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Review

Managing riparian buffer strips to optimise ecosystem services: A review

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ABSTRACT

Riparian buffer strips provide a wide range of ecosystem services in agricultural landscapes with benefits extending beyond those provided by non-riparian field margins. For example, in addition to enhancing the aesthetic value of the landscape and protecting biodiversity, riparian buffer strips also mitigate diffuse pollution and provide inputs to freshwater systems. Their multifunctional nature makes developing management prescriptions complex with the need to identify conflicts, interactions and synergies between the services offered. Here we explore how the placement, physical properties, management and vegetation structure influence the potential of riparian buffer strips to deliver a range of benefits.

Under high nutrient loadings, buffer strips can become saturated and act as a source of pollutants, thus limiting their long-term effectiveness. Furthermore, in saturated buffer strips nitrification can increase greenhouse gas emissions. Buffers should not therefore be viewed as an end-of-pipe solution, but within a wider management framework that controls pollutants at the source. Furthermore, biomass removal (e.g. via mowing) can prevent nutrient saturation, increasing the longevity of buffer strips; such operations should, however, be carefully timed to reduce adverse impacts of disturbance on water quality and biodiversity.

Wooded buffers can be less effective than grass buffers at intercepting sediments and sediment bound pollutants, but provide many benefits associated with mitigating the impacts of climate change (e.g. carbon capture and moderating aquatic temperatures). This highlights potential trade-offs between climate change and water quality objectives. Zoned buffers that combine strips of riparian woodland and grass, could therefore deliver benefits transcending these policy areas. With such buffers taking large areas of land out of production, they may not be financially sustainable, particularly in intensively managed catchments. In such catchments, to balance food production goals with other ecosystem services, vegetated and/or forested buffer strips, of variable width, should be selected based on objectives at the local or regional scale.

Catchment scale initiatives that support a diversity of unbuffered watercourses, vegetated buffers and wooded buffers could help address conflicts between policy areas. Furthermore, with riparian ecosystems being naturally dynamic and diverse, restoring riverbank heterogeneity will also enhance biodiversity. Catchment management plans should take an integrative approach to spatially target the placement of buffer strips to where the greatest benefits can be derived. This review aims to inform environmental managers, regulators and practitioners on how the multifunctionality of riparian zones can be optimised through targeting management actions at the local and catchment scale.

1. Introduction

Over the past century, degradation in the quality and function of riparian ecosystems resulting from changes in river and land management practices have simplified the physical structure of riparian habitats, altered river morphology and degraded water quality (Beechie et al., 2010; Larsen et al., 2009). With riparian habitats providing a range of ecosystem services, there is a need to protect and restore these vital habitats (Beechie et al., 2010; Gilvear et al., 2013). Subjected to disturbance from watercourses, riparian habitats are naturally dynamic and complex supporting many specialist flora and fauna species (Malmqvist, 2002; Naiman and Decamps, 1997; Rood et al., 2014).

In agricultural landscapes, riparian field margins exist in the transition zone (i.e. the ecotone) between agricultural and freshwater habitats. Watercourse protection is becoming increasingly embedded in

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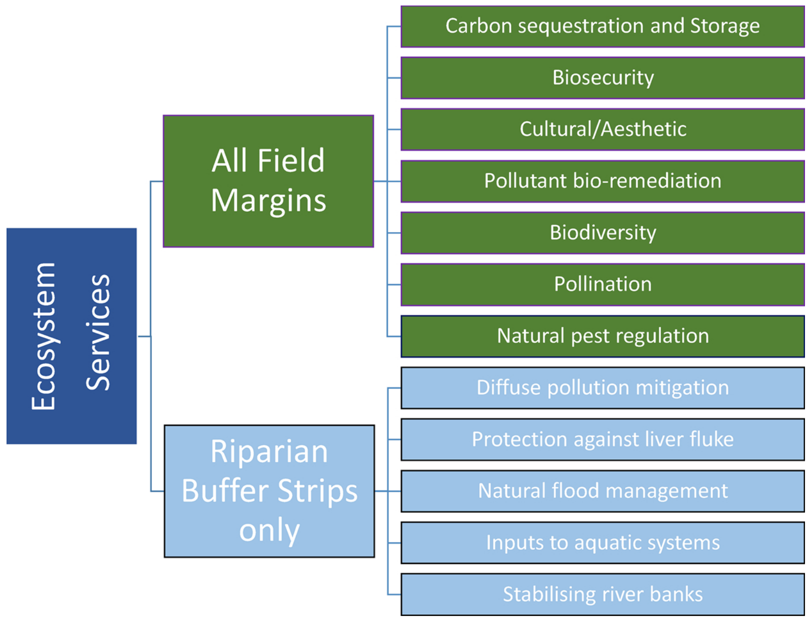

Fig. 1. Overview of ecosystem services provided by all field margins (i.e. irrespective of whether they are adjacent to watercourses or not) and additional ecosystem services provided specifically by riparian buffer strips.

industry standards and environmental legislation (e.g. The European Water Framework Directive, New Zealand's Resource Management Act 1991), restricting agricultural practices (e.g. cultivation, application of agrochemicals) adjacent to watercourses. Riparian field margins therefore provide unique ecosystems in agricultural landscapes with both the inherent complexity of riparian ecosystems, and historic lack of agricultural management, making them rich in biodiversity (Cole et al., 2015; Naiman and Decamps, 1997). Riparian buffer strips (i.e. riparian field margins managed for environmental benefits) have the potential to provide a much wider range of ecosystem services than non-riparian field margins (Fig. 1). Management prescriptions must therefore expand beyond the range of ecosystem services provided by regular field margins (Junge et al., 2015; Olson and Wäckers, 2007), to consider additional ecosystem services specific to riparian field margins (Beechie et al., 2010; Gilvear et al., 2013; McCracken et al., 2012; Stutter et al., 2019).

With riparian field margins providing a much wider range of ecosystem services than non-riparian field margins, developing effective management prescriptions is particularly complex. This review aims to explore how the spatial location, physical properties, vegetation composition and structure and management of riparian zones influences the services these zones provide. Through reviewing the scientific literature, we explore synergies and trade-offs between ecosystem service delivery and identify riparian management practices with the potential to enhance the services delivered. This review aims to inform environmental managers, regulators and practitioners on how the multifunctionality of riparian zones can be optimised through targeted management actions aimed to deliver local and catchment scale objectives.

2. Riparian buffer strips

As a result of the multifunctional nature of riparian buffer strips, management is pertinent to several, often disjointed, policy areas (e.g. food security, climate adaptation, biodiversity conservation and water quality). For example, in Europe, relevant policies include the Common Agricultural Policy (CAP), Habitats Directive, Floods Directive, Water Framework Directive and EU's Climate and Energy Policy. Formulating management prescriptions for riparian buffer strips is therefore complex and requires the integration of policy areas and consideration of multiple ecosystem services to identify conflicts, interactions and synergies (McCracken et al., 2012; Stutter et al., 2019; Valkama et al., 2019).

Agricultural practices degrade water quality through bank erosion and sediment run-off, and contamination by urine, faeces (i.e. both directly from livestock and via the application of organic fertilisers), agrochemicals (e.g. herbicides, pesticides and inorganic fertilisers) and veterinary antibiotics (Aguiar et al., 2015; Chu et al., 2010; Feld et al., 2018; Lin et al., 2011; Stutter et al., 2012). Establishing riparian buffer strips is widely advocated as a means of reducing diffuse pollution (e.g. Clean Water Act in the US and CAP in the European Union) (Aguiar et al., 2015; Poole et al., 2013; Stutter et al., 2012). In intensively grazed fields, riparian buffer strips are established by fencing land adjacent to the watercourse to exclude livestock, and thus prevent their excrement from contaminating the watercourse. Consequently, the resultant buffer, which is typically left unmanaged, becomes a (semi-) permanent landscape feature. Fences not only physically exclude livestock from the buffer strip, but in preventing livestock congregating at riverside watering points can reduce bank erosion and poaching (Aguiar et al., 2015; O'Callaghan et al., 2019; Poole et al., 2013). In arable fields, buffer strips are frequently established simply by removing production from riparian field margins. Such unfenced buffers can be subjected to annual cutting or aftermath grazing and are often relatively transitory landscape features with blurred boundaries.

Riparian buffer strips intercept pollutants through a range of physical, hydrological, chemical and biological processes (Dosskey et al., 2010; Knight et al., 2010; Tabacchi et al., 2000; Vought et al., 1995). These processes are influenced by sediment loading, weather conditions, geology, hydrology, soil type, vegetation type (i.e. both living and dead) and the physical structure and management of the buffer strip (e.g. width, cutting/grazing regime and spatial location) (Aguiar et al., 2015; Tabacchi et al., 2000). Riparian buffer strips create a barrier between field management practices and the watercourse and trap pollutants by diverting runoff through denser vegetation (Stutter et al., 2012). With most of the water entering receiving waterbodies via first and second-order headwater streams, targeting riparian buffers to protect these streams is particularly important (Correll, 2005).

2.1. The impact of buffer strip width on diffuse pollution

The effectiveness of buffer strips at mitigating diffuse pollution tends to be positively related to width (Knight et al., 2010; Rasmussen et al., 2011; Zhang et al., 2010), as a result of both an increased distance from in-field management practices and a greater area of vegetation to intercept pollutants. In addition to site specific factors (e.g. topography, soil type and vegetation structure), the impact of buffer width varies with pollutant type; particularly how a pollutant is transported (i.e. in the dissolved state or bound to sediments). Soluble pollutants are primarily reduced by infiltration and biological assimilation/ degradation, whereas, sediment bound pollutants are primarily removed by sediment trapping. Lin et al. (2011) found that increasing buffer strip width from 4 to 8 m did not reduce loadings of soluble herbicides (i.e. atrazine and s-metolachlorm) but effectively reduced loadings of the sediment bound herbicide, glyphosate. Similarly, a meta-analyses by Valkama et al. (2019) indicated no impact of width on nitrogen removal efficiency, contrary to the model predictions of Zhang et al. (2010). Schmitt et al. (1999) suggested widths of 7.5 m are effective at trapping sediments and sediment-bound pollutants (e.g. non-soluble phosphates), whereas to control soluble pollutants (e.g. nitrates and dissolved phosphorus) widths of 15 m are recommended.

Sediment trapping efficiency doesn't increase linearly with width, but instead increases sharply for the first 10 m with benefits then tailing off (Dorioz et al., 2006; Liu et al., 2008). Optimum width depends on site-specific factors (e.g. run-off sediment loads, pollutant type, soil type and slope), vegetation structure (influencing the ability of the vegetation to trap sediments and sediment bound pollutants) and species composition (influencing assimilation and biodegradation) (Dosskey et al., 2008, 2010; Schmitt et al., 1999). Coarser sediments are more likely to get trapped than fine sediments and, consequently, on gently sloping land (2 %) 4 m grass buffers were almost 100 % effective at trapping sandy loams but only 35 % effective at trapping silty clay loams (Dosskey et al., 2008). This makes it difficult to derive generic optimum widths, however, for gentle slopes (less than 10 % gradient), widths of 8–15 m are typically adequate (Dorioz et al., 2006).

Clearly optimum widths are context/pollutant specific. Widening buffers beyond what is necessary, particularly when the adjacent land is highly productive, is not economically viable and can result in resistance to adoption (Buckley et al., 2012; Dosskey et al., 2010). Variable width buffers (i.e. where width changes in relation to site specific risks to the waterbody) can help balance agronomic and environmental goals, however these can be difficult to administrate and regulate.

3. Vegetated buffer strips

3.1. The impact of vegetation composition on diffuse pollution mitigation

Morphological and functional traits of plants (i.e. physiological, structural and life history characteristics) can influence the efficiency of vegetated buffers to intercept and remove pollutants. Taller, denser, vegetation with less flexible stems increase hydraulic roughness, providing resistance that reduces flow velocity and increases the infiltration and trapping of fine sediments and pollutants (Dosskey et al., 2010; Krutz et al., 2005; Roberts et al., 2012; Tabacchi et al., 2000). Although stem flexibility reduces hydraulic roughness, plants with more flexible stems tend to resist and recover from flood disturbance (e.g. less susceptible to breakage, buckling or uprooting) (Puijalon et al., 2011; O'Hare et al., 2016). This trade-off between intercepting pollutants and tolerance to flood disturbance highlights the potential for riparian seed mixes to exploit inter-specific variation in stem flexibility.

Below ground plant traits also come into play, with both root depth and complexity influencing soil permeability and the plant's ability to stabilise sediments and prevent substrate mass movement. Complex fibrous roots are particularly good at binding surface soil, stabilising sediments (O'Hare et al., 2016; Perez et al., 2017). Deeper rooted species, however, can provide support to riverbanks therefore reducing bank erosion and improve soil permeability thus increasing infiltration rates (O'Hare et al., 2016; Perez et al., 2017).

The ability of species to survive throughout the year (i.e. during winter or the dry season), and seasonal changes in growth and biomass will impact the vegetation ability to reduce water flow, trap sediments and uptake and assimilate contaminants. For example, many annual or perennial species dieback during winter limiting their potential to mitigate diffuse pollution year-round (O'Hare et al., 2016). Plant life cycles must therefore be considered to ensure adequate vegetation coverage during periods when pollution risks are highest (e.g. cultivation, ice thaw and heavy rain: Vidon et al., 2010).

Grasses are quick to establish and present all year round. Their dense, flexible stems, increase hydraulic resistance, disperse concentrated overland water flows and quickly recover from disturbance (Dosskey et al., 2010). This makes them effective at trapping sediments (Erktan et al., 2013) and sediment bound pollutants (Lin et al., 2011), and densely tillered, deep rooted, species are particularly effective (Mekonnen et al., 2016). Even narrow strips (0.5–1.2 m wide) of stiff-stemmed grass species are effective at mitigating both dissolved and sediment bound pollutants (Wang et al., 2018). Grassy buffers, while being effective at protecting watercourses are, however, less effective at providing other ecosystem services (e.g. pollination services and carbon storage).

With species deviating in their response to environmental stress (e.g. extreme cold or flooding), mixes of carefully selected species could increase the stability of ground cover. Despite this, vegetated buffers are often established with a single species (Erktan et al., 2013). The impact of increasing the functional diversity of plant species has yielded both positive (Correll, 2005) and negative (Erktan et al., 2013) impacts on diffuse pollution mitigation. This disparity is driven by the morphological and functional traits of the species in question, alongside their spatial configuration, with clear zones of specific species appearing to outperform species grown together (Erktan et al., 2013). With the frequency of climatic extremes (i.e. drought and flooding) predicted to increase (Fischer and Knutti, 2015), exploiting interspecific differences to environmental stress to ensure the persistence of ground cover is likely to become increasingly important.

Exploring the interaction between plant species, their response to stress, their functional traits and the retention, assimilation, adsorption, mobilisation and removal of pollutants from buffer strips remains an interesting avenue for research. Determining whether functionally distinct plant species are synergistic in their actions could lead to the formulation of seed mixtures that enhance the performance of buffer strips and ensure continued effectiveness under environmental stress. Furthermore, to capitalise on the benefits derived from land taken out of production, seed mixtures should consider other regionally important ecosystem services (Fig. 1). For example, including nectar and pollen-bearing species in areas reliant on insect pollinated crops.

3.2. The impact that vegetated buffer strips have on biodiversity

Lack of management in riparian buffer strips results in changes to the vegetation structure, influencing its capacity to mitigate pollutants and to provide for terrestrial and aquatic biodiversity. Although riparian buffers can harbour a suite of wetland species absent from other agricultural habitats (Boutin et al., 2003), evidence on their effectiveness at increasing botanical diversity is conflicting (Cole et al., 2015; Feehan et al., 2005; Manhoudt et al., 2007; Stockan et al., 2012). Lack of management, however, typically results in an increase in the structural diversity of vegetation (e.g. leaves, flowers, seeds, stems and grass tussocks) thus increasing diversity at the micro-habitat and farm scale; with positive implications to a range of phytophagous invertebrates, pollinators, parasitoids and predatory arthropods (Cole et al., 2012a, 2015; Ó hUallacháin et al., 2014; Woodcock et al., 2009). Many predatory arthropods in intensive agricultural landscapes rely on unmanaged field margins to provide aerated and stable microhabitats (e.g. tussocky grass species) to aestivate or overwinter (Olson and Wäckers, 2007; Pfiffner and Luka, 2000; Cole et al., 2012a, 2012b). Protecting buffer strips from farm traffic will also reduce soil compaction and improve soil structure and biodiversity (Spooner et al., 2002).

Buffer strips support rich small mammal (Chapman and Ribic, 2002) and herpetofaunal communities (Maritz and Alexander, 2007) and provide foraging habitat for farmland birds (Cole et al., 2012a; Josefsson et al., 2013). Linear features such as buffer strips, however, can pose an increased predation risk to these birds (Morris and Gilroy, 2008; Schneider et al., 2012). The inherent connectedness of watercourses, and thus riparian habitats, facilitates the coordination of conservation efforts between farms in a catchment and consequently buffer strips have great potential to enhance ecological connectivity (Maritz and Alexander, 2007; McCracken et al., 2012).

Wider buffers have a higher interior to edge ratio and the greater distance from in-field practices (e.g. agro-chemical application, livestock grazing and cultivation) results in a more stable interior habitat. Their lower nutrient status supports plants adapted to nutrient-poor environments (Dybkjær et al., 2012). However, while plant species richness was similar in unbuffered grassland field margins, narrow buffers (< 3.5 m) and wide buffers (> 5 m), the abundance of flowers was greater in wide buffers thus increasing the availability of forage for pollinators (Cole et al., 2015). Wider buffers also support greater densities of predatory Heteroptera (Gilbert et al., 2014) and Carabidae beetles that overwinter as adults (Cole et al., 2012b). Being active early in the season, such beetles provide an important first line of defence against spring active crop pests such as aphids (Cole et al., 2012b). Wider buffer strips therefore have the potential to enhance both natural pest control and pollination services in agricultural systems. Although vegetated buffer strips appear to benefit a range of taxa, mobile taxa associated with disturbed habitats (e.g. Linyphiidae spiders and mobile Carabidae beetles) are favoured in unfenced riparian zones (Cole et al., 2012a, b; Stockan et al., 2014). Maintaining a diversity of fenced and unfenced riparian zones will therefore provide a divesity of habitats favouring a wider suite of species at the catchment scale.

The presence and management of riparian buffer strips also impacts aquatic biodiversity and, with the adult stages of many riverine invertebrates being terrestrial, this can impact on pollination and biological control services (Raitif et al., 2019). Buffer strips reduce sediments and contaminants entering the watercourse, improving the ecological status of waterbodies (Rasmussen et al., 2011; Stutter et al., 2012) and benefitting pollution-sensitive taxa such as mayflies, stoneflies and caddisflies (i.e. Ephemeroptera, Plecoptera and Trichoptera) over pollutant tolerant chironomids (Muenz et al., 2006; Orzetti et al., 2010). Effects are, however, context specific and influenced by the buffer strips physical properties (e.g. age, width, continuity), vegetation structure (e.g. species, canopy coverage), the physical attributes of the stream (e.g. width, substrate and hydrology) and the intensity of land use at the catchment scale (Orzetti et al., 2010; Sovell et al., 2000; Wahl et al., 2013).

The taller denser vegetation of buffer strips provides shade, with the positive impacts on aquatic invertebrates being linked to lower water temperatures (Parkyn et al., 2003). Vegetated buffers also introduce organic material (e.g. plant and invertebrate biomass) providing food for invertebrates and improving substrate structure, particularly when riverbeds have structurally poor, fine sediments (Sovell et al., 2000; Feld et al., 2018). However, inputs of organic matter and the provisioning of shade are typically greater in wooded buffers (see Section 4.2 below). The mitigating effects of buffer strips can, however, be overwhelmed by land use intensity at the catchment level, particularly when buffers are narrow, or discontinous, or water quality is degraded in the upper reaches of the watershed (Leps et al., 2015). In intensive agricultural catchments, extensive and/or targetted riparian restoration is recommended to restore sensitive freshwater communities (Wahl et al., 2013)

3.3. *Additional benefits provided by vegetated buffer strips*

Riverbanks provide habitat for the intermediate host (mud snail: *Galba truncatula*) of liver fluke (*Fasciola hepatica*), a disease with a significant negative economic impact to the livestock industry worldwide. By restricting livestock access to riverbanks, and thus to areas of high fluke density, buffer strips can reduce fluke prevalence (Olsen et al., 2015). The physical exclusion of livestock from watercourses also increases biosecurity by reducing the risk that livestock will come into contact with faecal pathogens such as Campylobacter and Cryptosporidium (Collins et al., 2007). Buffer strips therefore have the potential to reduce risks to not only human health, through protecting bathing and drinking waters, but also livestock health with positive implications for productivity (Collins et al., 2007; Lardner et al., 2005).

Riparian buffer strips provide habitat for beneficial insects such as ground beetles and insect pollinators (Cole et al., 2008, 2015). Supporting such insects can enhance ecosystem services that regulate and support agricultural production (e.g. pollination and pest control), with positive implications to yield (Pywell et al., 2015). Buffers straighten the often convoluted field edges that are typically found adjacent to riverbanks and, thereby, facilitate field management practices. There is, however, the perception that unmanaged field margins act as a source of economically damaging pests, weeds and diseases (Deschênes et al., 2003; Thomas et al., 2002; Buckley et al., 2012). Data on noxious weeds in riparian buffer strips was lacking, however, findings from non-riparian field margins indicate weed prevalence decreases with increasing margin width (Fritch et al., 2011).

4. Wooded buffer strips

4.1. *The impact of wooded buffer strips on diffuse pollution*

Planting trees in riparian buffer strips stabilises river banks, reducing erosion and sediment loss (Rood et al., 2014). Below ground, their deep roots provide strength to the soil matrix and increase the complexity, porosity and permeability of soils which increases infiltration and water retention (Dixon et al., 2015; Dosskey et al., 2010; Roberts et al., 2012; Thomas and Nisbet, 2007). As a result of their deeper roots, trees can recover nutrients leached below the rooting zone of herbaceous plants (Rood et al., 2014).

Above ground, the structural complexity of wooded buffers (i.e. upright trees, ground vegetation, leaf litter and fallen deadwood) slows water flow velocity and theoretical models predict they are more efficient at removing nutrients than vegetated buffers (Zhang et al., 2010). This is, however, dependent on the ground layer structure, velocity of surface run-off and microbial community (Knight et al., 2010). With these attributes varying considerably between locations, the relative efficiency of wooded and vegetated buffers at intercepting nutrients is context and pollutant specific (Sabater et al., 2003). Osborne and Kovacic (1993) found that wooded buffers were more efficient than vegetated buffers at reducing nitrate-nitrogen in shallow ground water but less efficient at reducing phosphates. Higher concentrations of organic matter in forested buffer strips can increase microbial activity resulting in increased denitrification rates (Osborne and Kovacic, 1993), but can also remobilise particulate phosphate to a more soluble form, potentially increasing phosphorus concentrations in overland and subsurface flow (Roberts et al., 2012). Ground vegetation in wooded buffers is frequently less densethan grassy buffers, resulting in grassy buffers having a greater hydraulic roughness. This makes grassy buffers more effective at slowing the flow of water, increasing their ability to intercept sediments, nutrients and other contaminants (Erktan et al., 2013; Knight et al., 2010; Lin et al., 2011; Osborne and Kovacic, 1993). When establishing wooded buffers, tree planting density should therefore be kept sufficiently low to allow the establishment of ground storey vegetation thus increasing hydraulic roughness and effectiveness (Dosskey et al., 2010; Fortier et al., 2011; Knight et al., 2010).

Tree age, species and management will influence the ability of wooded buffer strips to intercept pollutants. Fast growing trees, such as poplar (*Populus* spp.) and willow (*Salix* spp.), have a strong ability to uptake a range of pollutants, immobilising them in woody parts of the tree (Christen and Dalgaard, 2013). *Salix* spp. are particularly deep-rooted and are thus effective at stabilising river banks, reducing flood risk and recovering leached nutrients (Rood et al., 2014). Wooded buffers, however, take longer to establish than vegetated buffer strips and ground disturbance during planting and harvest can result in watercourse contamination (Forestry Commission, 2011).

4.2. The impact of wooded buffer strips on biodiversity

Riparian woodlands can have positive implications for terrestrial biodiversity, creating habitat for woodland specialists and enhancing woodland connectivity, moreover, trees provide shade and shelter, buffering climatic extremes (Boutin et al., 2003; Maisonneuve and Rioux, 2001; Paine and Ribic, 2002). Soil biodiversity is promoted by the presence of tree roots, with root inputs playing an important role in shaping soil food webs (Keith et al., 2009). When compared to vegetated buffers, wooded buffers support more native plant species (Paine and Ribic, 2002), and provide undisturbed habitats favouring woodland beetles (Stockan et al., 2014) and insectivorous mammals (Maisonneuve and Rioux, 2001). Wooded riparian habitats also provide better foraging habitat for bats, potentially due to higher invertebrate abundance or greater safety from predators (Scott et al., 2010; Todd and Williamson, 2019).

Early flowering trees, such as *Salix* spp., provide important early season food resources for pollinators. However, excessive shading decreases flowering plants at ground level, resulting in a lack of forage for insect pollinators when compared to vegetated buffer strips (Cole et al., 2017; Fortier et al., 2011; Stockan et al., 2012) which provide particularly profitable foraging habitats (Cole et al., 2017, 2015). Furthermore, the adult stages of many riverine invertebrates (e.g. damselflies and dragonflies) prefer open unshaded habitats and consequently their dispersal could be adversely influenced by afforestation (Petersen et al., 2004; Rouquette and Thompson, 2005).

Tree planting can influence aquatic organisms through a variety of mechanisms. By stabilising river banks, sediment entering the watercourse is reduced resulting in richer communities of benthic macroinvertebrates (Larsen et al., 2009). Woody debris entering the watercourse increases the diversity of hydraulic habitats and creates substrate and refuge with tree roots providing shelter for a range of aquatic organisms (Gilvear et al., 2013; Sovell et al., 2000; Thomas et al., 2016). Riparian woods, provide organic inputs (e.g. invertebrates, wood and leaves) to freshwater systems increasing the abundance and conservation value of aquatic macroinvertebrate (Malmqvist, 2002; Poole et al., 2013) and contributing to fish diets (Nakano and Murakami, 2001). The greatest benefits are derived from deciduous woodlands which support higher aquatic invertebrate biomass than coniferous plantations (Thomas et al., 2016).

Being strongly linked to both the hydrological cycle and atmospheric thermal regimes, riverine ecosystems are thought to be particularly vulnerable to climate change (Ormerod, 2009). Water temperatures influence growth rate, emergence, life history traits and community structure in a range of aquatic organisms (Broadmeadow et al., 2011; Durance and Ormerod, 2007; Thomas et al., 2016) and high water temperatures can reduce the dissolved oxygen content of streams creating hypoxic conditions for aquatic organisms (Null et al., 2017). Increased air temperatures are warming rivers (Kaushal et al., 2010; Pohle et al., 2019) and with climate change predictions indicating more frequent high-temperature extremes (Fischer and Knutti, 2015), buffer strips can play a vital role in mitigating the impacts on aquatic ecosystems. Wooded buffer strips not only improve water quality (thereby reducing additional stressors that can exacerbate the impacts of climate change), but, in providing shade, moderate diel fluctuations in stream water temperatures (Broadmeadow et al., 2011; Thomas et al., 2016).

As a result of the taller vegetation, wooded buffers are typically more effective at shading wider streams (e.g. over 6 m wide) than vegetated buffers which will only effectively shade narrow streams (i.e. < 2 m) (Davies-Colley and Quinn, 1998). Afforestation of buffers could therefore help build resilience to climate change in freshwater systems. The effectiveness of wooded buffers at stabilising water temperatures will depend on local conditions. Spatially targeting riparian tree planting to north-south facing river channels will maximise watercourse shading, and selecting channel sections with slower flow rates will increase the chance of cooling (Jackson et al., 2017). Excessive shade, however, can prevent streams from reaching optimum temperatures for brown trout growth (i.e. circa 80 % shade Broadmeadow et al., 2011), and a lack of light can reduce the growth rate and diversity of aquatic plants (i.e. > 60 % shade: Jusik and Staniszewski, 2019). There is, however, evidence that under high levels of shade (i.e. ≥ 79 %), native species are favoured over non-native species, indicating that shading may decrease the susceptibility of streams to invasive species (Kankanamge et al., 2019).

Clearly the biodiversity benefits derived from the afforestation of riparian buffer strip are dependent on woodland type, density and spatial configuration within the landscape, alongside target taxa for conservation. Greatest benefits are likely to be achieved by targeting afforestation to provide a diversity of open, forested and vegetated riparian buffer strips to retain moderate levels of shade (e.g. 20–40 %) for aquatic organisms (Broadmeadow et al., 2011). This will not only restore terrestrial habitat complexity, but also enable aquatic organisms to reduce thermal stress by moving between shaded and unshaded river stretches.

4.3. Additional benefits provided by wooded buffer strips

Through physically excluding livestock, wooded buffers will have similar benefits with respect to reducing the prevalence of liver fluke and biosecurity, positively impacting on livestock welfare and production. Planting trees and shrubs in buffer strips also provides livestock with shelter from adverse weather (e.g. sun, rain and wind), alleviating temperature stress, and further benefitting production and welfare (Broom et al., 2013; Maseyk et al., 2017; Renaudeau et al., 2012). With livestock often entering streams to cool down, providing shade is particularly important when access to watercourses is restricted.

The high soil moisture content and nutrient-rich status of riparian zones provide excellent conditions for tree growth and, consequently, wooded buffers can be extremely effective at carbon sequestration, particularly in the early stages of development (Borin et al., 2010; Udawatta and Jose, 2012). Wooded buffers also have a considerable capacity to store carbon (i.e. estimated at 80 t ha^{-1} $year^{-1}$) both as above and below ground biomass (Borin et al., 2010; Udawatta and Jose, 2012). If harvested for material biomass (e.g. timber and biofuel production), trees provide an additional source of income increasing the financial viability of wider buffer strips, particularly through managing high yielding trees (e.g. poplar and willow) as short rotational coppice (Borin et al., 2010; Christen and Dalgaard, 2013). In addition to being economically profitable, fast growing, high yielding trees, are particularly effective at sequestering carbon and in the uptake and immobilisation of pollutants (Christen and Dalgaard, 2013). Trees can, however, compete with pasture/crop for light, negatively impacting on production (Broom et al., 2013; Osborne and Kovacic, 1993).

Wooded buffers can contribute to natural flood management, slowing the flow of water from the land into receiving water bodies by increasing infiltration, reducing flow velocity and providing temporary water storage (Dixon et al., 2015; Tabacchi et al., 2000). The effectiveness of wooded buffers at reducing flow velocity depends on hydraulic roughness and thus on the structural composition (i.e. density, stiffness) of both living and dead material at ground level (Tabacchi et al., 2000). Densely planted, homogenous forested buffers, with poor understory development (e.g. poplar or sitka spruce plantations), are likely to be less effective at reducing surface flow than forested buffers with a well-developed herbaceous layer, or indeed grassy buffers. During the growing season, deciduous trees have a high evapotranspiration rate increasing the potential for water uptake and storage (Bracken and Croke, 2007). Deep-rooted species are associated with more porous soils increasing surface water infiltration (Tabacchi et al., 2000). Forested buffers introduce woody debris into watercourses, influencing stream morphology and dynamics with larger debris causing the formation of in stream pools that regulate channel flow and mitigate the impacts of flooding (Thomas and Nisbet, 2012; Dixon et al., 2015).

Riparian woodlands appear to be particularly valuable to flood defence under moderate flooding, however, effectiveness is dependent not only on the extent and location of the woodland, but also on age and density, with mature woodlands being more effective due to their increased structural complexity and greater volume of deadwood entering watercourses (Dixon et al., 2015). To optimise natural flood defence, forested buffer strips should be spatially targetted at the sub-catchment scale (i.e. 20–40 % of the total catchment) to de-sychronize pulses in flood waves between sub-catchments, reducing peak flow at the catchment level.

5. Zoned buffer strips

The differential uptake and retention of contaminants by forested and vegetated buffers indicates that zoned buffer strips, consisting of a strip of trees and a strip of grassy or herbaceous vegetation, are likely to be best at mitigating diffuse pollution (Knight et al., 2010, but see Valkama et al., 2019). Distinct zones of grass and trees are likely to outperform a buffer where the two are incorporated as a mixture (Erktan et al., 2013). Correll (2005) recommends three distinct zones of vegetation:

- Zone 1: A narrow strip of permanent native trees immediately adjacent to the watercourse to stabilise riverbanks, provide inputs to freshwater systems and stabilise water temperatures.
- Zone 2: A wider zone of native tree species that may be cropped for additional income. This zone will help reduce nitrates in ground water and reduce the acidity of ground water.
- Zone 3: A narrow zone of dense grass, or functionally similar vegetation, to trap sediments and sediment bound pollutants prior to them entering the wooded zone. This zone will assimilate nutrients and bind dissolved pesticides.

In productive agricultural catchments, or in smaller farms, zoned buffers may not be economically viable due to the amount of land taken out of production. Economic viability could be increased by harvesting the tree zone (e.g. for short rotational coppice); whilst ensuring that the permanent tree zone immediately adjacent to the watercourse is retained to prevent sedimentation following disturbance during harvest (Correll, 2005). Production benefits may also be increased by developing multi-functional seed mixes for the vegetated zone to enhance natural enemies and/or insect pollinators, in addition to mitigating diffuse pollution. However, where zoned buffer strips are not economically viable (e.g. productive landscapes with high densities of waterbodies), the selection of vegetated and/or forested buffer strips should be dependent on the water-quality issue at the catchment level, the relative risk of different pollutants at the field scale and other ecosystem services these buffers can deliver.

6. The fate of contaminants entering buffer strips

Agricultural contaminants (e.g. nitrates, phosphates, antibiotics and pesticides) are removed from buffer strips primarily via microbial mineralisation, biological assimilation and, in the case of nitrogen, denitrification. The microbial community strongly influences the rate that nutrients, pesticides and veterinary antibiotics are degraded and/or mineralised to biologically available forms (Chu et al., 2010; Krutz et al., 2006; Lin et al., 2011). Furthermore, conditions in buffer strips are typically ideal for microbial denitrification (i.e. damp, anaerobic and rich in organic matter); key to the removal of nitrogen as gaseous nitrogen (N_2). The ability of plants to uptake and store pollutants depends on the bioavailability of the contaminant (and thus microbial activity), physical and chemical properties of the soil which can vary with season, and plant age and species, with the greatest assimilation occurring during the growing season and initial establishment (Dosskey et al., 2010; Roberts et al., 2012).

Initially buffer strips can be more efficient at removing phosphates than nitrates because in addition to biological assimilation, dissolved phosphorus adsorbs to the soil particles (Vought et al., 1995). However, under high phosphate loadings, soil adsorption sites become saturated thus limiting the phosphorus retention capacity and decreasing the buffer strip's effectiveness over time. Phosphorus saturated soils pose a particularly high risk to watercourses (Vought et al., 1995; Stutter et al., 2012). Buffer strips also tend to have high microbial respiration rates which increases phosphorus solubilisation, particularly in the surface soils (top 5 cm), resulting in higher concentrations of bioavailable (i.e. dissolved reactive) phosphorus which is immediately available to algae and thus particularly harmful to aquatic environments (Roberts et al., 2012).

The rates that both nitrogen and phosphorus are immobilised and removed via biological processes (i.e. via microbial mineralisation, denitrification and plant assimilation) can be insufficient when loadings are high (Sabater et al., 2003; Stutter et al., 2012; Valkama et al., 2019; Vidon et al., 2010). Consequently, nutrient saturated buffer strips can act as a nutrient source rather than sink, posing particularly high risks to aquatic environments during periods of high flow, ice thaw and bank erosion (Dosskey et al., 2010; Vidon et al., 2010). Furthermore, decaying vegetation can produce considerable amounts of bioavailable phosphorus in springtime run off (Uusi-Kämppäet al., 2012).

Under nitrate saturated conditions, denitrification results in heightened levels of the greenhouse gas nitrous oxide being released (N_2O) (Hefting et al., 2003). Saturated buffers are therefore not only a threat to water quality, but can also conflict with climate change objectives. Such pollutant swapping remains largely unconsidered and could be particularly problematic where diffuse pollution mitigation targets the widespread fencing of watercourses (Kay et al., 2007).

To minimise risks associated with saturation, buffer strips should not be viewed as an "end-of-pipe" solution, but rather as a way to modify the transfer and conversion of pollutants with respect to extent, timing and chemical form (Roberts et al., 2012). To reduce the risk of saturation, it is important to control pollutants at their source (e.g. through nutrient budgeting and integrated pest management) and explore management options to prevent saturation.

7. Managing riparian buffer strips to retain long-term effectiveness

Management aimed at increasing the longevity of buffer strips should consider reducing contaminants entering the buffer strip and also the role that vegetation (e.g. species, age and season dynamics in growth), soil (e.g. type, moisture, organic content and temperature) and microbial (soil health and microbial activity) properties play in retaining, converting, storing and removing contaminants (Chu et al., 2010; Lin et al., 2011; Sabater et al., 2003; Vought et al., 1995).

The capacity of buffer strips to retain phosphorus can be increased through amendments that promote precipitation, or sorption, of phosphorus, thereby reducing its solubility (Kirkkala et al., 2012; Uusi-Kämppäet al., 2012). Lime–sand filters increase pH, and thus precipitation of soluble phosphorus and most metals, reducing total and bioavailable phosphorus and metals entering watercourses

(Kirkkala et al., 2012). These filters, however, became less effective overtime, potentially due to sediment clogging. Amending soils with iron-rich compounds that have a high affinity with phosphorus (i.e. ferric sulfate and Fe–gypsum) were again effective at reducing total and bioavailable phosphorus in runoff (Uusi-Kämppäet al., 2012). Both lime-sand filters and ferric sulfate alter the pH of runoff and thus can pose a threat to aquatic organisms, particularly when runoff enters the waterbody directly (Kirkkala et al., 2012; Uusi-Kämppä et al., 2012).

With nutrients building up in the upper layer of the soil (Roberts et al., 2012), mechanical removal of saturated soils could prolong buffer strip efficiency. Exposing bare ground, and the loss of vegetation, would however result in an increase in pollutants entering the watercourse and consequently it is critical that such practices are a last resort and timed to coincide with low risk periods. Alternatively, establishing soil bunds (Adimassu et al., 2014), or sediment fences (Vinten et al., 2014), on the edge of buffer strips, could help trap nutrients before they enter the buffer. In narrow, or steeply sloped, buffers, such options are more practical than the mechanical removal of soil or mowing (see below) due to the difficulty in manoeuvring machinery. Spreading the saturated soil (i.e. from bunds or soil removal) on the field would increase resource efficiency which is particularly important in light of phosphorus scarcity.

In vegetated buffer strips, annual mowing, with the offtake of cuttings, removes accumulated nutrients from the buffer strip and can reduce the risk of nutrient influxes due to decaying vegetation (Hefting et al., 2005; Kiedrzyńska et al., 2008; Uusi-Kämppä et al., 2012; Vought et al., 1995). Reducing the nutrient status of buffers in this way can help maintain nutrient uptake and is particularly effective in reducing phosphorus saturation (Stutter et al., 2012). Introducing machinery to buffer strips can, however, increase soil compaction and reduce soil porosity and organic debris, adversely influencing microbial activity and thus the rate contaminants are mobilised into biologically available forms and removed via microbial processes.

Annual cutting can help prevent scrub encroachment and control pernicious weeds (e.g. *Senecio vulgaris* L. and *Galium aparine* L.). With landowner concerns that buffer strips act as a source of weeds, permitting mowing may make this option more desirable (Buckley et al., 2012). Removing vegetation also opens up the vegetation structure, facilitating the natural regeneration of flowering plants (Fritch et al., 2011; Hille et al., 2018; Schippers and Joenje, 2001; Westbury et al., 2008). Reducing the soil nutrient status encourages the formation of structurally complex grass tussocks benefitting microbial biomass (Cooper et al., 1995) and providing overwintering habitat for predatory arthropods (Pfiffner and Luka, 2000; Woodcock et al., 2007). Mowing may, however, adversely affect rare arable weeds of conservation interest (Westbury et al., 2008), and can decrease the abundance of invertebrate prey for foliage gleaning birds (Perkins et al., 2002). Lower prey abundance does not, however, necessarily equate to poorer foraging grounds as tall, dense vegetation impedes the accessibility and detectability of prey (Douglas et al., 2009; Vickery et al., 2001) and increases predation risk (Morris and Gilroy, 2008; Schneider et al., 2012). On the whole, mowing is perceived to be beneficial to farmland birds (Douglas et al., 2009; Vickery et al., 2001).

Annual spring or autumn mowing tends to be more favourable than summer mowing for many terrestrial invertebrates (Bell et al., 2002; Vickery et al., 2001). Spring cutting, however, is deleterious to ground-nesting birds (Vickery et al., 2001) and may result in the loss of early season forage for insect pollinators (Cole et al., 2015). Autumn cutting may therefore favour a wider suite of species. The impact of mowing on diffuse pollution mitigation must, however, be considered. Removing vegetation in autumn could decrease the risk of nutrient influxes that can accompany decaying vegetation in winter (Hefting et al., 2005). While this highlights potential synergies between biodiversity and water quality goals, it is also important to consider the role that buffers play in trapping sediments following ploughing. Mowing reduces the density and structural complexity of vegetation, thereby decreasing the buffer strips efficiency, and consequently should not be implemented at a time when adjacent land is being cultivated. Autumn mowing would therefore be favourable from both a water quality and biodiversity perspective in spring sown cereals. In autumn sown cereals, however, spring would be the optimal time for mowing, indicating a conflict between biodiversity and water quality objectives.

In arable fields where unfenced buffer strips are common, mowing is easily implemented. In grassland situations buffer strips are typically fenced, and due to the difficulties in manoeuvring machinery in narrowly confined steeply sloped areas, grazing is often more cost effective and viable. Grazing intensity, timing and livestock species influence biodiversity benefits with optimum management being taxa specific (Humphrey and Patterson, 2000; Vickery et al., 2001; Woodcock et al., 2009). Grazing can help retain botanical diversity in field margins favouring insect pollinators (Carvell, 2002; Fritch et al., 2011), but can reduce the structural diversity of vegetation negatively influencing phytophagous insects, their predators and parasitoids (Woodcock et al., 2009; Ó hUallacháin et al., 2014). Grazing also decreases habitat stability, adversely affecting sedentary species (Cole et al., 2012b) and can result in soil compaction and poaching, albeit to a lesser extent than mowing. Furthermore, allowing livestock access to the buffer strip, and hence also the watercourse, directly conflicts with diffuse pollution mitigation (Stutter et al., 2012). To balance biodiversity and diffuse pollution benefits, grazing prescriptions for buffer strips should be extensive (with respect to both livestock density and time frame) and conducted late in the season when the majority of flowers have set seed. Additionally, in bathing water catchments grazing should only permitted outside of the bathing season to reduce risk to human health (McCracken et al., 2012).

Extrapolating best practice from non-riparian field margins indicates that establishing wild flower or bird seed mixtures in riparian buffer strips could increase their biodiversity value (Fritch et al., 2011; Holland et al., 2015; Vickery et al., 2002). Cultivation adjacent to watercourses can, however, destabilise riverbanks and result in siltation and, consequently, environmental legislation often prohibits deep cultivation in riparian zones. Furthermore, as watercourses can facilitate the spread of non-native species (Correll, 2005), seed mixtures should only contain species present in the local area. Given these constraints, management approaches that encourage natural regeneration without cultivation (e.g. through restricted grazing or mowing) may often be preferential.

The effectiveness of wooded buffers typically diminishes over time as the nutrient requirements of trees decrease with age (Valkama et al., 2019). Thus, in established wooded buffers there is an increased risk of nutrient saturation under high loadings. A potential solution is to harvest wood for biomass/timber with replanting to maintain growth, nutrient uptake and the buffer strips ability to mitigate diffuse pollution (Sheridan et al., 1999). Short rotational coppice (SRC) where fast growing species are harvested/coppiced regularly (i.e. every 1–5 years for willow and 4–10 years for poplar) could prolong nutrient uptake and provide an economic incentive for landowners to establish wider buffer strips (Christen and Dalgaard, 2013). The regular coppicing/harvesting of SRC will, however, disturb ground cover and expose bare ground temporarily resulting in an increase in pollutants entering the watercourse (Christen and Dalgaard, 2013). To make SRC both financially and environmentally viable it is suggested that SRC should be incorporated within a zonal buffer framework that combines a zone of coppiced trees with an undisturbed woodland zone immediately adjacent to the watercourse to intercept pollutants during harvest (Correll, 2005).

With the risk that over time buffer strips can become a source of pollutants rather than a sink, it is clear that management actions should focus on reducing contaminants entering buffer strips in the first place (e.g. more efficient nutrient budgeting, soil bunds, sediment fences) and biomass removal to prevent saturation. The removal of biomass, and thus nutrients (i.e. via grazing, mowing or tree harvest), is easily implemented and provides a cost-effective means of increasing, or extending, the nutrient retention capacity of buffer strips when compared to alternative methods involving the addition of soil amendment materials (Uusi-Kämppä et al., 2012). The timing and intensity of operations should consider the full range of ecosystem services provided by the buffer strip and the relative importance of these services at the farm and catchment level.

8. Taking a catchment based approach to riparian management

The ecological status of waterbodies is influenced by factors at the catchment scale (Larsen et al., 2009; Malmqvist, 2002; Osborne and Kovacic, 1993) and consequently riparian management should be implemented at this scale. To tackle diffuse pollution, narrow continuous buffers are thought to be more effective than wide intermittent buffers (Correll, 2005; Kay et al., 2007). Continuous buffers are also more effective at stabilising water temperatures and increasing ecological connectivity (Correll, 2005; Fischer and Fischenich, 2000). The widespread adoption of buffer strips at the catchment scale would, however, result in the simplification of riparian habitats which are inherently complex and dynamic in nature (Rood et al., 2014; Malmqvist, 2002). Consequently, the homogenisation of these habitat could conflict with biodiversity goals. Catchment management plans should therefore focus on restoring riverbank heterogeneity by providing a diversity of riparian zones (i.e. including unbuffered riparian zones, vegetated buffers and wooded buffers). Diffuse pollution risks will vary across a catchment in response to topographical, physiological, hydrological (e.g. river size, morphology and flow) and land use/management factors. Buffer strips provide a much wider range of ecosystem services than non-riparian field margins (Fig. 1) and management of such zones require greater consideration to ensure placement, management and structure increase multifunctionality of land taken out of production. Spatially targeting riparian management to take into account a range of ecosystem services, will also prevent the simplification of riparian habitats.

Headwater streams typically not only contribute most of the water entering receiving water bodies, but their large land to water interaction area also increases the risk that they will intake and transport pollutants (Correll, 2005). Furthermore, as contamination upstream impacts on water quality downstream, it's proposed that the greatest benefits at the catchment scale are achieved by protecting headwaters (Correll, 2005). However, due to harsher climatic conditions and geophysical constraints (e.g. shallow soils and steeply sloping land), upland areas are frequently less intensively managed and, as such, have a lower risk of diffuse pollution (Eliasson et al., 2010; Qasim et al., 2013). The spatial targeting of riparian buffers should therefore consider the location of the waterbody in the watershed, in addition to local factors that influence diffuse pollution risk, such as, soil type, slope and surrounding land use and management. Wider buffers should be targeted to high risk areas (i.e. intensively managed farms with steeply sloped banks) leaving low risk areas (e.g. areas of extensively managed upland grazing) unbuffered.

Spatial targeting should capitalise not just on mitigating diffuse pollution (Vidon and Hill, 2006), but also on promoting additional ecosystem services. For example, incorporating nectar and pollen-bearing flowers in buffers adjacent to insect pollinated crops can enhance pollination services and yield in the crop, balancing losses from land taken out of production (Pywell et al., 2015). In intensively grazed fields, establishing wide buffers consisting of a zone of tree species and a strip of grass would not only mitigate diffuse pollution, but also sequester and store carbon, and improve livestock welfare through providing shelter and shade. Furthermore, managing the woodland strip as short rotational coppice could reduce the risk of nutrient saturation, increasing the longevity of the buffer, and provide landowners with an additional source of income increasing economic viability (Christen and Dalgaard, 2013).

The inclusion of permanent riparian woodland will increase the catchments ability to sequester and store carbon, and stabilise water temperatures, mitigating the impacts of climate change (Thomas et al., 2016). Permanent wooded buffers can, however, become less effective over time at mitigating diffuse pollution, particularly when nutrient loadings are high, indicating a potential trade-off between diffuse pollution and climate change objectives (Valkama et al., 2019). Situating permanent riparian woodland adjacent to extensively managed areas with low influxes of pollutants, and spatially targeting to optimise natural flood defence and enhance woodland connectivity, will help to reduce conflicts between different policy areas.

Cost-effectiveness of buffer strips is dependent on the risk to waterbodies, the economic potential of land taken out of production and additional benefits derived, and therefore varies with local and regional site conditions and farming system. With the exception of food provisioning, ecosystem services are typically more efficiently delivered by wider buffers (i.e. over 5 m wide: Fig. 2). Wider buffers, however, take more land out of production, negatively influencing food production, and are thus more costly to sustain, especially in highly productive agricultural land where the greatest benefits can be achieved. To make informed decisions, cost benefit analyses should consider the wider societal (e.g. carbon sequestration and storage, natural flood management and biodiversity) and agronomic (e.g. increased biosecurity, improved livestock health and provisioning of pollination and pest control services) benefits riparian zones provide (Figs. 1 and 2). Furthermore, with the risk of pollutant swapping, the relative importance of different pollutants should also be considered alongside the risk that mitigation measures may decrease specific pollutants while increasing others (e.g. denitrification in riparian buffers reducing nitrates entering watercourses but increasing levels of greenhouse gases: Stevens and Quinton, 2009).

Catchment scale management requires a wide array of actions to be implemented at an appropriate spatial scale and location. To achieve this, we need decision support tools that are underpinned by detailed cost benefit analyses alongside spatial modelling to determine the optimum placement of measures. Robust cost benefit analyses will help explore how configuration and management of buffers influences the wide range of potential benefits and determine where synergies and trade-offs across policy areas occur.

Landscape-scale initiatives require collective decision making and thus co-operation between stakeholders. Stakeholders, however, have different objectives and therefore value benefits differently (Hein et al., 2006). This makes catchment scale initiatives difficult to implement (Adams et al., 2016). To reduce conflict, stakeholders should be engaged from the outset (e.g. in opportunity mapping) and funding is therefore needed to facilitate communication between stakeholders and coordinate action on the ground to achieve multiple benefits from strategically designed riparian areas.

9. Conclusions

Riparian zones are multifunctional and their management requires consideration of the wide range of ecosystem services to seek synergies and identify trade-offs between different policy objectives. Management prescriptions require an integrative approach to spatially target restorative actions to optimise the benefits gained and maintain the diversity of riparian ecosystems. Wood and grass buffer strips offer different services and zoned buffers, with distinct woody and grassy zones, can optimise multifunctionality. As a consequence of the land taken out of production, however, zoned buffers are unlikely to be economically viable in high value agricultural land. To balance agronomic and environmental goals, variable width buffers (i.e. where width changes in relation to site specific risks to the waterbody) and are likely to be perceived more favourably by land managers (Buckley et al., 2012).

Nutrient saturated buffer strips can act as a source of pollutants rather than a sink and can result in nutrient swapping. Buffer strips should therefore be integrated within a wider management framework rather than be viewed as an "end-of-pipe" solution. Management actions should focus on reducing contaminants entering buffer strips in

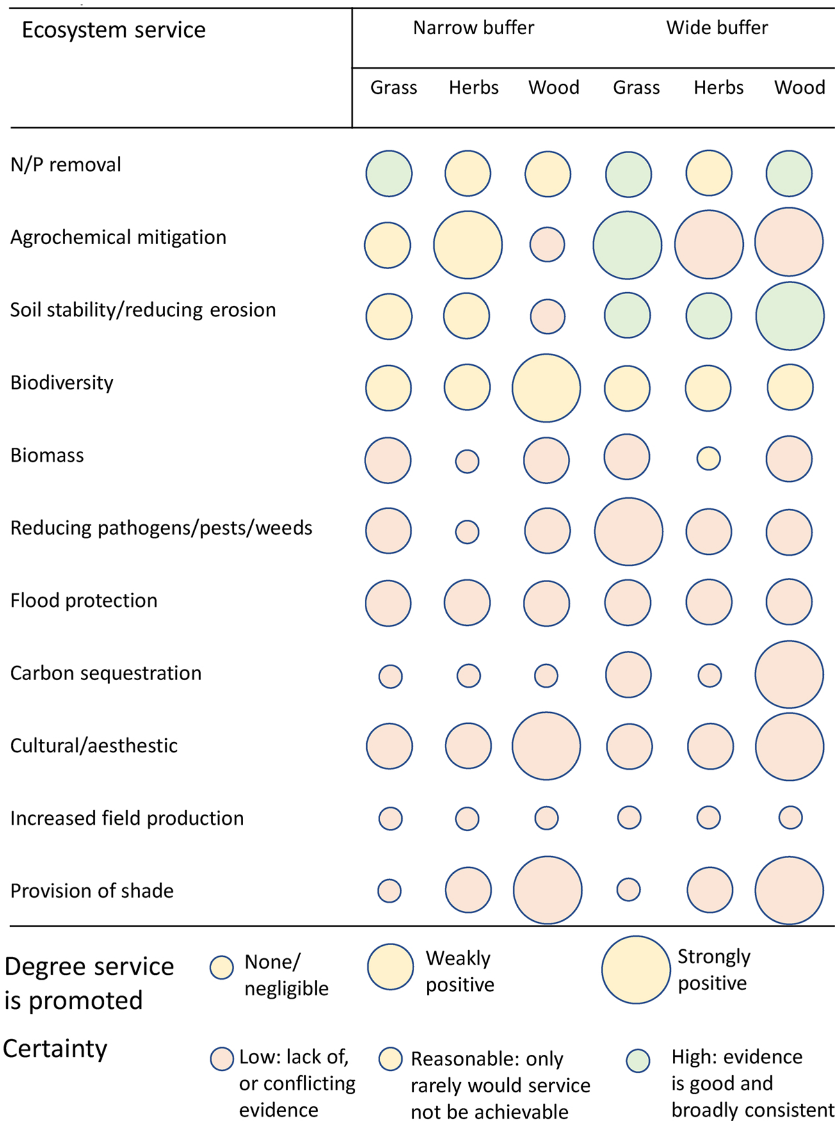

Fig. 2. Synthesis of evidence on the impact of buffer strip width and vegetation type on ecosystem service provisioning. This is based on the cited references including original articles and those cited in review papers. Evidence was scored on a 5-point scale from -2 (strong negative effect) to +2 (strong positive effect) and the mean values calculated. The level of certainty was scored on a 3-point score combining the number of studies and balance of beneficial effects from poor (red) (< 5 studies of which < 75 % show positive effects or < 10 studies of which < 50 % show positive effects) to green (5+ studies all showing positive effects or 5-9 studies of which 75 % show positive effects) (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article).

the first place, for example by reducing agrochemical applications at the source (e.g. through nutrient budgeting), and reducing the risk of saturation (e.g. via restricted grazing, mowing or tree harvest). A catchment scale approach is required to optimise the benefits gained and consequently cooperation between stakeholders is vital. To facilitate decision making, robust decision support tools are needed that are underpinned by comprehensive cost-benefit analyses and spatially modelling.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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