esa ECOSPHERE

FRESHWATER ECOLOGY

The effect of riparian woodland cover on ecosystem service delivery by river floodplains: a scenario assessment

Jan E. Vermaat[1,†] Martin Palt[2] Jeremy Piffady[3] Arturs Putnins[1,4] and Jochem Kail[2]

[1]Faculty of Environmental Sciences and Natural Resource Management, Norwegian University of Life Sciences, Ås 1432 Norway
[2]Department of Aquatic Ecology, University of Duisburg-Essen, Essen 45117 Germany
[3]INRAE, UR Riverly, EcoFlows, Villeurbanne F-69625 France

Citation: Vermaat, J. E., M. Palt, J. Piffady, A. Putnins, and J. Kail. 2021. The effect of riparian woodland cover on ecosystem service delivery by river floodplains: a scenario assessment. Ecosphere 12(8):e03716. 10.1002/ecs2.3716

Abstract. Sixteen ecosystem services were quantified for the riverine landscapes of the Nahe, Stever (Germany), Bresse plain, and Azergues (France), to assess the effects of riparian woodland cover. Future woodland cover in 2050 was modeled to reflect contrasting scenarios of river management aligned to the well-established shared socioeconomic pathways. The scenarios are labeled as current, pessimistic, best practice, and ambitious riparian management practices (RMPs). We linked services to floodplain land use and river morphology and quantified them separately for spatial segments (0.5–1 km in length, n = 118–3419, depending on river length), using an analytical framework, the "Mononen cascade." Conservative monetary value estimates were based on net producer income before tax and subsidy, a shadow market price for carbon, flood damage functions, or willingness to pay for recreation and non-use. Most services were linked to land use, some affected the value of other services through simple rules (woodland shade affected trout survival hence angling benefit, a minimum of woodland affected pest regulation, hence crop productivity). In the current landscape state, provisioning, regulating, and cultural services all showed optimum curves with woodland cover: Provisioning services and cultural services were maximal around 45%, whereas this was around 30% for regulating services. More woodland was present in steeper near-source segments. Averaged across rivers, mean total service provision was estimated at 1084 ± 4 €·ha^{-1}·yr^{-1}, with 40%, 36%, and 24% contributed by, respectively, provisioning, regulating, and cultural services. The three scenarios led to a limited change in total ecosystem service delivery, even if mean woodland cover was reduced from 27% to 17% in the pessimistic RMP and increased to 70% in the ambitious RMP for the most extreme case of the Stever. Provisioning services declined with increased woodland cover and cultural services increased. Regulating services did not change that much, because they are dominated by flood prevention in our assessment. The "best practice" scenario appeared to combine a modest increase in cultural services with a slight increase in provisioning service. An ambitious nature conservation objective as in the ambitious RMP appears possible without seriously compromising overall societal benefit.

Key words: ecosystem services cascade; riparian woodland; river restoration; shared socioeconomic pathways; Strahler river order; water quality.

Received 4 August 2020; revised 6 April 2021; accepted 20 April 2021. Corresponding Editor: Whitney S. Beck.

[4]Present address: Latvian State Forests (LVM), Vaiņodes 1 Riga LV-1004 Latvia.
† **E-mail:** jan.vermaat@nmbu.no

Introduction

The presence or absence of riparian woodland is thought to have a major influence on biodiversity and ecosystem functioning of streams and adjacent floodplains (Sweeney and Newbold 2014). Under natural conditions, most European rivers would be accompanied by woodland (Brown et al. 2018; a recent North American example in Whited et al. 2007). The establishment of woodland buffers is generally considered an effective restoration measure (Bernhardt et al. 2005, Stutter et al. 2012). The effect on flood buffering, however, is not straightforward (Leyer et al. 2012), and local conditions may determine a balance between biodiversity benefits and possibly adverse flooding effects upstream. Similar unforeseen trade-offs may occur among other functions as well, which calls for a comprehensive assessment of all possible effects of a measure, such as woodland restoration, across the whole extent of the current or historical floodplain of a river (Tockner et al. 2000).

The ecosystem services approach can be used as an integrating framework for such a comprehensive assessment, as it can link floodplain land use as well as river characteristics (together reflecting the ecosystem) to an exhaustive list of societal benefits (Burkhard et al. 2009, Bateman et al. 2013, Vermaat et al. 2020). A priori, it is important that critical methodological concerns are considered. This implies that the quantified services should be "final," hence directly contribute to human well-being (Boyd and Banzhaf 2007), that double counting is carefully checked, that different underlying assumptions for monetary value estimates or other rankings are understood (Wallace 2007, Bateman et al. 2011, Bouma and van Beukering 2015) and that its anthropocentricity is understood (i.e., "the benefits people obtain from ecosystems" or "nature's contribution to people"; MEA 2005; Braat 2018; Díaz et al. 2018; Kenter 2018).

Variation in woodland cover in the floodplain and the riparian area of rivers can thus be linked to variation in the provision of different services by the river and its floodplain. We used a modification of the ecosystem services "cascade" of Boerema et al. (2017) proposed by Vermaat et al. (2020) for this purpose. The view of ecosystem services as a cascade that flows from an ecosystem with structural components via intermediate functions to a final service that is of benefit to humans and thus can be valued economically is presented originally by Haines-Young and Potschin (2010). Mononen et al. (2016) and Boerema et al. (2017) summarize the debate on how the different elements of such a cascade can be understood. Variants of this "cascade" framework have been applied in decision support for the multiple use of landscapes (Dick et al. 2017), in regional and national assessments of the manifold of ways in which whole landscapes contribute to human well-being through ecosystem services (Martín-López et al. 2012, Mononen et al. 2016, Maseyk et al. 2018), and in integrated assessments of ecosystem restoration success (Vermaat et al. 2016).

In Europe, floodplain woodland cover varies substantially among and within river; it is often highest in the upper reaches, although this is under strong control of geomorphology and land-use patterns (Petts and Foster 1985). To explore the possible interactive effects of a future increase in woodland as a possible consequence of environmental policy or ongoing demographic processes such as the depopulation of the countryside, scenarios can be used. Scenarios are a common tool to systematically study the potential consequences of differences in policy focus and societal development (Lorenzoni et al. 2000; Berkhout et al. 2002, Busch 2006; O'Neill et al. 2017). We used the Shared Socioeconomic Pathway (SSP) scenarios of societal development developed by O'Neill et al. (2017) as they have become widely used benchmarks. The SSPs describe contrasting trajectories of societal change in terms of demography, economic development, technological advances, and national and global policy focus on issues of international cooperation, sustainability, and climate change. These SSPs have been used for projections of future land use, world energy markets, and climate modeling (O'Neill et al. 2017). We downscaled these SSP scenarios into a set of specific riparian management practices (RMPs) that describe measures taken by river management expressed as changes in floodplain land use and other river characteristics. Their effect was assessed with our ecosystem services assessment framework. Since the framework allows the tracing of separate services, identifying trade-offs among services (Martín-López et al. 2012) as a consequence of different scenarios is possible.

The underlying generic assumption would be that river restoration with increased woodland has an overall positive ecological effect, also measurable in ecosystem service provision, although the objective of a restoration effort is often implicit and inarticulate (Bernhardt et al. 2005, Jähnig et al. 2011). Gilvear et al. (2013) proposed that river restoration would generally lead to a decrease in provisioning services, whereas regulating and cultural services would increase. Increasing woodland cover can be seen as a form of restoration and then should lead to the same general pattern. However, different scenarios that involve substantial variation in the cover of riparian woodland may have opposing effects: More woodland will be negative for agricultural productivity in the floodplain but positive for in-stream water temperature mitigation and hence trout survival, as one trade-off (Broadmeadow et al. 2011). Since several such potential trade-offs may occur, phrasing a zero hypothesis is not straightforward, and we therefore chose to phrase more open questions:

1. What is the effect of riparian woodland cover on the suite of ecosystem services provided by rivers and their floodplains? Can we identify systematic patterns?
2. How do different riparian management practices (RMPs, linked to SSPs) perform in terms of ecosystem service delivery?
3. Can we generalize on trade-offs among different services that occur as a function of variable riparian woodland cover? Does the effect of increased woodland cover follow the prediction by Gilvear et al. (2013)?

Materials and Methods

Case study rivers

We selected two lower-mountain and two lowland river systems in, respectively, Germany and France: The Nahe, Stever, Azergues, and the Bresse plain. The latter actually combines three smaller rivers in a homogeneous landscape, and its riparian network has been studied by Van Looy et al. (2017). These mid-sized rivers are part of the drainage network of, respectively, the Rhine and Rhone. They differ among others in slope, underlying geology, land cover pattern, human population density, and intensity of agricultural land use (Table 1) and thus are considered to reflect a variety of riverine landscapes in the Northwest of Central Europe, with the lowland Stever and Bresse being under the most intensive agriculture.

Ecosystem service assessment framework

We used the ecosystem services framework that Vermaat et al. (2020) adopted from Mononen et al. (2016) and Boerema et al. (2017) and labeled the "Mononen cascade." Briefly, it is based on the ecosystem services classification CICES 5.1 and specifies the subsequent steps in the cascade for each service linking these to land cover and river morphology (Fig. 1). It uses the three MEA (2005) categories of provisioning, regulating, and cultural services to group the different final services. Each service is quantified in terms of biophysical units (benefit sensu Mononen et al. 2016; such as $kg \cdot ha^{-1} \cdot yr^{-1}$) and subsequent monetary units (societal benefit sensu Mononen et al. 2016, $€ \cdot ha^{-1} \cdot yr^{-1}$), which can be summed as an estimate of total economic value (TEV; 1 € ≈ 1.25 US$ median midmarket 2011–2021). The "Mononen cascade" originally consists of four elements: ecosystem structure, ecosystem function, societal benefit, and societal value. We use land-use cover as proxy for ecosystem structure. Then, we merge the two steps function and benefit into one element, the service in biophysical units since all underlying ecosystem functions that potentially contribute to a final service can be seen as intermediate and the final function is thus also the final service (as in Boerema et al. 2017 and Vermaat et al. 2020). Finally, we use a range of valuation methods from environmental economics (Brander et al. 2006, Bouma and van Beukering 2015) to arrive at a monetary estimate of societal "value," the third element in our adapted cascade. We want to stress that we use such monetary value estimates for final services and an aggregation of these into an estimate of TEV (as a rate per area and year) as a tangible indicator for comparative use in scenario evaluations and in communication with policymakers. They should not be interpreted as directly convertible to market prices or absolute "values."

Table 1. Characteristics of the four study rivers.

Characteristics	Nahe	Stever	Azergues (including Brevenne and Turdine)	Bresse (combines Chalaronne, Veyle, Reyssouze)
Drains into	Rhine	Lippe	Saone	Saone
Segment slope (%)†	3.66 ± 3.72 (0–35.83)	0.46 ± 0.59 (0–6.09)	4.84 ± 3.33 (0.25–15.03)	0.59 ± 0.53 (0.03–3.70)
(Sub-)segment width (m)‡	101.1 ± 79.0	90.1 ± 94.0	149.7 ± 142.7	112.8 ± 126.4
River length quantified (km)	3303	942	424	663
No. segments (sub-segments)	3499 (5638)	445 (1696)	119	171
Percentage woodland§ in floodplain (current)	39	27	36	19
Percentage agriculture in floodplain (pasture and cropland, current)	40	63	37	65
Percentage built-up in floodplain (current)	12	4	18	8
Nitrogen surplus (kg N·ha^{-1}·yr^{-1}, ˜2000–2010)¶	30	120	30	40
Human population density in catchment (No./km^2 ˜2010)#	170	194	131	147

† Values are expressed as mean ± standard deviation (SD) with range in parentheses.
‡ Values are expressed as mean ± SD.
§ Percentages land cover are averages across the (sub)segments.
¶ Based on Grizetti et al. (2007) and Poisvert et al. (2017).
# From regional statistics.

For all 16 quantified services, the assumptions and data sources are summarized in Table 2. A worked-out spreadsheet including all steps in the cascade is available as Data S1.

Deriving riparian management practices from benchmark shared socioeconomic pathway scenarios of societal change

We use a set of scenarios of societal change that have been derived from the benchmark SSPs (O'Neill et al. 2017) and were articulated for our specific purpose to reflect plausible, contrasting trajectories of riparian management in Europe (our RMPs). This articulation is documented separately in Vermaat et al. (2018). We set the starting year or baseline at approximately 2015 and label it as "current." Our chosen horizon in the future is 2050, as a compromise between a relevant time span for current policymakers and the time needed for policy to be fully implemented in landscapes. Furthermore, it is likely that by that time the trajectories of geophysical climate change grasped with the different Representative Concentration Pathways will not yet be markedly different beyond the projected uncertainty bands (IPCC 2014). This allows us to focus on the societal aspects of plausible futures described in the SSPs which simplifies the number of alternatives to be compared. It also excludes the possibly confounding uncertainty in, for example, the future hydrology of our study streams. As a consequence, our estimate of flood damage prevention is based on current flow regimes, which may well be a conservative underestimate.

We selected three out of the five SSPs (respectively SSP1, SSP2, and SSP3, also labeled "sustainability," "middle of the road," and "regional rivalry" in the literature; O'Neill et al. 2017; Popp et al. 2017). We downscaled these to reflect three different, contrasting overall pathways of change in society which then led us to three corresponding plausible ways in which European riparian management would develop: either with a stronger focus on environmental sustainability, or continuing along current lines, or moving away from and ignoring environmental concerns (Riparian Management Practices or RMPs labeled as "ambitious," "best practice," or "pessimistic"; Vermaat et al. 2018; characteristics in Table 3). Current inland water management in

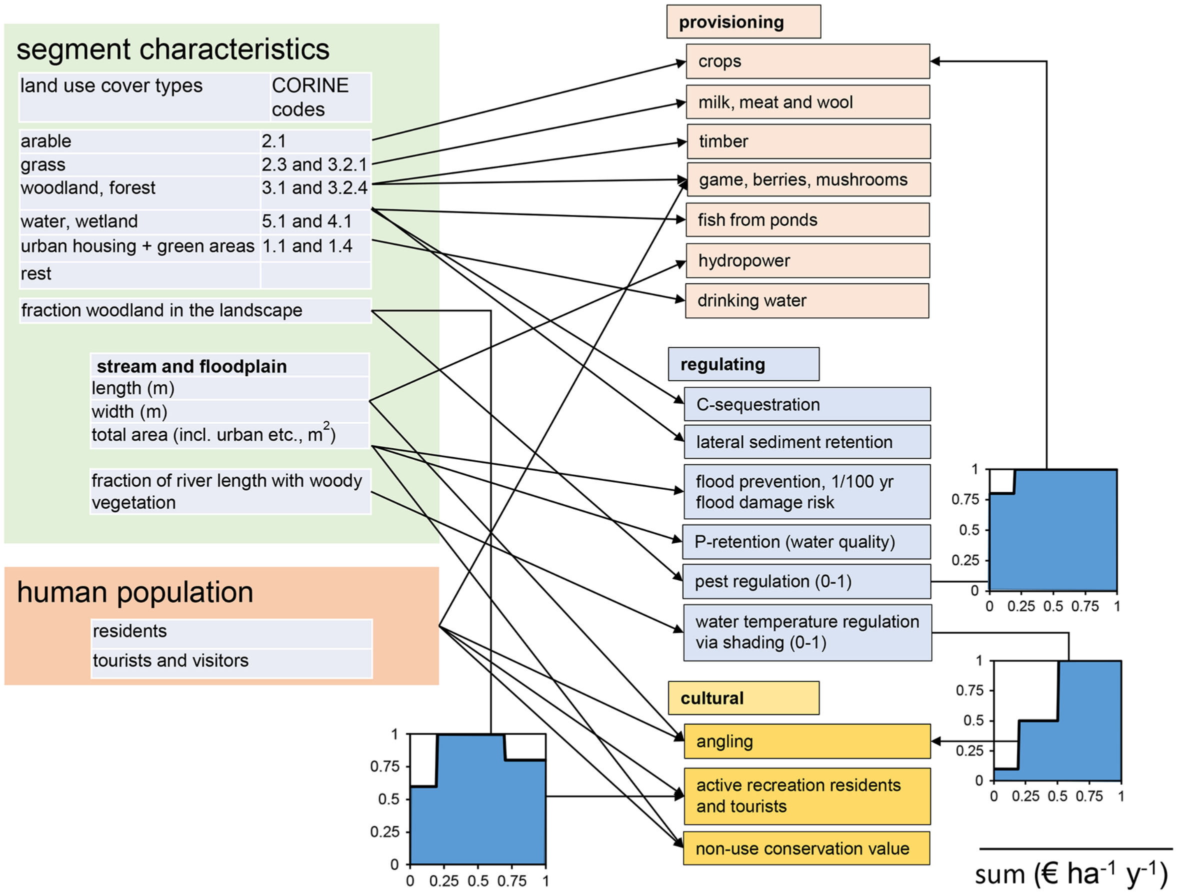

Fig. 1. Flow of 16 different ecosystem services from ecosystem structure (expressed as different types of land use, floodplain, and river metrics in the green box) to annual service flow in biophysical benefit and monetary value estimates. The elements benefit and value in the "Mononen cascade" are pooled here for simplicity. The box "riparian human population" provides population estimates from riparian municipalities through which the valley runs, used for the estimation of a number of services. Provisioning, regulating, and cultural services are indicated with different colors. Three small "knowledge rule" step diagrams indicate the effect of intermediate services, respectively, that of woodland cover on pest regulation expressed in crop revenue, that of riparian woodland cover on stream temperature, brown trout survival and hence value for angling, and that of landscape heterogeneity as the percentage woodland on attractiveness for recreation. Axis units of these diagrams are dimensionless fractions. Further explanation in Table 2.

the European Union is governed by the Water Framework Directive (WFD), and we aligned our RMPs with the currently known policy cycle and measures of this WFD (White and Howe 2003). Our first draft RMPs have been discussed with panels of institutional stakeholders for each of the four study river systems and adjusted when necessary. For the Nahe, we met on 20 November 2017 at the office of the federal state government of Rhineland-Palatinate in Mainz with eight participants; for the Stever, we met on 2 June 2018 in the office of the district government in Münster with 10 participants; and for the Bresse and Azergues, this occurred at IRSTEA in Lyon on 5 July 2018, with 13 participants from both rivers. In each workshop, at least two of the authors were present. An additional purpose of the workshops was to be informed of possible local sources of information for the quantification of the 16 ecosystem services.

Table 2. Relevant ecosystem services selected and aggregated when necessary from CICES 5.1 and quantified in the four river systems.

Service (CICES 5.1 codes)	Description	Explanation, sources
Provisioning		
Crops (1.1.1.1)	Net farm gate revenue arable farms (154–1152 €·ha^{-1} [cropland]·yr^{-1})	Income of farmer minus costs, but before taxes and subsidy, a benchmark statistic that is not market consumer price of a product, hence excludes any increases along the value chain. Data are from Mueller and Mueller (2017) from a standard set of representative and intensively monitored farms in Rheinland-Pfalz, Germany; from Boerman et al. (2015) for the Stever and from Agreste (2017) for the two French systems. In the Nahe, vineyards have not been included as they are outside the floodplain
Dairy, meat, hides, wool fleeces (1.1.3.1 and 1.1.3.2)	Net farm gate revenue dairy farms (201–1054 €·ha^{-1} [grassland]·yr^{-1})	Based on the same sources as crops. We have assumed dairy products to be the final service, and not cattle fodder. Sheep stocks reportedly are limited in the study areas and the value is based on a world market estimate per fleece of 19.5 €
Fish from ponds, mainly trout (1.1.4.1)	Gross income minus costs per km stream length (0–14 €·ha^{-1}·yr^{-1})	Several fish farms occur along the Stever and in the Bresse; productivity and net revenue estimated from Hiller and Wichmann (2010); values normalized per area floodplain
Timber (1.1.5.2)	Conservative annualized net present value estimate based on annual beech or fir productivity for Northern and Central Europe (138–218 €·ha^{-1} [forest]·yr^{-1})	We use a conservative low-end value for Germany based on Duncker et al (2012, different scenarios with different rates of interest, range of 0–800 €·ha^{-1}·yr^{-1}), Hastreiter (2017, 130 €·ha^{-1}·yr^{-1}, net revenue small scale forestry) and Boesch et al. (2018, 300 €·ha^{-1}·yr^{-1}). For France, the values were adjusted from Societe Forestiere (2018)
Berries and mushrooms, game (1.1.5.1 and 1.1.6.1)	Conservative estimate from a comparative European review, mainly Germany and France (12–24 €·ha^{-1} [woodland]·yr^{-1}, 90% due to game)	French and German data adopted from Schulp et al. (2014), which has a similar estimate as Boesch et al. (2018) for Germany
Hydropower (4.2.1.3)	Reported current locally generated hydropower (0–11 €·ha^{-1} [floodplain]·yr^{-1})	Values are normalized from length of 3rd-order streams to floodplain area. Consumer price is halved to reduce the benefits accumulating in the value chain and remain comparable with net farm gate revenues as for crops and dairy. Based on Anderer et al (2009) for the Nahe, on LANUV (2017) for the Stever, and stakeholder reporting on the Bresse. The Azergues currently has no hydropower generation
Drinking water (4.2.1.1)	Reported local extraction and use of surface water (0–5 €·ha^{-1} [floodplain]·yr^{-1})	This can be river water infiltrated into aquifers and then extracted again, or direct use. Market price is halved to reduce the benefits accumulated in the value chain and remain comparable to net farm gate revenue. Values are normalized to floodplain area. In the Nahe and the Bresse drinking water is mainly extracted from deep aquifers and no river water is used. A substantial fraction (crude estimate 40%) of the Stever flow is infiltrated at Haltern into a sandy aquifer, together with water from the Muehlenbach and natural groundwater recharge, at to produce drinking water for parts of the Ruhrgebiet region (data drinking water company Gelsenwasser AG and information service of Nordrhein-Westfalen www.elwasweb.nrw.de)
Regulating		
Greenhouse gas reduction (2.2.6.1)	Carbon sequestration in coniferous, deciduous woodland and riparian bushes at, respectively, 6, 5 and 4 ton C·ha^{-1}·yr^{-1} (based on Paul et al. 2009); we assume that mixed woodland is similar to deciduous	For the "current" state of riparian management, a low price of 5 euro per ton C is used (Elsasser et al. 2010, Loeschel et al. 2013), For the ambitious RMP, we assume a moderate increase due to the further development of a carbon credit market to 20 euro (Vermaat et al. 2016, Boesch et al. 2018). For "best practice," we use 10, and for "pessimistic," we use 1 euro per ton C

(Table 2. Continued.)

Service (CICES 5.1 codes)	Description	Explanation, sources
Erosion control: lateral sediment retention (2.2.1.1 and 2.2.1.2)	Expressed as riparian woodland P-loss prevention for erosion-derived material from the lateral zone adjacent to the stream (kg $P \cdot ha^{-1}$ [floodplain]$\cdot yr^{-1}$)	P is used as simple proxy for top-soil to avoid any possible double counting. Median low-end potential P loads for grassland and arable land (from Venohr et al. 2017) are reduced relative to the proportion of the river length that has riparian woodland. If this proportion is 1, all the potential load is retained. Grassland has 1 kg $P \cdot ha^{-1} \cdot yr^{-1}$ available for erosion, cropland 2 kg $P \cdot ha^{-1} \cdot yr^{-1}$. A low-end conservative value estimate for P is derived from an artificial fertilizer market price of 1.1322 €/kg P from a 2010 median market price at www.indexmundi.com
Flood prevention (2.2.1.3)	Damage function based on the risk of a 1/100 yr flood and a median distribution of different land-use types over the river corridor (0–7 €$\cdot ha^{-1}$ [floodplain]$\cdot yr^{-1}$)	Assumption is that one flooded upstream reach prevents the damage of flooding a median downstream reach of equivalent area, hence with the median distribution of land use across the whole river. Value of built-up land is particularly high (252 €/m^2, agricultural land has 7, and woodland has 1). This is conservatively down-adjusted to the height of the flood wave relative to property or crop (we use 0.2), normalized to an annual value with a factor 1/100. Based on De Moel and Aerts (2011) and then normalized to floodplain area. Duration and height of the 1/100 flood was estimated from locally available water authority data repositories and reports: for the Nahe from: http://www.gda-wasser.rlp.de, for the Stever from: www.elwasweb.nrw.de and www.luadb.it.nrw.de; for the Bresse from https://www.vigicrues.gouv.fr/niv2-bassin.php?CdEntVigiCru=18; and for the Azergues from the same website and the Plans Prevention des Risques d'Inondation at www.rhone.gouv.fr. A median flood duration of 7 d was used for all rivers except for the Azergues where we reduced it based on expert judgment of JP
Pest regulation (2.2.3.1 and 2.2.3.2)	Expressed as a modulation of crop productivity (provisioning service 1.1.1.1 above) linked to the presence of woodland and hedges as source of pest control. Modulation is a simple knowledge rule: if woodland cover <25%, then crop productivity reduced to 80%	Based on Tscharntke et al. (2012), who present a rule of thumb on a minimum woodland and hedge cover for central European landscapes
Water quality improvement: nutrient retention (2.2.5.1)	Waterborne phosphorus retention in stream and in riparian floodplain during a flood	Only phosphorus is used to conservatively prevent double counting. Different forms of nitrogen, BOD, or toxic substances are not addressed separately, and hence, this is likely a conservative underestimate. From load reduction per stream km as well as P sedimentation during a flood event and combined with a conservative low market price for P of 1.1322 €/kg P derived from artificial fertilizers in the same way as for erosion control. Load reduction per km of stream length is derived from De Klein and Koelmans (2011), and Olde Venterink et al. (2003) at around 200 kg P/km river length for low land rivers and conservatively reduced to 10 kg P/km river length for the steep Nahe and to 100 kg P/km for the other three rivers, because of a higher slope and flow in the current systems, and in accordance with unpublished MONERIS model estimates by Gericke and Venohr for the Nahe. Load reduction during flood is estimated at 0.14 kg $P \cdot ha^{-1} \cdot yr^{-1}$ for the Nahe, 0.05 for the Stever, 0.50 for the Azergues and 0.01 for the Bresse from local reported flood events and concentrations. The two retention mechanisms are normalized to floodplain area. Concentrations and loads for the Nahe from Ittel and Saelzer (2015), for the Stever from the ELWAS database (www.elwasweb.nrw.de), for the Azergues from Barry and Faure (2011), and for the Bresse from Gay Environnement (2016)

(Table 2. Continued.)

Service (CICES 5.1 codes)	Description	Explanation, sources
Water temperature regulation through riparian shading (2.2.6.2)	Shading affects the probability of trout survival and is expressed as a modulating effect on the cultural service angling. Knowledge rule: if 50% of the main river length is shaded by woodland, then 100% survival, else a stepwise decline in survival to a residual survival of 10%	The fish survival knowledge rule is directly linked to the value estimate for the cultural service recreative angling (Fig. 1), because trout is the most favored species for angling (Arlinghaus 2004). The trout survival knowledge rule is based on Broadmeadow et al (2011) who showed that in a stream in S England periods with water temperature over 25°C were effectively prevented if woodland cover of the stream exceeded 50% of its length. This temperature is the “incipient lethal water temperature” which, if maintained over 7 d, will cause 50% mortality
Cultural		
Recreative angling (taken separate from hunting, 1.1.6.1)	Angling days per km of stream	This is based on the proportion of households with one angler and the number of households in a catchment, and a low-end conservative estimate of their reported willingness to pay for angling per year from Arlinghaus (2004:275) and Federation Nationale de la Peche en France (2014) and Le Goffe and Salanie (2004; 130 €/yr per angler); value is normalized to river length and then floodplain area. Household numbers are derived from regional population statistics
Active recreation in the river and its floodplain corridor (all in CICES category 3.1 pooled)	Separate local estimates for the number of local and residents and tourist visitors that use and appreciate the area per year from local statistics. Multiplied with their willingness to pay for this and modulated by a knowledge rule on the appreciation of a scenic landscape: if forest cover declines below 20% tourist appreciation drops to 60%, if it is above 70% then appreciation drops to 80% (95–138 $€\cdot ha^{-1}$ [non-urban floodplain]$\cdot yr^{-1}$)	Knowledge rule on scenic landscape is based on Frank et al. (2013); willingness to pay of residents and visitors based on Elsasser et al. (2010) and Boesch et al (2018). Resident population and tourism data for the Nahe have been obtained from the public statistics of Rheinland-Pfalz: https://www.statistik.rlp.de/, those for the Stever from Wittkampf (2016), and those for Bresse and Azergues from Barry and Faure (2011)
Nature conservation non-use (all in CICES category 3.2 pooled)	Willingness to pay per valley household (5–162 $€\cdot ha^{-1}$ [non-urban floodplain]$\cdot yr^{-1}$)	Based on nationwide studies in Germany on household willingness to pay for nature conservation (Wuestemann et al. 2014; Boesch et al. 2018; 231 $€\cdot yr^{-1}\cdot household^{-1}$; 27% of households willing to pay, estimated household size 2 persons) and for France on Garcia et al. (2011), 50 $€\cdot yr^{-1}\cdot household^{-1}$; 58% of households willing to pay, household size 3 persons). Estimates adjusted to local population sizes from municipality national statistics and then normalized to floodplain area

Notes: Value estimates are expressed as euro per ha catchment per year, and monetary values can be considered approximately 2010–2015 values. An estimated biophysical service flow (e.g., $kg\cdot ha^{-1}\cdot yr^{-1}$), or a range for the monetary value estimate ($€\cdot ha^{-1}\cdot yr^{-1}$) is reported wherever it is a simple link to land use. A fully worked-out example of our data spreadsheet is provided as Data S1. In the Descriptions, the values in parentheses are ranges of monetary value estimates across the four catchments for the “current” state; or biophysical flow. RMP, riparian management practice.

Land use in floodplain segments

The four river networks were divided into river segments. These are homogenous with respect to national river types, Strahler order, and valley slope (based on the official river networks of the federal states in Germany and from the SYRAH CE network of Valette et al. 2012 in France). This resulted in river segments of different lengths up to several kilometers. To ensure comparability between segments, all segments with a length larger 1 km were subdivided into sub-segments with a length of 0.5–1.0 km and segments with a length less than 0.5 km were excluded from the analysis. The riparian area along the river sub-segments was demarcated using information on the river corridor or alluvial floodplain from local agencies or assuming it to be 12 times bank-full channel width, but at least 30 m on each side of the river. This corresponds to the functional definition of riparian

Table 3. Articulation of four Riparian Management Practices (RMPs) derived from the respective Shared Socioeconomic Pathways (SSPs, O'Neill et al. 2017; full downscaling of SSPs for the four study rivers in Vermaat et al. 2018).

Riparian management practice	Corresponding SSP label (from O'Neill et al. 2017)	In brief	Details: choices for implementation
Current, baseline	—	–	Current, the present situation in the four river systems, which approximately reflects the situation in 2015
Pessimistic	SSP3: "regional rivalry—a rocky road"	WFD no longer pursued, intensity of non-ecological agriculture is increased	• No additional WFD measures implemented, maintenance of structural measures stopped. • Woody vegetation along cropland removed
Best practice	SSP2: "middle of the road"	River management is continued in the period toward 2050 according to the current WFD regulations	• All woody buffer measures as planned in the first and second River Basin Management Plan† cycle are implemented. • In addition, similar measures were assumed to be implemented after the end of the WFD in 2027 to 2050: In the Nahe, 10 m wide woody buffers are developed along each side of all river segments that are classified as priority (Schwerpunktgewaesser) in the Nahe catchment. This is feasible for "best practice" since already between 2000 and 2015, about 1000 of the 8000 river km in Rhineland-Palatinate have been restored. In the Stever, all measures presently considered necessary to reach good ecological status are implemented. For French catchments, all the restoration programs involving riparian buffer management planned by the local stakeholders have been implemented. Furthermore, a sub-basin of Azergues and a sub-basin of Bresse had a dedicated management program for the riparian corridor, which also served as a basis for this scenario
Ambitious	SSP1: "sustainability—taking the green road"	A further development of the WFD toward a more sustainable water use	• Woody vegetation is developed in the whole riparian area, approximately corresponding to the meander belt width or active floodplain, at least a buffer of 30 m on each side of the river. • Except for the following areas: urban areas, transport lines (e.g., roads, railroads), electricity transmission corridors, open, non-forested nature reserves

† RMBP cycle is the policy cycle of the Water Framework Directive (WFD), water quality legislation across the European Union (White and Howe 2003).

areas of Ilhardt et al. (2000) and Verry et al. (2004). Since riparian woodlands provide ecosystem services such as nutrient retention and recreation in a larger landscape context, the whole floodplain was considered in addition, which was technically implemented by demarcating the official 100-yr flooding area, covering large parts of the valley floor, but at least including the riparian area. Land use in the riparian area and floodplain was described by quantifying the area covered by the following land cover classes: urban, urban green spaces, open mining, arable land, grassland, non-woody natural vegetation, woodland-shrubs, woodland-coniferous, woodland-mixed, woodland-deciduous, lakes, wetlands, rivers, and transport lines (roads, railroads). For the two German catchments, the most detailed official land-use data set ATKIS (covering woody vegetation up from a minimum size of 0.1 ha) was complemented by woody vegetation in the riparian area down to single lines of trees along rivers identified on orthophotos using remote sensing. For the two French catchments, a land-use data set with an even higher spatial resolution was already available (0.004 ha; Decherf et al. 2014).

In addition, land-use data were changed according to the RMPs (Table 3) and the area covered by the land-use classes was recalculated.

Results

In the current situation, the mean percentage of woodland in a floodplain segment was found to vary between 25% and 50%, but differences among individual rivers as well as river segments are substantial (Figs. 2, 3; Appendix S1: Fig. S1). Pooled across rivers, the three groups of services each show an "optimum curve" pattern with the available woodland in the floodplain (Fig. 2). River type, expressed as Strahler order, corresponds with the percentage woodland in the floodplain, with more woodland in lower order segments (Fig. 2). Visually estimated optima in woodland cover appears to be somewhat different for the three service groups (Fig. 2): Regulating services are maximal around 30% woodland cover (Strahler orders 4 and 5), provisioning, and cultural services around 45% (the mean woodland cover for Strahler order 2). In an overall analysis of variance (Table 4), segment area was the covariate explaining least, whereas the percentage woodland was more important than Strahler order for provisioning and cultural services, but not for regulating services. Here, Strahler order was more important, likely through the predominance of flood risk prevention. Also, both Strahler order and percentage woodland were independently significant, suggesting that they affect service delivery differently, despite the apparent underlying parallel trend in Fig. 2. Total explained variance of the model was particularly high for cultural services (48%), and this is likely due to the underlying optimum curve in the knowledge rule for the relation between recreation and woodland cover, which is supported when the individual segment estimates are inspected (Fig. 3).

For clarity, we have grouped the 16 services in the three MEA classes. Among the provisioning services, agricultural production and timber were generally most important in the current situation; among the regulating services, this was flood prevention; and among cultural services, recreation was predominant (Table 5). An exception was the Stever, where drinking water production was an important provisioning service, and non-use for biodiversity conservation was in the same order of magnitude as active recreation. The Stever was also distinctly higher in estimated agricultural value per ha than the other three rivers, likely reflecting the more intensive agricultural practice of lowland farming in Northwestern Europe (cf. Table 1).

The RMPs we outlined as plausible alternative future states of river management led to substantial differences in woodland cover in the river floodplain (Fig. 4). In all, we implemented the largest increases in woodland for the ambitious RMP. The overall effect for each river, however, was quite variable. Whereas for the Stever total ecosystem provision declined in the ambitious RMP, it increased for the other three rivers. However, within each river these differences in TEV among the RMPs are modest (maximal effect ratio ambitious/current = 1.15 for the Bresse). The effect ratio was often higher for cultural services, but this could coincide with a decline for provisioning services (e.g., 1.57 and 0.60 for the Stever, but 1.55 and 1.05 for the Bresse, see also Fig. 4). Overall, the absolute patterns were strongest for the Stever (Fig. 4), revealing a trade-off between provisioning and cultural services underlying the apparent flat response in TEV. Regulating services did not change very much across the different RMPs, particularly because they are dominated by our flood prevention estimate. Slight increases with the ambitious RMP (Fig. 4) are due to the increase in carbon sequestration with increasing woodland, and an assumed higher carbon price, and also due to a higher lateral sediment retention with increased woodland (Table 2).

Discussion

Our analysis suggests that in the current landscape configuration, all three service categories showed optimum curves with increasing woodland cover: Provisioning services and cultural services were maximal around 45%, whereas this was around 30% for regulating services. This apparent systematic pattern is more variable in the individual rivers (Appendix S1: Fig. S1). The river management scenarios (RMPs) we implemented led to major differences in riparian woodland cover, but the overall effect on total ecosystem service provision (TEV) was limited.

In three of the four rivers, an increase in cultural services was accompanied with a decrease in provisioning services. Among the predictions made by Gilvear et al. (2013), only the trade-off between cultural and provisioning services was supported by our findings, which we take as an argument for caution in generalizations.

When addressing the potential effect of woodland cover on ecosystem service provision, we must keep in mind that woodland cover in the studied river systems is not low (Table 1: 19–39%), compared to the riverine landscapes such as the one studied by Vermaat et al. (2016; average 25%, range 0–81%) or Maseyk et al. (2018). The latter authors found that an increase in wooded riparian buffers from zero to 7% led to only marginal changes in dairy production, landscape amenity, and three water quality variables. Actually, a substantial proportion of the segments of the lowland rivers Stever and Bresse have woodland cover below 25% (Fig. 3), but the large spatial variability in woodland cover along the stream becomes invisible in our aggregate means (compare Figs. 2, 3). Strahler order and woodland cover covaried, so that lower order upland stream segments have more woodland.

The estimated optimum in total ecosystem service provision at intermediate woodland cover (around 45%) and intermediate Strahler order (2–3) is comfortably close to the advice of 50% from an earlier qualitative review of multiple benefits of riparian woodland (Broadmeadow and Nisbett 2004). Overall patterns in regulating, provisioning, and cultural services suggest an increase in regulating services with increasing stream order, and a decrease in provisioning services and cultural services. This is likely the consequence of the geomorphological landscape configuration in these river networks, where floodplains become larger with higher stream order, and thus have more space for flood retention, but also for competing land-use forms other than agriculture. Steeper, first-order parts of the network often have more woodland due to the combination of suitability and demand for land, as in Tomscha et al. (2017). These are also the landscapes preferred for recreation and nature conservation. We interpret this as an overall, systematic pattern, which of course is subject to substantial local variation (Fig. 3). It must be noted

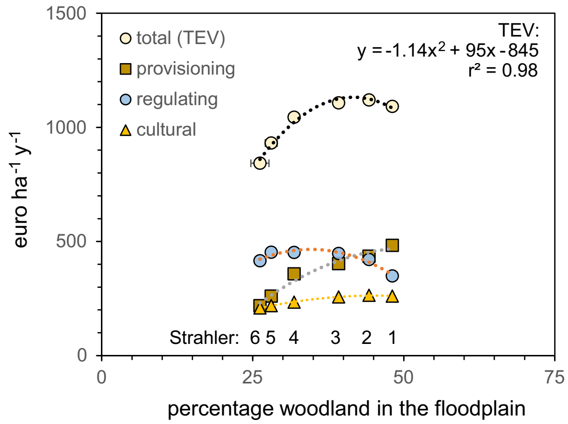

Fig. 2. Effect of the current percentage of woodland in the river corridor on ecosystem service delivery aggregated over all segments in the four river systems. The 16 services are pooled into the three MEA categories provisioning, regulating, and cultural. Strahler order is used as an ordering indicator of river type, with headwater streams having order 1. All polynomial fits are significant ($r^2 > 0.93$, $P < 0.01$), but only the one for total services is displayed. Note that vertical and horizontal standard errors are included but these are generally too small to be depicted due to the high number of segments included. Percentage woodland declined significantly with Strahler order but a regression explained a limited proportion of the variance ($y = -5x + 53$, $r^2 = 0.05$, $P < 0.001$, $n = 7622$). A similar figure broken down for the most important services and the four individual rivers is given in Appendix S1: Fig. S1.

that we did include first-order streams, contrary to Tomscha et al. (2017), because even though a floodplain may not be apparent in the landscape, these small upland streams do flood and the riparian woodland does provide all services we considered here. Our pattern in TEV does not correspond with the findings of Felipe-Lucia et al. (2014) for a Spanish river-and-floodplain system, who report a maximum in the diversity of services provided by the floodplain with a full riparian woodland coverage, but very different approaches make a direct comparison difficult.

The major change in woodland cover we realized in the ambitious RMP, which is based on the sustainability-oriented SSP1, did not lead to equally major shifts in total ecosystem service

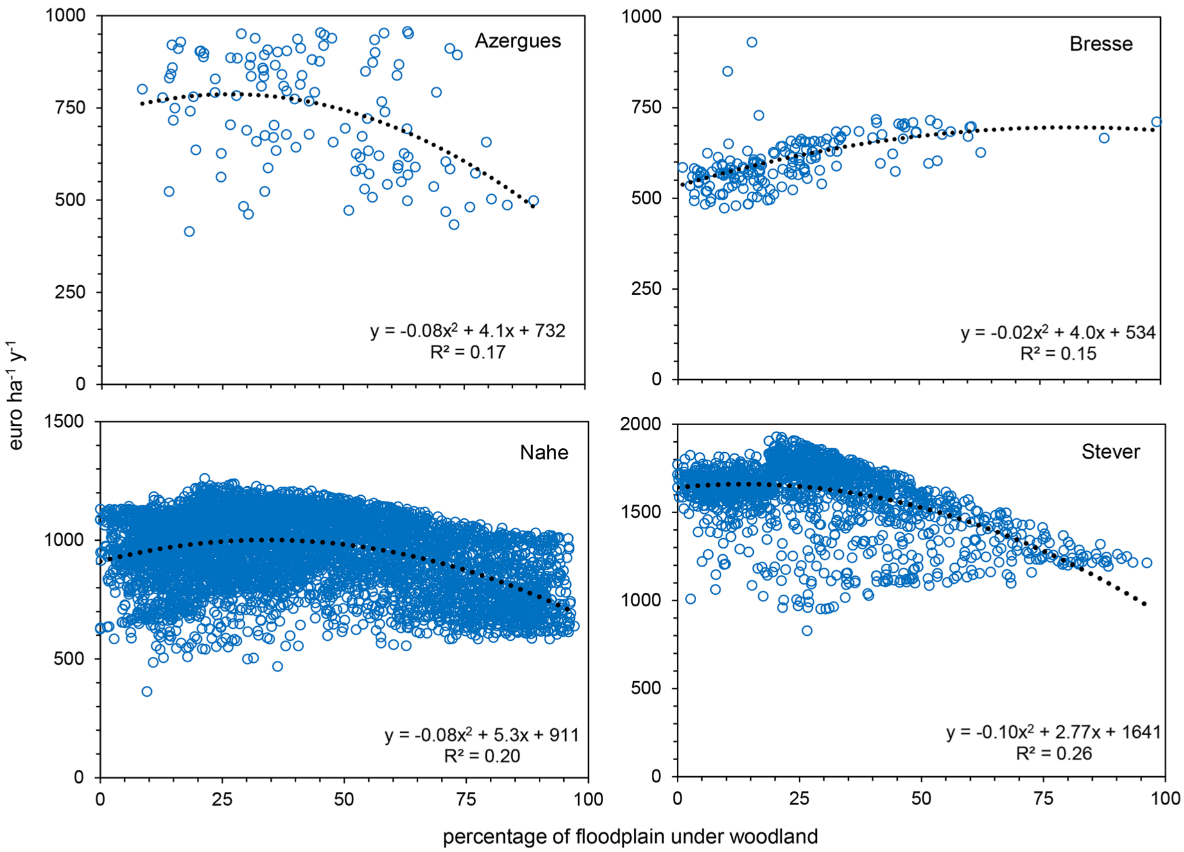

Fig. 3. Individual segment estimates of total economic value as sum of all provisioning, regulating, and cultural services quantified for all four rivers vs. current woodland cover.

delivery, but it led to an increase in cultural services at the expense of provisioning services (Fig. 4). For the Stever, it led to a decline in our value estimate of total ecosystem services, due to the replacement of intensively used agricultural land by woodland that has lower net returns and the predominance of these two provisioning services in the total value estimate (cf. Table 5). However, for the other three rivers total ecosystem service provision increased with woodland cover, particularly due to cultural services. The overall higher value of regulating services for the Nahe and the Bresse is due to a combination of absolute floodplain area (largest in the Nahe, Table 1) and the higher proportion of built-up areas (largest in the Bresse), as these contribute most to the value estimate of flood prevention (Table 2). The second most conspicuous pattern in our scenario outcomes is the limited difference between the remaining three RMPs. Both the pessimistic and the best practice RMP led to only slight changes in woodland cover with similar effects on the patterns in ecosystem service delivery. Notably in the Stever, the best practice RMP would already lead to an increase in cultural services without negatively affecting provisioning services, that is, farming output and drinking water production. It must be noted that the value estimate for nature conservation is derived from an overall appreciation of German citizens for nature protection, rather than a local appraisal of such a landscape change derived from choice experiment surveys as in, for example, Vermaat et al. (2016). Hence, this most likely is a low, conservative estimate, since local valuation studies for charismatic species, such as trout, may well elicit higher value estimates (cf. Martín-López et al. 2007). A third issue is the limited response in regulating services (Fig. 4c), which are dominated by our flood prevention estimate. This is

Table 4. Analysis of variance of the effect of Strahler order (1–6) with segment area and area woodland in the segment as covariates on total, provisioning, regulating, and cultural services value estimates.

Factor in the model	Total services	Provisioning	Regulating	Cultural
Intercept	42	9	55	20
Segment area	3	2	1	3
Area woodland in the segment	42	64	10	75
Strahler order	13	25	34	2
Total variance explained by the model (%)	19	21	23	48

Notes: Data pooled over the four river systems. Total degrees of freedom 7624. Presented are the percentage in the model sums of squares attributed to each factor and the total variance explained by the corrected model. All three factors included explained a highly significant part of the variance ($P < 0.001$).

Table 5. Most important (contributing >5 €·ha^{-1}·yr^{-1}) ecosystem services for each of the four study rivers under the current situation.

	Provisioning (€·ha^{-1}·yr^{-1})			Regulating (€·ha^{-1}·yr^{-1})			Cultural (€·ha^{-1}·yr^{-1})		
River (TEV)	Service	Mean	%	Service	Mean	%	Service	Mean	%
Nahe (935)	Dairy	140	53	Flood prevention	412	97	Recreation	115	46
	Timber	63	24	Carbon sequestration	10	2	Angling	70	28
	Crops	42	16				Conservation non-use	63	25
	Subtotal	264			423			248	
Stever (1590)	Crops	464	44	Flood prevention	299	91	Recreation	78	37
	Drinking water	353	34	Water quality: P-retention	13	3	Conservation non-use	73	35
	Dairy	168	16	Carbon sequestration	7	4	Angling	59	28
	Subtotal	1062			319			210	
Bresse (538)	Dairy	52	36	Flood prevention	279	96	Recreation	72	70
	Crops	51	35	Water quality: P-retention	6	2	Conservation non-use	17	17
	Timber	25	18	Carbon sequestration	5	2	Angling	13	13
	Fish culture	14	9						
	Subtotal	146			290			102	
Azergues (787)	Dairy	50	33	Flood prevention	487	97	Recreation	85	65
	Crops	48	31	Carbon sequestration	9	2	Angling	33	25
	Timber	42	28	Water quality: P-retention	8	2	Conservation non-use	13	10
	Subtotal	152			504			131	

Note: Presented ares mean estimated monetary value per ha, percentage contributed to its MEA class, as well as subtotals and grand totals, the latter an estimate of total economic value (TEV).

likely due to the fact that we have not varied population density, settlement policy or land and house pricing, or the location of settlements in our RMPs because we chose these RMPs to be limited to measures within the remit of European water management institutions. Thus, our estimate of flood damage value is likely both a conservative low-end value and unrealistically stable, but we think it is justified to limit the number of assumptions in our scenario articulations. Finally, Fig. 4 suggests a trade-off between provisioning and cultural services, in contrast to what we deduce from the pattern in the current situation in Fig. 2. So, if the current situation is pushed toward the occupation of agricultural land with woodland (ambitious RMP), this leads to a decline in overall value—an obvious "trade-off." Martín-López et al. (2012) also found a trade-off between provisioning and cultural services in an extensive study of societal preferences in eight areas across Spain. The apparent contradiction in our data is due to the fact that in the "current" situation we see a changing pattern along the length of the four rivers pooled, whereas when the comparison with the ambitious RMP is made, we see a change over time, and the separate pattern for each river is not equally intense.

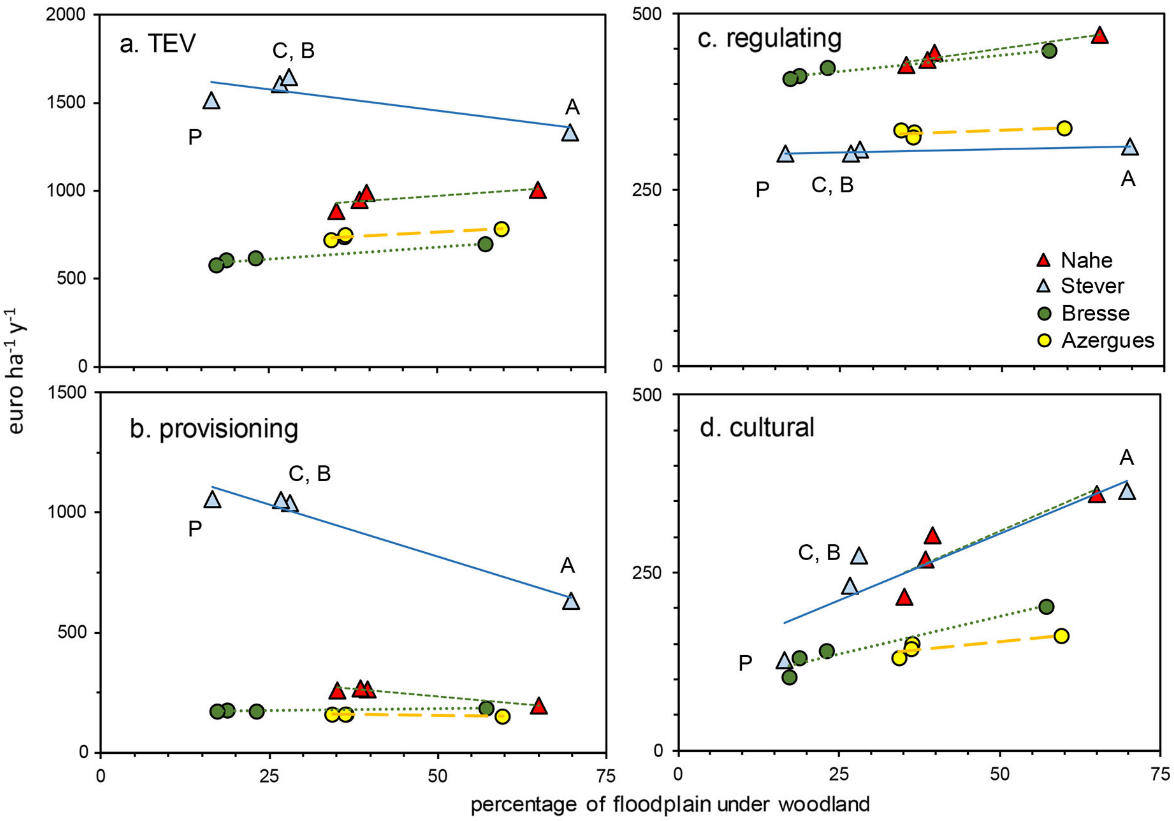

Fig. 4. Effect of the different Riparian Management Practices (RMPs) on (a) total ecosystem service delivery expressed as total economic value, (b) regulating, (c) provisioning, and (d) cultural services, all plotted against the percentage of woodland for each RMP. P, pessimistic; C, current; B, best practice and A, ambitious. The order of woodland cover of these four RMPs is the same for each river, from low to high: P, C, B, A. Different symbols indicate the four different rivers (see legend).

The "Mononen cascade" framework applied here was developed from the cascade model proposed by Mononen et al. (2016), which again has its roots in the cascade presented by among others Haines-Young and Potschin (2010). We will not reiterate the discussion whether "nature can be valued at all" (Gómez-Baggethun et al. 2010, Hermelingmeier and Nicholas 2017), but important premises of our approach are that one can attribute final services to land-use cover types and that monetary estimates of these services are consistent and "valid," though not necessarily "accurate" or "precise." Our compilation of different value estimates each with its underlying approaches and assumptions is a seriously disputed aspect of TEV estimates (among others Schröter et al. 2014). We think it allows comparison across scenarios or policy alternatives and services, if only framed carefully in a consistent study design (Boerema et al. 2017, Hanna et al. 2018), and thus can be used to inform policy. The valuation step, in principle, is not different from using a ranking scale which is summed, as applied in, for example, Burkhard et al. (2009) or Newton et al. (2012), but the monetary valuation causes a weighing of the different services, rather than treating all individual services as equal. Our weighing with a monetary ruler is equally traceable as using ranks or scores (Table 2), but it is based on expressed societal preferences, which indeed may lead to lower value estimates for nature conservation non-use than for active recreation (Table 5), although in three of our four rivers these are remarkably close. Vermaat et al. (2020) discuss the methodological strengths and weaknesses of the current framework in more detail.

Briefly, we see two important limitations of our current study. First, our approach would have benefited from a spatial linking of stream segments in the river network, so that we could have estimated flood prevention, but also nutrient and sediment retention in a more realistic way. We are not aware of a study that has succeeded in combining such hydrological realism with an assessment of the full suite of ecosystem services. Second, we have not done a formal uncertainty or sensitivity analysis, because estimating uncertainties without empirical basis would be mere guesswork. For example, a sensitivity analysis on the effect of our flood prevention estimate in the sum of regulating services and also that of our knowledge rule on the effect of woodland cover on recreational appreciation would have been useful. Vermaat et al. (2016) assessed changes in ecosystem service provision due to restoration of European rivers and their floodplains with a similar though less formalized approach. Our current TEV estimates are similar in order of magnitude (their median unrestored TEV 1000 €·ha^{-1}·yr^{-1}; ours 843 €·ha^{-1}·yr^{-1}). A final methodological point is our consistent choice for the most conservative low-end estimate, wherever we had the choice. Our justification is that we want to remain far from optimistic advocacy (Bouma and van Beukering 2015) and that we combine estimates based on highly different underlying approaches, but the consequence is that some of our estimates indeed are low. An obvious example is carbon sequestration: Other work, such as the natural capital accounting exercise for the UK (Trenbirth and Dutton 2020), uses 20 €/t C for non-traded carbon equivalents as a mid-level for 2010 which increases in the subsequent years, compared to our baseline of 5 €/t (Table 2). Overall, this implies that our value estimates best can be seen as indicative, but internally consistent, and then for an approximate time window of 2010–2015 for the baseline scenario.

If we equate our ambitious RMP to a major restoration effort, we can test the hypothesis of Gilvear et al. (2013). Increased woodland cover, however, only led to a substantial decrease in provisioning services in one of the four rivers, regulating services increased in two, whereas cultural services indeed increased in all cases. Hence, we cannot simply generalize along the lines of Gilvear et al. (2013) but must revert to more service- or landscape-specific hypotheses. For example, the market value of woodland linked to timber and an increasing demand for biomass to replace fossil fuel (Trømborg et al. 2020) should not be ignored, but also the intensity of adjacent land use, and hence, the land rent (cf. the Stever and Vermaat et al. 2016) is relevant when monetary value estimates of all possible services are of interest. At the landscape scale of a river and its floodplain, we see that greatly increasing the percentage covered with woodland, as in the ambitious RMP, may well lead to an increase in cultural services, hence appreciation by recreation including anglers, at the expense of provisioning services, here particularly agriculture.

Compared to previous assessments of ecosystem services provision along rivers, our study combines high spatial detail, a comprehensive and well-defined set of ecosystem services that includes a final monetary value estimate, and a verification stage with stakeholder representatives, rather than a limited selection of services or a rank-based scoring system. This largely corresponds with the five recommendations made by Hanna et al. (2018): assess multiple services, use reproducible data and methods, include service interactions, select extent study area relevant to question, and engage with stakeholders. The latter has been important in the verification of our scenario's, without these reflective workshops, our scenario articulations as RMPs would have been less realistic to river managers and land-use planners. At the same time, we experienced that we had to maintain a balance with our basis in the benchmark SSPs to ensure comparability with other scientific work on scenarios.

In conclusion, we have shown that our set of seven provisioning, six regulating, and three cultural services, as quantified with the "Mononen cascade" for four central European river systems, currently all show optimum curves with increasing woodland cover: Provisioning services and cultural services were maximal around 45%, whereas this was around 30% for regulating services. On average, river type, expressed as Strahler order, was found to correspond quite closely with the percentage woodland in the floodplain, with more woodland in steeper lower order segments.

Geomorphological and land cover variation among and within individual river systems is pooled into this average, but can be substantial (Table 1, Fig. 3; Appendix S1: Fig. S1). The three different modeled woodland cover scenarios led to a remarkably limited change in total ecosystem service delivery, even if mean woodland cover was reduced from 27% to 17% in the pessimistic RMP and increased to 70% in the ambitious RMP for the most extreme case of the Stever. We did, however, see a clear decline of provisioning service with increased woodland cover and an increase in cultural services. Regulating services did not change that much, because they are dominated by flood prevention in our assessment. It appears that the "best practice" scenario combines a modest increase in cultural services with a slight increase in provisioning services. Also, the outcome suggests that very ambitious nature conservation objectives can be met with a limited decrease in total societal benefit (TEV) only, and despite the low-end monetary value estimates for nature conservation non-use.

Acknowledgments

We thank all our colleagues in OSCAR for the positive collegial cooperation, and the workshop participants in Mainz, Münster, and Lyon for their constructive contributions. Work on this paper funded through the 2015–2016 BiodivERsA COFUND call for research proposals, which is supported by the national funders German Federal Ministry of Education and Research (01LC1618A), Agence National de Recherche (ANR-16-EBI3-0015-01), and Research Council of Norway (268704/E50). The first author also wants to acknowledge his colleagues in the ecosystem services team of the Nordic Centre of Excellence BIOWATER, which is funded by Nordforsk under project number 82263: Discussions with Bart Immerzeel, Eija Pouta, and Artti Juutinen were very useful. Erik Aschehoug helped with the writing style.

Literature Cited

Agreste. 2017. Report on Agricultural Statistics for the region Auvergne-Rhone Alpes. Ministere de l'agriculture et de l'alimentation, Paris, France. [In French.]

Anderer, P., U. Dumont, C. Linnenweber, and B. Schneider. 2009. The hydropower potential in Rhineland-Palatinate. Korrespondenz Wasserwirtschaft 2:223–227. [In German.]

Arlinghaus, R. 2004. Angling in Germany – a social and economical analysis. Report 18/2004. IGB Berlin: Institut fuer Gewaesserkunde und Binnenfischerei, Berlin, Germany. [In German.]

Barry, S., and J. P. Faure. 2011. Study of fish and crayfish in the catchment of the Azergues. Report. Federation du Rhone pour la Peche et la Protection du milieu aquatique, La Tour de Salvagny, France. [In French.]

Bateman, I. J., et al. 2013. Bringing ecosystem services into economic decision-making: land use in the United Kingdom. Science 341:45–50.

Bateman, I. J., G. M. Mace, C. Fezzi, G. Atkinson, and K. Turner. 2011. Economic analysis for ecosystem service assessments. Environmental Resource Economics 48:177–218.

Berkhout, F., J. Hertin, and A. Jordan. 2002. Socioeconomic futures in climate change impact assessment: using scenarios as 'learning machines'. Global Environmental Change 12:83–95.

Bernhardt, E. S., et al. 2005. Synthesizing U.S. river restoration efforts. Science 308:636–637.

Boerema, A., A. J. Rebelo, M. B. Bodi, K. J. Esler, and P. Meire. 2017. Are ecosystem services adequately quantified? Journal of Applied Ecology 54:358–370.

Boerman, J., U. Bodin, U. Lemke, and S. Schmitz. 2015. Time series for agriculture in Nordrhein-Westfalen 1960 to 2014. Landwirtschaftskammer Nordrhein-Westfalen, Münster, Germany. [In German.]

Boesch, M., P. Elsasser, K. Franz, M. Lorenz, C. Moning, R. Olschewski, A. Roedl, H. Schneider, B. Schroeppel, and P. Weller. 2018. Forest ecosystem services in rural areas in Germany: insights from the national TEEB study. Ecosystem Services 31:77–83.

Bouma, J. H., and P. J. H. van Beukering. 2015. Ecosystem services: from concept to practice. Cambridge University Press, Cambridge, UK.

Boyd, J., and S. Banzhaf. 2007. What are ecosystem services? The need for standardized environmental accounting units. Ecological Economics 63:616–626.

Braat, L. 2018. Five reasons why the Science publication "Assessing nature's contributions to people" (Diaz et al. 2018) would not have been accepted in Ecosystem Services. Ecosystem Services 30:A1–A2. https://doi.org/10.1016/j.ecoser.2018.02.002

Brander, L., J. E. Vermaat, and R. J. G. M. Florax. 2006. The empirics of wetland valuation: a meta-analysis. Environmental Resource Economics 33:223–250.

Broadmeadow, S. B., J. G. Jones, T. E. L. Langford, P. J. Shaw, and T. R. Nisbet. 2011. The influence of riparian shade on lowland stream water temperatures in Southern England and their viability for brown trout. River Research Applications 27:226–237.

Broadmeadow, S. B., and T. R. Nisbett. 2004. The effects of riparian forest management on the freshwater environment: a literature review of best management practice. Hydrology Earth System Science 8:286–305.

Brown, A. G., L. Lespez, D. A. Sear, J. J. Macaire, P. Houben, K. Klimek, R. E. Brazier, K. Van Oost, and B. Pears. 2018. Natural vs anthropogenic streams in Europe: history, ecology and implications for restoration, river-rewilding and riverine ecosystem services. Earth-Sciences Reviews 180:185–205.

Burkhard, B., F. Kroll, F. Müller, and W. Windhorst. 2009. Landscapes' capacities to provide ecosystem services – a concept for land-cover based assessments. Landscape Online 15:1–22.

Busch, G. 2006. Future European agricultural landscapes – What can we learn from existing quantitative land use scenario studies. Agricultural Ecosystems Environment 114:121–140.

De Klein, J. J. M., and A. A. Koelmans. 2011. Quantifying seasonal export and retention of nutrients in West European lowland rivers at catchment scale. Hydrological Processes 25:2102–2111.

De Moel, H., and J. C. J. H. Aerts. 2011. Effect of uncertainty in land use, damage models and inundation depth on flood damage estimates. Natural Hazards 58:407–425.

Decherf, A., N. Lalande, and F. Cernesson. 2014. Note on land cover in river corridors of the Saone basin. Technical Report. IRSTEA – UMR TETIS, Montpellier, France. [In French.]

Díaz, S., et al. 2018. Assessing nature's contributions to people. Science 359:270–272.

Dick, J., P. Verweij, E. Carmen, R. Rodela, and C. Andrews. 2017. Testing the ecosystem services cascade framework and QUICKScan software tool in the context of land use planning in Glenlivet Estate, Scotland. International Journal of Biodiversity Science, Ecosystem Services & Management 13:12–25.

Duncker, P. S., K. Raulund-Rasmussen, P. Gundersen, K. Katzenstein, J. De Jong, H. P. Ravn, M. Smith, O. Eckmueller, and H. Spiecker. 2012. How forest management affects ecosystem services, including timber production and economic return: synergies and trade-offs. Ecology and Society 17:50.

Elsasser, P., H. Englert, and J. Hamilton. 2010. Landscape benefits of a forest conversion program in North East Germany: results of a choice experiment. Annals Forestry Research 53:37–50.

Federation Nationale de la Peche en France. 2014. Annual Activity Report 2014. [In French.] www.federationpeche.fr

Felipe-Lucia, M. R., F. A. Comín, and E. M. Bennett. 2014. Interactions among ecosystem services across land uses in a floodplain agroecosystem. Ecology and Society 19:20.

Frank, S., C. Fuerst, L. Koschke, A. Witt, and F. Makeschin. 2013. Assessment of landscape aesthetics – validation of a landscape-metrics based assessment by visual estimation of the scenic beauty. Ecological Indicators 32:222–231.

Garcia, S., P. Harou, C. Montagne, and A. Stenger. 2011. Valuing forest biodiversity through a national survey in France: a dichotomous choice contingent valuation. International Journal of Biodiversity Science, Ecosystem Services & Management 7:84–97.

Gay Environnement. 2016. Water quality assessment of the Chalaronne, its tributaries and the Calonne. Rapport final pour le Syndicat des Rivières des Territoires de Chalaronne, Grenoble, France. [In French.]

Gilvear, D. J., C. J. Spray, and R. Casas-Mulet. 2013. River rehabilitation for the delivery of multiple ecosystem services at the river network scale. Journal of Environmental Management 126:30–43.

Gómez-Baggethun, E., R. de Groot, P. L. Lomas, and C. Montes. 2010. The history of ecosystem services in economic theory and practice: from early notions to markets and payment schemes. Ecological Economics 69:1209–1218.

Grizetti, B., F. Bouraoui, and A. Aloe. 2007. Spatialised European nutrient balance. Report EUR 22692 EN – DG Joint Research Centre, Institute for the Environment and Sustainability, Ispra, Italy.

Haines-Young, R., and M. B. Potschin. 2010. The links between biodiversity, ecosystem services and human well-being. Pages 110–139 *in* D. G. Raffaelli and C. L. J. Frid, editors. Ecosystem ecology: a new synthesis. Cambridge University Press, Cambridge, UK.

Hanna, D. E. L., S. A. Tomscha, C. O. Dallaire, and E. M. Bennett. 2018. A review of riverine ecosystem service quantification: research gaps and recommendations. Journal of Applied Ecology 55:1299–1311.

Hastreiter, H. 2017. Small private forest – does it pay off? LWF aktuell 2017:37–39. [In German.]

Hermelingmeier, V., and K. A. Nicholas. 2017. Identifying five different perspectives on the ecosystem services concept using Q methodology. Ecological Economics 136:255–265.

Hiller, J., and T. Wichmann. 2010. Results of a comparison between organic and conventional aquaculture enterprises in Germany. Report. LMS Landwirtschaftsberatung Mecklenburg-

Vorpommern/Schleswig-Holstein GmbH, Schwerin, Germany. [In German.]
Ilhardt, B., E. Verry, and B. Palik. 2000. Defining riparian areas in riparian management in forests of the Continental Eastern United States. Lewis Publishers, New York, New York, USA.
IPCC. 2014. Climate change 2014: synthesis report. Pages 151 *in* Core Writing Team, R. K. Pachauri, and L. A. Meyer, editors. Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. IPCC, Geneva, Switzerland.
Ittel, I., and J. Saelzer. 2015. Nutrients in the catchment of the Nahe, development of Nitrate, Ammonium and Phosphate loading, trends 1992-2012. Report. Landesamt für Umwelt, Wasserwirtschaft und Gewerbeaufsicht, federal state Rheinland Pfalz, Mainz, Germany. [In German.]
Jähnig, S. C., A. W. Lorenz, D. Hering, C. Antons, A. Sundermann, E. Jedicke, and P. Haase. 2011. River restoration success: a question of perception. Ecological Applications 21:2007–2015.
Kenter, J. O. 2018. IPBES: don't throw out the baby whilst keeping the bathwater; Put people's values central, not nature's contributions. Ecosystem Services 33:40–43.
LANUV. 2017. Assessment of the potential renewable energy in Nordrheinland-Westfalen. Potenzialstudie Erneuerbaere Energien NRW Teil 5 Wasserkraft. Fachbericht 40. Landesamt fuer Natur-, Umwelt und Verbraucherschuetz Nordrhein-Westfalen, Recklinghausen, Germany. [In German.]
Le Goffe, P., and J. Salanie. 2004. Organised angling in France: an economic analysis based on salmonids. La gestion associative de la peche en France: une analyse economique a partir du cas des salmonides. Bulletin Francais de Peche et Pisciculture 375:15–34. [In French.]
Leyer, I., E. Mosner, and B. Lehman. 2012. Managing floodplain forest restoration in European river landscapes combining ecological and flood-protection issues. Ecological Applications 22:240–249.
Loeschel, A., B. Sturm, and C. Vogt. 2013. The demand for climate protection – empirical evidence from Germany. Economy Letters 118:415–418.
Lorenzoni, I., A. Jordan, M. Hulme, R. K. Turner, and T. O'Riordan. 2000. A co-evolutionary approach to climate impact assessment: part I. Integrating socio-economic and climate change scenarios. Global Environmental Change 10:57–68.
Martín-López, B., et al. 2012. Uncovering ecosystem service bundles through social preferences. PLOS ONE 7:e38970.
Martín-López, B., C. Montes, and J. Benayas. 2007. The non-economic motives behind the willingness to pay for biodiversity conservation. Biological Conservation 139:67–82.
Maseyk, F. J. F., E. J. Dominati, and A. D. Mackay. 2018. Change in ecosystem service provision within a lowland dairy landscape under different riparian margin scenarios. International Journal of Biodiversity Science, Ecosystem Services & Management 14:17–31.
MEA. 2005. Millennium Ecosystem Assessment. General Synthesis Report. Island Press, Washington, D.C., USA.
Mononen, L., A. P. Auvinen, A. L. Ahokumpu, M. Rönkä, N. Aarras, H. Tolvanen, M. Kamppinen, E. Viiret, T. Kumpula, and P. Vihervaara. 2016. National ecosystem service indicators: measures of social-ecological sustainability. Ecological Indicators 61:27–37.
Mueller, G., and J. H. Mueller. 2017. Presentation of the economic situation in agriculture and viniculture in Rhineland-Palatine based on farm accounting statistics for 2016/2017. Report. Agricultural Statistics Rheinland Pfalz. Mainz, Germany. [In German.]
Newton, A. C., K. Hodder, E. Cantarello, L. Perrella, J. C. Birch, J. Robins, S. Douglas, C. Moody, and J. Cordingley. 2012. Cost–benefit analysis of ecological networks assessed through spatial analysis of ecosystem services. Journal of Applied Ecology 49:571–580.
O'Neill, B. C., et al. 2017. The roads ahead: narratives for shared socioeconomic pathways describing world futures in the 21st century. Global Environmental Change 42:169–180.
Olde Venterink, H., F. Wiegman, G. E. M. Van der Lee, and J. E. Vermaat. 2003. Role of active floodplains for nutrient retention in the river Rhine. Journal of Environmental Quality 32:1430–1435.
Paul, C., M. Weber, and R. Mosandl. 2009. The carbon sequestration in young afforestation areas, a literature survey. Karl Gayer Institut der Technische Universitaet Muenchen, Munich, Germany. [In German.]
Petts, G., and I. Foster. 1985. Rivers and landscape. Pages 274. Arnold, London, UK.
Poisvert, C., F. Curie, and F. Moatar. 2017. Annual agricultural N surplus in France over a 70-year period. Nutrient Cycles Agroecosystems 107:63–78.
Popp, A., et al. 2017. Land-use futures in the shared socioeconomic pathways. Global Environmental Change 42:331–345.
Schröter, M., E. H. Van der Zanden, A. R. E. Van Oudenhoven, R. P. Remme, H. M. Serna-Chavez, R. S. De Groot, and P. Opdam. 2014. Ecosystem

services as a contested concept: a synthesis of critique and counter-arguments. Conservation Letters 7:514–523.

Schulp, C. J. E., W. Thuiller, and P. H. Verburg. 2014. Wild food in Europe: a synthesis of knowledge and data of terrestrial wild food as an ecosystem service. Ecological Economics 105:292–305.

Societe Forestiere. 2018. The market of forests in France in 2018. Report Societe Forestiere, Paris, France. [In French.]

Stutter, M. I., W. J. Chardon, and B. Kronvang. 2012. Riparian buffer strips as a multifunctional management tool in agricultural landscapes: introduction. Journal Environmental Quality 41:297–303.

Sweeney, B. W., and J. D. Newbold. 2014. Streamside forest buffer width needed to protect stream water quality, habitat, and organisms: a literature review. Journal of the American Water Resources Association 50:560–584.

Tockner, K., F. Malard, and J. V. Ward. 2000. An extension of the flood pulse concept. Hydrological Processes 14:2861–2883.

Tomscha, S. A., S. E. Gergel, and M. J. Tomlinson. 2017. The spatial organization of ecosystem services in river-floodplains. Ecosphere 8:e01728.

Trenbirth, H., and A. Dutton. 2020. Woodland natural capital accounts, UK: 2020, Natural capital accounts containing information on ecosystem services for woodlands in the UK. Pages 47. Statistical Bulletin Office for National Statistics, London, UK.

Trømborg, E., E. O. Jåstad, T. F. Bolkesjø, and P. K. Rørstad. 2020. Prospects for the Norwegian forest sector: A green shift to come? Journal of Forest Economics 35:305–336.

Tscharntke, T., et al. 2012. Landscape moderation of biodiversity patterns and processes – Eight hypotheses. Biological Reviews 87:661–685.

Valette, L., J. Piffady, A. Chandesris, and Y. Souchon. 2012. SYRAH-CE: description of the modelled risks of hydromorphological change for river water quality. Report Onema/Irstea Lyon MAEP LHQ 104. [In French.]

Van Looy, K., T. Tormos, Y. Souchon, and D. Gilvear. 2017. Analyzing riparian zone ecosystem services bundles to instruct river management. International Journal Biodiversity Science and Ecosystem Service Management 13:330–334.

Venohr, M., P. Fischer, D. Pietras, and R. Arora. 2017. Retention of sediments, nutrients and pesticides in riparian buffer strips, summary and evaluation of a literature search. IGB report to the Ministry of Environment, Climate and Energy of the Federal State of Baden-Wuertemberg. IGB, Berlin, Germany. [In German.]

Vermaat, J. E., et al. 2016. Assessing the societal benefits of river restoration using the ecosystem services approach. Hydrobiologia 769:121–135.

Vermaat, J. E., S. Haaland, and J. Kail. 2018. Storylines articulating scenarios for an assessment of the importance of woody buffers along European streams. Report 49. Norwegian University of Life Sciences, Faculty of Environmental Sciences and Natural Resource Management, Ås, Norway. https://hdl.handle.net/11250/2648386

Vermaat, J. E., B. Immerzeel, E. Pouta, and A. Juutinen. 2020. Applying ecosystem services as a framework to analyse possible effects of a green bio-economy shift in Nordic catchments. Ambio 49:1784–1796.

Verry, E., C. Dolloff, and M. Manning. 2004. Riparian ecotone: a functional definition and delineation for resource assessment. Water, Air, Soil Pollution Focus 4:67–94.

Wallace, K. 2007. Classification of ecosystem services: problems and solutions. Biological Conservation 139:235–246.

White, I., and I. Howe. 2003. Policy and practice: planning and the European Union Water Framework Directive. Journal of Environmental Planning and Management 46:621–631.

Whited, D. C., M. S. Lorang, M. J. Harner, F. R. Hauer, J. S. Kimball, and J. A. Stanford. 2007. Climate, hydrologic disturbance, and succession: drivers of floodplain pattern. Ecology 88:940–953.

Wittkampf, P. 2016. Spatial foci of tourism in Westfalen. Report in Online-Documentation of the Geographischen Kommission für Westfalen im Landschaftsverband Westfalen-Lippe, Germany. [In German.] https://www.lwl.org/LWL/Kultur/Westfalen_Regional/Wirtschaft/Tourismus_Westfalen

Wuestemann, H., J. Meyerhoff, M. Ruehs, A. Schaefer, and V. Hartje. 2014. Financial costs and benefits of a program of measures to implement a national strategy on biological diversity in Germany. Land Use Policy 36:307–318.

Supporting Information

Additional Supporting Information may be found online at: http://onlinelibrary.wiley.com/doi/10.1002/ecs2.3716/full