Ecological Indicators 72 (2017) 131–141

Contents lists available at ScienceDirect

Ecological Indicators

journal homepage: www.elsevier.com/locate/ecolind

Integrating and extending ecological river assessment: Concept and test with two restoration projects

Amael Paillex[a,*], Nele Schuwirth[a], Armin W. Lorenz[b], Kathrin Januschke[b], Armin Peter[c], Peter Reichert[a]

[a] Eawag, Swiss Federal Institute of Aquatic Science and Technology, Department of Systems Analysis, Integrated Assessment and Modelling, Ueberlandstrasse 133, CH-8600 Duebendorf, Switzerland
[b] University of Duisburg-Essen, Faculty of Biology, Department of Aquatic Ecology, Universitaetsstrasse 5, D-45141 Essen, Germany
[c] Eawag, Swiss Federal Institute of Aquatic Science and Technology, Department of Fish Ecology and Evolution, Seestrasse 79, CH-6047 Kastanienbaum, Switzerland

ARTICLE INFO

Article history:
Received 30 July 2015
Received in revised form 16 July 2016
Accepted 26 July 2016
Available online 13 August 2016

Keywords:
Aquatic macroinvertebrates
Aquatic vegetation
Ecological state
Fish
Ground beetles
Multi-criteria decision analysis
Multi-attribute value theory
Thur and Töss rivers
Riparian vegetation
River restoration
Uncertainty

ABSTRACT

While the number of river restoration projects is increasing, studies on their success or failure relative to expectations are still rare. Only a few decision support methodologies and integrative methods for evaluating the ecological status of rivers are used in river restoration projects, thereby limiting informed management decisions in restoration planning as well as success control. Moreover, studies quantifying river restoration effects are often based on the assessment of a single organism group, and the effects on terrestrial communities are often neglected. In addition, potential effects of water quality or hydrological degradation are often not considered for the evaluation of restoration projects.

We used multi-attribute value theory to re-formulate an existing river assessment protocol and extend it to a more comprehensive, integrated ecological assessment program. We considered habitat conditions, water quality regarding nutrients, micropollutants and heavy metals, and five instream and terrestrial organism groups (fish, benthic invertebrates, aquatic vegetation, ground beetles and riparian vegetation). The physical, chemical and biological states of the rivers were assessed separately and combined to value the overall ecological state.

The assessment procedure was then applied to restored and unrestored sites at two Swiss rivers to test its feasibility in quantifying the effect of river restoration. Uncertainty in observations was taken into account and propagated through the assessment framework to evaluate the significance of differences between the ecological states of restored and unrestored reaches. In the restored sites, we measured a higher width variability of the river, as well as a higher width of the riparian zone and a higher richness of organism groups. According to the ecological assessment, the river morphology and the biological states were significantly better at the restored sites, with the largest differences detected for ground beetles and fish communities, followed by benthic invertebrates and riparian vegetation. The state of the aquatic vegetation was slightly lower at the restored sites. According to our assessment, the presence of invasive plant species counteracted the potential ecological gain. Water quality could be a causal factor contributing to the absence of larger improvements.

Overall, we found significantly better biological and physical states, and integrated ecological states at the restored sites. Even in the absence of comprehensive before-after data, based on the similarity of the reaches before restoration and mechanistic biological knowledge, this can be safely interpreted as a causal consequence of restoration. An integrative perspective across aquatic and riparian organism groups was important to assess the biological effects, because organism groups responded differently to restoration. In addition, the potential deteriorating effect of water quality demonstrates the importance of integrated planning for the reduction of morphological, water quality and hydrological degradation.

1. Introduction

Freshwater biodiversity is threatened worldwide (Vorosmarty et al., 2010), and scenarios for the current century predict a continu-

* Corresponding author.
E-mail address: amael.paillex@swissonline.ch (A. Paillex).

http://dx.doi.org/10.1016/j.ecolind.2016.07.048
1470-160X/

ous loss of biodiversity if human activities remain unchanged (Sala et al., 2000). To stop biodiversity loss in freshwater environments and to improve the ecological state of rivers, an increasing number of river restoration projects are undertaken (Bernhardt et al., 2005). However, the ecological effects of restoration are rarely assessed and only few of such assessments show an improvement (Palmer et al., 2010).

Studies quantifying the ecological state of rivers to assess the effect of river restoration often rely on one or just a few assessed organism groups, while studies using a comprehensive set of indicators are rare (despite the existence of such assessment protocols, see e.g. Hering et al., 2006; Woolsey et al., 2007), and terrestrial communities are often neglected. For this study, we selected five organism groups (fish, aquatic benthic invertebrates, aquatic vegetation, ground beetles, and riparian vegetation) to evaluate the ecological state of the rivers and to quantify the effect of restoration on aquatic and terrestrial communities. These organism groups reflect an important part of the biota living along and within rivers, and they are supposed to react to physical modifications in rivers induced by restoration measures (e.g. Jähnig et al., 2010; Januschke et al., 2011; Lorenz et al., 2012).

Integrative decision support methodologies for river management and for evaluating river restoration projects have been suggested (Beechie et al., 2008; Convertino et al., 2013; Corsair et al., 2009; Reichert et al., 2007, 2015), but are rarely applied in practice due to limited financial resources for monitoring. They rely on assessment protocols to evaluate the ecological state of rivers, which were not primarily designed for restoration (Bundi et al., 2000; Hering et al., 2004). Such protocols have the advantage that they can be applied to any kind of river management and support integrative river management by raising the awareness for other potential deficits than morphological ones, even when applied to restoration. On the other hand, modifications to such protocols may be needed to make them more sensitive to changes induced by restoration (Hering et al., 2006; Woolsey et al., 2007). Furthermore, uncertainty about observed or expected effects should be considered to evaluate the significance of changes.

To address these issues, we developed an integrative assessment framework based on value functions that transform environmental attributes measured in the field to degrees of fulfillment of a comprehensive, hierarchical set of ecological objectives. This framework provides a unified approach to all assessment areas, allowing us to quantify the overall ecological state of a river reach while still resolving assessments at lower hierarchical levels. The lower level objectives may be important for the analysis of causes of a bad state and for the design of measures for its improvement.

This study applies the approach developed for river morphology (Langhans et al., 2013) to a much more comprehensive set of assessment areas. With this approach we explicitly account for uncertainty about measured or predicted attributes of the river ecosystem by propagating this uncertainty to the calculated values. The framework is based on multi-attribute value theory, an important methodology for multi-criteria decision support (Dyer and Sarin, 1979; Eisenführ et al., 2010) that is increasingly used for environmental management (Reichert et al., 2015). Advantages of using this theory are that (i) it is based on axioms of "rational choice" which helps justifying decisions; (ii) it focuses explicitly on the objectives that should be achieved by a management decision (value focused instead of alternative focused thinking, Keeney, 1992); (iii) it is flexible regarding the mathematical formulation of preferences; (iv) it makes it possible to propagate uncertainty and to take risk attitudes into account (by extending value to utility functions, Dyer and Sarin, 1982); and (v) it naturally allows us to combine different assessment areas. In this unified framework, we combined physical, chemical and biological assessments to evaluate the overall ecological state of a river reach.

We addressed three goals in our study: 1) to construct a comprehensive ecological assessment procedure that considers the aquatic and the terrestrial parts of the river ecosystem based on physical, chemical and biological criteria, 2) to provide and apply a framework that considers uncertainty and can easily be integrated into decision support methodologies for river management, and 3) to test the suitability of the suggested approach for quantifying restoration effects on habitat diversity and on aquatic and terrestrial communities for two case studies in Switzerland.

According to ecological theory, habitat diversity promotes species diversity (Jähnig et al., 2009; Palmer et al., 2005, 2010). Thus, in the absence of other limiting factors, such as poor water quality and severe limitations to recolonization processes, we therefore expect that increasing instream habitat diversity by restoration promotes aquatic biodiversity (i.e. fish, benthic invertebrates, aquatic vegetation) (Lepori et al., 2005; Palmer et al., 2010). We also expect that enhanced flooding and morphological dynamics in the river floodplain by restoration increases the heterogeneity of the floodplain habitats and promotes terrestrial biodiversity (i.e. ground beetles, riparian vegetation) (Jähnig et al., 2009). In Europe, natural river corridors and floodplains are rare and threatened due to human alterations (Tockner and Stanford, 2002). Therefore, we expect that restoring alluvial habitats is favourable for threatened species living within and along rivers. However, dispersal limitations may be more severe for these species, preventing an increase in their presence at restored habitats. On the other hand, the creation of empty niches during the restoration process may lead to an increase in the presence of good colonizers, including alien species (Strayer, 2010). For these reasons, we included information about threatened and alien species in our integrative assessment procedure.

2. Material and methods

2.1. Study reaches and field study design

We studied the restored and unrestored reaches of two Swiss rivers (the Thur and Töss Rivers) to test the suitability of the assessment methodology described below for quantifying and valuing the effects of restoration. Both were highly channelized in the middle of the 19th century to increase flood security and to gain agricultural land. The natural river floodplain was disconnected and the rivers were reduced to straight channels. The restoration aimed at improving the hydromorphological conditions, first by lowering the floodplain by removing a layer of fine sediment, and second by removing the embankments to provide more space for the rivers (Pasquale et al., 2011; Schirmer et al., 2014). In both cases new gravel bars built-up (expected to be favorable to riparian vegetation and ground beetles) and the river shifted from a straight channel towards a more naturally braided river (in the Töss river, a short artificial structure was built at the upstream end of the widening to support the development of braided channels) with development of potential secondary channels absent before restoration (expected to be favorable to macroinvertebrates, fish and aquatic vegetation). Within each river, we studied a degraded and a restored reach. The restored reach at the Thur River is 1.5 km long and was restored in 2002. The restored reach at the Töss river is much shorter with 0.2 km length and was restored in 1999 (see Fig. A1 in the Supplementary material). In both cases, the degraded reach was representative of the river in the restored reach before restoration and located upstream (Fig. A1). The restored and degraded reaches were situated close to each other, with a distance of 650 m and 350 m for the Thur and Töss Rivers, respectively. In both rivers, no tributaries or differences in the chemical state occurred between the degraded and restored reaches. With this design, following standards developed for the European REFORM project (Poppe et al., 2012), the major factor expected to lead to differences between the biological communities of the reaches should be the morphological changes induced by restoration measures. For this study, we used data from the REFORM project (Hering et al., 2015; Poppe et al., 2012), from the regional authorities (AWEL, 2012), and from our own field campaigns. For the two rivers studied, we combined datasets for habitat conditions, water quality regarding nutrients, micropollutants and heavy metals in the sediment, as well as data on aquatic and terrestrial communities, leading to a data set with a comprehensive set of indicators.

2.2. Instream and floodplain habitat conditions

We evaluated habitat diversity and the effects of restoration by means of a principal component analysis (PCA) (Vaughan and Ormerod, 2005) using the R-package "ade4". Six variables were calculated to describe habitat types and conditions within the rivers and the floodplains (e.g. Lorenz et al., 2012). The data necessary to calculate the variables were measured in 2012 along ten transects distributed uniformly along the length of the reaches following procedures developed for the REFORM project (Poppe et al., 2012). The variables include the diversity of aquatic habitats within the channel (i.e. Simpson index), the variation of velocity conditions (i.e. ratio of the standard deviation to the mean), the variation of water depth, and the floodplain conditions: the bankfull width, diversity of meso-habitats, and the diversity of habitat composition. The habitat data were visualized along the two first axes of a centered PCA, and the diversity between the data from the different transects was quantified at all sites (i.e. beta diversity). Beta diversity was quantified as the distances of the data from the different transects to the centroid of each site. The significance of beta diversity differences between sites was tested by means of the Wilcoxon rank sum test using the R-package "stats".

2.3. Biological sampling

Five organism groups representative of instream and floodplain communities were sampled in each study reach in accordance with European standards, and following the methods described in Poppe et al. (2012). Below we give a short summary of the methods used within each reach studied and the expected responses to restoration:

1 Ground beetles were sampled in July 2012 with pitfall traps along vegetated areas, and by hand on open gravel bars. Ground beetles are good indicators for river restoration, and are expected to increase in richness with the recreation of gravel bars after widening the river (Jähnig et al., 2009; Januschke et al., 2011).
2 Riparian vegetation was sampled in July 2012 within 36 plots distributed along three out of the ten transects used for habitat description. Riparian vegetation is expected to increase in richness after the restoration of riparian habitats (Jähnig et al., 2009; Januschke et al., 2011).
3 Aquatic invertebrates were sampled in July 2012 within 25 quadrats according to a multi-habitat sampling approach (Poppe et al., 2012). In the absence of other limiting factors (e.g. lack of colonization sources, poor water quality), widening the river is expected to increase habitat diversity and in turn the benthic invertebrate richness (Jähnig et al., 2010).
4 Fish were monitored in September 2014 in accordance with a multi-habitat sampling approach by electrofishing. Fish are sensitive to habitat alterations and restoration increasing instream habitat diversity is expected to lead to higher fish richness in restored reaches (e.g. Pont et al., 2006; Schmutz et al., 2014).
5 Aquatic vegetation (i.e. macrophytes) was sampled in July 2012 by wading along the river banks, and by moving from one bank to the other along the river. Macrophytes are expected to increase in richness after the widening and the creation of more lentic zones (Lorenz et al., 2012).

2.4. Statistical analyses of differences in organism groups

Richness within restored and degraded reaches was calculated for the five organism groups described above. The aquatic invertebrate community was divided into groups of EPT (Ephemeroptera, Plecoptera, Trichoptera) and COH (Coleoptera, Odonata, Heteroptera) species to highlight the effect of restoration on the main species groups inhabiting rivers and requiring different habitat conditions as explained in Gallardo et al. (2014). Aquatic vegetation was divided into bryophytes and hydrophytes. Threatened species were identified as such according to well documented national red lists to account for conservation goals (see Appendix A1 in Supplementary materials). Alien species were identified as such according to a list of alien species established for Europe and Switzerland (DAISIE, 2009; Wittenberg et al., 2005). To estimate the effect of restoration on richness, a Wilcoxon signed rank test was applied between degraded and restored reaches.

2.5. Formulation of assessment protocols as value functions

To assess the ecological state of the river reach, in particular to identify differences between restored and unrestored sites, we translated existing assessment procedures for morphological, chemical and biological conditions (i.e. fish and macroinvertebrates) for Swiss rivers (Bundi et al., 2000) into so-called value functions (Langhans et al., 2013) using the R-packages "utility" (Reichert et al., 2013) and "ecoval" (Schuwirth and Reichert, 2014). These assessment protocols were developed to evaluate the state of the rivers at the scale of a river reach and not for the entire river, with the length of the studied river reach adjusted to the width of the river. With this method, the objective of reaching a good ecological state of a river reach is broken down into a hierarchy of sub-objectives, each covering a relevant aspect of the corresponding higher level objective (Reichert et al., 2015). The degrees of fulfillment of the lowest level objectives are then formulated as so-called value functions of observable attributes of the river. In general, a value function quantifies the degree of fulfillment of an objective measured by the attribute(s) on a scale between 0 (worst fulfillment of the objective) to 1 (best fulfillment of the objective). When defining these value functions, the choice of attribute ranges is crucial. For the purpose of river assessment, these ranges should span the range from worst case river reaches to near natural river reaches (reference conditions). In this case, a value of 0 always corresponds to the worst case (=no fulfillment of the objective) and a value of 1 corresponds to near natural state (=complete fulfillment of the objective). We can then derive five color coded quality classes analogous to the European Union Water Framework Directive (European Union, 2000): ([0-0.2[bad, [0.2-0.4[poor, [0.4-0.6[moderate, [0.6-0.8[good, and [0.8-1.0] high) (examples given in Fig. A3). At higher hierarchical levels, the values from the lower level are aggregated to the value at the higher level using aggregation functions as described by Langhans et al. (2014). For this study, we selected an additive-minimum aggregation technique for the higher level aggregation of the different assessment areas to emphasize the complementarity of these aspects of a good ecological state (Langhans et al., 2014). Finally, we get the degrees of fulfillment of objectives at all levels of the objectives hierarchy on the same scale between 0 and 1.

At the highest hierarchical level, the objective of reaching a good ecological state is broken down into the sub-objectives of a

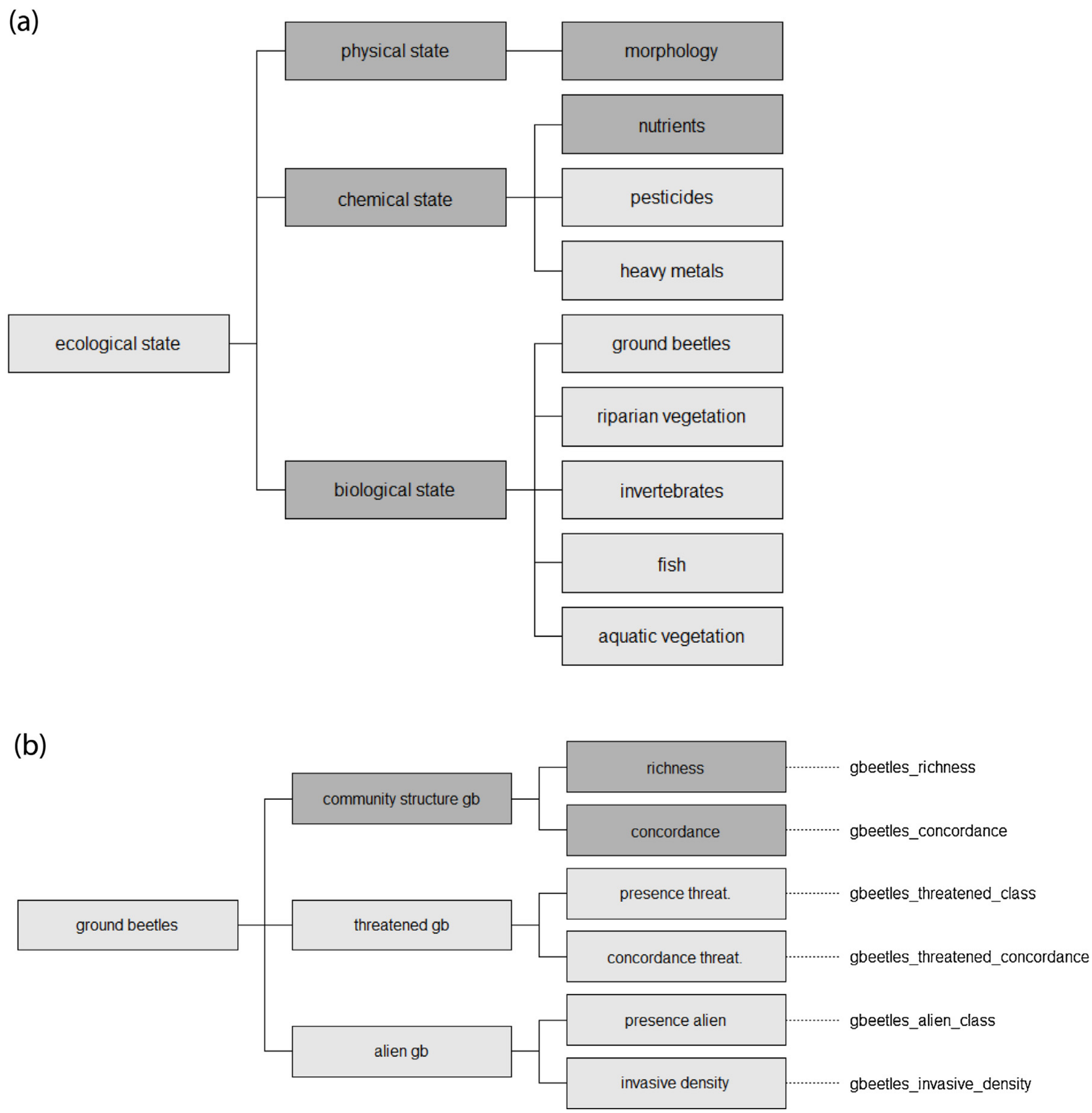

Fig. 1. Objectives hierarchy. (a) Topmost levels of the objectives hierarchy including a good physical state of the river, a good chemical state of the river water, and a good biological state in terms of five organism groups: 1) ground beetles, 2) riparian vegetation, 3) aquatic macroinvertebrates, 4) fish, and 5) aquatic vegetation. (b) Suggested objectives hierarchy for ground beetles. Objective hierarchies for floodplain vegetation and aquatic vegetation were structured analogously. The objectives hierarchy combined information about community structure, presence of threatened species, and presence of alien species. Mandatory endpoints are highlighted in dark grey. Attributes are given on the right side of the lowest-level sub-objectives (see Appendix A1 in Supplementary materials for details about the calculation of attributes).

good physical, chemical and biological state (Fig. 1a), which are all important aspects of a good ecological state, similar to Reichert et al. (2015). While the physical and biological sub-objectives are expected to respond to river restoration, the chemical state is used to evaluate potential limiting effects on biodiversity even if the physical state is improved by restoration. The physical state refers to the morphology of the river and is characterized by the variability of the river bed, degradation of the river bed and banks, and the width and vegetation of the riparian zone (Hütte and Niederhausser, 1998; Langhans et al., 2013). These indicators cover the full range of habitats that exist along rivers, and which influence biological communities. To assess water quality, we applied an existing assessment procedure for nutrients (Liechti et al., 2004) and extensions established for Swiss rivers for pesticides and heavy metals (AWEL, 2006) to measurements between 2010 and 2011. For nutrients (ammonia, nitrite, nitrate, phosphate, total phosphorus and dissolved organic carbon) the 90th percentile of 12 monthly grab samples is compared to ecological quality standards that were defined to prevent a negative impact on aquatic organisms. For pesticides, the concentrations of 12 herbicides and insecticides from eight monthly grab samples per year were grouped according to their mode of action and then evaluated depending on the number and timing of exceedances of ecological quality standards. For heavy metals (Cu, Zn, Pb, Cd, Hg, Ni, Cr), sediment concentrations in the fraction <0.063 mm were compared to ecological quality standards.

For the biological state, we used existing assessment procedures developed for aquatic macroinvertebrates and fish (Schager and Peter, 2004; Stucki, 2010), and developed new value functions for ground beetles, riparian vegetation and aquatic vegetation. The objectives hierarchies for riparian and aquatic vegetation follow the same scheme shown in Fig. 1b for ground beetles.

A good state regarding ground beetles considers the sub-objectives of a near natural community structure, the presence of threatened species, and the absence of alien species. For community structure, we quantify the concordance of observed taxa with expected taxa under natural conditions and assess the richness of the community. The attribute range is between 0 and 1 (0: not appropriate to reference conditions, 1: matching reference conditions), similar to the Ecological Quality Ratios developed for the European Union Water Framework Directive (WFD: annex V Section 1.4.1.ii). To estimate the list and number of species expected to be present in the reach under natural conditions and in the absence of predictive models, we developed a framework relating habitat types with species (see Appendix A1, Fig. A2 in Supplementary materials). According to our framework and literature about species present in Switzerland, we derived the list of species and the corresponding number of expected species for our reaches (Appendix A1, Tables A1–4 in Supplementary materials). The list of expected species should be as closely representative of the communities present in similar reaches under near natural conditions as possible. The same list of species expected to inhabit near natural river habitats was used to measure deviation to observed species in restored and unrestored reaches. Inaccuracy in our estimate of expected taxa was considered as a source of uncertainty for the assessment procedure (see next paragraph 2.6). For threatened species, we assess their presence and concordance with expected species. For alien species, we distinguish the presence of non-invasive and invasive species. To account for the complementarity of the sub-objectives at the lowest hierarchical level, we used the minimum-additive aggregation technique (Langhans et al., 2014) (see Appendix A1 and Fig. A3 for details in Supplementary materials). As mentioned in the introduction, due to dispersal limitations of threatened species, we cannot value their absence as a restoration failure. Similarly, the absence of alien species is not a particular success of restoration unless their elimination is specifically the target of the restoration project. To account for these concerns, we designed a specific technique for aggregating community structure with threatened and alien species. The presence of threatened species leads to a bonus and the presence of alien species leads to a malus, whereas the absence of threatened or alien species does not influence the basic assessment (see Appendix A1 in Supplementary materials). For this purpose, we applied an additive aggregation (weighted averaging with half the weight for the "bonus" and "malus" objectives compared to the main objective), but only considering the "bonus" objective if it is larger than the main objective and the "malus" objective if it is smaller than the main objective. Therefore, an increase of the number of alien species after restoration will reduce the fulfillment of the objective for the organism group and an increase of the number of threatened species will increase the fulfillment.

2.6. Quantification and propagation of uncertainty

Monitoring, taxa identification, and data analysis are all uncertain to different degrees (Hering et al., 2010). In our framework, we estimated uncertainty for physical-chemical and biological attributes separately. Uncertainty about attribute values was considered by formulating the uncertain knowledge as probability distributions based on expert judgment and literature, due to limited data to provide a data-based measure of uncertainty (e.g. Clarke and Hering, 2006; Lindegarth et al., 2013). For attributes that can only take positive values, we formulated this knowledge in the form of lognormal distributions. Probability distributions were truncated when attributes were known to be limited within a range of values. Standard deviations (coefficients of variation) were estimated according to current knowledge about uncertainty in monitoring of hydromorphological conditions, water quality and biological indicators (e.g. Clarke and Hering, 2006; Rode and Suhr, 2007). This lead to a relative standard deviation of 5% for attributes related to physical measurements, 15% for attributes of the biological samples (due to imprecision of samplings in the field, difficulties in taxa identification and inaccuracies in our estimate of expected taxa), and 20% for attributes related to chemical measurements (due to expected temporal variability and much less due to measurement imprecision). While uncertainty in the sampling techniques can vary between organism groups, a relatively conservative estimate of 15% was chosen for all cases in the absence of more precise information. The joint probability distribution of all parameters was constructed by assuming independence of the distributions for different attributes, and was propagated through the assessment procedure by Monte Carlo simulation. The resulting valuations were visualized as medians and 90% credibility ranges (Reichert et al., 2013). Since these distributions were based on expert opinion that is uncertain itself, we tested whether differences in the assessment between restored and degraded reaches were still significant when doubling the standard deviation of the attributes. Uncertainty was propagated in the framework with the use of the R package "utility" (Reichert et al., 2013).

3. Results

3.1. Important differences in habitat conditions and communities

Habitat conditions of restored reaches significantly differed from degraded reaches in both rivers (Fig. 2a–c), along the first axis of the PCA for the Töss River (Wilcoxon signed-rank test $P < 0.004$), and along the second axis for both rivers (Wilcoxon signed-rank test Thur $P = 0.02$; Töss $P < 0.001$). Aquatic habitat conditions mostly changed along the first axis of the PCA, and terrestrial conditions along the second axis (Fig. 2a). We observed an increase of velocity and depth conditions in the restored reach of the Töss (Fig. 2a, c), and an increase of the bankfull width and the diversity of meso-habitats within the terrestrial part of both river floodplains. Beta diversity of habitat conditions was significantly higher in the restored reach of the Thur River than in the degraded reach (Wilcoxon signed-rank test $P = 0.001$; Fig. 2d).

All organism groups showed a higher richness in restored reaches compared to the degraded reaches (Table 1), except for the aquatic vegetation in the Töss River, which decreased by three taxa

Table 1
Number of taxa per organism group measured within Töss and Thur Rivers. The number of taxa is given within degraded and restored reaches, and between reaches (i.e. difference: restored–degraded). * The number of taxa for the aquatic vegetation is divided into hydrophytes and bryophytes.

	Töss			Thur		
Organism group	degraded	restored	difference	degraded	restored	difference
Ground beetles	2	9	7	3	13	10
Riparian vegetation	36	39	3	20	29	9
Aquatic macroinvertebrates	47	48	1	39	47	8
Fish	4	4	0	7	10	3
Hydrophytes*	0	1	1	1	5	4
Bryophytes*	4	1	−3	2	4	2

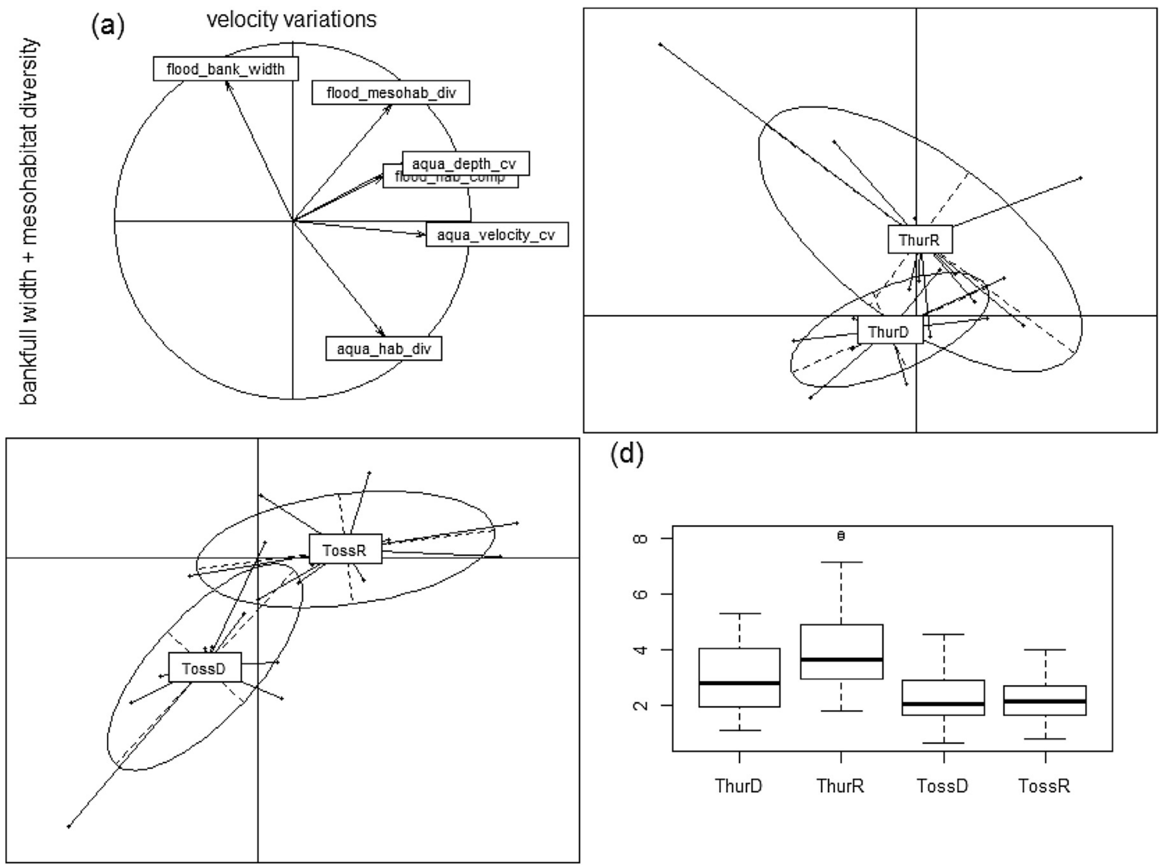

Fig. 2. Habitat differences between restored and degraded reaches in Thur and Töss Rivers measured with a principal component analysis. (a) Correlation circle of 6 variables, with names of driving variables highlighted along the axes: variability in velocity (coded: aqua_velocity_cv) along the first axis, and the bankfull width + mesohabitat diversity (coded: flood_bank_width and flood_mesohab_div) along the second axis. (b) Differences within Thur River (ThurD: Thur degraded, ThurR: Thur restored). (c) Differences within Töss River (TossD: Töss degraded, TossR: Töss restored). (d) Measure of beta diversity of habitat conditions within each reach. Boxes represent the interquartile range (Q75-Q25) around the median. Upper whisker represents (Q75 + 1.5*(Q75-Q25)) of the data, and the lower (Q25-1.5*(Q75-Q25)). Empty circles represent outliers.

of Bryophytes, and the fish in the Töss River, which did not show a difference. Overall differences were significant for the Thur River, and not for the Töss River (Wilcoxon signed-rank test Thur $P = 0.03$, Töss $P = 0.3$). The largest difference in richness was observed for ground beetles in both rivers (10 taxa in the Thur and 7 in the Töss), followed by riparian vegetation and aquatic macroinvertebrates in the Thur River (9 and 8 taxa, respectively; Table 1). The composition of aquatic macroinvertebrates in the main river channel of the Thur differed by 16 taxa between the degraded and the restored reach, including 6 EPT taxa (Table A5). In the Töss River, the composition of the degraded and restored reach differed only by 4 Taxa (Table A5). Differences in COH (Coleoptera, Odonata and Heteroptera) were smaller than those of EPT (Table A5).

3.2. Reflection on differences in assessment results

Morphology had a higher value for the restored reaches compared to the degraded ones. Moderate and poor morphological conditions were assessed for the degraded reaches of the Thur and the Töss respectively, and high conditions for the restored reaches of both rivers (Figs. 3a and 4a, or 3b and 4b with visualization of uncertainty). In both rivers, the width variability of the river bed and the width of the riparian zone were larger at the restored reaches (Figs. A4 –5). Along the restored reaches, the bank structure also showed a higher value following the removal of the embankments compared to the degraded reaches (Figs. A4–5). The biological state had a higher value for the restored reaches compared to the degraded ones (Figs. 3–4), due to higher values for all organism groups, with the exception of the aquatic vegetation (−0.13 in the Töss and −0.02 in the Thur, Table 2). The largest difference in value was reached for ground beetles in both rivers (0.66 in the Töss and 0.63 in the Thur, Table 2). The higher density and presence of invasive species (*Impatiens glandulifera* and *Elodea nuttallii*, Table 3) decreased the value of the riparian and aquatic vegetation for the restored reaches. The chemical state was higher for the Töss River compared to the Thur (Figs. 3–4), mostly due to the detection of photosynthetic inhibitors in the Thur River (see Fig. A4). A poor overall ecological state was assessed for the degraded reaches and a moderate state for the restored reach of the Thur, and a good state in the restored reach of the Töss (Figs. 3–4). The value of the ecological state for the restored reaches of the Töss and the Thur are 0.60 and 0.58 respectively, and 0.34 and 0.31, respectively, for the degraded reaches (1 being the near natural state and 0 the worst case).

3.3. Uncertainty

We found larger uncertainty ranges for results concerning aquatic macroinvertebrate communities and aquatic vegetation than for the other organism groups (Figs. 3b and 4b). A larger uncertainty was detected in the valuation of pesticides in the Thur River than for the uncertainty in the nutrients and the heavy metals (Fig. 4b). Values concerning physical and biological states for the restored reaches had similar levels of uncertainty compared to the degraded reaches (Figs. 3b and 4b). For both rivers, the difference in the ecological state between restored and degraded reaches was

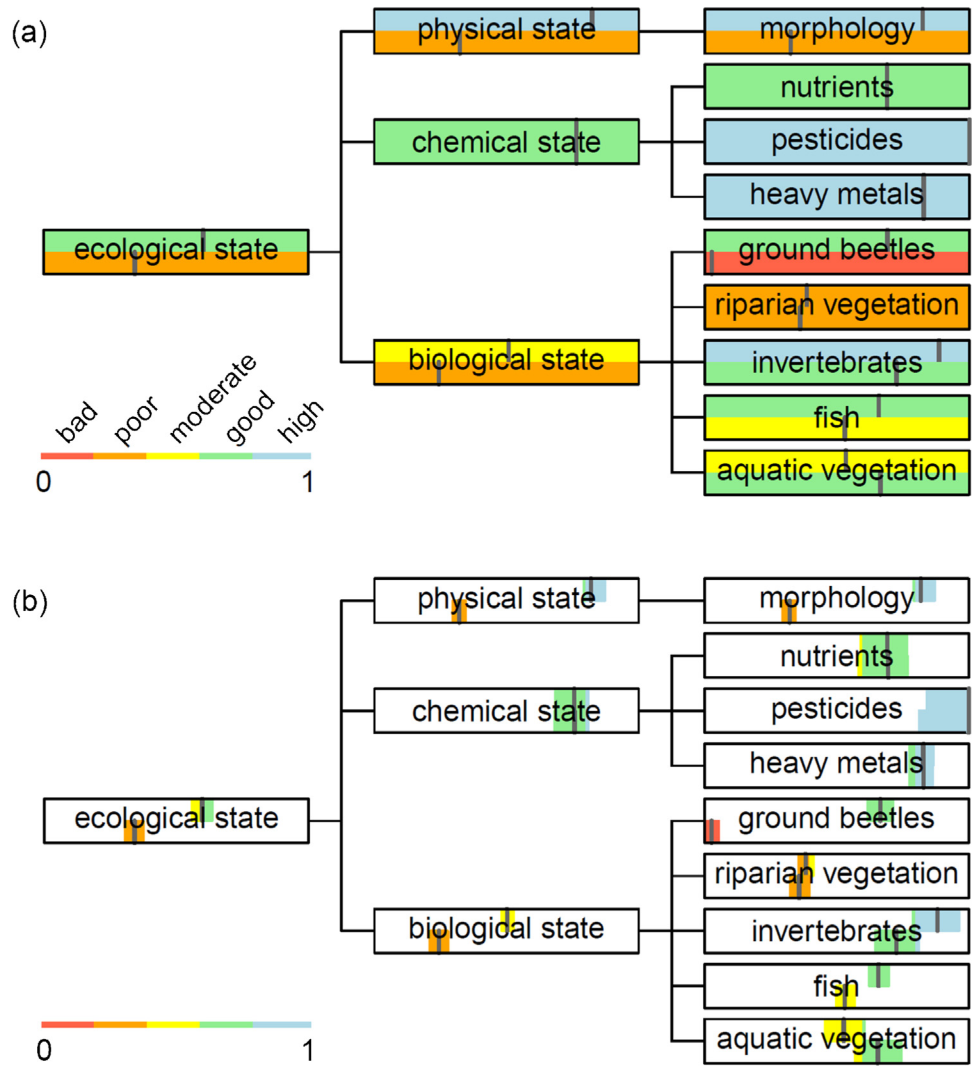

Fig. 3. Ecological valuation of the Töss River in degraded and restored reaches. The restored reach is represented in the upper parts of the boxes and the degraded reach in the lower part. (a) Version without uncertainty. Vertical grey lines represent the values corresponding to the mean of the attributes. Colors belong to five classes equally distributed from 0 (no fulfillment, worst case) to 1 (complete fulfillment, near natural state). (b) Version with uncertainty, vertical black lines represent median values, and colored areas represent 5–95% quantile intervals.

Table 2
Valuation of organism groups in Töss and Thur Rivers (on the value scale from 0 = no fulfillment, worst case to 1 = complete fulfillment, near natural state). Reaches are divided according to restoration efforts.

	Töss			Thur		
Organism group	degraded	restored	difference	degraded	restored	difference
Ground beetles	0.03	0.69	0.66	0.04	0.67	0.63
Riparian vegetation	0.36	0.39	0.03	0.35	0.47	0.12
Aquatic macroinvertebrates	0.73	0.89	0.16	0.68	0.73	0.05
Fish	0.53	0.66	0.13	0.46	0.61	0.15
Aquatic vegetation	0.66	0.53	−0.13	0.49	0.47	−0.02

Table 3
Alien species present in degraded and restored reaches of the Töss and Thur Rivers. Values represent the fraction of the alien species abundance (for macroinvertebrates) or coverage (for vegetation) relative to the rest of the community.

		Töss		Thur	
Organism group	Species	degraded	restored	degraded	restored
Macroinvertebrates	*Potamopyrgus antipodarum*	0	0	0.001	0.027
Riparian vegetation	*Solidago canadensis*	0.047	0.029	0.096	0.074
	Impatiens glandulifera	0	0.043	0.054	0.042
Aquatic vegetation	*Elodea nuttallii*	0	0	0	0.05

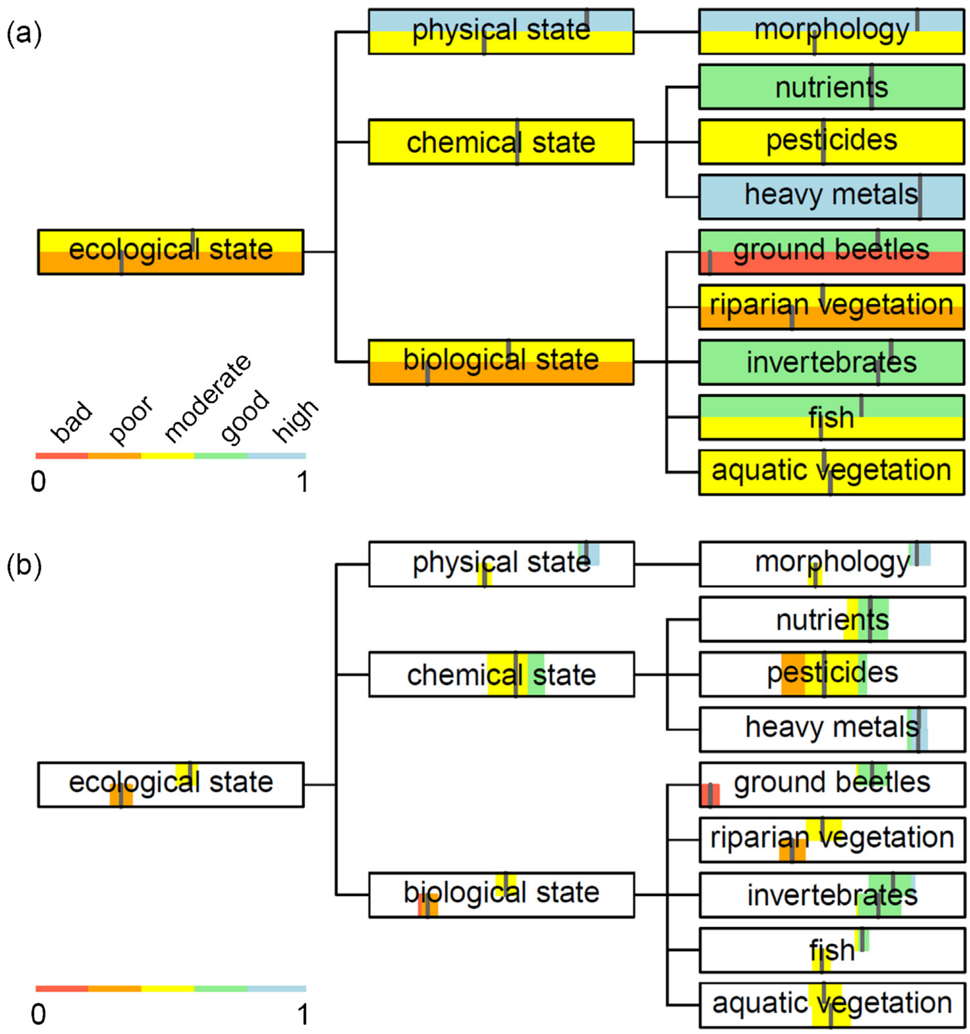

Fig. 4. Ecological valuation of the Thur River in degraded and restored reaches. Treatments are identified as in Fig. 3. (a) Version without uncertainty and (b) with uncertainty.

significant, even when taking uncertainty into account (Figs. 3b–4b, A6–7). When doubling the uncertainty of the attributes, the difference in the ecological state between restored and degraded reaches was still significant (change in standard deviation of the ecological state for each reach: Töss restored +0.009, Töss degraded +0.006, Thur restored +0.005, Thur degraded +0.004).

4. Discussion

4.1. Quantification of restoration effects on habitat conditions and biodiversity expressed as results of the ecological assessment

We observed significantly better physical and biological states for the restored reaches of both rivers. Even in the absence of comprehensive before-after data, based on the similarity of the reaches before restoration and mechanistic biological knowledge, this can be safely interpreted as a causal consequence of restoration. All organism groups benefited from restoration to different degrees, except the aquatic vegetation. Improvement was most pronounced for ground beetles, which benefited in both rivers from the creation of gravel bars following the river widening. The assessment procedure showed a significant difference from bad to good between degraded and restored sites. Riparian vegetation developing on gravel bars also benefited to a lesser degree from restoration (assessment results remained in the poor class for the Töss River and became moderate in the Thur River). A similar increase in the diversity of ground beetles was also observed for other restoration projects in Europe (Hering et al., 2015), which corroborated assumptions about the positive effect of restoration on ground beetles (Januschke et al., 2011; Lambeets et al., 2008). Two reasons may explain the lower improvement of the state of the riparian vegetation compared to the ground beetles. First, the close proximity of agricultural land and intensively managed forests could negatively influence the species composition by propagating seeds unrelated to floodplains that can germinate in the restored reaches. Unexpected species could also be present due to the inaccuracy in our methodology in establishing the list of expected plants in natural conditions. In both rivers, many of the observed species on the floodplain were not the expected ones, leading to a lower assessed state for the restored site. Second, the success of those unexpected species, if they are correctly not expected to occur at the sites, could also be an indication that the seed bank for floodplain species is impoverished (Brederveld et al., 2011) and that recolonization needs more time than the time span between restoration and investigation in both rivers.

In addition to these changes, restoration had measurable positive effects on instream conditions, with a higher variability in the water depth, river width and flow velocity. The removal of the embankments along the Thur River increased the width variability of the river. In the Töss River, the creation of a secondary connected channel increased the variability of flow velocity and water depth. Aquatic organism status also improved, especially for fish, followed by macroinvertebrates, but not for the aquatic vegetation. Fish and benthic invertebrates were both expected to increase in richness in the absence of other limiting factors (e.g. lack of colonization sources, poor water quality) (e.g. Jähnig et al., 2010; Pont et al., 2006; Schmutz et al., 2014), which was corroborated for fish and macroinvertebrates in the Thur River and to a lesser extent for the macroinvertebrates in the Töss River. At a higher spatial scale, the presence of many barriers within the Töss River could explain the lack of new fish species colonizing the restored reach. However, the species already present in the reach showed a higher density at the restored site compared to the degraded one, thereby increasing the assessment value for fish at the restored Töss River site.

While most indicators improved with restoration, the aquatic vegetation showed a negative response due to a lower richness in the Töss River and the occurrence of an invasive species at the restored reach of the Thur (i.e. *E. nuttallii*), both of which lowered the improvement by restoration. The increase in bed movement could explain the reduced number of aquatic plants in a restored reach compared to a degraded and more stable reach, but large widening of a river by restoration could also recreate slow flowing habitats, which in turn are favorable to specific species of aquatic plants (Madsen et al., 2001). This case is exemplified by the presence of *E. nuttallii* in the Thur River, indicating that shallow, slow flowing habitats have been created which fostered the establishment of this species (see for example Nichols and Shaw, 1986). Furthermore, missing indigenous species may be attributed to missing colonization sources upstream of the reaches (Barrat-Segretain, 1996) as the restored reaches are short compared to the many kilometers of degraded river reaches up- and downstream. Aquatic vegetation can be considered as potentially problematic for monitoring river restoration effects due to high uncertainty concerning the ability to detect effects at the site level (Demars et al., 2012), but see Lorenz et al. (2012).

4.2. Multi-attribute approach and uncertainty

To value the effects and success of river restoration, species richness is insufficient. We suggest also taking into account the presence of threatened and alien species. Few threatened species (for example *Bembidion prasinum*, ground beetles) and alien species (e.g. *E. nuttallii*, aquatic vegetation) occurred in restored reaches. Often, species living in floodplain habitats (aquatic and terrestrial) are threatened due to the high number of channelized rivers that lack natural banks and floodplain habitats. Restoration improved the structural conditions of the floodplain by increasing the space and the availability of the river to braid and recreate gravel bars, which in turn was favorable to threatened species such as *B. prasinum*. However, alien species (terrestrial and aquatic) also occurred in restored reaches, where the restoration process could have created empty niches susceptible to rapid colonization by alien species that are known to have high colonization abilities (DAISIE, 2009; Strayer, 2010). The colonization potential of alien species may prevent threatened species from establishing new populations. Long term monitoring will help to detect successes in creating new habitats for threatened species relative to the potential establishment of alien species.

While the river morphology assessment showed a large difference between restored and degraded sites in both rivers, the differences in the biological assessment were smaller but still indicate a positive effect of restoration. This is in part due to the presence of invasive species which, in our assessment procedure, lowers the ecological state. In addition, the lack of colonization sources and water quality problems can be potential limiting factors for the rivers and, as a consequence, for the recolonization of the restored sites. In the Thur River, a photosynthetic inhibitor (Diuron) was detected at concentrations exceeding environmental quality standards, resulting in a moderate assessment of pesticides (Fig. 4). Chronic exposure to photosynthetic inhibitors can change the biomass and composition of aquatic plants (McClellan et al., 2008), and therefore could limit the positive effect of morphological river restoration on biota. Nevertheless, our indicators for aquatic and terrestrial plants showed that the richness already reached a high value. Therefore, we expect that an improvement in water quality could only have a small positive effect on the populations already established at the sites. This exemplifies the importance of taking into account different types of abiotic indicators (physical indicators and water quality) in addition to biotic indicators to reveal potential deficits that can limit the effect of river restoration on the biota. Moreover, organism groups might respond differently to different stressors and mitigation measures, underlining the importance of combining several indicators and not relying on only one which could underestimate the effect of restoration on biodiversity. We argue that the combination of several organism groups increases the robustness of an ecological assessment (e.g. Hering et al., 2006) and thus provides more confidence in the quantification of restoration effects. In addition, multiple sampling campaigns could contribute to accounting for temporal variability and obtaining a more complete overview of species that are present at the different sites.

To determine whether an improvement of the ecological state due to restoration is significant or lies within the measurement error, the assessment and propagation of uncertainty is essential. Our results showed that the improvement by restoration was within the uncertainty range for some organism groups (e.g. for invertebrates), while for most groups (i.e. for ground beetles and fish) and for the river morphology the improvements were significant. The propagation of uncertainty through the framework allowed us to evaluate the confidence in the difference of the ecological state between the restored and the degraded reaches. In our study, even if we doubled the uncertainty of the measurements, the improvement due to restoration was still significant. Instead of doubling the uncertainty to test the significance under higher uncertainty, quantifying the uncertainty based on the variability of field measurements would be an improvement, but would require more data. Nevertheless, the effects still significant under higher uncertainty indicated that the measured effects were larger than our level of uncertainty regarding the observed attributes. It also shows that the selected attributes are sensitive enough to detect effects. The suggested framework (including the objectives hierarchy, attributes, value functions, and propagation of uncertainty) can be applied to other European rivers, with the need to adapt the expected species to the local reference conditions and to potentially modify the value functions to other national or local assessment procedures. We encourage river managers to use multiple indicators (physical, chemical, biological, terrestrial and aquatic) to comprehensively quantify the ecological state of the rivers and the success or failure of river restoration measures, and to include uncertainty in their assessment methods. Such an analysis at different hierarchical levels can also provide hints about potential causes of restoration failures.

5. Conclusions

The proposed framework based on multi-attribute value theory proved to be suitable to evaluate the ecological state of the rivers and, as a specific application, to quantify restoration success by comparing the ecological states of restored and unrestored river reaches. Extensions of traditional assessment procedures to include more organism groups seem useful to describe additional relevant aspects of the ecological state of rivers and to evaluate restoration success. A translation of existing methods and ecological judgement to value functions helps us assessing and visualizing results at all hierarchical levels on a unified continuous scale and to propagate uncertainty of observed or predicted attributes to the assessment results. A critical aspect in the design of ecological assessment protocols is to define the expected near natural state (diversity, taxa), which was done here based on the literature. More effort is needed to confirm and improve these results based on observations of community structures along a gradient of human influence and, if possible, including reference sites. Alternative metrics describing the functions performed by the communities could be developed and incorporated into the framework to enrich the assessment of the ecological state. To guide ecosystem management, the consideration of abiotic factors is important to detect possible deficits that limit biological success while an aggregation to higher levels is important for decision support, synthesis, and communication of results. In this regard, integrative valuation with value functions that is based on a hierarchy of objectives and allows an analysis of the degree of fulfillment of sub-objectives while also providing an overall valuation proved to be useful. The estimation and propagation of uncertainty helped us to evaluate the significance of differences between assessment results. Application of the proposed assessment method to other river restoration monitoring programmes can contribute to the identification of cause-effect relationships between physical and biological changes and the effect of chemical and hydrological degradation. This could support improvements in the effective design of river restoration measures and their integration into comprehensive river management frameworks that also address water quality and hydrological deteriorations.

Acknowledgments

We thank Tom Buijse for coordination of the Reform project and constant support, Daniel Hering and Reform partners for stimulating discussions, Benjamin Kupilas, Daniela Campana, Alexander Gieswein and Lisa Schülting for collecting field data, Rosi Siber, Jacqueline Schlosser and Hélène Mottaz for help in the field, Marion Junghans for constructive discussions about pesticides, Andrew Clarke and Bogdan Caradima for language corrections, anonymous reviewers for constructive comments, Marco Baumann, Christian Marti and Andri Bryner for providing photos, and Pius Niederhauser, Patrick Steinmann, Andreas Hertig and Heinz Ehmann for providing data. This paper results from the EU-funded Integrated Project REFORM (Restoring rivers FOR effective catchment Management), European Union's Seventh Framework Programme (FP7/2007-2013) under Grant Agreement No. 282656.

Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.ecolind.2016.07.048.

References

AWEL, 2006. Wasserqualität der Seen, Fliessgewässer und des Grundwasser im Kanton Zürich, Statusbericht, In: AWEL (Ed.), Zurich www.gewaesserqualitaet.zh.ch.
AWEL, 2012. Zürcher Gewässer 2012. Entwicklung—Zustand—Ausblick, In: AWEL (Ed.), Zurich www.gewaesserqualitaet.zh.ch.
Barrat-Segretain, M.H., 1996. Strategies of reproduction, dispersion, and competition in river plants: a review. Vegetatio 123, 13–37.
Beechie, T., Pess, G., Roni, P., 2008. Setting river restoration priorities: a review of approaches and a general protocol for identifying and prioritizing actions. North Am. J. Fish Manag. 28, 891–905.
Bernhardt, E.S., Palmer, M.A., Allan, J.D., Alexander, G., Barnas, K., Brooks, S., Carr, J., Clayton, S., Dahm, C., Follstad-Shah, J., Galat, D., Gloss, S., Goodwin, P., Hart, D., Hassett, B., Jenkinson, R., Katz, S., Kondolf, G.M., Lake, P.S., Lave, R., Meyer, J.L., O'Donnell, T.K., Pagano, L., Powell, B., Sudduth, E., 2005. Synthesizing US river restoration efforts. Science 308, 636–637.
Brederveld, R.J., Jahnig, S.C., Lorenz, A.W., Brunzel, S., Soons, M.B., 2011. Dispersal as a limiting factor in the colonization of restored mountain streams by plants and macroinvertebrates. J. Appl. Ecol. 48, 1241–1250.
Bundi, U., Peter, A., Frutiger, A., Hutte, M., Liechti, P., Sieber, U., 2000. Scientific base and modular concept for comprehensive assessment of streams in Switzerland. Hydrobiologia 422, 477–487.
Clarke, R.T., Hering, D., 2006. Errors and uncertainty in bioassessment methods—Major results and conclusions from the STAR project and their application using STARBUGS. Hydrobiologia 566, 433–439.
Convertino, M., Baker, K.M., Vogel, J.T., Lu, C., Suedel, B., Linkov, I., 2013. Multi-criteria decision analysis to select metrics for design and monitoring of sustainable ecosystem restorations. Ecol. Indic. 26, 76–86.
Corsair, H.J., Ruch, J.B., Zheng, P.Q., Hobbs, B.F., Koonce, J.F., 2009. Multicriteria decision analysis of stream restoration: potential and examples. Group Decis. Negot. 18, 387–417.
DAISIE, 2009. Handbook of alien species in Europe. In: Drake, J.A. (Ed.), Invading Nature: Springer Series in Invasion Ecology. Springer, Dordrecht, Netherlands.
Demars, B.O.L., Potts, J.M., Tremolieres, M., Thiebaut, G., Gougelin, N., Nordmann, V., 2012. River macrophyte indices: not the Holy Grail! Freshw. Biol. 57, 1745–1759.
Dyer, J.S., Sarin, R.K., 1979. Measurable multiattribute value functions. Oper. Res. 27, 810–822.
Dyer, J.S., Sarin, R.K., 1982. Relative risk-aversion. Manag. Sci. 28, 875–886.
Eisenführ, F., Weber, M., Langer, T., 2010. Rational Decision Making. Springer, Berlin, Germany.
European Union, 2000. Directive 2000/60/EC of the European Parliament of the Council of 23 October 2000 establishing a framework for Community action in the field of water policy. Off. J. Eur. Union, L327.
Gallardo, B., Doledec, S., Paillex, A., Arscott, D.B., Sheldon, F., Zilli, F., Merigoux, S., Castella, E., Comin, F.A., 2014. Response of benthic macroinvertebrates to gradients in hydrological connectivity: a comparison of temperate, subtropical, Mediterranean and semiarid river floodplains. Freshw. Biol. 59, 630–648.
Hütte, M., Niederhausser, P., 1998. Ecomorphology—Level I. Méthodes d'analyse et d'appréciation des cours d'eau en Suisse, in: FOEN (Ed.), L'environnement pratique n°27, Berne, p. 43.
Hering, D., Moog, O., Sandin, L., Verdonschot, P.F.M., 2004. Overview and application of the AQEM assessment system. Hydrobiologia 516, 1–20.
Hering, D., Feld, C.K., Moog, O., Ofenbock, T., 2006. Cook book for the development of a multimetric Index for biological condition of aquatic ecosystems: experiences from the european AQEM and STAR projects and related initiatives. Hydrobiologia 566, 311–324.
Hering, D., Borja, A., Carstensen, J., Carvalho, L., Elliott, M., Feld, C.K., Heiskanen, A.S., Johnson, R.K., Moe, J., Pont, D., Solheim, A.L., Van De Bund, W., 2010. The european water framework directive at the age of 10: a critical review of the achievements with recommendations for the future. Sci. Total Environ. 408, 4007–4019.
Hering, D., Aroviita, J., Baattrup-Pedresen, A., Brabec, K., Buijse, T., Ecke, F., Friberg, N., Gielczewski, M., Januschke, K., Köhler, J., Kupilas, B., Lorenz, A., Muhar, S., Paillex, A., Poppe, M., Schmidt, T., Schmutz, S., Vermaat, J., Verdonschot, P., Verdonschot, R., 2015. Contrasting the roles of section length and instream habitat enhancement for river restoration sucess: a field study on 20 European restoration projects. J. Appl. Ecol. 52, 1518–1527.
Jähnig, S.C., Brunzel, S., Gacek, S., Lorenz, A.W., Hering, D., 2009. Effects of re-braiding measures on hydromorphology, floodplain vegetation, ground beetles and benthic invertebrates in mountain rivers. J. Appl. Ecol. 46, 406–416.
Jähnig, S.C., Brabec, K., Buffagni, A., Erba, S., Lorenz, A.W., Ofenbock, T., Verdonschot, P.F.M., Hering, D., 2010. A comparative analysis of restoration measures and their effects on hydromorphology and benthic invertebrates in 26 central and southern european rivers. J. Appl. Ecol. 47, 671–680.
Januschke, K., Brunzel, S., Haase, P., Hering, D., 2011. Effects of stream restorations on riparian mesohabitats, vegetation and carabid beetles. Biodivers. Conserv. 20, 3147–3164.
Keeney, R.L., 1992. Value focused thinking. In: A Path to Creative Decisionmaking. Harvard University Press.
Lambeets, K., Hendrickx, F., Vanacker, S., Van Looy, K., Maelfait, J.P., Bonte, D., 2008. Assemblage structure and conservation value of spiders and carabid beetles from restored lowland river banks. Biodivers. Conserv. 17, 3133–3148.
Langhans, S.D., Lienert, J., Schuwirth, N., Reichert, P., 2013. How to make river assessments comparable: a demonstration for hydromorphology. Ecol. Indic. 32, 264–275.
Langhans, S., Reichert, P., Schuwirth, N., 2014. The method matters: a guide for indicator aggregation in ecological assessments. Ecol. Indic. 45, 494–507.
Lepori, F., Palm, D., Brannas, E., Malmqvist, B., 2005. Does restoration of structural heterogeneity in streams enhance fish and macroinvertebrate diversity? Ecol. Appl. 15, 2060–2071.
Liechti, P., Frutiger, A., Zobrist, J., 2004. Physico-chemical analyses—Level I and II. Méthodes d'analyse et d'appréciation des cours d'eau en Suisse, in: FOEN (Ed.), L'environnement pratique n°1005, Berne, p. 44.
Lindegarth, M., Carstensen, J., Johnson, R.K., 2013. Uncertainty of biological indicators for the WFD in Swedish water bodies: current procedures and a proposed framework for the future., WATERS Report no. 2013:1 Deliverable 2. 2-1, Havsmiljöinstitutet, Sweden.
Lorenz, A.W., Korte, T., Sundermann, A., Januschke, K., Haase, P., 2012. Macrophytes respond to reach-scale river restorations. J. Appl. Ecol. 49, 202–212.

Madsen, J.D., Chambers, P.A., James, W.F., Koch, E.W., Westlake, D.F., 2001. The interaction between water movement, sediment dynamics and submersed macrophytes. Hydrobiologia 444, 71–84.
McClellan, K., Altenburger, R., Schmitt-Jansen, M., 2008. Pollution-induced community tolerance as a measure of species interaction in toxicity assessment. J. Appl. Ecol. 45, 1514–1522.
Nichols, S.A., Shaw, B.H., 1986. Ecological life histories of the 3 aquatic nuisance plants, Myriophyllum-spicatum, Potamogeton-crispus and Elodea-canadensis. Hydrobiologia 131, 3–21.
Palmer, M.A., Bernhardt, E.S., Allan, J.D., Lake, P.S., Alexander, G., Brooks, S., Carr, J., Clayton, S., Dahm, C.N., Shah, J.F., Galat, D.L., Loss, S.G., Goodwin, P., Hart, D.D., Hassett, B., Jenkinson, R., Kondolf, G.M., Lave, R., Meyer, J.L., O'Donnell, T.K., Pagano, L., Sudduth, E., 2005. Standards for ecologically successful river restoration. J. Appl. Ecol. 42, 208–217.
Palmer, M.A., Menninger, H.L., Bernhardt, E., 2010. River restoration, habitat heterogeneity and biodiversity: a failure of theory or practice? Freshw. Biol. 55, 205–222.
Pasquale, N., Perona, P., Schneider, P., Shrestha, J., Wombacher, A., Burlando, P., 2011. Modern comprehensive approach to monitor the morphodynamic evolution of a restored river corridor. Hydrol. Earth Syst. Sci. 15, 1197–1212.
Pont, D., Hugueny, B., Beier, U., Goffaux, D., Melcher, A., Noble, R., Rogers, C., Roset, N., Schmutz, S., 2006. Assessing river biotic condition at a continental scale: a European approach using functional metrics and fish assemblages. J. Appl. Ecol. 43, 70–80.
Poppe, M., Stelzhammer, M., Seebacher, M., Januschke, K., Lorenz, A.W., Kupilas, B., Hering, D., 2012. Field protocols and associated database. Deliverable D.4.1. for EU collaborative project REFORM. - Restoring rivers for effective catchment management, p. 109, http://www.reformrivers.eu/field-protocols-and-associated-database-paired-river-restoration-comparison.
Reichert, P., Borsuk, M., Hostmann, M., Schweizer, S., Sporri, C., Tockner, K., Truffer, B., 2007. Concepts of decision support for river rehabilitation. Environ. Modell. Softw. 22, 188–201.
Reichert, P., Schuwirth, N., Langhans, S., 2013. Constructing, evaluating and visualizing value and utility functions for decision support. Environ. Modell. Softw. 46, 283–291.
Reichert, P., Langhans, S., Lienert, J., Schuwirth, N., 2015. The conceptual foundation of environmental decision support. J. Environ. Manag. 154, 316–332.
Rode, M., Suhr, U., 2007. Uncertainties in selected river water quality data. Hydrol. Earth Syst. Sci. 11, 863–874.
Sala, O.E., Chapin, F.S., Armesto, J.J., Berlow, E., Bloomfield, J., Dirzo, R., Huber-Sanwald, E., Huenneke, L.F., Jackson, R.B., Kinzig, A., Leemans, R., Lodge, D.M., Mooney, H.A., Oesterheld, M., Poff, N.L., Sykes, M.T., Walker, B.H., Walker, M., Wall, D.H., 2000. Global biodiversity scenarios for the year 2100. Science 287, 1770–1774.
Schager, E., Peter, A., 2004. Fish—Level I. Méthodes d'analyse et d'appréciation des cours d'eau en Suisse, in: FOEN (Ed.), L'environnement pratique n°44, Berne, p. 63.
Schirmer, M., Luster, J., Linde, N., Perona, P., Mitchell, E.A.D., Barry, D.A., Hollender, J., Cirpka, O.A., Schneider, P., Vogt, T., Radny, D., Durisch-Kaiser, E., 2014. Morphological, hydrological, biogeochemical and ecological changes and challenges in river restoration—The Thur River case study. Hydrol. Earth Syst. Sci. 18, 2449–2462.
Schmutz, S., Kremser, H., Melcher, A., Jungwirth, M., Muhar, S., Waidbacher, H., Zauner, G., 2014. Ecological effects of rehabilitation measures at the Austrian Danube: a meta-analysis of fish assemblages. Hydrobiologia 729, 49–60.
Schuwirth, N., Reichert, P., 2014. R package ecoval: Procedures for Ecological Assessment of Surface Waters. Version 1.0. The Comprehensive R Archive Network (CRAN): https://cran.r-project.org/web/packages/ecoval.
Strayer, D.L., 2010. Alien species in fresh waters: ecological effects, interactions with other stressors, and prospects for the future. Freshw. Biol. 55, 152–174.
Stucki, P., 2010. Macroinvertebrates—Level I. Méthodes d'analyse et d'appréciation des cours d'eau en Suisse, in: FOEN (Ed.), L'environnement pratique n°1026, Berne, p. 61.
Tockner, K., Stanford, J.A., 2002. Riverine flood plains: present state and future trends. Environ. Conserv. 29, 308–330.
Vaughan, I.P., Ormerod, S.J., 2005. Increasing the value of principal components analysis for simplifying ecological data: a case study with rivers and river birds. J. Appl. Ecol. 42, 487–497.
Vorosmarty, C.J., McIntyre, P.B., Gessner, M.O., Dudgeon, D., Prusevich, A., Green, P., Glidden, S., Bunn, S.E., Sullivan, C.A., Liermann, C.R., Davies, P.M., 2010. Global threats to human water security and river biodiversity. Nature 467, 555–561.
Wittenberg, R., Kenis, M., Blick, T., Hänggi, A., Gassmann, A., Weber, E., 2005. In: Wittenberg, R. (Ed.), An Inventory of Alien Species and Their Threat to Biodiversity and Economy in Switzerland. Federal Office for the Environment, Bern, p. 155.
Woolsey, S., Capelli, F., Gonser, T., Hoehn, E., Hostmann, M., Junker, B., Paetzold, A., Roulier, C., Schweizer, S., Tiegs, S.D., Tockner, K., Weber, C., Peter, A., 2007. A strategy to assess river restoration success. Freshw. Biol. 52, 752–769.