Science of the Total Environment 650 (2019) 1613–1627

Contents lists available at ScienceDirect

Science of the Total Environment

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

Ecological assessment of river networks: From reach to catchment scale

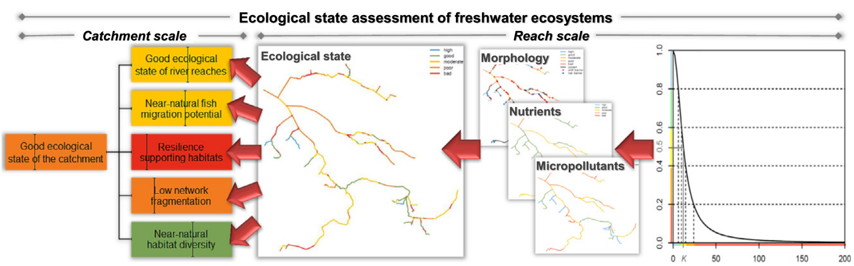
Mathias Kuemmerlen *, Peter Reichert, Rosi Siber, Nele Schuwirth

Eawag, Department of Systems Analysis, Integrated Assessment and Modelling, Ueberlandstrasse 133, CH-8600 Duebendorf, Switzerland

HIGHLIGHTS

- River restoration planning profits from assessment at the catchment-scale.
- The assessment is based on spatially-explicit reach-scale information.
- Spatial criteria are used to aggregate the assessment from reach to catchment-scale.
- Spatial criteria account for ecological network properties such as connectivity.
- The approach is tested in four different catchments of the Swiss plateau.

GRAPHICAL ABSTRACT



ARTICLE INFO

Article history:
Received 1 June 2018
Received in revised form 31 August 2018
Accepted 2 September 2018
Available online 06 September 2018

Editor: D. Barcelo

Keywords:
Ecological state assessment
Morphology
Nutrients
Micropollutants
Restoration strategy
Environmental management

ABSTRACT

Freshwater ecosystems are increasingly under threat as they are confronted with multiple anthropogenic impairments. This calls for comprehensive management strategies to counteract, or even prevent, long-term impacts on habitats and their biodiversity, as well as on their ecological functions and services. The basis for the efficient management and effective conservation of any ecosystem is sufficient knowledge on the state of the system and its response to external influence factors. In freshwater ecosystems, state information is currently drawn from ecological assessments at the reach or site scale. While these assessments are essential, they are not sufficient to assess the expected outcome of different river restoration strategies, because they do not account for important characteristics of the whole river network, such as habitat connectivity or headwater reachability. This is of particular importance for the spatial prioritization of restoration measures. River restoration could be supported best by integrative catchment-scale ecological assessments that are sensitive to the spatial arrangement of river reaches and barriers. Assessments at this scale are of increasing interest to environmental managers and conservation practitioners to prioritize restoration measures or to locate areas worth protecting. We present an approach based on decision support methods that integrates abiotic and biotic ecological assessments at the reach-scale and aggregates them spatially to describe the ecological state of entire catchments. This aggregation is based on spatial criteria that represent important ecological catchment properties, such as fish migration potential, resilience, fragmentation and habitat diversity in a spatially explicit way. We identify the most promising assessment criteria from different alternatives based on theoretical considerations and a comparison with biological indicators. Potential applications are discussed, particularly for supporting the strategic, long-term planning and spatial prioritization of restoration measures.

1. Introduction

Multiple methods have been implemented to assess freshwater ecosystems in different regions of the world (Buss et al., 2015; Ghetti, 1980; Metcalfe, 1989). Some aim at determining the quality of the water in

* Corresponding author.
E-mail address: mathias.kuemmerlen@eawag.ch (M. Kuemmerlen).

https://doi.org/10.1016/j.scitotenv.2018.09.019

streams and rivers, while others focus on the ecological state (i.e. integrity) of river reaches and their ability to sustain biodiversity. All have in common that they seek to determine the current state of the ecosystem, as to set a baseline against which past or future states can be compared to. This supports the identification of management alternatives to amend or overcome current impairments.

Most freshwater ecosystem assessment methods evaluate local (i.e. sampling site) or reach-wide (i.e. short river segment) properties of rivers, with a focus on water quality, hydromorphology or biota, often including both aquatic and riparian habitats (e.g. Kamp et al., 2007; Wernersson et al., 2015). Such assessments are useful to detect possible deficits stemming from the local surroundings and may even help identifying additional or alternative sources of impairment elsewhere in the catchment (Hynes, 1975). Land-use in the upstream subcatchment, for instance has a strong effect on water chemistry (Johnson et al., 1997), local habitats and freshwater biodiversity (Allan, 2004; Kuemmerlen et al., 2014). Reach-scale freshwater assessments are an important tool regularly implemented to test for compliance with laws, both in the European Union (2000) and elsewhere (e.g. Switzerland; Bundi et al., 2000).

For the purpose of prioritizing river restoration measures, particularly in the process of systematic long-term environmental planning, reach-scale freshwater assessments are, however, not sufficient unless placed in the context of the river network. The assessment of single reaches does not allow evaluating the potential benefit of joint restoration measures at different locations of the same river system. For example, if several longitudinal migration impediments (e.g. weirs) are removed jointly in a river system, individual assessments of the success of these measures will not be meaningful if the longitudinal connectivity between them and the rest of the stream network is not taken into account as well (Lake et al., 2007). Therefore, freshwater ecosystem assessments should ideally account for network properties, such as habitat fragmentation and size, as well as connectivity. Moreover, the overall success of river restoration measures has been linked to biotic and abiotic conditions of the nearby stream network (Stoll et al., 2014), as well as the upstream subcatchment (Sundermann et al., 2011; Tonkin et al., 2014). Also, recolonization processes (Brederveld et al., 2011) and how these interact with the structure of river networks (e.g. Altermatt et al., 2013; Grant et al., 2007), play an important role for the dispersal and distribution of freshwater biodiversity, and need to be taken into account when assessing river restoration success. In summary, spatially-explicit approaches that consider the arrangement of impaired and unimpaired reaches, as well as instream barriers, are necessary for an effective, long-term management of ecosystems. In the case of freshwater ecosystems, the catchment scale has been identified and implemented as the fundamental unit of ecosystem management (European Union, 2000) and conservation planning (Saunders et al., 2002). Throughout this publication, we use the term catchment to describe any hydrological unit such as river basins, drainage areas or watersheds, regardless of their size.

Catchment-scale assessments enable the optimization of ecological restoration efforts, by facilitating the prioritization of restoration measures. This is of utmost importance, as funding may be a limiting factor and maximum efficiency desirable. Also, trade-offs between environmental goals and societal needs frequently need to be evaluated, as to find a balance between the respective stakeholders (e.g. restoration of a river reach vs. production of hydropower). In this context, the possibility to construct different strategies and scenarios can be of great value to find a consensus solution. This is highly relevant in a European context and elsewhere, as large stretches of rivers are currently being restored to improve their ecological state and to allow the longitudinal movement of species. Switzerland, in particular, has committed to restore 25% of all river reaches currently in a bad morphological state until 2090 (Göggel, 2012).

Beyond river restoration, there is an increasing need for assessing the ecological state of entire catchments. On the one hand, there is a general interest in the conservation of freshwater ecosystems (Hermoso et al., 2012; Howard et al., 2018), as undisturbed catchments become rare and suitable habitats for many species disappear (Dudgeon et al., 2006; Malmqvist and Rundle, 2002). On the other hand, ecosystem functions and services delivered by freshwater ecosystems attract attention as valuable assets which are at risk and require active management (Flotemersch et al., 2016; Culhane et al., in review; Teixeira et al., in review). Research on these topics necessarily implies an analysis at the catchment scale and leads to management strategies designed and implemented from a catchment-wide perspective (Saunders et al., 2002).

Despite the advantages, not many ecological assessment methods have been put forward to evaluate the ecological state of entire catchments. Existing approaches rely primarily on either remote sensing (e.g. land use) or measured environmental data (e.g. river morphology), or a combination of both (Foerster et al., 2017; Hill et al., 2017; Huang et al., 2010; Paukert et al., 2011; Thornbrugh et al., 2018). While these data sources provide properties of the stream network, which are of utmost importance to the freshwater ecosystem, none of the existing catchment assessments takes into account the spatial arrangement of these properties, in the context of the catchment to be analyzed. For instance, the significance of the particular position of a culverted reach or a tall barrier in the stream network is not trivial. A single one of these structures is sufficient to significantly disrupt multiple ecosystem processes and, in consequence, some of its functions and services.

We present a method for an integrative assessment of the ecological state of entire catchments. It builds on individual, ecological assessments of single reaches, as well as on barriers to longitudinal movement. Due to limited data availability, the reach-scale integrative assessment in our case study is estimated from river morphology, nutrient and micropollutant (i.e. anthropogenic chemical compounds at low concentrations; sensu Schwarzenbach et al., 2006) concentrations in a spatially explicit way. This setup is needed to complement the reach-scale assessment by a catchment-wide assessment which reflects the hierarchical structure of the stream network, and is intended to account for key ecological processes. The main application targeted here, is to assist in the spatial prioritization of river restoration at the catchment scale, in coordination with other management activities (e.g. improving water quality).

As the assessment is intended to be used in the context of decision making in river management, we formulate the “assessment scores” as it is done in multi-attribute value theory by a “value function” that characterizes the “degree of fulfillment” of the goal of “achieving a good ecological state” as a function of system attributes (on a scale from 0 to 1 or 0 to 100%; Dyer and Sarin, 1979; Keeney, 1996; Reichert et al., 2015). This multi-criteria decision support method facilitates the incorporation of scientific knowledge to support decision makers in finding and evaluating suitable management alternatives.

The approach presented here is based on, and significantly extends the one introduced by Reichert et al. (2015), which was exemplified for the Mönchaltdorfer Aa catchment, one of four catchments used here for development and illustration of the suggested approach. In the present study, we also build on different existing assessment modules of the Swiss Modular Concept for Stream Assessment (SMC; Bundi et al., 2000; Hütte and Niederhauser, 1998). However, compared to Reichert et al. (2015), we extend the method in several aspects: a) in addition to morphology and nutrient concentration, we also consider micropollutant concentration for the assessment of individual river reaches (see Section 2.3.3); b) we introduce spatial criteria for further ecological principles and test various versions to quantify them (see Sections 2.4.4 and 2.4.5); c) we apply the method to three additional, larger catchments; and d) we use indices of biological condition for benthic macroinvertebrates and fish for comparison with the reach and catchment scale ecological state assessments.

2. Material and methods

2.1. Study region

The study region is located in the Swiss Plateau, one of four biogeographical regions identified for the country (FOEN, 2001). This region is encompassed by the Alps in the south-east and the Jura Mountains in the north-west, and has a mostly flat to hilly topography with elevations between 250 and 1300 m.a.s.l. The favorable landscape is one of the major reasons this area concentrates most of the country's population, as well as most of its agricultural, industrial and service activities.

We focus on the catchments of four rivers in the Swiss Plateau, distributed relatively evenly on a southwest to northeast extent: the Broye, the Suhre, the Toess and the Mönchaltdorfer Aa. The three former rivers were selected because they have large catchment areas that lie mostly within the Swiss Plateau (Fig. 1). The latter river was selected as it had been used as an illustrating example in the aforementioned study (Reichert et al., 2015). Their catchments areas range from approx. 50 to over 850 km^2 and are part of the Rhine River basin, discharging to the North Sea.

2.2. Data collection

Information for this study was drawn from different data sources and various monitoring programs. Land use data, political boundaries and river networks were provided by the Swiss Federal Office of Topography (swisstopo, 2016). Data on the biogeographical regions (FOEN, 2001) and river morphology (FOEN, 2016) were delivered by the Federal Office for the Environment (FOEN). The biological and chemical monitoring data were obtained from the federal monitoring programs NAWA (FOEN, 2013; Kunz et al., 2016), BDM (BDM Coordination Office, 2009) and NADUF (Jakob et al., 2002), as well as from monitoring programs conducted by the environmental agencies of the cantons of Zürich, Aargau, Luzern, Fribourg and Vaud. Data collection followed standard procedures according to the respective SMC modules for nutrients (Liechti, 2010), fish (Schager and Peter, 2004) and benthic macroinvertebrates (Stucki, 2010). The macroinvertebrate data from the BDM and NAWA programs were accessed through the MIDAT database (http://www.cscf.ch/cscf/de/home/wissenschaftliche-aktivitaten/makrozoobenthos/datenbanken-midat.html, last accessed 13 July 2017). From the benthic macroinvertebrate monitoring data, we used two indices: the Swiss stream macroinvertebrate index (IBCH; Stucki, 2010) and the species at risk index for pesticides (SPEAR; Liess and Ohe, 2005).

2.3. Reach scale evaluation

The evaluation of the ecological state at the reach scale was based on the SMC (Bundi et al., 1998), which includes assessment modules for biological components (macroinvertebrates, fish, aquatic macrophytes, diatoms), as well as abiotic components (morphology, hydrology, nutrients, micropollutants; Fig. 2). The original assessment modules provide results in five discrete quality classes (in the case of morphology four classes), similar to the Water Framework Directive (European Union, 2000; Table 1).

To derive an overall assessment of the ecological state at the reach scale, in a way that can be easily integrated in a decision support process for management, we translated the existing assessment modules into so called “value functions” based on multi-attribute value theory (e.g. Fig. 3; see Langhans et al., 2013). To this end, we first grouped the original assessment modules into an objectives hierarchy, were each assessment module represents one of the sub-objectives (Fig. 2). We then translated the outcome of the assessment modules into numerical values between 0 and 1, which quantify the degree of fulfillment of the management objective (i.e. to which degree the management goal is met). Here, 0 means 0% fulfillment of the objective, which corresponds to the worst case scenario for Swiss rivers. Analogously, 1 means 100% fulfillment of the objective, which corresponds to reference conditions that describe the natural (or near-natural) state. Furthermore, we assume that the five quality classes are evenly distributed across the value scale between 0 and 1 (Table 1; see Supplementary material Fig. A1 for an example).

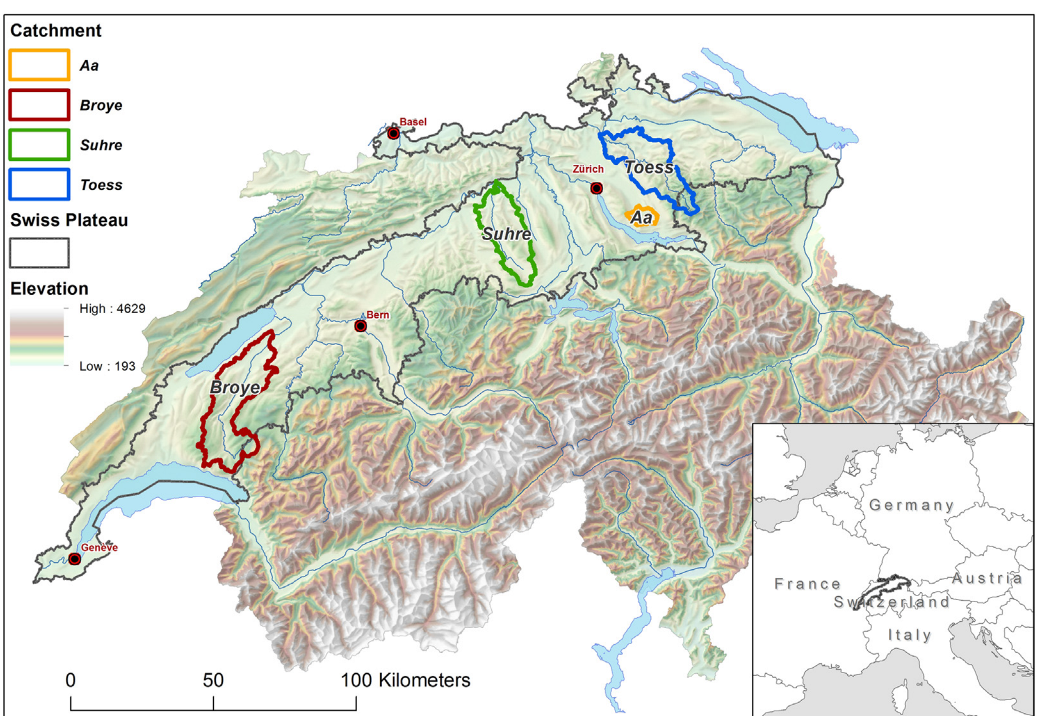


Fig. 1. Location of the Swiss Plateau within Switzerland and Europe (inset; ESRI© Esri Data world), with the catchments of the rivers Broye, Suhre, Mönchaltdorfer Aa and Toess (swisstopo, 2016).

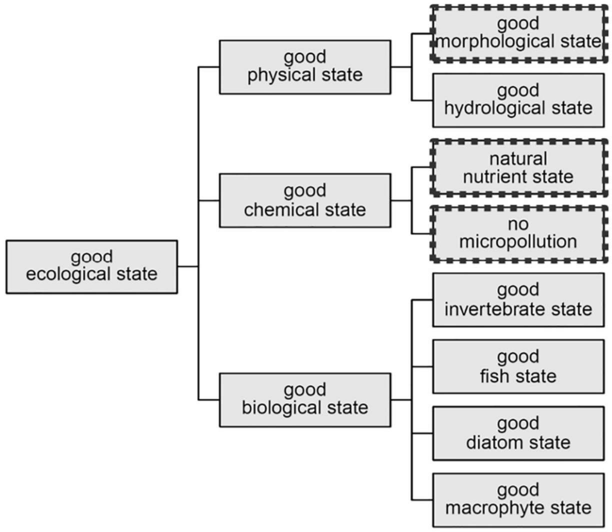


Fig. 2. Objectives hierarchy to describe the ecological state at the reach scale. Each of the lowest level objectives (boxes at the right) is related to an assessment module from the Swiss Modular Concept for Stream Assessment (SMC). Dashed boxes represent sub-objectives (i.e. assessment modules) for which data were available and which were included in this study.

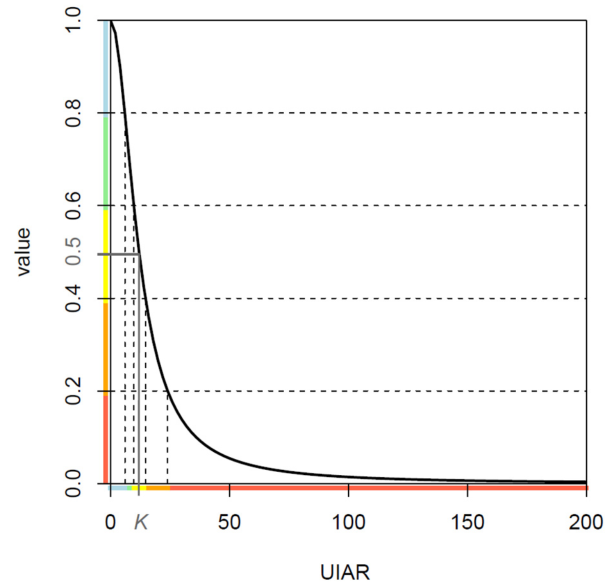


Fig. 3. Value function for the micropollutant state.

In the following, we distinguish between "good" reaches that fulfill the requirements and "bad" reaches that do not fulfill the requirements (note that the threshold between "good" and "bad" is at a value of 0.6; Table 1). The values of the assessment modules were then hierarchically aggregated to an assessment of the physical, chemical, biological state and ecological state (Fig. 2), as described below. In this study, we used three of the reach-scale assessment modules only (morphology, nutrients and micropollutants; Fig. 2), for which we were able to get or estimate reach-scale assessments for every reach in the stream network of all four catchments (as described below). The value function for the objectives hierarchy was implemented in R (R Core Team, 2017) using the R packages *utility* (Reichert et al., 2013) and *ecoval* (https://CRAN.R-project.org/package=ecoval).

2.3.1. Morphological state

The standardized morphological assessment has been carried out in much of the stream network of Switzerland, covering all rivers of order 3 and above and using the morphology module of the SMC (Hütte and Niederhauser, 1998). In some areas (e.g. Canton of Zurich), also smaller streams have been surveyed. The assessment is composed of 13 morphological properties of a river channel, that describe the width variability, modification of the riverbed, modification of embankments and condition of the riparian strips. These properties are aggregated to an overall morphological assessment (Hütte and Niederhauser, 1998) and translated into a value function following the method described by Langhans et al. (2013). For small order streams, where river morphology data were missing, we assumed a high morphological state as headwaters are often in near-natural, wooded areas. In the case of missing data on culverts for a specific reach, an absence of culverts was assigned.

Table 1
Color-coded quality classes of the original assessment modules and their translation to a continuous value scale between 0 and 1.

Quality class	Value scale	Legal requirements
Bad	0–< 0.2	Not fulfilled
Poor	0.2–< 0.4	
Moderate	0.4–< 0.6	
Good	0.6–< 0.8	Fulfilled
High	0.8–1	

2.3.2. Nutrient state

The nutrient assessment is based on the nutrient module of the SMC (Liechti, 2010), which evaluates the chemical state regarding orthophosphate (PO_4), nitrate (NO_3), nitrite (NO_2), ammonium (NH_4) and dissolved organic carbon (DOC; see Supplementary material Fig. A2). Additional optional nutrient parameters are total phosphorus (TP), total filtered phosphorus (TP filt.), total nitrogen (TN), total organic carbon (TOC) and biochemical oxygen demand (BOD_5). The assessment of the single parameters was translated into a value function and then aggregated using equal weights for each water quality parameter and an additive-minimum aggregation with equal contributions of the minimum and the arithmetic mean (Langhans et al., 2014).

Because the nutrient assessment was available for only some sites and not for every reach in the catchment, we developed a simple linear regression model to estimate the nutrient assessment for every reach based on land-use data and the fraction of treated wastewater (proportion of arable land, fraction of wastewater treatment plant effluents in river discharge volume and number of livestock units in the catchment; see Supplementary material A3). The model fitted with these variables had shown the highest explanatory power among all candidate models with all possible combinations of 91 candidate explanatory variables related to anthropogenic land use. The explanatory variables were extracted from various Swiss datasets (FOEN, 2014; FSO, 2008; swisstopo, 2010; swisstopo, 2016) and processed in ArcGIS (ESRI, Redlands, CA, 2016), as well as R (R Core Team, 2017).

The model assumed a linear effect of nutrient accumulation across the landscape from agriculture, cattle farming and disposal of treated wastewater. Preliminarily, we tested whether models calibrated for the entire Swiss Plateau and fitted to the individual catchments performed worse than those calibrated for and fitted to the individual catchments (see Supplementary material Fig. A3). The difference in model performance was negligible, which supported calibrating the model for the prediction of the nutrient state on all available data of the Swiss Plateau (mean correlation coefficient between observed and predicted nutrients for the four catchments = 0.579; correlation coefficient between observed and predicted nutrients for the Swiss Plateau = 0.648). Calibrating a single model for all catchments assures more transparency for replication and facilitates expanding the scope of this approach to other catchments in the Swiss Plateau not yet considered here. We extracted the coefficients from this model and applied them to predict the nutrient assessment for all stream reaches of the four catchments. Due to GIS processing, some reaches lacked the land-use data necessary for nutrient concentration prediction, in which case data from the next downstream reach were used.

2.3.3. Micropollutant state

A micropollutant module for the SMC is currently under development and sufficient micropollutant data are not yet available for catchment scale assessments. Therefore, an index based on estimated pesticide application rates by land use type (UIAR; Vermeiren et al., submitted) was used as a rough proxy for the micropollutant state, which quantifies the proportion of different crops in the catchment weighted by their average yearly number of crop-specific insecticide applications (Spycher et al., 2015) as well as micropollutant contributions from urban sources based on the urban area in the catchment (see Supplementary material A4). To translate the UIAR into a micropollutant assessment, we chose as value function the equation $v = \frac{K^2}{K^2+UIAR^2}$ based on the reasoning that the assessment should get worse with increasing UIAR and using the median UIAR of the reaches in the four catchments at the Swiss plateau as half-saturation constant K to assure that a bad state is reached within a typical range of values (Fig. 3; AWEL, 2012), because its known that intense agriculture and urban areas can lead to a bad micropollutant state in the Swiss Plateau. The proxy used here should be replaced by the SMC micropollutant assessment as soon as it becomes available.

Also in this case, some reaches lacked the land-use data necessary for the estimation of the micropollutant assessment due to GIS processing, in which cases data were drawn either from the next downstream reach if it contained the required data, or alternatively, from the upstream subcatchment (defined as portion of the catchment upstream of the reach or site of interest).

2.3.4. Aggregation of sub-objectives at reach scale

To quantify the degree of fulfillment of the higher-level objectives (physical, chemical, biological and ecological state, Fig. 2), the numerical values of the corresponding sub-objectives had to be aggregated. This can be done by different ways of averaging, while different aggregation methods have different properties (Langhans et al., 2014). The weighted arithmetic mean (also called additive aggregation) is the most commonly used aggregation method in the field of multi criteria decision support. However, it allows a compensation of a bad value of one sub-objective with a high value of another sub-objective, which does not seem appropriate for the aggregation of complementary sub-objectives in ecological assessments. For this reason, the minimum aggregation is often applied in ecological assessments (also called worst-case or one out-all out principle). However, this has the often undesired property, that it is only sensitive to changes in the worst of the aggregated values. A recent study confirmed that river managers favor a compromise between these two extremes (Haag et al., in press). We therefore applied the so-called additive-minimum aggregation, where an average of both methods is taken. We chose equal weights for the different sub-objectives and an equal contribution of the minimum and the arithmetic mean. The result is a reach-scale assessment of the ecological state of every reach in the stream network, reflecting the conditions given by river morphology, as well as its estimated nutrient and micropollutant concentrations (Fig. 4).

2.4. Spatial criteria for catchment scale evaluation

To prioritize restoration and find optimal trade-offs between conservation, restoration, and utilization (e.g. for hydropower production) of freshwater ecosystems, we need an ecological valuation at the catchment scale that builds on the reach-scale ecological valuation and describes how the spatial arrangement of reaches in various ecological states affects the ecological state of the entire catchment. To this end, we developed an objective hierarchy (see Fig. 5) composed of five sub-objectives at the intermediate level, each one representing a different approach to summarize the assessment of all reaches, which can be linked to an ecological principle. These objectives represent important goals in achieving a good ecological state of a river system. These goals, or objectives, are also highly relevant to the Swiss river restoration program and to catchment management initiatives in general (Göggel, 2012; Könitzer et al., 2012). Aggregated into the highest level objective (Fig. 5, left side), the sub-objectives help to describe the ecological state of the catchment, by considering the ecological state of each reach, as well as their spatial arrangement in the stream network. Moreover, some of the objectives consider the existence of natural and artificial migration barriers (e.g. drops and weirs) and their location in the stream network. Because they are all spatially explicit, in the following we refer to them as spatial criteria. For each of them, we propose a measurable attribute to quantify the fulfillment of the objective and, in a second step, define how the fulfillment of the objective depends on the attribute level. We used the R package *rivernet* (https://CRAN.R-project.org/package=ecoval) to read, analyze and plot the spatially explicit data of individual reaches and nodes, as well as their properties (e.g. river morphology, presence of a barrier) in a stream network context representing each one of the four catchments taken into account in this study. Subsequently, the R package *utility* (Reichert et al., 2013), was used to evaluate the value functions and plot the objectives hierarchies.

The five spatial criteria outlined below represent a range of different ecological properties and catchment possesses: a) the mean ecological state, b) the longitudinal connectivity, c) the resilience potential, d) the network fragmentation, and e) the habitat diversity. These spatial criteria are not an exhaustive list, as there are many alternative possibilities of aggregating and representing ecologically relevant information for an entire catchment. Further, each spatial criterion described here can be implemented using one, or more properties of the river reaches, in different combinations (see four examples in Fig. 5). Several such versions (n = 32; see list in Table 3) are tested for some of the spatial criteria presented here in order to identify the most appropriate one for the aggregation at the catchment scale. The spatial criteria and their versions were conceived from a combination of both the objectives pursued by the Swiss river restoration initiative and data availability. All suggested criteria had to fulfill the condition that any implemented restoration measure improving the state of a single reach (regardless of its length) or removing an artificial barrier, would not deteriorate the ecological state of the entire catchment (not even minimally). In addition, each restoration of a reach or removal of an artificial barrier had to improve the overall ecological state (small measures only minimally). Also, the overall ecological state had to reach a value of one if all reaches had an ecological state of value one (high ecological state) and there are no artificial barriers, or a value close to zero if all reaches are in a bad state and/or there are many artificial barriers.

For each of the above-mentioned criteria, we propose different implementation alternatives and quantify them with attributes that usually have a range from zero to one. Each of these attributes has subsequently to be transformed into a value (again from zero to one) that represents the degree of fulfillment of the ecological objective to be quantified with the attribute. The development of such value functions is usually done in a joint effort of scientists and stakeholders, as it was the case for the reach-scale assessment. As our spatial criteria are new, there is not sufficient experience available to derive such a value function. For this reason, we often use linear value functions that are preliminary approximations to value functions that may be refined later, if the suggested approach is implemented in practice.

2.4.1. Good ecological state of river reaches

In the context of a landscape, the basic ecological unit of freshwater ecosystems is the catchment (Hynes, 1975). Hence, the fate of these ecosystems, their resources (i.e. water) and their biodiversity, is defined by the properties of the catchment (Allan, 2004; Baron et al., 2002). One

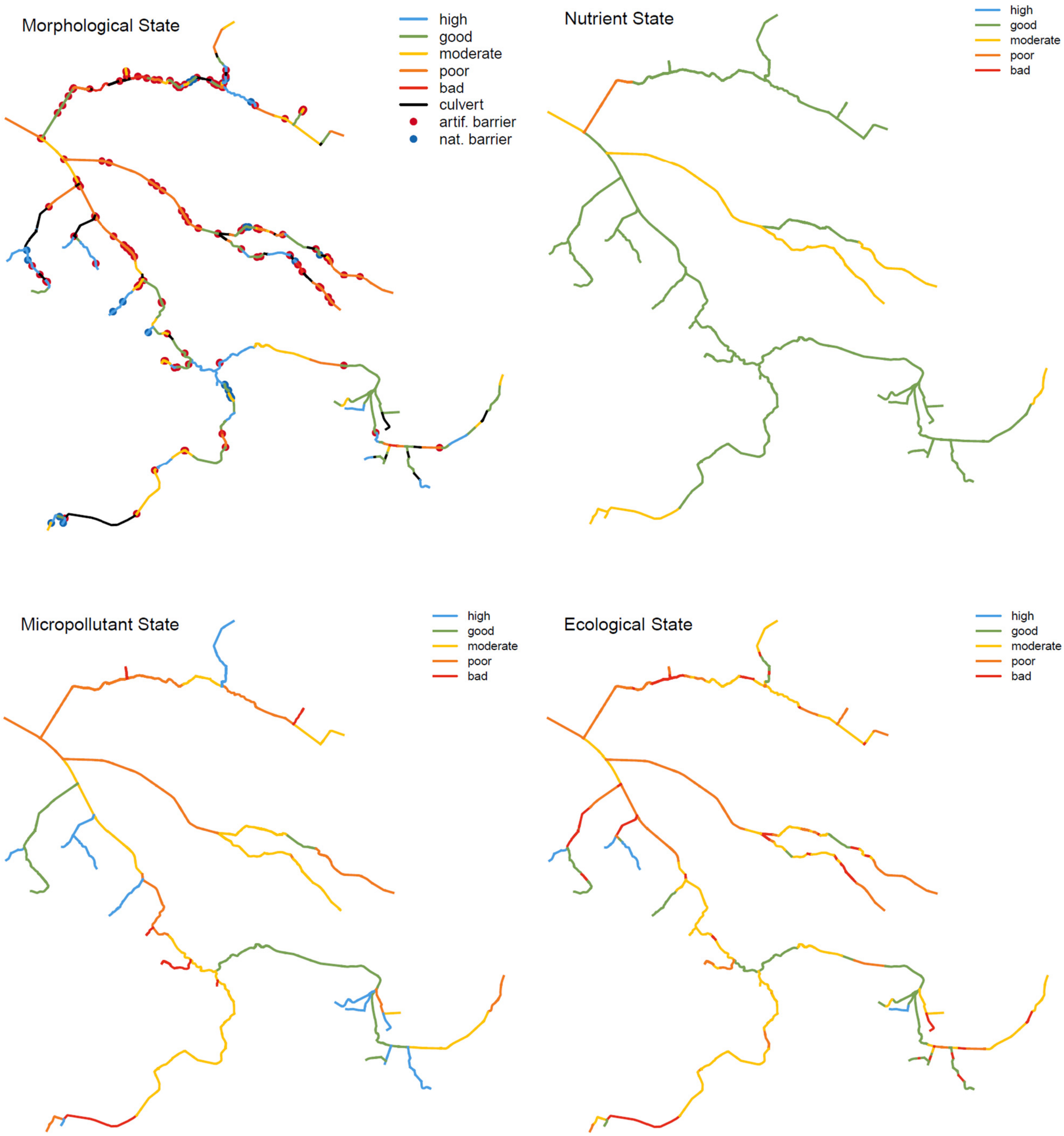


Fig. 4. Stream network of a subcatchment of the Mönchaltorfer Aa, showing reach scale assessments for the morphological, nutrient, micropollutant and ecological state.

fundamental management objective is a good ecological state of all reaches in the catchment, independent of their spatial arrangement. Thus, the mean (i.e. average) state across all reaches and the fraction of reaches in good state are two complementary ways of quantifying this objective.

2.4.1.1. Good mean state of reaches. We suggest aggregating the ecological assessment of all the reaches in the catchment through a weighted average. Weights are assigned to each reach in terms of their length and stream order (Strahler, 1957). The purpose of weighting by stream order is to increase the relative importance of reaches with high stream order because: these are proportionally underrepresented in a river network in terms of stream network length; b) they play an important role in connecting river sections of smaller stream order; and c) the habitat area they offer is proportional to their width, which increases with stream order, but which is not considered here explicitly.

Therefore, the fulfillment of the objective "good mean state of reaches" v_{MES} can be measured by the weighted mean ecological state of the reaches a_{MES}, which can be calculated as

$$a_{MES} = \frac{\sum_i l_i o_i v_{ES\,i}}{\sum_i l_i o_i}$$

where l_i is the length, o_i the stream order and $v_{ES\,i}$ the ecological state of reach i.

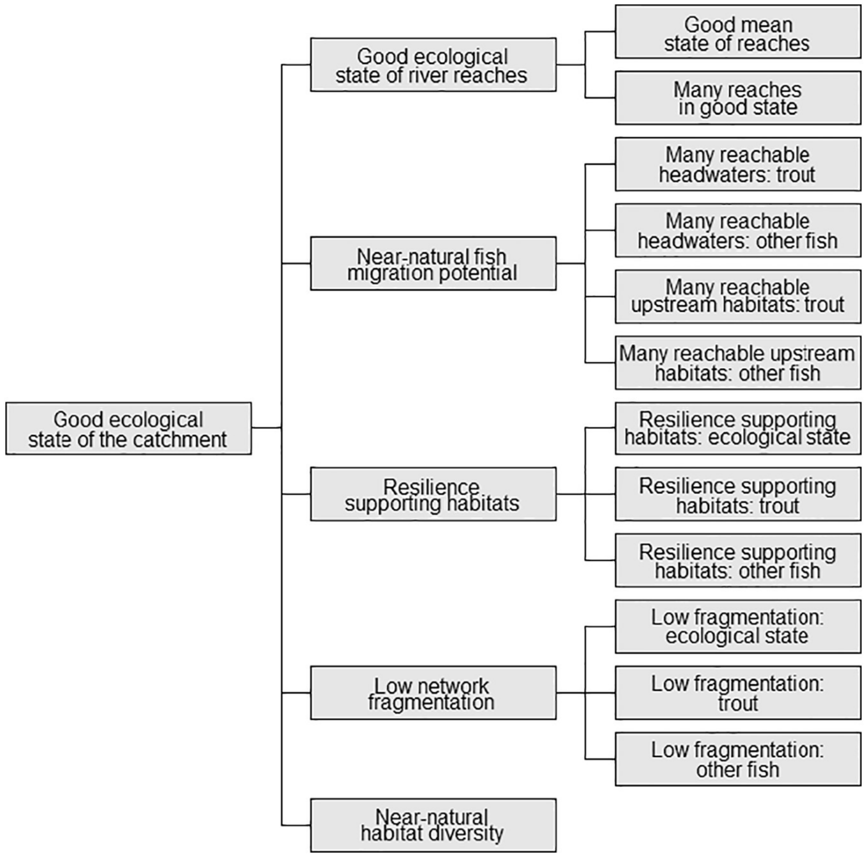


Fig. 5. Objectives hierarchy for the ecological assessment at the catchment scale.

As mentioned above, for our preliminary assessment, we assume a linear dependence of the value for the objective "good mean state of reaches" v_{MES} on the attribute a_{MES} with $v_{MES} = a_{MES}$ (See Fig. 6, upper left panel for an example).

2.4.1.2. Many reaches in good state. This sub-objective was inspired by freshwater policies regulating the ecological state of rivers in European countries (Bundi et al., 2000; European Union, 2000). According to these policies, legal thresholds generally have to be met in all river reaches and restoration measures have to be implemented where rivers are not in a good ecological state (classes bad, poor & moderate), with the exception of some socioeconomic or environmental constraints (e.g. groundwater recharge zone, presence of infrastructure, presence of endangered species population etc.). We therefore quantify the sub-objective "many reaches in good state" v_{FGS} by measuring the fraction of the river network that is in a good ecological state, a_{FGS}, where the river sections are again weighted by their length and stream order

$$a_{FGS} = \frac{\sum_i l_i o_i y_{ES\ i}}{\sum_i l_i o_i}$$

where l_i is the length, o_i the stream order of reach i, and $y_{ES\ i}$ is 0 if the ecological state of reach i is <0.6 (see Table 1 and related text for the justification of this threshold) and 1 otherwise.

Again, we assume a linear dependence of the value for the objective "many reaches in good state" v_{FGS} on the attribute a_{FGS} with $v_{FGS} = a_{FGS}$.

2.4.2. Near-natural fish migration potential

Longitudinal connectivity is an inherent property of freshwater ecosystems that plays an important role in structuring biotic communities (Cooper et al., 2017; Liermann et al., 2012), as well as in many ecological processes (e.g. migration, dispersal; Malmqvist, 2002). When this connectivity is interrupted, the biotic and abiotic conditions of the catchment can change significantly, in extreme cases even leading to speciation events (e.g. Gravena et al., 2015). However, many instream structures in non-mountainous regions are man-made and continue to be built in most river systems, with the consequence of partially, or even completely interrupting the natural connectivity of the stream network. The interruption of migration pathways through artificial barriers is a serious threat to species with this reproduction strategy and has been shown to cause genetic variation in migrating fish populations above and below such barriers (Limburg and Waldman, 2009; Wofford et al., 2005).

A catchment in a good ecological state should have a longitudinal connectivity as close as possible to the natural state. We propose quantifying the fulfillment of the objective "near-natural fish migration potential" v_{FMP} with several sub-objectives that a) consider different (groups of) fish species, which are important in the catchment to be assessed and may differ regarding their migration abilities, and b) either quantify the fraction of reachable headwaters from the catchment outlet, or the fraction of reachable upstream habitats, irrespective of whether they are considered headwaters or not.

The fraction of reachable headwaters a_{FRH} can be calculated as

$$a_{FRH} = \frac{n_{RH}}{n_{NRH}}$$

where n_{RH} is the number of headwaters that are reachable from the catchment outlet in the assessed state and n_{NRH} is the number of headwaters that would be reachable from the catchment outlet under natural conditions. With this formulation, natural barriers to fish migration

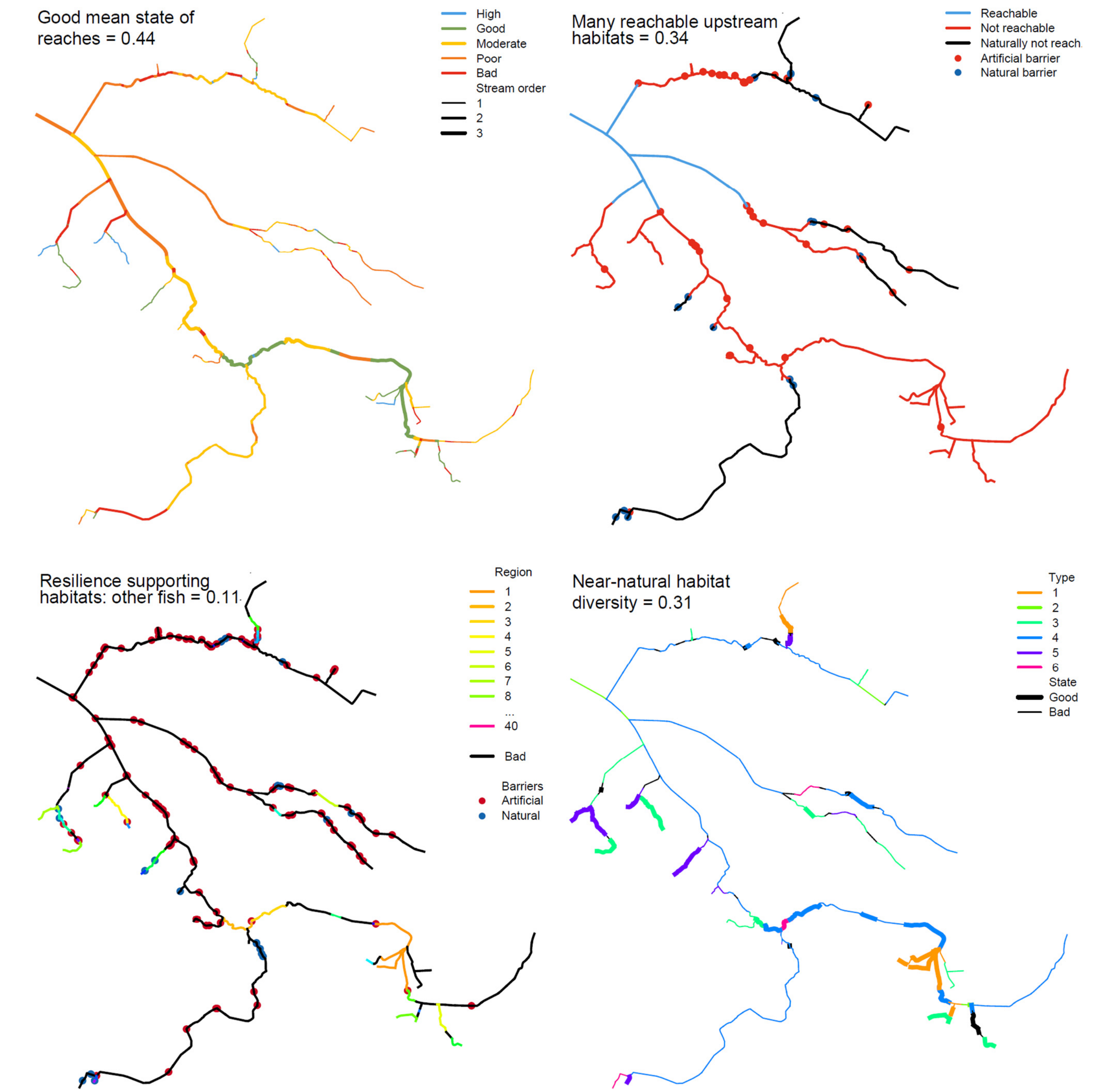


Fig. 6. A subcatchment of the Mönchaltorfer Aa with mapped properties of the stream network used to illustrate some of the spatial criteria for a catchment-scale assessment. Numbers indicate values of fulfillment of the objective on a scale from 0 to 1.

are taken into account. We define "headwaters" by stream order, o_{hw}, which can be adapted to the fish species as described below.

Similarly, the fraction of reachable upstream habitats a_{FRU} can be calculated as

$$a_{FRU} = \frac{l_{RU}}{l_{NRU}}$$

where l_{RU} is the length of the stream network that is reachable from the catchment outlet and l_{NRU} is the length of the stream network that would be reachable from the catchment outlet under natural conditions.

We explored several versions to define impediments to fish migration: a) transverse barriers with a height larger than a critical threshold h_{FRH}, b) culverts which are longer than a critical distance d_{FRHc}, and c) reaches in bad ecological state (e.g. due to low water quality or bad morphology), if they are longer than a critical distance d_{FRHs}. In addition, we considered a combination of a) and b), as well as of a), b) and c) (see full list of versions in Table 3). All parameters mentioned above have to be adapted to the (groups of) fish species that are considered in the sub-objectives (Fig. 6).

For our case study, we distinguish between trout and other fish, which are considered to have a lower ability to overcome barriers than trout. For the critical height, h_{FRH}, we chose 50 cm for trout and 10 cm for other fish. For the critical distance of culverts and reaches in bad state we chose 50 m for all fish species. For the definition of "headwaters" by stream order, o_{hw}, we chose a stream order of 2. We assume a linear dependence of the value for the sub-objectives for "near-natural fish migration potential" on the attributes a_{FRH} with $v_{FMP} = a_{FRH}$ (see comment at the end of the introduction to Section 2.4, and Fig. 6, upper left panel for an illustration).

2.4.3. Resilience supporting habitats

Short-term disturbances are common in natural systems and are one of the processes that help regulate ecological dynamics in river systems (McCluney et al., 2014; Resh et al., 1988). However, human activities can cause serious additional disturbances, which can lead to local extinctions of vulnerable organisms. Examples are activities such as local pollution events, hydropower generation that lead to direct effects on hydrological conditions, or carbon dioxide emissions that affect hydrological conditions via climate change. To assure the resilience of vulnerable freshwater biodiversity to any type of disturbance, refugia are likely to play an important role (Bisson et al., 2009). We assume that the resilience of ecosystems increases with the size of river fragments of adjacent stream reaches that are in a good ecological state, due to a larger probability of providing refugia to self-sustaining populations, which can act as sources for recolonization elsewhere in a catchment.

The fulfillment of the objective “resilience supporting habitats” v_{RSH} can be measured by the adjacency of good habitats a_{AH}, which can be calculated as

$$a_{AH} = \frac{\sum_j \frac{1}{r_{j\ \mathrm{good}}} l_{j\ \mathrm{good}}}{\sum_k \frac{1}{r_k} l_k}$$

where $l_{j\ \mathrm{good}}$ is the length of the stream fragment j that is in a good state (defined as adjacent reaches that are in a good state) and $r_{j,\ \mathrm{good}}$ is the ranking of these stream fragments by length (the largest fragment has rank 1, the second largest rank 2, etc.). Similarly, l_k is the length of the stream fragment k under natural conditions and r_k is the ranking of these stream fragments by length. Similar to the fish migration potential, we explored several versions that differ in the way these fragments are considered to be divided from each other (through reaches in a bad ecological state, culverts, artificial barriers, and combinations of these, with parameters to be adapted to the organism group, see Section 2.4.2). For the versions where just reaches in bad ecological state or culverts are considered, under natural conditions, the whole river network would consist of just one single “fragment” with the total length of the river network. However, in versions that consider fragmentation by barriers, we account for natural barriers that divide the river network into multiple fragments also under natural conditions. In these versions we cannot just divide by the total length of the river network in the equation given above but use the adjacency under natural conditions as benchmark.

The eight versions considered in the previous spatial criterion (near-natural fish migration potential) were applied here, along with the same parameters. We assume a linear dependence of the value for the objective “resilience supporting habitats” v_{RSH} on the attribute a_{AH} with $v_{RSH} = a_{AH}$ (see comment at the end of the introduction to Section 2.4, and Fig. 6, lower left panel for an example).

2.4.4. Low network fragmentation

Dispersal between suitable habitats in a catchment is crucial to recolonize reaches that have been impacted by natural (e.g. flooding) or anthropogenic disturbances (e.g. chemical spills, Lake, 2000), with the probability of successful recolonization being strongly related to the dispersal distance (Wiens, 2002). For the success of river restoration efforts, dispersal distance has been proven to be a crucial aspect, as the presence of upstream source populations has been suggested to lie within 5 km, ideally 1 km (Sundermann et al., 2011; Tonkin et al., 2014). To quantify the objective of low network fragmentation, we use a similar mathematical formulation as for the criterion resilience supporting habitats, measuring the fulfillment of the objective “low network fragmentation” v_{LNF} by the distance of fragment isolation a_{DFI}. For this purpose, we changed the critical distance d_{FRHs} from 50 m to 2000 m, while keeping the critical height h_{FRH} and the critical distance d_{FRHc}, as in previous criteria. The critical distance can be changed depending on the management objective or organism group being studied.

Also for this spatial criterion, we applied the same eight versions from the two previous spatial criteria (near-natural fish migration potential & resilience supporting habitats). We again assume a linear dependence of the value for the objective “low network fragmentation” v_{LNF} on the attribute a_{DFI}, with $v_{LNF} = a_{DFI}$ (see comment at the end of the introduction to Section 2.4).

2.4.5. Near-natural habitat diversity

Catchments are characterized by the unique mosaic of habitats they are composed of (Thorp et al., 2006). This habitat diversity is at least partially responsible for the biodiversity found in the catchment, the functions it performs and the services it offers (McCluney et al., 2014). The higher the habitat diversity in a catchment, the higher the total number of species to be expected (Cardinale et al., 2002). As a proxy for different freshwater habitats, we use the Swiss river typology, based on topographical, geological and climatic conditions (Schaffner et al., 2013) assuming that different river types offer different habitats. We propose to measure the fulfillment of the objective “near-natural habitat diversity” v_{HD} by calculating the mean of the river-type specific fractions of river lengths in a good ecological state

$$a_{RTF} = \frac{1}{n} \sum_{t=1}^{n} \frac{l_{t\ \mathrm{good}}}{l_t}$$

where n is the number of different river types represented in the catchment, $l_{t\ \mathrm{good}}$ is the total length of river reaches of river type t which are in a good or high ecological state and l_t is the total length of river reaches of river type t.

We assume linear dependence of the value for the objective “near-natural habitat diversity” v_{HD} on the attribute a_{RTF} with $v_{HD} = a_{RTF}$ (see comment at the end of the introduction to Section 2.4, and Fig. 6, lower right panel for an example).

2.4.6. Aggregation of sub-objectives at the catchment scale

To obtain the ecological state of the entire catchment, the spatial criteria and the selected versions are aggregated into an overall assessment (Fig. 5). Similar to the aggregation of modules on the reach scale, we use an additive-minimum aggregation with equal weights of the sub-objectives and an equal contribution of the arithmetic mean and the minimum. However, these weights can be modified if, for a specific management question in a given catchment, the different sub-objectives have different importance.

2.5. Data analysis

We compared the reach scale evaluations of morphology, nutrients and micropollutants, the aggregated chemical state and the aggregated ecological state based on these three modules with three biotic indices, two based on stream macroinvertebrates: the Swiss stream macroinvertebrate index (IBCH; Stucki, 2010), as well as the species at risk index for pesticides (SPEAR; Liess and Ohe, 2005); and the SMC assessment for fish (Schager and Peter, 2004). Biological data were available only for some of the reaches (stream macroinvertebrates between 182 and 154 sites; fish 23 sites) and in some cases, data were available from multiple monitoring events, in which case the index values were averaged. The comparison was intended to explore at the reach scale, whether the ecological state assessment based on abiotic factors is reflected in the biological assessments. We calculated Spearman's correlation coefficients for complete observation pairs. Furthermore, linear regression models were fitted with the biotic indices as the response variables and the reach scale assessments mentioned above as the explanatory variables; the coefficient of determination (R^2), the test significance (p-value), the slope and the intercept were extracted.

Furthermore, we compared the three selected biotic indices to the ecological state assessment for selected subcatchments, implementing the same tests and extracting the same data as for the reach scale. Here we refer to subcatchments as smaller hydrological units within the four assessed catchments, which are defined by sites of interest at their outlet (e.g. a biological sampling site). The spatial criterion 'near-natural fish migration potential' was compared to the fish index, as it is most relevant for this taxonomic group. All other criteria were compared to the IBCH index as it had the most sites and data available and was intended to reflect morphological and chemical impairments. In cases where subcatchments contained more than one monitoring site, we averaged the indices. Indices from repeated monitoring events were averaged as well, if present. The objective of the catchment scale comparison was to explore if the different versions of the spatial criteria differ in regard to their relation to the biotic indices to support the selection of some of them. For this comparison we computed catchment scale assessments for different subcatchments. Subcatchments were chosen following two different strategies: a) manually selected subcatchments of relatively similar size (n = 42; M. Aa = 6, Broye = 12, Suhre = 9, Toess = 15), for which we had at least one monitoring data point; b) subcatchments upstream of each of the 182 stream macroinvertebrate monitoring sites (M. Aa = 8, Broye = 93, Suhre = 45, Toess = 36; some sites with repeated observations).

We derive a recommendation for a selection of the versions for each of the spatial criteria based on expert opinion and on their comparison to the biotic indices. Basis for the recommendations were the suitability of each variant to match the ecological principle it was expected to represent and low redundancy with other spatial criteria. If results from statistical results were available, the recommendation was complemented based on: a) the significance of the test (p-value); b) the level to which variance was explained by each model (R^2); and c) the direction of the effect (slope).

All versions chosen to represent each spatial criterion were correlated against each other (Spearman's correlation) based on the manually selected subcatchments and those defined by monitoring sites. The goal was to identify possible redundancy or complementarity between the versions. All further statistical analyses were done using the R Base Package (R Core Team, 2017).

Finally, we evaluated the ecological state of the four study catchments based on the chosen versions for each spatial criteria and compare the assessments for the entire catchment, as well as of each spatial criterion.

3. Results and discussion

3.1. Reach scale evaluation

Reach-scale assessments of the ecological state showed significant and positive correlations with the two macroinvertebrate indices (Table 2). The IBCH index was significantly correlated with the chemical assessments (chemical, micropollutant, and nutrient state) in the same reach where the macroinvertebrates were sampled and to a lesser degree with the ecological state based on morphology, nutrients and micropollutants (Table 2). The IBCH was not correlated with the morphological state. For the SPEAR index all correlations were significant, but the correlation was markedly stronger for the chemical state and its components, nutrients and micropollutants, than for morphology. For the SMC assessment for fish we found no significant correlation (but note the low sample size).

3.2. Catchment scale evaluation

Manually selected subcatchments were more homogeneous in terms of river network length (min. = 17.2 km, mean = 64.9 km, max. = 309.9 km) than subcatchments defined by biotic monitoring sites (min. = 0.1 km, mean = 96.0 km, max. = 924.2 km).

Out of all the versions of the spatial criterion 'near-natural fish migration potential', we found one of them to be significantly correlated with the SMC fish assessment in the manually chosen subcatchments: 'many reachable headwaters: other fish' (Tables 3, A2, but note the low sample sizes). Both sub-objectives of the spatial criterion 'good mean ecological state' have a similar strong and positive correlation to the IBCH index. For the spatial criterion 'resilience supporting habitats', several versions showed significant correlation with the IBCH, particularly those that only considered reaches in bad state as separating elements or those that considered reaches in bad state in combination with barriers. Also for the spatial criterion 'low network fragmentation', the versions based on the reaches with bad state alone and in combination with barriers, showed a significant correlation with the IBCH. The spatial criterion 'near-natural habitat diversity' was significantly correlated to the IBCH index. A similar picture is seen when comparing the biotic indices with the evaluation of the subcatchments delineated by biotic monitoring sites (Table A2). In this comparison the same versions

Table 2
Statistics on the relationship between three biotic indices (IBCH, SPEAR, fish assessment) and reach scale estimates of abiotic assessments (morphology, nutrients, micropollutants) and aggregated assessments for higher levels of the objectives hierarchy (chemical and ecological state, grey color). Significant correlations (p-value < 0.05) are highlighted in bold.

Biotic index	Reach objectives	n	Correlation	R^2	p–Value	Slope	Intercept
Mean **IBCH** (stream macroinvertebrates)	**Ecological state**	**193**	**0.397**	**0.131**	**0.000**	**5.72**	**9.83**
	Chemical state	**193**	**0.583**	**0.316**	**0.000**	**10.22**	**7.51**
	Morphology	154	0.155	0.011	0.054	1.07	12.23
	Nutrients	**192**	**0.472**	**0.237**	**0.000**	**12.65**	**5.85**
	Micropollutants	**193**	**0.513**	**0.249**	**0.000**	**5.68**	**9.58**
Mean **SPEAR** (stream macroinvertebrates)	**Ecological state**	**193**	**0.560**	**0.282**	**0.000**	**35.79**	**23.92**
	Chemical state	**193**	**0.702**	**0.469**	**0.000**	**53.04**	**15.30**
	Morphology	**154**	**0.293**	**0.087**	**0.000**	**13.04**	**34.69**
	Nutrients	**192**	**0.620**	**0.365**	**0.000**	**66.94**	**5.91**
	Micropollutants	**193**	**0.629**	**0.383**	**0.000**	**30.08**	**25.69**
Mean **Fish** stream assessment	Ecological state	23	0.099	0.014	0.654	1.07	1.33
	Chemical state	23	−0.044	0.019	0.843	−0.89	2.08
	Morphology	22	−0.220	0.028	0.325	−0.83	1.87
	Nutrients	23	−0.343	0.138	0.109	−4.02	3.96
	Micropollutants	23	0.027	0.013	0.902	−0.52	1.91

Table 3
Statistical test results on the relationship between one of two biotic indices (IBCH or fish assessment) and catchment scale assessments of five objectives and their different sub-objectives (versions), at 42 selected subcatchments of the Toess, Mönchaltdorfer Aa, Suhre and Broye catchments. Recommended versions for each criterion are shown with grey background, significant correlations (p-value < 0.05) in bold font.

Biotic index	Objective	Sub-Objective	Version	n	r	R^2	p-value	Slope	Intercept
Mean Fish Stream Assessment	Near-natural fish migration potential	Many reachable headwaters: all fish	Ecological state	11	NA	NA	NA	NA	NA
		Many reachable headwaters: all fish	Culverts	11	0.22	0.03	0.523	0.64	1.43
		Many reachable headwaters: trout	Large barriers	11	0.50	0.00	0.172	0.09	1.76
		Many reachable headwaters: trout	Large barriers & ecological state	11	NA	NA	NA	NA	NA
		Many reachable headwaters: trout	Large barriers & culverts	11	0.57	0.01	0.113	0.30	1.72
		Many reachable headwaters: other fish	All barriers	11	0.31	0.00	0.459	-0.68	1.91
		Many reachable headwaters: other fish	All barriers & ecological state	11	NA	NA	NA	NA	NA
		Many reachable headwaters: other fish	**All barriers & culverts**	**11**	**0.79**	**0.42**	**0.021**	**20.45**	**1.54**
		Many reachable upstream habitats: all fish	Ecological state	11	NA	NA	NA	NA	NA
		Many reachable upstream habitats: all fish	Culverts	11	0.42	0.16	0.196	1.27	1.02
		Many reachable upstream habitats: trout	Large barriers	11	0.15	0.01	0.664	-0.20	1.70
		Many reachable upstream habitats: trout	Large barriers & ecological state	11	NA	NA	NA	NA	NA
		Many reachable upstream habitats: trout	Large barriers & culverts	11	0.22	0.00	0.512	-0.13	1.68
		Many reachable upstream habitats: other fish	All barriers	11	-0.14	0.01	0.676	-0.72	1.73
		Many reachable upstream habitats: other fish	All barriers & ecological state	11	NA	NA	NA	NA	NA
		Many reachable upstream habitats: other fish	All barriers & culverts	11	-0.14	0.01	0.676	-0.65	1.72
Mean IBCH (Stream Macroinvertebrates)	Good mean ecological state of river reaches	**Good mean state of reaches**	**-**	**42**	**0.67**	**0.42**	**0.000**	**11.95**	**6.51**
		Many reaches in good state	**-**	**42**	**0.69**	**0.42**	**0.000**	**5.34**	**10.80**
	Resilience supporting habitats	**Resilience supporting habitats: all fish**	**Ecological state**	**42**	**0.69**	**0.31**	**0.000**	**5.22**	**11.78**
		Resilience supporting habitats: all fish	**Culverts**	**42**	**0.45**	**0.20**	**0.003**	**4.51**	**10.10**
		Resilience supporting habitats: trout	Large barriers	42	0.03	0.00	0.850	0.30	13.13
		Resilience supporting habitats: trout	**Large barriers & ecological state**	**42**	**0.65**	**0.32**	**0.000**	**7.88**	**11.51**
		Resilience supporting habitats: trout	Large barriers & culverts	42	0.12	0.02	0.462	1.41	12.65
		Resilience supporting habitats: other fish	All barriers	42	0.25	0.04	0.117	1.92	12.38
		Resilience supporting habitats: other fish	**All barriers & ecological state**	**42**	**0.61**	**0.27**	**0.000**	**7.11**	**11.68**
		Resilience supporting habitats: other fish	All barriers & culverts	42	0.29	0.05	0.061	1.93	12.51
	Low network fragmentation	**Low network fragmentation: all fish**	**Ecological state**	**42**	**0.73**	**0.50**	**0.000**	**5.78**	**9.93**
		Low network fragmentation: all fish	Culverts	42	0.03	0.02	0.841	-36.11	49.20
		Low network fragmentation: trout	Large barriers	42	0.03	0.00	0.850	0.30	13.13
		Low network fragmentation: trout	**Large barriers & ecological state**	**42**	**0.57**	**0.25**	**0.000**	**6.34**	**11.21**
		Low network fragmentation: trout	Large barriers & culverts	42	0.03	0.00	0.856	0.32	13.12
		Low network fragmentation: other fish	All barriers	42	0.25	0.04	0.117	1.92	12.38
		Low network fragmentation: other fish	**All barriers & ecological state**	**42**	**0.57**	**0.27**	**0.000**	**6.66**	**11.36**
		Low network fragmentation: other fish	All barriers & culverts	42	0.25	0.04	0.104	1.84	12.44
	Near-natural habitat diversity	**-**	**-**	**42**	**0.54**	**0.31**	**0.000**	**5.40**	**10.53**

were correlated with the biotic indices, with exception of those compared to the SMC fish assessment. In addition, two further versions present significant correlations: the 'resilience supporting habitats' considering culverts and large barriers.

The comparison of spatial criteria with biological indices was a first attempt to explore how the biological indicators respond to the spatial criteria. This was intended to support the choice between the different versions proposed. For fish related criteria, however, the data base was clearly not sufficient. Collecting more data at new sampling locations could provide the required information, particularly if focused on covering many tributaries of few catchments. Moreover, the IBCH index is currently under revision to improve its sensitivity to anthropogenic impairments (Michel et al., 2017). Despite these shortcomings, all but three of the suggested versions are supported by significant relationships to biotic indices.

We considered conceptual advantages and disadvantages in addition to the comparison with biotic indices, to provide a recommendation for the versions to be used. For the spatial criterion 'near-natural fish migration potential' we recommend to include versions from both sub-objectives, because they integrate complentary aspects. Further, we consider the presence of long culverts an important impediment to longitudinal connectivity, alongside to large and small barriers and suggest to consider four versions (see Table 3). To characterize the mean ecological state of the catchment, we put forward both the weighted average of the ecological state of the river sections and the fraction of the river network that is in a good ecological state, because the former summarizes the catchment state well, while the second provides valuable information for catchment managers and is aligned to policy practice in most European countries. For 'resilience supporting habitats', we recommend the versions considering the ecological state of the reaches alone and in combination with large or all barriers. These versions consider all organism groups, regardless of their dispersal capacity. The same reasons are valid for putting forward the same versions for the spatial criterion 'low network fragmentation'. Here we have used a critical length that is assumed to be intermediate between the dispersal distances of macroinvertebrates and fish, in accordance to the literature (Sundermann et al., 2011; Tonkin et al., 2014). Finally, the 'near-natural habitat diversity' is a valuable addition to the full set of variants, as it is complementary to all others in its goals.

As the approach presented here has many potential applications in terms of management questions to be addressed, or specific taxonomic groups to be considered, we decided to recommend a broad variety of versions (12 sub-objectives on the right side of the objectives hierarchy in Fig. 5). Applications of this approach could chose to use all of the recommended versions, or only a relevant subset. For instance, if the management goal is to reestablish migration routes for headwater-spawning species (e.g. *Salmo trutta* m. *fario*), one or more versions of the spatial criterion 'near-natural fish migration potential' would have to be included. Additional versions of other spatial criteria would be optional. Furthermore, the parameters used in the different sub-objectives of the spatial criteria (e.g. height threshold, critical length) should also be adjusted to the management goal and the taxa of interest. On the other hand, in a mountainous catchment with already naturally strongly limited fish migration potential, the subobjective 'near-natural fish migration potential' may be irrelevant.

Among the recommended versions of the criteria, some are significantly correlated with each other (Table A3). This is due to the partial overlap of the underlying data used to calculate the values of the different spatial criteria. However, the correlation is dependent on the size and the properties of the catchment being investigated. In our study, manually chosen subcatchments and those defined by biotic monitoring sites had different patterns of correlation among the spatial criteria. Further, selecting smaller subsets of spatial criteria variants can avoid unnecessary duplication of information in the catchment assessment.

The overall ecological assessment for the four catchments, based on the recommended spatial criteria, shows that all four catchments are considered to be in an overall bad (Mönchaltorfer Aa, Suhre and Toess) or poor state (Broye; Fig. 7). However, this depends also on the choice of our value function which is not yet based on sufficient experience (see comment at the end of the introduction to Section 2.4). For this reason, it is of more interest to analyze the differences in the assessments between the catchments. The criterion resilience supporting habitats is the only one with a bad state in all catchments (Mean = 0.09, SD = 0.04), which is reflected in all but one of the versions, across all catchments. The Suhre catchment obtained the worst overall assessment (good ecological state of the catchment = 0.07), which can be attributed to the poor fulfillment of the objectives 'near-natural fish migration potential', 'near-natural mean habitat size' and ' low network fragmentation'. The same three criteria also result in a bad state in the Aa and in the Toess catchments, but overall, these catchments are in a better state (0.15 and 0.17, respectively). The Broye catchment, however, shows a poor state in 'low network fragmentation' and a moderate state in 'near-natural fish migration potential', leading to the best overall assessment (0.29). 'Near-natural habitat diversity' achieved the best results overall, with poor in the Mönchaltorfer Aa and Suhre catchments, moderate in the Toess catchment and good in the Broye catchment (Mean = 0.44, SD = 0.13). The objective 'good ecological state of river reaches' was the spatial criterion, which on average resulted in the second highest assessment, reaching a bad state in the Aa and the Suhre catchments, while in the Toess and the Broye catchments the state was moderate (Mean = 0.43, SD = 0.13).

The bad and poor states of the four catchments assessed is to be expected, considering the fact that these are located in the Swiss Plateau, where historic and current anthropogenic disturbance has been, and continues to be highest for Switzerland. The Mönchaltorfer Aa is a rather small catchment with strong anthropogenic influences due to densely populated areas and intensive agriculture. This leads to a high fragmentation of the network and low adjacency of good habitats (Fig. 7 and Supplementary material A5). The Broye catchment is an important agricultural region, which is reflected in the moderate to bad nutrient and micropollutant state of many reaches, especially of the lower parts of the catchment (see Supplementary material A6). Due to the large number of natural barriers, only a small part of the catchment is relevant for fish migration (see Supplementary material A6). This part has only few artificial barriers, leading to a good fish migration potential for trout and a moderate one for other fish. The Suhre catchment is a medium-sized, relatively flat catchment that contains the eutrophic Lake Sempach (Bürgi and Stadelmann, 2002). Insufficient water quality, especially in the main stems (see Supplementary material A7) and a heavily modified morphology lead to high network fragmentation. The existence of artificial barriers very close to the catchment outlet leads to a poor assessment of the fish migration potential. However there are numerous headwaters in hilly uplands with a high ecological state, which lead to an overall moderate mean ecological state. The Toess river flows through a landscape with a very strong gradient of anthropogenic disturbance: the lower parts of the catchment are in intense agricultural use and a city with an important industrial history is located in the middle section, while the upper reaches are very hilly and less intensely utilized or inhabited, which leads to a good to high water quality in this part of the catchment (see Supplementary material A8). A large number of natural barriers lead to the fact that only a small part of the catchment would be available for migratory fish even under near-natural conditions and makes the large number of artificial barriers less relevant in this catchment. The results of the four catchments highlight the importance of incorporating a fish migration criterion that does not only account for the migration potential from the catchment outlet but also for connectivity among regions within the catchment (i.e. 'low network fragmentation').

A catchment-scale assessment, as the one presented here, can make significant contributions to improving the design of long-term, large-scale river management strategies that jointly consider several management measures, such as morphological restoration and water quality improvements. While other societal objectives, such as costs and constraints (e.g. existing infrastructure, land availability) are important criteria to be considered for spatial planning as well, an integrative ecological assessment procedure, based on physical and chemical aspects and including important ecological principles at the catchment scale will support river managers to increase the efficiency of management strategies. The proposed approach also allows comparing dissimilar catchments (i.e. in size, topography, geomorphology, climate, etc.). In addition, the transparent structure of the objectives hierarchy allows identifying current deficits and supports the identification of efficient management strategies. Moreover, the consideration of diverse spatial criteria allows addressing several objectives in parallel which are important to different stakeholders related to river restoration, catchment management or environmental policy-making.

Our assessment is comparable to existing large-scale ecological assessment approaches, such as the one proposed by Flotemersch et al. (2016), where six key functions of catchments are identified, of which three have been implemented here: habitat provision, hydrologic connectivity and water chemistry regulation. Other catchment regulation functions described in that study associated to temperature, sediments and the hydrologic regime, remain challenging to address as relevant data are collected at only few sites per catchment. Such gaps in data coverage could be solved with models used to predict conditions for the entire river network, as done here with the nutrient and micropollutant assessments. The approach presented here also benefits from the

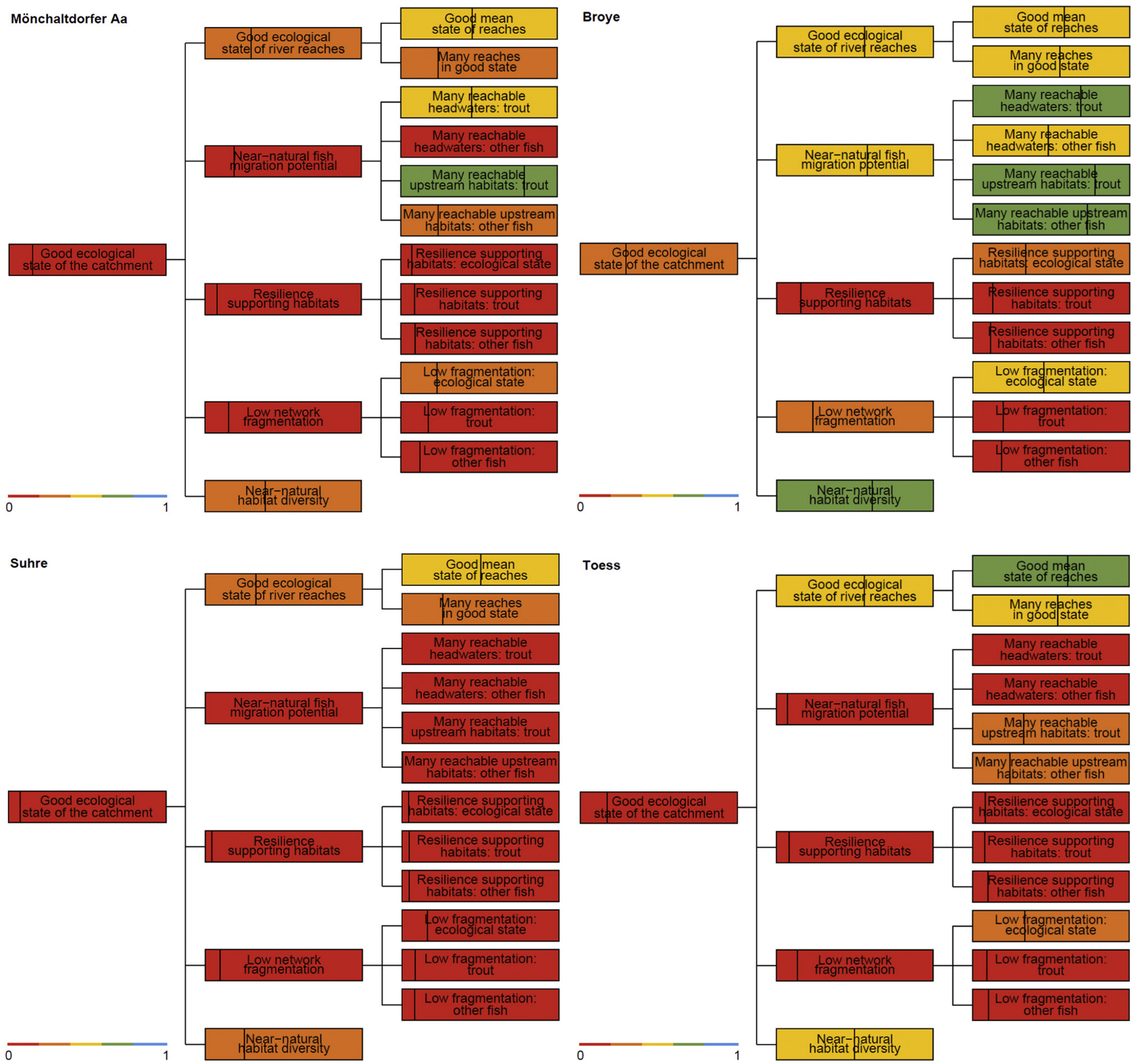


Fig. 7. Objective hierarchies with color-coded ecological assessments for the four catchments (red = bad, orange = poor, yellow = moderate, green = good, blue = high); vertical black lines indicate the ecological value on the scale between 0 and 1 (see legend). (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

catchment perspective it has been implemented with and is likely to deliver more specific results than other approaches implemented at much larger scales, across large river basins (Thornbrugh et al., 2018). Due to the comparably high abundance and high spatial resolution of abiotic datasets available in Switzerland, the approach presented here is able to include more aspects that are of high ecological relevance for freshwater ecosystems, than those which have been implemented elsewhere in similar assessments (Foerster et al., 2017; Hill et al., 2017; Huang et al., 2010; Paukert et al., 2011). However, we also illustrate how to deal with missing data (especially regarding water quality) by extrapolation based on land use data.

While the fact that we found a positive correlation between spatial criteria and biotic indices is encouraging, it does not indicate that the targeted ecological processes and functions were successfully captured. To test whether ecological processes and functions are represented by the spatial criteria suggested here, relevant data would have to be collected for statistical analysis. Such data could include detailed upstream and downstream movements of different species of fish at different life stages to estimate actual connectivity in the stream network (e.g. Calles and Greenberg, 2009), catchment-scale biodiversity assessments that take into account the niche breadth and dispersal ability of species (e.g. Heino, 2013) and the direct monitoring of certain ecological functions such as leaf-litter breakdown in rivers (Gessner and Chauvet, 2002). While studies collecting such data are rather scarce and dispersed, first attempts to quantify equivalent ecological functions are underway: for example, spawning sites for different species have been inventoried in the canton of Aargau (Breitenstein et al., 2017). Similar data from continuous and widespread inventories could help validate the spatial criteria used here, in a more appropriate way.

Moreover, currently lakes are not part of the assessment, despite being important parts of river basins that affect the physical, chemical and biological state of downstream rivers and can act as refugia for organisms such as migratory fish. Therefore, lakes should ideally be included in the ecological assessment and spatial prioritization within catchments.

4. Conclusions

Ecological assessment procedures for rivers developed in many countries are very useful tools for identifying deficits and supporting restoration planning. However, if there is insufficient funding for restoring all degraded river reaches, if restoration of all degraded reaches requires very long time frames, or if there are trade-offs between a good ecological state of the river and some societal services (such as hydropower generation or land required for settlements or agriculture), support for prioritizing reaches to be restored is needed. Reach-based ecological assessment procedures have only limited capabilities to support such a prioritization.

By combining reach-scale assessment procedures with a formalization of important ecological principles, such as supporting resilience (through considering the sizes of connected subsystems), migration potential (through considering barriers), and habitat diversity (through considering the fractions of different habitat types in good state), we suggest criteria for and implement a prototype of an ecological assessment at the catchment scale. Such a catchment-scale assessment allows us to compare the ecological value of different spatial arrangements of river reaches in good state and thus supports prioritization of restoration as well as conservation planning.

Due to the need of data for the whole river network, our current implementation of the reach-scale assessment used for the catchment-scale criteria is a simplified version based on river morphology and nutrient and pesticide pollution estimated from land use and sewage treatment plant effluents. Site-specific biological data were then used to validate the suggested criteria. The resulting correlations between the suggested spatial criteria and the site-specific biological data provided an empirical justification of the theory-based approach. Whenever better predictive models of the biological state would become available, the approach can easily be extended to include the reach-scale biological state.

The goal of this paper is to stimulate the discussion of the need for catchment-scale ecological assessment procedures and demonstrate the feasibility of this approach with a concrete prototype of such a procedure and its application to selected subcatchments in the Swiss Plateau. More testing of the ideas and its use within a formal optimization framework is planned in collaboration with representatives of cantonal authorities responsible for the implementation of the Swiss river rehabilitation program (which intends to rehabilitate 25% of the morphologically degraded rivers in Switzerland until 2090; Göggel, 2012).

More effort is needed with regard to improving the criteria (attributes and degree of fulfillment of the corresponding objective by a value function), directly considering the biological state indicators (in addition to the indirect indicators of morphology and water quality used here), and and in exploring the potential of applying formal optimization of the ecological state of the whole catchment for the spatial prioritization of restoration measures under given external constraints (such as budget and domestic areas).

Acknowledgements

We acknowledge the valuable data contributions from providers at federal monitoring programs (NAWA, BDM, NADUF) overseen by the Swiss Federal Office of the Environment (FOEN), CSCF, as well as at cantonal monitoring programs maintained by the environmental agencies of the cantons of Zürich, Aargau, Luzern, Fribourg and Vaud. We thank the BDM coordination office and the Centre Suisse de Cartographie de la Faune, CSCF, (especially Nadine Remund) for access to the macroinvertebrate data and support. We also thank Nanina Blank, Vinzenz Maurer and Gregor Thomas for the valuable discussions on earlier versions of the spatial criteria. This study is part of the AQUACROSS project, funded by the European Union's Horizon 2020 - Research and Innovation Framework Programme (Grant agreement No. 642317).

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2018.09.019.

References

Allan, J.D., 2004. Landscapes and riverscapes: the influence of land use on stream ecosystems. Annu. Rev. Ecol. Evol. Syst. 35, 257–284. https://doi.org/10.1146/annurev.ecolsys.35.120202.110122.

Altermatt, F., Seymour, M., Martinez, N., 2013. River network properties shape α-diversity and community similarity patterns of aquatic insect communities across major drainage basins. J. Biogeogr. 40, 2249–2260. https://doi.org/10.1111/jbi.12178.

AWEL, 2012. ZÜRCHER GEWÄSSER 2012 - Entwicklung Zustand Ausblick - Hauptbericht. Amt für Abfall, Wasser, Energie und Luft, Kanton Zürich, Zürich.

Baron, J.S., Poff, N.L., Angermeier, P.L., Dahm, C.N., Gleick, P.H., Hairston, N.G., Jackson, R.B., Johnston, C.A., Richter, B.D., Steinman, A.D., 2002. Meeting ecological and societal needs for freshwater. Ecol. Appl. 12, 1247–1260.

BDM Coordination Office, 2009. Biodiversity Monitoring in Switzerland (BDM) Coordination Office 2009: The state of biodiversity in Switzerland. Technical Report UZ-0911-E. Federal Office for the Environment (FOEN).

Bisson, P.A., Dunham, J.B., Reeves, G.H., 2009. Freshwater ecosystems and resilience of Pacific Salmon: habitat management based on natural variability. Ecol. Soc. 14.

Brederveld, R.J., Jähnig, 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. https://doi.org/10.1111/j.1365-2664.2011.02026.x.

Breitenstein, M., Flück, M., Kirchhofer, A., 2017. Inventar der Laichgebiete von Äsche, Nase, Barbe und Forelle in den grossen Fliessgewässern des Kantons Aargau. Wasser Fisch Natur AG.

Bundi, U., Frutiger, A., Göldi, C., Hütte, M., Kupper, U., Liechti, P., Meier, W., Niederhauser, P., Peter, A., Sieber, U., et al., 1998. Methoden zur Untersuchung und Beurteilung der Fliessgewässer: Modul-Stufen-Konzept. Mitteilungen Zum Gewässerschutz.

Bundi, U., Peter, A., Frutiger, A., Hütte, M., Liechti, P., Sieber, U., 2000. Scientific base and modular concept for comprehensive assessment of streams in Switzerland. Assessing the Ecological Integrity of Running Waters, Developments in Hydrobiology. Springer, Dordrecht, pp. 477–487 https://doi.org/10.1007/978-94-011-4164-2_37.

Bürgi, H., Stadelmann, P., 2002. Change of phytoplankton composition and biodiversity in Lake Sempach before and during restoration. Hydrobiologia 469, 33–48. https://doi.org/10.1023/A:1015575527280.

Buss, D.F., Carlisle, D.M., Chon, T.-S., Culp, J., Harding, J.S., Keizer-Vlek, H.E., Robinson, W.A., Strachan, S., Thirion, C., Hughes, R.M., 2015. Stream biomonitoring using macroinvertebrates around the globe: a comparison of large-scale programs. Environ. Monit. Assess. 187, 4132. https://doi.org/10.1007/s10661-014-4132-8.

Calles, O., Greenberg, L., 2009. Connectivity is a two-way street—the need for a holistic approach to fish passage problems in regulated rivers. River Res. Appl. 25, 1268–1286. https://doi.org/10.1002/rra.1228.

Cardinale, B.J., Palmer, M.A., Swan, C.M., Brooks, S., Poff, N.L., 2002. The influence of substrate heterogeneity on biofilm metabolism in a stream ecosystem. Ecology 83, 412–422. https://doi.org/10.1890/0012-9658(2002)083[0412:TIOSHO]2.0.CO;2.

Cooper, A.R., Infante, D.M., Daniel, W.M., Wehrly, K.E., Wang, L., Brenden, T.O., 2017. Assessment of dam effects on streams and fish assemblages of the conterminous USA. Sci. Total Environ. 586, 879–889. https://doi.org/10.1016/j.scitotenv.2017.02.067.

Culhane, F., Teixeira, H., Nogueira, A.J., Borgwardt, F., Trauner, D., Lillebø, A., Piet, G., Kuemmerlen, M., McDonald, H., O'Higgins, T., Barbosa, A.L., van der Val, J.T., Iglesias-Campos, A., Arevalo-Torres, J., Barbiere, J., Robinson, L.A., 2018. Risk to the supply of ecosystem services across aquatic realms. Sci. Total Environ. (in review).

Dudgeon, D., Arthington, A.H., Gessner, M.O., Kawabata, Z.-I., Knowler, D.J., Lévêque, C., Naiman, R.J., Prieur-Richard, A.-H., Soto, D., Stiassny, M.L.J., Sullivan, C.A., 2006. Freshwater biodiversity: importance, threats, status and conservation challenges. Biol. Rev. 81, 163–182. https://doi.org/10.1017/S1464793105006950.

Dyer, J.S., Sarin, R.K., 1979. Measurable multiattribute value functions. Oper. Res. 27, 810–822. https://doi.org/10.1287/opre.27.4.810.

ESRI, 2016. ArcGIS Desktop: Release 10.4. Environmental Systems Research Institute, Redlands, CA.

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. Communities L327, 1–72.

Flotemersch, J., Leibowitz, S., Hill, R., Stoddard, J., Thoms, M., Tharme, R., 2016. A watershed integrity definition and assessment approach to support strategic management of watersheds. River Res. Appl. 32, 1654–1671.

FOEN, 2001. Biogeographical Regions of Switzerland. Federal Office of the Environment, CH-3003 Bern.

FOEN, 2013. NAWA – National Surface Water Quality Monitoring Programme (NAWA). Konzept Fliessgewässer (No. 1327), Umwelt-Wissen. Federal Office of the Environment, CH-3003 Bern.

FOEN, 2014. Catchment River Segment Based. Federal Office of the Environment, CH-3003 Bern (including attribute: fraction of wastewater).

FOEN, 2016. Hydromorphology of Switzerland. Federal Office of the Environment, CH-3003 Bern.

Foerster, J., Halle, M., Müller, A., 2017. Entwicklung eines Habitatindex zur Beurteilung biozönotisch relevanter Gewässerstrukturen. Korresp. Wasserwirtsch. 8 pp. 466–471.

FSO, 2008. Statistics of Farming. Federal Statistical Office, CH-2010 Neuchâtel.

Gessner, M.O., Chauvet, E., 2002. A case for using litter breakdown to assess functional stream integrity. Ecol. Appl. 12, 498–510. https://doi.org/10.1890/1051-0761(2002)012[0498:ACFULB]2.0.CO;2.

Ghetti, P.F., 1980. Biological indicators of the quality of running waters. Ital. J. Zool. 47, 381–390.

Göggel, W., 2012. Revitalisierung Fliessgewässer. Strategische Planung. Ein Modul der Vollzugshilfe Renaturierung der Gewässer. Bundesamt für Umwelt, Bern. Umw.-Vollzug 1208, p. 42.

Grant, E.H.C., Lowe, W.H., Fagan, W.F., 2007. Living in the branches: population dynamics and ecological processes in dendritic networks. Ecol. Lett. 10, 165–175. https://doi.org/10.1111/j.1461-0248.2006.01007.x.

Gravena, W., Silva, D., F, V.M., Silva, D., F, M.N., Farias, I.P., Hrbek, T., 2015. Living between rapids: genetic structure and hybridization in botos (Cetacea: Iniidae: *Inia* spp.) of the Madeira River, Brazil. Biol. J. Linn. Soc. 114, 764–777. https://doi.org/10.1111/bij.12463.

Haag, F., Lienert, J., Schuwirth, N., Reichert, P., 2018. Identifying non-additive multi-attribute value functions based on uncertain indifference statements. Omega https://doi.org/10.1016/j.omega.2018.05.011 (in press).

Heino, J., 2013. The importance of metacommunity ecology for environmental assessment research in the freshwater realm. Biol. Rev. 88, 166–178. https://doi.org/10.1111/j.1469-185X.2012.00244.x.

Hermoso, V., Pantus, F., Olley, J., Linke, S., Mugodo, J., Lea, P., 2012. Systematic planning for river rehabilitation: integrating multiple ecological and economic objectives in complex decisions. Freshw. Biol. 57, 1–9. https://doi.org/10.1111/j.1365-2427.2011.02693.x.

Hill, R.A., Fox, E.W., Leibowitz, S.G., Olsen, A.R., Thornbrugh, D.J., Weber, M.H., 2017. Predictive mapping of the biotic condition of conterminous U.S. rivers and streams. Ecol. Appl. 27, 2397–2415. https://doi.org/10.1002/eap.1617.

Howard, J.K., Fesenmyer, K.A., Grantham, T.E., Viers, J.H., Ode, P.R., Moyle, P.B., Kupferburg, S.J., Furnish, J.L., Rehn, A., Slusark, J., et al., 2018. A freshwater conservation blueprint for California: prioritizing watersheds for freshwater biodiversity. Freshw. Sci. 37 (000–000).

Huang, P.-H., Tsai, J.-S., Lin, W.-T., 2010. Using multiple-criteria decision-making techniques for eco-environmental vulnerability assessment: a case study on the Chi-Jia-Wan Stream watershed, Taiwan. Environ. Monit. Assess. 168, 141–158. https://doi.org/10.1007/s10661-009-1098-z.

Hütte, M., Niederhauser, P., 1998. Methoden zur Untersuchung und Beurteilung der Fliessgewässer in der Schweiz, Ökomorphologie Stufe F (flächendeckend). Mitteilungen Zum Gewässerschutz.

Hynes, H., 1975. Edgardo Baldi memorial lecture. The stream and its valley. Verh. Int. Ver. Theor. Angew. Limnol. 19, 1–15.

Jakob, A., Binderheim-Bankay, E., Davis, J., 2002. National long-term surveillance of Swiss rivers. Verh. Int. Ver. Theor. Angew. Limnol. 28, 1101–1106.

Johnson, L., Richards, C., Host, G., Arthur, J., 1997. Landscape influences on water chemistry in Midwestern stream ecosystems. Freshw. Biol. 37, 193–208. https://doi.org/10.1046/j.1365-2427.1997.d01-539.x.

Kamp, U., Binder, W., Hölzl, K., 2007. River habitat monitoring and assessment in Germany. Environ. Monit. Assess. 127, 209–226. https://doi.org/10.1007/s10661-006-9274-x.

Keeney, R.L., 1996. Value-focused Thinking. Harvard University Press.

Könitzer, C., Zaugg, C., Wagner, T., Pedroli, J., Mathys, L., 2012. Wiederherstellung der Fischwanderung. Strategische Planung. Ein Modul der Vollzugshilfe Renaturierung der Gewässer. Bundesamt für Umwelt. Umw.-Vollzug, p. 54.

Kuemmerlen, M., Schmalz, B., Guse, B., Cai, Q., Fohrer, N., Jähnig, S.C., 2014. Integrating catchment properties in small scale species distribution models of stream macroinvertebrates. Ecol. Model. 277, 77–86. https://doi.org/10.1016/j.ecolmodel.2014.01.020.

Kunz, M., Schindler Wildhaber, Y., Dietzel, A., Wittmer, I., Leib, V., 2016. Zustand der Schweizer Fliessgewässer. Ergebnisse der Nationalen Beobachtung Oberflächengewässerqualität (NAWA) 2011–2014. Bundesamt für Umwelt, Bern. Umw.-Zustand 1620, p. 87.

Lake, P., 2000. Disturbance, patchiness, and diversity in streams. J. North Am. Benthol. Soc. 19, 573–592.

Lake, P.S., Bond, N., Reich, P., 2007. Linking ecological theory with stream restoration. Freshw. Biol. 52, 597–615. https://doi.org/10.1111/j.1365-2427.2006.01709.x.

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. https://doi.org/10.1016/j.ecolind.2013.03.027.

Langhans, S.D., Reichert, P., Schuwirth, N., 2014. The method matters: a guide for indicator aggregation in ecological assessments. Ecol. Indic. 45, 494–507. https://doi.org/10.1016/j.ecolind.2014.05.014.

Liechti, P., 2010. Methoden zur Untersuchung und Beurteilung der Fliessgewässer: Chemisch-physikalische Erhebungen, Nährstoffe. BAFU.

Liermann, C.R., Nilsson, C., Robertson, J., Ng, R.Y., 2012. Implications of dam obstruction for global freshwater fish diversity. Bioscience 62, 539–548. https://doi.org/10.1525/bio.2012.62.6.5.

Liess, M., Ohe, P.C.V.D., 2005. Analyzing effects of pesticides on invertebrate communities in streams. Environ. Toxicol. Chem. 24, 954–965.

Limburg, K.E., Waldman, J.R., 2009. Dramatic declines in North Atlantic diadromous fishes. Bioscience 59, 955–965. https://doi.org/10.1525/bio.2009.59.11.7.

Malmqvist, B., 2002. Aquatic invertebrates in riverine landscapes. Freshw. Biol. 47, 679–694. https://doi.org/10.1046/j.1365-2427.2002.00895.x.

Malmqvist, B., Rundle, S., 2002. Threats to the running water ecosystems of the world. Environ. Conserv. 29, 134–153. https://doi.org/10.1017/S0376892902000097.

McCluney, K.E., Poff, N.L., Palmer, M.A., Thorp, J.H., Poole, G.C., Williams, B.S., Williams, M.R., Baron, J.S., 2014. Riverine macrosystems ecology: sensitivity, resistance, and resilience of whole river basins with human alterations. Front. Ecol. Environ. 12, 48–58.

Metcalfe, J.L., 1989. Biological water quality assessment of running waters based on macroinvertebrate communities: history and present status in Europe. Environ. Pollut. 60, 101–139.

Michel, C., Schindler Wildhaber, Y., Leib, V., Remund, N., Schuwirth, N., 2017. Überarbeitung des Makrozoobenthos-Index. Natürliche Einflussfaktoren, Ursache-Wirkungsanalyse und Diskussion des Spear-Index. 97. Aqua Gas, pp. 70–77.

Paukert, C.P., Pitts, K.L., Whittier, J.B., Olden, J.D., 2011. Development and assessment of a landscape-scale ecological threat index for the Lower Colorado River Basin. Ecol. Indic. 11, 304–310.

R Core Team, 2017. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria.

Reichert, P., Schuwirth, N., Langhans, S., 2013. Constructing, evaluating and visualizing value and utility functions for decision support. Environ. Model. Softw. 46, 283–291. https://doi.org/10.1016/j.envsoft.2013.01.017.

Reichert, P., Langhans, S.D., Lienert, J., Schuwirth, N., 2015. The conceptual foundation of environmental decision support. J. Environ. Manag. 154, 316–332. https://doi.org/10.1016/j.jenvman.2015.01.053.

Resh, V.H., Brown, A.V., Covich, A.P., Gurtz, M.E., Li, H.W., Minshall, G.W., Reice, S.R., Sheldon, A.L., Wallace, J.B., Wissmar, R.C., 1988. The role of disturbance in stream ecology. J. N. Am. Benthol. Soc. 7, 433. https://doi.org/10.2307/1467300.

Saunders, D.L., Meeuwig, J.J., Vincent, A.C.J., 2002. Áreas Protegidas de Agua Dulce: Estrategias para la Conservación. Conserv. Biol. 16, 30–41. https://doi.org/10.1046/j.1523-1739.2002.99562.x.

Schaffner, M., Pfaundler, M., Göggel, W., 2013. Fliessgewässertypisierung der Schweiz. Eine Grundlage für Gewässerbeurteilung und-entwicklung. Bundesamt für Umwelt, Bern. Umw.-Wissen 63.

Schager, E., Peter, A., 2004. Fische Stufe F (flächendeckend): Methoden zur Untersuchung und Beurteilung der Fliessgewässer. Bundesamt für Umwelt, Wald und Landschaft, Dokumentation.

Schwarzenbach, R.P., Escher, B.I., Fenner, K., Hofstetter, T.B., Johnson, C.A., von Gunten, U., Wehrli, B., 2006. The challenge of micropollutants in aquatic systems. Science 313, 1072–1077. https://doi.org/10.1126/science.1127291.

Spycher, S., Hunkeler, J., Bosshard, A., Häni, F., 2015. Gewässerbelastung durch Pestizide–Ansätze zur Verminderung landwirtschaftlich bedingter Einträge in Oberflächengewässer. 12. Aqua Gas, pp. 56–71.

Stoll, S., Kail, J., Lorenz, A.W., Sundermann, A., Haase, P., 2014. The importance of the regional species pool, ecological species traits and local habitat conditions for the colonization of restored river reaches by fish. PLoS ONE 9, e84741. https://doi.org/10.1371/journal.pone.0084741.

Strahler, A.N., 1957. Quantitative analysis of watershed geomorphology. EOS Trans. Am. Geophys. Union 38, 913–920. https://doi.org/10.1029/TR038i006p00913.

Stucki, P., 2010. Methoden zur Untersuchung und Beurteilung der Fliessgewässer: Makrozoobenthos Stufe F. Bundesamt Für Umw, Bern Umw.-Vollzug 1026, p. 61.

Sundermann, A., Stoll, S., Haase, P., 2011. River restoration success depends on the species pool of the immediate surroundings. Ecol. Appl. 21, 1962–1971. https://doi.org/10.1890/10-0607.1.

swisstopo (2010), Vector25; DV 5704 000 000, reproduced by permission of swisstopo/JA100119.

swisstopo (2016), swissTLM3D, swissBOUNDARIES3D; DV 5704 000 000, reproduced by permission of swisstopo/JA100119.

Teixeira, H., Lillebø, A., Culhane, F., Robinson, L.A., Trauner, D., Borgwardt, F., Kuemmerlen, M., Barbosa, A.L., McDonald, H., Funk, A., O'Higgins, T., van der Wal, J.T., Piet, G., Hein, T., Arévalo-Torres, J., Iglesias-Campos, J., Barbière, J., Nogueira, A.J., 2018. Linking biodiversity to ecosystem services supply: integrating across aquatic ecosystems. Sci. Total Environ. (in review).

Thornbrugh, D.J., Leibowitz, S.G., Hill, R.A., Weber, M.H., Johnson, Z.C., Olsen, A.R., Flotemersch, J.E., Stoddard, J.L., Peck, D.V., 2018. Mapping watershed integrity for the conterminous United States. Ecol. Indic. 85, 1133–1148. https://doi.org/10.1016/j.ecolind.2017.10.070.

Thorp, J.H., Thoms, M.C., Delong, M.D., 2006. The riverine ecosystem synthesis: biocomplexity in river networks across space and time. River Res. Appl. 22, 123–147. https://doi.org/10.1002/rra.901.

Tonkin, J.D., Stoll, S., Sundermann, A., Haase, P., 2014. Dispersal distance and the pool of taxa, but not barriers, determine the colonisation of restored river reaches by benthic invertebrates. Freshw. Biol. 59, 1843–1855. https://doi.org/10.1111/fwb.12387.

Vermeiren, P., Reichert, P., Schuwirth, N. (2018) Pushing prediction boundaries by combining joint species distribution models with ecological trait information (submitted for publication).

Wernersson, A.-S., Carere, M., Maggi, C., Tusil, P., Soldan, P., James, A., Sanchez, W., Dulio, V., Broeg, K., Reifferscheid, G., Buchinger, S., Maas, H., Grinten, E.V.D., O'Toole, S., Ausili, A., Manfra, L., Marziali, L., Polesello, S., Lacchetti, I., Mancini, L., Lilja, K., Linderoth, M., Lundeberg, T., Fjällborg, B., Porsbring, T., Larsson, D.J., Bengtsson-Palme, J., Förlin, L., Kienle, C., Kunz, P., Vermeirssen, E., Werner, I., Robinson, C.D., Lyons, B., Katsiadaki, I., Whalley, C., den Haan, K., Messiaen, M., Clayton, H., Lettieri, T., Carvalho, R.N., Gawlik, B.M., Hollert, H., Paolo, C.D., Brack, W., Kammann, U., Kase, R., 2015. The European technical report on aquatic effect-based monitoring tools under the water framework directive. Environ. Sci. Eur. 27, 7. https://doi.org/10.1186/s12302-015-0039-4.

Wiens, J.A., 2002. Riverine landscapes: taking landscape ecology into the water. Freshw. Biol. 47, 501–515. https://doi.org/10.1046/j.1365-2427.2002.00887.x.

Wofford, J.E.B., Gresswell, R.E., Banks, M.A., 2005. Influence of barriers to movement on within-watershed genetic variation of coastal cutthroat trout. Ecol. Appl. 15, 628–637. https://doi.org/10.1890/04-0095.