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A systematic review of assessment and conservation management in large floodplain rivers – actions postponed

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• We review approaches to the assessment of ecological condition and conservation management of large floodplain rivers.
• The review highlights research gaps and emphasizes the importance of developing more holistic indicators of ecosystem condition.
• Indicators that better reflect landscape level changes in structure and functioning of floodplain rivers are needed.

• Studies that distinguish the role of different river floodplain habitat types in ecosystem services provision are needed.
• More effective spatial conservation prioritization tools are needed at the river floodplain scale.

Abstract

Large floodplain rivers (LFRs) are currently threatened by high levels of human alteration, and utilization is expected to grow. Assessments to determine ecological condition should address the specific environmental features of these unique ecosystems, while conservation management requires balancing maintenance of good ecological condition with the ecosystem services provided by LFRs. However, a systematic evaluation of the scientific literature on assessment of ecological condition of LFRs and trade-offs to guide conservation management is currently lacking. Here, we reviewed 153 peer reviewed scientific articles to characterize methodological patterns and trends and identify knowledge gaps in the assessment of LFRs. Our review revealed that most approaches used classical biotic indices for assessing ecological condition of LFRs. However, the number of articles specifically addressing the peculiarities of LFRs was low. Many studies used watershed level surveys and assessed samples from small streams to large rivers using the same methodological protocol. Most studies evaluated the status of main stem river habitats only, indicating large knowledge gaps with respect to the diversity of river-floodplain habitat types or lateral connectivity. Studies related to management were oriented toward specific rehabilitation actions rather than broader conservation of LFRs. Papers relating to ecosystem services of LFRs were especially few. Most importantly, these studies did not distinguish the different functional units of river-floodplain habitat types (e.g. eupotamon, parapotamon) and their role in ecosystem services provision. Overall, the number of articles was too low for meaningful analyses of the relationships and tradeoffs between biodiversity conservation, maintaining ecological condition, and use of ecosystem services in LFRs. Our review highlights research gaps and emphasizes the importance of developing more holistic indicators of ecosystem condition, which better reflect landscape level changes in structure and functioning of LFRs. As human use of water and land increases, the need to develop more effective spatial conservation prioritization tools becomes more important. Empirical research in this field can aid in solving conflicts between socio-economic demands for ecosystem services and nature conservation of LFRs.

key words: ecological condition, biological integrity, ecological status, rehabilitation, restoration, biodiversity, ecosystem services

Introduction

Large floodplain rivers (LFRs) are the lifelines of our landscapes. By draining large catchment areas, they integrate environmental, topographic and hydro-geomorphic conditions. LFRs are four dimensional systems, with longitudinal connectivity along the river gradient, lateral connectivity to the floodplain, vertical connections with the substrate and the groundwater layer, and having a temporal trajectory (Ward, 1989). Large river habitats can be considered hierarchically nested from regions down to river reaches, with quality and spatial arrangement of habitat units at the finer spatial scales controlled by processes at coarse spatial levels (Gurnell et al., 2016). Regularly occurring floods and droughts make rivers disturbance-driven systems subjected to periodic rejuvenation of habitats through erosion and deposition processes. As a result, LFRs provide a dynamic mosaic of habitats in various successional states that differ in complexity, connectivity and patchiness (e.g., Thorp et al., 2006), which is usually considered the foundation of their exceptionally high biodiversity (e.g., Tockner and Ward, 1999).

At the same time, LFRs are subject to intense use by humans, including transformation, reclamation, and degradation of the natural landscape (Tockner and Stanford, 2002; Peipoch et al., 2015). Ancient civilizations arose on floodplains by cultivating the fertile land. Increasing agriculture and urbanization, and the associated river regulation (e.g. channelization, building of dams, flood control by levees) over time have substantially reduced the area as well as the spatial and temporal complexity of LFRs. For example, more than 50% of the world's population currently lives within 3 km of freshwaters (Kummu et al., 2011), and more than 600,000 km of inland waterways have been altered for navigation worldwide (CIA, 2002). The net result is constriction of floodplains by more than 50% of the historical expanse (for details, see Tockner and Stanford, 2002). In Europe, which is the most human dominated continent, up to 90% of former floodplains have been degraded to functional extinction (Tockner et al., 2010). Modification and degradation is ongoing due to agriculture, urbanization, navigation and development of large hydropower projects, making LFRs the most threatened ecosystems on Earth (Arthington et al., 2010; Sommerwerk et al., 2010).

In sum, LFRs are highly complex natural systems of high biodiversity and societal value, but severely degraded and in urgent need of protection and rehabilitation. It shall be noted here that rehabilitation is used throughout this article to reference all measures and attempts to mitigate degradation and to improve ecosystem functions and processes. This acknowledges the persistence and irreversibility of certain uses and changes, respectively, and the corresponding impossibility to restore LFRs to historical or pristine states (i.e. restoration). Due to their size, inherent complexity and integrative nature, LFRs are costly to sample and assess (e.g. de Leeuw et al., 2007; Flotemersch et al., 2011). Broader challenges include the need to identify and prioritize the most pressing stressors on LFRs while balancing conservation and rehabilitation of ecological condition with the diverse benefits that LFRs provide to society (i.e. ecosystem services; see Fig. 1). Accordingly, examples of in-depth assessment of pressure effects, rehabilitation measures in or rehabilitation guidance for LFRs are rather scarce (e.g. Zajicek et al., 2018). Correspondingly, in Germany an analysis of the first river basin management plans implementing the European Water Framework Directive (WFD, 2000/60/EEC) revealed that huge knowledge gaps were evident (especially for large rivers), and mostly conceptual measures were planned (Kail and Wolter, 2011). Trade-offs and synergies between the spatial distribution of ecological condition and ecosystem services have to be understood and quantified. LFR management is expected to either spare the land for biodiversity conservation or for human use, or to share it between conservation and use for the joint benefit of both nature and the society (Cordingley at et al., 2016; Doody et al., 2016). This evaluation procedure requires scientifically robust methods that can assess the ecological or conservation status of LFRs and also identify optimal solutions for the allocation of resources (i.e. prioritization of the landscape for conservation/rehabilitation and/or for use).

This systematic review aims to evaluate status and progress in assessing and managing LFRs, defining research gaps and future research avenues. Several research and review articles emphasize the importance of natural patterns and processes in the effective conservation of LFRs (e.g. Jungwirth et al., 2002; Thorp et al., 2010). However, a systematic evaluation of assessment approaches for LFRs and how well they address societal goals of maintaining good ecological condition, conserving biodiversity, and capitalizing on ecosystem services is currently lacking. Consequently, we conducted a systematic review to summarize trends in the assessment of ecological condition, conservation and ecosystem services of LFRs. Specifically, we asked the following two questions: 1) how is ecological condition of LFRs assessed, and 2) how can maintenance of ecological condition be balanced with use of ecosystem services of LFRs?

Materials and Methods

We conducted a systematic evaluation of the peer-reviewed literature relating to the determination, conservation and rehabilitation of ecological condition, the conservation of biodiversity and/or the use of ecosystem services in LFRs. We performed a literature search in the Web of Science (WoS; http://apps.webofknowledge.com) database using the following keywords combination: („ecological status" OR „ecological condition" OR „ecosystem health" OR "ecological integrity" OR "biological integrity" OR conservation OR rehabilitation OR restoration OR biodiversity OR "ecosystem services") AND (river* OR floodplain* OR „floodplain-lake*" OR oxbow*). For simplicity, we selected English language articles only. The search was executed on 11 December 2017, and yielded 2426 articles in the time period from 1992 to 2017. All authors were assigned an equal number of articles to screen against review criteria. Because the definition of large rivers varied, we decided to incorporate all studies dealing with potamal floodplain rivers larger than 1000 km^2 in catchment size. Articles were excluded from the analyses if i) the main topic was not related to assessment of ecological condition, conservation or ecosystem services, ii) the focus was only on small streams and rivers, or iii) evaluations were performed at the level of sites or sub-catchments with unclear relation to LFRs. We also excluded review articles, except where they contained detailed case studies for effective evaluation (e.g. details of restoration projects in Jungwirth et al., 2002). This procedure resulted in a total of 153 papers matching our study criteria.

From each study, we extracted the location, spatial scale, year(s) of investigation, the floodplain habitat types studied and other circumstances of data collection (see Appendix I.). We paid special attention to evaluating the role of different river-floodplain functional habitat types (for details see Amoros et al., 1982; 1987; Ward and Stanford, 1995) in assessment and management goals. We distinguished five habitat types as follows: MR, main river or eupotamon habitats, which include the main channel and side arms that are connected to the main channel even at low flow; FP1, floodplain 1 or parapotamon, and plesiopotamon habitats, which are abandoned braided channels or backwaters blocked from upstream (parapotamon) and from both upstream and downstream direction (plesiopotamon), but often connected to the main arm depending on water level; FP2, floodplain 2 or paleopotamon habitats are oxbows in the floodplain area, which are only rarely connected to the river and to other side arm components by surface flow; FPA, flood protected area, which contains oxbows separated completely from the floodplain by dams; and R, riparian areas, which include all other terrestrial habitats belonging to the floodplain.

We characterized each study into six categories based on the main study objectives, as (1) assessment of ecological condition (EC; note that this broad term incorporates evaluation of ecological or ecosystem status, health, condition or ecological/biological integrity), (2) conservation (C), (3) rehabilitation or restoration (R, hereafter we use the term rehabilitation only, because – although the term is widely used – true restoration, e.g. of pristine or natural conditions of LFR is rarely intended), (4) ecosystem services (ES), (5) trade-off situation between C and ES (C/ES), and (6) biodiversity inventory or monitoring (BDM). Studies that addressed more than one topic were classified to more than one type (e.g., to both EC and BDM).

For ecological assessments (EC), we classified the taxonomic group(s), number and type of variables (metrics) used for the evaluation, the number and type of stressors measured, and the characterization of reference condition. For conservation (C), rehabilitation (R) and ecosystem service (ES) studies we examined the components of biodiversity and services, and whether and how trade-off relationships were handled. We also evaluated the reported involvement of stakeholders in achieving study objectives. Further details of the data collected and reviewed are provided in Appendix I.

Results and Discussion

General findings

Of the 153 articles reviewed, 60.0%, 24.7%, 9.5%, 4.2%, 1.6%, and 0.0% addressed EC, BDM, R, C, ES, and C/ES, respectively. The geographic distribution of the studies was highly unequal across continents and ecoregions (Fig. 2). A majority of the studies were conducted in Europe (32.0%) and North America (28.1%), whereas studies from Asia (16.3%), Africa (8.5%), South America (7.8%) and Australia and New Zealand (7.2%) were much less represented. Altogether 73 ecoregions were represented in studies. However, a relatively large proportion were conducted in just three ecoregions: Central & Western Europe 10.5% (Europe), the Upper-Danube 9.2% (Europe), and the Lower Mississippi 5.9% (North America).

Assessment of ecological condition

Evaluation of ecological condition (EC articles) was mostly performed (48.9% of the studies) using main river assemblages (i.e. in eupotamon habitats). In contrast, other floodplain habitats were assessed by a much lower number of studies (Fig. 3). Specifically, floodplain habitats type 1 (parapotamon, plesiopotamon) and type 2 (paleopotamon) were assessed by 22.6% and 18.9% of the studies, respectively, and flood protected areas and riparian systems were considered in only 6.3% and 3.2%, respectively. A majority of the studies (60.9%) incorporated only one habitat type for evaluating ecosystem status. Similar numbers of studies evaluated two (16.5%) and three (19.1%) habitat types; however, only 3.5% studies incorporated four habitat types. No study evaluated all five habitat types of LFRs.

The taxonomic groups most often used to assess ecological condition were fishes and benthic invertebrates, accounting for 45.6% and 35.0% of the studies, respectively. All other taxa (e.g. algae, macrophytes) were much less frequently used (Fig. 4). 83.0% of the papers used only a single taxonomic group for the assessment, 10% applied two groups, and only 7.0% of the studies used three or more groups. Taxonomic (e.g. species richness, number and/or abundance of specific taxa) and functional (e.g. % omnivores, % invertivores) metrics were the most frequently used biological response variables across all studies. In studies using fish as the response group, index-based approaches (i.e., scoring alteration metrics from a reference value and summing values into a single index) were most common (see e.g. Ganasan and Hughes, 1998; Sharma et al., 2017); however, it should be noted that this methodology was typically unchanged from how it is applied to assess site-level degradation in small streams and rivers (e.g., Karr, 1981). Assessments that focused on benthic invertebrates tended to rely on diversity indices (e.g. Shannon-Wiener, Simpson indices) and density metrics (individuals m^{-2}) (see e.g. Cabecinha et al., 2004; Raburu et al., 2009), which were only infrequently used in fish based studies. Though few in number, studies on macrophytes incorporated structural vegetation variables like maximum vegetation height. For example, in the San Pedro River, (Gila ecoregion, U.S.A.), Stromberg et al. (2006) examined how groundwater withdrawal influences the ecological condition of the floodplain system based on maximum vegetation height across the floodplain, % shrubland cover, and absolute as well relative cover of hydric perennial herbs. Interestingly, algae were also relatively rarely used in EA of LFRs. Utilizing algae as indicators, for example, Greiner et al. (2010) used classification algorithms (Self-Organizing Maps) to set up biotypes along an alteration gradient and to determine ecological thresholds for setting up the boundaries of condition classes.

Many studies, however, did not use biotic indices or any other quantitative assessment of ecological condition. These studies instead examined how the structure (i.e. presence/absence or relative abundance) of biological assemblages was associated with the degradation (i.e. ecological condition) of the habitats using multivariate community analyses (e.g. Pan et al., 2014). Further, some articles exclusively assessed habitat condition, which of course is an important component of overall ecological condition, but cannot be used *per se* for this purpose, if the biotic response to the habitats is not considered. For example, in Austrian rivers Muhar et al. (2000) concluded that only 43 km (5.9%) out of 731 km of large alluvial rivers remained in relatively intact condition using a scoring system that characterized the habitat quality based on morphological character, instream structures, longitudinal and lateral connectivity, and hydrological regime compared with reference conditions.

A surprisingly large number of papers did not provide a clear description of the methodology of ecological condition assessment by specifying the type of stressors or the response biotic metrics. In fact, many studies used only the biotic groups as indicators of ecological condition without evaluating the role of stressor variables (e.g. only 32.5% of the papers examined stressor metric relationships). When stressors were analyzed as part of the assessment, land use variables (e.g. percentage of forest, agricultural land) were the most frequently used, reported in 54.4% of the papers. Land use is not only easy to derive from thematic maps; it seemingly provides a good approximation for ecological degradation of large rivers. For example, Trautwein et al. (2012) found two simple land use metrics, % agriculture and % urbanization, were the best correlated stressor metrics with fish-based biotic indices (i.e. ecological condition) in the Upper Danube ecoregion, Austria; however, stream fish assemblages of lower mountain rivers were more sensitive to land use changes than fish assemblages inhabiting low gradient, large rivers. In the Paraiba do Sul ecoregion, Brasil, Pinto et al. (2006) found land use (especially % pasture, % urban area) and riparian condition closely associated with fish biotic indices.

Physical stressors were assessed in 34.2% of the papers. Among these, connectivity (effect of dams), instream and riparian habitat structure (flow regulation, channel modification) were most frequently measured. For example, in main stem rivers in the Central & Western Europe ecoregion, Czech Republic, Musil et al. (2012) demonstrated that weirs and dams affected the biotic status of fish assemblages. In the Upper Lancang (Mekong) ecoregion, China, Zhai et al. (2010) demonstrated how a series of hydropower dams affected the ecological condition due to alteration of flow, water quality and sediment transport. Chemical (i.e. water quality) stressors were utilized in 28.1% of studies and included primarily sediment pollution, point source pollution, concentration of nutrients and oxygen content. For example, in the Liao He ecoregion, China, basic physiochemical parameters, BOD5, CODcr, TN, TP, NH3-N, DO, petroleum hydrocarbon and conductivity were associated with an integrated ecological health index (Meng et al., 2009). This integrated index combines physical habitat quality, fecal coliform count, attached algae diversity, and a benthic index of biotic condition (Meng et al., 2009). Biological stressors appeared in only 7.0% of studies, and were largely comprised of the number or abundance of non-native species (fish) and livestock grazing. For example, in the Southern Iberia ecoregion, Spain, dominance of non-native fishes was an important determinant of ecological condition indicated by fish-based indices (Hermoso et al., 2010). In the Lake Victoria Basin ecoregion, Kenya, excessive grazing and deforestation affected fish-based ecological condition (Raburu and Masase, 2012). Nevertheless, most studies showed that a combination of stressors shape the structure and assemblages of biotic communities in large rivers (e.g. Weigel and Dimick, 2011; Sarkar et al., 2017), which corresponds well with findings from smaller streams and rivers (Hering et al., 2006; Feld and Hering, 2007).

Most assessments used either field intensive (50.0%) or field rapid (27.9%) data collection methodology (Fig. 5). This result clearly reflects a certain need for extensive sampling of biota to represent status of LFRs, and which can be only partially replaced by modern remote methods, even if collection of biological data is time consuming and resource intensive (e.g. Flotemersch et al., 2011). However, besides conventional methodologies, innovative methodological approaches became increasingly implemented. For example, Dzubakova et al., (2015) applied LiDAR imagery to evaluate the dynamics of lateral connectivity in river floodplain habitats, and similarly, Karim et al. (2014) developed a method to quantify connectivity (timing, duration) of floodplain wetlands over space and time using high resolution laser altimetry. A large majority of studies measured ecological condition against a reference; however, the method used to define reference conditions varied widely (Fig. 6), with designation of reference sites (29.8%) and modelling stressor-response relationships (29.8%) being equally most important. In contrast, half of the studies did not describe how natural variation was partitioned from human impacts (Fig. 7). When natural variation was addressed, most studies used site-based classifications (i.e. evaluation of sites in major typological classes) or focused on a single habitat type for filtering the role of natural environmental variation to detect perturbation effects (22.8%, Fig. 7). These approaches generally concur with those used in smaller streams and rivers (see Roset et al., 2007; Hermoso and Linke, 2012).

Conservation, rehabilitation and relationship with ecosystem services

Studies addressing management actions were more rehabilitation than conservation oriented. This is probably due to the typically high levels of human use throughout LFRs. Also, although systematic conservation planning exercises may be done at large spatial scales, selection of areas for conservation focus is typically at finer scales (i.e. among stream segments and their associated watersheds) within large river systems (Esselman and Allan, 2011; Hermoso et al., 2011; Dolezsai et al., 2015). These studies do not deal with the peculiarities of LFRs by addressing different scales, which are only indirectly related to the conservation management of LFRs. Our review suggests that systematic approaches that select among different reaches and floodplain habitats within the potamal section of LFRs are relatively rare. We also found that although floodplain habitats and their associated main stem section are often the focus of large scale rehabilitation projects (e.g. Tockner and Schiemer, 1997; Whalen et al., 2002), these areas are selected rather haphazardly or based on their ecological status relative to a small number of potential candidate sites (Buijse et al., 2002; Jungwirth et al., 2002; Sommerwerk et al., 2010; Hein et al., 2016). Most rehabilitation efforts targeted the enhancement of habitat at small spatial extents (e.g. hundreds of meters to a few kilometres; see e.g. Thomas et al., 2015; Morandi et al., 2017) or focused on increasing lateral connectivity between the main channel and the floodplain (see e.g. Jacobson et al., 2011; Riguier et al., 2015; Kozak et al., 2016). The emergent general conclusion of the studies is: although in many cases rehabilitation activities enhanced habitat conditions and increased biodiversity to some degree, the outcome of the rehabilitation depended greatly on the selected abiotic and biotic variables, the spatial scale of the rehabilitation activity and the temporal scales considered for evaluating rehabilitation effects (Bernhardt et al., 2005; Palmer et al., 2010; Muhar et al., 2016). Prime reasons for failure of rehabilitation activities in LFRs were: i) the overarching effect of catchment or landscape level alterations, ii) inadequate improvement of instream habitat quality, iii) limited recolonization potential of the species pool, and iv) the lack of a diverse species pool in the altered catchments (Palmer et al., 2010; Tonkin et al., 2014; Muhar et al., 2016; Stoll et al., 2016).

We found surprisingly few papers (1.6%) addressing ecosystem services in LFRs. Although the number of studies on ecosystem services of freshwaters is generally increasing, Hanna et al. (2018) concluded these are almost exclusively quantifying ecosystem services at the scale of watersheds or across multiple watersheds. Consequently, this review agrees with Hanna et al. (2018) that evaluation of ecosystem services at the scale of LFRs is still rare. Ecosystem services studies also did not distinguish between the different functional units of river-floodplain habitat types (i.e. eupotamon, parapotamon, plesiopotamon) and their potential role in ecosystem services provision. An important exception is Schindler et al. (2014), who reviewed the effects of 38 floodplain management interventions on 21 ecosystem services. The authors found that rehabilitation measures generally improved the multifunctionality of the riverscape and resulted in win-win situations for enhancing the overall supply of ecosystem services (Schindler et al., 2014, 2016). Overall, the number of studies is still too low for meaningful analyses of the relationships between biodiversity conservation, maintenance of ecological condition and ecosystem services in LFRs (but see e.g. Thorp et al., 2010 for a more general paper).

Conclusions and suggestions for future research

Our systematic review revealed a strong geographic bias in the literature toward developed countries in Europe and North America. Given systematically high levels of threat to rivers around the globe (Vörösmarty et al. 2010), this is a substantial research gap and further studies are clearly required in less examined continents to better understand the ecology and conservation management of LFRs. In fact, conservation management of LFRs could significantly benefit from intensive research in currently less studied and still relatively intact LFRs in terms of spatial organization of habitat patterns, functional connectivity between them and potential reference conditions. Europe and North America have a long history of intense, large scale river engineering and use and thus, largely lack stretches appropriate for use as natural references. Potential reference LFRs, however, may still exist in less developed areas, such as areas of South America, Asia and Africa. Even if they occur in markedly different biogeographic realms than more altered LFRs, which limits their applicability as reference for taxonomic evaluations, they can still provide reference for functional composition of species communities as well as functional connectivity between resources and thus, will enhance our understanding of ecological function and processes in LFRs. We acknowledge that ecology of LFRs has been investigated in some areas that our review indicates are understudied (e.g. in Russia and China), where results have simply not yet reached the English-dominated contemporary scientific literature.

Our review suggests that most ecological assessments to date have adopted use of classical biotic index based evaluations (e.g. Angermeier and Karr, 1994; Karr, 1999). Not surprisingly, these evaluations rely largely on fish and benthic invertebrate assemblages. Both taxa have a relatively long history of development and application as indicators (Karr, 1981), with established sampling guidance and diagnostic tools, particularly in small rivers (Herman and Nejadhashemi, 2015). However, it should be noted that the number of articles specifically addressing application of biotic indices in LFRs is low. Many studies applied sampling at the watershed level, where samples from small streams to large rivers were evaluated using the same methodological protocol. In addition, most studies evaluated the status of main stem river habitats only (see e.g. Flotemersch et al., 2006; Whittier et al., 2007; Birk et al., 2012a; Ruaro and Gubiani, 2013), but did not specifically consider the peculiarities of LFRs. The number of articles addressing the ecological assessment of the whole riverine landscape (i.e. all types of riverscape habitats) was very small (Fig. 3).

Most indices used to evaluate biotic condition were not specific to LFRs. A notable exception is the floodplain index, which was developed to assess ecological condition of and lateral connectivity between individual water bodies within a floodplain landscape (multiple riverine habitat types). The index is based on species specific habitat preferences, which were assigned to indicator values (Chovanec and Waringer, 2001; Chovanec et al., 2005; Illyova and Matecni, 2014; Šporka et al., 2016; Funk et al., 2017). The index is an effective biological indicator of spatial and temporal changes in the lateral hydrological connectivity of river-floodplain functional habitat types (Chovanec et al., 2005; Šporka et al., 2016). Since dynamic lateral hydrological connectivity is one of the most important determinants of river-floodplain systems (Bayley, 1995; Johnson et al., 1995; Ward et al., 2001), the floodplain index may serve as key measure for evaluating the ecological condition of LFRs at the landscape scale. However, the floodplain index cannot be related to specific stressors and thus, may not effectively indicate the summed effect of different physical, chemical and biological stressors on biota and the LFR system in general. Therefore, other metrics are also necessary for the effective evaluation of the ecological condition of LFRs, which we briefly review here to guide future assessment research.

To quantify the degree of landscape alteration and assess ecological condition it is necessary to determine how much area of the original landscape has been lost, and how structural components and functional processes have been altered (Beechie et al., 2010; Peipoch et al., 2015). However, most biotic indices quantify only site level alteration and consequently do not consider or provide information on habitat loss and alteration – including spatial configuration and diversity of different habitat types - at the landscape level. LFRs suffered most from large scale loss of their original habitat due to increasing agricultural land use (Tockner and Stanford, 2002). Therefore, we suggest that assessments of LFRs should explicitly incorporate landscape level metrics of habitat alteration. Patch based evaluations of habitat quantity, complexity (i.e. configuration, diversity, connectivity of patches) and quality are routinely used in terrestrial landscape ecology (Pascual-Hortal and Saura, 2006; Lausch et al., 2015). However, their application in riverscape ecology warrants greater consideration (Erős and Grant, 2015), particularly in ecological assessment and conservation management. For example, environmental history provides an excellent approach for quantifying spatial and temporal changes in habitat quantity, configuration and diversity in LFRs (see e.g. Hohensinner et al., 2004; Farkas-Iványi and Trájer, 2015). Further, graph theoretic and other network based methods are increasingly applied to quantify connectivity relationships (Erős et al., 2012; Wohl et al., 2018). In addition, since lateral diversity of habitats and the biota is a key component of LFRs, the floodplain index mentioned above can serve as a coarse measure for spatial and temporal changes in hydrologic connectivity and its effects on biota. Modelling stressor response relationships with more effective analytical tools (e.g. machine learning methods, Bayesian models) may lead to better predictive indices in the future (Kuehne et al., 2017). These tools could better incorporate both structural and functional parameters. In fact, measures of ecosystem function (e.g. water retention, organic matter decomposition, production of trophic levels) are still underutilized in river management (von Schiller et al., 2017). Overall, what is still missing is a more holistic approach, i.e. the effective integration of the different approaches in a unified assessment framework (but see Flotemersch et al., 2016 for an approach at the watershed level).

Classic indices are routinely used for determining quality of the biota (Birk et al., 2012a, 2012b; Ruaro and Gubiani, 2013). However, local, single habitat and single index based assessments may fail to correctly reflect the broader ecological condition of LFRs and the alteration of the riverscape (see also Moss et al., 2008), particularly if areas lost by water regulation, land use alteration and other kinds of habitat modification are not considered. For example, a riverscape that has lost 90% of its original area may show good ecological condition at the local scale, due to remnant river-floodplain segments with sufficient habitat quality and connectivity, while at the catchment scale the riverscape is seriously altered. This narrow focus on the site scale and single elements of the riverscape is standard in most environmental assessments of LFRs. For example, in Hungary the assessment of the ecological condition of large floodplain rivers (Danube, Tisza) is exclusively based on monitoring the main channel and the floodable area along the river. Oxbows and former side arms in the historic floodplain are treated as lakes in the ecological assessment procedure and their ecological condition is evaluated based on the criteria established for lakes. The formerly vast floodplain area cut off by levees for flood protection is considered terrestrial habitat and thus not evaluated at all. In the German environmental assessment system for the WFD, even the active floodplain is not considered part of the water body and thus not addressed by monitoring. Approaches that restrict the riverscape to the floodplain remaining between levees fall short in assessing the ecological condition, because they ignore the original extent of the riverscape as reference. Such an assessment largely underestimates the loss of habitats, neglects lateral fragmentation effects and consequently cannot estimate the full losses due to human alteration of LFRs. We are fully aware that many historical floodplain areas are irreversibly lost; however, we argue for their conceptual consideration as functional habitats. For fish in particular, small floodplain water bodies that are infrequently connected with the main channel have been identified as key habitats for floodplain specialists (Schomaker and Wolter, 2011). We argue that integrating landscape level and local scale evaluations will lead to more effective evaluation of the ecological condition of LFRs. The joint application of the different types of indicators of environmental quantity, complexity and quality together with information on ecological threat indices (Paukert et al., 2011; Tulloch et al., 2015) will allow development of more informed conservation and management decisions.

Limitations on conservation resources means that it is critically important to optimize solutions across multiple conservation/rehabilitation purposes and/or other ecosystem services. As indicated by the very low number of articles on ecosystem services of LFRs, this challenge remains widely unaddressed. Furthermore, studies that specifically quantify trade-off relationships between different ecosystem services and biodiversity conservation or the maintenance of ecological condition are virtually lacking for LFRs. Watershed level studies offer examples of how to optimize land use for the delivery of ecosystem services and for conservation and/or rehabilitation of biota (e.g. Doody et al., 2016; Terrado et al., 2016; Erős et al., 2018). Similar studies should be conducted in the segments of LFRs, because examining trade-off relationships at larger scales and spatial extents may require different approaches and result in different management outcomes (Erős et al., 2018; Hanna et al., 2018).

In LFRs, selecting areas for conservation or rehabilitation should focus on reaches sufficiently large to maintain a diverse array of floodplain habitat types and a diverse biotic community (Hein et al., 2016). Spatial prioritization and optimization approaches could help to define river segments 1) of priority for conservation and/or rehabilitation (e.g. biodiversity hotspots, regeneration potential, nutrient retention, ecotourism), 2) primarily for human use (e.g. infrastructure, housing, gravel mining), and 3) for both conservation functions and human use shared according to societal needs and intentions. Taking the "land sharing versus land sparing debate" (see Fisher et al., 2014; Shackelford et al, 2015) into the water would be useful for developing more effective conservation decisions that address societal concerns, especially for LFRs, where human needs for water seem to be in special conflict with conservation aims (Arthington et al., 2010; Sommerwerk et al., 2010).

In summary, our review of the ecological research identified substantial challenges in assessing and managing LFRs, primarily emerging from an insufficient recognition of the spatial (longitudinal and lateral) and temporal complexity of LFRs. This review highlights research gaps and emphasizes the importance of developing more holistic indicators and assessment schemes of ecological condition that can better reveal landscape level changes in the structure and functioning of LFRs. Improved assessment tools will help to effectively select areas for conservation and rehabilitation, and evaluate those areas which are rehabilitated. Indeed, as human use of water and land is increasing, developing effective spatial prioritization tools becomes more important. Empirical research in this field can aid in solving conflicts between socio-economic demands for ecosystem services and nature conservation in LFRs.

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Captions to figures

Figure 1. A schematic representation of the purpose of this study for exploring the assessment of ecological condition and its relationship with ecosystem services and for showing the balance between conserving and/or rehabilitating nature and utilizing it for human purposes appearing in peer-reviewed scientific articles.

Figure 2. The distribution of the studies among continents and ecoregions. Letters indicate the type of the article as follows. EC, assessment of ecological condition; C, conservation; R, rehabilitation/restoration; ES, ecosystem services; BDM, biodiversity inventory or monitoring; C/ES, trade-off between C and ES.

Figure 3. The percentage (%) distribution of the studies among the different river-floodplain habitat types. Abbreviations for the functional habitat types are as follows. MR, main river (eupotamon); FP1, floodplain 1 (parapotamon, plesiopotamon); FP2, floodplain 2 (paleopotamon); FPA, former riverscape habitats in the flood protected area (oxbows etc); RIP, riparian areas.

Figure 4. Representation (percentage % of all studies) of different taxonomic groups used to evaluate ecological condition in EC studies.

Figure 5. The percentage (%) distribution of the type of data collection methods for the assessment of ecological condition among the articles. Field-intensive (>0.5 day site^{-1}), field-rapid (<0.5 day site^{-1}), desktop (based primarily on spatial and/or remotely sensed data), expert (synthesis of expert knowledge).

Figure 6. The percentage (%) distribution of the methods of defining reference condition among the articles. Basis of comparison for ecological condition: Site, selection of reference sites; BPJ, best professional judgement or expert knowledge; Historical, based on empirically derived estimate of historical condition; Model, models reference conditions using empirical approach; Ambient, uses measured range of response.

Figure 7. The percentage (%) distribution of the methods among EC articles that partitioned natural variation from anthropogenic impacts. The categories used were as follows. Classification, categorization of sites based on their habitat characteristics; Untest, univariate tests of factors; Model, models which account for natural gradients; RGR, restricting geographic range.

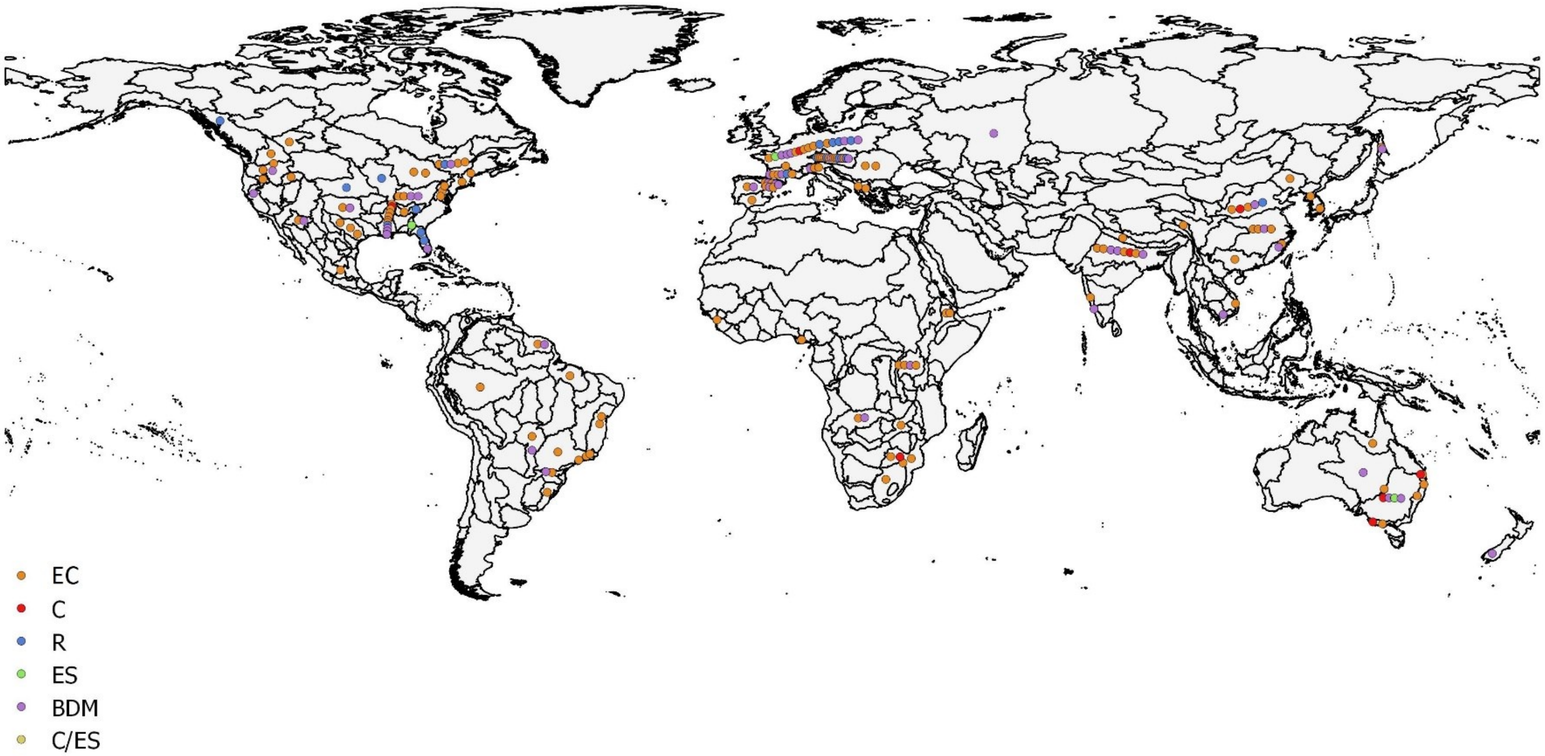

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Fig. 1.

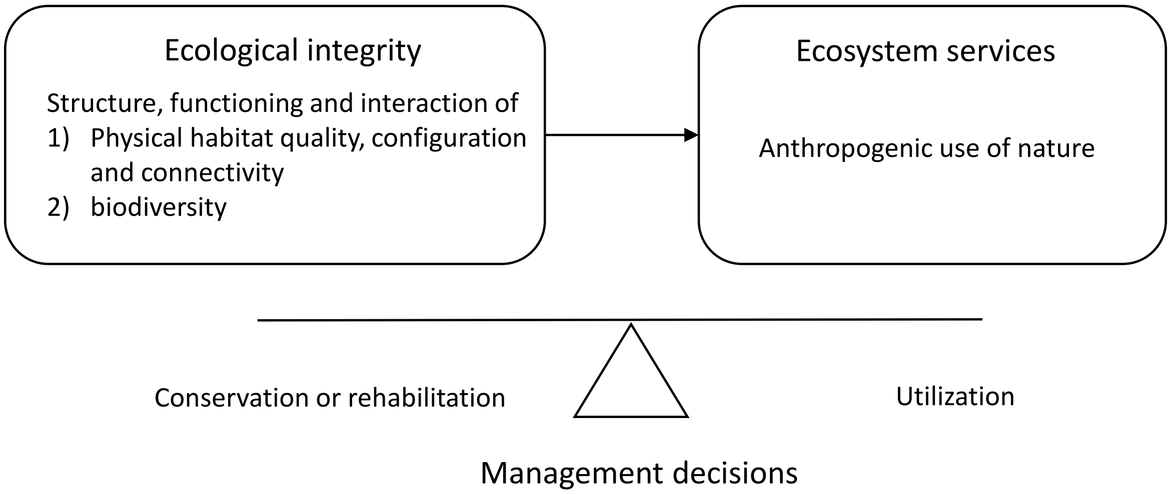

Ecological integrity
Structure, functioning and interaction of
1) Physical habitat quality, configuration and connectivity
2) biodiversity
Ecosystem services
Anthropogenic use of nature
Conservation or rehabilitation
Utilization
Management decisions

Fig. 3.

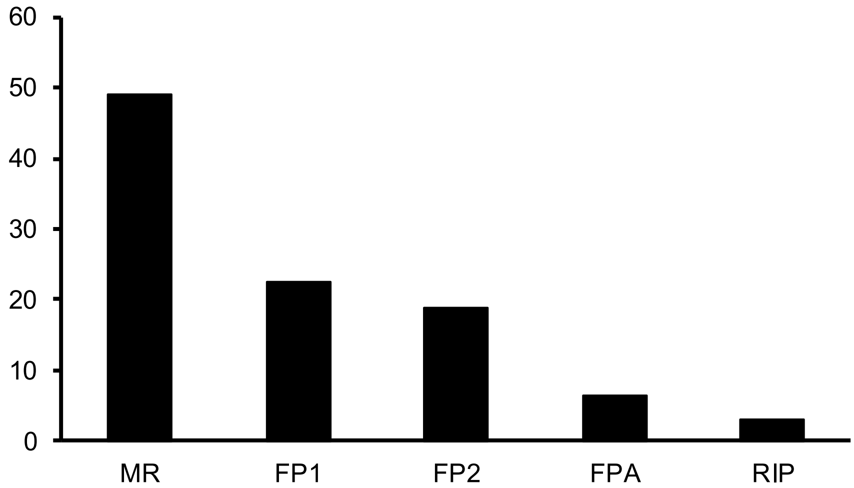


Fig. 4.

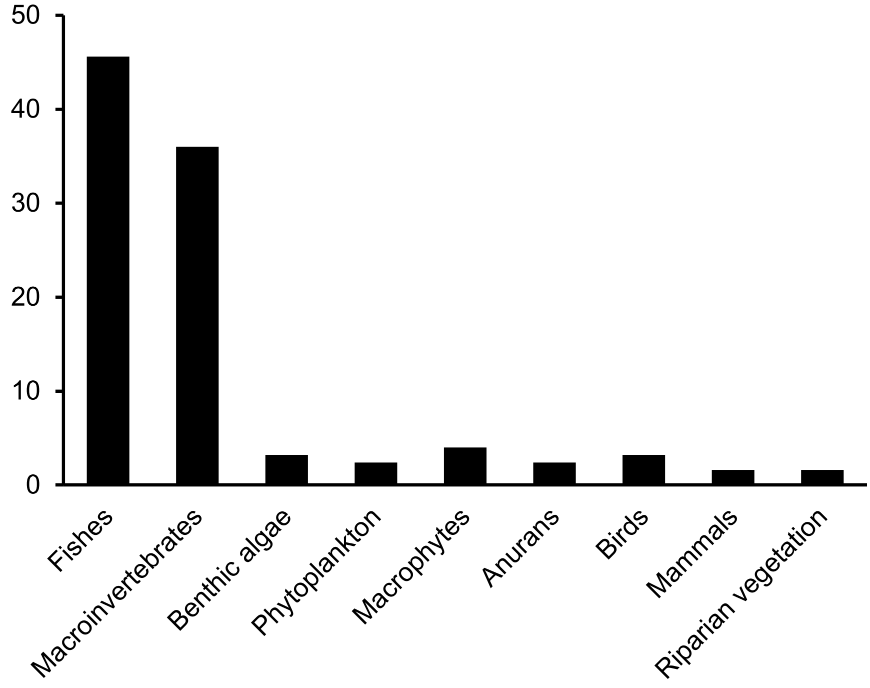
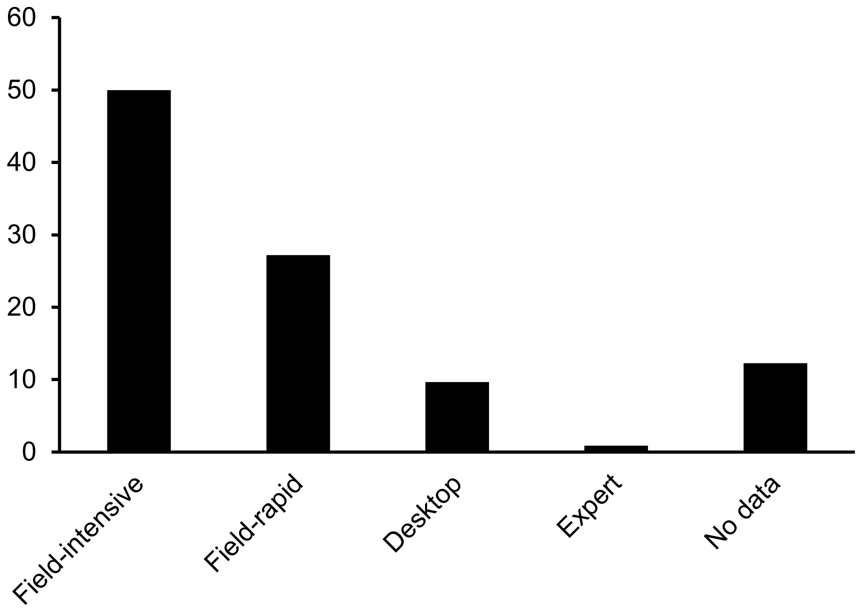

50
40
30
20
10
0
Fishes
Macroinvertebrates
Benthic algae
Phytoplankton
Macrophytes
Anurans
Birds
Mammals
Riparian vegetation

Fig. 5.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50
40
30
20
10
0
Field-intensive
Field-rapid
Desktop
Expert
No data

Fig. 6.

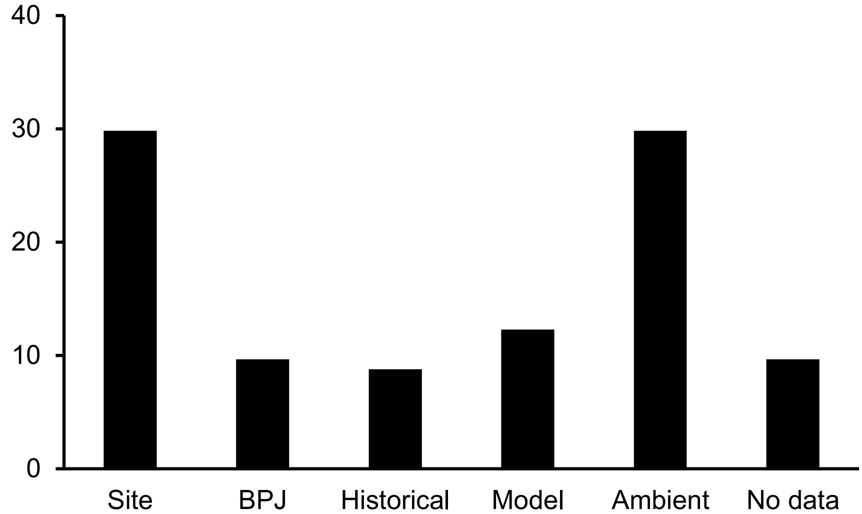
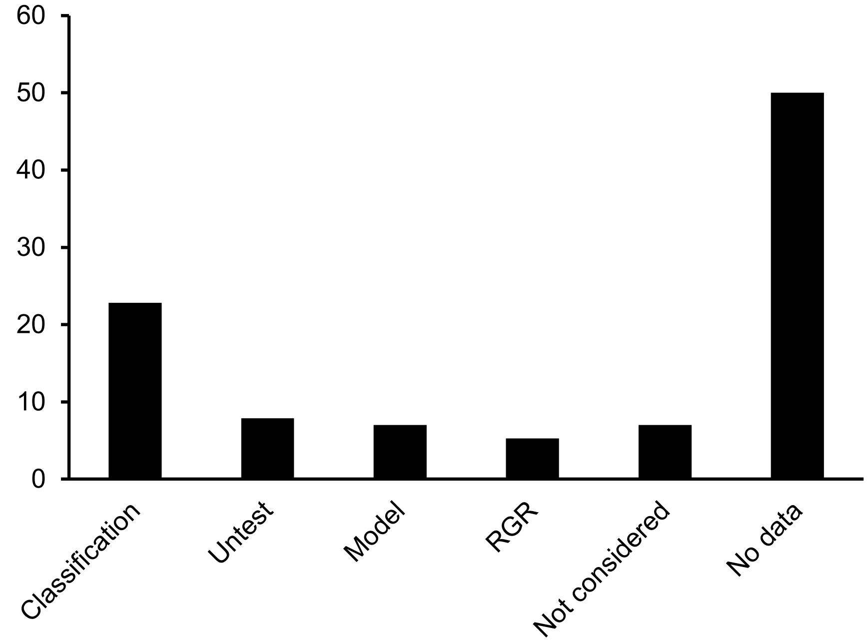


Fig. 7.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50
40
30
20
10
0
Classification
Untest
Model
RGR
Not considered
No data