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Reduction of nitrogen and phosphorus loads to European rivers by riparian buffer zones

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ABSTRACT

Key-words: *Riparian buffer zone, nutrient load reduction, buffer capacity, buffer permeability*

Riparian buffer zones play an important role as nutrient pollution controls for rivers. We provide a harmonized, explorative quantitative and spatially explicit continental assessment of nitrogen and phosphorus potential retention in riparian buffer zones along the European river network CCM2 (Catchment Characterization and Modellling). Diffuse emissions of nutrients from fertilized cropland and pasture, based on a statistical model (GREEN (Geospatial Regression Equation for European Nutrient Losses)), are partitioned into surface and subsurface flow pathways based on the innovative SUGAR (Surface water/Groundwatercontribution index) index. Surface flow N and P emissions are assumed to undergo attenuation as a function of riparian buffer width. In contrast, the attenuation of subsurface flow emissions and emissions from wetlands is assumed to be independent of buffer width. Buffer attenuation follows a nutrient-specific negative exponential decay function. For the study area, we estimate retention in surface runoff emissions of 33% for nitrogen and 65% for phosphorus. The results represent a valuable data source for water basin management with respect to water quality improvement, in particular buffer zone restoration.

RÉSUMÉ

Réduction des charges d'azote et de phosphore dans les rivières européennes par les zones tampons riveraines

Mots-clés : *Zone tampon riveraine, réduction de la charge en éléments nutritifs, capacité tampon, perméabilité de zone tampon*

Les zones tampons riveraines jouent un rôle important dans la régulation de la pollution en éléments nutritifs pour les cours d'eau. Nous proposons une évaluation harmonisée, exploratoire quantitativement et spatialement explicite de la rétention potentielle d'azote et de phosphore dans les zones tampons riveraines le long du réseau fluvial européen CCM2. Les émissions diffuses de nutriments provenant des terres cultivées fertilisées et des pâturages, basées sur un modèle statistique (GREEN), sont réparties en écoulements de surface et souterrain, voies basées sur l'indice novateur SUGAR. Les émissions de surface de N et de P sont supposées subir une atténuation en fonction de la largeur de la bande riveraine. En revanche, l'atténuation des émissions de flux de subsurface et les émissions des zones humides est supposée être indépendante de la largeur du tampon. L'atténuation dans la zone tampon suit une fonction négative de décroissance exponentielle spécifique du nutriment. Pour la zone d'étude, nous estimons la rétention des émissions de ruissellement à 33 % pour l'azote et 65 % pour le phosphore.

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Les résultats représentent une source précieuse de données pour la gestion des bassins hydrographiques en ce qui concerne l'amélioration de la qualité de l'eau, en particulier la restauration des zones tampon.

INTRODUCTION

Most pollutants entering surface waters result from non-point source pollution activities, including runoff from agricultural land (Klapproth and Johnson, 2009). Non-point source pollution originates from emissions of non-discrete points. It occurs mainly through runoff, erosion, tile drainage and groundwater discharge. Sources of origin include agricultural activities linked to the application of mineral fertilizers and manure, atmospheric deposition and scattered dwellings. Agriculture is the main contributor of nitrogen found in European streams (Grizzetti *et al.*, 2012) and the reduction of diffuse losses constitutes a major challenge for the improvement of water quality in European rivers.

Numerous studies have confirmed the crucial role of stream riparian zones (sometimes also called riparian buffer strips) in regulating sediment and nutrient transport into larger river courses (Castelle *et al.*, 1994; Gilliam, 1994; Hill, 1996; Jordan *et al.*, 1993; Phillips, 1989). Riparian forests are important both as filters and sinks for nitrogen, phosphorus, calcium, potassium, sulfur and magnesium (Lowrance *et al.*, 1984). For the particular case of nutrients entering riparian buffer zones/strips, a number of studies exist (Barling and Moore, 1994; Daniels and Gilliam, 1996; Dillaha *et al.*, 1989; Lowrance *et al.*, 2001; Mander *et al.*, 1995), amongst them two, recent, extensive review studies (Klapproth and Johnson, 2009; Mayer *et al.*, 2005).

Nutrient losses depend on a variety of factors, such as management practices, precipitation patterns, soil moisture, timing of fertilizer application, soil and vegetation characteristics, and others. Nutrients enter surface waters *via* subsurface or surface flows either in dissolved form or attached to soil particles (Gilliam *et al.*, 1997). Nitrogen is most commonly transported as dissolved nitrogen through subsurface flows, while phosphorus is often adsorbed to soil particles and organic materials in surface runoff after storm events (Pionke *et al.*, 1995).

Nitrogen subsurface removal efficiency is reported to be higher than surface removal efficiency (89.6% *vs.* 33.3% according to Mayer *et al.* (2005)) and, moreover, subsurface removal is not related to buffer width. Mayer *et al.* (2005) assume that the extent to which riparian buffers are able to attenuate nitrogen is at least partly a function of buffer width, referring to Vidon and Hill (2004) and to Phillips (1989), the latter estimating the nitrogen removal effectiveness portion due to buffer width to be 81%. However, additional factors also influence effectiveness.

Riparian forests remove nitrogen from agricultural runoff by different mechanisms. Estimates of nutrient retention quantities based on literature sources are reported by Pärn *et al.* (2012). Many studies support the idea that the primary mechanism is denitrification (Cooper *et al.*, 1987; Johnston *et al.*, 1984), for which Hedin *et al.* (1998) and Vidon and Hill (2005) describe the processes involved. Other removal mechanisms include uptake by vegetation and soil microbes and retention in riparian soils.

Key factors affecting the water quality benefits of riparian buffer zones are related to hydrology, soil and vegetation. Field slope and adjacent land use play an important role when assessing incoming pollutant loads as well as the frequency and magnitude of storm events (Osborne and Kovacic, 1993). An important requirement for buffer zone effectiveness is the existence of shallow sheet flow into and across the buffer strip (Klapproth and Johnson, 2009). Soil features that influence water quality include: soil depth to the water table, soil permeability and texture, soil chemistry, and organic matter content (EPA, 1993; Hedin *et al.*, 1998). These factors including soil temperature determine important soil biogeochemical processes. Phosphorus retention is highly dependent on overland flow conduits and barriers (Pärn *et al.*, 2012). Due to chemical-physical characteristics the removal of particulate phosphorus can be expected to be more effective than the removal of dissolved phosphate (Peterjohn and Correll, 1984). The COST (intergovernmental framework for European Cooperation in Science and Technology) Action 869, dealing with mitigation options for nutrient reduction in surface water and groundwaters, estimates for buffer widths from 3 to 10 m (slope $< 6\%$) a reduction of 70 to 90% in the form of particulate phosphorus, and 30–50% for dissolved nutrients (Gascuel *et al.*, 2010).

The retention of nutrients has been simulated using physically based approaches. Those process based models, such as REMM (Riparian Ecosystems Management Model) (Inamdar *et al.*, 1999) or VFSMOD (Vegetative Filter Strip MODel) (Munoz-Carpena and Parsons, 2005; Munoz-Carpena *et al.*, 1999) have been used at a local scale but require exhaustive data input which, on a European scale, becomes prohibitive. There is thus a need to develop new tools that can be applied at a larger scale in order to identify areas where riparian zones need to be maintained or restored in order to improve or maintain good water quality. In this study we propose a new simplified approach to estimate potential nutrient removal within riparian buffer zones using harmonized and readily available data. The expert model presented here is used to perform an explorative analysis: Screening or targeting of areas of interest for an improved water quality policy, providing first nutrient retention estimates, and delineation of areas which are particularly affected by nutrient emissions, or are in a particular good state. Similar screening techniques have been used in the past for large scale water pollution assessments (Giupponi and Vladimirova, 2006; Navulur and Engel, 1996).

MATERIAL AND METHODS

> STUDY AREA

The study area comprised most EU countries, but was limited by data availability. The study area covered EU27 with the exception of Sweden and Finland. The reference year was 2000. The area is best described by Figure 3.

> RIPARIAN AREAS

A riparian vegetation map covering EU27 was recently compiled from GIS and remote sensing data (Clerici *et al.*, 2013; Clerici *et al.*, 2011). From this data set the European riparian areas were mapped at a spatial scale of 25 m grid squares, combining hydrological and geomorphological data, land use data, including forest cover on a continuous scale of riparian-area-occurrence probability. The river network is that of the CCM2 (Catchment Characterization and Modelling) European river network database (Vogt *et al.*, 2007).

A pan-European wetland map of similar spatial detail does not yet exist. In the absence of this data, wetlands according to the Level 1 classification from CORINE land cover 2000 (EEA) were combined with the above riparian areas in order to provide a useful basis for calculating buffering capacity. For the present purpose, the riparian vegetation map was modified to also include riparian forest vegetation situated within agricultural areas (however, to avoid confusion no forest overlay occurred for the agricultural classes “fruit trees”, “vineyards”, “olive groves” and “complex cultivation patterns). The purely functional riparian areas (areas with no riparian vegetation observed but, nevertheless, of ecological value to wildlife) and areas situated over 2000 m a.s.l. were excluded from the analysis.

For this study, a 1 km^2 grid map of “average riparian buffer width” was created as follows: Riparian area per pixel of 1 km^2 was divided by river length per pixel. Assuming a symmetric distribution of riparian areas along both river banks, the value for a single river side can be obtained by dividing the value by 2.

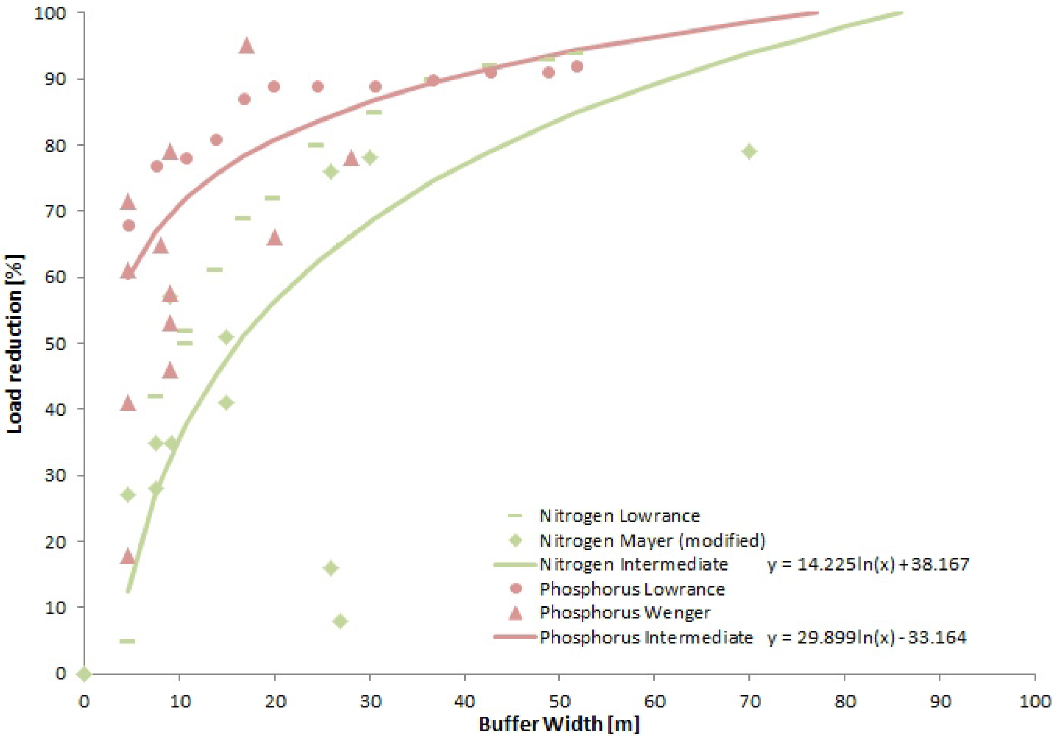


Figure 1
Literature-based load reduction rates for nitrogen (Lowrance et al., 2001 and Mayer et al., 2005) and phosphorus (Lowrance et al., 2001 and Wenger, 1999). The continuous lines represent the derived functions.

> NUTRIENT RETENTION

In the current literature, nitrogen load reduction rates are reported frequently but, for the underlying purpose of this study, two studies were particularly suited for the retrieval of a feasible model describing load retention as a function of buffer width. The first, a review study conducted by Mayer *et al.* (2005), distinguished surface flow from subsurface flow and collected 18 empirically gathered buffer width/load reduction data pairs. The second study was carried out by Lowrance *et al.* (2001) and contained modelled data by REMM for riparian buffers (comprising subsurface flow). The empirical data of the first study was more scattered than the modelled data but, by removing an outlier the coefficient of determination (R^2 = 0.40, using an exponential relationship) was improved remarkably. A considerably more linear distribution was observed for the REMM modelled data (R^2 = 0.95), although these data were not exclusively constrained to surface flow. To take advantage of both data sets and to mitigate their uncertainties, an intermediate function was derived to model the exponential relationship between buffer width and load reduction (see Figure 1).

The basis for the phosphorus load reduction model was derived, as for nitrogen, from both a model based simulation (Lowrance *et al.*, 2001) and from empirical data compiled in a literature review (Wenger, 1999). The scatter of the empirical values was high, and therefore an intermediate exponential model between the two major data sources was derived.

The basis for wetland occurrence is CORINE land cover (CLC) 2000. In the case of wetlands, the wetland body acts as a buffer itself. The relatively coarse scale of wetland mapping (25 ha minimum mapping unit), which results in few, large wetland bodies, led us to choose a buffer-width independent approach to model nutrient attenuation. This approach was based on the actual occurrence of a wetland only, regardless of its size. Natural wetlands are mapped only if they are predominant within a 1 × 1 km pixel, thereby filtering out smaller wetlands. Applying this procedure, only wetlands of a large extent are retained. The large wetland extent suggests

Table I

Widths and efficiency factors for riparian buffers and wetlands.

	Necessary buffer width for complete (100%) abatement [2]	Load reduction efficiencies (f)	
	Riparian buffer zones/strips		Wetlands
	Surface flow path	Subsurface flow path	
Nitrogen	86 m. Assuming cautious 75% efficiency (f) this results effectively as 86/0.75 ≈ 115 m	75%	Estimated 75% efficiency
Phosphorus	77 m. Applying cautious 75% efficiency (f) this results effectively as 77/0.75 ≈ 103 m	65%	Estimated 65% efficiency

[2] Estimated values applying a logarithmic function according to Figure 1.

the application of higher removal rates compared to those retrieved in literature, which are usually derived on small wetlands and relatively narrow vegetated ditches.

Nutrient removal rate for various constructed wetlands has been studied by Vymazal (2007), who reported mean efficiencies ranging between 41.2 and 54.8% for total nitrogen and values ranging between 41.1 and 59.5% for total phosphorus. The values for constructed wetlands are assumed to be comparable to those of natural wetlands. Fisher and Acreman (2004) collected data from 57 wetlands to study N and P removal efficiency and reported mean efficiencies of 67% for nitrogen and 58% for phosphorus. Due to the large extent of the CLC mapped wetlands we assume a slightly higher load reduction with fixed removal values of 75% and 65% for nitrogen and phosphorus, respectively.

Applied nutrient retention factors are listed in Table I.

> *BUFFER ABATEMENT MODEL*

The modelling approach adopted is depicted in Figure 2 and can be described as follows: firstly, diffuse emissions originating from mineral fertilizer and manure application, atmospheric deposition and scattered dwellings were calculated on a catchment scale by the GREEN model (Geospatial Regression Equation for European Nutrient Losses) (Grizzetti *et al.*, 2012), a simple statistical fate model. Retention processes within the catchment depend on annual rainfall rates, lumping together crop uptake, volatilization and denitrification. The utilized catchment database was based on CCM2 (Vogt *et al.*, 2007) where the average size of a sub-basin is 180 km^2. Land use data and crop shares were taken from the HYDE 3 database (Klein Goldewijk and Van Drecht, 2006), Global Land Cover 2000 (Bartholomé and Belward, 2005), the CAPRI database (Britz, 2004) and the SAGE database (Monfreda *et al.*, 2007). The CAPRI database was also the source of fertilizer application rates.

Then, the resulting diffuse nutrient loads were distributed to fertilized cropland and fertilized pasture (fCfP). The diffuse emissions were converted to 1 km^2 pixels, distributing each total catchment value of diffuse emissions into equal parts according to the number of a catchment's fCfP pixels. E.g. for a catchment value of 100 units containing 5 fCfP pixels: 100/5 = 20, each pixel gets a diffuse emission of 20 units. Since the aim of the study was an assessment of the impact of riparian vegetation on nutrient reduction, the contributing areas were assumed to be limited to the immediate vicinity of the river (within 1 km). Obviously, this assumption cannot be applied to subsurface flows, where nutrients can travel for long distances before reaching the watercourse.

To account for the different nutrient retention rates that can be applied in surface and subsurface flow, the flow was separated in different components using the SUGAR index (SUrface water/GroundwAtercontRibution index) (FOOTPRINT, 2008). SUGAR is an innovative hydrogeological index which provides information on whether water falling on a particular zone mostly contributes to groundwater recharge (*i.e.* infiltration areas) or to surface water runoff.

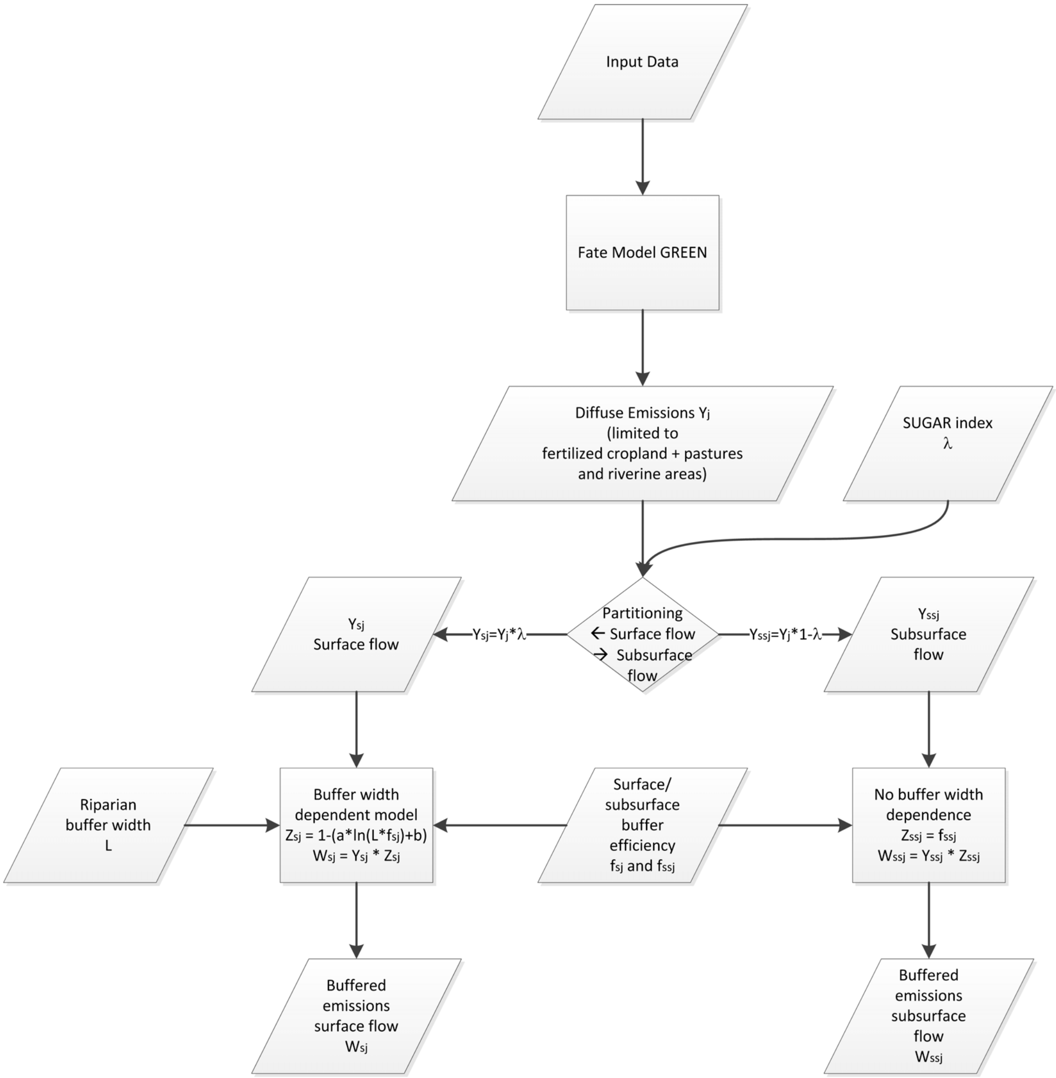


Figure 2
Flowchart depicting the data fluxes and the separate treatment of emissions for surface and subsurface flow pathways.

It is based on the combination of a Digital Elevation Model (DEM), a river network dataset, and soil data. The SUGAR index was used to allocate nutrient emission levels to water runoff or that infiltrating groundwater systems:

$$y_{sj} = Y_j \lambda \tag{1}$$

where Y is the diffuse load [kg ha^{-1} fCfP], s expresses the flow pathway (surface), j represents the specific nutrient and λ represents the SUGAR index [–], expressing the relative discharge to surface water.

Attenuation in surface flow is considered to be buffer-width dependent and is calculated according to equations (2) and (3) for nitrogen and phosphorus, respectively. Load reduction in subsurface flow is considered buffer-width independent and adopts a simple load reduction efficiency value summarized in Table I.

$$Z_{sN} = 1 - \frac{(29.899 \ln(Lf_{sN}) - 33.164)}{100} \tag{2}$$

$$Z_{sP} = 1 - \frac{(14.225 \ln(Lf_{sP}) + 38.167)}{100} \tag{3}$$

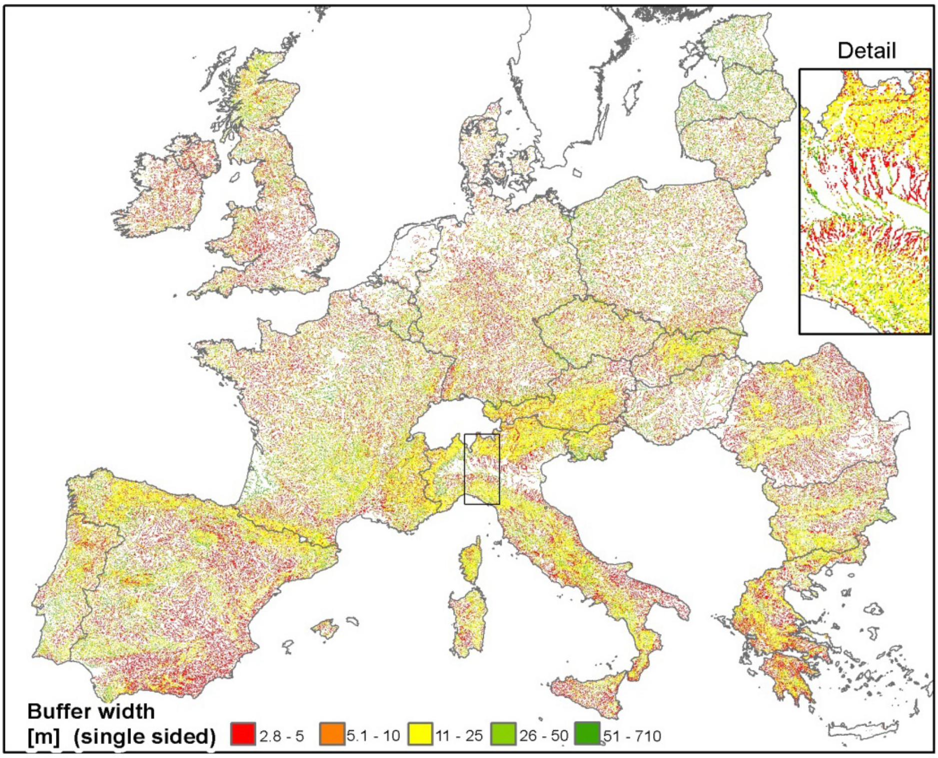


Figure 3
Average buffer width of riparian buffer zones/strips in m (single sided).

Z is the buffer zone permeability ranging between 0 and 1, s expresses the pathway (in this case s stands for surface pathway), N and P denote the specific nutrient, L stands for the buffer width in meters, and f is the nutrient specific efficiency factor. Potentially negative values are replaced by a minimum value of 1E-5 and values potentially greater than 1 are reset to 1.

The buffer zone permeability Z is equivalent to the fraction to which a buffer area (in this case a 1 km^2 pixel area) is permeable to nutrients. Z is a measure, to which degree a buffer area can be passed by nutrients to reach the adjacent river. A high Z value (close to 1) expresses high permeability and a low Z value (close to 0) corresponds to low permeability and hence strongly buffered areas. Buffer zone permeability Z is conversely related to buffer zone capacity B, which can be defined as a measure of a buffer area to abate incoming nutrient loads.

$$B_{sj} = 1 - Z_{sj}. \tag{4}$$

The streamside buffered emissions to the river can be calculated by simple multiplication.

$$W_{sj} = Y_{sj} Z_{sj} \tag{5}$$

where W is the total amount of nutrient j entering the river, and s, again, denotes the pathway (surface pathway).

The inclusion of all pixels i of a defined area allows the calculation of total nutrient reduction R or nutrient reduction rates r. For example, for the surface flow pathway the nutrient reduction R and the nutrient reduction rate r would be:

$$R_{sj} = \sum_{i=1}^{n} (Y_{sji} - W_{sji}) \tag{6}$$

$$r_{sj} = \sum_{i=1}^{n} \frac{Y_{sji} - W_{sji}}{V_{sji}} \tag{7}$$

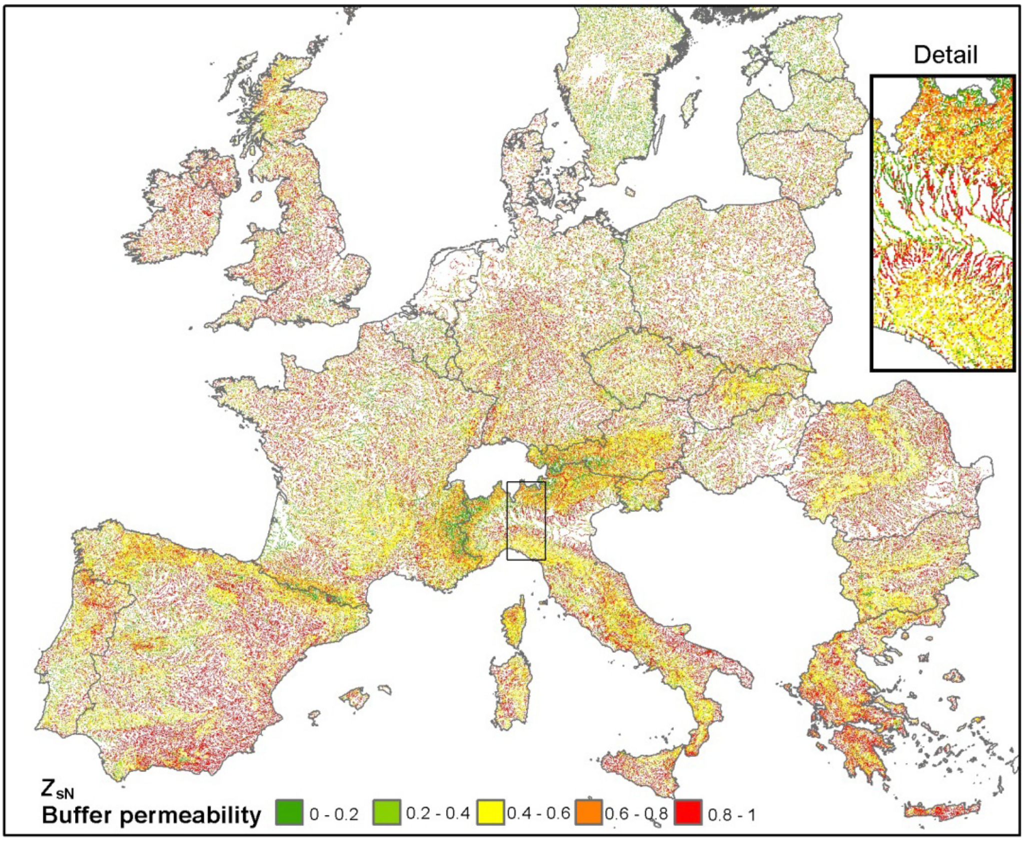


Figure 4
Buffer permeability for nitrogen following a surface flow pathway Z_{sN}.

RESULTS

The average buffer width along one bank for each km^2 of the study area is shown in Figure 3. Interestingly, buffer width in flat plains and hilly areas is either in the lower or in the higher ranges, while in mountainous areas intermediate values prevail. The extent of riparian buffer zones in flat areas is often restricted by cropland, whereas in mountainous areas riparian zones are usually constrained by geomorphological features. Gently undulating areas exhibit unexpectedly high observed buffer widths; these areas do not suffer from such intense agricultural pressures as flat plains, and the relatively forgiving geomorphology allows for wide buffer zones.

High values for buffer permeability *Z* (see Figure 4) were found in areas with a low presence of riparian vegetation, often intensely farmed areas or areas with a naturally low vegetation occurrence, such as semi-arid or arid zones. It should be noted that riparian zones were captured by a riparian area model (Clerici *et al.*, 2013) and needed to be of a minimum size in order to be recognised, *i.e.* larger than 25×25 m, which is the pixel size applied.

Comparing buffered nutrients entering streams *W* (Figure 5) and input loads *Y* (map not shown) highlights the importance of riparian buffers. The map of nutrient abatement (*B*), calculated as the difference between the loads (*Y*) *minus* nutrients entering the streams (*W*) is shown in Figure 6. For some regions of high input loads the buffering leads to a considerable reduction. This confirms the importance of considering riparian buffer zones when modelling nutrient emissions, as a significant proportion of nutrients can be retained by buffer zones.

A first indicative estimation of nutrient reduction regarding surface flow imissions for the study area is given in Table II. The table displays, for both nitrogen and phosphorus, indicative cumulative sums of diffuse emissions entering the riparian areas (*Y*), the fraction thereof directed to the surface flow pathway *(Ys),* the amount of buffered emissions entering the

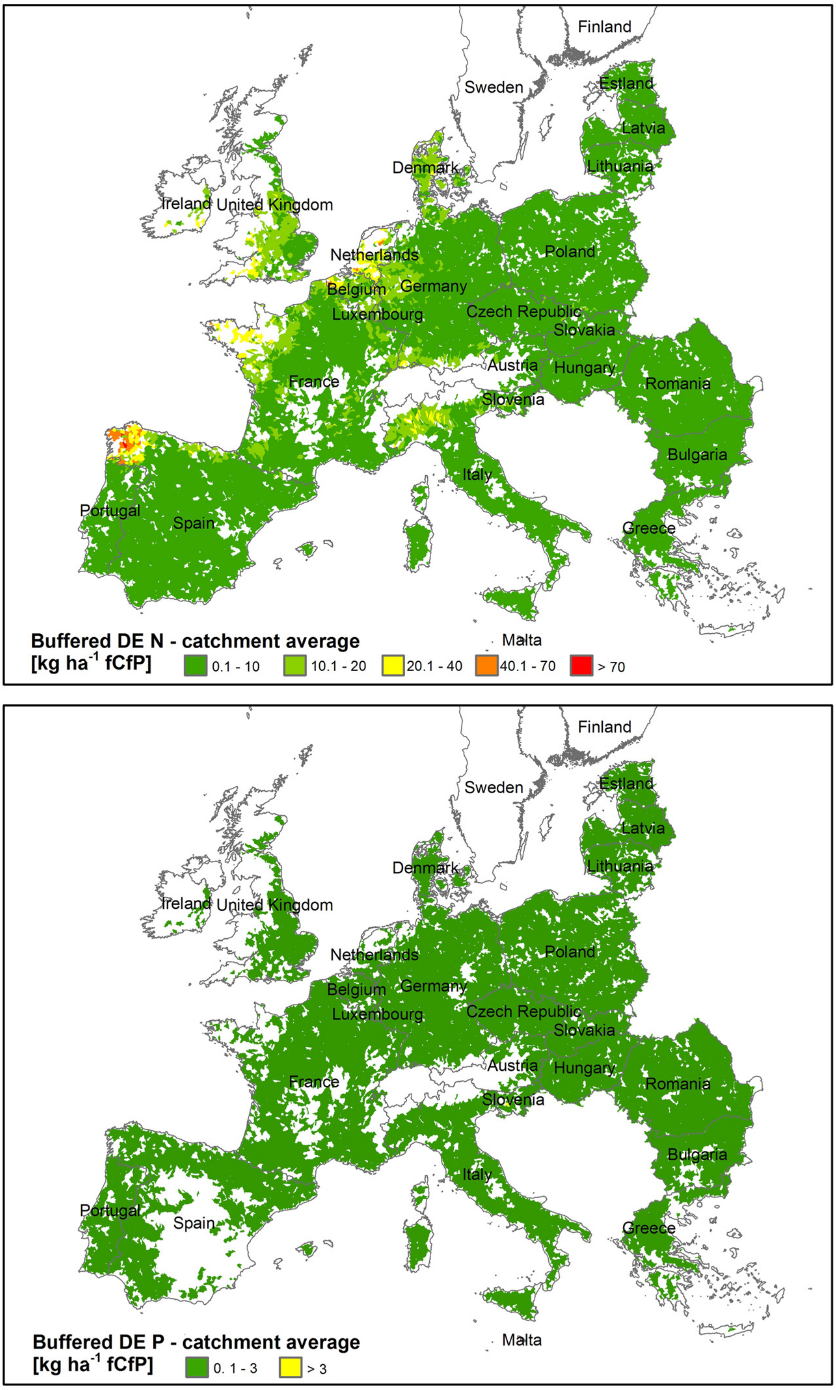
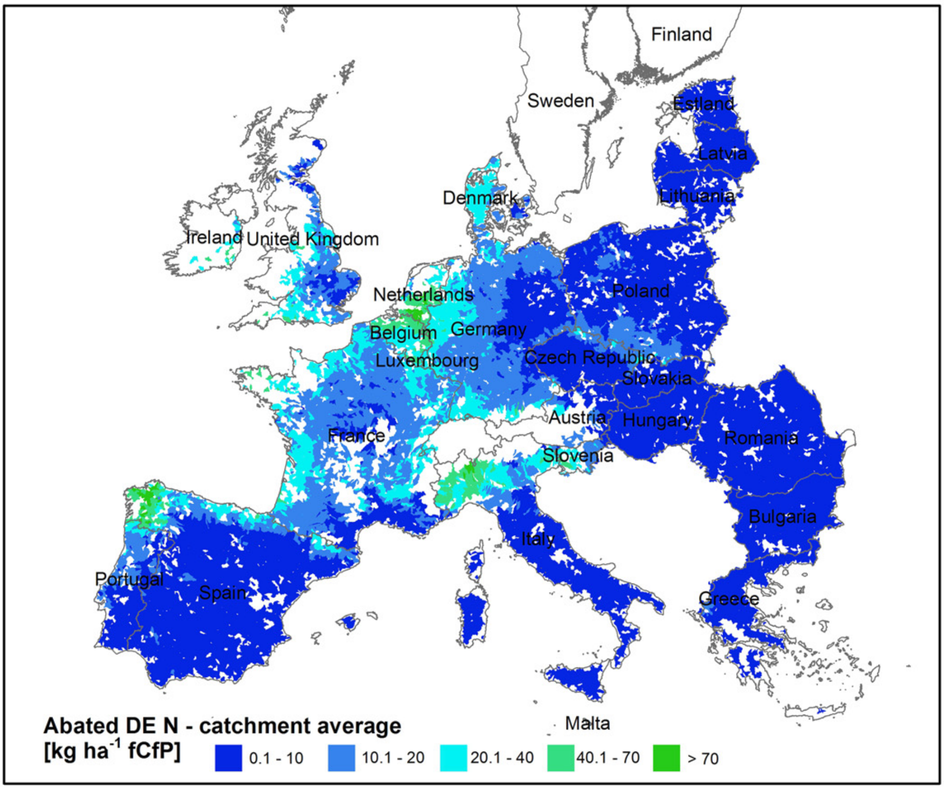


Figure 5
Buffered nitrogen and phosphorus emissions to streams per ha fertilized cropland + pastures in immediate vicinity of rivers (within 1 km*); averaged per catchment for better representation.*

watercourse *(Ws)*, and the reduction of surface pathway directed emissions as a percentage of *Ys*. The absolute numbers should be considered with caution as the sources of uncertainties are manifold and some of them cumulative (see Discussion section). In practice, it can be expected that due to concentrated flow paths the abatement values will range lower than the calculation shows (Table II). The reduction is lower for nitrogen (33%) than for phosphorus (65%), which reflects the distinct adsorption properties of each respective nutrient.



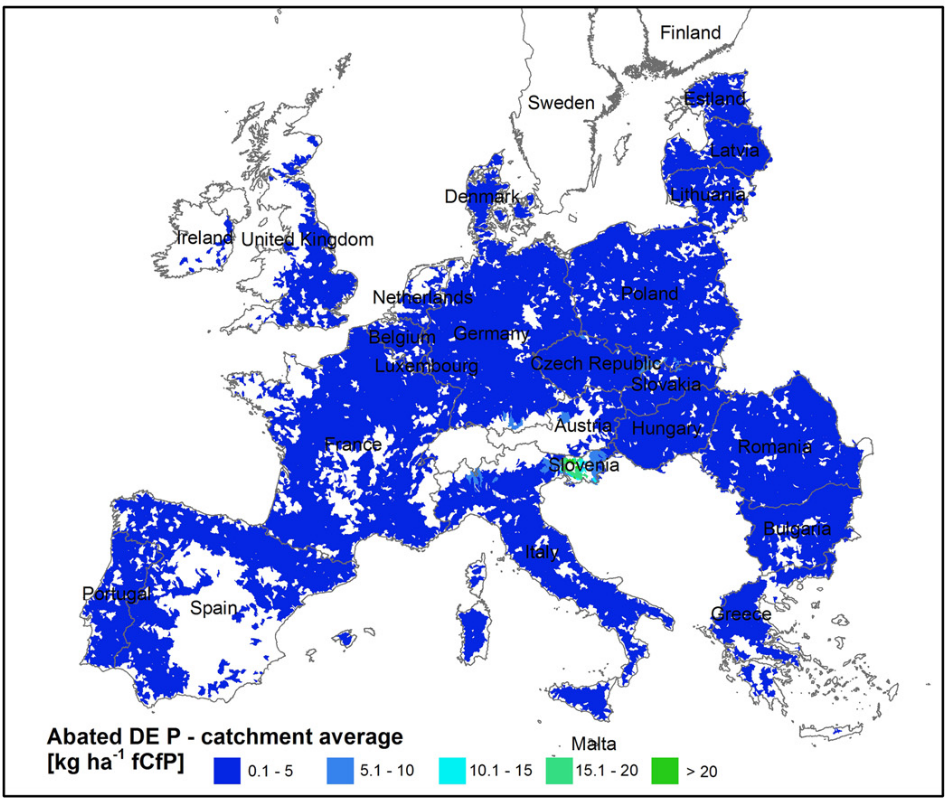


Figure 6
Nitrogen and phosphorus diffuse emission abatement via buffer zones, per ha fertilized cropland + pastures in immediate vicinity of rivers (within 1 km*); averaged per catchment for better representation.*

Table II
Cumulative statistics of nutrient input and retention (indicative values) when directed to the surface flow path-way, applied to the study area.

		Nitrogen	Phosphorus	Uncertainty
		tons		
Total diffuse emissions to riparian areas	*Y*	355 400	14 300	Moderate
Surface path way directed proportion thereof	*Ys*	161 000	6 300	Moderate, locally very high
Buffered emissions to streams (directed via surface pathway)	*Ws*	107 900	2 200	High[3], locally very high
Reduction of surface path way directed emissions		53 200	4 100	High[1], locally very high
		[%]		
		33.0	64.6	

[3] Uncertainty high mainly due to unknown presence of concentrated flow paths.

DISCUSSION

The presented results need to be discussed in the light of their accuracies, uncertainties and limitations. Obviously, calculations at this scale and extent require a high degree of simplifications, which in turn lead to inaccuracies that need to be considered when interpreting the results.
The following points illustrate main sources of uncertainties, which often are cumulative:

- Simplicity of buffer width based nutrient decay model: the spread of the reported literature based values is large: Pärn *et al.* (2012) talk about interquartile ranges of retention in riparian buffers of 66–89% for nitrogen and 3.8–84% for phosphorus. The data of Mayer *et al.* (2005) including others (Vought *et al.*, 1994) show remarkable spreads, *e.g.* Mayer's data with two independent nitrogen retention values for the same buffer width differing by 60 percentage points.
- Several data layers (*e.g.* soil data, slope) were not included in the model due to inappropriateness (*e.g.* due to coarse scale), unavailability or simply due to the fact that its impact would imply only minor changes compared to other shortcomings. Variable importance for modelling nutrient attenuation by riparian buffer zones was investigated by Bereitschaft (2007).
- A basic assumption of our model is that surface runoff occurs uniformly as laminar sheet flow through the riparian area. However, an important fraction of riparian areas are underperforming due to the presence of concentrated flow paths (CFP). Local studies (Knight *et al.*, 2010; Pankau *et al.*, 2012) showed CFPs may completely pass through the buffer in up to 69% of the cases, also depending on buffer width and vegetation cover. The non-inclusion of CFP data in the model will likely result in higher retention values.
- Riparian buffers are detected on pixels with a grid size of 25 m, which are then converted to a buffer width and pixels of 1 km^2. Buffers not captured by the riparian area model and very small buffers below detection limit are not considered. The pixel aggregation may level out a locally high or low presence of buffers. The river network is not including very small rivers. Also, the river network assumes higher drainage density in mountainous regions (Vogt *et al.*, 2007), which might have important implications for the detection of riparian areas, for the detected buffer width, and hence for the nutrient retention.
- The SUGAR index is derived from partly very coarse data (soil data, river network). Locally, the index might not be able to capture the real situation.
- The GREEN model is a statistical model with a high degree of simplifications. Although the model is designed for continental applications and has been validated its simplicity might bear inaccuracies.

The mentioned main sources of uncertainty should be seen in the frame of the model scale. It is clear that exact estimates of nutrient retention are almost impossible due to high uncertainties. The results should be seen with all its limitations, and it should be clear that those values are more of indicative or semi-quantitative nature, still allowing a targeting of critical areas, as the majority of shortcomings is occurring systematically. The uncertainty of the model outcome may be judged moderately high at a continental scale, but can be locally very high. This limits the use of data to targeting issues or as results of an explorative research.

The spatial resolution of the maps is a compromise between highly detailed riparian zones and nutrient load data which, at its best (after disaggregation), has a resolution of 1 km^2. Spatial detail is therefore determined by nutrient load data, making a downscaling of riparian zones necessary but, at the same time, reducing processing and handling time. However, identifying Europe's critical areas is achievable and such areas can be subject to further investigation on a more detailed scale. Much work has focused on the surface flow pathway of nutrients, while the subsurface flow pathway is treated with minor importance. Subsurface flow pathways are not determined by riparian buffer width and a comprehensive assessment of these imissions would require an assessment of all areas, including those beyond the immediate vicinity of river courses.

A validation *sensu stricto* of the modelled emissions entering streams requires surface water quality measurements to be taken both before entering and after leaving the buffer zone, which is problematic. To our knowledge, there is no database with these key data available. A validation using proxy water quality parameters, such as river quality data cannot be applied in this case since the modelled nutrient imissions reflect the outcome of a steady-state model with average precipitation, average nutrient fixation and deposition, and without considering the temporal distribution of rainfall or nutrient application dates, which in turn largely determine water quality parameters. However, the reported values stay within a plausible range: Dickson and Schaeffer (1997) have estimated the reduction potential for the Central Corn Belt Plains Ecoregion watersheds (USA). They assumed that for 35% of the study area (Illinois) subsurface drainage dominates, where riparian forests would have no further impact on the abatement. The reduction efficiency is assumed to be 85%, and the reduction potential for N for varying coverages of a fixed buffer of 25 m by riparian forest is reported. In comparison to our case, water falling on a site with relevant nitrogen input in the vicinity of a river contributes in average with 56.5% to subsurface drainage. The buffer width for the same sites is on average 16.13 m, which corresponds to 64.5% of a fixed 25 m buffer. According to Dickson and Schaeffer such a buffer coverage would result in approximately 55% abatement. Considering, that in our case the surface portion, which is abated by riparian areas, is 100 – 56.5 = 43.5% instead of Dickson and Schaeffer's 60% (100% – 35% – 10%/2; 10% is the assumed existing riparian area with no specification of drainage type, here assumed to be 50%: 50%), *i.e.* amounting at 72.5% of Dickson and Schaeffer's surface flow, the 55% abatement can be assumed to be only 72.5% of it, *i.e.* 39.9%. Our modelled average abatement for the study site and N is 33% and stays hence in the range of Dickson and Schaeffer's estimates.

Besides a first explorative quantification of buffered emissions entering streams, Figure 5 highlights critical areas for levels of diffuse pollution. These respective "hotspots" are zones where implementing additional measures aimed at improving water quality should be considered. Potential improvement measures could focus on the emission source (*i.e.* load reduction) or on increasing the buffering capacity of riparian zone, *e.g.* by creation or expansion of buffer zones. In other areas in Figure 5 no nutrient imissions are recorded at all, which is either due to a lack of fertilized cropland and pasture (*e.g.* in the Alps), a lack of diffuse emissions (*e.g.* large parts of arid/semi-arid Spain) or the complete abatement of emissions by riparian buffers.

CONCLUSIONS

The resulting maps allow not only a qualitative assessment of nutrient emissions entering watercourses, but an indicative and harmonized first estimate of how much, on average, riparian buffer zones can be expected to retain. Significant variations in imission patterns were observed for certain regions and we believe that by considering the effects of riparian buffer zones, an added value to the existing emissions data has been provided. Depending on the initial input load, in certain areas the buffering function of riparian zones is able to mitigate nutrient loads remarkably well.

The mapped results provide, in addition to a first explorative estimate of the amount of buffered emissions entering streams, indications of where to focus further efforts in order to improve water quality. These results represent a valuable data source for water-basin managers, allowing them to focus on priority areas, which is the most efficient course of action in terms of water quality improvements.

It is important to note that our estimation is not complete, since very small rivers and very narrow riparian areas are not captured by CCM2 or by the riparian area model. A model on a more detailed spatial scale would likely result in increased retention figures although we consider that this study already includes the most important retention areas.

The buffering capacity of a riparian zone is largely determined by the way nutrients enter the buffer strip. Shallow sheet-flow is the optimal way that nutrients can be captured within the buffer although suboptimal conditions prevail in many cases; a fact which can't be quantified so far but is partly being accounted for by the application of a cautious approach.

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