THEME: Environment (including climate change)
TOPIC: ENV.2011.2.1.2-1 Hydromorphology and ecological objectives of WFD Collaborative project (large-scale integrating project)
Grant Agreement 282656
Duration: November 1, 2011 – October 31, 2015

# REFORM

## REstoring rivers FOR effective catchment Management

Deliverable D3.4

Title Guidance on how to identify impacts of hydromorphological degradation on riparian ecosystems

Author(s) Annette Baattrup-Pedersen[1], Christopher Andrews[2], Barbara Belletti[3], Daniela Campana[3], Peter E. Carlson[4], Dan S. Chapman[2], Jaroslaw Chormaski[5], Francesco Comiti[3], Diego García de Jalón[6], Marta González del Tánago[6], Alan Gray[2], Stephen C. Ives[2], Richard K. Johnson[4], Adam Kiczko[5], Thomas R. Kjeldsen[2], Julia Kraml[7], Cedric L.R. Laize[2], Katharina Lebiedzinski[5], Helmut Mader[7], Judit Maroto[6], Vanesa Martínez-Fernández[6], Peter Mayr[7], Brendan G. McKie[4], Tomasz Okruszko[5], Massimo Rinaldi[3], Leonard Sandin[4], Mircea Staras[8], Adam J. Vanbergen[2], Ben A. Woodcock[2] & Matthew T. O'Hare[2]

[1] AU-NERI, [2] NERC-CEH, [3] UNIFI, [4] SLU, [5] WULS, [6] UPM, [7] BOKU, [8] DDNI

Due date to deliverable: 30 April 2015
Actual submission date: 24 August 2015

Project funded by the European Commission within the 7th Framework Programme (2007 – 2013)

Dissemination Level

| | | |
|---|---|---|
| PU | Public | X |
| PP | Restricted to other programme participants (including the Commission Services) | |
| RE | Restricted to a group specified by the consortium (including the Commission Services) | |
| CO | Confidential, only for members of the consortium (including the Commission Services) | |

## Summary

- The aim of this deliverable is to address the impact of hydromorphological degradation on floodplain and riparian ecosystems, with specific focus on vegetation, fish and invertebrate responses and to provide guidance on how to identify those impacts.

- An introductory chapter summarises the research context and reviews the lessons for managers and stakeholders. Based on the results of the analyses, and the river styles typology developed in Work Package 2 of REFORM, a generic process is recommended for assessing the impact on floodplain and riparian ecosystems, incorporating our key findings. It also highlights the usefulness and limitations of existing EU Directives in providing a suitable legislative framework.

- Assessments of instream impacts on riverine ecosystems make use of multi-site datasets, riparian and floodplain ecosystems are not subject to this type of extensive monitoring; hence, the results presented here are based primarily on case studies from across Europe.

- A key finding is that impacts to hydromorphological processes and that these impacts can take years to fully manifest themselves. The results can be dramatic with changes in river style and loss of riparian forest as unpredicted outcomes of human intervention.

- We spotlight vegetation, unlike other biota, it has a very direct influence on fluvial geomorphological and hydrological processes, by stabilising sediments and influencing flood conveyance. A number of case study contributions address various forms of this key interaction. A short summary chapter is provided to link these findings to this particular theme. Use of the vegetation-process model developed in WP2 is recommended to compare the post impact role of the physical processes of vegetation with the ‘natural’ conditions. This model is used in all the vegetation case studies and helps explain why impacts can take time to become fully manifested.

- In three Italian case studies, on the Magra, Panaro and Aurino rivers, the channels narrowed and the beds encised following a variety of impacts including gravel mining and catchment scale deforestation. This altered the patterns of riparian vegetation and tree growth. The three case studies also highlight the complexity of the relationships occuring between riparian vegetation and river hydromorphology in impacted rivers and how human disturbances may become dominant in structuring such relationships. The case study examples confirm that plant diversity alone cannot be considered a valid and exhaustive indicator to assess the health of a river system and its functioning.

- For the two Spanish case studies aerial photographic data was available before and after damming of the rivers. After damming, vegetation encroachment on downstream gravel bars stabilised the banks and the channel changed from braided to single threaded. Similar effects of flow regulation have been reported on other rivers. In the River Porma, the vegetation composition and structure changed from one dominated by young pioneer species to a mature forest with a dense overstory of late-seral species near the channel banks. In the case of the River Guadalete, the flow regulation reduced the recruitment potential of native species and favoured the exotic species *Eucalyptus camaldulensis*. As a direct consequence of flow regulation, areas affected by fluvial disturbances under pre-dam conditions have turned into areas dominated by hydrologic processes with negligible sediment dynamics during inundation. Based on these results key indicators of change are proposed.

- In Austria, the River Traun is regulated via a flood protection dam that cuts off the river from its floodplain and side arms and wet areas of the riparian forest have dried up. The river is deeply encised, and this has caused a significant lowering of the groundwater table with a consequent loss of riparian forests. The River Traun study site is representative of a large number of European rivers where the typical shruby pioneer vgetation and softwood riparian forests have disappeared and been replaced by mainly hardwood riparian forests that constitute the largest part of the remaining European riparian forests.

- In Poland, the low energy river Narew is one of the few remaining anastomising river systems in northern Europe. This river type was once common and widespread but is now confined and regulated across much of its historic distribution. The Narew has been subject to flow regulation in parts of its catchment. The analysis of inundation duration for the period 1978-2009 shows that the vulnerability to changes in the flood regime, induced by damming upstream, is habitat dependent and related to the duration of flooding. In the case of wetlands sedge *Phalaris* and *Carex-Phalaris* communities, their natural inundation periods are relatively short. These communities were affected by the change in the flood frequency, while other communities were unaffected. The study demonstrated that natural (or semi-natural) lowland river valleys can be quite resistant to a single pressure, in this case flood frequency changes.

- We also provide primary research on invertebrates and fish responses to riparian degradation. In Scotland, three rivers subject to varying degrees of flow regulation were studied, two of which are Special Areas of Conservation. The response of riparian invertebrates to flood inundation on mid-channel islands was studied. Overall, the insensitivity of these riparian invertebrate assemblages to flow peak or intensity floods suggests that the community structure is resilient. High abundance of Carabidae indicates a system unaffected by floods, suggesting that the system is hydrologically impaired. More important environmental factors were the size and habitat structure of the riparian habitat. The semi-natural habitat in the surrounding landscape provided a source of colonists.

- Swedish streams in catchments with natural (forest) and degraded (agriculture) riparian zones were compared. Degradation of the riparian zones had important effects on in-stream hydromorphology (riffle and pool sections) and instream invertebrate communities (changes in species traits composition). The agricultural streams characterised by long stretches of pool habitat are less likely to support insects with traits favouring greater dispersal than forested streams with a higher abundance of riffles. This in turn affects the subsidy of energy and nutrients to the riparian zone, in the form of aquatic insects emerging as adults and dispersing into the riparian zone. This implies that the few short riffle habitats in the agricultural landscape are important for the transfer of high quality food to terrestrial/riparian consumers.

- In the Danube Delta, Romania, fish communities have been significantly affected, locally, by loss of connectivity between the main stems of the river and floodplain lakes. The lakes have high species diversity due to the co-occurrence of rheophilic, eurytopic and limnophilic forms. Analysis of long-term data on commercial fishing and the history of hydrotechnical works indicates negative changes in the catch, which correlates well with the blocking of canals to alleviate siltation and nutrient inputs. Alternative solutions should now be considered. Reliable long-term commercial fishery data on migratory anadromous and potamodromous fish species can be used to indicate and explain effects of historical changes in the lateral or longitudinal connectivity of river systems. From a management point of view, maintaining the existing connectivity gradient in the delta lakes is vital for biodiversity conservation and economic needs.

## Acknowledgements

This document has been internally reviewed by Tom Buijse (Deltares), Judy England (EA), Martina Bussettini (ISPRA), María Isabel Berga Cano (CEDEX) and Fernando Magdaleno Mas (CEDEX).

REFORM receives funding from the European Union's Seventh Programme for research, technological development and demonstration under Grant Agreement No. 282656.

# Table of Contents

# 1 Introduction

## 1.1 Overview

This deliverable was produced under Task 3.4 of Work Package 3 of the REFORM project. The work package as a whole focuses on the effects of physical degradation on river ecosystems, acknowledging that many European rivers are subject to multiple pressures. This deliverable builds on our understanding of natural processes in river systems, developed in Work Package 2, and in combination with output from Work Package 4 it will contribute to the fundamental scientific that forms the basis of tools for end users, which are developed in Work Package 6.

The original aim of this deliverable was to provide knowledge on how hydromorphological degradation affects fish recruitment, invertebrates and floodplain/riparian vegetation. This aim reflects the understanding of biotic response to degradation, implicit in the Water Framework Directive and other EU Directives, as passive and one way. Riparian vegetation has a profound influence on river geomorphology, hence its response is not passive and its degradation dramatically impacts hydromorphological processes. The aims of the deliverable have been extended to encapsulate key interactions between riparian vegetation and fluvial processes. Effects are described across spatial and temporal hydromorphological gradients and in relation to both local and catchment processes, building on case studies from several European countries. In conformity with the description of work, this deliverable reports on effects of altered lateral connectivity, between the river channel and the adjacent floodplain, on key hydrological processes and how this is reflected in the biological communities. The hydrological processes encompass flow patterns and flooding regimes in the floodplain.

Across Europe and elsewhere, stream and river ecosystems have experienced long-term hydromorphological degradation. In headwaters, the construction of dams for the production of electricity /water supply has caused most damage (Ward & Stanford 1995), whereas lowland sections have been most affected by floodplain reclamation/riparian alterations and channel re-sectioning, for instance straightening, dredging, reinforcement of banks and building of lateral dikes (Figure 1.1) (Brookes, 1987; Brookes & Long, 1990; Verdonschot & Nijboer, 2002; Mattingly et al., 1993; Landwehr & Rhoads, 2003). This deliverable addresses the impacts of both dams and riparian alterations using case studies and comparisons between carefully selected systems. The study areas are situated in Sweden, Scotland, Poland, Austria, Italy, Romania and Spain. The Romanian case study addressed changes in the fish assemblage in the Danube delta; the case study conducted in Sweden looked into effects of agricultural land use in the catchment on trait characteristics of the macroinvertebrate community in streams; the case study in Scotland looked into effects of flood disturbance on the invertebrates living in the ecotone between land and water, and the studies conducted in Poland, Austria, Italy and Spain all examined effects of altered flooding regimes and groundwater levels on vegetation in the riparian zone. In the studies on riparian vegetation, a framework similar to that developed in Work Package 2 (Deliverables 2.1 and 2.2) was used, describing alterations in the riparian vegetation in relation to an altered flooding regime and groundwater table (Figure 1.2). Here, the riparian zone extends from the river across the floodplain, including the whole area influenced by the river channel (Gregory et al. 1991, Naiman and Décamps 1997, Naiman et al., 2005). The type of channel (in terms of confinement) determines the interaction between riparian vegetation and fluvial processes. The case studies range from entirely unconfined systems to highly confined systems subject to a wide range of hydrologic conditions. Chapter 2 of this deliverable provides a comprehensive introduction to the topic.

Figure 1.1 Two contrasting Danish lowland streams. Above: River Omme with a natural meandering planform and associated back-water habitats resulting from dynamic river processes. Below: River Brede being a typical agricultural stream with a channelized planform.

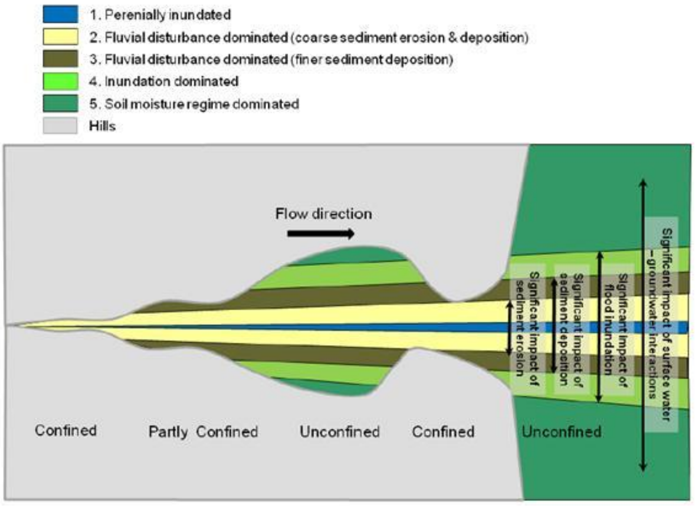


Figure 1.2 The framework used for the case studies describing effects of hydromorphological degradation on riparian vegetation.

Complex system responses

The case studies highlight what are best described as common degradation syndromes, where one or more hydromorphological pressures over time have unforeseen and complex impacts on river systems. A common impact is reduced hydrological disturbance in the riparian zone downstream dams, leading to vegetation encroachment and simplification of habitat and river style. The case studies from Spain highlight this impact.

Another relatively common impact, seen in both the Italian and Austrian case studies, is the inability of a river to dissipate its energy through normal pathways due to sediment starvation or confinement; the river cuts down into its bed instead. Thus, the river becomes deeply incised, causing local drops in the water table, which in turn impacts the riparian and floodplain vegetation.

The case studies on invertebrates show the possible influence of the wider catchment landscape on riparian and floodplain biota. The observed response by fish to reduced connectivity between the main river stem and floodplain lakes illustrates a complex response at fish community level, some species being favoured while others were not.

In the case studies a range of data sources was used. This hybrid approach, combining the use of historical data, remote sensing, field surveys and modelling, is especially powerful for identifying the impacts of hydromorphological degradation on riparian ecosystems.

## Implications for Managers & Stakeholders

### *Guidance for assessing impact*

Based on the successful application of the techniques used in the case studies we propose a generic approach to assessment (Figure 1.3).

Many stakeholders and managers will be familiar with Water Framework Directive assessments. The approach here differs somewhat by having a stronger emphasis on understanding the trajectory of change at a site. Data on the pre-impact state including the hydromorphological condition is often available from historic sources. Inclusion of historical evidence from maps and other sources, provides a clearer picture of how a river has changed over time. This is especially important for river systems as many river styles are dynamic and their natural state is variable, and historical analysis may help elucidate their development.

Many of the methods developed to assess hydromorphology for the Water Framework Directive do not collect data in a manner that allow straightforward interpretation of which hydromorphological processes are impacted by alterations. The methods have been developed as a supporting element describing the habitats of instream biota in order to characterise the riparian zones and their hydromorphology. We recommend simple modifications of some of these methods to improve their application to assessing the impacts on fluvial geomorphological processes. Deliverable D1.1 of REFORM reviews eco-hydromorphological methods and provides recommendations regarding their potential application.

The generic approach (Figure 1.3) advocates the use of modelling techniques to predict future trajectories. Detailed modelling approaches have improved in recent years, and many physical process models now include a vegetation component. Part 1 of Deliverable 2.2 of REFORM addresses the influence of natural hydromorphological dynamics on biota and ecosystem functioning and gives a comprehensive review of relevant modelling techniques.

The studies presented in this deliverable show that all the investigated biota responded strongly to floodplain and hydromorphological degradation. Vegetation, fish and invertebrates may therefore be successfully used as indicators of system degradation in combination with other indicators. As previously stated, it is important to collect the data in a manner which allows the establishment of links between pressures, hydromorphological processes and biotic responses. This is an active area of research as the case studies here illustrate. To develop our understanding to the point where recommendations could be made regarding generic assessment procedures for biota, a multi-site study of impacted systems, structured by river style and impact type is required.

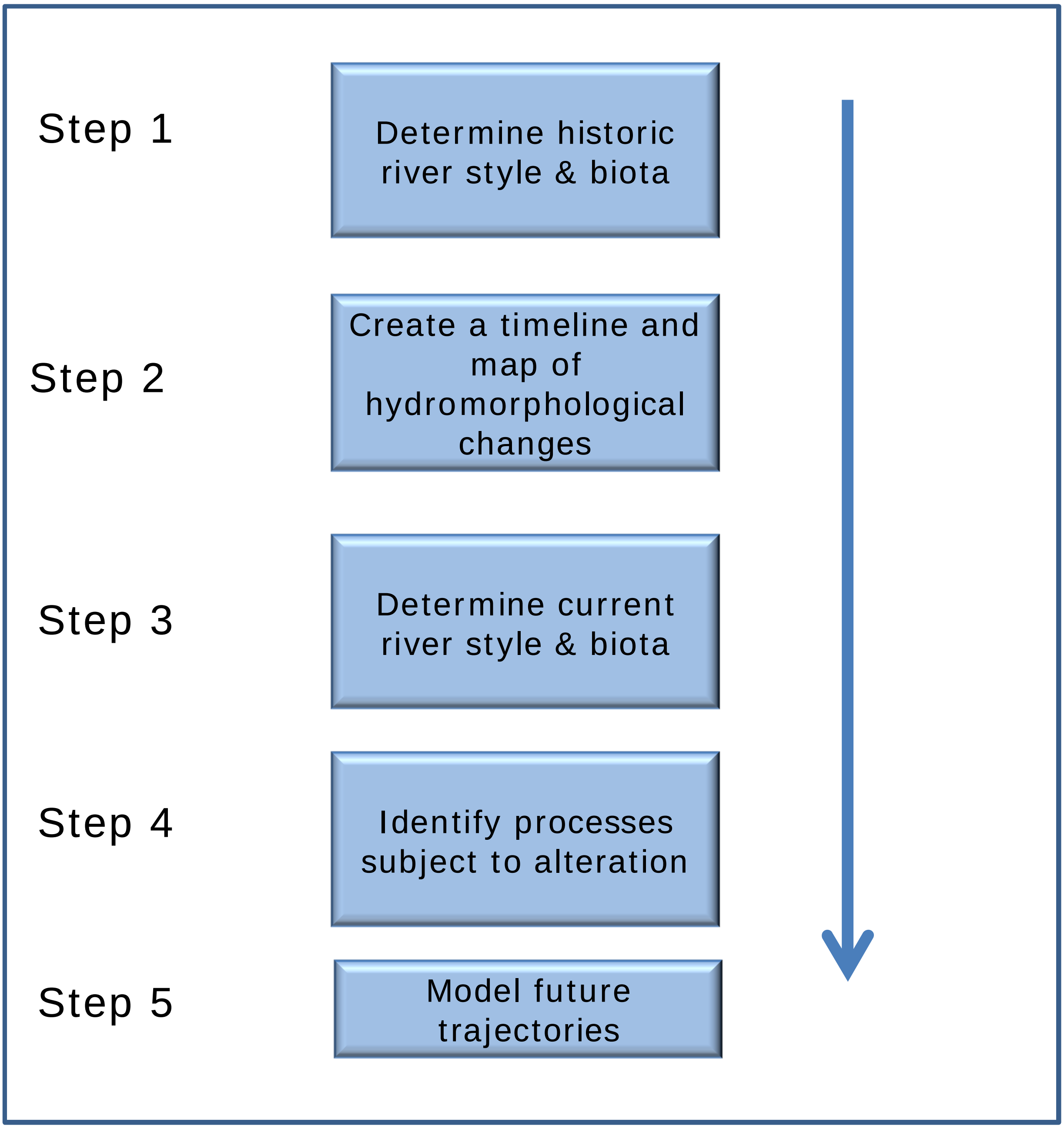


Figure 1.3 A diagram illustrating the steps in a generic approach to assessing the impact on floodplain and riparian systems. This approach can be considered a more refined version of the Forecaster approach.

*Reference conditions for riparian zones*

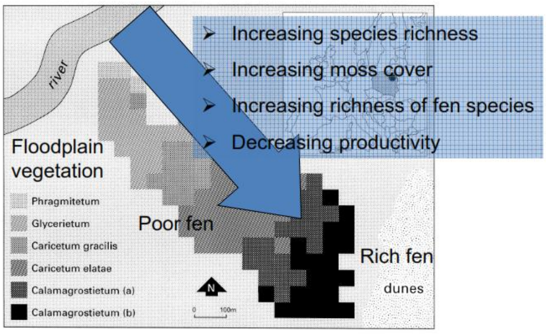
The case studies presented in this deliverable provides sufficient historical information to determine the environmental state of the systems before effects of major pressures occurred. The clearest examples are from Spain where the state of a river channel before and after establishment of a dam could be determined by using aerial photographs. WP2 identified the natural conditions in terms of processes for rivers but did not set reference conditions for the associated biota. This raises significant questions as the reference states of key elements of the riparian biota, playing strong ecosystem engineering roles, are not well defined. The issue is compounded by the low occurrence of systems in a natural state in Europe, (Figure 1.4). Studies of the few remaining systems in a near natural condition can provide useful data that should be considered when setting reference conditions. For example in the Biebrza valley, zonation of the riparian vegetation still exists, with a clear turnover in plant communities from floodplain vegetation with reed beds near the stream channel, over poor fen to rich fen communities at some distance from the channel. At the same time, there is an increase in species richness, moss cover and richness of protected fen species and a declining productivity when moving away from the stream.

Besides the few examples existing on the vegetation characterising undisturbed riparian areas, pollen and plant fossil records provide information on undisturbed conditions. In contrast to common opinion, multiple river floodplains in north-western Europe formerly exhibited widespread open vegetation in the temperate phase of the last interglacial (Svenning 2002). This is reflected by high percentages of non-arboreal pollen (NAP; often >40%), and both pollen and plant macrofossils show abundant occurrence of herbs in dry grasslands, wet meadows or open marshes, and disturbed ground as well as shrubs and small trees (mainly *Cornus sanguinea*, *Corylus avellana*, *Crataegus monogyna*, *Prunus spinosa*, *Acer monsspessulanum*, *Rubus fruticosus* and *Sambucus nigra*). Many of these habitat types likely have strong resemblance with those occurring in undisturbed areas today. In other floodplain sites, pollen and plant macrofossils indicate more tree-rich conditions, sometimes even closed forest, throughout or during part of the investigated period. Nevertheless, even in these cases there is still a clear evidence of open habitats (Svenning 2002).

While the river dynamics and high groundwater levels contributed to the existence of open vegetation in these periods, large herbivores could also have a decisive influence on the maintenance and extension of open vegetation, as the presence of water and herb-rich vegetation may have led many large herbivores to concentrate their foraging activities in the floodplains (see references in Svenning 2002).

Pre-agricultural Holocene pollen records from British floodplains suggest more forested conditions then than in earlier interglacials. NAP percentages are 10–20%, with grassland indicator species being very scarce (Waller 1993 1994). The fact that floodplains were more tree-dominated during the early Holocene than in the Pleistocene interglacials could be related to changes in the extensive herbivore fauna, notably the lack of large-sized herbivores (cf. Owen-Smith 1987; van Kolfschoten 1999) and/or perhaps lower densities of the surviving herbivores due to an increased human hunting pressure.

In Europe, hydromorphological degradation accelerated and intensified in the 1800s, reflecting processes initiated much earlier. The extensive drainage of wetland areas dates back to the middle ages. Introduction of ploughing tools in the middle Ages changed the sediment load to rivers, resulting in altered river forms across Europe. Over time, entire landscapes have been changed.



From: Wassen MJ; Peeters WHM; Venterink HO
PLANT ECOLOGY 165: 27-43

Figure 1.4 Very limited evidence exists on the vegetation characterising undisturbed river systems, reflecting that only few areas are in a natural or near-natural state. The photographs show the Ob River in Siberia and the Biebrza River in Poland that are both considered to be in a near-natural state. The gradient in biotic changes across the floodplain is show for The Biebrza River.

*Regulatory context*

Hydromorphological changes have been identified as the most widespread pressure on the ecological status of EU waters. Investigations into the status of the associated floodplains reveal a potentially even worse situation here; however, associated floodplains are not encompassed by river basin management plans even though they are encompassed by international legal regulations such as the Convention on Biological Diversity, the EU Water Framework Directive (WFD; Council Directive 2000/60/EC) and the EU Habitats Directive (HD; Council Directive 92/43/EC) that all outline targets for preservation, protection and improvement of the quality of the environment, including conservation of natural habitats and wild fauna and flora. The HD provides a useful framework for protection of riparian and floodplain zone, and, in fact, the most commonly cited habitat in the designation of Special Areas of Conservation is habitat 91E0, alluvial forests, occurring in 21% of all SAC citations. However, habitat 91E0 is only applied to systems of high conservation value and cannot be readily used to protect and upgrade degraded rivers. Thus, there is a gap in the EU legislation which needs to be addressed. Conceptually, an extension of the WFD to incorporate fully riparian biota and processes seems to be the most proper solution. By including riparian biota as one of the Biological Quality Elements (BQEs) *sensu* WFD, would remedy this legislative gap concerning the ongoing serious riparian degradation. Without change there is no legislative driver to overcome situations where riparian zones require remediation. Furthermore, without evidence linking riparian degradation to degraded instream BQE status, it may be difficult to imply measures targeting riparian degradation. The limited evidence that does exist reflects a patchy research effort rather than an absence of causal linkages. There is, good evidence available for fish, and some evidence for benthic invertebrates as can be seen in the studies presented here from Scotland, Romania and Sweden linking riparian degradation with degraded instream BQE status.

## 1.2 References

Brookes A. (1987) The distribution and management of channelized streams in Denmark. Regulated River, 1, 3–16.

Brookes A. & Long H.J. (1990) Chart Catchment Morphological Survey: Appraisal Report and Watercourse Summaries. National Rivers Authority Reading, UK.

Gregory ,S.V., Swanson , F. J., McKee A. W. &. Cummins, K. W. (1991) An Ecosystem Perspective of Riparian Zones: Focus on links between land and water. BioScience, 41, 540-551

Landwehr K. & Rhoads B.L. (2003) Depositional response of a headwater stream to channelization East Central Illinois USA. River Research and Application, 19, 77–100.

Mattingly R.L., Herricks E.E. & Johnston D.M. (1993) Channelization and levee construction in Illinois: review and implications for management. Environmental Management, 17, 781–795.

Naiman, R.J. & Decamps, H. (1997) The ecology of interfaces: riparian zones. Annual review of Ecology and Systematics, 28, 621-658.

Naiman, R. J. (2005). "Origins, Patterns, and Importance of Heterogeneity in Riparian

Systems" in Ecosystem Function in Heterogeneous Landscapes (0-387-24089-6, 978-0-387-24089-3), (p. 279). New York, NY: SPRINGER.

Owen-Smith, N., 1987. Pleistocene extinctions: the pivotal role of megaherbivores. Palaeobiology 13, 351–362

Svenning, J.-C. 2002. A review of natural vegetation openness in north-western Europe. Biological Conservation. 104, 133–48

van Kolfschoten, T., 1999. Pleistocene herbivores and their environment. In: Gerken, B., Gorner, M. (Eds.), Europaische Land-schaftsentwicklung mit großen Weidetieren-Geschichte, Modelle und Perspektiven. Natur- und Kulturlandschaft 3. Huxaria Druckund Verlag, Hoxter, Germany, pp. 138–145

Verdonschot P.F.M. & Nijboer R.C. (2002) Towards a decision support system for stream restoration in the Netherlands: an overview of restoration projects and future needs. Hydrobiologia, 478, 131–148.

Waller, M.P., 1993. Flandrian vegetation history of south-eastern England. Pollen data from Pannel Bridge, East Sussex. New Phytologist 124, 345–369.

Waller, M.P., 1994. Flandrian vegetational history of south-eastern England. Stratigraphy of the Brede valley and pollen data from Brede Bridge. New Phytologist 126, 369–392

Ward, J. V. and Stanford, J. A. (1995), Ecological connectivity in alluvial river ecosystems and its disruption by flow regulation. Regul. Rivers: Res. Mgmt., 11: 105–119. doi: 10.1002/rrr.3450110109

# 2 Riparian Vegetation Hydromorphology Interactions - Synopsis

Recently, the understanding of the relation between fluvial geomorphology and riparian plant ecology has progressed strongly (e.g. Hupp and Osterkamp 1985; Gurnell and Gregory 1995; Hupp and Osterkamp 1996; Bendix and Hupp 2000; Steiger et al. 2005; Gurnell et al. 2012; Gurnell 2014).

Based on existing literature, this chapter gives a brief summary of the relationship between riparian vegetation and fluvial processes in both near-natural and in human-impacted rivers. The intention is not to provide an exhaustive review but a basic background for the coming chapters reviewing these relations in impacted systems. More detailed information about the interaction between riparian vegetation and hydromorphology in near-natural river systems is provided by Gurnell et al. (2014a). Concerning river processes and functioning in near-natural as well as in impacted river systems, please see Gurnell et al. (2014b) and Rinaldi et al. (2015).

In the following chapters (6 to 9), the relationships between channel processes, changes and vegetation patterns will be reviewed at European scale and illustrated with examples from Italy, Spain, Austria and Polish rivers.

## 2.1 Interactions in near-natural rivers

The factors that influence the relationship between fluvial hydromorphology and vegetation include: (i) flow regime (directly responsible for the formation and turnover of channel features), (ii) channel geometry and its hydraulic properties, (iii) bank stability and (iv) variability in sediment characters (deposition, erosion rates, size) (Hupp and Bornette 2003). All of these factors interact, are interdependent and mutually contribute to determine the hydromorphology-vegetation patterns that occur in rivers.

Water is considered the most important factor in structuring fluvial patterns and features and thus also the vegetation patterns within a reach (e.g. Corenblit et al. 2007; Hupp and Osterkamp 2013). Water is meant both in terms of surface flow regime but also in terms of moisture availability related to groundwater depth (e.g. Osterkamp and Hupp 2010; Hupp and Osterkamp, 2013).

In particular, flow and flood regimes create and maintain specific geomorphic units that can support different plant communities. Several studies in different countries have shown that characteristic plant-species distributional patterns exist and are related to hydromorphological conditions in terms of type of geomorphic units, relative elevation and relative susceptibility to disturbance across the transversal gradient (e.g. Corenblit et al. 2007; Hupp and Bornette 2013; Osterkamp and Hupp 2010; Camporeale et al. 2013; Merritt 2013).

Additionally, different river types (i.e. sinuous, meandering, braided etc.) are characterised by different fluvial geomorphic units. Thus, vegetation types and patterns may also differ according to river type and channel dynamics. Beechie et al. (2006) found that the vegetation pattern, and in particular the turnover of vegetated patches, changes from single-thread straight, through meandering and island-braided to bar-braided planforms. Some river patterns (e.g. meandering) are characterised by active lateral dynamics that depend on flow and flood regime (catchment-based controls) but also on endogenous and local mechanisms (e.g. sediment type and cohesion).

Lateral dynamics and mainly bank erosion are amongst the most important fluvial processes affecting the riparian vegetation: they promote vegetation succession and large wood recruitment, create new habitats for vegetation and contribute to the building of in-channel and floodplain features (Rinaldi et al. 2013). An example of lateral channel dynamics is the episodic process of lateral accretion that occurs in temperate meandering river systems, starting during high flow events and continuing during low flows. This process of lateral accretion may allow for the development of ridges (crests) and swales (depressions) that, once integrated into the floodplain, over time are able to support different plant communities, by maintaining drier and more humid conditions, respectively (Hupp and Osterkamp 2013).

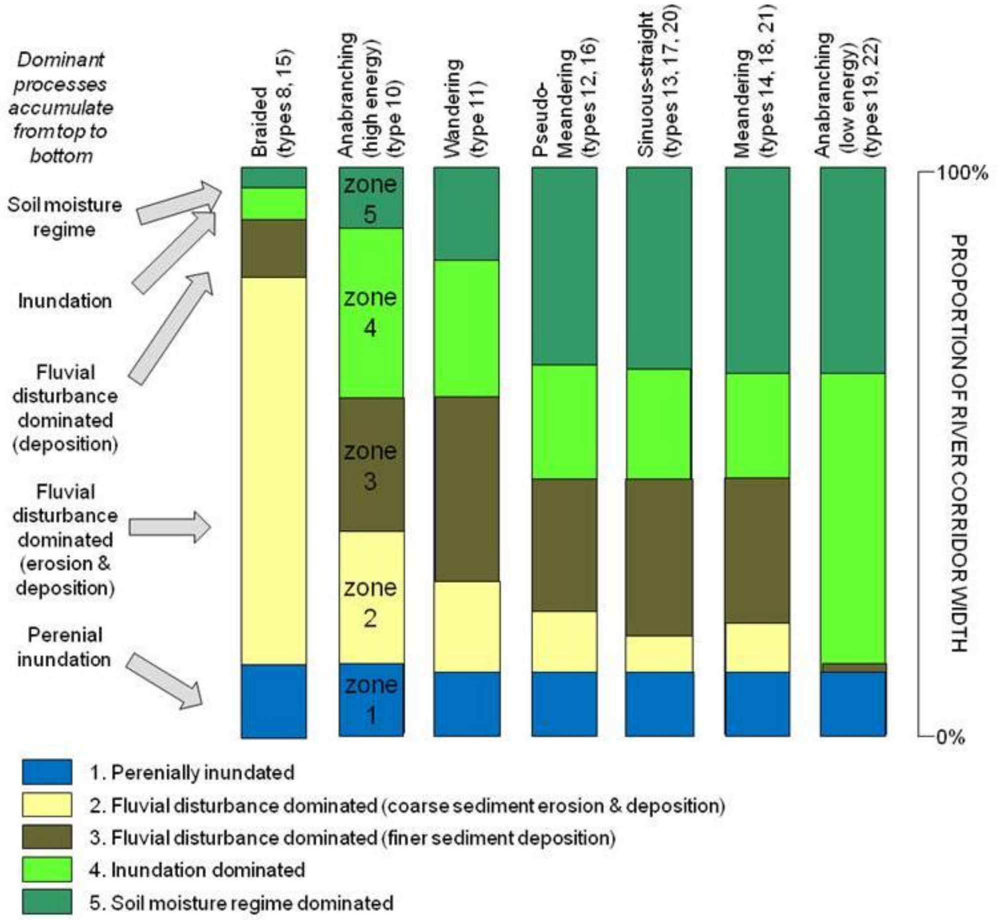


Figure 2.1 The relative proportions of the five river corridor lateral zones where vegetation might be dominated by the different hydrological and fluvial processes in unconfined reaches subject to different river types (river type numbers refer to the types defined in D2.1, section 7). Modified from Gurnell et al. 2014a (D2.2, Part 1).

Thus, vegetation, geomorphic features (or landforms) and hydrologic conditions interact with each other to determine vegetation and geomorphic patterns and processes (e.g. Corenblit et al. 2007; Hupp and Osterkamp 2013; Gurnell et al. 2014a; Gurnell 2014). In particular the distribution of plants across the river (i.e. from one side to the other) depends on the tolerance of each species to specific geomorphic processes and disturbance regime and on competition (Hupp and Osterkamp 2013).

In near-natural unconfined river systems, the interaction between vegetation and fluvial processes changes across the river corridor (laterally) (Gurnell et al. 2014a). In particular, the dominance of fluvial processes on vegetation decreases with increasing distance from and elevation above the low flow channel. The location and extent of the interaction zones differ amongst different river types (Figure 2.1).

## 2.2 Interactions in impacted rivers

The concept of evolutionary trajectory (e.g. Brierley et al. 2008; Dufour and Piégay 2009) emphasises the fact that a river is a complex system that continuously adjusts its morphology through time to changes in boundary conditions, such as flows and sediment flux variations. Adjustment processes can be categorised as follows: (a) bed level and bed slope changes; (b) changes in channel width; (c) lateral changes (e.g. channel migration and avulsion); (d) changes in channel patterns. These processes occur at a sufficiently wide spatial scale (from reach scale to the entire alluvial system) and within a significantly long temporal scale (at least 10–15 years).

The factors that influence the trajectory of channel change are various (Dufour and Piégay 2009). They can be naturally or human-induced and may act at both catchment and reach scale. In terms of temporal scale, they can be progressive (e.g. climate or land use change), impulsive (e.g. floods) or discontinuous, where the latter can be a transient (e.g. sediment mining) or a permanent disturbance (e.g. dam, bank protection).

In Europe, rivers have experienced a long history of human impacts and modifications. These mainly include changes in catchment and floodplain land use, mainly deforestation, because of pastoral and agricultural activities since the Neolitic; in-channel modifications such as channelisation, embankment, flow regulation, starting during the Roman Empire; the more recent sediment mining (Rinaldi et al. 2013). These disturbances have induced complex and multiple phases of channel changes (i.e. adjustments) as documented by several studies across Europe including France (Liébault and Piégay 2001, 2002; Liébault et al. 2013; Belletti et al. 2015a, 2015b), Poland (Wyżga 1993, 2001a, 2001b, 2008), Italy (Rinaldi and Simon 1998; Rinaldi 2003; Surian and Rinaldi 2003; Surian et al. 2009), Spain (Garcia-Ruiz et al. 1997; Rovira et al. 2005) and the United Kingdom (e.g. Winterbottom 2000).

These changes also have a number of detrimental environmental, ecological, and societal effects (Bravard et al. 1999).

Concerning riparian vegetation, the effects of human activities have manifested themselves both at reach and larger scales. The impact on vegetation has been direct, through local clearance, large-scale reforestation, burning, grazing and its abandonment and land use changes in general at both local and catchment scale. But it also includes all the indirect impacts that affect channel dynamics, such as activities of flood protection and flow regulation (e.g. dams in Spain, Lobera et al. 2015; hydropower plants in the Alpine regions), channel fixation (e.g. embankments, channelisation; e.g. for navigation and timber activities in northern and central Europe), torrential activity control (e.g. in France; Liébault and Piégay 2002) and sediment mining (e.g. in Italy; Surian and Rinaldi

2003).

The indirect impacts related to channel changes have had a strong impact on riparian vegetation. For example, in Europe bed incision has often caused disconnection of the riparian surfaces with the channel and the groundwater table, leading to drier riparian zones (e.g. Rinaldi et al. 2013). In North America and Australia, rivers were the first impacted landscape after the European settlement. Here, the combination of vegetation clearance and large wood removal induced a reduction in flow resistance which resulted in river bed vertical instability and consequent rapid and deep channel incision (Brierley and Fryirs 2005). Garofano-Gomez et al. (2012) demonstrated changes in riparian cover, structure and composition (i.e. increase in later successional species and increase in exotic species) following flow regulation in a Mediterranean river. They attributed encroachment of dense woody vegetation (and thus active channel narrowing) to the persistent low-flow conditions and low flow variability.

The intensity of human impacts on riparian vegetation has changed over time, in Europe varying with the succession of different populations and societal needs. The 19th century was one of the most severe impacting periods on the riparian environment because of the combination of direct impacts, such as grazing and agriculture activities, and climatic conditions, maximising river activity and sediment production (i.e. the Little Ice Age). During the 19th century the riparian areas were dominated by open areas with herbs and shrubby vegetation (Rinaldi et al. 2013). In the 20th century the trend has been reversed following reforestation of catchments starting from the uplands, mainly because of mountain and grazing abandonment, causing lowering of sediment load, consequent channel incision and channel narrowing stressed by vegetation encroachment. In some regions (e.g. the Mediterranean zone) the vegetation encroachment was mainly caused by flow regulation for irrigation purposes, as consequence of the increase in agricultural activities in the floodplains. The alteration of the flow regime occurred in terms of increased flows during dry season and decreased flows during wet seasons (Hooke 2006).

In summary, disturbances affecting flow and sediment transport regimes (dams, climate changes, diversions, gravel mining, weirs, banks reinforcement) can promote riverbed degradation and channel narrowing and, thus, vegetation encroachment on abandoned areas.

As a consequence, the interaction between vegetation and fluvial processes may also differ from the conceptual model for near-natural river systems (Figure 2.2). In the case of incised and narrowed rivers, for example as consequence of the alteration of the sediment regime, the width of the fluvial disturbance-dominated zones can be narrower, favouring soil moisture regime-dominated zones, or nearly absent (i.e. strongly incised rivers), causing complete disconnection of the floodplain with its channel (Figure 2.2). Indeed, narrowed and incised channels have in general lost part of their natural lateral connectivity, thus floodplains and also in-channel vegetation landforms are less frequently concerned by fluvial processes. In some extreme cases the channel is so incised that the former floodplain becomes a terrace and the water table level decreases, resulting in changed moisture conditions and potential establishment of unexpected vegetation communities.

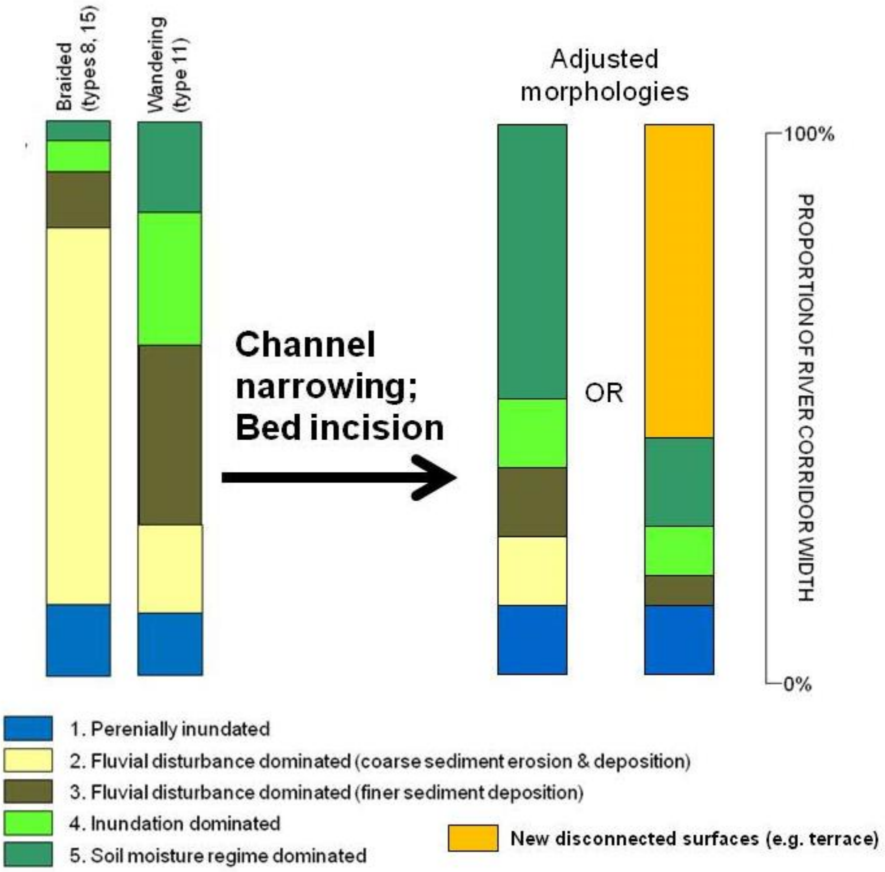


Figure 2.2 Change in the relative proportions of the five river corridor lateral zones in impacted river systems at unconfined reaches subject to braided and wandering river types (river type numbers refer to the types defined in D2.1, section 7). Modified from Gurnell et al. 2014a (D2.2, Part 1).

In case of highly regulated flow regime (e.g. because of dams and reservoirs for agricultural uses), the reduction of flood frequency combined with the increase of magnitude of low flows emphasize the channel narrowing by vegetation encroachment in favour of the soil moisture regime dominated zone (Figure 2.2; see, for instance, Chapter 7).

A strong detrimental effect on the interaction between vegetation and fluvial processes also occur as consequence of river channel fixation (i.e. for flood protection, hydropower plant, navigation). In that case strong channel pattern adjustments can occur in terms of narrowing and consequent bed incision (related to the increased flow energy and transport capacity in the narrowed, straightened and fixed channel). These lead the groundwater lowering with several consequences on the pattern and structure of riparian vegetation (see, for instance, Chapter 8).

The effects of regulated flow regime (e.g. reservoir) combined with land reclamation for agricultural uses may also have consequences on the riparian vegetation of low energy rivers (e.g. central Europe), where the riparian zone is naturally dominated by wetlands and associated riparian habitats (see, for instance, Chapter 9).

## 2.3 References

Beechie, T. J., M. Liermann, Pollock, M. M., Baker, S. and J. Davies (2006). "Channel pattern and river-floodplain dynamics in forested mountain river systems." Geomorphology 78: 124-141.

Belletti, B., Dufour, S., and H. Piégay (2015a). "What is the relative effect of space and time to explain the braided river width and island patterns at a regional scale?" River Research and Applications 31: 1-15. doi:10.1002/rra.2714.

Belletti, B., Rinaldi, M., Gurnell, A. M., Buijse, A. D. and E. Mosselman (2015b). "A review of assessment methods for river hydromorphological." Environmental Earth Sciences 73(5): 2079-2100. DOI 10.1007/s12665-014-3558-1.

Bendix, J. and C. R. Hupp (2000). "Hydrological and geomorphological impacts on riparian plant communities." Hydrological Processes 14: 2977-2990.

Bravard, J. P., Kondolf. G. M. and H. Piégay (1999). Environmental and societal effects of channel incision and remedial strategies. Incised Rivers Channels: Processes, Forms, Engineering, and Management. S. E. Darby, S. E. and A. Simon. Wiley, Chichester: 303-341.

Brierley, G. J. and K. A. Fryirs (2005). Geomorphology and River Management: Applications of the River Styles Framework. Oxford, UK, Blackwell Publishing.

Brierley, G. J., Fryirs, K. A., Boulton, A., and C. Cullum (2008). Working with change: the importance of evolutionary perspectives in framing the trajectory of river adjustment. River Futures: An Integrative Scientific Approach to River Repair. G. Brierley and K. A. Fryirs. Society for Ecological Restoration International, Island Press, Washington, DC, USA: 65–84.

Brookes A. (1987). "The distribution and management of channelized streams in Denmark." Regulated Rivers 1: 3-16

Brookes A. & Long H.J. (1990) "Chart Cathment Morphological Survey: Appraisal Report and Watercourse Summaries. National Rivers Authority Reading UK."

Camporeale, C., Perucca, E., Ridolfi, L. and A. M. Gurnell (2013). "Modelling the interactions between river morphodynamics and riparian vegetation." Review of Geophysics 51 DOI:10.1002/rog.20014.

Corenblit, D., Tabacchi, E., Steiger, J. and A. M. Gurnell (2007). "Reciprocal interactions and adjustments between fluvial landforms and vegetation dynamics in river corridors: a review of complementary approaches." Earth Science Reviews 84: 56–86.

Dufour, S. and H. Piégay (2009). "From the myth of a lost paradise to targeted river restoration: forget natural references and focus on human benefits." River Research and Applications 25: 568-581.

Garcia-Ruiz, J. M., White, S. M., Lasanta, T., Marti, C., Gonzalez, C., Errea, M. P., Valero, B. and L. Ortigosa (1997). Assessing the effects of land-use changes on sediment yield and channel dynamics in the central Spanish Pyrenees. Human Impact on Erosion and

Sedimentation. D. E. Walling and J. L. Prost. Proceedings of Rabat Symposium S6. IAHS Press, Publication no. 245, Institute of Hydrology, Wallingford: 151–158.

Garófano-Gómez, V., Martínez-Capel, F., Bertoldi, W., Gurnell, A. M., Estornell, J. and F. Segura-Beltrán (2102). "Six decades of changes in the riparian corridor of a Mediterranean river: a synthetic analysis based on historical data sources." Ecohydrology. DOI: 10.1002/eco.1330.

Gregory, S.V., Swanson, F. J. Mckee, W. A. and Cummins, K. W. (1991). "An ecosystem perspective of riparian zones." BioScience 41: 540-551.

Gurnell, A. M. and K. J. Gregory (1995) "Interactions between semi-natural vegetation and hydrogeomorphic processes." Geomorphology 13: 49-69.

Gurnell, A. M., Bertoldi, W. and D. Corenblit (2012). "Changing river channels: The roles of hydrological processes, plants and pioneer fluvial landforms in humid temperate, mixed load, gravel bed rivers." Earth-Science Reviews 111: 129-141.

Gurnell A. M. (2014). "Plants as river system engineers." Earth Surface Processes and Landforms 39(1): 4-25.

Gurnell, A. M., González Del Tánago, M., O'Hare M. T., Van Oorschot, M., Belletti, B, García De Jalón, D., Grabowski, R., Hendricks, D., Mountford, O., Rinaldi, M., Solari, L., Szewczyk, M. and A. Vargas-Luna (2014a). Influence of Natural Hydromorphological Dynamics on Biota and Ecosystem Function. Deliverable 2.2, Part 1, of REFORM (REstoring rivers FOR effective catchment Management), a Collaborative project (large-scale integrating project) funded by the European Commission within the 7th Framework Programme under Grant Agreement 282656.

Gurnell, A. M., Belletti, B., Bizzi, S. et al (2014b). A hierarchical multi-scale framework and indicators of hydromorphological processes and forms. Deliverable 2.1, a report in four parts of REFORM (REstoring rivers FOR effective catchment Management), a Collaborative project (large-scale integrating project) funded by the European Commission within the 7th Framework Programme under Grant Agreement 282656.

Hooke, J. M. (2006) Human impacts on fluvial systems in the Mediterranean region. Geomorphology 79: 311-335.

Hupp, C. R. and W. R. Osterkamp (1985). "Bottomland vegetation distribution along Passage Creek, Virginia, in relation to fluvial landforms." Ecology 66: 670-681.

Hupp, C. R. and W. R. Osterkamp (1996). "Riparian vegetation and fluvial geomorphic processes." Geomorphology 14: 277-295.

Hupp, C. R. and G. Bornette (2003). Vegetation as a Tool in the Interpretation of Fluvial Geomorphic Processes and Landforms in Humid Temperate Areas. Tools in fluvial geomorphology. G. M. Kondolf and H. Piégay. John Wiley and Sons, Chichester: Chapter 10.

Hupp, C. R. and W. R. Osterkamp (2013). Vegetation Ecogeomorphology, Dynamic Equilibrium, and Disturbance. Treatise on Geomorphology. J. Shroder (Editor in chief), D. R. Butler and C. R. Hupp. Academic Press, San Diego, CA. 12: 94-106.

Landwehr K. & Rhoads B.L. (2003). "Depositional response of a headwater stream to channelization East Central Illinois USA." River Research and Application 19: 77-100.

Liébault, F. and H. Piégay (2001). "Assessment of channel changes due to long term bedload supply decrease, Roubion River, France." Geomorphology 36(3–4): 167–186.

Liébault, F. and H. Piégay (2002). "Causes of 20th century channel narrowing in mountain and piedmont rivers of southeastern France." Earth Surface Processes and Landforms 27: 425-444.

Liébault, F., Lallias-Tacon, S., Cassel, M. and N. Talaska (2013). "Long profile responses of alpine braided rivers in SE France." River Research and Applications 29(10): 1253-1266. DOI: 10.1002/rra.2615.

Lobera, G., P. Besné, D. Vericat, J. A. López-Tarazón, A. Tena, I. Aristi and R. J. Batalla (2015). Geomorphic status of regulated rivers in the Iberian Peninsula. Science of the Total Environment 508: 101-114.

Mattingly R.L., Herricks E.E. & Johnston D.M. (1993). "Channelization and levee construction in Illinois: review and implications for management." Environmental Management 17: 781-795.

Merritt, D.M. (2013). Reciprocal Relations between Riparian Vegetation, Fluvial Landforms, and Channel Processes. Treatise on Geomorphology. J. Shroder (Editor in chief), D. R. Butler and C. R. Hupp. Academic Press, San Diego, CA. 9: 219-243.

Naiman, R. J., Decamps H (1997). "The ecology of interfaces: riparian zones." Annual review of Ecology and Systematics 28: 621-658.

Naiman, R. J., Decamps, H., McClain, M. E. (2005). "The role of riparian corridors in maintaining regional biodiversity" Ecological Applications 3: 209-212.

Osterkamp, W.R. and C. Hupp (2010). "Fluvial processes and vegetation - Glimpses of the past, the present, and perhaps the future." Geomorphology 116: 274-285.

Owen-Smith, N. (1987). "Pleistocene extinctions: the pivotal role of megaherbivores. " Palaeobiology 13: 351–362

Rinaldi, M. and A. Simon (1998). "Bed-level adjustments in the Arno River, central Italy." Geomorphology 22: 57-71.

Rinaldi, M. (2003). "Recent channel adjustments in alluvial rivers of Tuscany, central Italy." Earth Surface Processes and Landforms 28(6): 587-608.

Rinaldi, M., Wyżga, B., Gurnell, A. M., Dufour, S. and W. Bertoldi (2013). River processes and implications for fluvial ecogeomorphology: a European perspective. Treatise on Geomorphology. J. Shroder (Editor in chief), D. R. Butler and C. R. Hupp. Academic Press, San Diego, CA. 12: 37-52.

Rinaldi, M. et al. (2015). Final report on methods, models, tools to assess the hydromorphology of rivers. Deliverable D6.2 of REFORM (REstoring rivers FOR effective catchment Management), a Collaborative project (large-scale integrating project) funded by the European Commission within the 7th Framework Programme under Grant Agreement 282656.

Rovira, A., Batalla, R. J. and M. Sala (2005). "Response of a river sediment budget after historical gravel mining (the lower Tordera, NE Spain)." River Research and Applications 21: 829-847.

Steiger, J., Tabacchi, E,. Dufour, S., Corenblit, D. and J. L. Peiry (2005). "Hydrogeomorphic processes affecting riparian habitat within alluvial channel-floodplain river systems: a review for the temperate zone." River Research and Applications 21: 719-737.

Surian, N. and M. Rinaldi (2003). "Morphological response to river engineering and management in alluvial channels in Italy." Geomorphology 50(4): 307-326.

Surian, N., Rinaldi, M., Pellegrini, L., Audisio, C., Maraga, F., Teruggi, L. B., Turitto, O. and L. Ziliani (2009). Channel adjustments in northern and central Italy over the last 200 years. Management and Restoration of Fluvial Systems with Broad Historical Changes and Human Impacts. L. A. James, S. L. Rathburn and G. R. Whittecar. Geological Society of America Special Paper 451, Boulder, CO, USA: 83-95.

Svenning, J.C. (2002). "A review of natural vegetation openness in north-western Europe" Biological Conservation 104: 133-148.

von Koenigswald, W. (1991). "Exoten in der Großäuger-Fauna des letzten Interglazials von Mitteleuropa. " Eiszeitalter und Gegenwart 41: 70–84

Verdonschot P.F.M. & Nijboer R.C. (2002). "Towards a decision support system for stream restoration in the Netherlands: an overview of restoration projects and future needs. " Hydrobiologia 478: 131-148.

Waller, M.P. (1993). "Flandrian vegetation history of south-eastern England. Pollen data from Pannel Bridge, East Sussex. " New Phytologist 124: 345–369.

Waller, M.P. (1993). "Flandrian vegetational history of south-eastern England. Stratigraphy of the Brede valley and pollen data from Brede Bridge. " New Phytologist 126: 369–392

Winterbottom, S.J. (2000). "Medium and short-term channel planform changes on the Rivers Tay and Tummel, Scotland." Geomorphology 34: 195–208.

Wyżga, B. (1993). "River response to channel regulation: case study of the Raba River, Carpathians, Poland." Earth Surface Processes and Landforms 18: 541–556.

Wyżga, B. (2001a). "A geomorphologist’s criticism of the engineering approach to channelization of gravel-bed rivers: case study of the Raba River, Polish Carpathians." Environmental Management 28: 341–358.

Wyżga, B. (2001b). "Impact of the channelization-induced incision of the Skawa and Wisłoka Rivers, southern Poland, on the conditions of over- bank deposition." Regulated Rivers: Research and Management 17: 85–100.

Wyżga, B. (2008). A review on channel incision in the Polish Carpathian rivers during the 20th century. Gravel-Bed Rivers VI – From process understanding to river restoration, Developments in Earth Surface Processes. H. Habersack, H. Piégay and M. Rinaldi. Elsevier, 11: 525-555.

# 3 Riparian and Floodplain Vegetation Hydromorphology Interactions - Italian Case Studies

The three Italian case studies illustrate how channel changes (i.e. channel narrowing and bed incision), caused by severe in-channel sediment extraction and other human disturbances, have influenced riparian vegetation patterns in terms of landscape and plant species diversity, complexity of interactions, changes in lateral connectivity and riparian vegetation communities and decrease in the riparian tree growth rate.

## 3.1 Introduction

As illustrated in Chapter 5, European rivers have experienced a long history of channel changes caused by human impacts and modifications, with severe consequences for the overall riverine ecosystem including riparian vegetation.

In this chapter, three Italian case studies will be presented on the consequences of an altered fluvial hydromorphology for riparian vegetation, mainly caused by channel changes in terms of river channel narrowing and bed incision. The three case studies are based on recent studies carried out on this topic (Dufour et al. 2015, for the Magra River, for the Panaro River and the Aurino River).

The Magra River, a gravel-bed river located in the northern part of the Mediterranean biogeographical region (Central-Northern Western Italy). Here, changes have occurred in the riparian landscape structure and diversity since the 1930s induced by human-made channel changes, mainly in the form of catchment scale afforestation and in-channel sediment mining (Dufour et al. 2015). A comparative analysis of two unconfined, formerly braided reaches with different degrees of human disturbance and resulting channel changes was performed.

The second example is the Panaro River, a deeply incised and narrowed, formerly braided tributary of the Po River, located in the continental biogeographical region. The study focuses on existing vegetation types and indicator species in relation to the present environmental conditions (in terms of geomorphic units) and channel changes along the river (Gumiero et al. under revision). At this case study, we investigate if the use of indicator species, typical of vegetation types associated with specific geomorphic units, is helpful in understanding hydromorphological impacts on physical processes and riparian vegetation.

The Aurino River, a typical Alpine large mountain river, is the third and last example. The study uses dendroecology to show how changes in the vegetation relate to channel changes (i.e. incision). Dendroecology can help to understand the linkage between river hydromorphological changes and riparian vegetation. However, few dendrochronological studies have addressed riparian vegetation (Merigliano et al. 2013), only a limited number of studies have been performed which focus on the response of riparian vegetation to human-induced incision (e.g. Hupp and Simons, 1991), and no studies have described vegetation restoration works in mountain rivers.

## 3.2 The Magra River Case Study

### 3.2.1 Study area

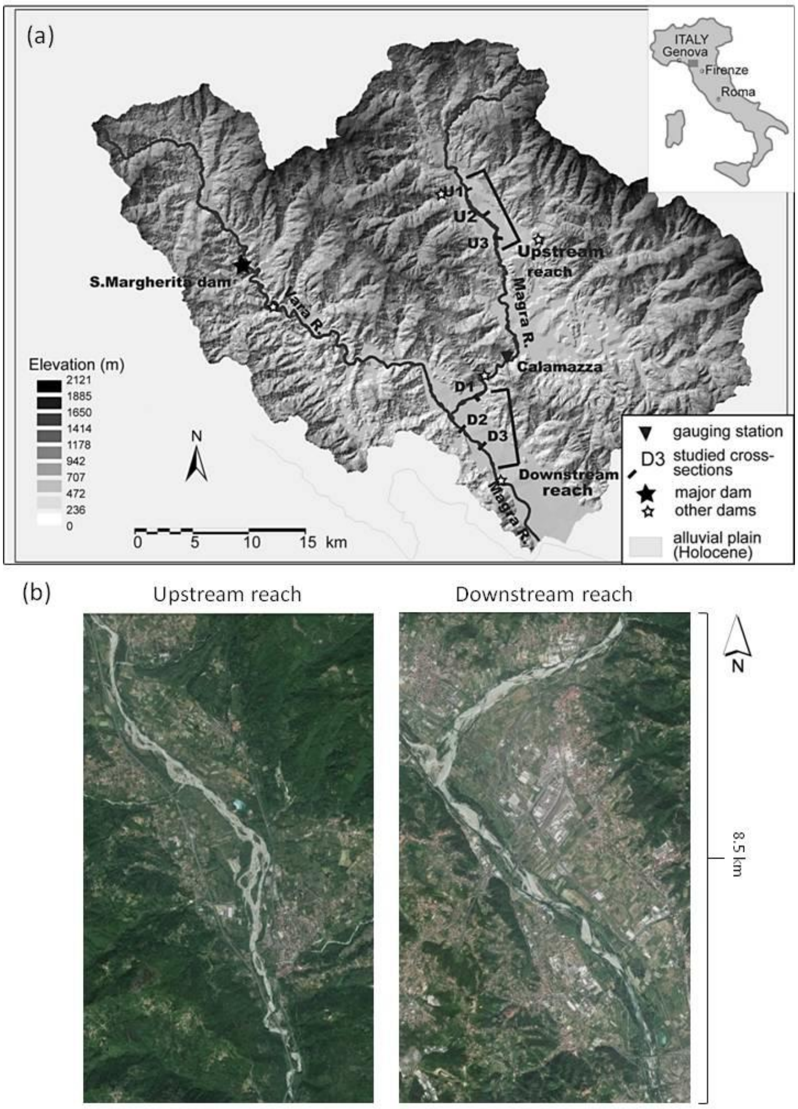
The Magra river catchment is located in the northern part of the Mediterranean biogeographical region, in Northern Tuscany and Liguria (Central-Northern western Italy) (Figure 3.1a). It is about 1700 $km^2$, and mainly dominated by hilly areas (from 200 to 600 m a.s.l.). The catchment is located in the temperate climatic zone with a dry summer season and a mean annual rainfall of 1707 mm. The Magra River is about 69.5 km long. The downstream part, about 43 km long, alternates between unconfined and partly confined reaches and is characterised by gravel bed, while the final reaches are dominated by sand. The channel gradient ranges from about 0.014 (upstream) to 0.0004 $mm^{-1}$ (near the sea). See D2.1 (Gurnell et al. 2014b, Part 3) for a more detailed description of the Magra River and its catchment.

The analysis is focused on two reaches located in the middle unconfined segments of the river (Figure 3.1). The reaches are characterised by comparable changes in channel pattern but the magnitude differs as a result of differences in the level of human impacts (see also Rinaldi et al. 2008, 2009). The upstream reach is 9.5 km long, flows in a relatively large alluvial valley and has an island-braided channel pattern. The downstream reach is a less steep 10 km long wandering channel.

### 3.2.2 Methods

The analysis is based on existing studies (e.g. Rinaldi et al. 2009) which describe changes in channel width and depth since the 1930s as a consequence of human impacts at catchment, segment and reach scale (mainly sediment mining and catchment afforestation). River corridor Floodplain and riparian structure is analysed for the period between 1937 and 1999 and its recent condition (in 2006).

The development of floodplain and riparian structure at the patch scale has been recorded over seven decades (i.e. 1937, 1954, 1971, 1981 and 1999) by the use of aerial photographs and remote sensing. The spatial units (patches) identified for this analysis were as follows: active channel (i.e. wetted channel and unvegetated bars), pioneer vegetation, units dominated by shrub, woodland, meadow, agriculture, human activity (i.e. roads, houses and sediment mining sites). Reaches were split into 250 m long sub-reaches in order to obtain replicates, and an index of Floodplain and riparian diversity was calculated for each (Shannon index). Additionally, from the overlay of all mosaics, the age of formation for all surfaces within the corridor was determined.



(c )

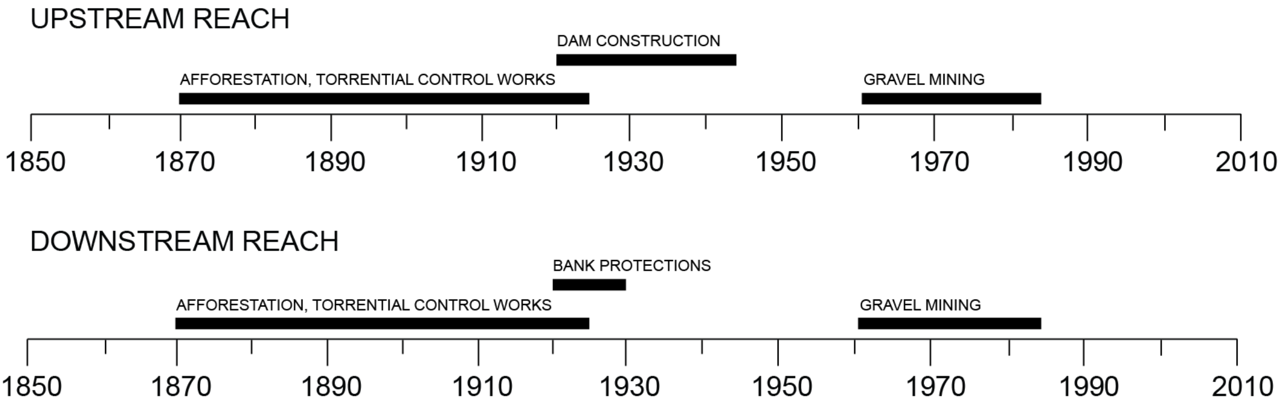


Figure 3.1(a) The Magra river catchment and location of the study reaches (from Dufour et al. 2015). (b) Detail of the study reaches and (c) timelines for alterations to the study reaches.

The current floodplain and riparian structure was classified into patches at two scales of increasing resolution (Table 3.1) by combining the results of remote sensing (on 2006 images) and a field survey conducted in 2006. During the field survey homogeneous patches of vegetation (named vegetation units) in terms of physiognomy (e.g. grassland, forest) and floristic composition were identified and surveyed (recorded physiognomic parameters and presence/absence of censed species) along six cross sections (i.e. 3 per reach). These data were analysed by correspondence analysis of 91 species and 11 types of vegetation units. The units described along cross sections were used to identify the nature of all patches (Table 3.1). Based on all the identified patches, the Floodplain and riparian structure was characterised by calculation of several metrics computed by V-LATE plug-in on ArcGIS (Lang and Tiede 2003; Table 3.2).

Table 3.1 List of units mapped in the current river floodplain and riparian analysis at two levels of detail (from Dufour et al. 2015).

| Level 1 | Description | Level 2 | Description |
|---|---|---|---|
| HUM1 | Human and non-vegetated units | ANT | Road, houses, villages |
| | | ANT_BG | Bare ground (secondary paths etc.) |
| HUM2 | Human origin and vegetated | CULT | Agriculture (crops) |
| | | VANT_T | Dominated by trees (along roads, around houses etc.) |
| | | VANT_S | Dominated by shrubs (along roads, between croplands etc.) |
| | | VANT_H | Dominated by herbaceous plants (football pitch, playground etc.) |
| FAL | Fallow land resulting from deforestation or cropland abandonment | FAL_S | Dominated by shrubs |
| | | FAL_H | Dominated by herbaceous plants |
| GRA | Grassland | QP | Wet grassland |
| | | MP | Mesic meadow (possible few isolated trees or shrubs) |
| | | DP | Dry meadow (possible few isolated trees or shrubs) |
| | | SHR | Shrub development in DP and MP |
| AQI | Aquatic area | PLA | Artificial lentic water (former gravel mining) |
| | | MC | Main and secondary channel (up-downstream connected) |
| | | SC_F | Lateral channel upstream or downstream connected |
| | | SC_W | Former channel non connected, located within the floodplain (e.g. surrounded by post-pioneer units) |
| GB | Gravel bars, sparsely vegetated | GB_1 | Vegetation cover < 5%, cobbles and pebbles |
| | | GB_2 | Vegetation cover > 5%, presence of small sandy patches |
| | | DSCF | Secondary channel, dry and vegetated by pioneer species |
| PIO | Pioneer patch dominated by shrubs (manly *Salix* sp. and *Populus nigra*) | PIO_D | Dense pioneer patch |
| | | PIO_C | Pioneer patch less dense with some GB 2 mixed |
| WOO | Woodland | FF_D | FF degraded (e.g. lower density) |
| | | FF_W | FF (probably) wet |
| | | FF_WD | FF (probably) wet and degraded (e.g. lower density) |
| | | FF_SA | FF dominated by *Salix alba* |
| | | FF_SAD | FF dominated by *Salix alba*, degraded (e.g. lower density) |
| | | FF_DD | FF dry and degraded (e.g. lower density) |

### 3.2.3 Results

*Channel changes and human impacts*

In 2006, the upstream reach is narrower (30%) and slightly incised (up to 4 m) compared to the 1950s. The downstream reach has undergone stronger channel changes in terms of narrowing (55%) and incision (up to 8 m), and consequent changes in channel pattern from braided to wandering have occurred since the 1950s. Both reaches have been affected by catchment-scale pressures, mainly afforestation, since the end of the 19th century until the 1920s, initiating a process of channel adjustment at catchment scale. The upstream reach is located downstream from an area affected by three small dams (i.e. 19% of the reach drainage area), built between the 1920s and the 1940s, which in part has contributed to the narrowing and incision of the channel. The Magra River was also impacted, as many others Italian rivers, by sediment mining during the 1950s-1970s, which was the main cause of the stronger alterations of the downstream reach (it is estimated that the mined sediment here was 3 orders of magnitude higher than the annual bedload transport; Rinaldi et al. 2009). In the downstream reach, bank protection is more extensive (building of bank protection starting during the 1920s) compared to the upstream reach.

*Floodplain and riparian habitat structure changes*

From the 1970s and onwards, a progressive shift in the floodplain and riparian habitat composition of the corridor occurred in both reaches, from open units to closed, wooded units (Figure 3.2a). After 1954, the floodplain and riparian diversity index increased (i.e. decrease of the active channel), with a maximum in 1981 for the upstream reach, whereas in the downstream reach, floodplain and riparian diversity was still increasing between 1981 and 1999 (Figure 3.2b). The increase in floodplain and riparian habitat diversity was, however, related to an increase in the diversity of anthropogenic units both upstream until 1981 and downstream at least between 1954 and 1971 (i.e. considering also anthropogenic units; Figure 3.2b).

*The current floodplain and riparian habitat structure*

The analysis of the current corridor structure confirms that floodplain and riparian habitat diversity is higher downstream than upstream, patches being larger and exhibiting lower density downstream than upstream for an equivalent form (MSI; Table 4.2). The overall edge density is also lower in the upstream reach. While natural units prevail in the corridor upstream (i.e. gravel bars and woodland), grasslands and anthropogenic units are frequent in the downstream corridor, but not significantly dominant. In the latter case, the higher proportion of anthropogenic features results from the presence of both larger and more numerous patches (data not shown). The downstream reach contains a lower proportion of surfaces younger than 25 years and of surfaces older than 69 years. This can be linked to the relatively higher narrowing intensity downstream than upstream.

The Correspondence Analysis of vegetation data highlights the relative influence of human and natural disturbances (Figure 3.3). The first axis (F1) separates young units (gravel bars, pioneer vegetation) characterised by pioneer and ruderal species, typical of the upstream reach, to more mature/stable units typical of the downstream reach, characterised by: post-pioneer floodplain forests and associated species; degraded forest (i.e. lower density due to wood cutting); some non-native (*Bamboo* sp., *Robinia pseudo-acacia*) and invasive species (*Arundo donax*) usually favoured by human disturbance. The second axis (F2) separates grasslands from riparian humid forest, with dry and degraded forest at an intermediate position. As expected, there is no clear difference between reaches in terms of dry and wet forests.

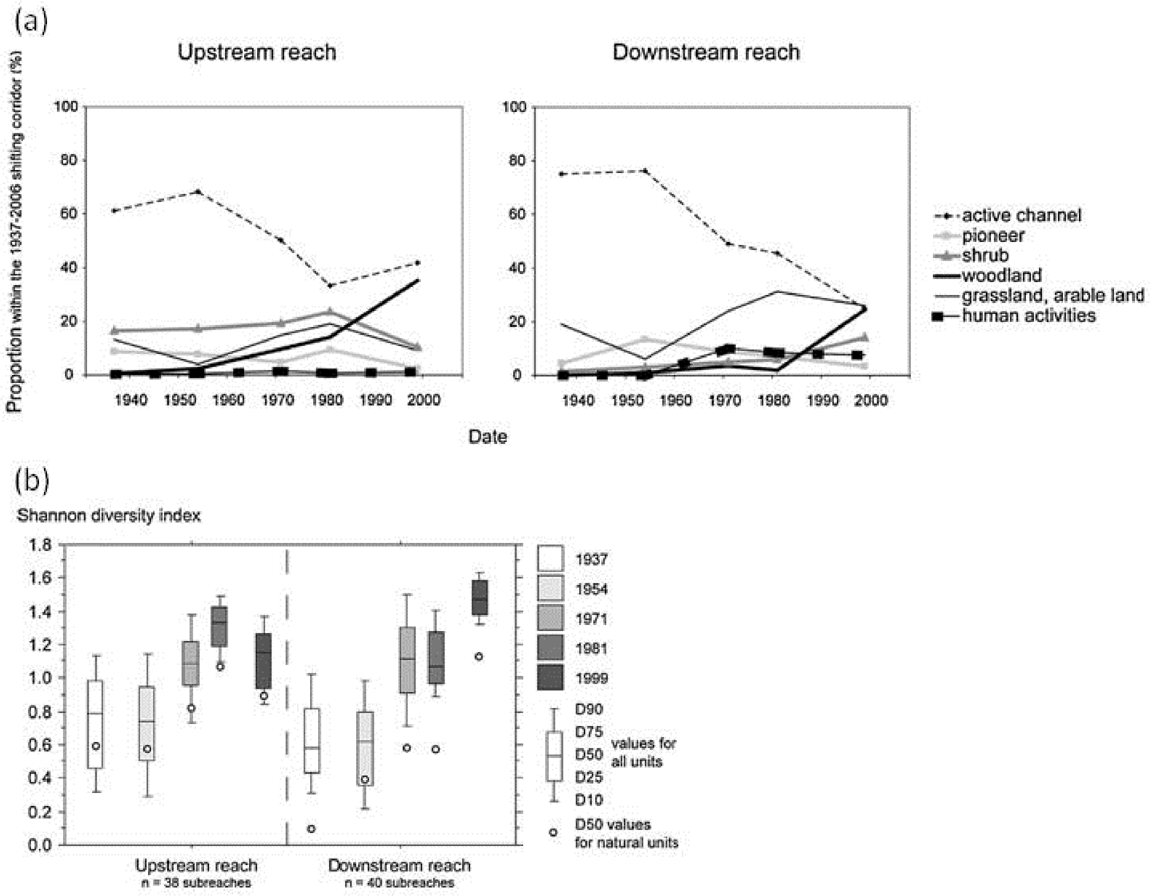


Figure 3.2 Floodplain and riparian habitat evolution in the Magra river corridor between 1937 and 1999 (modified from Dufour et al. 2015). (a) Changes in the proportion of each floodplain and riparian patch. (b) Change in floodplain and riparian habitat diversity within the two studied reaches between 1937 and 1999.

Table 3.2 Current floodplain and riparian pattern characteristics obtained by means of V-LATE plug-in on ArcGIS (Lang and Tiede 2003). For unit acronyms see Table 4.2. Modified from Dufour et al (2015).

| | Index | Upstream reach | Downstream reach |
|---|---|---|---|
| Area | Mean patch size (MPS) $m^2$ | 2429 | 3240 |
| | Density (n. patch $ha^{-1}$) | 4.1 | 3.1 |
| Form | Mean shape index (MSI) | 3.8 | 3.7 |
| Edge | Edge density (ED) (m $ha^{-1}$) | 2423 | 2037 |
| Diversity | Shannon's diversity (level1) | 1.5 | 2.0 |
| | Shannon's diversity (level 2) | 2.4 | 3.0 |
| Age distribution | 0 to 25 years | 28.9% | 10.7% |
| | 25 to 69 years | 58.9% | 85.2% |
| | More than 69 years | 12.2% | 4.1% |
| % floodplain and riparian features of level 1 | %HUM1 | 1.1 | 8.1 |
| | %HUM2 | 2.2 | 12.9 |
| | FAI | 0.3 | 7.5 |
| | GRA | 7.8 | 19.7 |
| | AQU | 8.9 | 10.9 |
| | GB | 33.0 | 13.6 |
| | PIO | 11.2 | 4.7 |
| | WOO | 35.4 | 22.5 |

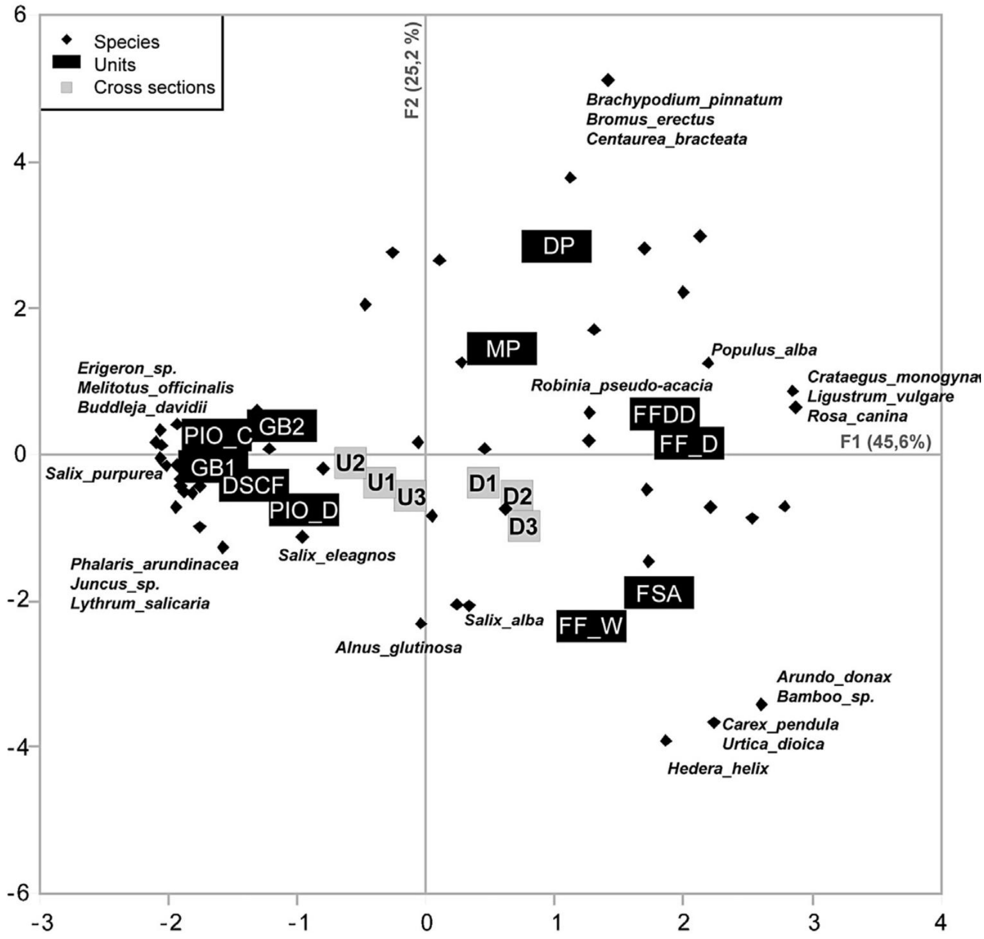


Figure 3.3 Results of the correspondence analysis performed on 91 species and 11 types of vegetation units surveyed along 6 cross sections (3 for each reach). Axis F1 and F2 explain, respectively, 45.6 and 25.2% of total inertia. Relevant species are named. For unit names see Table 3.1 (e.g. DSCF, FF_D, etc.). Position of the cross sections on the factorial map is given by the weighted mean based on the relative proportion of units along each cross section (U1, U2, U3: upstream cross sections; D1, D2, D3: downstream cross sections).

### 3.2.4 Discussion

The analysis of the riparian floodplain and riparian vegetation evolution in the last century showed that channel changes (i.e. narrowing and incision) are associated with a combination of human impacts (mainly afforestation and gravel mining), leading to enhanced floodplain and riparian diversity. This is also obvious when comparing the two reaches, human impact being greater in the downstream reach, resulting in greater riparian floodplain and riparian diversity.

This gives rise to questions concerning the management of river systems and which attributes to preserve or restore. The present study suggests that management of diversity should not necessarily be a management target for various reasons:

dplain and riparian diversity in the downstream reach is supported by the presence of more
enced units;

m reach, which has undergone less channel modification, due to lower impact intensity, the floodplain e dominated by two habitats, gravel bars and woodland that have been maintained or developed as a ve channel lateral dynamics, thus constituting a well-functioning river in terms of processes. esence of these two habitat types indicate a high naturalness of the system that should be preserved

Additionally, these habitats have a high ecological function related to the presence of pioneer units (e.g. maintenance of age and species diversity within the vegetation community, supporting wildlife, providing large wood and bank resistance).

## 3.3 The Panaro River Case Study

### 3.3.1 Study area

The Panaro River is a tributary of the Po River located in the northern Apennines (Northern Italy). The river is about 148 km long and has a catchment area of 1783 km$^2$. The upper part of the catchment is located above 2000 m a.s.l.; hence snow melt affects the flow regime, resulting in spring and autumn flow peaks and summer droughts.

In the present study, a 38 km long section of the river was selected corresponding to the central alluvial segments (Figure 3.4). The first 6 km concern a partly-confined pseudo-meandering channel located within a hilly physiographic unit (named segment 2), which is surrounded by near-natural vegetation and not-intensive agricultural lands. The central part (about 26 km) includes all of the alluvial unconfined segment where wandering channel morphology prevails, and is mainly surrounded by orchards (named segment 3).The last 6 km concern the upstream, unchannelised portion of the low alluvial plain with sinuous-meandering alluvial channel (segment 4), dominated by arable land (intensive agriculture).

The Panaro River and its catchment have been affected by several, relatively, recent human impacts (Table 3.3), including (1) widespread afforestation at the catchment scale since the early 20th century; (2) construction of levees (segment 4) and widespread bank protection measures between 1920-1960; (3) construction of several in-channel structures, i.e. numerous weirs and check dams within segments 2 and 3 (1970s-1980s), a large flood control retention basin in segment 3 (1980s), and a small dam on a tributary (1950s); (4) channel sediment mining (1960s-1980s), particularly within segment 3 (Figure 3.4).

The aim of the study was to document the changes to the channel bed and channel width in relation to the human pressures and to associate vegetation types with discrete geomorphic units. These associations make it possible to link changes to geomorphic units to riparian by vegetation.

### 3.3.2 Methods

Analysis of channel morphology and changes was performed by combining remote sensing analyses of historical and recent topographic maps (1980 and 1934-35, respectively) and aerial photos (1954-55, 1962-69, 1981, 1997, 2000, 2003; analysis of channel patterns and width changes), and existing information and field surveys carried out between 2006 and 2007 (analysis of bed level changes). The most recent aerial images were also used for delineation of the following six river geomorphic units along the study section according to Hupp and Osterkamp (1996), Hupp and Rinaldi (2007): B, active bar; HB, high bar; FP, floodplain; TB, terrace bank; T, terrace; W, wetland. The combination and interpretation of data on channel morphology and changes supported the delineation of six geomorphologically-homogeneous reaches. These geomorphic analyses (i.e. present geomorphic units and homogeneous reaches) provided the basis for the analysis of the riparian vegetation.

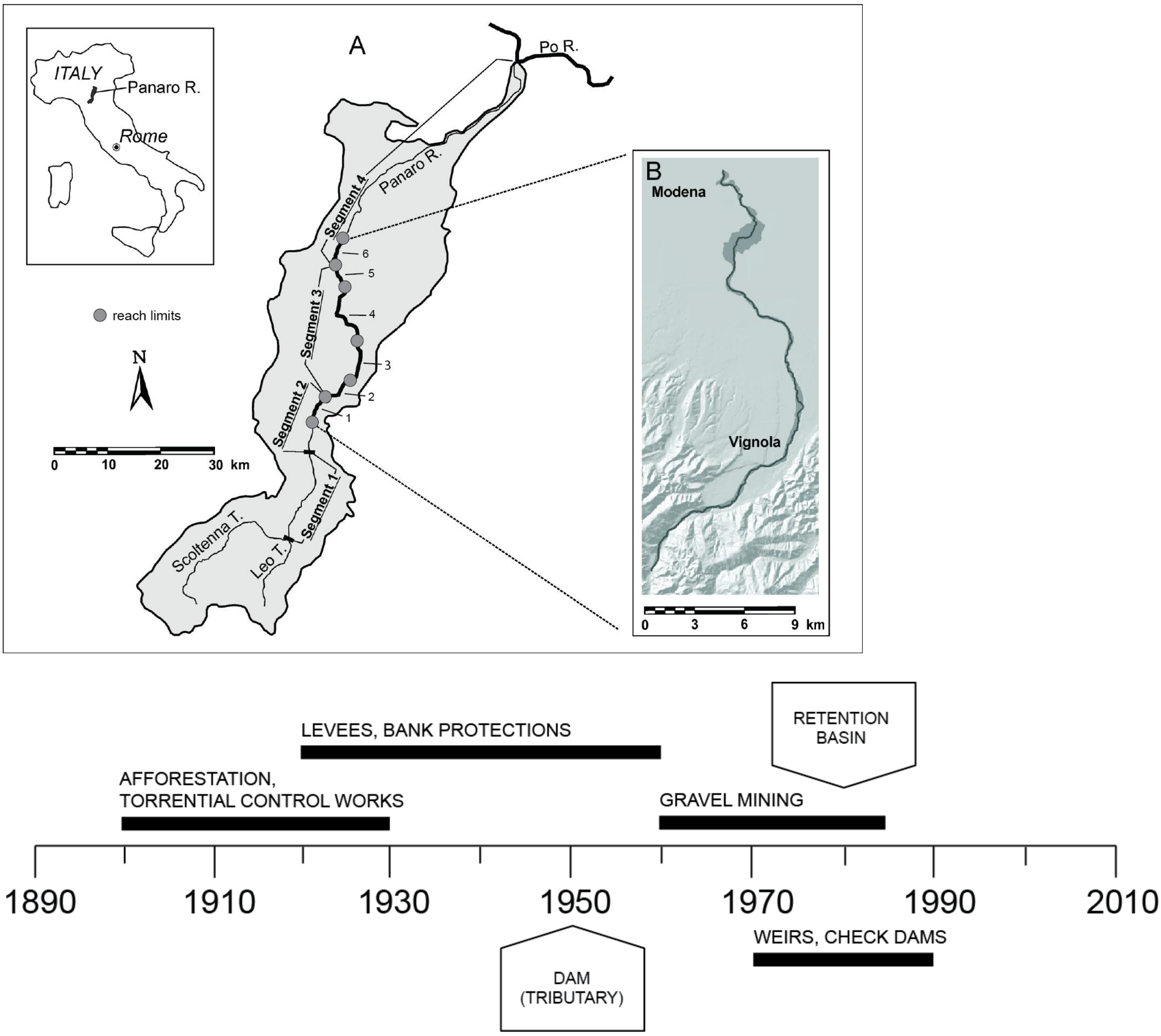


Figure 3.4 The Panaro river catchment subdivided into homogenous river segments and reaches (A); (B) details on the study area (from Gumiero et al. under revision) and (C) a timeline of impacts on the study area.

The riparian vegetation along the study section was surveyed between 2006 and 2009. Plant species were collected by means of about 120 phytosociological *relevés* (Braun-Blanquet 1965), including woody and herbaceous vegetation. The vegetation was analysed at two levels of resolution: (i) the *relevés* were classified into vegetation types (by means of average linkage and Jaccard similarity index, J). The relation between vegetation types and geomorphic units was investigated (Pearson correlation) and deviation from the expected relation was compared to a similar and near-natural river system (the Taro River: Biondi et al. 1997). (ii) Then, species being indicative of the vegetation types were identified (stenoecious or well-adapted) and used as indicators of specific ecological and thus environmental conditions of the various vegetation types (see: Landolt 1977; Ellenberg 1985). These species were then used to: (a) analyse in detail the relationship between vegetation types and geomorphic units along the study reach by means of Pearson correlations and PCA ordination between species and geomorphic units; (b) explore and understand plant species composition of the six geomorphologically homogeneous reaches, by means of Pearson correlations between species and reaches; additionally, the abundance of species within each river reach was also described by means of average values of the Braun-Blanquet coverage scale, i.e. S, sporadic and sparse species; W, widespread species; V, very common, abundant species).

### 3.3.3 Results

*Channel changes and human impacts*

Channel changes and the current morphological channel patterns and main human impacts at each reach are summarised in Table 3.3. Most of the reaches underwent strong channel changes in width and bed level (i.e. narrowing and incision). At reaches 1 to 5, there were also changes to the channel pattern. The most impacted reach was reach 2 where incision in some points reached 10 m together with widespread bedrock outcrop and a drastic change from a braided to a sinuous channel morphology (Figure 3.5). The most severe changes occurred during, or immediately after, the period of intense gravel mining (mainly incision).

*Vegetation analysis*

The analysis of riparian vegetation focused on reaches 1 to 5, and reaches 4 and 5 were analysed together (as they were homogeneous in terms of vegetation, morphology and channel changes). Reach 6 was excluded due to dominance of poplar plantations.

The classification of the 120 phytosociological *relevés* allowed identification of 7 main vegetation types (J= 6%; Table 3.4). As expected (e.g. Hupp and Rinaldi, 2007) and according to the model of near-natural riparian vegetation (Taro River: Biondi et al. 1997), each main vegetation type was strictly related to a particular geomorphic unit. However, some unexpected relationships were observed, i.e. VBR with T (Table 3.4; Figure 3.5, at the bottom).

In all, 412 plant species were identified (67 woody and 345 herbaceous) of which 51 are stenoecious, typical of a single vegetation type (named indicator species). Amongst the 51 species, 41 showed significant positive correlations with geomorphic units, most being strictly related to a specific unit (data not shown).

A PCA of indicator species and the geomorphic units showed that the first six components explain only 50% of total variance (Figure 3.6). These components seem to represent the main ecological factors influencing the composition of the riparian community relative to the type of geomorphic unit:

- The first component separates geomorphic unit B (i.e. bars) together with several ruderal weeds from the other species: the first factor represents a gradient in natural disturbance.
- The second component separates geomorphic units and species along a gradient of

human disturbance and dryness: the highest values of component 2 characterise a group of perennial herbs typical of anthropogenic semi-dry grasslands.
- The third and fourth components separate the floodplain (FP) from the other units and show the floristic diversification of the riparian woods growing in the floodplain. The third component could be identified as “soil richness” (presence of several nitrophilous species), while the fourth component could be interpreted as “soil evolution” towards humid and fine soils (presence of hygrophilous species).

The distribution of the 51 indicator species and their abundance support the environmental characterisation of the geomorphologically homogenous reaches (Table 3.5):

- Reach 1 is linked to species to less hygrophilous species (VSA1 and VBR); 1/5 of the indicator species are missing and there is a high number of widespread species compared to other reaches.
- Reach 2, sinuous and incised, is characterised by species typical of mature hygrophilous forests (VSA, VSA2) but also species characteristic of wetlands (VPH) and arid grasslands (VBR). Several indicator species are represented.
- Species typical of river bars are present in the wandering reach 3 (VBI). As for reach 2, several indicator species occur; reach 3 has the highest number of sporadic species.
- At reaches 4 and 5 the riparian vegetation is composed mainly of white willow and nitrophilous herbaceous species (VSAR) and the reaches are characterised by very low diversity and occurrence of very common and abundant species.

Table 3.3 Classification of the three studied segments of the Panaro River in relatively homogeneous reaches relative to present channel morphology and channel changes (since the 1890s). The main human impacts affecting each reach are summarised. S, segment; R, reach; L, reach length (km); PM, present channel morphology; CM, changes in channel morphology; WC, width changes; AC, altimetric changes (bed level); HI, existing human impacts on the reaches (modified from Gumiero et al. under revision).

| S | R | L | PM | CM | WC | AC | HI |
|---|---|---|---|---|---|---|---|
| 2 | 1 | 6 | Pseudo-meandering | Braided to pseudo-meandering | Narrowing >50 | Incision from 4 to 6 m | Weirs; bank protection |
| 3 | 2 | 6.2 | Sinuous | Braided to single-thread | Narrowing >50% | Incision >6 m (up to 10 m) | Weirs; bank protection; channelisation |
| 3 | 3 | 7 | Wandering | Braided to wandering | Narrowing >50% | Incision >6 m | Weirs; channelisation; bank protection |
| 3 | 4 | 9.7 | Sinuous | Wandering to sinuous | Narrowing >50% | Incision >6 m | Retention basin; channelisation; weirs |
| 3 | 5 | 3.5 | Sinuous | Wandering to sinuous | Narrowing >50% | Incision from 4 to 6 m | Levees; bank protection; small weirs |
| 4 | 6 | 5.9 | Meandering | No change | Narrowing <50% | Incision <4 m | Levees; bank protection |

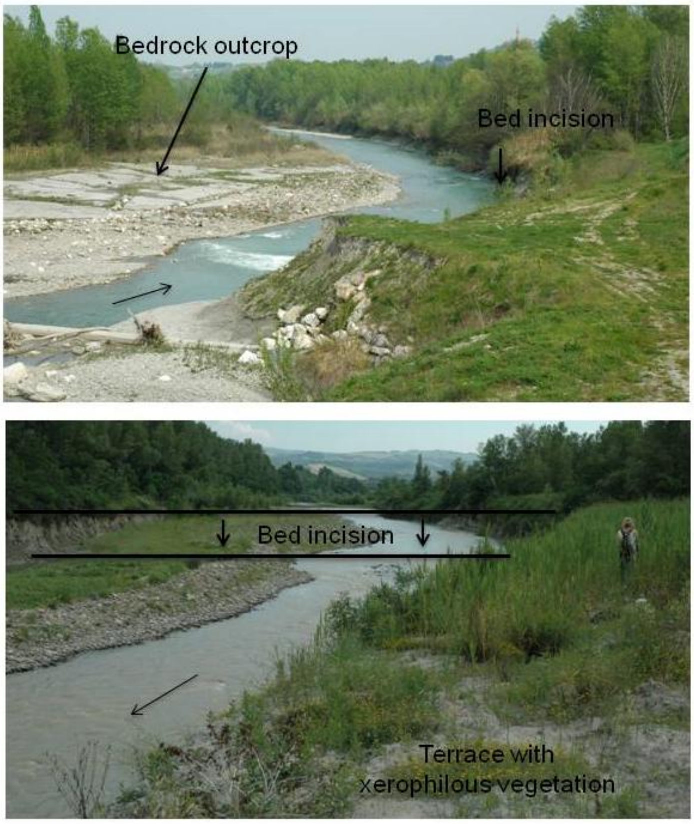


Figure 3.5 Bed incision, reach 2. Top photo shows the bedrock outcrop; bottom photo shows the emergence of a new surface (i.e. a recent terrace) completely disconnected from the main channel with an unexpected riparian vegetation community.

Table 3.4 Main vegetation types identified by cluster analysis of 120 phytosociological relevés and related (observed) geomorphic units (GUs) derived from the correlation analysis (Pearson). For each vegetation type, the expected geomorphic units according to the model of near-natural vegetation are shown (Taro River: Biondi et al. 1997). Geomorphic units: W*= channels; W= wetland; B= bar; HB= high bar; FP= floodplain; TB=terrace bank; T=terrace.

| Vegetation types | Description | Observed GUs | Expected GUs |
|---|---|---|---|
| VID | hydrophytic communities of the classes *Lemnetea minoris* and *Charetea fragilis* | W*; W | W* |
| VPH | helophytic reed beds of *Phragmito-Magnocaricetea* (*Phragmitetalia australis*) with *Phragmites australis* and *Typha* sp.pl. | W; B | W |
| VBI | sparse hygro-nitrophilous communities of *Stellarietea mediae* (*Bidentetalia*), with *Bidens frondosa*, *Polygonum* sp.pl. and *Echinochloa crus-galli* | W; B | B |
| VDM | nitrophilous communities of *Artemisietea vulgaris* (*Onopordetalia acanthii*) with *Melilotus officinalis* and *Artemisia verlotorum* | HB | HB |

| Vegetation types | Description | Observed GUs | Expected GUs |
|---|---|---|---|
| VSA | hygrophilous willow woods of *Alno-Populetea* (*Populetalia albae*) with *Salix alba*, *Populus nigra* and *Alnus glutinosa* | FP | FP |
| VLO | mixed woods (*Ostrya carpinifolia*, *Quercus cerris* and *Q. pubescens*) and scrubs of *Querco-Fagetea* (*Quercetalia pubescentis* and *Prunetalia spinosae*), mostly on hill slopes | TB | TB; T |
| VBR | xerophytic grasslands of *Festuco-Brometea* (*Brometalia erecti*) with *Bromus erectus*, *Sanguisorba minor*, *Euphorbia cyparissias* and *Artemisia alba* | T | |

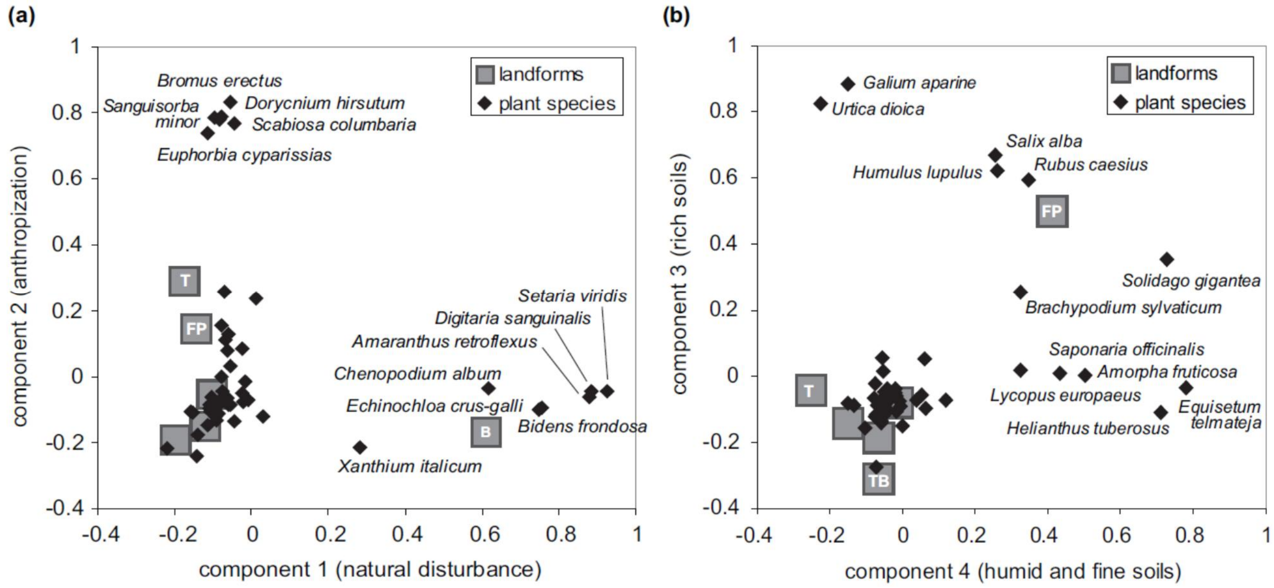


Figure 3.6 PCA ordination analysis showing the relationships between indicator species and the geomorphic units (landforms). (a) first (natural disturbance) and second components (anthropic disturbance); (b) third (soil richness-pollution) and fourth components (soil evolution) (extracted from Gumiero et al. under revision).

Table 3.5 Summary results of the relationship between the indicator species and the river reaches (see Table 3.3 for reach description). Only the indicator species showing significant relationships with specific reaches are included. Vegetation types and sub-types per reach are also shown. The total number of indicator species per reach is given as well as the number of species per class relative to abundance following the Braun-Blanquet coverage scale: S, sporadic and sparse species, with small populations occurring in a few or several sites, generally with low cover; W, widespread species, with several or many individuals present at most sampling points, sometimes in high abundances; V, very common, abundant and often dominant species (modified from Gumiero et al. under revision).

| | R1 | R2 | R3 | R4-5 |
|---|---|---|---|---|
| Indicator species | *Viburnum lantana*<br>*Quercus cerris*<br>*Ligustrum vulgare*<br>*Fraxinus ornus*<br>*Crataegus monogyna*<br>*Quercus pubescens*<br>*Euphorbia cyparissias* | *Rubus ulmifolius*<br>*Brachypodium sylvaticum*<br>*Alnus glutinosa*<br>*Alnus incana*<br>*Solidago gigantea*<br>*Alisma plantago-aquatica*<br>*Sanguisorba minor*<br>*Sedum sexangulare* | *Helianthus tuberosus*<br>*Echinochloa crus-galli*<br>*Digitalia sanguinalis*<br>*Amaranthus retroflexxus*<br>*Setaria viridis*<br>*Bidens frondosa*<br>*Typha latifolia*<br>*Fumana procumbens* | *Gallium aparine*<br>*Uritca dioica* |
| Vegetation types and sub-types | VLO/VSA1<br>VBR | VLO/VSA1<br>VSA<br>VSA2<br>VPH<br>VBR | VDM<br>VBI<br>VPH<br>VBR | VSAR |
| N. indicator species | 41 | 46 | 46 | 34 |
| Species abundance | | | | |
| S | 30 | 36 | 38 | 25 |
| W | 11 | 9 | 8 | 6 |
| V | 0 | 1 | 0 | 3 |

### 3.3.4 Discussion

The study was able to document the changes to the channel bed and channel width in relation to the human pressures and to associate vegetation types with discrete geomorphic units. These associations made it possible to link changes to geomorphic units to riparian by vegetation. The vegetation was an intimate part of the geomorphic response.

The case study of the Panaro River showed that human impacts severely affect channel morphology and its interaction with the riparian vegetation. Channel mining is considered the main reason for the strong channel changes; narrowing and severe channel incision (up to 10 m at some points). Complex and unexpected spatial and temporal patterns of the riparian vegetation may occur in cases of strong channel changes, which differs from the model of near-natural vegetation developed by Gurnell et al (2014a) (e.g. see Figs. 3.2 and 3.5). The results showed that:

(i) Most of vegetation types and indicator species are strictly related to a specific geomorphic unit, allowing the exploration of environmental conditions of this specific unit. In some cases the relationship between vegetation types and geomorphic units was unexpected and revealed changes in the environmental conditions. For example, a terrace was dominated by grassland vegetation instead of the expected woody forest, indicating that a strong and quite rapid channel incision took place, leading to disconnection of the former floodplain from the main channel and a drop in the water table (e.g. dieback of hygrophilous trees; field evidence; Figure 3.5).

(ii) Analysis of the indicator species gives further detail about ecosystem conditions (natural vs. human disturbance; Figure 3.6) and the direction of the environmental changes obtainable at geomorphic unit scale. For example, the analysis of the ecological value of indicator species (i.e. Landolt 1977) permitted identification, within the floodplain along the Panaro River, of sub-types of vegetation with different degrees of hygrophily, reflecting differences in channel changes and human impacts (data not shown). Human impact prevailed over natural disturbance.

(iii) The indicator species also provided useful information about differences in plant diversity and environmental conditions amongst river reaches (Table 3.5) subject to different channel evolution history (Table 3.3). In particular, (a) the distribution and abundances of indicator species confirm that reaches differ significantly in terms of physical and ecological conditions; (b) the plant diversity of an impacted river can be either high (reaches 2 and 3) or low (reaches 4 and 5); (c) the high plant diversity of a reach can be explained by natural disturbance or a consequence of human impacts (reach 3 versus reach 2, respectively). However, in the latter case it is expected that plant diversity will decrease since the coexistence of old and new species and communities is temporary; the new species appeared following channel changes and changes in habitat conditions and will progressively replace the existing species.

## 3.4 The Aurino River Case Study

### 3.4.1 Study area

The Ahr/Aurino River is located in the Eastern Italian Alps. The Ahr river basin has an extension of 629 km$^2$, with a maximum elevation of 3498 m a.s.l. and a present glacial cover of about 25 km$^2$.

The channel has suffered planimetric changes and bed degradation (Campana et al. under revision), as a consequence of sediment extraction from the river bed and sediment retention due to hydropower dams and several check-dams. The former floodplain is currently flooded only at a frequency of 30 to 50 years. In 2003, the Department of Hydraulic Engineering of the Autonomous Province of Bolzano started a river restoration programme involving removal of river bank protections, enlargement of the sections and raising of the riverbed.

The analysed riparian forests, dominated by *Alnus incana* (Figure 3.7), are located at an altitude between 830 and 850 m a.s.l. on the left bank of the restored stretch of Mühlen in Taufers and on both sides of the restored stretch of Gais.

### 3.4.2 Methods

The first sampling was carried out in October 2011 at the Mühlen reach where 8 trees were investigated: 7 grey alders (*Alnus incana*) and 1 ash (*Fraxinus excelsior*). The second survey took place in October 2012 and 35 trees were sampled from both Mühlen in Taufers and Gais forests (30 *Alnus incana* and 5 *Fraxinus excelsior*). Only dominant plants were selected to avoid intra- and inter-species competition effects on tree growth. A 5 mm increment borer was used and 2 cores were sampled from each tree in orthogonal direction to each other. The obtained wooden cores were subsequently measured and analysed using a Rinntech LINTAB 6 measuring station and Rinntech TSAP-Winsoftware. The obtained tree-ring series were cross-dated for each single tree and averaged into one single curve. Finally, the mean series were standardised through an exponential negative curve to obtain a Tree Ring Index (TRI).

### 3.4.3 Results

The tree ring index (TRI) for the ashes displays a marked decrease in growth rate during the 1970s (Figure 3.8) when bed incision was proceeding at the fastest rate, whereas alder trees exhibit a similar but milder pattern of reduced growth in these decade. Large variability in alders' TRI is evident until the 1950s and after the 1990s, with low growth rates in the years 2003-2004, characterised by very low precipitation amounts. Reduction in the growth of ash is less pronounced in these dry years, whereas a sharp increment in TRI is visible for both species in the most recent years, i.e. after restoration was carried out.

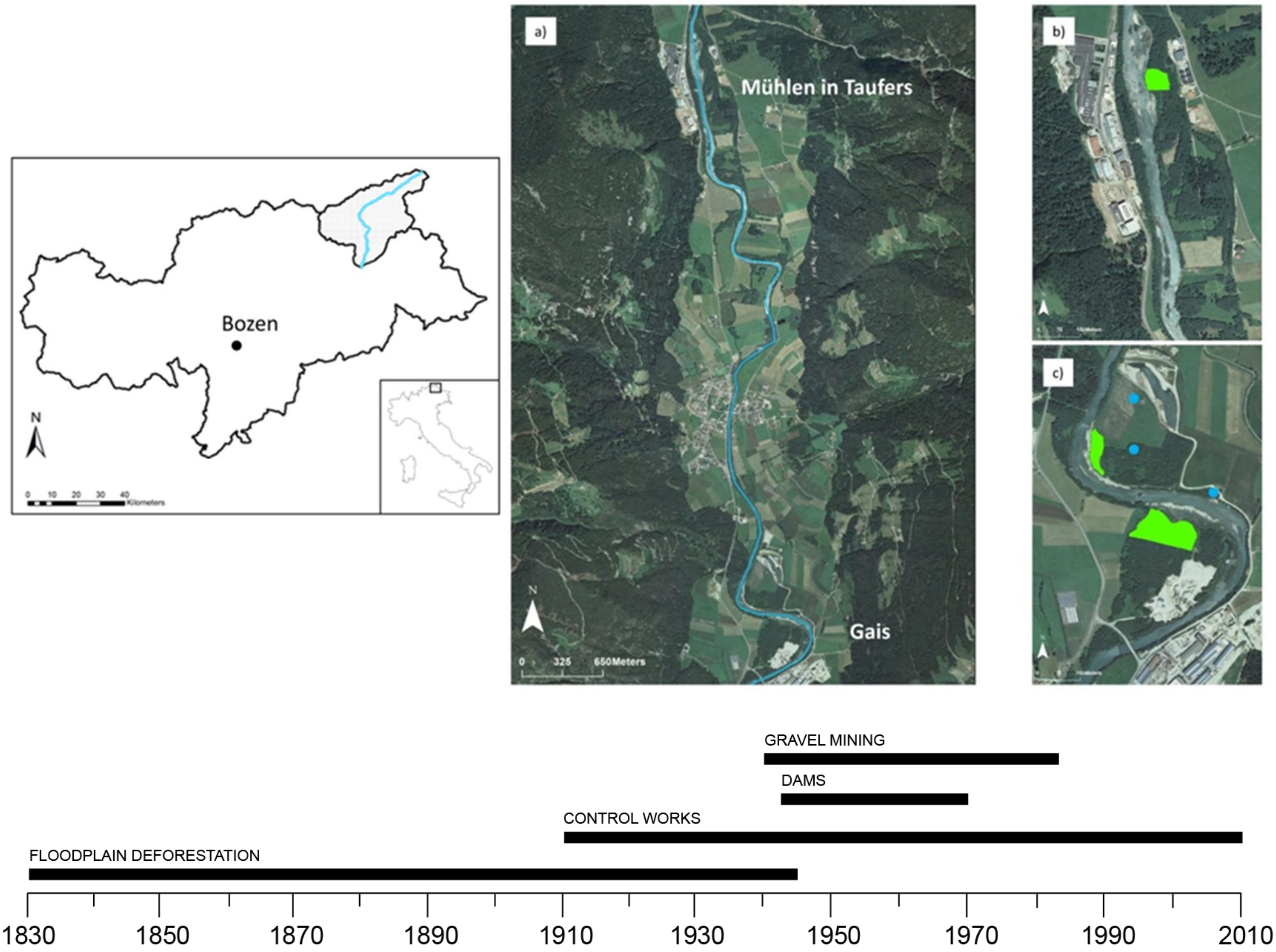


Figure 3.7 Location of the Ahr/Aurino basin within the Autonomous Province of Bozen/Bolzano (Italy, left) and orthophotos showing the restored reaches of Muhlen (b) and Gais (c) with the analysed riparian vegetation stands. (d) a timeline of impacts to the study reach.

### 3.4.4 Discussion

Variability in radial growth of ash and alder located in the floodplain of the Ahr River, a typical large mountain river in the inner Alps, appears to be mostly determined by variations in the groundwater table (tightly associated with water discharge oscillations in the river channel) rather than directly by precipitation. The dendrochronological results from the Ahr River seem to indicate that the effects of the past bed incision impacted the growth rates of trees already established in the floodplain, more severely for ash than alder, suggesting that alder responded less negatively to the fast lowering groundwater table associated with the degrading bed in the 1970s. Indeed, after the 1980s and until the mid-2000s also ash tree TRI values stabilised despite the still existing incised bed (and thus low groundwater table), but most probably the growth rate decline slowed down, probably as a consequence of the ability of the root system, at least partly, to follow the new, lower average level of the groundwater table.

The dendro-ecological study in the Ahr River points out the importance of both environmental factors and morphological channel changes for the tree growth rate of riparian forests and demonstrates that human activities interfering with sediment fluxes and river morphology (typically in-channel gravel mining and dam construction) can have severe consequences for riparian ecosystems.

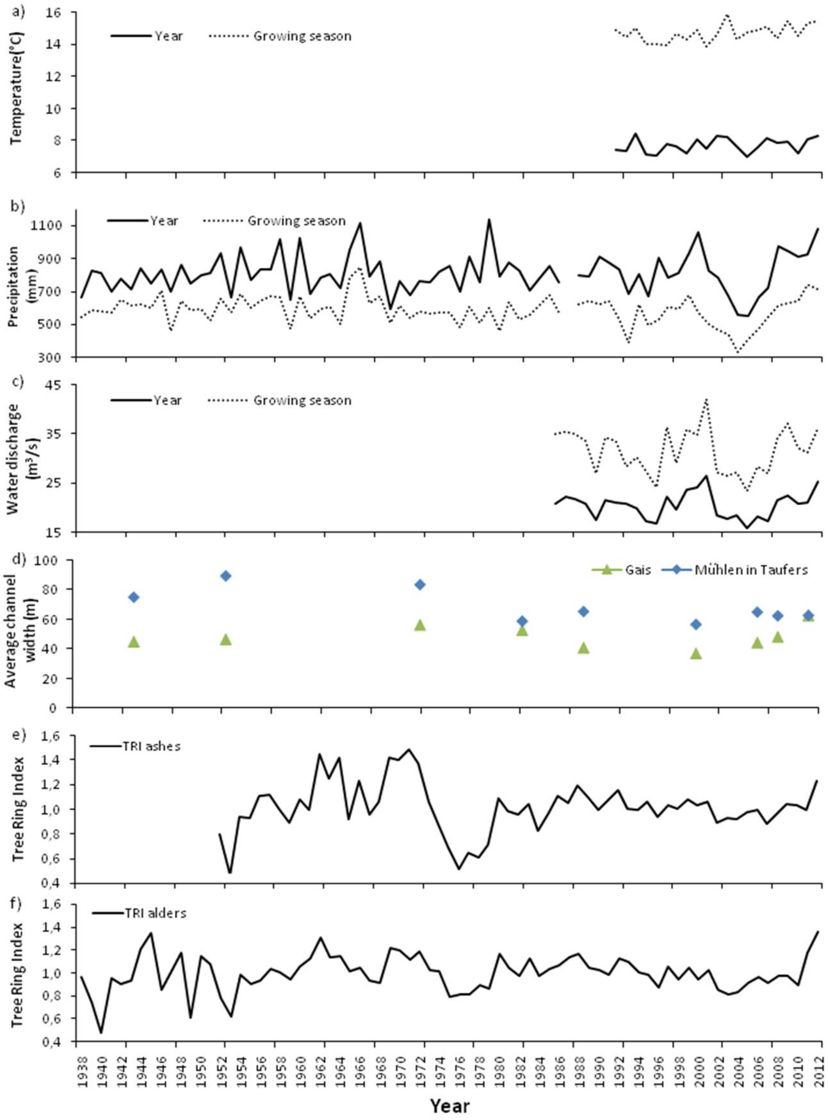


Figure 3.8 Trends during the years 1939-2012 in (a) mean temperatures; (b) cumulated precipitation; (c) average values of water discharge; (d) average channel width in Gais and Mühlen in Taufers; (e) mean standardised tree-ring growth for ash and (f) alder.

## 3.5 Discussion and conclusions

The three Italian case studies provide useful information on the relationship between hydromorphology in the case of impacted river systems and riparian vegetation.

As consequence of human impacts (mainly gravel mining, catchment-scale afforestation, etc.), the unconfined middle segments of the Magra, Panaro and Aurino rivers have adjusted to the modified conditions through changes in channel geometry and patterns (mainly narrowing and incision). This also affected the patterns of riparian vegetation including tree growth. Thus, unexpected interactions between riparian vegetation and river hydromorphology took place in comparison with the model of near-natural vegetation developed by Gurnell et al. (2014a).

The three case studies also highlight the complexity of the relationships that occurs between riparian vegetation and river hydromorphology in impacted rivers and how human disturbances may become dominant in structuring such relationships.

In summary, two key findings appeared:

1. Deviation from the conceptual model of near-natural vegetation patterns in the case of incised and narrowed river system, in terms of: (i) increased complexity in river floodplain and riparian structure and diversity (Magra); (ii) unexpected relationship between vegetation types and species versus novel geomorphic units and environmental conditions, as well as local and temporary increase in plant species diversity (Panaro); (iii) reduction of the growth rate of typical riparian trees (Aurino).

2. Despite the complexity of interactions between the riparian vegetation and the hydromorphology within impacted river systems, the riparian vegetation seems to be a good indicator of channel changes produced by human impact (at different spatial scales, i.e. geomorphic unit and reach scale). In particular, the use of index species and their ecological indicator value (including tree growth rate) seems to be significantly informative.

Finally, the case study examples confirm that plant diversity alone cannot be considered a valid and exhaustive indicator to assess the health of a river system and its functioning.

## 3.6 References

Biondi, E., Vagge, I., Baldoni, M. and F. Taffetani (1997). La vegetazione del Parco fluviale del Taro (Emilia-Romagna). Fitosociologia 34: 69-110.

Braun-Blanquet, J. (1965). Plant Sociology. The study of plant communities. Hafner, London: 439 pp.

Campana, D., Giammarchi, F., Tonon, G., Schmitt, A. O. and F. Comiti (under revision). "How sensitive is riparian vegetation to river morphological changes? A case study in the Italian Alps."

Dufour, S., Rinaldi, M., Piégay, H. and A. Michalon (2015). "How do river dynamics and human influences affect the landscape pattern of fluvial corridors? Lessons from the Magra River, Central–Northern Italy." Landscape and Urban Planning 134: 107–118.

Ellenberg, H. (1985). The ecological behaviour and the ecological grouping of species: Indicator values with regard to particular habitat factors. Vegetation of central Europe. H. Ellenberg. Cambridge University Press.

Gumiero, B., Rinaldi, M., Belletti, B., Lenzi, D. and G. Puppi (under revision). "Riparian vegetation as indicator of channel adjustments and environmental conditions: the case of the Panaro River (Northern Italy)."

Gurnell, A. M., González Del Tánago, M., O’Hare M. T., Van Oorschot, M., Belletti, B, García De Jalón, D., Grabowski, R., Hendricks, D., Mountford, O., Rinaldi, M., Solari, L., Szewczyk, M. and A. Vargas-Luna (2014a). Influence of Natural Hydromorphological Dynamics on Biota and Ecosystem Function. Deliverable 2.2, Part 1, of REFORM (REstoring rivers FOR effective catchment Management), a Collaborative project (large-scale integrating project) funded by the European Commission within the 7th Framework Programme under Grant Agreement 282656.

Gurnell, A. M., Belletti, B., Bizzi, S. et al (2014b). A hierarchical multi-scale framework and indicators of hydromorphological processes and forms. Deliverable 2.1, a report in four parts of REFORM (REstoring rivers FOR effective catchment Management), a Collaborative project (large-scale integrating project) funded by the European Commission within the 7th Framework Programme under Grant Agreement 282656.

Hupp, C.R. and A. Simon. (1991). "Bank accretion and the development of vegetated depositional surfaces along modified alluvial channels." Geomorphology 4: 111-124.

Hupp, C. R. and W. R. Osterkamp (1996). "Riparian vegetation and fluvial geomorphic processes." Geomorphology 14: 277-295.

Hupp, C. R. and M. Rinaldi (2007). "Riparian Vegetation Patterns in Relation to Fluvial Landforms and Channel Evolution Along Selected Rivers of Tuscany (Central Italy)." Annals of the Association of American Geographers 97: 12-30.

Landolt, E. (1977). Ökologische Zeigerwerte zur Schweizer Flora. Veroff. Geobot. Inst. ETH Stiftung Rubel, Zurich. 64: 1-208.

Lang, S. and D. Tiede (2003). (vector based tool for quantitative landscape analysis) vLATE Extension für ArcGIS—vektorbasiertes Tool zur quantitativen Landschaftsstrukturanalyse. In ESRI Anwenderkonferenz 2003. Innsbruck: CDROM. Available on https://sites.google.com/site/largvlate/home (June 2012) [In German].

Merigliano, M. F., Friedman J. M. and M. L. Scott (2013). Tree-Ring Records of Variation in Flow and Channel Geometry. Treatise on Geomorphology. J. Shroder (Editor in chief), D. R. Butler and C. R. Hupp. Academic Press, San Diego, CA. 12: 145-164.

Rinaldi, M., L. B. Teruggi, C. Simoncini and L. Nardi (2008). "Dinamica recente ed attuale di alvei fluviali: alcuni casi di studio appenninici (Italia Centro-Settentrionale)." Il Quaternario 21: 291-302.

Rinaldi, M., Simoncini, C. and H. Piégay (2009). "Scientific design strategy for promoting sustainable sediment management: the case of the Magra river (Central-Northern Italy)." River Research and Applications 25: 607-625.

# 4 Riparian and Floodplain Vegetation Hydromorphology Interactions - Spanish Case Studies

## 4.1 *Introduction*

Riparian and floodplain vegetation are created and maintained by hydrological and fluvial interactions including soil moisture supply and flood disturbances (Gurnell et al. 2014). Various characteristics of the flow regime affect assemblage patterns; such as seasonal occurrence of high and low flows, magnitude and frequency of flooding and its predictability (Nilsson and Svedmark 2002; Stromberg et al. 2007). Dams and reservoirs change the natural flow regime and consequently cause many alterations in riparian vegetation dynamics which may be assessed at different spatial and temporal scales (see Gurnell et al. 2014a; González del Tánago et al. 2015). Changes in riparian corridor dimensions and connectivity, including a reduction of diversity at river segment scale (Jansson et al. 2005; Uowolo et al. 2005), and changes in the distribution and complexity of vegetation patches and in germination and growth rates at river reach scale (Bunn and Arthington 2002; New and Xie 2008; Nilsson and Svedmark 2002), have been frequently associated with damming.

In Mediterranean regions flow regulation is frequently associated with irrigation (Hooke 2006; González del Tánago et al. 2012). This type of regulation changes the natural hydrologic regime of the river, reducing discharge during the wet season when the reservoir is being filled and increasing discharge during the dry season when irrigation takes place. This type of regulation frequently promotes growth of riparian vegetation as soil moisture levels are increased during summer as opposed to the natural drought that would otherwise limit growth. Additionally, flow regulation by large dams promotes the aging of riparian vegetation and decreases the potential recruitment of pioneer species. Spanish Mediterranean rivers are severely affected by flow regulation (Lobera et al. 2015), and future scenarios of global changes for water resources in the Mediterranean region suggest a progressive decline in average streamflow, including a decline in the frequency and magnitude of the most frequent floods (García Ruiz et al. 2011). Thus, understanding the effects of flow regulation on riparian ecosystems represents a big challenge in sustainable management of Spanish rivers.

In this work we present the case of two regulated Spanish rivers. Our aims were to:

i) describe the effects of flow regulation on floodplain and riparian vegetation

ii) Identify key factors/indicators that can be used to assess the level of degradation by damming

## 4.2 *Materials and Methods*

The study was conducted in the River Porma belonging to the Duero Basin (NW Spain) and in the River Guadalete that flows directly into the Atlantic Ocean in the Gulf of Cadiz (S Spain) (Figure 4.1). In each river a representative study site of the below dam hydro-morphological changes were selected.

*Site descriptions*

The River Porma has a drainage area of 1145 km$^2$ and a total length of 80 km. It has been regulated for irrigation purposes since 1968 by a large dam with a storage capacity of 106 % of the natural annual runoff (see Figure 4.1 for location). Our study site (Latitude 42º 49’ 41’’ Longitude 5º 19’ 32’’) encompassed a 2.5 km long reach, located 13 km downstream the dam. Here, the valley is partially confined and has an average longitudinal gradient of approximately 0.45%. The channel is artificially straight-sinuous (see Rinaldi et al, 2015 for planform typology) and has an average width of 25 m. Bed substratum is dominated by cobbles and coarse gravel.

The River Guadalete site (Latitude 36º 38’ 16’’ Longitude 5º 59’ 59’’) is located in the province of Cádiz, Southern Spain. This river has a total length of 159 km and a drainage area of 3,360 km$^2$. The flow regime is regulated by a series of large dams located in the main channel and in its main tributary, the River Majaceite (see Figure 4.1).

The study area is located downstream of the confluence of the River Majaceite and covers a river length of 2.5 km (Figure 4.1). Here, the river presents an average channel width of 15 m and exhibits a meandering pattern with fine sediment (i.e. fine sand and silt) bed substratum. The valley is partly confined and has an average longitudinal gradient of 0.06%.

*Hydrologic analysis*

Hydrologic analysis was carried out to characterise the flow regime based on daily mean flows from gauges close to the studied reaches. In the Porma River data came from gauging station id 2011 (http://hercules.cedex.es/anuarioaforos/default.asp), considering two periods: 1942-1967 as pre-dam period and 1968-2010 as post-dam. In Guadalete river data were taken by addition of flow data from Bornos dam (40 km upstream) and Guadalcacin dam (33 km). Pre- dam period considered in this case was 1965-1990 and post-dam 2000-2005.

Characteristics of mean, maximum and minimum annual flows were analyzed for the two periods in both rivers. Also, the recurrence interval of main floods (2-year, 5-year; 10-year flood values) was analyzed for the whole pre-dam and post-dam periods to assess the flow regulation effects on ordinary peak-flows. Criteria for the selection of hydrological indicators and procedures for analysing high-flow events are shown in Gonzalez del Tánago et al. (2015 a) and Gonzalez del Tánago et al. (2015b).

Floodplain and riparian vegetation changes were assessed by comparing pre-dam with post-dam conditions. Aerial photographs from 1956 and 2011 were analysed to evaluate the morphological evolution of the rivers. The 1956 photographs were black-and-white at 1:33 000 resolution scales and provided by the Technologic Agrarian Institute of Castilla y León, ITACyL (www.itacyl.es) in the case of the River Porma and by the Junta de Andalucía (www.ideandalucia.es) in the case of the River Guadalete. The 2011 aerial pictures were supplied by the National Geographic Institute of Spain (www.ign.es) with a resolution 0.25 and 0.5 m in all cases. Similar visual scales were used to cope with different air photographs resolution.

*Vegetation analysis*

We evaluated the total area of the river corridor, which includes the active channel and the riparian corridor, defined as the zone with distinct vegetation distinguishable from the surroundings. Within this corridor we measured area (i.e. total area, average width), longitudinal continuity (i.e. % of the length of the bankful channel margin abutting naturally functioning riparian vegetation Gurnell et al. 2014a), species composition, age structure and pioneer species. Woody vegetation community traits (abundance, composition and structure) were measured in each river.

In River Porma vegetation was sampled by three cross-transects (Martinez-Fernandez et al. 2014). All woody plants existing along each transect were analysed in terms of species composition and size class. For Salicacea species we distinguished seedlings and young individuals (height <1.5 m or diameter at breast height [d.b.h., 1.3 m] <2 cm), saplings (1.5 – 3 m height or 2 < d.b.h.< 10 cm) and adults (d.b.h.>10 cm). In River Guadalete five 150 m-long river reaches were selected downstream the dams where we assessed trends in composition and structure of woody riparian vegetation over the last decades. For this aim we identified all trees and shrubs and recorded their diameter at breast height within ten to twelve 10 x 10 m plots randomly distributed per reach. We applied the conceptual model of vegetation-hydromorphology interactions described by Gurnell et al. (2014b) (Figure 4.2) to assess the effects of flow regulation on the dominant hydrological and fluvial processes influencing riparian and floodplain vegetation.

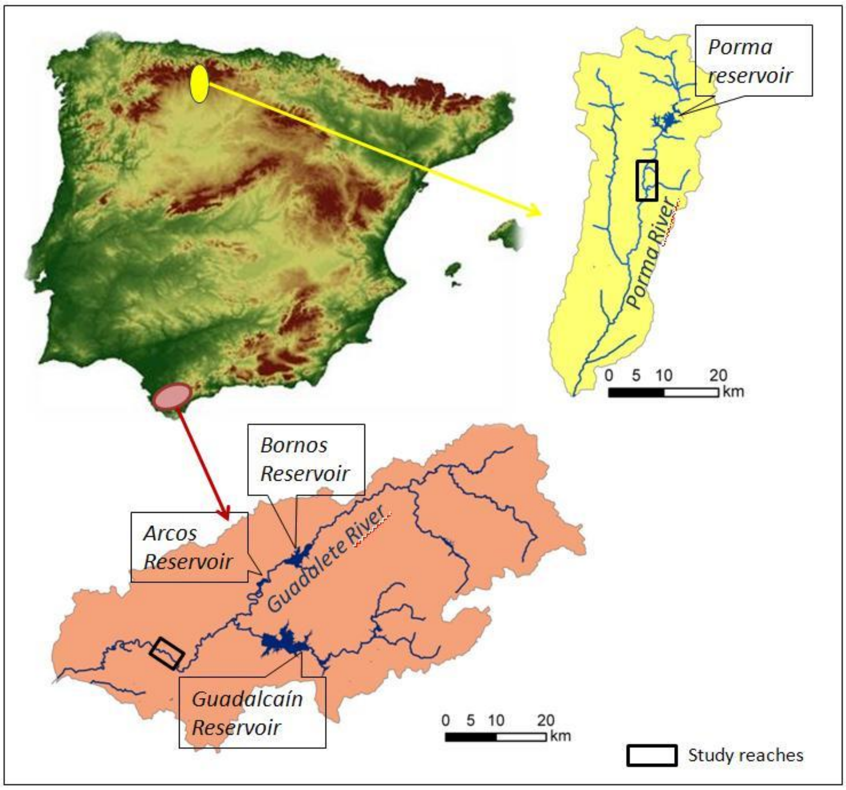


Figure 4.1 Location of the River Porma(North Spain) and the River Guadalete (South Spain).

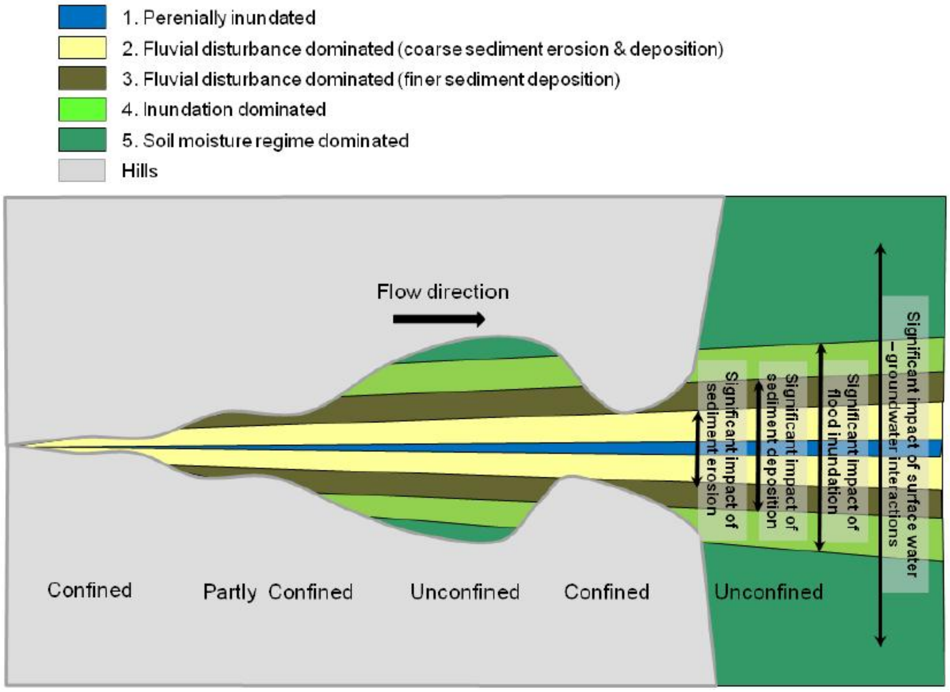


Figure 4.2 Lateral bands along the river continuum with variable confinement valley, indicating the dominant hydrological and fluvial processes influencing vegetation composition, growth performance and turnover (from Gurnell et al. 2014b).

## 4.3 Results

### 4.3.1 Case study 1: The River Porma

#### *Hydrologic alteration*

Figure 4.3 shows the hydrologic alteration caused by the dam in terms of seasonal timing, magnitude of maximum and minimum flows and recurrence intervals of peak flows. Since the operation of the dams the high-flow period corresponds to the irrigations months (June-August) and the low-flow period to the rainy season (November-May) (Figure 4.3A). The maximum annual discharge has decreased by 75.8%, and the minimum annual discharge has risen by 40.4% since the dam construction (Figure 4.3B)

The 2-years flood has dropped from 103.5 to 24.7 m3 $s^{-1}$ and maximum flow recurrence intervals have been drastically altered. The natural 10-year return period flood has been reduced by 74 % under the regulated regime (Figure 4.3C).

#### *Riparian and floodplain vegetation changes*

#### *Dimensions and longitudinal continuity*

In 1956, the river corridor of the studied reach represented a total area of 56 ha, which was mainly occupied by water surface and bare gravel bars, corresponding to a braided active channel. At this time the riparian corridor was almost negligible. In 2011 the river exhibited a single channel planform and the river corridor covered a total area of 27 ha, corresponding to 60% of the previous river corridor. The corridor was covered by woody riparian vegetation, forming a riparian corridor at both sides of the channel with an average width of 55.3 m.

Before the construction of the dam, only single trees and shrubs were present and the active channel was directly connected with the adjacent agricultural fields. The existing riparian corridor in 2011, despite being partially fragmented at both sides, presented a longitudinal coverage of woody vegetation of 67.5% on the left margin in the form of three vegetation patches and 72% on the right margin in the form of seven smaller vegetation patches.

### *Species composition and age structure*

Important changes in the relative abundance and age structure of the tree gallery have occurred. In 1956 the riparian vegetation was located in active gravel bars and likely represented pioneer species adapted to tolerate frequent disturbances (i.e. poplars and willows). At present, narrow mixed galleries border the existing channel with some old willows observed in sites that previously corresponded to channel banks, delineating the old river planform. Otherwise, the tree gallery near the channel banks is dominated by native species such as *Populus nigra, Salix fragilis*, *S. eleagnos* and *S. purpurea.* A relatively high occurrence frequency of late-seral species forming the overstory layer is found along the reach, such as *Fraxinus angustifolia, Crataegus monogyna*, *Ligustrum vulgare* and *Cornus sanguinea*. No invasive or exotic species have been observed within the riparian or floodplain area.

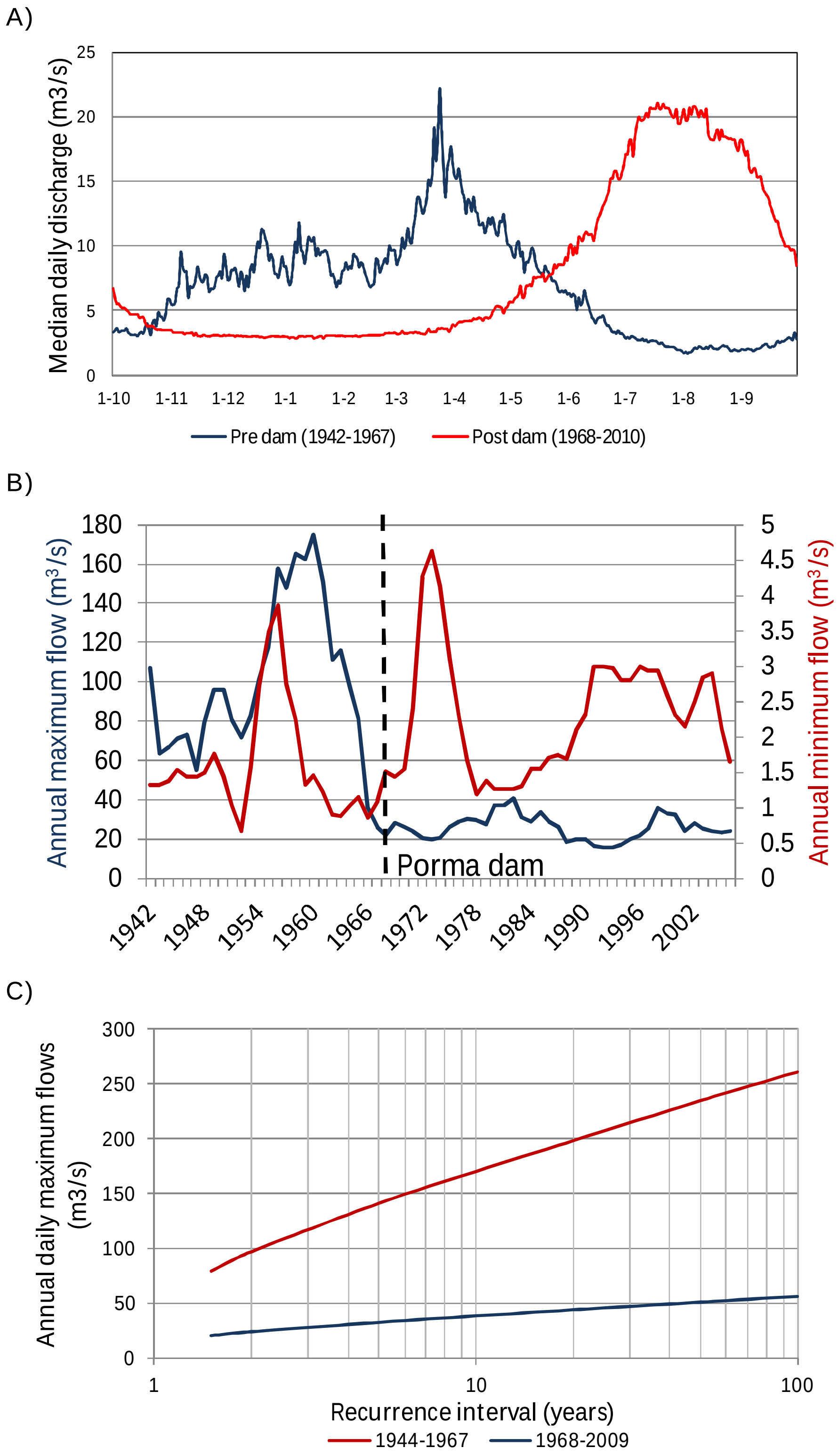


Figure 4.3 Changes in the flow regime of the River Porma. A) Median daily flows along the year. B) Annual maximum and minimum daily flows (3-year running average); vertical line indicates year when regulation started. C) Return periods of high-flow events.

Species diversity has increased as typically dryland species such as *Citysus scoparius* and *Thymus sp* have colonised the open spaces of the floodplain area (see Figure 4.4A), co-existing with the characteristic riparian flora. Mature forest occurs (see Figure 4.4B), with old pioneer species (*Populus nigra*, *Salix atrocinera*, *S. fragilis, S. eleagnos*) and late-seral species (*Crataegus monogyna*, *Ligustrum vulgare)* being the dominant species, and no recruitment of pioneer species was observed during the field surveys.

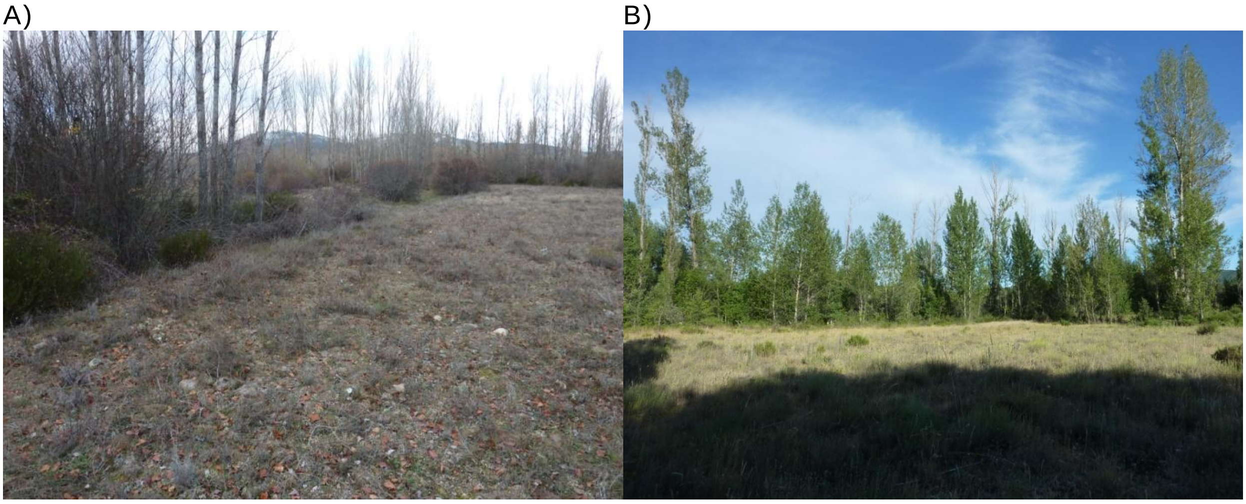


Figure 4.4 Partial views of the riparian and floodplain corridor of the studied reach.

*Application of the conceptual model of vegetation-hydro-morphology interactions*

Figure 4.5 shows changes in the extension and location of areas affected by hydrological processes. Comparing pre-dam and post-dam conditions, the area was dominated by perennially flowing water (Zone 1) and fluvial disturbance processes (Zones 2 and 3), representing 97% of the river corridor in 1956, whereas in 2011 these areas only occupied 48%. The area affected by high soil moisture levels showed the opposite pattern, representing 3% of the corridor in 1956 and 52% in 2011 (Figure 4.6).

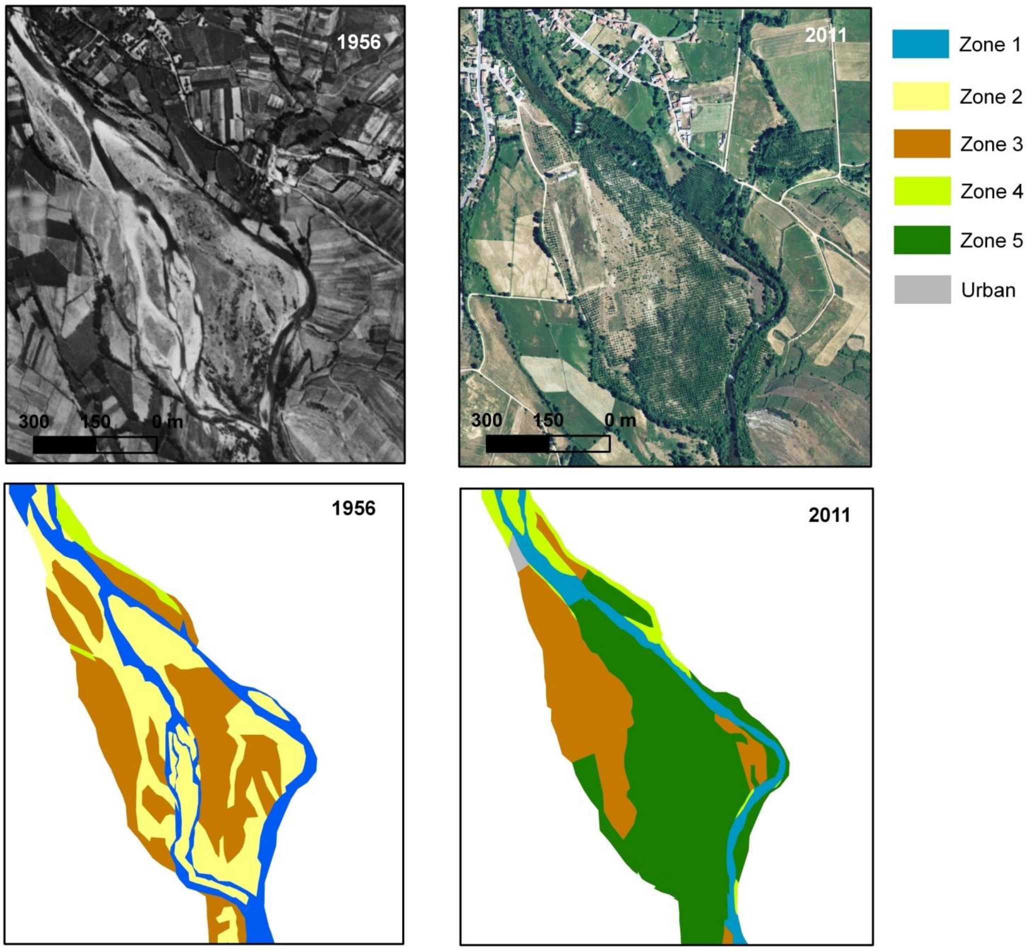


Figure 4.5 Partial view of the studied reach showing aerial photographs and interpretation of the results of the vegetation-hydrological interactions model (Gurnell et al. 2014b).

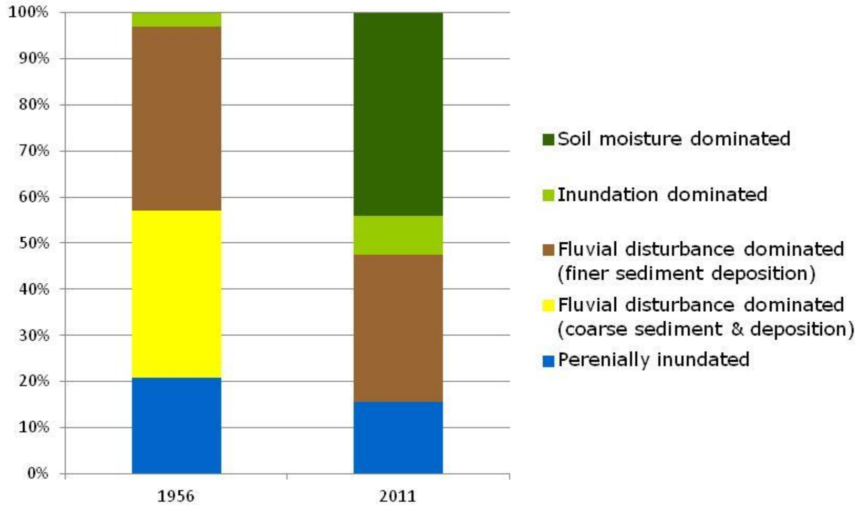


Figure 4.6 Temporal changes in the relative area of the distinct dominant processes within the river corridor (see Gurnell et al. 2014b for explanation of the dominant processes).

### 4.3.2 Case study II: The River Guadalete

#### *Hydrologic alteration*

The storage capacity of the reservoirs on the River Guadalete, and by inference the volume which can be regulated, equate to 420% of the annual natural runoff at the studied reach. The main purpose of the existing dams is to provide water for agricultural irrigation, similar to the situation on the River Porma. Consequently, a strong change in the seasonality of high and low flows is observed below the dams (Figure 4.7A). Magnitude and timing of the maximum and minimum flows have been altered as well. The 2 year return period flood at Las Juntas has decreased from 199.5 $m^3 s^{-1}$ (inflows to the Bornos plus Guadalcacín reservoirs) to 27.4 m3 $s^{-1}$ (outflows from Bornos plus Guadalcacín reservoirs), which is a reduction of nearly 85% compared to the unregulated situation (Figure 4.7B). The timing of the occurrence of the maximum annual discharge has also been modified. Before the operation of the dams (inflow values), the annual maximum flow occurred between December and March more than 80% of the time. Now, below the dams, 50% of the time the annual maximum flow occurred during the irrigation period between May and August.

### *Riparian and floodplain vegetation changes*

#### *Dimensions and longitudinal continuity*

The total area of the river corridor has decreased from 134 ha in 1956 to 64 ha today, corresponding to a reduction of approximately 50%.

Although the dimensions and average width of the riparian corridor have decreased, the river's longitudinal continuity has increased, which is likely related to the homogeneity of the flows. At present, a slender but continuous gallery of woody riparian vegetation is observed along both sides of the canal-like river channel, whereas in former times variability in width occurred and fragmented riparian bands were observed at both margins of the river, reflecting a much more active channel planform (Figure 4.8).

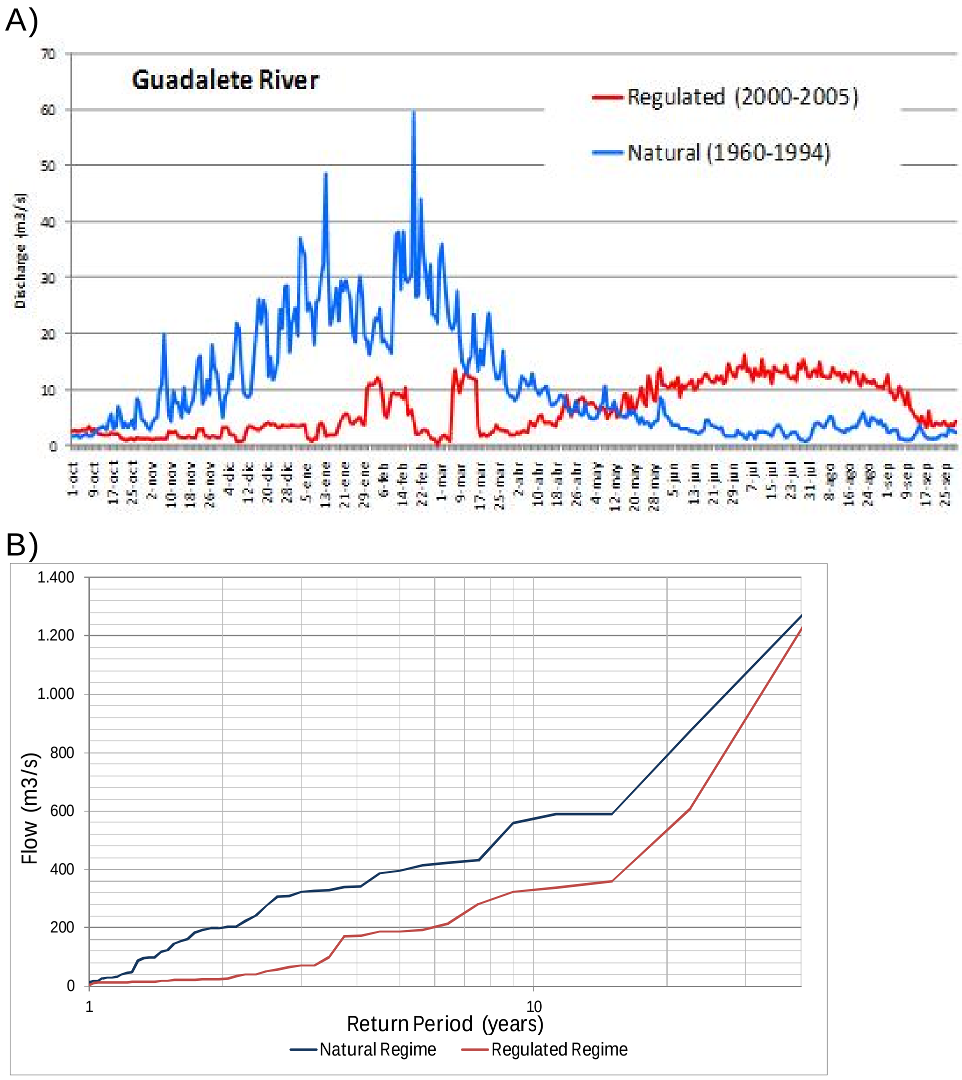


Figure 4.7 Changes in the flow regime in the River Guadalete. A) Mean daily flows along the year. B) Return periods of high-flow events.

### *Species composition and age structure*

A native riparian vegetation dominated by mixed forests of *Populus alba* and *Tamarix sp.* with presence of *Salix fragilis, Salix purpurea* and *Fraxinus angustifolia* has developed along discontinuous patches of the studied river reach, alternating with *Eucalyptus camaldulensis* stands. During the 1950s, *Eucalyptus camaldulensis* was often used in forestation programs around the River Guadalete; since then, this exotic species has gradually colonised the riparian zones and is now the dominant species along many river reaches.

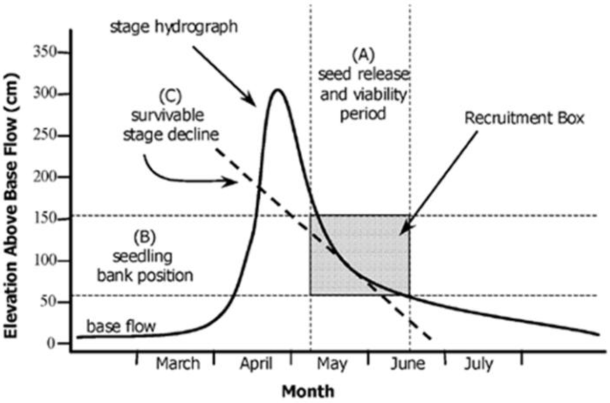
A mature, mostly dense, forest structure occurs along the riparian corridor, and recruitment has been only observed in the case of *Populus alba* and *Eucalyptus camaldulensis*.

Figure 4.8 Partial view of the River Guadalete showing changes in channel pattern and dimensions and the longitudinal continuity of the riparian corridor over time.

*Riparian Recruitment Assessment*

In order to understand the structure of the present riparian vegetation and the changes that have occurred after damming, we have studied the potential recruitment of the main riparian species associated with the current flow regime. For this purpose we used the 'recruitment box model' (Mahoney and Rood, 1998) to quantify more frequently flooded areas for representative river sections. We used flow records from the Bornos and Majaceite reservoirs adding daily in-flow data (natural regime) and out-flow daily data (regulated regime) for the period 1960-2005. The flow data series were transformed to a water level data series using a stage-flow curve for representative river sections.

For each annual series we only used data from March to July when riparian seeds of the reported species were available. Mahoney and Rood 1998 found that a water table decline of 4 cm per day was a threshold for the survivable of cottonwood seedlings in Canada. However, the survivable rate of water table decline varies with cottonwood species (Amlin & Rood, 2002) and is influenced by temperatures that control growth and the texture of the riparian substrate. Gonzalez et al. (2010a) found a survival of 25 % samplings under a drawdown of 5 cm/day on sandy substrate. We hypothesised that potential seed emergence and thus recruitment success in the Guadalete River are dependent on: a) a water level recession ratio less than 5 cm/day; b) maintenance of this ratio during at least five consecutive days. Therefore, a potential recruitment event took place only when smooth recession rates lasted more than five days (Figure 4.9).



Daily flows: 1960-2005

**Recruitment criteria** for flow events:

- Period of seed viability: **March-July**
- Recession ratio: **< 5cm/day**
- Occurrence occasion: more than **5 continuous days**

Figure 4.9 Application of the recruitment box model (Mahoney and Rood 1998) to the River Guadalete.

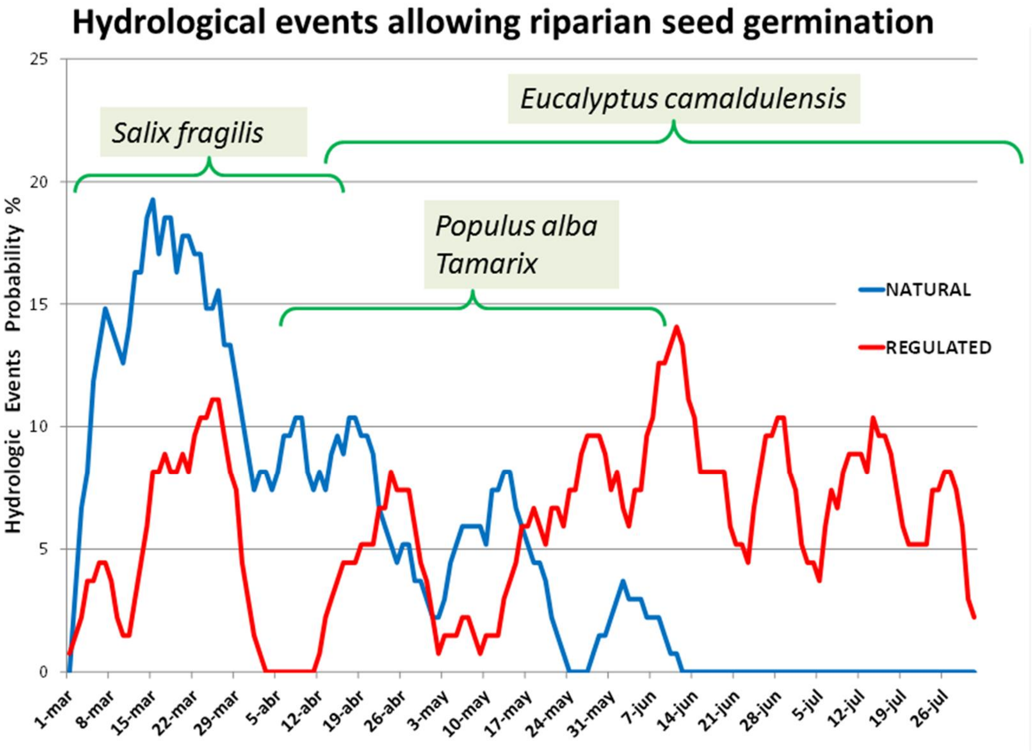


Figure 4.10 Probability of occurrence of potential recruitment events for riparian species in the River Guadalete under natural and regulated flow regimes. Green brackets indicate the seed dispersal period of the species. Recruitment probability of native species such as *Salix fragilis* is reduced, whereas recruitment probability of the exotic *Eucalyptus* is highly promoted under the current flow conditions.

In Figure 4.10 the probability of occurrence of riparian recruitment events are shown. Under non-regulated (i.e. natural) conditions, recruitment probability peaked in March and decreased sharply towards mid-June. Native species disperse along this period: willows in early spring and white poplars at the end of spring. Under regulated flows, there is recruitment potential the whole year round. As *Eucalyptus camaldulensis* flourishes several times throughout the year, even in summer, this particular species is able to benefit from the expanded potential recruitment period.

*Application of the conceptual model of vegetation-hydromorphology interactions*

Figure 4.11 shows the temporal changes that occurred in the riparian corridor. In 1956 perennially-flowing water (Zone 1) and fluvial disturbance processes (Zones 2 and 3) represented 54% of the riparian zone, but in 2011 this percentage had declined to 10%. In contrast, the area affected by soil moisture regime, occupying in 1956 46% of the area, has expanded to 90% (Figure 4.12).

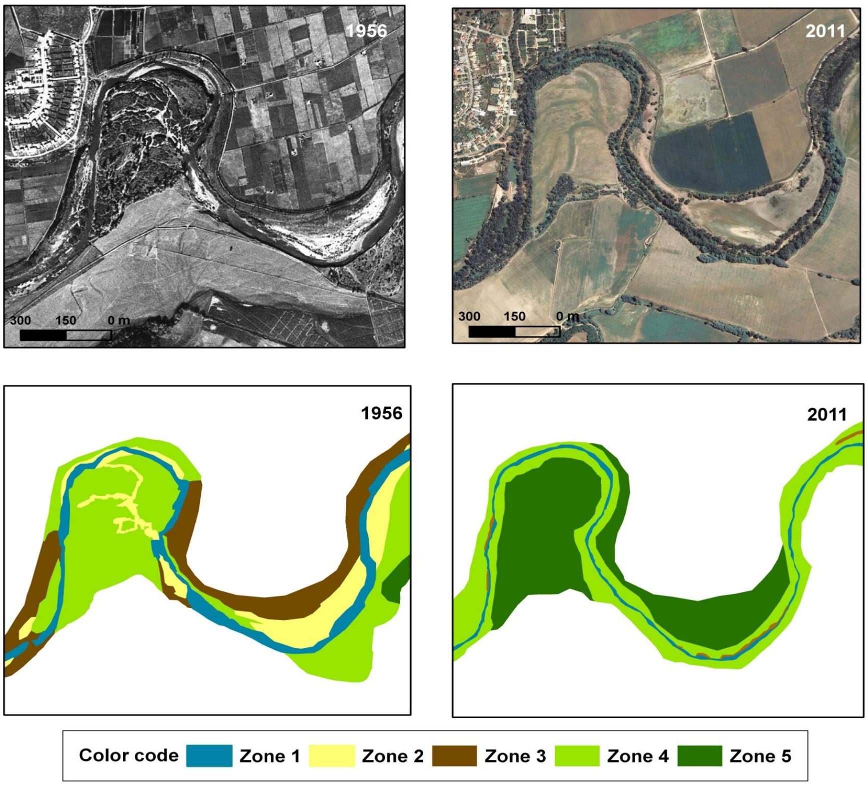


Figure 4.11 Partial view of the Guadalete reach in the studied periods (above) and interpretation of the conceptual model results following Gurnell et al. (2014b) of the interactions between vegetation and hydrological processes (below).

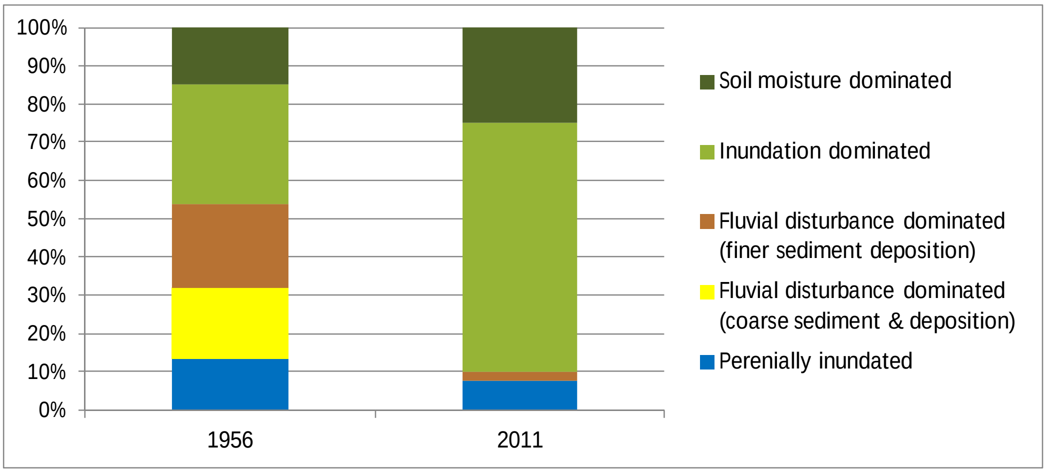


Figure 4.12 Temporal changes in the relative area occupied by the different zones in the studied River Guadalete River (see Gurnell et al. 2014b for explanation of the zones).

## *4.4 Discussion and conclusions*

Riparian and floodplain vegetation has been affected by flow regulation in the two case studies in a relatively similar way. The dimensions and continuity of the riparian corridor have increased under regulated flows conditions as a consequence of vegetation encroachment on the existing gravel bars, probably because these areas are less frequently affected by floods. Similar effects of flow regulation have been reported in other rivers under flow regulation schemes (Gordon and Meentemeyer 2006; Poff and Zimmerman 2010; Belmar et al. 2013), but also in Spanish regulated rivers (González *et al.* 2010b, Bejarano et al. 2012, Garofano et al. 2013, Martinez-Fernandez *et al.*, 2014).

In the River Porma, the species composition and structure have changed from a more active planform dominated by young pioneer species to a mature forest with a dense overstory of late-seral species near the channel banks. This tendency may be associated with the simultaneous geomorphic changes observed in the channel where the reduction of flood disturbance frequency below the dam has produced more stable and disconnected river banks.

In the case of the River Guadalete, the temporal changes associated with flow regulation have promoted the exotic riparian species *Eucalyptus camaldulensis*. In this case, the reduced recruitment potential under regulated flows during the seed dispersal period of native species may be associated with the expansion of the *Eucalyptus*, which has also been favoured by its plantation along the adjacent roads. Similar results of significant changes in the potential of cottonwood recruitment associated with changes in timing, magnitude and frequency of flows have been found by many authors (Williams and Cooper 2005; Burke et al. 2009).

As peaks flows have decreased their intensity and frequency in both rivers, the potential areas being affected by relatively high velocities and shear stress and by sufficient flowing water to mobilise, transport and deposit sediments have decreased (Zones 1, 2 and 3 in the applied conceptual model). As a direct consequence of flow regulation, areas affected by fluvial disturbances under pre-dam conditions have turned into areas dominated by hydrologic processes with negligible sediment dynamics during inundation

(Zones 4 and 5).

The variables used to characterise riparian and floodplain vegetation changes reflect the geomorphic development of the channels since the operation of the dams. Dimensions and continuity of the riparian corridor may be primarily linked to the reduced frequency of peak flows, whereas changes in species composition and structure may be associated with changes in the timing and magnitude of flows.

The conceptual model proposed by Gurnell et al. (2014b) acts as an additional, very useful tool for quantifying riparian and floodplain vegetation changes associated with flow regulation as it explicitly includes changes in the main hydrological processes that are likely related to regulated flows (i.e. decrease in magnitude and frequency of flood disturbance with associated effects on soil moisture supply and erosion and deposition of coarse and fine sediments).

Finally, some key factors or indicators that could be used to assess the level of degradation from flow regulation are: a) absence or decrease of recruitment; b) presence of old individuals of pioneer species at sites of the riparian zone away from water margins, reflecting changes in the position of active channel banks; c) abundance of late-seral species in the proximal parts of the riparian zones; d) mature forest structure dominating the riparian vegetation; e) presence of nonnative invasive species; f) areas of former riparian zones occupied by agriculture, poplar plantations or other anthropic uses taking benefit of less frequently flooded conditions.

## 4.5 References

Amlin, N. M., & Rood, S. B. (2002). Comparative tolerances of riparian willows and cottonwoods to water-table decline. Wetlands, 22(2), 338-346.

Bejarano, MD, M. Gonzalez del Tanago, D. Garcia de Jalon, M. Marchamalo, A. Sordo-Ward & J. Solana-Gutierrez. (2012) Responses of riparian guilds to flow alterations in a Mediterranean stream. Journal of Vegetation Science 23, 443-58. Doi: 10.1111/j.1654-1103.2011.01360.x

Belmar, O., D. Brunoa, F. Martínez-Capel, J. Barquín, and J. Velasco (2013). Effects of flow regime alteration on fluvial habitats and riparian quality in a semiarid Mediterranean basin. Ecological Indicators 30: 52–64.

Burke, M., K. Jorde and J. M. Buffington (2009). Application of a hierarchical framework for assessing environmental impacts of dam operation: changes in streamflow, bed mobility and recruitment of riparian trees in a western North American river. Journal of Environmental Management 90: S224-S236.

Bunn, S. E. and A. H. Arthington (2002). Basic principles and ecological consequences of altered flow regimes for aquatic biodiversity. Environmental management 30: 492-507.

Hooke, J. M. (2006) Human impacts on fluvial systems in the Mediterranean region. Geomorphology 79: 311-335.

Garófano□Gómez, V., Martínez□Capel, F., Bertoldi, W., Gurnell, A., Estornell, J., & Segura□Beltrán, F. (2013). Six decades of changes in the riparian corridor of a

Mediterranean river: a synthetic analysis based on historical data sources. Ecohydrology, 6(4), 536-553.

García-Ruiz, J. M., J. I. López-Moreno, S. M. Vicente-Serrano, T. Lasanta–Martínez and S. Beguería (2011). Mediterranean water resources in a global change scenario. Earth-Science Review, 105: 121-139.

González del Tánago, M., D. García de Jalón, Román, M. (2012). River restoration in Spain: Theoretical and practical approach in the context of the European Water Framework Directive. Environmental Management 50, 123-139.

González del Tánago M, Martínez-Fernández V, García de Jalón D (2015a) Diagnosing problems produced by flow regulation in Southern European rivers: The Porma and Curueño Rivers (Duero Basin, NW Spain). Aquatic Sciences, accepted.

González del Tánago, M., Bejarano, M.D., García de Jalón, D. & Schmidt, J. (2015b). Biogeomorphic responses to flow regulation and fine sediment supply in Mediterranean Streams (The Guadalete River, Southern Spain). Journal of Hydrology, DOI: 10.1016/j.jhydrol.2015.06.065

González, E., Comín, F. A., & Muller, E. (2010). Seed dispersal, germination and early seedling establishment of Populus alba L. under simulated water table declines in different substrates. Trees, 24(1), 151-163.

González, E., González-Sanchis, M., Cabezas, Á., Comín, F. A., & Muller, E. (2010). Recent changes in the riparian forest of a large regulated Mediterranean river: implications for management. Environmental Management, 45(4), 669-681.

Gordon, E. and R. K. Meentemeyer (2006). Effects of dam operation and land use on stream channel morphology and riparian vegetation. Geomorphology 82: 412-429.

Gurnell A. M., B. Belletti, S. Bizzi, B. Blamauer, G. Braca, T. Buijse, M. Bussettini, B. Camenen, F. Comiti, L. Demarchi, D. García de Jalón, M. González del Tánago, R. C. Grabowski, I. D. M. Gunn, H. Habersack, D. Hendriks, A. Henshaw, M. Klösch, B. Lastoria, A. Latapie, P. Marcinkowski, V. Martínez-Fernández, E. Mosselman, J. O. Mountford, L. Nardi, T. Okruszko, M. T. O'Hare, M. Palma, C. Percopo, M. Rinaldi, N. Surian, C. Weissteiner and L. Ziliani (2014a) A hierarchical multi-scale framework and indicators of hydromorphological processes and forms. Deliverable 2.1, a report in four parts of REFORM (REstoring rivers FOR effective catchment Management), a Collaborative project (large-scale integrating project) funded by the European Commission within the 7th Framework Programme under Grant Agreement 282656.

Gurnell, A. M., M. González Del Tánago, M. T. O'Hare, M. Van Oorschot, B. Belletti, T. Buijse, D. García De Jalón, R. Grabowski, D. Hendriks, O. Mountford, M. Rinaldi, L. Solari, M. Szewczyk and A. Vargas-Luna (2014-b) Influence of Natural Hydromorphological Dynamics on Biota and Ecosystem Function. Deliverable 2.2, a report in two parts of REFORM (REstoring rivers FOR effective catchment Management), a Collaborative project (large-scale integrating project) funded by the European Commission within the 7th Framework Programme under Grant Agreement 282656.

Jansson, R., U. Zinko, D. M. Merritt and C. Nilsson (2005). Hydrochory increases riparian plant species richness: a comparison between a free-flowing and a regulated river. Journal of Ecology 93: 1094-1103.

Lobera, G., P. Besné, D. Vericat, J. A. López-Tarazón, A. Tena, I. Aristi and R. J. Batalla (2015). Geomorphic status of regulated rivers in the Iberian Peninsula. Science of the Total Environment 508: 101-114.

Mahoney, J.M. and S. B. Rood (1998). Streamflow requirements for cottonwood seedling recruitment—an integrative model. Wetlands 18: 634-645.

Martínez-Fernandez V, Gonzalez del Tanago M, Maroto J, Garcia de Jalon D. (2014) Flow regulation and vegetation changes in gravel bed rivers (Upper Esla river, NW Spain). Procs. 10th International Symposium on Ecohydraulics. Trondheim.

New. T. and Z. Xie (2008). Impacts of large dams on riparian vegetation: applying global experience to the case of China's Three Gorges Dam. Biodiversity Conservation 17: 3149–3163.

Nilsson, C. and M. Svedmark (2002)- Basic principles and ecological consequences of changing water regimes: riparian plant communities. Environmental Management 30: 468-480.

Poff, N. L. and J. K. Zimmerman (2010). Ecological responses to altered flow regimes: a literature review to inform the science and management of environmental flows. Freshwater Biology 55: 194-205.

Rinaldi, M., A. M. Gurnell, M. González del Tánago, M. Bussettini and D. Hendriks (2015) Classification and characterization of river morphology and hydrology to support management and restoration. Aquatic Sciences, submitted

Stella, J. C., Battles, J. J., McBride, J. R., & Orr, B. K. (2010). Riparian seedling mortality from simulated water table recession, and the design of sustainable flow regimes on regulated rivers. Restoration Ecology, 18(s2), 284-294.

Stromberg, J. C., V. B. Beauchamp, M. D. Dixon, S. J. Lite and C. Paradzick (2007) Importance of low-flow and high-flow characteristics to restoration of riparian vegetation along rivers in arid south-western United States. Freshwater Biology 52: 651-679.

Uowolo, A. L., D. Binkley and E. C. Adair (2005). Plant diversity in riparian forests in northwest Colorado: effects of time and river regulation. Forest Ecology and Management 218: 107-114.

Williams, C. A. and D. J. Cooper (2005) Mechanisms of riparian cottonwood decline along regulated rivers. Ecosystems 8: 382–395.

# 5 Riparian and Floodplain Vegetation Hydromorphology Interactions - Austrian Case Studies

## 5.1 Introduction

After the last Ice Age, approximately 10,000 years ago, riparian forests arose all over Europe as highly dynamic transitional environments between terrestrial and freshwater ecosystems on river floodplains. Riparian zones acted as forested corridors through the European landscape, enhancing connectivity within and between other forest patches. Human colonisation of river valleys all over Europe started several thousands of years ago. Forest clearance and agricultural use within the upper catchments resulted in long-lasting alterations of hydrological regimes and sediment transport. Over centuries, human activities along the river corridors focused on the reclamation of floodplain areas for agricultural use and on the control of floods achieved mainly by the construction of embankments and dams. The improvement to timber floating and navigation was a result of the straightening and channelisation of water courses. Systematic river channelisation and dyking in Europe reached its zenith during the 18th and 19th century and finally led to a loss of connection between rivers and floodplain areas.

Today, 90% of the area of riparian forests in Europe has disappeared from the landscape (Hughes (ed.), 2003). Remaining riparian forest fragments all over Europe often are in a critical condition, mainly because of reduced flooding frequency and the increased depth of the water table as a result of river channelisation and bed erosion.

The natural processes of flood disturbance, sedimentation and erosion processes are now disrupted. Riparian forests are considered to be one of Europe's most threatened natural ecosystems and are listed in Annex I of the European Habitats Directive as being a 'priority forest habitat type'. European initiatives such as NATURA 2000, the European Green Belt and the European Umbrella Initiative “RIVERS NEED SPACE” (INTERREG III B CADSES and the PHARE Programme) mainly focus on an integrated river management policy to give rivers more space.

A map based on data provided by the United Nations Environment Programme-World Conservation Monitoring Centre (UNEPWCMC) illustrated in Figure 5.1 shows that most remaining riparian forests are located in central and eastern Europe (Moss and Monstadt 2008). Only a few riparian forests have survived as highly dynamic natural ecosystems with periodic floods, channel movement and natural sedimentation and erosion processes. One of Europe’s best remaining examples is located in the area of the former Iron Curtain, the Danube-Morava-Thaya National Park. The study site is located there, in the lower Traun valley in Upper Austria.

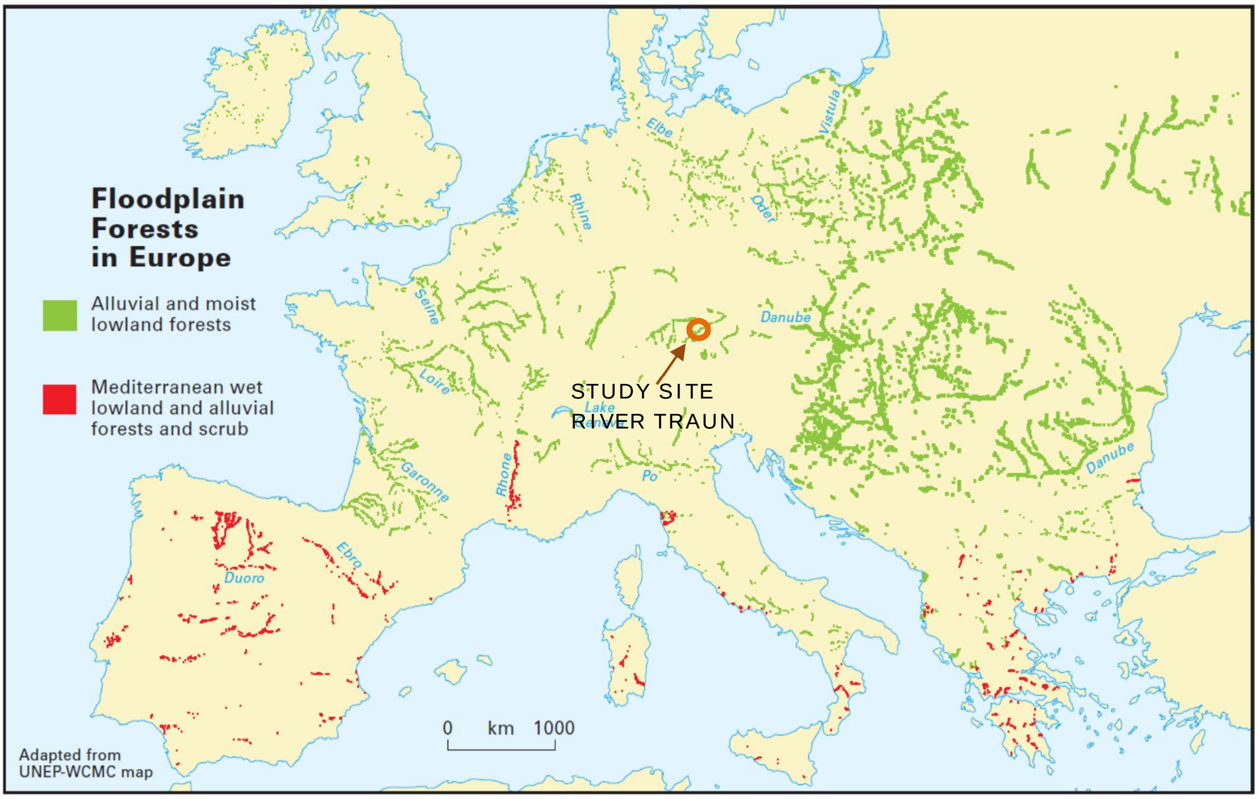


Figure 5.1 Floodplain forests in Europe (modified from Moss and Monstadt 2008).

The study site is part of ecoregion 9 (central highlands) and the bioregion of the Bavarian-Austrian alpine foothills. According to System A of the EU-Water Framework directive, 5% of the water bodies within this bioregion are classified as large rivers with a catchment area larger than 1000 km². Over 80% of the large rivers within this bioregion belong to the same river type as the study site at the River Traun (Wimmer et al. 2007). Consequently, the River Traun study site is typical for the area and valuable knowledge can therefore be generated on how anthropogenic disturbance affects large rivers in the foreland areas of Alpine regions.

Applying the river typology classification developed within D2.1, Part 1, chapter 7 (Gurnell et al. 2014a), the degraded river is classified as type 13, sinuous – straight gravel-bed river, whereas the historical river is classified as type 9, island braided gravel-bed river.

Vegetation zones of rivers of the alpine foothills of Central Europe have been classified by (Ellenberg and Leuschner 2010) and are applicable to the Traun. These zones have a high degree of similarity to the hydromorphologically defined zones presented in Reform Deliverable 2.2, part 1, where different hydromorphological characteristics of the river corridor have been used to define five zones based on the magnitude of inundation, fine sediment deposition, or sediment erosion and deposition. The five zones were characterised as follows: 1 perennially inundated, 2 fluvial disturbance dominated (coarse sediment erosion and deposition), 3 fluvial disturbance dominated (finer sediment deposition), 4 inundation dominated and 5 soil moisture regime dominated (Gurnell et al. 2014b). This classification system allows fluvial geomorphological and hydrological processes to be linked to vegetation changes.

We described how it was possible to identify the impacts of hydromorphological degradation on riparian ecosystems on the Traun system, by first characterising the historic evidence of channel changes in response to human alterations and combining it with evidence of associated vegetation changes and numeric models.

## 5.2 Materials and Methods

### 5.2.1 Study site

With a total length of 73 km, the River Traun drains a catchment of 2770 km². The study site of a 6th order alpine stream is part of the Colline altitude level (Moog and Wimmer 1990). It has a total length of approx. 5.5 km and a total project area of approx. 730 ha (Wimmer et al. 2007).

Data are available from the beginning of the 17th century and onwards. A detailed technical survey, local maps, cross section data and a longitudinal profile of the river in 1885 were used to generate a digital terrain model of the natural river almost unaffected by human impacts at this time.

Figure 5.2 Study site at the River Traun, 2014.

### 5.2.2 Hydromorphological data

*Hydrology*

The main reference of hydrological data is the Austrian Hydrographic Yearbook of the Austrian Federal Ministry of Agriculture, Forestry, Environment and Water Management. Hydrological data were recorded daily from the gauging station Wels-Lichtenegg and formed the basis for further calculations and evaluations. The observation period was set from 1890 to 2010. Records from 1938 to 2010 were directly available as flow data. Data prior to 1938 were available as water level records that were converted into flow data using revised rating curves of 1904 (already after river regulation), 1947, 1948 and 1949. The hydrological classification corresponds with a moderate Nival Flow Regime with a distinct annual character. The key process is snowmelt with a maximum in May or June. Characteristic discharges were used as input for hydraulic modelling and basis for further analysis.

Table 5.1 Characteristic discharges and link to applied vegetation zones (cf. Table 5.3).

| Abbr. | [$m^3s^{-1}$] | Definition |
|---|---|---|
| $NQ_t$ | 27.7 | lowest mean daily flow value in the observed period |
| $MJNQ_t$ | 41.4 | mean annual daily low flow in an annual series t (veg. zone 1) |
| MQ | 134 | mean flow value in the observed period (veg. zone 2) |
| $HQ_1$ | 500 | flood event with a 1-year return period (veg. zone 3) |
| $HQ_{100}$ | 1500 | flood event with a 100-year return period (veg. zone 4) |
| $HQ_{300}$ | 1750 | flood event with a 300-year return period |

*Fluvial geomorphology*

A detailed technical survey, local maps, cross section data and a longitudinal profile of the natural situation of 1885 before river channelization were available which were used to generate a digital terrain model of the natural river situation. The basis for the current hydromorphological situation is provided using echo-sounding data, terrestrial measuring data as well as analogue plans of the study site area.

*Groundwater*

Mainly glacial and post glacial quaternary deposits are superimposed on the aquiclude consisting of molasses sediment in the study site (Blaschke 2001). In the study site, many springs rise in the southwest of the weir, originating from the fluvioglacial terraces on the marl at the boarder of the Traun-Enns-Plate. These springs keep the groundwater table within the area of the meadow area called Fischlhamer Au higher than in the other parts of the study site. The lower part of the Fischlhamer Au is influenced by the weir that is located just a few hundred metres downstream. The groundwater table of this meadow is stabilised by the tailback caused by the weir and, in consequence, characteristic fluctuations and low water levels do not occur in this area. Increasing impoundments in the Fischlhamer Au still take place due to high flow in the tributary running throughout this meadow. Heighten backwater of the weir or overflow of upstream dams also result in a higher water level in this area. (Schuster 1992).

### 5.2.3 Vegetation data

It is known, that in the 19th century a major part of the riparian forest within the project area consisted of soft wood, mostly willows. Even though the former distribution of habitat types cannot be assessed exactly, the abundance and approximate distribution of the habitat types can be determined using historical data in terms of herbaria, chronicles, taxa lists, reports and the Franziscan Cadastre. In 1817 the emperor Franz I. of Austria introduced a patent for property taxing. I The 'Franziscean Cadastre' recorded accurate property boundaries and detailed land use information, including Figure 8.8 important information about the vegetation structure. The Franziscean Cadastre show the natural vegetation situation of the riparian area at the Traun river up to1885 (Strauch 1992c).

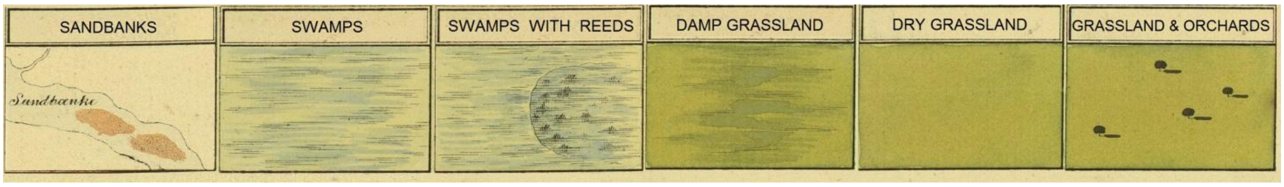


Figure 5.3 Extract from the key of the Franziscean Cadastre

The current vegetation is well documented in several field surveys and studies (Essl and Schuster 2001; Hauser 1998; Strauch 1992c; Strauch 1992d; Hüttmeir 1992; Muhar et al. 1987 and others. The report of Essl and Schuster (vegetation list see Annex of this deliverable) referred explicitly to the Natura 2000 Directive and in particular the habitat types listed in the Habitats and Birds Directives.

*Natural Habitat Types and Species of conservation interest*

Natural habitat types of Community interest, whose conservation requires the designation of special areas of conservation according to Annex I of the Habitats Directive (Council directive on the conservation of natural habitats and of wild fauna and flora), appear in the study site. The same coding system can also be found as part of Annex B “Riparian and Aquatic Plant Communities of Europe” in D2.2, part 1 (Gurnell et al. 2014b).

The flagship species for nature conservation of specific habitat types in the project area were defined by Essl and Schuster (2001).

The main characteristics of these species are their occurrence in one or a few specific habitats, which makes them valuable species for assessments.

Table 5.2 Natural habitat types of the study site (Habitats Directive 1992; Essl and Schuster 2001).

| Code | Habitat Type |
|---|---|
| Priority habitat types | |
| 3140 | Hard oligo-mesotrophic waters with benthic vegetation of *Chara* spp. |
| 6210 | Semi-natural dry grasslands and scrubland facies on calcareous substrates (*Festuco-Brometalia*) (important orchid sites) |
| 91E0 | Alluvial forests with *Alnus glutinosa* and *Fraxinus excelsior* (*Alno-Padion*, *Alnion incanae*, *Salicion albae*) |
| Non-priority habitat types | |
| 3150 | Natural eutrophic lakes with *Magnopotamion* or *Hydrocharition* — type vegetation |
| 3220 | Alpine rivers and the herbaceous vegetation along their banks |
| 3230 | Alpine rivers and their ligneous vegetation with *Myricaria germanica* |
| 3240 | Alpine rivers and their ligneous vegetation with *Salix elaeagnos* |
| 6410 | Molinia meadows on calcareous, peaty or clayey-silt-laden soils (*Molinion caeruleae*) |
| 6430 | Hydrophilous tall herb fringe communities of plains and of the montane to alpine levels |
| 6510 | Lowland hay meadows (*Alopecurus pratensis, Sanguisorba officinalis*) |
| 7230 | Alkaline fens |
| 9170 | *Galio-Carpinetum* oak-hornbeam forests |
| 91F0 | Riparian mixed forests of *Quercus robur*, *Ulmus laevis* and *Ulmus minor*, *Fraxinus excelsior* or *Fraxinus angustifolia*, along the great rivers (*Ulmenion minoris*) |

*Vegetation data compilation*

Riparian forests form complex vegetation mosaics that are subject to high diversification and constant renewal. A compilation of taxa lists, herbaria, chronicles and other literature prepared by Strauch (1992d) formed the basis for the historical and current characterisation of the vegetation in the study site. The data was expanded and supplemented using vegetation mapping, field surveys and studies of the project area conducted by Muhar et al. (1987), Hüttmeir (1992), Hauser (1998) and Essl and Schuster (2001).

To cover the riparian zone and the floodplain of the study site area, two datasets were created. The first dataset included data for 41 specific sample points within the study site area. An overview of the location of the sampling sites is given in Figure 5.4. The second dataset included relevant data from an extensive vegetation survey published by Strauch (1992d), including also historical (19th century) vegetation records from the lower Traun valley. Taxa were categorised as belonging to the main areas of heathland, higher and lower riparian forests and/or the lower terraces of the River Traun in the southeast of the study site. Furthermore, the frequency of the taxa, risk of extinction and risk according to the Red List classification as well as anthropogenic influence on their distribution are reported in addition to additional relevant information.

All taxa registered can be found in the appendix to this deliverable.

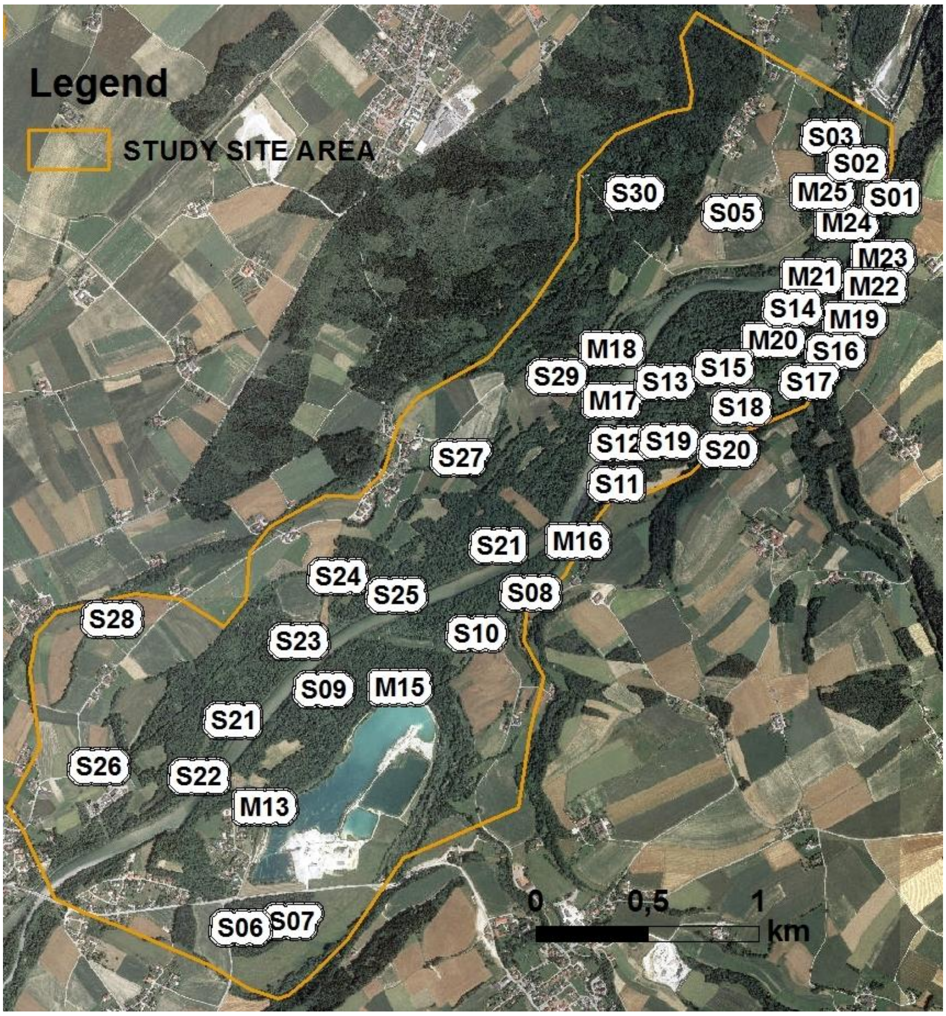


Figure 5.4 Vegetation sampling points within the River Traun study site.

The species composition recorded in the latest vegetation surveys within the study site show broad variability within the large inundated areas at a flood event with a high return period. Due to the river regulation measures the inundated area at mean flow and at an annual flood event was reduced dramatically. Therefore, only very few points of vegetation surveys and detailed data are available for wetted areas at an annual flood or mean flow within the study site area.

Table 5.3 Study site-related flagship species for vegetation habitats.

| Habitat | Flagship species |
|---|---|
| Semi-dry grassland | *Anacamptis pyramidalis, Ophrys holosericea, Orchis tridentata* |
| Humid grassland | *Carex davalliana, Epipactis palustris, Taraxacum palustre agg., Ophioglossum vulgatum* |
| Pioneer bushes of flowing waters | *Myricaria germanica* |
| Forests | *Cypripedium calceolus* |

### 5.2.4 Hydraulic modelling

Datasets described in 5.2.2 form the basis of a digital terrain model and were used to generate a hydraulic model using the software SMS (Surface-Water Modelling System) and Hydro_AS_2D. The model was adjusted using observations and records from the flood in 2002 (1568 $m^3s^{-1}$). Numerous simulations were performed at flows from low flow (41.4 $m^3s^{-1}$) up to flood events with a 300 years return period (1750 $m^3s^{-1}$).

## 5.3 Results

### 5.3.1 Pressure timeline

Rich riparian forests began to develop around the River Traun after the last ice age. The river bed with its multiple branches and the riparian landscape were continuously changed by floods over several thousand years. The lower Traun valley has been inhabited and used by humans for thousands of years. To a certain degree the anthropogenic-induced changes have been ameliorated by nature. On the lower Traun the river widths ranged up to 700 metres and several hundred side arms were formed throughout the valley. The multiple branches of the river in the lower riparian zone made agricultural use more difficult than in , the higher riparian zones which were already cultivated by 1825 (Strauch 1992b).

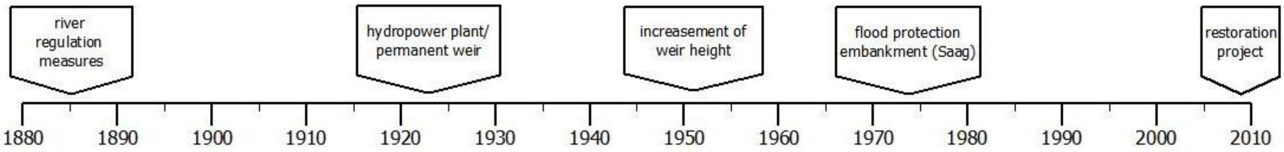


Figure 5.5 Timeline of significant changes of the study site at the River Traun.

After 1885 human influence increased on the river system, in terms of river regulations, change of land use, gravel extraction and hydropower plants, the river has changed substantially. In summary, the request for full use and the technical possibilities of the 20st century have led to a dramatic hydromorphological degradation (Speta 1992). Especially areas of riparian forests and grassland were affected by the developments in the whole lower Traun valley as illustrated in Figure 5.6.

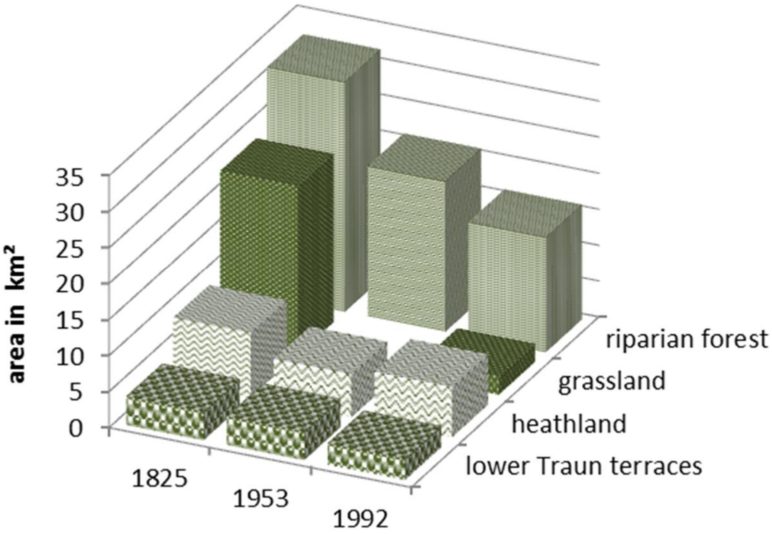


Figure 5.6 Development of areas of riparian forest, grassland (no data available for 1953), heathland and the lower terraces of the River Traun of the lower Traun valley from 1825 until 1992 according to Strauch (1992b, modified).

The rapid alterations of the riparian forests are described in an article of a hunting journal written by Gemböck in 1886. Due to the increasing technological progress of river regulation, side arms are cut off the main channel and periodical inundations, erosion and sedimentation processes have therefore disappeared. The areas have lost their wide variety of plant species. In 1924 a newspaper from the nearby town of Wels described the regulation of the River Traun as a serious disadvantage for the river flora and fauna and referred to the loss of side branches and wetted areas. The lowering of the groundwater table and the consequent reduction and partial die-off of willows are also described in this article (Roth 1924). Nowadays, a monotonous, straightened single bed river characterises the landscape of the lower Traun valley. Starting in the 1880s, additional, radical river regulation was performed in the study site area over the whole length of the River Traun. The construction of a hydropower plant and the installation of a permanent weir at the lower end of the study site area were completed in the early 1920s. A flood protection embankment was established in the 1970s to protect a settlement area called Saag.

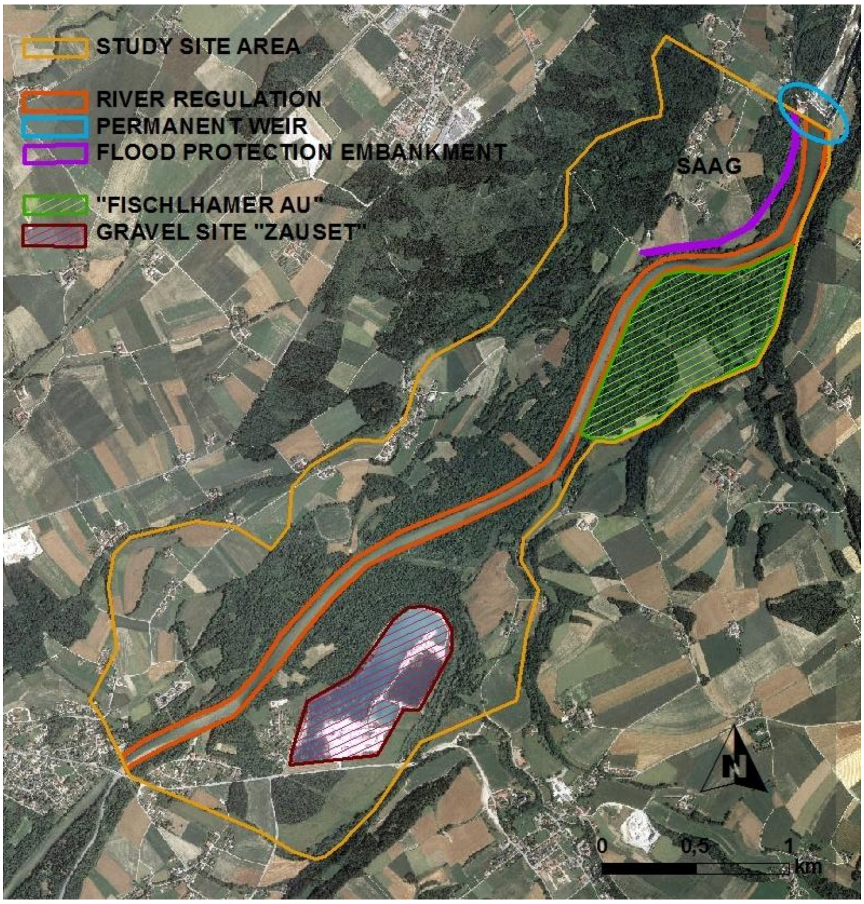


Figure 5.7 Study site overview showing significant changes caused by anthropogenic influences and relevant areas (Fischlhamer Au).

In addition to the three major hydromorphological alterations shown in Figure 5.5 and Figure 5.7, other multiple-pressure effect include the changes in land used for agriculture and forestry, extensive gravel and sediment extraction. Gravel sites cover 3.7% (= 6.5 $km^2$) of the total area of the lower Traun valley.

*Changes to hydromorphology*

Before the extensive river regulation measures at the end of the 19th century, the riverbeds of the multiple branches of the River Traun were embedded in the floodplain to a depth of approximately 2 metres. Even small floods caused large inundations of the floodplain. This situation changed rapidly after the river regulation in the 1980s. Until 1943 the water level dropped by 4.55 metres at the gauge in the town Wels, which is located downstream just a few kilometres northeast of the study site area. The water level of the stream is strongly related to the groundwater table (Müller 2002). The recess caused by river regulation measures and bed erosion processes can be determined as the reason for the drop of the water level. High floods, such as the two major flood events in 1897 and 1899, flooding the centre of the town Wels, do no longer occur in this area (Kohl 1992).

*Changes to groundwater*

The deepening of the river bed in the study site area has generated periodically low water levels over the last century as shown in Figure 5.8. The alteration of the groundwater table is almost similar to the change in water level within the River Traun in the study site area (Müller 2002). Figure 5.8 depicts how the groundwater table has shown an ongoing and steady drop within the 20st century. Furthermore, the stable water level in the backwater of the weir can be clearly identified between river kilometre 37 and 38 for the measurements undertaken after the completion of the weir and commissioning of the hydropower plant.

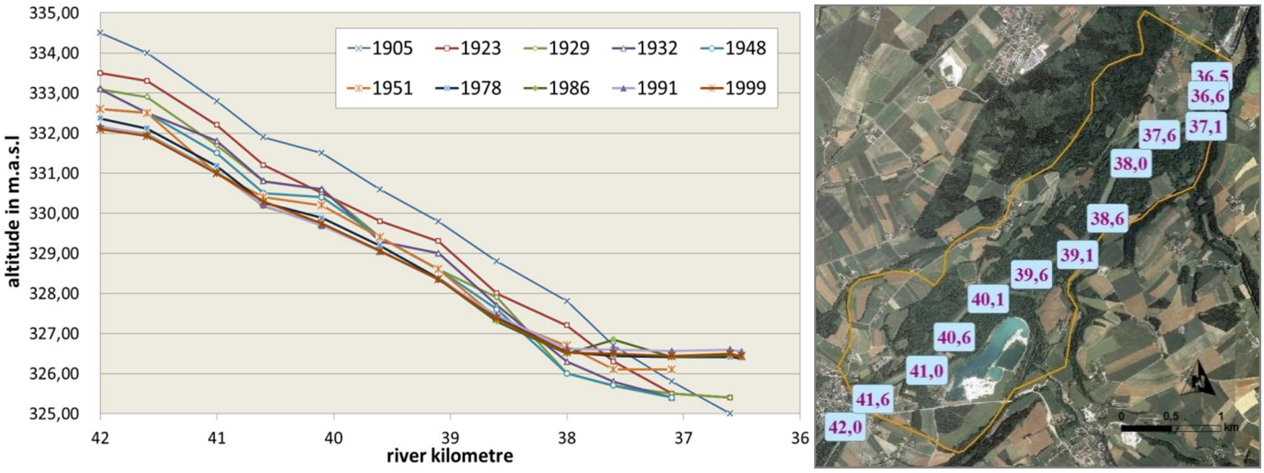
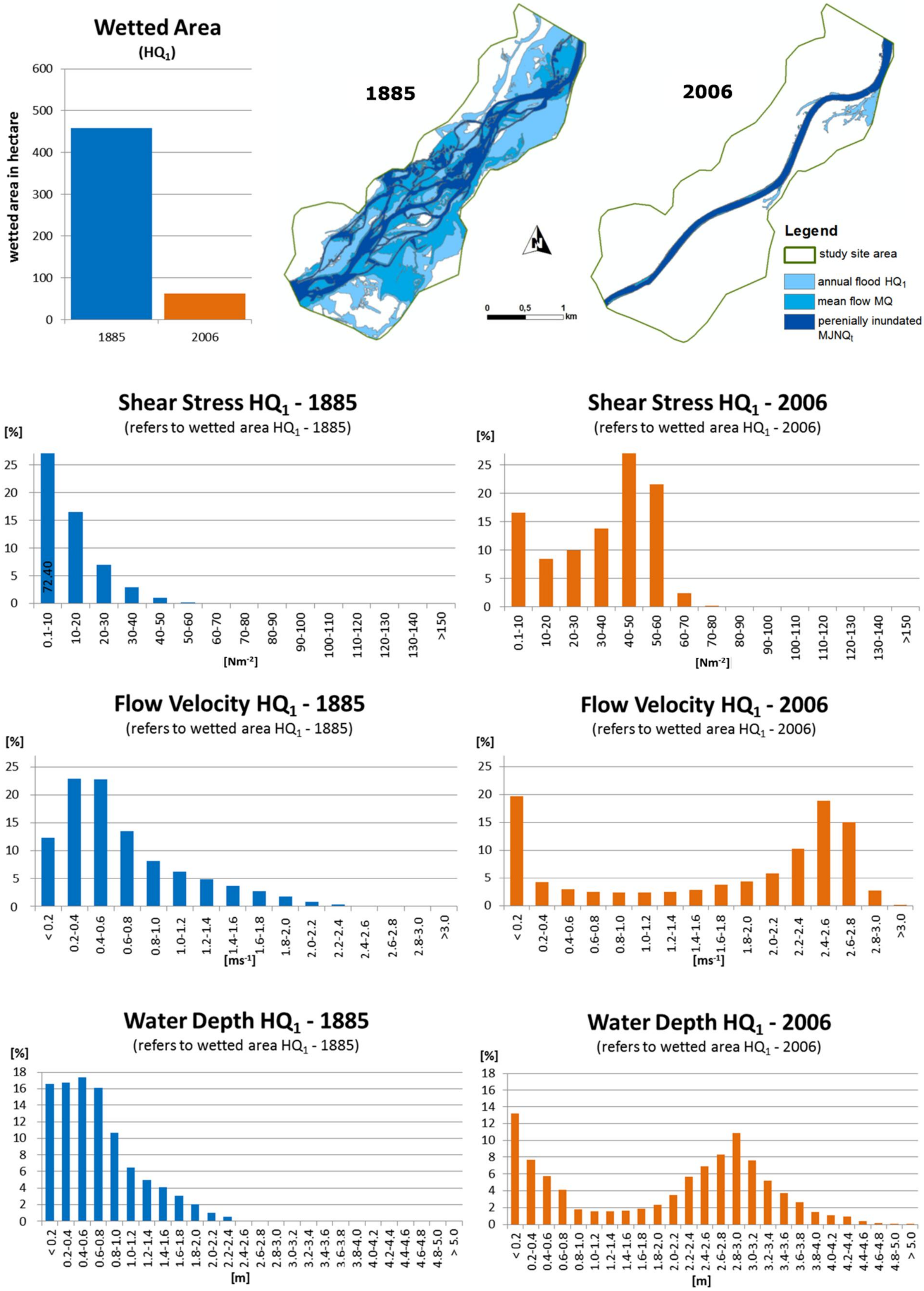


Figure 5.8 Tendency of river bed deepening as listed in low stream water level records (highly corresponding with ground water table) for the time period 1905 to 1999 (Mader et al. 2002) and an overview of associated river kilometre of the study site area (right).

### 5.3.2 Modelling results

Figure 5.9 displays an annual flood event at the study site. The analysis shows a significant reduction in the wetted area from more than 450 hectares down to 60 hectares over the last century. The flood event in 2006 mainly affected the area of the Fischlhamer Au in the north-eastern part of the project area. 89% of the wetted area in 1885 had shear stresses below 20 $Nm^{-2}$, while only 0.05% exceeded 40 $Nm^{-2}$. The wetted area covered broad sections of the floodplain. Today, the river does not overflow the regulated river banks. The largest flooded areas can be found along a right-side tributary in the lower part of the study area. Furthermore, the distribution of flow velocities and water depths is shifting to the right compared to the historical flood in 1885 (see Figure 5.9). Flow velocities in the range of two to three metres per second increased from approximately 1% in historical times to 52% of the total wetted area in 2006. In terms of shear stress, the annual flood also causes a clear shift, with 49% of the wetted area having shear stresses in the range of 40 to 60 $Nm^{-2}$ in 2006

The alterations in flooding depth, velocity and shear stress are completely different in 2006 compared to 1885 at the high flood event. In 2006, which is comparable to the current situation, significantly greater areas of the floodplain were inundated at an annual flood event. As seen in Figure 5.10, the distribution of shear stress and flow velocity for the flood with a 100-year occurrence probability is similar to the distribution for the historical annual flood event, and only a slight increase in the proportion of areas with very shallow and shallow water depths can be observed. In the current situation, in contrast, both flooding depth, velocity and shear stress show a clear shift to the left at high floods, implying that large areas of the floodplain are inundated and that shear stresses, flow velocities and water depths are low. Similar percentage for distribution for shear stress, flow velocity and water depth can be found when comparing the annual flood with the high flood event for the analysis of the historical dataset. When assessing a flood event in 1885 (natural condition) compared with a flood event with the same occurrence probability in 2006 (channelized river) significant changes occur in the distribution of all parameters.


Wetted Area (HQ$_1$)

Shear Stress HQ$_1$ - 1885 (refers to wetted area HQ$_1$ - 1885)

Shear Stress HQ$_1$ - 2006 (refers to wetted area HQ$_1$ - 2006)

Flow Velocity HQ$_1$ - 1885 (refers to wetted area HQ$_1$ - 1885)

Flow Velocity HQ$_1$ - 2006 (refers to wetted area HQ$_1$ - 2006)

Water Depth HQ$_1$ - 1885 (refers to wetted area HQ$_1$ - 1885)

Water Depth HQ$_1$ - 2006 (refers to wetted area HQ$_1$ - 2006)

Figure 5.9 Comparison of shear stress, flow velocity and water depth in the wetted area of an annual flood event in 1885 and 2006.

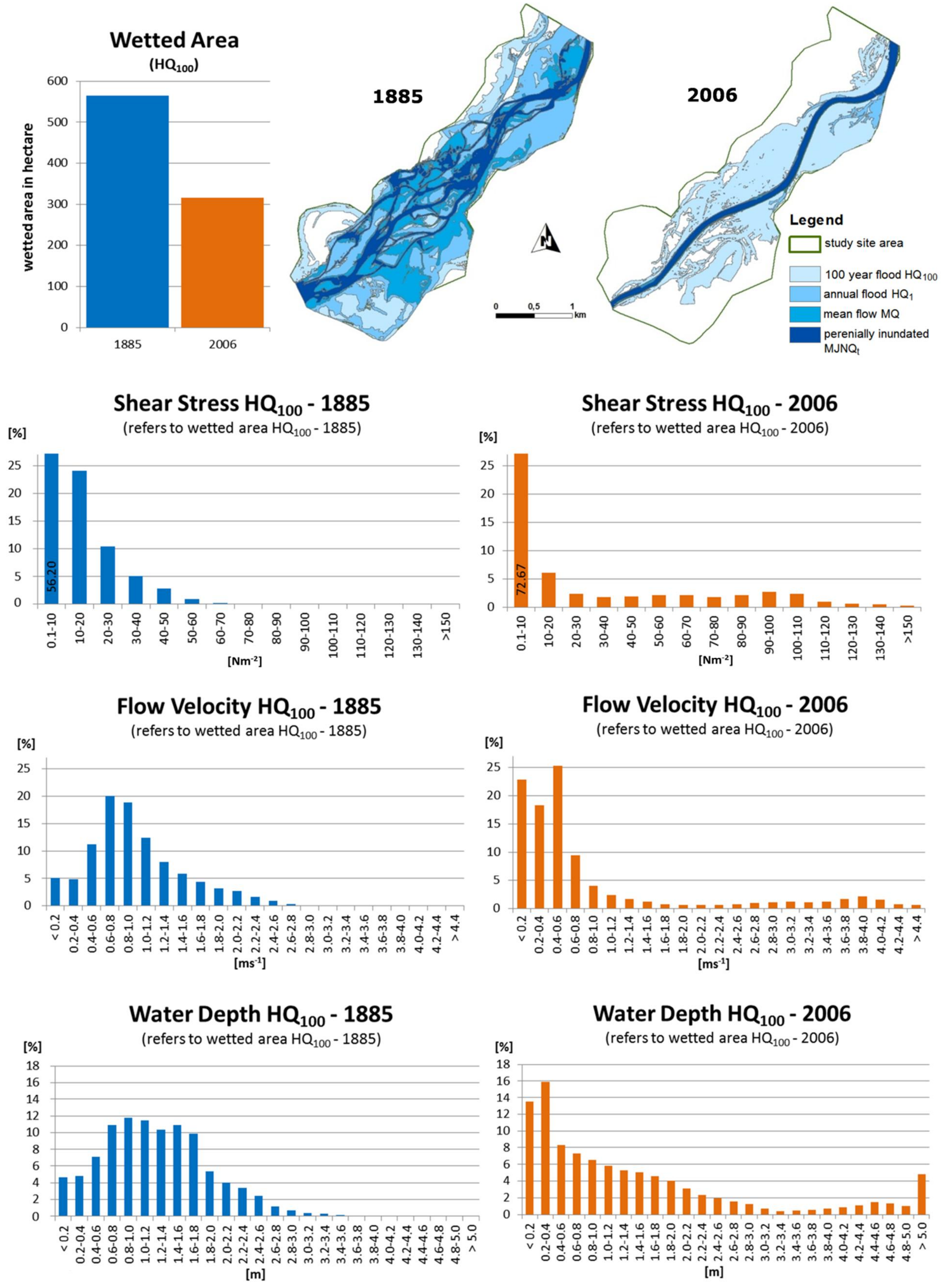


Figure 5.10 Comparison of shear stress, flow velocity and water depth in the wetted area of a flood with a 100-year return period in 1885 and 2006.

### 5.3.3 Distribution of vegetation

Figure 5.11 shows the distribution of vegetation as recorded for the Franziscan Cadastre compared to the situation of vegetation distribution and structural diversity 100 years after implementation of the first major river regulation measures (Muhar et al. 1987). The change in the distribution of the different types of vegetation corresponds to the changes in inundation frequency caused by anthropogenic alterations and the concomitant drop in the groundwater table.

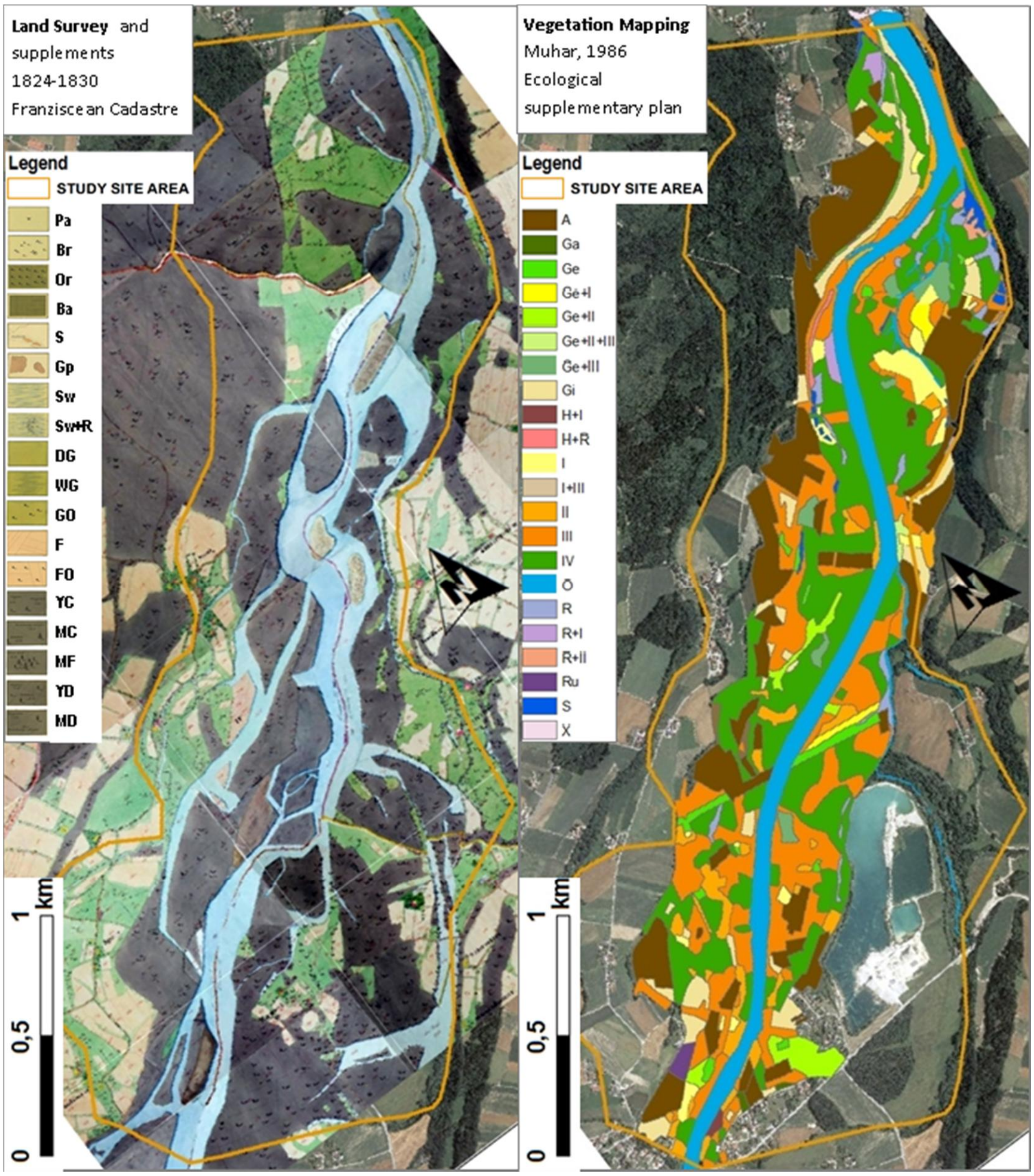


Figure 5.11 Historical (left) and current (right) vegetation distribution maps. Legend key can be found in the appendix of this deliverable. Map basis provided by the national archive of the province of Upper Austria (2014)

## 5.4 Discussion

### 5.4.1 Hydromorphological development

Within the study site, the River Traun is characterised as a lowland river. Typical characteristics of this river type are low flow velocities, a meandering and strongly braided river course, high relocating capacity and self-steering processes and high inundation dynamics in the area of riparian forests (Strauch 1992c). In former times, the River Traun had several branches and side arms in its natural unregulated condition. The shear stresses and flow velocities were much lower than after the major anthropogenic alterations. In the historic situation, groundwater lowering took place only at a very small scale (Strauch 1992b).

After the river regulation measures were introduced, flow velocities in the narrow single bed river increased dramatically, causing extensive erosion of the river bed. Thus, the groundwater table has decreased significantly and is still subject to this ongoing recess of the riverbed. The lowering subsequently caused an extended die-back of the riparian forests, but at the same time the nearby historical town centre of Wels was protected from floods (Kohl 1992). The backwater area in the north-eastern part of the study site is dominated by sedimentation processes and in contrast to the erosion-dominated area of the main channel. This backwater area also leads to a stable and monotonous groundwater table in the remaining parts of the riparian forests of the Fischlhamer Au. Although the groundwater table in this area is not lowered, it is still not in a natural condition as typical natural fluctuations are missing. Due to the absence of periodical inundations in this area, deposition of fine sediments, and with it natural fertilisation and corresponding humidity, no longer takes place in the Fischlhamer Au.

The distribution of defined classification zones is presented in Figure 5.12 as a proportion of the total area of the study site. The approach and definition of the five classes can be found in chapter 5.2.4 of this Deliverable. Figure 5.11 shows the zonal distribution for a natural, island braided gravel-bed river corresponding to the situation in 1885 and an anthropogenic-influenced sinuous straight gravel-bed river corresponding to the situation today. These two types correspond to the extended river typology classification developed within D2.1, part 1 (Gurnell et al. 2014a) – type 9 and 13, respectively. The distribution graphs of both conditions of the River Traun study site should be seen as complementary to the approach of hydrological and fluvial process gradients that drive the lateral zonal mosaic for different river types introduced in the topic “Non-linear Interactions between Vegetation and Physical Processes” D2.2, part 1, chapter 2.2.3 (Gurnell et al. 2014b).

It is of special interest that zones where fluvial disturbances dominate (zones 2 and 3) have been reduced drastically or even disappeared almost completely due to river regulation measures and their associated consequences for the hydrology of the system.

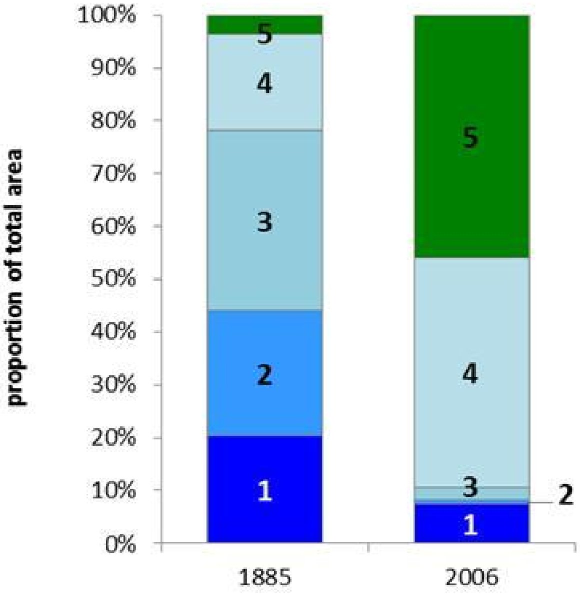


Figure 5.12 Distribution of classification zone areas (see Table 5.2 for legend key) in proportion of total study site area.

### 5.4.2 Vegetation development

Softwood riparian forest, which is the characteristic vegetation type for the natural undisturbed condition of the River Traun, is strongly affected by the anthropogenically induced alterations occurring over the last century. One of the reasons for this is that growth of riparian species is not only dependent on periodical floods but also on a high groundwater table. One of the last remaining patches of riparian forest of the lower Traun valley is located in the Fischlhamer Au. Here, a small right-side tributary joins the River Traun and the groundwater table is higher than the rest of the study site (Schuster 1992). Nowadays, ash trees (*Fraxinus excelsior*) cover more than 70% of the forested area in the river meadow Fischlhamer Au. Due to their strong growth, other native and typical species like *Populus nigra*, *Populus alba*, *Tilia cordata*, *Quercus robur* and *Ulmus glabra* are declining in abundance caused by a cumulation of changes of hydromorphological processes.

The significant lowering of the river bottom caused by the initial river channelization and the ongoing erosion process yield in a significant drop of the ground water table, less flooding of the riparian area and a significant change of the sedimentation processes in the river forelands with a loss of fine sediment. Considering the whole study area, the former river meadows were dried out and turned into fields of moor grass and semi-arid grasslands, which have been partially preserved until today. From the perspective of species conservation, these dry areas are considered highly valuable as they provide habitats for many endangered species (e.g. orchids). Nevertheless, at the same time natural riparian forests have been lost in the area.

Information about the historical and current distribution of vegetation zones in the study site is summarised in Table 5.4. Additional information about alteration processes and ecosystem functions is provided for the individual zones referring to the measures affecting the individual areas of interest based on the findings of Schuster and Essl (2001) and Hughes (2003) within this table.

Table 5.4 Classification of vegetation zones.

| Flowing water | |
|---|---|
| Historical distribution | Present distribution |
| island braided river, high gradient, heavy sediment transport, various changing side arms, no clearly discernible main channel. | sinuous-straight main channel, no side arms, bank reinforcement and bed stabilisation, river bed erosion of several metres, typical structures are missing. |

| Stagnant water | |
|---|---|
| Historical distribution | Present distribution |
| stagnant water areas were probably strongly influenced by hydrological dynamics (water level fluctuations, probably partial desiccation, periodical inundations, sedimentation, erosion as decisive factor), high dynamic stagnant waters. | high dynamic stagnant waters with typical character disappeared, only few stagnant waters, rarely inundated, mostly fed by groundwater (but water level of stagnant waters regulated by the water level of River Traun), tendency to silt up (due to deepening of river bed and increasing sedimentation), area called Fischlhamer Au is particularly affected; additionally, some fish ponds and ponds at the gravel site Zauset. |

| Gravel pioneer sites | |
|---|---|
| Historical distribution | Present distribution |
| extensively developed (gravel and sandbanks and gravel islands), flood events caused relocation at varying intensity and in different forms, strongly linked to riparian forests and various side arms. | no extensive gravel pioneer sites at the lower Traun; small sites still exist, larger patches are not visible except at low flow conditions; typical species of this habitat (*Myricaria germanica*, *Calamagrostis pseudophragmites*, *Epilobium dodonae*) are missing or have become extinct; secondary habitats are located at the gravel site Zauset. |
| The system is closely linked to the elevation above mean stream water level and sediment patterns. Gravel pioneer species are dependent on ongoing periodical relocation processes of the gravel banks. | |

| Pioneer sites of shrubbery | |
|---|---|
| Historical distribution | Present distribution |
| with the decreasing relocating capacity the succession of gravel sites and therefore first shrubs could establish; initially *Salici-Myricarietum*, then *Salicetum incano-purpureae*; based on literature, the pioneer shrubbery was a very characteristic element in riparian forests. | typical shrubbery pioneer sites are completely missing in the study site; fragments of *Salix eleagnos* are retained locally at the riverbank; small atypical patches are located at the gravel site Zauset; the strong regional decline continues through the state and national levels. |

| Softwood riparian forest | |
|---|---|
| Historical distribution | Present distribution |
| pioneer sites of shrubbery develop into softwood riparian forests (primarily *Salix alba*, *Alnus incana* and *Populus nigra*); periodically inundated at undisturbed flow regime; according to historical maps, this habitat was most spatially significant in the study site; it is assumed that there was a large variability in softwood riparian forest types (*Populus nigra* dominated on drier and higher coarse gravel; *Salix alba* and *Alnus incana* in moist habitats). | decreased significantly due to river regulations and the establishment of the weir and hydropower plant downstream; became partly extinct; remaining sites are located in the Fischlhamer Au and Saager Au (*Salix alba*); some very few mature *Populus nigra* can also found in these areas; rejuvenation of the *Salix* and *Populus* population is not possible at present due to the lack of relocation dynamics. |

| Hardwood riparian forest | |
|---|---|
| Historical distribution | Present distribution |
| only small areas or on the edge of the riparian forest; on dry sites dominated by *Quercus* and *Tilia*; on fresh locations dominated by *Fraxinus*. | due to river regulation and lowering of the river bed, hardwood riparian forest constitutes the largest part of riparian forest; *Tilia* in higher regions (distant from groundwater and never flooded), *Fraxinus* in the lower, moister areas; within the study site an increase of hardwood riparian forest can be assumed. |

| Vegetation of riverbanks | |
|---|---|
| Historical distribution | Present distribution |
| diverse and strongly characterised by flow dynamics; mainly pioneer species on gravel sites, small reeds and tall-forb communities; no detailed historical data available on distribution of reeds, however, local occurrences were concluded based on historical data of birds with a strong binding to specific reeds. | continuously regulated riverbanks (rip-rap) are a uniform and less valuable from a nature conservation point of view; riverbanks in the Fischlhamer Au and other remaining riparian forests are characterised as high quality; large sedge-reeds, reeds and tall-forb communities are well developed; pioneer communities are missing completely. |

| Aquatic vegetation | |
|---|---|
| Historical distribution | Present distribution |
| little information available; it is assumed that aquatic vegetation was not able to establish significantly in areas influenced by dynamic relocating processes and species-rich aquatic vegetation communities. | still exists in the few permanently auriferous side channels; *Potamogeton natans* occurring regularly; rare occurrence of *Lemna minor*. |

| Tall-forb communities | |
|---|---|
| Historical distribution | Present distribution |
| little information available; it is assumed that a few tall-forb communities were located in humid sites with fine soil (islands, gravel banks). | occur in humid sites of the riparian forest and close to the side channels; in the course of dehydration of the riparian forest within the study site the areas of tall-forb communities declined. |
| Tall-forb communities occur mainly on deep soils. They are rich in species diversity and very productive (Murray et al. 1991). Areas with tall-forb communities provide humid and cool microclimate conditions and high nutrient levels. Good progress in succession of growth usually results in the development of shrubby vegetation. | |

| Semi-dry grassland | |
|---|---|
| Historical distribution | Present distribution |
| these areas were the main components of the mosaic of habitats in historical times; semi-dry grassland (often grazed areas on the edge of the riparian forests, pioneer dry grassland (on young and higher situated alluvion deposits); fluent transition to forests; no detailed information about area size of semi-dry grassland available; records of impressive dry sites with *Juniperus communis* and other species characteristic for this habitat from the 19th century. | most areas are lying fallow or are mown grasslands, dry moor grass fallows and semi-dry grasslands; mainly small areas on the edge of forests or on the edge of intensively used fields; areas decreased significantly; remaining patches are of high relevance from a botanical point of view as there are some very rare species; some grassland types disappeared completely like extensively grazed semi-dry grassland or gravel pioneer grassland; the distribution centre is located on the left river side in the higher areas of the Saager Au; the habitat is sensitive to the change of cultivation practices (abandonment as well as intensification). |

The number of extinct pontic-pannonian species of the lower Traun valley is remarkable. Known occurrences of endangered plant species are often dated back to the 19th century. Species disappeared after the beginning of intensive agriculture. From the detailed knowledge of historical data of this area, the species loss is well documented. It can be reasonably assumed that the loss of species occurred similarly around other rivers of the alpine foothills. This assumption could not be verified due the lack of comprehensive historical vegetation datasets

| Humid grassland | |
|---|---|
| Historical distribution | Present distribution |
| little or not present in the central, dynamic processes dominated area of the riparian forest; common in border areas of riparian forests, along saturated areas on the foot of the slopes; nutrient-rich and nutrient-poor (*Molinion*, *Caricion davallianeae*) occurrence. | only small areas remaining; special emphasis should be placed on small humid and nutrient-poor grassland in the southeastern part of the Fischlhamer Au; some *Molinia*-dominated assemblages can be found in the former drainages of the riparian forest; locally also nutrient-rich patches of humid grassland. |

| Conifer forests | |
|---|---|
| Historical distribution | Present distribution |
| none; possibly few small anthropogenic-influenced patches for raw material production. | Constitute a major part of some areas in the study site (mainly on sites of formerly hardwood riparian forests) and continue to expand. |

| Grassland, intensive use | |
|---|---|
| Historical distribution | Present distribution |
| are not found according to the present meaning. | fertilised, partly temporary grassland, only small areas on the edge of the study site, little importance from a botanical perspective. |

| Fields and ruderal vegetation | |
|---|---|
| Historical distribution | Present distribution |
| Occurrence only possible in fringe areas of higher riparian forests as lower areas were frequently flooded; only extensive use as intensive agricultural use was not possible due to the less profitable and shallow soil layer. | intensive agricultural use of large parts (mainly on the left side of the River Traun) of the formerly riparian forests after introduction of river regulation measures (instead of grassland and by clearing woods). |

| Settlements and infrastructure | |
|---|---|
| Historical distribution | Present distribution |
| no buildings in the area of the riparian forests due to the flood risk and high self-steering dynamic processes; mainly farmhouses were established close to the outer boarder of the riparian forest (flood-safe sites). | increasing building development within the study site due to river regulation and flood protection measures; mainly in the southwestern part of the study site; a power line intersects the area at river-kilometre 40. |

## 5.5 Conclusions

Today, river channelisation is one of the major types of degradation of riparian ecosystems in north-western and central Europe, and natural riparian forests are declining rapidly as a consequence of this. This decline is also of importance for the microclimate influencing the distribution and abundance of other organism groups as well as the biogeochemical cycles.

The River Traun river regulation at the turn of the 19th to the 20th century marked the beginning of a major structural transformation of the study area. Nowadays the entire study area is affected by the river regulation. Furthermore, the lower part of the area is influenced by a permanent weir of a downstream hydropower plant. A flood protection dam cuts off the river from its floodplain in the northwest part of the study area. Side arms and wet areas of the riparian forest are dried up and have partly been turned into fields. The deepening of the nowadays single bed river has also caused a significant lowering of the groundwater table. The loss of riparian forests is mainly due to the drop of groundwater and the cut-off of side arms due to river regulation measures initiated in the 1880s. In summary, the multi-stressors at the River Traun caused a fundamental alteration of the riparian forests, a dramatic loss of valuable vegetation habitats and a significant loss of species. Especially species dependent on dynamic fluvial processes (pioneer sites on gravel banks, stagnant water vegetation) are affected by the regulations and several alterations. The number of species in the lower Traun valley has decreased dramatically. Within the 20th century more than 300 species like *Astragalus onobrychis*, *Orlaya grandiflora*, *Euonymus latifolia*, *Cicuta virosa*, *Utricularia vulgaris*, *Rumex hydrolapathum* and *Iris pseudacorus* became extinct or strongly endangered within this area (see annex of this deliverable for taxa list, Strauch,1992d). Nevertheless, the lower Traun valley is still one of the most species-rich areas of Upper Austria in terms of vegetation. This diversity also results from a high diversity in many dry ruderal and gravel sites. These areas are important for protection of endangered species even though they are not part of the natural vegetation of the area (Strauch 1992a).

Results of the very detailed analyses of historical and actual vegetation distribution in combination with hydromorphological data of the 2D hydraulic model at the Traun river site clearly show that floodplain forests are unable to persist in channelised single bed rivers with embankments because of a significant drop in the water table. Furthermore, the dynamic forces of flood events are completely lost, and species relying on periodic flooding and channel dynamics disappear. The River Traun study site is representative of a large number of European rivers where the typical shrubbery pioneer sites and softwood riparian forests have disappeared and been replaced by mainly hardwood riparian forest that constitute the largest part of the remaining European riparian forests.

## 5.6 References

Austrian Hydrographic Yearbook / Hydrographisches Jahrbuch von Österreich (1895-2011). Hydrograph. Zentralbüro - Abteilung VII 3 - Wasserhaushalt im Bundesministerium für Land- und Forstwirtschaft, Umwelt und Wasserwirtschaft (Austrian Federal Ministry of Agriculture, Forestry, Environment and Water Management), Wien.

Blaschke, A. P. (2001). Hydrologisches Gutachten zur Festlegung des Zustrombereiches zum Brunnen Hochholz der Gemeinde Gunskirchen

Council Directive 92/43/EEC of May 1992 on the conservation of natrual habitats and of wild fauna and flora, Annex I (1992L0043-EN-01.01.2007-005001)

Ellenberg, H. (1986). Vegetation Mitteleuropas mit den Alpen.

Ellenberg, H. and C. Leuschner (2010). Vegetation Mitteleuropas mit den Alpen: in ökologischer, dynamischer und historischer Sicht.

Essl, F. and A. Schuster (2001). Konzeptstudie Renaturierung Traun.

Franziscean Cadastre (1825). Oberösterreichisches Landesarchiv, Land Oberösterreich.

Gemböck, R, (1886). Naturpürschen in unserem Heimatlande. Hugos Jagdztg. 23: 721-724.

Gurnell, A. M., M. Bussettini, B. Camenen, M. G. D. Tánago, R, Grabowski, D, Hendriks, A. Henshaw, M. Rinaldi and N. Surian (2014a). A multi-scale framework and indicators of hydromorphological processes and forms. Deliverable 2.1, Part 1, of REFORM (REstoring rivers FOR effective catchment Management), a Collaborative project (large-scale integrating project) funded by the European Commission within the 7th Framework Programme under Grant Agreement 282656.

Gurnell, A.M., Tánago, M.G.D., O´Hare, M.T., Van Oorschot, M., Belletti, B., Buijse, García De Jalón, D., Grabowski, R., Hendriks, D., Mountford, O., Rinaldi, M., Solari, L., Szewczyk, M. and Vargas-Luna, A. (2014b). Influence of Natural Hydromorphological Dynamics on Biota and Ecosystem Function. Deliverable 2.2, Part 1, of REFORM (REstoring rivers FOR effective catchment Management), a Collaborative project (large-scale integrating project) funded by the European Commission within the 7th Framework Programme under Grant Agreement 282656.

Hauser, E. (1998). Biotopkartierung der Traun - Auen bei Fischlham und Saag._Naturkundl. Jahrb. Stadt Linz 42/43: 343-400

Hughes, F. M. R. (ed.) (2003). The Flooded Forest: Guidance for policy makers and river managers in Europe on the restoration of floodplain forests. FLOBAR2, Department of Geography, University of Cambridge, UK. 96pp.

Hüttmeir, S. (1992). Planzensoziologische und vegetationsökologische Studien in den Auwäldern der Traun im Raum Lambach – Wels._ Dipl.Univ.Salzburg, 103pp.

Kohl, H. (1992). Kataloge des OÖ Landesmuseum N:F:054b; Die Entwicklung des Traunflusses und seines Tales im Laufe der Erdgeschichte, 5-25.

Moog, O., Wimmer, R. (1990). Grundlagen zur typologischen Charakteristik österreichischer Fließgewässer. In: Wasser und Abwasser, Wien, Bd. 34: 55-211.

Moss, T. and J. Monstadt (2008). Restoring Floodplains in Europe; Policy Contexts and Project Experiences.

Muhar, S., G. Schramayr and S. Muhar (1987). Landschaftsökoligische Begleitplanund Kraftwerk Edt/Traun, Fachbereich Botanik.

Murray, R., H.F. Mayland and G. E. Shewmaker (1991). Response of montane tall-forb communities to 2,4-D and mixtures of 2,4-D and picloram. Journal of Range Management 44(4).

Müller (2002)- Projekt „Lebenswerte Traun“, Abstimmung mit den wasserwirtschaftlichen Verfügungen und Grundsätzen.

National Archives of the Province of Upper Austria (2014). Geographical data weblink: http://doris.ooe.gv.at.

Roth, J. (1924). Unsere einheimische Vogelwelt – Welser Ztg. Vom 18th April, 1924.

Schuster, A. (1992). Kataloge des OÖ Landesmuseum N:F:054b; Botanische Forschungen entlang der Traun seit mehr als zwei Jahrhunderten als Beitrag zum Schutz der Natur, 409-429.

Speta, F. (1992). Kataloge des OÖ Landesmuseum N:F:054b; Die Amphibien der unteren Traun, 79-92.

Strauch, M. (1992a). Kataloge des OÖ Landesmuseum N:F:054a; Flora und Vegetation im Unteren Trauntal , 54-62.

Strauch, M. (1992b). Kataloge des OÖ Landesmuseum N:F:054b; Entwicklung der Wald- und Wiesenflächen sowie der Besiedelung im Unteren Trauntal seit 1825, 251-262.

Strauch, M. (1992c). Kataloge des OÖ Landesmuseum N:F:054b; Gewässerzustandskartierung am Beispiel einiger Fliessgewässer im Unteren Trauntal, 205-215.

Strauch, M. (1992d). Kataloge des OÖ Landesmuseum N:F:054b; Die Flora im Unteren Trauntal (Oberösterreich), 277-329.

Wimmer, R., H. Wintersberger and G. Parthl (2007). Hydromorphologische Leitbilder (BMLFuW).

# 6 Riparian and Floodplain Vegetation Hydromorphology Interactions - Polish Case Study

## 6.1 Introduction

The case study was conducted in North East Poland in the Narew River valley. The upper Narew valley covers more than 18 thousand hectares and is located in the central part of Podlaskie Voivodship, between the Siemianówka reservoir and Suraż (Figure 6.1). To the east it borders with Białowieża Forest and to the west with the Marshy Narew Valley. The upper Narew Valley is one of the best preserved Polish river valleys and, next to the Biebrza Marshes, it constitutes one of the largest wetlands in central European; with habitats of the greatest biological diversity for the moderate climate zone.

This type of river was common in the lowland plains of Europe. Low longitude gradient as well as low erosion potential in the catchment made their hydromorphology and biota stable for centuries. This type of river has been physical modified for many centuries, a process which accelerated in the 19th century modified due to rapid agricultural development and urbanisation. Large scale land reclamation, flood protection hydraulic structures as well as river training for navigation caused significant changes in the river bed structure in-stream biota as well as floodplain vegetation. The Narew River in its upper part has not undergone any significant morphological changes and remains a unique example and good reference for number of restoration purposes on European lowlands.

The Narew River has a natural bed and the relatively narrow, parallel located valley is shaped by the regular floods. Due to these environmental characteristics, most of the valley area is covered with non-forest plant communities, mostly rushes from *Phragmitetea Cl.*, the presence of which depends on the annual inundations. Communities of *Carex acuta.* and *Glyceria maxima* are predominant here and around the oxbows – reedbeds of *Phragmites australis*. In the periphery of the valley, there are meadows inundated with ground waters, and species characteristic of both fresh and dry habitats can be found on the sandy uplifts to which access is difficult during floods. More than half of the area is used as meadows and pastures.

The Upper Narew valley is within the boundaries of two Nature 2000 network sites: Ostoja w Dolinie Górnej Narwi (PLH200010) i Dolina Górnej Narwi (PLB200007). In the valley are 10 types of natural habitats, protected under the Directive of the Council 92/43/EEC f.e. 91F0 riparian mixed forest, 91E0 alluvial forest, 7230 alkaline fens, but this habitats occupy small areas in the valley - wetland vegetation from *Phramitetea* class, which dominates in the valley, is not protected The area is also a sanctuary for multiple species of birds, especially water and predatory species that reach record-high numbers. At least 35 bird species have been recorded in the area covered by Annex I of the Directive of the Council 79/409/EEC as well as minimum 12 animal species listed in Annex II of the Directive of the Council 92/43/EEC such as otter and beaver as well as wolf, the European priority species.

The case study focuses on changes in riparian vegetation that are induced by the operation of the water reservoir located in the upstream part of the valley, which reduces flood water in spring.

Figure 6.1 Study area

The Siemianówka water reservoir was put into operation in 1992. Prior to this, there were no pressures on the riparian vegetation except of extensive grazing and mowing, and the area was categorized as a semi-natural wetland ecosystem. After 1992 no significant changes have occurred in catchment land use, implying that the observed shifts in vegetation structure are due to the altered hydrology (Figure 6.2).

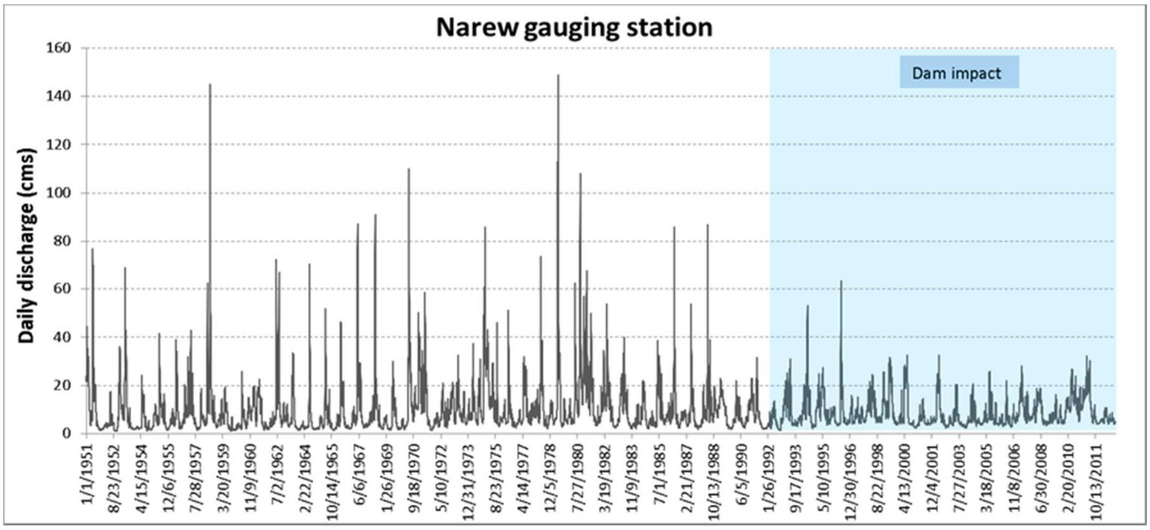


Figure 6.2 Daily discharge at the Narew gauging station 1951-2012 (Institute of Meteorology and Water Management – National Research Institute).

The hydroecological case study included a hydrological study using the HEC-Ras model, remote sensing evaluation of the flood extent for the specific types of vegetation, assessment of changes in flood frequency at selected locations and assessment of changes in vegetation structure after 20 years of reservoir operation.

## 6.2 Materials and Methods

### 6.2.1 Site description

The case study covered a section of the Narew River, is located between the Bondary and Suraż gauge station (Figure 6.3). Until the 1980s, when the Siemianówka reservoir was constructed and waters were accumulated, the valley had not undergone any significant fluvial geomorphological modifications.

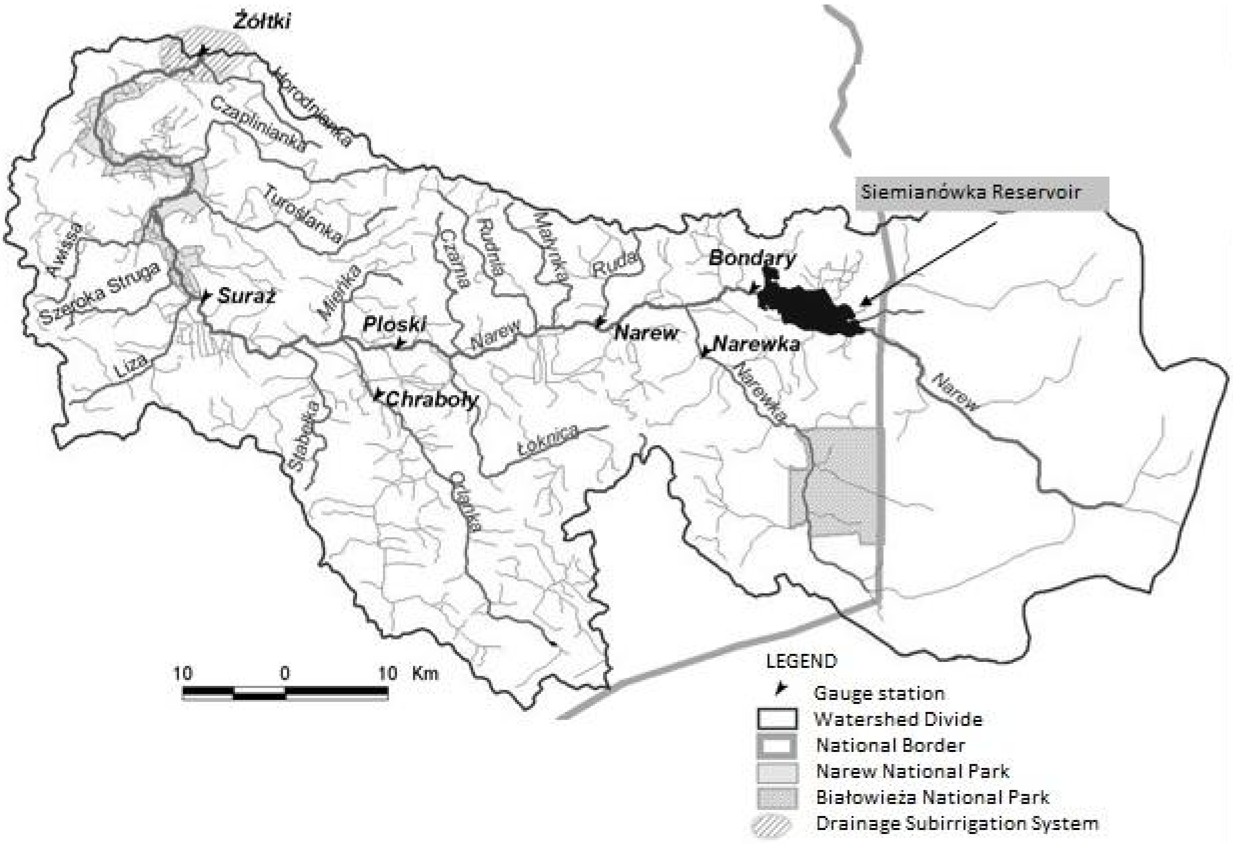


Figure 6.3 Hydrographic map of upper Narew

The hydrological regime of the Upper Narew is typical of lowland rivers. It is characterised by a single spring flood, resulting from snow melt, and a quite substantial summer low flow (Figure 6.2). Summer floods are sporadic. Since the Siemianówka reservoir was put into operation in 1990, the river regime has undergone some modifications, especially in the upper part of the valley. The exact impact of hydro-engineering objects, including the Siemianówka reservoir is unclear. The significant drop in the maximum flows since 1990 is a fact, but except for the reduction of the flood peak by the reservoir, it could be also caused by different climatic conditions in that period. The factors that directly affect the height of the spring freshet are a sum of the precipitation and the snow accumulation. Maksymiuk et al. (2008) presented the analysis of the basic climatic variables for the Biebrza basin, the river Narew's sub-catchment. It indicates that there is no significant change in the annual precipitation before and after 1990. However it is different in the case of the snow cover, which amount fall after 1990. The answer for the question what is the main cause of the reduction of maximum flows in the Upper Narew is not straightforward, as it is probably a joint effect of the reservoir operation and changes in climatic conditions.

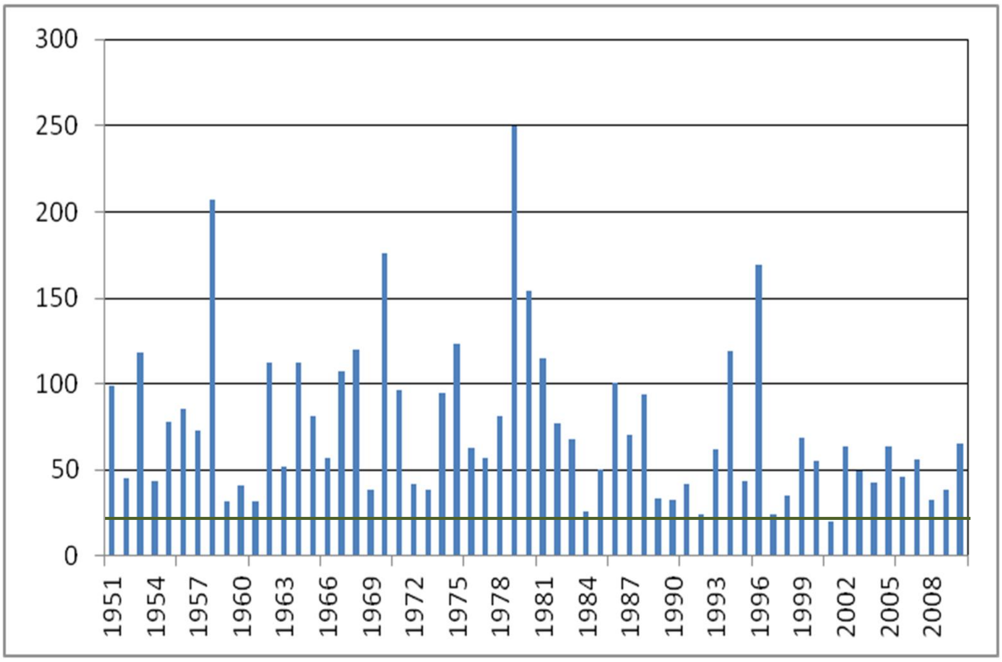


Figure 6.4 Maximum annual flow in Suraż during 1951-2010, river bank flow indicated with a green line (based on data provided by the Narew National Park).

### 6.2.2 Vegetation description

The area of the upper Narew valley is dominated by non-forest plant communities characteristic of wetlands with a fluviogenic type of hydrological feeding (Oświt, et al. 1996; Okruszko, et al. 2011). Detailed maps of the actual vegetation from 2008 (Wpływ 2008) were used to classify the vegetation into groups of plant communities occurring under similar conditions, as a means to examine the effects of altered flooding frequency on the vegetation. We classified the following eight groups:

Common reeds of *Phragmition*, occupying 13.6% of the study area, are dominated by phytocoenosis of *Phragmitetum communis* and *Glycerietum maximae*. They occur within the vicinity of the old river beds and oxbow lakes, on periodically exposed areas of the river bed and in periodically submerged floodplains.

Swamp sedge communities whose presence indicates soligenic alimentation. This group include *Caricetum rostratae, Caricetum appropinquatae*, *Caricetum acutiformis* as well as a part of the phytocenoses of *Calamagrostis canescens*. Their areal extension does not exceed 2% of the studied area. Because of their small extension, their patches are included in the wetland sedge communities category.

Wetland sedge communities occur in intensely flooded areas and thus mainly in the areas that are less drained by the river. Compared to swamp sedge communities, they stand out because of the descending groundwater table after the flood wave has drained out. This community is dominated by forms of *Caricetum gracilis* often in a mosaic with *Phalaris-Carex* communities, *Phalaridetum arundinaceae*, wet meadows and *Phragmites* communities. This vegetation type occupies just over 10% of the total study area.

*Phalaris* and *Phalaris-Carex* communities occupy significant areas along the entire length of the valley, mainly in the immediate vicinity of the river, under conditions of marked drainage which facilitates the flow of flood waters. This vegetation type is mainly represented by *Phalaridetum arundinaceae* along with drier varieties of the *Caricetum gracilis* community. They occupy 25.1% of the total area included in the current study.

The above-mentioned three groups of *Magnocaricion* communities, occurring in different habitat conditions, occupy the largest part of the study area, approx. 38%.

Wet meadows of the *Molinietalia* appear in a valley located higher than the area occupied by the sedge and *Phalaris* communities. Depending on the location and the depth of the summer groundwater levels, the moisture levels of lowland hay meadows can be very different. The largest areas are occupied by *Alopecuretum pratensis* and *Deschampsietum caespitosae*. Because of the great hypsometric diversity of the valley, they often comprise a mosaic of *Carex* spp. and *Phalaris* communities, but they are clearly dominant in some areas of the valley and occupy 27.5% of the total surface of the studied area.

Sedge-moss communities and grass-sedge meadow of the *Molinietalia* are inundated by groundwater (soligenic type of hydrological feeding) and confined to parts of the valley where the land is characterised by lateral groundwater flow and poor river drainage. In total, this vegetation type occupies 1.2% of the study area.

Fresh and grassland meadows are relatively rare in the studied area. They mainly appear in marginal parts of the valley and are dominated by productive grasses. Sand and rock grasslands in the late stage of development ,represented by the communities of *Vicio lathyroidis-Potentillion argenteae* are included in this group due to their physiognomic and floristic similarity as well as similarity in satellite images. In total, they occupy 7.0% of the study area.

Sand grasslands in the early stage of development, represented by the communities of *Corynephorion canescentis*, with loose turf, appear on the dry mineral islands of sandy oak-hornbeam forests, rising at least a few feet above the heavily flooded surfaces of the valley. Their areas are characterised by total isolation during floods (Oświt 1996). In total, they occupy 1.5% of the area of the current study.

Forests and scrubs mainly occur in marginal parts of the valley. Alder bogs of the *Alnion glutionosae* occupy the habitats with a stronger and more durable hydration – and riparian forests of the *Alno-Padion* occupy habitats with a more variable humidity. Forest and thicket communities occupy a total of 8.5% of the area within the confines of the current study.

1

2

3

4

5

6

7

8

Figure 6.5 Plant communities: 1 - common reeds; 2 - common wetland sedge; 3 - Phalaris comm.; 4 – grass segde comm.; 5 – wetland meadows; 6 – fresh and grassland meadows; 7 – sand grassland. 8 – scrubs and forest (taken by M. Szewczyk).

### 6.2.3 Flow model

For the Upper Narew river reach, several successful applications of flow routing models are reported. Kubrak et al (2005) applied a one-dimensional model to identify interactions between the Siemianówka reservoir and river discharge within protected areas of the Narew National Park (NNP). A similar model was used by Mirosław-Świątek et al. (2008) to analyse how modifications to reservoir operation rules might affect the flow conditions in NNP. Romanowicz et al. (2010) and Kiczko et al. (2012) analysed the applicability of different flow routing methods for the Upper Narew case, in a reservoir control problem. Okruszko and Kiczko (2008) conducted a preliminary study of the Upper Narew River using a one-dimensional model to identify the inundation pattern for NNP, their study being performed immediately downstream the presently analysed river reach.

A common assumption of these studies is the one-dimensional form of the river flow. This is justified by the semi-natural character of the Upper Narew valley, causing no significant disruption of the flow direction, being as it is shaped by high waters. This approach finds additional support in studies by, for instance, Horritt and Bates (2002) and Chatterjee et al. (2008) who stated that one-dimensional models might provide just as good a reconstruction of the inundation extent as two-dimensional models. Here, the crucial feature is an inundation delineation procedure that depends on the quality of the digital terrain model (DTM). Of course, the accuracy of methods depends on site-specific conditions.

To reconstruct the inundation pattern of and changes occurring in the Upper Narew reach, the UNET model (One Dimensional Unsteady Flow Through a Full NETwork of Open Channels, Barkau, 1993) was employed. It is included in the HEC-RAS (Hydrologic Engineering Centers River Analysis System) modelling software, developed by the U.S. Army Corps of Engineers.

The model is a numeric implementation of the Saint-Venant equations:

$$\frac{\partial A}{\partial t}+\frac{\partial Q}{\partial x}=0$$

$$\frac{1}{A}\frac{\partial Q}{\partial t}+\frac{1}{A}\frac{\partial}{\partial x}\left(\frac{Q^2}{A}\right)+g\frac{\partial h}{\partial x}-g(S_o-S_f)=0 \qquad (1)$$

where: A– flow area, Q – flow discharge, h – water level, x – channel length, $S_f$ – friction slope in terms of Manning formula, $S_o$ – bed slope and t – time. To reflect the dynamics of the river, where the flow area is non-uniform, the water velocity in the main channel is significantly higher than in floodplains, the equations are solved, taking into account a compound character of the cross-section. This is achieved by providing a separate representation of the flow in the main channel and floodplains. The flow partitioning is computed on the basis of the conveyance coefficient for flow zones in the following way:

$$Q_c=\frac{K_c}{K_c+K_f}Q \qquad (2)$$

where $Q_c$ – the channel discharge, $K_c$ and $K_f$– the conveyance coefficients of the main channel and floodplains, respectively. The conveyance coefficients are defined by the Manning formula:

$$K_c = \frac{1}{n_c} A_c R_{hc}^{2/3}$$
$$K_f = \frac{1}{n_f} A_f R_{hf}^{2/3} \tag{3}$$

$n_c$ and $n_f$ are the Manning coefficients and $R_{hc}$ and $R_{hf}$ the hydraulic radius for the main channel and floodplains.

To improve the model performance for meandering rivers, where the length of the flow is different for each flow zone (the left and right floodplain, the main channel), the channel length x, between computational nodes - Δx, is computed in terms of an equivalent flow path:

$$\Delta x = \frac{\overline{A_c}\Delta x_c + \overline{A_f}\Delta x_f}{\overline{A_c} + \overline{A_f}} \tag{4}$$

where $\overline{A_c}$ , $\overline{A_f}$ and $\Delta x_c$ , $\Delta x_f$ stand for the averaged flow areas and lengths between computational nodes for the main channel and floodplains, respectively.

The model was elaborated for the 90 km river reach between the outflow from the Siemianówka reservoir in the village of Boundary and upstream borders of the NNP in Suraż. The data used to build a geometric representation of the river valley consisted of 50 cross-sections, acquired during the field measurements.

For the subcritical flow conditions, which characterised the analysed river reach, the model requires the single upstream and downstream boundary conditions to be specified. For the Upper Narew river the upstream condition was defined as an inflow hydrograph (reservoir outflow), while the downstream condition was an assumption of a normal depth. The water inflow along the river reach was represented as a point inflow for two controlled tributaries: Narewka and Orlanka. The supply from the uncontrolled subcatchments was computed on the basis of a simple mass correction between gauge stations and introduced to the model as the uniform lateral inflow. For the present case study, the feasibility of this approach was proven by Kubrak et al. (2005).

In contrast to the previous studies on flow conditions in the Upper Narew river reach, the simulation horizon was significantly extended. This emphasised the problem of observation scarcity for this river reach. In 1983 the measurements at the river gauge in Ploski were suspended. This measurement point was important as it provided information about the lateral inflow to the reach up to the Narew river gauge. Therefore, it was necessary to identify the relationship between the Ploski and Narew river gauges. This was done using a simple linear regression of daily discharge records with a delay operator. Mass correction for the model was derived from the single measurement point, with complete time series. To obtain the same model accuracy for the whole simulation time, all computations were performed with this empirical discharge relationship.

The model parameters that had to be identified were the Manning coefficients for the main channel and floodplains. Water level observations, to be used for identification, were available for thee gauging stations: Narew, Ploski and Suraż. Between them, the roughness coefficients were assumed to be constant, which resulted in three river sections, for which the coefficient for the left and right floodplain and the main channel had to be specified. The identification was done by trial and error in order to minimise the difference between the computed and observed water levels for all three river gauges.

It was assumed that the roughness coefficients do not change within a year. Such an assumption would be wrong if a vegetation cycle had a significant seasonal effect on the inundation extent. This is not the case for the Upper Narew where almost all significant high-water events are the result of snowmelt in spring. As a result, the roughness of the floodplain, shaped by the vegetation cover, should have similar properties for almost all freshets being modeled.

### 6.2.4 Inundation delineation

The inundation delineation was elaborated based on a digital elevation model obtained from Gheodesy Survey. The model is a high quality Digital Elevation Model (DEM) created using Airborne Laser Scanning (ALS) technology. ALS is being increasingly used worldwide as the most accurate source for DEMs. Currently in Poland, a country-wide estimation of flood-prone areas is undertaken based on ASL data. The DEM for Narew River valley was elaborated in 2013.

In general, digital elevation models are required for hydrodynamic studies of inundation extent and frequency (Bates et al. 2003; Chormański et al. 2009; Mason et al. 2006; van der Sande et al. 2003). Hydrodynamic models always require topographic data for cross-section elaboration (if not measured) and flood extent determination. The ALS LiDAR delivers dense elevation information; yet, with lower accuracy than provided by ground surveys, but out-performing interpolation of elevation data derived from topographic maps. Therefore, typically, the use of LiDAR-derived DEM displays terrain geometry more realistically than other sources and enables a more accurate simulation of water flow. The precision of the DTM is particularly important for flat lowland valleys where small differences in the height and shape of the ground imply inundation of a wide area. Application of ALS DEM data for inundation analysis and modelling has grown rapidly in recent decades.

The water level values calculated using the numerical model of flood flow in the cross-sections are used for determination of the water extent of a particular flood using DEM. The digital model of the floodwater table is interpolated between cross-sections, and then raster map algebra in ArcGIS is processed on the DEM and water table elevation layer, resulting in an inundation map.

Inundation delineation was done based on satellite image analysis performed according to the method set forth by Chormański (2003), which was applied in studies of delineation of flooding of wetlands (Świątek, Chormański 2007; Chormański et al. 2011). The Landsat images have been found to be a valuable data source for inundation area identification in natural environments (e.g. Profetti and Mcintosh 1997; Townsend and Walsh 1998; Świątek, Chormański 2007).

Delineation of the flooded areas has employed the images of satellites Landsat 4 TM (Thematic Mapper), Landsat 5 TM and Landsat 7/ETM+ (Enhanced Thematic Mapper Plus), registered within the period of up to 10 days from the maximal water level noted on the water level indicator in Suraż, during the spring freshet of the Narew river (Table 1). In principle, they were registered at scene 187/23, except for year 1987 (186/23).

The information recorded in the Landsat TM satellite images is contained in seven channels covering various ranges of the electromagnetic spectrum: six multi-spectrum channels (blue: 0.450-0.515μm, green: 0.525-0.605μm, red: 0.630-0.690μm, near infrared: 0.750-0.900μm, middle infrared: 1.55-1.75μm and 2.09-2.35μm) with a resolution of 30 m and a single thermal channel (10.40-12.50μm) with a resolution of 120 m. The Landsat 7/ETM+ images are also contained in the panchromatic channel (0.520-0.900μm) with a spatial resolution of 15 m and a thermal channel with 60 m resolution. The images have been obtained from the resources of the Department of Hydraulic Engineering of the Warsaw University of Life Sciences (SGGW) as well as resources of USGS/EROS – a free portal Glovis (http://glovis.usgs.gov). All tasks were performed using Erdas Imagine and Erdas ErMapper software The images were radiometrically and geometrically corrected. The satellite image adjustment was carried out by means of the nearest neighbourhood method on the basis of 8 control points determined on the basis of a topographic map of the studied region, scale 1:25,000, in the national coordinate system 1965, zone 2. The root mean square (RMS) error for 8 GCP ranged from 0.37-0.95 pixel (11.1-28.5 m) (Table 6.1).

Table 6.1 Dates with maximum water levels and registration of satellite images as well as accuracy of geometric correction.

| Flood number | Date of observed maximum water level of the water level indicator in Suraż | Date of satellite image registration | Satellite type | Scene | Root mean square error of geometric adjustment[pixel/m] |
|---|---|---|---|---|---|
| 1 | 03-04-1986 | 16-04-1986 | MSS | 187/23 | 0.54/16.2 |
| 2 | 08-04-1987 | 12-04-1987 | TM | 186/23 | 0.37/11.1 |
| 3 | 11-03-2002 | 19-03-2002 | ETM+ | 187/23 | 0.56/16.8 |
| 4 | 27-03-2005 | 04-04-2005 | ETM+ | 187/23 | 0.91/27.3 |
| 5 | 16-03-2003 | 22-03-2003 | ETM+ | 187/23 | 0.95/28.5 |

The next images were processed to enhance the interpretation possibilities. The NDVI index created on the basis of the bands from infrared and near-infrared ranges was calculated (bands TM/ETM+4: 0.750-0.900 μm and TM/ETM+3: 0.630-0.690 μm (Lillesand et al. 2008)) the use of which emphasises well in the analysed image the areas covered with plant life, areas deprived of it as well as water. Another transformation of images used in the preparation of data for the purpose of analysis was the index TM/ETM+7/4 (TM/ETM+7: 2.09-2.35μm and TM/ETM+4: 0.75-0.90μm), allowing identification of various types of soils, being particularly sensitive to water separation (Lillesand et al. 2008). In addition, principal component analysis (PCA) was conducted, allowing reduction of the multi-spectrum image data redundancy resulting from the existence of correlations between the particular bands (Lillesand et al. 2008). Only the multi-spectrum channels (TM/ETM+1; 2, 3, 4, 5, 7) were selected for the PCA analysis, thus omitting the thermal and panchromatic band. The images presenting the first and second principal component (PC1 and PC2), transferring ca. 95% of information from 6 channels, were selected as the layers for further analysis.

The transformed image channels, presented as channel composition: NDVI, TM/ETM+7/4, PC1 and PC2, underwent supervised classification by means of the maximum likelihood method, separating flooded areas on the basis of juxtaposition of the test fields separated individually for each image. Upon classification, the results for images presenting only the flooded area were recorded in ESRI-ASCI format and the coordinate system was transformed to PUWG1992.

## 6.3 Results

### 6.3.1 Modelling

The model was applied to the period January 1972 – April 1980 period, for which observations at the Plosk river gauge were available. This period includes high water flows, which resulted in significant flooding events. Data available for verification was limited due to lack of registration at Ploski after 1983. Finally, the model performance was tested on observations from 1980-1983. The mean deviation of computed and observed water levels for the river gauges: Narew, Ploski and Suraż was 0.15 m.

In Figure 6.6 a comparison of computed and measured water levels for the river gauge Narew are presented. The model provides a satisfactory representation of the water level dynamics. The only event with a rather poor representation is a short-lasting peak in February 1982, where the model underestimates water levels. This is probably due to an ice-jam effect, not included in the model. The same explanation applies to peaks during winter 1980-1981.

For the Ploski river gauge, the peaks are reproduced with a similar level of accuracy, while the low water profile is noticeably underestimated. This might be explained by morphological changes in channel geometry occurring after the cease of registration. In Suraż, small discrepancies occurred for the year 1980, probably as a result of inaccuracy of the lateral inflow description arising from the untypical hydrological pattern in this period, which was characterised by multiple small rain-driven freshets.

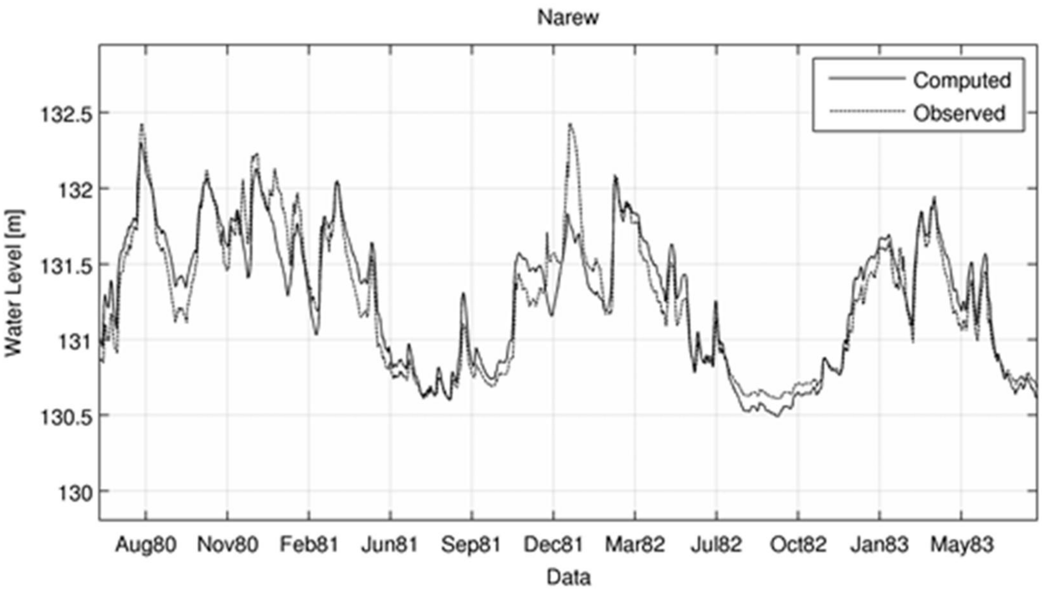


Figure 6.6 Model verification for the Narew river gauge; mean deviation 0.15 m.

### 6.3.2 Inundation delineation

The inundation extent of river waters was computed on a daily basis. The water level values calculated by the model for the cross-sections are used to determine the water extent using DEM. The digital model of the floodwater table is interpolated between cross-sections, and then raster map algebra in ArcGIS was processed on the DEM and water table elevation layer, resulting in an inundation map.

The verification period for the flow routing model is characterised by low annual flows (see Figure 6.2) compared to the mean flows in previous years and the data used for model identification. Therefore, the verification gives an insight into how the flow routing model represents the main channel dynamics rather than true floodplain flows. The ability of the applied methodology to reproduce the inundation pattern was validated by comparing the computational results with satellite data.

The tests were performed for high water events captured by satellite imagery during 03-Apr-1986, 08-Apr-1987, 11-Mar-2002, 16-Mar-2003 and 27-Mar-2005. The events had a different magnitude. The first two sets refer to periods of significant flooding, while the remaining represent very moderate freshets.

In the first stage it was analysed how the model fit changes along the river length in order to identify the potential errors in geometric representation of the terrain elevations, which should be site dependent. In Figures 6.4 and 6.6 the example variability of the inundation along the river reach is given for two plan communities as well as for the model output and the satellite data. To calculate how the inundation extend changes with the river length, the area was divided into 1 km wide strips perpendicular to the valley. It is seen that the model explains the inundation pattern well. The residuals do not depend significantly on the distance, but magnitude is different for the different plant communities.

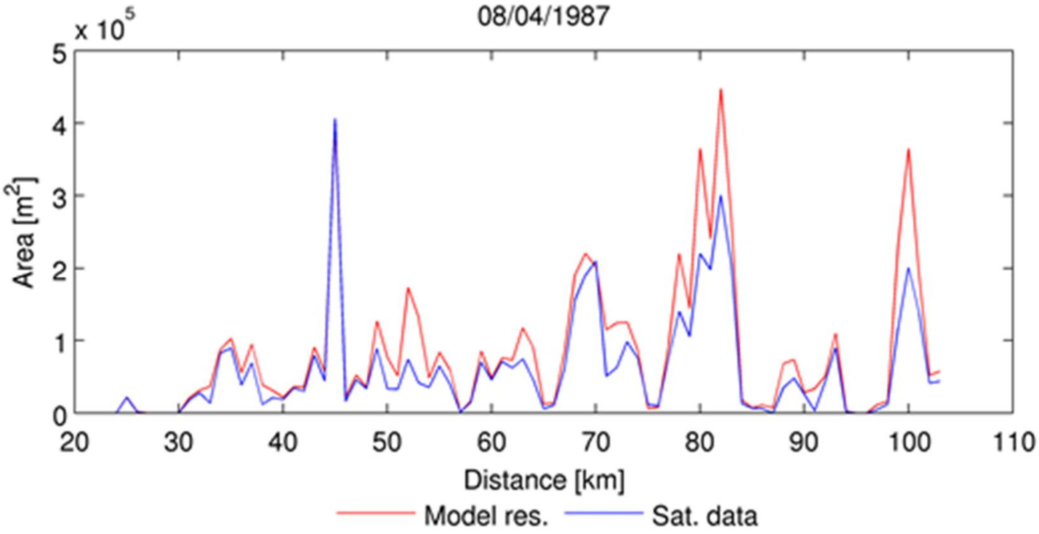


Figure 6.7 Validation of the computed inundation extent for the *All. Phragmition* community using remote sensing data; 1987 flood event; distance is measured from the Siemianówka outflow.

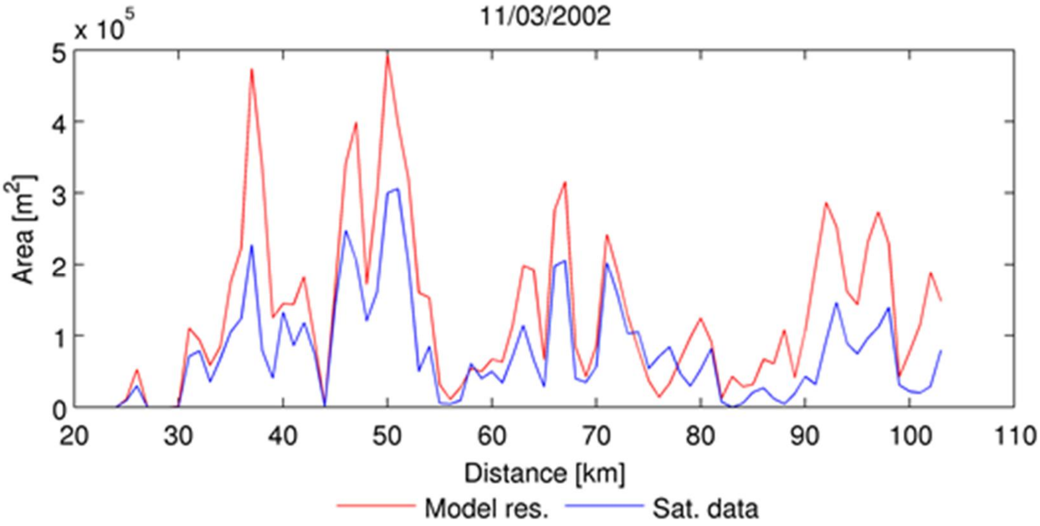


Figure 6.8 Validation of the computed inundation extent for the *All. Phragmition* community, using remote sensing data; 2002 flood event; distance is measured from the Sieminówka outflow.

Using the satellite data on the inundation extent, the relative error of the computed area was elaborated. As shown in figures 6.4 and 6.6, the error depends on the type of community. The lowest error was reported for communities of *Magnocaricion* and for all satellite data sets. The highest error occurred for the fresh grasslands for the 2003 freshet. Here, the error is not constant among the satellite sets. The highest error was found for the lower flood event (2002 and 2003). These years are in the period when less agriculture practice were elaborate in the valley. Better fit of model and satellite data was observed in flood event of 1986 and 1987. The flood of 1986 was the highest from observed flood set. High water level is easier to detect using remote sensing method, and consequently it minimize the estimation error. Better fit is also a function of management practice in the valley – inundation on the area of the cut grass is always accurately observed as open water. The highest error is observed on the area covered by natural plant communities.

### 6.3.3 Inundation variability for selected plant communities

The plant community was considered as inundated if in given time step more than half of its area was covered with water. Then, for each year, the inundation frequencies, defined as the ratio of days with inundation to days in a year, was computed.

In Figures 6.9 and 6.10 the frequencies for two plant communities are presented: *Phalaridetum arundinaceae* and *All. Phragmition*. It can be seen that floods were much more frequent in the beginning of the analysed period. The period 2000-2009 are characterised by the lowest amount of days with inundation. Moreover, in 1992, 1997 and 2001, as a result of long periods of low flow, the communities of *Phalaridetum arundinaceae* and *All. Phragmition* were totally or almost cut off from the direct river supply, respectively.

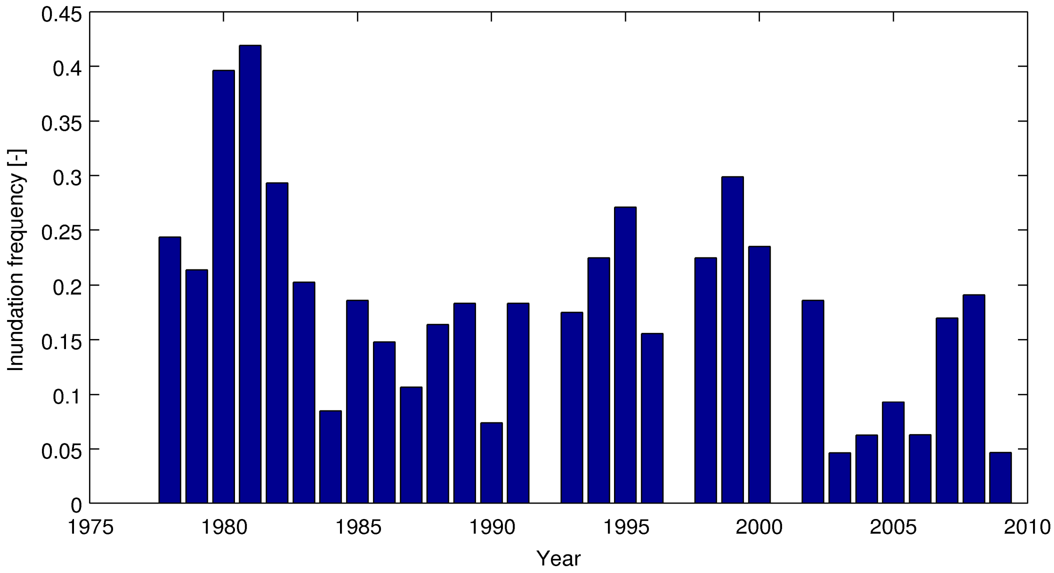


Figure 6.9 Inundation frequencies for Phalaridetum arundinaceae.

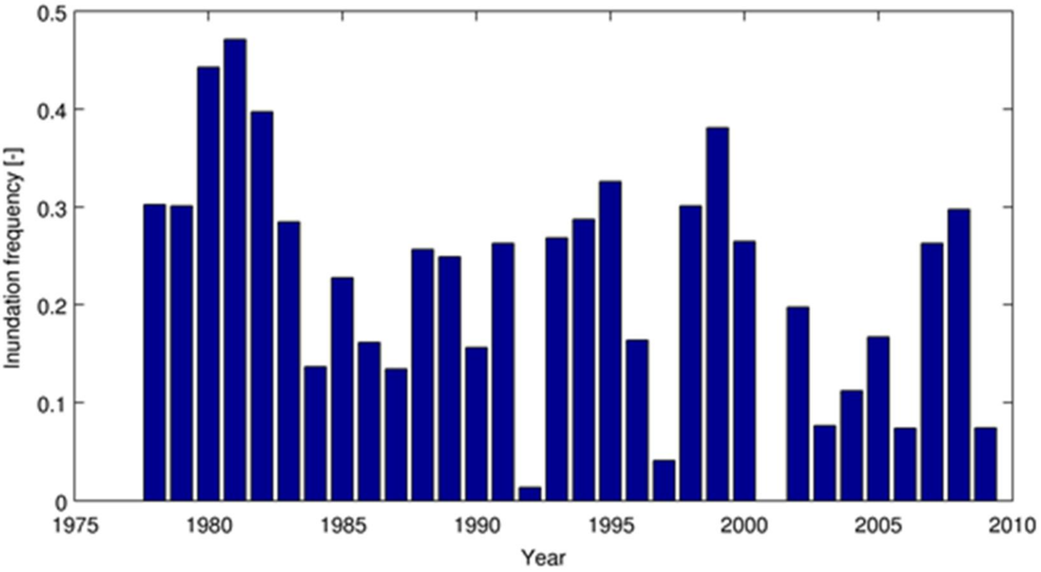


Figure 6.10 Inundation frequencies for the All. Phragmition communities.

Apart from the inundation frequencies, a map presenting the change in mean number of days in a year with inundation for periods before and after the construction of the Siemianówka reservoir is presented in Figure 6.11. As shown, large areas were affected by reduced inundation frequency.

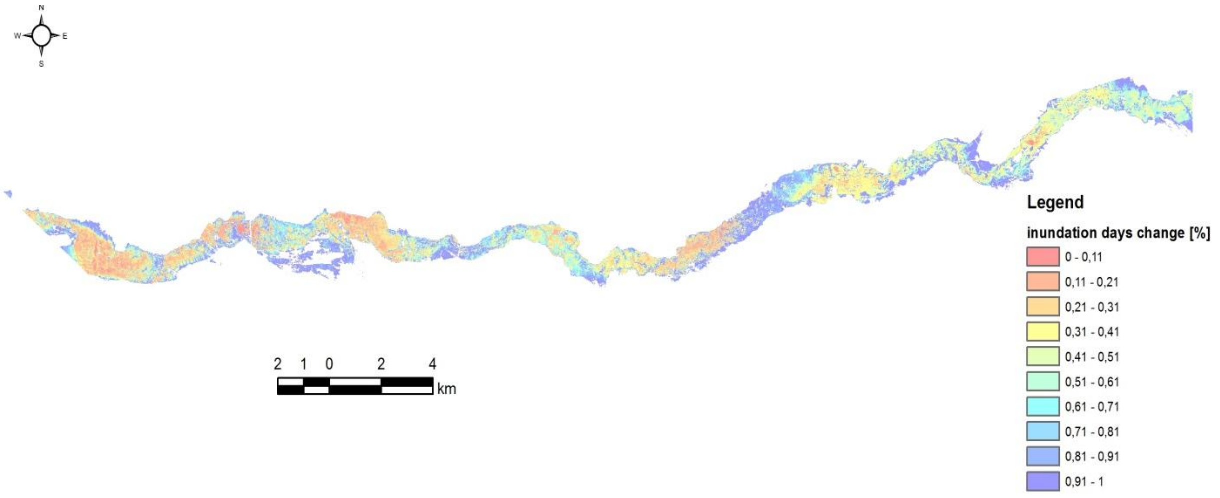


Figure 6.11 Decrease in mean number of inundated days between pre- and post-1990 periods in the upper Narew river valley.

### *6.3.4 Changes in the vegetation structure*

The analysis of changes in the vegetation in the upper Narew valley is based on existing maps of actual vegetation (Oświt 1996; Wpływ 2008). As the maps were developed using different mapping methods, the possibility of undertaking a detailed comparison was limited. Instead we used the pre-defined vegetation categories described earlier. A general overview of estimated changes in the surface areas of individual groups of vegetation is shown in Table 6.2 the difference in the number of inundation days is illustrated in Figure 6.12.

Table 6.2 Changes in the areas covered by different vegetation groups

| Vegetation group | Phytosociological unit | Trend in the coverage area |
|---|---|---|
| Common reeds | *Phragmition* | ↑ |
| Swamp sedge communities | *Magnocaricion* | ↓ |
| Wetland sedge communities | *Magnocaricion* | ↓ |
| *Phalaris* and *Phalaris-Carex* communities | *Magnocaricion* | ↑ |
| Wet meadows | *Molinietalia* | ↑ |
| Sedge-moss communities and grass-sedge meadows | *Scheuchzerio-Caricetea nigrae, Molinietalia* | ↓ |
| Fresh and grassland meadows | *Arrhenatheretalia, Vicio lathyroidis-Potentillion argenteae* | undefined |
| Sand grasslands | *Corynephorion canescentis* | undefined |
| Forests and scrubs | *Alnion glutionosae, Alno-Padion Alno-Padion* | ↑ |

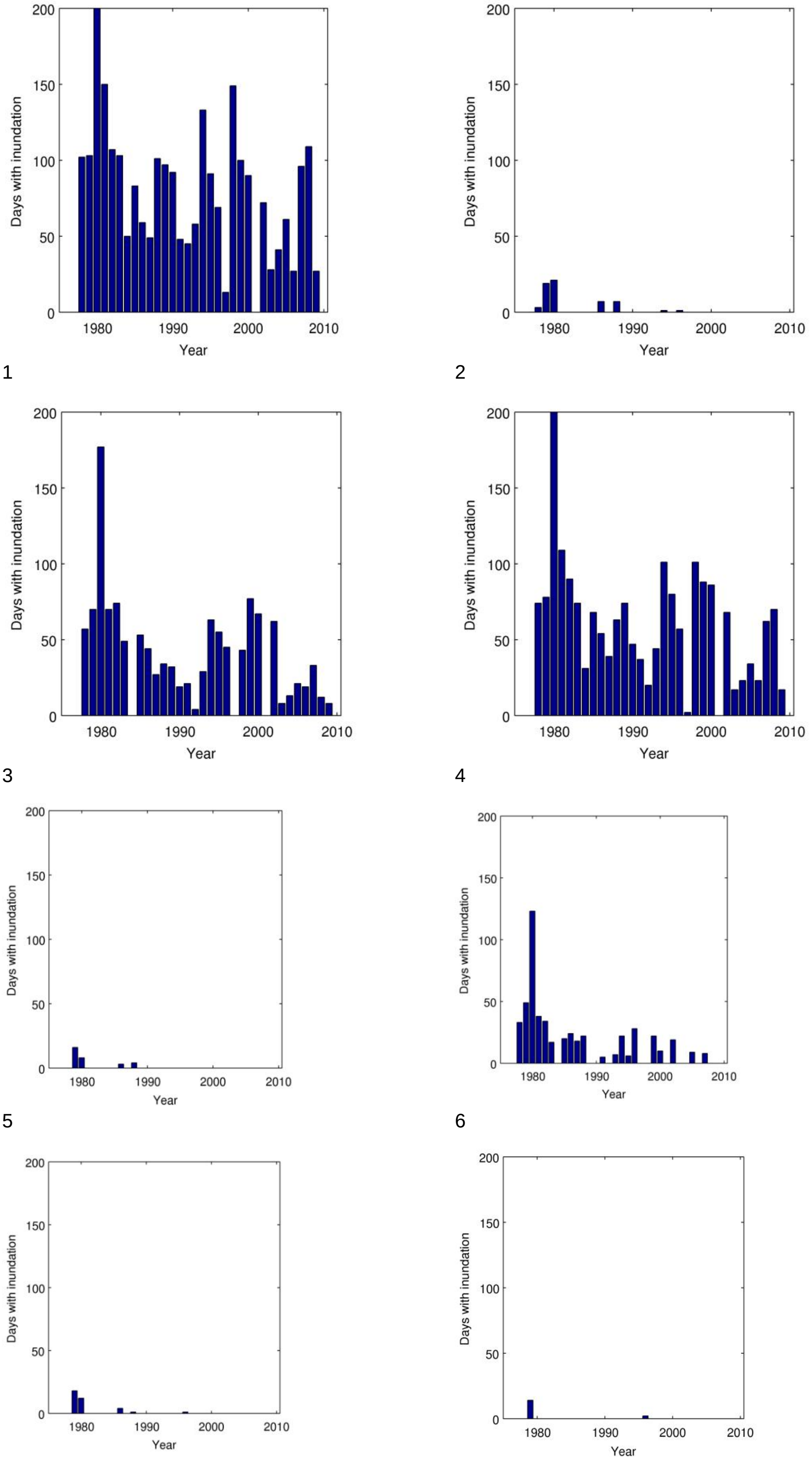
Common reeds of *Phragmition*. An increased distribution area has been observed mainly due to *Phragmites communis* encroachment on intensely flooded communities of wetland sedge communities.

Swamp sedge communities. A reduced surface area has been observed due to:(1) intense reed coverage, (2) transition into *Caricetum gracilis* or wet meadows of the *Molinietalia* order.

Wetland sedge communities. In general, their surface area has decreased. Now parts of their former surface area are covered by *Phalaris* and *Phalaris-Carex* communities, reedbeds or meadow communities. New patches of wetland sedge communities have been observed in the former habitats of swamp sedge communities or meadows.

*Phalaris* and *Phalaris-Carex* communities. General rise in surface area. They have been observed along the entire length of the valley, including areas in which they were not observed in 1990s.

Wet meadows of the *Molinietalia*. A general rise in surface area has been observed due to a number of reasons including changes in hydrology as well as reclassification of certain areas in the vegetation maps.



1 2

3 4

5 6

7 8

Figure 6.12 The number of days with inundation in a year for communities of: 1. *Phragmition*, 2 swamp sedges, 3 wetland sedges, 4 *Phalaris*, 5 Sedge-moss, 6 wet meadows, 7 fresh and grassland, 8 sand grassland.

Sedge-moss communities and grass-sedge meadows of the *Molinietalia* order. Reduction of surface area mainly and replacement by *Carex* and *Phalaris* communities as well as scrubs and forests.

The changes in the range of fresh meadows as well as rock and sand grasslands are primarily a result of differences in mapping methods. Some cases of atrophy in patches of sand grasslands in the valley result from forestation and the process of succession towards meadow communities.

Forests and scrubs. A minor rise in surface area was observed. Forest communities encroach on *Carex nigra* grass-sedge meadows flooded with groundwater and on sedge-moss communities of the *Scheuchzerio-Caricetea nigrae* class.

An analysis of the distribution of areas with the greatest changes in mean number of inundated days (80-100%) and distribution of plant communities indicates that the largest changes in the number of flooding days mainly affected areas located on the edge of valley where meadow communities occur, mainly wet meadows (Figure 6.12).

Three parts of the valley, where the greatest changes were observed (Figure 6.11), are dominated by meadows: (1) the section of the valley between the mouth of Orlanka and road bridge in the Strabla village, (2) the area around the villages of Wojszki and Ploski (3) the area around the village of Narew. These are areas where no significant changes have occurred relative to the distribution of plant communities before the construction of the Siemianówka reservoir.

## 6.4 Discussion

The presented study is based on a flow routing model that was used to describe the dynamics of the Upper Narew River. The model provides a satisfactory representation of flow dynamics; however, its precision is reduced by simplifications, especially concerning the assumption of the one-dimensional character of the flow and the inaccuracy of data. The performed analysis does not allow us to conclude whether the one-dimensional form of the model has an impact on the inundation mapping. Yet, based on the reasons described in section 3, we may assume with some degree of certainty that it does not have a significant impact on the overall result. On the other hand, application of the one-dimensional model allowed us to undertake computations for a very long period, which otherwise would require enormous computational costs, i.e. if a two-dimensional model was used.

A much more important source of error results from processes like ice jams, which were not included in the model. Here, however, an effect is expected in winter when the exchange of water between river and floodplain is limited. There is an impact on calculated inundation frequencies, which are constant for the whole simulation period, though.

The shortage of data on lateral inflow also affects the overall result. The inaccuracies in water levels, which were demonstrated in the verification and expectedly are related to/and expectedly will influence the description of the lateral inflow, were not significant.

There is no reason to believe that this should be different for other simulation periods. The local water level dynamics might also be disrupted by changes in the river geometry, which are impossible to reconstruct within such a wide time horizon. This has an impact on the main channel flow, and for the verification set it was seen only in a low water profile of the Ploski river gauge. The high water profile, shaped by the floodplain flow, should be insensitive to this effect.

The more important test of the applied methodology, considering the objectives of the study, was the performed comparison of the computed inundation extent with the extent judged from satellite imagery. It was found that accuracy depends on the land cover, i.e. plant communities, and is in opposite relation to the overall inundation extent (the lowest values of the error were obtained for the largest floods). The resulting error varies from 30% up to more than 100%. This is due to inaccuracy of the modelling method and of the classification of satellite images. The error for the different types of reeds communities can be considered acceptable. This is not the case for the grasslands where the error is roughly above 100% of the inundation area.

The remote sensing method was elaborated for the spring period, with a positive influence on the results, when leaves were not fully developed, a fact that was also noted by Profetti and Mcintosh (1996) and Townsend and Walsh (1998). A comparison of flooded areas derived from model estimates with local images shows much more significant errors. The relatively large errors can be explained by the limitations of the remote sensing method, especially when using archive data without ground truth data collected during periods with maximum flood. The method was developed based on data from 2003 and 2002 and was used for other years where verification datasets for these seasons were unavailable. The Landsat images of maximum water stage did not capture the exact maximum stage every year – and a few days’ variation from the flood peak may also influence the results. Different vegetation communities in the area that were not managed by farmers and characterised by a more developed biomass could complicate the determination of the inundation here.

Another problem influencing the results is the quality of the DEM. Relative to natural river valleys, especially in the National Parks and the Natura 2000 areas, the accuracy of ALS DEM declines significantly and overestimates the elevation by up to 0.3-0.5 m. The terrain elevation overestimation appears in natural riparian valleys where dense vegetation can influence LIDAR penetration. In case of natural floodplains of central European lowland rivers, *Carex* species of different types are dominant. This applies in particular to *Carex cespitosa* and *Carex appropinquata*, which for communities of *Caricetum cespitosae* and *Caricetum appropinquatae.* Otherwise, in a natural state and when not mown, develop tussocks protruding above the surface of the ground for dozens of centimetres. Such a microtopography feature is developed to ensure plant survival in the systematically flooded valley (Opdekamp et al. 2012) and can strongly influence ALS DEM quality and, consequently, the calculation of the inundation extent and depth (Brach and Chormanski 2014).

Among the observed changes, significant tendencies occurring in the vegetation of the Narew valley include a rise in the area of *Phalaris* and *Phalaris*-*Carex* communities as well as a rise in the surface area of communities dominated by *Phragmites communis*. Other changes in the valley vegetation structure include a decrease in the area of swamp and wetland sedge communities and a reduction in the area of *Carex nigra* grass-sedges meadows and sedge-moss communities of the *Scheuchzerio-Caricetea nigrae* class.

The pressure on river valleys, which began in the 19th century because of the demographic boom in the Polish lands, which resulted in a demand for farmland and reclamation of soil with poor workability, continued in the 20th century, especially until the 1960s. This process was halted in 1980s, and from 1990 no agricultural withdrawal of water has occurred from the marginal lands. This process has to some extent been reverted by introducing the agro-environmental schemes of EU Common Agricultural Policy.

In the case of the upper Narew valley, the greatest interference was the construction of the Siemianówka reservoir. At the same time the water regime was indirectly influenced by the drainage of the tributaries of the Narew with accelerated water flow while rising, having a draining effect on the main valley area. An increase in the surface area of *Phalaris* and *Phalaris-Carex* communities in the upper Narew valley in parts of the areas occupied by swamp and wetland sedge communities may be an indirect proof of changes in hydrological feeding.

However, in comparison with other lowland rivers, the natural environment of the upper Narew valley has avoided the occurrence of any major transformation such as implementation of intensive drainage schemes. This means that old river beds and branches of the river still form a significant impediment to farming in the valley, forcing long detours and the construction of crossings; the micro relief of the valley is an obstacle to agricultural mechanisation, and the possibility of flooding in summer complicates the rhythm of work in the fields and hay harvesting (Dembek and Danielewska 1996). The analyses performed in the first decade of the 21st century have shown that approx. 50% of the valley area is not used for agricultural purposes.

However, apart from *Phalaridetum arundinaceae*, which is associated with high fluctuations in groundwater due to flow oscillation (Oświt 1973), *Caricetum gracilis* in its meadow form has also been included in the *Phalaris-Carex* communities category. An increase in its area may point to a drop in the groundwater level and to habitat depletion characteristic of extensive reclamation (Grynia 1975). However, in wetter years, when reaping is postponed or impossible, the growth of helophytes is stronger, which makes them similar to wetter, and also swamp, variants of *Caricetum gracilis* (Oświt 1996).

Thus, a rise in the surface area of one type of reed does not provide grounds to conclude that there are permanent changes in the valley's vegetation and may merely indicate a period of drier years characterised by shorter and less intense flooding and a greater tendency of the valley to become dry in summer, which enhances the draining effect of river waters. The occurrence of *Phalaris* and *Phalaris-Carex* communities depends to a greater extent on the use of the valley.

To a very limited extent, the process of succession in the upper Narew valley leads to the development of scrubs and forests. More frequently, reed encroachment is observed. Reed expansion is related to both the fact that the land is not used and to a change in water feeding conditions, lack of long-term floods and a decrease of the humidity of the valley (Hayball and Pearce 2004).

## 6.5 Conclusions

The analysis of inundation duration for the period 1978-2009 shows that the vulnerability to changes in flood regime, induced by damming upstream, is habitat dependent and related to the duration of flooding. In the case of wetlands sedge Phalaris and Carex-Phalaris communities, their natural inundation periods are relatively short. These communities were affected by the change in the flood frequency. In contrast, habitats which are inundated for long periods are mainly associated with common reeds, wetlands sedge Phalaris and Carex-Phalaris communities, as well as wet meadows. The changes in flood frequency has not affected those sites.

Swamp sedge communities, sedge-moss communities and grass-sedge meadows as well as forests and scrubs are not directly flooded by the river. The high water table and saturated soil moisture conditions are caused by groundwater flow. As the aquifer system has been affected neither by pumping nor by intensive drainage in the valley, we assume the water condition is not affected. It suggests the ground water fed part of the valley plays an important buffering role in preservation of the protected habitats and ecosystems.

Inundation of wet meadows is very irregular. They underwent the biggest shift in hydrological patterns, however the vegetation has not been drastically changed. Over the same period vegetation structure also changed as mowing was abandoned. Regular mowing of wet meadows is more important than inundation for maintaining its stability. Extensive agriculture plays an important role in the preservation of the open landscape in this lowland river valley.

Our study has proved that natural (or semi-natural) lowland river valleys can be quite resistant to a single pressure, in this case flood frequency changes. It means that unaffected features (hydromorphology, aquifers) or long lasting, extensive human disturbance (mowing) play an important role in buffering the changes in surface hydrology.

## 6.6 References

Barkau, R. L. (1993). “UNET, one-dimensional flow through a full network of open channels”, user’s manual version 2.1., Publication CPD-66, U.S. Army Corps of Engineers, Davis, CA, Hydrologic Engineering Center.

Bates, P.D., M. D. Wilson, M. S. Horritt, D. C. Mason, N. Holden and A. Currie (2006). Reach scale floodplain inundation dynamics observed using airborne synthetic aperture radar imagery: Data analysis and modelling. Journal of Hydrology 328: 306-318.

Chatterjee, C., S. Forster and A. Bronstert (2012). Comparison of hydrodynamic models of different complexities to model floods with emergency storage areas. Hydrological Processes 22(24); 4695-4709, doi:10.1002/hyp.7079.

Chormański, J. (2003). Methodology for the flood extent determination in the Lower Biebrza Basin. Warsaw. Warsaw Agricultural University, Ph.D. Thesis.

Chormański, J., D. Świątek and R. Michałowski (2009). A hydrodynamic model coupled with GIS for flood characteristics analysis in the Biebrza riparian wetland. Oceanological Hydrobiological Studies 38(1): 65-73

Chormański, J., T. Okruszko, S. Ignar, O. Batelaan, K. T. Rebel and M. J. Wassen (2011). Flood mapping with remote sensing and hydrochemistry: A new method to distinguish the origin of flood water during floods. Ecological Engineering 37 (9): 1334-1349, DOI: 10.1016/j.ecoleng.2011.03.016.

Dembek, W. And A. Danielewska (1996). *Zróżnicowanie siedliskowe doliny Górnej Narwi od zbiornika Siemianówka do Suraża.* (w:) Rolniczo i ekologicznie zrównoważone zasady gospodarowania zasobami wodnymi i walorami przyrodniczymi w dolinie Górnej Narwi. Zesz. Probl. Post. Nauk Rol. 428, Falenty.

Grynia, M. (1975). *Przekształcanie się zbiorowisk łąkowo-pastwiskowych w ostatnich dziesiątkach lat, jako wskaźnik zmian w środowisku przyrodniczo-rolniczym.* Zesz. Probl. Nauk Roln. z 169, Warszawa.

Hayball, N. and M. W. Pearce (2004). *Influences of Simulated Grazing and Water-depth on the Growth of Juvenile* Bolboschoenus caldwellii*,* Phragmites australis *and* Schoenoplectus validus *Plants*. Aquatic Botany 78(3).

Horritt, M. S. and P. D. Bates (2002). Evaluation of 1D and 2D numerical models for predicting river flood inundation, Journal of Hydrology 268(1): 87-99, doi:10.1016/S0022-1694(02)00121-X.

Kubrak, J., T. Okruszko, D. Mirosław-Świątek and I. Kardel (2005). Recognition of Hydraulic Conditions in the Upper Narew River System and Their Influence on the Wetland Habitats in the River Valley, Publ. Inst. Geophysics Polish Academy of Sciences, E-5 (387), 209-238

Miroslaw-Swiatek, D., T. Okruszko, J. Kubrak and I. Kardel (2008). The Use of Hydrological Characteristics for Wetland Habitats Protection in Water Management of the Upper Narew River System. In Integrated Water Management (pp. 283-293). Springer Netherlands.

Kiczko, A., and T. Ermolieva (2012). Multiple-Criteria Decision Support System for Siemianówka Reservoir under Uncertainties. In Managing Safety of Heterogeneous Systems (187-201). Springer Berlin Heidelberg.

Maksymiuk, A., K. Furmanczyk, S. Ignar, J. Krupa and T. Okruszko (2008). Analiza zmienności parametrów klimatycznych i hydrologicznych w dolinie rzeki Biebrzy. Przegląd Naukowy. Inżynieria i Kształtowanie Środowiska 17(3 [41]).

Mason, D.C., D. M. Cobby, M. S. Horritt and P. D. Bates (2003). Floodplain friction parameterization in two-dimensional river flood models using vegetation heights derived from airborne scanning laser altimetry. Hydrological Processes 17: 1711-1732

Okruszko, T. and A. Kiczko (2008). Assessment of water requirements of swamp communities: the river Narew case study. Publ. Inst. Geophysics Polish Academy of Sciences, E-9 (405): 27-39.

Okruszko, T., H. Duel, M. Acreman, M. Grygoruk, M. Flörke and C. Schneider (2011).

Broad-scale ecosystem services of European wetlands—overview of the current situation and future perspectives under different climate and water management scenarios. Hydrological Sciences Journal 56(8): 1-17.

Opdekamp, W., J. Teuchies, D. Vrebos, J. Chormański, J. Schoelynck, R. van Diggelen., P. Meire and E. Struyf (2012) Tussocks: Biogenic Silica Hot-Spots in a Riparian Wetland. Wetlands 32: 1115-1124.

Oświt, J. (1973): *Naturalne łąki mozgowo-mannowe w dolinie górnej Narwi*. Zesz. Probl. Post. Nauk Rol. z. 134, Warszawa.

Oświt, J. (1996). *Roślinność, wartość rolnicza i produkcyjność łąk łęgowej części doliny Górnej Narwi.* (w:) Rolniczo i ekologicznie zrównoważone zasady gospodarowania zasobami wodnymi i walorami przyrodniczymi w dolinie Górnej Narwi. Zesz. Probl. Post. Nauk Rol. 428, Falenty.

Plit, J. (2008). *Zarządzanie krajobrazem dolin rzecznych*. Prace Komisji Krajobrazu Kulturowego Nr 10. Komisja Krajobrazu Kulturowego PTG, Sosnowiec.

Profeti, G. and H. Macintosh (1997). Flood management through Landsat TM and ERS SAR data: a case study. Hydrological Processes 11: 1397-1408.

Romanowicz, R. J., A. Kiczko and J.J. Napiórkowski (2010). Stochastic transfer function model applied to combined reservoir management and flow routing. Hydrological Sciences Journal–Journal des Sciences Hydrologiques 55(1): 27-40.

Świątek, D., Chormański, J., 2007. The verification of the numerical river flow model by use of Remote Sensing. In: Okruszko, T., Maltby, E., Szatyłowicz, J., Świątek, D., Kotowski, W., (Eds), Wetlands: Monitoring, Modelling, Management. Taylor and Francis/Balkema, The Netherlands, pp. 173-180.

Townsend, P. A. and S. J. Walsh (1998). Modeling floodplain inundation using integrated GIS with radar and optical remote sensing. Geomorphology 21: 295-312.

*Wpływ redukcji zalewów oraz regresu rolnictwa na siedlisko łęgu rozlewiskowego*. Projekt KBN nr 2 P06S 043 28, Instytut Melioracji i Użytków Zielonych, Falenty 2008.

van der Sande, C.J., S. M. de Jong and A. P. J. de Roo (2003). A segmentation and classification approach of IKONOS-2 imagery for land cover mapping to assist flood risk and flood damage assessment. International Journal of Applied Earth Observation 4: 217-229.

# 7 Riparian and Floodplain Invertebrates Hydromorphology Interactions - Scottish Case study

## 7.1 Introduction

In line with the Description of Work, described for D3.4, in the text of Task 3.4 we measured the influence of hydromorphological degradation on riparian zone invertebrate fauna and analysed the relationship in the context of both local and catchment processes. We address reduced hydrological connectivity (the study sites have regulated flows) using existing data sets. Spatial (along the river valley) and temporal (seasonal) surface water dynamics in the riparian zones, including flood dynamics, were included in the analysis. We address the relevance of such studies to the implementation of the Habitats and Species Directive by including Special Area of Conservations in our study, the Rivers, Tay (EU Code UK0030312) and Tweed (EU Code UK0012691), and place our findings in the context of 'natural - reference' processes described in WP2. The riparian sites we use are actually mid-channel bars 'islands', which on the four study rivers represent areas of riparian vegetation that reveal the influence of regulation and landscape management more clearly as they tend to be less subject to other local management activities (e.g. grazing or mowing) than the river margins. The effects of flood inundation and catchment land use on invertebrate biodiversity and trait assemblages are addressed and placed in the context of other process-related drivers. Below we describe the theoretical background to the work.

The frequency and intensity of inundation by flood waters driven by the management of discharge, or as predicted with anthropogenic global climate change (IPCC, 2013), are known to affect invertebrate diversity and abundance of riparian and floodplain habitats [Gerisch et al 2012, Lambeets, et al 2008; Rothenbucher & Schaefer et al 2006; Brose, 2003; Ellis et al 2001). Episodic or periodic flooding can directly eliminate organisms and/or shape local habitat structure to influence population and community dynamics (Moretti et al 2009; Jonsson et al 2009; Herrando et al 2002; Perdomo et al 2012). Invertebrate species will persist, perish or migrate when the environment is flooded, whilst immigration re-establishes as floods recede. This can influence the diversity of frequently flooded habitats, for example, it has been shown that the species richness of riparian spider assemblages was reduced by increased flood intensity, whereas carabid beetle species richness peaked at intermediate levels of flooding. In this deliverable, river flow was used as a proxy for flooding of riparian habitat and characterised using the median annual maximum flood peak (QMED) and specific stream power (SSP) as a descriptor of the stream energy at a particular flow and geographic co-ordinates (Environment Agency 2008).

Modification of river channels or flows will affect the hydrological deposition of sediments and subsequent stabilization by vegetation and ultimately the area and stability of riparian and in-stream terrestrial habitats (e.g. islands or mid-channel bars) (Deliverables 2.1 and 2.2). Island biogeography, metapopulation and metacommunity predict that habitat size and isolation from sources of colonists lead to the dynamic extinction, recolonisation or turnover of species, variable population sizes and modified community structure [MacArthur & Wilson 1967; Vandermeer & Carvajal 2001). Subsequent and repeat flooding of these riparian environments through anthropogenic regulation of discharge (or as a consequence of climate change) are expected to produce gradients in habitat structure that filter the composition of the ecological community (Sydenham et al 2014, Hoiss et al 2012). For instance, disturbance from floodwaters will over time alter the local plant community structure and diversity and this is widely understood to have ramifications for animal diversity (Tews et al 2004, Brose et al 2003, Vanbergen et al 2007, Woodcock et al 2009) and the biological interactions that underpin ecological processes such as pollination, herbivory and predation (Vanbergen et al 2006, 2014).

In addition to the disturbance from flood waters at a local habitat scale, it is likely that the context of the flooded environment in the wider landscape will also have a role in shaping invertebrate diversity and abundance. Land use and management including conservation are driven by policy (e.g. CAP, Birds and Habitats Directives), which can produce gradients in landscape structure. Such anthropogenic modification of landscape structure can as a consequence of niche requirements and species dispersal among habitats affect population and community dynamics (Driscoll & Weir 2005, Ducatez et al 2014). Landscape heterogeneity is known to influence the species richness and abundance of soil invertebrates (Eggleton et al 2005, Sousa 2006), carabid beetles and spiders (Vanbergen et al 2005, 2010, Billeter et al 2008, Purtauf et al 2005) , pollinators (Oliver et al 2010, Kennedy et al 2013), birds and trophic interactions (Roland & Taylor 1997, Thies et al 2003). Thus in addition to the direct (e.g. mortality and extinction) and indirect (e.g. habitat area and vegetation structure) effects of floods, it is expected that the invertebrate biodiversity of riparian habitat islands would be partly governed by the composition of the surrounding landscape.

Species extinctions or other biodiversity changes tend to be non-random with species possessing certain eco-evolutionary traits (e.g. higher trophic level, rarity and low dispersal ability) prone to be vulnerable to particular environmental stressors (e.g. habitat fragmentation) (Raffaelli 2004). Species functional traits are morphological, physiological or behavioural adaptations to biotic interactions and abiotic conditions, which influence survival, fitness and ultimately ecosystem functions (Pewtchey & Gaston 2006). Traits can predict species sensitivity to environmental stress , hence the diversity and dominance of functional traits in the community may be altered by disturbance (Ribera et al 2001) or landscape structure (Woodcock et al 2010, Vandewalle et al 2010, Barbaro & van Halder 2009, Tscharntke et al 2008). For example, floods have been reported to induce shifts in the trait diversity of predatory invertebrate communities, and low levels of bee and beetle functional diversity have been associated with intensive agriculture at the scale of a national landscape (Woodcock et al 2013). Shifts in functional diversity are likely to have implications for communities especially if functional redundancy is reduced or removed, thereby influencing the resilience or resistance of ecosystem processes (e.g. predation, pollination) to environmental changes (Gerisch 2014). Therefore, we can expect that the response of invertebrate biodiversity to the combined impacts of floods, habitat size and structure and landscape context will differ among taxa or species according to traits governing sensitivity to environmental gradients.

In this deliverable, we hypothesised that the response of river island invertebrate communities (millipedes–Diplopoda; ground beetles–Carabidae; spiders–Araneae) to *i*) river flow (QMED, SSP), *ii*) island size and habitat structure and *iii*) landscape structure would be contingent on dispersal ability (Holt et al 1999). Specifically, we predicted that the species richness and abundance of invertebrate taxa with low dispersal potential (i.e. terrestrial locomotion only) would depend on the presence of certain habitat features (e.g. tree cover), which provide refuges from island inundation by flood waters. In contrast, the abundance and richness of taxa with high dispersal potential (beetles capable of flight or spiders 'ballooning' on silk threads) were predicted to be primarily governed by niche space (indicated by correlation with island area or vegetation structure) and/or landscape structure, indicating recolonisation events from the surrounding habitat (Plum et al 2005). Finally, using the carabid beetles, the taxon with the greatest potential functional diversity, we hypothesised that the disturbance from floods, and its effects on island size or habitat structure, would filter the prevalence of species traits, thereby altering functional diversity.

## 7.2 Materials and Methods

### *Island sites*

Twenty-eight islands were surveyed in 2010 and 2011 across four river reaches (R. Earn = 6 islands, R. Tay = 6, R. Tummel = 5 and R. Tweed =11) within three catchments (Earn, Tay-Tummel, Tweed) in central and southern Scotland (Figure 7.1 and 7.2). All are upland catchments. Most of the Tay-Tummel catchment is steeply sloping with impermeable, metamorphic bedrock overlain by superficial deposits, flows are highly regulated by hydroelectric power generation and reservoirs. Land use in the Tay-Tummel catchment is mainly rough grazing and forestry. The Earn catchment is less regulated, although there is some hydroelectric production on certain stretches, the main channel follows a major geological fault; sandstone (40%) to its south, metamorphic (60%) to its north and is mostly (60%) overlain by boulder clay. Land use is dominated by rough grazing and forestry. The Tweed catchment is only modified by public water abstraction with some reservoirs and is underlain by impervious igneous formations or Silurian shale with substantial superficial deposits in the valleys. Land use is dominated by livestock grazing with some forest, improved grassland and arable land. Islands were mid-channel bars formed by hydrological deposition of sediments and subsequent stabilisation by vegetation. The perimeter coordinates of each island were mapped with a GPS (Garmin 12) and subsequently the area ($m^2$) of each island was determined using ArcGIS™ (version 9.3.1, ESRI®).

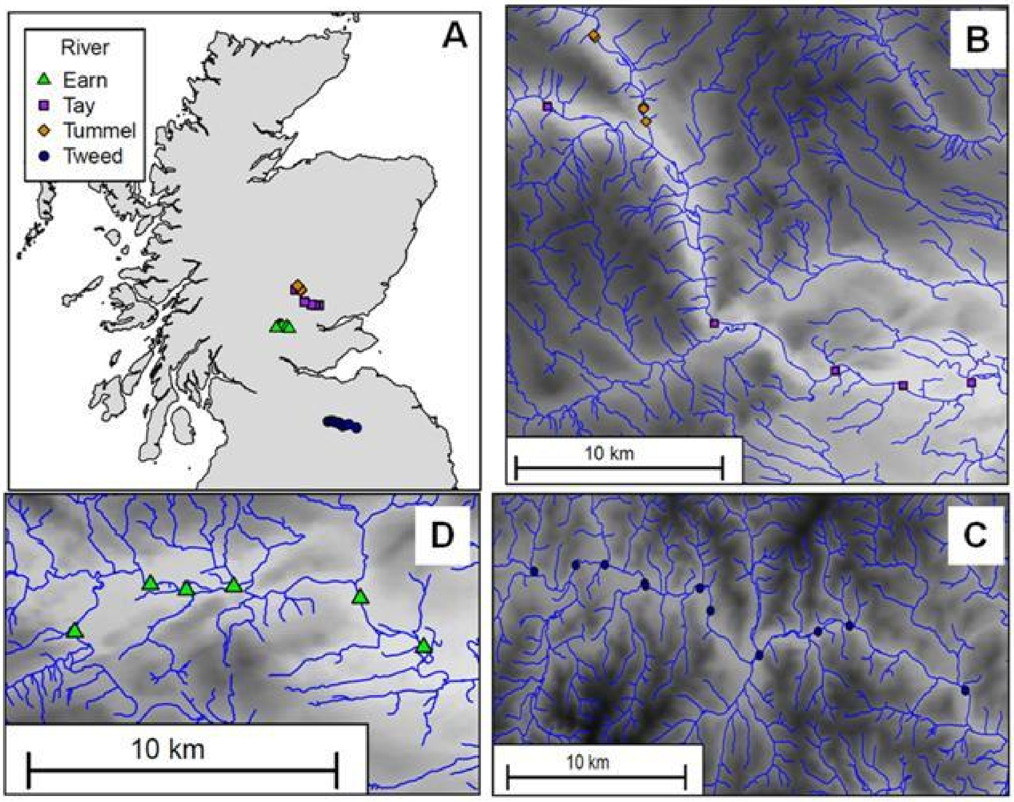


Figure 7.1 Geographic distribution of 28 river islands situated within the Rivers Tay, Tummel, Earn and Tweed in Scotland. Panels B-D are digital elevation maps (SRTM 50x50m) of catchments showing the spatial distribution of islands within the rivers (B) Tay (n= 6 islands) and Tummel (5), (C) Tweed (11) and (D) Earn (6), increasing elevation (metres above sea level) indicated by darker shading.

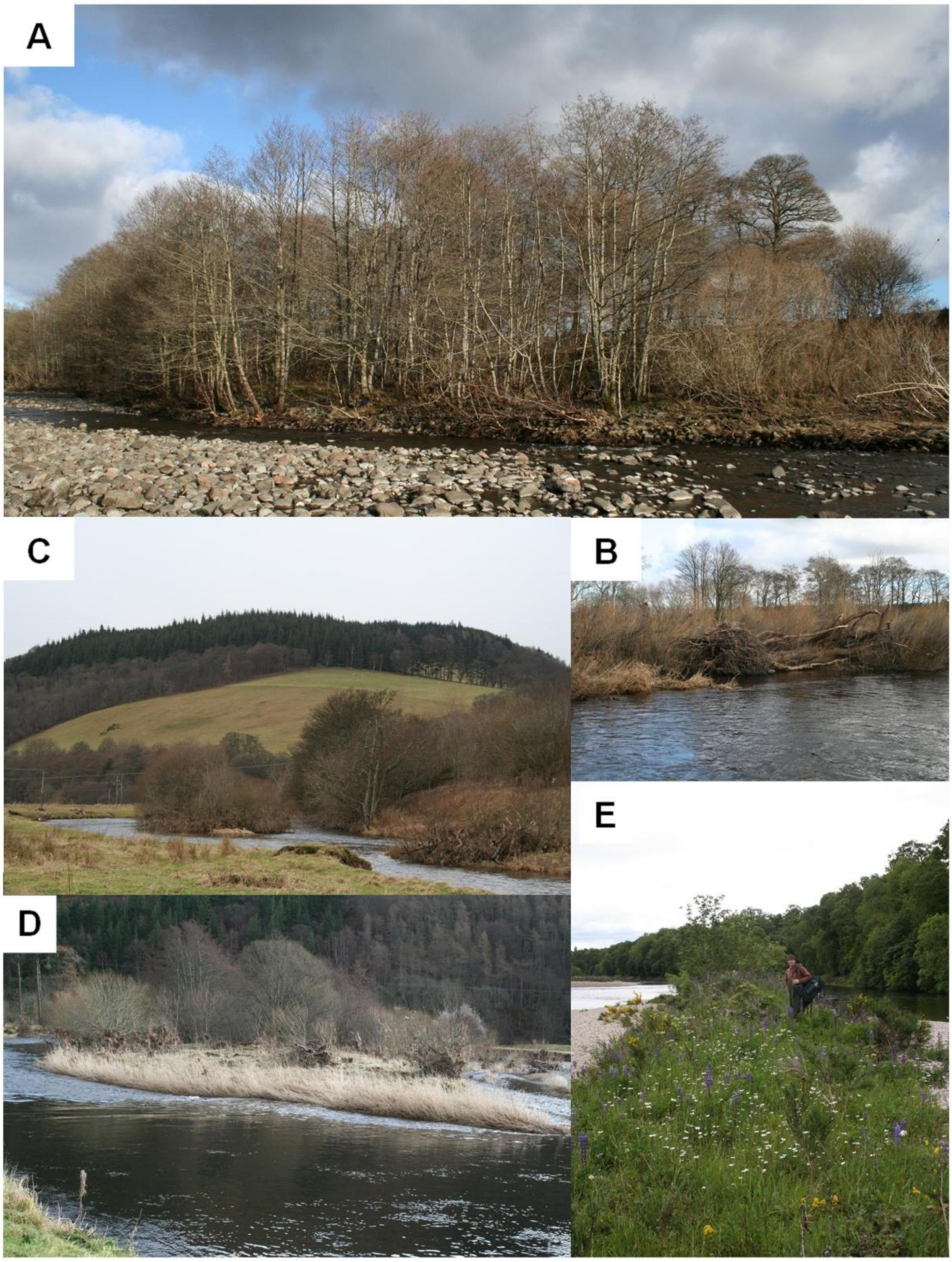


Figure 7.2 Examples of Scottish river island sites on the Rivers Earn (A and B), Tweed (C and D) and Tay (E). In 2010 and 2011, terrestrial invertebrate communities were surveyed in 27 islands that differed in habitat (unvegetated gravel to forest), size (66-78,158 $m^2$), landscape context (6-50 % cover of semi-natural habitat) and flow regulation and flooding (median annual flood peak (QMED) = 35-665 $m^{-1}$ $s^{-1}$).

*Island flood characteristics*

The disturbance to islands from river flow was characterised using the median annual maximum flood peak (QMED) and specific stream power (SSP) as a descriptor of the stream energy at a particular flow and given set of geographic co-ordinates.

Total stream power is defined as:

$\Omega = \gamma QS$

where Ω is total stream power per unit length of channel ($Wm^{-1}$), γ is the specific weight of water (9807 $Nm^{-2}$), Q is discharge ($m^3s^{-1}$) and S is the energy slope (Barker et al 2009, Lawler 1999). The total stream power was evaluated for discharge values *S* equal to the median annual maximum flood peak, QMED (i.e. the 1 in 2 year flood) characterised the high flow for each river (Knighton et al 1999). Estimates of QMED were obtained for each site using a published equation (Environment Agency 2008) that predicts QMED for ungauged sites using four different catchment descriptors (catchment area, annual average rainfall, degree of flow attenuation from upstream lakes and reservoirs and base flow characteristics as predicted from soil data). The initial estimates of QMED were subsequently refined by the degree to which the equation under- or over-estimates at similar, preferably local, gauged catchments (Kjeldesen & Jones 2010).

As a surrogate for energy slope we derived valley slope measured over 500m upstream to 500m downstream of each site. Again, this derivation was automated using established methods [54] and applied to a digital terrain model derived from interpolation of Ordnance Survey of Great Britain contour data, with a resolution of 50m x 50m x 0.1m (Morris & Flavin 1990). We screened the derived slopes for outliers, arising for example from artefacts in the digital terrain model and presence of dams within 500m upstream. As a measure of stream energy and hence flood intensity across river channels of different size, we calculated specific stream power (SSP) across the bankfull channel width at each island location:

$\omega = \Omega/W$

where ω is specific stream power (SSP = $Wm^{-2}$) and W is the bankfull width of the channel (m). Both QMED and SSP were fitted as predictor variables in subsequent GLMMs (see below).

*Island vegetation*

The community composition and structure of the herbaceous plant understory was surveyed in each island (July 2010 and 2011) in a series of six quadrats ($1m^2$) placed randomly on six of the available 10 sampling points. Within each quadrat, the identity (Stace 1997) and percentage cover of the vascular and non-vascular plant species and the mean height (mm) understorey sward (4-random points per quadrat) were determined. Tree canopy density over each quadrat was estimated using a concave spherical densiometer (Forestry suppliers Inc. USA).

*Terrestrial invertebrates*

A 20m transect was randomly situated in the centre of each island along the up- and downstream axis with sampling points located at 2m intervals. To sample the invertebrate assemblages, 10 pitfall traps comprising a polypropylene cup (8.5 cm diameter, 10 cm deep), part filled with 70% propylene glycol as a preservative and killing agent, were run for 16 weeks of continuous trapping in both 2010 and 2011 (3-7 May and 30 August) with fortnightly collections. The length of the trapping period was set to provide as complete a sample of the communities as logistically possible. Adult beetle (Carabidae), spider (Araneae) and millipede (Diplopoda) specimens were identified to species (Roberts 1987, Luff 2007, Blower 1985) and counted to provide activity density per species (juvenile spider counts were included in the overall spider density estimate, but not species richness). Activity density is proportional to the interaction between abundance and activity and this is used as a proxy of true abundance (Thiele 1977).

### *Dispersal ability, species traits and functional diversity*

From the literature, the capacity for aerial or terrestrial dispersal for each recorded invertebrate species was determined (Table S1). For the Carabidae, aerial dispersal ability was scored according to the presence (macropterous or dimorphic) or absence (brachypterous) of wings (Kotze & O'Hara, 2003, Ribera et al 1999) For the Araneae, species were scored by their ability to disperse as adults or juveniles by ballooning on silk (Lambeets et al 2008), where information on ballooning was lacking (18% of total) then species were conservatively classified as being only capable of terrestrial locomotion. Diplopoda are only capable of terrestrial locomotion (Dauber et al 2005).

Using an existing dataset on carabid species traits, we also explored the relationship between functional traits describing carabid species' ecology which influence responses to disturbance, habitat heterogeneity and ability to persist in anthropogenic landscapes (Cole et al 2002). A full description of the traits and their relevance to an assessment of community responses to disturbance from floods and local and landscape structure are given in Table 1. For each island assemblage, the functional diversity of carabid beetles was determined using the 'Functional Dispersion' index (*FDis*) (Lailberte & Legrendre 2010) implemented in package 'FD' (R version 2.14.1). The *FDis* index represents the average distance of species in multidimensional space from a centroid defined by a distance matrix, weighted by the relative abundance of the individual species. Species with greater abundance will have a greater effect on the value of *FDis*, but this index is not affected by species richness . All traits in the analysis were given equal weighting. As the traits for carabid beetles (Table 1) were represented by a mixture of continuous and categorical variables, the Gower method was used to calculate the distance matrix and all traits scores were standardised to have a range of 0–1 (Gower 1971).

### *Landscape structure*

As a single measure of this landscape structure we derived the combined percentage cover of semi-natural habitats from the UK Land Cover Map (LCM 2007). This land cover map is derived from satellite-based multispectral scanners combined with ground-truthing of habitat types and represents a comprehensive and high resolution land use map for the UK [68]. Semi-natural habitat was defined here as the sum percentage cover of the broad habitat classes: bog, marsh, acid grassland, rough low productivity grassland, heather grassland, heather and dwarf shrub, deciduous woodland. Many or all of these habitats are utilised by the studied invertebrate taxa for feeding, breeding or overwintering habitat [59, 60]. Semi-natural habitat cover was quantified using ArcGIS™ (version 9.3.1, ESRI®) in a 1km buffer around each island.

Table 7.1 Carabid traits used in calculation of functional dispersion (FDis) and community–weighted mean traits (mT).

| Species trait | Trait metric | Description | Example literature |
|---|---|---|---|
| Sensory acuity | Ratios of eye : head width and antennae : body length | Defines the relative size of key sensory organs used during hunting and foraging and hence adaptations to particular habitats | [41, 62] |
| Foraging range | Femora width: length ratio | Used as an index of walking speed and so potential to avoid disturbance or migrate | [42, 43, 62] |
| Flight | Wings full, absent or dimorphic | Presence of wings affects ability to disperse and colonise habitat in fragmented landscapes | [41-43, 61] |
| Body size | Body mass (mg) *sensu Roger 1976* | Species of larger body size are often more sensitive to disturbance | [41-43, 61] |
| Diet specialisation | Collembola specialists, obligate predators or facultative herbivores or omnivores | Trophic level affects sensitivity to disturbance/landscape structure, increased habitat heterogeneity is expected to increase niches for more trophic levels | [30, 33, 40, 41, 43] |
| Life-cycle duration | Annual or biennial | Species with longer life cycles more sensitive to disturbance | [40, 60, 65] |
| Adult activity period | Summer only versus Summer and Autumn | Species with summer activity may avoid the greater frequency and intensity of floods in autumn/winter | [40, 43, 65] |
| Breeding season | Spring/Summer or Autumn/Winter | Species that breed in summer may avoid the greater flood frequency/intensity in autumn/winter | [40, 43, 60, 65] |
| Overwintering stage | Adults only or Adults and larvae/larvae only | Species that overwinter as less mobile larvae are likely to be more sensitive to disturbance from intense winter floods | [40, 43, 60, 65] |

*Statistical analysis*

The response of invertebrate species richness and abundance to flooding, local island and landscape-scale habitat structure was analysed using linear mixed models (proc mixed, SAS v9.1) with a Gaussian distribution and island site fitted as a random effect (Littell et al 1996). Where required data were log or (for carabid species traits scaled 0-1) arcsine square-root transformed to ensure that model assumptions of residual homogeneity of variance and normality were met (proc univariate SAS v9.1).

To allow our analyses to account for spatial autocorrelation mediated by river network distances, we adjusted the island spatial coordinates so that pairwise Euclidean distances calculated from the adjusted coordinates preserved both the along-river distances within catchments and the geographic distances between catchments. The mixed models accounted for residual spatial autocorrelation by assuming that correlation decays exponentially in relation to the adjusted Euclidean distance between islands within each catchment in each year.

Rarefaction (package 'vegan' R version 2.14.1) was used to standardise invertebrate species richness estimates (set to 20 individuals) and thereby control for the varying number of individuals sampled across different island sites (Gotelli & Colwell 2001). The candidate list of explanatory variables was limited in view of the limited sample size (56 observations: 28 islands observed in each of 2 years). The following 10 fixed effects were fitted to detect effects of i) floods (annual average flood peak – QMED, and specific stream power - SSP); ii) island size and habitat structure (island area, total plant species richness, % cover of herbaceous and graminoid plants and tree canopy density) and iii) landscape structure (% cover of semi-natural habitat within 1 km radius). The 'sampling year' (2010 or 2011) and 'river' (Tay, Tummel, Earn or Tweed as categorical variable) were included to capture inter-annual and spatial structure in data according to the stretch of river. Model selection was by stepwise backward elimination starting from a model that contains all ten of these variables, with relative model performance evaluated using AICc [71]. F-ratios and p–values reported are adjusted (SAS type III) for the other significant parameters retained in the final model. Degrees of freedom were estimated using Sattherthwaite's approximation [69]. Partial residual plots were produced from GLMMs to show the effect of the significant explanatory variables conditional on other fixed and random effects in the final model for each analysis.

## 7.3 Results

*Observed patterns in invertebrate diversity and abundance*

During 2010 and 2011, respectively, we collected 5101 and 6177 individual millipedes, 8577 and 5437 carabid beetles and 6037 (6869-including juveniles) and 3614 (4565) spider individuals. The species richness of all taxa tended to increase with the vegetation successional stage of the island with maximal richness attained in islands with shrubs and trees (Figure 7.3a). The carabid beetle and millipede communities were the most and least species rich, respectively (Figure 7.3a, Appendix-Table A1). Millipede abundance was greatest in vegetated islands, especially where woody shrubs and trees were present (Figure 7.3b). The abundance of carabid assemblages tended to be greatest in gravel bars and forested islands, whereas spiders were slightly more abundant in earlier successional stages (Figure 7.3b). All the spider and millipede species were common or widespread. There were, however, carabid beetle species recorded that, while not meeting Red Data Book criteria, are designated as being nationally scarce at the scale of

Great Britain (occurrence recorded in the range of 31-100 10km squares of the GB National Grid or, for less-well recorded groups, between eight and twenty vice-counties), although they can be locally abundant. These species were in order of abundance in the sample: *Bembidion saxatile* (n=1), *Bembidion stomoides* (n=1), *Bembidion bipunctatum* (n=3), *Pterostichus aethiops* (n=4), *Pterostichus cristatus* (n=20), *Trechus rubens* (n=28), *Pterostichus oblongopunctatus* (n=2003) (Appendix-Table A1).

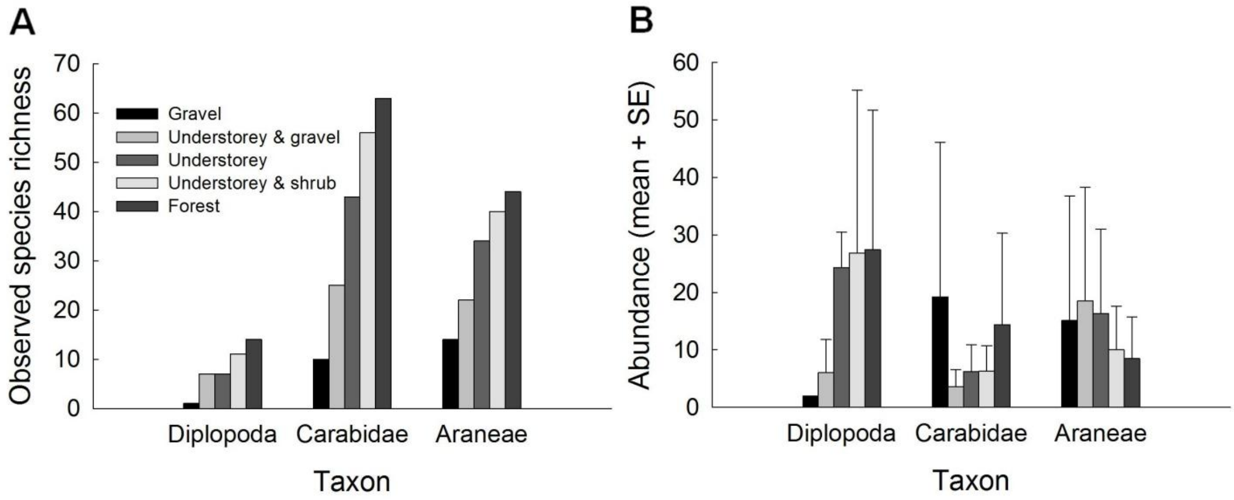


Figure 7.3 Observed species richness (A) and abundance (B) of millipedes (Diplopoda), ground beetles (Carabidae) and spiders (Araneae) in broad classes of riparian island habitat.

*Impact of flood peak and intensity on riparian invertebrates*

The response to inundation by floods varied between the different invertebrate groups. The cross-taxon (Diplopoda, Carabidae, Araneae) analysis revealed that flood peak (QMED) or intensity (SSP) had little effect on the abundance or species richness of invertebrates, irrespective of whether they dispersed either overland or aerially (Table 7.2, Appendix-Table A2). Similarly, there was no effect of either flood metric on the abundance or richness of the spiders (Araneae) or millipedes (Diplopoda) (Appendix-Table A6 &7). In contrast, an analysis of carabid beetles revealed that this community was impacted by the median annual flood peak (QMED), but not the flood intensity (SSP). Overall, the abundance of carabid beetles was negatively related to QMED (Appendix-Table A8) and this effect was greater for flightless carabid species compared with winged species (Table 7.3, Figure 7.4a). Carabid species richness was unaffected by variation in flooding (QMED or SSP) whether or not the species were classified according to dispersal (flight) capacity (Table 7.3 and Appendix-Table A4 and 5). Carabid functional diversity (*FDis*) was unaffected by QMED, SSP or the stretch of river (Appendix-Table A9). In contrast to overall functional diversity, species that overwintered as larvae were correlated negatively with QMED ($F_{1, 23}$=6.46, P =0.0183).

The abundance of invertebrates capable of aerial dispersal, flightless carabids, millipedes and spiders were all significantly affected by the stretch of river, in all probability reflecting local geographic structuring of the species pool and population sizes (Table 7.3, 7.4, and Appendix-Table A6 and 7). Catchment-specific trait assemblages, and possibly limits to the regional pool of species traits, were indicated by the river system being a significant predictor of the community-weighted mean traits (*mT*) for sensory acuity ($F_{3,24} = 4.90$, $P = 0.0085$), foraging range ($F_{3,24} = 6.00$, $P = 0.0034$), flight ability (wings full: $F_{3,25} = 19.82$, $P < 0.0001$; wings dimorphic: $F_{3,24} = 4.11$, $P = 0.0170$; wingless: $F_{3,27} = 9.61$, $P = 0.0002$ ), body mass ($F_{3,25} = 6.51$, $P = 0.0054$) and overwintering life-history stage ($F_{3,24} = 3.96$, $P = 0.0202$).

### *Influence of area and habitat structure of islands on riparian invertebrates*

The size of the island was positively related to the abundance of species capable of aerial dispersal (Araneae and Carabidae) (Figure 7.4b, Table 7.2), but was more important as a driver of carabid beetle abundance than spider abundance (Table 7.3 and Appendix-Table A7). Island area did not influence the species richness of any taxonomic group (Appendix-Tables A6-8) nor that of invertebrates grouped according to dispersal ability (flight vs terrestrial locomotion only) across taxa (Table 7.2, Appendix-Table A2 and 3) or within the taxon, the Carabidae (Table 7.3, Appendix-Table A4 and A5).

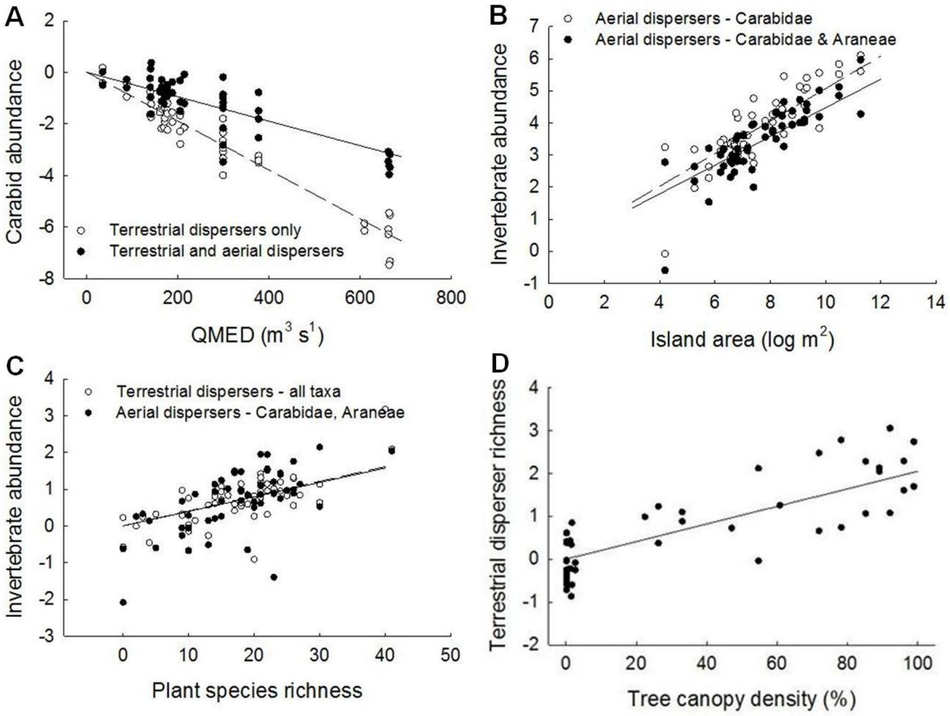


Figure 7.4 The effects on invertebrate abundance or species richness, according to aerial or terrestrial dispersal mode, of: (A) annual average flood peak (QMED) – for carabid beetles only; (B) island area; (C) local plant species richness; (D) tree canopy density (%). Plots are partial residuals on the linear predictor scale accounting for other predictors and random effects. Dashed fitted line = open symbols, solid line = closed symbols.

Carabid functional diversity (*FDis*) and most community mean traits (*mT*) were unaffected by island size (Appendix-Table A9). One exception was the preponderance of the trait for an annual or a biennial life-cycle which was respectively positively and negatively related to island area ($F_{1,52} = 9.50$, $P = 0.0049$).

The local vegetation structure of the islands was an important determinant of invertebrate abundance species richness and functional traits. The presence of a dense tree canopy was positively related to the species richness of invertebrates with limited dispersal ability (terrestrial locomotion only) (Figure 7.4d). Tree canopy cover was also positively related to the abundance of species capable of aerial dispersal across taxa (Araneae and Carabidae -Table 7.2), within a taxon (Carabidae -Table 7.3) and for different taxa (Diplopoda, Carabidae) irrespective of dispersal ability (Appendix-Table A6 and A8). The presence of understorey vegetation was also a predictor of invertebrate abundance. The plant species richness of the island was positively related to the abundance of aerial and terrestrial dispersers (Table 7.2, Figure 7.4c) and spider abundance (Appendix-Table A7), whilst the abundance of carabid beetle species that could disperse through flight was positively affected by the percentage cover of graminoid (grasses and sedges) vegetation (Table 7.3). The species richness of all invertebrate groups and dispersal modes was unaffected by the understorey vegetation (Table 7.2; Table 7.3, Appendix-Tables A2-A8).

Habitat structure in the form of understorey vegetation was also important for the species traits and functional diversity of the carabid beetle communities. Plant species richness was related positively to functional diversity ($F_{1, 53}$ =6.52, P= 0.0136) and the community prevalence of an annual life cycle trait ($F_{1, 45}$ =5.70, P= 0.0212). The percentage cover of herbaceous vegetation was also positively related to functional diversity ($F_{1, 27}$ =10.20, P = 0.0024), but also larger body mass ($F_{1, 51}$ =6.28, P =0.0154) and the preponderance of annual life cycle ($F_{1, 47}$ =8.67, P =0.0050), autumn breeding habit ($F_{1, 54}$ =5.12, P =0.0276) and larval overwintering ($F_{1, 49}$ =5.54, P =0.0226) traits. Herbaceous plant cover was also positively related to the prevalence of wing dimorphism and negatively related to the trait for fully developed wings ($F_{1, 45}$ =13.32, P=0.0007) ($F_{1, 45}$ =11.25, P=0.0016) in the carabid beetle community.

### *Influence of landscape structure on riparian invertebrate assemblages*

When species were classified according to ability to disperse aerially or solely by terrestrial locomotion, there was no detectable effect of the structure of the landscape (% cover of semi-natural habitat) surrounding each island (Table 7.2; Table 7.3, Appendix-Table A2-A5). However, analysis of each taxon separately revealed some effect of landscape structure. Spider and millipede species richness were negatively and positively related to landscape cover of semi-natural habitat, respectively (Figure 7.5a, Appendix-Table A6 and 7), whereas carabid species richness was unaffected (Appendix-Table A8). In contrast, carabid beetle abundance was positively related to the landscape cover of semi-natural habitat (Figure 7.5b, Appendix-Table A8), but spider and millipede abundance was unaffected (Appendix-Table A6 and 7). Landscape structure has no influence on the functional diversity or preponderance of species traits in the carabid beetle community (Appendix-Table A9).

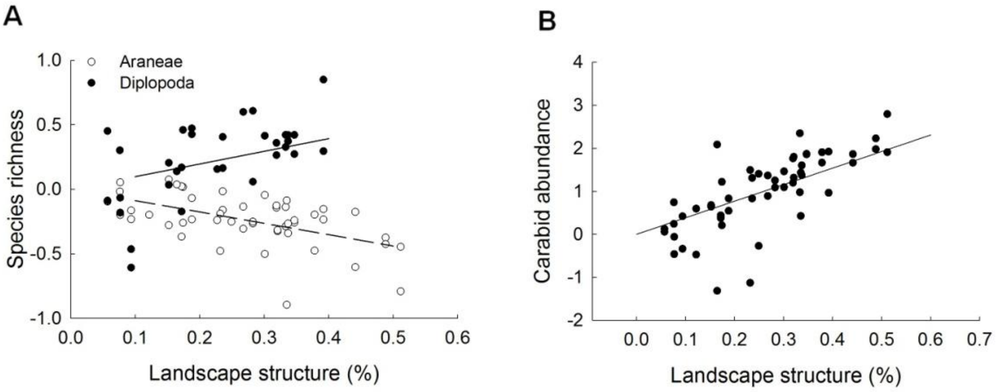


Figure 7.5 The effects of landscape structure (% cover of semi-natural habitat) on (A) spider (Araneae) and millipede (Diplopoda) species richness and (B) carabid beetle abundance. Plots are partial residuals on the linear predictor scale accounting for other predictors and random effects. Dashed fitted line = open symbols, solid line = closed symbols.

### *Spatial autocorrelation*

In all models (Table 7.2; Table 7.3, Appendix-Tables A6-9), spatial autocorrelation was explicitly accounted for, yet it was always either zero or very close to zero. This suggests either that autocorrelation was not a significant influence on these data or that the sample size is too small to meaningfully estimate the actual magnitude of it.

Table 7.2 Final linear mixed models of abundance and species richness of river island invertebrates by dispersal mode in response to floods, island size and habitat structure and landscape structure. Twenty-seven islands were sampled over two years.

| Dispersal mode (Taxa) | Predictor | Estimate | F | df | P |
|---|---|---|---|---|---|
| Aerial dispersers (Carabidae, Araneae) | | | | | |
| Abundance (log) | Year | | 10.40 | 1,27 | 0.0033 |
| *Random effect* = 0.04817 | 2010 | 1.3047 | | | |
| *Autocorrelation* = 0.0000 | 2011 | 1.6347 | | | |
| *Residual* = 0.5652 | River | | 6.05 | 1,21 | 0.0039 |
| | Earn | 0.7446 | | | |
| | Tay | 0.5383 | | | |
| | Tummel | 0.4308 | | | |
| | Tweed | 1.6347 | | | |
| | Island area (log) | 0.4471 | 21.68 | 1,21 | 0.0001 |
| | Plant species richness | 0.03938 | 7.14 | 1,35 | 0.0113 |
| | Tree canopy | 0.006772 | 5.27 | 1,20 | 0.0324 |
| Species richness | NULL | | | | |
| *Random effect* = 2.0260 | | | | | |
| *Autocorrelation* = 0.0000 | | | | | |
| *Residual* = 2.0214 | | | | | |
| Terrestrial dispersers (Diplopoda, Carabidae, Araneae) | | | | | |
| Abundance (log) | Plant species richness | 0.04035 | 5.78 | 1,35 | 0.0216 |
| *Random effect* = 2.6302 | | | | | |
| *Autocorrelation* = 0.0000 | | | | | |
| *Residual* = 0.3407 | | | | | |
| Species richness | | | | | |
| *Random effect* = 0.02047 | Tree canopy | 0.02047 | 5.97 | 1,20 | 0.0238 |
| *Autocorrelation* = 0.0000 | | | | | |
| *Residual* = 0.6934 | | | | | |

Table 7.3 Final linear mixed models of abundance and species richness of river island carabid beetles according to dispersal mode in response to floods, island size and habitat structure and landscape structure. Twenty-seven islands were sampled over two years.

| Dispersal mode (Taxon) | Predictor | Estimate | F | df | P |
|---|---|---|---|---|---|
| Aerial dispersers (Carabidae) | | | | | |
| Abundance (log) | Year | | | | |
| *Random effect* = 0.4067 | 2010 | 1.1508 | 9.87 | 1,26 | 0.0041 |
| *Autocorrelation* =*0.000* | 2011 | 0.4992 | | | |
| *Residual* = 0.5727 | QMED | -0.00474 | 13.60 | 1,23 | 0.0012 |
| | Island area (log) | 0.5068 | 14.91 | 1,23 | 0.0008 |
| | Graminoid plant | 0.02713 | 5.48 | 1,43 | 0.0240 |
| | Tree canopy | 0.01655 | 14.09 | 1,23 | 0.0010 |
| Species richness | NULL | | | | |
| *Random effect* = 2.6855 | | | | | |
| *Autocorrelation* =1.11x10$^{-16}$ | | | | | |
| *Residual* = 1.3599 | | | | | |
| Terrestrial dispersers (Carabidae) | | | | | |
| Abundance (log) | River | | 8.85 | 3,23 | 0.0004 |
| *Random effect* =0.8041 | Earn | 3.7714 | | | |
| *Autocorrelation* =*0.000* | Tay | 8.2294 | | | |
| *Residual* = 0.3874 | Tummel | 4.3776 | | | |
| | Tweed | 5.4372 | | | |
| | QMED | -0.00948 | 11.59 | 1,23 | 0.0024 |
| Species richness | NULL | | | | |
| *Random effect* = 0.09077 | | | | | |
| *Autocorrelation* = 0.0000 | | | | | |
| *Residual* = 0.01924 | | | | | |

## 7.4 Discussion

There was partial support for our hypothesis that species dispersal ability (aerial or terrestrial) would govern the response of invertebrate species richness and abundance to variability in the riparian environment (flood disturbance, local habitat area and vegetation structure or landscape structure). Furthermore, there was some evidence that the direct and indirect (island vegetation) effects of disturbance from floods played a role in filtering and/or sorting species traits in the carabid community (Moretti & Legg 2009). Below we describe the implications of our findings for both management and scientific understanding of these systems.

*Impact of flood peak and intensity on riparian invertebrates*

The river systems studied are subject to flow regulation to varying degrees. Typically the storage of water in reservoirs for water supply tends to reduce flood peaks. Discharge from hydropower schemes while tending to also reduce flood peaks can also increase the intensity of lower magnitude floods during power generation. Therefore, in terms of management recommendations it was important to consider if there were clear relationships between the peak or intensity of floods and the abundance, diversity and assemblage structure of these riparian invertebrates. In the cross-taxon analysis of invertebrate dispersal ability (Diplopoda, Carabidae and Araneae), there was no evidence of flow regulation (flood peak -QMED or intensity-SSP) directly impacting the abundance or species richness according to dispersal mode (aerial vs terrestrial). In taxon-level analysis only the carabid beetles were affected, with flood peak (QMED) - and hence the potential level of inundation of the riparian habitat - negatively related to beetle abundance. Moreover, within this taxon, the strength of this negative flood-biodiversity relationship varied according to dispersal mode with species limited to terrestrial dispersal most affected. Therefore, from a management perspective there is the potential for flow regulation to influence the assemblage of riparian invertebrates in these study systems, but it is likely to impact directly on only a component of the assemblage, the sensitive carabids, which can be considered as a potential indicator group. As flood peak is the component of the hydrograph which induces the biotic response, management of both hydropower schemes and water supply reservoirs are implicated.

This negative relationship between flood peak and carabid beetle abundance is best explained by the composition of these island communities. Of the 25 species that comprised 95% of the total carabid catch, 48% were common, eurytopic species associated with dry conditions (e.g. *Pterostichus niger*, *P. oblongopunctatus*, *Bembidion tetracolum*), another 20% and 8% were eurytopic (e.g. *P. strenuus*, *P. nigrita, Clivina fossor*) or woodland (e.g. *Calathus spp.*, *Platynus assimilis, Cychrus caraboides*) species, sometimes associated with moist habitat. Whereas, only 24% of this dominant group were hygrophilic species, often present in riparian habitats (e.g. *Agonum fuliginosum, A. micans, Patrobus atrorufus*) or habitat specialists associated with riparian shingle and gravel bar areas (i.e. *Bembidion atrocaeruleum, B. geniculatum, B. prasinum, B. punctulatum*).

This overall dominance of these riparian carabid communities by habitat generalists from the wider countryside explains the observed negative relationship with flood peak. It also suggests that in this dynamic ecosystem mass effects predominate, whereby the community structure is maintained following flood episodes by repeated immigrations, aerially or transported downstream on woody debris (Braccia & Batzer 2001), from mainland source habitats. The carabid assemblage as a whole can therefore not be considered as a specialist riparian community and its conservation value must be considered in the context of the value that such assemblages contribute to biodiversity and ecosystem services in the wider terrestrial landscape (Stockan et al 2014). Carabid beetles are key predators of herbivorous insects and slugs and as such fulfil an important function in the food web, including the biocontrol of many agricultural and horticultural pests (Symondson et al 2002, Lang et al 1999) and as food for birds (Orros et al 2014). Nonetheless, there were species recorded in these riparian habitats that are of conservation concern (nationally scarce in Great Britain). Of these scarce species *Bembidion bipunctatum, Bembidion stomoides* and *Bembidion saxatile* are riparian habitat specialists [58].

In contrast to earlier studies we found less evidence that flood disturbance altered the trait structure or functional diversity of the carabid beetle community. Nonetheless, overwintering life history strategy predicted species sensitivity to flood peak: carabid species that overwintered as larvae or adults were correlated negatively and positively with annual median flood peak, respectively. This reflects the general intolerance of carabids to submersion and hence the greater vulnerability of sedentary larvae overwintering in soil or litter to the greater severity/frequency of winter floods. Concomitantly, it may also reflect the ability of species that overwinter as adults in the landscape to be in position to rapidly re-colonise these highly disturbed areas in spring .

While some earlier studies of riparian or floodplain habitats have revealed decreases in spider abundance or richness with inundation , others have reported no effect, in agreement with our findings (Bonn et al 2002). Many spider species can tolerate submersion (Lambeets et al 2008) and immediately post-flood the spider community can attain a high level of abundance as populations rapidly re-colonise the habitat by aerial ballooning on silk threads or on rafts of debris (Ballinger et al 2005). The composition of the spider communities lends support to the view that re-colonization processes provide resilience by subsidising the spider communities. In these riparian islands, of the species comprising 95% of the spider abundance, 54% are known to be capable of ballooning (i.e. *Pardosa amentata*, *Erigone atra/dentipalpis*, *Leptorhoptrum robustum, Pardosa agricola, Bathyphantes gracilis, Bathyphantes nigrinus* and *Oedothorax spp.*) and hence can rapidly re-colonise flooded habitats. Furthermore, three of these aeronaut species (*P. amentata, L. robustum* and *O. apicatus*) are also known to be adapted to disturbed and/or moist habitats such as wet woodland, fen and marshes. In contrast to the carabid beetle assemblages where habitat generalists dominated, 47% of the spider species recorded from these islands are known to inhabit riparian habitat, and the most abundant spider species in this study (*Halorates distinctus* – 22% of total spider abundance) is a riparian or wetland specialist (Lambeets et al 2008).

Flood peak did not affect the richness and abundance of species restricted to terrestrial dispersal, this was largely due to a similar lack of an effect on the millipede (Diplopoda) assemblages which comprised the majority of this dispersal class. The restricted mobility, range size and intolerance of submersion of species within this taxon (Uetz et al 1979) mean that they are very susceptible to floods and lack the ability to quickly re-colonise habitats. Hence their occurrence was strictly limited to stable riparian habitats where flood incidence was very low and habitat features (see below) existed which enabled species persistence.

Aside from the tested flood metrics, the identity of the stretch of river was a significant predictor of abundance for carabid and spider species that disperse by air, carabid species that disperse solely through ground locomotion and for millipedes and spiders generally. Furthermore, the trait structure (*mT*) of the carabid beetle assemblages was principally affected by the identity of the river catchment (in 7/10 models). Taken together the influence of the river stretch on the abundance of these different taxa, functional groupings and species traits may reflect geographic differences in the pool of available species and traits, implying the occurrence of regional patterns in community assembly.

*Influence of area and habitat structure of islands on riparian invertebrates*

Hydro-geomorphological processes govern the development and size of mid-channel bars or islands. In many river systems the impact of flow regulation is to reduce disturbance, which allows riparian vegetation to reach later successional stages. The role of vegetation in engineering geomorphic features can become particularly important with channels becoming less dynamic. In the studied river island system the island features appear to go through a development process from initial colonisation of gravel bars by herbaceous flora, which then encourages deposition of fine sediment, which over time becomes colonised by shrubs and trees. Over time, the islands elongate and sediment accrues increasing island height above the water line, thereafter the islands are narrowed through erosion from the sides and eventually disappearing. If flow regulation has any effect on the habitat structure of these islands, it is most likely to encourage their development to later vegetation successional stages and in theory it could alter the rate of sediment accrual too. In addition to the effects of flow regulation, other management impacts on the biodiversity of these islands include the effects of channel maintenance works by government agencies and cutting of island vegetation/trees by angling associations. Management should be considered in relation to its impact on the invertebrate communities.

In this study, island vegetation structure was an important predictor of the abundance of different taxa and species dispersal groupings. Tree cover was positively related to the abundance of aerial dispersers across (Araneae and Carabidae) and within (Carabidae) taxonomic groups as well as overall for the carabid beetles and millipedes. Tree cover was also positively related to the species richness of the terrestrial dispersal group (across all taxa). Another component of vegetation structure, total plant species richness of the understorey, was also positively related to the abundance of both aerial and terrestrial dispersers. The diversity and cover of island understorey vegetation, including plant species richness, graminoid cover but especially herbaceous plant species cover (in 7/10 models), was a highly significant and frequent predictor of carabid trait structure and functional diversity (*FDisp*). Functional diversity is known to be affected by landscape structure or disturbance, but hitherto few studies have tested or shown (Woodcock et al 2012) how animal functional diversity can be affected by fine-scale vegetation structure.

These positive relationships between components of riparian vegetation and the abundance and species richness of terrestrial invertebrates imply that the concentration of plant or prey resources and/or niche space combine to support both riparian specialists and repeatedly immigrating habitat generalists (carabid beetles particularly) (Root 1973). In particular, millipedes were very reliant on woody habitat. Forest or woodland species dominated the millipede assemblage with 80% of the most abundant species (95% of the total) being forest or tree-climbing specialists (e.g. *Ommatoiulus sabulosus, Tachypodoiulus niger*). Millipedes were therefore concentrated in forested islands which either ameliorated or provided physical refuges from flood events. Millipedes are limited to terrestrial dispersal so the most likely mode of millipede immigration to these wooded islands was through downstream transportation on floating woody debris. Whilst not quantified here, substantial rafts of woody debris were deposited by flood water on these islands, particularly where it could become lodged in forest and shrub vegetation.

Island area was positively related to the abundance of species that could disperse by air (most spider and carabid beetles here), this concurs with other studies that have shown a variety of population density responses to island area e.g. Connor et al (2000). Larger islands may simply be more apparent to actively flying beetle species or for passively ballooning spiders represent a higher stochastic probability of landfall. Contrary to predictions of island biogeographical theory, we found no effect of island area on species richness, but this is consistent with other studies that have shown neutral or negative effects of island area on species richness in island ecosystems (Wardle et al 2003). One likely explanation is that these river islands are simply insufficiently isolated for area effects to prevail over dispersal processes. Another is that some of the un-vegetated gravel bars, which tend to support low species richness, were as large as the forested islands and it is habitat heterogeneity not area that is paramount in maintaining diversity in these systems.

Ecological niche differentiation can promote species coexistence and hence diversity by partitioning food and habitat resources, producing spatial and temporal heterogeneity and facilitating biological interactions (Liebold & mcPeek 2006). Fine-scale habitat heterogeneity or environmental gradients are known to affect animal taxonomic diversity .

### *Influence of landscape structure on riparian invertebrate assemblages*

Rivers across Europe are protected under various EU legislation and although this legislation is focused on particular species or habitats, the particular wording of the habitat types is very inclusive. It is not surprising therefore that *Alnus glutinosa* (alder) riparian vegetation is one of the most common habitat forms used in designating Special Areas of Conservation across Europe and across all landscape forms. The islands studied support riparian vegetation structures which are complicit with that habitat type. As the EU Habitat and Species Directives place no spatial limit on any factor which impacts on the designation status of a site, it is important, from a legislative perspective, to view the invertebrate assemblages supported by this habitat in a broader spatial context. Moreover, from a conservation perspective the invertebrate assemblages inhabiting this riparian habitat has to be assessed in the broader context of landscape management of terrestrial and freshwater habitats.

Our expectation was that the more dispersive organisms would be most sensitive to gradients in surrounding landscape structure because their abundance or richness on island sites was predicted to be a function of immigration from source habitats. However, we found that landscape structure defined as the percentage cover of semi-natural habitat had no influence on abundance or species richness when species were grouped according to their dispersal mode either across taxa (Diplopoda, Carabidae and Araneae) or within taxon (Carabidae). Furthermore, there was no influence of landscape structure

on any functional traits or functional diversity of the carabid assemblages.
In contrast to functional traits or dispersal mode, there were some significant effects of landscape structure on different taxa. In contrast to earlier studies millipede species richness was unaffected , but landscape structure did exert a weak positive influence on millipede abundance. As detritivores, millipedes are often closely associated with forest habitats that provide a dense litter layer (Emmerling 1995) and forest specialists adapted to climb trees dominated the wooded islands (see above). Proximity to source habitat is known to influence rates of recolonisation and the direction and end point of community recovery following disturbance for millipedes (Gongalsky & Persson 2013, Redi et al 2005) and soil microarthropods. Landscape cover of semi-natural habitats included marsh, grassland and moorland (73% of the total land-cover class) and deciduous forest (range 0-30%, mean = 12% of 1km buffer surrounding each island) with the latter being a particular source of forest-adapted millipede colonists.

The positive relationship between carabid abundance and the cover of semi-natural habitats reflected the dominance of island assemblages by habitat generalists (see above) able to use a variety of forest and open, semi-natural habitats and rapidly colonise the islands through flight from these different habitats in the wider landscape. Species richness of spiders was related negatively to the cover of semi-natural habitat, most likely due to the predominating riparian species (47%) in this assemblage being ill-adapted to the drier grassland, woodland and moorland habitats that comprised much of this semi-natural habitat class.

## 7.5 Conclusions

Only one of the three studied invertebrate taxa (Carabidae) was affected negatively by floods. High abundance of Carabidae could be used to indicate that a system is not affected by floods, an indication that the system is hydrologically impaired. More important as environmental factors, for all the studied invertebrate taxa, were the size and habitat structure of the riparian habitat. Habitat features (trees, species-rich plant communities) either directly ameliorated the effects of floods or provided sufficient habitat or food resources to offset any impact from floods. While less important than the size and local structure of the riparian habitat, the amount of semi-natural habitat in the surrounding landscape provided a potential source of colonists to subsidise the invertebrate populations on the islands. Overall, the insensitivity of these riparian invertebrate assemblages to flow peak or intensity floods suggests that the community structure is resilient due to the capacity for rapid colonisation and re-establishment by species either actively or passively (air and water transport). The diversity and preponderance of functional traits in the carabid beetle assemblages differed by river catchment, indicating geographic structuring of species trait pools, and were further filtered by local vegetation structure and, for one specific trait, flood peak. The taxonomic and functional assembly of these riparian communities appears to be governed by multiple processes (e.g. mass effects, extinction-recolonisation, resource concentration) arising from the combination of disturbance from episodic floods, local habitat heterogeneity and wider landscape structure. These river islands represent patches of semi-natural habitats in an otherwise intensively managed agricultural and silvicultural landscape. The biodiversity of these riparian habitats can be best maintained by management plans which promote island successional processes and maintain the cover of vegetation, especially woodland. Furthermore, these riparian islands are part of a network of semi-natural woodland, grassland and marsh habitats which require protection and enhancement for the conservation of biodiversity, including some scarce species, and ecosystem services in the wider landscape.

## 7.6 References

Atkinson, P.W., R.J. Fuller, J.A. Vickery, G.J. Conway, J.R.B. Tallowin, R.E.N. Smith, K.A. Haysom, T.C. Ings, E.J. Asteraki, and V.K. Brown 2005. Influence of agricultural management, sward structure and food resources on grassland field use by birds in lowland England. Journal of Applied Ecology, 42: 932-942.

Ballinger, A., R. Mac Nally, and P.S. Lake, 2005. Immediate and longer-term effects of managed flooding on floodplain invertebrate assemblages in south-eastern Australia: generation and maintenance of a mosaic landscape. Freshwater Biology, 50: 1190-1205.

Barbaro, L. and I. van Halder, 2009. Linking bird, carabid beetle and butterfly life-history traits to habitat fragmentation in mosaic landscapes. Ecography, 32: 321-333.

Barker, D.M., D.M. Lawler, D.W. Knight, D.G. Morris, H.N. Davies, and E.J. Stewart, 2009. Longitudinal distributions of river flood power: the combined automated flood, elevation and stream power (CAFES) methodology. Earth Surface Processes and Landforms, 34: 280-290.

Billeter, R., J. Liira, D. Bailey, R. Bugter, P. Arens, I. Augenstein, S. Aviron, J. Baudry, R. Bukacek, F. Burel, et al., 2008. Indicators for biodiversity in agricultural landscapes: a pan-European study. Journal of Applied Ecology, 45: 141-150.

Blower, J.G., Millipedes: Keys and notes for the identification of the species. Synopses of the British Fauna 1985 The Linnean Society of London/The Estuarine and Brackish Water Sciences Association.

Bonn, A., K. Hagen, and D. Wohlgemuth-Von Reiche, 2002. The significance of flood regimes for carabid beetle and spider communities in riparian habitats - A comparison of three major rivers in Germany. River Research and Applications, 18: 43-64.

Braccia, A. and D.P. Batzer, 2001. Invertebrates associated with woody debris in a southeastern US forested floodplain wetland. Wetlands,. 21: 18-31.

Brose, U., 2003. Bottom-up control of carabid beetle communities in early successional wetlands: mediated by vegetation structure or plant diversity? Oecologia, 135: 407-413.

Brose, U., 2003. Regional diversity of temporary wetland carabid beetle communities: a matter of landscape features or cultivation intensity? Agriculture Ecosystems and Environment, 98: 163-167.

Burnham, K.P. and D.R. Anderson, 2002: Model selection and multimodel inference: a practice information-theoretic approach Springer Verlag, New York.

Cole, L.J., D.I. McCracken, P. Dennis, I.S. Downie, A.L. Griffin, G.N. Foster, K.J. Murphy, and T. Waterhouse, 2002. Relationships between agricultural management and ecological groups of ground beetles (Coleoptera : Carabidae) on Scottish farmland. Agriculture Ecosystems and Environment, 93: 323-336.

Connor, E.F., A.C. Courtney, and J.M. Yoder, 2000. Individuals-area relationships: the relationship between animal population density and area. Ecology, 81: 734-748.

Dauber, J., T. Purtauf, A. Allspach, J. Frisch, K. Voigtlander, and V. Wolters, 2005. Local vs. landscape controls on diversity: a test using surface-dwelling soil macroinvertebrates of differing mobility. Global Ecology and Biogeography, 14: 213-221.

Dawson, F.H., D.D. Hornby, and J. Hilton, 2002. A method for the automated extraction of environmental variables to help the classification of rivers in Britain. Aquatic Conservation-Marine and Freshwater Ecosystems, 12: 391-403.

Driscoll, D.A. and T. Weir, 2005. Beetle responses to habitat fragmentation depend on ecological traits, habitat condition, and remnant size. Conservation Biology, 19: 182-194.

Ducatez, S., A. Humeau, M. Congretel, H. Fréville, and M. Baguette, 2014. Butterfly species differing in mobility show different structures of dispersal-related syndromes in the same fragmented landscape. Ecography, 37: 378-389.

Eggleton, P., A.J. Vanbergen, D.T. Jones, M.C. Lambert, C. Rockett, P.M. Hammond, J. Beccaloni, D. Marriott, E. Ross, and A. Giusti, 2005. Assemblages of soil macrofauna across a Scottish land-use intensification gradient: influences of habitat quality, heterogeneity and area. Journal of Applied Ecology, 42: 1153-1164.

Ellis, L.M., C.S. Crawford, and M.C. Molles, 2001. Influence of annual flooding on terrestrial arthropod assemblages of a Rio Grande riparian forest. Regulated Rivers-Research and Management, 17: 1-20.

Emmerling, C., 1995. Long-term effects of inundation dynamics and agricultural land-use on the distribution of soil macrofauna in fluvisols. Biology and Fertility of Soils, 20: 130-136.

Environment Agency, 2008.Improving the FEH statistical procedures for flood frequency estimation : Science Report: SC050050, Environment Agency, Bristol, UK,

Fordham, D.A., K.T. Shoemaker, N.H. Schumaker, H.R. Akçakaya, N. Clisby, and B.W. Brook, 2014. How interactions between animal movement and landscape processes modify local range dynamics and extinction risk. Biology Letters, 10. doi: 10.1098/rsbl.2014.0198

Gerisch, M., V. Agostinelli, K. Henle, and F. Dziock, 2012. More species, but all do the same: contrasting effects of flood disturbance on ground beetle functional and species diversity. Oikos, 121: 508-515.

Gerisch, M., 2014: Non-random patterns of functional redundancy revealed in ground beetle communities facing an extreme flood event. Functional Ecology. 28: 1504–1512.

Gongalsky, K.B. and T. Persson, 2013. Recovery of soil macrofauna after wildfires in boreal forests. Soil Biology and Biochemistry, 57: 182-191.

Gotelli, N.J. and R.K. Colwell, 2001. Quantifying biodiversity: procedures and pitfalls in the measurement and comparison of species richness. Ecology Letters, 4: 379-391.

Gower, J.C., 1971. General coefficient of similarity and its properties. Biometrics, 27: 857-&.

Heino, J. and J. Alahuhta, 2014: Elements of regional beetle faunas: faunal variation and compositional breakpoints along climate, land cover and geographical gradients. Journal of Animal Ecology. 84: 427–441

Herrando, S. and L. Brotons, 2002. Forest bird diversity in Mediterranean areas affected by wildfires: a multi-scale approach. Ecography, 25: 161-172.

Hoiss, B., J. Krauss, S.G. Potts, S. Roberts, and I. Steffan-Dewenter, 2012. Altitude acts as an environmental filter on phylogenetic composition, traits and diversity in bee communities. Proceedings of the Royal Society B: Biological Sciences, 279: 4447-4456.

Holt, R.D., J.H. Lawton, G.A. Polis, and N.D. Martinez, 1999Trophic rank and the species-area relationship. Ecology, 80: 1495-1504.

Hudson, L.N., T. Newbold, S. Contu, S.L.L. Hill, I. Lysenko, A. De Palma, H.R.P. Phillips, R.A. Senior, D.J. Bennett, H. Booth, et al., 2014. The PREDICTS database: a global database of how local terrestrial biodiversity responds to human impacts. Ecology and Evolution. 4(24):4701-35.

IPCC, 2013: Summary for Policymakers. In Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change D.Q. T.F. Stocker, G.-K. Plattner, M. Tignor, S. K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex and P.M. Midgley, Editor Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Jonsson, M., G.W. Yeates, and D.A. Wardle, 2009. Patterns of invertebrate density and taxonomic richness across gradients of area, isolation, and vegetation diversity in a lake-island system. Ecography, 32: 963-972.

Kennedy, C.M., E. Lonsdorf, M.C. Neel, N.M. Williams, T.H. Ricketts, R. Winfree, R. Bommarco, C. Brittain, A.L. Burley, D. Cariveau, et al., 2013. A global quantitative synthesis of local and landscape effects on wild bee pollinators in agroecosystems. Ecology Letters, 16: 584-599.

Kjeldsen, T.R. and D.A. Jones, 2010. Predicting the index flood in ungauged UK catchments : On the link between data-transfer and spatial model error structure. Journal of Hydrology, 387 1-9.

Knighton, A.D., 1999. Downstream variation in stream power. Geomorphology, 29: 293-306.

Kotze, D.J. and R. O'Hara, 2003. Species decline—but why? Explanations of carabid beetle (Coleoptera, Carabidae) declines in Europe. Oecologia, 135: 138-148.

Laliberté, E. and P. Legendre, 2010. A distance-based framework for measuring functional diversity from multiple traits. Ecology, 91: 299-305.

Lambeets, K., F. Hendrickx, S. Vanacker, K. Van Looy, J.P. Maelfait, and D. Bonte, 2008. Assemblage structure and conservation value of spiders and carabid beetles from restored lowland river banks. Biodiversity and Conservation, 17: 3133-3148.

Lambeets, K., M.L. Vandegehuchte, J.P. Maelfait, and D. Bonte, 2008. Understanding the impact of flooding on trait-displacements and shifts in assemblage structure of predatory arthropods on river banks. Journal of Animal Ecology, 77: 1162-1174.

Lambeets, K., J.-P. Maelfait, and D. Bonte, 2008. Plasticity in flood-avoiding behaviour in two congeneric riparian wolf spiders. Animal Biology, 58: 389-400.

Lang, A., J. Filser, and J.R. Henschel, 1999. Predation by ground beetles and wolf spiders on herbivorous insects in a maize crop. Agriculture Ecosystems and Environment, 72: 189-199.

Lawler, D.M., J.R. Grove, J.S. Couperthwaite, and G.J.L. Leeks, 1999. Downstream change in river bank erosion rates in the Swale-Ouse system, northern England. Hydrological Processes, 13: 977-992.

Leibold, M.A., M. Holyoak, N. Mouquet, P. Amarasekare, J.M. Chase, M.F. Hoopes, R.D. Holt, J.B. Shurin, R. Law, D. Tilman, et al., 2004. The metacommunity concept: a framework for multi-scale community ecology. Ecology Letters, 7: 601-613.

Leibold, M.A. and M.A. McPeek, 2006Coexistence of the niche and neutral perspectives in community ecology. Ecology, 87: 1399-1410.

Littell, R.C., G.A. Milliken, W.W. Stroup, and R.D. Wolfinger, 1996. SAS system for mixed models, SAS Institute Inc.: Cary, North Carolina, USA.

Luff, M.L., The Carabidae (ground beetles) of Britain and Ireland. 2007: 2 ed. Handbooks for the Identification of British Insects. Vol. 4 Royal Entomological Society/Field Studies Council. Part 2.

MacArthur, R. and E.O. Wilson, The Theory of Island Biogeography 1967: Princeton University Press.

Moretti, M., F. de Bello, S.P.M. Roberts, and S.G. Potts, 2009. Taxonomical vs. functional responses of bee communities to fire in two contrasting climatic regions. Journal of Animal Ecology, 78: 98-108.

Morris, D.G. and R.W. Flavin. A digital terrain model for hydrology. 1990. in Proceedings of the Fourth International Symposium on Spatial Data Handling. Zurich, Switzerland

Oliver, T., D.B. Roy, J.K. Hill, T. Brereton, and C.D. Thomas, 2010. Heterogeneous landscapes promote population stability. Ecology Letters, 13: 473-484.

Perdomo, G., P. Sunnucks, and R.M. Thompson, 2012. The role of temperature and dispersal in moss-microarthropod community assembly after a catastrophic event. Philosophical Transactions of the Royal Society B-Biological Sciences, 367: 3042-3049.

Sydenham, M.A.K., S.R. Moe, Ø. Totland, and K. Eldegard, 2014: Does multi-level environmental filtering determine the functional and phylogenetic composition of wild bee species assemblages? Ecography.

Tews, J., U. Brose, V. Grimm, K. Tielborger, M.C. Wichmann, M. Schwager, and F. Jeltsch, 2004. Animal species diversity driven by habitat heterogeneity/diversity: the importance of keystone structures. Journal of Biogeography, 31: 79-92.

Rothenbucher, J. and M. Schaefer, 2006. Submersion tolerance in floodplain arthropod communities. Basic and Applied Ecology, 7: 398-408.

Sousa, J.P., T. Bolger, M.M. da Gama, T. Lukkari, J.F. Ponge, C. Simon, G. Traser, A.J. Vanbergen, A. Brennan, F. Dubs, et al., 2006. Changes in Collembola richness and diversity along a gradient of land-use intensity: A pan European study. Pedobiologia, 50: 147-156.

Thies, C., I. Steffan-Dewenter, and T. Tscharntke, 2003. Effects of landscape context on herbivory and parasitism at different spatial scales. Oikos, 101: 18-25.

Petchey, O.L., K.L. Evans, I.S. Fishburn, and K.J. Gaston, 2007. Low functional diversity and no redundancy in British avian assemblages. Journal of Animal Ecology, 76: 977-985.

Plum, N., 2005. Terrestrial invertebrates in flooded grassland: A literature review. Wetlands, 25: 721-737.

Raffaelli, D., 2004. How extinction patterns affect ecosystems. Science, 306: 1141-1142.

Petchey, O.L. and K.J. Gaston, 2006. Functional diversity: back to basics and looking forward. Ecology Letters, 9: 741-758.

Purtauf, T., J. Dauber, and V. Wolters, 2005. The response of carabids to landscape simplification differs between trophic groups. Oecologia, 142: 458-464.

Ribera, I., S. Doledec, I.S. Downie, and G.N. Foster, 2001. Effect of land disturbance and stress on species traits of ground beetle assemblages. Ecology, 82: 1112-1129.

Roland, J. and P.D. Taylor, 1997. Insect parasitoid species respond to forest structure at different spatial scales. Nature, 386: 710-713.

Roberts, M.J., The spiders of Great Britain and Ireland: Linyphiidae and check list. Vol. 2. 1987, Colchester, England: Harley Books.

Morton, D., C. Rowland, C. Wood, L. Meek, C. Marston, G. Smith, R. Wadsworth, and I.C. Simpson, Final Report for LCM2007 - the new UK land cover map., in Countryside Survey Technical Report No 11/072011. p. 112.

Moretti, M. and C. Legg, 2009. Combining plant and animal traits to assess community functional responses to disturbance. Ecography, 32: 299-309.

Orros, M., R. Thomas, G. Holloway, and M.E. Fellowes, 2014. Supplementary feeding of wild birds indirectly affects ground beetle populations in suburban gardens. Urban Ecosystems, 1-11.

Redi, B.H., R.J. van Aarde, and T.D. Wassenaar, 2005. Coastal dune forest development and the regeneration of millipede communities. Restoration Ecology, 13: 284-291.

Ribera, I., D.I. McCracken, G.N. Foster, I.S. Downie, and V.J. Abernethy, 1999. Morphological diversity of ground beetles (Coleoptera : Carabidae) in Scottish agricultural land. Journal of Zoology, 247: 1-18.

Root, R.B., 1973. Organization of a plant-arthropod association in simple and diverse habitats - fauna of collards (Brassica-Oleracea). Ecological Monographs, 43: 95-120.

Stace, C., 1997. New Flora of the British Isles, Cambridge: Cambridge University Press.

Stockan, J.A., J. Baird, S.J. Langan, M.R. Young, and G.R. Iason, 2014. Effects of riparian buffer strips on ground beetles (Coleoptera, Carabidae) within an agricultural landscape. Insect Conservation and Diversity, 7: 172-184.

Symondson, W.O.C., D.M. Glen, A.R. Ives, C.J. Langdon, and C.W. Wiltshire, 2002. Dynamics of the relationship between a generalist predator and slugs over five years. Ecology, 83: 137-147.

Thiele, H.U., 1977Carabid Beetles In Their Environments, Berlin: Springer Verlag.

Tscharntke, T., C.H. Sekercioglu, T.V. Dietsch, N.S. Sodhi, P. Hoehn, and J.M. Tylianakis, 2008. Landscape constraints on functional diversity of birds and insects in tropical agroecosystems. Ecology, 89: 944-951.

Uetz, G.W., K.L. Vanderlaan, G.F. Summers, P.A.K. Gibson, and L.L. Getz, 1979. Effects of flooding on floodplain arthropod distribution, abundance and community structure. American Midland Naturalist, 101: 286-299.

Vanbergen, A.J., A.D. Watt, R.J. Mitchell, A.M. Truscott, S.C.F. Palmer, E. Ivits, P. Eggleton, T.H. Jones, and J.P. Sousa, 2007. Scale-specific correlations between habitat heterogeneity and soil fauna diversity along a landscape structure gradient. Oecologia, 153: 713-725.

Vanbergen, A.J., R.S. Hails, A.D. Watt, and T.H. Jones, 2006. Consequences for host-parasitoid interactions of grazing-dependent habitat heterogeneity. Journal of Animal Ecology, 75: 789-801.

Vanbergen, A.J., B.A. Woodcock, A. Gray, F. Grant, A. Telford, P. Lambdon, D.S. Chapman, R.F. Pywell, M.S. Heard, and S. Cavers, 2014. Grazing alters insect visitation networks and plant mating systems. Functional Ecology, 28: 178-189.

Vanbergen, A.J., B.A. Woodcock, M. Koivula, J. Niemela, D.J. Kotze, T. Bolger, V. Golden, F. Dubs, G. Boulanger, J. Serrano, et al., 2010. Trophic level modulates carabid beetle responses to habitat and landscape structure: a pan-European study. Ecological Entomology, 35: 226-235.

Vanbergen, A.J., B.A. Woodcock, A.D. Watt, and J. Niemela, 2005. Effect of land-use heterogeneity on carabid communities at the landscape scale. Ecography, 28: 3-16.

Vandermeer, J. and R. Carvajal, Metapopulation dynamics and the quality of the matrix. American Naturalist, 2001. 158: 211-220.

Vandewalle, M., F. de Bello, M.P. Berg, T. Bolger, S. Dolédec, F. Dubs, C.K. Feld, R. Harrington, P.A. Harrison, S. Lavorel, et al., 2010. Functional traits as indicators of biodiversity response to land use changes across ecosystems and organisms. Biodiversity and Conservation, 19: 2921–2947.

Wardle, D.A., G.W. Yeates, G.M. Barker, P.J. Bellingham, K.I. Bonner, and W.M. Williamson, 2003. Island biology and ecosystem functioning in epiphytic soil communities. Science, 301: 1717-1720.

Woodcock, B.A., D.B. Westbury, A.J. Brook, C.S. Lawson, A.R. Edwards, S.J. Harris, M.S. Heard, V.K. Brown, and S.R. Mortimer, 2012. Effects of seed addition on beetle assemblages during the re-creation of species-rich lowland hay meadows. Insect Conservation and Diversity, 5: 19-26.

Woodcock, B.A., J. Redhead, A.J. Vanbergen, L. Hulmes, S. Hulmes, J. Peyton, M. Nowakowski, R.F. Pywell, and M.S. Heard, 2010. Impact of habitat type and landscape structure on biomass, species richness and functional diversity of ground beetles. Agriculture, Ecosystems and Environment, 139: 181-186.

Woodcock, B.A., C. Harrower, J. Redhead, M. Edwards, A.J. Vanbergen, M.S. Heard, D.B. Roy, and R.F. Pywell, 2013. National patterns of functional diversity and redundancy in predatory ground beetles and bees associated with key UK arable crops. Journal of Applied Ecology, 51: 142-151.

Woodcock, B.A., S.G. Potts, T. Tscheulin, E. Pilgrim, A.J. Ramsey, J. Harrison-Cripps, V.K. Brown, and J.R. Tallowin, 2009. Responses of invertebrate trophic level, feeding guild and body size to the management of improved grassland field margins. Journal of Applied Ecology, 46: 920-929.

## 7.7 Appendices

Table A1. List of invertebrate species (Araneae – spiders, Carabidae – ground beetles, Diplopoda – millipedes) and their potential for long distance dispersal (flight or ballooning vs terrestrial locomotion) sampled from 27 Scottish river islands during 2011 and 2012. Values are sum abundance.

| Taxon | Binomial | Dispersal | River | | | |
|---|---|---|---|---|---|---|
| | | | Earn | Tay | Tummel | Tweed |
| Araneae | *Agyneta conigera* | Aerial | | | 1 | |
| | *Allomengea scopigera* | Aerial | | | 3 | 1 |
| | *Alopecosa pulverulenta* | Aerial | | 1 | | |
| | *Anelosimus vittatus* | Terrestrial | | | 1 | |
| | *Anyphaena accentuata* | Terrestrial | | 3 | | |
| | *Arctosa cinerea* | Terrestrial | | 4 | 3 | |
| | *Bathyphantes approximatus* | Aerial | | | | 7 |
| | *Bathyphantes gracilis* | Aerial | 40 | 11 | 32 | 105 |
| | *Bathyphantes nigrinus* | Aerial | 10 | 43 | 4 | 41 |
| | *Centeromerita bicolor* | Aerial | | | 14 | |
| | *Clubonia compta* | Terrestrial | 1 | 2 | 2 | |
| | *Clubonia lutescens* | Terrestrial | | 14 | 4 | 12 |
| | *Clubonia reclusa* | Terrestrial | | 1 | | |
| | *Diplocephalus latifrons* | Aerial | 9 | 213 | 15 | 111 |
| | *Diplostyla concolor* | Aerial | | | | 1 |
| | *Dismodicus bifrons* | Aerial | | 1 | | 1 |
| | *Erigone atra/dentipalpis* | Aerial | 133 | 81 | 64 | 911 |
| | *Gongylidiellum vivum* | Aerial | 2 | 14 | | 4 |
| | *Gongylidium rufipes* | Aerial | | 54 | 14 | 4 |
| | *Halorates distinctus* | Aerial | 83 | 9 | 3 | 2034 |
| | *Halorates repobus* | Aerial | | 43 | 8 | |
| | *Hypomma bituberculatum* | Aerial | 6 | | | 4 |
| | *Juvenile spider* | Aerial | 248 | 359 | 235 | 694 |
| | *Lepthyphantes minutus* | Aerial | | 2 | 6 | |
| | *Lepthyphantes obscurus* | Aerial | | | 1 | |
| | *Lepthyphantes tenuis* | Aerial | 33 | 77 | 51 | 45 |
| | *Leptorhoptrum robustum* | Aerial | 219 | 38 | 99 | 741 |
| | *Linypha tringularis* | Aerial | 2 | | 2 | 2 |
| | *Lophomma punctum* | Aerial | | | | 1 |
| | *Lycosidae (Pardosa) juveniles* | Aerial | 19 | 51 | 66 | 51 |
| | *Micrargus subequalis* | Aerial | | 20 | | |
| | *Microlinypha pusilla* | Aerial | | 1 | | |
| | *Microneta viaria* | Aerial | | 1 | 1 | |
| | *Oedothorax apicatus* | Aerial | 134 | 415 | 252 | 25 |
| | *Oedothorax fuscus/retusus* | Aerial | 20 | 12 | 1 | 141 |
| | *Oedothorax gibosa* | Aerial | | | | 5 |
| | *Pachygnatha clercki* | Aerial | 5 | 3 | | 72 |
| | *Pachygnatha degereii* | Aerial | 20 | 4 | 1 | 12 |

| Taxon | Binomial | Dispersal | River | | | |
|---|---|---|---|---|---|---|
| | *Pardosa agricola* | Aerial | | 211 | 195 | 60 |
| | *Pardosa amentata* | Aerial | 325 | 551 | 102 | 854 |
| | *Pardosa pallustris* | Aerial | 1 | 9 | 2 | 2 |
| | *Pardosa proxima* | Aerial | 2 | | | 3 |
| | *Pocadicnemis juncea* | Aerial | | 5 | | 1 |
| | *Porrhomma errans* | Aerial | | 1 | 1 | |
| | *Savigyna frontana* | Aerial | 17 | 1 | 3 | 61 |
| | *Scytodes thoracica* | Terrestrial | 1 | | | |
| | *Segestria senoculata* | Terrestrial | | 2 | | |
| | *Tapinopa longidens* | Aerial | | | 3 | |
| | *Tetragnatha extensa* | Aerial | | | 1 | 1 |
| | *Tiso vegans* | Aerial | | | | 2 |
| | *Trichopterna thorelli* | Aerial | | 4 | | |
| | *Trochosa ruricola* | Aerial | 9 | 52 | 41 | 2 |
| | *Trochosa terricola* | Aerial | | | | 2 |
| | *Troxochorus scabriculus* | Aerial | 36 | 120 | 20 | 295 |
| | *Walckaneria alticeps* | Aerial | | | | 2 |
| | *Walckaneria nudipalpis* | Aerial | | | | 1 |
| | *Xytiscus cristatus* | Aerial | | | | 2 |
| | *Zora spinimana* | Terrestrial | | 1 | | |
| | *Amaurobius fenestralis* | Terrestrial | | | 1 | |
| Carabidae | *Abax parallelepipedus* | Terrestrial | 26 | 57 | 14 | |
| | *Agonum fuliginosum* | Aerial | 26 | 1 | 1 | 1047 |
| | *Agonum gracile* | Aerial | 1 | | | |
| | *Agonum micans* | Aerial | | | | 532 |
| | *Agonum muelleri* | Aerial | 13 | 12 | 2 | 330 |
| | *Agonum piceum* | Aerial | | | 1 | |
| | *Amara aenea* | Aerial | 1 | | | |
| | *Amara communis* | Aerial | | | 1 | 5 |
| | *Amara eurynota* | Aerial | | | | 1 |
| | *Amara familiaris* | Aerial | | | 1 | |
| | *Amara fulva* | Aerial | | 1 | | |
| | *Amara lunicollis* | Aerial | 3 | 1 | 1 | 2 |
| | *Amara ovata* | Aerial | | | | 6 |
| | *Amara plebeja* | Aerial | 1 | | | 2 |
| | *Amara similata* | Aerial | | | | 3 |
| | *Anchomenus dorsalis* | Aerial | 17 | 1 | | 285 |
| | *Asaphidion flavipes* | Aerial | 22 | 2 | 1 | 13 |
| | *Bembidion aeneum* | Aerial | | | | 1 |
| | *Bembidion atrocaeruleum* | Aerial | 181 | | 200 | |
| | *Bembidion biguttatum* | Aerial | 1 | | | |
| | *Bembidion bipunctatum* | Aerial | | 2 | 1 | |
| | *Bembidion bruxellense* | Aerial | 1 | | | 8 |
| | *Bembidion decorum* | Aerial | 1 | | 18 | |
| | *Bembidion femoratum* | Aerial | 2 | 17 | | 3 |

| Taxon | Binomial | Dispersal | River | | | |
|---|---|---|---|---|---|---|
| | *Bembidion geniculatum* | Aerial | 7 | | 48 | 96 |
| | *Bembidion guttula* | Aerial | 1 | | | 85 |
| | *Bembidion lampros* | Aerial | 5 | 3 | 1 | 12 |
| | *Bembidion mannerheimii* | Terrestrial | | | | 80 |
| | *Bembidion obtusum* | Aerial | | | | 3 |
| | *Bembidion prasinum* | Aerial | | | 27 | |
| | *Bembidion punctulatum* | Aerial | 17 | | | |
| | *Bembidion saxatile* | Aerial | | 1 | | |
| | *Bembidion stomoides* | Aerial | | | 1 | |
| | *Bembidion tetracolum* | Terrestrial | 117 | 34 | 42 | 632 |
| | *Bembidion tibiale* | Aerial | 1 | 2 | 40 | 6 |
| | *Calathus erratus* | Aerial | | | 5 | |
| | *Calathus fuscipes* | Terrestrial | 1 | | | |
| | *Calathus melanocephalus* | Aerial | 4 | 1 | 1 | 155 |
| | *Calathus rotundicollis* | Aerial | 20 | 385 | 21 | 51 |
| | *Carabus nemoralis* | Terrestrial | 5 | | 4 | |
| | *Carabus problematicus* | Terrestrial | 4 | 44 | 2 | |
| | *Carabus violaceus* | Terrestrial | 2 | 75 | 3 | 1 |
| | *Clivina collaris* | Aerial | | | | 7 |
| | *Clivina fossor* | Aerial | 26 | 7 | 1 | 199 |
| | *Curtonotus aulicus* | Aerial | 1 | 1 | 1 | 5 |
| | *Cychrus caraboides* | Terrestrial | 5 | 159 | 6 | 17 |
| | *Dromius quadrimaculatus* | Terrestrial | | | | 1 |
| | *Dyschirius (Damaged)* | Terrestrial | | 1 | | |
| | *Elaphrus cupreus* | Aerial | | | | 1 |
| | *Harpalus latus* | Aerial | | 1 | | |
| | *Harpalus rufipes* | Aerial | 2 | 8 | | |
| | *Laemostenus terricola* | Terrestrial | 1 | | 3 | |
| | *Leistus rufomarginatus* | Aerial | | | 1 | |
| | *Leistus terminatus* | Aerial | | 17 | | |
| | *Loricera pilicornis* | Aerial | 24 | 52 | 19 | 99 |
| | *Nebria brevicollis* | Aerial | 92 | 54 | 57 | 96 |
| | *Nebria rufescens* | Aerial | 31 | | 5 | 11 |
| | *Nebria salina* | Aerial | 2 | | | 12 |
| | *Notiophilus biguttatus* | Aerial | | 2 | 3 | 2 |
| | *Ocys harpaloides* | Aerial | 1 | | | 1 |
| | *Paranchus albipes* | Aerial | 183 | | 11 | 48 |
| | *Patrobus assimilis* | Terrestrial | | | 1 | 1 |
| | *Patrobus atrorufus* | Terrestrial | 1 | 1 | | 499 |
| | *Platynus assimilis* | Aerial | 688 | 128 | 261 | 369 |
| | *Poecilus versicolor* | Aerial | 1 | | | |
| | *Pterostichus adstrictus* | Aerial | | 1 | | 2 |
| | *Pterostichus aethiops* | Terrestrial | | | | 4 |
| | *Pterostichus cristatus* | Terrestrial | 7 | | 13 | |
| | *Pterostichus diligens* | Aerial | | 6 | 1 | 1 |

| Taxon | Binomial | Dispersal | River | | | |
|---|---|---|---|---|---|---|
| | *Pterostichus macer* | Aerial | | 1 | | |
| | *Pterostichus madidus* | Terrestrial | 51 | 2 | 1 | 20 |
| | *Pterostichus melanarius* | Aerial | 95 | 54 | 16 | 125 |
| | *Pterostichus niger* | Aerial | 241 | 1342 | 412 | 60 |
| | *Pterostichus nigrita* | Aerial | 16 | 7 | 14 | 234 |
| | *Pterostichus oblongopunctatus* | Aerial | | 1558 | 445 | |
| | *Pterostichus strenuus* | Aerial | 47 | 46 | 12 | 693 |
| | *Pterostichus vernalis* | Aerial | 25 | 3 | | |
| | *Stomis pumicatus* | Terrestrial | 1 | 5 | | 3 |
| | *Synuchus vivalis* | Aerial | 4 | 3 | 3 | 3 |
| | *Trechoblemus micros* | Aerial | | | | 4 |
| | *Trechus obtusus* | Terrestrial | 7 | 80 | | 174 |
| | *Trechus rubens* | Aerial | 11 | | 3 | 14 |
| | *Trichocellus cognatus* | Aerial | 1 | | | |
| | *Trichocellus placidus* | Aerial | | | | 2 |
| Diplopoda | *Brachydesmus superus* | Terrestrial | 18 | | 1 | 23 |
| | *Brachyiulus pusillus* | Terrestrial | 15 | 84 | 27 | 417 |
| | *Craspedosoma rawlinsii* | Terrestrial | 68 | 30 | 4 | 54 |
| | *Cylindroiulus britannicus* | Terrestrial | 28 | 82 | 3 | 123 |
| | *Cylindroiulus nitidus* | Terrestrial | | 18 | 3 | |
| | *Cylindroiulus punctatus* | Terrestrial | 5 | 45 | 6 | 14 |
| | *Julus scandinavious* | Terrestrial | 2 | 202 | 79 | 53 |
| | *Nanogona polydesmoides* | Terrestrial | 2 | 2 | | 15 |
| | *Nemasoma varicorne* | Terrestrial | | 1 | | |
| | *Ommatoiulus sabulosus* | Terrestrial | 306 | 5518 | 1170 | 649 |
| | *Ophyiulus pilosus* | Terrestrial | 76 | 804 | 189 | 305 |
| | *Polydesmus angustus* | Terrestrial | | 2 | 34 | |
| | *Polydesmus inconstans* | Terrestrial | | 3 | | |
| | *Tachypodoiulus niger* | Terrestrial | 48 | 569 | 89 | 92 |
| Total | | | 3987 | 13972 | 4582 | 14125 |

Table A2. Full linear mixed models of abundance and species richness of river island invertebrates that can disperse by flight or ballooning (Carabidae, Araneae) in response to floods, island size and habitat structure and landscape structure. Twenty-seven islands were sampled over two years. Parameters in bold were retained in final model, other fixed effects were eliminated by step-wise backwards simplification.

| Dependent | Predictor | Estimate | F | df | P |
|---|---|---|---|---|---|
| Abundance (log) | | | | | |
| | Year | | 10.40 | 1,27 | 0.0033 |
| *Random effect* | 2010 | 1.3047 | | | |
| Site = 0.04817 | 2011 | 1.6347 | | | |
| | River | | 6.05 | 1,21 | 0.0039 |
| *Spatial autocorrelation* | Earn | 0.7446 | | | |
| Catchment x site = 0.0 | Tay | 0.5383 | | | |
| | Tummel | 0.4308 | | | |
| *Residual* = 0.5652 | Tweed | 1.6347 | | | |
| | QMED | -0.00106 | 0.16 | 1,19 | 0.6966 |
| | SSP | -0.00220 | 0.20 | 1,19 | 0.6601 |
| | Island area (log) | 0.4471 | 21.68 | 1,21 | 0.0001 |
| | Plant species richness | 0.03938 | 7.14 | 1,35 | 0.0113 |
| | Herbaceous plant | 0.01573 | 0.42 | 1,37 | 0.5219 |
| | Graminoid plant | 0.004939 | 0.20 | 1,33 | 0.6598 |
| | Tree canopy density | 0.006772 | 5.27 | 1,20 | 0.0324 |
| | Landscape structure | -0.8655 | 0.25 | 1,17 | 0.6245 |
| Species richness | | | | | |
| | Year | | 0.02 | 1,6 | 0.9031 |
| *Random effect* | 2010 | 11.6778 | | | |
| Site = 2.0260 | 2011 | 11.7449 | | | |
| | River | | 0.08 | 3,22 | 0.9697 |
| *Spatial autocorrelation* | Earn | 11.3541 | | | |
| Catchment x site = 0.0 | Tay | 9.6862 | | | |
| | Tummel | 10.7127 | | | |
| *Residual* = 2.0214 | Tweed | 10.5636 | | | |
| | QMED | -0.00019 | 0.00 | 1,25 | 0.9552 |
| | SSP | 0.000438 | 0.00 | 1,15 | 0.9812 |
| | Island area (log) | -0.3762 | 2.45 | 1,25 | 0.1302 |
| | Plant species richness | -0.03721 | 1.67 | 1,12 | 0.2197 |
| | Herbaceous plant | 0.000157 | 0.00 | 1,11 | 0.9978 |
| | Graminoid plant | -0.00224 | 0.01 | 1,51 | 0.9227 |
| | Tree canopy density | 0.004822 | 0.23 | 1,24 | 0.6369 |
| | Landscape structure | -0.8634 | 0.07 | 1,28 | 0.7995 |

Table A3. Full linear mixed models of abundance and species richness of river island invertebrates that disperse by terrestrial locomotion only (Carabidae, Araneae, Diplopoda) in response to floods, island size and habitat structure and landscape structure. Twenty-seven islands were sampled over two years. Parameters in bold were retained in final model, other fixed effects were eliminated by step-wise backwards simplification.

| Dependent | Predictor | Estimate | F | df | P |
|---|---|---|---|---|---|
| Abundance (log) | | | | | |
| | Year | | 0.99 | 1,27 | 0.3280 |
| *Random effect* | 2010 | 3.2905 | | | |
| Site = 2.6302 | 2011 | 3.4577 | | | |
| | River | | 2.15 | 1,23 | 0.1209 |
| *Spatial autocorrelation* | Earn | 2.5370 | | | |
| Catchment x site = 0.0 | Tay | 4.4698 | | | |
| | Tummel | 2.4957 | | | |
| *Residual* = 0.3407 | Tweed | 3.2781 | | | |
| | QMED | 0.000518 | 0.01 | 1,18 | 0.9414 |
| | SSP | 0.009111 | 0.58 | 1,22 | 0.4542 |
| | Island area (log) | -0.01762 | 0.01 | 1,20 | 0.9441 |
| | Plant species richness | 0.04035 | 5.78 | 1,35 | 0.0216 |
| | Herbaceous plant | -0.00201 | 0.00 | 1,30 | 0.9477 |
| | Graminoid plant | 0.009399 | 0.47 | 1,35 | 0.4974 |
| | Tree canopy density | 0.01432 | 3.16 | 1,26 | 0.0873 |
| | Landscape structure | 0.1093 | 0.00 | 1,21 | 0.9748 |
| Species richness | | | | | |
| | Year | | 0.79 | 1,20 | 0.3850 |
| *Random effect* | 2010 | 4.6240 | | | |
| Site = 0.02047 | 2011 | 4.8568 | | | |
| | River | | 0.02 | 3,13 | 0.9968 |
| *Spatial autocorrelation* | Earn | 4.9321 | | | |
| Catchment x site = 0.0 | Tay | 4.8250 | | | |
| | Tummel | 4.7786 | | | |
| *Residual* = 0.6934 | Tweed | 4.6275 | | | |
| | QMED | -0.00240 | 2.01 | 1,20 | 0.1714 |
| | SSP | -0.01140 | 0.69 | 1,17 | 0.4189 |
| | Island area (log) | 0.1690 | 0.40 | 1,19 | 0.5366 |
| | Plant species richness | -0.01146 | 0.11 | 1,26 | 0.7428 |
| | Herbaceous plant | 0.06010 | 1.42 | 1,37 | 0.2401 |
| | Graminoid plant | 0.007809 | 0.10 | 1,32 | 0.7525 |
| | Tree canopy density | 0.02047 | 5.97 | 1,20 | 0.0238 |
| | Landscape structure | -2.0562 | 0.46 | 1,18 | 0.5059 |

Table A4. Full linear mixed models of abundance and species richness of river island carabid beetles (Carabidae) that can disperse by flight in response to floods, island size and habitat structure and landscape structure. Twenty-seven islands were sampled over two years. Parameters in bold were retained in final model, other fixed effects were eliminated by step-wise backwards simplification.

| Dependent | Predictor | Estimate | F | df | P |
|---|---|---|---|---|---|
| Abundance (log) | | | | | |
| | Year | | | | |
| *Random effect* | 2010 | 1.1508 | 9.87 | 1,26 | 0.0041 |
| Site = 0.4067 | 2011 | 0.4992 | | | |
| | River | | 0.56 | 3,19 | 0.6484 |
| *Spatial autocorrelation* | Earn | 0.0610 | | | |
| Catchment x site = 0.0 | Tay | -0.0067 | | | |
| | Tummel | 0.0024 | | | |
| *Residual* = 0.5727 | Tweed | -0.6478 | | | |
| | QMED | -0.00474 | 13.60 | 1,23 | 0.0012 |
| | SSP | -0.00552 | 0.75 | 1,21 | 0.3965 |
| | Island area (log) | 0.5068 | 14.91 | 1,23 | 0.0008 |
| | Plant species richness | 0.02506 | 1.92 | 1,43 | 0.1725 |
| | Herbaceous plant | 0.01320 | 0.16 | 1,36 | 0.6907 |
| | Graminoid plant | 0.02713 | 5.48 | 1,43 | 0.0240 |
| | Tree canopy density | 0.01655 | 14.09 | 1,23 | 0.0010 |
| | Landscape structure | 1.9696 | 1.92 | 1,24 | 0.1783 |
| Species richness | | | | | |
| | Year | | 0.02 | 1,23 | 0.8776 |
| *Random effect* | 2010 | 8.2705 | | | |
| Site = 2.6855 | 2011 | 8.3370 | | | |
| | River | | 1.97 | 3,19 | 0.1523 |
| *Spatial autocorrelation* | Earn | 6.8831 | | | |
| Catchment x site $=1.11\text{x}10^{-16}$ | Tay | 7.4469 | | | |
| | Tummel | 5.8891 | | | |
| *Residual* = 1.3599 | Tweed | 8.8435 | | | |
| | QMED | 0.000339 | 0.00 | 1,15 | 0.9724 |
| | SSP | 0.006818 | 0.22 | 1,17 | 0.6443 |
| | Island area (log) | -0.06290 | 0.04 | 1,17 | 0.8471 |
| | Plant species richness | -0.00080 | 0.00 | 1,33 | 0.9854 |
| | Herbaceous plant | 0.007695 | 0.02 | 1,35 | 0.9013 |
| | Graminoid plant | 0.007332 | 0.10 | 1,38 | 0.7594 |
| | Tree canopy density | -0.00944 | 0.95 | 1,18 | 0.3438 |
| | Landscape structure | -0.4270 | 0.02 | 1,23 | 0.8913 |

Table A5. Full linear mixed models of abundance and species richness of river island carabid beetles (Carabidae) that are limited to terrestrial dispersal by walking in response to floods, island size and habitat structure and landscape structure. Twenty-seven islands were sampled over two years. Parameters in bold were retained in final model, other fixed effects were eliminated by step-wise backwards simplification.

| Dependent | Predictor | Estimate | F | df | P |
|---|---|---|---|---|---|
| Abundance(log) | | | | | |
| | Year | | 3.09 | 1,27 | 0.0899 |
| *Random effect* | 2010 | 5.5783 | | | |
| Site = 0.8041 | 2011 | 5.2961 | | | |
| | River | | 8.85 | 3,23 | 0.0004 |
| *Spatial autocorrelation* | Earn | 3.7714 | | | |
| Catchment x site = 0.0 | Tay | 8.2294 | | | |
| | Tummel | 4.3776 | | | |
| *Residual* =0.3874 | Tweed | 5.4372 | | | |
| | QMED | -0.00948 | 11.59 | 1,23 | 0.0024 |
| | SSP | -0.00637 | 0.69 | 1,17 | 0.4159 |
| | Island area (log) | 0.1609 | 1.03 | 1,19 | 0.3222 |
| | Plant species richness | 0.01206 | 0.53 | 1,45 | 0.4692 |
| | Herbaceous plant | 0.02812 | 0.86 | 1,43 | 0.3600 |
| | Graminoid plant | 0.000037 | 0.00 | 1,41 | 0.9978 |
| | Tree canopy density | 0.008145 | 1.18 | 1,21 | 0.2890 |
| | Landscape structure | 2.0442 | 0.83 | 1,22 | 0.3714 |
| Species richness | | | | | |
| | Year | | 0.09 | 1,13 | 0.7685 |
| *Random effect* | 2010 | 0.7295 | | | |
| Site = 0.09077 | 2011 | 0.7565 | | | |
| | River | | 1.26 | 3,9 | 0.3437 |
| *Spatial autocorrelation* | Earn | 1.4774 | | | |
| Catchment x site =0.0 | Tay | 0.5062 | | | |
| | Tummel | 0.8485 | | | |
| *Residual* = 0.01924 | Tweed | 1.3713 | | | |
| | QMED | 0.000399 | 0.59 | 1,11 | 0.4573 |
| | SSP | 0.001571 | 0.51 | 1,2 | 0.5523 |
| | Island area (log) | 0.09371 | 1.83 | 1,6 | 0.2265 |
| | Plant species richness | -0.01335 | 3.21 | 1,26 | 0.0848 |
| | Herbaceous plant | -0.02210 | 2.90 | 1,10 | 0.1210 |
| | Graminoid plant | -0.00591 | 2.16 | 1,26 | 0.1533 |
| | Tree canopy density | 0.001748 | 0.90 | 1,13 | 0.3589 |
| | Landscape structure | 1.0293 | 1.64 | 1,6 | 0.2488 |

Table A6. Full linear mixed models of abundance and species richness of river island millipedes (Diplopoda) in response to floods, island size and habitat structure and landscape structure. Twenty-seven islands were sampled over two years. Parameters in bold were retained in final model, other fixed effects were eliminated by step-wise backwards simplification.

| Dependent | Predictor | Estimate | F | df | P |
|---|---|---|---|---|---|
| Activity density (log) | | | | | |
| | Year | | 0.75 | 1,26 | 0.3953 |
| *Random effect* | 2010 | 1.7406 | | | |
| Site = 3.1936 | 2011 | 1.9725 | | | |
| | River | | 3.12 | 3,23 | 0.0458 |
| *Spatial autocorrelation* | Earn | 2.4939 | | | |
| Catchment x site = 0.0000 | Tay | 4.7869 | | | |
| | Tummel | 1.9479 | | | |
| *Residual* = 1.0270 | Tweed | 1.9829 | | | |
| | QMED | -0.00572 | 0.61 | 1, 17 | 0.4457 |
| | SSP | 0.005809 | 0.18 | 1,16 | 0.6779 |
| | Island area (log) | 0.2694 | 0.88 | 1,20 | 0.3597 |
| | Plant species richness | -0.00323 | 0.01 | 1,39 | 0.9140 |
| | Herbaceous plant | 0.05674 | 1.84 | 1,41 | 0.1827 |
| | Graminoid plant | -0.00812 | 0.15 | 1,39 | 0.7006 |
| | Tree canopy density | 0.02111 | 4.58 | 1,23 | 0.0432 |
| | Landscape structure | -6.1419 | 2.26 | 1,22 | 0.1467 |
| Species richness | | | | | |
| | Year | | 0.03 | 1,12 | 0.8619 |
| *Random effect* | 2010 | 1.3771 | | | |
| Site = 0.001686 | 2011 | 1.3953 | | | |
| | River | | 1.72 | 3,16 | 0.0816 |
| *Spatial autocorrelation* | Earn | 1.4406 | | | |
| Catchment x site = 3.7410 | Tay | 1.1268 | | | |
| | Tummel | 1.2941 | | | |
| *Residual* =0.06607 | Tweed | 1.2837 | | | |
| | QMED | 0.000620 | 0.34 | 1,18 | 0.5689 |
| | SSP | -0.00086 | 0.98 | 1,9 | 0.3477 |
| | Island area (log) | -0.02594 | 2.16 | 1,9 | 0.1764 |
| | Plant species richness | 0.002415 | 0.10 | 1,15 | 0.7591 |
| | Herbaceous plant | -0.01236 | 3.67 | 1,15 | 0.0739 |
| | Graminoid plant | 0.004109 | 1.24 | 1,21 | 0.2784 |
| | Tree canopy density | 0.001739 | 2.56 | 1,20 | 0.1255 |
| | Landscape structure | 0.9814 | 5.09 | 1,29 | 0.0317 |

Table A7. Full linear mixed models of abundance and species richness of river island spider (Araneae) in response to floods, island size and habitat structure and landscape structure. Twenty-seven islands were sampled over two years. Parameters in bold were retained in final model, other fixed effects were eliminated by step-wise backwards simplification.

| Dependent | Predictor | Estimate | F | df | P |
|---|---|---|---|---|---|
| **Abundance (log)** | | | | | |
| | **Year** | | **3.83** | **1,27** | **0.0607** |
| *Random effect* | **2010** | **2.5447** | | | |
| Site = 0.1464 | **2011** | **2.0423** | | | |
| | **River** | | **4.49** | **3,22** | **0.0133** |
| *Spatial autocorrelation* | **Earn** | **0.7158** | | | |
| Catchment x site = 0.000 | **Tay** | **0.9463** | | | |
| | **Tummel** | **0.8313** | | | |
| *Residual* = 0.8558 | **Tweed** | **2.0423** | | | |
| | QMED | 0.004249 | 3.93 | 1,20 | 0.0610 |
| | SSP | -0.00567 | 0.81 | 1, 19 | 0.3808 |
| | **Island area (log)** | **0.3381** | **6.34** | **1,23** | **0.0193** |
| | **Plant species richness** | **0.04090** | **4.25** | **1,40** | **0.0458** |
| | Herbaceous plant | -0.01408 | 0.19 | 1,29 | 0.6632 |
| | Graminoid plant | 0.009248 | 0.42 | 1,36 | 0.5232 |
| | Tree canopy density | -0.00110 | 0.03 | 1,17 | 0.8714 |
| | Landscape structure | -1.3907 | 0.60 | 1,20 | 0.4473 |
| **Species richness** | | | | | |
| | Year | | 2.75 | 1,25 | 0.1098 |
| *Random effect* | 2010 | 2.0760 | | | |
| Site = 0.03516 | 2011 | 1.9813 | | | |
| | River | | 1.31 | 3,21 | 0.2980 |
| *Spatial autocorrelation* | Earn | 2.1253 | | | |
| Catchment x site =0.000 | Tay | 2.6348 | | | |
| | Tummel | 2.3755 | | | |
| *Residual* = 0.04069 | Tweed | 2.1316 | | | |
| | QMED | -0.00022 | 0.70 | 1,25 | 0.4101 |
| | SSP | -0.00090 | 0.20 | 1,20 | 0.6614 |
| | Island area (log) | 0.02102 | 0.23 | 1,18 | 0.6396 |
| | Plant species richness | -0.00530 | 0.78 | 1,38 | 0.3830 |
| | Herbaceous plant | 0.007342 | 0.79 | 1,41 | 0.3801 |
| | Graminoid plant | 0.002442 | 0.43 | 1,39 | 0.5167 |
| | Tree canopy density | -0.00021 | 0.01 | 1,16 | 0.9205 |
| | **Landscape structure** | **-0.8819** | **5.37** | **1,28** | **0.0281** |

Table A8. Full linear mixed models of abundance and species richness of river island ground beetles (Carabidae) in response to floods, island size and habitat structure and landscape structure. Twenty-seven islands were sampled over two years. Parameters in bold were retained in final model, other fixed effects were eliminated by step-wise backwards simplification.

| Dependent | Predictor | Estimate | F | df | P |
|---|---|---|---|---|---|
| Abundance (log) | | | | | |
| | **Year** | | **9.08** | **1,26** | **0.0057** |
| *Random effect* | **2010** | **0.6937** | | | |
| Site = 0.2606 | **2011** | **0.6626** | | | |
| | River | | 0.28 | 3,18 | 0.8214 |
| *Spatial autocorrelation* | Earn | 0.6227 | | | |
| Catchment x site = 1.11 x $10^{-16}$ | Tay | 1.2198 | | | |
| | Tummel | 0.7598 | | | |
| *Residual* =0.5855 | Tweed | 0.3174 | | | |
| | **QMED** | **-0.00345** | **8.68** | **1,22** | **0.0075** |
| | SSP | -0.00628 | 1.34 | 1,20 | 0.2596 |
| | **Island area (log)** | **0.4146** | **12.17** | **1,22** | **0.0021** |
| | Plant species richness | 0.03158 | 3.58 | 1,41 | 0.0657 |
| | Herbaceous plant | 0.01878 | 0.48 | 1,33 | 0.4954 |
| | Graminoid plant | 0.01841 | 2.77 | 1,38 | 0.1040 |
| | **Tree canopy density** | **0.01630** | **16.52** | **1,22** | **0.0005** |
| | **Landscape structure** | **3.8451** | **10.12** | **1,22** | **0.0043** |
| Species richness (log) | | | | | |
| | Year | | 1.21 | 1,8 | 0.3025 |
| | 2010 | 2.2730 | | | |
| *Random effect* | 2011 | 2.3585 | | | |
| Site = 0.06857 | River | | 0.94 | 3,21 | 0.4388 |
| | Earn | 2.5429 | | | |
| *Spatial autocorrelation* | Tay | 2.7974 | | | |
| Catchment x site = 0.4995 | Tummel | 2.5038 | | | |
| *Residual* = 0.02245 | Tweed | 2.7944 | | | |
| | QMED | 0.000533 | 0.17 | 1,16 | 0.6860 |
| | SSP | 0.001526 | 0.51 | 1,13 | 0.4870 |
| | Island area (log) | -0.04278 | 1.66 | 1,19 | 0.2132 |
| | Plant species richness | -0.00127 | 0.08 | 1,11 | 0.7794 |
| | Herbaceous plant | 0.008334 | 1.47 | 1,15 | 0.2442 |
| | Graminoid plant | 0.003659 | 1.60 | 1,43 | 0.2127 |
| | Tree canopy density | -0.00132 | 0.92 | 1,18 | 0.3498 |
| | Landscape structure | -0.4083 | 0.82 | 1,24 | 0.3750 |

Table A9. Full linear mixed models of carabid beetle (Carabidae) functional diversity (FDisp) and prevalence of community-weighted mean traits (mT) in response to floods, island size and habitat structure and landscape structure. Twenty-seven islands were sampled over two years. Parameters in bold were retained in final model, other fixed effects were eliminated by step-wise backwards simplification.

| Diversity/trait metric | Predictor | Estimate | F | df | p |
|---|---|---|---|---|---|
| Functional diversity | | | | | |
| | Year | | 0.77 | 1,31 | 0.3861 |
| *Random effect* | 2010 | 0.2004 | | | |
| Site = 0.001948 | 2011 | 0.2125 | | | |
| | River | | 1.15 | 3,25 | 0.3483 |
| *Spatial autocorrelation* | Earn | 0.08555 | | | |
| Catchment x site = 1.11 x $10^{-16}$ | Tay | 0.10755 | | | |
| | Tummel | 0.04691 | | | |
| *Residual* = 0.002064 | Tweed | 0.09357 | | | |
| | QMED | -0.00033 | 2.17 | 1,21 | 0.1550 |
| | SSP | -0.00001 | 0.00 | 1,18 | 0.9799 |
| | Island area (log) | 0.01173 | 1.76 | 1,19 | 0.2002 |
| | Plant species richness | 0.002553 | 6.52 | 1,53 | 0.0136 |
| | Herbaceous plant | 0.005288 | 10.20 | 1,52 | 0.0024 |
| | Graminoid plant | -0.00005 | 0.00 | 1,40 | 0.9506 |
| | Tree canopy density | -0.00026 | 2.85 | 1,22 | 0.3653 |
| | Landscape structure | -0.1400 | 1.38 | 1,24 | 0.2516 |
| Species traits | | | | | |
| Sensory acuity | | | | | |
| *Ratio of eye : head width* | Year | | 0.67 | 1,30 | 0.4195 |
| | 2010 | -1.4440 | | | |
| Random effect | 2011 | -1.4493 | | | |
| Site = 0.003108 | River | | 2.48 | 3,24 | 0.0855 |
| | Earn | -1.4075 | | | |
| Spatial autocorrelation | Tay | -1.4952 | | | |
| Catchment x site = 0 | Tummel | -1.4325 | | | |
| | Tweed | -1.4406 | | | |
| Residual =0.001459 | QMED | 0.000153 | 0.91 | 1,23 | 0.3488 |
| | SSP | -0.00022 | 0.22 | 1,20 | 0.6456 |
| | Island area (log) | 0.001301 | 0.01 | 1,18 | 0.9058 |
| | Plant species richness | 0.000306 | 0.08 | 1,44 | 0.7746 |
| | Herbaceous plant | -0.00146 | 0.78 | 1,49 | 0.3810 |
| | Graminoid plant | -0.00052 | 0.42 | 1,47 | 0.5220 |
| | Tree canopy density | 0.000359 | 0.77 | 1,21 | 0.3905 |
| | Landscape structure | 0.09430 | 0.45 | 1,20 | 0.5086 |
| *Antennae : body length* | Year | | 0.29 | 1,31 | 0.5959 |
| | 2010 | -0.5057 | | | |
| Random effect | 2011 | -0.5098 | | | |
| Site = 0.000704 | River | | 4.90 | 3,24 | 0.0085 |
| | Earn | -0.4494 | | | |
| Spatial autocorrelation | Tay | -0.5195 | | | |
| Catchment x site = 0 | Tummel | -0.4854 | | | |
| | Tweed | -0.4868 | | | |
| Residual =0.000600 | QMED | 0.000048 | 0.16 | 1,22 | 0.7577 |
| | SSP | -0.00008 | 0.10 | 1,20 | 0.7577 |
| | Island area (log) | -0.00111 | 0.04 | 1,18 | 0.8502 |
| | Plant species richness | 0.000653 | 1.32 | 1,50 | 0.2564 |
| | Herbaceous plant | -0.00055 | 0.27 | 1,47 | 0.6025 |
| | Graminoid plant | 0.000385 | 0.74 | 1,48 | 0.3939 |
| | Tree canopy density | 0.000184 | 1.29 | 1,23 | 0.2671 |
| | Landscape structure | 0. 01535 | 0.04 | 1,20 | 0.8531 |
| Foraging range | | | | | |
| *Femora width: length* | Year | | 0.05 | 1,30 | 0.8167 |

| Diversity/trait metric | Predictor | Estimate | F | df | p |
|---|---|---|---|---|---|
| *ratio* | | | | | |
| | 2010 | -1.4525 | | | |
| Random effect | 2011 | -1.4498 | | | |
| Site = 0.002460 | River | | 6.00 | 3,24 | 0.0034 |
| | Earn | -1.6267 | | | |
| Spatial autocorrelation | Tay | -1.5022 | | | |
| Catchment x site = 0 | Tummel | -1.5595 | | | |
| | Tweed | -1.5182 | | | |
| Residual =0.001746 | QMED | -0.00022 | 2.02 | 1,23 | 0.1691 |
| | SSP | 0.000082 | 0.03 | 1,18 | 0.8643 |
| | Island area (log) | 0.003454 | 0.15 | 1,20 | 0.7033 |
| | Plant species richness | -0.00021 | 0.03 | 1,42 | 0.8572 |
| | Herbaceous plant | 0.002775 | 2.62 | 1,50 | 0.1115 |
| | Graminoid plant | -0.00017 | 0.04 | 1,44 | 0.8354 |
| | Tree canopy density | -0.00051 | 4.23 | 1,23 | 0.0514 |
| | Landscape structure | 0.005099 | 0.00 | 1,19 | 0.9750 |
| Flight | | | | | |
| *Wings full* | Year | | 4.99 | 1,29 | 0.0335 |
| | 2010 | 0.7433 | | | |
| | 2011 | 0.6529 | | | |
| Random effect | River | | 19.82 | 3,25 | <.0001 |
| Site = 0.01369 | Earn | 1.1416 | | | |
| | Tay | 0.9612 | | | |
| Spatial autocorrelation | Tummel | 1.3149 | | | |
| Catchment x site = 1.3601 | Tweed | 0.6529 | | | |
| | QMED | 0.000636 | 1.06 | 1,21 | 0.3153 |
| Residual =0.02071 | SSP | 0.000667 | 0.32 | 1,20 | 0.5782 |
| | Island area (log) | -0.00650 | 0.06 | 1,17 | 0.8049 |
| | Plant species richness | 0.000756 | 0.06 | 1,40 | 0.8123 |
| | Herbaceous plant | -0.02104 | 13.32 | 1,45 | 0.0007 |
| | Graminoid plant | 0.006771 | 7.49 | 1,46 | 0.0009 |
| | Tree canopy density | 0.001583 | 4.21 | 1,22 | 0.0520 |
| | Landscape structure | 0.5051 | 2.69 | 1,20 | 0.1160 |
| *Dimorphic* | Year | | 1.70 | 1,30 | 0.2027 |
| | 2010 | 0.2988 | | | |
| Random effect | 2011 | 0.3578 | | | |
| Site = 0.006455 | River | | 4.11 | 3,24 | 0.0170 |
| | Earn | 0.1641 | | | |
| Spatial autocorrelation | Tay | 0.2193 | | | |
| Catchment x site = 0 | Tummel | 0.082 | | | |
| | Tweed | 0.3511 | | | |
| Residual = 0.02739 | QMED | -0.00008 | 0.02 | 1,21 | 0.9008 |
| | SSP | 0.000020 | 0.00 | 1,18 | 0.9861 |
| | Island area (log) | 0.01766 | 0.70 | 1,19 | 0.4132 |
| | Plant species richness | -0.00034 | 0.01 | 1,32 | 0.9212 |
| | Herbaceous plant | 0.01797 | 11.25 | 1,45 | 0.0016 |
| | Graminoid plant | -0.00101 | 0.17 | 1,33 | 0.6819 |
| | Tree canopy density | -0.00021 | 0.08 | 1,22 | 0.7807 |
| | Landscape structure | -0.3874 | 1.62 | 1,22 | 0.2165 |
| *Wingless* | Year | | 0.79 | 1,28 | 0.3813 |
| | 2010 | 1.0778 | | | |
| | 2011 | 1.1355 | | | |
| Random effect | River | | 9.61 | 3,27 | 0.0002 |
| Site = 0.01196 | Earn | 0.3035 | | | |
| | Tay | 0.4042 | | | |
| Spatial autocorrelation | Tummel | 0.1495 | | | |
| Catchment x site = 0.2353 | Tweed | 0.6242 | | | |
| | QMED | -0.00068 | 1.27 | 1,49 | 0.2649 |
| Residual =0.03247 | SSP | -0.00093 | 0.48 | 1,12 | 0.5018 |
| | Island area (log) | -0.01052 | 0.13 | 1,10 | 0.7234 |
| | Plant species richness | -0.00245 | 0.57 | 1,46 | 0.4543 |
| | Herbaceous plant | 0.004710 | 0.56 | 1,20 | 0.4642 |

| Diversity/trait metric | Predictor | Estimate | F | df | p |
|---|---|---|---|---|---|
| | Graminoid plant | -0.00401 | 2.42 | 1,34 | 0.1294 |
| | Tree canopy density | -0.00118 | 2.03 | 1,13 | 0.1774 |
| | Landscape structure | -0.3430 | 0.78 | 1,45 | 0.3817 |
| Body mass (mg) | | | | | |
| | Year | | 0.13 | 1,29 | 0.7174 |
| *Random effect* | 2010 | 0.5658 | | | |
| Site = 0.2689 | 2011 | 0.5128 | | | |
| | River | | 6.51 | 3,25 | 0.0054 |
| *Spatial autocorrelation* | Earn | 1.8754 | | | |
| Catchment x site = 1.11 x $10^{-16}$ | Tay | 2.2246 | | | |
| | Tummel | 1.5051 | | | |
| *Residual* =0.2236 | Tweed | 0.9146 | | | |
| | QMED | -0.00177 | 1.12 | 1,23 | 0.3020 |
| | SSP | 0.000058 | 0.00 | 1,18 | 0.9915 |
| | Island area (log) | 0.07818 | 0.64 | 1,21 | 0.4326 |
| | Plant species richness | 0.02074 | 3.69 | 1,50 | 0.0604 |
| | Herbaceous plant | 0.05021 | 6.28 | 1,51 | 0.0154 |
| | Graminoid plant | 0.01107 | 1.54 | 1,47 | 0.2205 |
| | Tree canopy density | -0.00243 | 0.30 | 1,19 | 0.5888 |
| | Landscape structure | -0.7608 | 0.23 | 1,19 | 0.6340 |
| Life-cycle duration | | | | | |
| *Annual vs Biennial* | Year | | 0.03 | 1,28 | 0.8558 |
| | 2010 | -0.2469/1.8177 | | | |
| | 2011 | -0.2546/1.8254 | | | |
| | River | | 1.23 | 3,23 | 0.3200 |
| *Random effect* | Earn | -0.2648/1.8356 | | | |
| Site = 0.04638 | Tay | -0.1397/1.7105 | | | |
| | Tummel | -0.3033/1.8741 | | | |
| *Spatial autocorrelation* | Tweed | -0.4221/1.9929 | | | |
| Catchment x site = 1.11 x $10^{-16}$ | QMED | -0.000580/0.000580 | 0.79 | 1,21 | 0.3842 |
| | SSP | -0.000304/0.000304 | 0.03 | 1,18 | 0.8751 |
| *Residual* =0.01688 | Island area (log) | 0.0891/-0.0891 | 9.50 | 1,27 | 0.0049 |
| | Plant species richness | 0.0089/-0.0089 | 5.70 | 1,45 | 0.0212 |
| | Herbaceous plant | 0.0169/-0.0169 | 8.67 | 1,47 | 0.0050 |
| | Graminoid plant | 0.00150/-0.00150 | 0.26 | 1,43 | 0.6137 |
| | Tree canopy density | -0.00154/0.00154 | 0.84 | 1,20 | 0.3699 |
| | Landscape structure | -0.05183/0.05183 | 0.01 | 1,17 | 0.9358 |
| Adult activity period | | | | | |
| *Summer only vs* | Year | | 0.00 | 1,28 | 0.9820 |
| *Summer and Autumn* | 2010 | 0.3929/1.1779 | | | |
| | 2011 | 0.3915/1.1793 | | | |
| *Random effect* | River | | 0.98 | 3,21 | 0.4208 |
| Site = 0.06648 | Earn | 0.5139/1.0569 | | | |
| | Tay | 1.2484/0.3224 | | | |
| *Spatial autocorrelation* | Tummel | 0.7168/0.8540 | | | |
| Catchment x site = 0 | Tweed | 0.4392/1.1316 | | | |
| | QMED | 0.000426 | 0.97 | 1,23 | 0.3348 |
| *Residual* =0.03948 | SSP | 0.00014/-0.00014 | 0.00 | 1,17 | 0.9580 |
| | Island area (log) | 0.03361/-0.03361 | 0.85 | 1,25 | 0.3650 |
| | Plant species richness | 0.01149/-0.01149 | 5.73 | 1,51 | 0.0204 |
| | Herbaceous plant | -0.02686/-0.02686 | 11.19 | 1,53 | 0.0015 |
| | Graminoid plant | 0.00267/-0.00267 | 0.40 | 1,44 | 0.5279 |
| | Tree canopy density | 0.00005/-0.00005 | 0.00 | 1,47 | 0.9776 |
| | Landscape structure | 0.05493/-0.05493 | 0.00 | 1,18 | 0.9446 |
| Breeding season | | | | | |
| *Spring and Summer vs* | Year | | 0.47 | 1,30 | 0.4967 |
| *Autumn* | 2010 | 1.3413/0.2295 | | | |
| | 2011 | 1.3030/0.2678 | | | |
| | River | | 2.13 | 3,24 | 0.1226 |
| *Random effect* | Earn | 1.2085/0.3623 | | | |

| Diversity/trait metric | Predictor | Estimate | F | df | p |
|---|---|---|---|---|---|
| Site = 0.05195 | Tay | 1.0200/0.5508 | | | |
| | Tummel | 1.3006/0.27015 | | | |
| *Spatial autocorrelation* | Tweed | 1.3219/0.2489 | | | |
| Catchment x site = 0 | QMED | 0.00071/-0.00071 | 1.10 | 1,22 | 0.3061 |
| | SSP | 0.00037/-0.00037 | 0.03 | 1,16 | 0.8698 |
| *Residual* =0.03277 | Island area (log) | -0.02535/0.02535 | 0.37 | 1,19 | 0.5479 |
| | Plant species richness | -0.004720/0.004720 | 1.20 | 1,47 | 0.2786 |
| | Herbaceous plant | -0.0164/0.0164 | 5.12 | 1,54 | 0.0276 |
| | Graminoid plant | -0.002908/0.002908 | 0.65 | 1,48 | 0.4240 |
| | Tree canopy density | 0.00082/-0.00082 | 0.19 | 1,18 | 0.6685 |
| | Landscape structure | 0.3610/-0.3610 | 0.29 | 1,17 | 0.5953 |
| Overwintering life stage | | | | | |
| *Adults only vs* | Year | | 0.38 | 1,32 | 0.5405 |
| *Adults and larvae/larvae only* | 2010 | 0.7948/0.7759 | | | |
| | 2011 | 0.7596/0.8111 | | | |
| *Random effect* | River | | 3.96 | 3,24 | 0.0202 |
| Site = 0.06516 | Earn | 0.9051/0.6656 | | | |
| | Tay | -0.3404/1.9112 | | | |
| *Spatial autocorrelation* | Tummel | 0.5054/1.0654 | | | |
| Catchment x site = 0 | Tweed | 0.8301/0.7407 | | | |
| | QMED | 0.0020/-0.0020 | 6.46 | 1,23 | 0.0183 |
| *Residual* =0.03591 | SSP | 0.00106/-0.00106 | 0.18 | 1,19 | 0.6742 |
| | Island area (log) | -0.01945/0.01945 | 0.15 | 1,20 | 0.6990 |
| | Plant species richness | -0.00683/0.00683 | 2.15 | 1,47 | 0.1490 |
| | Herbaceous plant | -0.0193/0.0193 | 5.54 | 1,49 | 0.0226 |
| | Graminoid plant | 0.00102/-0.00102 | 0.06 | 1,41 | 0.8068 |
| | Tree canopy density | 0.00203/ -0.00203 | 0.98 | 1,22 | 0.3316 |
| | Landscape structure | -0.1204/0.1204 | 0.02 | 1,17 | 0.8886 |

# 8 Riparian Hydromorphology influences on instream invertebrates - Swedish Case Study

Cross-habitat resource exchanges are highly important components of ecosystem functioning in many natural systems (Likens and Bormann 1974; Polis et al. 1997; Lamberti et al. 2010). In particular, the integrity of habitat boundaries, such as those between aquatic and terrestrial ecosystems (i.e. the riparian habitat and the stream or river), are often heavily degraded by human activities, with potentially strong effects on linkages and subsidies between these adjacent habitats. Thus reduced connectivity across the riparian land-use gradient will have important effects on water-land fluxes of energy. Here we test if degradation of riparian land-use (change from forest to agriculture) has potential effects on how much energy and nutrients are transported from the aquatic to the terrestrial environment and thus on riparian and catchment biodiversity and processes.

Adult aquatic insects with a winged terrestrial life stage are important vectors transferring resources to terrestrial consumers, thereby supporting different organisms such as spiders and insectivorous vertebrates in riparian communities. Incorporation of this aquatic subsidy in terrestrial food webs depends heavily on the dispersal and life history traits of aquatic insects. Anthropogenic stressors (such as land use change) are unlikely to affect single dispersal and life history traits of a community but rather affect entire suites of traits simultaneously. Here, we assessed how traits related to the dispersal and life history of adult stream insects are affected by in-stream habitat (riffles vs pools), seasonality (spring and autumn) and catchment land use (agricultural vs forested). Aquatic insect abundances were overall higher in the agricultural streams compared to the streams with forested riparian zones, and at the same time the agricultural streams were characterised by traits associated with reduced dispersal capacity (e.g. small body size, low flight strength, very short adult life spans). Traits associated with reduced dispersal capacity were also more characteristic of pool than riffle assemblages in our study, which has implications at larger scales given the scarcity of riffle habitats in agricultural landscapes. This change in habitat types affects terrestrial consumers through truncated dispersal and potentially an asynchrony (including seasonal changes) between subsidy availability and consumer needs. The dispersal potential of aquatic insects emerging from streams are affected by agricultural activities, which may have knock-on effects on cross-system subsidies and affect food availability and quality for many terrestrial organisms such as birds, bats and invertebrates. This study showed the necessity of using life history and dispersal traits to detect potential changes in resource subsidies with changes in anthropogenic disturbance levels.

## 8.1 Introduction

Cross-habitat resource flows are recognised as important components of virtually all ecological network, since ecosystems are rarely closed or non-interactive with other systems (e.g. Polis et al. 1997, Lamberti et al. 2010). The transfer of subsidies from terrestrial to aquatic environments has been particularly well studied (Nakano and Murakami 2001; England and Rosemond 2004), with a strong focus on the importance of leaf litter inputs for stream detrital foodwebs (Richardson 1991; Wallace et al. 1997). The role of anadromous fish, such as salmon, in transferring resources from marine into freshwater and ultimately terrestrial foodwebs through consumption by terrestrial predators is also well documented (Willson and Halupka 1995). Currently, there is an increasing interest in the importance of flow of aquatic resource subsidies into terrestrial food webs via aerial movement of adult stages of aquatic insects (Moldenke and Linden 2007; Hoekman et al. 2011). Not only do aquatic insects often emerge with very high abundance and biomass, but they also carry nutrients and essential lipids that are produced within aquatic environments (Baxter et al. 2005; Gladyshev et al. 2009; Torres-Ruiz and Wehr 2010), and they may therefore support a multitude of organisms in riparian communities, including arthropods (Collier et al. 2002; Kato et al. 2003), birds (Gray 1993; Chan et al. 2008) and bats (Dejong and Ahlen 1991; Fukui et al. 2006). However, an understanding of the factors regulating the transfer of aquatic insect subsidies into terrestrial food webs, and especially how anthropogenic effects alter these linkages, is presently limited.

A key factor regulating the incorporation of aquatic subsidies in terrestrial food webs is the timing and extent of dispersal by aquatic insects, which in turn is dependent on their species-specific dispersal and life history traits (e.g. body size, flight ability, life span, number of life cycles per year and synchronisation of emergence). Anthropogenic disturbances often have particularly strong effects on the species trait composition of communities, favouring smaller, short-lived organisms with more flexible life cycles and generalist feeding behaviours (Statzner and Beche 2010). These traits are not independent entities, but because of ecological and evolutionary trade-offs they are tightly linked in ‘syndromes’, clusters (Reich et al. 2003) or ‘life history strategies’ (Verberk et al. 2008), i.e. a group of tightly linked traits strongly associated with particular genera or families (Poff et al. 2006). Accordingly, anthropogenic stressors are unlikely to affect single traits in isolation; rather stressors related to land use changes typically affect entire suites of traits simultaneously.

In this study, we assessed how traits related to the dispersal and life history of stream insects are affected by a pervasive anthropogenic disturbance, agricultural land use, and how these changes are further moderated by in-stream habitat and seasonal variation. We characterised the species trait composition of benthic invertebrates from both agricultural and reference forested streams to test the hypothesis that streams in agricultural landscapes are characterised by traits associated with disturbed environments, smaller body size, faster life cycles and greater life history flexibility. Shifts in these traits are likely to affect aquatic – terrestrial landscape-level linkages via truncated dispersal (e.g. small organisms does not travel as far into the terrestrial landscape) and asynchrony between subsidy availability and consumer needs (e.g. emergence of adults from the aquatic environment does not match needs or the presence of terrestrial consumers).

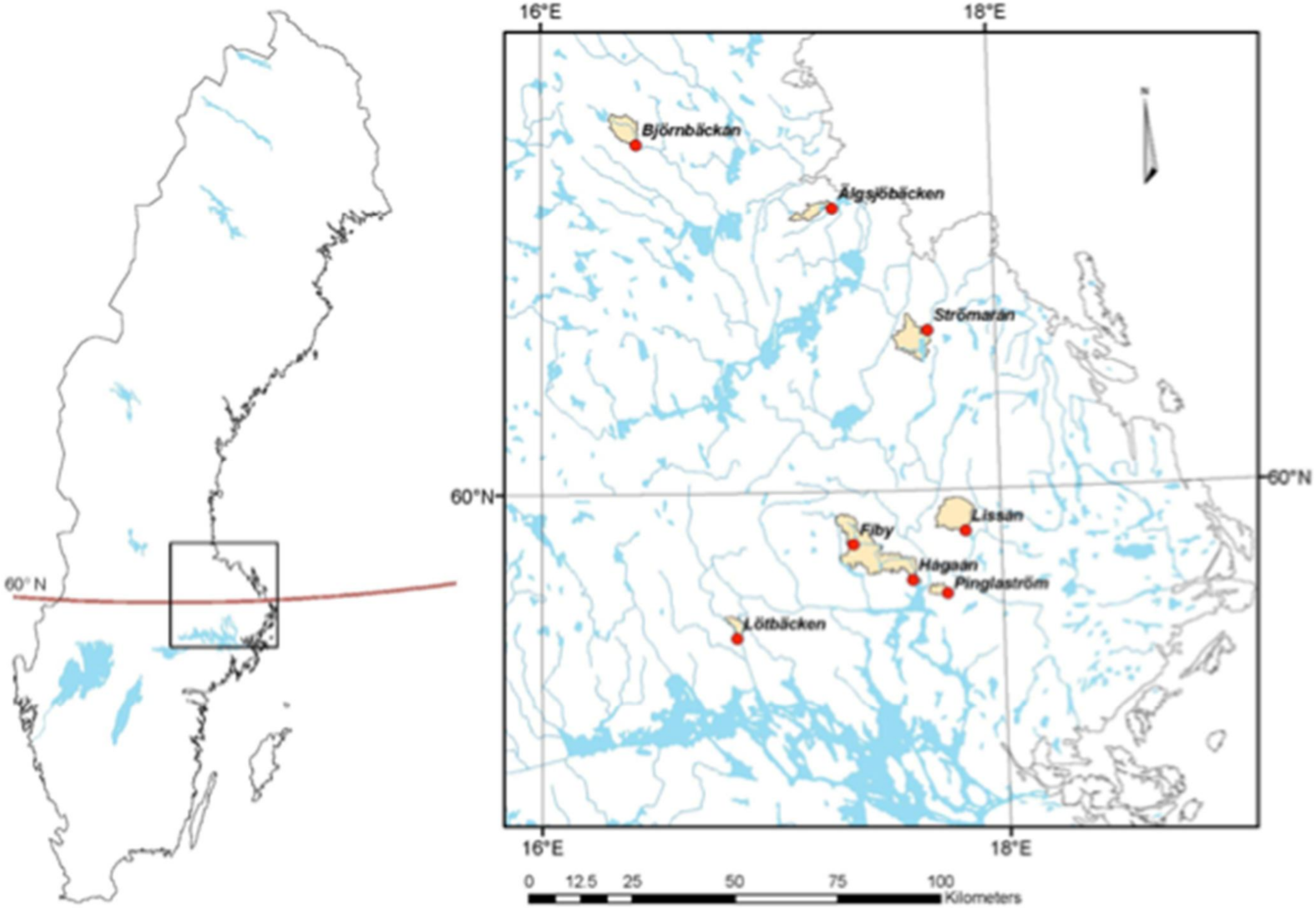


Figure 8.1 The eight sampled streams.

## 8.2 Materials and Methods

### *Study area and sites*

The eight study streams situated in central Sweden (Figure 8.1) were chosen to represent two major land use types (forest versus agriculture; Figure 8.2). Initial selection was based on GIS information of catchment and riparian vegetation (i.e. predominance of coniferous forests or agriculture/open grassland and deciduous forests). The final selection was based on on-site examination of the presence of riffle and pool habitats. Information of catchment land use for all sites was obtained using the Corine land cover database (http://sia.eionet.europa.eu/CLC2000), delineated according to topographic maps with a scale of 1:100 000 (Swedish Geodata; roadmap from Lantmäteriet), and digitised using ArcGIS 9 (ESRI, Redland, CA.; http://www.esri.com/).


a) b)

Figure 8.2 Two of the eight streams sampled in the study. a) The agricultural Strömarån and b) the forested Björnbäcken

All forested streams were predominated by coniferous forest (73 – 98%), with very little influence of agriculture (0 - 0.7% arable land/pastures). Catchments of the four agricultural streams consisted of 8 – 43% agriculture/pasture and 36 – 61% mixed deciduous-coniferous forest. Width of the riparian vegetation at the individual sampling reaches also differed between the two groups: the four forest-dominated streams had at least 100 m riparian vegetation perpendicular to the streams, while average widths of riparian vegetation for the four agricultural streams ranged from 5 - 34 m. Furthermore, the streams differed in the main type of riparian vegetation, with the forested streams having 93 – 98% forest, whilst the agricultural streams had 40 – 95 % agriculture/grassland. The streams were all lowland (< 191 m a.s.l.), small-medium sized (Strahler order 2-4, 1.4 – 10.6 m average stream width) and shallow (0.1 – 1.1 m average depth). The streams were circumneutral (pH 6.3 to 8.3, conductivity 2 – 65 mS $cm^{-1}$) had a total organic carbon (TOC) content of 15 – 52 mg $L^{-1}$ and ranged from nutrient poor (18 µg $NO_2$-$NO_3$ $L^{-1}$ and 9 µg TP $L^{-1}$) to nutrient rich (1824 µg $NO_2$-$NO_3$ $L^{-1}$

and 198 µg TP $L^{-1}$).

Table 8.1 Basic data on the eight study streams.

| Stream | Type | Catchment area ($km^2$) | Width riparian corridor (m) | Agricultural land-use in catchment (%) | Forested land-use in catchment (%) |
|---|---|---|---|---|---|
| Björnbäckån | Forest | 34.8 | 100 | 0.0 | 76.4 |
| Fibyån | Forest | 23.9 | 100 | 0.7 | 75.9 |
| Pinglaström | Forest | 9.1 | 100 | 0.0 | 98.1 |
| Älgsjöbäcken | Forest | 19.9 | 100 | 0.1 | 72.7 |
| Husbyån | Agriculture | 156 | 5.2 | 25.2 | 44.5 |
| Lissån | Agriculture | 55.3 | 33.5 | 32.2 | 50.0 |
| Lötbäcken | Agriculture | 12.9 | 5.9 | 42.7 | 36.4 |
| Strömarån | Agriculture | 53.9 | 7.4 | 8.4 | 60.5 |

### *Invertebrate sampling*

Benthic invertebrate assemblages were quantitatively sampled in autumn 2008 and spring 2009 from one pool and riffle habitat per stream. Three replicate samples were collected from each pool and riffle habitat using a 40 cm high, 500 µm mesh Hess sampler covering a bottom area of 0.086 $m^2$ (Wildlife Supply Company http://www.wildco.com). Collected organisms and detrital material were preserved in the field in 70% ethanol and returned to the laboratory for processing. Organisms were sorted from the debris, identified to the lowest possible level of taxonomic resolution (mainly to species and genus, but some to higher taxonomic levels such as Chironomidae, Simuliidae and Oligochaeta) counted. Samples with a relatively high number of organisms (>300/sample) were subsampled by sorting a minimum of 300 individuals in a fraction of the sample, with the remainder of the sample being searched for large, rare taxa not present in the subsample.

### *Characterisation of species traits and statistical analyses*

In total, 105 taxa (out of 144 taxa in the dataset) were identified as insect taxa with an adult winged stage and were thus included in the analyses. Data for these genera were extracted from Poff et al. (2006) for the following traits: (1) adult flying strength (weak/strong), (2) voltinism (semi/uni-, bi- or multivoltine), (3) size at maturity (small/medium/large), (4) synchronisation of emergence (poorly/well synchronised), (5) adult life span (very short/short/long), (6) organism development (fast seasonal/slow seasonal/no seasonal) and (7) female dispersal ability (low/high), yielding in total 18 states (2 - 3 trait states for each of the seven traits). The traits database by Poff et al. (2006) was developed for North American insect genera, but we have successfully used it in an earlier study of Swedish stream invertebrates (Göthe et al. 2012). Shifts in trait composition in relation to the main environmental factors (land use, habitat and season) were graphically assessed using nonmetric multidimensional scaling (nMDS), with square root transformed trait abundance data (i.e. number of individuals possessing a given trait and where each trait is treated as a "species" in the analysis) and the Bray-Curtis similarity index using the *vegan* routine (Oksanen et al. 2013) in the R statistical software (R Development Core Team 2012) Release 3.0.1.

Individual traits were further analysed using a split-plot Analyses of Variance model applying the *proc mixed* routine in SAS Release 9.3 statistical software (SAS Institute Inc. 2011). Land use (forested versus agriculture), habitat (pool versus riffle) and season (autumn versus spring) were fitted as fixed effects. As we wanted to directly contrast habitats and land use between the two sampling periods, we chose to fit season as a fixed rather than random factor in our ANOVA models, while acknowledging that details of these responses may have differed in other years with different seasonal trajectories. Our analysis has multiple error terms: stream(landuse), stream(landuse*habitat), stream(landuse*season) and stream(landuse*habitat*season), reflecting the hierarchical nature of our sampling design. The "main plots" were the eight streams, fitted as a random block factor and used for testing the effects of land use (four replicates per category). All other fixed factors and interactions were tested against error terms incorporating the stream block factor. ANOVAs were conducted for all traits using both trait abundance data and trait proportions (percentage of all individuals possessing a given trait). Data were either square-root or arcsine transformed to correct got violations of parametric assumptions. The error terms were estimated using residual maximum likelihood (REML).

## 8.3 Results

*Total abundance of individuals with a winged adult life stage*

Total abundance of taxa with a winged adult life stage was higher in the agricultural than the forested streams (Figure 11.2$$; ANOVA $F_{1,6}$= 8.66; $p < 0.05$) and differed between the two habitat types, with a higher number of individuals in riffle (15,889 ± 2,412 individuals $m^{-2}$; mean ± SE ) compared to pool (5,252 ± 1,466 individuals $m^{-2}$) habitats (Figure 8.3, ANOVA $F_{1,6}$ = 44.31; $p < 0.001$). There were no differences in the abundance of individuals with a winged adult life stage when comparing season or any of the interaction terms ($p > 0.05$ in all cases).

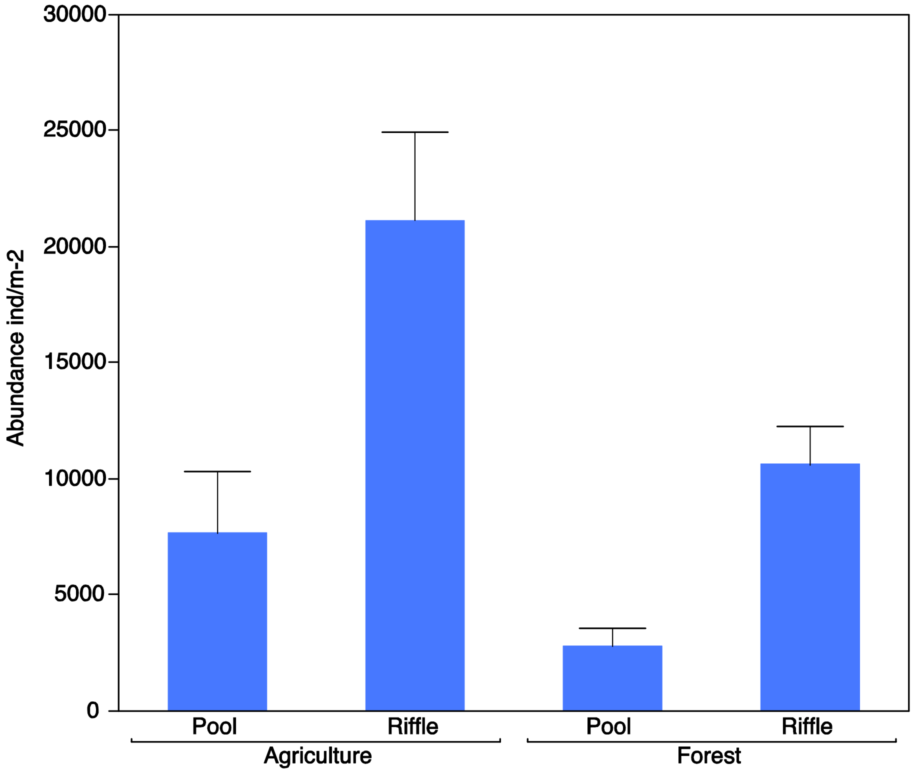


Figure 8.3 Total abundance (mean ± SE) of individuals with a winged adult life stage from pool and riffle habitats sampled in agricultural versus forested streams.

*Trait abundances and community composition*

The nMDS analyses revealed five clear clusters or “syndromes” related to life history strategies of closely associated traits (here denoted A-E; each cluster containing several traits closely related in ordination space) as well as two single traits (strong adult flying strength and large size at maturity). These syndromes varied between land use types (forested versus agriculture), and more strongly between habitats (riffles versus pools), but were not related to the sampled season (spring versus autumn) (Figure 8.3). Two trait syndromes (hereafter called trait syndrome A and B) were characteristic of communities found in forested streams. Trait syndrome A included semivoltine taxa with a non-seasonal development and long adult life span (taxa in this group were all river beetles, e.g. *Elmis aenea*, *Limnius volckmari* and *Oulimnius* sp.). Trait syndrome B included taxa with a short adult life span, medium size at maturity, slow seasonal development and poorly synchronised emergence (taxa in this group were all large dipterans such as *Dicranota* sp., *Ptychoptera* sp. and *Tipula* sp.). Two other trait syndromes characterised agricultural streams (hereafter trait syndrome C and D). Trait syndrome C included taxa with a small size at maturity, a very short adult life span, fast seasonal development and well synchronised emergence (typical taxa in this group were stoneflies such as *Amphinemura* sp., *Leuctra* sp. and *Nemoura* sp, as well as small non-biting midges such as Orthocladiinae, Tanypodinae and Chironomini). Trait syndrome D included univoltine taxa with high female dispersal and weak adult flying strength (typical taxa were caddisflies such as *Athripsodes* sp., *Mystacides* sp. and *Polycentropus* sp., as well as non-biting midges such as Orthocladinae, *Prodiamesa olivacea* and Tanypodinae).

Invertebrate traits also differed between pool and riffle habitats. Taxa with a large size at maturity were often associated with pool habitats (caddisflies such as *Halesus* sp., *Limnephilus* sp. and *Potamophylax* sp., as well as dragonflies and damselflies such as *Calopteryx* sp., *Somatochlora metallica* and Libellulidae). One trait syndrome (E) and a single trait were related to riffle habitats. Trait syndrome E was characterised by bi- and multivoltine taxa with low female dispersal (almost exclusively mayflies such as *Baetis* sp., *Centroptilum* sp. and *Caenis luteola*), whereas the single trait included taxa with a strong adult flying strength (mainly consisting of beetles such as Dytiscidae, *Haliplus* sp. and *Platambus maculatus*, and dragon- and damselflies such as Libellulidae, *Calopteryx* sp, and *Somatochlora metallica*).

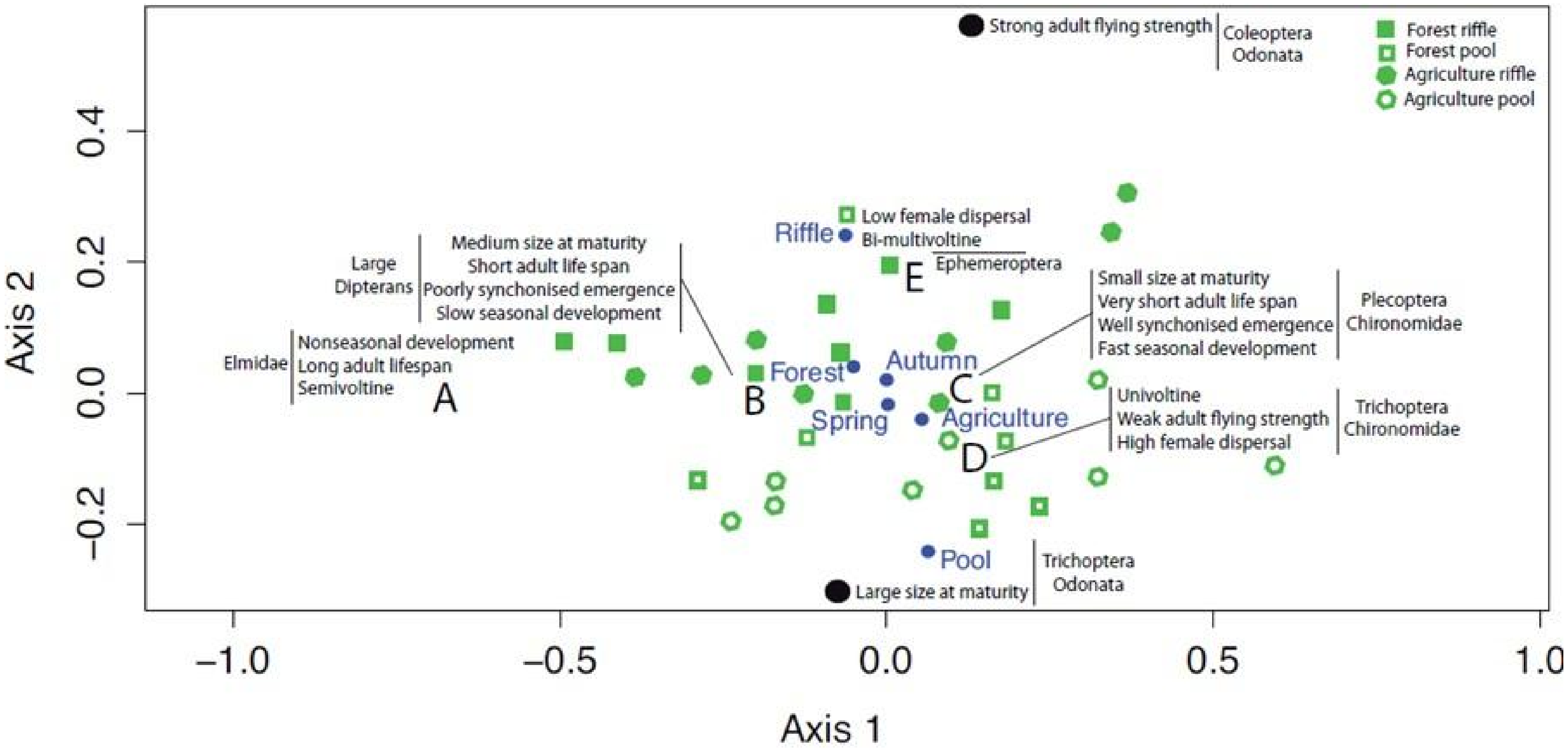


Figure 8.4 Nonmetric multidimensional scaling of square root transformed trait abundances. Blue circles = the three environmental factors: land use (agriculture – forest), habitat (pool – riffle) and season (spring – autumn). Letters A-E denote clusters of traits where each cluster contains several traits closely related in ordination space (each trait treated as species in the analyses; see text).

*Trait abundances, percentages in relation to environmental factors*

Significant interaction effects between land use and season, habitat and season and/or all three factors were apparent for only one trait syndrome (D) as well as for taxa with a large size at maturity (Odonata and Trichoptera). Abundances of these traits were higher in the autumn of 2008 compared to spring 2009 in the agricultural streams, whereas abundances were higher in spring compared to autumn in the forest streams. There were few statistically significant relationships when analysing the trait data as percentages rather than abundances, and these were all related to habitat type and were similar to results observed in analyses of trait abundances (data not shown).

## 8.4 Discussion

Although aquatic insect abundances were overall higher in the agricultural streams in our study, several lines of evidence from our species traits analyses suggest that their dispersal capacity as adults may be lower. Firstly, at the local reach scale, benthic insect assemblages from agricultural streams were characterised by trait syndromes associated with reduced dispersal capacity (e.g. small body size, low flight strength, very short adult life spans) than those from forest streams. Such traits were also more characteristic of pool rather than riffle assemblages in our study, which has implications for the dispersal capacities of insects in agricultural landscapes at larger scales. Furthermore, Carlson (2014) found that the total abundance of flying adult aquatic insects was significantly higher at the four agricultural sites (near the stream) compared to the forested sites. The abundance of the two main insect orders Diptera and Trichoptera declined more strongly with distance from the stream in the agricultural compared to the forested landscape (number of individuals caught was more than halved at 50 m from the stream edge in the agricultural landscape and remained > 75% in the forested landscape). Riffle habitats are typically greatly reduced in extent in agricultural relative to forested streams (Petersen 1992; Burcher et al. 2007), also our study region (pers. obs.), which is largely attributable to the loss of riparian integrity and its consequences for erosion and channel stability. If adult assemblages emerging from agricultural streams are dominated by taxa from pools, then the increase in traits associated with reduced dispersal evident at local scales within our study reaches is likely to be magnified at landscape scale. These effects of land use change on the innate dispersal capacities of adult aquatic insects have important implications for where adult aquatic insects are most likely to subsidise terrestrial food webs in agricultural compared to forested landscapes.

The export of insect adults from aquatic to terrestrial habitats can be a major component of the flux of energy and nutrients between the two ecosystems (Wesner 2010). The quantity exported is, however, dependent on the exporting aquatic ecosystem type (lakes versus streams; Gratton and Vander Zanden 2009), habitat type (pools versus riffles) (Iwata et al. 2003) and productivity of the donor ecosystem (Statzner and Resh 1993; Huryn and Wallace 2000). The overall pattern in production of potential emergent stream invertebrates was strongly affected by land use in our study, with abundances twice as high in the agricultural compared to the forested streams. This likely reflects higher autotrophic productivity associated with reduced shading and increased nutrient inputs in agricultural landscapes (Delong and Brusven 1998; Harding et al. 1999).

Since algae represent a high quality food source, this in turn favours greater secondary production of benthic macroinvertebrates (Allan 2004). While we did not quantify biomass production directly, it is likely that increased abundances in the agricultural streams translate into a higher biomass of adult aquatic insect since assemblages in both stream types were numerically dominated by small-sized Diptera.

Greater abundance of aquatic insects emerging from agricultural streams implies an increased potential subsidy for terrestrial food webs, but the spatial extent of this subsidy may be limited if adults do not disperse far from their natal habitat. Insect dispersal is regulated both by the innate dispersal traits characterising any given assemblage and by environmental factors. We found that traits affecting dispersal differed between the two land use types (forest versus agriculture) as well as between the two habitat types in the study (riffles versus pools). Moreover, we found that many traits with strong taxonomic affinities were intercorrelated into tight trait syndromes. This finding agrees with that of Poff et al. (2006) in their analysis of traits of North American lotic invertebrates and the review by Verberk et al. (2008) on life history strategies of freshwater macroinvertebrates. The two trait syndromes having a strong affinity to agricultural streams were characterised by trait syndrome C with a small size at maturity, a very short adult life span and well-synchronised development as well as trait syndrome D with univoltine taxa having a weak adult flying strength. Such traits have previously been associated with agricultural disturbance and represent traits associated with greater population resilience (e.g. Townsend and Hildrew 1994; Doledec et al. 2006; Poff et al. 2006; Tullos et al. 2009). Individuals characterised by these traits will generally stay close to the stream edge following emergence, and their short adult life span will limit the time available for extensive dispersal. Together, these attributes are likely to limit the potential for emerging aquatic insects to subsidise terrestrial foodwebs in agricultural landscapes over a large spatial area. The invertebrate taxa most associated with agricultural streams could further affect uptake into terrestrial food webs through interactions with terrestrial consumers and near-stream environments. For example, short adult life spans of the aquatic taxa limit the time available for each prey item to be consumed by terrestrial consumers, at least as a living prey item. Small prey size will also be an important factor for the terrestrial consumers as according to optimal foraging theory there is a trade-off between prey density, handling time and prey energetic value (Pyke et al. 1977; Stein et al. 1984). Accordingly, these prey traits should benefit relatively small-sized, sit-and-wait predators along the agricultural stream edges, such as web building and some ground-hunting spiders which have been shown to receive most of their carbon from aquatic insects (Collier et al. 2002, Sanzone et al. 2003; Akamatsu et al. 2004). Furthermore, many of the small-sized taxa, for instance weak-flying chironomid midges (Delettre and Morvan 2000), tend to aggregate in vegetated areas near the stream and are less likely to move from the preferred riparian habitat into open habitat in an agro-stream environment. In these agricultural streams, it is therefore probable that aquatic prey will decrease in importance as a subsidy for terrestrial consumers with distance from the stream (Lynch et al. 2002; Dudgeon 2007).

The traits related to the forested streams contrasted strongly with those in the agricultural landscape, with a higher proportion of individuals having a medium size (9-16 mm) at maturity, poorly synchronised emergence and longer adult life span (e.g. Elmidae and large dipterans). Accordingly, adult insects in the forested landscape have a greater potential to disperse further from the stream channel, and they also live longer, increasing the time available for longer dispersal and consumption by terrestrial consumers. Their larger size increases detectability as well as handling time by terrestrial consumers, but also probably improves the trade-off between handling time and the gain of nutrients and energy up to a certain size level (e.g. Hjelm and Persson 2003; Brose et al. 2008). Thus, the likelihood that the subsidy associated with forested streams will be dispersed over a large area seems higher than those emerging from agricultural streams, and the energetic return for terrestrial consumers may also be greater when feeding on larger- sized insects. We found no clear seasonal patterns in trait composition, with mainly interaction effects between land use and season for the Chironomidae and Trichoptera (univoltine, weak adult flying strength and high female dispersal). The two sampled seasons (late autumn 2008 and early spring 2009) did not differ markedly in trait composition, presumably because most taxa had not yet commenced emergence. Thus, while it is notable that land use had a substantially stronger effect on species trait composition in our study than variation between seasons, stronger effects of seasonality may have been detected if we had sampled later in the spring or during the summer.

In addition to the overall differences between forested and agricultural streams, abundance of invertebrates with an adult winged life stage also differed between in-stream habitats. Riffle habitats generally harboured about three times as many individuals with an adult winged life stage as pool habitats (within the same land use type), and riffle habitats in the agricultural landscape harboured about twice as many individuals per square meter as riffle habitats in the forest habitat (the interaction effect between land use and habitat was non-significant, however). A higher number of individuals in riffle versus pool habitats were also found in all studies evaluated in a review of the fauna of these two habitat types (Logan and Brooker 1983). Higher abundances of benthic insects in riffle habitats may reflect higher oxygenation concentrations and light penetration in the shallow, fast flowing waters, favouring greater biological activity and secondary production (Rosenfeld and Hudson 1997). Regardless of the underlying mechanism, these results highlight the particular importance of riffle habitats in supporting insects with a winged adult life stage and thus the transfer of matter and energy from aquatic to terrestrial ecosystems.

## 8.5 Conclusions

In this study we tested if degradation of riparian land-use (change from forest to agriculture) have effects on how much energy and nutrients are transported from the aquatic to the terrestrial environment and thus on riparian and catchment biodiversity and processes. Here the degradation effects goes both ways as degradation of the riparian zones has important effects on in-stream hydromorphology (riffle and pool sections) and biological communities (changes in species traits composition). This in turn affects the relationship between the in-stream biology and the riparian zone biodiversity through potential changes in fluxes of energy and nutrients from the aquatic environment to the terrestrial riparian and catchment level.

The differences in abundances of aquatic insects associated with riffle and pool habitats have important implications for the total potential production of aquatic subsidy, given variation in the spatial distribution of the two habitat types among different streams (Gratton and Vander Zanden 2009). Agricultural streams are often strongly modified relative to forested reference sites in the same region, reflecting both hydromorphological changes arising from losses of riparian vegetation, and direct anthropogenic modification of channel forms, such as dredging and agricultural channelisation (Burcher et al. 2007).

Consequently, agricultural streams often have long pool sequences separating short riffle sequences without meanders (Petersen 1992), in contrast to the more regular pool-riffle sequence which characterises many natural meandering systems flowing through forest (Leopold et al. 1964). This contrast was apparent in our study, where riffle habitats were very rare and short in length in the agricultural streams (P. Carlson *pers obs*), whereas in the forested streams there was a more even distribution between pool and riffle habitats. This implies that the overall reduction in traits associated with stronger dispersal at local scales within our agricultural study reaches (i.e. the significant land use effects in our analyses) is likely to be further strengthened by the overall decline in riffle habitats in agricultural landscapes, given that riffles both support higher abundances of winged adults and that those adults are likely to be better dispersers. Our trait-based analysis of benthic invertebrate assemblages indicates that agricultural streams characterised by long stretches of pool habitat are less likely to support insects with traits favouring greater dispersal than forested streams with a higher abundance of riffles. This implies that the relatively few and short riffle habitats in the agricultural landscape are potentially of great importance for the transfer of high quality food to terrestrial consumers. Loss of these habitats potentially jeopardises an important resource, and resilience of food availability, for many terrestrial organisms such as birds, bats and invertebrates. Future research should focus on how differences in innate dispersal capacity, as revealed in our analysis of species traits, interact with the environmental differences in the terrestrial habitats of both forested and agricultural landscapes to affect not only the dispersal of adult aquatic insects but also their consumption by terrestrial consumers.

## 8.6 References

Akamatsu, F., H. Toda, and T. Okino. (2004). "Food source of riparian spiders analyzed by using stable isotope ratios." Ecological Research 19: 655-662.

Allan, J. D. (2004). “Landscapes and riverscapes: The influence of land use on stream ecosystems.” Annual Review of Ecology Evolution and Systematics 35: 257-284.

Baxter, C. V., K. D. Fausch, and W. C. Saunders. (2005). “Tangled webs : reciprocal flows of invertebrate prey link streams and riparian zones”. Freshwater Biology 50: 201-220.

Brose, U., Ehnes, R. B., Rall, B. C., Vucic-Pestic, O., Berlow, E. L. and Scheu, S. (2008). “Foraging theory predicts predator–prey energy fluxes.” Journal of Animal Ecology 77: 1072–1078.

Burcher, C. L., and E.F. Benfield. (2007). “The land-cover cascade: Relationships coupling land and water.” Ecology 88: 228-242.

Carlson, P. (2014). Land use effects on ecological linkages between small streams and their surrounding terrestrial habitats. Doctoral Thesis No. 2014:18. Faculty of Natural Resources and Agricultural Sciences. Swedish University of Agricultural Sciences, Uppsala, Sweden.

Chan, E. K. W., Y. T. Yu, Y. X. Zhang, and D. Dudgeon. (2008). “Distribution patterns of birds and insect prey in a tropical riparian forest.” Biotropica 40: 623-629.

Collier, K. J., S. Bury, and M. Gibbs. (2002). “A stable isotope study of linkages between stream and terrestrial food webs through spider predation.” Freshwater Biology 47: 1651-1659.

Dejong, J. and I. Ahlen. (1991). F”actors Affecting the Distribution Pattern of Bats in Uppland, Central Sweden.” Holarctic Ecology 14: 92-96.

Delettre, Y. R. and N. Morvan. (2000). “Dispersal of adult aquatic Chironomidae (Diptera) in agricultural landscapes.” Freshwater Biology 44: 399-411.

Delong, M. D. and M. A. Brusven. (1998). “Macroinvertebrate community structure along the longitudinal gradient of an agriculturally impacted stream.” Environmental Management 22: 445-457.

Doledec, S., N. Phillips, M. Scarsbrook, R. H. Riley, and C. R. Townsend. (2006). “Comparison of structural and functional approaches to determining landuse effects on grassland stream invertebrate communities.” Journal of the North American Benthological Society 25: 44-60.

Dudgeon, D. (2007). “Going with the flow: global warming and the challenge of sustaining river ecosystems in monsoonal Asia.” Sustainable and Safe Water Supplies 7: 69-80.

England, L. E. and A. D. Rosemond. (2004). “Small reductions in forest cover weaken terrestrial-aquatic linkages in headwater streams.” Freshwater Biology 49: 721-734.

Fukui, D., M. Murakami, S. Nakano, and T. Aoi. (2006). “Effect of emergent aquatic insects on bat foraging in a riparian forest.” Journal of Animal Ecology 75: 1252-1258.

Gladyshev, M. I., M. T. Arts, and N. N. Sushchik. (2009). Preliminary estimates of the export of omega-3 highly unsaturated fatty acids (EPA+DHA) from aquatic to terrestrial ecosystems. Pages 179-210 *in* M. Kainz, M. T. Brett, and A. M.T., editors. Lipids in Aquatic Ecosystems. Springer, New York.

Gratton, C. and M. J. Vander Zanden. (2009). “Flux of aquatic insect productivity to land: comparison of lentic and lotic ecosystems.” Ecology 90: 2689-2699.

Gray, L. J. (1993). “Response of Insectivorous Birds to Emerging Aquatic Insects in Riparian Habitats of a Tallgrass Prairie Stream.” American Midland Naturalist 129: 288-300.

Göthe, E., Angeler, D.A. and Sandin, L. (2012). “Metacommunity structure in a small boreal stream network.” Journal of Animal Ecology 82: 449-458.

Harding, J. S., R. G. Young, J. W. Hayes, K. A. Shearer, and J. D. Stark. (1999). “Changes in agricultural intensity and river health along a river continuum.” Freshwater Biology 42: 345-357.

Hjelm, J. and Persson, L. (2003). “Size-dependent attack rate and handling capacity: inter-cohort competition in a zooplanktivorous fish.” Oikos 95: 520–532.

Hoekman, D., J. Dreyer, R. D. Jackson, P. A. Townsend, and C. Gratton. (2011). “Lake to land subsidies: Experimental addition of aquatic insects increases terrestrial arthropod densities.” Ecology 92: 2063-2072.

Huryn, A. D. and J. B. Wallace. (2000). “Life history and production of stream insects.” Annual Review of Entomology 45: 83-110.

Iwata, T., S. Nakano, and M. Murakami. (2003). “Stream meanders increase insectivorous bird abundance in riparian deciduous forests.” Ecography 26: 325-337.

Kato, C., T. Iwata, S. Nakano, and D. Kishi. (2003). “Dynamics of aquatic insect flux affects distribution of riparian web-building spiders." Oikos 103: 113-120.

Lamberti, G. A., D. T. Chaloner, and A. E. Hershey. (2010). "Linkages among aquatic ecosystems." Journal of the North American Benthological Society 29: 245-263.

Leopold, L. B., M. G. Wolman, and J. P. Miller. (1964). Fluvial Processes in Geomorphology. Freeman and company, New York.

Likens, G.E, and F.H. Bormann. (1974) "Linkages between terrestrial and aquatic ecosystems." Bioscience 24: 447-456.

Logan, P. and M. P. Brooker. (1983). "The Macroinvertebrate Faunas of Riffles and Pools." Water Research 17: 263-270.

Lynch, R. J., S. E. Bunn, and C. P. Catterall. (2002). "Adult aquatic insects: Potential contributors to riparian food webs in Australia's wet-dry tropics." Austral Ecology 27: 515-526.

Moldenke, A. R. and C. V. Linden. (2007). "Effects of Clearcutting and Riparian Buffers on the Yield of Adult Aquatic Macroinvertebrates from Headwater Streams." Forest Science 53: 308-319.

Nakano, S. and M. Murakami. (2001). "Reciprocal subsidies: Dynamic interdependence between terrestrial and aquatic food webs." Proceedings of the National Academy of Sciences of the United States of America 98: 166-170.

Oksanen, J., F. G. Blanchet, R. Kindt, P. Legendre, P. R. Minchin, R. B. O'Hara, G. L. Simpson, P. Solymos, M. H. H. Stevens, and H. Wagner. (2013). vegan: Community Ecology Package.

Petersen, R. C. (1992). "The Rce - a Riparian, Channel, and Environmental Inventory for Small Streams in the Agricultural Landscape." Freshwater Biology 27: 295-306.

Poff, N. L., J. D. Olden, N. K. M. Vieira, D. S. Finn, M. P. Simmons, and B. C. Kondratieff. (2006). "Functional trait niches of North American lotic insects: traits-based ecological applications in light of phylogenetic relationships." Journal of the North American Benthological Society 25: 730-755.

Polis, G. A., W. B. Anderson, and R. D. Holt. (1997). "Toward an integration of landscape and food web ecology: The dynamics of spatially subsidized food webs." Annual Review of Ecology and Systematics 28: 289-316.

Pyke, G. H., H. R. Pulliam, and E. L. Charnov. (1977). "Optimal Foraging - Selective Review of Theory and Tests." Quarterly Review of Biology 52: 137-154.

R Development Core Team. (2012). R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

Reich, P. B., I. J. Wright, J. Cavender-Bares, J. M. Craine, J. Oleksyn, M. Westoby, and M. B. Walters. (2003). "The evolution of plant functional variation: Traits, spectra, and strategies." International Journal of Plant Sciences 164: S143-S164.

Richardson, J. S. (1991). "Seasonal Food Limitation of Detritivores in a Montane Stream - an Experimental Test." Ecology 72: 873-887.

Rosenfeld, J. S. and J. J. Hudson. (1997). "Primary production, bacterial production and invertebrate biomass in pools and riffles in southern Ontario streams." Arch. Hydrobiol. 139: 301–316.

Sanzone, D. M., J. L. Meyer, E. Marti, E. P. Gardiner, J. L. Tank, and N. B. Grimm.

(2003). "Carbon and nitrogen transfer from a desert stream to riparian predators." Oecologia 134: 238-250.

SAS Institute Inc. 2011. Base SAS® 9.3 Procedures Guide. Cary, NC: SAS Institute Inc.

Statzner, B. and L. A. Beche. (2010). "Can biological invertebrate traits resolve effects of multiple stressors on running water ecosystems?" Freshwater Biology 55: 80-119.

Statzner, B. and V. H. Resh. (1993). "Multiple-Site and Multiple-Year Analyses of Stream Insect Emergence - a Test of Ecological Theory." Oecologia 96: 65-79.

Stein, R. A., C. G. Goodman, and E. A. Marschall. (1984). "Using Time and Energetic Measures of Cost in Estimating Prey Value for Fish Predators." Ecology 65: 702-715.

Torres-Ruiz, M. and J. D. Wehr. (2010). "Changes in the nutritional quality of decaying leaf litter in a stream based on fatty acid content." Hydrobiologia 651: 265-278.

Townsend, C. R. and A. G. Hildrew. (1994). "Species Traits in Relation to a Habitat Templet for River Systems." Freshwater Biology 31: 265-275.

Tullos, D. D., D. L. Penrose, G. D. Jennings, and W. G. Cope. (2009). "Analysis of functional traits in reconfigured channels: implications for the bioassessment and disturbance of river restoration." Journal of the North American Benthological Society 28: 80-92.

Wallace, J. B., S. L. Eggert, J. L. Meyer, and J. R. Webster. (1997). "Multiple trophic levels of a forest stream linked to terrestrial litter inputs." Science 277: 102-104.

Verberk, W.C.E.P., Siepel, H. and Esselink, H. (2008). "Life-history strategies in freshwater macroinvertebrates." Freshwater Biology 53: 1722-1738.

Wesner, J. S. (2010). "Aquatic predation alters a terrestrial prey subsidy." Ecology 91: 1435-1444.

Willson, M. F. and K. C. Halupka. (1995). "Anadromous Fish as Keystone Species in Vertebrate Communities." Conservation Biology 9: 489-497.

# 9 Riparian hydromorphology influences on fish - Danube Delta Case Study

## 9.1 Introduction

Riverine floodplains are among the most endangered landscapes worldwide (Olson and Dinerstein 1998; Tockner and Stanford 2002). In headwaters, the construction of dams has caused most damage (Ward and Stanford 1995), whereas lowland sections have been most affected by channelization, i.e. shortening of the channels and bed incision and floodplain reclamation. Floodplain reclamation has affected the processes and functions of the river-floodplain systems of Europe’s large rivers (Junk et al. 1989; Diester 1994; Tockner et al. 1998; Junk 1999; Tockner et al. 1999; Tockner et al. 2000; Schneider 2002; Tockner et al 2009). For example, the Lower Rhine floodplain is lined by high embankments isolating the former floodplain from the river, and the width of the floodplain has been reduced up to 10 km with 1 km on average (Buijse et al. 2002). The Danube floodplain losses are also severe, being 95%, 75%, 72% of the upper, middle and lower sections, respectively (Schneider 2002). This damage is not limited to Europe; for example along the Mississippi (Gore and Shields 1995), only about 10% of the former floodplains are in a near natural state (cf. Jungwirth, Muhar and Schmutz 2002). In addition, dams have been constructed to facilitate navigation during low flow. Due to these drastic impacts, the hydrological connectivity between the river and its floodplain is restricted to groundwater pathways since geomorphological dynamics are mostly absent (Marchand 1993; Heiler et al. 1995). This has large consequences for the migration of permanent aquatic organisms such as fish and aquatic molluscs, affecting the overall biodiversity of the systems (Schiemer 1999; Grift 2001; Robinson Tockner and Ward 2002).

Hydromorphological degradation is widespread throughout Europe and for the first time the extent of different types of hydromorphological degradation have been quantified. In the European Union, the first River Basin Management Plans (RBMP) from the Member States has designated a total of 104,311 river water bodies with a total length of 1.17 million km. About 56% of the river water bodies, which equates to 64% of their total length, have been reported as having less than good ecological status or potential (EEA 2012). Hydromorphological changes have been identified as the most widespread pressure on the ecological status of EU waters. Hydromor-phological pressures and altered habitats account for 48.2% and 42.7% of their river water bodies, respectively (Fehér et al. 2012).

The consensus among scientists is that floodplain loss is a major cause of the decline in fish populations along large rivers (Bayley 1991; Cowx and Welcomme 1998; Regier 1989; Schiemer 2000; Schiemer et al. 2004; Schlosser 1991). Studies of endangered fish species show that both species preferring flowing water and still-water species are at risk (Lelek 1987; Schiemer and Spindler 1989; Schiemer and Waidbacher 1992; Guti 1995; IUCN 2014; de Leeuw et al. 2007). An assessment of eight European and North American large rivers indicated that the fish species that depend for all or some of their life stages on fluvial conditions are most imperiled (Galat and Zweimuller 2001).

Classification of fish species into ecological guilds based on their habitat requirements in different life stages (Banarescu 1964; Schiemer and Spindler 1989; Schiemer and

Waidbacher 1992; Guti 1995) has become a common used tool for planning of river floodplain restoration works and indicators for assessment of ecological integrity, connectivity status and restoration success in large river systems such as the Danube, the Rhine and the Oder river systems (Schiemer et al. 1991; Schiemer 1999; Schiemer et al. 1999; Schmutz and Jungwirth 1999; Schmutz et al. 2000; Bischoff and Wolter 2001; Grift 2001; Chovanec et al. 2002; Noble et al. 2007; Schomaker and Wolter 2011). Among the ecological guilds considered for the biological assessment, diadromous and potamodromous fish species are particularly sensitive to continuum disruptions as an interruption of migrations severely affects these species (Bacalbasa-Dobrovici 1989; Northcote 1998).

The aim of this case study is to describe the impact of reduced hydrological connectivity between river and floodplain on fish in the large rivers reaches such as Danube Delta, supporting activities within the task 3.4. The embankment of the floodplain in the Middle and Lower Danube is the main cause of the decline of eurytopic fish species but also of rheophilic species that spend some life stages in the floodplain wetlands (Bacalbasa-Dobrovici 1989; Schmutz and Jungwirth 1999; Staras 1999; Buijse et al. 2002; Schiemer and Waidbacher 1992; Schiemer 1999; Schiemer et al. 1999; Schiemer et al. 2004; Navodaru et al. 2005; Navodaru et al. 2008).

Here we investigate how this disruption of the hydrological connectivity between the river and floodplain affects 1) the floodplain vegetation, riparian invertebrate fauna and fish recruitment dynamics and 2) building on existing datasets, the linkage between riparian area, biodiversity and hydromorphological channel degradation across relevant scales, from the reach to the catchment scale.

## 9.2 Materials and Methods

*Catchment scale: Site description. Danube Delta*

The Danube River is the second largest European river and the world's most international river basin with a length of 2,857 km and a catchment size of 801,463 km$^2$. It includes the territories of 19 countries and is home to 83 million people. At its end, the Danube Delta is located on the coast of the Black Sea and includes the area between its three arms located in Romania and the secondary delta of the Chilia arm, which is Ukrainian territory (Table 9.1).

The valley form is plain floodplain and the channel form is diverse due to the complexity of the delta: braided arms (braiding intensity 65%); split, sinuous and composite anabranching; meandering channel (degree of sinuosity >1.26). The average width of the main arms is 450 m at Kilia, 400 m at Sulina and 450 m at Sf. Gheorghe. The mean depth of the three branches amounts to 13 m. The average width of the floodplain is about 100 km. This section is characterised by a medium current velocity of 0.7 m/s (Chilia 0.7 m/s, Sulina 0.65 m/s, Sf. Gheorghe 0.68 m/s). Multiple wood collections are present on the river bank; fallen trees are sporadic. Lotic side arms and dead arms, cut-off channels and oxbow lakes, temporary side arms and standing water bodies fed by the tributaries are water bodies found in the floodplain (UNDP/GEF 2003). Slope values vary and the dominant substrates are a mixture of river stretch types 19, 20 and 21 developed within the FP 7 /REFORM project (D2.1).

Both Romanian and Ukrainian Danube Delta territories are parts of the UNESCO-MAB Programme-designated Biosphere Reserves and form a common Transboundary Biosphere Reserve. The Romanian Danube Delta is also included in the EU Natura 2000 network.

Figure 9.1 Danube Delta within the Danube River basin.

Table 9.1 General characteristics of the Danube Delta. Sources: www.icpdr.org; Gastescu and Stiuca 2008; Sommerwerk et al. 2009.

| Characteristics | | |
|---|---|---|
| Geographical coordinates | Latitude interval (N) | 45.490000; 45.834000 |
| | Longitude interval (E) | 28.741000; 29.790000 |
| Ecoregion | | Pontic province |
| Catchment area (km$^2$) | | 4,560 |
| Surface, total (km$^2$) | | 4,180 |
| In Romania (km$^2$) | | 3,510 |
| Climate | | temperate |
| Geology | | Siliceous, organic |
| Slope (m km $^{-1}$) | Sulina arm | 0.045-0.001 |
| (max-min range) | Chilia arm | 0.035-0.001 |
| | Sf. Gheorghe arm | 0.029-0.001 |
| Discharge (m$^3$ s$^{-1}$) | maximum | 15,540 |
| | average | 6,515 |
| | minimum | 1,350 |
| Altitude (m, above sea level) | maximum | +12.4 |
| | average | +0.52 |
| | minimum | -3.00 |
| Inhabitants (number) | | 15,000 |

In the Danube Delta, about 30% of the natural wetlands have been lost due to embankments, but near-natural reference habitats have been preserved (Buijse 2002; Schneider 2002). The human impacts are consequences of different land use policies and drivers, which have changed the pristine features. At the end of the 19th century, measures were taken to improve the navigability of the middle arm of the delta, without major impact on the other delta's functions. Between 1903-1960, in the so-called "capture fishery period", new channels were built or the older natural ones enlarged to activate water circulation inside the delta, aiming to improve fish production.

The "fish culture period", between 1971-1980, followed by the "agriculture period", mostly between 1983-1989, altered the network of water courses. The embankments increased from 24,000 ha to 97,000 ha and resulted in cut-off from the Danube river pulse system, and the total length of canals in the Romanian delta increased from 1743 km to 3496 km (Gastescu et al. 1983). As a result of political changes, the Danube Delta was declared a Biosphere Reserve in 1991 and the conservation of natural values and recovery of wetlands functions became priority objectives. A restoration programme of damaged ecosystems was initiated in 1993 as a marking of the new period in the Danube Delta history as a Biosphere Reserve.

*Catchment scale – historical analysis methods*

Data from scientific papers, grey literature, reports on historical hydromorphological changes in the Danube Delta and available long-term statistical data on capture fishery were used to identify and characterise the responses of fish at catchment scale. The use of commercial fishing data might affect reliability and should always be checked (Schmutz et al. 2000). Uncertainty of fish data is lower for the data derived from the fully state-controlled system in Romania before 1990.

Additional data on hydromorphological changes upstream of the Danube Delta with effects on semi-migratory and migratory fish species were also used in this case study (Figure 9.2).

As a result of canal development and embankments in the Danube Delta (Figure 9.3), a lake typology induced by the connectivity gradient from the river to the floodplain lakes was developed and described, including the type of the fish community (Figure 9.4) (Oosterberg et al. 2000).

Table 9.2 Summary characteristics of the lake types related to fish

| Variable | Type 2 | Type 1 | Type 3 |
|---|---|---|---|
| Water residence time | Low | Intermediate | High |
| P total (summer av. mg/l) | 0.10-0.15 | 0.10-0.15 | 0.10-0.15 |
| Transparency | CLEAR | TURBID | CLEAR |
| Fish composition | Eurytopic | Intermediate | Limnophilic |

The cumulative effect of increased water inflow by dredging new canals and water pollution between 1960 and 1989 was increased nutrient fluxes from the river to its delta (Staras 2001) (Figure 9.5).

*Reach scale: Site description – the Isac-Uzlina lake complex.*

At reach scale, an example of the impact of interruption of lateral connectivity on the fish population is given based on the blocking the canal between the Danube River and a group of lakes in 2002 (Navodaru et al. 2005). These lakes are shallow and characterised

by a similar gradient in connectivity and cumulative residence time (residence time plus travel time) in all three types of lakes described for the Danube Delta (Oosterberg et al. 2000). The canal was blocked in early summer 2002 as a water management solution to mitigate the intensive siltation of Lake Uzlina.
Sampling data from a survey on fish seasonality carried out in the years 2001-2002 in the Isac-Uzlina complex of lakes have been used to assess the effect of interruption of the lateral connectivity on fish (Figure 9.6; Table 9.3).

*Reach scale: Sampling method*

Sets of three gillnets for each sampling sites were used. They were set at 7 p.m. and lifted again at 7 a.m.. A gill net unit had 14 mesh sizes (range 6 – 75 mm bar mesh) and a total length of 42 m, which is in accordance with Nyman (1990) as described by Navodaru et al. (2002).

Species composition was recorded by mesh size in terms of abundance, individual total length (accuracy 1 mm) and weight (accuracy 1 g). Fish were identified to species level and grouped by their preference for flowing or stagnant conditions, feeding type, food preference, reproductive substrate requirements, tolerance of low dissolved oxygen levels and body colour (Schiemer and Waidbacher 1992). Catch per unit of effort was calculated for 100 m gillnet fishing effort by species and lakes.

Table 9.3 Coordinates of the sampling sites, June 2001-2002 and September 2001-2002

| Lake | Geographical position |
|---|---|
| Isac | N45°06'31" E29°17'45" |
| Isac | N45°07'16" E29°16'23" |
| Isac | N45°07'30" E29°15'56" |
| Cuibul cu lebede | N45°07'53" E29°20'36 |
| Cuibul cu lebede | N45°07'13" E29°20'42 |
| Cuibul cu lebede | N45°07'10" E29°20'18" |
| Chiril | N45°05'06" E29°18'11" |
| Chiril | N45°04'57" E29°19'30" |
| Gherasimova | N45°07'22" E29°18'41" |
| Uzlina | N45°05'01" E29°15'38" |
| Uzlina | N45°04'48" E29°14'52" |
| Uzlina | N45°05'32" E29°15'47" |
| Uzlina | N45°06'05" E29°15'60" |

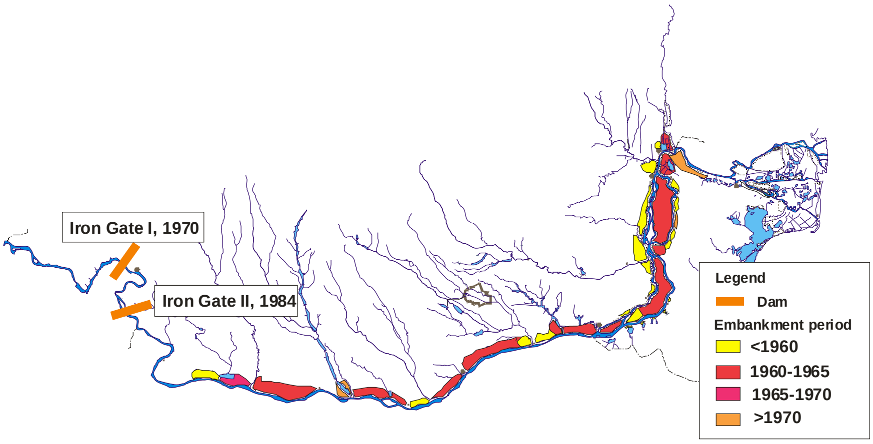


Figure 9.2 Hydromorphological changes in the Lower Danube.

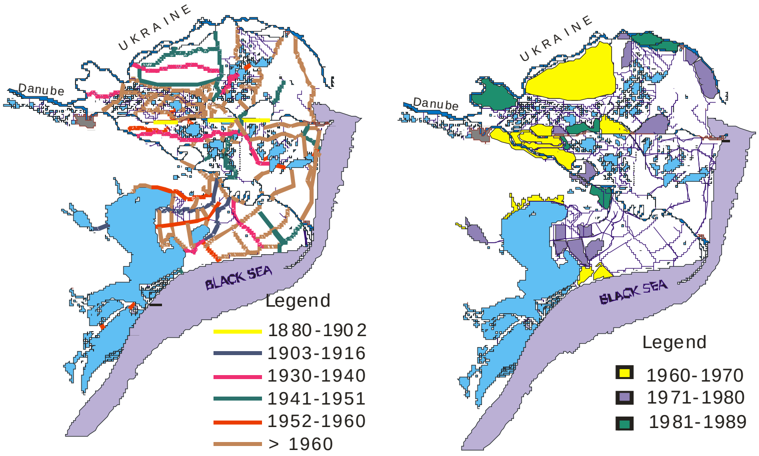


Figure 9.3 History of embankments (left) and man-made canals (right) in the Danube Delta.

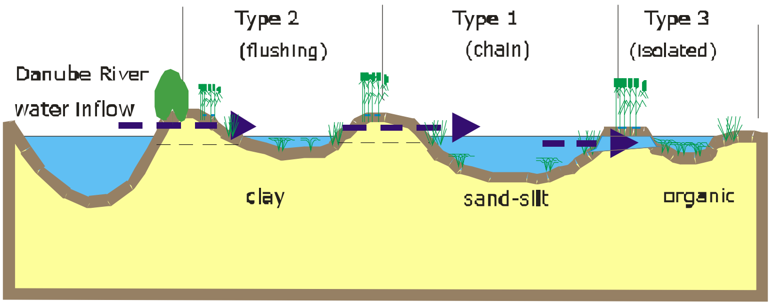


Figure 9.4 Lake typology along the connectivity gradient in the Danube Delta.

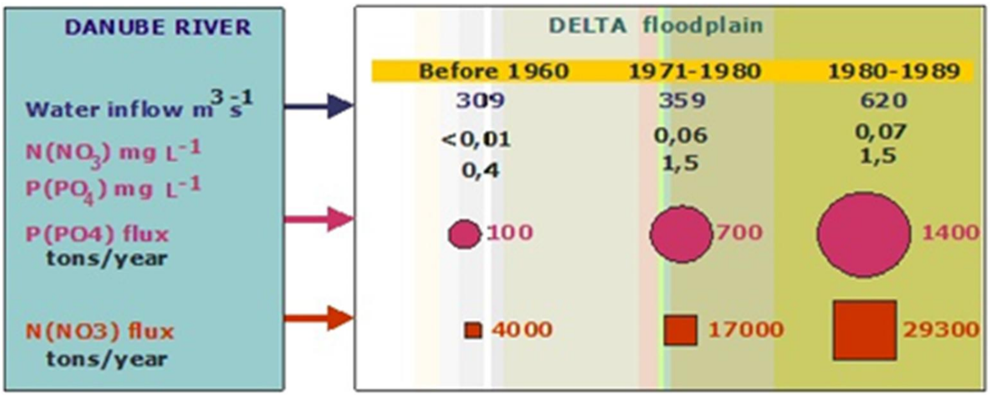


Figure 9.5 Nutrient fluxes as a cumulative effect of increased water inflow and pollution.

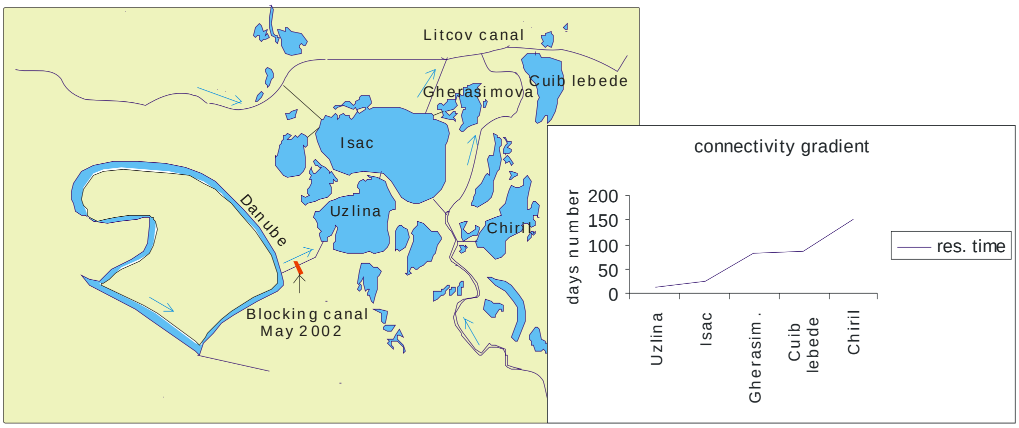


Figure 9.6 The Isac-Uzlina lake complex

## 9.3 Results

### 9.3.1 Catchment scale

Fishery statistics indicate that catches in the 1920s were dominated by common carp (*Cyprinus carpio*) (21%), pike (*Esox lucius*) (20%) and roach (*Rutilus rutilus*) (16%), and one third of the catches comprised piscivorous fish (pike, zander, *Stizostedion lucioperca* and wels catfish, *Silurus glanis*). In total, about 10 species contributed significantly to the catches, which is regarded as being a diverse composition (Daia 1926). A similar picture continued until 1960s before construction of the embankments and dams.

*Embankment and eutrophication impacts*

The embankment of 85% of the Lower Danube floodplain, upstream from the delta, was undertaken mainly between 1960-1965. A dramatic decline was seen in the fishery of common carp in the delta and also in the Lower Danube in general (Figure 9.7) following the embankment, when the semi-migratory fish species like carp (*Cyprinus carpio*) lost key spawning and nursing areas. Opportunistic species such as roach, on the other hand, which do not need migration for spawning grounds, increased in abundance.

The embankments in the Danube Delta itself also had an impact on non-migratory fish species. The most affected fish species were the group of cyprinids, which lost a significant part of their spawning and nursing grounds. This explains the decline of the fishery in general and of cyprinids in particular (Figure 9.7; Figure 9.8) (Staras 1999).

The response of the fish community to eutrophication was typical for European temperate waters (Ligtvoet and Grimm 1992) and was reflected in a reduction in the number of species and proliferation of generalist species with broader habitat requirements (Figure 9.9) (Navodaru et al. 2002).

### *Dam impacts*

The Iron Gates I and II hydropower dams that were built in 1970 (at km 943) and 1984 (at km 863) interrupted the migration of sturgeon for spawning upstream, which contributed to the collapse of the sturgeon fishery in the whole Lower Danube-Black Sea system (Bacalbasa 1989; Bacalbasa 1991) (Figure 9.10).

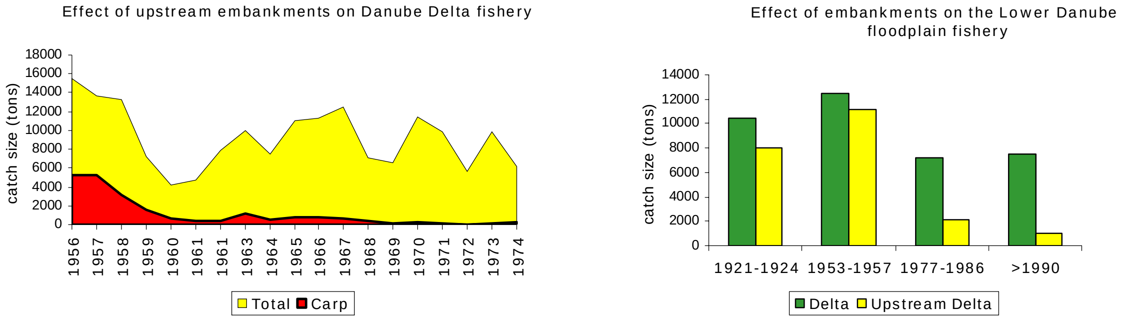


Figure 9.7 Effects of embankments on fishery in the Lower Danube.

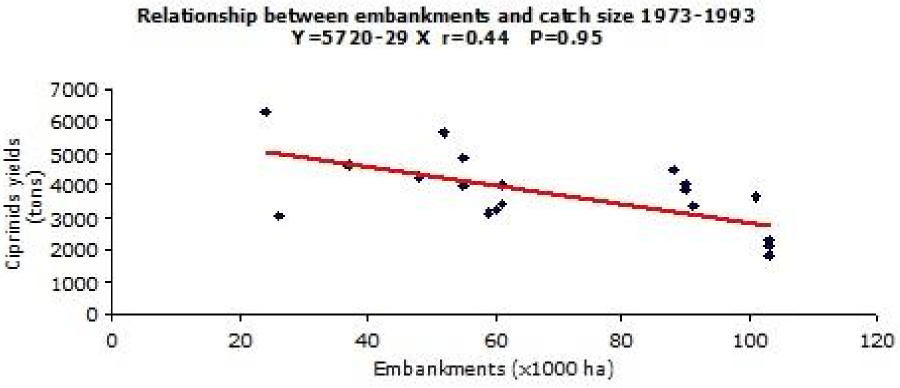


Figure 9.8 The impact of embankments on cyprinids yields.

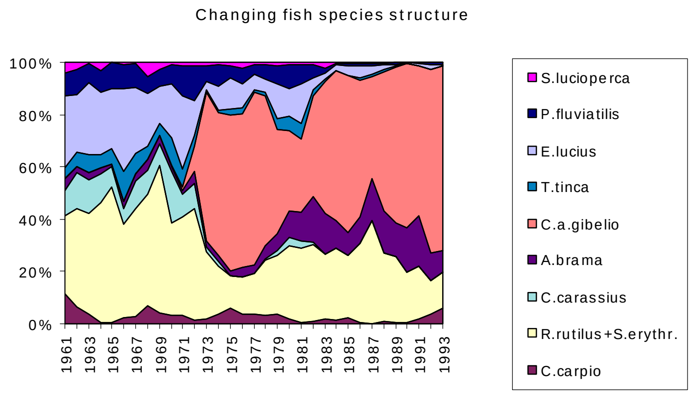


Figure 9.9 Temporal changes in fish community structure in the commercial catches (1961 – 1993).

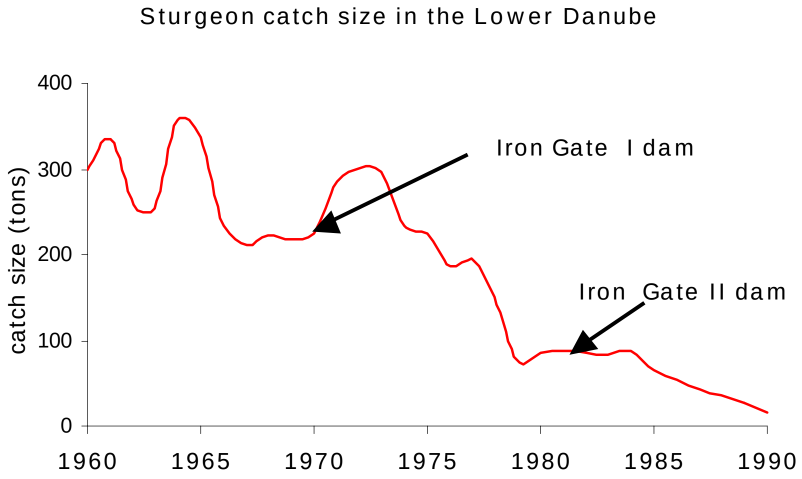


Figure 9.10 Dynamics of the sturgeon catches in the Lower Danube.

### 9.3.2 Reach scale

In the case of the Isac-Uzlina lake complex, these lakes were characterised in 2001 before the connectivity was interrupted by blocking the inflowing waters. The highest abundances were recorded in lakes close to the river and the lowest abundances in the remote lakes. Eurytopic fish species dominated in the lakes with high connectivity, whereas limnophilic species dominated in the remote lakes.

A sharp decline in fish abundance was recorded in June 2002, after blocking the canal in all lakes, except for the most remote lake. The impact of this on fish abundance increased with increasing connectivity between the river and the lake.

The strongest decline was recorded for eurytopic species, but also the limnophilic species group declined, mostly in lakes directly connected with the river (Figure 9.11). The rheophilic species were represented in small numbers by asp (*Aspius aspius*), which was present as adults and young specimens in all lakes in early autumn 2001 before blocking the Uzlina canal. After blocking, asp was found only in lakes with indirect connection with the Danube via the Litcov canal, and all specimens found were at immature stages. A sharp decrease in abundance and biomass was also recorded for the eurytopic species pike, which is the main target for sport fishing in the area (Figure 9.12).

Data on species composition and abundance before and after connectivity interruption are shown in Appendix 1.

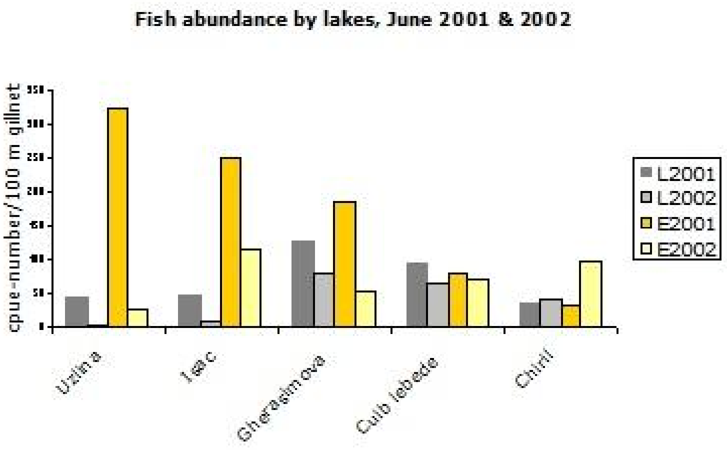


Figure 9.11 Fish abundance before (2001) and after (2002) blocking canal relative to lakes and fish guilds (L=limnophilic; E=eurytopic).

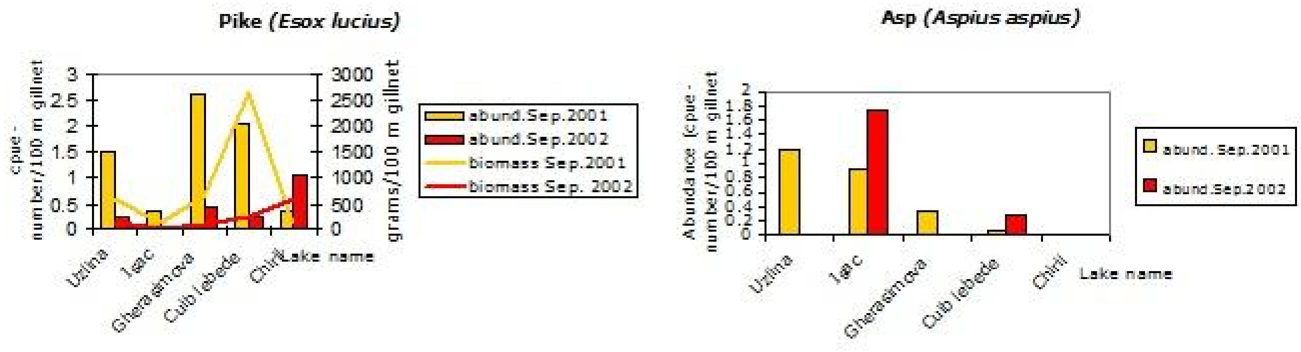


Figure 9.12 Abundance of asp and pike before and after blocking canal

## 9.4 Discussion

In many European rivers as well as along the Danube River upstream of the delta, floodplains have been isolated from the main channel either for safety reasons or land reclamation, mostly for agricultural purposes. As a consequence, the lateral connection between the river and its floodplain has been lost (Ward and Stanford 1989). The resulting landscape is a main channel with a fringe of floodplain or riparian zone. Such zones are still dynamic and frequently inundated, but large areas of former floodplains and remote lakes are seldom flooded as they have been amputated from the river through dikes or levees. Limnophilic/”black fish” species are highly vulnerable to the loss of lateral connectivity. These species are typical for floodplain water bodies both during the dry and the wet season and table communities may occur even under completely isolated conditions (Schomaker, Wolter 2011). In particular, limnophilic species are physiologically adapted to life in floodplain water bodies, especially low dissolved oxygen levels (Regier et al. 1989). For example, limnophilic species may have supplementary breathing organs, the dominant sensory organs are tactile, scales are reduced or absent, and the skin is dark-coloured. In water with low levels of dissolved oxygen, European weatherfish (*Misgurnus fossilis*) can breathe through their skin and intestines and *U. krameri* through their swimming bladder (Banarescu 1964). In addition to limnophilic species, eurytopic/”grey fish” species, which use the floodplain for spawning, and rheophilic/”white fish” species, which search the floodplain for food during the wet season, are also negatively affected by loss of lateral connectivity.

The general pattern of species richness in the European large river floodplain like the Danube, Rhine and Oder is a decline in fish species richness as a consequence of the loss of connectivity. Our findings confirm that the fish fauna is an excellent indicator of the ecological quality of large European river, because of the ontogenetically variable requirements of the various species belonging to these systems (Schiemer and Spindler 1989; Table 10.4). The ecological groups of limnophilic, eurytopic and rheophilic species have been impacted by interruption of the lateral connectivity in most of the system except for the remote lakes. Today, quite a number of limnophilic as well as rheophilic species are listed on the IUCN red list of threatened species (IUCN 2014). Of the species recorded in the Danube Delta crucian carp (*Carassius carassius*) and European weatherfish are listed as ‘lower risk’ and *U. krameri* as ‘vulnerable’. According to Banarescu (1994) *U. krameri* is little vulnerable and *T. tinca* is vulnerable in Romania. Isolated water bodies still host communities of several endangered species, however, with an increasing number of species from river to remote or isolated water bodies, reflecting that these species are more tolerant to lack of connectivity (Schiemer 1999; Ward et al. 1999; Grift 2001; Navodaru et al. 2003; Schomaker and Wolter 2011). Particularly for large European rivers, rheophilic type B fish species like asp (Habitats Directive, Annex II) are excellent indicators of lateral connectivity since they require connectivity between the river and the floodplains to have a complementary habitat for feeding and as a winter refuge (Schiemer et al. 2004).

The long-term data set of the commercial fishery provides a good indication of a shift in the ecological character of lakes in the Danube Delta and illustrates the consequences of upstream floodplain impoundments (Bacalbasa-Dobrovici 1989; Navodaru et al. 2002). The historical data from the 1920s suggest that the fish community was characteristic of lakes with clear water with aquatic vegetation and shallow areas flooded during the wet season. Prior to impoundment, the floodplains upstream of the delta were used by species such as potamodromous common carp (*Cyprinus carpio)* wild form for spawning (Bacalbasa-Dobrovici 1989; Navodaru et al. 2002). Land reclamation of almost 5000 km$^2$ of floodplain along the Lower Danube in the 1950s resulted in the collapse of the common carp fishery, not only in the Lower Danube but also in the Danube Delta. The situation worsened in the 1960s when another area of approximately 1000 km$^2$ was embanked and reclaimed for agriculture in the Danube Delta itself, and the catches of all native cyprinid species declined.

Canalisation of the Danube Delta for various purposes, for instance to improve water circulation and increase fish production, followed by navigation, reed harvesting and construction of freshwater reservoirs for irrigation significantly affected the fish community. In the beginning of the 20th century, the increased water circulation in the delta and lagoon lakes resulted in an increase in fish production (Antipa 1932). But at the same time, the changes in the hydrological infrastructure with enhanced connectivity between the Danube River and its lakes in the delta most certainly were the reason behind the decline in the populations of limnophilic species. Starting from the 1970s, a general nutrient enrichment of Danube River took place, which in combination with enhanced water circulation mediated a connectivity/eutrophication gradient creating different types of lakes with different fish biota (Oosterberg et al. 2000). Consequently, over the years the fish fauna has shifted from dominance of limnophilic/"black fish" species towards a fauna characterised by eurytopic/"grey fish". In the case of the Isac-Uzlina lake complex studied, all lakes are connected, but the remote lakes are slightly dominated by limnophilic/"black fish" species, while those near the river are dominated by eurytopic/"grey fish". This pattern reflects the connectivity gradient characteristic of the lakes described above and is similar in all major lake complexes in the delta. In addition to this shift in the fish fauna, the fish community has been simplified and is now dominated by a few opportunistic species (*Abramis brama*, *Sander lucioperca* and the invasive exotic *Carassius gibelio*), while catches of clear-water fish species (*Carassius*, *Esox lucius* and *Tinca tinca*) has declined. The prey-predator ratio also declined from 60:40 to 80:20 during this period (Navodaru et al. 2002).

Table 9.4 Indicators of fish responses to the connectivity gradient between river and flooplain

| | Connectivity status | | | |
|---|---|---|---|---|
| Indicator | Directly connected | Intermediate | Remote | Isolated[2] |
| Residence time[1] | Small | intermediate | large | very large (underground) |
| Fish abundance and biomass | High | medium | low | low |
| Rheophilic fish species abundance | low (VR) | low (VR) | low (VR) or absent | absent or very low |
| Eurytopic fish species abundance | high (C) | intermediate | low (C) | low (C) |
| Limnophilic fish species abundance | low (RC) | low (RC) | high(C) | high (C) |
| L/Eu ratio | Low | low | high | high |
| Fish community type | Eu | intermediate | L-Eu | L-Eu |
| Species richness | High | intermediate | low | Low, but highest limnophilic |
| Fishery status | Good | good | fair/ bad | collapsed |

[1] cumulated: travel time + lake residence time (Oosterberg et al. 2000)

[2] (Schiemer 1999; Schomaker and Wolter 2011)

Abundance status: C=common (>10%); RC=relatively common (5-10%); R=rare (1-5%); VR=very rare (<1%);

Preference for current guild: L=limnophilic; Eu=eurytopic; Rh= rheophilic

Despite hydromorphological changes, the recent studies show that the Danube Delta is still in a rather pristine state and that there is still a highly diverse fish community throughout the delta (Oosterberg et al. 2000; Buijse et al. 2002; Navodaru et al. 2002). This is of great value in a European context and reflects that the Danube Delta and its lakes has remained a hydrological interconnected system despite the hydromorphological impairments. The high diversity in the delta lakes itself is due to the co-occurrence of limnophilic, rheophilic and eurytopic forms.

*Anadromous sturgeons*

The access of anadromous sturgeons to the spawning grounds of the Middle Danube River has been interrupted since 1972 by the construction of the Iron Gate I Hydropower dam (rkm 943). The migratory route was further shortened by the Iron Gate II dam (rkm 863) since 1984. This contributed to the collapse of sturgeon fishery in the Lower Danube, including the Danube Delta. The restoration of connectivity is believed to be a high priority requirement in the recovery strategy for the Danube sturgeons (Bloesch et al. 2006).Successful passing of the Iron Gate dams across future fish pass facilities and technical solutions for successful upstream and downstream migration is questionable according to the present knowledge. The homing behaviour is another issue requiring further research. Quick implementation of measures for the conservation of anadromous sturgeons should focus on the downstream end of the Iron Gate dams where viable populations need to be gradually built up to ensure the necessary number of individuals migrating upstream in the Lower Danube River (Guti 2014).

## 9.5 Conclusions

The Danube Delta proved to be an appropriate site for assessment of the impact of hydromorphological degradation on fish within a river-floodplain system. An explanation is that the Danube Delta with its lakes has remained a hydrological interconnected system. In many other European river floodplains as well as along the Danube River upstream of the Delta, floodplains have been isolated from the river, rendering distinction between factors influencing the fish more difficult.

Fishes are the only group of aquatic organisms for which historical information is available. Long-term data on commercial fishing and the history of hydrotechnical works gave valuable information on the impact at catchment scale. The development of the capture fishery in the Danube Delta correlates very well with habitat alterations showing a decline in some species and increases in others, changes in fish fauna composition and biodiversity loss. Reliable long-term commercial fishery data on migratory anadromous and potamodromous fish species may be useful to correlate and explain effects of historical changes in the lateral or longitudinal connectivity of the river systems.

Despite hydromorphological changes, our recent study showed that the Danube Delta is in a rather pristine state and that there is high diversity in fish community structure throughout the delta. The high fish species diversity in the lakes is due to the co-occurrence of rheophilic, eurytopic and limnophilic forms. From a management point of view, maintaining the existing connectivity gradient in the delta lakes is vital for biodiversity conservation and also social economic needs. The research results show that blocking of canals as a measure to mitigate siltation and nutrient inputs is not appropriate. Rather lengthening of the distance between the river and lakes by meandering and reducing the slope should be taken into consideration.

The results from the case study confirm that fish are excellent indicators of the lateral connectivity between large European rivers and their floodplains. A eurytopic-limnophilic community with presence of some rheophilic species indicates a connectivity gradient, whereas a limnophilic-eurytopic community type indicates low connectivity. Long-term isolation and stagnant water bodies are indicated by the presence of only limnophilic specialist species adapted to hypoxic conditions. Absence of rheophilic and decline of eurytopic-limnophilic species could be an indicator of recent or short-term connectivity interruption.

The acquired knowledge of the relationship between lateral and longitudinal river connectivity and fish species in their life stages is a sound basis for planning restoration measures in the Lower Danube system, including floodplain restoration and Iron Gate fish passes.

## 9.6 References

Antipa Gr. (1932). Les principes de l' amélioration de la productivite du bas Danube. Bulletin de la Section Scientifique. Academie Roumaine. Monitorul Oficial si Imprimeriile Statului. Imprimeria Nationala, Bucharest. Vol. XV (3-4), 19 p.

Bacalbasa-Dobrovici, N. (1989). The Danube River and its Fisheries. In: Proceedings of the International Large River Symposium (Ed. D.P. Dodge). Canadian Special Publication of Fisheries and Aquatic Sciences 106: 455-468.

Bacalbasa, N. (1991). Fish population evolution (1985-1988) in the last built Reservoir Iron Gate II. Verhandlungen der Internationale Vereinigung für Limnologie 24: 2524-2527.

Banarescu P. (1964). Pisces-Osteichthyes. Vol XIII, Ed. Acad. Romane, 962 p.

Banarescu P. (1994). The present-day conservation status of the freshwater fish fauna of Romania. In: Protection of nature and surroundings environment. Tom 38(1): 1-16.

Bayley, P.B. (1991). The flood pulse advantage and the restoration of river-floodplain systems. Regulated Rivers: Research and Management 6: 75-86.

Bischoff, A. and C. Wolter (2001).The flood of the century on the river Oder: Effects on the 0+ fish community and implications for floodplain restoration. Regulated rivers. Research and Management 17: 171-190.

Bloesch J., T. Jones, R. Reinartz and B. Streibel (2006). Action Plan for Conservation of Sturgeons (*Acipenseridae*) in the Danube River Basin. Nature and Environment. Council of Europe Publishing 144: 121.

Buijse, A.D., H. Coops, M. Staras, L.H. Jans, G.J. Van Geest, R.E. Grift, B.W. Ibelings, W. Oosterberg, F.C.J.M Roozen. (2002). Restoration strategies for river floodplains along large lowland rivers in Europe. Freshwater Biology 47: 889-907.

Chovanec, A., Schiemer, F., Weidbacher, H. and Spolwind, R. (2002). Rehabilitation of a

Heavily Modified River Section of the Danube in Vienna (Austria): Biological Assessment of Landscape Linkages on Different Scales. International Review of Hydrobiology 87: 183-195.

Cowx J. G. and Welcomme R.L. (1998). Rehabilitating of Rivers for Fish. Fishing News Books, Oxford.

Daia P.P. (1926). Exploatarea pescariilor statului (Exploatation of state fisheries). Tipografia Copuzeanu. 201 p (in Romanian).

de Leeuw J.J., Buijse A.D., Haidvogl G., Lapinska M.,Noble R., Repecka R., Virbickas T., Wisniewolski W. and Wolter C. (2007). Challenges in developing fish-based ecological assessment methods for large floodplain rivers. Fisheries Management and Ecology 14, 483–494.

Dister E. (1994). The function, evaluation and relicts of near-natural floodplains. In: Biologie der Donau (Ed. R. Kinzelbach). Limnologie Aktuell 2. Gustav Fischer Verlag, Stuttgart: 317-328.

EEA – European Environmental Agency (2012). European Environmental Agency, Copenhagen, EEA Report 8/2012.

Fehér, J., J. Gáspár, K. Szurdiné-Veres, A. Kiss, P. Kristensen, M. Peterlin, L. Globevnik, T. Kirn, S. Semerádová, A. Künitzer, U. Stein, K. Austnes, C. Spiteri, T. Prins, E. Laukkonen, A.-S. Heiskanen (2012). Thematic assessment for EEA Water 2012 Report. European Topic Centre on Inland, Coastal and Marine Waters, Prague, ETC/ICM Technical Report 2/2012.

Galat, D.L. and I. Zweimuller (2001). Conserving large-river fishes: is the highway analogy an appropriate paradigm?. Journal of the North American Benthological Society 20: 266–279.

Galat, D.L., L.H. Fredrickson, D.D. Humburg et al. (1998). Flooding to restore connectivity of regulated, large river wetlands. Bioscience 48: 721–733.

Gastescu, P., B. Driga, C. Anghel (1983). Caracteristici morfohidrografice ale Deltei Dunarii, Hidrobiologia 17, Academia Romana, Bucuresti (in Romanian).

Gastescu P. and Stiuca R. (eds.) (2008). *Delta Dunarii, Rezervatie a Biosferei (Danube Delta, Biosphere Reserve)*. Editura CD Press, ISBN 978-973-1760-98-9, 399 pp. (in Romanian, with English summary).

Gore, J.A. and F.D. Jr. Shields (1995). Can large rivers be restored ?. Bioscience 45: 142–152.

Grift, R. E. (2001). How fish benefit from Floodplain Restoration along the Lower River Rhine. PhD Thesis. Wageningen University, Wageningen. ISBN 9058084884.

Gunther-Diringer D. (2001). Evaluation of wetlands and floodplain areas in the Danube River basin. In: River Restoration in Europe (Eds H.J. Nijland and M.J.R. Cals). RIZA report 2001.023. RIZA, Lelystad, The Netherlands: 91-94.

Guti G. (1995). Conservation status of fishes in Hungary. Opusc. Zool. Budapest XXVII–XXVIII, 153–158.

Guti G. (2014). Can Anadromous Sturgeon Populations be Restored in the Middle Danube River ?. Acta zoologica bulgarica, Suppl. 7: 63-67.

Heiler, G., T. Hein, F. Schiemer and G. Bornette (1995). Hydrological connectivity and flood pulses as the central aspects for the integrity of a river–floodplain system. Regulated Rivers: Research and Management 11: 351–361.

ICPDR (2005). *Roof Report*. www.icpdr.org.

IUCN (2014). International Union for the Conservation of Nature, Gland. (www.redlist.org.)

Jungwirth, M., S. Muhar and S. Schmutz (2002). Re-establishing and assessing ecological integrity in riverine landscapes. Freshwater Biology 47: 867–887.

Junk, W.J., P. B. Bayley and R.E. Sparks (1989). The flood pulse concept in River-Floodplain Systems. Can. Spec. Publ. Fish. Aquat. 106: 110-127.

Junk, W.J. (1999). The flood pulse concept of large rivers: learning from the tropics. Archiv fur Hydrobiologie 115 (Suppl.) (Large Rivers, 11): 261–280.

Lelek, A. (1987). The Freshwater fishes of Europe,9. Threatened Fishes of Europe. Aula-Verlag, Wiesbaden. 343 p.

Lelek, A. (1989). The Rhine River and Some of its Tributaries Under Human Impact in the Last Two Centuries 106: 469-487.

Ligtvoet, W, M.P. Grimm (1992). How an estuary changed into freshwater lake. Proceedings and Information, CHO-TNO 46: 69-84.

Marchand, M. (Ed.) (1993). Floodplain Rheabilitation Gemenc: Main Report. Delft Hydraulics, RIZA/Vituki, Delft, Lelystad, The Netherlands/Budapest, Hungary.

Navodaru, I., A. D. Buijse and M. Staras (2002). Effects of Hydrology and Water Quality on the Fish Community in Danube Delta Lakes. International Review Hydrobiology 87: 329-348.

Navodaru, I., A.D. Buijse, M. Staras and J.D. Leeuw (2003). Seasonal patterns in fish distribution in Danube Delta lakes. RIZA work document 2003.132x.

Navodaru, I., M. Staras, A. D. Buijse and J. J. de Leew (2005). Changes in fish populations in Danube Delta lakes: effects of hydrology and water quality change. Review of results and potential for rehabilitation. Ecohydrology and Hydrobiology 5 (3): 245-256.

Navodaru, I., M. Staras and I. Cernisencu (2008). Evolution of the ihtiofauna- Results of sampling and monitoring. In: Evolution of Babina polder after restoration works (Eds. E. Schneider, M. Tudor and M. Staras). WWF Auen Institute, Germany / Danube Delta National Institute, Romania: 42-46.

Noble, R., Cowx, I., Goffaux, D. and Kestemont, P. (2007). Assessing the health of European rivers using functional ecological guilds of fish communities: standardising species classification and approaches to metric selection. Fisheries Management and Ecology 14: 381–392.

Northcote T.G. (1998). Migratory behaviour of fish and its significance to movement through riverine fish passage facilities. In: Fish Migrations and Fish Bypasses (Eds. M. Jungwirth, S. Schmutz and S. Weiss). Oxford, Fishing News Books, Blackwell Science: 3-18.

Nyman, L., (1990) Surveying fish and crustacean resources in tropical reservoirs. Fisheries Development Series 42: 1–28.

Olson, D.M. and E. Dinerstein (1998). The global 200: a representation approach to conserving the earth's most biologically valuable ecosystems. Conservation Biology 12: 502–515.

Oosterberg, W., M. Staras, L. Bogdan, A.D. Buijse, A. Constantinescu, H. Coops, J. Hanganu, B.W. Ibelings, G. A. M. Menting, I. Navodaru, L. Torok (2000). Ecological gradient in the Danube Delta; present state and man-induced changes. RIZA rapport nr. 2000.15. Netherlands.

Regier, H.A., R. L. Welcomme, R. J. Steedman, H.F. Henderson (1989). Rehabilitation of degraded river ecosystems. Canadian Special Publication of Fisheries and Aquatic Sciences 106: 86-97.

Robinson, C.T., K. Tockner and J.V. Ward (2002). The fauna of dynamic riverine landscapes. Freshwater Biology 47: 661-677.

Schiemer, F. (1999). Conservation of biodiversity in floodplain rivers. Archiv für Hydrobiologie 115 (Suppl.) (Large Rivers, 11): 423-438.

Schiemer, F. (2000). Fish as indicators for the assessment of the ecological integrity of large rivers. In: Assessing the Ecological integrity of Running Waters (Eds. M. Jungwirth, S. Muhar and S. Schmutz). Hydrobiologia, 422/423: 271-278.

Schiemer, F. and T. Spindler (1989). Endangered fish species of the Danube River in Austria. Regulated Rivers: Research and Management 4: 397-407.

Schiemer, F., T. Spindler, H. Wintersberger, A. Schneider and A. Chovanec (1991). Fish fry associations: Important indicators for ecological status of large rivers. Verh. Internat.Verein. Limnol. 24: 2497-2500.

Schiemer, F. and H. Waidbacher (1992). Strategies for conservation of the Danubian Fish Fauna. In: River Conservation and Management (Eds.P.J. Boon, P. Calow and G.E. Pets). Willey. Chichester: 364-382.

Schiemer, F., C. Baumgartner and K. Tockner (1999). Restoration of floodplain rivers: the ,,Danube restoration project". Regulated Rivers: Research and Management 15: 231-244.

Schiemer, F., G. Guti, H. Keckeis and M. Staras (2004). Ecological status and problems of the Danube River and its fish fauna: a review. In: Proceedings of the second symposium on the management of large rivers for fisheries, Vol I, (Eds. Welcomme R. and Petr T.). FAO Regional Office for Asia and the Pacific, Bangkok, Thailand. RAP Publication 2004/16: 273-279.

Schlosser, I. J. (1991). Stream fish ecology: a landscape perspective. Bioscience, 41: 704-712.

Schomaker, C., C. Wolter (2011). The contribution of long-term isolated water bodies to floodplain fish diversity. Freshwater Biology 56: 1469-1480.

Schmutz, S. and M. Jungwirth (1999). Fish as indicators of large river connectivity: the Danube and its tributaries. Archiv für Hydrobiologie 115 (Suppl.) (Large Rivers, 11): 329-348.

Schmutz S. and M. Kaufmannn, B. Vogel, M. Jungwirth and S. Muhar (2000). A multi-level concept for fish-based, river-type-specific assessment of ecological integrity. Hydrobiologia 422/423: 279-289.

Schneider, E. (2002). The ecological functions of the Danubian floodplains and their restoration with special regard to the Lower Danube. Large Rivers, 13, 1-2, Arch. Hydrobiol. Suppl. 141, 1-2: 129-149.

Sommerwerk N., Baumgartner C., Bloesch J., Hein T., Ostojic A., Paunovic M., Schneider-Jacoby M., Sieber R. and Tockner K. (2009). The Danube River Basin. In: Rivers of Europe (Eds. K. Tockner, Uehlinger U. and Robinson C. T.), pp. 59-112. Elsevier Academic Press. ISBN-13: 978-0-12-369449-2.

Sparks, R. E. (1995). Need for ecosystem management of large rivers and their floodplains. Bioscience 45: 168–182.

Staras, M. (1999). Fishery in relation with environment in the Danube Delta Biosphere Reserve. In: Dealing with nature in deltas (Ed. Nijland H.L.). RIZA nota 99.011 RIZA, Lelystad, The Netherlands: 157-168.

Staras, M. (2001). Restoration programme in the Danube Delta: achievements, benefits and constraints. In: River Restoration in Europe (Eds H.J. Nijland and M.J.R. Cals). RIZA report 2001.023. RIZA, Lelystad, The Netherlands: 95-101.

Tockner, K., F. Schiemer and J. V. Ward (1998). Conservation by restoration: the management concept for a river floodplain system on the Danube River in Austria. *Aquatic Conservation: Marine and Freshwater Ecosystems* 8: 71-86.

Tockner, K., F. Schiemer, C. Baumgartner, G. Kum, E. Weigand, I. Zweimuler and J. W. Ward (1999). The Danube restoration project: species diversity patterns across connectivity gradients in the floodplain system. Regulated Rivers: Research and Management 15: 245-258.

Tockner, K., F. Malard and J. V. Ward (2000). An extension of the flood pulse concept. Hydrological Processes 14: 2861-2883.

Tockner, K. and J. A. Stanford (2002). Riverine flood plains: present state and future trends. Environmental Conservation 3: 308-330.

Tockner, K., U. Uehlinger and C. T. Robinson (2009). Rivers of Europe. ISBN-13: 978-0-12-369449-2. Elsevier Ltd.

Van Urk, G. and H. Smit (1989). The lower Rhine geomorphological changes. In: Historical Change of Large Alluvial Rivers: Western Europe (Eds G.E. Petts, H. Moller and A.L. Roux). John Wiley and Sons Ltd, Chichester: 167-182.

Ward J.V. and J.A. Stanford (1989). Riverine ecosystems: the influence of man on catchment dynamics and fish ecology. In: Proceedings of the International Large River Symposium (Ed. D.P. Dodge). Canadian Special Publication of Fisheries and Aquatic Sciences 106: 56-64.

Ward, J.V. and J. A. Stanford (1995). The serial discontinuity concept: extending the model to floodplain rivers. Regulated Rivers: Research and Management 10:159–168.

Ward J.W., K. Tockner and F. Schiemer (1999). Biodiversity of floodplain river ecosystems: ecotones and connectivity. Regulated rivers. Research and Management 15: 125-139.

Appendix 1, Species composition and abundance (number fish/100 m gillnet) before and after connectivity interruption

| | June 2001 | | | | | June 2002 | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | Uzlina | Isac | Gherasimova | Cuibul cu lebede | Chiril | Uzlina | Isac | Gherasimova | Cuibul cu lebede | Chiril |
| <u>Limnophilic</u> | | | | | | | | | | |
| *Carassius carassius* | 0.00 | 0.43 | 1.15 | 4.83 | 15.44 | 0.00 | 0.00 | 0.36 | 0.65 | 3.03 |
| *Leuciscus borysthenicus* | 1.19 | 0.07 | 0.00 | 0.00 | 0.00 | 0.29 | 0.58 | 4.33 | 8.73 | 7.97 |
| *Leucaspius delineatus* | 0.00 | 0.00 | 0.00 | 0.79 | 0.79 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| *Pungitius platygaster* | 0.79 | 0.00 | 0.00 | 1.59 | 0.00 | 0.00 | 0.79 | 0.79 | 0.00 | 0.00 |
| *Rhodeus sericeus amarus* | 34.13 | 20.49 | 65.08 | 4.76 | 6.35 | 1.59 | 3.57 | 19.84 | 1.59 | 9.52 |
| *Scardinius erythrophthalmus* | 8.26 | 25.54 | 50.58 | 52.24 | 9.02 | 3.32 | 3.82 | 45.02 | 47.76 | 15.66 |
| *Tinca tinca* | 0.18 | 2.42 | 9.45 | 29.22 | 3.17 | 0.29 | 0.65 | 9.52 | 7.29 | 5.78 |
| Total limnophilic | 44.55 | 48.95 | 126.26 | 93.43 | 34.78 | 5.48 | 9.42 | 79.87 | 66.02 | 41.96 |
| <u>Eurytopic</u> | | | | | | | | | | |
| *Abramis brama* | 0.00 | 0.69 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| *Alburnus alburnus* | 176.98 | 70.79 | 34.13 | 29.37 | 5.56 | 8.01 | 30.09 | 16.16 | 1.59 | 21.57 |
| *Blicca bjoerkna* | 19.84 | 30.74 | 23.16 | 4.98 | 0.07 | 0.58 | 6.24 | 2.24 | 3.39 | 2.53 |
| *Carassius auratus gibelio* | 0.76 | 1.80 | 7.29 | 0.65 | 0.72 | 2.09 | 2.89 | 1.01 | 0.43 | 1.73 |
| *Cobitis sp.* | 5.56 | 8.73 | 3.17 | 3.97 | 5.56 | 0.00 | 1.59 | 1.59 | 0.79 | 3.97 |
| *Cyprinus carpio* | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.14 | 0.04 | 0.07 | 0.79 | 0.00 |
| *Esox lucius* | 0.14 | 0.18 | 0.07 | 0.22 | 0.00 | 0.07 | 0.29 | 0.58 | 0.87 | 0.97 |
| *Gobiidae* | 0.40 | 0.79 | 0.79 | 1.59 | 1.59 | 0.00 | 0.00 | 0.00 | 0.0.0 | 0.00 |
| *Gymnocephalus cernuus* | 1.59 | 7.29 | 3.17 | 0.79 | 0.00 | 4.33 | 6.49 | 2.45 | 0.00 | 0.11 |
| *Perca fluviatilis* | 42.39 | 33.30 | 23.88 | 10.39 | 3.39 | 4.11 | 34.74 | 12.84 | 5.12 | 13.49 |
| *Rutilus rutilus* | 69.41 | 82.90 | 86.51 | 28.57 | 15.95 | 7.14 | 30.99 | 15.51 | 59.88 | 54.95 |
| *Silurus glanis* | 0.11 | 0.29 | 0.29 | 0.14 | 0.07 | 0.00 | 0.00 | 0.07 | 0.14 | 0.11 |
| *Stizostedion lucioperca* | 4.37 | 11.15 | 0.79 | 0.00 | 0.00 | 0.00 | 1.59 | 0.00 | 0.00 | 0.00 |
| Total eurytopic | 321.54 | 248.64 | 183.26 | 80.66 | 32.90 | 26.48 | 114.94 | 52.53 | 73.01 | 99.43 |
| Other sp. | | | | | | | | | | |
| Cyprinid hybrid | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.14 | 0.00 | 0.07 | 0.07 |
| *Hypophthalmichthys molitrix* | 0.00 | 0.07 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| *Lepomis gibbosus* | 0.00 | 0.00 | 0.0722 | 0.1443 | 0.00 | 0.00 | 0.00 | 0.07 | 0.00 | 0.11 |
| *Alosa caspia nordmanni* | 0.00 | 0.4329 | 0.00 | 0.00 | 0.00 | 0.79 | 1.33 | 0.00 | 0.00 | 0.04 |