulated reach upstream (to 31.8% and 50.4% respectively). More importantly, lower flows have increased the average number of units per km in these stretches. It is possible to interpret this as a positive effect, with increased number of units representing greater physical diversity and therefore likely to support enhanced biodiversity. However, we suggest the overall effect is a negative one, because although regulated reaches are dominated by more units, but these are significantly smaller (narrower and shorter) and are more isolated or fragmented. This effect is illustrated in the Figure 4 where the regulated reach plots in the lower right-hand corner, but increasing abstraction and reduction in flow creates narrower and shorter units and hence reach results plot in the upper left-hand corner.

It is highly likely that there will be a relationship between the diversity (number of types) of CGU's present and flow, the exact nature of which will be partly controlled by local geomorphology. At high flows, reaches will be dominated by a small number of fast and turbulent CGU's (e.g. rapids and runs). At intermediate flows, diversity will higher, with the additional presence of riffles (formerly submerged at high flows), glides and possibly some pools. As flow decline to relatively low flows, CGU diversity will decrease again, with slow flowing and non-turbulent types (glides and pools) dominating, interspersed with runs and riffles at isolated locations where local geomorphology creates an increased gradient. The exact relationship will clearly be controlled by the valley gradient and local geomorphology.

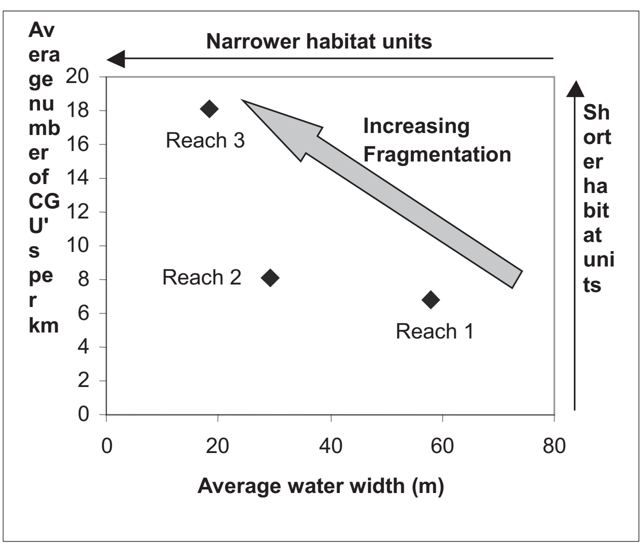

Figure 4. Average width and length relationships as an indicator of habitat fragmentation in each reach.

The preliminary results presented here provide a basis on which to interpret habitat mapping data to compare habitat size and fragmentation along continuous stretches at the intermediate scale. This study suggests that in the So a River under the flow conditions present during the survey, flow regulation alters the dominant types of CGU's present (to slower flowing and less turbulent features), significantly reduces the size of CGU's, and affects the longitudinal distribution of types by reducing habitat connectivity and creating greater habitat fragmentation.

Further research that examines the temporal dynamics of habitat composition along the same reach (and hence negates the impact of different geomorphological controls operating on different reaches) at a range of flows would be very valuable. This may identify critical parts of the flow regime when significant changes in habitat diversity (i.e. how many types of CGU's are present), size and fragmentation occur. This in turn may be useful for environmental flow determination. The objective identification of units is also clearly important in any such assessment and this relies on reliable and repeatable assessment methods. Whilst visual identification from the bankside goes some way to accomplishing this, it is likely that technological advances in the use of remote sensing and airborne multispectral digital imagery (Whited et al. 2002) will increase the speed of data collection. Subsequent image analysis could also enable improved and more robust classification of hydraulic and geomorphic units. More fundamentally, ecological validation of CGU's and the exact requirements of stream communities in terms of habitat size, diversity and fragmentation is required to ensure the relevance of the habitat units being mapped, and to strengthen our knowledge of flow-habit-biota relationships.

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DATS - Data Availability- and Transfer System of COST 626

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ABSTRACT: DATS - Data Availability- and Transfer System of COST 626 is a meta-database search engine for technical, physical and biological data, specially for river and freshwater based data.

The DATS system is running "self-administering" through web interface. It is open to the public. Researchers, end users, authorities and other users have to apply for an account for free. Data input and data search is running form based. The meta-database search engine is based on open source technology (PHP, MySQL, PEAR, htdig).

DATS is a searchable meta-database, were the availability of data of certain specifications can be established. The data itself will not be stored. As a result of the search, a contact to the holder of the data of the search result will be supplied.

Entering the system, the “Start Page” will give first information about DATS. As a first step, every user has to apply for an account for free. After logging in into the DATS system, the “New data” and the “Search” mode will be available. Extendable search- and input forms are based due to a modular concept. The “Members” module give information about account holders and their affiliation.

The “New Data” module incorporates 5 modules for data input. The “Common” module contains input such as country, ecoregion, river, purpose, …. Further on 4 more modules are available for subject areas “Biotic”, split into fish and macro-invertebrate sub-module, “Riverine Vegetation”, “Abiotic” and “Combined”. To run a search for a certain data specification by using the “Search” module, each data input in each of the 5 modules of the data input can be defined as a search key.

As a result, the availability of data sets with data set-specific contents and specifications and a contact address to the holder of the data set will be presented at the end of the search process.

1 GENERAL

1.1 *What is DATS*

DATS – The **D**ata **A**vailabilty and **T**ransfer **S**ystem is a web based meta data base search engine for technical, physical and biological data, specially for river and freshwater based data, which should improve the availability and access of habitat data. DATS is a searchable meta-database, were just the availability of data of certain specifications can be established. The data itself will not be stored. As a result of the search, a contact to the holder of the data of the search result will be supplied. DATS should not provide the data itself, but contact information to persons and organizations, who did the research and store the data. So as to say it is an address-database for scientific purposes.

The DATS system is running "self-administering" through web interface. It is open to the public. Researchers, end users, authorities and other users have to apply for an account for free. Data input and data search is running form based. The meta-database search engine is based on open source technology.
DATS is a result of COST 626 group activities of the last 5 years (2001 – 2005).

1.2 *Technical data of DATS*

DATS is web based and can be used without the need of special skills, neither for administrating user data, nor for data input or data search.

The website is coded using only Open-source technology. The following software and libraries are used:

Apache 2.x webserver http://www.apache.org
PHP 4.x http://www.php.net
MySQL 4.x http://www.mysql.org
SMARTY templates http://smarty.php.net
PEAR library http://pear.php.net
htdig search- and indexengine http://www.htdig.org/

The site is coded in XHTML 1.x and should reach WAI level AA by minimum.

Info: The WAI is the Web Accessibility Initiative of the World Wide Web Consortium W3, which is the standardization organization founded by Tim Berners Lee, one of the founders of the WWW. It's mission is to lead the Web to its full potential includes promoting a high degree of usability for people with disabilities. WAI, in coordination with organizations around the world, pursues accessibility of the Web through five primary areas of work: technology, guidelines, tools, education and outreach, and research and development.

The page is optimized for search engines and should work with the most common browser like Internet Explorer 6.x, Firefox, Mozilla, Netscape 7.x, Opera, Safari, Lynx on most of the operating systems like MS Windows, MacOS, Linux, Solaris.

No java script is used, all the formats are done using Cascading Stylesheets, CSS. The site is done in English only and doesn't use any foreign page encoding algorithm.

1.3 *Licenses*

DATS is published under the GNU General Public License http://www.gnu.org/copyleft/gpl.html and the Mozilla Public License 1.1 http://www.opensource.org/licenses/mozilla1.1.php

2 DATA AVAILABILITY AND TRANSFER SYSTEM - DATS

2.1 *How to use DATS*

Entering the system, the "Start Page" will give first information about DATS. As a first step, every user has to apply for an account for free. After logging in into the DATS system, the "New data" and the "Search" mode will be available. Extendable search- and input forms are based due to a modular concept. The "Members" module give information about account holders and their affiliation.

The websites comes with the following sections:

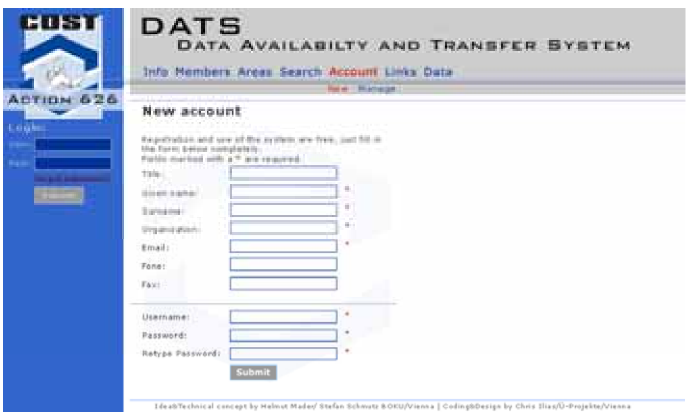
Info:
    Holds the main information about DATS
Members:
    A list of the participants and registered users of this system (searchable)
Areas:
    A list of probed areas and regions (searchable)
Search:
    Search for available data
Account:
    Apply for an account and manage account data in case of changes
Links:
    Web links to other sites (searchable)
Data:
    Input new, change and delete existing data
Admin (hidden):

Section to manage all input data like user accounts, only for members of the group admins. This menu item will only be displayed if you log in as an admin! This section uses encripted transfer under SSL.

You don't have to be logged in to use the system except you want to manage your account data or contribute new data.

2.2 Apply for an account

If you want to contribute data to the system, you have to apply for an account. Just fill in the required data marked with an *. After the form has been submitted, there will be sent an email to the given address, containing all the given data and the chosen username and password. The email also contains a web-link which has to be visited in order to activate the account, otherwise the account is locked and you can't login. This has to be done to validate the email address and to impede fraud and data theft. There can only be one person registered per email address!



Users can only change their own data unless they are members of the group admins.

2.3 Data input

Users have to be logged in to contribute new data. The “New Data” module incorporates 5 modules for data input. The “Common” module contains input such as country, ecoregion, river, purpose, …. Further on 4 more modules are available for subject areas “Biotic”, split into fish and macro-invertebrate sub-module, “Riverine Vegetation”, “Abiotic” and “Combined”.

Users have to fill in all the forms, or just parts of it. For example if there are only biotic data available, users just have to fill in the required fields, marked with an *, in the subsections "Common" and "Biotic" and skip the other subsections.

Users can only change the data, which they themselves contributed to the systems. Data of other users can only be changed by the administrators. All data inputs and changes will be logged in the database. These data contain the IP-address, the timestamp, the username and the action taken.

If a user has forgotten his password, there should be selected the link "Forgot password?". After entering the email address there will be a new password set and sent by email, if the supplied email address was listed in the database. Attention: Data fields may change during the design process.

2.4 Data Search function

To run a search for a certain data specification by using the “Search” module, each data input in each of the 5 modules of the data input can be defined as a search key.

Users don't have to be logged in to search the database. Every subsection can be searched for itself, or there can be done a combined search defining criteria for all subsections.

After submitting the search criteria, a list of results will be displayed ordered best matching the given criteria. In the list will be displayed the contact data of the person, who contributed the data to the system.

As a final result, the availability of data sets with data set-specific contents and specifications and a contact address to the holder of the data set will be presented at the end of the search process. Users have to contact the holder of the data set by email given in the result page and ask for submission of required data sets.

3 SEARCH AND INPUT CRITERIAS

To run a search out of or an input of data in DATS, the following criterias in each of the 5 modules are defined as a input or search key.

3.1 Meta-Database "Common"

The meta-database common will contain the following criteria for data input and search mode:

Country
Date of probe
River/water
Section/reach
Ecoregion
Purpose
River status
- natural reference
- modified hydrology
- modified morphology
- modified water quality

Publication
Data form

Analogue
Digital

3.2 *Meta-Database "Biotic"*

Meta-Database Biotic is split into fish and macro-invertebrate sub-database.

3.2.1 Sub-Meta-Database FISH

The sub-meta-database fish will contain the following criteria for data input and search mode:

Type of study
- Experimental (e.g. experimental channels, treatments)
- Empirical (e.g. monitoring)

Sampling method
- Single species (Taxa list of FAME) – pull down menu (multiple selections possible)
- Community pull down menu for dominant species (multiple selections possible)
- Exotic species (box to click)

- Snorkeling Scuba
- Visually from above water
- Viewing box
- Video camera
- Electrofishing
- Drift nets
- Radio/Acoustic telemetry
- PIT tagging
- Echo sounding
- Seines, traps, drop nets, explosives
- Experimental channel

Sampling area
- Point (point abundance sampling)
- Partial (e.g. mesohabitat)
- Total profile

Sampling intensity
- <100m
- >100m
- >1 km
- < 10 min
- < 1 h
- > 1 h

Type of data (select box)
- Presence/absence

CPUE (catch per unit effort) - semi quantitative
Quantitative data - density
Quantitative data – biomass
Fish length data
Fish morphology
Abnormalities (diseases, parasites, deformations)
Life stage (eggs, larval, 0+, juvenile, adult, spawning)

Sampling period
< week
months
season
1 year
> 1 year

3.2.2 Sub-Meta-Database MACROINVERTEBRATES

The sub-meta-database macroinvertebrates will contain the following criteria for data input and search mode:

Type of study
Experimental (e.g. experimental channels, treatments)
Empirical (e.g. monitoring, survey)
Sampling method (e.g. RIVPACS, AQEM; Surber; drift; artificial samplers)
Qualitative
Semi-quantitative
Quantitative data
Mesh size
Single habitat or multi habitat
Single family
Community pull down menu for dominant families
Exotic species
Abnormalities (deformations)
Species morphology
Life stage (larval/nymph, pupae, adults)
Season (all year; winter; spring; summer; autumn)

3.3 Meta-Database"Riverrine Vegetation"

The meta-database riverrine vegetation will contain the following criteria for data input and search mode:

Quality & applied methods
Potential vegetation
Real vegetation
Vegetation map
Relevee
Plant association
Vegetations types

List of species
Detailed ecophysiological information (scientific name of species)
Detailed investigations on abiotic conditions
Soilmoisture/Water balance
Soil and substrate types
Light
Water quality
Inundation

3.4 Meta-Database "Abiotic"

The meta-database abiotic will contain the following criteria for data input and search mode:

Scale
- Macro
- Meso
- Micro

River size
Catchment area
Altitude
Hydrologic regime
Discharge
- Mean
- Flood
- Low

Geology
Stream order
Maximum flow length
River type
- Lowland
- Sub-alpine
- Alpine
- Meandering
- Straight
- Braiding

Temperature
Meteorological data
Slope
Cover
Velocity
- Measuring method
- Instrumentation
- Time period
- Resolution
- Dimension (1D, 2D, 3D)
- Post processing (point, cell, equivelocity contours, equivelocity volumes)

Substrate

Method
Shear stress
Bed load
Suspended Load
Grain size
Morphology
Instrumentation
Regular grid
Irregular grid
Transects
Depth
Measuring method
Instrumentation
Resolution
Width
Measuring method
Instrumentation
Resolution
Morphodynamic data
Hydraulics
Measured
Modelled

3.5 Meta-Database "Combined"

The meta-database combined will contain a combined search defining criteria for all subsections.
For data input every subsection has to be used for itself.

4 RESULT

DATS only gives the information about the availability of data sets with data set-specific contents and specifications and the contact address of the holder of the data set.

Users have to contact the holder of the data set by email given in the result page and ask for submission of required data sets.

Data set holders have to define rights on data use by themselves in direct contact to interested users.

Weighted Usable Volume Habitat Modeling - The Real World Calculation of Livable Space

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ABSTRACT: Habitat research goes back for several decades. The importance of diversify available habitat for aquatic biota has been shown by many authors throughout the last years. In order to understand complex natural river structures and to analyze functional interactions between aquatic biota and abiotic components of the environment many methodologies were developed with the main goal to provide practical tools for river management.

Within these models normally the "Weighted Usable Area" (WUA) for aquatic organisms under specified environmental conditions are calculated. But, weighted usable area as a result of mean velocity data in verticals or transects do not describe the real world situation of velocity distributions in rivers cause of data smoothing.

Instream flow modelers are nowadays increasingly utilizing 2D and 3D hydrodynamic modeling approaches to assess aquatic habitat versus discharge relationships. Out of technical increase of instrumentation, resolution and sampling strategies, new possibilities out of the use of acoustic Doppler or laser 3D profilers open new vistas within hydraulic and habitat modeling. As an example the linkage between energy intake and use, which is used in bio-energetic models, necessitates a very detailed description of the study site with a 3D velocity field.

For further development in habitat modeling, the use of WUV (Weighted Usable Volume) as a result of a special velocity distribution, which describes the real world of living space without data smoothing consequentially is a must.

The new **HA**bitat **MO**delling **SOFT**ware (HaMoSoft) has been developed to analyse basis data (like measured velocity fields or velocity results from 3D hydraulic models, geometry, substrate and cover) with a 3D output value termed "Weighted Usable Volume" (WUV). The great difference to the existing habitat models is, that the measures or simulated velocity basis data will not be reduced to mean values in verticals or transects.

The spatial velocity distribution (measured or simulated point or cell velocities) will be used with the whole information. Equi-velocity contours in transects and/or horizontal layers will be calculated out of measured or simulated point/cell velocities for further use in the new habitat calculation model. Within this 4 parametric habitat analyzing software, using and combining velocity distribution, depth, substrate and cover, the real world of habitat distribution is analyzed spatially. The analyzed WUV describe the real world situation of living space within nature like rivers and streams.

1 INTRODUCTION

In the last decades the demand for ecological studies of flowing waters grew significant. The aim and the purpose of such studies is on the one hand to assess the human impact on flowing water ecosystems and on the other hand to make an attempt for fully understanding the complex natural river structures. Habitat (or „liveable space") as the local physical, chemical and biological features that provide an environment for the instream biota is affected by instream and surrounding topographical features and is a major determinant of aquatic community potential. Hydro-morphological and hydraulic parameters are the major factors influencing abundance and distribution of aquatic organisms in rivers and streams. Habitat model output is as good and correct as the model input (hydraulics, morphology). Mainly used parameters for describing fish habitat are mean velocity, depth, substrate and cover. Traditionally the description of the habitat quality (PHABSIM approach) is via WUA (weighted usable area) and SI (suitability index). Mean cross section velocities nowadays are often/mostly used as key factors at meso habitat scale. This reduction of measured or simulated point or cell velocities (2 D or 3 D vectors) to mean values for cells or transects results in a dramatic loss of information caused by data smoothing.

Bain and Stevenson (1999) give an overview of the sampling strategies, the data need, the limitations and the possibilities of abiotic data sampling for further use in fish habitat studies. Instream flow modelers are nowadays increasingly utilizing 2 D and 3 D hydrodynamic modeling approaches to assess aquatic habitat versus discharge relationships (Addley, R.C., Hardy, T.B. 2002). Out of technical increase of instrumentation, resolution and sampling strategies, new possibilities out of the use of acoustic Doppler or laser 3 D profilers open new vistas within hydraulic and habitat modeling. For further development in habitat modeling, the use of WUV (weighted usable volume) as a result of a special velocity distribution, which describes the real world of living space without data smoothing consequentially is a must.

Recent research demonstrates the need of reasonably accurate habitat models based on the real world situation of habitat abiotics for both, a better understanding of the interactions between aquatic biota and their environment and to provide practical tools for river management.

2 PROJECT AIM

The main object of the project is to develop new modeling software components for a further combination with hydraulic- and habitat models to analyze the basis data (like measured velocity fields or velocity results from 3 D hydraulic models, geometry, substrate and cover) for sophisticated habitat modeling with a 3 D output value termed "Weighted Usable Volume" (WUV).

The principal object is to set up new software components for standardized calculation methods of physical parameters to simulate and analyze available living space.

The great difference and step forward to existing habitat models is, that the measures or simulated velocity basis data will not be reduced to mean values in verticals or transects. Equivelocity contours in transects and/or horizontal layers will be calculated out of measured or simulated point/cell velocities for further use in the new habitat calculation model. The analyzed WUV describe the real world situation of living space within nature like rivers and streams (Mader 1999).

Spatial organization of abiotic habitat parameters, specially the spatial distribution of the flow velocity in a river system or reach is very important to fish population dynamics as fish utilize different habitats at different life history stages and during different seasons (Kocik and Ferreri 1998, Zauner and Eberstaller 1999).

The important and decisive difference between the existing habitat models and the new strategy in habitat modelling is the integration of the whole information of spatial distribution of the flow velocity within a river system or reach into a habitat model by using WUV (weighted usable volumes).

3 METHODOLOGY AND APPLICATION OF THE TOOL „HAMOSOFT"

The habitat modelling tool "HaMoSoft" to analyse the real world situation of liveable space consists of the following modules:

Geo-statistic Data Processing Module
Input Data Module
Equi-Area /Equi-Volume Calculation Module

The individual modules are embedded in an "Excel-Macro-Program-Code" written in "Visual Basic for Application". To use and execute / apply the tool "HaMoSoft" it is necessary that the software Microsoft Excel and Surfer (Golden Software) is installed on the computer.

3.1 Geo-statistical data processing module

New techniques of simultaneous velocity measurement in irregular grid (Mader et.al 2003) makes it necessary, that irregular spatial distributed data have to be post processed and brought into line for creating a transect oriented input data file for further analysis (see capt. 3.2). Specially the velocity data as a result of very different measurement strategies (standard strategies s.a. point sampling, profile sampling or advanced strategies s.a. simultaneous sampling) and the results of 3 D hydraulic models have to be processed before further calculation (DELFT3 D, SSIIM, …).

The aim of processing of geo-statistical data is to show, in our case, some possible statistical solutions of a given problem. This process concentrates on the numerical studies and solution of the algorithm that results in finding the estimators of parameters, which sufficiently describe the model of a stochastic distribution of flow of water in the river basin from which various conclusions can be drawn. In order to cope with such a complex problem, a "powerful" part of the statistical theory, which is called nonlinear statistical models, will be utilized whose outcomes predict, on a certain level of probability, a behavior of the studied problem in which we are interested in.

The whole process of determining the velocity distribution can be generally based on the following model (1.1):

$$Y = f(\beta) + \varepsilon\,, \beta \in R^k$$

or equivalently described by (1.1.a):

$$Y \approx_n [f(\beta), \Sigma], \beta \in R^k$$

In both cases, a describing function $f(\beta)$, where β is a random vector, represents the nonlinear function which characterizes the general behavior of the liquid material. From the statistical point of view, the main goal of the above given problem is to find the mean value and the covariance matrix of the nonlinear vector function $f(\beta)$. In fact it is the nonlinear error propagation law. Generally, this task does not need to have a solution. There are two possible ways how to find the proper answer.

- The first one is to linearize the given model (1.1) or (1.1.a) with the help of an acceptable transformation.
- The second way is to linearize the above given model (1.1) or (1.1.a) by finding a Taylor expansion in an arbitrary point.

Our geo-statistical data processing will follow the latter method. The objective of this method is based, as mentioned before, on a construction of the Taylor expansion of the nonlinear model (1.1.) or (1.1.a) in an arbitrary point. The whole process is given by a following procedure.

We have given the model in the form

$$Y = f(\beta) + \varepsilon\,, \beta \in R^k$$

If β_0 is a known point, we can then make the Taylor expansion in the point β_0 in which we will neglect the terms of the second and higher order. This implies that we are able to make a linear estimation of unknown parameters and in case of normality, we can also use the standard principles for the construction of the confidence areas and hypothesis testing.

For the function above, we will generate the Taylor expansion in an arbitrary point β_0 which is given by the next expression (1.2.):

$$f(\beta_1) = f(\beta_0) + F(\beta_1 - \beta_0)\,,\ \beta_i \in R^k\,,\ i = 0,1$$

According to the theory of measurement, we have to define a matrix *F* which is given by the following formula (1.3.):

$$F = \left(\frac{\partial f}{\partial \beta'} \right)$$

Now we have a linear approximation of the given problem and we can use the standard data processing. Based on the above given expressions, we can rewrite our model in the form of (1.4.)

$$Y = f^{(O)} + F\delta\beta + \varepsilon \cong f(\beta) + \varepsilon\,,\ \beta_i \in R^k\,, i = 0,1\,,$$
$$\operatorname{var}(\varepsilon) = \Sigma\,.$$

The best linear unbiased estimator of the vector of parameters β can be achieved with the help of the next formula (1.5.):

$$\delta\hat{\beta} = \left(F'\Sigma^{-1}F\right)^{-1} F'\Sigma^{-1}\left(Y - f^{(0)}\right)$$

By using (1.2.) and (1.5.) we can write (1.6.)

$$\hat{\beta} = \beta^{(0)} + \delta\hat{\beta}$$

To make this process clearer, we will use all these results to the given problem of the determining a stochastic distribution of a flow of water in the river basin. Firstly, we have to create a suitable model which will describe the behavior in the river basin. The first step is to find the “river line” which will represent the places with the highest water speed. We will arrive at the following expressions (1.7.):

$$x = x,$$
$$y = \beta_1(x) = \sum_{i=1}^{n} \gamma_i x^{i-1},$$
$$z = \beta_2(x) = \sum_{j=1}^{m} \delta_j x^{j-1}.$$

Using (1.7.), we are able to construct the model describing the velocity distribution, i.e. (1.8.)

$$E[Y_i] = v_0 - \left(\frac{y_i - \hat{y}}{\cos\beta} \right)^2 \kappa_1 - \left(\frac{z_i - \hat{z}}{1} \right)^2 \kappa_2$$

where

Y_i …is the measured velocity in the point (x_i, y_i, z_i);
v_0 …is the velocity measured for the x-projection point of the „river line" according to the coordinate x_i of the point (x_i, y_i, z_i). This value could be also estimated separately;

$y = p_1(x)$, where $\widehat{x}$ is the x-coordinate of the projection of the point (x_i, y_i, z_i) on the „river line";

$z = p_2(x)$, where $\widehat{x}$ is the x-coordinate of the projection of the point (x_i, y_i, z_i) on the „river line";

$$\cos\beta = \frac{1}{\sqrt{1+\left(\frac{\mathrm{d}\beta_1(\widehat{x})}{\mathrm{d}\widehat{x}}\right)^2}}$$

and κ_1 and κ_2 denote the coefficients of the viscosity of the water.

This represents the model of the form of

$$Y = f(\beta) + \varepsilon, \beta \in R^k$$

where

$$v_0 - \left(\frac{y_i - \widehat{y}}{\cos\beta}\right)^2 \kappa_1 - \left(\frac{z_i - \widehat{z}}{1}\right)^2 \kappa_2$$

is the nonlinear function $f(\beta)$. To write the whole model more digestedly, we will use all the above-mentioned formulas and finally arrive at the model described by an expression of the form of

$$E[Y_i] = v_0 - \left(\frac{ y_i - \left(\frac{ x_i + y_i \frac{d \sum_{i=1}^{n} \gamma_i \widehat{x}^{i-1}}{d\widehat{x}} + z_i \frac{d \sum_{j=1}^{m} \delta_j \widehat{x}^{j-1}}{d\widehat{x}} }{ 1 + \left(\frac{d \sum_{i=1}^{n} \gamma_i \widehat{x}^{i-1}}{d\widehat{x}} \right)^2 + \left(\frac{d \sum_{j=1}^{m} \delta_j \widehat{x}^{j-1}}{d\widehat{x}} \right)^2 } \right) \sum_{i=1}^{n} \gamma_i \widehat{x}^{i-1} }{ \frac{1}{\sqrt{1 + \left(\frac{d \sum_{i=1}^{n} \gamma_i \widehat{x}^{i-1}}{d\widehat{x}} \right)^2}} } \right)^2 \kappa_1 - \left(\frac{ z_i - \left(\frac{ x_i + y_i \frac{d \sum_{i=1}^{n} \gamma_i \widehat{x}^{i-1}}{d\widehat{x}} + z_i \frac{d \sum_{j=1}^{m} \delta_j \widehat{x}^{j-1}}{d\widehat{x}} }{ 1 + \left(\frac{d \sum_{i=1}^{n} \gamma_i \widehat{x}^{i-1}}{d\widehat{x}} \right)^2 + \left(\frac{d \sum_{j=1}^{m} \delta_j \widehat{x}^{j-1}}{d\widehat{x}} \right)^2 } \right) \sum_{j=1}^{m} \delta_j \widehat{x}^{j-1} }{1} \right)^2 \kappa_2$$

For lucidity, we will use the notation

$$E[Y_i] = f(v_0, \kappa_1, \kappa_2, \gamma_i, \delta_j) = f(\beta), i = 1 \ldots n, j = 1 \ldots m$$

With the help of formula (1.6.) and (1.7.), we can estimate the value of the parameter $\hat{\beta} = \beta^{(0)} + \delta\hat{\beta}$, where we have to define the following matrix:

$$F = \left(\frac{\partial f}{\partial \beta'} \right) = \left(\frac{\partial f}{\partial v_0}, \frac{\partial f}{\partial \kappa_1}, \frac{\partial f}{\partial \kappa_2}, \frac{\partial f}{\partial \gamma_i}, \frac{\partial f}{\partial \delta_j} \right)', i = 1 \ldots n, j = 1 \ldots m$$

Finally, the whole process of determining a stochastic distribution of a flow of water in the river basin can be summarized into four steps:

With the use of the measured data, we will estimate the values of unknown parameters appearing in the vector

$$\beta = (v_0, \kappa_1, \kappa_2, \gamma_i, \delta_j), i = 1 \ldots n, j = 1 \ldots m$$

We will generate a new grid of points in which we will subsequently calculate further values of the velocity;
According to data acquired in the second step, we will apply the described statistics on these data in order to find out a stochastic distribution of a flow of water in the river basin;
Finally, we will depict all these results graphically.

3.2 Input data module

To start the modelling process a so called "Input-File" is necessary for feeding the computer with raw data. This special transect oriented data file includes all required information and consists of geodesy data of the river section (3 D terrain model), flow velocity data (point or profile sampling, velocity data from 3 D hydraulic models s.a. SSIIM), substrate and cover data (longitudinal and lateral extension).

At the one hand side, input data is taken out of transect measured field data or simulated 3 D hydraulic model data with transect oriented output (s.a. SSIIM). On the other hand, input data file has to be created out of irregular measured grid as a result of geo-statistic data processing (see capt. 3.1).

According to the available raw data in the input-data module and the desired type of output information of the habitat modelling tool there are several possibilities to control and choose the way of the modelling process (Fig. 1).

Inputdata - Velocity		**Analysing Method**
Cross-section Data	→	UA (usable Areas) out of mean vertical Velocities
Cross-section Data	→	UA & UV (usable Volumes) with 1, 2 or 3-D Velocities
Irregular distributed Data	→	UA & UV (usable Volumes) with 1, 2 or 3-D Velocities
3-D Hydraulic Model Data	→	UA & UV (usable Volumes) with 1, 2 or 3-D Velocities

Figure 1: Example of the combination of velocity input-data and possible analysing methods

3.3 *Equi-Area /Equi-Volume Calculation Module*

After starting the tool the user interface of the program on the one hand needs further information like

velocity- and depth classification (v-range, depth-cell),
velocity analysis (v_x, $v_{x,y,z}$),
gridding method (Triangulation with linear interpolation, Kriging),
grid-spacing and

on the other hand it allows to choose options like

analysing only the flow velocity,
analysing the combination of velocity and depth,
analysing the combination of velocity, depth and substrate and
analysing the combination of velocity, depth, substrate and cover.

Figure 2 shows the systematic design of the 1 to 4 D matrix with possible parameter combinations (velocity, depth, substrate, cover).

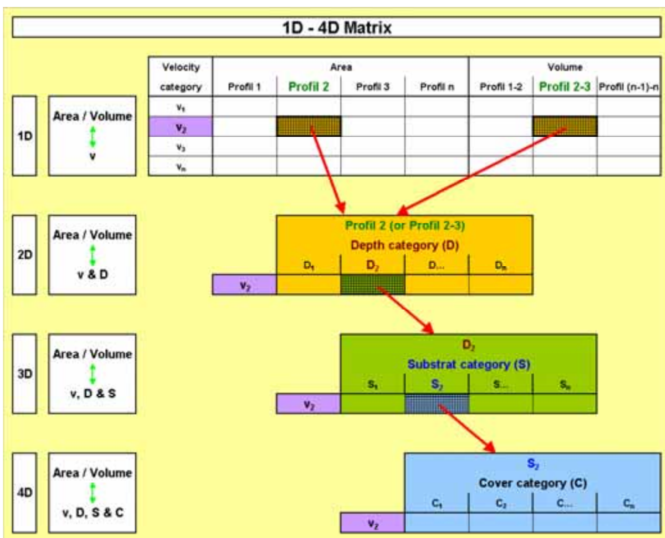

Figure 2: “Output-File” for the usable areas and volumes, 1-D to 4 D matrix

Combining the usable volumes with existing biotic data (species & life stage specific SI – curves) within existing habitat models, the “Weighted Usable Volume” for single or combined parameter calculations will be calculated for pico, micro or meso habitat structures as the real world of liveable space.

Further on, abiotic parameter at different discharges out of field measurements or 3 D hydraulic simulation processes will be analyzed (WUV_{Q1}, WUV_{Q2}, … WUV_{Qn}).

As a final step of the calculation the real world of liveable space and its within-year fluctuation at present situation (natural river, reconstructed river, channel like river, minimum flow demand, ...) is shown by analysing different discharge situations (annual or seasonal hydrograph) out of the hydrograph of the test site (Fig. 3).

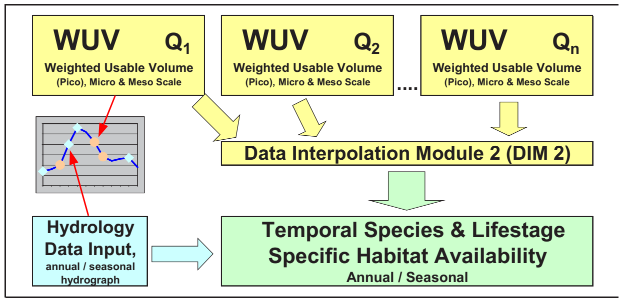

Figure 3: Final result of the sophisticated habitat modelling – the real world of liveable space and its within-year fluctuation

4 RESULTS

The sophisticated habitat modeling output is evaluated within data analyses (measured versus simulated) at the "laboratory creek", a 1 : 1 physical scale model of the Myra River south of Vienna with a river length of 20 m, a river width of up to 4 m, water depth 0,2 – 0,8 m with controlled flow.

Interpolated habitat describing parameters are calculated as equi – value areas in transects and/or horizontal layers and further on volumes of equi - value classes of parameters will be calculated stretch-wise or for natural habitat structures. Further results are graphic data output files as illustrate in figure 4.

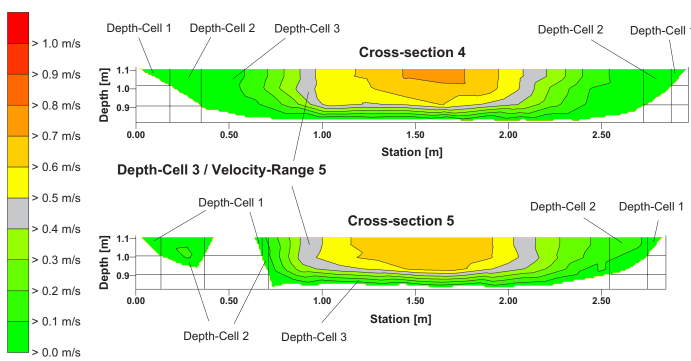

Figure 4: Graphic data output - velocity distribution in cross-section plots

The calculation of the volume of all cross-sections (function of area and distance) for each depth-cell and velocity-range combination results in the total volume [m³ and %].

The 1 to 4 D matrix of usable volumes of single parameters (velocity, depth, cover or substrate) or parameter-combinations (velocity & depth & cover &

substrate) is the most important output file of the new habitat modeling software "HaMoSoft" (Fig. 5).

vdsc-Volume Cross-section 4-16

Depthcell	Substrate	Cover	Velocity range [m/s]										Σ
[m/s]			0,00 - 0,10	0,10 - 0,20	0,20 - 0,30	0,30 - 0,40	0,40 - 0,50	0,50 - 0,60	0,60 - 0,70	0,70 - 0,80	0,80 - 0,90	0,90 - 1,00	[m³]
0,2 - 0,3	Psammal	no	0,0203	0,0275	0,0276	0,0390	0,0448	0,0705	0,1233	0,0417			0,3947
		low	0,0075	0,0128	0,0160	0,0147	0,0131	0,0156	0,0431	0,0034			0,1263
		medium	0,0073	0,0108	0,0009								0,0190
		high	0,0714	0,0208	0,0016	0,0026	0,0052	0,0085					0,1101
		only	0,0278	0,0011	0,0014	0,0016	0,0002						0,0321
	Σ	[m³]	0,1343	0,0730	0,0475	0,0579	0,0633	0,0946	0,1665	0,0452	0,0000	0,0000	0,6822
	Akal	no	0,0043	0,0061	0,0062	0,0062	0,0062	0,0089	0,0643	0,0037			0,1059
		low	0,0071	0,0112	0,0237	0,0055							0,0476
		medium	0,0414	0,0566	0,0377	0,0203	0,0195	0,0154	0,0097	0,0039			0,2045
		high	0,0902	0,0350	0,0161	0,0285	0,0205	0,0084					0,1988
		only	0,0267	0,0695	0,0004	0,0009							0,0975
	Σ	[m³]	0,1697	0,1785	0,0841	0,0615	0,0462	0,0327	0,0740	0,0076	0,0000	0,0000	0,6543
	Mikrolithal	no	0,1275	0,1425	0,0946	0,0792	0,0701	0,0845	0,0759	0,0305			0,7048
		low	0,0047	0,0050	0,0045	0,0059	0,0220	0,0240					0,0660
		medium	0,0016	0,0019	0,0038	0,0061	0,0001						0,0135
		high											0,0000
		only											0,0000
	Σ	[m³]	0,1338	0,1494	0,1028	0,0912	0,0922	0,1085	0,0759	0,0305	0,0000	0,0000	0,7843
	Mesolithal	no	0,1077	0,0027	0,0024	0,0024	0,0035	0,0073	0,0068				0,1327
		low	0,0094	0,0104	0,0097	0,0182	0,0171	0,0126					0,0773
		medium	0,0444	0,0518	0,0040								0,1002
		high	0,0070	0,0056									0,0126
		only											0,0000
	Σ	[m³]	0,1685	0,0704	0,0160	0,0206	0,0206	0,0198	0,0068	0,0000	0,0000	0,0000	0,3227
	Makrolithal	no	0,3088	0,0556	0,0066	0,0013							0,3722
		low	0,0700	0,0394	0,0122	0,0104	0,0074						0,1395
		medium	0,0145	0,0209	0,0056								0,0410
		high	0,0282	0,0079									0,0361
		only											0,0000
	Summe	[m³]	0,4215	0,1238	0,0244	0,0117	0,0074	0,0000	0,0000	0,0000	0,0000	0,0000	0,5888
	Megalithal	no											0,0000
		low	0,0081										0,0081
		medium	0,0384	0,0227									0,0611
		high	0,0318	0,0014									0,0332
		only	0,0040										0,0040
	Σ	[m³]	0,0823	0,0241	0,0000	0,0000	0,0000	0,0000	0,0000	0,0000	0,0000	0,0000	0,1064
Σ		[m³]	1,1101	0,6192	0,2748	0,2429	0,2297	0,2556	0,3231	0,0833	0,0000	0,0000	3,1386

Figure 5: Matrix output of useable volumes of parameter combinations Velocity/Depth/Cover/Substrate. Example for depth-cell 0.2 - 0.3 m.

Out of the combination cell by cell with species & life stage specific SI – curves the Weighted Usable Volume" WUV will be calculated for each required discharge of the hydrograph.

The comparison of the results of the new way of habitat calculation using the WUV method against the old system dealing with WUA (weighted usable areas) show the dramatically loss of information referring to specific habitats (stagnophilic species, life history stage, spatio-temporal habitat requirement) within conventional models.

The new way of a sophisticated habitat modeling is a very important and useful tool of the implementation of the recent demand of the European Water Framework Directive (reestablishing good quality of rivers, monitoring strategies for regarding the effects of diverting water, minimum flow analyzes, effects of river restoration measures, deficit analyzes).

4.1 Usable Area vs. Usable Volume method

First results demonstrate significant differences between the old habitat modelling method dealing with **U**sable **A**reas (UA) and the new HaMoSoft method dealing with **U**sable **V**olumes (UV), particularly the combination of velocity and depth.

Figure 6 shows the output of the usable area within the test reach (UA in %), figure 7 shows the result of usable volumes (in %).

Area [%]

Depthcell [m]	Velocity range [m/s] 0,00 - 0,10	0,10 - 0,20	0,20 - 0,30	0,30 - 0,40	0,40 - 0,50	0,50 - 0,60	0,60 - 0,70	0,70 - 0,80	0,80 - 0,90	0,90 - 1,00	> 1,00	Summe [%]
0 - 0,1	11,3	0,5	0,2	0,1	0,0	0,0	0,0	0,0	0,0	0,0	0,0	**12,1**
0,1 - 0,2	8,5	2,2	0,9	0,3	0,1	0,0	0,0	0,0	0,0	0,0	0,0	**11,9**
0,2 - 0,3	8,2	6,0	2,7	3,0	4,2	5,5	0,0	0,0	0,0	0,0	0,0	**29,5**
0,3 - 0,4	5,2	3,4	0,9	0,6	1,4	3,2	0,0	0,0	0,0	0,0	0,0	**14,7**
0,4 - 0,5	3,0	1,6	1,3	0,7	1,1	1,8	0,0	0,0	0,0	0,0	0,0	**9,5**
0,5 - 0,6	2,7	1,5	1,0	2,9	0,2	0,0	0,0	0,0	0,0	0,0	0,0	**8,3**
0,6 - 0,7	1,9	2,5	2,4	1,4	0,1	0,0	0,0	0,0	0,0	0,0	0,0	**8,4**
0,7 - 0,8	0,4	2,0	1,2	1,2	0,4	0,0	0,0	0,0	0,0	0,0	0,0	**5,1**
0,8 - 0,9	0,0	0,2	0,1	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	**0,3**
Summe	**41,2**	**20,0**	**10,6**	**10,1**	**7,5**	**10,5**	**0,0**	**0,0**	**0,0**	**0,0**	**0,0**	**100,0**

Figure 6: Result of test reach, **U**sable **A**rea (UA) in %

Volume [%]

Depthcell [m]	Velocity range [m/s] 0,00 - 0,10	0,10 - 0,20	0,20 - 0,30	0,30 - 0,40	0,40 - 0,50	0,50 - 0,60	0,60 - 0,70	0,70 - 0,80	0,80 - 0,90	0,90 - 1,00	> 1,00	[%]
0 - 0,1	1,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	**1,1**
0,1 - 0,2	2,9	0,7	0,2	0,1	0,0	0,0	0,0	0,0	0,0	0,0	0,0	**3,8**
0,2 - 0,3	8,3	4,8	2,1	1,9	1,8	2,0	2,5	0,7	0,0	0,0	0,0	**24,0**
0,3 - 0,4	5,8	3,2	0,9	0,6	0,7	1,5	1,5	0,3	0,0	0,0	0,0	**14,6**
0,4 - 0,5	4,3	2,0	1,5	0,9	0,7	1,4	1,6	0,0	0,0	0,0	0,0	**12,5**
0,5 - 0,6	3,7	2,4	2,1	3,5	2,0	0,2	0,0	0,0	0,0	0,0	0,0	**14,0**
0,6 - 0,7	5,2	3,7	2,9	2,9	1,3	0,1	0,0	0,0	0,0	0,0	0,0	**16,0**
0,7 - 0,8	1,5	4,3	1,5	1,7	2,0	0,3	0,0	0,0	0,0	0,0	0,0	**11,3**
0,8 - 0,9	0,4	0,5	1,2	0,6	0,0	0,0	0,0	0,0	0,0	0,0	0,0	**2,7**
	33,1	**21,6**	**12,5**	**12,2**	**8,6**	**5,4**	**5,7**	**1,0**	**0,0**	**0,0**	**0,0**	**100,0**

Figure 7: Result of test reach, **U**sable **V**olume (UV) in %

Calculating **U**sable **A**reas (UA), the sum of habitats within depth-cell 1 is 12.1% of the total habitat.

Usable **V**olume (UV) calculation shows for depth-cell 1 just 1.1% of the total habitat of the test reach.

Looking at the distribution of available habitat in relation to depth, UA – method is overestimating shallow areas and underestimating deep areas of the test flume (figure 8).

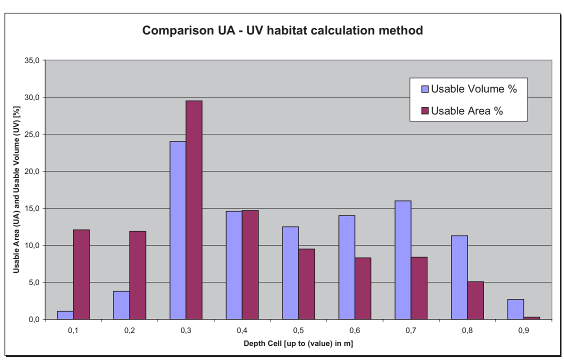

Figure 8: UA & UV method, available habitat in relation to depth.

Habitat availability in relation to flow velocities show overestimations for lowest and highest value at UA – method. The available habitat at highest velocities between 0.6 and 0.8 m/s are lost cause of data smoothing during the process of mean vertical velocity calculation with UA method. Results of UV method show the whole range of available habitat up to the highest measured velocity values (figure 9).

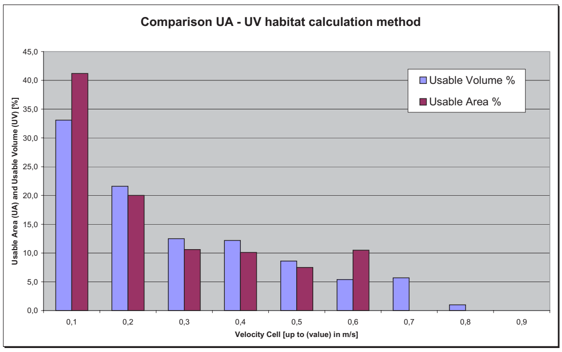

Figure 9: UA & UV method, available habitat in relation to velocity.

Further simulations are necessary for model validation and will be done in the next step of the research-project.

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Invertebrates and near-bed hydraulic forces: combining data from different EU countries to better assess habitat suitability.

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ABSTRACT: In this work we have combined in a common data base a lot of existing data relating invertebrates and near- bed forces measured with the FST-hemisphere method (Statzner & Müller, 1989) from French and German rivers. We could generate hydraulic preference curves for more than 250 invertebrate taxa from 23 reaches of 17 rivers. However, for most taxa preference curves were not transferable spatially and/or temporally. This could be explained by species characteristics (preferences varying with ontogenesis) but also by river characteristics (e.g. geomorphology, size, discharge). We have compared models developed both in France (FSTress) and in Germany (CASIMIR) used to predict invertebrate population changes as a consequence of an alteration in hydraulic conditions. Most of the FST distributions obtained by these two models showed contradicting results. Even for the most similar results, FST distributions calculated with FSTress were always bimodal, whereas they were unimodal with CASiMiR. FSTress might not be universal and should be improved by the collection of additional data sets in a wide range of river types. CASiMiR model should also be improved by using generalised numerical approach to calculate FST-hemisphere numbers based on velocity/FST-hemisphere correlations.

Future works will 1) further extend the hydraulic/invertebrate data base to other EU countries; 2) make all these data available on a web site and 3) integrate spatial and temporal parameters in the biological models to make them transferable. All these data are deeply needed throughout Europe to aid decisions on flow management.

1 INTRODUCTION

The importance of stream hydraulics to benthic organisms has long been recognised in running water ecology (see reviews by Newbury 1984 and Hart & Finelli 1999). Therefore, understanding how invertebrate species respond to near-bed hydraulic conditions is of primary importance in explaining community composition but also for the design of measures concerning river management in terms of flow regulations or hydromorphological enhancements.

Statzner & Müller (1989) developed a simple standard method (FST-hemispheres) to measure near-bed flow forces directly (and shear stress via calibrations) integrating much of the flow structure complexity close to the bottom. Although there are several institutions in the EU using this method for river investigations there has hardly been an exchange of data. Furthermore, two different approaches for a prediction of FST-hemisphere distributions with modelling tools have been developed in Germany (CASiMiR) and in France (FSTress). These models also provide a module for assessing the habitat suitability for invertebrates based on FST-hemispheres and then for near-bed constraints.

In this context our aims were 1) to combine data on invertebrates and near-bed hydraulic variables and to develop a common data base; 2) to compare hydraulic preferences of species encountered in both German and French rivers to check their transferability throughout Europe and 3) to use these curves to predict invertebrate population changes as a consequence of an alteration in hydraulic conditions by using and comparing models developed both in Germany (CASiMiR) and in France (FSTress).

2 BUILDING A COMMON DATA BASE

2.1 Data base organization

We could combine data and model preference curves for FST-hemispheres for more than 250 taxa from 23 reaches of 17 German and French rivers. Beside fauna characteristics and their hydraulic preference curves, we have also integrated river and sampling method characteristics (Table 1). River characteristics are very important in the data base to enable the selection of the right data set for model application (i.e. the right preference curve if not transferable, see Table 1 and § 2.3).

Table 1: River, Fauna and Sampling characteristics available in the hydraulic/invertebrate data base.

River	Fauna	Sampling
Country	Systematic [1]	Sampler type
River name	Feeding habits [2]	Area sampled
Geographic position	Preference curves for FST	Mesh size
Altitude	Preference categories [3]	Season
Stream order	Transferability [4]	
River reach name		
Mean annual discharge		
Mean reach width		

[1]phylum to species.
[2]absorber, deposit feeder, shredder, scraper, filter-feeder, piercer or predator.
[3]limnobiontic, limnophilic, rheophilic or rheobiontic.
[4]tells if the curve is transferable throughout a country or Europe or not transferable.

2.2 Hydraulic preference curves

We used a method proposed by Schmedtje (1995) based on non linear regression analysis to define preference curves as this method has been already used on some of the German data and because we needed comparable curves.
This non linear model is described by the following function:

$$N(FST) = a * FST^{b} * e^{c*FST}$$

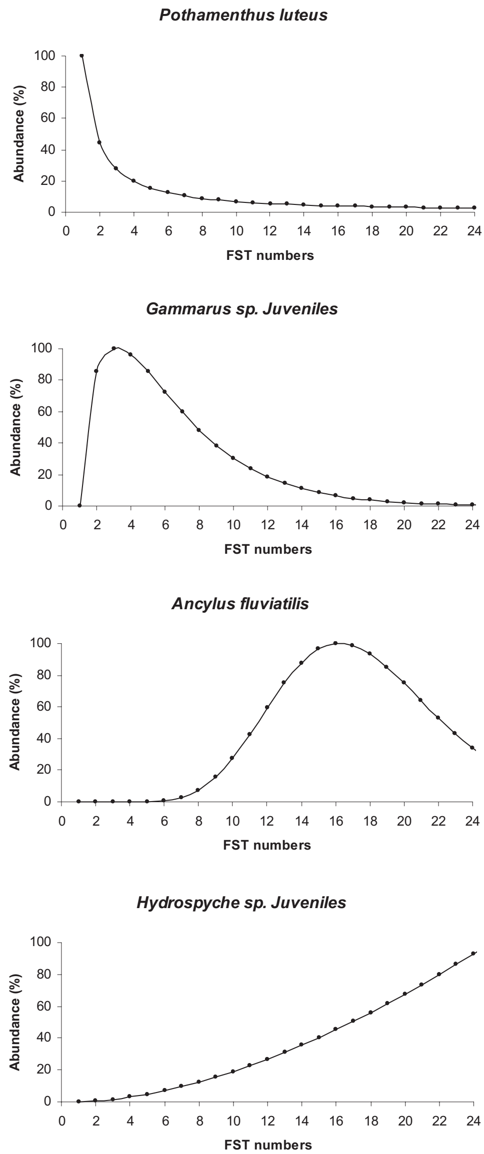

Figure 1. Hydraulic preference curves for 4 taxa encountered in the Rhône River illustrating the 4 Schmedtje's preference categories. First graph, a limnobiontic species (Chautagne reach, winter), model parameters: $a = 36.1$, $b = -1.2$, $c = 0.5 * 10^{-8}$ and $R^2 = 0.43$, a limnophilic taxon (Belley, spring), $a = 2574.3$, $b = 0.7$, $c = -0.3$ and $R^2 = 0.77$, a rheophilic species (Chautagne, spring), $a = 9.4.10^{-8}$, $b = 11$, $c = -0.7$ and $R^2 = 0.72$ and a rheobiontic taxon (Brégnier-cordon, spring), $a = 0.1$, $b = 1.7$, $c = 1.2 * 10^{-8}$ and $R^2 = 0.64$)

With N, the density of the taxa, FST, the FST-hemisphere number and a, b, c the three parameters of the model $a > 0$ and $c < 0$. Depending on b and c values and thus on the shape of the curves, Schmedtje defined 4 preference categories (limnobiontic, limnophilic, rheophilic and rheobiontic, see Figure 1 for examples). We used the non linear regression module (iterative Gauss Newton method) of Systat-10 statistical software (Wilkinson et al. 1996) to determine the three parameters and the fitting of each model.

An excel macro was then developed to transform the raw data into percentages so that all the preference curves were in the same format and therefore comparable. So, in the data base, for each taxa encountered in each reach at each season, an excel file describing its preference curve is available.

2.3 Hydraulic preference curve transferability

If one wants to model habitat availability or habitat suitability classes for a given species, the appropriate hydraulic preference curve should be integrated into the model. The ideal situation would be to have transferable preference curves throughout Europe so that only one curve would be available for a species. The preference curves available in our data base showed that for most taxa, curves are not transferable even within a country. For instance, the Mollusc species *Ancylus fluviatilus*, showed different hydraulic preferences within the same River reach (Rhur, Germany) being limnophilic in summer and rheophilic in winter (Fig. 2). Moreover, in the Rhône River (France) *Ancylus fluviatilus* was also rheophilic in spring in Chautagne reach but with a maximum of individuals for a FST number 16 whereas in the Rhur in summer the maximum was 9 (Figs 1 & 2).
The modeller after all has to choose the right preference curve that was define in a river reach having the same characteristics and that was sampled with the same method at the same period of the year than the river reach in which he wants to assess habitat suitability. This is why it was so important to add river and sampling characteristics in our data base.

3 COMPARISON OF THE GER MAN AND FRENCH MODELS

3.1 FSTress and CASiMiR models

FSTress links a biological model and a statistical hydraulic model. Local invertebrate samples and local FST measurements are used to predict species densities at a given shear stress. At the reach scale, the statistical hydraulic model predicts FST frequency distribution at a given discharge from very simple input data such as mean width and mean depth at two different discharges. By coupling the two models, FSTress predicts the densities of species for different discharge scenarios at the reach scale
(http://www.lyon.cemagref.fr/bea/lhq/stathab.htm).
The CASiMiR module BHABIM (Jorde 1997, Jorde & Bratrich 1998) is based on FST hemispheres as an indicator for near-bottom flow forces. The model uses measured distributions of FST-hemispheres and adapts statistical distributions. At the same time the portions of classified flow forces in relation to the wetted surface are calculated. By linking this information with hydraulic preference

functions of benthic organisms the available habitats as a function of the discharge can be assessed.

The idea to compare these two models was to check if FSTress could be used in any river so that a lot of time could be saved for the collection of field data (measuring FST distributions for CASiMiR is time consuming).

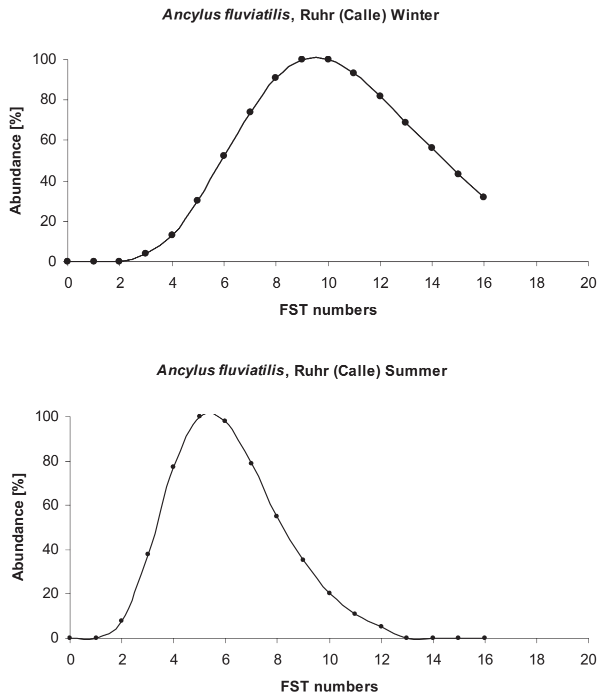

Figure 2. Preference curves for FST hemispheres for the Mollusc species *Ancylus fluviatilis* for the reach « Calle » from the Rhur River in Germany at two different seasons (winter and summer).

3.2 Modelling FST distributions

We compared the modelling of the FST distributions with these two approaches using data of 5 different river reaches of the German rivers (Alz and Weisse Elster and three morphologically different reaches in River Kocher). This comparison showed that in some of the investigated reaches, the results of FSTress were similar to those received with CASiMiR but not for all discharges (see an example Fig. 3). However, in some reaches model outputs were more different. Even for the most similar results, FST distributions calculated with FSTress were always bimodal whereas they were unimodal with CASiMiR (Fig. 3).

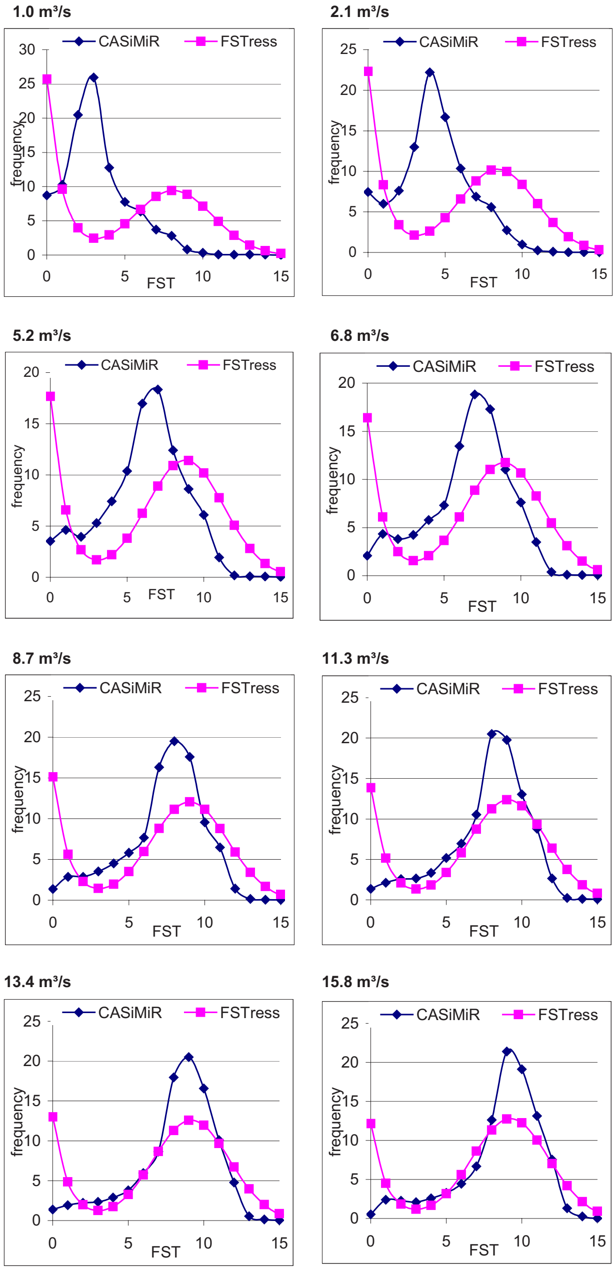

Figure 3. Comparison of FST hemisphere frequencies gained by CASiMiR and FSTress models in a 1 km reach long of River Alz close to Trostberg, Bavaria, Germany

3.3 *Modelling habitat assessment*

The above mentioned calculation of frequency distributions of FST-hemispheres is only the first step in assessing habitat suitability for macroinvertebrates. The final output of the computer models FSTress and CASiMiR (module BHABIM) are habitat values in terms of integrated habitat availability or habitat suitability classes. The comparison of the two models showed that CASiMiR outputs deliver additional information in terms of the portion of different habitat suitability classes.

4 DISCUSSION AND PERSPECTIVES

In this work we have combined a lot of existing data on hydraulic and macroinvertebrates from French and German rivers. The numerous preference curves now available in the common data base will be very useful for modellers dealing with habitat assessment. However, our results showed that in most cases preference curves were not transferable spatially (between reaches or rivers) or temporally (between seasons). This can be explained by species but also by river characteristics. For many taxa, hydraulic preferences vary during ontogenesis, in respect to changes in oxygen needs (Collier 1994) and/or abilities to withstand flow (Buffagni et al. 1995), leading to the non transferability of the curves seasonally (Sagnes & Mérigoux, in prep). In the same way, hydraulic preferences can vary for species depending on the availability of hydraulic conditions at different flow rates (Mérigoux & Dolédec 2004). Our future work is aiming at developing biological models that will integrate these spatial and temporal parameters so that the biological models will become transferable. Transferability of these models is deeply needed throughout Europe to accurately predict responses of invertebrates to hydraulic changes when rivers are regulated or when discharges are enhanced (river restoration programs).

Two hypotheses were proposed to explain the contradicting FST distributions obtained from FSTress and CASiMiR models. First, FSTress that was developed from hydraulic data sets of about 20 small mountainous German rivers might not be universal. To improve this model performance, additional data sets in a wider range of river types are deeply needed. The second hypothesis is that FST-hemisphere measurements can only be performed for a water depth higher than about 10 cm and this might explain why there was so few low hemisphere numbers using CASiMiR model in rivers with a high portion of shallow areas. To cope with these measurement problems, future works will use velocity/FST-hemispheres correlations (Mader & Meixner 1995, Schneider & Ortlepp 2003) or a generalised numerical approach dealing with the balance of forces on the FST-hemispheres to calculate FST-hemisphere numbers (Kopecki & Schneider 2005).

Comparing FSTress and CASiMiR (module BHABIM) in assessing habitat suitability for macroinvertebrates, it was shown that CASiMiR outputs deliver additional information in terms of the portion of different habitat suitability classes. Therefore, CASiMiR outputs will be used in the future with all the French data. It can easily be done since a part of the FSTress model outputs can be transferred to CASiMiR..

Finally, in this work we have combined hydraulic preference curves for invertebrates from many Germany and France rivers in a common data base. The idea is to further extend this data base by information from other EU countries and to make all these data available on a web site. These data will be very useful throughout Europe to run habitat suitability models that aid decisions on flow management.

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Application of MesoCASiMiR: assessment of *Baetis Rhodanii spp.* habitat suitability

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ABSTRACT: The assessment of the ecological status of running waters at a mesohabitat scale is commonly based on fish. Nevertheless, due to their strong dependence on a good physical and chemical habitat, macroinvertebrate species can also be used for this assessment. A new approach is presented by applying the MesoCASiMiR module of the CASiMiR modeling system. In this way, the habitat suitability for the mayfly *Baetis Rhodanii spp.* was modeled at a mesohabitat scale in the river Zwalm (Flanders, Belgium). The model inference system was based on fuzzy logic. Fuzzy variable sets and rules were derived from expert knowledge and from a database of biological samples. The suitability of the different mesohabitats for *Baetis Rhodanii spp.* in the river Zwalm could be reliably described by three morphohydraulical variables (depth, velocity and dominating substrate) and by the oxygen concentration. As a result, the habitat suitability could be calculated and a habitat map of the studied reach was generated. Validation of this map was performed by biological samples at different sites along the reach and indicated that predicted habitat suitability was closely correlated to the observed abundances in most of the sampling sites. Due to the universality of the MesoCASiMiR module, the presented approach is applicable on other rivers and can be used for prediction of the impact of restoration options at a mesohabitat scale.

1 INTRODUCTION

Since the eighties, river management in Flanders is mostly conducted at the basin level, using instruments as wastewater treatment plants and enforced effluent standards. Although these measures resulted in a significant improvement of the chemical and the ecological river quality (VMM, 2003), lots of small-scale efforts as remeandering, flood plane restoration and fish passages are still needed to meet the aim set by the Water Framework Directive (EU, 2000). To allocate these efforts in an efficient way, good river management should be based on a reliable assessment of the ecological bottlenecks in the river basin, at the micro- or mesoscale level.

Several microhabitat models only describe a small reach of the river (Alfredsen, 1997) and extrapolation of this microhabitat model to the river basin scale introduces a high level of uncertainty (Maddock, 1999). Furthermore, ecological assessment at the micro scale level is very time consuming and thus less suitable for river basin management. Therefore, intermediary methods between the micro- and the macroscale level were developed (Borsányi, 2002), assuming that the river consists of hydromorphological units (HMUs) (Thickner, 2000). The assessment of the ecological river status at this mesoscale level avoids the problems of time efficiency and upscaling (Maddock, 1999) and is therefore a suitable approach for good river basin management (Parasiewicz, 2003).

The ecological status of rivers is assessed by several mesohabitat models based on fish: MesoHABSIM (Parasiewicz, 2001), Habitat Mapping (Maddock & Bird, 1996), MesoCASiMiR and Habitat (Alfredsen, 1997). Unfortunately, due to severe disturbance of the aquatic ecosystem by human activities, fish communities are severely reduced in the Zwalm River basin and in the rest of Flanders. Furthermore, monitoring of ecological river quality in Flanders is done by the Flemish Environmental Agency (VMM) based on macroinvertebrates while the Water Framework Directive aims at a good quality of this community in itself.

Therefore, this paper attempts to apply MesoCASiMiR, one of the present mesohabitat models, on macroinvertebrates, in order to create a first step towards the assessment of the ecological status of Flemish running waters at the mesoscale. MesoCASiMiR is a module of the CASiMiR modelling system (Jorde, 1996; Schneider, 2001), based on fuzzy logic (Zadeh, 1965). Besides this modelling approach, a practical method to represent the results is proposed.

2 MATERIAL AND METHODS

2.1 Study site

The Zwalm river basin is part of the Scheldt river basin (Carchon & De Pauw, 1997). The Zwalm River has a length of 21.75 km and its river basin has a total surface of 11.650 ha. (Fig. 1). The average water flow (at Nederzwalm, very near the River Scheldt) is about one m^3s^{-1}. The water quality in the Zwalm river basin improved in the last years, due to investments in sewer systems and wastewater treatment plants (VMM, 2003). Nevertheless, most parts of the river are still polluted by untreated urban wastewater discharges and by diffuse pollution originating from agricultural activities. Also structural and morphological disturbances are numerous (Carchon & De Pauw, 1997). Weirs for water quantity control obstruct fish migration and are one of the most important ecological problems within the river basin. Therefore an in-depth study has been made on the development of fish migration channels and also natural overflow systems to reach an ecologically friendly water quantity management in the near future (Soresma, 2000). Some upper parts of the watercourses in the Zwalm river basin are colonized by very rare fish species and several vulnerable macro-invertebrates.

Figure 1. Location of the Zwalm River basin in Flanders

2.2 *Data collection and processing*

Fuzzy rules describing the habitat suitability for *Baetis Rhodanii spp.* were derived from biological data collected in the Zwalm River. All data were gathered during 5 consecutive years between August and September (2000 – 2004). At each of the 323 studied sites, 10 m of the present mesohabitat was sampled by means of 5 minutes kick sampling, using a standard handnet with mesh size 500 µm (NBN, 1984) and by in situ exposure of artificial substrates (De Pauw et al., 1983). The number of present *Baetis Rhodanii spp*. was expressed as absolute presence. In order to correct for different river width and thus for different sampling areas, the presence of *Baetis Rhodanii spp*. was expressed as weighted presence using:

$$weighted\ presence = \frac{absolute\ presence}{10 \times W_{avg}} \qquad (1)$$

in which W_{avg} is the average width of the sampled river stretch. This weighted presence was used to express the habitat suitability for *Baetis Rhodanii spp.*

Structural and physical variables were measured to describe the different mesohabitats (Table 1). Flow velocity was determined using a propeller flow velocity meter (Höntzsch ZS25GFE). For each 10 m stretch, flow measurements were performed at 40 % of depth on 15 points, divided over 5 transects. Each transect consisted of 3 equidistant points, forming a uniform grid. The dominating substrate was visually assessed and expressed in 4 classes. Field measurements were performed for dissolved oxygen (OXI 330/SET).

Table 1. Measured variables at each sampled river stretch.

Variable	Unit
Flow velocity (m/s)	m/s
Dissolved oxygen	mg/l
Dominating substrate	4 classes (from 1 = pebble to 4 = loam/clay)
Depth	m

2.3 *Software*

The constructed fuzzy rules were implemented in the MesoCASiMiR module of the CASiMiR modelling system (Jorde, 1996; Schneider, 2001; Schneider et al., 2001). This module in its current version was developped as an extension of ArcView GIS 3.3 (ESRI) and is not restricted to commonly used habitat parameters as flow velocity, depth and substratum but is designed to be used more universally. It can handle any habitat parameter, which can be defined as a property of a GIS polygon, for the assessement of habitat suitability. The model is adaptable in the way that habitat parameters itself and also their classfication in terms of fuzzy sets (see also 0) can be defined specific to the site, the investigation goals or the data availability. While e.g. for the investigation of benthic habitats flow velocity substratum and water quality parameters can be chosen, for fish habitat investigations morphological properties can be of higher importance for habitat assessment.

Any rule combining the classified habitat parameters with a habitat suitability (also classified by the use of fuzzy sets) can be used for the description of habitat preferences. An example of a rule is "IF flow velocity is 'high' AND concentration of dissolved oxygen is 'low' AND … is 'very high' AND … is 'very low' … THEN habitat suitability is 'medium' ". The number of five fuzzy sets, available for definition in the model (e.g. 'very low', 'low', 'medium', 'high', 'very high'), is assumed to be sufficient to describe most habitat preferences adequately.

2.4 *Model evaluation*

Evaluation of the results was based on two criteria, the percentage of Correctly Classified Instances (CCI) and the weighted Kappa ( ) (Cohen, 1960; Fleiss & Cohen, 1973), which were derived from the confusion matrix (Fielding & Bell, 1997). The CCI is defined as the number of sites where the modelled habitat suitability class was the same as the monitored one, divided by the total number of sites. The weighted Kappa is a simply derived statistic that measures the proportion of all possible habitat suitability classes that are predicted correctly by a model after accounting for chance.

3 RESULTS

3.1 *Fuzzy sets and rulebase*

Fuzzy sets and rules were derived from expert knowledge (Adriaenssens et al., in prep.) and from the collected data based on three fold cross validation, using two third of the dataset, randomly chosen. Each variable was divided in a number of fuzzy sets (Table 2), while 192 IF…THEN rules were constructed based on these sets.

Table 2. Number of fuzzy sets in which each variable was divided.

Variable	Number of fuzzy sets
Flow velocity (m/s)	4
Dissolved oxygen	3
Dominating substrate	4
Depth	4
Habitat suitability	4

The rules and sets were implemented in the MesoCASiMiR module of the CASiMiR modelling system. The output of this module, the weigthed presence of *Baetis Rhodanii spp*., was converted to the habitat suitability class to which it belonged the most. The fuzzy rules were evaluated by comparing the modelled and the actual habitat suitability class for the remaining one third of the dataset, containing 107 records (Table 3). Based on this confusion matrix, a percentage of CCI of 64.5 % and a weighted Kappa of 0.410 were obtained.

Table 3. Confusion matrix based on the constructed rule base (HS= Habitat suitability).

Predicted HS class / Monitored HS class	1	2	3	4
1	62	20	1	0
2	6	4	0	0
3	3	3	2	1
4	3	0	1	1

3.2 Habitat suitability mapping

The created fuzzy rules were applied to predict the habitat suitability for *Baetis Rhodanii spp.* in a 5 km reach of the Zwalm River. This reach was divided in different stretches, each stretch being defined by one HMU. No distinction was made between different mesohabitats within one stretch, as in most stretches, variation of the mesohabitat was not significant. Along the studied reach, 3 hydromorphological variables (depth, flow velocity, dominant substrate) and 1 physical chemical variable (dissolved oxygen) were assessed in each stretch. No significant flow change occurred during the measuring period. The habitat suitability for *Baetis Rhodanii spp*. was modelled for each mesohabitat based on the created fuzzy rules and these measurements. Visualisation of this habitat suitability was done by means of a habitat suitability map consisting of different polygons, each describing one mesohabitat (Fig. 2).

Validation was performed by comparison of the modelled results with biological samples of 24 mesohabitats. These samples were not included in the fuzzy rule development process. The resulting percentage of CCI and the weighted Kappa were respectively 62.5 % and 0.554.

Table 4. Confusion matrix based on the validation of the Habitat suitability (HS) map

Predicted HS class / Monitored HS class	1	2	3	4
1	12	3	1	0
2	2	1	1	0
3	0	2	1	0
4	0	0	0	1

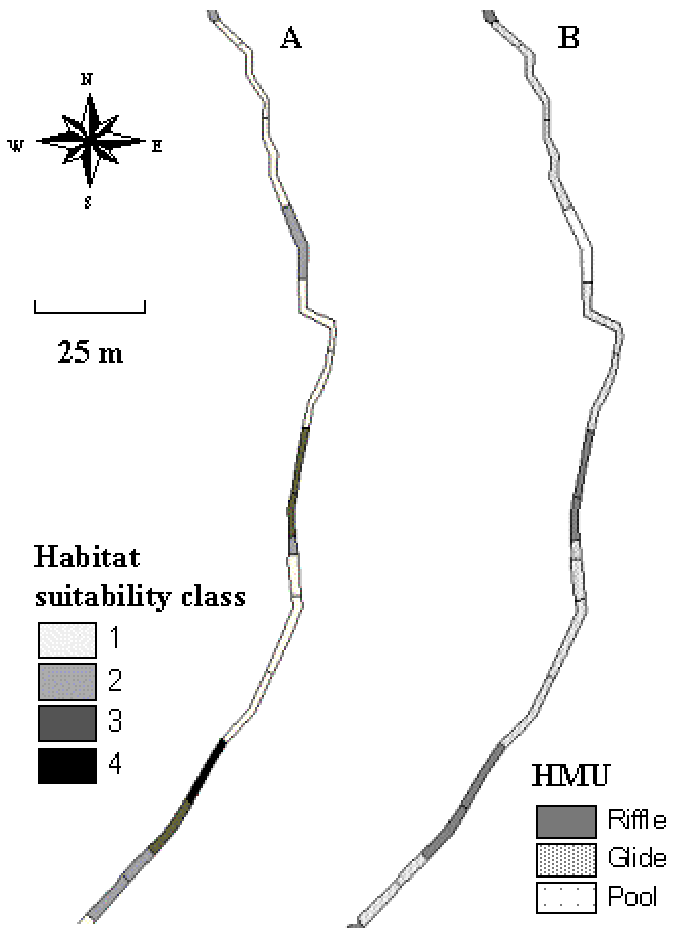

Figure 2. Habitat suitability map for *Baetis Rhodanii spp*. (A) of a 250 m reach of the studied area and hydromorphological units (HMUs) in this reach (B)

4 DISCUSSION

Most prediction errors occurred due to the prediction of a higher habitat suitability for *Baetis Rhodanii spp*. than the monitored one. Absence of *Baetis Rhodanii spp*. can be determined by other variables that were not included in the fuzzy rules, but as well by the limited monitoring efficiency and (re)colonization by this species. Therefore, overestimation of the habitat suitability indicates that the concerned habitat is suitable for *Baetis Rhodanii spp*. concerning the 4 studied variables, and this is not necessarily a prediction error. For instance, the dissolved oxygen was included in the variable set to take into account the trophic status of the river. Previous research states that conductivity could also have an important effect on macroinvertebrate presence (D'heygere, 2003; Adriaenssens et al., in prep.). Furthermore, biotic interactions were not included in the fuzzy model although these can also play an important role. As a result, the percentage of correctly classified instances would rise up to 85.1% when the predictions overestimating the habitat suitability would also be considered correct.

The data were unequally divided over the four suitability classes, which is also reflected in the confusion matrix resulting from rule validation. This disproportion is due to the fact that a significant part of the study site is severely impacted by human activities. Only a small part of the river basin contains reference sites, situated in some of the least disturbed brooks in Flanders. This results in more prediction errors in the lower HS classes (1&2) than in the higher ones.

Comparison of the derived rules with expert knowledge derived from literature (Adriaenssens et al., in prep.) indicated that the used sets and rules are not necessarily transferable between different rivers. It is clear that each river consists of specific conditions as geomorphology, (micro)climate, typology,… In that way, the set of key variables determining habitat suitability for macroinvertebrates can be different for each river. Furthermore, the range of the concerned variables changes for different rivers, resulting in other sets and rules.

Generalisation of rules might only give an indication of the impact of some variables on macroinvertebrate habitat suitability. In that way data collection is inevitable in order to establish fuzzy rules, which can be used as a river "blueprint". Once this template is constructed, maintenance can easily be performed by regular rule and set validation. This theory is emphasized by the fact that there is a small difference between the percentage of CCI resulting from rule validation and from map validation, although the sites used for map validation were not included in the rule development process.

In the modelling process, data collection and development of rules and sets require the most efforts. In order to increase efficiency, rules could be derived from the data in a faster model driven way, for instance using Artificial Neural Networks to derive the most important parameters (Dedecker et al., 2004) and Hillclimbing to set the optimal rule base (Van Broekhoven et al., 2004).

In a next step, the presented approach can be used for prediction of the ecological impact of different river restoration options. Restoration decisions are nowadays often based on intuition rather than on rigorous science (Muotka & Laasonen, 2002). If an objective of a river restoration option is not obtained, efforts are lost because the measures are already taken. This justifies the need for approaches that can give a reliable indication of the effect on river biology. Ecological models are a powerful tool and can be used for this purpose. Furthermore, modelling will also allow comparing the effects of alternative mitigation options. This will aid a river managers in selecting an optimal set of restoration options to obtain a desired ecological quality in a river system. Moreover, implementing such ecological models for macroinvertebrates in a Decision Support System will allow river managers to weigh conflicting demands of different stakeholders such as households, farmers, nature developers, water quantity managers,…

5 CONCLUSION

Due to the universality of the MesoCASiMiR module, the presented approach is applicable on other rivers and can be used for prediction of the impact of restoration options at a mesohabitat scale. To achieve this, biological sampling and expert knowledge have to provide fuzzy rules, which can act as a blueprint for the studied river stretch. Furthermore, macroinvertebrates are particularly interesting for assessment of the ecological river status in Flanders, as river degradation has severely reduced fish populations. Other important variables describing this ecological status should be revealed by further research on other macroinvertebrate indicator species. This could result in better and more reliable decisions in integrated river management.

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Climate change and possible impacts on fish habitat. A case study from the Orkla river in Norway

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ABSTRACT: The aim of the study was to determine possible impacts of the climate change on the habitats of Atlantic salmon in the river Orkla. The topography of an approximately 250m long study reach of Orkla in Meldal was measured to simulate velocity and depth habitat using the SSIIM model, a 3 dimensional flow model, at a wide range of flows between 15 m^3/s and 100 m^3/s. Velocities at two different discharges (26 m^3/s and 62 m^3/s) were used for validation. The results of the hydraulic simulations were used as input files for the HABITAT model to assess the impact on fish habitat. A mesohabitat mapping of the Orkla river was previously carried out in order to scale the results to a larger reach of the river. The modeled river reach does not contain all mesohabitats found in Orkla, but the results from HABITAT give a good indication on how large areas of the river will be altered in a future climate. Time series for discharge for the period of 1980-2040 was simulated using a rainfall-runoff model given predictions of air temperature and precipitation. A hydropower production planning model was also used to determine the power production and the final flow going into Orkla. Time series of habitat was compared between the control period (1980-2009) and the future climate period (2010-2040). The results show lower frequency of large floods, higher frequency of small and intermediate floods, higher winter runoff and an overall increased flow of about 10 per cent in the future. The long-term analysis of habitat conditions for Atlantic salmon show a decrease of approximately 10% in suitable areas as a yearly average.

1 INTRODUCTION

1.1 Background

During the last decades, river and water resource management measures had widespread impacts on the ecology of rivers. The habitats of many rivers were altered dramatically through river channelization, the creation of flood defence structures, irrigation and the construction of dams and weirs for power production. Furthermore the flow regimes were altered through urbanization and the change of land use. During the last decades we have very strong indications on climate change. Predicted changes in climate will impact the ecosystem. This study is a part of a research program to assess possible future impacts on the riverine ecosystem from the combined effect of climate changes and hydropower operation.

These changes affected riverine ecosystems in a negative way. Fish species were reduced and extinct, but also 'non-visible' organisms were affected by these changes. By measuring physical habitat conditions and modelling of the rivers, reasons for the repression of the animals can be found.

1.2 *The river Orkla*

The study site was an approximately 250 m long stretch of the river Orkla 60 km south of Trondheim, Norway. Orkla has a catchment area of 3053 km² and an average annual discharge of 71 m³/s.

After 1978 five hydro power plants were built along the river, which led to significant changes in the hydrology. At the study site, no water is abstraced but the hydrological regime is altered due to hydropower production further upstream.

1.3 *Physical habitat*

The life of all animals is affected by various biotic and abiotic factors. Physical, biotic and chemical features that provide an environment for plants and animals can be defined as habitat. Physical habitat is dependent on two factors. It is the combination of a riverbed structure with a certain discharge. Hence a physical habitat is dynamic in space and time.

By measurements the optimal physical habitats can be found. By using hydraulic modelling it can be predicted if there are enough habitats for the fish at a certain discharge.

According to studies, only four parameters are important to describe habitat conditions. These are water depth, flow velocity, substrate and cover (Heggenes, J. 1996). But the importance of these parameters declines, if the water quality is not good.

Because fish are heterothermic animals, their preferences are different in summer and winter, normally with a shift in preferences and behaviour at 6-10 degrees of water temperature.

2 MEASUREMENTS

To generate a hydraulic model using the 3 dimensional flow model SSIIM (Olsen 2000), the topography of the study reach and the flow velocities had to be measured.

Measurements were performed at two days with different discharges (26 m³/s and 65 m³/s) to verify the model with two different data sets.

2.1 *Discharge and velocity*

There are many different ways to measure discharge and velocity. For this study an Acoustic Doppler Currant Profiler (NORTEK Qliner) was used. The instrument transmits a short acoustic impulse, which is reflected by particles in the water and the flow velocity in 3 dimensions can be calculated from the shift in the frequency. By using a current profiler the velocity profile over the whole depth can be measured.

2.2 Topography

To obtain the topography of the river bed, we used a geodetic survey with a Leica theodolite and electronic disatance meter. The measured points were randomly spread over the whole study stretch. The survey at Orkla was performed in a local coordinate-system.

3 SIMULATIONS

The measured data was used to generate a hydraulic model using the 3 dimensional flow calculation program SSIIM to obtain the distribution of the flow velocity and water depth and analyze the results for habitat conditions for fish.

3.1 SSIIM

The measured topography data of both days was used to interpolate a smoother and regular grid with the Surfer software, and this grid was used in the further work.

The size of the calculation grid was set to 150x40x4, and the grid was brought into the shape of the river using the measured coordinates of the waterline.

With the two different waterlines at different discharges two independent models were generated and calibrated. The calibration was done by changing the value for the roughness in the input file in order to achieve the correct water surface level and flow velocities. (Table 1)

Table 1. Measured and calculated flow velocities (examples)

Point number	Measured [m/s]	Calculated [m/s]
3	0.33	0.38
6	1.30	1.22
8	1.04	1.02
9	0.44	0.38
11	0.14	0.16

After the models were calibrated, they were adapted to other discharges between 15 m^3/s and 100 m^3/s. Each time, the discharge was changed, the waterline also had to be adapted to the new discharge, because SSIIM does not work with dry cells.

From the calculation the flow velocity and water depth in each grid point was gained.

3.2 *HABITAT*

The results of the hydraulic calculation were used as input files for the HABITAT (Harby and Heggenes 1995) model. The other input to the model were preference curves for Atlantic salmon for water depth, flow velocity and substrate. These preference curves are normally calculated on the base of actual fish observations. In Orkla we had no local fish observations. We used the preferences given in table 2 after Harby and Heggenes (1995), Heggenes (1996) and Harby et al (1999), without separating summer and winter conditions.

Table 2. Fish preferences used in Orkla

	Preferred	Indifferent	Avoided
Depth [cm]	40-80	20-40, 80-150	<20, >150
Velocity [cm/s]	30-60	20-30, 60-80	<20, >80
Substrate [cm]	1.6-25.6	0.2-1.6, >25.6	<0.2

HABITAT compares the values for the velocity, the depth and the substrate in every grid point of the SSIIM model with the preference curve, and indicates the cell as preferred, indifferent or avoided. The results of HABITAT are a plot with coloured grid cells according to the preference class (figure 1) and files with the size of each preference class. This analysis was done for each discharge

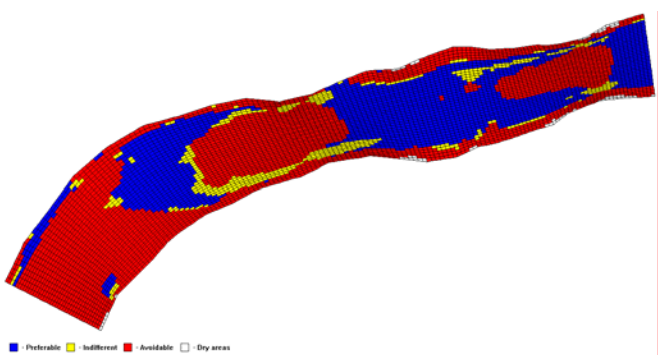

Figure 1. HABITAT result plot for the combination of velocity and depth at 10 m^3/s. Blue cells indicate preferred areas, and red indicate avoided areas.

3.3 *Time series*

HABITAT is able to make a time series of habitat conditions. To evaluate the impact of the climate change on the habitat availability, two time series were modelled, a 'today' scenario and a 'future' scenario. For both scenarios, simulated time series were used. The 'today' time series starts on the 1st of January 1980 and ends on the 24th of November 2009, while the 'future' series starts on the 1st of January 2010 and ends on the 24th of November 2040. Both series contain a daily average discharge.

The time series of natural flow were generated with the rainfall-runoff model HBV (Bergstrøm 1992). Three subcatchments were calibrated with observed data of precipitation, rainfall and flow. Then downscaled time series from climatic change scenarios were used in the simulations. The scenarios were taken from the RegClim project of the Norwegian meteorological office (www.regclim.no). To create time series of regulated flow at the study site, the ORP model (ref) was applied to the hydropower system and the whole drainage basin.

From the simulated time series, it could be recognized that the 'today' series has a decreasing trend, while the discharge in the 'future' series steadily increases. But not only the mean annual discharge changes in the future, the discharge also changes in the year, specially in the winter months.

4 RESULTS

From the HABITAT analysis of the different modeled discharges can be seen, that the areas with a suitable combination of depth and velocity (figure 2) are decreasing. Even though also the areas with preferred depth are decreasing, depth is not the limiting factor in the study stretch. With increasing discharge areas with avoided velocity outbalance suitable areas.

Using the input data (the time series and the results of the HABITAT models) two time series were modeled for the two parameters depth and velocity to compare the habitat availability today and in the future.

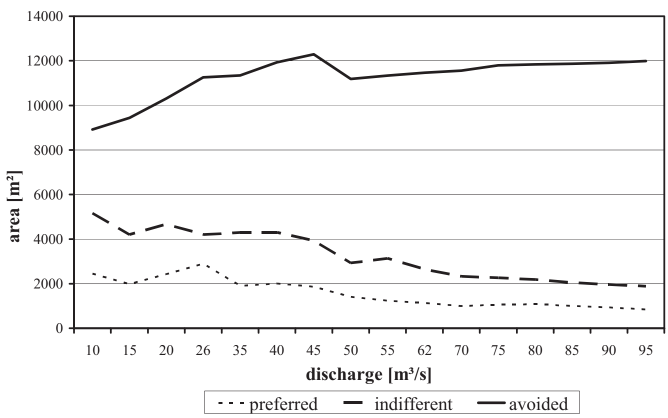

Figure 2. Area of each class with changing discharge. (The drop in the overall area is caused by the two different calibrated starting models.)

The results of the HABITAT time series were time series for the size of each parameter (depth and velocity) and class (preferred, indifferent and avoided).
The loss of suitable habitat area for salmon is indicated by the decrease of the average size of preferred velocity areas from 1511 m² today to 1377 m² in the future. Figure 3 shows the size of areas with avoided velocity. In the today time series, the size of avoided areas has a slight decrease, while the future time series shows a trend to bigger areas with avoided flow velocities, in accordance to the discharge, which also increases in the future.

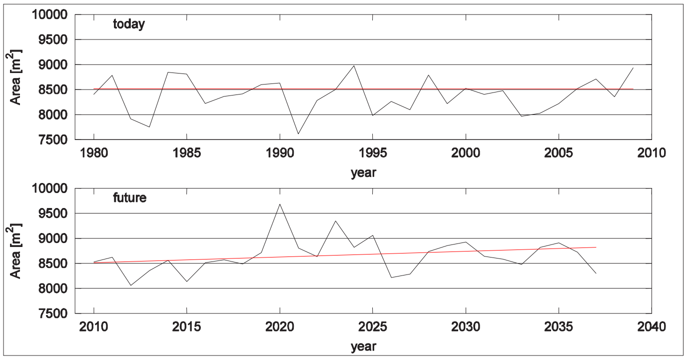

Figure 3. Time line of the size of annual average avoided velocity areas.

At a discharge of 95 m^3/s the water covered area in the study stretch is about 12.000 m^2. In the future time series (30 years) 130 periods with an avoided area larger than 10.000 m^2 occur.

To be able to apply the results of the study stretch to the rest of the river, the results of the HABITAT analysis were compared to a mesohabitat mapping (Borsanyi et al 2003), performed earlier by Alfredsen et al, (pers. comm. 2004). During the field work the river was divided into the ten mesohabitat classes and the hand sketches digitized with a GIS software.

Comparing the results from HABITAT and the mesohabitats at a similar discharge, the preference of the salmon for the different mesohabitats could be seen. It could be seen, that the mesohabitat classes C (*pool*) and D (*walk*) fit best to the preferences of the young salmon. In the main stream line, the river was classified as a *shallow glide* (B2) and there the flow velocity is too fast for the salmon, so these areas are avoided like the *deep glide* areas (B1).

With this information it should be possible to apply the results of this study to other stretches of the river.

5 CONCLUSIONS

Triggered off by the climate change, which cannot be doubted, a change in the discharge regime from Orkla will occur in the future. Warmer temperatures in the mountain regions of the catchment cause snowmelt during the whole year, and during winter more precipitation will fall as rain.

But not only is the annual average discharge increasing, also the distribution of the run-off shifts in the course of the year. The spring peak appears earlier in the year and is not so distinct due to the higher run-off in the winter.

The life of the Atlantic salmon is highly influenced by the discharge, and changes in the annual discharge distribution also have effects on the fish. For instance is the start of the migration to the sea triggered off by the discharge and the hatching time is highly dependent on the water temperature. If the discharge is high at the time of the *swim up* the small parr might not be able to stay at the desired place but is drifted away by the high flow velocity. Furthermore the invertebrates, young salmon feed on, retreat into the gravel at the bottom of the river or they area also drifted away.

The change of the habitat availability is directly linked to the change in the discharge. A higher discharge automatically causes a higher flow velocity in most parts of the river, and an increased depth. However the depth is no problem in the study stretch. Due to the shallow gravel bank on the right side of the river, there are always enough areas of suitable depth for the salmon.

The shallow bank can also be disadvantage for the fish. If the discharge drops drastically in a short period of time, what is quite possible in a regulated river, the fish could be trapped in small pools on the gravel bank or even strand on the gravel banks. (Saltveit et al 2001).

The much bigger effect on the habitat availability has the flow velocity. With increasing discharge, the flow velocity increases simultaneously and the area with high flow velocity pushes the areas with suitable velocities further and further to the edges. This is also reflected in the fact that the areas of suitable velocity are approximately 12% lower in the future than today.

5.1 Possible improvements

To get more reliable results from the simulations, some improvements could be made during the whole process.

If more velocity points were measured in the field, the models could have been calibrated better. The velocity points measured during the first field trip are also not spread very well in the whole survey area. No velocity was measured on the left side of the river or in the middle, and at the inflow of the stretch.

It would have been useful for the comparison of the HABITAT results, if all models had the same start and end profile. So the decrease in the water covered area between 45 and 50m^3/s (figure 2) could have been avoided, and the development of the water covered area could have been described better.

Probably the models with a high discharge could have been improved, if another calibrated model existed, or if at least an additional water line was measured at a discharge around 90m^3/s. Now, the waterline between 65 and 95m^3/s is always only adapted from the calibrated model, and so inaccuracy can arise.

The preference curves, that were used in HABITAT were general applicable. It would have been best, if the preference curves were generated from fish observations at Orkla itself. The results are not inaccurate for this reason, but otherwise, the preferences of the local salmon population could have been considered. The preferences for winter and summer conditions should also have been used in stead of general all-season preferences.

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Norwegian mesohabitat method used to assess minimum flow changes in the Rhône River, Chautagne, France. Case study, lessons learned and future developments - impacts on the fish population

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ABSTRACT: Eight bypass sections of the Rhône River are being rehabilitated in order to improve ecological status. At Chautagne, since 1980, most of the flow is diverted to a canal leading to a hydropower plant. In July 2004, the environmental flow of the approximately 8 km old branch of the Rhône River was increased from 10 (winter) – 20 (summer) m^3/s to 50 (winter) – 70 (summer) m^3/s. A scientific study program will investigate changes in the fish and invertebrate populations. The Norwegian method of classifying the river section into maximum ten different physical meso scale morphological classes (mesohabitat classes) was applied at 10 and 70 m^3/s. The results showed a change in mesohabitats after flow increase. Before 1980, the upstream part of the Rhône River used to braid in the floodplain and fish community was dominated by large rheophilous fish such as cyprinids, grayling and brown trout. A long term survey available at Chautagne showed that since discharge reduction, densities of some populations decreased greatly and can be considered endangered. These species are grayling (*Thymallus thymallus*), brown trout (*Salmo trutta fario*), nase (*Chondrostoma nasus*) and dace (*Leuciscus leuciscus*). Meanwhile, population densities of species like gudgeon (*Gobio gobio*), minnow (*Phoxinus phoxinus*) and stone loach (*Barbatula barbatula*) have increased. The decrease of some population densities can be explained by reduced availability of habitat types for specific ontogenic stages at low discharge. The 2004 flow increase in this bypass section has modified both the arrangement and the proportion of mesohabitats. Habitat preferences according to the "Norwegian mesohabitat" method's classification have been coded using habitat preferences from Lamouroux et al. (1999a) and Mallet et al. (2000) (for grayling). Most of the endangered species need high velocity mesohabitats and both high and low depths during all or part of their development. The increase of such mesohabitat proportions at 70 m^3/s indicates that their population should recover in the next years.

1 INTRODUCTION

The Rhône is one of the biggest European rivers (812 km, watershed area = 95 500 km^2, mean annual discharge at the mouth = 1710 m^3/s). The hydrologic regime is characterised by a high discharge in spring (snow melting) and summer (ice melting). As the river flowed from the Alps, with a steep slope gradient, the Rhône was mostly braided form the headwaters to its upper delta (Bravard et al. 1992). Until the early 19th century, like the Rhine and the Danube, the human pressure on the river has progressively increased from channelisation for the improvement of navigation to hydropower production (Bravard et al. 1992, Persat et al. 1995).

Downstream from Lake Geneva, the Rhône River was one of the first rivers used for hydraulic power exploitation (years 1871, 1886, 1892). Presently, there are 22 dams along the river, and among them, 18 (from Chautagne to Vallabrègues) are low-fall power plants.

Such equipments divert a maximum flow ranging from 700 to 2300 m^3/s (depending on their position along the longitudinal gradient) into a headrace canal to the power station. The former river bed (called the bypass section) receives a compensation flow outside the period of spate, but almost all the discharge during spates. After more than 50 years of intensive river regulation, a large restoration program has been recently initiated. This 10 year program has two main objectives: to increase the minimum flow in bypass sections, and to restore several side-arms with varying connections to the main channel. Currently, the program will tackle only 8 sections of the upper and lower parts of the Rhône. One early task was to determine the ecological value of the minimum flow within economic constraints. Hydraulic and biological models have been used to answer this question, focusing on long term data on fish community and habitat preferences. These studies aimed to predict changes in fish community composition related to hydrological changes (river width, depth and flow velocity) associated with discharge. River managers have appealed to scientists (including ecologists, geomorphologists, hydraulic engineers, sociologists and economists) to develop and test scientific survey methods to evaluate the success of restoration. These include:

- collation of relevant data,
- data management (creation of databases and GIS),
- identifying critical gaps in the knowledge base,
- developing new protocols for pre-restoration assessments,
- analysing pre-restoration data and developing predictive models of the effects of restoration,
- measuring post-restoration responses
- explaining the success or otherwise of the restoration procedures.

This is an excellent opportunity to integrate the long-term responses (e.g. fish) with short-term recolonisation patterns. River engineers can use this to develop protective policies for effective river management and to assess the success of river restoration at several temporal scales.

At Chautagne, since 1980, the compensation flow was about 10 m^3/s (from 1st December to 31st May) and 20 m^3/s (form 1st June to 30th November). From July 2004 on, the minimum flow was increased, the present values are 50 m^3/s (from 1st of September october to 30th of Avril) – 70 m^3/s (from 1st of May to 31st of August). Klingeman et al. (1998) have reviewed the dewatering impacts and concluded that "the small amount of water left in natural streams after bypass development or large-scale water diversion may appear to be sufficient initially, but may over time be insufficient to keep the former aquatic ecosystem intact. Longer-term changes in aquatic biota and the stream and flood-plain plant life may include: 1) decrease in numbers for many populations; 2) shifts of populations; 3) intrusion of monopolistic populations in niches and gaps; 4) arrival of new species; and 5) appearance of new types of communities." Such conclusions highlight the complexity of the ecosystem changes and the wide range of possible impacts, direct or indirect, immediate or delayed, on aquatic populations. In this context, it is no easy to link the observed changes in populations and the responsible factors. Using local habitat preferences of organisms can help to understand how human activities can modify population or community structures and provide tools to improve river management. In this paper we will focus on the interpretation of the data concerning fish community collected since 1989 and the first results obtained by using the "Norwegian mesohabitat method " as described by Harby et al. (2005).

The "mesohabitat method" is used to evaluate the amount of habitat available for fish according to their hydraulic characteristics. Using this approach, the aim of the study was to evaluate both the proportion and availability of each mesohabitat type and the potential modifications in fish community or populations in the next years.

2 METHODS

2.1 Fish sampling

Point abundance electro-fishing sampling (P.A.S.) (Nelva et al. 1979, Persat & Olivier 1991) has been performed in the bypass section from 1989 to 2004 in order to evaluate the long-term effect of off-channel hydropower bypass scheme. Only data collected during autumn periods have been taken into account in the present study to describe seasonal variations due to the presence of a large amount of 0+ fish at that season compared to spring. The sampling effort was: 50 P.A.S. from 1989 to 1998 and 100 P.A.S from 1999 to 2004. Data were not collected during autumn 2001 and 2002 because power station management problems (large discharge fluctuations in the bypass section). Data are given as fish abundance for 100 P.A.S.

2.2 Mesohabitat mapping and classification

The Norwegian method (Borsányi et al. 2003) of classifying the river section into maximum ten different physical meso scale morphological classes (mesohabitat classes) was applied at 10 and 70 m^3/s. The classification is carried out by manual observations and estimations of four key factors: surface pattern (wave height), surface longitudinal gradient, water velocity and water depth. The factors are generalised for a wet area of at least one river width in the length direction. It is possible to make maximum 3 classes across the river (see details in Harby et al. 2005).

2.3 Habitat models for fish

The mesohabitat method may be used in combination with habitat-hydraulic models (see Harby et al. 2005).

In this part of the study, we have chosen to assign a preference to each mesohabitat class for each taxa (fish species or size groups).

Fish habitats preferences have been summarised from data available in Lamouroux et al. (1999a) for 18 species and in Mallet et al. (2000) for grayling.

For 14 species, the authors have defined size classes and habitat preferences were computed for each species size class (table 1).

3 STUDY SITE

Among the eight bypass sections which are being rehabilitated, the Chautagne bypass section is the first one on the upper-Rhône (between Geneva and Lyon). Bypass scheme operations affect all hydrology aspects, but the main effects are 1) the duration of the period with minimum instream flow (almost 80% of time) because the canal capacity (700 m^3/s) exceeds highly the mean annual discharge (400 m^3/s) and 2) the fast increase of the discharge during spates, changing the hydraulic conditions in the bypass section suddenly. This artificial hydrological regime induces specific ecological conditions. The impacts of such conditions are difficult to assess and such questions need long term surveys of populations. Most of the time, the bypass section looks like a small river but large floods and spates play a major role on the sediment transport processes and the shaping of aquatic habitat.

Fish were sampled in four stations considered as representative (Fig. 1).

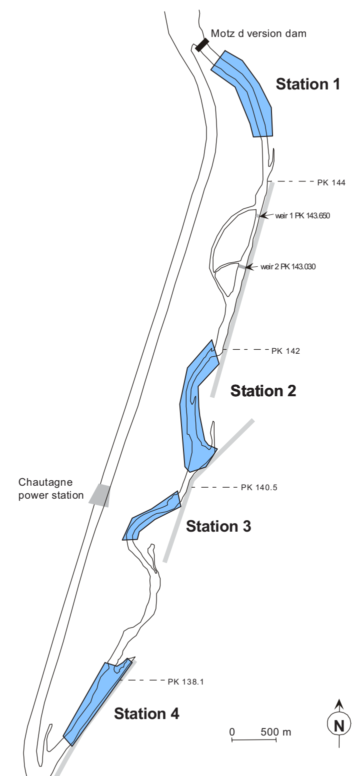

Figure 1. Chautagne bypass section and location of the 4 fish sampling stations

The 12th of July 2004 the minimum flow was increased from 20 to 70 m^3/s (Fig. 2).

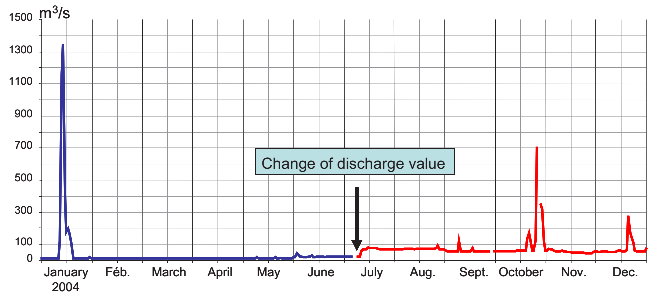

Figure 2. Hydrograph for the Rhône River at Chautagne 2004 (mean daily discharge in the bypass section).

4 RESULTS

A total of 28 fish species have been sampled from 1989 to 2004 in the Chautagne bypass section. Among them, 5 are exotic species and 11 were regularly sampled during the studied period: stream bleak (*Alburnoïdes bipunctatus)*, brown trout (*Salmo trutta fario)*, grayling (*Thymallus thymallus)*, tench (*Tinca tinca)*, chub (*Leuciscus cephalus)*, gudgeon (*Gobio gobio)*, perch (*Perca fluviatilis)*, barbel (*Barbus barbus)*, minnow (*Phoxinus phoxinus)*, stone loach (*Barbatula barbatula)* and ruffe (*Gymnocephalus cernuus)*.

4.1 Fish mesohabitat preferences

For the 18 most common species mesohabitat preferences using the classes defined by Harby et al (2005) have been established using Lamouroux et al (1999a) and Mallet et al. (2000) data. Classification has been made mainly by using fish preferences for flow velocity and water depth. Results are presented in Table 1.

26 size classes or species show preferences for mesohabitats C or D (lentic section more or less deep). Among these, 12 are the smaller size class of several species. Only young grayling and stone loach prefer fast flowing habitats.

Older rheophilous fish usually shift from slow flowing habitat to fast flowing habitat: blageon (*Leuciscus soufia)*, bleak (*Alburnus alburnus)*, stream bleak, barbel and nase (*Chondrostoma nasus)*.

4.2 Long term trends in fish abundance

The analysis of fish abundance variability from 1989 to 2004 showed that several groups of species could be identified according to their ability to maintain population after discharge reduction since 1980.

Species such as barbel or chub showed a relative stability of their abundance during this period, and moreover these species were able to maintain healthy populations in this part of the river (Fig. 3). The size classes defined above have been used to see the evolution of each size class.

Table 1. Mesohabitat preferences for the 18 more common species sampled at Chautagne from 1989 to 2004.

Species scientific name	Size class limits (cm)	Mesohabitat preferences
Salmo trutta fario	-	B2
Thymallus thymallus	[0-19[	B2G2
Thymallus thymallus	[19-30[	B1G1
Thymallus thymallus	>30	B1G1
Rutilus rutilus	[0,6[	D
Rutilus rutilus	[6,11[	C
Rutilus rutilus	>11	C
Leuciscus soufia	[0,8[	D
Leuciscus soufia	>8	B2G2
Leuciscus leuciscus	[0,9[	CD
Leuciscus leuciscus	[9,19[	CD
Leuciscus leuciscus	>19	CD
Leuciscus cephalus	[0,9[	D
Leuciscus cephalus	[9,17[	D
Leuciscus cephalus	>17	C
Phoxinus phoxinus	[0,4[	D
Phoxinus phoxinus	>4	D
Tinca tinca	-	C
Alburnus alburnus	[0,8[	D
Alburnus alburnus	[8,12[	B1G1E
Alburnus alburnus	>12	B1G1E
Alburnoides bipunctatus	[0,7[	D
Alburnoides bipunctatus	>7	B2
Blicca bjoerkna	-	C
Abramis brama	-	C
Chondrostoma nasus	[0,8[	D
Chondrostoma nasus	[8,19[	C
Chondrostoma nasus	>19	B1
Gobio gobio	[0,10[	D
Gobio gobio	>10	D
Barbus barbus	[0,9[	D
Barbus barbus	[9,22[	B2G2
Barbus barbus	>22	B1G1
Barbatula barbatula	[0,6[	B2G2
Barbatula barbatula	>6	B2G2
Lepomis gibbosus	[0,8[	C
Lepomis gibbosus	>8	C
Perca fluviatilis	[0,10[	D
Perca fluviatilis	>10	C

Abundances of minnow, stone loach, gudgeon and ruffe increased during the last 15 years (Fig. 4) while abundances of nase, dace (*Leuciscus leuciscus*) (Fig. 5), stream bleak, grayling and trout decreased progressively. For these last species, young-of-the-year (Y-O-Y) fishes are still regularly caught but usually in very low number.

The size class distribution of nase (Fig. 6) show that abundances have greatly decreased since 1992 and that adult fish are now almost absent in this bypass section. Only few Y-O-Y are still caught now, revealing the existence of reproduction, 1+ or 2+ fish have not been caught since 1994. Size class structure evolution during the same period was almost the same for dace.

Trout and grayling maintain natural populations but densities were very low. Grayling reproductive success can be relatively high and then recruitment seems to be effective in the population, allowing the survival of the species in this bypass section.

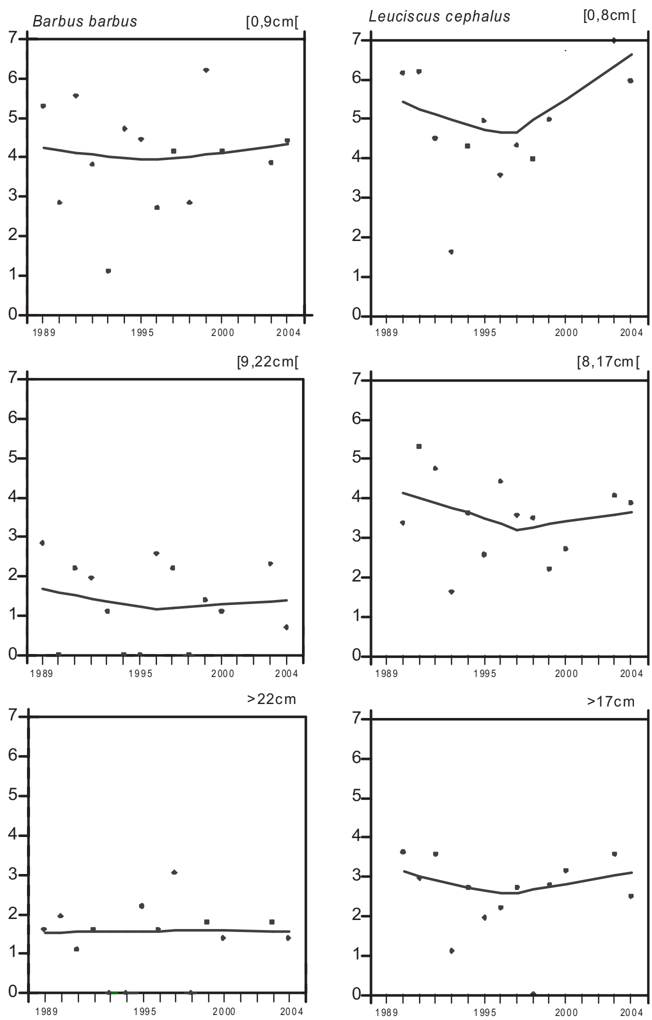

Figure 3. Barbel and chub abundances calculated for 100 P.A.S. from 1989 to 2004. Data are presented by size classes (see table 1). Points represent the data, lines represent smoothed data.

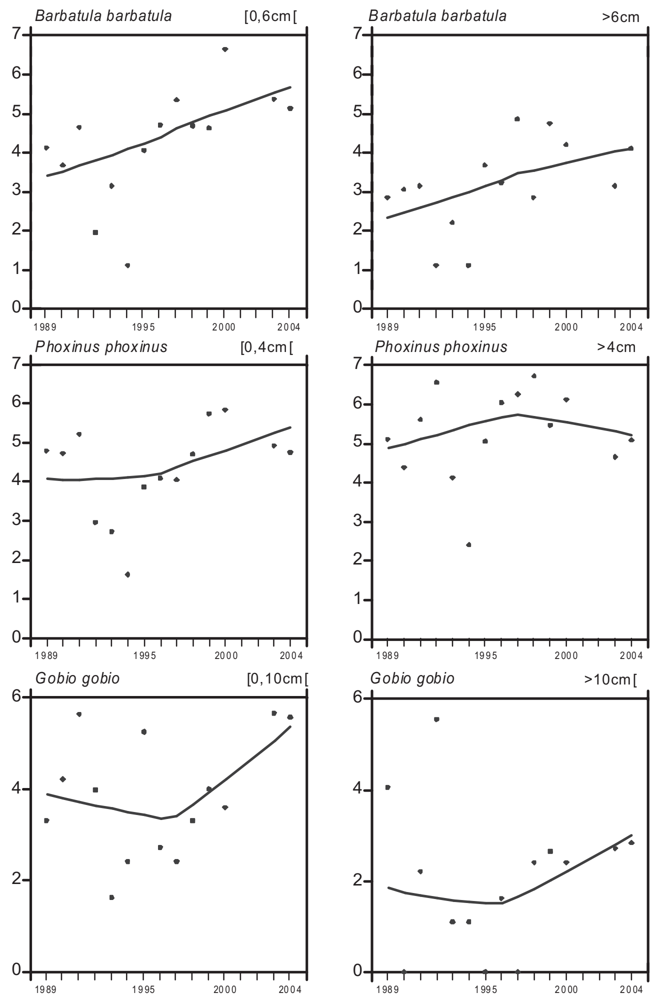

Figure 4. Stone loach, minnow and gudgeon abundances calculated for 100 P.A.S. from 1989 to 2004. Data are presented by size classes (see table 1). Points represent the data, lines represent smoothed data

4.3 Mesohabitat classification at 10 and 70 m^3/s.

Here we only summarise results presented by Harby et al. (2005) (see table 1-2 in Harby et al. for mesohabitat class definition):

- the total wetted area increases to 173 903 m^2 as the discharge increases from 10 m^3/s to 70 m^3/s,
- mesohabitat C (deep and slow) dominates in the bypass section at both 10 and 70 m^3/s, probably because the river bed is naturally calibrated for a higher discharge (400 m^3/s),

- at low flow, classes A and E (steep, deep and fast) are missing, and B1-G1 (moderate surface gradient, deep and fast) and H (moderate surface gradient, shallow and slow) cover a very small area,
- at 70 m^3/s, all classes are present, mesohabitats A, E, G2 (moderate surface gradient, shallow and fast) and H cover small area ; area of mesohabitats B1, B2 (moderate surface gradient, shallow and fast), C and G1 increase significantly, mesohabitat D area (moderate surface gradient, shallow and slow) is considerably reduced,
- the diversity of mesohabitats (Shannon-Weaver and Simpson indices) is higher at 70 m^3/s than at 10 m^3/s,
- the Chautagne bypass section is a relatively short section and the spatial distribution of mesohabitats show that the spots of high mesohabitat diversity are concentrated at the upstream and downstream riffles of meanders.

4.4 Mesohabitat changes between 10 and 70 m^3/s and implications for fish.

Table 2 shows the changes of mesohabitat availability for each size-class or species as the minimum flow increases from 10 to 70 m^3/s. Changes are expressed in a qualitative form, 4 categories ranging from + + (significant increase) to - - (significant loss).

The most important loss concerns mesohabitat class D which is the preferred habitat of almost all juvenile fish.

The strong reduction of mesohabitat D area at 70 m^3/s should be analysed carefully because large gravel bars provide shallow, sunny gravel habitats along the shore with a velocity gradient from the shore to the channel. Such habitats are very suitable for most of Y-O-Y fish species and should be considered as equivalent to mesohabitat D. The proportion of such gravel beaches will probably not be affected by minimum flow change. The increase of the availability of classes B1, B2, G1, G2, E at 70 m^3/s is favourable for rheophilous species like trout, grayling, blageon, nase, stream bleak, barbel, stone loach and bleak.

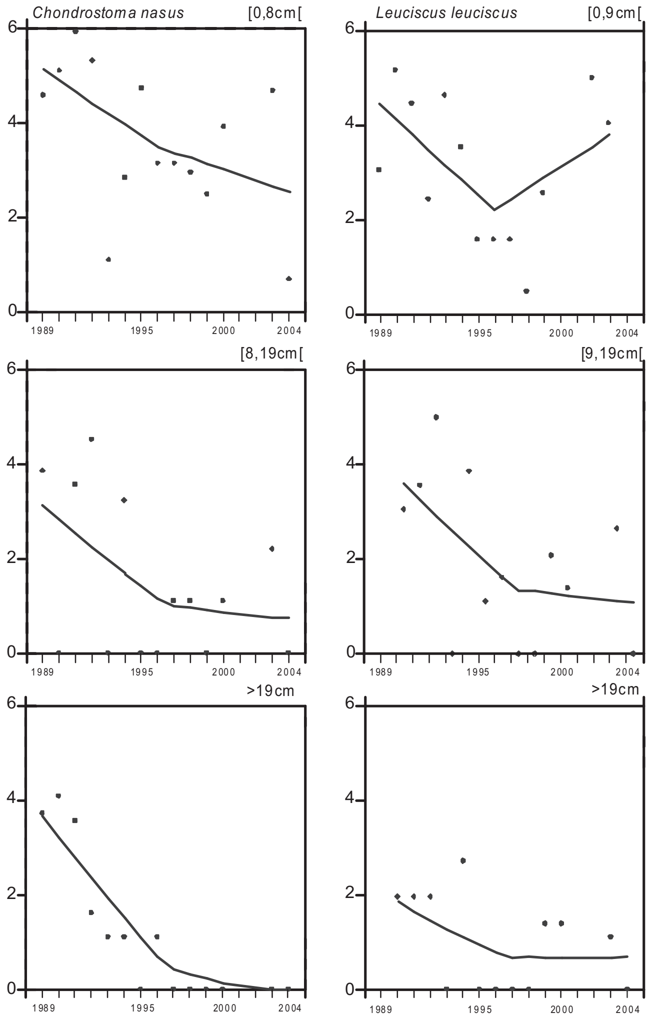

Figure 5. Nase and dace abundances calculated for 100 P.A.S. from 1989 to 2004. Data are presented by size classes (see table 1). Points represent the data, lines represent smoothed data

5 DISCUSSION

Bypass operations create particular environments quite different form natural rivers. The bypass section of the Rhône River at Chautagne receives since 1980 a residual flow which corresponds most of the time to the discharge of a small river. Nevertheless, during spate and floods, this part of the river receives large quantities of water and during these periods, hydraulic forces govern sediment transport processes and then determine the nature of habitat for aquatic organisms. The mesohabitat method used in this study does not take into account the dynamic aspect of habitat patterns, and at this stage we can only compare static flow situations (i.e. “low” and “high” flow). The method does not take substrate size or cover into account directly. Very often there is a link between the other physical factors and substrate, but this link may be altered in regulated rivers.

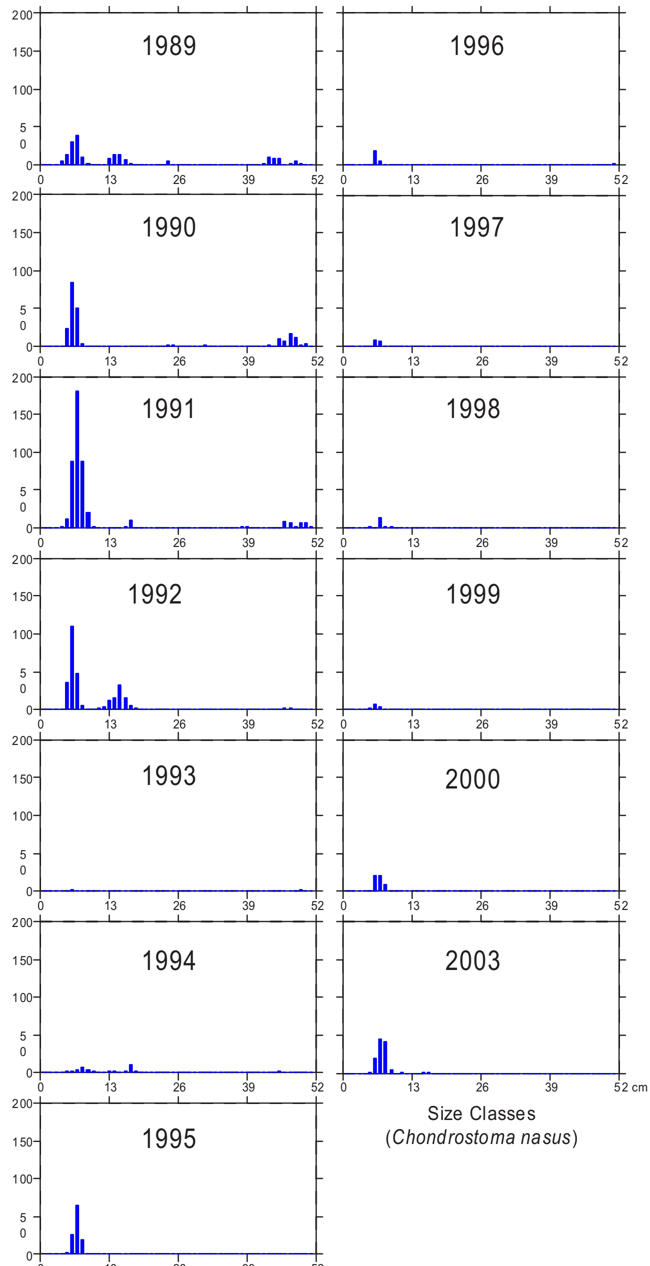

Figure 6. Size class distribution (in cm) of nase caught during autumn in Chautagne bypass section from 1989 to 2004.

For these reasons, linking fish community or populations parameters to the availability of different habitat classes at a given discharge has to take into account long term trends in community (at least integrating the period of hydroelectric development).

Table 2. Changes in mesohabitat availability for each species or size-class as the discharge increased from 10 to 70 m^3/s. Changes are expressed as follows: + + large increase, + small increase, empty cell: no change, - loss, - - important loss of mesohabitat area.

Species	Size-class limits (cm)	Mesohabitat preferences	Change
Salmo trutta fario	-	B2	+
Thymallus thymallus	[0-19[	B2G2	+
Thymallus thymallus	[19-30[	B1G1	++
Thymallus thymallus	>30	B1G1	++
Rutilus rutilus	[0,6[	D	- -
Rutilus rutilus	[6,11[	C	
Rutilus rutilus	>11	C	
Leuciscus soufia	[0,8[	D	- -
Leuciscus soufia	>8	B2G2	+
Leuciscus leuciscus	[0,9[	CD	
Leuciscus leuciscus	[9,19[	CD	
Leuciscus leuciscus	>19	CD	
Leuciscus cephalus	[0,9[	D	- -
Leuciscus cephalus	[9,17[	D	- -
Leuciscus cephalus	>17	C	
Phoxinus phoxinus	[0,4[	D	- -
Phoxinus phoxinus	>4	D	- -
Tinca tinca	-	C	
Alburnus alburnus	[0,8[	D	- -
Alburnus alburnus	[8,12[	B1G1E	+ +
Alburnus alburnus	>12	B1G1E	+ +
Alburnoides bipunctatus	[0,7[	D	- -
Alburnoides bipunctatus	>7	B2	+
Blicca bjoerkna	-	C	
Abramis brama	-	C	
Chondrostoma nasus	[0,8[	D	- -
Chondrostoma nasus	[8,19[	C	
Chondrostoma nasus	>19	B1	+ +
Gobio gobio	[0,10[	D	- -
Gobio gobio	>10	D	- -
Barbus barbus	[0,9[	D	- -
Barbus barbus	[9,22[	B2G2	+
Barbus barbus	>22	B1G1	+ +
Barbatula barbatula	[0,6[	B2G2	+
Barbatula barbatula	>6	B2G2	+
Lepomis gibbosus	[0,8[	C	
Lepomis gibbosus	>8	C	
Perca fluviatilis	[0,10[	D	- -
Perca fluviatilis	>10	C	

The analysis of abundance evolution of the most common species during the last 15 years has shown that for several species, especially large rheophilous species (nase, dace, grayling and trout), highlight severe problems threatening population survival (low abundance and species size-class structures). Because large rheophilous cyprinid species in large rivers are long-lived (Schlosser 1990), consequences of river regulation may appear several years later, when the older individuals of the population are affected. These species were considered as typical from this part of the Rhône River before damming.
On the contrary, species such as minnow, stone loach and gudgeon which are more specific to smaller streams have developed strong populations with high reproductive success under the hydrologic conditions until summer 2004, The barbel has developed a strong population too. These results agree with the mesohabitat availability at low flow (10 m^3/s) (mesohabitats D, B2, G2). For many species found in this bypass section, and probably for all of them, large pools (mesohabitat C) play an important role as refuge against high flow velocities or predators.

In such a context, the "mesohabitat method" used here can be considered as very useful to explain the present state of the community or populations, for example the rarity of class B1 at 10 m^3/s could be, at least to a certain extend, responsible of the decline of the nase population. On the other hand, it is difficult to explain the dace population decline with similar arguments, as the C and D classes were widely represented at low flow. Such difficulties in data interpretation highlight the complexity to assess ecosystems evolution. As usually in ecology, regulation of population parameters depends on several factors and it is not easy to identify key factor(s) which could be considered as a limited factor for a given population or a given developmental stage at a given moment.

The increased proportion of mesohabitats B1 and G1 after the increase of minimum flow, should lead to the recovery of rheophilous populations. These predictions agree with those obtained using other habitat models like ESTIMAB (Lamouroux et al 1999b). The observed distribution of mesohabitats at 70 m^3/s provides a direct and immediate picture of the effects of flow increase on the availability of instream habitat.

The mesohabitat classification method used in this study was originally designed for the assessment of the physical habitat for Atlantic salmon (*salmo salar*) in small and medium-sized streams (several thousands of km^2 catchment area or several hundreds of m^3/s as mean flow) (Borsanyi 2003). Our study in the Rhône River shows that it is possible to apply the method to a large river, even though the bypass section at Chautagne may be seen as a medium-sized river.

To keep the method simple and comparable, we preferred to use the same physical limits between different meoshabitat classes as in Borsanyi et al (2003). As long as we can assign fish preferences to the different mesohabitat classes, the physical limits may be reasonable. However, they may also be better fitted to the actual species found in the Rhône River at Chautagne.

Some of the on-going studies in Norway on the use of this method for Atlantic salmon (Forseth, Halleraker, Harby, Ugedal, pers.comm.), show only minor differences in preference between several mesohabitat classes. In the study of the Rhône River, there are clear differences in preference for mesohabitats especially between rheophilous and other species, but also between different size classes within some species, probably leading to more confident results.

The development of the fish population at Chautagne and other sites in the Rhône River after the change in the minimum flow will be followed carefully and also give important validation of our mesohabitat method used with preferences developed from a large number of studies reported in the literature. A regular establishment of such an evaluation of mesohabitat distribution together with a long term survey of fish community should help in the understanding of future evolution of the ecological status of this part of the Rhône.

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Northeast Instream Habitat Program

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ABSTRACT: The Northeast Instream Habitat Program (NEIHP) is a multidisciplinary initiative that aims to improve the scientific and methodological foundation for the ecologically sound, sustainable management of running waters. It is an integrated research, teaching and extension program with a strong commitment to research and development, sustained outreach, training and technical assistance. It addresses the needs of resource conservationists and stewards, receiving feedback from stakeholder advisory committees. The NEIHP employs a mesohabitat-scale approach to river management assessment using the habitat modeling system MesoHABSIM. The result of the model is a seasonal description of the magnitude, frequency and duration of critical habitat events as well as a quantitatively evaluated selection of restoration scenarios (e.g. channel improvements, flow augmentation). NEIHP tools have been applied to multiple projects in the Northeastern US, and the techniques have become part of legislative recommendations for the development of statewide instream flow standards in Connecticut, Massachusetts and New Hampshire.

1 INTRODUCTION

There are two important issues facing instream flow management in the Northeastern United States: an increasing demand for water to meet human needs while protecting aquatic ecosystems; and the lack of adequate planning tools for management of running waters. The Northeast Instream Habitat Program (NEIHP) is a multidisciplinary research and outreach initiative that aims to improve the scientific and methodological foundation for the ecologically sound, sustainable management of running waters (www.neihp.org).

The NEIHP was founded as a partnership among the University of Massachusetts at Amherst, the US Geological Survey, US Fish and Wildlife Service, and the Environmental Protection Agency. It addresses the needs of resource conservationists and stewards, and receives feedback from advisory committees of state and federal agencies and NGO's. The NEIHP is part of the New England Regional Water Quality Program and is funded by USDA §406 Water Quality funds, UMass Extension, state and federal natural resource agencies and NGO's.

NEIHP is an integrated research, teaching and extension program with a strong commitment to research and development, sustained outreach, training and technical assistance, and development of expanded graduate and undergraduate education. NEIHP tools have been applied at multiple projects in the Northeast region (CT, MA, NH and NY). The techniques have become part of legislative recommendations for the development of state-wide instream flow standards in Connecticut, Massachusetts and New Hampshire.

As a core tool the NEIHP employs a mesohabitat-scale approach to river management assessment using the habitat modeling system MesoHABSIM (Parasiewicz 2001).

2 MESOHABSIM METHOD

Mesohabitat Simulation Model (MesoHABSIM) addresses the requirements of watershed-based management of running waters. It builds upon pre-existing physical habitat simulation models (e.g. PHABSIM) to predict an aquatic community's response to habitat modification. The Physical Habitat Simulation System (PHABSIM) is a central part of the US Government's method of setting minimum stream flow requirements (Stalnaker 1995). MesoHABSIM is an enhancement of this system, developed during a restoration study on the Quinebaug River, located in Massachusetts and Connecticut. Biological responses to change in a river, along with physical attribute data, are used to provide the basis for assessment. MesoHABSIM modifies the data acquisition technique and analytical approach of similar models by changing the scale of resolution from micro- to meso-scales. Earlier methods were limited to a few short sampling sites, and this resulted in the simplification of hydraulic conditions. Due to an increase in scale, the MesoHABSIM model takes variations in stream morphology along the river into account and is more applicable to large-scale issues. We believe that habitat and fish measurements at larger spatial units are more practical, more relevant to river management, and more conducive to habitat modeling.

Mesohabitat types are defined by their hydromorphological units (HMUs, such as pools and rapids), geomorphology, land cover and other hydrological characteristics. Mesohabitats are mapped under multiple flow conditions at extensive sites along the river. Fish data are collected in randomly distributed mesohabitats and detailed habitat surveys are conducted in the fishing sites. This allows modeling of available fish habitat at a range of flows. Rating curves represent the changes in relative area of suitable habitat in response to flow and allow for the determination of habitat quantity at any given flow within the range of surveys. These rating curves can be developed for river units of any size making them useful for drawing conclusions about the suitability of channel patterns or habitat structures for various species of fish. This can be done for specific sections or for the entire river. Rating curves can also be used to evaluate the benefits of various restoration measures on the entire fish community (Figure 1). In combination with hydrologic time series, rating curves are used to create Continuous-Under-Threshold (CUT) curves for the analysis of frequency, magnitude and duration of significant habitat events. The CUT curve technique described by Capra et al. (1995) helps us define critical thresholds and determine what habitat variability and availability is necessary to support the target river fauna. CUT curves evaluate durations of unsuitable habitat under a specified threshold by comparing continuous durations in days under this threshold to the cumulative durations in the study period. A highly useful product of the CUT curves are reference tables that managers can use to determine how long a given species can tolerate unsuitable conditions depending on its life stage (Table 1).

2.1 Validation of MesoHABSIM

During the Quinebaug River study (see below) we performed additional experiments to determine how the use of different habitat models (micro-habitat models versus MesoHABSIM) could influence the results of instream-habitat assessment and therefore the conclusions for river management. We also wanted to determine the sources of potential disagreement between the models by addressing the following issues:

- Use of microhabitat versus mesohabitat scales for habitat-data collection
- Use of multivariate versus univariate habitat-suitability criteria
- Spacing of cross-sections and cell lengths
- Extrapolation of site-based models to the study area
- Species-specific bias of the models

Table 1. An example of summer flow augmentation criteria developed for a hypothetical example.

July – September	Typical	Critical	Trigger	Minimum
Habitat threshold (% suitable area)	54	51	34	15
Corresponding flow (ls^{-1})	600	400	350	200
Allowable duration below (days)	18	10	5	0
Duration of pulse (days)	2	2	2	2
Duration before triggering catastrophe (days)	37	31	28	1
Allowable recurrence of catastrophe (years)	7	15	30	0

The habitat should never get below minimum. When it stays under the trigger threshold, management action is required in the form of a flow pulse to the next threshold level. The corresponding flow depends on the restoration scenario (more restoration, less flow is allowable). The catastrophic durations should not occur more frequently than specified in the table.

The experiment conducted for accomplishing these goals was to develop three types of models for a given portion of the Quinebaug River: a microhabitat model with univariate habitat-suitability criteria (PHABSIM), a microhabitat model using multivariate criteria (HARPHA, Parasiewicz et al. 1999), and a mesohabitat model with multivariate criteria (MesoHABSIM). Both microhabitat models were computed using two different calculation methods for cell length. The rating curves produced by each model were compared at the site scale as well as at the study-segment scale. The validation of model predictions was conducted with the help of separate fish collections in the study area. The following conclusions were made:

- Of the three models, only MesoHABSIM passed the validation test.
- The HARPHA model produced results that were much closer to those of MesoHABSIM than standard PHABSIM.
- Univariate habitat suitability criteria that are combined into composite suitability indices using algorithms defined *a priori* are a main source of error, especially for non-salmonid fishes.
- The second largest source of inaccuracy is second-stage error introduced during extrapolation of microhabitat observations.
- Use of different algorithms for the determination of cell length for microhabitat models have only a marginal influence on the model results.

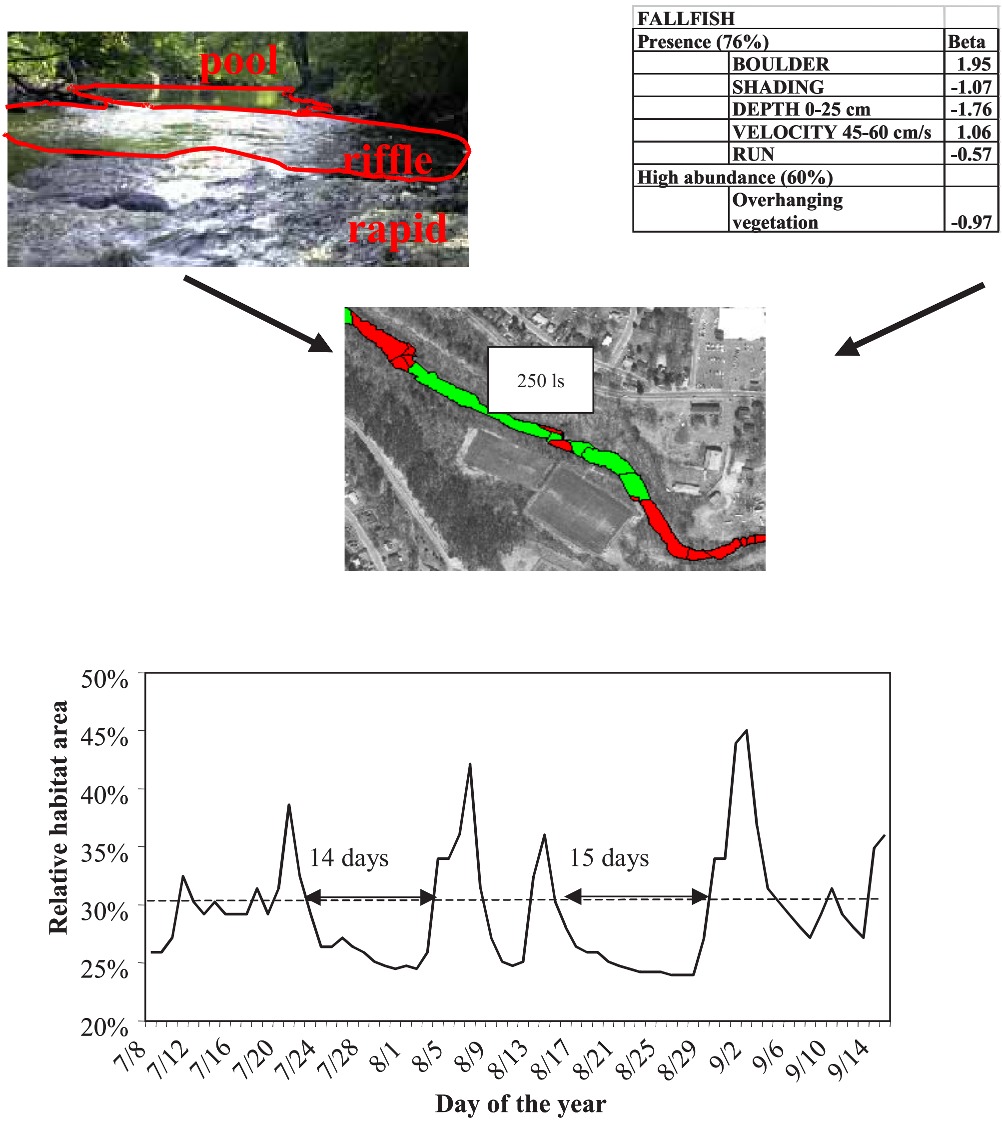

Figure 1: Schematic representation of the MesoHABSIM process. The habitat survey delineates hydromorpologic units and their physical attributes (top left). The fish survey identifies key habitat attributes affecting fish (top right). The model calculates the probability of fish presence within each habitat and delineates suitable areas (in green). A habitograph (bottom) combines the rating curve with flow time series and is used to analyze durations and frequency of specific habitat events with the help of the CUT curve technique.

2.2 *Reference fish community*

To use physical habitat models to analyze and predict ecosystem potential, we must also determine the composition of the native fish community and select a subset of species for model development. Our development of a Reference Fish Community (RFC) is based on the Target Fish Community (TFC) approach described by Bain & Meixler (2000). A comprehensive list of species is generated from literature sources and available regional data collected on relatively intact river reaches. The species are ranked on the basis of abundance in long-term fish collection data from multiple rivers of similar character. Securing habitat for naturally occurring dominant species should preserve the most profound characteristics of the ecosystem, providing survival conditions for the majority of the aquatic community and therefore a reference for restoration efforts. The simplest way to create a river habitat model is to select the five to ten highest-ranking species for model development. We can assume that community structure reflects habitat structure, so the most common species should indicate the most common habitat. Since habitat availability forms the structure of the aquatic fauna, the affinity between the structure of the river habitat and the structure of the fish community can be used as a measure of habitat quality.

The results of MesoHABSIM and Reference Fish Community create the framework for integrative analyses of many aspects of the ecosystem. This also allows managers to recreate reference conditions and evaluate possible instream and watershed restoration measures or alterations, such as dam removals or changes in water withdrawals. From the perspective of resource managers, this methodology not only allows for quantitative measures of ecological integrity, but also creates a basis for making decisions where trade-offs between resource use and river restoration need to be considered.

3 PROJECT EXAMPLES

3.1 Ecohydrology study of the Quinebaug River

The Ecohydrology Study of the Quinebaug River in Massachusetts and Connecticut focuses on assessing the river's biophysical conditions, habitat deficits, and potential improvement measures (Parasiewicz 2005). It is part of a multidisciplinary investigation required by the US Army Corps of Engineers Section 404 permit program and by the Massachusetts Department of Environmental Protection (DEP) Section 401 Water Quality Certification for the Millennium Power Project in Charlton, Massachusetts. The study began in Fall 1999 and was conducted by the Instream Habitat Program of the Department of Natural Resources at Cornell University. The results of the study provide a basis for future decision-making processes and the design of an implementation plan.

The MesoHABSIM model for the target fish community is one of the principal tools used in this investigation. Hydromorphology, fish habitat, fish density, invertebrate samples, and temperature data were analyzed in every section to determine the present condition of the river and its restoration potential (Parasiewicz 2003).

The Quinebaug River is a fourth-order river with multiple impoundments and a history of industrial use. Within the study area, the different river sections demonstrate a wide range in condition, type, and degree of environmental impact. The study identified a number of ecological deficits with regard to fish habitat, river morphology, flow and thermal regime, and also noted the presence of pollution. We presented a series of recommendations, such as dam removals and channel restoration, as essential measures to strengthen the resistance of the fish community to environmental stressors. This does not, however, remove the stress factors, which are most likely low flows, high temperatures, and pollution. The main factors controlled in this analysis are flow levels and continuous durations of low flows. Long lasting, persistent droughts reduce available habitat and also dramatically increase the impact caused by thermal stresses and pollution. With low base flows, the effects of sudden flow reductions and peaks are more dramatic.

Because of the limited amount of water available in the system, we developed a dynamic flow-augmentation scheme to reduce impacts on fish habitat, water temperature, and pollutant concentrations attributable to those consistently low flows. Using a CUT-curve technique, we determined allowable periods of low-flow conditions and ecosystem-specific release procedures by comparing a simulated habitat time series representing the flow regime in an unaltered watershed with a time series observed on the Quinebaug Gauge between 1948 and 1994. As a result, we determined a series of seasonal habitat thresholds together with durations and frequencies of naturally occurring habitat events for the intra- and inter-annual scale. We also developed a set of experimental rules for a habitat augmentation strategy that applies short "pulses" of water to increase habitat availability levels to mitigate the effects of persistent drought conditions. The amount of water necessary to achieve the required habitat levels was calculated for various river corridor restoration scenarios (Table 1). To test our augmentation procedures, we performed two evaluation tests, one using a hypothetical habitat time series and one applying our rules to flows that occurred in 1994 (a drought year). The latter test showed that the number of flow interventions necessary was fairly limited, underlining the feasibility of a pulse-based approach (Fennessey 2004). A set of flow-chart templates was developed to guide the planning of an augmentation scheme by utilizing flow releases from dams.

3.2 Instream flow studies and watershed management plan for the Souhegan River designated reach

A goal of the instream flow studies and watershed management plan for the Souhegan River is the determination of Protected Instream Flow (PISF) values for designated reaches. PISF values must be established that protect legislatively mandated Instream Public Uses, Outstanding Characteristics, and Resources Entities, which may constrain water use by Affected Water Users in the Souhegan River basin. Consideration of PISF levels in relation to current and projected water use patterns in the basin will be an integral component of the Water Management Plan for the Souhegan River.

Under leadership at the University of New Hampshire (UNH) and in collaboration with Normandeau Associates Inc., the NEIHP is mapping the Souhegan River for specific fish habitats to develop a water management plan. Besides habitat mapping, the study includes SCUBA diving in impoundments, monitoring for mussels, dragonfly nymphs and fish as well as computation of a physical habitat model. Scientists from Normandeau Associates are modeling the influence of water levels in riparian and emergent wetlands on aquatic habitat and endangered species. The hydrological analysis is being conducted by

UNH hydrologists, and includes concurrent flow measurements, simulation of pre-colonial time series and ground water monitoring. The collected data and models will support a multi-criteria decision analysis, which is a foundation for the Water Management Plan for the Souhegan River. There are many groups of stakeholders involved in developing the plan and the study team coordinates with these parties and with the state government.

The State of New Hampshire's pilot program for the determination of instream flows for designated river segments is the culmination of years of discussion about the need for, geographic scope of and method of instream flow regulation statewide. The pilot program will evaluate both the scientific methodology for establishing an instream flow as well as the institutional framework developed to provide technical oversight and to solicit input from all stakeholders. The Souhegan River, which has been affected by recreation, water withdrawals, and returns from a wastewater treatment plant, is the first project area within this program.

3.3 Developing a stream management plan for the Stony Clove watershed

The New York City Department of Environmental Protection has expressed interest in integrating the above methodology with the presently used geomorphic approach to develop a Stream Management Plan for the Stony Clove watershed in Greene and Ulster Counties. MesoHABSIM and the Target Community Approach were incorporated into the habitat assessment work performed by the Greene County Soil and Water Conservation District. This study was conducted by the Instream Habitat Program at Cornell University. Our overall goal was to assist in the development of sophisticated grass-roots environmental resource protection programs and to enable local communities to see beyond single issues and consider the total ecological health of their landscape. To accomplish this we used a target fish community-based habitat assessment approach combined with geomorphological and fish biology studies to develop a stream management plan. This project demonstrated the application of MesoHABSIM as a means of integrating aquatic habitat management, flood protection and water quality protection. In this process of transferring expertise to local governments and state agencies, we provided a knowledge base of instream habitats, whereby ecological goals for the integrated management concept of the given watershed could be established (Parasiewicz et al. 2003).

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Defining Spatial and Temporal hydromorphological sampling strategies for the Leigh Brook river site

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ABSTRACT: Detailed surveys of depth and velocity are undertaken to describe hydro-ecological status of rivers. Fieldwork for these surveys is time consuming and expensive. This paper aims to describe the methodology applied in order to determine the most suitable depth sampling strategy for effective field data collection and river representation in time and space at the Leigh Brook river site, Worcester, UK.

The accuracy of three different sampling strategies for predicting depth at non-measured points has been compared and the mesohabitats that better characterise depth changes due to variations in discharge have been identified. The results show that depth changes due to discharge change are mainly located at shallow and deep glide mesohabitat types. The analysis for the comparison of sampling strategies indicates that grid sampling strategies give better results than regular transects. Since the results also show that higher errors in predictions are obtained in the deepest areas, higher sampling densities should be applied in these locations.

1 INTRODUCTION

Hydromorphological sampling strategies have been applied over time in order to obtain data sets for different purposes, such as assessing the habitat quality of rivers, environmental planning, appraisal and impact assessment, habitat modelling, hydraulic modelling, planning habitat improvements and hydromorphological assessment of rivers among others.

Different methodologies have been applied for collection of hydromorphological data according to the spatial scale of the objective, the accuracy required in the measurements, the country where the study is being carried out and the economic and temporal factors. Currently, the sampling strategies applied for hydromorphological data collection are those described in the following methodologies: River Habitat Survey (RHS) from the UK, the Systeme d'Evaluation de la Qualite du Milieu Physique (SEQ-MP) from France, the Landerarbeitsgemeinschaft Wasser (LAWA-vor-Ort) from Germany and the Physical Habitat Simulation Model (P-Habsim) from USA.

Fieldwork for these surveys is time consuming and expensive, yet little work has examined the most suitable sampling strategy for effective field data collection and river representation in time and space. Furthermore, some of the existing methodologies are very broad and they are not accurate enough to be applied for many objectives, there is no consensus on the appropriate length of river that needs to be sampled, the spatial variability between measured cross-sections is not considered in some simulation processes, the temporal variation (change of hydromorphological characteristics according to the discharge) is also not considered and there is a cumulative accuracy problem when trying to define the mesohabitats present in the study areas.

We can define spatial pattern of a hydromorphological parameter in a river as the rate of variance between points in relation to the distance between them, and temporal pattern as the hydromorphological changes of the river site due to variations in discharge. Recognition of spatial and temporal pattern of hydromorphological parameters can contribute to solve the limitations mentioned above by the implementation of effective and efficient sampling strategies. This will allow us to obtain the maximum information at the river site with lower time investment by positioning the sampling points at those locations that (1) provide more information about hydromorphological changes over time and (2) best allow the implementation of interpolation techniques (kriging) for the prediction of hydromorphological values at non-measured locations.

Emery *et al* (2003) studied spatial and temporal patterns in rivers by exploring the hydraulic functioning of riffle-pool bedforms, particulary the variations in the hydraulic performance of different bed oscillation morphologies. The study addressed the need for an integrated understanding of the hydraulic perfomance of riffle-pool sequences across a range of flows to complement existing physical habitat assessment methods.

Habersack (2000) described the river scaling concept and contributed to the understanding of river spatial patterns by summarizing various tools available to obtain data at non-measured sites. The river scaling concept (RSC) is an integration of two procedures: down and up-scaling. The down-scaling procedure starts at a catchment scale and goes through sectional and local scale to point scale. The approaches available for the river-scaling concept are deterministic models, stochastic models, dimensional analysis (Habersack, 2000) or even habitat mapping surveys (Maddock & Lander, 2002). Deterministic and stochastic techniques aggregate the results in order to derive average values for larger time-space scales. Deterministic models describe relations between different scales by distributing small-scale modelled values, resulting in a spatial or temporal pattern whilst stochastic models apply covariance and distribution functions. Dimensional analysis uses fractals (or tessellation methods) to determine spatial and temporal structure (Nestler & Sutton, 2000).

The objectives of our study were (1) to identify spatial pattern at the Leigh Brook river site by establishing a relationship between the spatial location of sampled points and the accuracy obtained when predicting values with stochastic models (geostatistics) at non measured locations and (2) to determine the temporal hydromorphological changes of the river site due to variations in discharge and link them to the mesohabitats and flow types present at the site. The study is part of a long term project whose aim is to provide a guidance tool for effective and efficient hydromorphological data collection for lowland river sites.

2 THE RIVER SITE

The Leigh Brook is a tributary of the river Teme and rises on the Malvern Hills in Worcestershire. Data were collected in a 198 m reach within the Knapp and Papermill Nature reserve, which is managed by the Worcestershire Wildlife Trust. The reach is 10 m to 15 m wide and 2 m to 3 m deep at bankfull discharge. The catchment area upstream of the reach is approximately 80 km^2 (Maddock and Lander 2002).

3 METHODS

3.1 Data Collection

Hydromorphological data were collected at the Leigh Brook by Ian Maddock and colleagues from Worcester College. Quantitative (velocity, depth) and qualitative (i.e. mesohabitat type, and flow type) variables were observed at 5429 georeferenced points at the site. Data was collected for two different discharges for a period of three days following dry weather and with steady flow conditions. The first survey was at 0.517 m^3s^{-1} (Q82), whilst the second was at 0.344 m^3s^{-1} (Q93) (Maddock and Lander, 2002). Derived variables, e.g. Froude number, were calculated from the quantitative data.

The observations were taken at 200 cross-sections located at 1m intervals in the streamwise direction. Points were distributed across each cross-section at a 0.5m interval. The topographical survey at each of the measured points was carried out with a Nikon NPL-820 Reflectorless total station. Average water velocities were recorded at 0.6 x depth at those points located inside the wetted width of each transect.

Mesohabitat types were defined for each point following the typology developed by Maddock and Bird (1996). The categorisation of the mesohabitat types was obtained by determining the dominant type in each cross section (Table 1). Flow types were assigned according to the typology used within the River Habitat Survey (Raven et al., 1998).

Table 1. Definition of mesohabitat types (after Maddock (1996)).

Mesohabitat Type	Description
Riffle	Relatively steep water gradient, coarser bed material than local vicinity, some broken water. Usually of limited downstream extent with deeper water evident both upstream and downstream.
Shallow glide	Relatively smooth and low gradient water surface compared to riffle. Differentiated from deep glide by the majority of water (<0.5m deep). Visible flow: clearly evident.
Deep glide	Relatively smooth and low gradient water surface. Differentiated from shallow glide by a greater proportion of water (>0.5m deep). Visible flow: clearly evident.
Pool	Smooth, low gradient water surface. Usually of limited downstream extent with shallower water evident both upstream and downstream.

3.2 Spatial patterns at the Leigh Brook river site

This study aimed to (a) identify the differences between sampling strategies for eight different indicators when predicting values at non measured locations with geostatistical techniques, (b) study the spatial distribution of the error encountered for the predicted values and (c) identify if there is a relation between mesohabitat or flow types and the difference encountered between observed and predicted values. 2583 points were used for the data analysis as these had measured values of velocity and depth and calculated Froude number for the two discharges considered.

Three sampling strategies have been compared for the Leigh Brook river site; random grids, stratified grids and regular transects. The points selected to represent the sampling strategies were obtained from the 2583 point original data set. Random grids were created by random selection without replacement, stratified grids by selection of points included at a specific depth, velocity or Froude number interval and regular transects by selection of equally spaced cross-sections. A preliminary study was carried out in order to identify the number of points to be included for the comparison of sampling strategies. Subsets of data with different number of points were extracted from the original data sets in order to compare the range, sill and nugget (Figure 1) of the variograms obtained for each sampling strategy and variable (depth, velocity and Froude Number). Differences between the variograms of the sampling strategies were identified when decreasing the data sets down to 428 points and thus, this was defined as the size of the sampling data sets.

It was also studied how (a) the random selection of points, (b) the intervals for the stratified sampling strategies and (c) the location of the regular transects affected the characteristics of the variogram by replicating the selection process of the subsets three times. Regular transects and stratified grids presented differences between replicated subsets whilst no differences were encountered for random grids. Hence, it was decided to include one replicate for the stratified grid and regular transect sampling strategies. The study was developed for measurements obtained at both discharges ($Q=0.344\ m^3s^{-1}$ and $Q=0.517\ m^3s^{-1}$).

Qualitative (mapping resolution, standard error maps) and quantitave (variogram assessment, mean squared error, standard error, r-squared, frequency distributions and predictions for cross-section/longitudinal profiles), indicators were analysed in order to characterise the differences between sampling strategies in terms of accuracy of the predicted values obtained, accuracy of mapping resolution and accuracy of water volume in the channel. The description for these indicators and the criteria for sampling strategy selection have already been described in previous publications (Rivas *et al*, 2004).

The variogram is a measure of the variance between data as a function of distance (Figure 1). Three different variables describe the variogram; the range, the sill and the nugget (Figure 1). The range is the maximum distance over which pairs of observations remain correlated, the sill is the maximum value of variance encountered in the semivariogram and the nugget represents the spatial dependence occurring over smaller distances than the smallest sampling interval. These variables combine together in the variogram equation, which defines the spatial behaviour of the variance of the parameter under observation. The values of range, sill and nugget have been calculated for each sampling strategy and compared to those obtained for the complete data set collected. The higher the difference between values is, the worse is the sampling strategy.

The study of the spatial distribution of errors (difference between observed and predicted values) has been carried out by mapping and visually comparing the georeferenced error at all the measured points. A General Linear Model (GLM) analysis has been carried out in order to identify if the spatial distribution of errors is related to the spatial distribution of mesohabitats and flow types.

3.3 Temporal pattern at the Leigh Brook river site

The objective of this study was to investigate the extent to which hydromorphological changes occurred in different mesohabitat types and flow types due to variations in discharge.

The variables included in the analysis were defined by the rate of change encountered, and were mesohabitat change, depth change, velocity change, flow change and Froude number change. 2583 points were used for the data analysis as these had measured values of velocity, depth and Froude number for the two discharges considered.

Two different studies were carried out: one identifying the changes from Q=0.517 m^3s^{-1} to Q=0.344 m^3s^{-1} and a second characterising the hydromorphological changes for the opposite direction (from Q=0.344 m^3s^{-1} to Q=0.517 m^3s^{-1}). The method employed was to take the categorical variables of the first discharge as a reference. The relationships between the different quantitative variables (dependent variable: velocity, depth and Froude change) measured and the categorical variables (independent variable: mesohabitat and flow type) were established using analysis of variance and covariance.

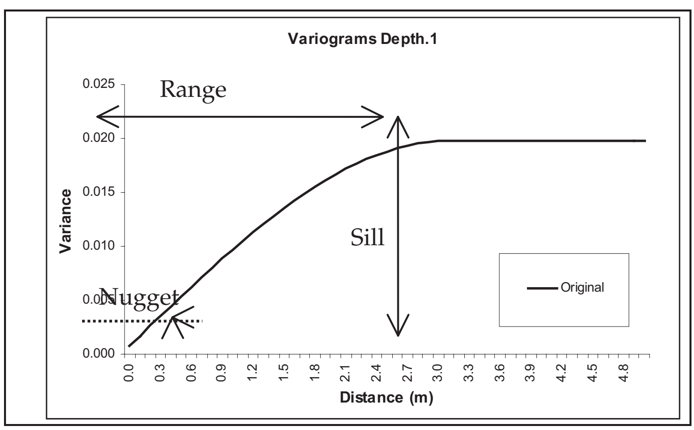

Figure 1. Variogram obtained for the original depth data set collected at the Leigh Brook river site (first discharge measured). Range, sill and nugget have been drawn to illustrate the concept.

The data set was analysed with multivariate techniques. A K-means clustering algorithm was applied to group the data into different clusters. After experimenting with different numbers of clusters, this was set to four as the patterns of clusters showed most correspondence with patterns of flow types. This is the same as the number of mesohabitat types identified on the Leigh Brook site. Each parameter was standardised prior to cluster analysis. This was because the parameters were measured at different scales, categorical and continuous parameters were used and the K-means algorithm is not invariant under scaling.

Histograms representing the percentage of points contained within each mesohabitat type included in each cluster were obtained to determine which clusters corresponded to each mesohabitat type. Since the number of points attributed to each mesohabitat was not comparable because not all mesohabitat categories had the same number of points, relative values to the size of the mesohabitat type sample were obtained to create a complementary set of histograms.

The spatial distribution of cluster classes was compared to the spatial distribution of mesohabitat types for both flow rates. Maps of velocity, depth, Froude number, mesohabitat and flow type changes were created to compare the spatial distribution of these variables with that of the cluster classes.

4 RESULTS

4.1 Spatial pattern at the Leigh Brook river site

The analysis of the variograms obtained for depth, velocity and Froude number shows that the variogram has the same behaviour for the two directions analysed (across and along the river). This indicates that the spatial pattern is comparable for both directions and therefore, sampling densities should be equal for both directions. Grid sampling strategies presented the best approximations of range to the original variogram and thus, these sampling strategies can be considered as the ones that better define the variogram (Table 2 and Figure 2). Regular transects give the worst approximations for range, sill and nugget and therefore are considered the worst sampling strategies for the recognition of spatial pattern. In addition, the results obtained are highly dependent on the selected location of the transects.

The values of range obtained for the variograms of the original data set (2583 points) indicate that the correlation between points is lost for distances higher than 3m for depth, 5m for velocity and 9m for Froude number (Table 2). This gives an approximation of the sampling distances that should be applied when collecting data at the Leigh Brook river site for these three hydromorphological parameters.

Table 2. Range, sill and nugget identified for some of the sampling strategies studied (Rand.G= Random Grid, Strat.G= Stratified Grid, Reg.T= Regular Transects and Original = 2583 points original data set).

		Rand.G	Strat.G	Reg.T	Original
Depth.1*	Range	2.570	3.150	1.700	3.090
(m)	Sill	0.0190.023	0.018	0.019	
	Nug	0	0	$6*10^{-4}$	$8*10^{-4}$
Depth.2**	Range	3.220	3.260	2.020	3.330
(m)	Sill	0.0190.017	0.018	0.019	
	Nug	0	$7*10^{-4}$	$1.7*10^{-4}$	$6*10^{-4}$
Vel.1	Range	3.160	8.730	1.790	5.000
(ms^{-1})	Sill	0.0420.048	0.041	0.052	
	Nug	0.0150.019	0.008	0.016	
Vel.2	Range	2.730	2.112	6.880	6.260
(ms^{-1})	Sill	0.048	0.041	0.100	0.048
	Nug	0.003	0.015	0.013	0.048
Froude.1	Range	9.800	7.040	95800	9.160
	Sill	0.022	0.022	176.0	0.042
	Nug	0.013	0.010	0.0080	0.008
Froude.2	Range	9.240	10.13	50326.1	9.170
	Sil	0.027	0.041	126.00	0.050
	Nug	0.017	0.006	0.011	0.009

*.1- For the first discharge measured.
**.2- For the second discharge measured.

The analysis of the spatial distribution of prediction errors indicates that the difference between observed and predicted values increases from riffle to pool mesohabitat types for depth, velocity and Froude number. Higher errors are also located at unbroken and broken standing waves for velocity and Froude number. The location of extreme and outlier errors is approximately the same for all the sampling strategies when mapping their values and coincides with the location of unbroken-broken standing waves for velocity or Froude number and Pool-deep glides for depth measurements. Higher sampling densities should be applied in deep areas when measuring depth and broken-unbroken standing waves when measuring velocity-Froude number in order to identify the peaks of variability in the semivariograms.

4.2 Temporal pattern at the Leigh Brook river site

The two discharges analysed are both towards the lower end of the flow duration curve and the changes that they produce on the hydromorphological variables are not large (Table 3). Only 9% and 39% of the data points experienced a change in their mesohabitat type and flow type respectively. Results indicate that shallow glides and riffles were the most frequent mesohabitat types identified, and rippled flow types were the most common.

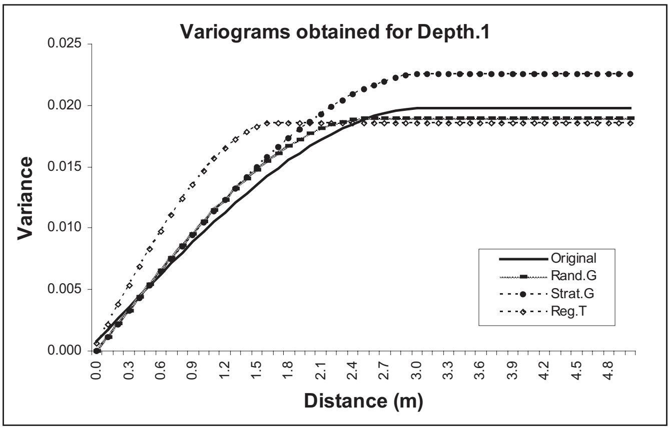

Figure 2. Comparison of variograms of depth (first discharge measured) obtained for some of the sampling strategies analysed.

Figure 3 shows the mesohabitat types and flow types that were more affected by the change in discharge; mesohabitat changes have been considered equal to 1 when a change appeared and equal to 0 when no change was registered whilst the rate of discharge change was calculated as the difference of flow type codes (positive changes indicate increase of flow type intensity whilst negative changes indicate decrease of the flow type intensity). Deep glides accounted for the majority of the changes and were significantly different to the other mesohabitat types. Three significantly different groups can be distinguished for the flow types: unbroken standing waves and broken standing waves group, smooth and rippled group and no perceptible flow group. Higher changes are identified in no flow and rippled and smooth flow types, which are strongly associated to the mesohabitat changes that experienced the highest rates of change (deep-shallow glide and pool mesohabitat types).

ANOVAs were used to test the following null hypothesis (Ho): the classes of each categorical variable have the same rate of change. The ANOVA tested the hypothesis between one categorical parameter and one quantitative parameter, creating the combination of tests presented in Table 4. The results indicate that the null hypothesis can be rejected for the following analysis: depth vs. mesohabitat type, velocity vs. mesohabitat type, velocity vs. flow type and Froude number vs. mesohabitat type.

The mean plots with 95% confidence intervals of depth change vs. mesohabitat showed that there are significant differences between deep glides and shallow glides. There were no significant differences between pools and riffles. Shallow glides presented the greatest depth changes followed by deep glides. Deep glides cannot be considered significantly different to riffles and pools due to the overlapping of the 95% confidence intervals. The mean depth changes were located between 0.037 m (riffles) and 0.048 m (shallow glide).

Table 3. Descriptive statistics for the continuous parameters measured.

Variable	max	Min	mean	std dev	median
Depth.1*(m)	0.94	0.02	0.27	0.15	0.25
Vel.1(ms^{-1})	1.53	-0.39	0.30	0.25	0.26
Froude.1	1.20	0.00	0.21	0.19	0.17
Depth.2** (m)	0.85	0.00	0.23	0.14	0.20
Vel.2 (ms^{-1})	1.51	-0.25	0.24	0.23	0.20
Froude.2	1.35	0.00	0.20	0.21	0.14
DepthChange (m)	0.58	-0.37	-0.04	0.06	-0.04
VelocityChange (ms^{-1})	0.98	-1.13	-0.06	0.16	-0.05
FroudeChange	1.27	-0.92	-0.01	0.13	-0.01

*.1- For the first discharge measured.
**.2- For the second discharge measured.

Since (a) the group constituted by pool-riffles did not experience high depth changes with discharge, (b) deep and shallow glides present higher depth changes than the rest of mesohabitat types and (c) the overlapping between confidence intervals of deep glides and the riffle-pools group is very small, it is possible to conclude that depth changes for the Leigh Brook site between the two measured discharges were mainly located at deep and shallow glides, which usually are a link between pools and riffles.

Velocity changes followed an inverse pattern to depth changes; mesohabitat types that present higher changes in depth did not have significant changes in velocity as all the energy was dissipated with the depth change. The opposite situation was encountered when analysing mesohabitat types with small depth changes. The results indicate that shallow and deep glides are those mesohabitat types that alternatively represent the maximum and minimum change for the different quantitative parameters analysed. Since they are never included in the same group, it is possible to consider them as the extremes of the hydromorphological changes. Pools and riffles switch between extremes according to the parameter analysed, and therefore it is difficult to establish a pattern on their grouping preferences.

The analysis of variance indicated that changes in Froude number are different for at least two mesohabitat types. Pools, riffles and shallow glides did not present significant differences as their mean values were very similar and the 95% confidence intervals overlap between them. Deep glides are different to other mesohabitat types although there is a small overlap between this mesohabitat and pools. Maximum Froude number changes were identified in deep glides and minimum changes in shallow glides, which supports the observations noted above regarding the representation of extreme values by these two mesohabitat types.

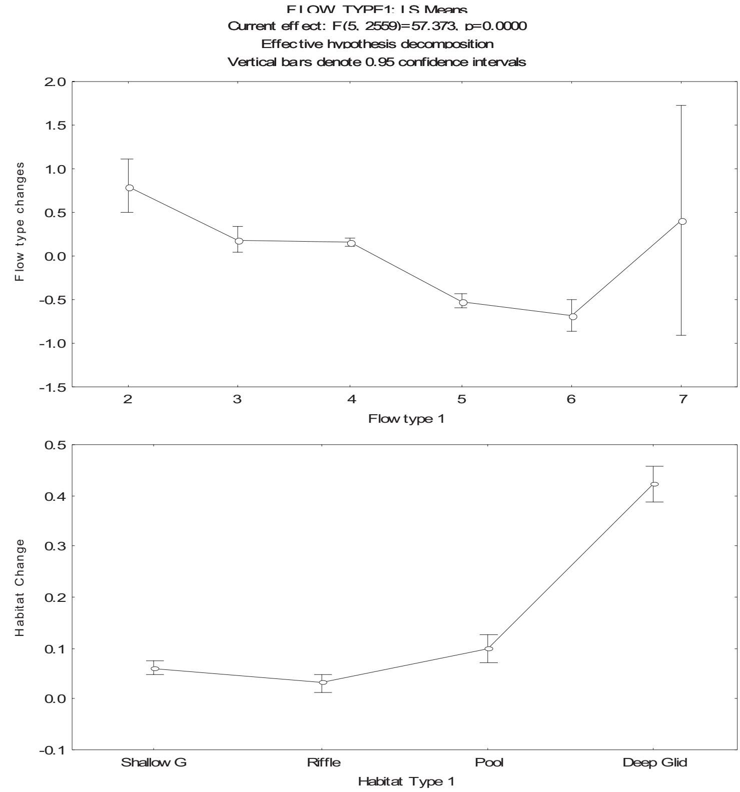

Figure 3. Mesohabitat types and flow types that were more affected by change in discharge. Mesohabitat change is between 0 (no change) and 1 (change), discharge change was calculated as the difference of flow type codes (1 = No flow i.e. dry, 2 = No perceptible flow, 3= Smooth, 4 = Rippled, 5 = Unbroken Standing waves, 6 = Broken standing waves and 7 = Boil)

Table 4. P-values for analysis of variance between categorical and continuous parameters after standardisation.

	Velocity change	Depth change	Froude change
Habitat Type	<0.001	<0.001	0.0029
Flow Type	<0.001	0.1456	0.571

Depth and Froude number changes were not significantly different between flow types. However, there were significant differences between flow types for the velocity change; unbroken standing waves were significantly different to the rest of the flow types.

5 CONCLUSIONS & DISCUSSION

5.1 Spatial pattern at the Leigh Brook river site

Three sampling strategies were tested at the Leigh Brook river site in order to study the spatial pattern of three hydromorphological parameters: depth, velocity and Froude number. Results indicate that grid sampling data are best for prediction of velocity and depth using geostatistical techniques. Spatial pattern for Froude number was difficult to identify and no preference between sampling strategies could be stated.

Results obtained for the variogram assessment indicate that the correlation between points is lost at distances equal to 3m, 5m and 9m for depth, velocity and Froude number. This provides an indication for the sampling distance that should be applied at the Leigh Brook river site.

Spatial pattern of prediction errors indicate that higher sampling densities should be applied in deep areas when measuring depth and broken – unbroken standing waves when measuring velocity or Froude number, as these are the locations that present higher errors in the predicted values.

Future work will focus on the analysis of 16 more rivers, the majority of which are lowland sites, with the methodology described in order to identify if a specific spatial pattern can be identified according to the physical characteristics of the river site.

5.2 Temporal pattern at the Leigh Brook river site

Shallow and deep glides present higher changes in depth and velocity, respectively as discharge changes. Sampling for the monitoring of hydromorphological change with discharge should therefore be located in these areas.

Flow type data provided a good indication of where to monitor for hydromorphological change.

Mesohabitat data were only collected at cross-section level for this study and thus proved less useful. Future work will look at mesohabitat data at individual points.

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Flow variability to preserve fish species critically at risk: a Spanish case study.

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ABSTRACT: The Jucar River Basin (Spain) harbors a high number of endemic mussel and fish species that are declining or going extinct at an alarming rate. Factors affecting *Chondrostoma arrigonis* (endemic fish specie critically at risk) were analyzed to explain the population decline. After develop and evaluate a list of candidate causes, flow modification was selected as the major factor governing the threat processes. To understand local extinctions across the historical range of the cyprinid, physical habitat degradation was assessed by comparing river reaches with different flow modification histories. A geographical information system was developed and time series of habitat were generated by the PHABSIM simulator. Our findings indicate that presently, heavily flow modification activities may cause the rapid extinction of the remaining populations, regardless of setting minimum flows for the rivers.

1 INTRODUCTION

The degree of endemicism of primary and secondary freshwater fishes in Spain is remarkable (Elvira, 1995). Twenty-five of the 29 (86%) Spanish native species and subspecies belonging to Cyprinidae, Cobitidae and Cyprinodontidae (Blanco & González, 1992) are endemic to the area.

Chondrostoma arrigonis (Steindachner, 1866), was a endemic fish specie locally common in the Júcar River Basin (JRB) at the beginning of this century, mainly in middle reaches (Doadrio et al. 2004). Historically, there were approximately 10 streams that supported *Chondrostoma arrigonis* (CA) populations. By the late 1980s, this figure had dropped to approximately 5 streams. Individual populations recently have gone extinct (or are about to go extinct) in 3 streams. The recent Technical Summary for *Chondrostoma arrigonis* in the Júcar River Basin, done by the National Museum of Natural Sciences (CSIC) (Doadrio et al. 2004), shows that populations of CA presently inhabit 5% or less of its historic range.

CA populations continue to decline at an alarming rate with most remaining populations small and fragmented (Blanco & González, 1992; Elvira, 1995; Doadrio et al, 2004). Demographic stochasticity presently is important because total population size has become very small, and the loss of genetic diversity decreases the specie ability to adapt to changed environmental conditions and increases the risk of genetic extinction. A variety of proposed activities and ongoing projects and the absence of a comprehensive conservation strategy to protect and restore aquatic ecosystems probably will cause the irrevocable extinction of this specie in a short term (Elvira, 1995; Doadrio et al, 2004).

Within the Júcar River Basin, dams have significantly affected a substantial amount of stream habitat altering flow conditions since 1912 (CHJ, 1997). Currently there are at least 19 dams with storage capacity in excess of 1 hm^3. The extent to which these dams affect the deterioration of aquatic ecosystems and the imperilment of the related biota has not been documented (Elvira, 1995; Doadrio et al. 2004).

Preliminary exploratory analyses were conducted by the Stressor Identification process (Cormier et al. 2000) to identify the causal evidence of the biological impairment. After develop a list of candidate causes and analyze the information related to chemical data, biotic surveys, habitat analyses and hydrologic records, physical habitat condition was selected as the major factor limiting the target population.

The assessment of the risks of adverse effects on *Chondrostoma arrigonis* populations needs to evaluate both the causes of the habitat degradation (flow modification) and the response of aquatic ecosystems. According to stakeholder expectations and interests, the purpose of this investigation is to evaluate whether flow modification is responsible for the severely restricted current distribution and abundance of the specie in concern.

2 METHODS

2.1 Study area and site selection

The study area, a subset of the larger area including in the historical range of CA, was comprised of the Júcar and Cabriel rivers, located in the eastern Spain (Figure 1). The drainage basin for the watershed of the former is 21,578 km^2, being approximately 500 km in length and 43 m^3/s of median average flow. The Cabriel River (the main tributary of the Júcar River) has an average historical flow of about 12.2 m^3/s, 262 km in length and 4754 km^2 of watershed.

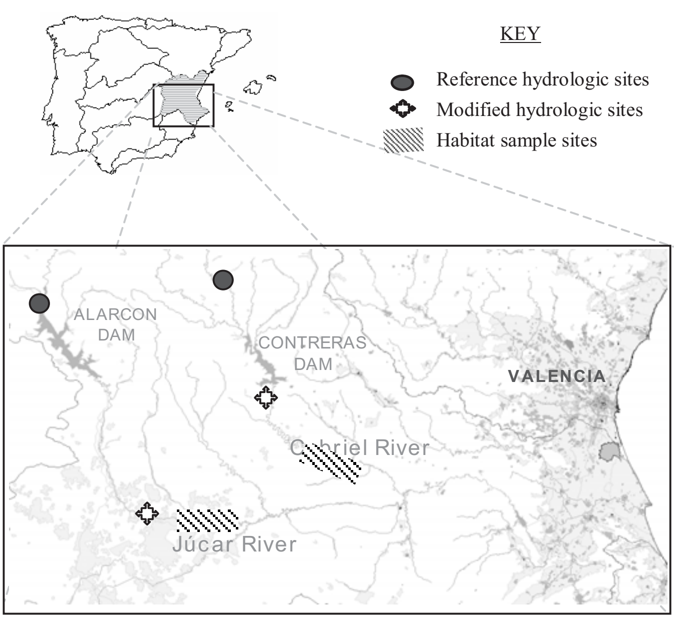

Figure 1. Study area location.

Annual discharge pattern shows a short and low duration flood in spring and a more important flood in fall, usually between October and November. During summer, the river is normally at the low-water period.

Efforts to categorize dam operations have focused in the bigger structures Alarcón Dam (build in 1954 and 1114 hm^3 of storage capacity) and Contreras Dam (build in 1970 with 800 hm^3 of storage capacity). The storage ability of them, allows runoff to be retained for subsequent controlled release, with an unlimited range of management options.

Preliminary study sites for the IFIM study were selected according to a probability design to ensure that analytical results have clear inference for the larger population. Physical habitat data from sampled sites were condensed to stream reach summaries that are spatially representative of the habitat characteristics within the reach.

Measurement of water depth, water velocity, substrate composition and cover were made at various intervals along each transect. Substrate composition and cover were assessed by visually estimating the percent of the two main particle size classes and type of cover. Site specific data on fish preferences for depth, velocity, substrate, and cover was gathered by National Museum of Natural Sciences (CSIC) biologists.

Streamflow metrics were determined for gaged sites within the study area from daily discharges provided by the Water Agency (Confederación Hidrográfica del Júcar). Different periods were analyzed depending of the availability and quality of the streamflow records.

2.2 Data analysis

One of the work hypothesis followed in the present study is based in the supposition that those managed rivers where the value of daily flows falls outside the range of natural variation involve poor habitat conditions for *Chondrostoma arrigonis*. As a consequence, different factors may act synergistically to exacerbate problems to the remaining populations, increasing the threats and being more prone to extinction.

Inspired in the Indicators of Hydrologic Alteration (Ritchter, 1996), the state of the perturbed system was assessed to identify differences from what it would have been in the absence of the perturbation. The natural flow regime was statistically characterized and compared with the human-made flow regime. The temporal variability in the hydrological regime was characterized using for every day the probability distribution of N-year daily discharge. Data were extracted at various percentages for which specified discharges are equalled or exceeded in a water year defining the range of natural variation.

In order to establish an adequate understanding of these habitat limiting factors, basic information on habitat requirements and life history of CA was provided (Doadrio et al. 2004). Based on the concept of a habitat "bottleneck" (Hall and Field-Dodgson 1981), spawning and early stages have been defined as the freshwater life stages most restricted by the limiting habitat.

A Threshold Analysis (TA) was developed to estimate the critical level. The critical level (derived from expert opinion) was defined as the minimum viable amount of habitat conditions which can support the fish life-stage without permanent damage. Based in the habitat reduction of natural conditions versus the artificial conditions, two threshold categories were established. Reduction in a 75% or more was considered a “critical impact”, while a reduction from 50 to 75% was considered a “great impact”. The IFIM methodology (Bovee, 1982) was selected as the best method for predicting available fish habitat in response to incremental changes in streamflow.

3 RESULTS

Hydrological analysis

Many reservoirs and diversions have been constructed in the century within the endangered-fish habitat, with the largest impact coming from Alarcón Reservoir (Júcar river since 1954) and Contreras Reservoir (Cabriel river since 1970).

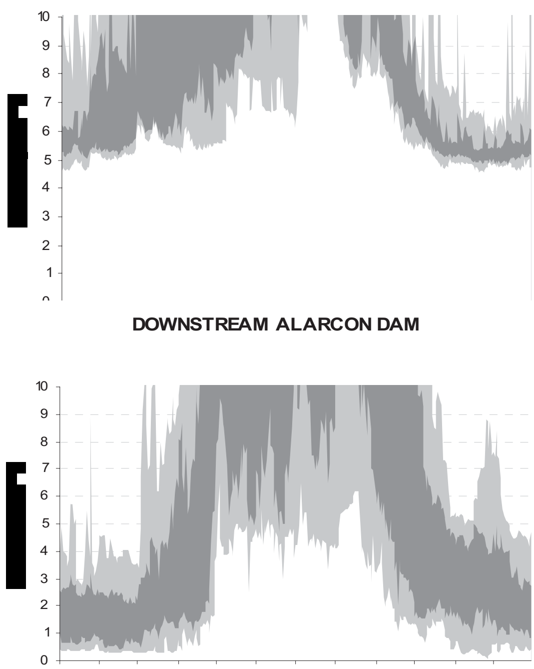

Figure 2. Júcar River hydrographs upstream Alarcón Dam (above) and downstream Alarcón Dam (below). Dark grey indicates values between 25 and 75 % of probabilities of flow excedence. Clear grey indicates values between 90-75% and 25-10% of probabilities of flow excedence.

In order to maximize the potential irrigation diversion downstream, the dams have been operating to store high flows in the fall, winter and spring, and then make constant releases in the summer growing season. As a result, the operation of these reservoirs substantially alters the flow regime of the Júcar and Cabriel rivers downstream. The net effect on flows is to increase flows in the river in the summer, and reduce them in the spring, fall and winter.

For Alarcón Reservoir (Figure 2), impoundment has reduced average flows in the months of October to May from about 7 to 2,5m^3/s, and has increased average flows in the month of August from about 12 to 20m^3/s .

The annual minimum flows have been drastically reduced; in downstream, flows below 2 m^3/s occurred 30% of the time, with a minimum of 0,15 m^3/s, while in natural conditions, the minimum flow registered was 4,5 m^3/s and flows in excess of 5 m^3/s occurred 92% of the time.

The alteration of the hydrograph caused by Contreras Dam impoundment (Figure 3) resulted in considerable change in the seasonal pattern in discharge also, resulting in a large increase in summer flows (from 8 to 20 m^3/s), and reducing dramatically average flows in the months of October to May; in upstream, flows in excess of 4 m^3/s occurred 95% of the time (minimum flow registered of 2,8m^3/s), whereas downstream, the river can be dewatered immediately below this dam; flows below 1 m^3/s occur at a frequency of greater than 65%, with 5% of zero flows.

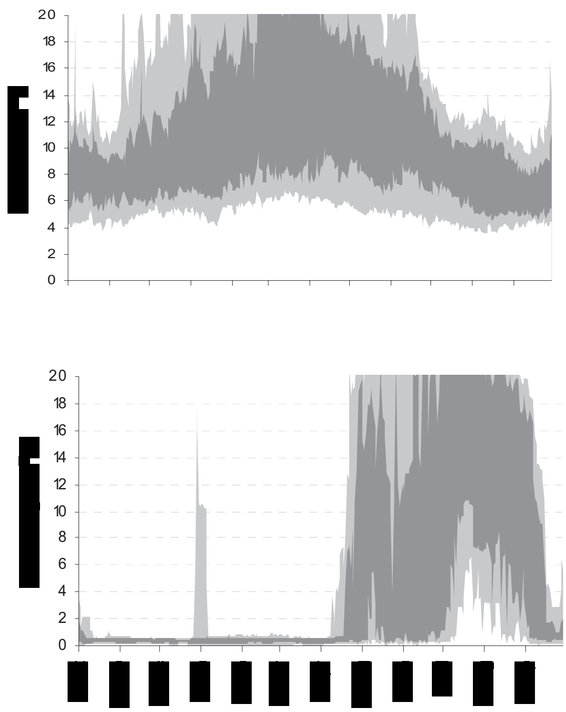

Figure 3. Cabriel River hydrographs upstream Contreras Dam (above) and downstream Contreras Dam (below). Dark grey indicates values between 25 and 75 % of probabilities of flow excedence. Clear grey indicates values between 90-75% and 25-10% of probabilities of flow excedence

Habitat analysis

The habitat impact was computed by comparing the total weighted usable area under natural conditions versus the artificial conditions, and extended for the whole period of records on a daily basis scale.

From the results of the hydrological analysis presented above and according to the physical habitat simulation model, historical changes in dam operations have resulted in dramatic differences in the extent and quality of endangered-fish habitat.

For the Júcar River (Figure 4), dam operations have decreased largely the availability of suitable habitat for about ten months of the year. Critical impacts have occurred in 20% of the time while only the 50% of the time can be considered "habitat not impacted" downstream Alarcón Dam.

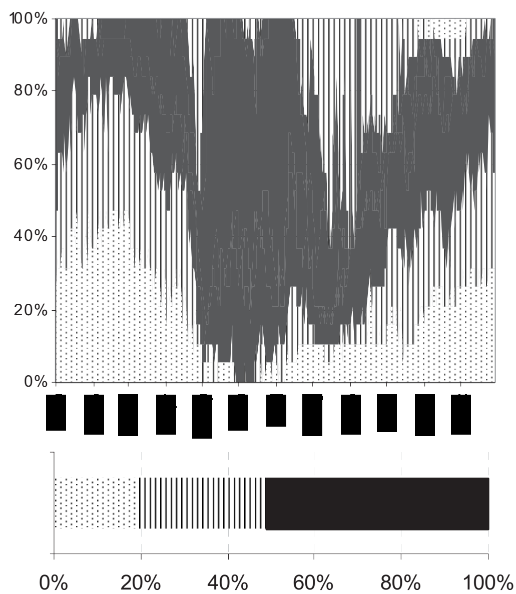

Figure 4. Habitat impact for the Júcar River applying the Threshold Methodology. Cumulative daily habitat impacts (above) and total impacts (below) for the period of record. Pointed area indicates critical impacts, area with bars indicates great impacts and the black area indicates no impact.

In the case of the Cabriel River (Figure 5), suitability habitats for *Chondrostoma arrigonis* were drastically reduced for every months of the year. Critical impacts occurred 64% of the time of the period of record. Only 17% is habitat not impacted.

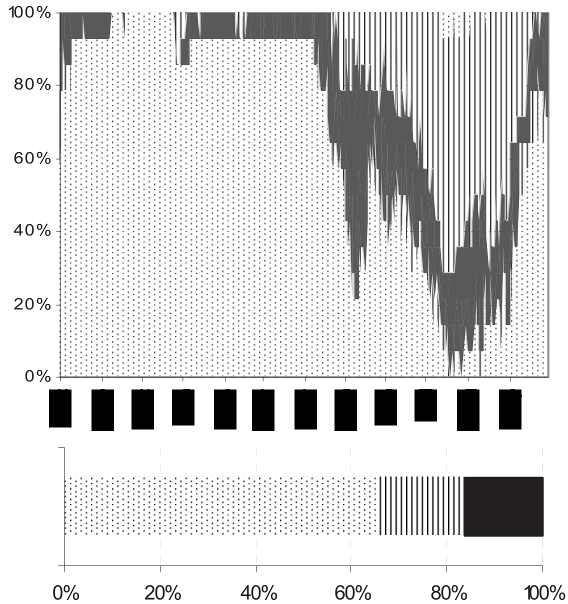

Figure 5. Habitat impact for the Júcar River applying the Threshold Methodology. Cumulative daily habitat impacts (above) and total impacts (below) for the period of record. Pointed area indicates critical impacts, area with bars indicates great impacts and the black area indicates no impact.

4 DISCUSSION

The analysis provided in this paper documents the initial investigation into how biological assessments may be influenced by hydrological conditions. The primary finding of this case study is that natural flow analysis appears to have merit for evaluating hydrologic changes in the Júcar River Basin and for identifying possible impacts on *Chondrostoma arrigonis* stocks.

In our review we have found a number of important impacts that have been better determined when contemporary hydrologic and habitat analytic methods (PHABSIM, TA and another one similar to IHA) have been used. According to the repeated observation in different places and times, a consistent association of flow modification, physical habitat and CA stocks decline is likely to indicate true causation.

In the JRB, the relationship between the degree of flow modification and biological response of *Chondrostoma arrigonis* is not simple. While resilience is a very useful theoretical concept, there is still insufficient knowledge to be able measure a system's resilience and to use this to set thresholds for particular disturbances. In spite of this, the Threshold Analysis (theoretical critical level in habitat reductions respect natural conditions) based on expert opinion could be a suitable tool to explain the decline of the specie in concern.

Finally, biological communities integrate the effects of different stressors reflecting overall ecological integrity. The extinction of individual species usually does not take place in isolation. Therefore, local extinctions of the Iberian cyprinid directly assess the status of some waterbodies of the Júcar River Basin, key aspect related with the primary goal of the Water Framework Directive (WFD).

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Morphohydraulic quantification of non spatialized datasets with the "Hydrosignature" software

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ABSTRACT: A general problem with habitat modelling is finding a relationship between the physical habitat and the aquatic fauna. One way to describe the habitat is to quantify the hydraulic diversity by calculating the hydrosignature. The hydrosignature of a river unit at one discharge quantifies the hydraulic variety by calculating the surface (or volume) percentages in a depth and current velocity cross classification.

The data sets used for quantification are the hydraulic measurements of verticals described by their depth and average velocity, which can be spatialized or not.

The hydrosignature of a river unit that has been spatialized is more significant than a hydrosignature for which the measurement points are unknown. The results discussed in this article aim to improve hydrosignature calculations in terms of non-spatialized data sets.

Three different methods designed to deal with such data can be distinguished:

NOXY: The field protocol consists in vertically measuring depths and mean velocities without recording location (no X and Y co-ordinates). NOXY directly takes into account the discrete information of verticals that have no spatial relationship between themselves.

NOXY2: The field protocol in this case is the same as NOXY. The difference between these two methods lies in the mode used to calculate the hydrosignatures assimilating the verticals to continuous information.

NOXY3: Concerning this protocol, the relative position of the verticals is recorded on-field, which then makes it possible to break down the described unit into cross sections that are perpendicular to the flow direction. This type of semi-spatialization also enables the assimilation to continuous hydraulic information.

A total of 250 river units spatialized in a triangular irregular network (TIN) composed the test data set.

This data set was degraded into a non-spatialized description to which NOXY, NOXY2 and NOXY3 were applied.

Hydrosignatures calculated by the three different methods were compared with hydrosignatures resulting from the original TIN data set that was used as a reference.

An index based on the same principle as that used for filters in spatial analysis applications was created for the purpose of comparing the two hydrosignatures.

When it is not possible to apply geo-referencing methods, the NOXY3 semi-spatialized approach is recommended.

1 INTRODUCTION

Both the scientific relevance and basic principle of a hydrosignature used to quantify the hydraulic diversity of a section of the aquatic environment at a given flow rate – referred to as a unit – was in the article "HydroSignature software for hydraulic quantification" by Le Coarer. It is therefore preferable the above-mentioned document be read prior to reading this article.

A unit can be hydraulically described by measuring verticals defined by their depth and mean velocity. During field measurements, these verticals are either geographically referenced or not. For a given vertical density, spatialization makes it possible to improve the descriptive accuracy as the hydraulic variables between the verticals can be interpolated using linear approximations. Even though spatialization makes it possible to obtain precise volume and surface measurements, this technique nevertheless takes longer to complete and tends to be more technical. This applies both on field and during post-processing phases: topographical calculations, meshes, etc.

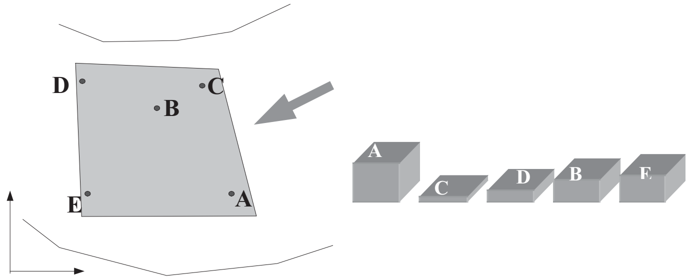
Non-spatialized data represent discrete information and the relative distances between the verticals in the hydraulic space are lost. The continuous hydraulic reality is degraded when using this type of representation. Methods that assimilate non-spatialized verticals to a continuous universe by "pseudo" or semi-spatialization are therefore applied to minimise the effect of such a problem. The objective was therefore to define an effective methodology designed to hydraulically quantify such data based on the hydrosignature of non-spatialized depth and velocity measurements.

2 METHODS

2.1 NOXY

With regard to this field protocol, the depth (hw) and average velocity (v) of the verticals are recorded without reference to their position, neither are they expressed in terms of the co-ordinates x and y (no x no y = NOXY). The mean length and width describe the unit dimensions.

As the relative positions of the verticals are unknown, the precise representativeness of these verticals in terms of horizontal surface is also unknown. Each vertical is therefore considered to represent an equal proportion of surface. Each vertical produces a right prism with constant depth and velocity values. The unit is broken down into right prisms with discrete physical values and constant horizontal surfaces devoid of hydraulic continuity (Fig. 1).



Unit expressed in NOXY
view in the horizontal plane

The verticals in NOXY,
represent discrete information.

```
[ NOXY]

[HEAD]
unit number 77
[/HEAD]
[hw v]

[UNIT 77]
[LENGTH(m) 20.19]
[WIDTHS(m) 1]
1.3      1.25
0.2      0.2
0.5      0.25
0.8      0.7
1.2      0.8
 [/UNIT]

[/NOXY]
```

Input file for Hydrosignature software

Figure 1. NOXY method.

The greater the density of the verticals, the more accurate the hydrosignature is in terms of description and quantification.

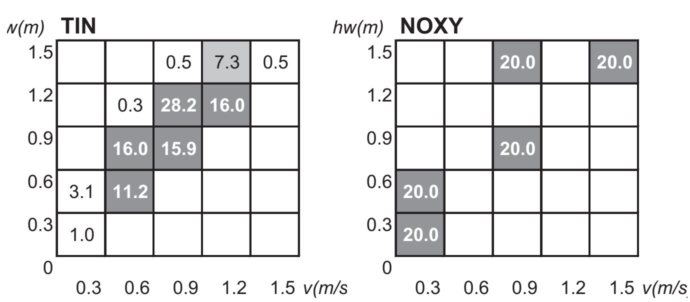


Figure 2. Schematic comparison of Hydrosignatures based on a TIN mesh and NOXY method.

Figure 2 here-above illustrates that for a given number of verticals, a hydrosignature produced by a triangular irregular network (TIN) mesh provides a better representation of the hydraulic continuity than that generated by the NOXY method.

This observation led to the development of other methods designed to assimilate non-spatialized verticals to a continuous data set.

2.2 NOXY2

The field protocol used to describe units in the NOXY2 method is identical to that of NOXY. The difference between the two methods lies in the hydrosignature calculation mode. A pseudo spatialization was applied to assimilate verticals to the cross section of an arbitrary representative width of 1 metre and a length such that the horizontal surface of the unit is retained.
This virtual cross section aims at interconnecting all the verticals in order to more closely resemble the hydraulic reality in a way that random spatialization could not achieve.

The construction of the NOXY2 cross section is carried out in three stages (Fig. 3):

– The first stage involves sorting the NOXY verticals according to an ascending variable. The user is free to choose this variable:
  Depth,
  Velocity,
  Velocity x depth,
  Froude number.
– The second stage consists in constructing a cross section based on a set of sorted verticals.

The first part of the cross section is composed of every second vertical whereas the second part is composed of the remaining verticals that have been sorted in a descending manner (according to the variable selected in stage 1).

- With regard to the third stage, the first vertical of the cross section is once again annexed at the end of the cross section. This last stage makes it possible to produce the same representativeness for all verticals in the horizontal plane.

At this stage, the space between two verticals is considered constant and the NOXY2 cross section is established.

The NOXY2 hydrosignature is therefore calculated based on this virtual cross section using formulas applied to real cross sections.

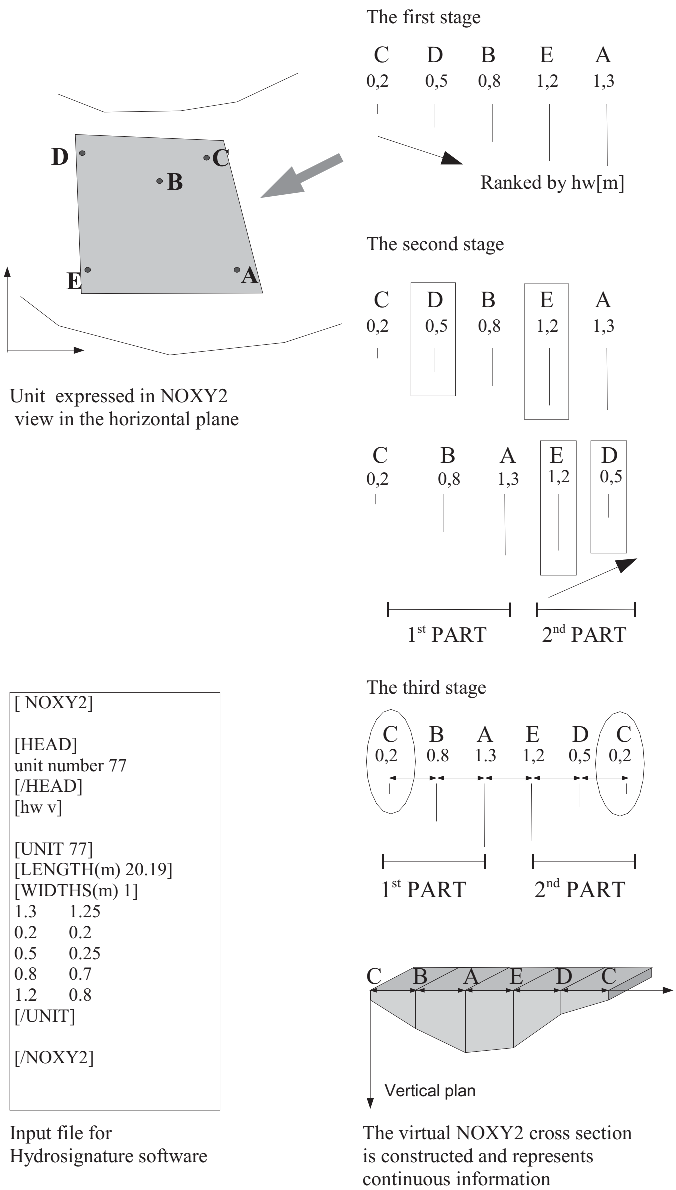
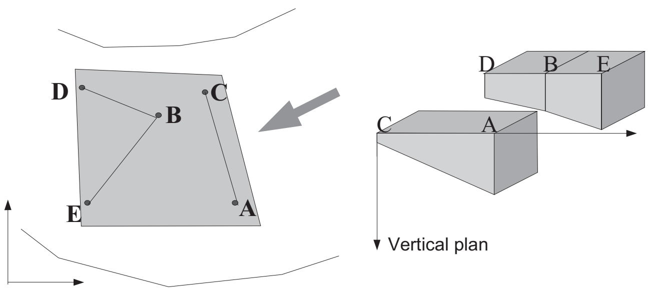


Figure 3. NOXY2 method.

2.3 NOXY3

Regarding the NOXY3 method, the unit description is based on semi-spatialized verticals. This semi-spatialization can be obtained by applying a field protocol identical to that of NOXY. The only difference between these two methods lies in the fact that the description units are divided/ broken down into identified sequences of verticals positioned perpendicularly to the flow direction. This is referred to as intra-cross section sequences.

The depth and velocity are recorded sequentially for each intra-cross section when covering the cross section from one edge to the other. The relative position of the verticals is therefore known according to this semi-spatialization. The end of an intra-cross section is indicated by "/", which is also recorded in the input file required by the HydroSignature software.



Unit expressed in NOXY3 view in the horizontal plane

The NOXY3 intra-cross sections represent continuous information.

```
[[NOXY3]

[HEAD]
unit number 77
[/HEAD]
[hw v]

[UNIT 77]
[LENGTH(m) 20.19]
[WIDTHS(m) 1]
1.3     1.25
0.2     0.2  /
0.5     0.25
0.8     0.7
1.2     0.8
[/UNIT]

[/NOXY3]
```

Input file for Hydrosignature software

Figure 4. NOXY3 method.

2.4 Comparison of two hydrosignatures

An index of comparison between two hydrosignatures was developed in order to compare the relative precision of the different methods.

In terms of the velocity/ depth plane, the proximity of the crossed classes directly expresses their hydraulic proximity and a hydrosignature represents spatialized data within this plane.

A filter technique used in spatial analysis was therefore employed to construct a hydrosignature comparison index.

The following values were expressed as such:

j: number of a crossed class of mean water depth and velocity.

ccj: percentage of the crossed class j

For a hydrosignature $\sum cc_j = 100$ (1)

Filter

A filter as defined below was applied to each signature (Fig. 5).

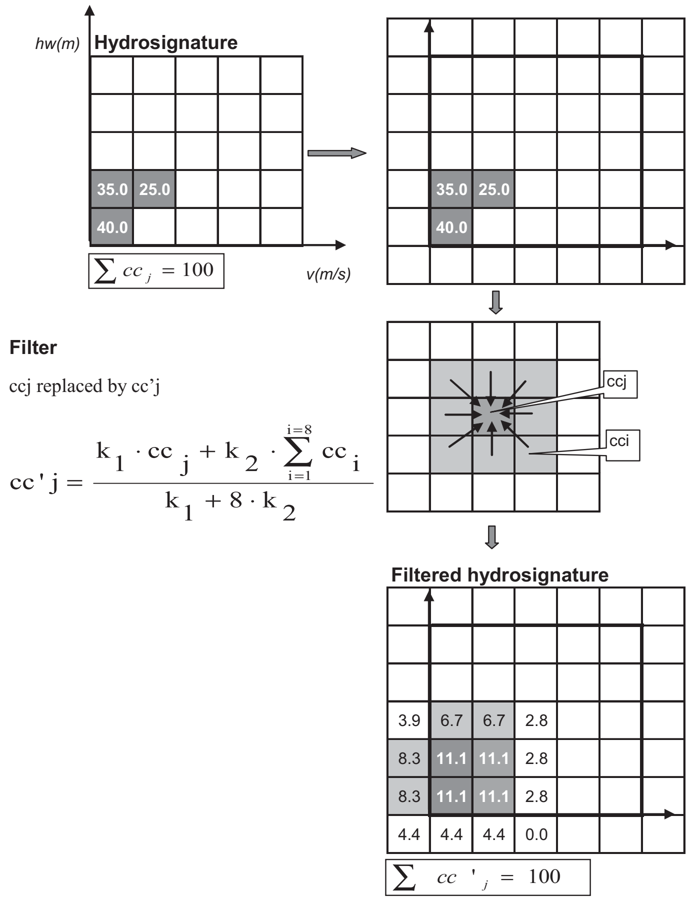


Figure 5. Filtering of a hydrosignature.

The hydrosignature was first placed in an infinite grid of crossed classes and the ccj retained their values whereas all the other cells were equal to zero.

It was then question of constructing a filtered hydrosignature whose crossed classes of the index j were recorded as cc'j and calculated using equation 2 below.

$$cc'j = \frac{k_1 \cdot cc_j + k_2 \cdot \sum_{i=1}^{i=8} cc_i}{k_1 + 8 \cdot k_2} \tag{2}$$

with
ccj: initial percentage of the crossed class j
cci: percentage of the 8 cells surrounding j
k1 > 0; weighting factor for the cell j
k2 ≥; weighting factor for the cells i

By construction, the sum of the crossed classes belonging to the filtered hydrosignature equalled 100.

$$\sum cc_j' = 100 \qquad (3)$$

HSC Index
Provided that definitions are based on the same depth and velocity classes, it is possible to compare two filtered hydrosignatures a and b by calculating the hydrosignature comparison index (HSC) using the following formula (Fig. 6):

$$\mathrm{HSC}(a,b) = \frac{\sum \left| cc'_j(a) - cc'_j(b) \right|}{2} \qquad (4)$$

By construction, the HSC index can reach values between 0 and 100.
HSC[k1,k2](a,b)=0 if two signatures are identical.
HSC[k1,k2](a,b)=100 if two signatures are not identical, that is, if:

$$\forall\ j\ \ si\ cc_j(a) \neq 0 = \ c_j(b) = 0 \qquad (5)$$

Comparisons using the Friedman Test
The Friedman test makes it possible to compare series containing more than 3 hydrosignatures (Sprent 1992) . This non-parametric statistical test compares the ranks of crossed class without comparing their values. Furthermore, this test neither represents all spatial data nor the distribution of values within the velocity/ depth plane. Subsequently, with reference to the theoretical case illustrated in Figure 6, this test does not find any differences between the hydrosi-gnatures a, b and c.

In this case, the HSC index obtains a greater hydraulic proximity between a and b than between a and c.

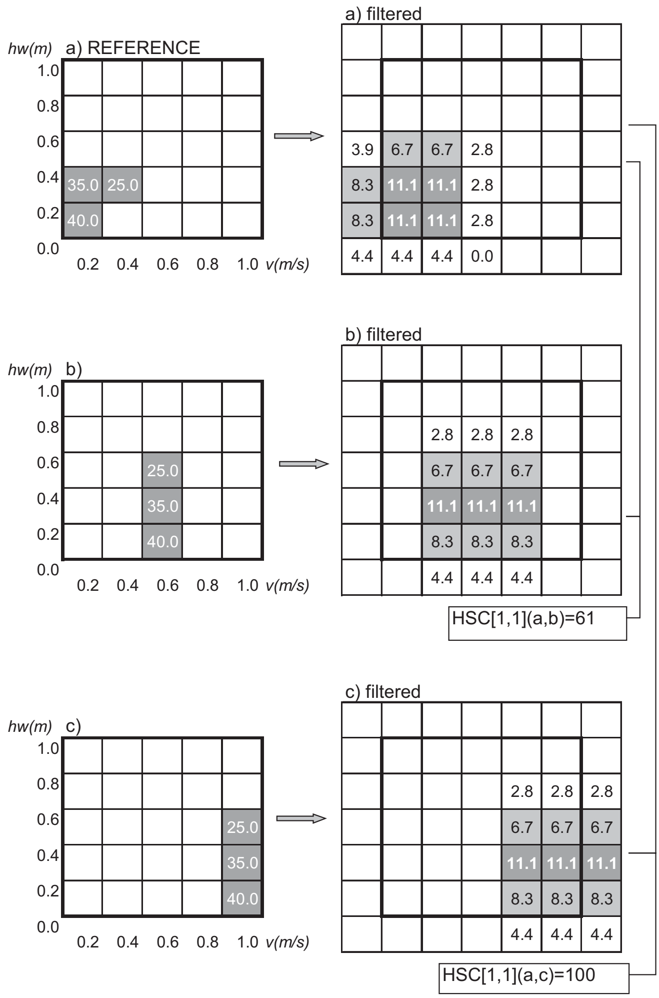


Figure 6. Comparison of theoretic hydrosignatures using a filter k1=1, k2=1.

2.5 Comparison of the three different methods

The data set

In order to test the performances of the NOXY, NOXY2 and NOXY3 methods, it was decided to apply a database containing 248 spatialized river units that were meshed using a triangular irregular network (TIN). This data set was originally compiled for research in fish / habitat modelling. These units were defined by selecting polygons within the horizontal plane representing homogeneous descriptive parameters in terms of hydraulics, shelter and morphology (border areas, etc.).

Table 1 below illustrates the general characteristics of the units. The hydrosignatures were defined using classes of velocity and depth with respective values of 0.2 m/s and a depth of 0.2 m.

Table 1. Mean morphohydraulic characteristics of 248 units.

	surface [m²]	volume [m³]	water depth [m]	velocity [m/s]
Min	1,83	0,25	0,05	0,00
Max	214,43	128,00	1,09	1,61
Mean	42,91	16,56	0,38	0,49
SD*	36,03	18,60	0,21	0,35

* Standard Deviation.

For three test units, the spatialized hydraulic descriptions were repeated five times using the same number of verticals and under the same flow conditions. The maximum HSC [1,1] value – obtained for surface and volume hydrosignatures calculated using meshes (TIN) and describing the same unit – was equivalent to 14.9, with an average of 9.2 and a standard deviation of 3.8.

Table 2. Mean morphohydraulic characteristics of 3 repeated units.

	surface [m²]	volume [m³]	water depth [m]	velocity [m/s]
Min	13,08	3,78	0,19	0,47
Max	30,85	10,74	0,36	1,33
Mean	22,06	6,27	0,28	0,74
SD*	6,75	2,94	0,07	0,37

* Standard Deviation.

The TIN meshed units degraded in non-spatialized descriptions corresponding to the NOXY, NOXY2 and NOXY3 methods (Fig. 7).

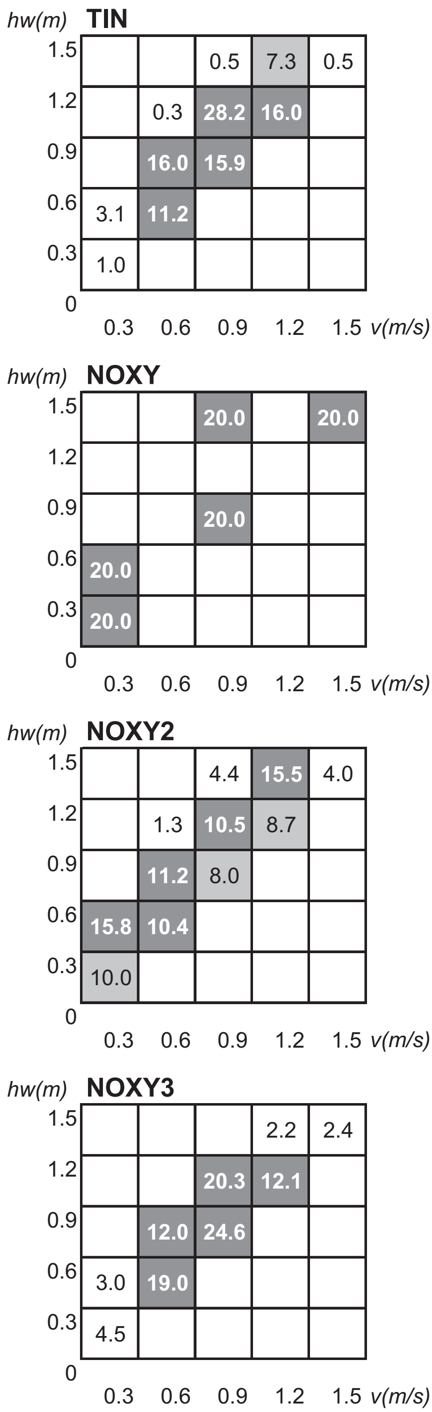


Figure 7. Example of hydrosignatures describing the same unit based on the TIN, NOXY, NOXY2 & NOXY3 methods.

The hydrosignatures calculated using these three methods were compared to those calculated using a TIN approach which was used as a reference.

Three types of comparisons using the indexes HSC (TIN, NOXY), HSC (TIN, NOXY2) and HSC (TIN, NOXY3) were carried out for each unit. The index values were sorted by comparison type and specified in Figure 8.

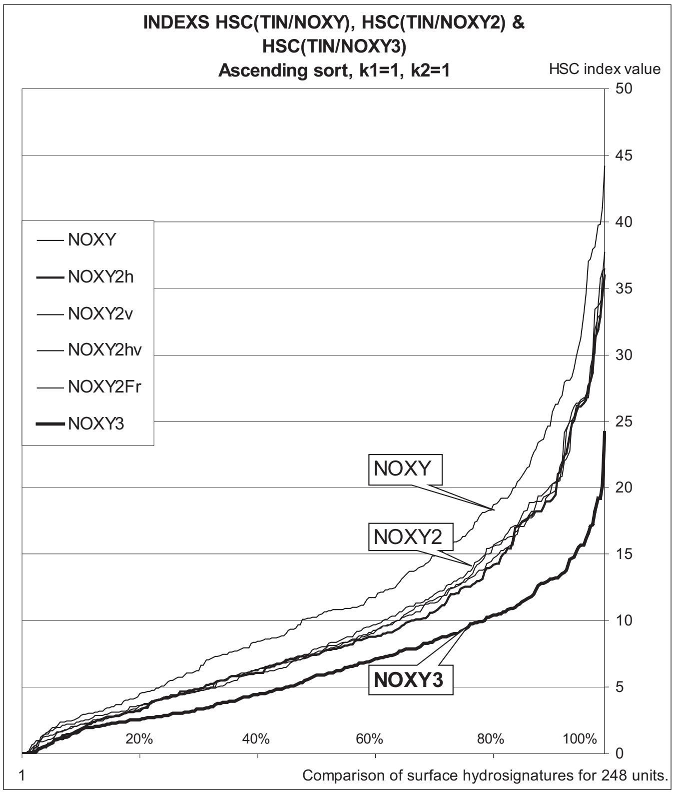


Figure 8. Representation in ascending sort order of the indexes HSC (TIN, NOXY), HSC (TIN, NOXY2) and HSC (TIN, NOXY3) for 248 units.

Table 3. Comparison of surface hydrosignature indexes HSC (TIN, NOXY), HSC (TIN, NOXY2) and HSC (TIN, NOXY3) for 248 units. [*u*] : HSC unit

HSC(TIN/	<10 [u]	<15 [u]	<20 [u]	<25[u]	Mean
	%	%	%	%	[HSC]
NOXY)	*48,0	71,0	84,3	90,7	** 12,0
NOXY2h)	**66,5**	**82,3**	**91,5**	**94,4**	**9,2**
NOXY2v)	62,1	79,4	90,7	93,5	9,7
NOXY2hv)	64,1	81,5	90,3	94,0	9,5
NOXY2Fr)	64,1	79,4	91,5	94,0	9,5
NOXY3)	**79,0**	**94,8**	**99,2**	**100,0**	**6,5**

* For example, 48% of compared surface hydrosignatures have an HSC index under 10 [HSC].

** HSC[TIN/NOXY] mean = 12 [HSC].

Table 4. Comparison of volume hydrosignature indexes HSC (TIN, NOXY), HSC (TIN, NOXY2) and HSC (TIN, NOXY3) for 248 units. [u] : HSC unit

HSC(TIN/	<10 [u]	<15 [u]	<20 [u]	<25[u]	Mean
	%	%	%	%	[HSC]
NOXY)	42,3	66,5	79,0	89,1	13,5
NOXY2h)	**61,7**	**78,6**	**87,5**	**94,4**	**10,3**
NOXY2v)	60,9	76,6	89,5	95,2	10,4
NOXY2hv)	60,5	81,0	88,3	94,0	10,3
NOXY2Fr)	61,7	80,2	90,3	95,6	9,8
NOXY3)	**79,4**	**95,6**	**99,2**	**99,6**	**6,5**

3 DISCUSSION

Hydrosignatures make it possible to quantify hydraulic diversity. To calculate such hydrosignatures, the most accurate mathematical solution consists in using a TIN mesh approach based on verticals of spatialized depth and velocity measurements. Three different methods were developed for the HydroSignature software and non-spatialized input data types: NOXY, NOXY2 and NOXY3.
In order to compare these methods with each other, the HSC index was developed, enabling the comparison of two hydrosignatures. The values of this index directly depend on the choice of depth and velocity classes defining the hydrosignature grids.

In the case of a data set comprising 248 hydraulic units with a mean of 43 m2 and a standard deviation of 36 m2, Table 3 and Figure 8 highlight the following observations:

– The NOXY method is not as efficient as the NOXY2 method when calculating hydrosignatures,
– The NOXY3 method – requiring a semi-spatialized but rather unrestricted protocol – makes it possible to obtain hydrosignatures that come the closest to mesh results (TIN).

The NOXY3 method is therefore recommended to users when collecting non-spatialized data. This task may even be carried out using former data for which a site plan of verticals is available.

The NOXY2 method is recommended when the relative position of the verticals cannot be reconstructed. The variants of the NOXY2 method offer similar numerical performance levels, with NOXY2h generally achieving the best surface results and NOXY2Fr obtaining the best volume results.

Nevertheless, comparison of the different methods must be performed on hydraulic units of greater size such as channel unit types (i.e. pools, riffles) in order to validate such conclusions on a larger scale.

Furthermore, there is talk of testing data sets obtained at different flow rates in order to check whether HSC index values increase with the flow rate.

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WEB SITE

http://hydrosignature.aix.cemagref.fr/

Predicting reference fish communities of European rivers

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ABSTRACT: Within the EU-project FAME (Development, Evaluation and Implementation of a Standardised Fish-based Assessment Method for the Ecological Status of European Rivers) predictive reference models were developed in order to fulfil fish monitoring requirements of the Water Framework Directive.

About 15 000 fish samples of 12 countries, 17 ecoregions and 2700 rivers have been collected and divided into unimpacted (no or only slight anthropogenic pressure) and impacted sites. Two different methods were used to predict fish community composition of unimpacted sites: (1) Fish samples (i.e. relative species composition) were clustered within ecoregions and resulting fish types (60 types) were clustered again at the European level revealing 15 European Fish Types (EFT). Discriminant analysis was used to select environmental characteristics in order to predict EFT. (2) Multi-logistic and multi-linear regression models were developed to predict functional components of fish communities (i.e. tolerance, trophic status, reproduction, habitat and migration requirements) of individual sites using environmental characteristics as independent variables.

The discriminant model explained 69 % of the variability of fish community composition at the type level. The regression models were able to explain about 31 % to 48 % of variability in functional fish community composition at the site level. Both methods revealed that the longitudinal position within the catchment, i.e. distance from source, altitude, channel width and slope is as important as the geographical position in Europe.

The results show (1) that by means of an adequate database it is possible to predict riverine fish communities across Europe and (2) that modelling techniques are useful tools for predicting reference conditions as required for the implementation of the Water Framework Directive

1 INTRODUCTION

Assessing the ecological status of running waters according to the EU Water Framework Directive (WFD) requires characterization of reference conditions. For developing target fish communities most authors have used either expert judgment or *best available* sites. Both are not in accordance with the WFD as reference conditions are strictly defined herein as *no or minor deviation* from undisturbed conditions. Independent pressure criteria are necessary to distinguish between unimpacted and impacted sites. Reference fish communities can be predicted based on models using environmental characteristics of undisturbed sites.

Within the EU-project FAME (Development, Evaluation and Implementation of a Standardised Fish-based Assessment Method for the Ecological Status of European Rivers, EVK1-CT-2001-00094) predictive reference models were developed coincidencing with the WFD requirements. We tested two different methods to predict fish community composition of unimpacted sites. For the *spatially-based* approach we clustered sites into homogeneous groups and described the reference species composition and environmental characteristics for each *fish type*. For the *site-specific* approach we used multi-logistic and multi-linear regression models to predict functional components of fish communities (i.e. tolerance, trophic status, reproduction, habitat and migration requirements) of individual sites using environmental characteristics as independent variables.

2 METHODS

2.1 Data

All data used for analyses were extracted from the FIDES database developed within the FAME project (see http://fame.boku.ac.at). Data were obtained from fisheries surveys conducted between 1978 and 2002. A total of 5252 sites of 11 European ecoregions, for which all the obligatory fish and environmental variables were fulfilled, were retained. All sites were sampled by electric fishing, only the first pass was considered.

2.2 Defining reference sites

The following 4 main variables to assess anthropogenic pressures, i.e. modification of morphology, hydrology, presence of toxic substances or acidification, and nutrient loading were used to separate unimpacted from impacted sites. Each of these variables is semi-quantitative, with five modalities ranging from 1 (almost no pressure) to 5. Only sites of class 1 and 2 were retained for reference sites. Reference sites (N=1608) were thus not pristine or totally undisturbed (i.e. only rated 1 for each of the considered impact variables), but rather sites where physical and chemical alterations were not sufficient to notably affect the fish fauna.

2.3 Environmental variables

Abiotic variables were measured in the field or from topographical maps, or estimated using GIS at each site: altitude, distance from source, catchment area reach slope, wetted width, mean annual air temperature, presence/absence of a natural lake upstream, geological type (calcareous, siliceous), flow regime (permanent or temporary), two map coordinates, latitude and longitude and river groups (group of drainage basins).

These variables were chosen because (1) they are not likely to be influenced by human activities and (2) they describe the spatial position in both dimensions the geographic position within Europe and the location along the longitudinal continuum of rivers.

In addition, 3 variables describing sampling methods and effort were considered. The description of the sampling effort was summarized by two variables: sampling technique (boat or wading) and sampling method (complete or partial width of the river).

2.4 Developing spatially-based model

In the first step fish samples (i.e. relative species composition) were clustered (hierarchical cluster analyses, Ward's method, SPSS 10.0®) within ecoregions and resulting fish types were clustered again at the European level resulting in *European Fish Types* (EFT). In the second step discriminant function analysis (DFA) was used to select environmental characteristics in order to predict EFT.

2.5 Developing site-specific models

Multi-logistic and multi-linear regression models were developed to predict functional components of fish communities (i.e. metrics for tolerance, trophic status, reproduction, habitat and migration requirements) of individual sites using environmental characteristics as independent variables. Metrics were selected due to their capacity of being predicted by environmental characteristics and to respond to human pressures (for further details see Pont et al. in press).

3 RESULTS

3.1 Type-specific models

Clustering relative species composition resulted in 60 fish types within the 11 ecoregions. Re-clustering the 60 fish types reduced the number of types to 15 (Table 1).

Table 1: Dominating fish species of the 15 European Fish Types (EFT)

Fish species	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15
Salmo trutta fario	x	x	x	x											
Phoxinus phoxinus					x	x									
Thymallus thymallus							x								
Salmo salar								x							
Cottus poecilopus									x						
Leuciscus carolitertii										x					
Salmo trutta lacustris											x				
Salmo trutta trutta												x			
Barbus meridionalis													x		
Gasterosteus aculeatus														x	
Rutilus rutilus															x

All 7 environmental parameters were significant ($p<0.001$) in predicting fish types based on DFA. In total 7 significant functions were defined by DFA. The first 3 functions explained 87 % of total variance covered by the DFA. Main environmental variables (standardized coefficients > |0.4|) are longitude, latitude, slope, mean air temperature and distance to source (Table 2).

The discriminant model explained 69 % of the variability of fish community composition.

Table 2: Standardised coefficients of environmental parameters for the first 3 discriminant functions (highest coefficient per function in bold)

	1	2	3
Longitude	-,145	**,695**	,329
Latitude	**,828**	-,176	-,218
Mean air temperature	,426	-,407	**,770**
Altitude	-,251	,029	,337
Slope	-,587	-,189	-,181
Distance to source	,079	,137	,475
Channel width	-,037	,250	,149

3.2 *Site-specific models*

For site specific models in total 13 environmental variables (including methods) were selected by multilinear regression analyses to predict 10 metrics characterizing reference conditions. Ten variables were describing the physiographical conditions, 3 were related to the sampling method. In most cases 7 (6-12) environmental variables were selected by the regression models. The number of insectivorous increases with altitude, slope, but also temperature and decreases with stream width. The number of omnivorous and phytophilous and the relative number of tolerant species is higher if a lake is situated upstream, while the relative number of lithophilic and intolerant species declines. The number of potamodromous species increases with size of catchment and decreases with elevation. Lithophilic occur mainly in catchments ranging from 100 to 10 000 km^2 but prefer comparable small rivers. The relative number of intolerant species deceases in rivers with temporary flow regimes and with distance from source. All community attributes but long distance migrants show a positive trend with increasing elevation. While the number of insectivorous and lithophilic increases with slope the number of omnivorous and the relative number of tolerant species declines. All but omnivorous individuals and tolerant species have preferences for particulate river groups (Table 3).

The regression models were able to explain about 31 % to 48 % of the variability in functional fish community composition at the site level.

Table 3: Site-specific models for the ten metrics. Lin=linear model, Log=logistic model. FISH=sampling area; TECH=sampling technique (B:boat, W:wading); METH=sampling method (P:partial sampling, W:whole sampling); CAT=catchment class area (1:< 10 km^2, 2:10-99 km^2, 3:100-999, 4:1000-9999 km^2, 5:>10000 km^2); LAK=presence of a natural lake upstream; GEO=geology (C:calcareous, S:siliceous); FLOW:flow regime (P:permanent, T:temporary); ELE=elevation; DIST=distance from source; WID=river width; TEMP=mean air temperature; SLOP=river slope; RIVG=river groups (D:Danube, E:Ebro River, MC:Mediterranean rivers from Catalunya, MF:Mediterranean rivers from France; Meuse-group rivers, NP:North Portugal rivers, NE:European Northern plain rivers, R:Rhône river, SE:West South Sweden rivers, UK:United Kingdom rivers, WF:West France rivers).

	Ni-INSE	Ni-OMNI	Ni-PHYT	Ns-BENT	Ns-RHEO	Ns-LONG	Ns-POTA	%NiLITH	%NsINTO	%NsTOLE
Model	Lin	Lin	Lin	Lin	Lin	Lin	Lin	Log	Log	Log
Intercept	4.242	1.399	1.950	-0.200	-0.491	0.841	-0.656	0.373	0.751	-1.875
FISH	-	-	-	0.061	0.080	-0.038	0.086	-	-0.122	-
TECH-B	0.000	-	-	-	-	-	-	0.000	0.000	0.000
TECH-W	1.038	-	-	-	-	-	-	0.816	0.250	-0.297
METH-P	0.000	0.000	0.000	0.000	0.000	-	-	0.000	-	-
METH-W	-0.550	-0.766	-0.533	-0.135	-0.064	-	-	0.433	-	-
CAT-1	-	-	-	-	-	-	0.000	0.000	-	-
CAT-2	-	-	-	-	-	-	0.035	-0.002	-	-
CAT-3	-	-	-	-	-	-	0.228	0.398	-	-
CAT-4	-	-	-	-	-	-	0.574	0.431	-	-
CAT-5	-	-	-	-	-	-	0.473	0.148	-	-
LAK-N	-	0.000	0.000	-	-	-	-	0.000	0.000	0.000
LAK-Y	-	0.500	0.450	-	-	-	-	-0.835	-0.322	0.316
GEO-C	-	-	-	-	-	0.000	-	0.000	-	0.000
GEO-S	-	-	-	-	-	0.083	-	0.025	-	0.181
FLOW-P	-	-	-	-	-	0.000	0.000	0.000	0.000	-
FLOW-T	-	-	-	-	-	-0.178	0.179	0.297	-0.710	-
ELE	0.674	1.356	0.702	0.427	0.472	-0.063	0.326	1.566	-	0.627
ELE2	-0.098	-0.137	-0.077	-0.055	-0.051	-0.012	-0.034	-0.161	-	-0.087
DIS	-	0.276	-	0.069	0.079	-	-	-0.399	-	-
DIS2	-	0.000	-	-	-	-	-	0.031	-	-
WID										

	Ni-INSE	Ni-OMNI	Ni-PHYT	Ns-BENT	Ns-RHEO	Ns-LONG	Ns-POTA	%NiLITH	%NsINTO	%NsTOLE
WID2	-2.046	0.898	0.313	-	-	0.079	-	-1.168	-	-0.147
TEMP	0.319	-0.264	-0.106	-	-	0.008	-	0.208	-	0.000
TEMP2	0.324	-0.017	-	0.036	0.038	-	0.010	-0.396	-0.205	0.115
SLOP	-0.029	0.016	-	0.000	0.000	-	0.002	0.005	-	0.000
SLOP2	2.263	-3.347	1.842	-0.248	0.102	0.036	-0.087	1.083	0.958	-1.047
RIVG-D	-0.354	0.468	0.274	0.020	0.000	-	-	-0.059	-0.092	0.148
RIVG-E	0.000	-	0.000	0.000	0.000	0.000	0.000	0.000	0.000	-
IVG-MC	1.601	-	-0.080	-0.196	-0.312	0.244	-0.465	12.13	6.824	-
IVG-MF	0.839	-	0.284	0.027	-0.469	0.037	-0.235	1.514	-1.488	-
IVG-MN	1.724	-	-0.144	-0.027	-0.215	0.254	-0.569	1.259	0.707	-
RIVG-NP	1.116	-	0.022	0.425	0.232	0.054	-0.080	0.057	-0.016	-
RIVG-NE	0.531	-	1.243	-0.029	-0.575	0.088	-0.444	0.742	-0.206	-
RIVG-R	-0.281	-	-0.015	0.106	-0.057	-0.114	-0.263	-0.155	-0.296	-
RIVG-SS	1.314	-	0.080	0.144	0.012	0.163	-0.402	1.955	0.706	-
	-0.022	-	0.175	-0.422	-0.325	-0.218	-0.442	1.528	-0.458	-

4. CONCLUSIONS

The results show that for both type- and site-specific methods a set of about 7 environmental variables is sufficient to explain a large proportion of the natural variability of European fish communities.

In principle two spatial dimensions are structuring fish assemblages:

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Concept for integrating morphohydraulics, habitat networking and water quality into MesoCASiMiR

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ABSTRACT: The transfer of habitat modeling approaches into larger scales is one of the main issues that has to be worked on within the next years. As the use of habitat models so far is mainly restricted to local investigations such as minimum flow problems and in the meantime also increasingly for river restoration measures this will be crucial for the wider application and propagation of habitat related models. Particularly for an effective implementation of the Water Framework Directive in Europe there is an urgent need for tools that enable the assessment of ecological river status.

The "RIVERTWIN" project, partly funded by the European Commission aims to support the goals of the Global Water Initiative by adjusting, testing and implementing an integrated regional model for the strategic planning of water resources management in twinned river basins. Part of this project is the intensification of the development of MesoCASiMiR, a mesoscale version of the model system CASiMiR. This system originally has been developed for the assessment of fish and benthic habitats in the microscale. The fish habitat module uses a fuzzy rule - based approach for the description of habitat requirements. While until now the approach was mainly focused on commonly used habitat parameters as water depth, flow velocity and river bottom structure it is also highly suitable for the integration of additional parameters such as the networking of habitats and water quality. Especially the networking is of major interest for the assessment of the ecological status of rivers in the mesoscale.

The fuzzy-rule based approach meets the demands of habitat description in the mesoscale in two ways: Leaving the microscale it becomes increasingly difficult to define physical conditions accurately. On the other hand the knowledge about linkages of habitats is not as far developed as it is for several physical parameters in the microscale.

1 INTRODUCTION

The upscaling of habitat models is one of the most important tasks in the near future in order to establish them as a standard tool for the assessment of river status. Currently the application of habitat models is mainly concentrated on the local scale and the investigation of spatially delimited problems like ecological flow regulations or morphological enhancements. However for many water management issues and strategic planning, modeling tools have to cover larger areas respectively river sections. E.g. in the Water Framework Directive of the

European Union (European Commission 2000) the management units of river basins are waterbodies with an extension of about 200 km².

Consequently the project RIVERTWIN, funded by the European Commission, was built up to support the aims of the Global Water Initiative, that was launched in 2003 to apply the principles of the WFD to other continents. Main goal of the project is the development of an integrated regional model for the strategic planning in water resources management (http://www.rivertwin.org, Gaiser & Dukhovny 2004).

This integrated model named MOSDEW comprises different submodels for e.g. landuse, agroeconomics, water demand, hydrology etc.. One important element of the whole system is the submodel MESOCASiMiR that together with the submodel for water quality (QUAL2K) gives information on changes in river ecology caused by different scenarios as object of investigation. MesoCASiMiR is further stage of the habitat model CASiMiR (Jorde 1996, Schneider 2001).

2 MAPPING METHOD

Providing suitable habitat mapping methods is one of the crucial aspects in the development mesohabitat models. Without an objectified method, fast enough to cover large systems but detailed enough to enable habitat assessment the broader practical application of habitat models in the mesoscale can be questioned.

Since the model was not supposed to be based on descriptive parameters but rather on really quantitative information it was decided to use a compromise method. Mesohabitats are described by representative values for hydraulic and morphological parameters. However the hydraulic parameters as flow velocity and water depth are classified and classes are partly overlapping in order to facilitate the assignment of a mesohabitat to one of the classes (Eisner et al. 2005).

But another crucial aspect is the support of the mapping by use of a PDA stored in a waterproof box (Fig.1). A commercial software was adapted to enable a fast registration of habitat parameters. Not only hydromorphological information is collected but also information on e.g. current flow rate, closest gaging station, migration barriers, water extraction etc. . Furthermore the shape and size of mesohabitats is drawn as a sketch directly on an interface of the mapping software. These sketches together with coordinates registered by a GPS are used later on to draw habitat polygons in a GIS shape file.

Since all habitat parameters stored in the PDA can easily be appointed to these polygons the generation of habitat maps as input for MesoCASiMiR is accelerated considerably.

Fig.1: PDA in waterproof case used while Mesohabitat mapping for MesoCASiMiR

3 MODELLING STEPS

Habitat modeling is planned to be performed in three steps within MesoCASiMiR.

3.1 Hydromorphological habitats

As in "conventional" physical habitat models in the first step morphological and hydraulic habitat parameters are considered. It is well known that e.g. flow velocity, water depth, substratum at the river bottom and available cover have strong influence on the habitat use of many fish species. Usually habitat modeling is performed for selected indicator species. Since the habitat requirements of different life stages are quite different the output of this modeling step is a "patchwork" of habitats. In most cases the availability of habitats for all life stages is regarded when interpreting model results and additionally seasonal aspects are considered. However the interaction between different habitat types is mostly neglected.

3.2 Networking -"Living space" and Migration

The expression "habitat" is often used by modelers when talking about physical conditions for a certain fish life stage as target of the modeling process. But the suitability of a river stretch depends on the availability of different habitat types and their spatial context.

To give two examples: A good spawning habitat will only be useful for a fish population if good habitats for juvenile fish are close and located downstream of the spawning ground. A feeding habitat by many fish species will only be frequented if a certain cover type is "not too far away".

This is why the expression "living space" is introduced. It is meant to indicate that even though in a river stretch there might be optimum habitats for different life stages of a species this doesn't necessarily mean that this species can survive in that stretch. This is because distances, orientation and connectivity between different habitat types are defining the suitability of a river stretch.

Another aspect of habitat networking is the passability of migration barriers. Usually fish tries to overcome migration barriers mainly during the spawning period to make its way to suitable spawning grounds. So even though a "living space" as defined above can cover a wide range of different habitat types and provide good conditions for any life stage throughout the year it can hardly contribute to the survival of a fish population if at a certain time of the year suitable spawning grounds are not reachable. In that sense the suitability of a whole living space is affected by the disconnection to reproduction areas and has to be considered when developing management concepts for rivers in the larger scale. An advantage of the mesoscale is that the cumulative effect of migration barriers can be considered. Furthermore it is not only the barrier and the migration facility itself but additional factors like the location of the facility in lateral direction and the extension of the backwater zone upstream of the barrier that are effecting fish passage. These effects can be integrated and visualized in a mesocale model (Fig. 2).

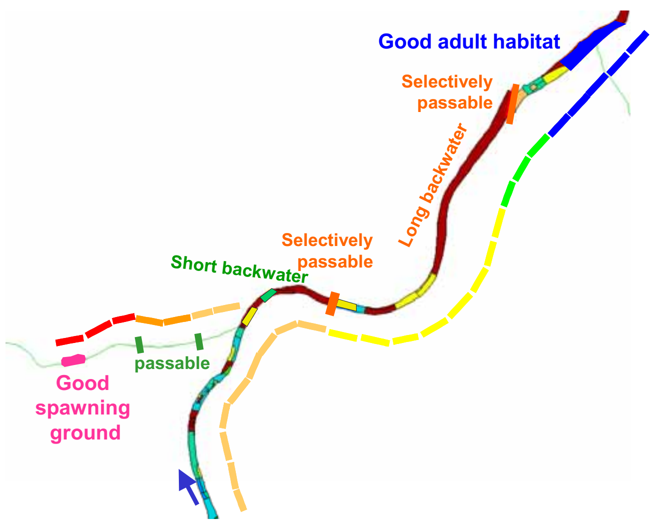

Fig.2: Aspects of migration, cumulative effect of migration barriers and visualization in a mesohabitat model

As the spatial relations between different habitat types can only be investigated after these habitat types are known consequently this step is the second one within the modeling process.

3.3 Water quality

Water quality can be seen as governing factor for the usability of habitats. As mentioned before fish habitats can be optimum from the hydromorphological point of view, but as long as they are not in a certain spatial context they won't make up a suitable "living space".

But even if the available "living space" comprises all kinds of habitat types - with a high connectivity and spatial connectivity - insufficient water quality will defeat these principally highly suitable conditions.

One problem in integrating water quality parameters in fish habitat modeling is that for many parameters the effects on fish physiology are hardly known. This is why in the current model development the focus is on two parameters, that comparatively most information is available for: water temperature and oxygen concentration. Even more than for other habitat parameters the transition between suitable and unsuitable ranges is fuzzy. Several authors distinguish between an optimum range and an lower and upper restricted, critical and lethal range.

Beside the absolute values of water temperature and oxygen concentration the duration of the impact is of high importance. Many fish can resist unfavourable conditions as long as they are not present for a longer time period.

These issues are hardly to describe by exact numbers. This is why a fuzzy rule based approach, successfully applied in the microscale, was chosen to describe these linkages (see example in Fig. 4).

Since water quality is more or less independent of the other two steps but dominating the overall habitat suitability it is considered in the third model step. Fig. 3 gives an overview of the model principle.

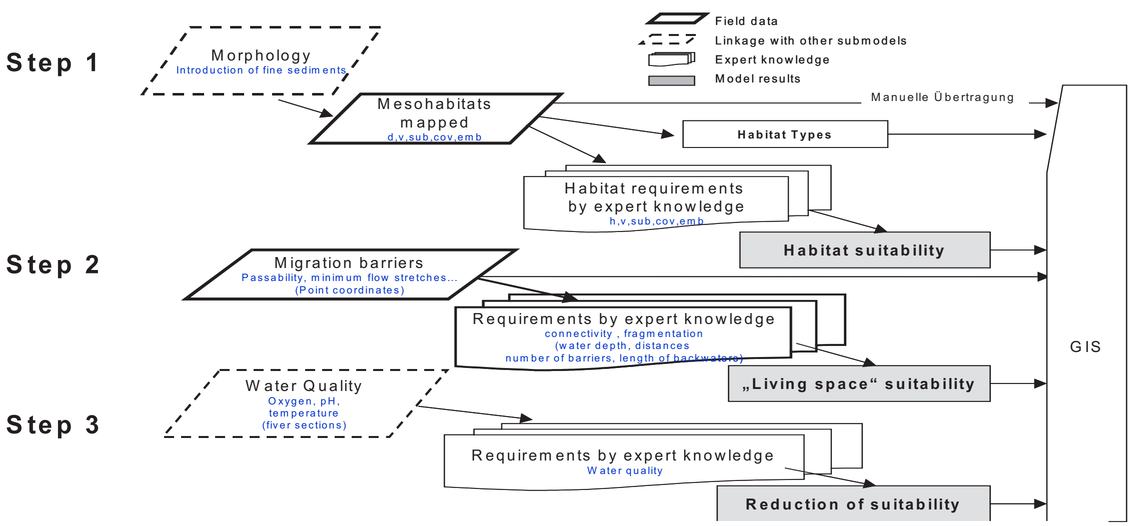

Fig. 3. Three step modelling principle of MesoCASiMiR integrating hydromorphology, connectivity and water quality aspects.

4 EXPERT KNOWLEDGE BASED APPROACH

Most of the knowledge about fish habitat requirements is more or less qualitative. Compared to the microscale, knowledge in the mesoscale is even more imprecise. What is a tolerable distance between a spawning and a juvenile habitat? What is the temperature a fish can resist for how many days?

It is impossible to answer these questions exactly, however many fish experts have an idea about what e.g. a “short” or a “long” backwater zone is. Investigations on temperature behaviour of fish as e.g. performed by Küttel et al. (2002) give an impression on what temperature ranges fish prefer, resist, avoid or cannot survive.

Fig. 4 gives an example of observed temperature ranges, assigned fuzzy sets and experts rules for the definition of habitat suitability based on the sets.

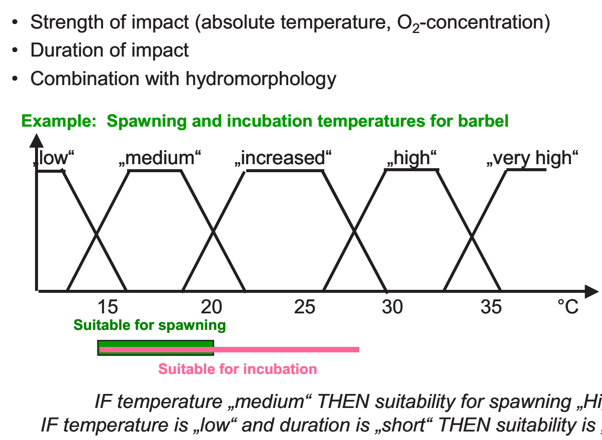

Fig. 4: Temperature ranges and related expert rules defining habitat suitability.

Instead of looking at single habitat parameters separately and integrating them afterwards the combination of parameters (e.g. temperature value and duration) is considered right from the start.

Another example for an expert rule describing the importance of habitat connectivity could be e.g.

IF habitat suitability of spawning habitat is “High” And suitability of juvenile habitat is “High” AND distance between spawning habitat and juvenile habitat is “medium” AND juvenile habitat is downstream AND there is no deep backwater zone in between THEN suitability is “High”.

The example shows that there is some more or less evident linkages between habitat types that can be used for the definition of habitat suitability. For other linkages rule definition might be a much bigger challenge.

However, expert knowledge in many cases is the only information that can be used for modeling. This is mainly because the efforts for finding out about habitat requirements in the mesoscale are quite high or just because there is no more target fish present in the investigated rivers.

So in consequence the proposed approach is not only suitable in the way that it uses knowledge as it is available – imprecise - but it might be the only approach practicable.

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Movement of juvenile salmonids on diel and seasonal scales in response to experimental hydropeaking in Newfoundland, Canada

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ABSTRACT: A five year experimental study was undertaken in insular Newfoundland, Canada to investigate effects of 'simulated' hydro peaking power generation on juvenile salmonids. Juvenile Atlantic salmon (*Salmo salar*) and brook trout (*Salvelinus fontinalis*) were implanted with radio transmitters, released into a experimental reach, and flow was manipulated over the range of 1.0 to 4.2 m^3s^{-1} during a series of 5 day 'experiments', representing water storage during day and generation at night, and the reverse, with gradual (2 hour transition) between minimum/ maximum flow. Fish were manually tracked, located in 2–D space, and movement examined. Salmon exhibited either high site fidelity or considerable movement during trials. Both species were more active during fall experiments. Salmon moved greater distances and had larger home ranges than trout under all experimental conditions, and during both seasons. Trout moved more in relation to dynamic events (up and down ramping) than at steady state flows. Additional experiments were conducted summer of 2002 and winter of 2003 to contrast the behavioural response of salmon between summer and winter periods. Protocols were similar to previous experiments except that the magnitude of change was greater (0.5 to 5.2 m^3s^{-1}) and flow changes were made more rapidly. Salmon were more mobile during all flow conditions and adopted greater home ranges and utilized more of the stream reach in summer than winter. Large home ranges and distances moved suggest hydro peaking regimes can be energetically costly which may affect production and survival, and winter movements, although less than summer, may be particularly important as this is a period when fish are conserving energy. Results are discussed in the context of peaking flow power generation on juvenile fish and it is anticipated these studies will assist hydro producers to design and operate hydro peaking regimes to minimize ecological impact.

1 INTRODUCTION

In Canada, and indeed globally, there is an increasing trend towards deregulation and variable pricing in the electrical energy market and this has provided hydroelectric producers with an economic incentive to respond to rapidly changing electrical demand, often on daily or hourly time scales (Morrison and Smokorowski 2000). Consequently more hydro power installations are producing electricity using hydropeaking, or pulse power, generation where water is stored to generate electricity during times of peak demand. This results in variable water pulses in a river below the power station reflecting unnatural flow patterns involving alterations to magnitude, duration, sequence, and frequency of flow. Rapid changes in river discharge and associated habitat conditions can occur over very short time scales (less than a day, or multiple peaks per day) and can be moderate to several orders of magnitude. Hydrological modifications of river character include alterations to stream bank and channel morphology, water depth, wetted area, velocity distribution, substrate composition, suspended matter, temperature, habitat structure and heterogeneity, and patterns and dynamics of ice formation (Gore and Petts 1989).

Hydropeaking may influence quantity and quality of habitat available to fish (Moog 1993; Valentin et al. 1996). Effects can be direct (e.g., stranding, mortality or habitat abandonment) or indirect (e.g., downstream displacement, volitional movement, depleted food production, increased physiological stress) (Moog, 1993; Valentin et al. 1996; Bradford 1997). Effects depend on capacity of fish to respond to temporary, often severe habitat alterations, and the ability to find and exploit hydraulic refugia (Valentin et al. 1996). Effects can include sub-lethal behavioral response and metabolic cost of holding station or moving from established territories (e.g., Flodmark et al. 2002). Rate of change and duration, time of day (light), season and/or temperature, behavior of fish, fish species and life stage (size), and the morphology and substrate character of the stream appear to be the most influential factors determining the effect of pulse power generation on fish (Halleraker et al. 2003, Steele and Smokorowski 2000). The mobility of fish ensures that they are less impacted by hydropeaking than sedentary species, including some invertebrates (Heggenes et al. 1999).

Relatively few studies have described the quantitative effects of hydro peaking power production on aquatic systems and, in particular, the non-lethal responses of fish have largely been unstudied. Most early studies involving fish examined severe impacts such as stranding (e.g. Bradford 1997; Valentin et al, 1996) and much recent research has been focused on sub-lethal impacts to non-stranded fish including behavioural and physiological responses (e.g Halleraker et al. 1999; Floodmark et al. 2002; Murchie and Smokorowski 2004; Berland et al. 2004). This paper summarizes a series of experiments conducted in insular Newfoundland, Canada, to determine the behavioral response of juvenile salmonids to experimental hydro peaking regimes Response was examined by determining number of movements and relative distance moved in relation to base (low) flow, transitional flow (up-ramping and down-ramping), and peak (high) flows conditions. Seasonal (summer, fall, and winter) and diel influences on movement patterns and home range were examined. Responses of two species, juvenile Atlantic salmon (*Salmo salar*) and brook trout (*Salvelinus fontinalis*), were compared and contrasted in initial experiments (1999). Later experiments (2002/2003), focused on Atlantic salmon, included a greater absolute change in discharge (between high and low flow conditions) as well as more rapid transition (higher ramping rate). Knowledge gained from these studies will assist producers of hydroelectricity, and regulatory agencies, help reduce the impacts of hydro peaking operations on fish and fish habitat.

2 STUDY SITE

Experiments were conducted at the Upper Salmon Hydroelectric Development (48^0 30′ N, 56^0 20′ W), constructed in 1979-80 by Newfoundland and Labrador Hydro in south-central insular Newfoundland, Canada (Figure 1). The study river, West Salmon River, is regulated with a two-level flow regime of 40% of the pre-project mean annual flow (MAF) (2.6 m^3s^{-1}) from June 1 to November 30 and 20% of the MAF (1.3 m^3s^{-1}) for the remainder of the year (Tennant 1975). This flow regime has been in effect since construction (circa 1981) with the exception of the experimental regimen associated with this study. Flow is regulated through a control structure (gate) in the West Salmon Dam, and the ability to control the flow on the West Salmon River, in an experimental fashion, made it an optimal site for this study.

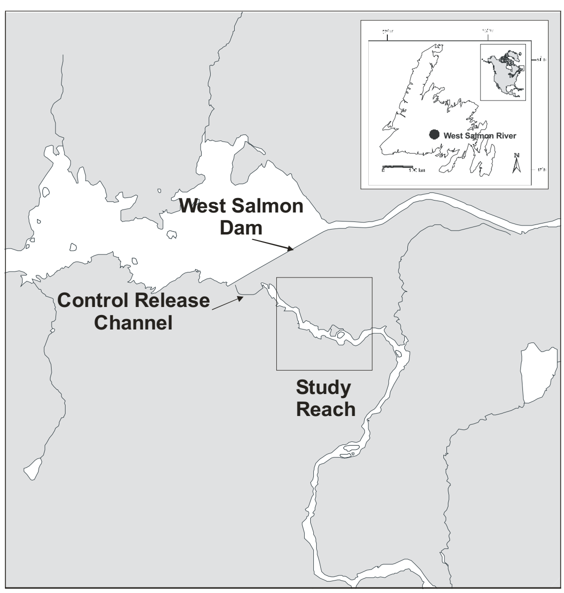

Figure 1. Map of the West Salmon River, Upper Salmon Hydroelectric Development, Newfoundland, Canada.

The river contains diverse high-quality habitats for juvenile landlocked Atlantic salmon, and brook trout and there are extensive data on hydraulics, habitat conditions and fish populations (e.g., LeDrew et al. 1996; Scruton and LeDrew 1997; Scruton 1998). The selected study reach, about 0.5 km below the West Salmon Dam, had a wetted width from 20 - 45 m at 2.6 m^3s^{-1} and a gradient from 0.2 % (downstream) to 0.8% (upstream) and included riffle, run, and pocket water meso-habitats (Scruton and Gibson 1995). The reach contains many small and large boulders, with minimal compaction from smaller substrates, leaving interstitial spaces that are used as refugia by juvenile fish. The reach, being broad and shallow, underwent substantial hydraulic changes in response to small changes in discharge.

3 MATERIALS AND METHODS

3.1 Topograghy and Hydraulics of the Study Reach

Initially, in 1998 and 1999, a series of transects (n = 57) were established to detail the topographic and hydraulic characteristics of the site and to provide a grid system for 2-D positioning of fish. End pins of each transect were positioned with a Differential Global Positioning System (DGPS) and registered with an aerial photograph in a Geographic Information System (GIS) (MapInfo, Version 7). Hydraulic (depth, velocity, substrate, cover) and river bottom topography data were collected at verticals (0.5 m spacing along transects) and at points of topographic change (boulder edges) using total station surveying methods. Additional topographic data were collected in 2002 as required for River2D modeling.

3.2 Experimental Animals

Juvenile Atlantic salmon and brook trout were captured from the study reach by electrofishing (Smith-Root Model Type 12, operating at 1000 V, frequency 60-120 Hz) and surgically implanted with radio transmitters (Lotek Wireless Inc., Newmarket, Ontario, Canada). In 1999, a total of 8 salmon and 7 trout in the summer, and 16 salmon and 9 trout in the fall, were surgically implanted with transmitters and their position monitored throughout the hydropeaking experiments. Experiments conducted in the summer 2002 and winter 2003 focused on Atlantic salmon only with 15 individuals included in each experiment, respectively. Surgical protocols for implantation of transmitters are described in Scruton et al. (2002). The lengths, weights and ratios of transmitter to fish weight for all experiments are detailed in Table 1.

Table 1. Fish length (cm), weight (g), and ratio of transmitter weight to fish weight (%) for each hydropeaking experiment.

Experiment		*n*	*min*	*max*	*mean*
Summer 1999, salmon	length	8	11.3	16.5	13.1
	weight	8	14.8	38.0	20.8
	ratio	8	2.0	5.1	3.97
Summer 1999, Trout	length	7	12.0	15.7	14.2
	weight	7	17.0	45.0	33.3
	ratio	7	1.7	4.4	2.75
Fall 1999, salmon	length	16	10.6	15.6	13.1
	weight	16	11.0	36.0	20.1
	ratio	16	2.1	6.8	4.22
Fall 1999, Trout	length	9	9.7	15.3	12.4
	weight	9	8.0	30.0	15.8
	ratio	9	2.5	9.4	5.55
Summer 2002, Salmon	length	15	12.0	21.1	13.7
	weight	15	18.0	81.0	27.4
	ratio	15	1.1	4.7	3.6
Winter 2002, Salmon	length	15	13.5	16.8	14.7
	weight	15	22.0	47.0	29.3
	ratio	15	3.8	6.9	5.4

3.3 Summer and Fall 1999 Experiments

Four experiments were conducted in 1999 to examine fish response to experimental hydro peaking (Table 2). Each experiment was conducted over a 5 day period and included two (2) summer experiments, June 26-30 and September 24-28, and two (2) fall experiments, October 21-25 and November 11-15. Discharge was experimentally varied over a range of 1.0 m^3s^{-1} to 4.2 m^3s^{-1} through the flow release structure in the West Salmon dam. Each experiment consisted of four 24 hour periods, representing two possible hydropeaking scenarios. The first scenario consisted of water storage during the day (low flow) and power generation at night (high flow). Low flow (1.2 m^3s^{-1}) was maintained for 10 hours, followed a transitional period of two hours (up ramping). During this transitional period flow was increased incrementally, to 1.8 m^3s^{-1}, through 3.4 m^3s^{-1}, and then to 4.2 m^3s^{-1} (maximum). High flow was then maintained for ten hours followed by another 2 hour transitional period during which flow was decreased from high to low (down ramping). The second scenario consisted of storing water during the night (low flow) and generating power during the day (high flow). Each scenario was repeated twice during each experiment.

Table 2. A summary of the dates, daytime and nighttime flow conditions, transition period (ramping rate), and water temperatures for each of the hydropeaking experiments.

Exp.	Day (m^3s^{-1})	Night (m^3s^{-1})	Trans. period	Water temp.
Summer 1999 (June 26-30)	High (4.2)	Low (1.0)	2 h	16. 0 to 21.6 ^{0}C (μ=18.38)
	Low (1.0)	High (4.2)	2 h	
Summer 1999 (September 24-28)	Low (1.0)	High (4.2)	2 h	11.5 to 19.7^{0}C (μ=15.25)
	High (4.2)	Low (1.0)	2 h	
Fall 1999 (October 21-25)	High (4.2)	Low (1.0)	2 h	7.1 to 8.7 ^{0}C (μ=7.5)
	Low (1.0)	High (4.2)	2 h	
Fall 1999 (November 11-15)	Low (1.0)	High (4.2)	2 h	1.3 to 5.2 ^{0}C (μ=3.49)
	High (4.2)	Low (1.0)	2 h	
Summer 2002 (August 20-26)	High (5.0)	Low (0.5)	30 min	17.4 to 22.7 ^{0}C (μ=19.72)
	Low (0.5)	High (5.0)	30 min	
Winter 2003 (March 15-25)	High (2.9)	Low (0.6)	30 min	-0.1 to 1.6 ^{0}C (μ =0.90)
	Low (0.6)	High (2.9)	30 min	

3.4 Summer 2002 and Winter 2003 Experiments

A second set of experiments were conducted in 2002 and 2003 to contrast fish response to flow changes between summer and winter conditions (Table 2). In the summer 2002 (August 20-26), discharge was experimentally varied from 0.5 to 5.0 m^3s^{-1}. The intention was to replicate these conditions in the winter 2003 experiments (March 15-25), but ice disabled the control structure; the resulting range was 0.62 to 2.95 m^3s^{-1}. Each experiment included four 24-h periods, representing the two operational scenarios as described for the 1999 experiments. The first scenario represented water storage during the day (low flow) and power generation at night (high flow). Low flows were maintained for approximately a 12-h period, and flow then was quickly changed by increasing discharge (up ramping) to the desired level. High flow was maintained for a 12-h period, followed by another rapid transition to low flow (down ramping). The second scenario was the reverse, involving storage at night (low flow) and power generation during the day (high flow). Each scenario was repeated twice during each experiment. The experimental conditions in 2002/2003, in contrast to the 1999 study, reflected a greater absolute change in discharge between high and low flow conditions as well as a rapid transition (higher ramping rate).

3.5 *Fish Tracking*

Radio telemetry was used to collect data on precise 2-D positioning of fish relative to the experimental protocols. After capture and surgical implantation of the radio transmitter, fish were held for a 24 hour recovery period, released close to the initial point of capture, and discharge remained unchanged for 24 hours allowing fish to distribute and adopt territories (Scruton *et al.*, 2002). Flow was then varied according to the experimental design and fish locations were monitored through manual tracking with a sequential scanning receiver (Lotek Model SRX_400) and a hand-held H- or dip antennae. The general location of each fish was initially determined from the shoreline (H-antennae) and the precise location of each fish was then determined using a dip antennae (stripped coaxial cable) and receiver at reduced gain (Bunt *et al.*, 1999). The precise location was verified, where possible, visually (viewing tube or snorkeling).

In 1999, a marker was placed at the fish location and position, to the nearest centimeter (cm) on a fiberglass measuring tape, was recorded relative to the nearest transect in the study reach. Once distance along the transect was measured, then the vertical distance to the fishes position was measured to provide precise positioning in 2-D space. In 2002 and 2003, fish locations were determined by total station surveying and use of DGPS. During winter, when the transect end pins were obscured by snow and ice, positions were determined by total station surveying relative to known benchmarks and DGPS. In all experiments, fish were located a minimum of four times daily, just before transition between high/low flows and after the transition.

3.6 *Data Analyses*

Fish locations were entered into a GIS for movement and home range analyses. The relative movement (frequency, distance) was evaluated as an indicator of response to the experimental peaking flows. Distance was determined from the position at the start of each experimental manipulation and end of the manipulation and any movement recorded within that time period, with each manipulation being (i) constant low flow, (ii) transitional flows (either up or down ramping), and constant high flow. In the 1999 experiments, seasonal (summer and fall) and diel influences on movement patterns were examined as well as inter-specific differences in response of the two study species, juvenile Atlantic salmon and brook trout. In 2002/2003, similar seasonal (summer and winter) and diel influences on movement patterns were examined for Atlantic salmon only. Home ranges were calculated as the minimum polygon to include all recorded positions, given at least three positions (Bachman 1984).

4 RESULTS

During the 1999 experiments, a total 27 salmon and 22 brook trout were released into the study area and, of these, 24 salmon and 16 trout were subsequently tracked within the study area and their movement patterns and distances observed. Nine (9) fish exited the study reach immediately after release, and were subsequently located in upstream and downstream areas, however movement distances were not included in the analysis. In the 2002/2003 experiments, 15 juvenile salmon for each experiment were released into the study reach, and 11 fish in summer and 11 fish in winter were subsequently tracked continuously. In the summer 2002 experiment, 2 fish were found dead, likely due to stranding, while 2 other fish yielded too few records to track movements and home ranges. In the winter 2003 experiment, one fish left the study reach and three others provided insufficient records for detailed analyses.

In 1999, the mean daily temperatures in the study reach ranged from 16.0 to 21.6 °C (mean of 18.38), 11.5 to 19.7 °C (mean of 15.25), 1.3 to 5.2 °C (mean of 3.49) in the June, September and November experiments, respectfully. Temperatures in November were below 6 to 7 °C, considered the threshold blow which juvenile salmon adopt winter habitat behaviour (e.g. Rimmer at al. 1983). Similarly the average day length was 16.0, 12.0 and 9.5 during the June, September and November experiments, respectfully. In the summer of 2002 (August), temperatures ranged from 17.4 to 22.7 °C (mean of 19.72) while in the winter of 2003, temperatures ranged from 0.1 to 1.6 °C (mean of 0.9) and the average day length was 14.9 and 12.1 days, respectively.

In 1999, there was a distinct pattern to the movements demonstrated by Atlantic salmon, with individual variations, however there were two groups of fish; those that showed considerable movement over the course of the experiments and those that remained within their territories for most of the experimental trials. Brook trout showed similar number of movements as salmon but less site fidelity. Movement differences between salmon and trout during the experimental manipulations in summer and fall is shown in Figure 2. Salmon demonstrated a greater distance moved than trout for all manipulations, and in both seasons, however these differences were not significant (Kruskal-Wallis test, $p < 0.05$). Both species were considerably more active during the fall hydro peaking experiments although no patterns were apparent in relation to up and down ramping or continuous low or high flows. Fish generally did not move long distances and most often stayed on the same side (bank) of the river reach and moved longitudinally more so than laterally.

Figure 2. Differences in movement numbers and distance moved between juvenile Atlantic salmon and brook trout, by season.

Patterns in diel and seasonal movements during the 1999 experiments for brook trout is demonstrated in Figure 3. In all cases, fish moved greater distances in the fall as compared to summer. Trout also moved greater distances at night during the up and down ramping events and also at low flows, but not at high flows. Trout demonstrated considerably more movements associated with dynamic events (up and down ramping) as compared to steady state conditions (low and high flows) in the fall.

Figure 3. Seasonal and diel differences in movement numbers and distance moved for juvenile brook trout for up and down ramping (top panel) and high and low flows (lower panel).

In 2002/2003, distances (mean ± S.E.) moved in each experiment were compared for combinations of flow and time of day (day time high [DH], day time low [DL], night time high [NH], and night time low [NL]) using Kruskal-Wallis and Dunn's multiple comparison tests ($\alpha = 0.05$) (Figure 4). There was no significant difference in the distances for diel flow conditions within the summer or winter data, but they were significantly greater (Mann-Whitney rank sum test, $P < 0.001$) in summer for all four flow and time combinations (DH, DL, NH, NL).

Distances (mean ± S.E.) moved were examined relative to increased (up-ramping) and decreased (down-ramping) flow in each of the experimental periods (seasons) using a 2-way analyses of variance (ANOVA) (Figure 5). Distances moved were significantly greater ($P = 0.023$) in summer than in winter, while the differences related to ramping regime (up and down ramping) were not significant ($P = 0.705$) after allowing for effects of season. There was no significant interaction ($P = 0.486$) between season and ramping.

Figure 4. Movement distances (mean +/- S.E.) for tagged juvenile Atlantic salmon in relation to flow (high/low) and diel (day/night) conditions for each of the experiments.

Figure 5. Movement distances (mean +/- S.E.) for tagged juvenile Atlantic salmon in relation to the experimental flow changes (up and down ramping) in the Summer 2002 and Winter 2003 experiments.

The home ranges of tagged fish in the summer and winter experiments are shown in Table 3. In 1999, summer home ranges for salmon varied from 196.4 to 1,479.0 m^2 ($\mu = 686.9$) while fall home ranges ranged from 0.58 to 3,208.0 m^2 ($\mu = 836.3$). Similarly brook trout summer home ranges in 1999 ranged from 3.0 to 45.5 m^2 ($\mu = 15.3$) and fall home ranges from 31.5 to 1,582.0 m^2 ($\mu = 536.3$). In 1999, brook trout home ranges were significantly different (less) in the summer than fall (Mann-Whitney rank sum test, $P=0.011$), and there was a significant difference between trout and salmon summer home ranges (Mann-Whitney rank sum test, $P=0.027$). In 2002/2003 salmon home ranges in summer varied from 0.56 to 3,177.0 m^2 ($\mu = 700.0$) and in winter, they were less, from 0.12 to 247.8 m^2 ($\mu = 65.1$) (Table 3). The difference was significant (Mann-Whitney rank sum test, $P < 0.001$).

Table 3. Home ranges (m^2) for juvenile salmonids for each of the hydropeaking experimental periods.

Exp.	n	Min.	Max.	Mean	S.D.
Brook Trout					
Summer 1999	4	3.0	45.5	15.3	20.2
Fall 1999	9	31.5	1,582.0	536.3	612.7
Atlantic salmon					
Summer 1999	7	196.4	1,479.0	686.9	497.9
Fall 1999	13	0.58	3,208.0	836.3	1,131.2
Summer 2002	11	0.56	3,177.0	700.0	1,045.3
Winter 2003	11	0.12	247.8	65.1	89.1

5 DISCUSSION

Competition for habitat by juvenile salmonids may be highly influenced by inter- and intra-specific competition and propensity to move from preferred habitats, as flows change, may be related to species, size and related social hierarchy (Shirvell 1994). Dominant and sub-dominant fish may react differentially as flow conditions change, and less dominant fish may more readily respond than dominant fish who hold territories (Mati-Petays et al. 1999). Mobile sub-dominants may be more profitable under changing discharge conditions such that under long term hydropeaking conditions, there may be a selective advantage for individuals with a mobile habitat selection strategy (Hutchings 1986). Individuals used in our study were larger and likely dominant juveniles, owing to transmitter size, therefore the stimulus for movement may not be reached until reduction in habitat increased intra-specific competition between dominants. In the 1999 study, salmon demonstrated a distinct movement pattern in response to hydro peaking as one group demonstrated strong site fidelity while another group moved considerably, suggesting a possible dominance based behavioral component to the observed responses (Scruton 2002b; Scruton et al. 2003).

Substrate provides a number of requisites for stream salmonids including velocity refuge and visual isolation (cover) thereby integrating biological functions such as foraging, competition and predation risk. Under changing flows, fish experience confinement and discomfort related to potential exposure to predators, and stream morphology and substrate character, in consideration of magnitude of discharge change, may be important in determining propensity of fish to move (Debowski and Beall 1995; Floodmark et al. 2002). Simple stream reaches experience incremental loss of preferred microhabitats, with changing flow, until a threshold is reached and fish passively (through displacement) or actively abandon the reach. In Norwegian studies, juvenile salmonids waited for exposure of their dorsal fin before moving and then followed the waters edge during receding and increasing flow. Substrate in the study river was predominantly cobble with little interstitial space affording fish no opportunity to exploit velocity refugia (Harby et al. 2001; Saltveit et al. 2001). In our study, fish were able to find ample refugia as the reach was hydraulically heterogeneous, characterized by boulder and cobble substrates with large interstitices, providing salmonids with extensive shelter from potential predators and increasing velocity. Physically and hydraulically complex stream reaches may afford better hydraulic and behavioural shelters during both low and high flow conditions (Pearson et al. 1992; Valentin et al. 1996).

Energetic considerations for highly territorial juvenile Atlantic salmon, in response to changes in discharge, are important as fish hold station and establish territories to maximize profitability under one flow condition, and as flows change these positions become less profitable (Armstrong et al. 1998). Energy expenditure associated with foraging can be reduced by restricting the distance traveled to reach prey, and even short travel distances can be costly at high velocities (Metcalf et al. 1986). Benthic species such as salmon use morphological and behavioral adaptations to utilize higher velocity habitats with less energetic constraints (Facey and Grossman 1992). Salmon parr resist downstream displacement in a current by holding station by clinging to the substrate with enlarged pectoral fins, acting as hydrofoils, generating negative lift as water flows over them (Arnold et al. 1991). This important behavior allows parr to avoid swimming while maintaining position adjacent to fast currents. Atlantic salmon parr are less buoyant and have greater maximum sustained swimming speeds than other salmonids, allowing them mre efficient access to drift resources (Sosiak 1982; Peake et al. 1997). In our studies, salmon that demonstrated high site fidelity did so by exploiting refuges and demonstrating the above behavioral modifications to flow increases.

Atlantic salmon parr demonstrate marked seasonal changes in habitat use and as temperatures decline in the fall, fish shelter in interstices in coarse substrates (Rimmer et al. 1983; Clarke and Scruton 1999) and, in winter, this is more pronounced with fish sheltering in substrate during the day and becoming active at night (Cunjak 1996; Heggenes et al. 1999). This behavioural shift is considered an adaptation to avoid predation, minimize energy expenditure and avoid harsh environmental conditions (Cunjak et al. 1998; Valdimarsson and Metcalf 1998. In winter, fish must maximize energy intake during the growing season to maintain lipid reserves to survive winter (Cunjak and Power 1987; Cunjak 1988) and movement and foraging in winter may be costly both in terms of predation risk and consumption of stored reserves (Valdimarsson and Metcalfe 1998; Metcalfe et al. 1999; Hiscock et al. 2002). Differences observed in movement by salmon in summer and winter in this study may indicate fish are sheltering in the substrate in the winter and are not exposed to ambient flow conditions, as in summer, when they would be actively foraging. In winter, fish are more concerned with conserving energy and are less active (Cunjak 1996), although previous studies in Newfoundland have suggested juvenile salmon are feeding in winter to augment energy reserves to improve over-winter survival (Robertson et al. 2004).

A common behavioral strategy of salmonids is to adopt an area over which fish normally move, including the established territory, termed a 'home range', and this can vary by species, life stage, degree of inter- and intra-specific competition, season, and environmental conditions (Gerking 1953; Bachman 1984). Saunders and Gee (1964) reported home ranges of 36 m^2 for Atlantic salmon parr in a small natural stream while Hesthagen (1990) reported home ranges of about 40-50 m^2 for age 1+ salmon parr in a small coastal Norwegian river. Armstrong et al. (1997) also reported small home ranges for 1+ salmon parr in a Scottish stream. Berland et al. (2004) found large home ranges (average of 2770 m^2) for salmon parr in a Norwegian River operated under a peaking hydropower regime. In the 1999 experiments, salmon had larger home ranges that brook trout in both the summer and fall (and indeed in all 4 individual experiments), however the differences were only significant in the summer. In the 2002/2003 experiments, there was marked variation in individual movements for salmon, and this was reflected in the size and variability in summer home ranges, from 0.56 to 3177 m^2 (μ=700 m^2), which were significantly different from the home ranges demonstrated in winter (from 0.12 to 248 m^2, μ= 65 m^2). Summer home ranges in our study are greater than those reported for small, natural, un-regulated streams but are less than those reported for a Norwegian hydropeaking river which had a much greater rate and magnitude of change. The large home ranges reported by Berland et al. (2004) and in our study in summer, suggest that hydropeaking regimes may potentially be energetically costly. The apparent greater movement under variable flow (hydropeaking) regimes could affect growth, production and survival. This may be particularly important in winter as fish are less active, conserving energy, and the stored energy reserves from the summer may be a critical factor in determining over winter survival.

6 CONCLUSIONS

Studies reported in this paper used an experimental approach to investigate the effects of hydropeaking power generation, and associated hydraulic and habitat alterations, on the home range and movement of juvenile Atlantic salmon brook trout on seasonal and diel scales. Salmon demonstrated two behavioural responses to flow change; individuals either demonstrated high site fidelity or a tendency for considerable movement between flow changes. Both species were more active in fall experiments than in summer, fish generally moved more in longitudinal direction that laterally across the stream, and salmon generally moved greater distances than trout under all experimental conditions. Trout demonstrated a greater movement response to dynamic (ramping) as opposed to steady state flow conditions. Salmon had larger home ranges and were more mobile during all flow conditions and over diel cycles in summer than winter and there was anecdotal evidence of stranding in isolated pools in summer. Stream morphology, in consideration of the magnitude of discharge change and season, can be an important determinant of the propensity (need and opportunity) to move and hence the likelihood of potential stranding.

The energetic consequences of movement, or holding station, in relation to hydropeaking is a critical aspect of the ecological consequences of this type of hydroelectric generation. Studies on stress (cortisol) indicate short term response to flow fluctuations with rapid habituation, and the period of time between stressors (ramping rate) may determine whether the response is cumulative (Floodmark et al. 2002). Energy demands during summer may affect stored energy reserves by the end of annual growth which could in turn affect over winter survival. Behavioural response to hydropeaking in winter as observed in this study, for fish to remain relatively sedentary, may increase the likelihood for dewatering, stranding, and freezing which may lead to higher mortality. In winter, energy reserves are depleted and any stress related to hydropeaking may potentially affect production and survival. This results of this research, conducted under experimental conditions, will assist utilities and regulators develop hydropeaking regimes that minimize impacts on resident fish.

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Environmental flow assessment for Slovenian streams and rivers

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ABSTRACT: In last 20 years, the problem of Environmental Flow (EF) assessment has been tackled through the development of a number of different methods around the world. EF provision requires definition of the quantity and quality of water, which is needed to preserve the ecological balance in the river and in the riparian zone.
The diversity of hydro-morphological types of rivers in Slovenia (karst, lowland and alpine rivers) and great biological diversity demand special treatment and determination of EF for each individual section of the river system.
On the basis of hydrological, morphological and ecological criteria, hydrological and ecological methods in Slovenia are applied for the determination of EF. From 1992 more than 180 study sites in streams and rivers have been examined for research and applications. The values of EF were determined mostly for existing water users. The EF assessment demonstrated that most water users abstract too large quantities of water in low-flow periods. We required water flow in the river to be increased to facilitate an improvement of conditions for instream biota and in the riparian zone. This means that especially in the low-flow periods smaller quantities of water may be abstracted from the rivers.
This paper describes the criteria and methods used to determine EF in Slovenia and experiences with their application.

1 INTRODUCTION

Natural flow regimes in many Slovenian running waters are modified by dams, transfers and a variety of hydraulic uses. Licensing the water uses demands the determination of Environmental Flows (EF) for water abstractions from running waters because of the need for protection of the natural environment and the enforcement of the Environmental Protection Act and the new Water Act (2002) in Slovenia.
In last 15 years, the problem of EF assessment has been tackled through the development of a number of different methods. A range of methods has been developed in various countries that can be employed to define ecological flow requirements. In broad terms, these can be classified into four categories: look up tables, desk top analysis, functional analyses and habitat modelling (Acreeman and Dunbar, 2004). The review of worldwide approaches for setting River Flows objectives were presented by Dunbar *et al*. (1998). The rapid assessment methods based on hydrological and hydraulic calculations are applied less often when the demands of ecology, river morphology, forestry and other branches are considered more frequently.

2 EVALUATION OF ENVIRONMENTAL FLOW IN SLOVENIA

2.1 Characteristics of Slovenia

There are numerous hydro-morphological types of running waters in Slovenia. This includes lowland, karst and mountain running waters to torrents. On average, in every km^2 of Slovenia there is 1.3 km of running waters, which means that Slovenia is very rich with running waters. The pattern of rainfall and the shape of the area creates high flow variability. Water regime is very variable and sensitive to all kinds of human impacts.
On the running waters there are water abstractions for drinking water, energetic use, fish farming, irrigation and technological purposes and many more are still planned. Most localities of the existing and planned water abstractions are in small river basins where no information about the quantity of water exists. Because of specifically hydro-geological conditions in particular sections of running waters, first of all, low-flow values should be determined by means of simultaneous measurements of water flows in the low flow period (Burja *et al*, 1995). To understand the nature of a environmental impacts a knowledge of the structure and function of river ecosystems is essential (Vrhovsek & Smolar, 1997). Such a notion of EF, of course, requires that each part of the running water is treated individually and that EF is determined with interdisciplinary co-operation (Vrhovsek *et al*., 1994).

2.2 Definition of Environmental Flow

The first definition of minimum flows of running waters in Slovenia (Uradni list SRS, 1976) was determined as a quantity of water, which enables the survival of water organisms. This formed the basis for granting permission, according to specific regulations, to ensure the availability of water supply for drinking and economic purposes. The evaluation of minimum flow was usually given by angling societies, but in many cases, the water users paid damages and too high abstraction resulted in no water in the river during low flow period. For this reason, Ministry of the Environment and Spatial Planning financed the research project beginning in 1992 to define the criteria for evaluating the provision of the quantity and quality of water to remain in the riverbed. The project was completed in 1994 (Vrhovsek *et al.,* 1994) and upgraded in 2002 (Smolar-Zvanut *et al.,* 2002).

Environmental flow provision in Slovenia (also referred to as ecologically acceptable flow) requires a definition of the quantity and quality of water which is needed to preserve the ecological balance in the running water and in the riparian zone. The EF is the quantity of water which enables the survival and reproduction of water organisms in different hydraulic habitats.

2.3 Criteria and methods for Environmental Flow Assessment

On the basis of hydrological, hydraulic, morphological and ecological criteria in the section concerned, hydrological and ecological methods have been developed in Slovenia in 1994. Since then, both methods were applied and improved through numerous assessments of EF in different types of running waters and for different water users. In 2002 (Smolar-Zvanut *et al.,* 2002) both methods were completed and are now used in practise.

The starting points for the definition of an EF by both methods are the basic hydrological and hydraulic parameters, such as the mean annual flow, the mean minimum flow, the minimum flow, etc. In some cases, a special analysis of flow in the months of low flow is required and consequently, a flow duration curve is constructed. In addition to the hydrological data, ecological data (for example, the ecological estimation of the river reach), an inventory of habitats and hydro-morphological estimation are needed by the hydrological method for the determination of EF.

In the application of the ecological method for the EF assessment, the samples of zoobenthos and phytobenthos at the chosen sampling points in the affected river sections are carried out. In the affected section, hydrological and morphological measurements are made; at the sampling points, the river depth, local velocities and the size of substrata are measured. The inventory and diversity of water organisms, the changes in biomass of phytobenthos are determined as well as an inventory of macrophytes, and flora and fauna of the riparian zone. It is suggested that ichthyologic research is also carried out. On the basis of the analysis, the existing situation is described. Depending on the quantity, length and duration of water abstraction, and the characteristics of the running waters, the research can be reduced or extended. Because of the seasonal dynamics of organisms and different flows during the year, the analysis should be performed during the whole year in different seasons, according to seasonal appearance of water organisms. The frequency of sampling is higher in the low flow periods when the effect of water abstraction on water organisms is higher.

In the research project (Smolar-Zvanut *et al.,* 2002) we proposed that EF should be assessed primarily by the ecological method if either, a) the running water is in a preserved or legally protected area, or, b) the abstraction lowers the flow to less than the mean minimum flow. For small running water (e.g. mean annual flow < 1 m^3/s) the EF should be mostly determined by the hydrological method.

Both methods applied are river assessment techniques. Our ecological method treats the community of water organisms and not only target species, while the main deficiency is, like all methods, the lack of evidence that biota responds to changes in flow regime (Petts and Maddock, 1995).

The EF is assessed according to biotic and abiotic parameters, which are critical with respect to the EF, where the ecological balance is still preserved. A decision about the EF is defined at a workshop using the "expert panel method" (Young *et al.*, 2004).

3 APPLICATION OF ENVIRONMENTAL FLOW ASSESSMENT

From 1992 on more than 180 study sites the EF have been examined using to hydrological and ecological methods. The values of EF were determined mostly for existing water users, where the tolerance-limit of the user economy was considered. In the running waters where hydroelectric power plants have existed for many years, high changes in the EF are questionable. In such cases the concept of EF in Slovenia is, that ecology and economy have their tolerant limits, within it is possible to find a compromise solution for determination of EF.

During the determination of the EF the results demonstrate that most water users abstract too large quantities of water in low-flow periods. We required the water flow in the riverbed to be increased and an improvement of conditions for instream biota and in the riparian zone. This means that especially in the low-flow periods smaller quantities of water may be abstracted from the running waters.

3.1 *Case study: The Rizana River*

Like most of the Mediterranean region, the area of Slovenian Istria has limited water resources. The only major spring is the karst spring of the Rizana River, which is also the most important source of water supply for the Slovenian coastal area (delivering a maximum of 0.35 m^3s^{-1}). The Rizana River is 14 km long and drains a watershed area of 204.5km^2. It has a mean flow of 4.1 m^3s^{-1} and a minimum daily flow of 0.01 m^3s^{-1}.

Downstream, the increasing density of population, and related negative impacts, such as settlements, agriculture, industry, trade, traffic, and landfill-sites can be observed. The demand for drinking water, as well as the industrial and agricultural exploitation of the Rizana River exceeds the water supply capability. The Rizana River has been a source of water supply since 1935. Directly below the Rizana River source is a water abstraction for a fish farm and for irrigation. There are a further 15 water abstractions downstream, and in summer some uncontrolled irrigation abstractions. The consequences can be observed primarily in the summer period when, due to the deterioration of the aquatic environment, there have been several cases of fish kills. A major cause of these has been the lack of dilution of pollutants.

Because of this major water exploitation and an almost complete absence of water in summer in the lower parts of the Rizana River, an EF was determined. An interdisciplinary approach was undertaken in order to achieve an improvement in conditions for water organisms.

Taking into consideration the hydrological, ecological, landscape and morphological characteristics and habitat evaluation, we proposed an EF value for the dry summer period of 0.160 $m^{-3}s$. This would dilute the pollution levels, and enable the maintenance of the ecological balance both in the river and in the riparian zone. The level of EF was decided by “expert panel” approach, taking into account the information detailed above, including the historical levels of abstraction (Smolar-Zvanut and Vrhovsek, 2003).

4 CONCLUSIONS

In the last 10 years there have been strong efforts to improve the ecological characteristics of Slovenian running waters. One important step of river basin management is the determination and assurance of an EF. The EF should be determined at the outset, before the proposals for the water development project are complete and the final decision on the construction is reached. The water should be abstracted only on the sections of the running water where this is ecologically and economically acceptable.

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Comparison of habitat modeling approaches to find seasonal environmental flow requirements in a Norwegian national salmon water course

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ABSTRACT: Several habitat hydraulic models are available for detailed studies of shorter river reaches. Most of these models are based on the same basic assumptions of the relationship between hydraulics and biology, but use different modeling approaches, input data and data analysis for the hydraulics and similar biological input. The objective of the article is to compare the hydraulic habitat models regarding their relevance in setting seasonal environmental flow requirements for river reaches. This approach focuses on the modeling process in total, how model variables vary as a function of flow in addition to the connection between the physical variables and the biological model. With this as a basis a comparison of three habitat hydraulic approaches were conducted. The programs in focus were HABITAT (SSIIM hydraulics), RIVER2D and ESTIMHAB. As part of a flow optimization project in the Surna river system, a 40 x 500 meter reach has been surveyed and measured in detail regarding physical and biological data. The habitat hydraulic programs were set-up and calibrated, and the reach was simulated using general biological preferences. The model results were compared. The results regarding environmental flow requirements from each model are presented and discussed in relation to the total habitat modeling process regarding each of the compared modeling tools.

1 INTRODUCTION

1.1 The Sande river reach

The river system Surna is located in the western parts of mid-Norway, in the counties of Rindal and Surnadal. The natural drainage basin in the Surna river system is 1200 km^2 in total (Bævre 1995, Svelle 2003).

Of this the regulated drainage area is approximately 560 km^2. The Surna river system has been regulated for hydro power since 1968 and both Follsjø and Gråsjø which drains to a power station in the middle parts of Surna are artificially dammed up lakes.

Salmon migrate 54 km from the river outlet to the Lomundsjøen Lake. In the middle parts of the migration route several stations for registration of fish habitat quality have been created for tracking, collection and registration of fish and habitat data, including biological and physical on-site parameters. The Sande river reach is located around habitat station 11 as viewed in Figure 1. The reach is located upstream the power station outlet and is semi-regulated as more upstream parts of the river is drained to the Follsjø reservoir.

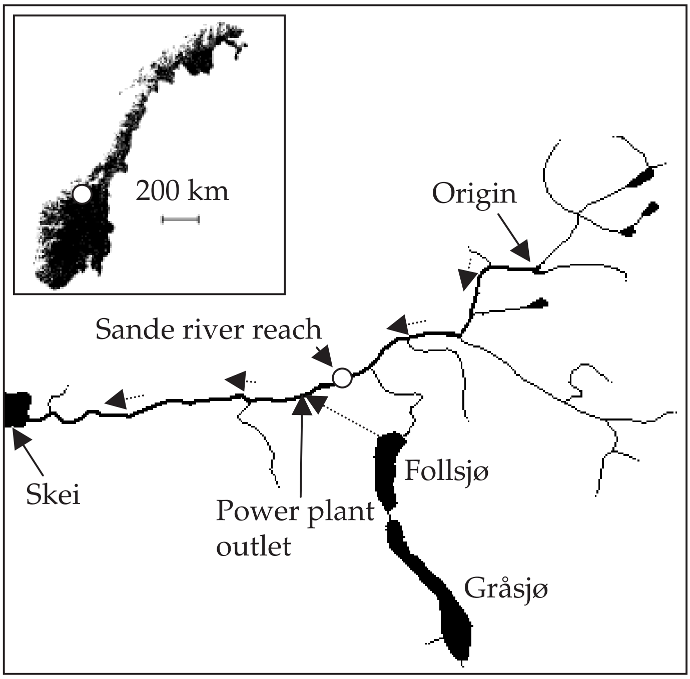


Figure 1: The Surna river system from origin (the rivers Rinna and Sunna) to the river outlet by Skei and the Follsjø reservoir.

1.2 *Background and study objective*

The regulation of the Surna river system has lead to redistribution of water in different parts of the basin. The river reach upstream the power station outlet has had water diverted into Follsjø reservoir, which have lead to a reduction of the median all-year-discharge from more than 20 m^3/s to approximately 10 m^3/s. As a result the average water covered area on the reach has been reduced by 20 % (Sundt et al, 2005).

A worldwide survey of available environmental flow methods showed that habitat-hydraulic modeling still are among the most widespread extensive methods applied in river systems with only a few target key species (Tharme, 2003). As a part of a future plan for revising the power company’s use of the Surna River System, detailed habitat modeling at micro scale at several locations in the river is planned. The process of field measurement and micro habitat modeling is time and cost consuming and new methods for conduction of such work are therefore considered. The objective of this study was to test and compare different approaches to micro habitat modeling for a small river reach in a semi regulated area of the river Surna. The method for the study was to utilize different model tools with either similar or dissimilar calculation or analysis of the reach of interest and compare the results in between the models.

In addition a comparison of the time and cost efficiency in addition to general utilization when using the different models was conducted. The idea was to, in future projects, be able to distinguish and choose the most appropriate method for the work in question. The comparison will work as a foundational tool which can be related to the extent of the work and the resources available.

2 MODELS AND METHODS

2.1 *Habitat-hydraulic and statistical models*

Predictive models for mapping of fish position choice are becoming more relevant as more model tools are becoming available for commercial use. The use of habitat-hydraulic and statistical models must be conducted at the relevant level of scale to be comparable with on-site fish preferences (Scruton 1998, Wiens 1989).

Several habitat-hydraulic and statistical models are available for modeling of micro stations.

ESTIMHAB (Lamouroux 2005) is a statistical model tool for estimation of fish habitat within a determined reach. The model tool makes use of topographical and hydraulic averages on the reach to predict (by inter- and extrapolation) habitat quality as a function of the local inflow discharge.

HABITAT (Alfredsen 2005) is a two-Dimensional habitat-hydraulic model tool, originally a part of the River System Simulator, for prediction of fish habitat quality within a river reach of predetermined length and extent. It requires input of water depth, water velocity and alternatively substrate size and composition, preferably at several contrasting discharges. The model is calibrated by adapting substrate size and composition within the area. The model tool determines local fish habitat quality within a mesh set by the topographical detail in the area, based on local fish preferences, if available. If not, general fish preferences are used.

RIVER2D (Steffler, Blackburn 2002) is a two-Dimensional depth averaged model of hydraulics and fish habitat similar to the HABITAT model. It uses water depth and flow as input and is calibrated by adapting substrate size and composition within the area. The model tool determines local fish habitat quality by Weighted Usable Area (WUA). The WUA is calculated as an aggregate of the product of a composite suitability index (CSI – range 0.0 – 1.0) (based on given preferences) evaluated at every point in the domain and the "tributary area" associated with that point (Steffler, Blackburn 2002).

2.2 *The study of micro sites*

Simulation on micro scale means working on the most relevant scale for fish. Micro scale parameters include topographical detail level with cm accuracy, water depth, water velocity, substrate size, composition and distribution, water temperature in addition to biological factors. As mesohabitat registration (Borsanyi, 2005) becomes more utilized, the fundamental data within each mesohabitat is of great importance. Further focus on the topic is vital as the area of interest is expanding.

3 RESULTS

3.1 *Preferences*

The HABITAT and River2D models use on-site registrations of fish placement and / or spawning locations as function of the depth, velocity, substrate in addition to riverbed cover (alternative). These data are given in preference files which connect parameter intervals to preferences of different habitat suitability (preferred, indifferent and avoidable). For this study general fish preferences (for habitat quality) were used due to the lack of on-site registrations of fish positions. The general preferences are based on fish position registrations in several Norwegian rivers with the same physical attributes as the Surna River (Harby et al, 1999).

Table 1: General summer fish preferences – Juvenile Salmon, based on snorkeling observations in many Norwegian rivers (based on Heggenes, 1996 and Harby et al, 1999).

Variable	[ ]	Preferred	Indifferent	Avoidable
Depth	cm	20–100	10–19 / 101-120	<10 / > 120
Velocity cm/s	cm/s	20-50	5-19 / 51-70	<5 / > 70
Substrate	cm	3-25	25-38	<3 / >38

The EstimHab model use internal fish preferences but have no registration of Salmon habitat and therefore (Lamouroux 2005) Trout preferences have been used as they are likely to be closest to the Salmon preferences.

3.2 ESTIMHAB

EstimHab input include average depth and width within the reach of interest at two (if achievable) distinctly different flows, in addition to average particle size and the median natural flow. Through inter- and extrapolation habitat suitability curves are calculated statistically as a function of the discharge with the included fish preferences.

The results from the statistical calculation indicates that optimum flow for adult fish (Trout – Salmo Trutta) is between 5 and 6 m^3/s, while it for juvenile fish (Trout – Salmo Trutta) reaches optimum flow around 2,5 m^3/s, as viewed in Figure 2.

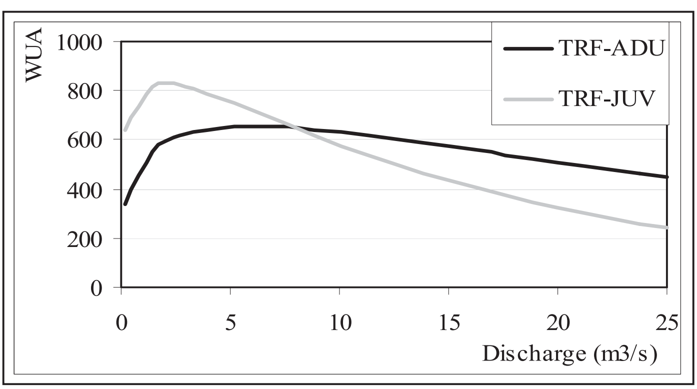
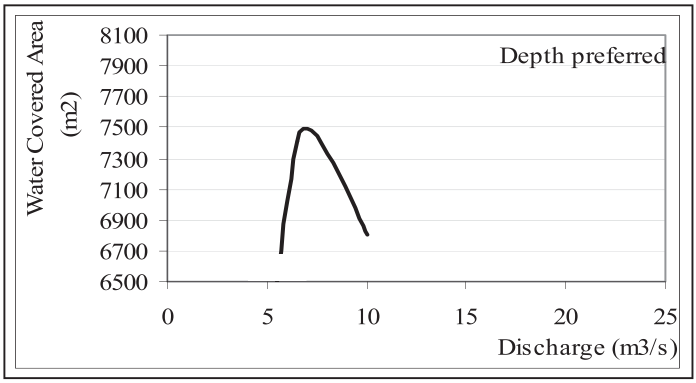


Figure 2: Weighted Usable Area from EstimHab calculations with Trout (Salmo Trutta) preferences (Adult and Juvenile).

3.3 HABITAT (SSIIM)

Habitat input includes a detailed topographical surface file, based on random or distributed points, often supplied through interpolation in surface mapping programs (Surfer). In addition a hydraulic base file created using hydraulic modeling tools (SSIIM or Hec) is necessary to connect the preferences to local velocities, depths and substrate size and composition.



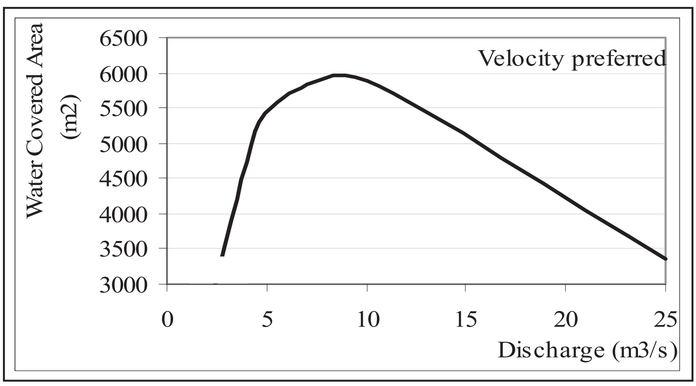


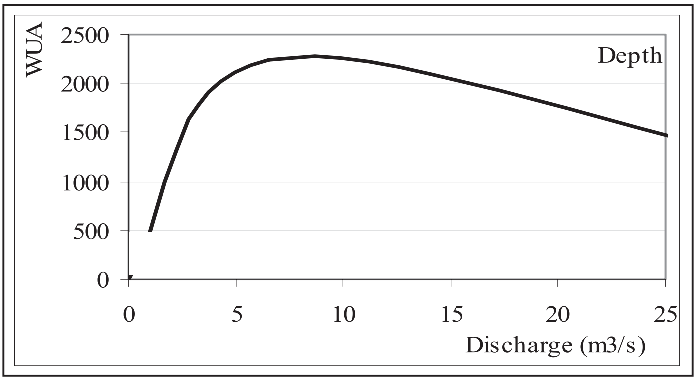
Figure 3 and 4: Preferred depth and velocity results from HABITAT (SSIIM) calculations with general summer preferences for young Salmon (Salmo Salar).

Figure 3 and 4 show the HABITAT results when connecting the hydraulic model results with general salmon preferences (Harby et al, 1999) as viewed in Table 1. Figure 3 shows preferred depth as a function of the discharge (m^3/s) and indicate an optimum between 7 and 8 m^3/s. Figure 4 shows preferred velocity as a function of the discharge (m^3/s) and indicate an optimum around 9 m^3/s.

3.4 RIVER2D

River2D input are fundamentally much of the same as for HABITAT (SSIIM) and includes detailed topographical surface file, based on random or distributed points, often supplied through interpolation in surface mapping programs (Surfer).

Figure 5 and 6 show the RIVER2D results when connecting the hydraulic model results with general salmon preferences (Harby et al, 1999) as viewed in Table 1. Figure 5 shows WUA for depth as a function of the discharge (m^3/s) and indicate an optimum at 9 m^3/s. Figure 6 shows WUA for velocity as a function of the discharge (m^3/s) and indicate an optimum around 7 m^3/s.



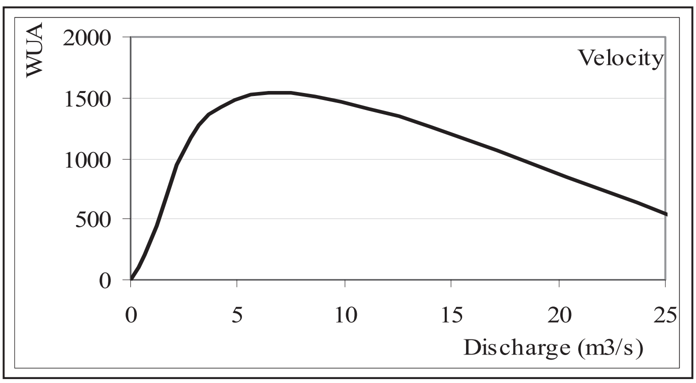


Figure 5 and 6: Preferred depth and velocity results from RIVER2D calculations with general summer preferences for young Salmon (Salmo Salar).

Table 2: Comparison of optimal habitat flow (the Sande reach)

Parameter	ESTIMHAB	HABITAT	RIVER2D
Depth	6 (2) m^3/s	7-8 m^3/s	9 m^3/s
Velocity	6 (2) m^3/s	9 m^3/s	7 m^3/s

OBS! ESTIMHAB values are for Trout (Salmo Trutta). HABITAT and RIVER2D values are for young Salmon (Salmo Salar) (summer).

4 DISCUSSION

Conducting field measurements in relation to micro scale modeling can be a time and cost consuming experience. Though field work is of great relevance, a simplification of the field process would in many cases aid to increase the focus on other essential parts of the project, i.e. the analysis of the resulting data including discussion and conclusion. A simplification must be of such an extent that it does not interfere with the necessary basis data needed for the main analysis. Thus models with an easier field campaign often will be to prefer as more work can be done in the latter parts of the project.

Regarding ESTIMHAB, as a rule, ten or more widths must be measured equally distributed along the reach of interest at each distinct flow. In addition the average depth must be calculated from at least 40 random points within the reach area, again at each of the flows. This indicates that no advanced topographical outside the 40 random depth points need to be conducted.

Regarding HABITAT, the collection of points to obtain a topographical map of high detail can be an extensive campaign. All topographically extreme points (those of importance when conducting inter- or extrapolation) must be registered to secure a correct representation of the area. This often means entering areas of high difficulty, available only when the flow is low. HABITAT also needs to be calibrated on several distinctly different flows (ideally three or more). This indicates that an equal amount of representative water lines (including downstream water level), water depths and water velocities must be collected in the reach area.

Regarding RIVER2D, The model tool consists of several sub programs. The RIVER2D Bed program establishes a topographical surface file for further use in the main program using distinct coordinate fixed points with height values to create a grid onto which the simulations will progress. During simulation the grid can be adjusted to aid to the calibration of the model. Calibration can be conducted changing the riverbed substrate size and composition, equally to the HABITAT (SSIIM) model. The calibration is checked against observed local values for depth and velocity for different flows.

When comparing these three habitat models regarding time and cost efficiency and general utilization, one has to consider important factors like the accuracy of the results (relevance and precision) in addition to the field and model work extensiveness. The model results in this study indicate (if only vaguely) that the models are comparable when it comes to habitat determination as function of flow, (Table 3). If assumed that juvenile trout follows much of the same patterns as salmonids, the optimal habitat flow for young fish is situated between 6 and 9 m^3/s for the Sande river reach. Importantly, these data are based on a minor of flows for calibration. Additional data must be collected for further verification of the three habitat models to ensure a correct comparison. It is suggested that the earlier version of ESTIMHAB – STATHAB - is utilized with the Salmon preferences.

Table 3 gives an overall overview for the three habitat models in focus in this study, based on rating of different parameters of interest and relevance when focusing on the whole habitat modeling process from start to end. Beware of objectivity.

Table 3: Comparison table for habitat modeling approaches – model utilization rating from 1 to 6 (best)

Parameter	ESTIMHAB	HABITAT	RIVER2D
Hydraulic precision	3	5	5
User-friendliness	6	3	4
Field work cost	5	3	3
Modelling cost	5	3	4
Sustainability (proving)	4	5	5
Relevance for Norway	2	5	5
Average	4.2	4.0	4.3

OBS! Beware of objectivity!

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Coupling water quality and fish habitat models for river management: simulation exercises in the Dender basin

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ABSTRACT: During the last ten years, a lot of research on pollution loads has been done on the Dender river. This research revealed that the pollution of the Dender river is highly variable, since there are many types of direct discharges (industry, treated and untreated sewage from households, …) in combination with nutrient releases from agricultural activities. The high nutrient loads result in severe algae blooms during summer, leading to complex diurnal processes. Additionally, a lot of structural habitat modifications were established to ease flood control and guarantee boat traffic. These modifications have however a severe impact on the habitat characteristics and induced a completely different fish community compared to the natural conditions. To get a better understanding of these combined effects, water quality models of the Dender river were developed in ESWAT. Additionally, fish species models were generated on the basis of data driven methods. These latter models allow predicting communities on the basis of the outcomes of the water quality model simulations and habitat data. This paper presents the methodology used to develop both type of models and their combination, and shows their practical use to make simulation exercises relevant for the implementation of the European Water Framework Directive in Flanders.

1 INTRODUCTION

Pike, Essox Lucius is a fish that did occur in our rivers a lot. Pikes can be found in all of northern freshwaters and can be found in cold deep lakes as well as in warm shallow ponds and muddy rivers. Having a broad range of tolerances for water temperature, clarity, and oxygen content allows E. Lucius to be "one of the most adaptable freshwater species" (Sternberg,1992) Due to pollution of our river waters and the change in natural habitats, pike community decreased of even disappeared in our Belgian rivers. To fulfill the Water Framework Directive (2000) it will be necessary to undertake actions to bring back pike. What those actions have to be is however not so easy. Is it only a reduction in pollution load towards the river, is it acute pollution or chronicle impacts that is mostly influencing, or is the habitat like river bottom, flow velocity, etc. In this research it is the aim to link the factors that are most influencing to pike species to water quality predictions and this way predicting the occurrence of pike. Therefore the model outcome of a river water quality model is linked with a model that predicts biological life in rivers.

2 DENDER RIVER BASIN AND SITE DESCRIPTION

The catchment of the river Dender has a total area of 1384 km^2 and has an average discharge of 10 m^3/s at its mouth. As about 90% of the flow results from storm runoff and the sources make very little contribution, the flow of the river is very irregular with high peak discharges during intensive rain events and very low flows during dry periods. To allow for navigation and to temper the high flows, the Dender is canalized and regulated by 14 sluices between. Due to this, during dry periods, the Dender acts as a series of reservoirs with a typical depth of 3 to 5 m, a width of 12 to 50 m and a length of 2 to 8 km. During periods of high flows, all locks are opened and the river regains a more natural stream profile.

In 1999 and 2000, 17 sampling places were described on the Dender. Next to the measurements that were done on those places also a description of all the structural characteristics were given (D'heygere et al., 2002). Three of those measurement places are used in this study. It are Aalst, pollare and Denderbelle (Fig. 1).

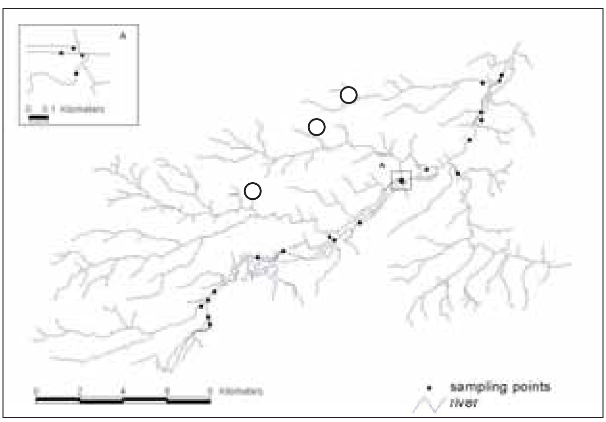

Figure 1 The river Dender with indication of the measurement places

3 ESWAT MODEL DEVELOPMENT

A water quantity and quality model for the river Dender for 1999 and 2000 was implemented in ESWAT. ESWAT is an extension of SWAT (van Griensven and Bauwens, 2001), the Soil and Water Assessment Tool developed by the USDA (Arnold et al., 1998). ESWAT was developed to allow for an integral modelling of the water quantity and quality processes in river basins. The water quality model is dynamic and QUAL2E based. Output can be generated on an hourly, daily or monthly basis.

Inputs for the model are evaporation, relative humidity, temperature, solar radiation and rainfall data every 10 minutes for Pollare, a city situated in the centre of the catchment, flow data upstream, daily water quality data upstream for DO, BOD, NO_3, NH_4. Point pollution inputs comprise wastewater treatment plants outlets, industries and untreated inhabitants. For the diffuse pollution the fertiliser use, land use and management practices are introduced in the ESWAT model. Ground and crop processes are taken into account to calculate diffuse pollution towards the river.

4 FISH HABITAT MODEL DEVELOPMENT: CASE-STUDY OF PIKE

Classification trees were constructed on the basis of the Weka software (Witten and Frank, 2000). Weka is a collection of machine learning algorithms for data mining tasks. The algorithms can either be applied directly to a dataset or called from your own Java code. Weka contains tools for data pre-processing, classification, regression, clustering, association rules, and visualization. It is also well suited for developing new machine learning schemes.

The applied algorithm to grow classification trees is 'weka.classifiers.trees.J48'. This is an algorithm to construct pruned or unpruned C4.5 classification trees (Witten and Frank, 2000). Default settings were applied (based on trial and error these gave relative good results that are difficult to improve in a general manner for all used datasets), except for the confidence factor that was set at 0.50 (default 0.25), which had a very important effect on the selected variables and the number that were used for the classification.

A dataset was constructed on the basis of electro fishing data, collected in rivers of Flanders. In total, 168 measurements were used, of which in 50% of the cases pike was present. A training set of 112 instances was used for classification tree development, while 56 instances served for validation of the constructed model. In both subsets, 50% of the instances were characterized by pike presence. In addition to the presence/absence of pike, eight variables (river characteristics) were available that could serve for the prediction of pike: width, slope, depth, electrical conductivity, dissolved oxygen, pH, water temperature.

5 RESULTS

5.1 ESWAT model

The model is first calibrated for the hydraulics. Then the water quality parameters are considered for calibration. The ESWAT model contains 33 water quality parameters that can be tuned. Here a manual calibration is performed with monthly measurements taken by the VMM, on two places, one in Pollare and one in Aalst. The parameters changed to obtain a calibrated model were based on the parameters found in Vandenberghe et al. (2001). Figure 2 and 3 show the time series for DO concentrations in the river at Pollare and Aalst for 1999 and 2000 together with the VMM measurements. The model was on the same time also calibrated for temperature (fig. 4), NO_3 and NH_4. The model simulations capture not all the measurement points. This is because of the few measurement points available on the one hand and because of uncertainties in input and parameters. However all dynamics are well covered and it is assumed that the model simulations are accurate enough to link them to biological prediction models.

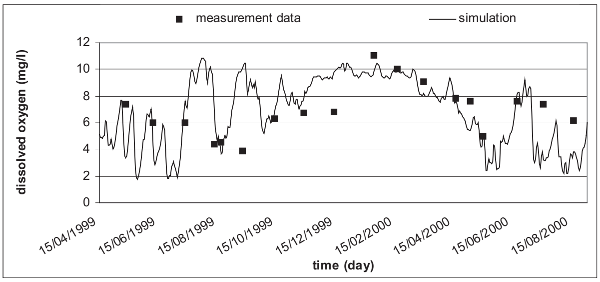

Figure 2. Simulation time series and measured data of DO at Pollare from March 1999 till August 2000

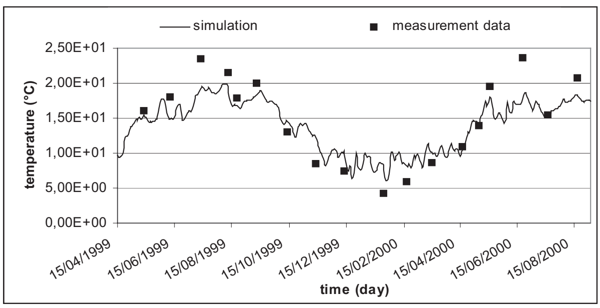

Figure 3. Simulation time series and measured data of DO at Aalst from March 1999 till August 2000

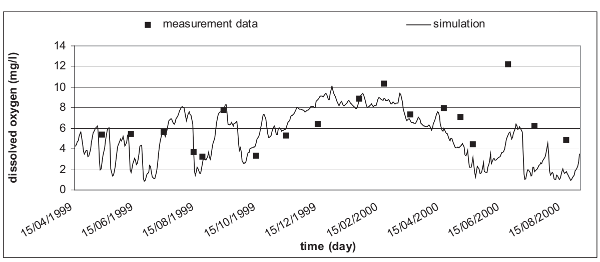

Figure 4. Simulation of water temperature at Pollare with measurements from March 1999 till August 2000

5.2 Fish habitat model pike based on classification trees

The reliability of the model was proven by the prediction assessment in the validation dataset. About 71 % of the instances was correctly predicted (CCI of 71 and Cohen's Kappa of 0.43). The tree consisted of the following rule set (D'heyghere et al., 2003):

```
WIDTH <= 2.54
          SLOPE <=0.8 : PIKE PRESENT
          SLOPE >0.8 : PIKE ABSENT
WIDTH > 2.54
          SLOPE <=0.3 : PIKE PRESENT
          SLOPE >0.3
                    EC <= 419 : PIKE PRESENT
                    EC > 419
                              EC <= 607 : PIKE ABSENT

                              EC > 607
                                        DO <= 7.1 : PIKE ABSENT
                                        DO  > 7.1
                                                  DEPTH <= 0.5
                                                         SLOPE <= 2.1: PIKE PRESENT
                                                            SLOPE  > 2.1: PIKE ABSENT
                                                  DEPTH > 0.5 PIKE PRESENT
```

Results are then obtained according to the methodology describeded in Goethals (2005).

There exist long periods in DO concentration below critical value of pike. This is also found back looking at the results of the predictions were it is shown that pike is absent. Pike is endangered based on the water quality. This is mainly related to algae blooms, as a result of nutrient inflow. On top of this also the habitat quality is very poor in the stem river, while the tributaries are characterized by a very bad water quality. The remaining population was based on fish stockings, but when water quality is not improved, these activities seem to be useless.

6 DISCUSSION

Not only is the dissolved oxygen content of the river Dender changing day by day, also during the day a typical dissolved oxygen profile can be seen caused by algae blooms, where the oxygen content increases during the day and lowers during the night (Fig.5) (Vandenberghe et al., in press). If simulations of water quality based on an hourly basis would be introduced, together with more than 1 electro fishing event, the model results could be more accurate. However the data nowadays available don’t allow us to have good model predictions on an hourly basis. To be able to have such databases, more communication between physico-chemical and biological measurement campaigns and a good database set-up and management will be needed.

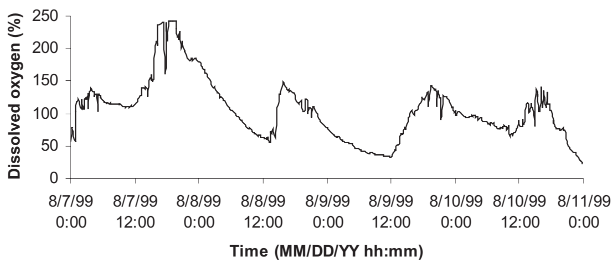

Figure 5. Diurnal variation of the dissolved oxygen in the Dender river at Geraardsbergen during the period 07/08/99-10/08/99 (DD = day, MM = month, YY = year, hh = hour, mm = minute)

7 CONCLUSIONS

This research is a first step into the direction of the coupled prediction of physico-chemical water predictions and biological life prediction models. They show us the key factors for the occurrence of fish species and can help in bringing forward solutions to come to the pristine situation for surface water as described in the Water Framework Directive. At the same time, such a research also shows the gaps in the data nowadays available and can give guidelines towards better data gathering and management.

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Local scale factors affect fish assemblage in a short-term regulated river reservoir

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ABSTRACT: Short-term streamflow regulation (hydropeaking) affects the ecology of regulated rivers. We examined the longitudinal and temporal changes occurring in fish assemblages in a hydropeaking single river reservoir between two power plants by using electrofishing along the shoreline, hydroacoustics and test fishing in the open water. A longitudinally changing fish community was found among bottom-dwelling fish in the fast-flowing and highly disturbed upstream part of the reservoir progressing to generalists and pelagic fish in the lentic and most stable environment at the downstream end. The fish assemblage showed temporal patterns as fish density increased during night-time darkness and also towards autumn. Our work provides evidence for gradient effects of flow regulation and contributes to awareness of the effects of disturbance (flow/habitat variability) on biological systems.

1 INTRODUCTION

Short-term regulation (hydropeaking) is a potent form of flow regulation (Harby et al., 2002). Discharge from the power plant usually follows the demands of the electricity markets. Changes in discharge occur rapidly and can be large: within only a few hours the discharge can become many-fold higher or lower. Even more important than the magnitude, from the standpoint of the regional fauna, is the unpredictability of flow changes. Although some general patterns do occur, e.g. flow is usually lower at night when energy demand is lower, flow changes usually are irregular. Little is known about how these unpredictable changes in flow modify the animal communities (but see Bain et al., 1988), and therefore the effects of hydropeaking on the aquatic ecosystem are not well understood.

Fish assemblage in the river is determined both large scale and local scale processes. Generally large scale processes determine the pool of species available whereas local scale factors, e.g. flow regime, tend to diminish the number of species actually present. Longitudal changes in fish community together with the upstream-downstream gradient of a river are consequences of both biological zonation and increase in river size (Huet, 1959, Schlosser, 1982). Flow regime, which largely contributes to how stable the environment is in streams, and physical habitat characteristics (water depth, current velocity, substrate) are important determinants of stream fish community structure (Schlosser, 1985, Bovee, 1986). In a hydropower reservoir flow regime and habitat characteristics usually change in a relative short distance without actual biological zonation. The main aim of this study was to examine how fish community responds to these artificial man made changes in physical factors, water velocity, depth and their short-term fluctuation. Our objective was especially to study the gradient effects of flow regulation downstream from the point of regulation. Most of the previous work on flow regulation is from river habitats downstream of a dam, whereas the present paper considers a flowing reservoir habitat between two dams affected by flow releases from another upstream reservoir. Our work contributes to awareness of the effects of disturbance (flow/habitat variability) on structure and function of biological systems.

2 MATERIAL AND METHODS

2.1 Study area

The River Oulujoki is a 110-km-long stream flowing from Lake Oulujärvi into the Gulf of Bothnia in the Baltic Sea. The river has seven power plants, which are used for hydropower production (Fig. 1).

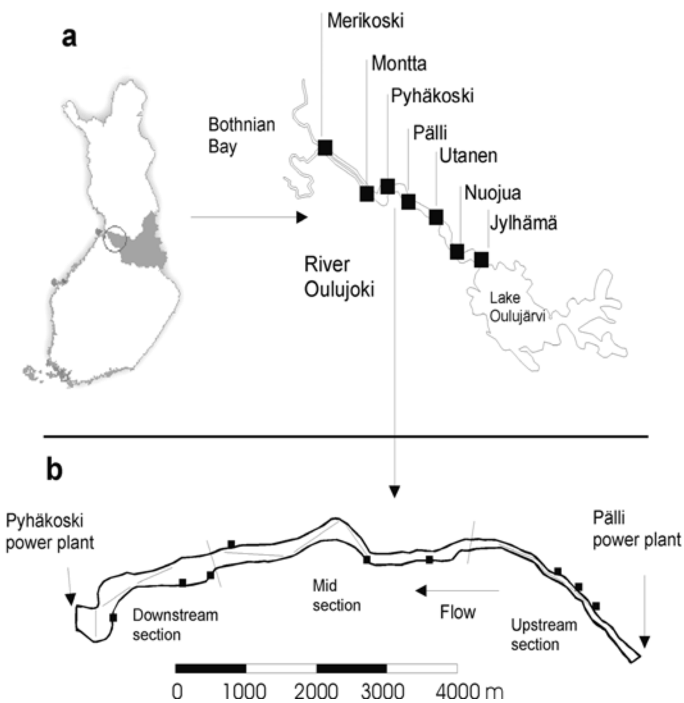

Figure 1. a) Location of the study area. Darkly shaded area in Finland shows the area of the Oulujoki River drainage basin and black squares the locations of power plants along the Oulujoki River. b) The Pyhäkoski River Reservoir. Black squares show the location of electrofishing areas, and the dotted lines the location of hydroacoustic sampling lines. Solid lines separate the reservoir into 3 sampling areas: upstream, mid- and downstream sections.

The study was conducted in the Pyhäkoski Reservoir, which is 8.7-km-long impoundment between the Pälli and Pyhäkoski Power Plants (Fig. 1). The mean daily discharge during the study period, May 5, 2000-October 31, 2000, was 208 (range 27-508) m3 s-1. As a result of peaking power production, the discharge in the reservoir also varies widely during a day, according to energy demand. The river water is mesotrophic and slightly humic, its colour being about 75 mg Pt l-1 and pH around 6.7.

2.2 *Physical habitat mapping and habitat modelling*

Habitat measurements were made using a combination of an echo sounder for bottom topography points together with differential GPS (Trimble 4000 SSI) position points. An acoustic Doppler device (Acoustic Doppler Current Profiler™) was used for water velocity sampling. A small boat able to move in shallow water carried all equipment and data were saved on laptop computers (Kylmänen et al., 2001). The data obtained were processed and input to a 2-D hydraulic model RMA2 (US Army Corps of Engineers) and SMS (Surface Water Modelling System) graphical interface. Eight different discharge situations were simulated: 30, 70, 120, 200, 300, 400, 500 and 700 m3 s-1. The 2-D flow model consisted of 19 698 rectangular elements, corresponding to 60 215 calculation points. There were 29 elements corresponding to 57 calculation points across the river. With this density, there is approximately 1 point per 5 m across the river and about 1 point per 12 m along the river.

2.3 *Sampling areas for fish data*

Corresponding to changes in the geomorphology and current velocities of the reservoir, the impoundment was a priori divided into three areas for fish data sampling (Fig. 1). Current velocities and water depths in the open water were significantly different between the areas (ANOVA; $P < 0.001$, df = 2). The upstream sampling area (hereafter, upstream section) of the reservoir is the shallowest and current velocities highest there. The downstream sampling area (downstream section) is the most lentic and with the lowest current velocities and highest water depths. The midsampling area (midsection), is where the physical characteristics of the river reservoir gradually change from the lotic conditions of the upstream section towards the almost lentic ones in the downstream section, with intermediate current velocities and water depths.

The upstream section has the most unpredictable current velocity regime, whereas the downstream section is the most stable. The range of mean current velocities when discharge changes from 30 m3s-1 to 500 m3s^{-1} is 1.05 ms^{-1} (0.07-1.125 ms^{-1}) in the uppermost section, 0.44 ms^{-1} (0.03-0.47 ms^{-1}) in the midsection and only 0.13 ms^{-1} (0.009-0.14 ms^{-1}) in the lowermost section. Correspondingly changes in mean water depth are higher in the upstream section and decrease downstream.

To describe the physical conditions along the shore, the water depth, current velocity and substrate size were measured from three lines (1 m, 2m and 3 m from the shoreline) across each electrofishing area (3 in each of the 3 sampling areas). The overhanging vegetation (percentage cover) was also assessed for each area. The current velocities and water depths differed significantly between sampling areas (ANOVA, $P < 0.001$, $df = 2$); current velocity and water depth were highest (i.e. shoreline was steepest) in the upstream section and lowest in the downstream section. Cover of overhanging vegetation was densest in the upstream section, variable in the midsection and lowest in the downstream lentic area. Substrate distribution in the two upstream areas consisted of gravel and large cobbles, whereas it was mostly fine substrates with small cobbles in the lowermost area.

2.4 *Electrofishing and hydroacoustic sampling*

Three permanent electrofishing sites and three hydroacoustic sampling lines were selected from each of the three areas, and marked with landmarks (Fig. 1). The electrofishing sites were randomly selected after dividing each area into 100 zones in the map. Each sampling site was placed in the middle of the selected zone. At night, small fires were used to mark the lines used for hydroacoustic sampling and electrofishing was done using handheld halogen lamps. In each case an area of 27 m2 (9 x 3 m) was electrofished. The area was approached quietly from upstream, using a small boat, and it was surrounded with a small-meshed stop net. The area was then electrofished using a Honda EX 1000 W power supply operated from the shore with a pulsed DC current of 0.2 amperes at 1200 V, square wave-form and 50-Hz pulse frequency.

Hydroacoustic sampling was done with a zig-zag survey across the river (Fig. 1). A vertically aimed 120 kHz split-beam echosounder (Simrad EY500, ES120-7F transducer, 7o beam width, 2 pulses per second) was used. The transducer was carried at about a 20-cm depth, at 1 m distance from the side of the boat. The data were processed with an EP500 program to calculate fish densities (fish ha^{-1}).

Both electrofishing and hydroacoustic sampling were done both day and night on three occasions: May 15-30, August 15-30 and October 10-20, 2000. The water temperature was 10-13° C in May, 15-17° C in August and 10-12 ° C in October, respectively. In these latitudes night-time in late May is only a short period of twilight (sun about 20 h per day above the horizon), longer in August (about 15 h) and October (about 9 h). Electrofishing and hydroacoustics were in each occasion begun at least 30 min. after the sunset or sunrise.

The vertical distribution of fish in the water column was during hydroacoustic sampling examined by dividing the sampled water column into two layers: surface layer (4-16 m in the downstream section and 4-8 m in the midsection) and bottom layer (from 16 m or 8 m to bottom). The upstream section was not deep enough to allow similar division. Each sampling line was divided into 10 sections, each equal in length, and fish density (fish ha-1) calculated as the mean across the sections.

To study the species composition echosurveyed, purse seining (height 5 m, length 92 m, mesh sizes 5-40 mm) and gill netting (length 40 m, height 1.8 m, mesh sizes 12-75 mm) were carried out. Test fishing was done in areas near the sampling sites and dates, using gill nets parallel to the shorelines at depths of 2-9 m and purse seining in the pelagic area at depths above 6 m. The gill nets were emptied after 7-10 h and fishing was carried out for a total of 142 gill net series fishing periods (68 at day and 74 at night). The purse seine net was used 50 times (35 at day and 15 at night). Fishing effort was kept constant between sampling areas. After fishing the lengths and weights of all fish caught were measured.

1.1 Statistical methods

We used a factorial design (repeated measures Analysis of Variance, ANOVAR, SAS®: PROC MIXED) to test for possible differences in fish densities between fixed factors: sampling areas (mid- and downstream section) and water layers (surface vs. bottom layer) during different study periods (May, August, October) and time of day (day vs. night) ($\alpha = 0.05$). If higher-order interactions are significant in a design, interpretation of the results should be based on highest-order one (e.g. Winer et al., 1991, Kirk, 1995). Therefore a posteriori test (T-test) was used to breakdown the significant interactions. Hydroacoustic data were used in the pelagic area (number of fish ha^{-1}) and data from electrofishing in the shoreline area (number of fish per 27 m^2). The effect of sampling area on fish density in the hydroacoustic data was separately analysed with two-way ANOVA, because division to surface and bottom layers was not possible in one of the sampling sections (the upstream section) due to the relative shallowness of the area. Data were log-transformed to stabilise variances.

2 RESULTS

2.1 Fish densities

The hydroacoustic data show that the overall fish densities in the pelagic area differed significantly among the three sampling areas (ANOVA, $F = 17.2$, $df = 2$, $P < 0.001$). In the upstream lotic sampling section the pelagic zone was scarce of fish (mean density ± SE, 9 ± 8 fish ha^{-1}). In every study period, fish densities were higher in the midsection (38 ± 16 fish ha-1), but highest in the downstream section of the impoundment (181 ± 72 fish ha^{-1}).

Fish densities in the mid- and downstream sections were further analysed with ANOVAR to test for differences between surface and bottom water layers. There were no overall differences in fish densities between water layers, but water layer interacted significantly with study month and time of day (Table 1, Fig. 2). A posteriori t-tests showed fish densities were significantly lower ($p<0.05$) in May in the surface water layer compared to other combinations of study month and water layer. There was also a significant interaction between time of day and water layer (Table 1, Fig. 2). According to a priori tests fish densities were always significantly lower ($p<0.05$) during the day at the surface water layer compared to other combinations of time of day and water layer whereas only once a significant difference was found in the bottom layer (bottom layer at day vs. surface layer at night).

Table 1. Summary of the results from repeated measures analysis of variance for the effects of water layer (surface layer vs. bottom layer), study month (May, August and October) and time of day (day vs. night) on the fish density (fish ha-1, log-transformed) in the pelagic area of the impoundment.

Source of variation	df	F	P
Between subject			
Water layer	1	1.72	0.1943
Within subject			
Month	2	11.14	<0.001
Time of day	1	18.11	<0.001
Month*water layer	2	3.41	0.0394
Time of day*water layer	1	4.96	0.0297
Month*time of day	2	2.54	0.0876
Month*time of day*water layer	2	3.00	0.0575

Shoreline fish densities did not differ among the three sampling areas in the electrofishing survey (Table 2). A significant interaction between time of day and sampling month was found (Fig. 3, Table 2). A posteriori tests illustrated ($p<0.05$) that fish densities were always higher at night compared to day but the nighttime fish densities in May were always smaller compared to day or nigttime densities during August and October.

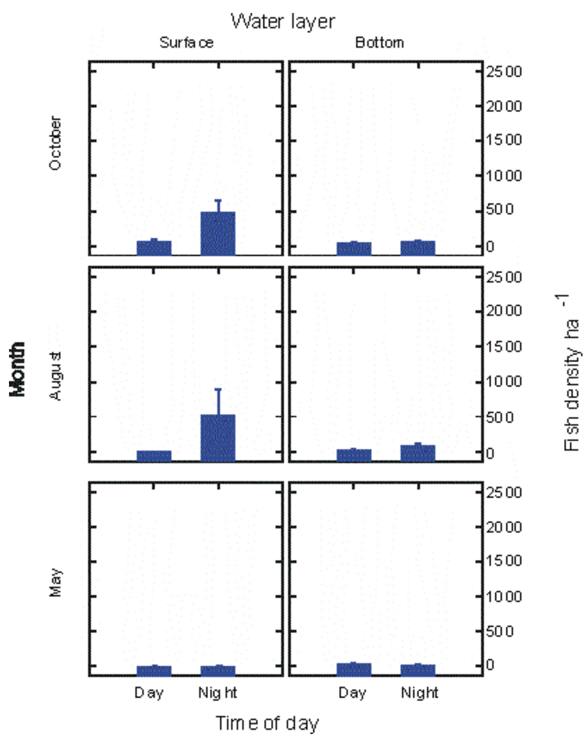

Figure 2. Fish densities (mean ± SE) from the hydroacoustic survey in the open water of the Pyhäkoski River Reservoir in surface and bottom water layers during different study months and at night and day.

Table 2. Summary of the results from repeated measures analysis of variance for the effects of sampling area (upstream, mid and downstream sections), study month (May, August and October) and time of day (day vs. night) on the electrofishing fish densities (fish per 27 m3, log-transformed) in the shoreline of the impoundment.

Source of variation	df	F	P
Between subject			
Sampling area	2	1.68	0.219
Within subject			
Month	2	3.53	0.042
Time of day	1	83.15	<0.01
Month*sampling area	4	0.48	0.749
Time of day *sampling area	2	2.98	0.081
Month*time of day	2	3.84	0.033
Sampling area*month*time of day	4	0.73	0.578

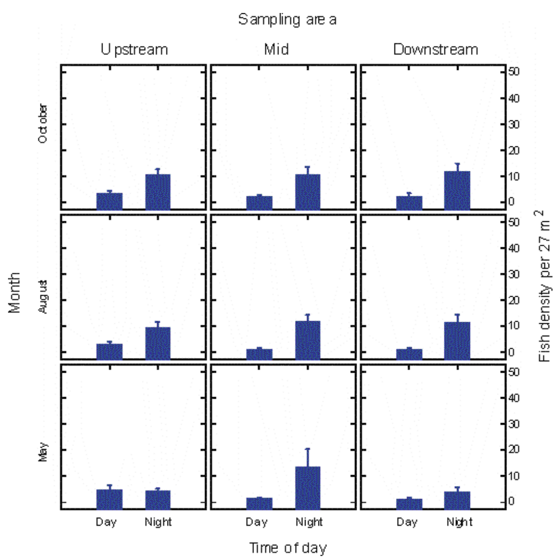

Figure 3. Fish densities (mean ± SE of the area 27 m2) from the electrofishing survey along the shore of the Pyhäkoski River Reservoir in three sampling areas during different study months and at night and day.

3.2 Fish assemblage and habitat relationships

A total of 647 fish (213, 243 and 191 from upstream, mid- and downstream sections, respectively) from ten species were caught, using electrofishing from the shoreline of the reservoir. All ten species were caught in the upstream fluvial section, eight from the mid section and six from the downstream lentic section.

These six latter species appeared in all three sections: roach (*Rutilus rutilus* (L.)), bullhead (*Cottus gobio* L.), ruffe (*Gymnocephalus cernuus* (L.)), pike (*Esox lucius* L.), perch (*Perca fluviatilis* L.), and burbot (*Lota lota* (L.)). Dace (*Leuciscus leuciscus* (L.)) and stone loach (*Barbatula barbatula* (L.)) where caught only in mid- and upstream sections and smelt (*Osmerus eperlanus* (L.)) and ten-spined stickleback (*Pungitius pungitius* (L.)) only from the upstream section. However, only one specimen each of smelt and ten-spined stickleback were caught in the uppermost electrofishing sampling area near the Pälli Power Plant (see Fig. 1). These fish originated most likely from the lake area upstream and drifted in the water current due to power plant operations.

Test fishing in the open water captured 3489 fish out of 12 species. Seven species were the same as those caught along the shoreline (perch, pike, ruffe, smelt, burbot, roach and dace), while five species were captured only from the open water: vendace (*Coregonus albula* (L.)), whitefish (*Coregonus lavaretus* s.l.), brown trout (Salmo trutta L., only one specimen), bleak (*Alburnus alburnus* (L.)) and bream (*Abramis brama* (L.)). Most of the fish (2098 specimen) were captured in the lowermost lentic section of the reservoir compared to 778 in the midsection and 445 in the lotic section, respectively.

4 DISCUSSION

The three physically distinct macrohabitats in the Pyhäkoski River Reservoir were the fast-flowing and highly disturbed (by flow changes) upstream section, the midsection with intermediate physical features (velocity, depth) and disturbance effects, and the most stable downstream section with the greatest water depth. The macrohabitats corresponded with large differences in fish assemblage and fish density in the open water. The upstream reach with extreme changes in water velocity due to flow changes had low numbers of fish in the open water. The overall fish densities in the open water increased downstream as the disturbance effects declined and water volume increased. Our results are consistent with earlier findings that systems with large, anthropogenic flow fluctuations have suppressed fish densities (Bain et al., 1988). In the highly unpredictable upstream section, the mean current velocities during high flows were occasionally high, even above the range (> 1 m s-1) suitable for many stream salmonids (e.g. Heggenes & Dokk, 2001). This together with rapid flow changes is the obvious reason for the low number of fish in this macrohabitat.

Previous studies have shown that fish assemblages within a stream system are largely determined by longitudinal changes in habitat structure (Gorman, 1988; Peterson & Rabeni, 2001) and disturbance regime (Bain et al., 1988). River reservoirs, such as the Pyhäkoski Reservoir, show longitudinal changes in water depth and current velocities over short distances, and thereby offer excellent opportunities for studies on the distribution of species along spatial habitat gradients (see Ward & Stanford, 1982). Disturbance effects of hydropeaking decreased downstream as the distance from the upstream dam increased, and channel profile deepened and widened (see also Bosco-Imbert & Stanford, 1996; Valentin et al., 1996). Current ecological theory predicts that fish assemblage recovery from artificial fluctuations of flow can be measured with the recovery gradient, especially by the presence or abundance of fish species largely restricted to flowing water habitats (fluvial specialists; Kingsolving & Bain, 1993; Stanford & Ward, 1995, Ward & Stanford 2001). In the Pyhäkoski Reservoir, the fluvial specialists (e.g. grayling, *Thymallus thymallus* L., juvenile brown trout and salmon, *Salmo salar* L.), common in unregulated rivers of similar size (Jutila, 1992; Romakkaniemi et al., 2000), were totally absent. In the case of the Pyhäkoski Reservoir, the river is so intensively regulated and used for hydropower production that recovery to anything even near natural conditions is not possible without removing some of the dams (see also Stanford & Ward, 2001). However, a gradient of changing fish assemblage, paralleled by changes in physical conditions, was found in the present study.

Fish in the Pyhäkoski Reservoir underwent distinct diurnal migrations as the number of fish increased both along the shore and in the upper water layers of the open-water area during the night. This was obviously caused by migrations related to night-time darkness, because similar increases in fish densities were not observed in May, when the sun is below the horizon for only a few hours and night-time darkness is only a short period of twilight. Kubecka & Duncan (1998) stated that fish in the Thames River appeared to move into deeper water layers during the daytime and became undetectable with echo-sounding. This was also the case in the Pyhäkoski Reservoir because we frequently observed dense scatterings, obviously of fish, very close to the bottom in the steep and deep underwater slopes, which, however, was undetectable with hydroacoustics. We suggest that fish went deeper and closer to the bottom during the day. Similarly, Rakowitz & Zweimüller (2000) observed higher fish densities at night than during day in the open-water of The Danube River, Austria. The diurnal migrations of fish in the Pyhäkoski Reservoir resembled the general pattern found in lakes (e.g. Jurvelius & Heikkinen, 1988; Lieschke & Closs, 1999).

The reasons suggested for fish moving in shore or closer to the bottom are related to the avoidance of predators or use of food resources. Jacobsen & Berg (1998) showed experimentally that predation risk increased diel variation in habitat use of perch: fish migrated from the open-water habitat at night to machrophyte shelters in the morning when predators were present. There is also evidence that small fish move inshore at night due to predators (Copp & Jurajda, 1993,1999). Fish predation can also change the diel habitat use by fish prey animals: for example, zooplankton (Daphnia) moved deeper during day or towards shallower water at night due to fish predation (Gonzalez & Tessier, 1997). These changes in diel habitat use of important prey animals are also likely to affect the fish preying on them. Given the general tendency of zooplankton to migrate upwards in the water column at night (e.g. Ringelberg, 1999), and a distinct density peak in invertebrate drift and night-time activity (Price et al., 1991; Waters, 1972; Brittain & Eikeland, 1988; Kreivi et al., 1999), we suggest that both predation threat and food availability may affect the diel habitat use by fish in the Pyhäkoski Reservoir.

Our results have also management implications. The differing conditions and related fish community composition in different parts of the reservoir needs to be considered in management actions, e.g. in fish stocking and mitigation of the physical habitat for fish. The whole Oulujoki river system is heavily regulated, consisting mainly of subsequent river reservoirs, making the results exploitable in the whole river system. On a global scale our results stress especially the importance of gradient effects of flow regulation on fish communities and adds to the awareness of disturbed rivers by providing an example of the longitudinal effects of flow regulation on fish assemblages composition and on temporal changes in fish abundance and habitat use.

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NoWPAS ~ Nordic Workshop for PhD students on Anadromous Salmonid research

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Background

During the last decade's thorough research on anadromous salmonids in different aspects has been conducted and will most likely increase in the future. In this connection, PhD and post-doctoral students with related focus play an important and central role in gaining new knowledge and by developing new methods in order to cope with the issues and the future challenging problems. In order to increase and further improve the profit obtained from the PhD study's a new network has been established. As an element of building a European network a Nordic workshop over three days with title NoWPAS (*Nordic Workshop for PhD students on Anadromous Salmonid research*) was held in Agdenes, Norway, April 2005.

Objectives

The objective with such a network is two fold: Firstly, we wish to arrange an annual independent workshop where the participants can gather, exchange knowledge and ideas and to have discussions in an interdisciplinary forum. Secondly, we wish to establish connections with the "outer world" by inviting key researchers to give lectures and short courses within the sphere. On the basis of this we mean that the utility value is potentially larger for both the participants and the community.

Today's and future PhD students are representing the recruitment of researchers within the science of anadromous salmonids. Therefore it will be very important that younger scientists establish connections with thoughts of future collaboration in an international environment. The future network has objective to both act as an informal basis and as a serial of annual workshops. It will emphasize both the opportunity for future collaboration through existing and future projects. As an overview the future workshops will have following main objectives:

Participating PhD and Post-doctoral students shall present their work and/or obtained results. In this way they will have the opportunity to get feedback on their own work and to be oriented of other people's work and findings within the sphere.

Invited external scientists within the sphere will present actual problem issues in addition to take part in discussions.

Presented materiel and outcomes will make basis for a final report which will be published and send to all participants and members of the network. In addition it will be accessible on the network's homepage.

Future work

As mentioned NoWPAS is a part of a future European network. Primary it will include PhD and Post doctoral students from European countries that do research on anadromous salmonids. Further, the network aims on providing short courses for the members in addition to the mandatory oral presentations However, in order to prevent “artificial growth” followed by increased risk of “negative selection” the network will during the two first years aim at being a Nordic network. The first annual workshop has already been held in Norway, and was considered being a success. 27 participants gave presentations whereas 4 guest lectures were invited. Based on this several members have already established contacts and developed ideas for future collaborative project. Today we are about 40 members. In order to preserve and improve the network a committee has been established and sustains of following people:

- Morten Stickler, Norway, coordinator.
- Anders Finstad, Norway
- Mikko Kiljunen, Finland
- Olle Calles, Sweden
- Lasse Fast Jensen, Denmark

Within first of May this year a homepage will be established (*www.nowpas.org*). Here, new members can gain information of the network and its members, and to apply for membership. Previous and future events of upcoming meetings and workshops will also be presented here. In addition the homepage will provide a data base of papers and articles made by the members.

The next year workshop is planned to be held in Karlstad, Sweden. The committee of NoWPAS will have a meeting during the upcoming fall to plan the event, to make arrangements and to send out invitations. We will most probably have an upper limit of 30-40 participants due the budget and to preserve the intimacy of the workshop. A short course for the members is also to be discussed. Since NoWPAS-2006 will also be a Nordic workshop only students from the Nordic countries will be invited. However, we encourage interested people to take contact in case we have available “seats”. People who are interested in participating either as a member or external participant can either go to our homepage or contact Morten Stickler (see below) directly.

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