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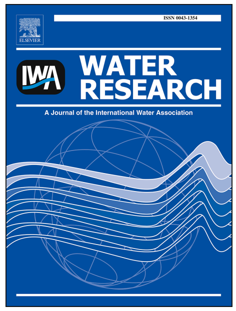

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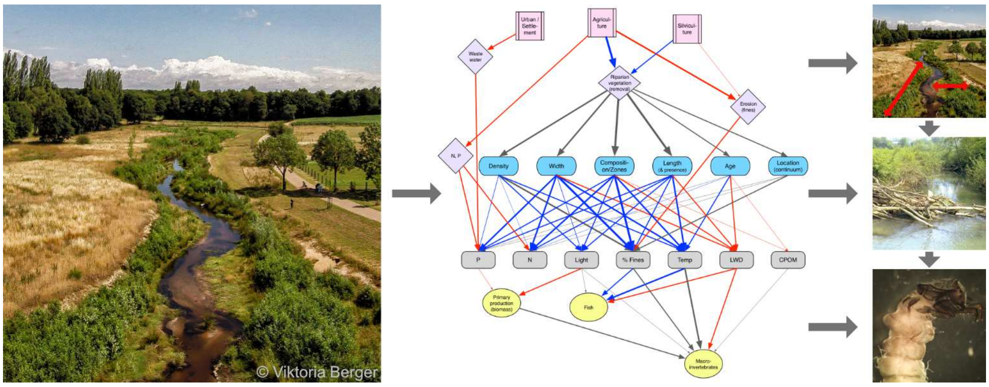
© Viktoria Berger
Urban / Settle- ment
Agricul- ture
Silvicul- ture
Waste water
Riparian vegetation (removal)
Erosion (fines)
N, P
Density
Width
Compositi- on/Zones
Length (& presence)
Age
Location (continuum)
P
N
Light
% Fines
Temp
LWD
CPOM
Primary production (biomass)
Fish
Macro- invertebrates

Evaluating riparian solutions to multiple stressor problems in river ecosystems — a conceptual study

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Abstract

Rivers are among the most sensitive of all ecosystems to the effects of global change, but options to prevent, mitigate or restore ecosystem damage are still inadequately understood. Riparian buffers are widely advocated as a cost-effective option to manage impacts, but empirical evidence is yet to identify ideal riparian features (e.g. width, length and density) which enhance ecological integrity and protect ecosystem services in the face of catchment-scale stressors. Here, we use an extensive literature review to synthesise evidence on riparian buffer and catchment management effects on instream environmental conditions (e.g. nutrients, fine sediments, organic matter), river organisms and ecosystem functions. We offer a conceptual model of the mechanisms through which catchment or riparian management might impact streams either positively or negatively. The model distinguishes scale-independent benefits (shade, thermal damping, organic matter and large wood inputs) that arise from riparian buffer management at any scale from scale-dependent benefits (nutrient or fine sediment retention) that reflect stressor conditions at broader (sub-catchment to catchment) scales. The latter require concerted management efforts over equally large domains of scale (e.g. riparian buffers combined with nutrient restrictions). The evidence of the relationships between riparian configuration (width, length, zonation, density) and scale-independent benefits is consistent, suggesting a high certainty of the effects. In contrast, scale-dependent effects as well as the biological responses to riparian management are more uncertain, suggesting that ongoing diffuse pollution (nutrients, sediments), but also sources of variability (e.g. hydrology, climate) at broader scales may interfere with the effects of local riparian management. Without concerted management across relevant scales, full biological recovery of damaged lotic ecosystems is unlikely. There is, nevertheless, sufficient evidence that the benefits of riparian buffers outweigh potential adverse effects, in particular if located in the upstream part of the stream network. This supports the use of riparian restoration as a no-regrets management option to improve and sustain lotic ecosystem functioning and biodiversity.

Keywords

Agriculture, Aquatic biota, Fine sediments, Nutrients, Riparian buffer, River management

1. Introduction

Growing evidence suggests that rivers are among the most sensitive of all ecosystems to the effects of global change. As the major terrestrial expression of the global water cycle, they are at risk from major anthropogenic modifications to the atmospheric, catchment and riparian environments from which they receive drainage (Durance & Ormerod, 2007; Palmer et al., 2008; Woodward et al., 2012; Beketov et al., 2013; Bussi et al., 2016). Already well over half of the World's river discharge is appropriated for human use, while pollution, climate change and habitat modification interact among a suite of multiple stressors on river ecosystems that now incur some of the most rapid biodiversity losses on Earth (Matthaei et al., 2010; Gutiérrez-Cánovas et al., 2013). These effects are not only of intrinsic ecological significance, but also pose major risk to rivers as some of the World's most valuable natural capital assets and as the sources of ecosystem services of vital importance to human survival (Vörösmarty *et al.*, 2010; Maltby & Ormerod, 2011). The degradation of river environments is now a pressing policy priority, and in Europe the Water Framework Directive (2000/60/EC) aims to return almost 60% of Europe's rivers to 'good ecological status' by 2027 (EEA, 2012).

Among the multiple stressors affecting European freshwaters, agricultural intensification, hydromorphological alteration and climate change are among the main causes of river deterioration, increasing nutrients and sediment in waters, reducing habitat quality and modifying thermal and hydrological regimes (EEA, 2012; Hering et al., 2015). However, protecting rivers, arresting degradation and restoring ecological damage in the face of global change is a challenging task, and requires some combination of i) cessation or prevention of damaging activities (e.g. Wilcock et al., 2009; Vaughan & Ormerod, 2014); ii) mitigation of ecological effects of stressors (e.g. Bednarek & Hart, 2005); iii) enhanced resilience by adapting river ecosystems to further change (e.g. Thomas et al., 2016) and iv) restoration to accelerate ecological recovery (e.g. Hickford et al., 2014; Hering et al., 2015). So far, there is only limited information to underpin the implementation of the most effective and practicable combination of these strategies at relevant scales and at low cost. While case studies exist, there is an urgent need to synthesise the extant evidence, which is often local, fragmentary or arises from studies with limitations in study sample size and design.

Among the restorative and management strategies to improve ecological status, the establishment of riparian buffers has been most frequently utilised to mitigate diffuse pollution by agriculture (Feld et al., 2011; Collins et al., 2012) and thermal deterioration induced by climate change (Palmer et al., 2009). However, empirical evidence from studies assessing the effects of planting or restoring riparian buffers is unclear because of the many features that characterise riparian buffers and ultimately determine their ecological effects, for example buffer length, width, density, or the planted species and its zonation (i.e. single *vs.* multi-zone buffers) (Dosskey, 2001). Practitioners therefore face a lack of clear guidance about the dimensions and composition required for riparian buffers to be effective. Additionally, individual local river characteristics and upstream catchment can all mediate the ecological effects of riparian buffers (Feld et al., 2011). For example, thermal effects of riparian shade are limited at wide and deep river sections (i.e. by tree height and water body volume), while reach-scale water quality effects can be constrained by the degree of land use further upstream in the catchment. Therefore, knowledge of the interplay of riparian buffer effects and related catchment features is critical to render river restoration ecologically successful in the long term.

Here, we present a synthesis of studies performing river restoration and management actions for mitigating the impacts of agricultural intensification, hydrological alteration and climate change across a range of regions, climates and management features. We introduce a conceptual model to visualise the effects of agriculture, urbanisation and silviculture on riparian degradation, instream nutrient and fine sediment concentrations, and eventually on aquatic biodiversity. We hypothesise that some riparian buffer restoration effects will be consistent across a wide range of spatial scales, i.e. they are ‘scale-independent’; in contrast, other restoration benefits are ‘scale-dependent’ as they can only be gained by simultaneous actions across scales such that the effects are large enough to offset or mitigate the impact of stressors at the catchment-scale (e.g. tile-drainage, extensive agriculture). Second, we hypothesise that riparian buffer restoration effects are negatively related to catchment size and thus conditional on the longitudinal position along the river continuum. Riparian buffers at headwater sections thus would be more likely to give rise to positive outcomes as compared to buffer restoration in the middle and lower parts of the river network.

2. Material and methods

2.1 Literature review

We focused our synthesis on evidence about real outcomes from management intervention and related recovery trajectories, because biological responses to restoration are not necessarily the reverse of responses to degradation (Feld et al., 2011). For example, hysteresis effects or alternative endpoints may prevent ecosystems to recover its pre-disturbance properties after a restoration action (Verdonschot et al., 2013). We searched the peer-reviewed literature using the Web of Science and Scopus using the following combinations of search terms ('*' truncation to include similar versions of the same word such as singular/plural):

catchment* OR watershed* OR land use* OR riparian OR riparia* vegetation OR buffer AND manage* OR enhance AND rive* OR strea*

riparian* AND catchment* AND manage* AND rive* OR strea*

riparia* AND land us* AND catchmen* AND manage* AND rive* OR strea*

rive* OR strea* AND land us* AND catchmen* AND restor* AND manage*

rive* OR strea* AND land us* AND manage* AND spatial scal*

riparia* AND catchmen* AND stress* AND rive* OR riparia* AND catchmen* AND stress* AND strea*)

riparia* AND basin* AND stress* AND rive* OR riparia* AND basin* AND stress* AND strea*

The terms resulted initially in 219–998 hits for each search that were scanned (title, keywords and abstracts) to exclude irrelevant references, which led to 711 candidate studies. The candidates were then grouped into i) studies addressing riparian *and* catchment-scale management simultaneously; ii) studies solely addressing management at riparian *or* catchment scale; iii) studies addressing mechanistic modelling or literature reviews of management effects at either scale. Studies that did not fit into any of the groups were omitted, which eventually resulted in 138 references to enter a review database.

2.2 Review database

To allow for a structured review including some qualitative meta-analysis of the reviewed body of literature, we defined several criteria to extract information from the reviewed papers, which was compiled into a database (Table 1). These were: i) general study characteristics (e.g. study origin, spatial scale and year), ii) information on the main drivers and related catchment-scale pressures impacting the study area (e.g. agricultural land use, eutrophication), iii) riparian management characteristics (e.g. type and spatial extent of a restoration), iv) catchment management characteristics (e.g. type and spatial extent of a modelled or actually implemented management option) and v) the instream abiotic and biological effects of management (e.g. changes in nutrient concentrations or biological indices). The database assisted the conceptualisation and synthesis of the evidence of cause-effect relationships (i.e. management-recovery effects), which resulted in a conceptual model.

Table 1

2.3 Conceptual model of riparian and catchment-scale management effects

Our conceptual model represents the multi-layer relationship between riparian-scale and catchment-scale management effects on the instream environmental and biological conditions (Fig. 1). The model follows the Driver-Pressure-State terminology, as part of the DPSIR scheme (EEA, 1999). In this context, we use the term 'stressor' to refer to either a pressure (e.g. diffuse pollution) or an environmental state (e.g. nitrogen concentration) that adversely affects biodiversity or ecosystem functioning (sensu Townsend et al., 2008).

Figure 1

First, we considered all potentially relevant cause-effect links for our study and distinguished positive, negative and indifferent (i.e. no clear sign definable) potential relationships. Second, to provide a qualitative measure of the support for each link, we counted the number of papers showing significant and consistent effects for each relationship and whether the relationship was positive or negative. The sign and strength of effects were derived from a study's model coefficients or ANOVA results. Third, we assigned arrow colours (sign) and thickness (strength) to visualise the sign and strength of the evidence of model linkages. Red and blue arrows in the model mark linkages that were consistently reported as positive or negative in the literature; indifferent linkages are marked grey. Arrow thickness is linearly related the number of evidence items in the literature that support that link.

Unfortunately, a quantitative meta-analysis was impracticable, because we addressed numerous and often multi-layered links, for which in several cases only qualitative information was available. Further, the many effect-response variables addressed in the studies were of very different nature, including various kinds of abiotic and biological indicators.

3. Results

3.1 Reviewed literature

Of the 138 studies reviewed in detail, only 55 provided evidence of statistically significant management and restoration effects on the instream abiotic and biological states addressed. These 55 references constituted the core evidence, either based on monitoring surveys after the implementation of management or restoration options, through experiments or through (sub-) catchment-scale mechanistic modelling. The remaining references encompassed review papers and empirical studies, the latter of which usually addressed statistical relationships among stressors and biological responses to progressively degraded riparian environments.

The 55 core studies were published between 1990 and 2017 and originated mainly from the USA (36%), Europe (32%), New Zealand (24%) and Canada (7%). Experimental studies (52%) dominated over modelling studies (26%), statistical analysis of environmental gradients (17%) and reviews (17%) (NB: percent values do not necessarily sum up to 100% as some studies addressed several criteria simultaneously, for example, if data originated from several countries).

Only about 15% of the studies addressed *in situ* monitoring following intervention, highlighting a potentially important shortcoming in evaluating river restoration and management. This reveals another shortcoming in that poor experimental design often limits the quantification of net buffer effects. To calculate net effects, ideally the conditions before and after buffer management would be compared against control or reference locations, to isolate the effects of management action from natural variation. This design is referred to as the "BACI design", i.e. the before-after-control-impact comparison that allows the estimation of type II errors in the statistical analysis (Conner et al., 2016). In our sample, the gold standard approach involving the BACI design had been applied in only six studies (11%).

Most studies focussed on small streams (66%) and addressed headwater and upstream sections (66%), while the middle (32%) and downstream sections (10%) were less frequently addressed. Fewer than 2% of the studies addressed catchment areas >1,000 km^2. Regarding elevation, 56% of the studies were conducted in lowland streams (<200 m a.s.l.), 41% in piedmont streams (200–500 m a.s.l.), 7% in mountainous streams (500–800 m a.s.l.) and only 3% in alpine streams (>800 m a.s.l.). This suggests that riparian management, but presumably also riparian degradation, is fairly limited to riverscapes at altitudes below 500 m.

3.2 Riparian management studies

Riparian management studies most often addressed the reach (61%) and segment scales (42%), as compared to sub-catchment (16%) and site scales (7%). More specifically, the length of the management section was generally less than 1 km (31% of the studies) or 2–10 km long (27%), while studies addressing longer segments (>10 km) were very rare (7%) (Fig. 2a). We should note, however that this information was absent from roughly a third of the studies. Most riparian buffer widths were <10 m (34%), followed by buffer widths of 10–20 m (22%) and >20 m (20%), respectively (Fig. 2b). Buffer height varied, but again two thirds of the studies provided no usable information on this feature. Buffer vegetation age was usually <5 years (49%), although long-term management effects were also represented (5–10 a: 18%, 10–20 a: 12%, >20 a: 18%). The type of vegetation managed in the studies were mainly trees (74%), followed by grass/forbs (57%) and shrubs (34%). The plant combinations used in the buffers were mostly single trees (27%) or multi-zone configurations (25%), while trees and grass (9%), single grass (10%), shrubs and grass (6%) and trees and shrubs (4%) were less common combinations.

3.3 Common abiotic and biological management effects

Studies almost equally addressed pollution by nitrogen (total N, soluble inorganic N, nitrate-N, nitrate; 41%), diffuse sediments (41%), phosphorous (37%) and thermal effects (31%). Shade (18%) and the provision of large woody debris (LWD; 8%) were less frequently addressed (Fig. 3).

Only about half of the studies (55%) addressed management effects on instream and/or floodplain biota. Of these, macroinvertebrates (26%) and fish (25%) were most commonly addressed, followed by instream primary producers (8%) and riparian vegetation (8%) (Fig. 4a). Most often, community diversity was used to quantify biological effects (23%), followed by various biotic indices (e.g. national water quality status, multi-metric assessment indices; 19%), trait-based community metrics (e.g. feeding types, substrate preferences; 17%), measures of abundance (15%) and community composition (e.g. the number of Ephemeroptera, Plecopera and Trichoptera taxa; 9%) (Fig 4b).

Figure 2–4

3.4 Conceptual model of riparian and catchment-scale management effects

We found evidence for altogether 58 links (arrows) of our conceptual model in the reviewed literature (Fig. 5). Most of this evidence was consistent with regard to the sign of the relationship: 25 negative, 16 positive and 17 indifferent links. Notably, the evidence of the effects of riparian configuration (density, width, zonation, length, age, but not location, see Table 1 for an explanation) on instream water quality and habitat conditions was fairly consistent. In particular, the arrows that connect riparian buffer width, zonation and length with instream water quality and habitat variables were supported, on average, by 6–10 evidence items (Fig. 5). Biological response to riparian management was consistent only for primary producers (although evidence was rare), while fishes and macroinvertebrates revealed a fairly unpredictable response.

While riparian management studies almost exclusively addressed real management interventions, the majority of catchment management-related studies presented the outcome of mathematical models. The models were based on catchment-wide management scenarios and represented 14 out of the 55 core studies reviewed here. Notably, only a single study addressed the effects of a real sub-catchment-scale management intervention (Hughes & Quinn, 2014). The authors presented results from a 13-year integrated catchment management plan, investigating the Figure 5
effects of cattle exclusion from and land use change in the riparian zone (total area: 153 ha) of a headwater catchment in western Waikato, New Zealand.

The dominant drivers of riparian degradation that preceded management and restoration in the reviewed studies were agriculture and silviculture (30% each of the studies). Although there was evidence for direct effects of both these land uses on the erosion of fine mineral sediments (11 and 8% of the studies, respectively; Fig. 5), many studies reported that riparian vegetation influenced interactions between land use and instream sediment and nutrient conditions, particularly through buffer density (15% of the studies), width (15%), composition (30%) and length (26%), but less so for buffer age (4%).

Biological effects have been reported mainly from riparian management studies, whereas only a single catchment-scale modelling study addressed biological response variables (Guse et al., 2015). The effects are detailed below.

3.5 Evidence of riparian and catchment management effects

3.5.1 Nutrient pollution

About 75% of the studies reported effects of riparian restoration on nitrogen and/or phosphorous retention in surface and sub-surface waters (Fig. 3). Restorations typically consisted in planting riparian buffers, promoting vegetated buffer strips or fencing, to manage riparian degradation through livestock. Well-developed riparian buffers can retain up to 100% of total nitrogen from the sub-surface groundwater flow before entering the stream network (Feld et al., 2011; Aguiar et al., 2015), but retention capacities for nitrate usually range over 50–75% (Dosskey, 2001; Broadmeadow & Nisbet, 2004; Mankin *et al.*, 2007; Krause *et al.*, 2008; Dodd *et al.*, 2010; Collins *et al.*, 2012). Phosphorous retention by riparian buffers was slightly lower, at 40–70% (Dosskey, 2001; Dodd *et al.*, 2010; but see Kronvang *et al.*, 2005) and mainly associated with particles retained from surface runoff (Dosskey, 2001).

Several features, such as buffer length, width, zonation and density, seem to influence nutrient retention (Fig. 5). Buffer width was positively related to N and P retention (Dosskey, 2001; Feld et al., 2011; Sweeney & Newbold, 2014; King et al., 2016) and, together with buffer zonation, they can control the amount of nutrients retained from surface runoff and upper groundwater layer (Dosskey, 2001). A buffer width of 30 m was reported to effectively retain N and P from surface and sub-surface groundwater runoff, if buffers consisted of multiple zones of mature wooded vegetation and grass strips (Feld et al., 2011; Sweeney & Newbold 2014). King et al. (2016) found that 15 m wide buffers retained 2.5 times more nitrogen from the sub-surface groundwater than 8 m wide buffers, while buffer vegetation type had no significant effect. Denitrification plays an important role in the overall nitrogen retention capacity. It is promoted by carbon-rich soils with high microbial activity, which usually occur in wetlands (Mayer *et al.*, 2005). Lowrance at al. (1995) found denitrification rates in forested riparian buffers to be significantly lower than those measured in adjacent grassy riparian buffers, while denitrification rates in hydrologically intact wetlands can resemble those of mature riparian forests. The authors concluded that denitrification rates in their study were due to factors other than riparian reforestation itself. Total phosphorous was primarily and effectively retained by grass strips ranging 1–3 m in width that mechanically filter phosphorous compounds adhered to fine sediment particles (Dosskey, 2001; Yuan et al., 2009). The role of buffer length and density was less often quantified, but buffer strips >1,000 m in length appeared to support nutrient retention (Feld et al., 2011).

The role of riparian buffer tree age for nutrient management remains unclear. Trees and shrubs, with deep and dense root systems can retain nitrogen more effectively at intermediate ages (ca. 15 a), whereas mature stands of woody vegetation (ca. 40 a) were found to be less effective (Mander *et al.*, 1997). However, due to the shade that trees and shrubs cast on the stream banks, dense wooded buffers can suppress the understory vegetation and hence negatively influence stream bank stability and filtering effects of the understory vegetation, with adverse effects on sediment and phosphorous retention (Hughes & Quinn, 2014).

In the absence of riparian vegetation planting, riparian livestock exclusion by fencing appears to be less effective an option to retain nutrients if compared to vegetated riparian buffer strips (Parkyn *et al.* 2003; Collins et al. 2012; Muller et al. 2015). However, fencing is a prerequisite for the establishment of vegetated buffers where livestock grazing occurs in the riparian area.

Irrespective of the kind of riparian intervention to reduce nutrient pollution, there is a common shortcoming in the design of studies that prohibits the calculation of net retention effects taking into account the type II errors. Net retention effects can be quantified by comparing the conditions before and after buffer management with those of unmanaged (control) sites. There is evidence that agricultural control sites without riparian buffer structures attenuate already 27–35% of nitrate-N (Clausen et al., 2000; King et al., 2016), which points at the need to include control effects in the quantification of management effects. The mere comparison of managed and unmanaged sites after buffer instalment, however, although a common design in many studies, does not fulfil the criteria of the BACI design, as the conditions at the managed site before management may deviate substantially from those at the unmanaged (control) site considered, which then may lead to an overestimation of the effect size attributable to the management intervention.

At the broad scale, simulations of different land use intensities and agri-environmental schemes suggest that catchment-scale management might reduce nutrient loads in stream systems by 25–50% for nitrogen and 8–50% for total phosphorous (Krause et al., 2008; Lam et al. 2011; Hughes & Quinn, 2014; Weller & Baker, 2014). However, the direct comparison of nitrogen reduction levels requires a harmonisation of the different N compounds considered (e.g. nitrate, nitrate-N, total nitrogen). In addition, the broad-scale models also revealed that part of the variability in the nutrient reduction is explained by other environmental co variates such as temperature, precipitation or soil characteristics.

3.5.2 Fine sediment pollution

In general, riparian buffers can retain between 60–100% fine sediment from surface runoff (Dosskey, 2001; Hook, 2003; Mankin *et al.,* 2007; Yuan et al., 2009; Feld *et al.*, 2011; Sweeney & Newbold, 2014), although once again BACI designs have been rarely applied. Retention capacity was higher for sand-sized particles (up to 90%) than for silt and clay-sized particles (20%) (Dosskey, 2001). Sediment retention has primarily been linked to grass strips, which act as mechanical filters at widths between 3 and 8 m (Hook, 2003; Mankin *et al.,* 2007). However, Dosskey (2001) found that riparian stiffgrass almost completely retained sand-sized sediments already at a width <1 m. In contrast, riparian trees and shrubs have been found much less effective in the retention of fine sediments (Sovell *et al.,* 2000, Yuan et al., 2009). Shading can suppress the understory vegetation and thus reduce the buffer's sediment filter functionality (Hughes & Quinn, 2014). Consequently, buffer tree age and height might negatively affect sediment buffer functionality, as close-to-mature tree stands with their wider and dense canopies cast more shade than less developed woody vegetation. However, evidence on negative buffer effects and the role of buffer tree age in this context is still scarce.

The role of riparian vegetation length and density has not been assessed frequently in riparian management studies, although both aspects are frequently discussed with regard to the limitations of vegetated riparian buffers. Some studies suggest that gaps in the riparian buffer system, together with insufficient buffer width (3–8.5 m) or length cause a weak sediment retention (Parkyn *et al.* 2003; Collins *et al.* 2012). In addition, riparian actions to control lateral sediment inputs are likely to not reduce instream sediment content when the upstream area is already exposed to sediment inputs (Collins *et al.* 2012). This points at the role of buffer longitudinal location as an important determinant of its effectiveness, as riparian buffers cannot mitigate sediment pollution that occurs further upstream in the continuum. Instead, riparian management should cover the entire stream network subjected to lateral sediment inputs, in order to effectively control sediment pollution.

The effects of riparian fencing on sediment retention are similar to those reported for nutrients, since fencing primarily induces the establishment of riparian grass vegetation as a mechanical filter strip. Furthermore, fencing reduces fine sediment and nutrient input by cattle activity. The effects of fencing are detectable shortly after instalment of fences (Carline & Walsh, 2007), since grass strips grow fast and may already provide full functionality after one or a few years. In general, however, the evidence of the effects of fencing appears to be less consistent as compared to planting buffer vegetation, which renders fencing alone rather insufficient to guarantee the establishment of a functional riparian buffer strip.

Buffer strips need to be thick and wide enough to prevent gully erosion (Dosskey, 2001), which can occur because of damage from agricultural activities such as ploughing at the riparian zone. Removing vegetation cover and ploughing perpendicular to the stream can initiate gully erosion and thus can easily counteract the effect of riparian buffers. In contrast, ploughing along the contour line can help reduce gully erosion (Dosskey, 2001). Surprisingly, tile drainages, and their effects on riparian buffer performance did not figure in the literature reviewed, although there is evidence of their importance in pollutant flux (e.g. Jacobs & Gilliam, 1985).

Four catchment-scale studies addressed management effects on fine sediment pollution, two of which detected fairly limited reductions ranging 0.8–5.0% following the simulation of management interventions (Lam et al., 2011; Panagopoulos *et al*., 2011). In contrast, the other two studies by Gumiere et al. (2014) and Nigel et al. (2014) found vegetated riparian buffers to effectively reduce sediment loss by 32–93% and 40%, respectively. The major determinant of sediment trapping efficiency in the case study model by Gumiere et al. (2014) was buffer density (and with a minor role also buffer location; model area <1 km^2), while Nigel et al. (2014) defined a variable buffer width (5–120 m) conditional on the topography (i.e. slope) and economic restrictions (i.e. agricultural land use) in their model catchment (model area: 108 km^2). The results of these studies suggest that the potential of riparian buffers to reduce instream annual sediment loads can be fairly limited and influenced by catchment features, yet in general bear a great potential to reduce fine sediment pollution, if buffer density in the catchment achieves 70%.

3.5.3 Shade and water temperature

Most studies report a cooling effect linked to the width of riparian wooded vegetation (Collier *et al.,* 2001; Broadmeadow & Nisbet, 2004; Whitledge *et al.,* 2006; Broadmeadow *et al.,* 2011; Sweeney & Newbold, 2014). Accordingly, a buffer width of 20 m on either bank side has been found sufficient to keep water temperature within 2 °C of a fully forested watershed, while 30 m wide buffers on either side are required for full protection from measureable temperature increases (Beschta *et al.,* 1987; Sweeney & Newbold, 2014). Thermal damping by riparian vegetation was most effective at streams <5 m wide (Whitledge *et al.,* 2006) and at shading levels within 50–80% (Broadmeadow *et al.,* 2011), which points at stream width and buffer density as key controls of riparian shade and water temperature.

Surprisingly, we found limited evidence showing the effects of buffer length on water temperature. A rare example is provided by Collier *et al.* (2001), who found the first 150 m of a planted (15 m wide) riparian buffer to reduce water temperature already by 3 °C. Yet, in the absence of riparian trees, reheating may occur immediately. Riparian tree harvesting along stretches of 185 m–810 m length of alpine headwater streams led to an increase of 4–6 °C in water temperature (Macdonald et al., 2003). Based on modelling studies, Parkyn *et al.* (2003) concluded that at least 1–5 km of shaded stream length was required for first-order streams and 10–20 km for fifth-order streams to reduce water temperature to reference conditions. A width-length function of shading effects was illustrated by Broadmeadow & Nisbet (2004) and could help estimate required buffer width-length combinations to limit the maximum summer water temperature.

For tree age, the reviewed evidence suggests that mature riparian vegetation is required to maximise thermal damping (Broadmeadow *et al.*, 2011; Feld *et al.*, 2011; Sweeney & Newbold, 2014). Our synthesis clearly shows that buffer cooling effects, at least in summer, are related to the presence of tree cover (Fig. 5). Besides buffer characteristics, it is important to note that instream water temperature is controlled too by natural geo-climatic co-variates such as latitude, precipitation, stream size and current velocity (Collier *et al.*, 2001; Hook, 2003; Arora et al., 2016). This raises the need to put riparian buffer management into a regional geographical and climatic context. For instance, best practice buffer management is likely to differ between the temperate central European and the summer-dry Mediterranean region. More generally, there is a need for better heat budgets, to understand the physical mechanisms through which cooling, warming and insulating effects occur under different riparian canopies, with or without the influence of groundwater resurgence; radiative heating is only one component alongside sensible heat transfer or advection, yet has received most interest.

3.5.4 Large Woody Debris (LWD)

The presence and quantity of in-stream LWD is linked to riparian buffer width, zonation, length, density and buffer tree age. Opperman & Merenlender (2004) showed that fencing riparian vegetation over periods of 10–20 years increased the amount of LWD and subsequently enhanced the conditions of river biota. In this study, the density of trees, their basal area and the number of LWD pieces was higher in restored reaches than in unrestored reference reaches. This study also found debris dams were five times as numerous at restored reaches. McBride et al. (2008) revealed that passive restoration of the riparian zone, over a course of >40 years increased the presence of LWD. Yet, although forested reaches had 40% more pieces of LWD as compared to non-forested reaches, total LWD volume and number of debris dams remained similar between both groups of reaches. Other studies showed that forested reaches and reaches buffered by a 15 m-wide tree zone have almost four times as much LWD volume per bottom surface area unit as compared to pasture reaches, although there was a very strong seasonal variation (e.g. Lorion & Kennedy, 2009).

3.5.5 Coarse Particulate Organic Matter (CPOM)

Our review includes only one study that explicitly addressed the effect of riparian management on instream CPOM (Thompson & Parkinson 2011), investigating the effect of a planted multi-zone riparian buffer compared with open-canopy reaches. Leaf litter input was about 40–50% higher along restored reaches, accompanied by an increase in the richness of macroinvertebrate shredders due to the increased availability of litter, while open reaches showed a greater abundance and biomass of invertebrates feeding on autochthonous resources such as algae. Algal biomass showed no significant differences between restored and unrestored reaches.

3.5.6 Primary producers

There is evidence that aquatic primary producer biomass can be managed effectively by means of riparian shading. Notably, Hutchins *et al.* (2010) found riparian shade to be even more effective than nutrient reduction through sewage treatment. In combination, both management options led to a reduction of phytoplankton peak biomass by 44%, as compared to 11% at unshaded reaches. Shading can also effectively reduce periphyton and macrophyte growth (Davies-Colley & Quinn, 1998; Parkyn *et al.*, 2003). However, as a negative consequence, dissolved nutrients might be transported further downstream, thus extending the nutrients spiralling.

3.5.7 Benthic macroinvertebrates

We found evidence of both positive and negative responses of macroinvertebrates to fine sediment and temperature reduction at the catchment scale. For example, sediment retention by riparian buffers can increase macroinvertebrate density, but not diversity (Carline & Walsh, 2007). On the other hand, water temperature reduction in response to catchment-wide riparian shading was linked to the increase of several macroinvertebrate biotic indices (Collier *et al.*, 2001; Parkyn *et al.*, 2003; Quinn *et al.*, 2009; Dodd *et al.*, 2010), thus reflecting the dominance of organisms showing preferences for clean and cool water. Other studies report no changes

(Quinn *et al.*, 2009) or even a decrease in macroinvertebrate diversity and production in response to reduced water temperature (Weatherley & Ormerod, 1990).

3.5.8 Fish

Similar to macroinvertebrates, the response of fish to riparian restoration was inconsistent and in part species-specific. Fish density or growth rates may decline through riparian shade (Sovell *et al.*, 2000; Weatherley & Ormerod, 1990) or increase (Whitledge *et al.*, 2006). Melcher et al. (2016) observed consistent beneficial effects of riparian shading on water temperature and fish community composition in two piedmont streams, particularly supporting species adapted to cool water such as brown trout (*Salmo trutta*) and grayling (*Thymallus thymallus*).

Positive effects of LWD arise through an increased pool-riffle heterogeneity, which benefits some species such as trout (Sievers et al., 2017) and eel (Jowett *et al.*, 2009). After LWD addition, for example, trout density on average increased by 87.7% (Sievers et al., 2017). However, other species may benefit from more homogenous habitats without LWD (Lorion & Kennedy, 2009), which implies that beneficial effects of LWD are not universal, but species-specific.

4. Synthesis and recommendations

Riparian management offers a promising management option to recover and protect lotic species adapted to clear, cold, well-oxygenated and flowing water (e.g. Elliot & Elliot, 2010; Verberk et al., 2016). In fact, in comparison to open-canopy conditions, aquatic environments with reduced light and water temperatures, and at the same time enhanced amounts of LWD and CPOM are associated with unique and often diverse lotic communities of benthic algae (Potapova &

Charles, 2002; Hering et al., 2006), macroinvertebrates (Gutiérrez-Cánovas et al., 2013; Thomas et al., 2016) and fish (Jowett et al., 2009; Sievers et al., 2017) in temperate regions. A higher CPOM availability can diversify trophic links offering food for macroinvertebrate shredders, in particular during late autumn and winter, when primary production is limited by low temperatures (e.g. Wallace et al., 1997; Thomas et al., 2016). A higher abundance of LWD on the stream bottom increases habitat heterogeneity and thus the in-stream retention of nutrients and sediments (Gurnell & Sweet, 1998; Pusch *et al.*, 1998; Mutz, 2000; Gurnell et al., 2002).

Our study provides the first conceptual model based on published evidence, which links different anthropogenic drivers and pressures affecting riparian characteristics to the features that mediate anthropogenic impact on the freshwater ecosystem. The reviewed evidence, however, provided consistent results only for a limited number of relationships outlined in the conceptual model (Fig. 5). It is these well-evidenced cause-effect relationships that can help water managers design efficient schemes for riparian management and restoration. Our conceptual model discriminates four variables, namely light, water temperature, LWD and CPOM that can be considered largely scale-independent and thus point at management options with rather positive effects at the local scale, irrespective of other co-occurring stressors operating at the same or broader scales.

These variables are, however, conditional on the flow regime, which will largely determine the age, structure and complexity of riparian buffers even in altered situations. For example, reduced and homogenised flow is likely to promote dense and old buffer vegetation, with more shade casted and LWD accumulated on the stream bed. Consequently, riparian buffer management too requires the integration of flow and riparian vegetation dynamics (Egger et al., 2013).

Contrastingly, the beneficial effects of riparian management on nutrient and sediment retention are scale-dependent and thus often limited by particular adverse conditions at broader scales, such as extensive agriculture (Table 2) or environmental co-variates linked to topography and topology (Gumiere et al. 2011). Then, both the riparian and the catchment scale require consideration, to effectively manage and restore a stream reach or segment. Numerous studies provided evidence that the riparian and floodplain land use conditions upstream of a managed stream section can largely influence and even counteract site or reach-scale restorations (Mayer et al., 2005; Richardson et al., 2010; Feld et al., 2011; Lorenz & Feld, 2013; Giling et al., 2016). Such broad-scale adverse impacts, for example, imposed by intensive land use may operate up to 5–10 km upstream (Lorenz & Feld, 2013) or even further (Feld et al., 2011). Riparian management without broader-scale land use management thus is unlikely to be sufficient to protect lotic ecosystem integrity and diversity.

Table 2

In light of the evidence synthesized in this study, we recommend that riparian buffers should be i) at least 20–30 m wide (Dosskey, 2001), ii) consist of multiple continuous zones with trees, shrubs and grass strips (Weller & Baker, 2014) and iii) cover the entire stream reach or segment impacted by lateral diffuse nutrient and sediment inputs (Parkyn et al., 2003). Future research in this field is urgently required to evaluate sub-catchment and catchment-scale management options, in particular the effects of real (i.e. not modelled) agri-environmental measures such as land use abandonment and fertilizer management at broader scales. Future research should also address two widespread shortcomings in the evaluation of riparian buffers. Firstly, management studies should apply the BACI (i.e. before-after-control-impact) design, to be able to reliably quantify the net effects of management interventions and restoration measures. The comparison of managed (impact) and unmanaged (control) sites after the intervention (also referred to as “space-for-time-substitution”) may provide useful short-term estimations of the management effects, and may be the only option where decadal time periods are required for buffer development. However, these study designs do not replace controlled comparison with the conditions at the managed site *before* the intervention. Secondly, riparian management studies are often short-term (Feld et al., 2011) and thus do not allow of a reliable estimation of long-term effects, for example, in course of the development of riparian forests. Longer-term BACI assessments of riparian buffer effects are extremely scarce in the scientific literature. Only two field studies conducted in North Carolina and Pennsylvania, United States have reported nitrogen attenuation potential of riparian buffers using a 12 and a 15-year data set (Newbold et al., 2010; King et al., 2016). Computer simulation models can help quantify the long-term performance of riparian buffers for nutrient and sediment retention (see Tilak et al., 2014; 2017 for an example), yet require sound data to set-up and calibrate the models. Such data might be derived from a limited number of long-term field surveys, for instance, linked to or alike the network of Long Term Ecological Research (LTER) sites (https://lternet.edu/site/).

With regard to the location of riparian management in the stream continuum, our synthesis implies that scale-independent benefits are common in the upstream parts of the network. Indeed, almost 60% of the core studies addressed 1st and 2nd order streams, which points at a bias towards headwater studies in the reviewed body of literature. We may infer that this bias is owed to the fact that headwater and upstream sections are much more influenced by terrestrial and riparian vegetation (Nakano & Murakami, 2000), as opposed to wider and deeper sections further downstream in the continuum. Then, scale-independent management effects through shade, and CPOM and LWD recruitment are more likely to occur upstream in the network. Recent research on meta-community theory suggests that habitat improvements in the upstream part of the network are much more likely to enhance lotic biodiversity as opposed to stream sections further downstream (Swan & Brown, 2017). Hence, if biodiversity improvement is the goal of lotic ecosystem management, riparian restoration should start upstream in the network and then continue further downstream, to aid the subsequent recolonization of restored reaches.

5. Conclusions

Riparian management constitutes a widely-applied option to restore and protect stream ecological functioning and biology, yet with often variable and sometimes inconsistent effects. Management effects not only are controlled by physical buffer characteristics, but are subject to other environmental co-variates (e.g. slope, soil particle size, precipitation). Therefore, it is not trivial to provide general guidance for those in charge of the management and restoration of stream ecosystems towards a good ecological status. A critical synthesis of the available evidence, if embedded within a useful structural framework, can help identify generalisable management options that are likely to be beneficial for the instream biota. The conceptual model provided with this study constitutes such a framework and allows of the following statements, provided that the minimum demands (e.g. buffer length, width, zonation; see section 4 Synthesis) are met:

1. Consistent beneficial effects arise from the supply of coarse particulate organic matter, large woody debris and shade (and thus thermal damping) to the stream system. These effects are largely independent of the conditions further upstream in the continuum, i.e. the effects are scale-independent.
2. Inconsistent and sometimes even adverse effects are evident for the riparian buffer function, i.e. the retention of nutrients and fine sediments in the riparian area before both can enter the stream system. These effects are scale-dependent and conditional on the situation further upstream in the continuum.

3. To be beneficial, scale-dependent effects require concerted management efforts at both the riparian and the (sub-)catchment scale. Riparian buffer management thus needs to be accompanied by nutrient and erosion control measures at broader scales.
4. Evidence of the effects of (sub-)catchment-scale management options to reduce nutrient and fine sediment pollution is scarce and largely derived from modelling case studies of lowland catchments. The models' outcome, however, suggests that riparian management alone can buffer only up to 50% of the nutrients that enter the stream system. The other half requires nutrient reduction options (e.g. fertiliser management) at the broad scale.
5. Riparian management effects on aquatic biota are less often addressed and largely inconsistent, thus pointing at the poor and incomplete knowledge in the biological domain. However, biological effects implicitly require consideration, if the ultimate goal of stream management is to improve and sustain biodiversity and ecological status. Future studies should address biological effects of riparian management, to provide the scientific basis for an effective riparian management.

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Tables

Table 1: Criteria, variables and variable classification extracted from 138 references to form the review database and to draft the conceptual model of cause-effect relationships (Fig. 1).

Criterion	Variable	Variable classification
General study characteristics, meta-data	Study origin and location	Country, latitude, longitude
	Altitude (m a.s.l.)	Lowlands (<200), uplands (200–500), mountainous (500–800), alpine (>800)
	Catchment area at management site/reach (km^2)	Headwater (<10), small (10–100), medium (101–1,000), large (>1,000)
	Stream network position of management site/reach (Strahler order)	Upstream (1–2), middle (3–4), downstream (>4)
Drivers and pressures	Drivers	Agriculture, silviculture, urbanisation
	Diffuse pressures	Nutrient pollution, fine sediment pollution,
	Point-source pressures	Waste water pollution
	Riparian pressures	Vegetation removal, vegetation alteration
	Pressure spatial scale (km)	Site (<0.5), reach (0.5–2), segment (2–5), sub-catchment (>5), catchment (entire catchment)
Riparian management characteristics	Active Passive	Planting Fencing
	Riparian management spatial scale (km)	Site (<0.5), reach (0.5–2), segment (2–5), sub-catchment (>5), catchment (entire catchment)
	Riparian management spatial extent	Length (m), width (m), density (%), vegetation age (a)
	Vegetation zonation (Dosskey, 2001)	Single-zone (trees or shrubs or forbs or grass), multi-zone (any combination thereof)
Catchment management characteristics	Agricultural	Crop rotation, conservation tillage, livestock density, fertiliser application, land use change/abandonment
	Silvicultural	Afforestation
	Catchment management spatial scale	Sub-catchment or catchment (no further classification)

Instream environmental effects	Physico-chemistry	Nitrogen (Total Nitrogen, , NO_3, NH_4), phosphorous (Total Phosphorous, -ortho-PO_4, ortho-PO_4-P, Soluble Reactive Phosphorous), water temperature, light, conductivity, turbidity
	Habitat	Fine sediments, large woody debris (LWD), coarse particulate organic matter (CPOM), habitat quality index
Instream biological effects	Targeted organism groups	Fish, macroinvertebrates, aquatic macrophytes, benthic algae, riparian vegetation, ground beetles
	Diversity	Species richness, Shannon (community) diversity
	Composition/density	EPT taxa (Ephemeroptera-Plecoptera-Trichoptera), abundance, biomass
	Functions/traits	Primary production, feeding types

847 [a)] Available at the ArcGIS Online Resources Center.

Table 2: Evidence of riparian management effects in light of potential limiting factors operating at broader spatial scales. The table summarises the reviewed riparian management literature that reports weak or no effects after the implementation of management and restoration measures, and that attributes the lack of effects to broad-scale stressors/pressures that continue to impact the restored river sites/reaches.

Riparian management option	**Abiotic effect**	**Biological effect**	**Limitation**	**Reference [type of study]**
Wooded multi-zone riparian buffer strips, 5–30 m wide and >1,000 m long	Retention of nutrients (up to 100% N/P) and fine sediments (up to 100%), reduction of stream temperature, habitat improvement (LWD, CPOM)	Increase of macroinvertebrate and fish diversity, improvements of functional traits, improved community composition, enhanced fish biomass, less studies effects of riverine plants	Land use further upstream in the continuum continues to limit restoration success; poorly designed buffers (too narrow, too short) are not functional	Feld *et al.* (2011) [review of 57 riparian management papers, various regions and stream types worldwide]
Scenario 1 covers partial land use change on sensitive floodplain areas (e.g. hydromorphic soils, erodible soils) and 20 m-wide riparian forested buffers along the river course; scenario 2 covers full land use change on sensitive areas and 50 m-wide riparian forested buffers	Reduced nitrate leaching from the root zone (43–85% for scenarios 1 and 2, respectively); reduced nitrate contribution from the floodplain (70–100%); floodplain can even constitute a sink for river-derived nitrate.	--	Floodplain nitrate contribution constitutes only about 1% of total river nitrate loads per year; hence modelled management effects are negligible	Krause *et al.* (2008) [modelling of land use and management effects of two scenarios within a ca. 1,000 km^2 sub-catchment of River Havel, Germany]
Comparison of pasture sites with unlimited livestock access and fenced sites without livestock access and riparian trees/shrubs present	Bank erosion processes vary throughout catchments (with particular reference to their scale dependence); only two	--	The exclusion of livestock from riparian areas is generally reported as the principal factor in the measured improvements or differences; planting of riparian vegetation in headwater	Hughes (2016) [review of various studies with and without livestock access to river banks and riparian trees/shrubs]

	studies specifically attributed reduced stream bank erosion to the presence of riparian vegetation		streams and the subsequent shading of stream banks can reduce bank stability and promote channel widening (and hence a release of sediment; see also Hughes & Quinn 2014)	
Riparian management targeting the provision of riparian habitat that fulfils critical functions for fish (e.g. bank stability, shade/temperature, large wood, water clarity, sediment retention)	Riparian habitat is crucial for the provision of shade, control of channel complexity and sediment inputs through bank stabilization, input of large wood and allochthonous energy sources, and filtering of nutrients and toxins from adjacent land	Riparian habitat should be considered biologically critical for most species of freshwater fish, unless the habitat requirements of individual species indicate insensitivity to the ecological functions associated with riparian zones	Protecting the riparian zone alone may not be sufficient to maintain stream ecosystem integrity or species at risk, if the development within the watershed (e.g. agriculture or urbanization) significantly alters hydrology or water quality	Richardson *et al.* (2010) [review of various riparian management studies in light of habitat demands of fish]
Riparian land use in buffers of 100–200 m width and 500–10,000 m length upstream, and riverine hydromorphology 500–10,000 m upstream of biological sampling sites	--	Upstream land use and hydromorphology are stronger determinants of ecological recovery after restoration than local land use and hydromorphology at restored sites	Land use and hydromorphological degradation in the sub-catchment upstream can limit the success of local restorations	Lorenz & Feld (2013) [analysis of biological effects of riverine hydromorphology and riparian land use at several distances upstream of restored and unrestored lowland and mountainous stream sites in Germany]
Comparison of modelled nitrogen loads from cropland conditional on the amount of buffered stream length and streamflow	In the entire watershed, croplands release 92.3 t of nitrate nitrogen, 19.8 t of which is removed by riparian buffers; 29.4 t more might be	--	47% of cropland nitrogen load cannot be reduced by riparian buffers and must be addressed by other management options	Weller & Baker (2014) [modelling of riparian buffer effects on cropland nitrate loads at 1,964 sub-basins of Chesapeake Bay, USA]

	removed with all buffer gaps closed; the remaining 43.1 t of cropland load cannot be removed by riparian buffers			
Analysis of the response of aquatic macroinvertebrate assemblages to riparian replanting (8–22 a before monitoring) at agricultural streams	--	Macroinvertebrates did not respond to replanting over the time gradient, probably because replanting had little benefit for local water quality or in-stream habitat; invertebrate assemblages were influenced mainly by catchment-scale effects, but were closer to reference condition at sites with lower total catchment agricultural land cover	Reach-scale replanting in heavily modified (agriculturally-used) landscapes may not effectively return biodiversity to pre-clearance condition over decadal time-scales	Giling *et al.* (2016) [analysis of riparian vegetation replanting of different ages at streams in south-eastern Australia]
Meta-analysis of the effects of riparian buffer width and buffer vegetation type on the removal of nitrogen from surface runoff and sub-surface groundwater flow paths	Riparian buffers effectively remove nitrate through uptake and denitrification (mean: 74%), but the relation to buffer width is not strong	--	Riparian buffers are a best-practice management option, but only in concert with other management options at the watershed scale; soil characteristics can promote denitrification (high organic content, water-saturated soils)	Mayer *et al.* (2005) [review of the effects of riparian buffers on nutrient and fine sediment retention]
Passive ecological restoration (excluding livestock by fencing along an entire stream, 1 m from the stream bed) with the assumption that recovering riparian habitat will restore ecological processes (e.g. filtration, soil stabilization)	After eight years, the restored stream had complex riparian banks, similar to those of reference streams (more trees, less bare soil, increased habitat heterogeneity)	--	Water quality did not improve; the same low water quality in the reference stream demonstrated the need for a whole watershed-scale approach and for actions to improve agricultural practices before implementing restoration practices at a smaller scale	Muller *et al.* (2015) [monitoring of water quality and riparian habitat heterogeneity of an entire stream in France, eight years after lifestock exclusion through fencing]

Analysis of the capability of longitudinally restricted riparian forest buffers to enhance in-stream nutrient retention in nutrient-enriched headwater streams.	Riparian forested buffers can increase instream ammonia (but not phosphate) uptake through enhanced hydrologic retention (reduced flow) induced by LWD on the bottom	--	Already highly eutrophied streams seem to have a limited retention capacity for N and P components; instream nutrient retention cannot compensate for deficits in riparian nutrient retention when the nutrient supply exceeds the demand significantly	Weigelhofer *et al.* (2012) [experiment and modelling of the effects of riparian forested buffers on instream nutrient uptake]
Measurement of water quality along four Australian tropical streams in two catchments with similar agricultural development (mainly sugarcane growing) but contrasting riparian vegetation (intact native rainforest vs. exotic weeds).	Nitrate and nitrite (NOx) concentrations and loads were significantly lower in streams with greater riparian vegetation; yet, NOx concentration significantly increased with distance downstream (i.e. with the amount of fertilized agricultural land in the catchment)	--	An adequate reduction in NOx in streams can only be achieved by reduced fertilizer application rates in the catchments	Connolly *et al.* (2015) [comparison of N reduction along buffered and unbuffered streams in four agricultural catchments in Australia]

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Figure captions

Figure 1: Conceptual model showing the hypothesised hierarchical relationships between catchment drivers of impact (land-use), catchment pressures, riparian buffer management, instream environmental and biological states. Blue arrows represent assumed negative relationships, red arrows assumed positive relationships and grey arrows assumed unclear effects, i.e. both positive and negative relationships are possible. (See Supplementary Table S1 for the linkage of arrow numbers and core references.)

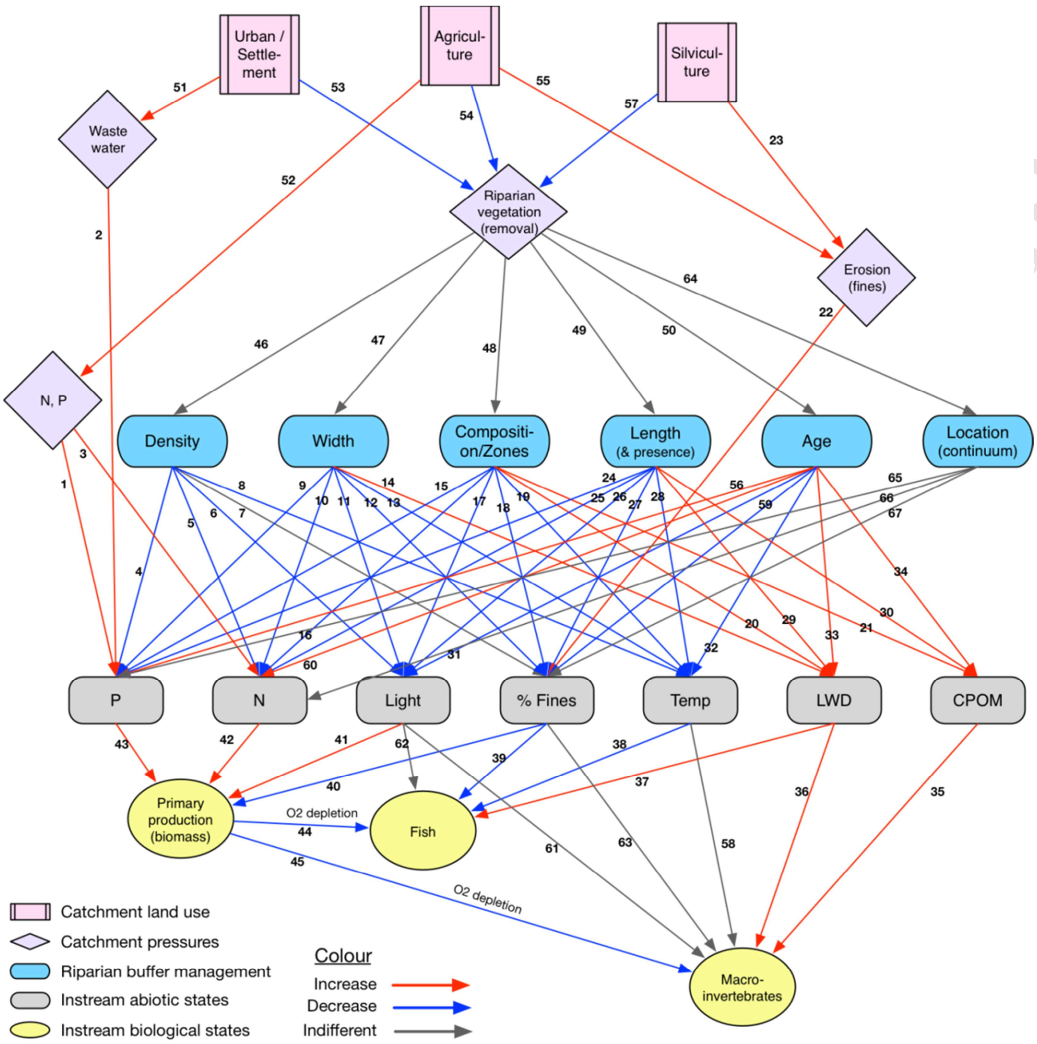
Figure 2: a) Length (km) and b) width classes (m on either side of the stream) of riparian management areas addressed by the 55 core studies.

Figure 3: Common abiotic state variables (stressors) addressed in the 55 core management papers (N=**n**itrogen, P=phosphorous, Organic=organic matter).

Figure 4: a) Common biological response variables and b) community attributes addressed by the 55 core management studies (Riparian=riparian **i**nvertebrates, Indices=various assessment indices).

Figure 5: Conceptual model showing the meta-analysis results through hierarchical relationships between catchment land-use, catchment pressures, riparian buffer management, instream abiotic states and instream biological states. Arrows represent consistent evidence of negative (blue) and positive (red) relationships, or unclear evidence (grey) with both positive and negative effects reported in the literature. Arrow thickness is proportional to the number of studies supporting a significant relationship between two elements of the model. (See Supplementary Table S1 for the linkage of arrow numbers and core references.)

Figures



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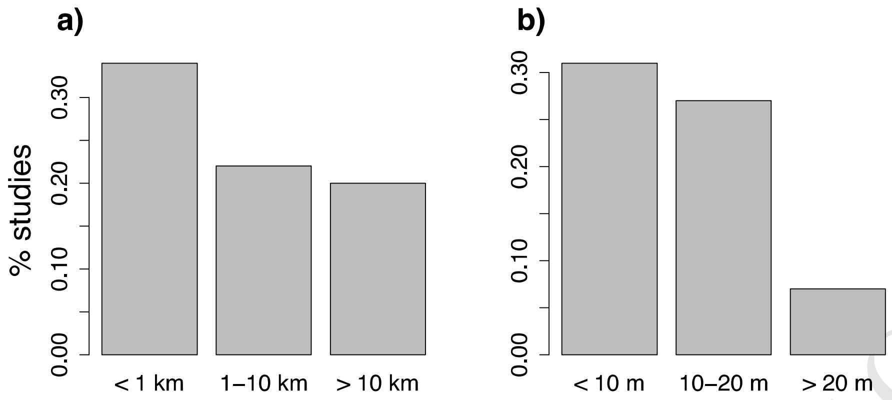

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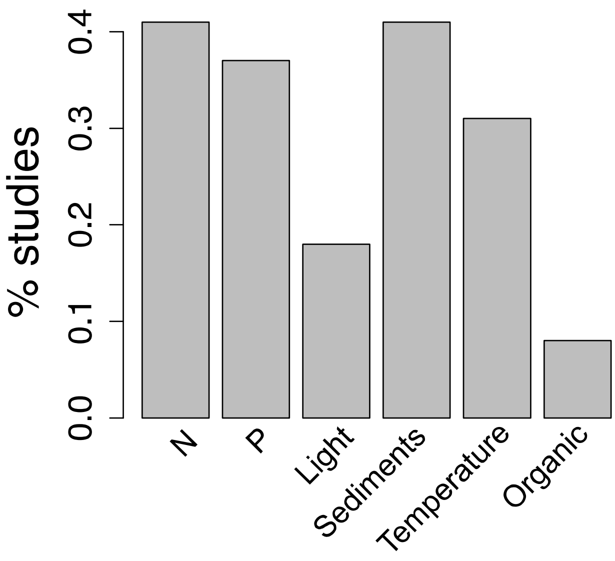

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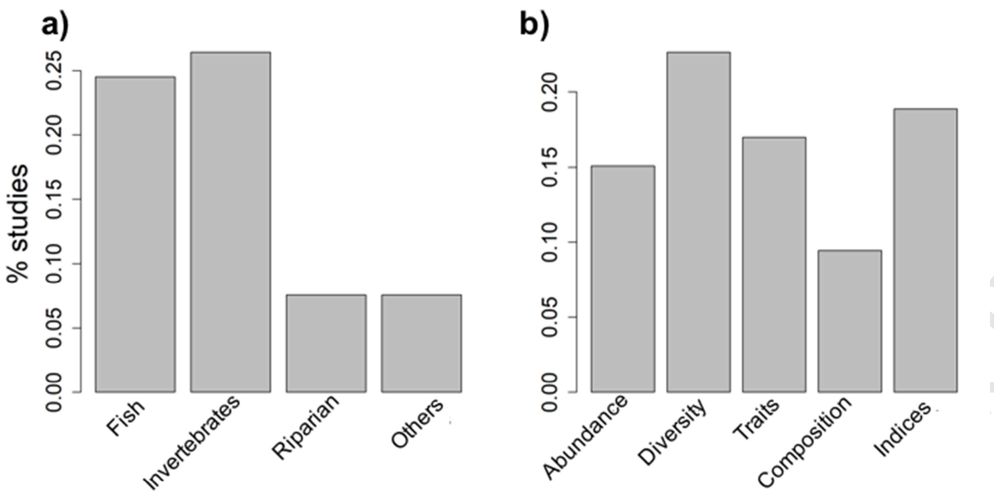

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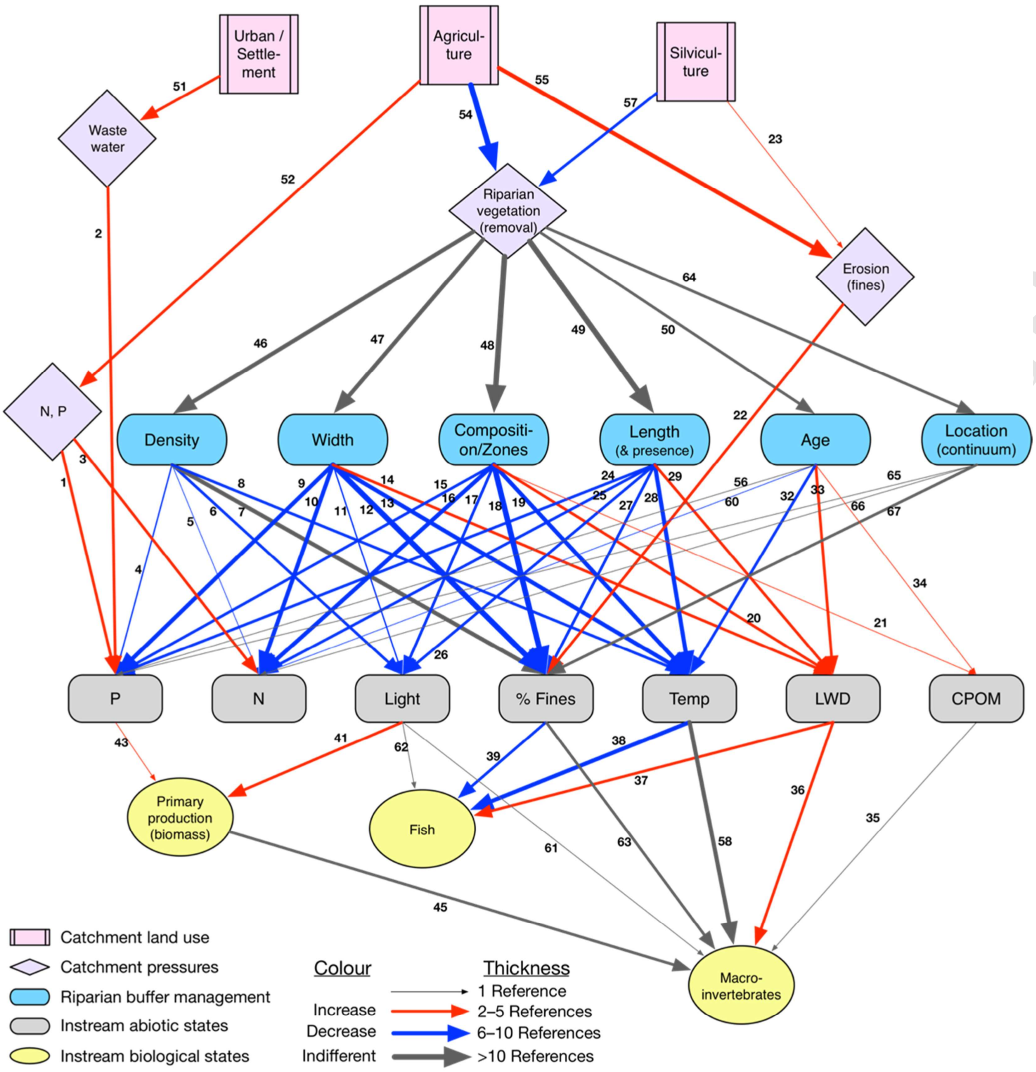

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Highlights

- A conceptual framework to evaluate riparian management options is presented.
- The framework is tested against the evidence in the management literature.
- Consistent beneficial effects on the instream environment are detectable.
- For full ecosystem protection, management beyond the riparian scale is required.