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Original Research Article

Modelling the role of ground-true riparian vegetation for providing regulating services in a Mediterranean watershed

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A B S T R A C T

Intensive agricultural and industrial activities are often considered major sources of water contamination. Currently, riparian vegetation (RV) is increasingly being promoted as a solution to balance the potentially adverse effects that agriculture may have on water quality. Nonetheless, existing RV is often overlook in recent modelling efforts, failing to capture the current amount of ecosystem services provide. Here, we used the Soil and Water Assessment Tool ecohydrological model to simulate the influence of ground-true RV on i) nutrient (nitrate and total phosphorus) and sediment exports from agricultural areas and ii) its effect for in-stream concentrations. These results are further compared against a set of hypothetical scenarios of different RV widths and different land-uses. Our results point to a great relevance of existing RV in controlling in-stream concentration of sediments and nutrients where pressure from agriculture is highest, preventing them to surpass limits set in the EU Water Framework Directive. On the other hand, in areas with industry discharges, the role of RV is limited and model results suggest that restoring RV would have limited impacts. We illustrate how existing RV may already provide strong but not acknowledged water quality regulation services, how these services can differ substantially between nearby streams, and that effective strategies to improve water quality using RV must acknowledge existing patterns of vegetation, land use and contamination sources.

1. Introduction

The presence of high levels of contaminants in water bodies is still one of the most serious environmental concerns worldwide (Galloway et al., 2008). In Europe, even after decades of research and political efforts such as the EU Water Framework Directive (WFD), water quality status is often classified as insufficient (EEA, 2018). This is often due to loads of chemicals into the streams from human activities, such as urban and industrial waste waters and agriculture. Chemicals reaching the water bodies are usually classified to their origin as point sources (e.g. discharges from wastewater treatment plants - WWTP), and nonpoint sources (e.g. resulting from agricultural activities (Carpenter et al., 1998)); nonpoint inputs are usually considered the main source of water pollution (Wang et al., 2012). Hence, agricultural systems play a major role as a source of water bodies contamination (Hildebrandt et al., 2008; Rasmussen et al., 2015), by frequently relying on the intensive use of fertilizers. Moreover, agricultural fields and poor land management practices promote runoff and soil erosion (Cerdà et al., 2009). Generally, chemicals from nonpoint sources are transported to nearby water bodies either in a dissolved form or coupled with sediments during heavy rains and erosion events (Holtan et al., 1988), even in areas with a gentle slope (Arreghini et al., 2005).

In the Mediterranean region, the low organic matter content,

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low levels of mineral nutrient and highly erodible soils, together with long periods of droughts, make the agricultural sector dependent on the high use of agrochemicals to meet production demands (Prosdocimi et al., 2016; Ryan et al., 2009). Alongside with the intensive use of machinery, these agronomic practices have degraded the quality of soil and water (Debolini et al., 2018; Zalidis et al., 2002), with negative consequences to the ecosystems (Ochoa-Hueso et al., 2011). Additionally, nonpoint sources of pollution rarely occur in isolation, and point sources can add pressure to an already unbalanced ecosystem (Jarvie et al., 2006). WWTP add-up to the non-point sources with continuously discharged effluents having a high proportion of contaminants immediately available to the ecosystem, even during the dry season, when the dilution effect is at its lowest (Bowes et al., 2009) and aquatic ecosystems are particularly sensitive (Rolls et al., 2012). There is a wide range of consequences for pollutants in the streams, which include the increase of the associated costs of water purification for human consumption (Parris, 2011), and the salinization and eutrophication of water bodies (Smith, 2003), leading to changes of plant and animal composition (Foy, 2005), followed by a restructuring of communities and a decrease in ecosystem productivity over time (Isbell et al., 2013), resulting in a lower capacity to sustain biodiversity and to provide ecosystem services (Sala et al., 2000).

Several solutions to tackle water pollution have been discussed in the literature. Nowadays, there is a vast array of nature-based solutions and best management practices (BMPs) that can be applied to reduce surface runoff to nearby streams. BMPs are known to regulate soil and water quality (decreasing contaminants and sediments from runoff), sequester carbon, support biodiversity, among others, generating high levels of ecosystem services in areas otherwise characterized by its disservices (Power, 2010). BMPs can either be applied to the soil itself to prevent contaminant or sediment mobilization, or on areas between fields and streams where the connectivity of water, sediment and nutrients can be limited (Keesstra et al., 2018). Riparian Vegetation (RV) act as a connectivity break, making it one of the most common and effective BMPs (and nature-based solution) available (Inamdar et al., 2001; Lee et al., 2003).

RV can be defined as “vegetated area set-aside from the main cropping regime … installed for the purposes of benefiting native biota, water and air quality, socio-economics, and yield” (Haddaway et al., 2018); and are known to remove sediments, nutrients and pesticides from surface water runoff by increasing surface roughness and hence limit surface runoff velocity, promoting filtration, deposition, adsorption and infiltration (Dillaha et al., 1989; Gumiere et al., 2011; Oshunsanya et al., 2019). RV areas are considered to be open to surrounding areas as they interact with physical, biological and human processes; and to be a hybrid system, since they are shaped by human and natural processes (Dufour et al., 2019). In agricultural fields, the riparian zone is characterized by high spatial and temporal variability, where human activities are seen as a major driver that shapes it (e.g. Brown et al., 2018; Dufour et al., 2015). Recently, Riis et al. (2020) reviewed and ranked the ecosystem services provided by these natural structures, concluding that the capacity to filter particles and control erosion figures in the top of most important ecological services provided. Moreover, such services are delivered in a disproportionately high amount relative to their extent in the landscape (see Sweeney & Newbold, 2014), making RV a highly effective BMP, particularly in agricultural watersheds (Chase et al., 2016).

A large number of field experiments have evaluated the local-specific effectiveness of RV. Recently, ecohydrological models incorporated this knowledge and have been successfully used to assess water quality and sediment transport from agricultural areas in multiple contexts (Rocha et al., 2020; Roebeling et al., 2014; Serpa et al., 2015), including the effectiveness of RV (Xie et al., 2015). The large majority of these studies focus on hypothetical scenarios of ecosystem restoration. Although evaluating such scenarios can arguably weight in decision making and promote awareness (Xie et al., 2015), they often overlook the existing distribution of RV in the landscape and, consequently, the ecosystem services they are already delivering.

Here, we applied the ecohydrological model Soil and Water Assessment Tool (SWAT) to a Mediterranean watershed characterized by the presence of extensive agricultural and agroindustrial areas. SWAT is considered to be an agriculture-oriented model (Xie et al., 2015), and has been applied to investigate multiple research questions related with agricultural and ecological field, including the effectiveness of a wide range of BMPs (e.g. Briak et al., 2019; Dechmi & Skhiri, 2013). Unlike previous applications of SWAT model that exclusively focus on hypothetical scenarios of best management practices (e.g. Giri et al., 2020; Jeon et al., 2018; Qiu et al., 2020), we specifically aim to characterize the importance of existing RV. More precisely, we assess the effect of existing RV in: i) the concentrations nutrients and sediments in the streams; ii) controlling nutrient and sediment exports from irrigated farmlands. We further compared the effectiveness of different management practices and RV expansions in exports from fields and in-stream concentrations.

2. Material and methods

2.1. Study area

This study was conducted in the Almonda Valley, including the Almonda and Alviela rivers, both tributary of Tagus River, located in central Portugal (Fig. 1). The climate of the study area is considered Mediterranean, with dry and hot summers (Koppen, Csa). The average rainfall exceeds 1100 mm in Serra de Aires, north-west of the basin (Arrimal and Minde meteorological stations) and a minimum of around 600 mm in the southern part of the basin (Chamusca meteorological station). Precipitation is irregularly distributed along the year, with more than 80% occurring between October and April. The mean temperature is approximately 15 °C (IPMA, 2018). The elevation in the study area ranges from 7 to 675 m, with an average slope of around 8% (derived from European Digital Elevation Model - EEA, 2017).

The study area can be divided into two distinct major classes regarding the land-use: the centre-north part, characterized by a more forested used, with olive groves (24%), Mediterranean shrublands (16%), forest (9%) and eucalyptus plantations (7%); and the centre-south part, characterized by agricultural uses, with relevance for irrigated farmlands (corn), winter wheat and oats (15, 10 and 7% of the total area, respectively) (DGT, 2018). Likewise, the northern part of the catchment is dominated by a karst formation and Chromic Luvisols soils (loams), while Calcic Cambisols (loams) and Eutric Fluvisols (sandy loams) soils dominate the centre and southern part, respectively (Cardoso et al., 1973) (Fig. 1). Karsts are formed by the dissolution of carbonate rocks by water, developing large pipes, interconnected cavities and cave systems where groundwater flow tends to concentrate (EC, 2003). These subterranean water flow paths can vary greatly in size, from tiny conduits to underground rivers (Hughes, 2018), which usually end as a spring (Smart & Worthington, 2004).

The study area is further characterized by an important tanning industry; and by several WWTPs across the main streams (Fig. 1 and Table A1). These WWTP vary widely in size, from just 2000 in population equivalent, to 400 000 (Alcanena) in population equivalent. Alcanena WWTP is located in a tributary of the Alviela

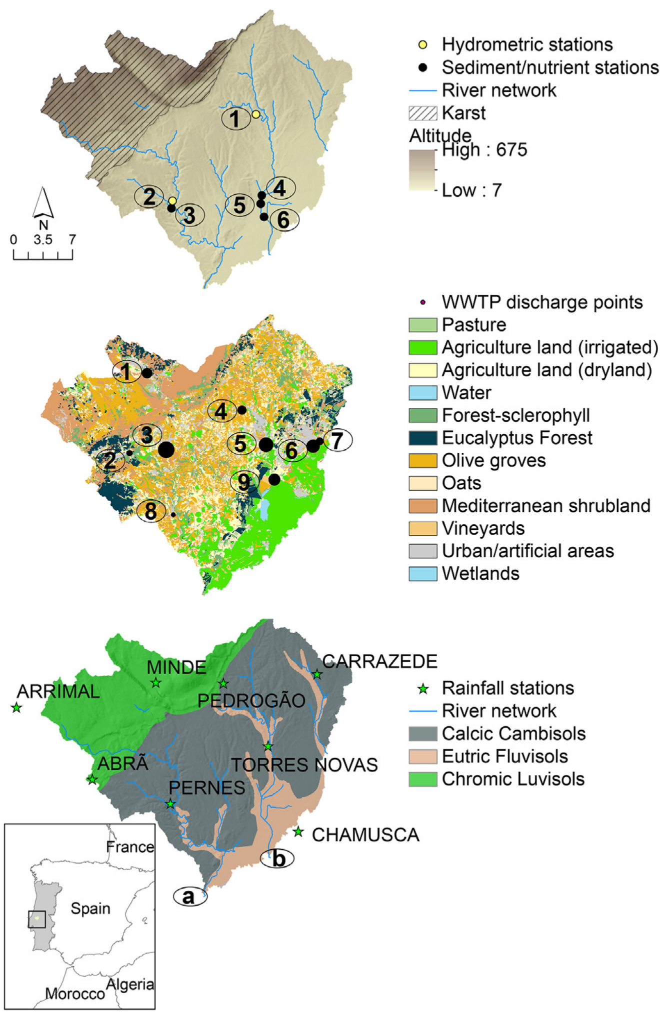

Fig. 1. Location of the study-area, with the representation of streamflow and water quality stations (top), land-cover (middle) and soil classification (bottom). Flow and water quality stations are identified in the left as follows: 1 – C.N.F.T – Torres Novas; 2 - Ponte Ribeira Pernes; 3 - Ponte Ribeira; 4 - Ponte Himalaia; 5 - Braço Cortiço; 6 - Quinta Broa (Norte). Waste water treatment plants (WWTP) are identified in the centre panel with size reflecting population equivalent (details on WWTP are provided in Table A1); Streams are identified in the right panel as: A) Alviela; B) Almonda. Rainfall stations are also identified in the right panel.

stream, and is responsible for the treatment of the effluents from the tanning industry.

2.2. Hydrological model: soil and Water Assessment Tool (SWAT)

Here, we applied the Soil and Water Assessment Tool (SWAT) (Arnold et al., 1998, 2012) to study nutrient and sediment exports from agricultural fields and in-stream concentrations. SWAT is a conceptually-based, semi-distributed and continuous hydrological model (Arnold et al., 1998), able to estimate surface and subsurface flow, sediment and nutrient exports and erosion at a daily scale, in a given catchment (Gassman et al., 2007).

SWAT divides the study watershed into multiple sub-basins, which are in turn subdivided into hydrologic response units (HRUs). HRUs can be labelled as smaller spatial modelling units that are dependent on the heterogeneity of land use, soil types and slope classes (Cibin et al., 2010). Runoff, sediment and nutrient outputs are simulated for each HRU; they depend on the input of climate data, topography, soil proprieties, land-use and land management practices. Hence, the total loading of a given sub-basin results from the underlying processes inside each HRU it encompasses.

2.3. Model setup

SWAT 2018 version was used and implemented using the QGIS interface and SWAT Calibration and Uncertainty Program (SWAT-CUP) (Abbaspour, 2015) with the stochastic method Sequential Uncertainty Fitting 2 (SUFI-2) (Abbaspour et al., 2004) (see below). Table 1 lists the inputs to SWAT model. Land-use was derived from the Portuguese land-use and land cover maps (DGT, 2018), which has a minimum mapping unit of 1 ha. Land-use attributes used for the simulation process were based on crop and urban database from SWAT and in previous simulations for nearby regions (Nunes et al., 2017; Serpa et al., 2015). Soil data was originated from the FAO 1:1,000,000 soil map (Cardoso et al., 1973) and combined with lithology maps by da Silva (1982) to enhance the delimitation between soil units. Topography was taken from EEA, at a resolution of 25 m (EEA, 2017), and later used to generate slope. Regarding climate, precipitation data was obtained from meteorological stations within the area of study (APA, 2019) and was harmonized and corrected by linear regression with data from E-OBS version 19 (Cornes et al., 2018), which is a valid infilling method for hydrological modelling (Ruman et al., 2020). Temperature was entirely derived from E-OBS version 19 dataset (Cornes et al., 2018). Potential evapotranspiration (PET) was calculated using the Hargreaves method (Neitsch et al., 2011).

Point source pollution from WWTP was introduced as continuous outflow throughout the year, apart from the Alcanena WWTP, where the annual outflow was disaggregated into seasonal patterns by comparison with water quality data downstream from the station. The generated load (population equivalent) of wastewater treatment plants were obtained from EEA (EEA, 2019; Table A1). Since no data regarding sediment and nutrient exports from WWTP is available for the region, these were estimated based on reference data (Economopoulos, 1993) and in accordance with known exports from other WWTP in the country (INSAAR, 2008).

Fertilizers were applied in the land-uses classified as corn, winter pasture, oats, winter wheat, vineyard and olive grove,

Table 1
Data used for SWAT setup and the respective sources.

Data	Source
DEM	European Digital Elevation Model (EEA, 2017)
Land-cover	Portuguese land use and land cover maps (DGT, 2018)
Soil	Portuguese Soil Map (Cardoso et al., 1973; da Silva, 1982)
Meteorological data (Temperature and Precipitation)	Portuguese Water Resources Information System (APA, 2019); E-obs v.19 (Cornes et al., 2018)
Hydrometric data	Portuguese Water Resources Information System (APA, 2019)
Sediment data	Portuguese Water Resources Information System (APA, 2019)
Nutrient data	Portuguese Water Resources Information System (APA, 2019)
Location and population equivalent of WWTP	Urban Waste Water Treatment Directive (EEA, 2019)

considering a known initial fertilization input at the start of the growing season and automatic fertilization to compensate nutrient deficits throughout the year (see Table A2).

2.3.1. Mapping and input of RV

RV was mapped for the areas where irrigated agriculture is dominant, using aerial photographs and land-use maps (DGT, 2018) (Figure A.1, A.2 and A.3). The RV was manually digitized over a high resolution ESRI World Imagery layer (ArcGIS Online data, Copyright © ESRI), obtained in 2018, with a spatial resolution of 0.6 m. Multipart polygons were digitized at a 1:1000 scale, with a maximum intra-patch aggregation distance of 10 m and a minimum mapping unit of 200 m^2. Field validation took place in late spring/early summer of 2019, to ensure that the digitized patches correspond to the ground-truth. A comparison between the RV mapped and the RV that existed between the period 2004–2006 (DGT, 2021) was carried out (Appendices C). This analysis allowed us to verify that the location of RV did not change between the periods, and hence, that RV mapped in 2018 can be used during the calibration and validation period.

SWAT model allows the user to introduce RV using a certain width of the RV, using the FilterW module (detailed in Appendices A). This method has been proven to be useful in previous application (e.g. Jang et al., 2017). Because SWAT is a semi-distributed model, it is not possible to place RV on its actual location. In order to define as precisely as possible the existing RV structure, we matched each area of RV with the corresponding HRU by discriminating between land-use and soil types individually assessing its width (and consequently the value used in FilterW). Hence, the value of FilterW can differ between sub-basins, but also within HRUs of the same sub-basin.

2.3.2. Model calibration and validation

The model was calibrated and validated for streamflow, sediment and nutrient (nitrate and total phosphorus) exports. The validated model was then used to evaluate in-stream concentrations of nutrients, following previous applications (e.g. Nunes et al., 2017). The calibrated model parameters for streamflow, nitrate and TP are available in Table B1. The simulated results were compared against the observed values in six different stations (identified in Fig. 1): two for streamflow (identified in yellow: 1 - C.N.F.T Torres Novas; and 2 - Ponte Ribeira Pernes); two for sediment exports (3 – Ponte Ribeira; and 6 – Quinta da Broa); and three for nutrient exports (4 – Ponte Himalaia; 5 – Braço Cortiço; and 6 - Quinta da Broa). The calibration and validation period was dependent on the data availability: for streamflow, Ponte Ribeira Pernes was calibrated for 1980–1989 and validated for 2002–2009 period; and C.N.F.T. Torres Novas was calibrated for 2002–2005 and validated for 2006–2009; for sediments Ponte Ribeira was calibrated for 2002–2004 and validated for 2006–2008 and Quinta da Broa was calibrated for 2002–2003 and validated for 2004–2006; finally, for nutrients, due to the limited number of samples, multiple stations were combined (but keeping the match between the watershed of the simulated values and the corresponding water quality station) and calibrated for the period of 1999–2003 and validated for 2004–2008. This multi-site and multi-variable calibration allows for more robust models (Daggupati et al., 2015). Prior to calibration, a warm-up period of 5 years was set to ensure model stability. The model was calibrated and validated with the existing RV areas; and the effect of RV areas was assessed by re-running the calibrated model without such areas.

The calibration process was divided into different steps: first, a manual calibration was conducted for streamflow and sediment yield (in accordance with the values reported in Cerdan (2010); see Figure B1), and later a smaller range of values for the parameters was given as input to the SUFI2 algorithm implemented in the SWAT-CUP automatic calibration software (Abbaspour, 2015), in order to refine the calibration. We ran 800 simulations for the individual automatic calibration process for streamflow and sediments (1600 in total), divided in four interactions of 200 runs each. Finally, nutrients were manually calibrated. We have used a temporal split-sample calibration and validation approach (Daggupati et al., 2015) where the parameters adjusted in the calibration are fixed for the validation. The model performance was assessed using the R^2, Nash-Sutcliffe efficiency (NSE) coefficient and the Percentage Bias (PBIAS), following guidelines provided by Moriasi et al. (2007, 2015).

2.3.3. RV and management scenarios

In order to characterize the influence of existing RV on nutrient and sediment in-stream concentrations and to compare the relevance of agricultural sources, a set of different scenarios were created:

- ‘Current’, representing the calibrated and validated model with the existing RV and urban and industrial effluents;
- ‘Current with only RV’, representing the existing RV but without urban and industrial effluents;
- ‘Current with only WWTP’, representing the existing urban and industrial effluents but without RV in the agricultural areas;

Table 2

Calibration and validation results at a daily and monthly time step for streamflow, nitrate (NO_3) and total phosphorus (TP). The value for NO_3 and TP were obtained with a combination of samples between station due to the lack of observations. Statistical performance measures are identified following Moriasi et al. (2015) **as not satisfactory (in bold)**; satisfactory (with single underline); good (with dashed underline); and *very good (in italic)*. Nutrients lack classification towards daily observations.

Parameter	Station	Time step	Calibration			Validation		
			R^2	NSE	PBIAS (%)	R^2	NSE	PBIAS (%)
Streamflow	Ponte Ribeira de Pernes	Monthly	0.84	0.80	**-24.7**	0.82	0.74	**-41.7**
		Daily	0.83	0.81	-	0.74	0.7	-
	Torres Novas	Monthly	0.79	0.61	**-61.7**	*0.93*	0.79	**-55.1**
		Daily	0.65	0.54	-	*0.87*	0.75	-
Sediment-flow	Ponte Ribeira de Pernes	Monthly	*0.88*	*0.88*	-6.9	0.69	0.64	-12.8
	Qt. da Broa	Monthly	*0.81*	*0.81*	*-0.11*	*0.88*	0.66	-3.9
NO_3	Multiple	Daily	0.56	0.47	-22.3	0.70	0.5	-34.3
TP			0.28	0.1	52.4	0.55	0.54	6.3

- 'No WWTP, No RV', representing a scenario without RV in the agricultural areas and urban and industrial effluents;
- 'Naturalized', representing a scenario without agriculture and human activities (pasture, timber, industrial and urban areas).

These scenarios were complemented by another set to evaluate the potential losses and gains from destruction or expansions of RV. Unlike the previous case, these scenarios were used both to characterize the influence in nutrient and sediment concentrations in the streams, and to study the influence of RV's width in controlling nutrient and sediment exports from irrigated farmlands. These scenarios consist of changing the width of RV to 0 (i.e. without RV), 0.25, 0.5, 1, 5, 10 and 20 m. We selected dense intervals at the lower end to test the effect of narrow RV.

3. Results

3.1. Model performance

At the stream network scale, SWAT predicted reasonably well daily and monthly streamflow, sediment-flow and nutrient-flow according to the performance statistics of R^2 and NSE defined by Moriasi et al. (2007, 2015). Model performance showed, however, a poor result for PBIAS in streamflow, which as considered not satisfactory for the validation period in both stations and during the calibration period for Torres Novas (Figure B2 and B3). Apart from nutrients, the model shows a better fitting for the calibration period than for the validation period. Sediment flow showed a constant underestimation in both stations during the periods of calibration and validation, although higher in Ponte Ribeira de Pernes (Fig. B5 and B6). The different nutrient compounds revealed divergent tendencies for over or underestimation of monthly exports (Table 2). In the case of nitrate, SWAT consistently underestimated monthly exports for both the calibration (PBIAS = −22.3%) and validation period (PBIAS = −34.3%; Figure B7, B8, B9 and B.13). The reverse was true for total phosphorus, for both the calibration and validation periods (PBIAS 52.4% and 6.3%, respectively; Fig. B10, B.11, B.12 and B.14). Overall, the model performs worse during low flows, as a result of the underestimations of simulated streamflow.

At the HRU (or hillslope) scale, average erosion predictions for different land covers in the study-area are within the intervals of those estimated by Cerdan et al. (2010) for Mediterranean Europe, despite tracking the lower end (Figure B1). These lower values of erosion are likely to be justified due to the low slopes: approximately half of the percentage of slope in the study area (8%) when compared with the study-areas in Cerdan et al. (2010) (15%).

3.2. Impacts of currently existing RVs on nutrient and sediment concentrations

RV were found to occupy an already important area in irrigated farmlands, frequently with a width higher than 10 m (Figure A3). Fig. 2 shows the concentration of nitrate, total phosphorus and sediment in the two streams of the study-area, under different scenarios. Overall, and as expected, the presence of industrial and agricultural activities showed a strong influence on the amount of nutrient exports: in all watersheds, scenario 'Current with only WWTP' presented the highest nutrient concentrations, followed by the 'Current' scenario', 'No RV, No WWTP', 'Current with only RV' and by 'Naturalized'. The fact that the scenarios with RV led to lower nutrient and sediment concentrations reflects the importance of the existing RV in balancing the exports from agricultural fields and counteract (to a certain degree) the influence of WWTP loads.

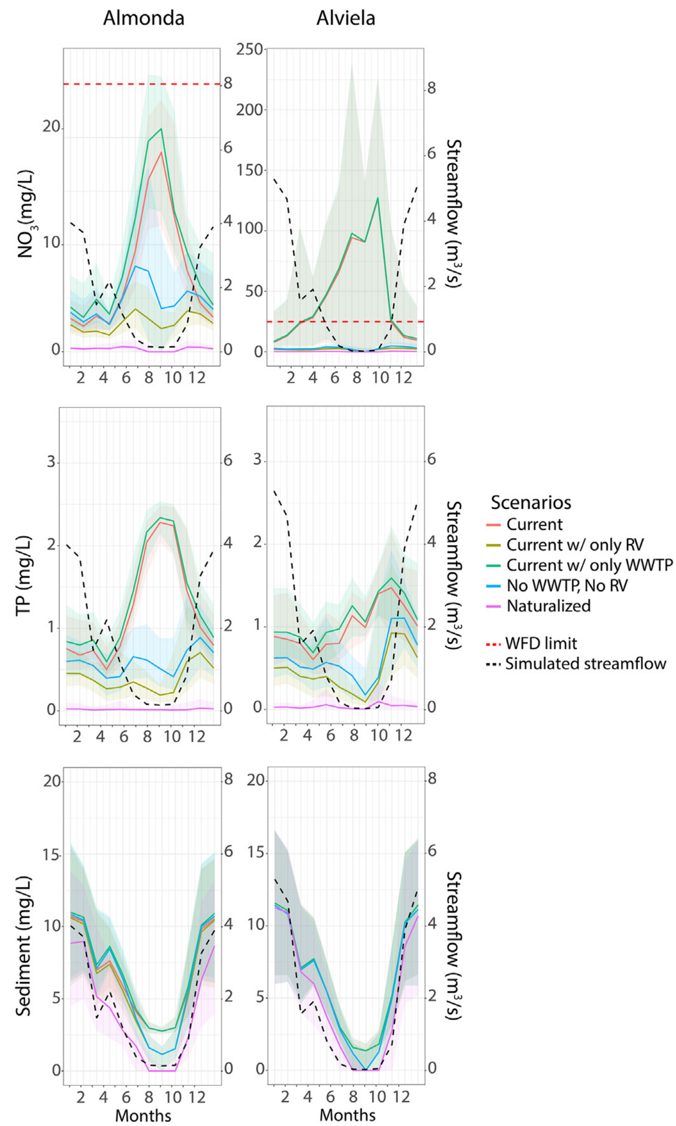

Fig. 2. Median nitrate (top), total phosphorus (middle) and sediment (bottom) concentration in Almonda and Alviela outlets per month, per scenario considered. Shaded areas indicate the Percentile 25 and 75 of each scenario. Dashed black line represents the median simulated streamflow; red dashed line represents the limit of 'good ecological status' of WFD. Note that all graphics have the same scale, apart from nitrate (NO_3) concentration for Alviela watershed.

Nitrate exports in the Almonda watershed, where intensive agricultural activities occupy a large area, showed a strong response to the presence of RV. In this case, the 75th percentile of nitrate concentrations are above the WFD limit for good quality status in the dry season for the 'Current with only WWTP' scenario, but they are below it for the 'Current' scenario. This result may indicate that existing RV is preventing recurrent and severe contamination problems. For the Alviela watershed, the presence of tannery industry dominated nitrate concentrations, especially during the dry season, and the efficiency of RV in removing these contaminants was residual. Finally, in the absence of urban and industrial activities (and hence without WWTP), nitrate concentrations showed two peaks in all streams, in spring and in autumn/winter, corresponding to the fertilization at the start of the growing period for summer and winter crops; instead of the present peak in the summer dry season due to the low dilution capacity for WWTP loads.

The patterns of TP concentration were similar to those of nitrates. For scenarios that do not consider urban and industrial pressure, TP concentration peaks at the beginning of each hydrologic year, during the initial rainstorms after the dry season; in this period, enhanced erosion transports remaining fertilizer phosphorus to the water bodies by adsorption to sediment particles.

The analysis of sediment exports reveals a considerably lower influence of WWTP than for nutrient exports. In the Alviela the existing RV areas do not seem to influence sediment exports. This can indicate the sediment transport capacity of streamflow is already at its maximum for Alviela river, and changes in HRUs exports will not alter the 'Current' scenario, as the increased sediment deposits in the streambed. Moreover, the overall low slope in the lower reaches of the streams decreased the potential capacity of the river to transport sediments. Another factor, also derived from the low slope in the agricultural area, is that the sediments in the streams were mainly originated from erosion in the uplands and hence far from the influence of RV. As expected, sediment concentrations were higher during the wet season due to erosion and sediment transport promoted by precipitation; and WWTP influence for sediment exports is more relevant during drier months.

3.3. Impact of RV width on nutrient and sediment concentrations

Fig. 3 shows the influence of different widths of RV on in-stream nutrient and sediment concentration. The width of RV impacted the concentration of nutrients and sediments found in the water bodies, with scenarios with narrower RV leading to higher in-stream nutrient and sediment concentration. Regarding nitrate concentration, RV showed a higher importance for Almonda watershed, although relatively small. Furthermore, RV efficiency (i.e. capacity to reduce component concentration) was found to be constant throughout the year. On the contrary, Alviela watershed showed little change in nitrate concentration, which is related with the large volume of effluents from industrial activities.

Regarding TP, the increase in RV width was found to have small effect in reducing TP concentration in Almonda and Alviela during the rainy season. This is, again, related with the association between soil erosion and TP transport adsorbed to sediment particles. For this reason, wider RV also led to a small decrease in sediment concentration during the wet season in Almonda; this was not the case in Alviela, possibly due to sediment transport saturation.

3.4. Effect of RV on nutrient and sediment exports from farmlands

Fig. 4 shows the effect of the real and hypothetical scenarios of RV width on nutrient and sediment exports. Generally, existing RV are present in most of the landscape, with a width often greater than 10 m (vertical dashed lines in Fig. 4; see Figure A1 and A3). The creation of hypothetical scenarios allowed the delineation of trend lines and the evaluation of the rate of change in nutrient and sediment exports following changes in RV width. As shown in Fig. 4, wider RV promote lower exports, with a value of around 90% reduction for all variables found for 20 m wide RVs. As expected, the importance of RV varies between HRUs, as some have higher nutrient and sediment exports than others (e.g. due to higher slope). Moreover, the impact of RV width decreases with larger widths, with an inflexion point between 2 and 5 m; even an RV of 0.5 m seems to have a large impact compared with no RV.

4. Discussion

4.1. Model performance

The SWAT model was successfully calibrated and validated for

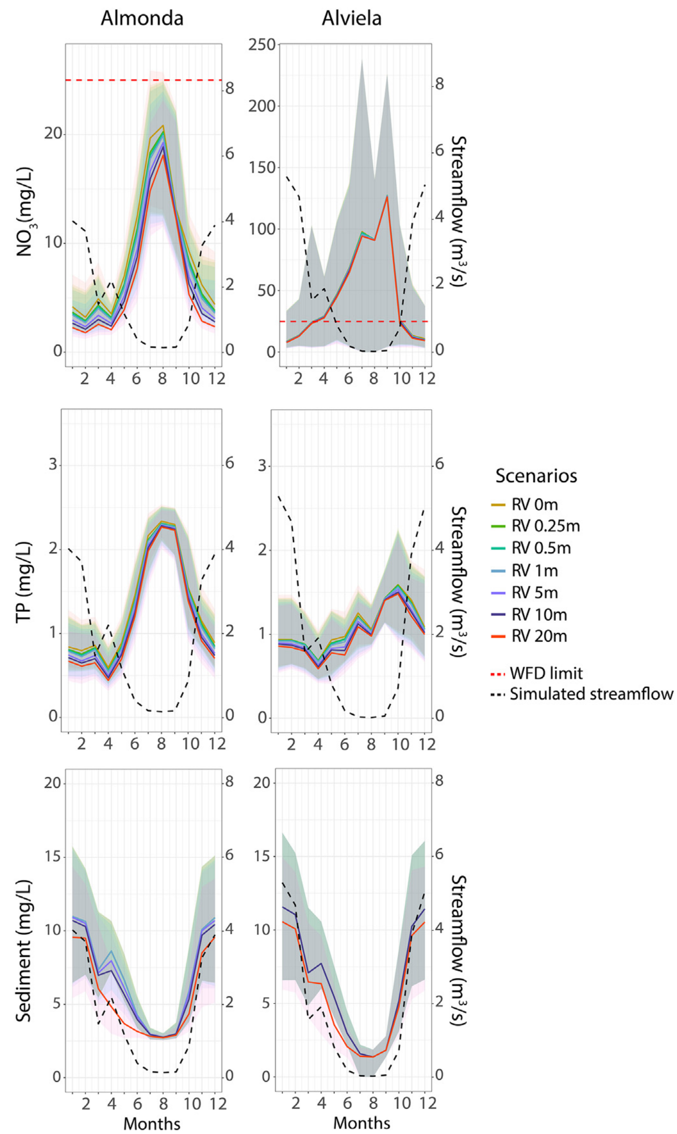

Fig. 3. Median nitrate (top), total phosphorus (middle) and sediment (bottom) concentration in Almonda and Alviela outlets per month, per scenario of riparian vegetation width considered. Shaded areas indicate the Percentile 25 and 75 of each scenario. Dashed black line represents the median simulated streamflow; red dashed line represents the limit of 'good ecological status' of WFD. Note that all graphics have the same scale, apart from nitrate (NO_3) concentration for Alviela watershed.

nutrient and sediment flow, despite important underestimation of streamflow. This underestimation may be explained by a set of different factors. The most important is related with the underlying karst in the upper reaches of the watershed, as previous studies have found similar challenges in using hydrological models in karst formations (Hartmann et al., 2014). Although some methodologies exist to deal with such challenges (e.g. Baffaut & Benson, 2009; Malagò et al., 2016), they often rely on the extensive characterization of the landscape, with the mapping of sinkholes and springs. Other factors include the resolution of the DEM, which may have a large impact on the underestimation of streamflow in flat areas such as the Almonda agricultural valley (Rocha, Duarte, et al., 2020); and the short time series available for streamflow (namely for the Torres Novas station), which may limit the variability testing of model parameters (Lespinas et al., 2014; Piras et al., 2014). Despite these challenges, a multi-site and multi-variable calibration and validation strategy can allow for a greater robustness and lesser

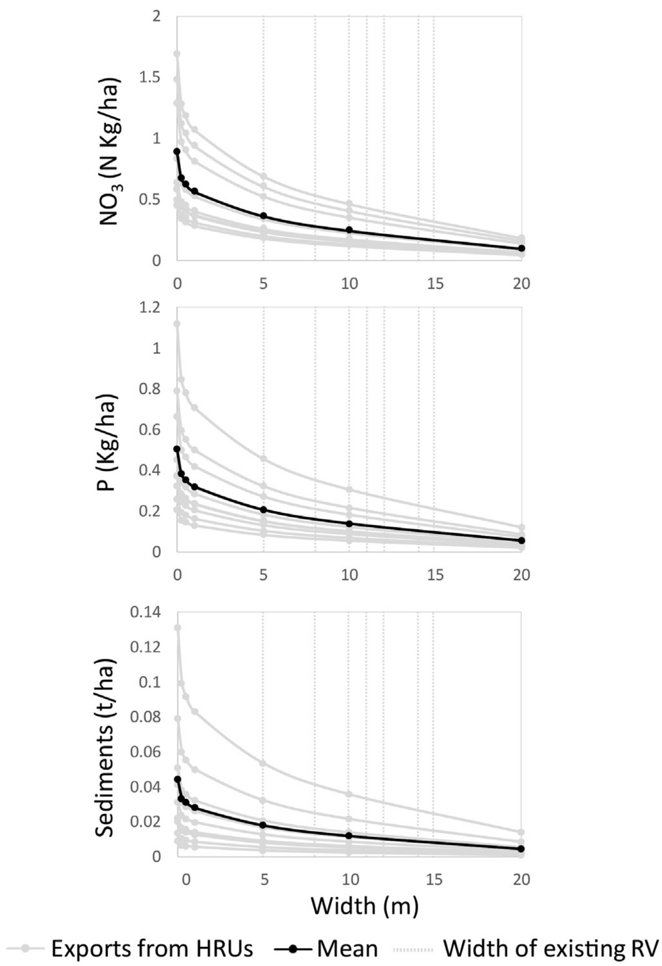

Fig. 4. Monthly mean values of nitrate (top), total phosphorus (middle) and sediment (bottom) exports in surface runoff per HRU (solid grey lines) and the mean for the study-area (solid black line). Values are showed in kilograms per hectares for nitrate and total phosphorus and in tons per hectare for sediments. Dashed vertical lines represent the width of the existing riparian vegetation in multiple HRUs.

equifinality in areas with a diversity of characteristics, such as this study area (Bergström et al., 2002; Beven, 2006; Briak et al., 2019; Daggupati et al., 2015) In particular, this complex calibration and validation strategy increases the robustness of the parameterization of RV effects.

Regarding nutrient and sediment flow, the calibration and validation performed satisfactorily. We were able to capture the annual patterns of nitrate and TP in both with the presence of WWTP (nitrate and TP peaking during summer – see Shore et al., 2017; Duncan et al., 2017) and in their absence (nitrate peak twice a year - in summer and in winter; and TP peak once a year, following start of a new hydrological year).

4.2. Scenario analysis

The system's characterization showed that existing RV occupy a fairly large extension of the interface land-streams in irrigated farmlands, with a width generally larger than 10 m. This network of RV is a remnant of the original RV vegetation but also likely results from the creation of drainage ditches and agricultural practices from irrigated cropping and rice pad waterlogging, insuring an increase in wetness across the agrolandscape. As it is, existing RV probably plays an important role in controlling nutrient and sediment exports from farmlands as runoff and consequently, their concentration in the nearby streams, as indicated by the modelling results with different RV widths. A meta-analysis conducted by Zhang et al. (2010) revealed that a 20 m wide RV would increase nitrogen and phosphorus removal efficiency by around 90%, while for sediment the same value of removal efficiency would be reached with 10 m wide RV. The values obtained when applying the SWAT model were similar to the ones reported in Zhang et al. (2010) for nitrate and TP removal; for sediment, we obtained around 90% of removal efficiency for 20 m wide RVs, which can be linked with the lower slopes on farmlands and hence lower erosion potential. Even acknowledging some diversity in reported values (e.g. Mayer et al., 2007), 20/30 m wide RV are suggested as reasonably effective for removal of nutrients from surface runoff (Sweeney & Newbold, 2014), supporting the applicability of the RV module in the SWAT model.

However, the impact of existing RVs on nutrient and sediment concentrations in the stream network was only relevant in the Almonda watershed, where results indicate that they might be preventing a recurrent summer exceedance of WFD reference values. This is due to the large area occupied by intensive agriculture combined with limited WWTP discharges. In the Alviela watershed, however, the strong presence of tannery industry dominated nutrient concentrations, despite the presence of intensive agriculture, corroborating the conclusions of Jarvie et al. (2006). For this watershed, nitrate concentrations were above WFD limits throughout the majority of the year, reflecting a well-known and long-lasting problem of pollution (Aquanena, 2019). Finally, we found a relative small effect of both RV and WWTP on sediment concentration in streams. This may be due to the fact that agricultural areas have a small slope (and hence small erosion rate) when compared with upland areas that, for this reason, are the major contributors to in-stream suspended sediments (Huisman et al., 2013); or due to saturation of the sediment transport capacity of streamflow. Moreover, sediment transport capacity is lower in flat areas (Xiao et al., 2017), which is the case of the study-area.

These results are limited by our inability to model the structure of RV (e.g. grass or forest), density growth and vigour (e.g. degraded or restored, plant biodiversity), or other critical chemical factors for RV on spatially continuous terms (Adel et al., 2018; Zhao et al., 2020). This distinction would have likely influence the magnitude of results presented, since both factors have different degrees of importance in filtering particles and controlling erosion (Aguiar et al., 2015; Nóbrega et al., 2020; Riis et al., 2020). Moreover, the SWAT module used does not spatially allocate the mapped RV to their exact site, but to the HRU (resulting from the combination of land-use and soil types) inside each subbasin. Moreover, SWAT assumes that each HRU is linked with a channel, forcing the allocation of RV to individual HRU. This abstract representation is consistent with SWAT's semi-distributed approach to spatial discretization. We acknowledge that relevant information may be lost in the process of modelling the effect of RV. This reflects the need for a more comprehensive module of SWAT (or other tools) that specifically target these limitations, offering more options when modelling riparian buffers.

4.3. Implications

Despite a good understanding by the farmers of the ecosystem services generated in their farmlands, and their recognition of the major threats (see Smith & Sullivan, 2014), pressure from a wide range of anthropogenic activities (e.g. pollution, land-use change; see Riis et al. (Riis et al., 2020)) threaten and jeopardize such services. It is estimated that around of 80% of natural RV have been destroyed in Europe over the past two centuries (Naiman et al., 1993), and existing RV ecosystems may be further exposed to the observed and projected increased frequency of streamflows below ecological thresholds (Pascual et al., 2015). Hence, the protection of existing riparian habitats (e.g. by resorting to schemes paying for ecosystem services - Smith & Sullivan, 2014) and reforestation actions are needed to face such threats (Riis et al., 2020). Our results illustrate this issue for agricultural farmlands, where already existing RV provide a neglected and unrecognized ecosystem service by maintaining water quality below WFD thresholds in the Almonda watershed.

Our results also support the case to focus future restoration efforts on sites without or with narrow RV, following their high potential to deliver ecosystem services. In fact, even though restoration often requires higher investment than preservation, the ecosystem services provided by those restored RV areas often exceed the restoration cost (Daigneault et al., 2017; Uggeldahl & Olsen, 2019).

5. Conclusions

The Soil and Water Assessment Tool was applied to the Almonda Valley, including the Almonda and Alviela rivers, highly impacted by agriculture and tannery industry, to determine the effectiveness of existing RV to control nutrient and sediment concentrations in streams and their exports in the runoff from irrigated farmlands. RV were found to occupy a fairly large extension of the land-stream interface, with a width generally larger than 10 m. Our results show that existing RV may already provide strong but not acknowledged water quality regulation services by preventing recurrent and severe contamination problems. For instance, in the absence of RV, the simulated 75th percentile of nitrate concentrations during the summer in the Almonda river would rise above the WFD limit for good quality status, highlighting the importance of existing RV for water quality in areas of intensive agriculture. On the other hand, RV showed a low influence in areas with large impacts of industry discharges on water quality, regardless of the presence of intensive agriculture, and these problems should be addressed with specific measures.

Our results emphasize how sources of pollution can differ substantially between nearby streams and that effective strategies must acknowledge them. Hence, and following recommendations from previous works (e.g. Gücker et al., 2006; Shore et al., 2017), we exemplify the need for both efficient and diluted discharges from industrial facilities (and urban areas) and for refinement of agricultural practices (with the expansion and promotion of RV) to reduce nutrient loads.

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Author contributions

BAA, JPN, LFD and TF contributed to the planning of the paper. AF mapped the riparian vegetation. BAA and JPN led the work. BAA applied the SWAT model. JPN, LFD and LB supervised the model application. BAA and JPN prepared the figures and the associated analysis. BAA and JPN wrote a first version of the manuscript. All authors reviewed the manuscript.

Declaration of competing interest

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

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.iswcr.2022.07.005.

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