Agriculture as a source of phosphorus for eutrophication in southern Europe

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Abstract

Large areas of the southern European countries possess a Mediterranean climate, which influences soil properties, land use, fertilizer application practices and pathways of phosphorus (P) loss from agricultural soils. On average, inputs of fertilizer P exceed P exports from the agricultural areas in these countries; however, large differences in P surplus/deficit and soil P test values exist among regions. Losses of P in drainage water are modest except in some irrigated areas and in those regions where intensive animal production is concentrated. Losses of P in runoff water, whether as dissolved reactive or particulate P, can be substantial as a result of the significance of erosive processes under the land uses typical of the Mediterranean region, where extreme rainfall events contribute disproportionately to such losses. Eutrophication due *sensu lato* to agricultural P sources affects a relatively high proportion of rivers, lakes and reservoirs. The typical Mediterranean climate and patterns of land use result in marked seasonality in the concentration of P in surface waters. Despite the growing number of studies conducted, the contribution of agricultural P to eutrophication in southern European countries remains largely unassessed and thus warrants pertinent research at the soil, field and catchment scales.

Keywords: Eutrophication, mediterranean, phosphorus runoff, southern Europe

Introduction

This study is concerned with those European countries that possess a Mediterranean climate at least in part of their territories (i.e. countries bordering or close to the Mediterranean Sea). Relatively little attention has been paid to areas in such countries possessing a humid temperate climate because the patterns of eutrophication by agricultural P in such areas resemble those observed in central and northern Europe. The Mediterranean climate, with its typical warm dry summers and cool wet winters, prevails over an area of about 700 000 km^2 in southern Europe and encompasses a variety of subclimates; thus, summer droughts can last for 2–5 months, mean annual temperatures can range from 9 to 20 °C and the mean annual rainfall from 300 to 1500 mm. While, strictly speaking, the areas bordering the Mediterranean Sea possess such a climate, the presence of various mountain ranges introduces changes in atmospheric circulation resulting in certain areas not far from the sea having a humid temperate climate (e.g. the Po Valley in northern Italy). Indeed several major rivers flowing into the Mediterranean (e.g. River Rhone) have a humid temperate climate over most of their basins.

Mediterranean and neighbouring European regions constitute a complex climatic, orographic and soil mosaic that exhibits a wide array of land uses. Generally, agriculture is not as intensive as that in central and western Europe, which suggests a decreased risk of eutrophication by P; however, areas where traditional low-input, extensive agriculture is practised coexist with areas of intensive horticultural or livestock production and the magnitude of the eutrophication problem has increased over the last decades in parallel with rapid changes in agricultural technology and production systems. The increasing use of mineral fertilizers in areas of extensive agriculture, the increase in irrigated land area, the concentration of intensive livestock production in small geographical areas,

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and the failure to curb water erosion and runoff in vulnerable areas have played significant roles in this respect.

Public awareness of the importance of eutrophication by agricultural P in Europe was stimulated by the EU 'Nitrates Directive' of 1991 (Council Directive 91/676/EEC), given that pollution by nitrates and phosphate are related. Although the degree of implementation of this Directive varied between southern European countries (European Commission, 2002), the prescribed measures were useful towards mitigating agricultural P loss to water. One further step in this direction was the adoption of the 'Cross Compliance' measures of the EU Common Agricultural Policy (CAP), the degree of implementation of which also differs greatly among countries – and so certainly do farmers' attitudes towards the adoption of such measures.

In this cursory review, we examine available data on the sources of agricultural P, pathways for agricultural P loss to water and water quality in the southern European countries with a view to assessing the magnitude and trends of the eutrophication problem, the nature of the corresponding mitigation possibilities and future research needs.

Soils and soil use

Despite their high diversity, the soils in Mediterranean areas can be distinguished from soils in other European climatic regions, which justifies designating them 'Mediterranean soils' (Torrent, 2005). For various reasons, these soils tend to be more clayey than soils in humid temperate areas, and hence exhibit a higher water holding capacity than the latter. Stoniness and structural instability are widespread in cultivated soils, which can be partly ascribed to their low contents of organic matter. pH is mostly in the slightly acid to moderately alkaline range as a result of limited leaching. More than half of the soils in the Mediterranean region are calcareous or exhibit accumulation of secondary calcium carbonate in the soil profile. In contrast to many soils in the humid temperate regions where poorly crystalline Fe and Al oxides and organic complexes usually predominate in the P-sorption complex, crystalline Fe oxides, carbonates and, to a lesser extent, silicate clays are the most important P-reactive minerals in Mediterranean soils (Peña & Torrent, 1990).

In southern European countries, permanent pasture and permanent crops occupy about 30 and 15%, respectively, of the agricultural area. Grapevines and olive trees are the most important permanent crops in non-irrigated areas and cereals (mostly wheat and barley) are grown in rotation with legumes, sunflower or fallow on non-irrigated arable land. The irrigated area has increased steadily over the last few decades, with Spain (3.8 Mha), Italy (2.8 Mha), France (2.6 Mha) and Greece (1.4 Mha) accounting for more than 90% of the total area (FAO, 2005). Major crops in these areas include maize, sugar beet, rice, citrus, peach, pears and apples.

Sources of agricultural phosphorus

Except in France, where most agricultural land has a humid climate, the consumption of P fertilizers in the last few decades has been relatively modest (Table 1). Compared with the 1961–2002 average, P consumption is currently greater in Portugal, Spain and Greece (by 20–40%), and smaller in France and Italy (by 40–50%); therefore, current consumption figures are relatively similar in all countries (approx. 8–12 kg P ha^{-1} agricultural area $year^{-1}$). The amount of P in animal excreta (Table 2) is currently of the same order of magnitude as that in mineral fertilizers. The data in Tables 1 and 2, and the current P fertilizer application rates, allow us to estimate surplus P inputs in the region of about 5 kg P ha^{-1} $year^{-1}$ for Greece, 10 kg P ha^{-1} $year^{-1}$ for Portugal, Spain and the former Yugoslavia; and 15–20 kg P ha^{-1} $year^{-1}$ for France and Italy. These average figures, however, conceal the important fact that greater surpluses occur in areas of intensive horticulture and where animal production is concentrated or composts and agro-industrial residues (e.g. olive-mill waste) are applied. Thus, Sacco *et al.* (2003)

Table 1 Average use of phosphorus fertilizers in some southern European countries (1961–2002)[a]

Country	(Gg P_2O_5 $year^{-1}$)	Agricultural area in 1990 (Mha)	Average application rate (kg P ha^{-1} $year^{-1}$)[b]	Crop removal (kg P ha^{-1} $year^{-1}$)[c]
Portugal	69	3.96	7.6	7.8
Spain	467	30.47	6.7	7.3
France	1414	30.57	20.2	16.6
Italy	562	16.84	14.6	12.1
Former Yugoslavia	169	14.08	5.2	7.6
Albania	14	1.12	5.4	7.9
Greece	141	9.22	6.7	5.7

[a]Data from FAO (2005). [b]Based on the 1961–2002 yearly average and the 1990 agricultural area. [c]Harvested and grazed in 1989 (Sibbesen & Runge-Metzger, 1995).

Table 2 Livestock density[a] in some southern European countries

Country	Cattle (ha^{-1})	Chickens (ha^{-1})	Pigs (ha^{-1})	Sheep (ha^{-1})	Turkeys (ha^{-1})	P in excreta ($kg\ ha^{-1}\ year^{-1}$)[b]
Portugal	0.34	8.45	0.54	1.33	1.69	12
Spain	0.22	4.24	0.79	0.79	0.03	9
France	0.65	6.77	0.51	0.31	1.18	15
Italy	0.44	6.48	0.60	0.52	1.62	13
Former Yugoslavia	0.24	3.22	0.60	0.29	0.16	8
Albania	0.61	3.77	0.10	1.58	0.46	12
Greece	0.07	3.32	0.11	1.07	0.01	3

[a]Based on the number of livestock in 2004 and the total agricultural land in 2002 (FAO 2005). [b]Based on 16, 0.7, 6, 0.15, and 0.45 kg ha^{-1} $year^{-1}$ for cattle, sheep, pig, chicken, and turkey, respectively.

Table 3 Estimated phosphorus surplus for different farm types in the Po Valley, Mugello (central Italy) and Sardinia

Region	Farm type	P surplus ($kg\ ha^{-1}\ year^{-1}$)	Stocking rate ($t\ ha^{-1}$ live weight)
Po Valley	Non-livestock farms[a]	22	0
Po Valley	Cattle farms[a]	82	2.63
Po Valley	Traditional livestock farms[a]	104	3.36
Po Valley	Dairy farms[a]	112	3.55
Po Valley	Pig farms[a]	103	4.27
Mugello	Cattle farms[b]	6	0.43
Mugello	Dairy farms[b]	11	0.69
Sardinia	Sheep farms[c]	24	< 1.0

[a]Sacco *et al.* (2003). [b]Argenti *et al.* (1996), [c]Caredda *et al.* (1997).

reported surplus P inputs of up to 112 kg P ha^{-1} $year^{-1}$ for intensive farms in the Po Valley, where 4 million units of cattle, 7.1 million of pigs and 126 million of poultry are concentrated. In contrast, Argenti *et al.* (1996) reported much lower values for Tuscany, and so did Caredda *et al.* (1997) for Sardinia (Table 3). Generally, areas of extensive grazing or cereal dry-farming have a modest P surplus or are in balance.

The phosphorus status of soils

The differences in P surplus between the agricultural areas in southern European countries are due, among other factors, to differences in land use and climate, and are obviously reflected in the results of soil P tests. For instance, the proportion of Spanish soils with very high Olsen *P* values was higher for regions with a humid temperate climate and/or where the soils were irrigated than for dry-farming areas (Figure 1); also, in some regions of Northern Italy, the soil

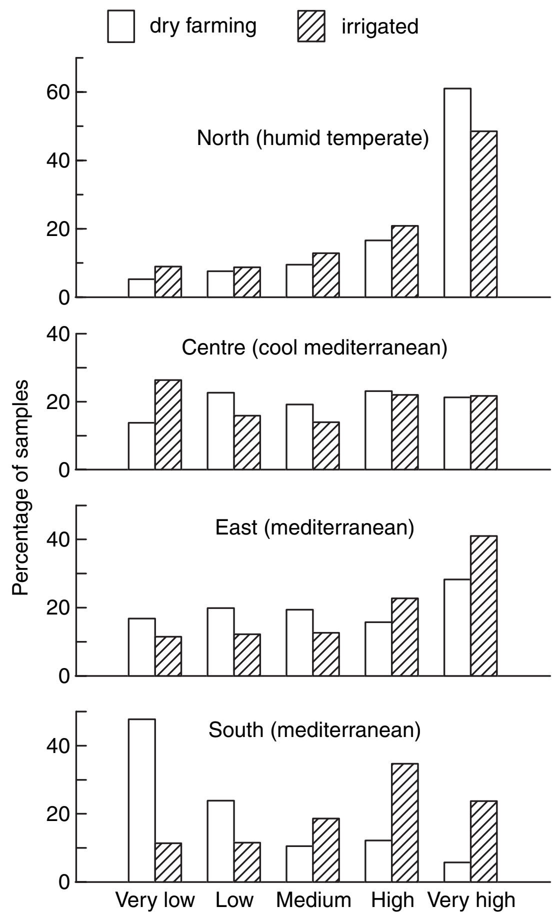

Figure 1 Proportion of agricultural soil samples falling in the very low, low, medium, high and very high fertility classes according to the Olsen P test in various regions of Spain under dry farming or irrigated agriculture (2000–2005 period). The limits of each class depend on soil type, crop and land use; as average values, < 10 = very low, 10–15 = low, 15–20, 20–35 = high and > 35 mg P kg^{-1} = very high. The prevailing climate in each region is stated in parenthesis. Based on data supplied by Fertiberia SA.

Table 4 Proportion of soils falling in the different fertility classes in various agricultural regions of northern Italy

		Piemonte[a]					
Fertility class	Olsen P (mg kg^{-1})	Fruit orchards	Herbaceous	Rice	Grapevine	Lombardia[b]	Emilia Romagna[c]
Low	≤8	21	15	2	47	12	22
Medium	8.1–14	14	13	9	20	14	25
High	14.1–21	14	15	20	12	16	22
Very high	> 21	51	57	68	21	58	32

[a]Unpublished data from LAR–Torino, n > 20 000, 1995–2005. [b]Unpublished data from MAC–Vertemate con Minoprio–Como, n = 5552. [c]Unpublished data from Servizio Sviluppo Sistema Agro-alimentare dell'Assessorato Agricoltura-Emilia Romagna, n > 43 000.

status was strongly dependent on the particular crop (Table 4). In agricultural areas of Portugal, the proportion of soils falling into high P classes was generally greater than that of soils in low or medium P classes; and levels consistently increased from 1980–1988 to 1990–2002, with significant differences among regions (Table 5). In all cases, the proportion of soils testing high for P indices was relatively high.

Estimation of the degree of phosphate saturation

The degree of phosphate saturation (DPS), defined as the ratio of sorbed P to the P sorption capacity (PSC) of the soil, is a measure of the ability of the soil solid phase to adsorb/desorb P from/into the soil solution in contact. Following the seminal idea discussed by van der Zee *et al.* (1987), many authors have used the molar $P_{ox}/\alpha(Al_{ox} + Fe_{ox})$ ratio, where the subscript 'ox' is short for 'oxalate extractable' and α is generally equal to 0.5, as an index for DPS. This is based on the assumptions that the P-sorption capacity of the soil is determined by poorly crystalline Fe and Al phases and that P_{ox} represents the P reversibly sorbed by the soil. These assumptions are reasonably accurate for acid and coarse-to-medium textured soils, but fail to explain the behaviour of soils that are calcareous and/or contain substantial amounts of lithogenic apatite (Uusitalo & Tuhkanen, 2000), or in which, like many Mediterranean soils, P sorption is determined by crystalline Fe oxides, carbonates and silicate clays.

Table 5 Percentage of soils falling in the different P fertility classes in various agricultural regions of Portugal[a]

	Very low and low[b]	Medium[b]	High and very high[b]
Açores	18	43	40
Beja	20 (45)	50 (23)	30 (32)
Braga	3	28	68
Bragança	16	46	38
Castelo Branco	17 (51)	39 (24)	43 (25)
Coimbra	13 (43)	32 (20)	56 (37)
Évora	22 (65)	53 (17)	25 (19)
Faro	32	30	38
Guarda	15 (34)	45 (23)	39 (44)
Leiria	17 (45)	33 (16)	50 (38)
Lisboa	15	32	53
Porto	3	22	75
Santarém	12	30	58
Setúbal	12 (38)	34 (18)	54 (44)
Viana do Castelo	10	39	51
Vila Real	28	40	32
Viseu	17	40	43

[a]Data from Adubos de Portugal (private communication) for the 1993–2002 period and Soveral Dias *et al.* (1989) for the 1980–1988 period (in parenthesis). [b]As determined using the Egner-Riehm method (Egner *et al.*, 1960): < 12.5, 12.5–25, 25–100, 100–200, and < 200 mg kg^{-1} correspond to the very low, low, medium, high, and very high P fertility classes.

In practice, one can use the ratio of the value of a soil P test which is reasonably related to the amount of desorbable P, to the PSC of the soil [estimated by direct measurement of the P sorption curve (e.g. Borling *et al.*, 2004) or from appropriate pedotransfer functions (Peña & Torrent, 1990; Borggaard *et al.*, 2004)] as an index for DPS. Various soil P tests have been used for this purpose. Thus, Borling *et al.* (2004) used Olsen P (Olsen *et al.*, 1954 – the test most widely used in several Mediterranean countries), together with the PSC obtained from the sorption curve, to establish an index for DPS. It should be noted that, on the basis of the soil P test alone, one can also estimate a 'change-point' (viz. the threshold value above which losses to water are significant), as previously done by some authors following the proposal of Heckrath *et al.* (1995). A recent example of this approach was provided by Horta (2005), who estimated Olsen P change points for a group of Portuguese soils. The change points ranged between 20 and 56 mg kg^{-1} depending on the pathway of soil P loss (which was mimicked in the laboratory) and soil properties, which suggests the need for calibration in each soil P loss scenario.

Based on P sorption/desorption curves, Torrent & Delgado (2001) provided evidence that the concentration of dissolved reactive phosphorus (DRP) in the 1:1 soil:water extract is useful in predicting the release of soil P to water in various P desorption scenarios and, more important, across a wide range of European soils with a low to medium P sorption capacity, including some in the Mediterranean region. The potential of these and other approaches needs to be investigated in Mediterranean soils, where the use of the oxalate extraction-based DPS generally lacks a sound basis.

Pathways of phosphorus loss

Losses of P in drainage water from agricultural soils are generally small in the Mediterranean region because leaching is limited and the subsurface horizons of many soils are relatively rich in effective P sorbents (clay, Fe oxides and calcite). It should be noted, however, that the winter surplus of water can be substantial some years because of the great variability in winter rainfall. Thus, in Córdoba (southern Spain; 37.8°N, 4.8°W; mean annual temperature = 18 °C; mean annual rainfall = 630 mm), the calculated amount of water leached from a soil with a water holding capacity of 125 mm was > 100 mm in 44 years and > 300 mm in 15 years of the 1901–2000 period. This makes P losses possible, especially when preferential water movement occurs. Indeed, significant P losses in drainage water can also be expected from irrigated soils or in areas with a more humid climate. Thus, the water from underground drainage systems on typical farms of the Eastern Po Valley was found to contain from 0.021 to 0.162 mg DRP L^{-1} (Rossi *et al.*, 1992).

Table 6 shows such losses for a sprinkler-irrigated, reclaimed marsh soil near the estuary of River Guadalquivir (south-western Spain). As can be seen, an increase in the leaching fraction resulted in a significant increase in the losses of total P (TP), dissolved reactive P (DRP), total dissolved P (TDP), and particulate P (PP = TP − TDP). Losses can be relatively high in cracking soils (Vertisols and Vertic Cambisols), particularly when irrigation water flows into the cracks.

Substantial losses of organic P forms – usually greater than those of DRP – occur in areas of high animal density or where manure, cattle/pig slurry and/or biosolids/composts are applied to the soil. A study of the Terra Chá area (province of Lugo, north-western Spain), where the climate is humid and cattle slurry is commonly applied to pasture, showed the presence of significant amounts of organic P, but low DRP concentrations, in well water (López Periago, 1993), as a result of the soil profile possessing a high P sorption capacity (García-Rodeja & Gil-Sotres, 1995). In Cambisols and Luvisols [the World reference base for soils (FAO & ISRIC, ISSS, 1998) is used throughout] of the western Po Valley, the TDP concentration in the soil water was relatively high (0.045–0.081 mg L^{-1}) in the deepest horizons, where organic P was the most abundant form (up to 92% of TDP). This suggests that organic P can make a significant contribution to the P load to groundwater. Both DRP and organic P concentrations reached a maximum in June and a minimum in January, which was ascribed to seasonal variation in manure use and in P mineralization and immobilization rates combined with seasonal changes in the solubility of organic matter (Barberis *et al.*, 2001).

Unfortunately, few data exist on the concentration of organic P in the groundwater of many critical areas. However, the presence of nitrate and various potability indices have raised concern about the presence of organic P in groundwater. Based on such indirect indices, organic P is probably present at significant concentrations in many areas of intensive livestock production. For instance, pollution cause by agricultural activities affects the groundwater of about 430 000 ha in Catalonia, north-eastern Spain (Departament de Medi Ambient i Habitatge, Generalitat de Catalunya, 2006), where more than a quarter of the Spanish pig production is concentrated.

There are few studies on the losses of P from rice fields (approx. 420 000 ha in southern European countries in 2004) or their dependence on the quality and management of irrigation water, soil characteristics and P status. Low DRP concentrations (0.02 ± 0.06 mg L^{-1}) were found in water at the outlet of rice fields south of the Albufera fresh water lake near Valencia (Spain), an area where irrigation water is not significantly affected by urban or industrial effluents

Table 6 Concentration of forms, and losses of phosphorus, in drainage water in a pipe-drained marsh soil (Eutric Gleysol)[a] near the estuary of the River Guadalquivir (south-western Spain) in three different growing seasons with different irrigation water management methods (D. Hurtado and A. Delgado, private communication)

Crop	Drainage fraction[b]	DRP (mg L^{-1})	TDP (mg L^{-1})	TP (mg L^{-1})	DRP (g ha^{-1})	TDP (g ha^{-1})	PP (g ha^{-1})
Sugar beet	0.03	0.068 ± 0.008 (0.024–0.165)[c]	0.116 ± 0.026 (0.039–0.451)	0.123 ± 0.066 (0.032–0.479)	16	19	4
Cotton	0.07	0.046 ± 0.026 (0.002–0.204)	0.054 ± 0.027 (0.003–0.215)	0.081 ± 0.07 (0.003–0.744)	34	37	17
Cotton	0.35	0.039 ± 0.024 (0–0.215)	0.053 ± 0.024 (0.014–0.261)	0.077 ± 0.048 (0–0.551)	113	154	98

[a]0 to 30 cm, the soil contains 70% clay, 24% calcium carbonate equivalent, and 19 mg kg^{-1} Olsen P. 30 to 90 cm, 47% clay, 35% calcium carbonate equivalent, and 12 mg kg^{-1} Olsen P. Before planting, 70 kg P ha^{-1} was applied to the soil. [b]Drainage fraction is the fraction of irrigation water lost through drainage during the growing season. [c]Mean ± standard deviation (range). DRP, dissolved reactive P; TDP, total dissolved P; TP, total P; PP, particulate P (TP − TDP)

(Soria *et al*., 2002). Rossi *et al*. (1997) monitored TDP and TP inputs and effluents from rice fields in a large area in northern Italy and reported that the water leaving the fields exhibited no increase in TDP relative to the inflowing water in about half the monitored area. In Italy, where about 40% of the rice cropping area is on sandy soils, Maio *et al*. (2003) calculated that, for a total annual input of 8500–11 000 m^3 ha^{-1} water, 56–61% was lost by evapotranspiration, with surface water outflow and net percolation accounting for 6–10 and 29–39%, respectively, of applied water. Therefore, there is a need to model leaching and runoff of P from paddy fields with a view to assessing environmental impacts, as already done for some pesticides (Capri & Maio, 2002; Maio *et al*., 2003).

In general, P losses via erosion and surface runoff markedly outweigh those in drainage water because of various climatic, topographic and agronomic factors. In cultivated areas, relatively few annual species survive after the dry summer season, so the early autumn rains, often as intense showers, fall on unprotected soil and cause runoff and soil erosion. In addition, the scarcity of water during the growing season forces farmers not only to remove weeds that compete with crops for water and nutrients, but also serve to protect the soil surface against erosion. The long history of human settlement has moved croplands to areas of steeper slopes, where the risk of erosion is high. Soil loss rates are highly variable; in the Iberian Peninsula they range between <10 and >100 t ha^{-1} $year^{-1}$. In a catchment in the Po valley, with a mean slope of 0.61%, runoff has caused relatively large soil (6.7 t ha^{-1} $year^{-1}$) and P (3.8 kg ha^{-1} $year^{-1}$) losses (Zavattaro *et al*., 2003). At river basin scale, sediment yields are less because of the gradual deposition of the detached material along water flow paths, with values estimated to range from 0 to 20–30 t ha^{-1} $year^{-1}$ (J.V. Giráldez, private communication).

Soil management has a strong influence on erosion, runoff and P loss. For instance, Gómez *et al*. (2004) recorded annual soil losses in an olive grove of 8.5 t ha^{-1} in non-tilled weed-free soil versus 4.0 t ha^{-1} from a conventionally tilled soil, and only 1.2 t ha^{-1} from soil under a grass cover. Adopting conservation tillage systems, in which the soil surface is protected by at least 30% residue cover, limits soil and P losses. These systems, which are slowly gaining acceptance among farmers, are practised in 1, 6, 14 and 17% of the total agrarian surface of Portugal, Italy, Spain and France, respectively (ECAF (European Conservation Agriculture Federation), 2006).

Heavy rains result in much runoff and loss of particulate P, not only in autumn, but also in late winter or early spring, a time when soils are moist or saturated in wetter years, if the soil does not have crop cover. As can be seen in Table 7, the losses of P caused by one rain event in a Chromic Vertisol catchment in southern Spain were close to 10 kg P ha^{-1} (mainly PP), because the soil was saturated and 90% of the rainfall was transported in overland flow. Losses were smaller for a similar event on a Calcic Luvisol catchment (Table 8), because soil moisture content was lower and topsoil infiltration rates were greater than in the Vertisol. In

Table 7 Overland flow, sediment and phosphorus losses during various rain events in a Chromic Vertisol catchment[a] in southern Spain (Data from Jódar 2003).

Date	Rainfall (mm)	Runoff (mm)	Sediment loss (kg ha^{-1})	Total P loss in runoff (kg ha^{-1})	TDP loss in runoff (kg ha^{-1})
11 January 2002	24	3.3	290	0.18	0.001
21 March 2002	55	50	14 500	9.66	0.022
17 April 2002	31	13	635	0.46	0.009

[a]Catchment surface: 61.5 ha. Topsoil properties: clay content 440–500 g kg^{-1}; pH 7.4–7.9, Olsen P 10–85 mg kg^{-1} (10–30 mg kg^{-1} in 80% of the catchment); crop: sunflower; Unfertilized. TDP, total dissolved P.

Table 8 Overland flow, sediment and phosphorus losses during various rain events in a Calcic Luvisol catchment[a] in southern Spain (Data from Díaz de la Torre 2002 and Jódar 2003)

Date	Rainfall (mm)	Runoff (mm)	Sediment loss (kg ha^{-1})	Total P loss in runoff (kg ha^{-1})	TDP loss in runoff (kg ha^{-1})
16 January 2001	59	27	131	0.24	0.055
22 January 2001	23	2	12	0.02	0.004
13 March 2001	162	47	1820	1.35	0.132
30 January 2002	11	0.6	32	0.02	0.001
21 March 2002	55	54	395	0.35	0.097

[a]Catchment surface: 24.5 ha. Topsoil properties: clay content 60–330 g kg^{-1}; pH 7.4–8.0, Olsen P 5–70 mg kg^{-1} (20–30 mg kg^{-1} in 80% of the catchment); crop: sunflower in 2001, wheat in 2002; fertilizer rate: 0 and 30 kg P ha^{-1} in 2001 and 2002, respectively; TDP, total dissolved P.

both cases, the concentration of P in the sediment (data not shown) was greater than in the soil because the sediment was rich in clay – the fraction commonly richest in P. The total P enrichment ratio (ER, the total P content in sediment/total P content in soil) was close to 1 for the Vertisol, a value typical for clay-rich soils (Barberis & Withers, 2002). ER values ranged from 2.7 to 6.4 in the Luvisol, with the lower values corresponding to events that caused more runoff. The differences in enrichment ratio by offset is the large difference in TP content of the soils. Thus, the TP concentration in the sediments from both the soils was 500–700 mg kg^{-1}, although the TP concentration was much higher in the Vertisol than in the Luvisol (approx. 600 vs. 220 mg kg^{-1}).

The data in Tables 7 and 8 summarize the importance of single events in the losses of P, mainly as PP, with further release of such P into water depending on the DPS of the sediment. It is important to recognize that the native mineralogical and chemical characteristics of a soil influence the speciation of applied P into forms that differ in the rate of release of phosphate into the solution. Such forms can indeed occur in different proportions in the eroded sediment, as shown by Saavedra & Delgado (2005, 2006) in sediment suspended in rainfall simulator-produced runoff from 17 typical Mediterranean soils cropped to wheat, sunflower, beans, peas and chickpea. These authors reported substantial differences in the proportions of Ca phosphates and Fe oxide-related P, the concentration of the latter being greater in the sediment than in the soil because Fe oxides are usually concentrated in the fine soil fractions. Another result of these experiments was that, despite the small organic carbon content of the topsoil (<15 g kg^{-1}), there were significant concentrations of organic P in the runoff (0.04–0.41 mg P L^{-1}), even though only a small fraction of this seemed to be bioavailable.

Extreme rainfall events are an important climatic feature in the Mediterranean region. The rainfall rates comparable to those of other regions in Europe are 50–100 mm h^{-1} for a 100-year return period (Casas *et al.*, 2004; Hand *et al.*, 2004). Relative to similar events in western and Central Europe, those of Mediterranean areas have a greater tendency to occur between late summer and early winter (Casas *et al.*, 2004; Houssos & Bartzokas, 2006; Neppel *et al.*, 1997), a period when many soils are bare or have only sparse vegetative cover. The most extreme autumn rainfall events in the Mediterranean area occur around Italy (Paeth & Hense, 2005). There is a trend in this region towards increased annual precipitation, in contrast to the trend for less rainfall in the eastern Mediterranean areas (Kostopoulou & Jones 2005). Such a 'paradoxical' increase in torrential rainfall despite the decrease in annual rainfall has also been detected in the western Mediterranean region (Alpert *et al.*, 2002).

It should be noted that the spread of extreme rainfall events – mainly those of convective origin – is relatively limited. Thus, in the Languedoc–Roussillon region (south-eastern France), Neppel *et al.* (1997) estimated the 24-h isohyet-bounded areas for a return period of 100 years to be 24 000 km^2 for a 50-mm rainfall, but only 1500 km^2 for a 250-mm rainfall. Thus, large local losses of P are 'diluted' in large catchment areas. At the local level, they can cause losses equivalent to the 'normal' P losses of several decades. Thus, a rainfall of 205 mm that fell in 2 h 15 min on a vinyard soil in north-eastern Spain on 10 June 2000 caused a loss of 109 kg P ha^{-1} (Ramos & Martínez-Casanovas, 2004). In this area, extreme rainfall events account for more than 60% of the annual soil and nutrient losses (Ramos & Martínez-Casanovas, 2006).

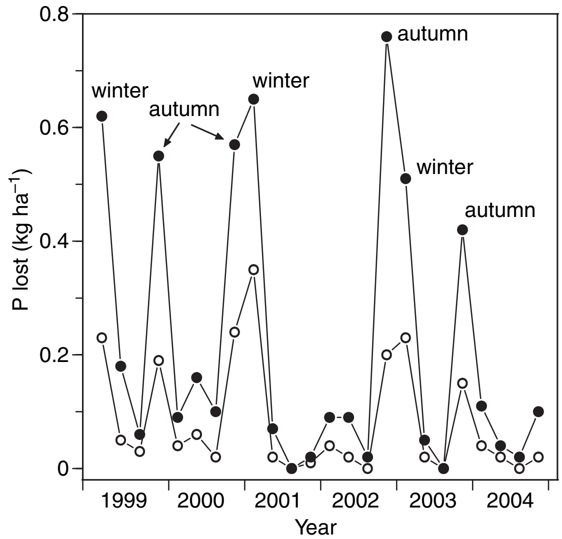

Figure 2 Seasonal losses of TP (closed circles) and DRP (open circles) in a 36-km^2 catchment near A Coruña (north-western Spain) during 2000–2004 (data from Sade Fouz, 2005). Data points are for winter (January-March; marked as 'winter'), spring (April–June; not marked), summer (July–September; not marked), and autumn (October–December; marked as 'autumn').

'Incidental' losses of P [i.e. those in runoff when rainfall interacts directly with fertlizers and manures spread on the soil surface (Withers *et al.*, 2003)], can be substantial from arable land because P fertilizer granules are often applied in autumn, a time when soils are bare and rainfall erosivity peaks (Renschler *et al.*, 1999).

Areas of southern Europe with a humid climate exhibit a more regular temporal pattern of P losses via runoff – such losses can be substantial, however. Figure 2 shows the seasonal losses of TP and DRP for a 36-km^2 catchment near A Coruña (north-western Spain), where land use is 20% grazing, 35% agriculture and 35% forestry. Winter and autumn, the seasons with the greatest rainfall and runoff, account for most of the P loss, which is approximately 1 kg TP ha^{-1} $year^{-1}$.

Surface water quality

Currently, no general statement can be made on the relative contribution of agricultural non-point sources (NPS) (we include here inorganic fertilizers, animal excreta/manure and agricultural residues/composts) to the P load of rivers, lakes and reservoirs in southern Europe. It is worth noting that the upper courses of many rivers and related lakes/reservoirs lies in an area that is, mainly for orographic reasons, more humid and cooler than the climates of the lower rivercourses. As a result, such land is preferentially used for grazing and forestry, so runoff in periods of heavy rains can incorporate P from animal excreta or manure rather than from inorganic fertilizers. Thus, Geraldes & Boavida (2003) studied two reservoirs on tributaries of the River Douro (north-eastern Portugal) and found pasture areas to be the most important source of P for the reservoir at the higher altitude (1300 m), and arable land and pasture to contribute similarly to the reservoir at the lower altitude (500 m).

The relative contribution of agricultural NPS to the load of P in river water generally decreases downstream as the density of urban and industrial point sources (PS) increases. The magnitude of this decrease varies widely. Thus, the P load of reservoirs located on the middle course of rivers in Galicia (north-western Spain), which collect water from areas where population density and industrial activity are low, originates mainly from arable land and pasture NPS (50–60%) and intensive livestock production (20–30%) (Alfaro Monge, 2004). By contrast, 88% of the P load in the central sector of the River Ebro (north-eastern Spain) derives from urban and industrial PS (Torrecilla *et al*., 2005). These sources contribute less P in some tracts of the lower course of the River Guadalquivir (south-western Spain), which drains areas of intensive dry-farming and irrigated agriculture (Mendiguchia *et al*., 2004). The substantially decreased P load in waters flowing into some estuaries, coastal lagoons and lakes near the sea, when action was taken to curb the P content of domestic and/or industrial effluents, confirms that diffuse agricultural sources may have a weaker impact than non agricultural P sources (La Jeunesse *et al*., 2002; Romo *et al*., 2005). Nevertheless, the Po Authority (Autorità di Bacino del fiume Po, 2001) has estimated that $>90\%$ of P loads in the river are due to agricultural sources, with estimated contributions of about 10 000 t P year^{-1} from municipal waste, 1000 t P year^{-1} from industry, 50 000 t P year^{-1} from animal husbandry, and 90 000 t P year^{-1} from arable land.

Surface waters that are mainly, or exclusively, affected by agricultural NPS of P differ widely in their SRP and TP concentrations. Several cases illustrate this point, namely: (i) the reservoirs in the Geraldes & Boavida (2003) study were classified as meso-eutrophic on the basis of TP concentration in water (25–100 μg L^{-1}), with part of their P loading ascribed to decomposition of flooded terrestrial vegetation, exposure of littoral sediment to wetting and drying cycles and land fires; (ii) the reservoir of Bort-les-Orgues (Dordogne River, French 'Massif Central' region), where only 20% of the external P load is due to domestic effluents and industry, tends to be mesotrophic (Brigault & Ruban, 2000); (iii) the Guadalmellato reservoir (southern Spain) is eutro-hypertrophic (Marín Galvín, 1993), even though it is affected by no urban or industrial P sources; (iv) Lake Banyoles (north-eastern Spain), a lake of karstic origin receiving 90% of its P load from agricultural NPS, is, by contrast, essentially oligotrophic (ILEC (International Lake Environment Committee Foundation), 2005); (v) the concentration of TP in water of temporary and shallow lakes in the Castilla y León plateau (Spain) that were not affected by cattle/pig slurry/manure ranged from 10 to 1090 μg L^{-1} (median = 105 μg L^{-1}; n = 73) in the 2003 and 2004 summer months (F. García Criado, unpublished data). These differences result not only from differences in land use intensity and pathways of P transport, but also in part from, the presence of flooded terrestrial vegetation in reservoirs and contents/forms of native P in lake bottom and reservoir sediments (Camargo *et al*., 2005).

The concentration of P from agricultural NPS in river and lake/reservoir waters in Mediterranean regions is markedly seasonal – the result of the climatic regime and consequent influence on land use and agricultural practices. For instance, Kotti *et al.* (2005) reported high concentrations of SRP (>1000 μg L^{-1}) in waters of some rivers in north-western Greece in winter and spring which were ascribed to runoff water from agricultural areas. By contrast, P from urban and industrial PS was responsible for much of the P load in such rivers in autumn. Seasonality is especially marked in lakes (e.g. Petaloti *et al*., 2004), one reason being that they are generally shallow, so changes in water level have a strong impact on mixing and enhance the influence of bottom sediments. The dramatic changes in reservoir levels that occur during long drought periods nearly always result in substantial changes in P concentration in water. Such changes tend obviously to be smaller when the number and capacity of the reservoirs on a river course increase.

Mitigation possibilities

Agricultural P inputs to surface waters can be reduced by adopting various management practices (Withers & Jarvis, 1997). Because of the climatic constraints in southern Europe, conserving soil and intercepting runoff are critical in this respect, where water erosion is severe (Poesen & Hooke, 1997). To date, the adoption of measures intended to curb erosion has been slow or non-existent in many vulnerable areas, even though the problem has been recognized by farmers, planners and the public at large. The lack of coercion in implementing such measures appears to be a major cause.

The mid term reform of the EU CAP introduced in 2003, incorporated the Cross Compliance (CC) policy, which is a significant step towards the adoption of soil conservation practices. CC (i.e. attaching conditions to the receipt of agricultural subsidies) can be viewed as a mechanism intended to improve the standards of environmental conservation in farming. It requires the farmer to accept some Statutory Management Requirements (SMRs) (e.g. the Nitrates Directive) and a number of standards aiming to ensure the 'good agricultural and environmental conditions' (GAEC) of agricultural land. (e.g. standards on minimum soil cover or on retaining terraces). Currently, the Member States are in the process of implementing such requirements. A recent report (Farmer and Swales, 2004) exposed significant differences among the southern European countries and between regions within these countries, in the emphasis placed on the various soil conservation practices. The success of the CC policy will be dependent on the efficacy of the control measures, which are being developed by the Member States.

Different public institutions responsible for water quality have already taken or are in the process of taking steps to develop plans intended to mitigate eutrophication by P in some countries. For instance, the Basin Authority of the river Po has promoted the creation of a Consultation Committee including a number of stakeholders and adopted a strategic plan to control eutrophication for implementation by the Regional Councils. As in other instances, the implementation of mitigation options is lagging behind recognition of the problem.

Conclusions and proposals for further research

The inputs of fertilizer P into the agricultural soils of southern European countries are smaller than those in other European regions, but significantly greater, on average, than the amounts of P removed from the soil through cropping and grazing. Therefore, high soil P test values are recorded in many areas, which are influenced, among other factors, by land use, crop type and livestock density. The characteristics of the Mediterranean climate and soils, as well as land use, result in P transfer from agricultural soils through runoff rather than in drainage water. In this context, extreme rainfall/erosion events appear to contribute substantially to P losses. Available data on the quality of water in rivers, lakes and reservoirs suggest that eutrophication caused by agricultural P is common. In this respect, the effective implementation and adoption by farmers of soil conservation measures and other good agricultural practices is crucial.

This paper underlines the pressing need for research on various issues including (i) the development and evaluation of soil P tests useful for agronomic purposes and for estimating the degree of P saturation in Mediterranean soils and eroding sediments; (ii) the characterization of the magnitude and seasonality of P losses from benchmark soils in high risk catchments under the typical land uses of the Mediterranean region; and (iii) the characterization of loss pathways and the significance of P losses in areas of irrigated agriculture and intensive animal production.

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