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VULNERABILITY OF AQUATIC ECOSYSTEMS

Responses of riparian plant communities and water quality after 8 years of passive ecological restoration using a BACI design

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Abstract This study investigated the consequences of passive ecological restoration on a riparian habitat and on water quality. The restoration plan consists of excluding livestock by constructing fences along an entire stream 1 m from the stream bed, with the assumption that recovering riparian habitat will restore their ecological processes (e.g., filtration, soil stabilization). We measured responses of riparian plant communities and physico-chemical water quality. We presented data from an 8-year before-after control-impact design across a reference stream and a restored stream in a rural landscape in Normandy, France. Restoration appeared to modify plant communities. After 8 years of restoration, the restored stream had a complex riparian bank, similar to that of the reference stream, with an increase in the number of trees, a decrease in bare soil, and an increase in habitat heterogeneity. Despite this modification, water quality did not improve. The same low water quality in the reference stream demonstrated the need for a watershed-scale approach and for actions to improve agricultural practices before implementing restoration practices at a smaller scale. Nonetheless, the lack of improved water quality does not necessarily mean that the restoration failed. Other functions and services can be provided by excluding livestock.

Keywords BACI design · Headwater · Bundles of ecosystem services · Passive ecological restoration · Riparian plant communities · Watershed · Water quality

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Introduction

In a dense river network, headwaters (first- and second-order streams) can be abundant and represent therefore major components (Lowe & Likens, 2005; Finn et al., 2011). Due to their interlinked position between terrestrial and aquatic systems, riparian zones play a crucial role not only in the ecological quality of headwaters but also beyond the watershed, by cumulative effects. Plant communities in this ecotone fulfill essential functions. Many studies demonstrated that riparian vegetation (1) improves water quality by filtering sediments, pesticides, and particulate organic matter (e.g., Lowrance et al., 1984; Schilling & Jacobson, 2014) and decreases nitrate, phosphorous, and other nutrient concentrations in groundwater (e.g., Pinay & Decamps, 1988; Skłodowski et al., 2014) (2) maintains biodiversity by providing heterogeneous habitats and a physical structure for fauna (e.g., Naiman et al., 1993; Suurkuukka et al., 2014), (3) reduces bank erosion and stabilizes the bank (e.g., Abernethy & Rutherfurd, 1998; Rood et al., 2014), (4) affects river morphology and fluvial processes (Camporeale et al., 2013), (5) increases in-stream aquatic diversity, and (6) provides recreational opportunities (Tunstall et al., 2000). Through these functions, ecosystems can provide goods and services for human societies and shift toward a more utilitarian role by providing ecosystem services (Millennium Ecosystem Assessment, 2005; Palmer et al., 2014).

Watershed disturbance caused by agricultural expansion and intensification in the twentieth century may affect headwater health and ecosystem services provided by riparian zones (Harding et al., 1998; Richardson & Danehy, 2007; Palmer et al., 2014). Ecological restoration appears a promising response to this degradation to maintain the essential role of riparian ecosystems (Clewell & Aronson, 2006; Naiman et al., 2010). Through the diversity of existing methods to restore streams (McIver & Starr, 2001), from gravel deposits in stream beds to channel reconfiguration, a recent meta-analysis of 644 river restoration projects concluded that most focused on physical manipulation of channels, while few (17%) implemented riparian restoration (Palmer et al., 2014). For rivers degraded by intensive livestock grazing, however, restoration of riparian zones is essential. Livestock grazing directly increases nutrient inputs (from manure), alters vegetation due to trampling and consumption, compacts soil, and collapses stream banks due to trampling (Armour et al., 1991; Belsky et al., 1999; del Rosario et al., 2002; Sweeney et al., 2004). These effects negatively affect upland erosion, turbidity, nutrient concentrations in stream, stream shading, stream temperature, stream channel morphology, and aquatic and riparian wildlife (Knapp & Matthews, 1996; Belsky et al., 1999; Sweeney et al., 2004; Hansen & Budy, 2011).

Riparian zone restoration can be “passive,” in which the disturbance is identified and removed and spontaneous succession is used, or “active,” in which technical intervention such as planting or weeding is performed (Society for Ecological Restoration, 2004). Frequently, riparian restoration projects plant riparian vegetation and/or remove non-native riparian vegetation (Holmes et al., 2005; Palmer et al., 2014). However, passive ecological restoration (PER), such as livestock exclosure, is promoted (McIver & Starr, 2001; Prach & Hobbs, 2008) because it saves time, effort, and money and can be implemented along an entire stream, whereas active restoration often occurs only on a short river section (Jähnig et al., 2010). Many studies report the success of PER on plant communities, with a decrease in bare soil, an increase in tree cover, and recovery of riparian communities (Kauffman et al., 1997; Hansen & Budy, 2011; Forget et al., 2013; O’Donnell et al., 2014), even if PER is not appropriate in all situations and is less successful in multifactor contexts (McIver & Starr, 2001; Sarr, 2002). Although it is well documented that excluding livestock improves riparian habitat recovery, the few studies assessing ecosystem function after PER have conflicting results. Some studies observed a decrease in nutrient concentrations after PER, suggesting recovery of the filtering function of riparian zones (Van Velson, 1979; Li et al., 1994; Sweeney et al., 2004; Hansen & Budy, 2011). Other studies observed little or no difference in stream nutrient concentrations after PER (McKergow et al., 2003; Hughes & Quinn, 2014; Summers et al., 2014).

In general, all restoration projects fail to assess fauna components and ecosystem processes, and questions remain about the effectiveness of restoration for fish, invertebrates, and water quality, because the assumption is that by restoring the riparian ecosystem, ecological functions and processes associated with riparian zone recovery will follow (Palmer et al., 2014). Responses of fish communities, invertebrate communities, and water quality have not shown consistent trends and remain problematic. Several studies observed a positive effect of river restoration on fish (Paller et al., 2000; Raposa, 2002; Whiteway et al., 2010; Lorenz et al., 2013), invertebrates (Muotka et al., 2002; Raposa, 2002; Sudduth & Meyer, 2006; Sarriquet et al., 2007), and water quality (Osborne & Kovacic, 1993; Sarriquet et al., 2007; Hansen & Budy, 2011; Richardson et al., 2011), but others observed little or no difference on fish (Moerke & Lamberti, 2003; Shields et al., 2003; Moerke et al., 2004; Baldigo & Warren, 2008; Stoll et al., 2013, 2014), invertebrates (Moerke et al., 2004; Jähnig et al., 2010), or water quality (Hansen & Budy, 2011; Kail et al., 2012). Recent reviews (Roni et al., 2002; Nilsson et al., 2014; Palmer et al., 2014) mention this lack of response by fauna components and water quality. This failure can be explained by the fact that (1) habitat improvement offers few benefits for functional restoration; thus, the previous assumption is false (Jähnig et al., 2010; Palmer et al., 2010; Kail et al., 2012), (2) restoration projects are implemented too locally without considering watershed effects and integration at this large scale (Wohl et al., 2005; Stoll et al., 2013, 2014; Pander & Geist, 2013; Pander et al., 2014), and (3) monitoring occurs at an inappropriate temporal scale (Bash & Ryan, 2002; Roni et al., 2008). Since natural variability makes each stream unique, it is meaningless to evaluate success of riparian restoration based only on comparing the number of species, composition, and structure of a stream's plant community to those of a non-existent reference stream. In contrast, focusing on the recovery of processes and functions of the riparian ecosystem may be a better indicator of restoration success (Wohl et al., 2005).

In the present study, we investigate biological responses through plant communities, and functional responses, through water quality, of riparian restoration. This paper examines the influence of riparian PER on (1) riparian plant communities and (2) water characteristics in headwaters in Normandy (France) and tests the assumption that restoring a stream's riparian ecosystem will restore its ecological functioning. The goal is to assess how PER can benefit riparian plant communities and water quality after several years. We compared data from an 8-year before-after control-impact (BACI) design on a restored stream and a reference stream in the same watershed. Since riparian habitat heterogeneity is crucial for in-stream biodiversity (Le Pichon, 2006), evaluation focused not only on responses of plant community species richness and its similarity to that of the reference stream after PER, but also on habitat heterogeneity along the stream. We hypothesized that PER will (1) enable recovery of plant communities similar to those of the reference stream, with an increase in tree recruitment and a decrease in bare soil, (2) increase riparian habitat heterogeneity by increasing natural tree recruitment, (3) reduce water stream temperature by increasing riparian shading, and (4) reduce nutrient input in stream due to the absence of cattle and recovery of the riparian habitat's filtering function.

Methods

Study site

The study site is located on the Oir River watershed, 20 km east of Mont Saint-Michel Bay (Fig. 1, southern Normandy, France). This watershed has been intensively studied by the French National Institute of Agricultural Research (INRA) because it is a large nursery for wild Atlantic salmon and brown trout (Baglinière et al., 2005) offering a long-term series of ecological datasets. We focused on two second-order tributaries of the Oir River: Vallée-Aux-Berges (VAB), the restored stream, and La Roche, the reference stream. These streams have similar widths (mean = 1 m) and lengths (mean = 4 km). Both occur in landscapes traditionally used for pasture and crops. Land use and agricultural practices changed little over the study period: 2004 (when the PER began) to 2012 (end of monitoring) (Bal et al., 2011).

Approach for riparian passive ecological restoration

VAB was highly degraded by cattle pressure (intensive trampling and grazing on the banks). In August 2004, river managers from local organizations and public agencies collaborated on a riparian restoration project to conserve migrating fish populations and to reach a good ecological state of surface water. A PER approach was suggested and tested on the VAB stream. Fences were erected along the entire stream 1 m from the stream bed and drinking troughs for cattle were installed along the stream. Since the farmers own the stream bank, the distance for the implementation of the fences from the stream bed is set only after debates between farmers and river managers. One meter from the stream bed is the maximum distance obtained by river manager; a wider buffer zone is not that easy to implement, because it would be a grazing land loss to farmers. The PER goal was to stop trampling and allow bank vegetation to recover without initiating or accelerating its recovery (i.e., no planting, seeding, or soil treatment). Unlike VAB, the entire length of La Roche has always been,

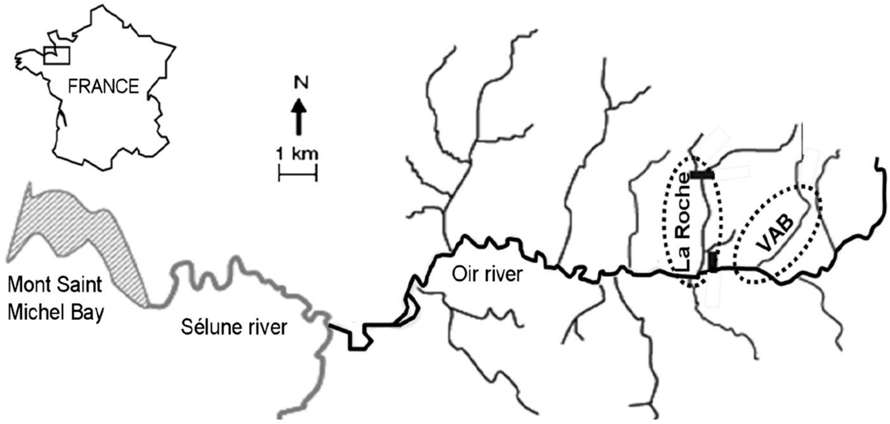

Fig. 1 Location of the study sites Vallée-Aux-Berges (VAB) and La Roche in the Oir River watershed

historically and due to current landowner practices, protected by fences from the impact of cattle, and its riparian vegetation has been relatively undisturbed by livestock pressure; thus, it was considered as a reference stream for riparian plant communities in the watershed.

Riparian vegetation monitoring

To test effects of PER on establishment of riparian plant communities, vegetation was monitored in May (corresponding to period where most species are present) 2004 (3 months before restoration), May 2005 (after 1 year of restoration), May 2006 (after 2 years), May 2010 (after 6 years), and May 2012 (after 8 years). To be able to compare community over years, the same monitoring and the same period are crucial. To compare riparian vegetation between the restored VAB stream and the reference stream, La Roche was monitored with the same vegetation monitoring protocol in May 2010 and May 2012 to evaluate variability in vegetation response to potential climate variations over these 2 years. For each stream, vegetation was monitored on the river bank at points equally distributed along the stream. A total of 36 permanent 1 m × 15 m plots were sampled along each stream. In each plot, we recorded the number of trees, the percentage of plot surface area covered by bare soil, and the percentage of plot surface covered by each species using the following scale: 0.5 for species covering less than 1%, 1 for species covering between 1 and 5%, 2 for 5 and 25%, 3 for 25 and 50%, 4 for 50 and 75%, and 5 for more than 75%.

Physico-chemical water sampling and analysis

VAB and La Roche water quality was monitored every month at the same location, downstream of each stream, from January 2004 (8 months before VAB restoration) to December 2012 (after 8 years of restoration). Each month, water level (cm), pH, temperature (°C), and electrical conductivity ($\mu S\ cm^{-1}$) were measured directly in the field. Two hundred and fifty mL of surface water was taken and stored in a refrigerator until laboratory analysis for NH_4^+, NO_3^-, and PO_4^{3-}. In the laboratory, NH_4^+, NO_3^-, and PO_4^{3-} were determined by colorimetric methods with WTW kit (NH_4^+: 14752, NO_3^-: 14773, and PO_4^{3-}: 14848).

Data analysis

To analyze effects of riparian PER on riparian plant communities, we compared means of species richness, percentage of bare soil, and the number of trees of VAB in 2004 (3 months before restoration), VAB in 2012 (after 8 years), and the La Roche reference in 2012. Since no obvious changes in the plant community were observed at La Roche between 2010 and 2012, we only performed analysis with La Roche 2012 (see supplementary material, no significant difference in species richness, in bare soil, and in number of trees in La Roche in 2010 and in 2012). Since the data did not conform to parametric conditions, we performed Kruskal–Wallis and pairwise Wilcoxon tests with a *P* value adjustment according to the simple Bonferroni method, in which the p values are multiplied by the number of comparisons.

Changes in community structures following riparian PER were examined with correspondence analysis (CA) of the vegetation data from VAB in 2004, VAB in 2012, La Roche in 2010, and La Roche in 2012 (144 plots $\times$ 135 species). To analyze differences between VAB before restoration (VAB 2004) and VAB after 8 years of restoration (VAB 2012), we applied non-parametric multivariate analysis of variance (MANOVA) to the vegetation data (Anderson, 2001).

To assess riparian habitat modification after restoration, a measure of riparian vegetation heterogeneity along the stream was calculated for VAB in 2004, VAB in 2012, and the La Roche reference in 2012. According to Anderson et al. (2011), heterogeneity represents a difference in species composition between two plots and can be estimated by the Bray–Curtis dissimilarity index, based on species abundances (Raup & Crick, 1979). For each plot surveyed, the mean Bray–Curtis index was calculated between it and the other plots in the stream. An index of zero means that two plots have the same species composition (i.e., no heterogeneity between plots), while an index of 1 means that they have no species in common (i.e., high heterogeneity between plots).

To test impacts of PER on water quality, we conducted an 8-year BACI (Green, 1979) design. BACI design is used to measure effects of perturbation (i.e., restoration) on an ecosystem (i.e., the stream) by following two sites, "control" (i.e., the reference La Roche) and "impacted" (i.e., the restored VAB), before and after the perturbation. This approach considers natural variability among years by simultaneously monitoring water quality at the La Roche reference site and the restored VAB site before and after restoration. Since the natural variability among years at the restored site is estimated by that measured at the reference site, it is essential to select a reference site that is as similar as possible to the restored site to ensure that both sites have the same responses to natural variations in their environment and that the differences observed at the restored site before and after restoration are due to this (Duhaime & Pinel-Alloul, 2005). Since they lie within a mean of 2 km of each other in the same watershed, we assumed that VAB and La Roche were similar and equally influenced by any natural variability.

To analyze effects of PER on water quality variables, data were divided into a pre-PER period, January to August 2004 (i.e., 8 months), and a post-PER period, January 2006 to December 2012 (i.e., 7 years). The same period names ("pre-PER" and "post-PER") are used for La Roche even though it was not restored. Data from September 2004 to December 2005 were excluded from analysis because it was considered a transitional period.

Many studies have demonstrated the dependence of water quality variables on river water level (Helsel & Hirsch, 2002; Hughes & Quinn, 2014). To exclude effects of water level, we developed a linear model of each water quality variable (pH, conductivity, temperature, NH_4^+, NO_3^-, PO_4^{3-}, and turbidity) as a function of water level. We then compared means of model residuals (i.e., each water quality variable adjusted for water level among pre-PER La Roche, post-PER La Roche, pre-PER VAB, and post-PER VAB) using Kruskal–Wallis and pairwise Wilcoxon tests because data did not comply parametric conditions. Evidence of a restoration effect required observing a significant change at the restored VAB site pre- and post-PER and an absence of change at the La Roche reference site pre- and post-PER. If the same temporal changes were observed on both rivers, natural variability was considered as the agent of change rather than restoration.

All tests were performed using R 2.12.0 (R Development Core Team, 2010) with a $P = 0.05$ threshold using the "ade4" package (Dray et al., 2007) and the "vegan" package (Oksanen et al., 2008).

Results

Effect of restoration on plant communities

We found no significant difference in plant species richness at VAB after 8 years of PER (25.0 $\pm$ 0.7 in 2004 vs. 22.8 $\pm$ 1.4 in 2012, Fig. 2A), and the species richness of VAB was similar to that of the reference (22.3 $\pm$ 1.4 in La Roche 2012). However, after 8 years of PER, VAB had significantly less bare soil than before PER (7.7 $\pm$ 1.5% in 2004 vs. 0.7 $\pm$ 0.7% in 2012, Fig. 2B) and than the reference (2.5 $\pm$ 1.0% in La Roche 2012). The number of trees significantly increased after 8 years of PER (1.0 $\pm$ 0.2 in 2004 vs. 1.9 $\pm$ 0.2 in 2012, Fig. 2C) but was still lower than that present in the reference stream (3.2 $\pm$ 0.4 in La Roche 2012). The Bray–Curtis dissimilarity index was significantly higher in VAB after 8 years of PER than before PER (0.1 $\pm$ 0.01 in 2004 vs. 0.3 $\pm$ 0.02 in

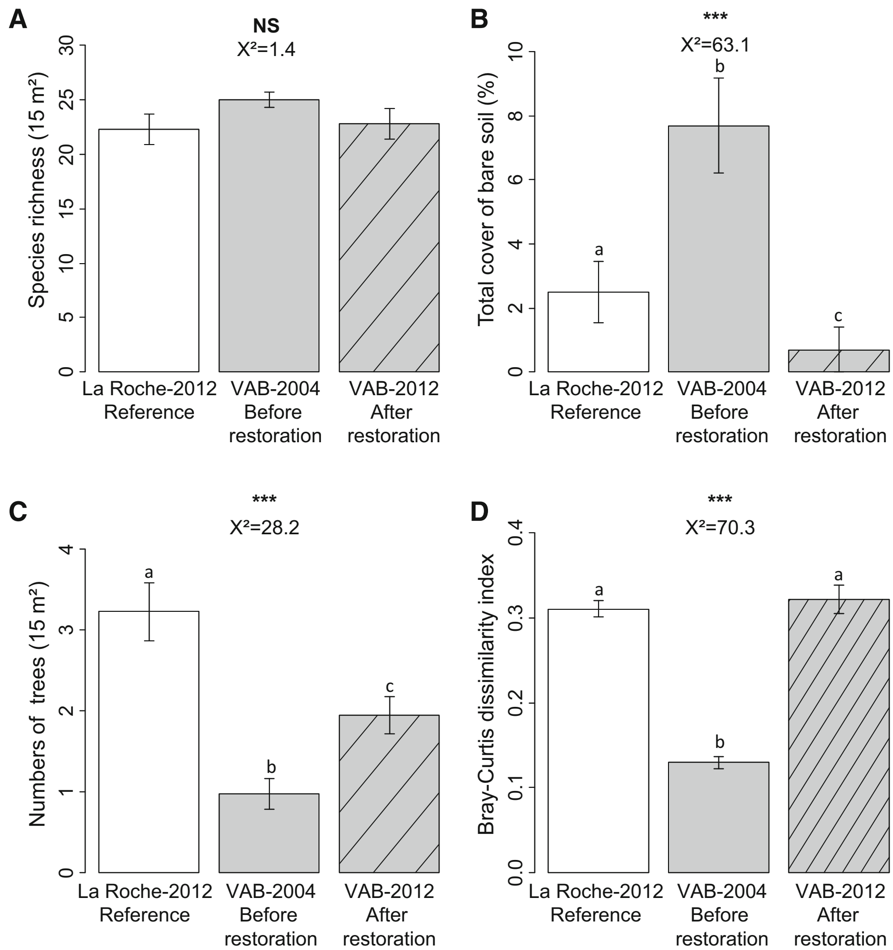

Fig. 2 Mean and standard error of **A** species richness (15 m^2), **B** bare soil (%), **C** number of trees (15 m^2), and **D** Bray–Curtis dissimilarity index in the La Roche reference in 2012 (*white bars*, $n = 36$ plots) and the restored VAB stream in 2004 (before restoration, *gray bars*, $n = 36$ plots) and in 2012 (after 8 years of restoration, *shaded gray bars*, n = 36 plots). The X^2 of Kruskal–Wallis tests performed are shown above the bars (***$P < 0.001$, *NS* non-significant); *bars* with the *same letters* have no significant differences according to pairwise Wilcoxon multiple comparisons with Holm P adjustment

2012, Fig. 2D) and approached the values of the reference (0.3 ± 0.01 in La Roche 2012), indicating an increase in vegetation heterogeneity along the stream after PER.

Non-parametric MANOVA showed that restoration significantly modified plant community composition after 8 years of restoration, between VAB in 2004 and VAB in 2012 (df $= 1$, $F = 27.8$, $P = 0.001$). The CA based on species abundance highlights these results. The first axis of the CA (8%; Fig. 3) discriminated VAB in 2004 from the other three communities (the reference stream in 2010 and 2012 and VAB in 2012, after 8 years of PER). The VAB stream in 2004 was composed of mesophyllous meadow species, such as *Trifolium repens* L. and *Lolium perenne* L. Species composition of VAB in 2012 was similar to that of the reference stream in 2010 and in 2012 and was characterized by more ruderal species (e.g., *Rubus fruticosus* L., *Geum urbanum* L., and *Galium aparine* L.), tree species (e.g., *Alnus glutinosa* (L.) Gaertn., *Quercus robur* L., and *Salix atrocinerea* Brot.) and forest species (e.g., *Hedera helix* L.). No obvious differences between La Roche in 2010 and in 2012 were observed. The barycenters of VAB's plant community between 2004 and 2012 suggested that the community converged towards that of the reference stream.

Effect of restoration on water quality

Temperatures in the restored VAB stream exhibited patterns similar to those in the La Roche reference, with a mean of 11.4°C (Fig. 4A). Likewise, pH and conductivity were similar between periods (ca. 7.02

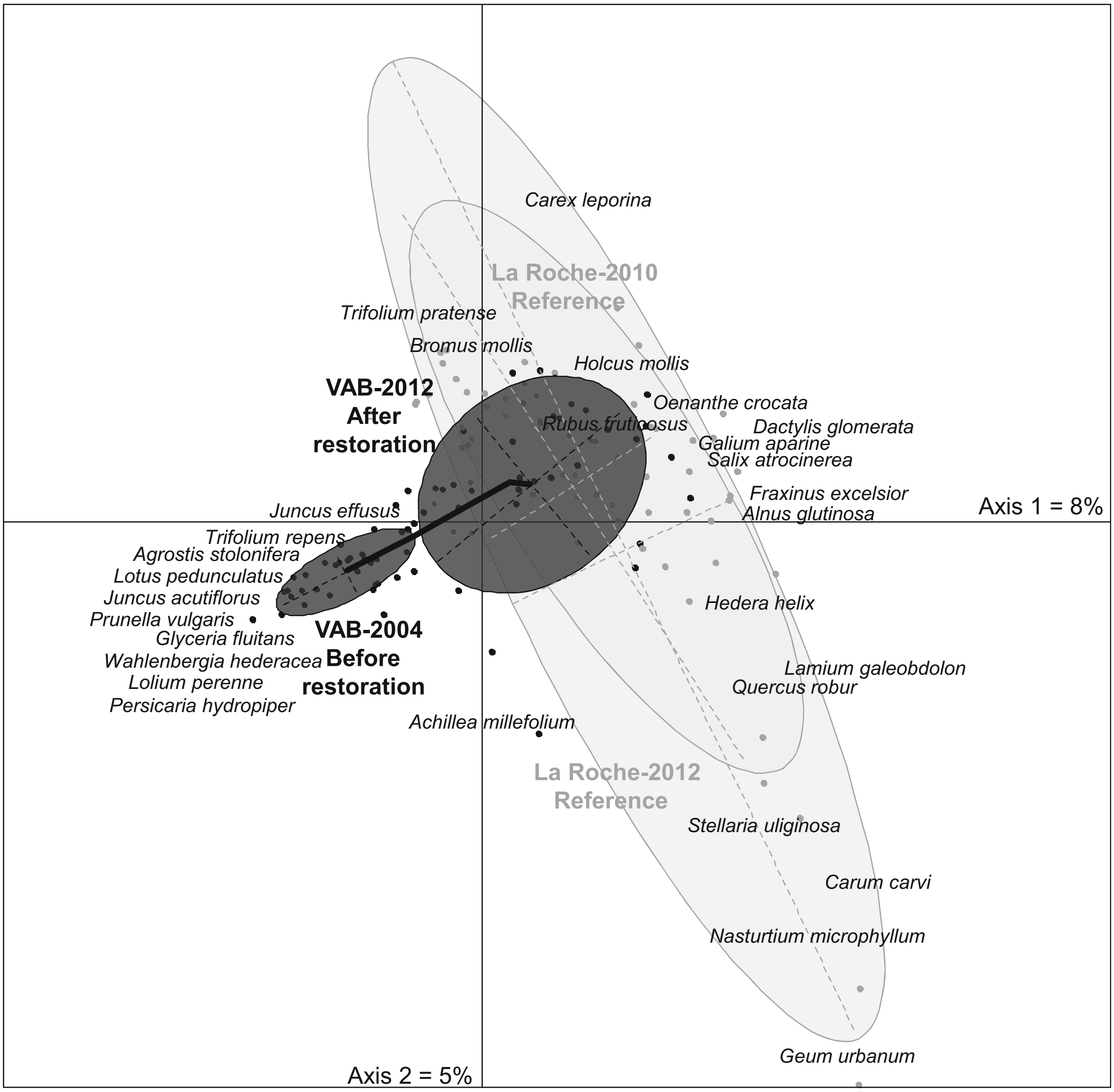

Fig. 3 Ordination plot of a correspondence analysis based on species abundance of the 135 species present in at least five plots (144 plots × 135 species) on the La Roche reference in 2010 (*light gray*, 36 plots) and in 2012 (*light gray*, 36 plots) and on restored VAB stream in 2004 (before restoration, *dark gray*, 36 plots) and in 2012 (after 8 years of restoration, dark gray, 36 plots). *Dark lines* represent the succession of vegetation in the restored stream from 2004 to 2005, 2005 to 2006, 2006 to 2010, and 2010 to 2012, according to the position of their barycenters. In the interests of clarity, only the 29 species with the highest contributions to axes are shown

and 201.8 μS cm^{-1}, respectively) in both streams (Fig. 4B, C). No significant difference was observed in pre- and post-PER adjusted NH_4^+, NO_3^-, and PO_4^{3-} concentrations (Fig. 4D–F). In both streams and both periods, PO_4^{3-} concentrations exceeded 0.12 mg l^{-1}, NH_4^- concentrations exceeded 0.07 mg l^{-1}, and NO_3^- concentrations exceeded 35.3 mg l^{-1}.

Discussion

After 8 years, PER changed the riparian plant communities. Livestock exclusion enabled an increase in tree recruitment, a decrease in bare soil, and a modification of the plant community toward riparian and ruderal species. The increase in heterogeneity

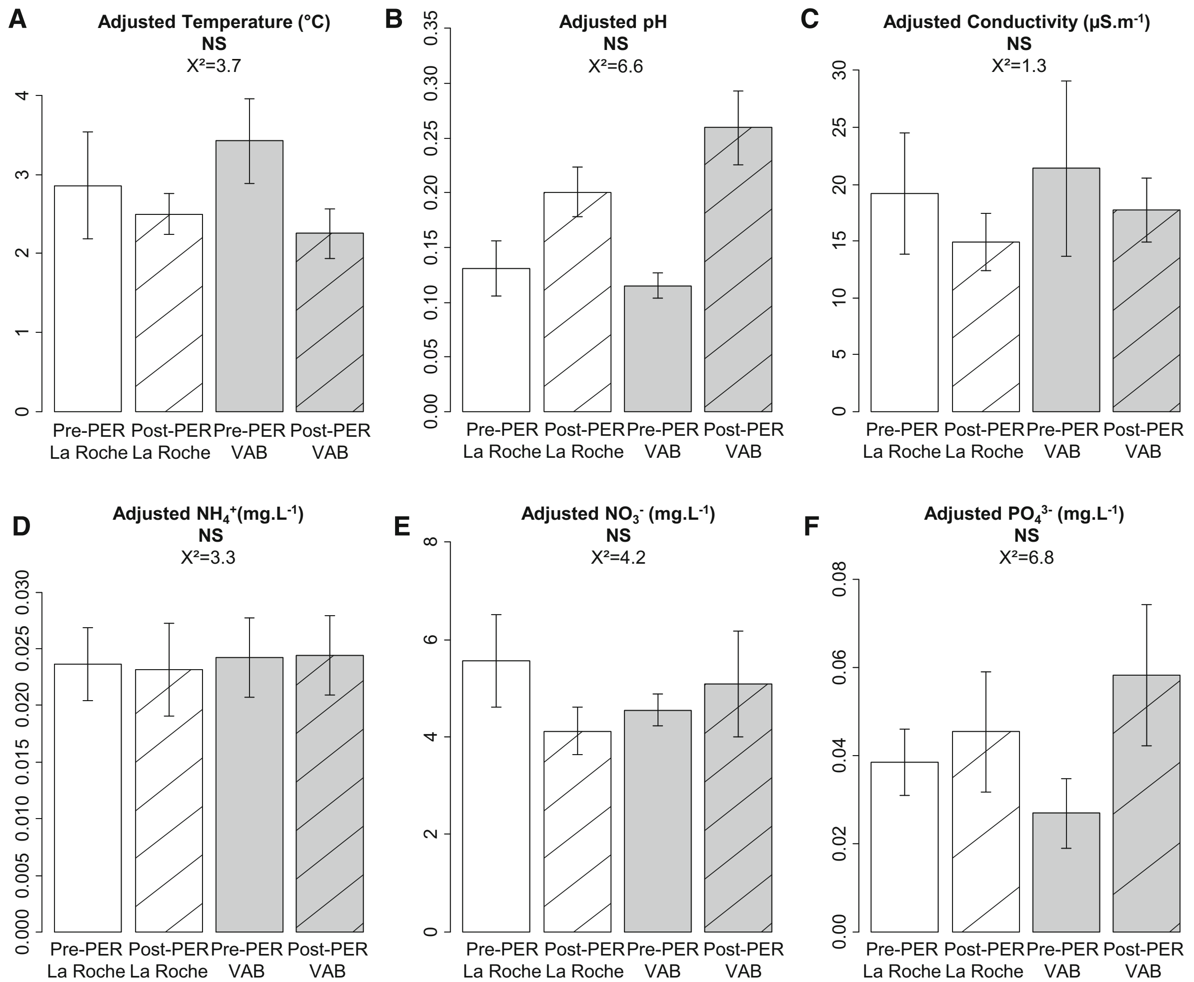

Fig. 4 Mean and 1 standard error (SE) of pre-passive ecological restoration (PER) (*unshaded bars*) and post-PER (*shaded bars*) water-level-adjusted **A** temperature, **B** pH, **C** conductivity, **D** NH_4^+, **E** NO_3^-, and **F** PO_4^{3-} in the La Roche reference stream (*white bars*) and in the restored VAB stream (*gray bars*). The X^2 of Kruskal–Wallis tests performed are shown above the bars (NS: non-significant)

clearly reflects the change in plant community composition and tree recruitment; the latter is not as homogenous and evenly spaced and linear along the stream as tree planting, as it is recommended in active restoration (McIver & Starr, 2001). Since livestock grazing reduces riparian vegetation and the riparian stream bank surface (Armour et al., 1991) and decreases tree germination (Dembélé et al., 2006), cattle exclusion and the subsequent absence of trampling and foraging promote natural plant regeneration from the seed bank and seed rain. Similar responses have been observed in other studies that observed significant recovery of riparian plant communities after fencing the riparian zone (Kauffman et al., 1997; McIver & Starr, 2001; Sarr, 2002; Hansen & Budy, 2011; Forget et al., 2013; O'Donnell et al., 2014). The larger area of bare soil in the reference stream than the restored stream can be a long-term consequence of excluding livestock. Trees in La Roche are older than those in VAB (Forget et al., 2013; Sawtschuk et al., 2014), creating more shade and thus reducing viability of groundcover vegetation. This argument is supported by previous work that highlights the loss of dense grass groundcover due to increased shading (Quinn et al., 2009; Hughes & Quinn, 2014). The convergence towards ruderal species after 8 years of PER can be explained by the recovery of filtration functions in the riparian zone, which helps to retain nutrients. Indeed, the riparian zone, by retaining more sediment and nutrients, seems to favor nitrophilous species.

However, water quality measurements in the restored stream did not have the expected results; thus, the attempt to recover the filtration function by restoring riparian habitat was not successful. No significant difference in water quality variables was observed between pre- and post-PER. Width of the riparian zone can affect filtration function (Osborne & Kovacic, 1993), and in our PER, fences may have been implemented too close to the stream bed to ensure filtration. Effects of excluding livestock on water quality in the literature were equivocal and did not reach a consensus. Some authors observed a decrease in stream nutrient concentrations after PER (Van Velson, 1979; Li et al., 1994; Sweeney et al., 2004; Hansen & Budy, 2011). Other studies observed no differences in stream nutrient concentrations after PER (McKergow et al., 2003; Hughes & Quinn, 2014). The variability in responses illustrates the complex nature of the water quality, which is not a result of local process but a watershed-scale process dynamics, explaining why in some situations and not in others, nutrient concentration changes in water following PER. Moreover, water quality measurement in the La Roche reference stream remained the same as those in the restored VAB stream, revealing a slightly alkaline pH and poorly mineralized water. La Roche and VAB have high nutrient concentrations, especially nitrates near the upper threshold (low water quality) according to regulations of the Water Authorities and Ministry of Ecology. This water quality is representative of most of the human-impacted and rural streams in Normandy and Brittany (Baglinière et al., 2005; Sarriquet et al., 2007) as a result of intensive agriculture (crops and livestock) in the watershed. In our experiment, PER undertaken to improve water quality was not effective on an intensively cultivated watershed, confirming that water quality is a result of watershed-scale process (Mitsch & Gosselink, 2000; Shields et al., 2010).

In contrast, in a healthy watershed with a relatively short agricultural history and low-intensity land use, Hughes & Quinn (2014) noticed a positive and rapid effect on stream water clarity after excluding livestock from the banks. Water quality and external nutrient inputs to streams are strongly influenced by watershed disturbance (Mitsch & Gosselink, 2000; Shields et al., 2010); therefore, an intensive watershed strongly decreases the ability of riparian filtration to reduce nutrient inputs.

Previous studies showed no effect after stream restoration measures (Roni et al., 2002, 2008; Jähnig et al., 2010; Kail et al., 2012; Stoll et al., 2013; Nilsson et al., 2014; Palmer et al., 2014), indicating that local restoration actions cannot compensate for watershed-scale degradation (Sudduth & Meyer, 2006). Before focusing on local restoration actions, the watershed scale should receive more attention to reduce external nutrient inputs and improve water quality by improving agricultural practices. Previous studies emphasize the need to consider the watershed-scale restoration approach, potentially constraining the effectiveness of local restoration measures (Roni et al., 2002; Jähnig et al., 2010; Ouyang et al., 2011; Kail et al., 2012). This new approach challenges implementation of the European Water Framework Directive (European Commission, 2000), which aims to restore a "good ecological status" of water bodies in all Member States, which is often realized through local restoration measures. The European Water Framework Directive requires appropriate prioritization of stream restoration planning, and practitioners should consider the watershed scale more often. Even if watershed scale was not integrated for restoration projects, the filtering function of the riparian zone could however be tested by sampling nutrient concentrations across the riparian zone, before and after PER, to assess the evolution of nutrient input.

Our study is consistent with many other studies on stream restoration measures, and suggested that even if plant communities change, there is no guarantee that functions will be restored (Jähnig et al., 2010; Kail et al., 2012; Nilsson et al., 2014; Palmer et al., 2014). Our findings highlighted the importance of using multiple indicators to assess restoration success and the caution about the interpreting of only one indicator, not reflecting ecosystem functionality. This also highlights the importance of having a functional perspective in restoration to regain the full suite of biogeochemical, ecological, and hydrogeomorphic processes (Palmer et al., 2014). As Wohl et al. (2005) state, "because natural variability is an inherent feature of all river systems, restoration of process is more likely to succeed than restoration aimed at a fixed end point."

This does not necessarily mean that PER in an intensive watershed will fail, even if improvement in water quality is not expected and the watershed scale is not addressed. Riparian zones are not only a tool to reduce nutrient uptake, but can also offer other services. By restoring the riparian habitat, PER may potentially contribute to a more diverse and functional stream ecosystem. Riparian habitat restoration can improve stream bank stability and reduce erosion. This bank stability function could be tested by measuring water turbidity. Riparian habitat heterogeneity is usually associated with high habitat diversity, which affects in-stream structure (Kail & Hering, 2009; Lorenz et al., 2013) and may lead to a more heterogeneous in-stream habitat. Tree recruitment and the presence of roots in the stream can create meanders, increase sinuosity, and create refugia and diversify hydromorphology and flow facies. Riparian habitat may also provide an ecological corridor for aquatic and terrestrial wildlife. The concept of amenities, which evokes pleasant aspects of the environment, should also be a positive effect of PER. Research should be performed on the bundle services associated with restoration.

These bundle services may also help implement restoration, especially if functions are beneficial to farmers. Livestock exclosure needs to be sociologically and economically acceptable to farmers and can only be successful with their involvement. This PER plan is an initial step in the dialog between farmers and river managers, and since the farmers own the small stream banks, acceptance of a fence and a natural edge along the stream is a positive result. Evidence from the literature suggests that excluding livestock from streams can be economically beneficial for farmers, leading to increases in cattle weight gain (Willms et al., 2002; Zeckoski et al., 2007) or milk production (Landefeld & Bettinger, 2002; Zeckoski et al., 2007). Additionally, studies have shown a decrease in cattle disease (Pfost et al., 2007). Despite these benefits, Zeckoski et al. (2007) noticed that many farmers in Virginia, USA, were slow to adopt stream exclusion systems. A similar study to investigate farmers' benefits and constraints in the intensive agricultural watershed of the Oir River could improve restoration actions and consider farmers as a key to restoration success.

Conclusion

This study evaluates effectiveness of PER on a riparian habitat and on water quality. Livestock exclusion can successfully restore riparian habitat, but in this study, it did not decrease nutrient inputs to the stream because they are directly related to the level of disturbance in the watershed. Although the watershed scale should be considered and agricultural practices need to change, PER may potentially provide bundles of ecosystem services by restoring riparian habitat, and this should be tested and validated in further studies. Since farmers are essential for implementing PER, evidence of its ecological and economic benefits is needed.

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References

Abernethy, B. & I. D. Rutherfurd, 1998. Where along a river's length will vegetation most effectively stabilise stream banks? Geomorphology 23: 55–75.

Anderson, M. J., 2001. A new method for non-parametric multivariate analysis of variance. Austral Ecology 26: 32–46.

Anderson, M. J., T. O. Crist, J. M. Chase, M. Vellend, B. D. Inouye, A. L. Freestone, N. J. Sanders, H. V. Cornell, L. S. Comita, K. F. Davies, et al., 2011. Navigating the multiple meanings of β diversity: a roadmap for the practicing ecologist. Ecology Letters 14: 19–28.

Armour, C. L., D. A. Duff & W. Elmore, 1991. The effects of livestock grazing on riparian and stream ecosystems. Fisheries 16: 7–11.

Baglinière, J.-L., F. Marchand & V. Vauclin, 2005. Interannual changes in recruitment of the Atlantic salmon (*Salmo salar*) population in the River Oir (Lower Normandy, France): relationships with spawners and in-stream habitat. ICES Journal of Marine Science: Journal du Conseil 62: 695–707.

Bal, G., E. Rivot, E. Prévost, C. Piou & J. L. Baglinière, 2011. Effect of water temperature and density of juvenile salmonids on growth of young-of-the-year Atlantic salmon *Salmo salar*. Journal of Fish Biology 78: 1002–1022.

Baldigo, B. P. & D. R. Warren, 2008. Detecting the response of fish assemblages to stream restoration: effects of different sampling designs. North American Journal of Fisheries Management 28: 919–934.

Bash, J. S. & C. M. Ryan, 2002. Stream restoration and enhancement projects: is anyone monitoring? Environmental management 29: 877–885.

Belsky, A. J., A. Matzke & S. Uselman, 1999. Survey of livestock influences on stream and riparian ecosystems in the

western United States. Journal of Soil and Water Conservation 54: 419–431.
Camporeale, C., E. Perucca, L. Ridolfi & A. M. Gurnell, 2013. Modeling the interactions between river morphodynamics and riparian vegetation. Reviews of Geophysics 51: 379–414.
Clewell, A. F. & J. Aronson, 2006. Motivations for the restoration of ecosystems. Conservation Biology 20: 420–428.
Del Rosario, R. B., E. A. Betts & V. H. Resh, 2002. Cow manure in headwater streams: tracing aquatic insect responses to organic enrichment. Journal of the North American Benthological Society 21: 278–289.
Dembélé, F., N. Picard, M. Karembé & P. Birnbaum, 2006. Tree vegetation patterns along a gradient of human disturbance in the Sahelian area of Mali. Journal of Arid Environments 64: 284–297.
Dray, S., A. B. Dufour & D. Chessel, 2007. The ade4 package II: two-table and K-table methods. R News 7: 47–52.
Duhaime, L. & B. Pinel-Alloul, 2005. Méthode de sélection de lacs de référence dans le cadre d'une étude Before-After Control-Impact (BACI) évaluant les effets des coupes forestières sur le zooplancton des lacs de la forêt boréale. Revue des sciences de l'eau/Journal of Water Science 18: 199–220.
European Commission, 2000. Council Directive 2000/60/EC of 23 October 2000 on framework for Community action in the field of water policy. Official Journal of the European Communities 327: 72.
Finn, D. S., N. Bonada, C. Múrria & J. M. Hughes, 2011. Small but mighty: headwaters are vital to stream network biodiversity at two levels of organization. Small 30: 963–980.
Forget, G., C. Carreau, D. Le Coeur & I. Bernez, 2013. Ecological Restoration of headwaters in a rural landscape (Normandy, France): a passive approach taking hedge networks into account for riparian tree recruitment. Restoration Ecology 21: 96–104.
Green, R. H., 1979. Sampling Design and Statistical Methods for Environmental Biologists. Wiley, New York.
Hansen, E. S. & P. Budy, 2011. The potential of passive stream restoration to improve stream habitat and minimize the impact of fish disease: a short-term assessment. Journal of the North American Benthological Society 30: 573–588.
Harding, J. S., E. F. Benfield, P. V. Bolstad, G. S. Helfman & E. B. D. Jones, 1998. Stream biodiversity: the ghost of land use past. Proceedings of the National Academy of Sciences 95: 14843–14847.
Helsel, D. R. & R. M. Hirsch, 2002. Statistical Methods in Water Resources: Techniques of Water Resources Investigations. US Geological Survey Report, Reston.
Holmes, P. M., D. M. Richardson, K. J. Esler, E. T. F. Witkowski & S. Fourie, 2005. A decision-making framework for restoring riparian zones degraded by invasive alien plants in South Africa: review article. South African Journal of Science 101: 553–564.
Hughes, A. O. & J. M. Quinn, 2014. Before and After Integrated Catchment Management in a Headwater Catchment: Changes in Water Quality. Environmental Management 54: 1288–1305.
Jähnig, S. C., K. Brabec, A. Buffagni, S. Erba, A. W. Lorenz, T. Ofenböck, P. F. Verdonschot & D. Hering, 2010. A comparative analysis of restoration measures and their effects on hydromorphology and benthic invertebrates in 26 central and southern European rivers. Journal of Applied Ecology 47: 671–680.
Kail, J. & D. Hering, 2009. The influence of adjacent stream reaches on the local ecological status of Central European mountain streams. River Research and Applications 25: 537–550.
Kail, J., J. Arle & S. C. Jähnig, 2012. Limiting factors and thresholds for macroinvertebrate assemblages in European rivers: empirical evidence from three datasets on water quality, catchment urbanization, and river restoration. Ecological Indicators 18: 63–72.
Kauffman, J. B., R. L. Beschta, N. Otting & D. Lytjen, 1997. An ecological perspective of riparian and stream restoration in the western United States. Fisheries 22: 12–24.
Knapp, R. A. & K. R. Matthews, 1996. Livestock grazing, golden trout, and streams in the Golden Trout Wilderness, California: Impacts and management implications. North American Journal of Fisheries Management 16: 805–820.
Landefeld, M. & J. Bettinger, 2002. Water Effects on Livestock Performance. Ohio State University Agriculture and Natural Resources, Columbus.
Le Pichon, C., 2006. Une approche"paysage aquatique" pour une meilleure connaissance du fonctionnement des écosystèmes fluviaux et l'amélioration de la conservation des peuplements de poissons. Université Paris VI-Diversité du Vivant-Ecologie, Paris.
Li, H. W., G. A. Lamberti, T. N. Pearsons, C. K. Tait, J. L. Li & J. C. Buckhouse, 1994. Cumulative effects of riparian disturbances along high desert trout streams of the John Day Basin, Oregon. Transactions of the American Fisheries Society 123: 627–640.
Lorenz, A. W., S. Stoll, A. Sundermann & P. Haase, 2013. Do adult and YOY fish benefit from river restoration measures? Ecological Engineering 61: 174–181.
Lowe, W. H. & G. E. Likens, 2005. Moving headwater streams to the head of the class. BioScience 55: 196–197.
Lowrance, R. R., R. L. Todd & L. E. Asmussen, 1984. Nutrient cycling in an agricultural watershed: I. phreatic movement. Journal of Environmental Quality 13: 22–27.
McIver, J. & L. Starr, 2001. Restoration of degraded lands in the interior Columbia River basin: passive vs. active approaches. Forest Ecology and Management 153: 15–28.
McKergow, L. A., D. M. Weaver, I. P. Prosser, R. B. Grayson & A. E. Reed, 2003. Before and after riparian management: sediment and nutrient exports from a small agricultural catchment, Western Australia. Journal of Hydrology 270: 253–272.
Millennium Ecosystem Assessment, 2005. Ecosystems and Human Well-Being. Island Press, Washington.
Mitsch, W. J. & J. G. Gosselink, 2000. The value of wetlands: importance of scale and landscape setting. Ecological Economics 35: 25–33.
Moerke, A. H. & G. A. Lamberti, 2003. Responses in fish community structure to restoration of two Indiana streams. North American Journal of Fisheries Management 23: 748–759.
Moerke, A. H., K. J. Gerard, J. A. Latimore, R. A. Hellenthal & G. A. Lamberti, 2004. Restoration of an Indiana, USA, stream: bridging the gap between basic and applied lotic

ecology. Journal of the North American Benthological Society 23: 647–660.
Muotka, T., R. Paavola, A. Haapala, M. Novikmec & P. Laasonen, 2002. Long-term recovery of stream habitat structure and benthic invertebrate communities from in-stream restoration. Biological Conservation 105: 243–253.
Naiman, R. J., H. Decamps & M. Pollock, 1993. The role of riparian corridors in maintaining regional biodiversity. Ecological Applications 3: 209–212.
Naiman, R. J., H. Decamps & M. E. McClain, 2010. Riparia: Ecology, Conservation, and Management of Streamside Communities. Elsevier Academic Press, London.
Nilsson, C., L. E. Polvi, J. Gardeström, E. M. Hasselquist, L. Lind & J. M. Sarneel, 2014. Riparian and in-stream restoration of boreal streams and rivers: success or failure? Ecohydrology. doi:10.1002/eco.1480.
O'Donnell, J., K. Fryirs & M. R. Leishman, 2014. Can the regeneration of vegetation from riparian seed banks support biogeomorphic succession and the geomorphic recovery of degraded river channels? River Research and Applications. doi:10.1002/rra.2778.
Oksanen, J., R. Kindt, P. Legendre, B. O'Hara, G. L. Simpson, P. Solymos, M. H. H. Stevens, & H. Wagner, 2008. Vegan: Community Ecology Package. R Package Version 1. 13-1
Osborne, L. L. & D. A. Kovacic, 1993. Riparian vegetated buffer strips in water-quality restoration and stream management. Freshwater Biology 29: 243–258.
Ouyang, N. L., S. L. Lu, B. F. Wu, J. J. Zhu & H. Wang, 2011. Wetland restoration suitability evaluation at the watershed scale: a case study in upstream of the Yongdinghe River. Procedia Environmental Sciences 10: 1926–1932.
Paller, M. H., M. J. M. Reichert, J. M. Dean & J. C. Seigle, 2000. Use of fish community data to evaluate restoration success of a riparian stream. Ecological Engineering 15: S171–S187.
Palmer, M. A., H. L. Menninger & E. Bernhardt, 2010. River restoration, habitat heterogeneity and biodiversity: a failure of theory or practice? Freshwater biology 55: 205–222.
Palmer, M. A., K. L. Hondula & B. J. Koch, 2014. Ecological restoration of streams and rivers: shifting strategies and shifting goals. Annual Review of Ecology, Evolution, and Systematics 45: 247–269.
Pander, J. & J. Geist, 2013. Ecological indicators for stream restoration success. Ecological Indicators 30: 106–118.
Pander, J., M. Mueller & J. Geist, 2014. A comparison of four stream substratum restoration techniques concerning interstitial conditions and downstream effects. River Research and Applications 31: 239–255.
Pfost, D. L., J. R. Gerrish, M. Davis & M. Kennedy, 2007. Pumps and Watering Systems for Managed Beef Grazing. University of Missouri Office of Extension, Missouri.
Pinay, G. & H. Decamps, 1988. The role of riparian woods in regulating nitrogen fluxes between the alluvial aquifer and surface water: a conceptual model. Regulated Rivers: Research & Management 2: 507–516.
Prach, K. & R. J. Hobbs, 2008. Spontaneous succession versus technical reclamation in the restoration of disturbed sites. Restoration Ecology 16: 363–366.
Quinn, J. M., G. F. Croker, B. J. Smith & M. A. Bellingham, 2009. Integrated catchment management effects on flow, habitat, instream vegetation and macroinvertebrates in Waikato, New Zealand, hill-country streams. New Zealand Journal of Marine and Freshwater Research 43: 775–802.
R Development Core Team., 2010. R: A Language and Environment for Statistical Computing. R Development Core Team, Vienna.
Raposa, K., 2002. Early responses of fishes and crustaceans to restoration of a tidally restricted New England salt marsh. Restoration Ecology 10: 665–676.
Raup, D. M. & R. E. Crick, 1979. Measurement of faunal similarity in paleontology. Journal of Paleontology 53: 1213–1227.
Richardson, C. J., N. E. Flanagan, M. Ho & J. W. Pahl, 2011. Integrated stream and wetland restoration: a watershed approach to improved water quality on the landscape. Ecological Engineering 37: 25–39.
Richardson, J. S. & R. J. Danehy, 2007. A synthesis of the ecology of headwater streams and their riparian zones in temperate forests. Forest Science 53: 131–147.
Roni, P., T. J. Beechie, R. E. Bilby, F. E. Leonetti, M. M. Pollock & G. R. Pess, 2002. A review of stream restoration techniques and a hierarchical strategy for prioritizing restoration in Pacific Northwest watersheds. North American Journal of Fisheries Management 22: 1–20.
Roni, P., K. Hanson & T. Beechie, 2008. Global review of the physical and biological effectiveness of stream habitat rehabilitation techniques. North American Journal of Fisheries Management 28: 856–890.
Rood, S. B., S. G. Bigelow, M. L. Polzin, K. M. Gill & C. A. Coburn, 2014. Biological bank protection: trees are more effective than grasses at resisting erosion from major river floods. Ecohydrology. doi:10.1002/eco.1544.
Sarr, D. A., 2002. Riparian livestock exclosure research in the western United States: a critique and some recommendations. Environmental Management 30: 516–526.
Sarriquet, P.-E., P. Bordenave & P. Marmonier, 2007. Effects of bottom sediment restoration on interstitial habitat characteristics and benthic macroinvertebrate assemblages in a headwater stream. River Research and Applications 23: 815–828.
Sawtschuk, J., M. Delisle, X. Mesmin & I. Bernez, 2014. How past riparian management practices can affect composition and structure of vegetation for headwater ecological restoration projects. Acta Botanica Gallica 161: 309–320.
Schilling, K. E. & P. Jacobson, 2014. Effectiveness of natural riparian buffers to reduce subsurface nutrient losses to incised streams. Catena 114: 140–148.
Shields Jr, F. D., S. S. Knight, N. Morin & J. Blank, 2003. Response of fishes and aquatic habitats to sand-bed stream restoration using large woody debris. Hydrobiologia 494: 251–257.
Shields Jr, F. D., R. E. Lizotte Jr, S. S. Knight, C. M. Cooper & D. Wilcox, 2010. The stream channel incision syndrome and water quality. Ecological Engineering 36: 78–90.
Skłodowski, M., E. Kiedrzyńska, M. Kiedrzyński, M. Urbaniak, K. M. Zielińska, J. K. Kurowski & M. Zalewski, 2014. The role of riparian willows in phosphorus accumulation and PCB control for lotic water quality improvement. Ecological Engineering 70: 1–10.
Society for Ecological Restoration, 2004. The SER International primer on ecological restoration. Society for Ecological Restoration International.

Stoll, S., A. Sundermann, A. W. Lorenz, J. Kail & P. Haase, 2013. Small and impoverished regional species pools constrain colonisation of restored river reaches by fishes. Freshwater Biology 58: 664–674.

Stoll, S., J. Kail, A. W. Lorenz, A. Sundermann & P. Haase, 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.

Sudduth, E. B. & J. L. Meyer, 2006. Effects of bioengineered streambank stabilization on bank habitat and macroinvertebrates in urban streams. Environmental Management 38: 218–226.

Summers, R., D. Weaver, N. Keipert, & J. Steele, 2014. Does riparian filtration reduce nutrient movement in sandy agricultural catchments? Environment and Natural Resources Research 4: 155 pp.

Suurkuukka, H., R. Virtanen, V. Suorsa, J. Soininen, L. Paasivirta & T. Muotka, 2014. Woodland key habitats and stream biodiversity: does small-scale terrestrial conservation enhance the protection of stream biota? Biological Conservation 170: 10–19.

Sweeney, B. W., T. L. Bott, J. K. Jackson, L. A. Kaplan, J. D. Newbold, L. J. Standley, W. C. Hession & R. J. Horwitz, 2004. Riparian deforestation, stream narrowing, and loss of stream ecosystem services. Proceedings of the National Academy of Sciences of the United States of America 101: 14132–14137.

Tunstall, S. M., E. C. Penning-Rowsell, S. M. Tapsell & S. E. Eden, 2000. River restoration: public attitudes and expectations. Water and Environment Journal 14: 363–370.

Van Velson, R., 1979. Effects of livestock grazing upon rainbow trout in Otter Creek. Forum–grazing and riparian/stream ecosystems. Edition Trout Unlimited, Denver.

Whiteway, S. L., P. M. Biron, A. Zimmermann, O. Venter & J. W. Grant, 2010. Do in-stream restoration structures enhance salmonid abundance? A meta-analysis. Canadian Journal of Fisheries and Aquatic Sciences 67: 831–841.

Willms, W. D., O. R. Kenzie, T. A. McAllister, D. Colwell, D. Veira, J. F. Wilmshurst, T. Entz & M. E. Olson, 2002. Effects of water quality on cattle performance. Journal of Range Management 55: 452–460.

Wohl, E., P. L. Angermeier, B. Bledsoe, G. M. Kondolf, L. MacDonnell, D. M. Merritt, M. A. Palmer, N. L. Poff & D. Tarboton, 2005. River restoration. Water Resources Research 41: W10301.

Zeckoski, R. W., B. L. Benham & C. Lunsford, 2007. Water Quality and Economic Benefits of Livestock Exclusion from Streams: Experiences from Virginia. Watershed Management to Meet Water Quality Standards, San Antonio.