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SEVENTH FRAMEWORK PROGRAMME

WISER

Water bodies in Europe: Integrative Systems to assess Ecological status and Recovery

DELIVERABLE

Deliverable D5.1-4: Guidance on management options and measures of pressure reduction to improve the ecological status of rivers with emphasis on the implications of global/climate change

Lead contractor: **ALTERRA Green World Research**

Contributors: **Piet Verdonschot (Alterra), Christian Feld, Armin Lorenz, Veronica Dahm (UDE), Anahita Marzin, Didier Pont, Maxime Logez, Gérôme Belliard (Irstea), Andreas Melcher, Helga Kremser, Martin Seebacher (BOKU)**

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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)	

Non-technical summary

The WISER WP5.1-4 deliverable provides guidelines, a synthesis and key questions. All three chapters are based on the results of studies performed in WP5.1.

Guidelines

The demand to restore riverine ecosystems and improve their overall ecological quality has greatly increased since 2000. Restoration successes can be strongly enhanced when both empirical knowledge and ecological theory are being combined and used during the planning and implementation of restoration schemes. This "River restoration and management guidance for practitioners" guidance document summarises the knowledge obtained during the WISER project and identifies principles from project experiences and ecological theory that have been, or could be, used to guide practical riverine restoration. Ten key guidelines are presented.

Synthesis

In the Synthesis chapter all results were translated to specifically address the practitioners. The main messages were:
Sensitivity of BQEs to pressures; the four BQEs bring complementary information on the river ecological status and, therefore the use of multiple biological groups is considered appropriate for monitoring programs.
Conceptual framework of potential effects of restoration; a general conceptual framework to analyse abiotic effects (= states) and biological recovery in the course of restoration (= society's response to degradation) is presented (DPSIRR) and shown to be of value for river restoration.
Actual effects of restoration; there is sufficient evidence for the chief role of broad-scale stressors that may act at the scale of entire catchments and control environmental conditions at finer spatial scales. Consequently, local restoration is considered ineffective as long as broad-scale stressors continue to impact a site or reach.
Scale and hierarchy in degradation processes; in general agricultural land use cannot be used as proxy for river degradation. More in detail, stressors from a larger scale have a stronger impact on BQEs than local stressors. The highest impact is linked to agriculture in the catchment, meaning eutrophication and alkalinisation (indicated by diatoms), while the main local stressor is agriculture along the river leading to structural degradation (indicated by fish and diatoms).
Indicators of degradation and restoration; short-term indication of restoration effects is going to provide the basis for adaptive management, but in turn required a short-term application, i.e. annual or bi-annual monitoring events within the first decade after restoration.
Ecological and environmental thresholds; ecological thresholds refer to change points or transition zones along a stressor gradients at which a dramatic change of biological characteristics is detectable. Hence, such thresholds may mark a critical value of the stressor variable, or ranges thereof, yet they may not be obvious for all stressors and biological responses, respectively.

Effect of climate change (temperature) and global change; shifts in metrics related to 'cold' and 'warm' water fish species clearly indicated that over the period of the Water Framework Directive implementation, it will be necessary to revise the multimetric indices based on functional traits, which are commonly used now, such as for instance the European Fish Index.

Key messages

- Riverine assemblages respond differently to individual stressors and stress levels
- Environmental stressors act hierarchically
- Catchment and riparian land use control local habitat conditions
- Restoration is more likely to be successful, if upriver physical habitat degradation and land use impacts are low
- Local restoration is often unsuccessful
- River Basin Management Plans insufficiently account for research and monitoring demands
- Climate change alters fish assemblage structure and function distribution in Europe
- Projections of European fish distribution under Climate Change implies the loss of cold water-adapted species

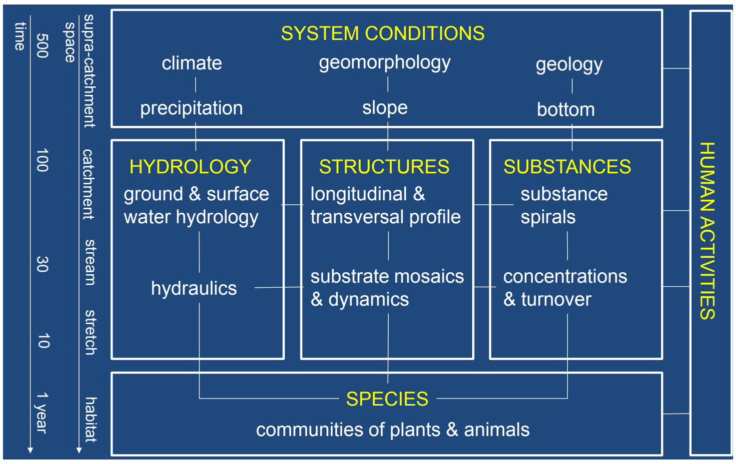
GUIDE TO BEST MANAGEMENT PRACTISE

River restoration and management guidance for practitioners

The demand to restore riverine ecosystems and improve their overall ecological quality has greatly increased since 2000. Restoration successes can be strongly enhanced when both empirical knowledge and ecological theory are being combined and used during the planning and implementation of restoration schemes. This guidance document summarises the knowledge obtained during the WISER project and identifies principles from project experiences and ecological theory that have been, or could be, used to guide practical riverine restoration.

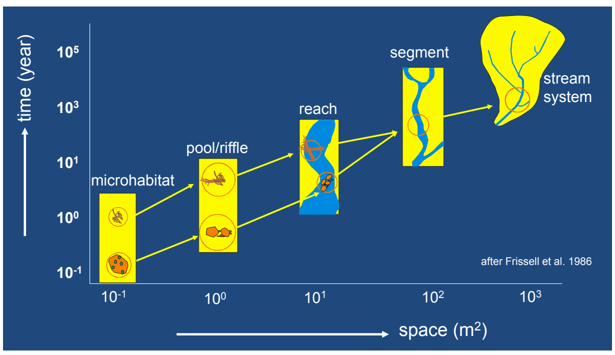
Ten key principles in river restoration

1. The key factors governing the ecosystem



River restoration must take into account all hydrological, morphological, physical-chemical and biological parameters, including anthropogenically altered (stressor) as well as natural environmental (landscape descriptor) variables. Restoration measures should address the instream habitats, but also the riparian areas and in particular the land use in the floodplain. Furthermore, restoration must fit the geomorphological and hydrological settings of the catchment.

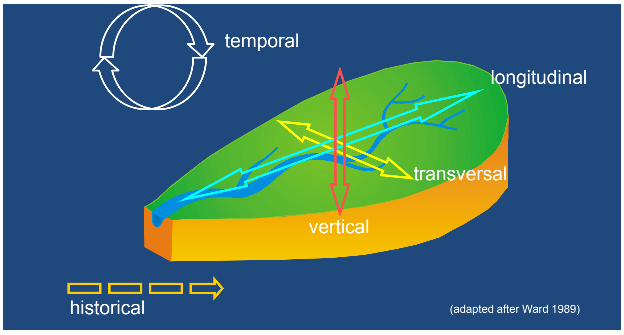
2. The appropriate scale and hierarchy



The spatial and temporal scales of restoration projects are critical to success. Success is more likely with integrated large-scale or catchment-scale restoration projects, but those often are infeasible in terms of the available resources and conflicts of stakeholder interests. Small-scale restoration may remedy specific local problems. Restoration should occur at the appropriate spatial scale such that restoration is not reversed by the prevailing (multiple) disturbances.

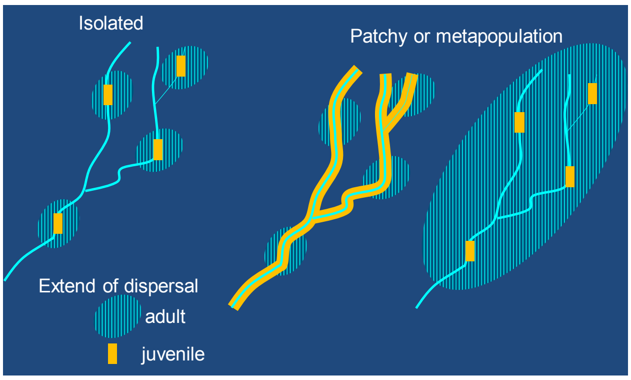
There is a top-down hierarchy or dominance in key factors and between large and fine scales. Ideally restoration is implemented top-down, i.e. the main stressors at the large scale is being addressed first, before less important stressors and/or finer scales are being addressed.
Integrating different scales in space and time is the key to successful restoration.

3. Connectivity in four directions



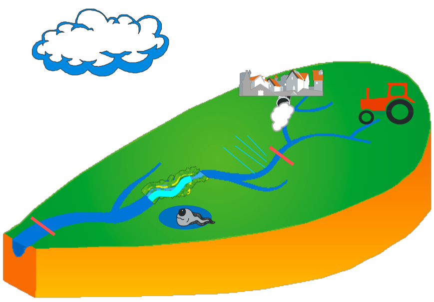
Riverine ecosystems are connected to their catchment in four major directions: longitudinal, lateral, vertical and temporal. Restoring connectivity, especially longitudinal connectivity, has been a major restoration objective. Restoring lateral connectivity will re-establish the role of the riparian zone as a critical transition zone between rivers and their catchments and as the centre of high productivity, biochemical processes and biodiversity. Restoring the vertical connection re-introduces refugia in the hyporheic zone and provides both temperature regulation through shade, allochtonous carbon input as primary food source for the river community and habitat for the aerial life stages. Restoring the temporal connectivity provides time for cyclic and developmental processes to evolve.

4. The role of biological interactions and processes



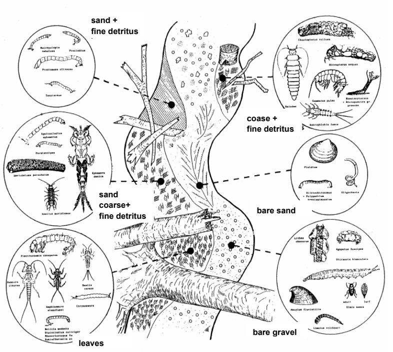
Knowledge of the species' life history traits, habitat templates and spatio-temporal ranges is crucial to set the environmental demands for restoration. More often dispersal is a critical process in restoring viable populations, thus one should check for the presence of source or donor populations of targeted species to inform practitioners about the recolonisation potential of restored sites. Recovery of biological processes needs time, often decades, which should be taken into account when evaluating restoration progress.

5. The effect of multiple stressors



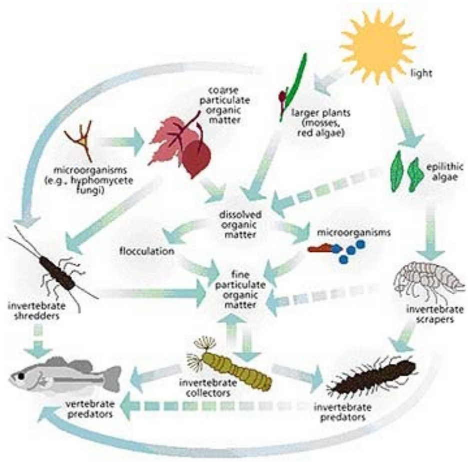
River restoration success is expressed as the rehabilitation of riverine ecosystems. Apart from dealing with site or stretch-specific stressors, one has also to check for adverse large-scale stressors upstream. These stressors can 'spoil' restoration, for catchment scale stressors overrule local stressors.

6. The role of habitat heterogeneity



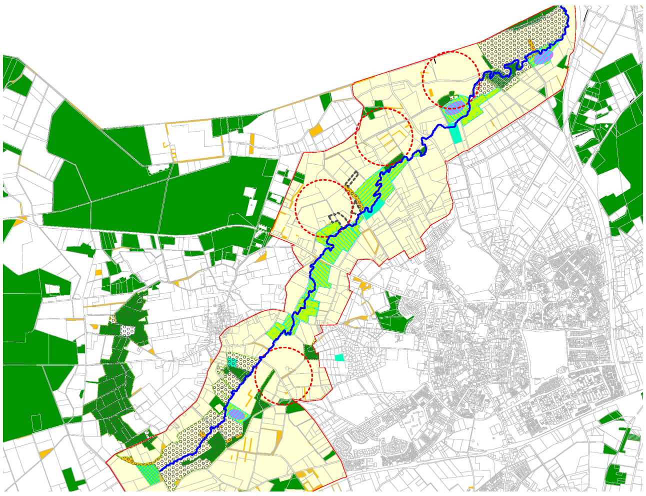
River organisms survive disturbances by using refugia. A heterogeneous and diverse river bottom and a well-structured river bank offer a variety of habitats where organisms can find shelter, food and suited substrates. In restoring refugia the re-colonisation and establishment of target species populations can be fostered. Refugia facilitate resistance and resilience features to remaining anthropogenic disturbance regimes even after restoration.

7. River restoration must consider ecosystem processes.



The cost-effectiveness and predictability of river restoration will improve with an increased understanding of the processes by which lotic ecosystems develop. One of the key processes is the increase in ecosystem functioning and thus improved ecosystem services when biodiversity increases. Another important component which until now got little attention are the food-web interactions. For example, nutrient stress is often underestimated in its effects on stream ecosystems but are easily recognised in the food-web structure.

8. Identify restoration priority areas



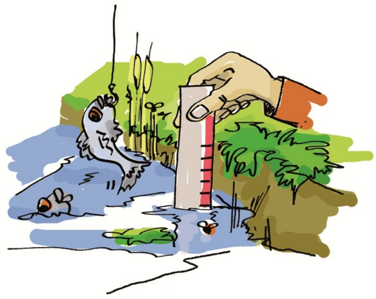
River restoration must be coordinated at the River Basin Scale. Not every river stretch can be restored in a cost-effective way. Land and water uses can be of great economic importance and continue to slow down or block recovery. Chances for cost-effective restoration must be evaluated from a catchment perspective. Areas with high potential for successes and low cost must prevail. Thus the selection of restoration sites is a coordinated top-down process.

9. Communicate

River restoration needs consensus among a variety of stakeholders including the inhabitants of the floodplain as well as the (sub-) catchment. Because large-scale restoration is infeasible without broad stakeholder support, future opportunities to advance restoration depend on future demand for the ecosystem amenities that restoration can potentially provide.

Therefore a crucial educational role for managers to communicate the many ways in which functional ecosystems enhance stakeholders' quality of life, as well as how delivery of those benefits or river ecosystem services is impaired when rivers are degraded.

10. Tailor-made monitoring



Restoration projects need to define clear, well-defined goals or endpoints. Furthermore, the succession pathways and processes by which these endpoints can be achieved must also be seriously considered and preferably as much as possible quantified. Restoration needs a before-after-control-impact (BACI) design of monitoring with site or measure specific, tailor-made indicators capable of indicating both short-term changes towards recovery and long-term recovery of the riverine assemblages. Furthermore, monitoring must involve several BQEs depending on the stressors and monitoring should address all stressors, as knowledge about them is crucial.

SYNTHESIS

Translation of results to specifically address the practitioners

Sensitivity of BQEs to pressures

Biological Quality Elements (BQEs) respond differently, depending on the type of human pressure (Figure 1). Macro-invertebrate and diatom metrics appear to be the most sensitive to global degradation. Fish metrics generally show a strong (= medium to high intensity) but 'late' (low to medium sensitivity) response to pressures (Figure 1). By contrast, macrophyte metrics reveal the least intense, but comparatively sensitive response to pressures, in particular to morphological degradation. However, a common hierarchy pattern stands out for the four BQEs (Figure 1). Global and water quality degradations of the river appear to be better detected by BQE metrics than morphological and hydrological degradations (stronger responses and higher sensitivity). Moreover, although numerous metrics respond only to high levels of human-induced degradations, in particular the functional trait-based metrics are most sensitive and respond generally to lower levels of pressure.

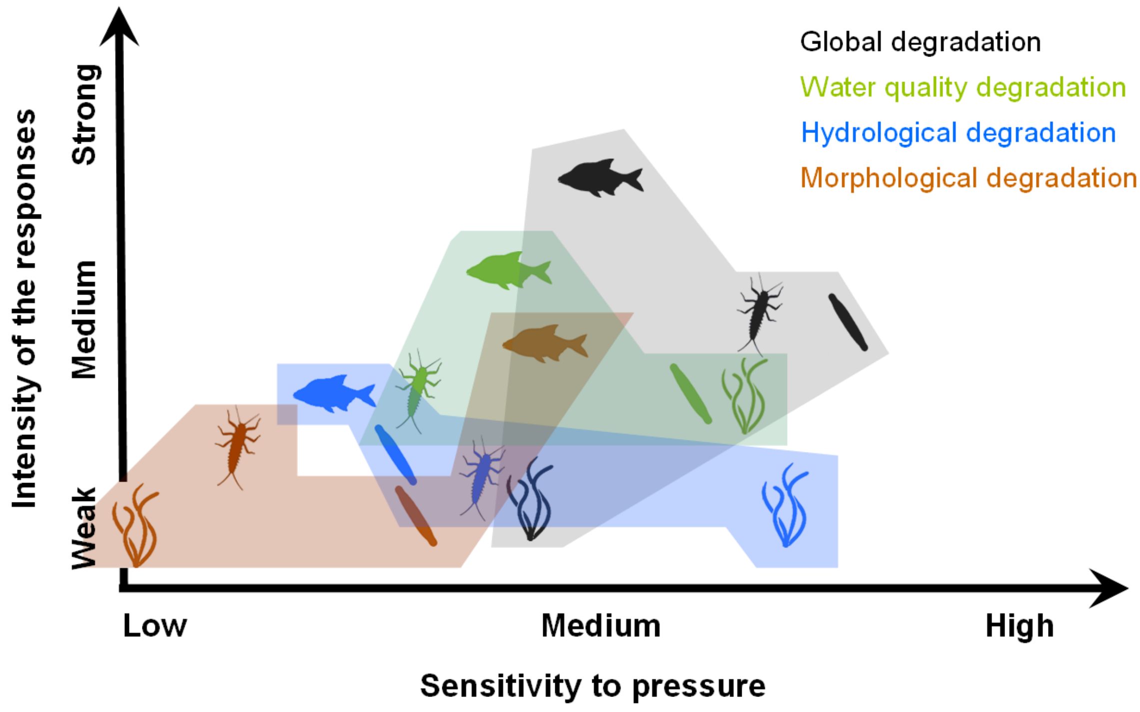


Figure 1. Mean response sensitivity and intensity of fish, macro-invertebrate, macrophyte and diatom metrics to human-induced pressures after removing the effect of physiographical factors (e.g. climate, geology, and geomorphology).

The WISER studies clearly support the use of ecological and biological functional trait metrics to build multi-metric indexes in order to assess river biotic integrity. Consequently, knowledge on river assemblage's traits needs to be improved, particularly for macrophytes. In addition, according to our findings, the four BQEs bring complementary information on the river ecological status and, therefore the use of multiple biological groups is considered appropriate for monitoring programs. Finally, the common response pattern imply that water quality problems are still notable and need to be solved before morphological or hydrological restorations can have the desired effects.

Conceptual framework of potential effects of restoration

Human impact on aquatic ecosystems has been subject to numerous studies that aimed to develop and test equally numerous indicators to assess and monitor the various environmental impacts on aquatic assemblages. This knowledge about the linkages between environmental stressors and aquatic community characteristics is now used to derive appropriate measures of restoration in order to initiate ecological recovery. Restoration ecology often assumes that communities start to recover as soon as the stressors are reduced or removed. However, the simple reversal of degradation alone often does not show the desired and anticipated ecological effects. The biota continues to stay 'degraded'.

Conceptual models for rivers illustrate the relationships between restoration measures, their effects on environmental key variables and, finally, the responses of benthic algae, macrophytes, benthic invertebrates and fish (Figure 2). Such conceptual frameworks were developed and tested in WISER in order to review the scientific evidence of three common river restoration measures: i) instalment of riparian buffers to improve water and habitat quality, ii) placement of in-river structures to improve the mesohabitat, and iii) removal of weirs to restore connectivity, hydrology and geomorphology.

Overall, riparian buffer instalment can be considered an appropriate restoration measure to reduce fine sediment entry, and nutrient and pesticide inflow from riparian and floodplain areas (Feld et al. 2011a). Thereby, buffer width and length strongly determine the retentiveness; 5–30 m width are recommended in the literature (e.g. Wenger 1999), while buffer length should be at least one to several kilometres (Feld et al. 2011b). Among instream mesohabitat enhancement, the introduction of large woody debris (logs, tree trunks), boulders and gravel are the most common single measures, yet the often detectable and immediate enhancement of habitat quality was equally often found to be superimposed by large-scale geomorphological and physico-chemical effects from further upriver (e.g. fine sediment entries due to row-crop agriculture, flood peaks due to impervious areas in urban landscapes). Restoration studies that reported long-term biological recovery after habitat enhancement were missing. In contrast, weir removal is reported to immediately enhance sediment and flow diversity upriver, and to re-establish the longitudinal connectivity (Bednarek 2001).

Biological recovery, then, however, might lag behind for several years, as often vast amounts of fine sediment have accumulated upriver of the former barrier, which are transported downriver after weir removal. This deposition of fines has temporarily adverse effects on habitat availability for macroinvertebrates and fish.

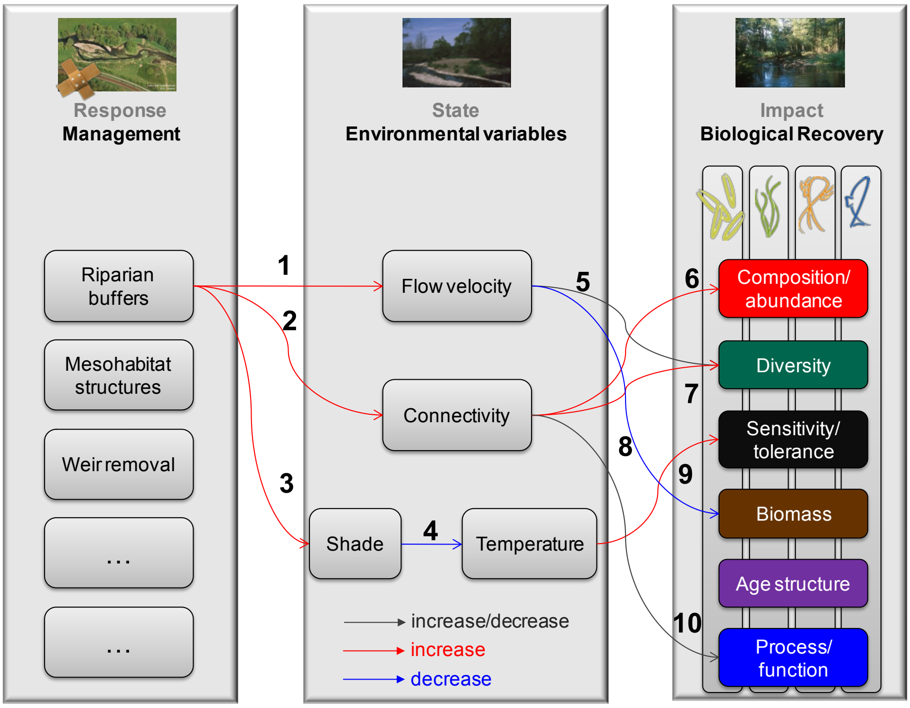


Figure 1: General conceptual framework to analyse abiotic effects (= states) and biological recovery in the course of restoration (= society's response to degradation). The linkages between restoration and recovery are mostly of indirect nature (i.e. via changing environmental states), but direct linkages may occur, for instance, with bio-manipulation. Each link in the conceptual framework should refer to evidence reported in the restoration literature. The relationship may either be positive (red arrows), negative (blue arrows) or ambiguous (black arrow). The recovery characteristics in the framework on the right hand side refer to those defined by the WFD (Annex V), amended by the group of biological measures of processes and functions. For further explanation, see Feld et al. (2011a).

In conclusion, conceptual modelling can help structure the state-of-the-art of restoration and, thereby, assists the identification of both suited environmental variables to track abiotic changes and applicable biological indicators to track ecological recovery after restoration. Its application to three common restoration measures also revealed three major key factors, whose neglect may easily become a drawback in an attempt to design river restoration ecologically successful also in the long term.

1. Degradation acts at different spatial scales, thus restoration must account for all scales that impose impact on riverine ecology. Local restoration is insufficient response to catchment-wide degradation.
2. Recovery after restoration may take decades and is rarely detectable in the course of scientific graduation studies (e.g. during 3–4 year PhD projects). Restoration monitoring requires long-term efforts in order to track its changes. This knowledge can become the key to the design of successful restoration schemes in the future.

3. Ecological functions and processes are important characteristics of river integrity, but rarely considered in restoration. Ecological malfunction may inhibit recovery and requires consideration in restoration monitoring.

Actual effects of restoration

Due to the remarkable lack of restoration monitoring data, empirical analysis of actual effects of river basin restoration on lotic assemblages was restricted to about 47 restored river reaches within the frame of WISER (all sites located in Germany). In addition, actual effects were derived from a review of restoration studies (Feld et al. 2011a).

With regard to riparian buffer instalment, there is evidence for the effectiveness of mixed riparian buffer strips (trees, shrubs, grasses) in the retention of fine sediments and adhering colloidal phosphorous through surface runoff. Riparian trees can also effectively reduce nitrogen in the upper groundwater layer. More detailed specifications on the minimum width and lengths required for effective buffer function vary in the literature, yet there is consensus about a minimum width of 30 m, while the length should be of several kilometres (small to medium-sized rivers) up to 10 km (larger rivers).

In-river habitat enhancement by additions of woody debris, boulders, spawning gravel and other substrates can lead to immediate habitat improvements for fish (e.g. gravel as spawning habitat) and benthic invertebrates (e.g. wood as foraging habitat). However, there is clear evidence for the ineffectiveness of local habitat enhancement, especially when it does not fit the regional geomorphic and wider landscape context. Excessive fine sediment loads from agriculture upriver or flood peak discharges induced by large amounts of impervious (urban) areas above a restoration stretch can easily render habitat additions nonsense, if substrates are being flushed even by small floods and deposited further downriver or if spawning gravels get lost under layers of fine sediment.

Weir-removal, in contrast, constitutes a restoration measure that immediately and sustainably restores the longitudinal connectivity. Formerly impounded (i.e. lentic or even stagnant) reaches immediately switch back to flowing rivers. The side effect, however, is that immediately after removal of a weir, the vast amounts of fine sediments that have been accumulated in front of a weir get mobilised, too. This can be considered a severe disturbance and may impose adverse effects on river ecology in the segment downriver, which may last a decade or even longer until full recovery.

For the evaluation of the effects of restoration the choice of indicators is crucial. Biological indicators based on the four BQEs are scale-dependent and respond stressor-specific. Hence, evaluations can be different with regard to the targeted BQE (compare Figure 1). The knowledge about the long-term effects of restoration including biological recovery, however, is very limited due to the lack of such long-term restoration monitoring data. This shortcoming largely restricts the testing of the conceptual framework, as was aimed by

WISER. Empirical data were available only for measures of habitat improvement, yet which often come along with the removal of artificial substrates (e.g. rip-rap, sheet piles) or with modifications of the river course (e.g. re-meandering). A comparison of biological and environmental effects of restoration at 47 pairs of (restored and unrestored control) sites supported the dominant impact of broad-scale (catchment-wide) stressors. Percent area as agriculture in the catchment above the restoration as well as the overall physical habitat conditions in the river segment up to 10 km above the restoration were significantly correlated with the ecological status of fish, benthic invertebrates and macrophytes at the restored reaches. In contrast, only little effects were attributable to the restoration measures itself. It should be stressed, however, that this relationship also holds true for favourable conditions upstream, which promote restoration and ecological recovery.

In summary, there is sufficient evidence for the chief role of broad-scale stressors that may act at the scale of entire catchments and control environmental conditions at finer spatial scales. Consequently, local restoration is considered ineffective as long as broad-scale stressors continue to impact a site or reach. Furthermore, indicators of actual effects differ dependent on scale and pressure.

Scale and hierarchy in degradation processes

Eleven years after the implementation of the Water Framework Directive (WFD), there is tremendous evidence on the effects of different types of degradation (i.e. environmental stressors) on riverine assemblages. Numerous studies have statistically related lotic community characteristics of all biological quality elements to pollution, eutrophication, land use, hydrological and morphological degradation, acidification and other stressors (e.g. Furse et al. 2006, Pont et al. 2007). WISER supports the findings of previous research projects on rivers, and puts particular emphasis on two important aspects of degradation, i.e. the matter of spatial scaling of stressors and its hierarchical linkage.

Natural environmental factors as well as stressors can act at different spatial scales, from the local (site) to the entire catchment scale (mostly upriver of a site). Catchment-scale geomorphic settings determine substrate particle size and thus control the local habitat of the river network in the catchment. Runoff from upriver adjacent agricultural land will result in siltation of the substrate mosaics at sites downriver. In turn, local habitat composition or degradation affects local fish or invertebrate communities that are somehow related to or dependent on the either or not modified habitat. So, the broader scale can determine the conditions at the finer scale.

This hierarchical scaling of the environment is crucial in bioassessment. If environmental stress is imposed by broad-scale stressors, such as (agricultural) land use, its effects can be dominant over large river stretches. Intensive forms of (row-crop) agriculture often involve the application of pesticides and fertilizers, which then enter the river networks through surface, drainage and groundwater infiltration (Allan 2004). Monocultures can destroy the soil texture and lead to excessive entries of fine sediments, while they do not leave space for the development of riparian vegetation to buffer these impacts. Groundwater management to maximize agricultural production led to canalisation and regulation of rivers. It is quite obvious that agriculture has led to severe modifications of entire watersheds in many regions of Europe (see CORINE 2000 land cover, EEA Data Service: http://www.eea.europa.eu/themes/landuse/interactive/clc-download)). The designation of many lowland rivers as heavily modified water bodies, for example in the Netherlands and Germany, shows the dominant role of agricultural activities in European river degradation. All of this is likely to adversely affect the lotic communities.

Empirical analysis in WISER confirmed the leading role of agriculture as a catchment-wide stressor that influences all BQEs and often significantly reduces the ecological quality (see key messages below for detailed results). Agriculture, together with its main detectable effect of eutrophication, was found to be the chief stressor variable using river monitoring data from Austria, France, Germany and the Netherlands. A correlation of assemblage metrics with agriculture would support its adverse effects on the water and habitat quality but the complexity of interactions prevents clear direct links. So, in general agricultural land use cannot be used as proxy for river degradation. More in detail, stressors from a larger scale have a stronger impact on BQEs than local stressors. The highest impact is linked to agriculture in the catchment, meaning eutrophication and alkalinisation (indicated by diatoms), while the main local stressor is agriculture along the river leading to structural degradation (indicated by fish and diatoms).

Indicators of degradation and restoration

The use of indicators in river assessment and monitoring schemes is framed by the WFD, which in case of river ecosystems requires the use of fish, benthic invertebrates, aquatic macrophytes and benthic algae. Accordingly, measures of richness, abundance, sensitivity/tolerance, age structure (only fish) and biomass (only benthic algae) are being used in European monitoring schemes.

Biological indicators often require fastidious and time-consuming field sampling and lab preparation. However, they are applicable in standard monitoring schemes and have been proven numerous times to respond predictably to the impacts of water quality perturbations, morphological and hydrological disturbances, the presence of impoundments and the overall (global) degradation of river systems.

Based on 93 metrics tested in the frame of WISER (all BQEs, Marzin et al. submitted; see also Feld et al. 2011b). Multimetric indices in general were independent from the natural variability, while single richness, abundance and diversity metrics turned out to be sensitive to (natural) longitudinal gradients.

The same set of BQEs and indicator characteristics is often being applied to the assessment and monitoring of restoration, most likely in the course of regular monitoring events within the six-year cycle of a River Basin Management Plan (RBMP). This approach can lead to the conclusion that, for example, river (habitat) restoration does not increase the richness of benthic invertebrates, as reviewed by Palmer et al. (2009). Yet, although recovery after restoration should be mirrored and, thus, detectable by regular monitoring indicators, the application of the same set of indicators is flawed for two reasons:

- First, monitoring indicators aim to assess various sources of degradation, which include long-term (legacy) degradation, e.g. through land use, deforestation or drainage. It is unlikely that the same indicators detect changes due to restoration in the presence of long-term and broad-scale stressors that continue to impact the restored reach, even less in the short term.
- Second, ecological recovery requires the presence of donor populations of the targeted BQE(s) in the catchment that can serve as sources for recolonisation. If such donors lack for a species, the species cannot recolonise a restored river reach; recovery then is limited or even impossible.

Therefore, supplementary indicators are required, capable of tracking not only an ecological status after restoration, but also more incremental changes towards recovery (Matthews et al. 2010). This includes the application of environmental indicators capable of tracking water quality and habitat changes due to restoration, which can be considered a prerequisite for recolonisation. Eventually, indicators are required to track changes also in the short-term. Short-term indication of restoration effects is going to provide the basis for adaptive management, but in turn required a short-term application, i.e. annual or bi-annual monitoring events within the first decade after restoration.

Ecological and environmental thresholds

Ecological thresholds refer to change points or transition zones along a stressor gradients at which a dramatic change of biological characteristics (e.g. assemblage metric values, ecological quality ratios) is detectable. Hence, such thresholds may mark a critical value of the stressor variable, or ranges thereof, yet they may not be obvious for all stressors and biological responses, respectively.

Using land use variables and biological assemblage metrics from 500 river reaches in France and Germany, notable differences were detectable between mountain and lowland ecoregions that were consistent among BQEs. For catchment agriculture, many assemblage metrics were found to change at 5–15% in mountain, and at 15–75% in lowland ecoregions. Hence, in particular the assemblages in mountain rivers and rivers turned out to be rather prone to the impacts from arable (crop) land. In contrast, a minimum catchment forest cover of 45–55% in mountain and 15–25% in lowland systems was necessary to keep assemblage metrics close to 'natural' values. Similar trends were detectable for near-river agriculture in mountains, but not in lowlands and neither for near-river forest cover in both ecoregion types.

Effect of climate change (temperature) and global change

Using species distribution models calibrated with a data set covering 15 European countries, we were able to project the future distribution of 23 fish species. These projections showed that fish distributions will be greatly modified by climate change. Coldwater species, such as salmonids will experience massive extinctions of local populations. Grayling (*Thymallus thymallus*), brown trout (*Salmo trutta*) and sculpin (Cottus gobio) are expected to go extinct in the whole Seine basin (France), as revealed by a detailed case study analysis of the Seine river. These predictions are supported by a case study analysis of data from the Traun River (Austria). This study demonstrated the shift of fish assemblage composition during the last three decades, because of increasing water temperatures. Historically dominated by grayling, which strongly declined, species with warmer thermal tolerances (e.g. barbel) increased and have replaced grayling.

On contrary, warm water species will clearly "benefit" from climate change as revealed by the projected maps of their distribution in 2050-2060. These species are expected to expand their distributions to locations that were climatically unsuitable before. This is the case for bitterling (*Rhodeus amarus*) and bleak (*Alburnus alburnus*). For other species with intermediate thermal tolerances, the response patterns are more contrasted and we expect mostly a shift of their distributions both at the watershed and distribution area levels.

By relating the variability of functional metric (e.g. number of intolerant species) to the environmental conditions using statistical models, we were also able to predict the effect of climate change on the functional fish assemblage structure. Several metrics commonly used to assess ecological status in lotic systems turned out to be notably affected by global warming, for instance, oxygen-depletion intolerant and habitat-demanding species.

These shifts clearly indicated that over the period of the Water Framework Directive implementation, it will be necessary to revise the multimetric indices based on functional traits, which are commonly used now, such as for instance the European Fish Index. Some of the metrics included in these indices would not be represented in future fish assemblages. This is especially the case of metrics based on species intolerance which are largely used due to their responses to human degradation. Therefore the computation of index scores will be done on metrics that will become naturally absent or only slightly represented, leading to inconsistent assessment of river ecological conditions. Moreover, of the wish to intercalibrate indices is maintained over the Water Framework Directive, it would be also necessary to revise the common index used for this process. Indeed, this index is a combination of metrics based on species intolerance, which are expected to be strongly affected by climate change.

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KEY MESSAGES FOR POLICY MAKERS

Riverine assemblages respond differently to individual stressors and stress levels

Key message

Fish, benthic invertebrates, macrophytes and benthic diatoms are differently affected by environmental degradation. While hydraulic alterations, for instance, may impose a strong negative impact on fish assemblages, this may be less strong for macrophytes: the **intensity** of responses varies across riverine assemblages and environmental stressors. In selected cases response to stress may even be positive; hence the **sign** of response varies too. Eventually, the stress levels at which a response can be detected vary notably and reveal a dissimilar **sensitivity** of assemblages to stress (Figure 1).

Evidence

There is empirical evidence that river biota are almost always sensitive to general degradation (mixture of non-distinguishable stressors), land cover and water quality degradation *(low uncertainty)*, as opposed to hydrological and morphological degradation which affects could be less reproduced (link to Table 1). The response of fish to agricultural land use in the catchment, for instance, depends on the spatial scales considered for the calculation of percent land cover (link to Figure 2).

Diatoms and macroinvertebrates respond most strongly to general degradation already at low stress levels. This renders both organism groups weak indicators of local habitat improvement in degraded catchments, i.e. both groups are unlikely react to restoration unless broad-scale impacts are being remedied. Besides general and water quality degradation, fish and macroinvertebrates respond most intensively to morphological degradation, structural modification and catchment land use. Fish respond strongly to hydrological degradation, too. Hence, river fauna reveals a more intense, but not necessarily more sensitive, responses to stress, compared to the flora. Overall, aquatic macrophytes were found to be comparatively weak indicators of the stressors considered.

Implication

Assessment and monitoring systems must account for the different capabilities of river biota in the detection and indication of single and multiple stressors. If multiple stressors act in a catchment, the use of a single assemblage only is likely to be insufficient and may lead to the wrong conclusions regarding the appropriateness of management or restoration measures.

River Basin Management must address and reduce *all* stressors relevant for ecological status. In agricultural or otherwise widely degraded watersheds, the impact of fertilizer and pesticide application, soil degradation and runoff modification is often omnipresent and can easily superimpose other, rather local impacts of structural and habitat degradation. Consequently, *any* local restoration in agricultural catchments must account for such large-scale impacts upriver of a restored site to initiate biotic recovery.

Further reading

In depth analysis of empirical data is available through WISER Deliverable 5.1-2. The conceptual linkages of environmental variables and riverine biota is available through WISER Deliverable 5.1-1 and Feld et al. (2011)

(http://www.sciencedirect.com/science/article/pii/B9780123747945000031).

The conceptual models of linkages can be accessed and used interactively at http://www.wiser.eu/programme-and-results/management-and-restoration/conceptual-models/.

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Paul, M.J., and Meyer, J.L. (2001). Rivers in the urban landscape. Annu. Rev. Ecol. Syst. 32, 333–365.

Table 1. Intensity and sensitivity of BQE's (riverine assemblages) response to different stressor groups.

BQE		general degradation	physico-chemical	hydrological	morphological	land use
Diatoms	Intensity	high	medium	low	low	medium
	Sensitivity	high	high	low	medium	high
Macrophytes	Intensity	low	medium	medium	low	low
	Sensitivity	medium	high	low	low	low
Benthic Invertebrates	Intensity	high	medium	low	medium	medium
	Sensitivity	high	medium	low	low	medium
Fish	Intensity	high	high	medium	high	high
	Sensitivity	medium	medium	medium	medium	low

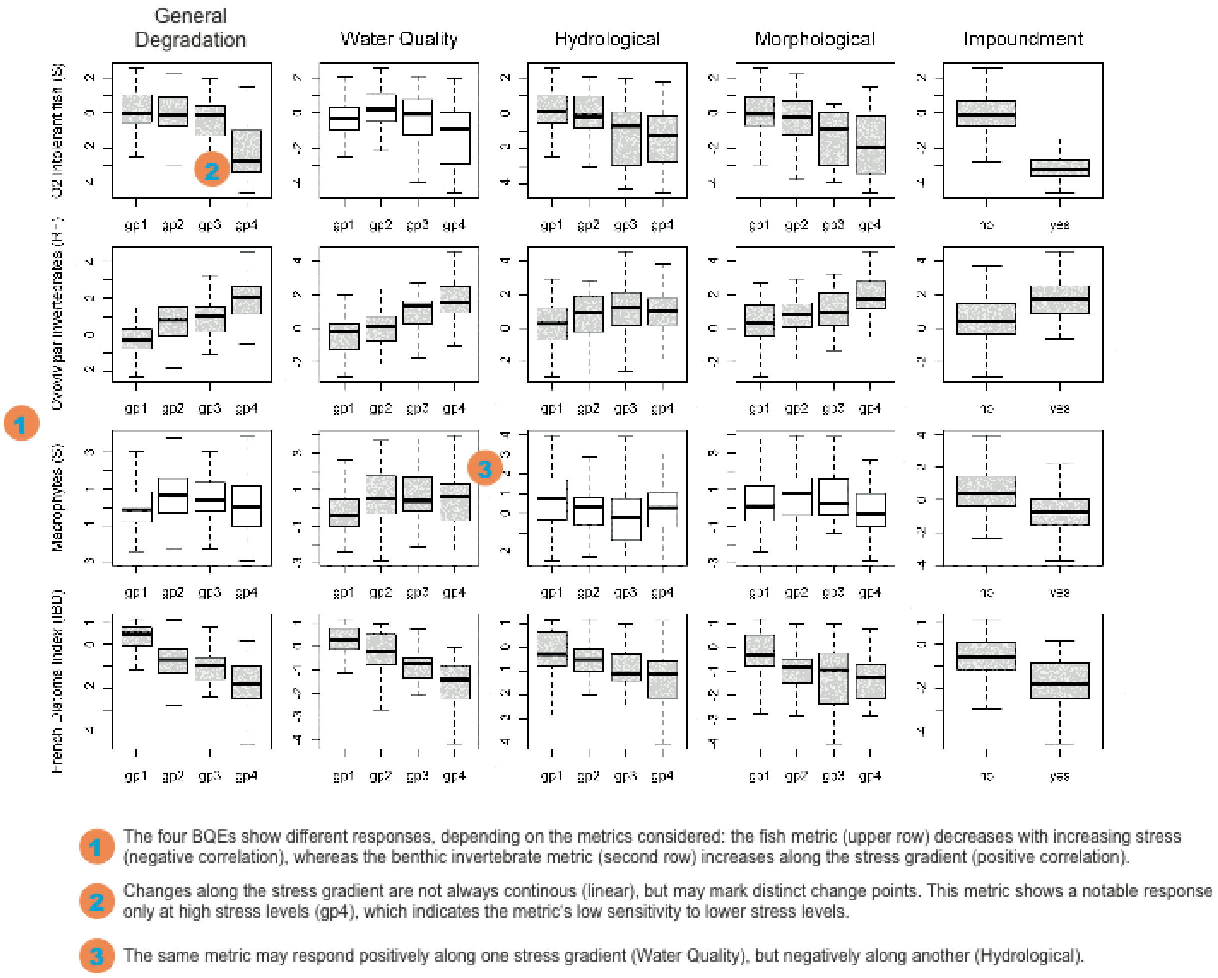

Figure 1. Response patterns of four selected metrics along five stress gradients. The metrics represent fish, benthic invertebrates, macrophytes and benthic diatoms at ca. 290 stations in France. General degradation is a mixture of all other single stressors. Stress increases from left (gp1) to right (gp4) in all plots.

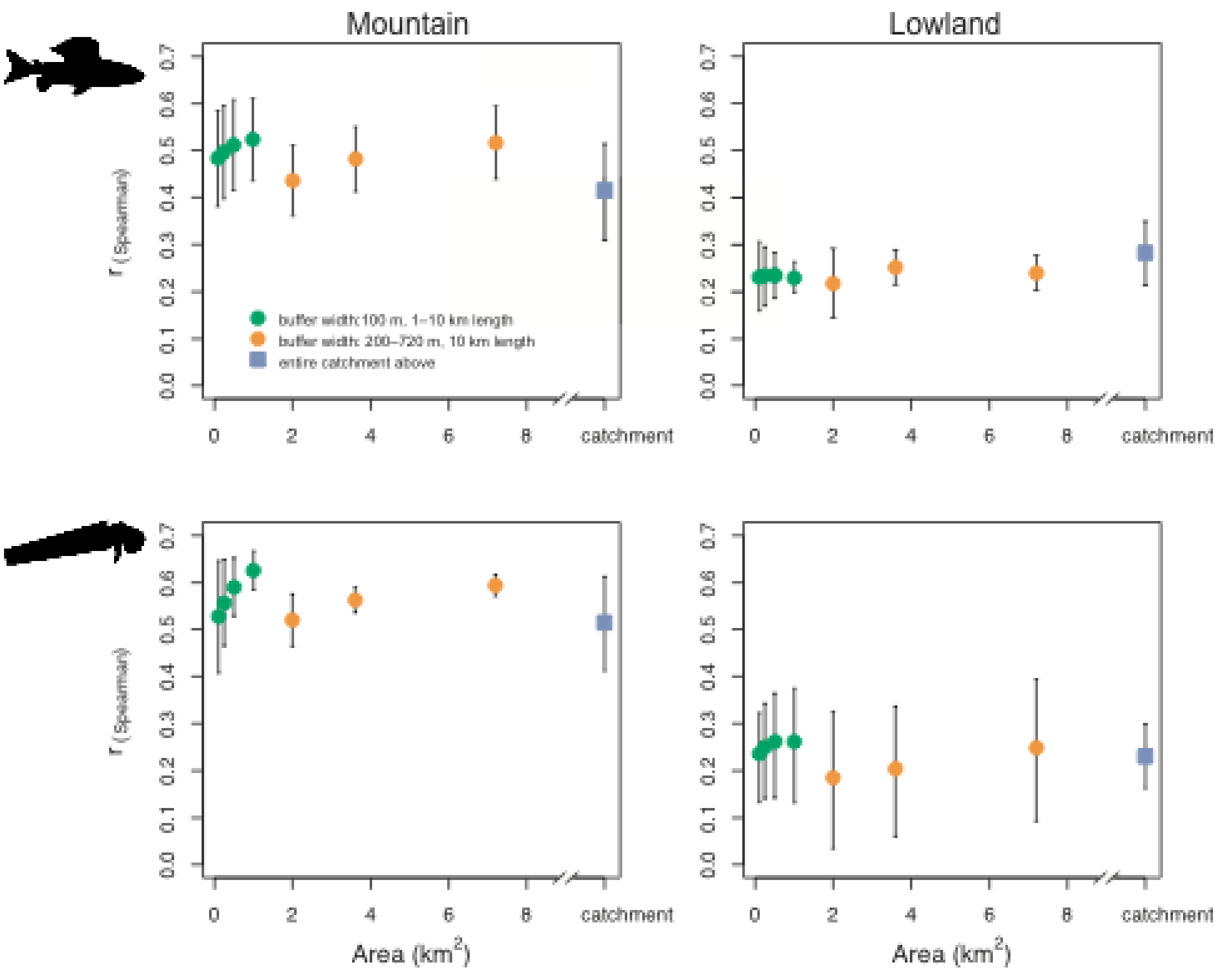

Figure 2. Correlation between the proportion of pollution intolerant fish (upper row), the number of Ephemeroptera-Plecoptera-Trichoptera taxa (benthic invertebrates, lower row) and % area as agriculture or forest (absolute mean ± SD) in different buffers upriver. The analysis was based on 500 sampling stations in France and Germany.

Environmental stressors act hierarchically

Key message

Landscapes and riverscapes are organised hierarchically (Table 2). For example, catchment geology and geomorphic features determine substrate particle quality and size, whereas current velocity is a function of river slope and discharge. Hence, broad-scale landscape factors can largely control local habitat conditions. This hierarchical relationship also applies to stressors, such as catchment agriculture or urbanisation, both of which can largely determine segment-, reach- and habitat-scale water quantity and quality.

Besides these spatial hierarchies, there is also a qualitative hierarchy of stressors, for instance, water quality problems can superimpose hydrological and morphological conditions.

Evidence

There is a tremendous body of literature, which has been reviewed, summarised and illustrated by Paul and Meyer (2001), Allan (2004) and Feld et al. (2011). In brief, catchment-wide (often referred to as watershed-scale) urbanisation can severely degrade river hydrology and lead to enhanced frequencies and strengths of peak discharges (*low uncertainty*). The percentage as impervious area in the catchment is a good proxy measure to estimate the degree of degradation. The percentage of agriculture in the catchment is a suitable indicator of nutrient and fine sediment pollution, both of which severely degrade riverine habitats, down to the estuaries and coastal waters (*low uncertainty*).

Further empirical evidence shows that broad-scale agriculture and its direct impacts (e.g. eutrophication) are superior over stretch- to local-scale stressors (e.g. structural degradation; Table 3). Percent of agriculture in the catchment (and in riparian buffers along river stretches) and eutrophication revealed the strongest relationships to compositional and functional metrics of all tested assemblages (*low uncertainty*).

In another context, hierarchy refers to the strength of one stressor over another, irrespective of the spatial scale considered. Water quality deterioration (pollution, eutrophication) always overrules the impacts of hydrological and morphological degradation (Feld et al. 2011) (*low uncertainty*). In part, this is supported by empirical results presented in Table 3. Agriculture (through fertiliser application and erosion) can lead to eutrophication, which directly affects the water quality and was found to be strongly related to all tested BQEs, in particular the percentage of arable (crop) land in the entire watershed above a site.

Implication

The hierarchical relationship between broad-scale stressors and reach-scale habitat conditions implies that restoration at the local scale is unlikely to initiate ecological recovery unless broad-scale impacts upriver are managed in parallel. As catchment-wide restoration is unrealistic within the time-scale of the WFD, practitioners plan measures bottom-up. Smart and adaptive concepts can help design measures so that multiple local measures can synergistically combine to ecological recovery at the larger segment scale (compare Table 2). Thereby, river basin management must not neglect water quality problems when they are obvious; they may continue to impact rivers even after extensive hydromorphological improvements.

Table 2. Spatial scales and their extents in riverscapes (colour coding according to Table 2.2).

Spatial scale (classification)	Longitudinal spatial extent [m]
Microhabitat (local)	10^{-1}–10^{0}
Habitat (or site, local)	10^{0}–10^{1}
Reach (or stretch, local)	10^{1}–10^{2}
Segment (or buffer, intermediate)	10^{2}–10^{3}
River network (broad)	10^{3}–10^{4}
Catchment (watershed, broad)	10^{4}–10^{5}

Table 3. Ranking (hierarchy) of stressor impacts on fish (285 stations), benthic invertebrates (227) and benthic diatoms (85), based on data from Austria. Broad-scale (sub-/catchment) stressors are marked green, fine-scale stressors (local/reach/riparian buffer scale) orange.

Rank order	Fish	Benthic Invertebrates	Benthic Diatoms
1	Catchment arable	Catchment arable	Catchment arable
2	Eutrophication	Eutrophication	Alkalisation
3	Catchment urban/fabric	Buffer arable	Eutrophication
4	Buffer agriculture	Alkalisation	Habitat structure
5	Barriers up-/downriver	Catchment heterogeneous agriculture	Barrier upriver
6	Habitat structure	Buffer urban fabric	Barrier downriver
7	Catchment heterogeneous agriculture	Buffer heterogeneous agriculture	Catchment heterogeneous agriculture
8	Buffer urban and heterogeneous agriculture	Habitat structure	Riparian vegetation modified
9	Alkalisation	Catchment urban fabric	Buffer heterogeneous agriculture
10	Buffer urban fabric	Riparian vegetation modified	Buffer urban fabric
11	Riparian vegetation modified	Barrier downriver	Catchment urban fabric
12	Barrier upriver	Barrier upriver	Buffer arable

Further reading

Detailed results can be derived from the sections by Marzin et al. and Dahm et al. in WISER's Deliverable 5.1-2. The importance of land use for structuring riverscapes is reviewed in Paul and Meyer (2001) and Allan (2004).

Allan, J.D. (2004). Landscapes and riverscapes: The Influence of Land Use on River Ecosystems. Annu. Rev. Ecol. Evol. Syst. 35, 257–284.

Feld, C.K., Birk, S., Bradley, D.C., Hering, D., Kail, J., Marzin, A., Melcher, A., Nemitz, D., Petersen, M.L., Pletterbauer, F., Pont, D., Verdonschot, P.F.M. & Friberg, N. (2011) From natural to degraded rivers and back again: a test of restoration ecology theory and practice. Adv. Ecol. Res. 44, 119–209.

Paul, M.J., and Meyer, J.L. (2001). Rivers in the urban landscape. Annu. Rev. Ecol. Syst. 32, 333–365.

Catchment and riparian land use control local habitat conditions

Key message

The hierarchical order of landscapes and riverscapes implies a hierarchical order of stressors, too. Stressors, such as land use or river regulation, are ubiquitous in large parts of the world because of the multifaceted land and water uses. Flood protection is usually linked to severe modifications of hydrological and morphological (structural) characteristics. Agriculture increasingly dominates entire regions due to society's growing demand for food, resources and energy.

Broad-scale stressors impose serious problems for restoration and recovery. Not only do, for instance, agriculture and urban settlement control habitat conditions at finer scales, but land use impacts have often been present for decades or even centuries in many regions, e.g. in Central and Western Europe. Thus, the legacy of land use past may continue to impact entire river basins or sub-basins as long as such impacts are not being mitigated by appropriate (broad-scale) management schemes.

Evidence

Urban settlement and agriculture in the catchment upriver of a site largely influence and control the physical habitat conditions at the respective site (*low uncertainty*). Urban settlements can influence water retention and storage through the percent as impervious area in the catchment, which in turn affects the hydrograph and can lead to severe flash floods following stormwater release. Less than 10% urban settlements in the catchment are frequently reported to significantly reduce biological and ecological quality (Paul and Meyer 2011).

The major impact pathways of intensive agriculture are nutrient enrichment (eutrophication) and excessive fine sediment entries (habitat loss). While nutrient enrichment can directly affect algal and plant communities, the loss of coarse substrates (pebbles, cobbles and larger stones) affects fishes and invertebrates.

Naturally vegetated riparian buffer strips not only can buffer impacts from agriculture, but also provide habitat (woody debris, leaves), shelter (root wads, shade), food (wood, leaves, terrestrial insects) and energy (carbon and nitrogen) to the riverine assemblages (Allan 2004, Feld et al. 2011, *low uncertainty*).

Aquatic assemblages (e.g. fish and macroinvertebrates) significantly change their structural and functional composition, when the percent area as agriculture upriver exceeds 20% in mountain ecoregions (Figure 3) (*low uncertainty*). Lowland assemblages seem to respond less sharp to agriculture and significantly change values at 30–50% (*medium uncertainty*). These findings are in line with the thresholds reported by previous studies (e.g. Allan 2004).

Near-river buffer areas along several kilometres upriver can help maintain biological diversity and functionality at a site, if a minimum of 40–50% within the buffer area is covered by forest (*medium uncertainty*). Ecological recovery may be promoted already by a minimum of 25%

forested buffers upriver (*high uncertainty*). Yet it is important to note that the increase of forest cover alone is unlikely to mitigate the impacts of land use.

Implication

Intensive agriculture and other land uses characterise large parts of Europe (Figure 4) and constitute potential broad-scale stressors for riverscapes and its ecology. This in particular applies to the agricultural lowlands of Eastern, Central and Western Europe. Without appropriate mitigation and management, the impacts of land uses (e.g. eutrophication, habitat degradation, pollution) are likely to continue to impact rivers and hence hinder recovery, irrespective of hydrological and morphological improvements that may be achieved at the site, reach or segment scale.

Consequently, restoration and river basin management *must* adequately address land use impacts. That is, restoration measures are required that i) are capable of mitigating land use impacts and that ii) address the appropriate scale of impact. Riparian buffers can be considered best practice. For instance, mixed riparian buffer strips (trees, shrubs, grass) have been proven to effectively retain nutrients and fine sediments from adjacent crop fields (see Feld et al. 2011 for a review). Buffer strips require several kilometres of length (segment scale) rather than tens or hundreds of metres (Figure 2).

Eventually, given the omnipresent character of agriculture, it may be the right time to start thinking about a re-organisation of land uses, i.e. future river basin management may involve measures of land use management. Conversion to less intensive land use forms in riparian areas will be most effective. This would require the reorganisation of agricultural policies in parallel.

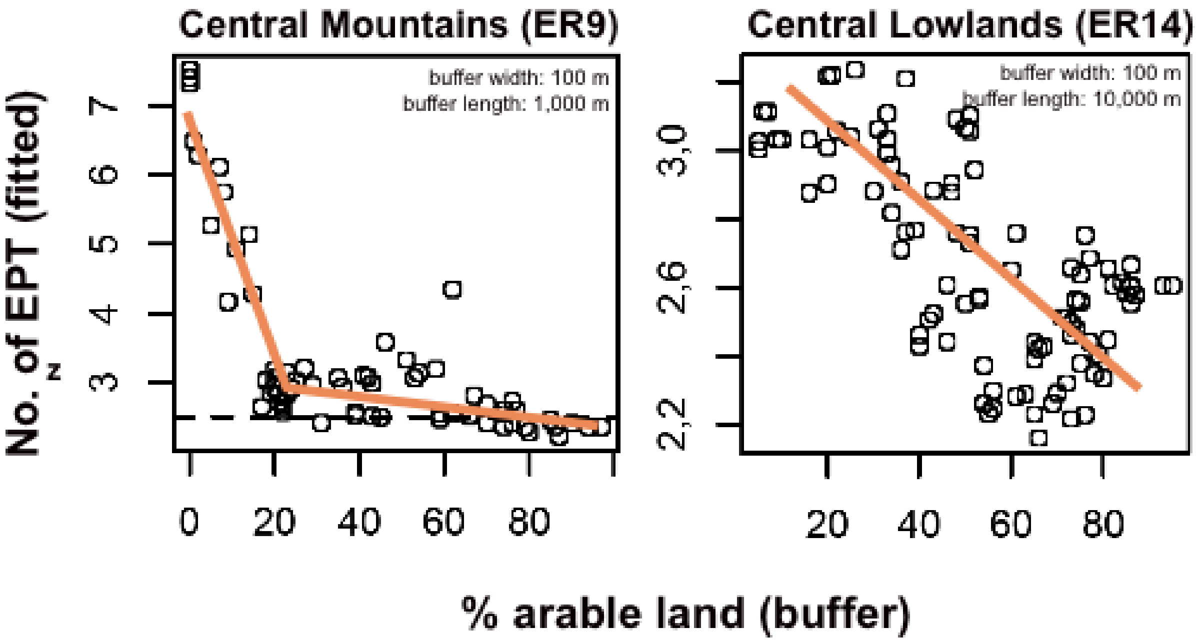

Figure 3. Boosted regression models identified the number of Ephemeroptera-Plecoptera-Trichoptera taxa (No. of EPT) to significantly decrease with increasing arable land in the riparian buffer of mountain rivers. A sharp decrease was obvious between 0 and 20% arable land. This decreasing trend is obvious too, although with less sharp the change, for lowland rivers. Note that the fitted values for EPT richness in lowland rivers mark a short gradient of one taxon difference only. The analysis

was based on ca. 200 German macroinvertebrate samples in ecoregion (ER) 9 and 14. More in-depth results including fish and macrophytes are provided with WISER Deliverable D5.1-2.

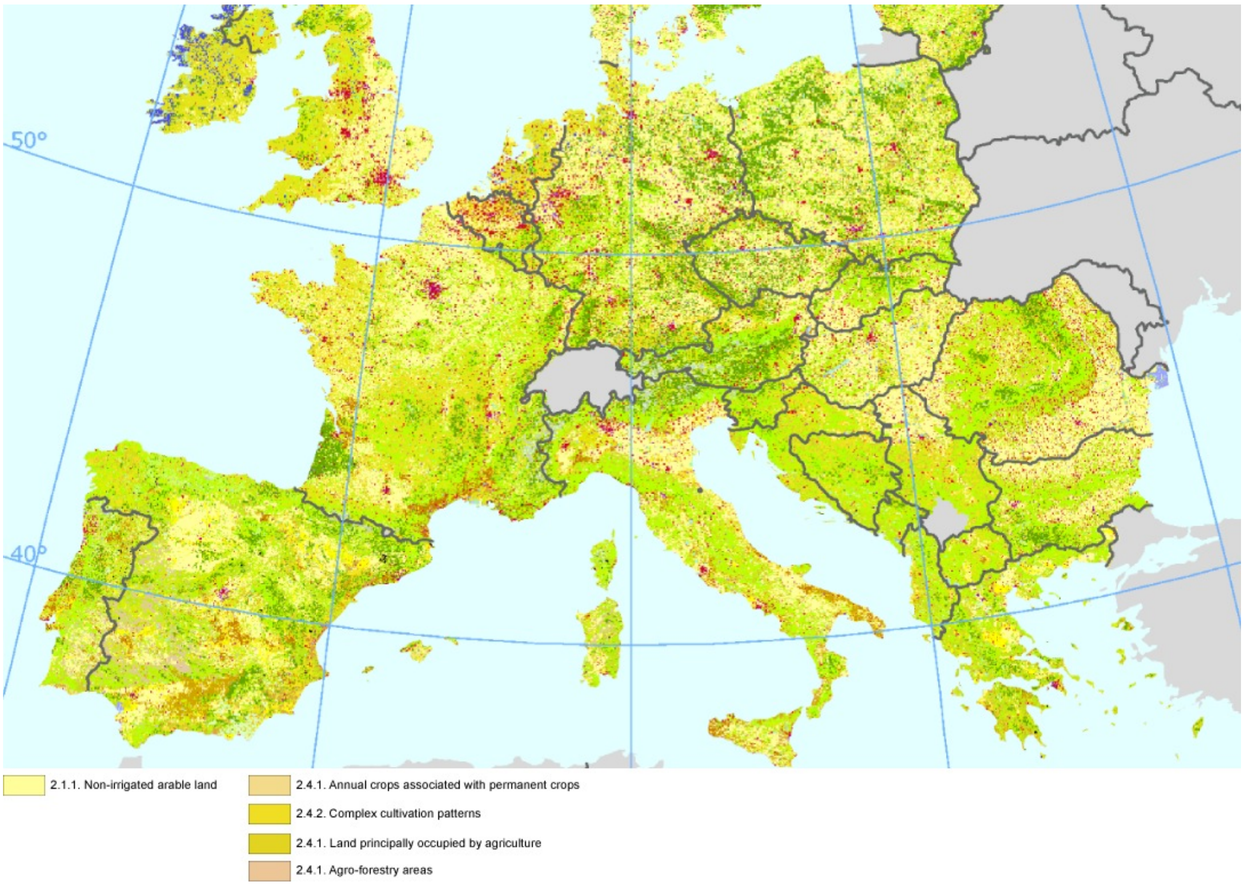

Figure 4. Intensive forms of agriculture (yellowish colour) prevail in large parts of Eastern, Central, Western and Southern Europe, particularly in the lowlands. (Data source: CORINE 2000 land cover, EEA Data Service: http://www.eea.europa.eu/themes/landuse/interactive/clc-download.)

Further reading

Detailed results can be derived from the sections by Feld and Lorenz in WISER's Deliverable 5.1-2. The importance of urban and agricultural land use for structuring riverscapes is reviewed in Paul and Meyer (2001) and Allan (2004). The role of land use for river restoration and its potential prevention of recovery was reviewed by Feld et al. (2011).

Allan, J.D. (2004). Landscapes and riverscapes: The Influence of Land Use on River Ecosystems. Annu. Rev. Ecol. Evol. Syst. 35, 257–284.

Feld, C.K., Birk, S., Bradley, D.C., Hering, D., Kail, J., Marzin, A., Melcher, A., Nemitz, D., Petersen, M.L., Pletterbauer, F., Pont, D., Verdonschot, P.F.M. & Friberg, N. (2011) From natural to degraded rivers and back again: a test of restoration ecology theory and practice. Adv. Ecol. Res. 44, 119–209.

Paul, M.J., and Meyer, J.L. (2001). Rivers in the urban landscape. Annu. Rev. Ecol. Syst. 32, 333–365.

Restoration is more likely to be successful, if upriver physical habitat degradation and land use impacts are low

Key message

Statement 2 and 3 address the predominant role of broad-scale stressors that may act at the scale of entire (sub-) catchments and consequently may impact any site within the catchment.

Consequently, river restoration is more likely to initiate and maintain biological recovery, if such broad-scale impacts are either completely missing or being mitigated in parallel to restoration at the fine (local) scale.

Evidence

There is empirical evidence (*medium uncertainty*) from restoration monitoring that restoration measures can initiate biological recovery, if the physical habitat conditions several kilometres upriver of the restoration are only moderately modified or in better condition. In particular the fish and macrophyte assemblages were found to be strongly influenced by habitat quality up to 10 km upriver (Table 4). Macroinvertebrate ecological quality was related to shorter stretches upriver (up to 2.5 km). Empirical analyses imply that about 1 km length upriver in a moderate or better physical habitat quality might suffice to promote biological recovery (*high uncertainty*) (WISER Deliverable D5.1-2).

Implication

Where broad-scale stressors impact ecological quality after restoration and may hinder recovery, such stressors require mitigation. Practitioners need to know the multiple stressors that may impact restoration candidate sites. They should prioritise those stretches that are least impacted by broad-scale stressors and thus may constitute stepping-stones within a broader restoration scheme. Rather local restoration measures need to be integrated into restoration schemes at the broad scale.

This broad-scale and integrated restoration is well referred to by the WFD and termed ‘River Basin Management’. Yet, it seems as if this broad-scale approach deserves more attention by scientists and practitioners in order to better use the limited resources available for river restoration and management.

Table 4. Spearman rank correlation and significance levels of the relationship between ecological quality ratios (EQRs) of three BQEs at unrestored and restored sites and the physical habitat quality in several distances upriver of the sites (N = number of valid cases; significant correlations in bold). The correlations reveal a notable relationship of fish EQRs with physical habitat conditions up to 10 km upriver (maximum values at 2.5–5 km upriver) of the sampled river sites. Macrophytes showed a similar relationship up to 7.5 km upriver, while the relationship with invertebrate EQRs was significant up to 2.5 km upriver only.

	Fish		Invertebrates		Macrophytes	
Distance upriver	Unrestored	Restored	Unrestored	Restored	Unrestored	Restored
500 m	**-0.37** N=32 p=0.035	**-0.44** N=34 p=0.010	**-0.36** N=33 p=0.038	**-0.50** N=35 p=0.002	-0.27 N=34 p=0.128	**-0.49** N=35 p=0.003
1,000 m	**-0.35** N=32 p=0.048	**-0.41** N=34 p=0.002	**-0.38** N=33 p=0.027	**-0.42** N=35 p=0.013	-0.25 N=34 p=0.150	**-0.46** N=35 p=0.005
2,500 m	**-0.51** N=32 p=0.003	**-0.52** N=34 p=0.002	**-0.45** N=33 p=0.008	**-0.40** N=35 p=0.017	-0.32 N=34 p=0.068	**-0.54** N=35 p=0.001
5,000 m	**-0.47** N=32 0.007	**-0.51** N=34 p=0.002	-0.32 N=33 p=0.071	-0.31 N=35 p=0.066	**-0.37** N=34 p=0.034	**-0.45** N=35 p=0.006
7,500 m	**-0.47** N=32 p=0.007	**-0.42** N=34 p=0.014	-0.23 N=33 p=0.208	-0.24 N=35 p=0.165	**-0.36** N=34 p=0.036	**-0.38** N=35 0.023
10,000 m	**-0.50** N=32 0.004	**-0.35** N=34 p=0.043	-0.22 N=33 p=0.229	-0.25 N=35 p=0.147	-0.33 N=34 p=0.060	-0.29 N=35 p=0.089

Further reading

For a detailed analysis of the effects of upriver physical habitat quality and land use conditions on ecological quality assessment at restored and unrestored sites see Lorenz in WISER's Deliverable D5.1-2.

Local restoration is often unsuccessful

Key message

Local restoration refers to the scale of single river sites or reaches, i.e. the scale of several tens up to hundreds of metres of river length (Table 2). This fine scale is typical for habitat enhancement (e.g. wood, boulder or gravel addition) or the removal of bank and bed fixation structures, re-meandering or re-braiding.

Although such fine-scale measures typically result in quantifiable improvements of habitat quality and diversity, they rarely result in notable improvements of the ecological quality. Neither structural community characteristics (e.g. richness and diversity measures) nor functional attributes (e.g. feeding types) show general improvements after habitat enhancement or other morphological restoration.

Furthermore, fine-scale habitat enhancement is often found to be 'spoiled' by natural dynamic processes, such as excessive erosion and sedimentation after floods, which easily can reset habitat conditions back to the state prior to restoration. This is the case when hydrological and morphological restoration targets do not fit the landscape characteristics (e.g. discharge dynamics, sediment type, slope).

Evidence

Restoration at the fine scale often failed to improve ecological quality (*low uncertainty*) (e.g., Palmer et al. 2010, Feld et al. 2011). There is less evidence for the underlying reasons. Frequently, it is assumed that local habitat improvement does not address broad-scale stressors and thus cannot initiate ecological recovery as (*high uncertainty*). This assumption may sound trivial, yet remains untested.

'Larger', more extensive restoration measures, which in parallel improve the in-river habitat heterogeneity and overall channel patterns at the segment scale, are more likely to improve fish, macroinvertebrate and macrophyte assemblages at the same time (WISER Deliverable D5.1-2).

Methodological drawbacks may render habitat enhancements ineffective (see Feld et al. 2011 for example references). Wood additions, for instance, were found to be destroyed, buried under fine sediments or removed and transported further downriver even after moderate flood events (5 years recurrence interval). Spawning gravel additions were rapidly covered by fine sediment layers and made them useless for gravel-spawning fish (e.g. salmon, trout, nase).

Implication

River restoration practice is a story of both success and failure. The manifold examples of restoration failure studies suggest that the design and planning of local habitat enhancement and other fine-scale restoration requires careful consideration of broader landscape characteristics (Figure 5). Not only must practitioners consider the stressors in the catchment upriver, but also the geomorphic (natural) landscape features, such as precipitation, geology and slope. The discharge dynamics, for instance, will largely control riverine processes such as erosion and deposition. Excessive fine sediment loads originating from agricultural land use on the floodplain will inevitable affect the availability of coarse substrates on the river bottom.

A step forward to overcome these broad-scale constraints from natural landscape settings and human landscape modifications might be to better adapt restoration to these constraints. Individual measures should be designed 'broader', probably at the scale of river segments of several kilometres or even tens of kilometres of lengths. This might be supported by rather technical mitigation measures such as the fixation of large pieces of wood with wire or the trapping of fine sediments at specifically designed (wide and flat) river sections above. Where landscape characteristics potentially disturb specific habitat restoration measures, such measures should not be implemented, unless the mitigation of adverse landscape effects is appropriately addressed.

Further reading

Feld, C.K., Birk, S., Bradley, D.C., Hering, D., Kail, J., Marzin, A., Melcher, A., Nemitz, D., Petersen, M.L., Pletterbauer, F., Pont, D., Verdonschot, P.F.M. & Friberg, N. (2011) From natural to degraded rivers and back again: a test of restoration ecology theory and practice. Adv. Ecol. Res. 44, 119–209.

Palmer, M.A., Menninger, H., & Bernhardt, E.S. (2010) River restoration, habitat heterogeneity and biodiversity: A failure of theory or practice? Freshwat. Biol. 55 (Suppl. 1), 205–222.

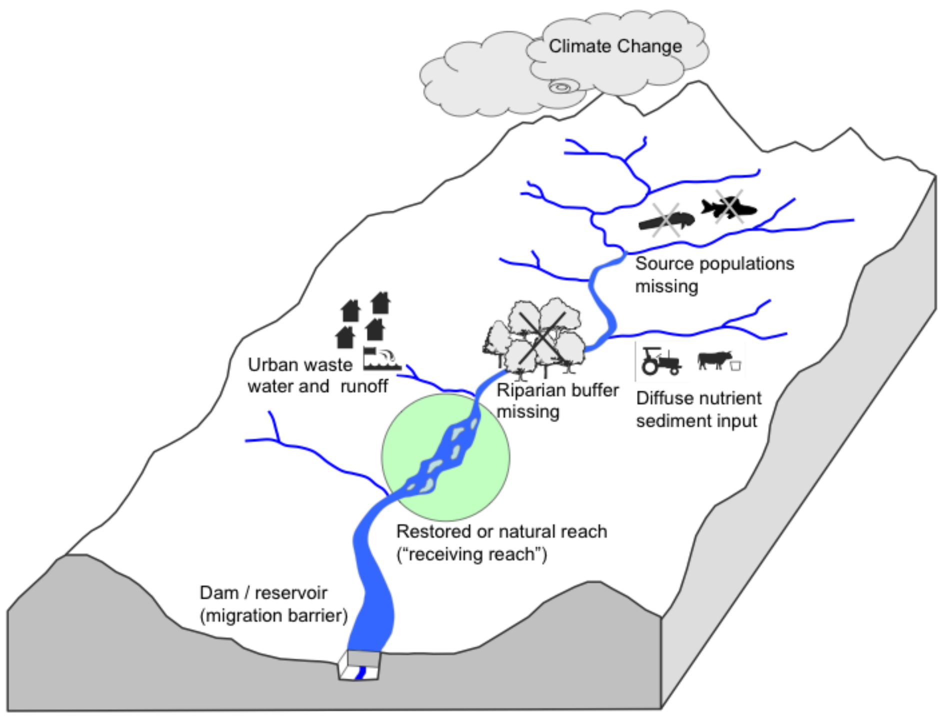

Figure 5. Multiple stressors may act upriver and downriver of a restoration candidate river site or stretch and hence induce numerous impacts on the site (green circle). Rather local restoration (e.g. re-braiding) is unlikely to initiate ecological recovery unless the stressors further above in the catchment are being addressed and their impact mitigated. The same applies to impacts further downriver, such as dams, weirs and other migration barriers, which may inhibit the colonisation of restored sites by fish and invertebrates. (Source: http://www.impact.igb-berlin.de/research-program/basic-idea, by courtesy of Jochem Kail, IGB Berlin, Germany)

River Basin Management Plans insufficiently account for research and monitoring demands

Key message

The assessment and monitoring of the ecological status of rivers and other surface waters is explicitly referred to in the WFD and hence constitutes a basis for all River Basin Management Plans (RBMPs). River Basin Managers and practitioners are informed about the stressors to be assessed, the BQEs to be used for monitoring and the frequency of monitoring events with regard to each individual BQE.

In contrast, the monitoring of restoration and management measures is neither specifically referred to in the WFD, nor is it sufficiently defined elsewhere. The general approach to date is to apply operational monitoring to assess restoration effects. Changes due to restoration often remain dubious as practitioners miss to sample and record the ecological status of a restoration candidate prior to the implementation of measures. Consequently, the knowledge about the specific requirements of restoration measures that determine restoration success or failure is humble due to the lack of appropriate restoration monitoring schemes

Evidence

The lack of appropriate monitoring schemes is obvious (*low uncertainty*). A review of 160 restoration studies revealed two major shortcomings (Feld et al. 2011): First, restoration monitoring is often poorly designed and hence inappropriate to reliably assign any detected change (or non-change) to restoration. And second, the status before restoration is rarely being monitored, while the monitoring duration is limited to 3–4 years: Thus, long-term effects (>7–10 years) of restoration remain unknown for the majority of studies (Feld et al. 2011).

The lack of restoration monitoring is likely to continue within the first management period of the WFD (until 2015) (*medium uncertainty*). This, in part, becomes evident from the selection of River Basin Management Plans analysed for WISER Deliverable D5.1-2 (see Verdonschot et al. therein). Although the selection represents only a small part of Europe, the considered RBMPs concordantly prove that little attention has been assigned to additional research and monitoring until 2015. Moreover, the RBMPs imply that practical restoration is primarily planned for the second and third monitoring period (i.e. until 2021 and 2027, respectively), which means that the existing knowledge gaps with regard to the reasons for success and failure of restoration remain presumably persist.

Implication

River Basin Management involves huge efforts for and investments in restoration and mitigation measures in the future, presumably for the next couple of decades. As these investments in the environment compete with other society's demands, it is necessary that any bit of these investments is being spent efficiently.

However, the ongoing lack of appropriate restoration monitoring schemes hinders the detection of effects. Consequently, practitioners do not know whether a specific measure is going to support ecological recovery

Sufficiently simple, but 'smart' monitoring designs might help scientists and practitioners fill the knowledge gaps (compare Feld et al. 2011 and WISER's Deliverable D5.1-2). First, as a minimum requirement, at least one sampling event prior to the implementation of measures is required to define the ecological status *before* restoration. Furthermore, an unrestored river stretch close (upriver) to the restored section is required as *control* in order to be able to detect the degree of temporal variability within the river system, which may superimpose the effects purely assignable to restoration. The full design is called *BACI* (before-after-control-impact) and can be considered the method of choice in restoration monitoring (Feld et al. 2011). Second, in addition to the WFD assessment and monitoring tools, more thorough records of hydrological, morphological and biological changes after restoration are required to better detect the multiple effects of individual restoration measures as well as their interactions. And third, restoration monitoring must help inform practitioners about both the short- and long-term changes after restoration. This will help better design sustainable restoration measures.

Further reading

The comparison of selected RBMPs in Austria, France, Germany and the Netherlands is presented by Verdonschot et al. in WISER's Deliverable 5.1-2.

Feld, C.K., Birk, S., Bradley, D.C., Hering, D., Kail, J., Marzin, A., Melcher, A., Nemitz, D., Petersen, M.L., Pletterbauer, F., Pont, D., Verdonschot, P.F.M. & Friberg, N. (2011) From natural to degraded rivers and back again: a test of restoration ecology theory and practice. Adv. Ecol. Res. 44, 119–209.

Climate change alters fish assemblage structure and function distribution in Europe

Key message

Species distributions are driven by environmental conditions, be it natural landscape settings or environmental stress induced by human activities including Climate Change. The Intergovernmental Panel on Climate Change predicted changes in temperature and precipitation in Europe for the periods 2020–2030 and 2050–2060. These changes are expected to greatly alter the distribution of fish, by providing more suitable habitats for species tolerating or preferring warm water, and by restricting species adapted to cold water habitats; the latter are expected to decline or even go extinct some regions of Europe. As these changes may also affect fish assemblage metrics in use at present for assessment and monitoring purposes, this implies that the reference condition baselines use to assess the ecological status of rivers based on fish would not be adequate in the future.

Evidence

Empirical evidence of th.ese changes was shown by the study conducted downstream of lake outlet flow in the Traun river. During the last three decades the water temperature increase by on average 2.2 °C in August. This increase lead to unsuitable thermal conditions for the grayling (*Thymallus thymallus*) which was historically present in this area. Consequently the grayling population greatly decline in favour of more adapted species such as barbel (*Barbus barbus*, Figure 6).

Depending of the individual species considered, the accuracy of species distribution models (SDMs) may be very variable (Figure 7). In general, the models on species with a narrow and distinct temperature niche, i.e. both cold water and warm water-adapted species (e.g. bleak, *Alburnus alburnus*) are more accurate.

In lowland catchments (e.g. the Seine basin in France), the absence of possible thermal refugia in the upstream part of the catchment may amplify the risk of regional species extinctions (Figure 8).

Implication

Climate Change effects have to be taken into account in River Basin Management, for instance when using reference conditions as baselines for assessment or when designing restoration measures. If salmonid species, for example, go extinct in particular catchments, this requires consideration when setting the biological assessment reference in that catchment, or when defining the biological goals for restoration. Without consideration of Climate Change impacts, assessment runs the risk of misclassification. To evaluate such potential shifts, a monitoring network of reference sites in Europe may help inform the practitioners about potential consequences of global warming and its effects on both the biota and its abiotic environment.

Further reading

The climate change effects on fish BQE (species and metrics) is presented by Logez et al. in WISER's deliverable 5.1-3.

Logez, M., Bady, P. and Pont, D. (2011), Modelling the habitat requirement of riverine fish species at the European scale: sensitivity to temperature and precipitation and associated uncertainty. Ecology of Freshwater Fish. doi: 10.1111/j.1600-0633.2011.00545.x.

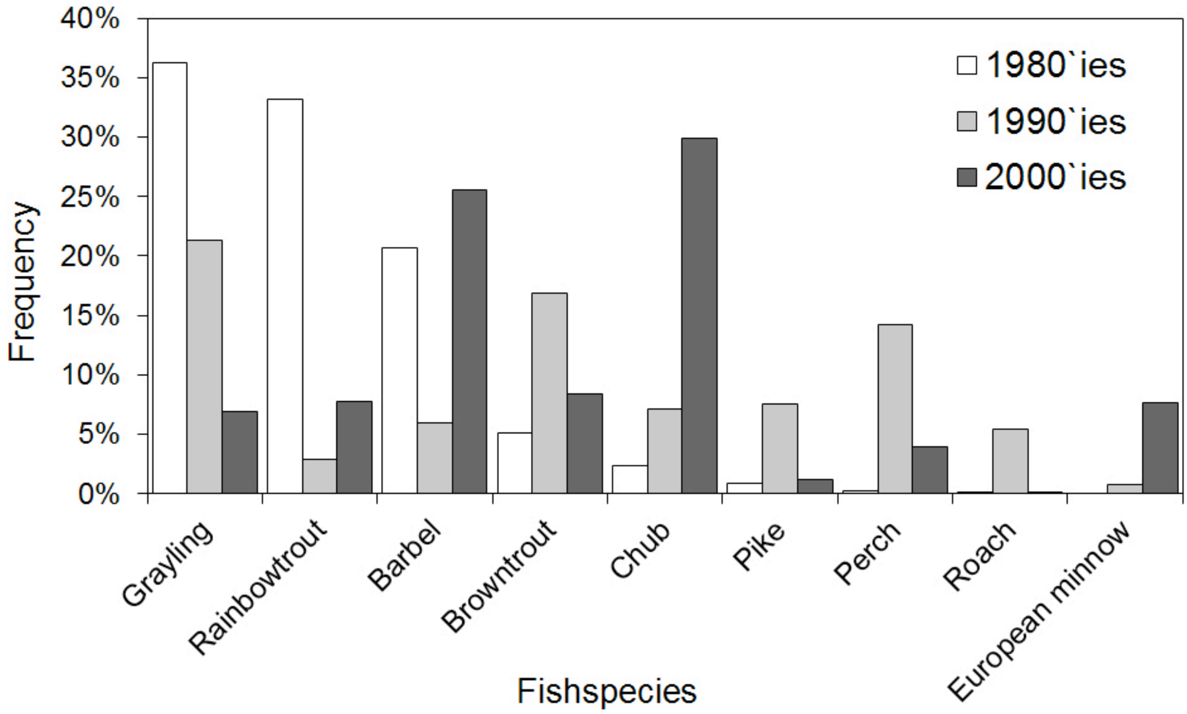

Figure 6. Shift of species composition from the 1980'ies until the 2000'ies in the River Traun in relation with an increase of water temperatures (on average +2.2°C).

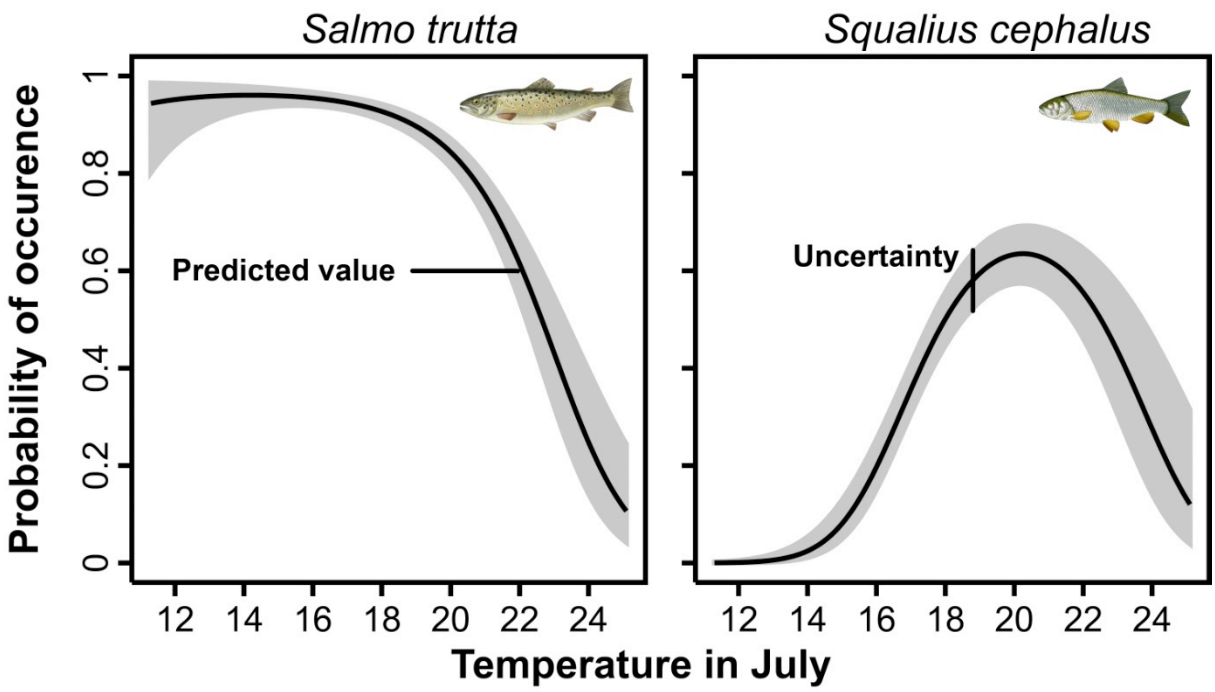

Figure7. Marginal effect of mean air temperature in July on species probability of occurrence of two fish species, brown trout and chud, predicted with species distribution models (Logez et al. 2011). The black curve represents the predicted values and the prediction confidence bands are in grey. These representations could have been obtained by fixing the other environmental values (stream power, thermal amplitude between July and January, upstream drainage area) to their median.

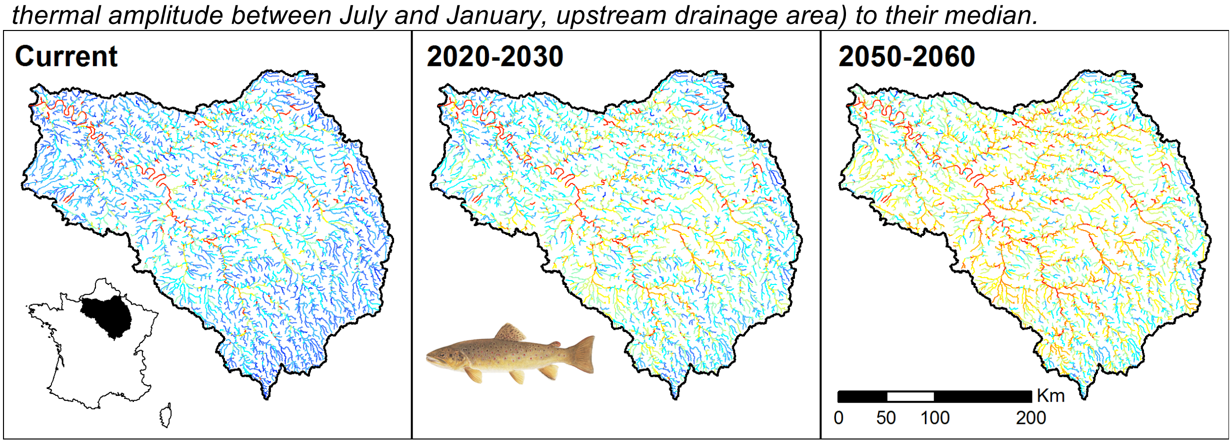

Figure 8. Probability of presence of the brown trout (Salmo trutta, L.) in the Seine river basin (France) derived from the species distribution models (Logez et al. 2011) for the (a) the current environment conditions, (b) projected climatic conditions for 2020-2030 and (c) for the projected climatic conditions for 2050-2060. Probabilities are computed for each stream reach of the CCM2 network (probabilities: — 0-0.1, — 0.1-0.2, — 0.2-0.3, — 0.3-0.4, — 0.4-0.5, — 0.5-0.6, — 0.6-0.7, — 0.7-0.8, — 0.8-0.9, — 0.9-1).

Projections of European fish distribution under Climate Change implies the loss of cold water-adapted species

Key message

The Intergovernmental Panel on Climate Change (IPCC) predicted that temperatures and precipitations in Europe abnormally change in the future; predictions were made for the periods 2020–2030 and 2050–2060. These changes are expected to largely alter the distribution of lotic fish species.

Evidence

The use of four climatic scenarios to project the distribution of 23 widespread fish species (Logez et al. in WISER’s D5.1-3) using species distribution models (Logez et al. 2011) revealed important changes in species distribution due to global warming. Numerous local populations, for instance, of brown trout (*Salmo trutta*) and grayling (*Thymallus thymallus*) are predicted to go extinct as early as 2050–2060 (Figure 9) due to the projected increases in water temperature. Implication

Implications

Current ecological status assessment systems may require future adaptations, for instance, a revision of reference conditions due to the climate-induced shift of baselines. Also, climate change may interfere with the goals of restoration schemes, especially where restoration targets the recovery of cold water-adapted species.

Further reading

The climate change effects on fish BQE (species and metrics) is presented by Logez et al. in WISER's deliverable 5.1-3 (link to D5.1-3).

Logez, M., Bady, P. and Pont, D. (2011), Modelling the habitat requirement of riverine fish species at the European scale: sensitivity to temperature and precipitation and associated uncertainty. Ecology of Freshwater Fish. doi: 10.1111/j.1600-0633.2011.00545.x.

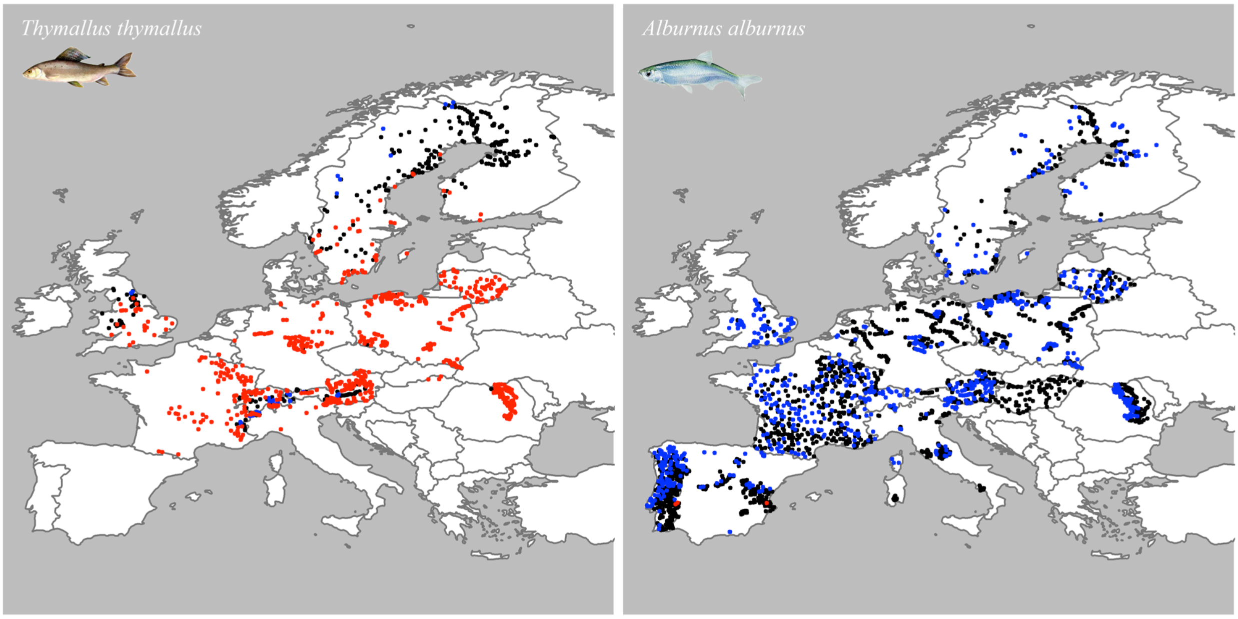

Figure 9. Projected distributions of grayling (left plot) and bleak (right plot) in the period 2050–2060, based on predictive modelling of French monitoring data. Black dots represent unchanged suitable conditions (compared to current climatic conditions), blue dots represent locations with climatic conditions becoming suitable, and red dots locations with climatic conditions becoming unsuitable by the target period.