Surface Water Quality of Intensive Farming Areas Within the Santa Lucia River Basin of Uruguay

Authors: Barreto, Patricia, Dogliotti, Santiago, and Perdomo, Carlos

Source: Air, Soil and Water Research, 10(1)

Published By: SAGE Publishing

URL: https://doi.org/10.1177/1178622117715446

Surface Water Quality of Intensive Farming Areas Within the Santa Lucia River Basin of Uruguay

Air, Soil and Water Research
Volume 10: 1–8

DOI: 10.1177/1178622117715446
SAGE

Patricia Barreto[1], Santiago Dogliotti[2] and Carlos Perdomo[1]

[1]Departamento de Suelos y Aguas, Facultad de Agronomía, Universidad de la República, Montevideo, Uruguay. [2]Departamento de Producción Vegetal, Facultad de Agronomía, Universidad de la República, Montevideo, Uruguay.

ABSTRACT: The Santa Lucia River basin provides drinking water to 60% of Uruguay population. The input of excessive amounts of nitrogen and phosphorus both from point and nonpoint sources could impair surface water quality and prevent its use for human consumption. The objective of this work was to evaluate surface water quality in small catchments under agricultural use located within this river basin. To this end, 5 streams and 4 small polders were surveyed between September 2008 and December 2009. The median concentrations of total phosphorus and total nitrogen in the streams across all sites and periods were 770 and 1659 μg L^{-1}, respectively, exceeding in the case of total phosphorus national and international thresholds. Furthermore, soluble phosphorous, the most readily available form to algae, represented 88.3% of total phosphorus. Concentrations of phosphorous and nitrogen in polders were also high. The eutrophication process of this water bodies could be linked to the intensive agricultural land use in the area because the nutrient input from cities and industries was not relevant in these catchments.

KEYWORDS: Eutrophication, soluble phosphorus, nonpoint pollution

RECEIVED: February 5, 2017. **ACCEPTED:** May 16, 2017.

PEER REVIEW: Five peer reviewers contributed to the peer review report. Reviewers' reports totalled 545 words, excluding any confidential comments to the academic editor.

TYPE: Original Research

FUNDING: The author(s) disclosed receipt of the following financial support for the research, authorship, and/or publication of this article: This project was partially funded by the European Union, Project INCOCT-2006-032387, and by INIA-Uruguay, Project FPTA 209.

DECLARATION OF CONFLICTING INTERESTS: The author(s) declared no potential conflicts of interest with respect to the research, authorship, and/or publication of this article.

CORRESPONDING AUTHOR: Patricia Barreto, Departamento de Suelos y Aguas, Facultad de Agronomía, Universidad de la República, Garzón 780, C.P. 12900 Montevideo, Uruguay. Email: pbarreto@fagro.edu.uy

Introduction

The Santa Lucia River basin, located in the south of Uruguay, provides drinking water to 60% of the country's population.[1] In the northern part of this watershed (zone A), about 50% of the area is occupied by grazing-based dairy farms, whereas in the southern area (zone B), vegetable farms associated with poultry and swine production predominate.[2] These farms use large amounts of organic and chemical fertilizers to increase production. However, the perception that an increase in nitrogen (N) is needed to increase agronomic productivity has resulted in an excessive input of phosphorus (P) to the system. This is due to the low N:P ratio of the organic fertilizer used (poultry manure), resulting in a positive balance of P and in a soil P enrichment.[3] Nutrient enrichments from point or nonpoint sources may lead to eutrophication, degrading the aesthetic and recreational value of surface water.[4]

International nutrient thresholds can be used as guidelines for water quality assessments. According to Daniel et al,[5] to control eutrophication, total phosphorus (TP), and soluble phosphorus (SP) in streams should not exceed 20 and 10 μg L^{-1}, respectively. Dodds et al[6] compiled a database with TP and total nitrogen (TN) log-mean concentrations in temperate streams and set the boundary between mesotrophic and eutrophic conditions in the lowest third of the cumulative frequency plots (70 and 1500 μg P L^{-1}, respectively). In Uruguay, Decree 253/79 established a TP critical level of 25 μg P L^{-1} for surface waters,[7] but recently, the National Water Authority has been considering raising this limit to 70 μg P L^{-1}.[8]

The results of several surveys conducted in this basin have shown that the water quality in the area has been deteriorating, affecting the possibility of using this water resource as a source for drinking water. One of these surveys reported TP values that ranged between 80 and 735 μg P L^{-1}, whereas those of N in the forms of nitrate and ammonium fluctuated from 0.014 to 0.25 and 0.01 to 0.25 μg N L^{-1}, respectively.[9] In another report of 15 sampling points, the TP and TN annual means in 9 of these points were above the eutrophic limit (TP: 170-440 μg P L^{-1} and TN: 4200-10 500 μg N L^{-1}), whereas 4 of the remaining 5 were classified as supereutrophic, and 1 as hypereutrophic.[10] In 2013, a harmful algal bloom in this area caused changes in the odour and taste of the water supplied by the water authority and generated concern and alarm among the public.[11] Because of this event, an 'Action Plan' based on several best management practices (BMPs) was implemented by the relevant ministry.[2]

According to the latest report, this situation has not changed substantially; the TP and TN values at a measurement station located at the confluence of the Santa Lucia and San Jose rivers were 310 and 1280 μg P L^{-1}, respectively.[8] There are conflicting opinions about the main sources of this pollution; although some stakeholders primarily hold the point sources of cities and industries responsible for this situation, others blame the agriculture-related sources of contamination.

Due to the steady and worrying decline in water quality of some watersheds in the country and the scarce availability of information in horticultural areas, the aim of this work is

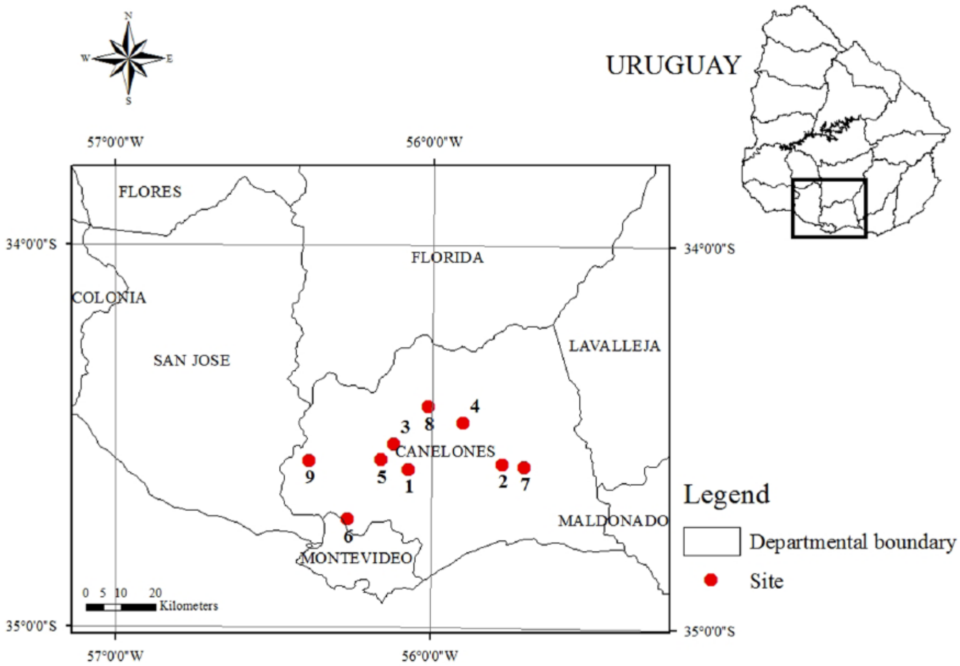

Figure 1. Geographical distribution of the sampling sites.

Table 1. Location and production system at each site.

SITE NUMBER	GLOBAL POSITIONING SYSTEM LOCATION		SYSTEM OF PRODUCTION
	LATITUDE	LONGITUDE	
1	34°35′16.96″S	56°4′19.12″W	V, L, P
2	34°34′24.28″S	55°46′25.21″W	V
3	34°31′18.89″S	56° 7′9.23″W	V
4	34°27′52.41″S	55°54′6.53″W	V, L
5	34°33′48.36″S	56° 9′26.83″W	V, L
6	34°43′7.74″S	56°15′55.02″W	V
7	34°34′48.46″S	55°42′15.54″W	V, L, S
8	34°25′26.09″S	56° 0′41.25″W	V, L
9	34°33′57.34″S	56°23′14.38″W	V, L

Abbreviations: V, vegetable; L, livestock; P, poultry; S, swine.

to evaluate the impact of agricultural land use on surface water quality in horticultural catchments located within or near zone B of the Santa Lucia River basin, where there were no point sources of pollution (cities and industries). In addition, with this information it is possible to identify the effect on the water quality of this productive system and to implement a national control plan and an evaluation of the effectiveness of the control mechanisms and control systems based on the results of official water controls.

Materials and Methods

A total of 9 'pilot' farms located within zone B of the Santa Lucia River basin[2] were selected for surface water quality monitoring (Figure 1 and Table 1), including 5 natural streams and 4 artificial polders (Table 2). Sampling was conducted from September 2008 to December 2009 with a variable frequency.

Soils in this area are mostly fertile-clay soils with a relatively low textural differentiation (Pachic or Vertic Argiudolls and Typic Hapluderts) developed above 2 Pleistocene formations (Libertad and Raigón). These are mixed with less-fertile and more differentiated loamy-clay soils (Abruptic Argiudolls) formed on top of a late Oligocene formation (Fray Bentos).[12] Further details of the soil are given in Table 3.

Table 2. Number of samples collected per site, type of water, and year.

SITE NUMBER	STREAMS			POLDER			STREAMS AND POLDER
	2008	2009	2008–2009	2008	2009	2008–2009	2008–2009
1	2	3	5	ND	ND	ND	5
2	ND	ND	ND	6	2	8	8
3	4	2	6	ND	ND	ND	6
4	3	1	4	ND	ND	ND	4
5	ND	ND	ND	3	ND	3	3
6	5	4	9	ND	ND	ND	9
7	ND	ND	ND	2	1	3	3
8	ND	ND	ND	4	3	7	7
9	4	5	9	ND	ND	ND	9
Total across sites	18	15	33	15	6	21	54

Abbreviation: ND, no data.

Table 3. Information about the predominant geological formations and soil types at each site.

SITE NUMBER	GEOLOGY	SOIL CLASSIFICATION	SOIL TEXTURE
1	FBF, MgF, MF	AA	L, LS
2	FBF, MgF, MF	AA	L, LS
3	LF, FBF, MgF	TH	SC
4	LF, FBF, MgF	TH	SC
5	LF, FBF, MgF	TH	SC
6	RF, MgF	PVA	CS, LC, SC
7	RF, MgF	PVA	CS, LC, SC
8	RF, MgF	TH	SC
9	FBF, MgF, MF	AA	L, LS

AA, Abruptic Argiudolls (Brunosoles subútricos lúvicos); C, clay; FBF, Fray Bentos Formation; L, loam; LF, Libertad Formation; MgF, Migues Formation; MF, Mercedes Formation; PVA, Pachic and Vertic Argiudolls (Brunosoles éutricos/subéutricos típicos); RF, Raigón Formation; S, silt; TH, Typic Hapluderts (Vertisoles rúpticos); Durán et al.[12]
Adapted from García et al.[13]

The main land use in this area is intensive vegetable production, mixed with poultry and swine farms. The climate is classified as temperate, with a mean annual temperature and rain of 17°C and 1200 mm, respectively.[14]

Grab samples were collected from mid-stream into polyethylene bottles (1 L). To prevent biological changes to the water composition, all bottles were previously spiked with 1 mL of H_2SO_4 5.5 M. All samples were analysed for TN, nitrate ($N\text{-}NO_3^-$), TP, soluble reactive phosphorus (SRP), and turbidity (Tb). Total nitrogen was analysed using H_2SO_4 digestion, according to the method of Bremmer and Mulvaney[15] and colorimetry determination according to the method of Rhine et al.[16] Before digestion, a 100-mL water subsample was acidified with 2 mL of H_2SO_4 5.5 M and concentrated to 10 mL by evaporation at 90°C. Analysis of $N\text{-}NO_3^-$ was conducted according to the method of Keeney and Nelson.[17] Total phosphorus and SRP were analysed using the method of Pote and Daniel.[18] All nutrient colorimetric determinations were made with a Unicam 5675 Spectrometer. Turbidity was determined using an Oakton T-100 portable turbidity meter.

Concentrations of TP and TN measured in water samples were compared with thresholds used in different countries (including Uruguay) to assess the freshwater eutrophication risk (Table 4).

Nitrate concentration levels were compared with the Uruguayan drinking water threshold[7] enforced by Decree

Table 4. International reference values for total nitrogen (TN), total phosphorus (TP), and turbidity (Tb) in streams.

REGION	TN, $\mu g\,L^{-1}$	TP, $\mu g\,L^{-1}$	TURBIDITY, NTU
US pristine ecoregion[19]	120	10	1
US agricultural ecoregion[19]	2180	130	18
Australia and New Zealand[20]	1200	100	50
Europe[a, 21]	250	30	50
Uruguay[7]	ND	30	50

Abbreviation: ND, no data.
[a]33 member countries.

253/79. This parameter, however, only allows the identification of non-drinkable water – it cannot be used to evaluate potability.

Regression procedures (Microsoft Excel, 1997) were used to evaluate the relationships between the water quality variables. Nutrient concentrations were not normally distributed according to the Shapiro-Wilks test, so differences between sites were evaluated by the Kruskal-Wallis nonparametric median test using the statistical package InfoStat version 2011.[22] Differences were considered statistically significant when $P \leqslant .05$.

Results and Discussion

Streams

The median concentration across all sites and periods (MCAS) of TP in streams was $770\,\mu g\,P\,L^{-1}$; a value higher than all national and international thresholds presented in Table 1. The median concentrations for the study sites (MCSS) varied between 27 and $3080\,\mu g\,P\,L^{-1}$ (Figure 2). In most cases, these MCSS values were above the previously mentioned thresholds but comparable with TP concentrations reported from international studies performed under similar land use conditions. For example, in commercial citrus groves in the United States, Yu et al[23] reported that mean TP concentrations in surface water ranged from 510 to $2640\,\mu g\,P\,L^{-1}$, whereas He et al[24] found that this range was between 1510 and $3920\,\mu g\,P\,L^{-1}$ for vegetable farms. In turn, in our study, the MCAS of SRP was $554\,\mu g\,P\,L^{-1}$, whereas MCSS ranged between 10 and $3078\,\mu g\,P\,L^{-1}$, within the ranges of 200 to 2130 and 470 to $1980\,\mu g\,P\,L^{-1}$ reported by Yu et al[23] and He et al,[24] respectively. In the 2 studies mentioned above, SRP, which is the most bioavailable form of phosphorus, represented between 74% and 45% of TP, respectively. In our study, this range was between 40% and 99%.

In the case of TN, MCAS was $1659\,\mu g\,N\,L^{-1}$: higher than the limits established for Europe but lower than those established for New Zealand and the United States (Table 4). The range of MCSS values varied from 87 to $8724\,\mu g\,N\,L^{-1}$, exceeding the limits established for the US Environmental Protection Agency (USEPA) agricultural ecoregion at only 2 sites (#3 and #4) (Table 4). These values were lower than the median TN concentration obtained by Bartley et al[25] for Australian basins used for horticulture ($32\,000\,\mu g\,N\,L^{-1}$), but the MCSS range of TN found here was wider than that cited by Smith et al[26] for US dairy basins (690-3270 $\mu g\,N\,L^{-1}$). The average N: P relation for streams (across all sites and periods) was 16:1, equal to the Redfield ratio.[27] This implies that neither N nor P was limiting primary productivity, or in another words, that TN and TP were in equilibrium with the phytoplankton demand for both nutrients. This equilibrium, however, was more apparent than real – in four of these sites (#1, #3, #4, and #6), this ratio was lower than 16:1 (1.2:1, 2.8:1, 4.2:1, and 4.9:1, respectively), indicating that P was excessive, whereas in the other site (#9), this relation was higher than 16:1 (20:1), indicating that P was limiting.

In addition to the TN and TP values presented above, the MCAS of Tb (6.19NTU) was lower than most of the international reference values; however, it was just above the threshold for the most pristine of USEPA's ecoregions (Table 4). This indicates that the visual aspect of 'clear water' does not necessarily indicate the absence of nutrient pollution problems. In the case of $N\text{-}NO_3^-$, MCAS was $340\,\mu g\,N\,L^{-1}$, and this value fell within the expected range, given that in the absence of artificial drainage, this anion would preferentially move to groundwater.[28]

Among the sites examined here, there were significant statistical differences in MCSS values in the cases of both TP and SRP. The highest concentrations of both P forms were found at sites #4 and #6, and the lowest at sites #1 and #9, respectively. In the cases of TN, $N\text{-}NO_3^-$, and Tb, these differences were not statistically significant, although the probability value ($P > .074$) for Tb was close to that of the significance limit.

Polders

The MCAS of TP was $350\,\mu g\,P\,L^{-1}$ (Figure 2) – higher than the threshold suggested by Dodds et al[6] to separate oligotrophic from mesotrophic waters in lakes and reservoirs (lentic systems). This value also exceeded the maximum allowable concentration in the United States and New Zealand for different ecoregions.[19,20] As in surface waters, the MCAS for

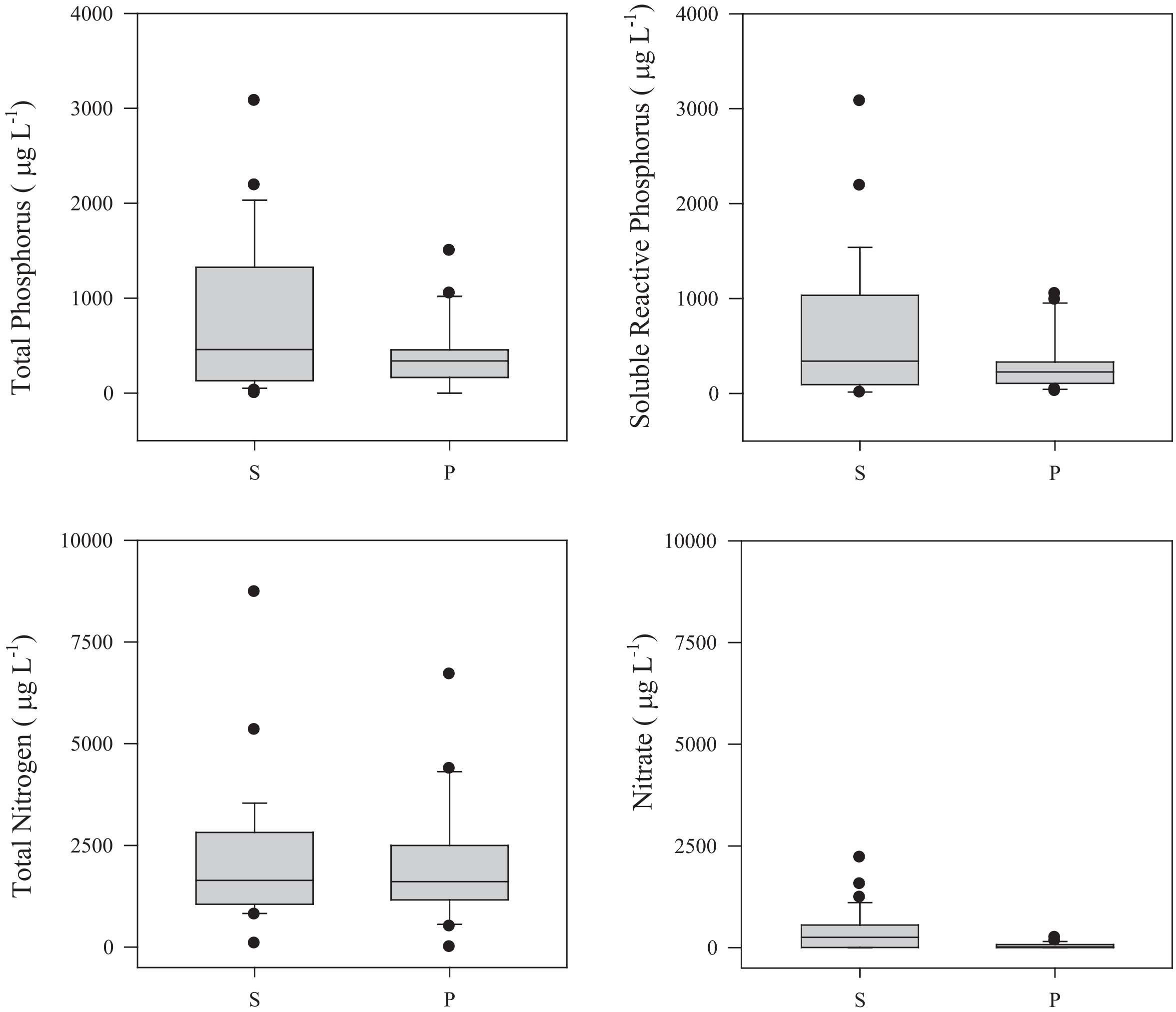

Figure 2. Box plots of total phosphorus, soluble reactive phosphorus, total nitrogen, and nitrate in streams (S) and polders (P) from all sites and years. Centre lines show the medians, box limits indicate the 25th and 75th percentiles, and whiskers indicate minimum and maximum values, and outliers are represented by dots.

SRP was high (230 µg P L^{-1}), and the greater part of TP was in soluble form. Therefore, as in streams, the polders were enriched with P, although in a lesser extent. This is expected because there is more sediment deposition in dams (lentic systems) than in streams (lotic systems). This is why lotic systems have higher thresholds, which are translated into less demanding values than those established for lentic systems.[19] It may be noted that international thresholds established for lentic systems[19] are the same as the Uruguayan national water quality standards for freshwater (25 µg P L^{-1}), which suggests that this threshold is too demanding and more applicable to lakes and reservoirs.

In the case of TN, observed MCAS (1567 µg N L^{-1}) was higher than the less demanding reference value (1270 µg N L^{-1}) for ecoregion #XIII in the United States.[19] In Uruguay, there is no reference for this variable. These waters were more enriched with P than N (N:P ratio < 9:1) but to a lesser extent than surface waters. The value of MCAS for Tb (20.6 NTU) was higher than the reference value for the US agricultural ecoregion but lower than other international reference values (Table 4). However, this variable is not included in the standards for lentic systems.[19] The MCAS of NO_3 (14.5 µg N L^{-1}) was lower than the international and local critical level concentrations for drinking water (10 000 µg N L^{-1}). Moreover, all samples were below this threshold.

No significant relationships between the analysed water quality parameters were observed, but when the MCAS values from streams and polders were grouped together, there was a statistically significant positive relationship between TN and Tb ($r = 0.85$, $P < .0076$). Vidon et al[29] showed similar results gathered from surface water monitoring sites located both upstream and downstream from a cattle grazing area. They reported that when cattle had direct access to the stream channel (during summer-fall), Tb values tended to be higher downstream than those observed upstream at a 13:1 ratio, but NO_3 concentrations were not significantly affected. According to these authors, the reason for these high Tb values could be related to the deposition of cattle manure in surface water sources. Part of the organic matter from the manure remains as a suspension in the water.

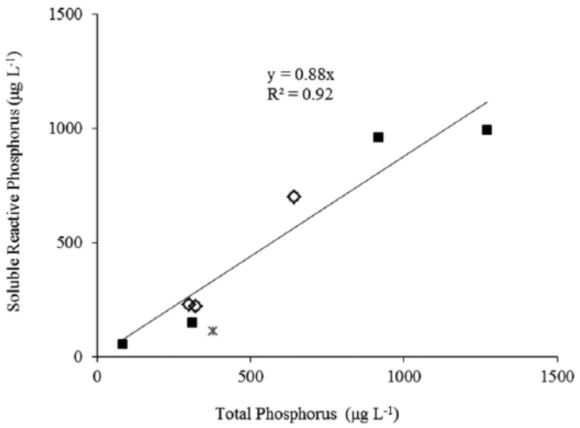

Figure 3. Relationship between median concentration of total phosphorus and soluble reactive phosphorus in streams and polders by site. The filled boxes represent streams and the empty boxes represent the polders. The grey cross represents a stream value that was not included in the regression.

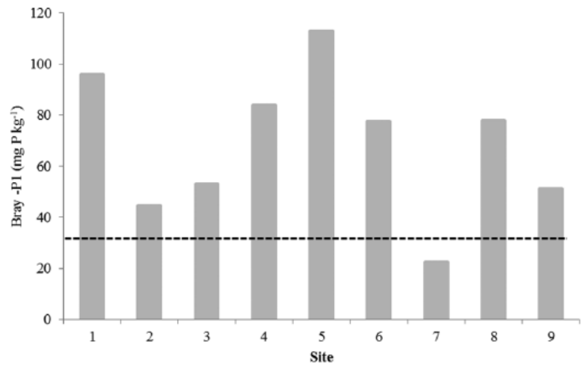

Figure 4. Soil test values for phosphorus (Bray I) at each site. The dotted line represents the critical value required to maximize lucerne (*Medicago sativa*) yield.

Relationships between TP and SRP concentrations in surface waters

The concentration ranges of TP and SRP observed in the 2 types of surface water (streams and polders) evaluated in this study were relatively similar. Therefore, these 2 water sources were grouped together, and the relationship between TP and SRP in this new group was analysed, resulting in a linear and statistically significant relationship between them (Figure 3). The linear equation did not include an intercept term because it was not significantly different from 0. Based on this equation, and on the legal Uruguayan threshold (25 µg P L^{-1}) established in Decree 253/79, an SRP threshold value of 22 µg P L^{-1} was estimated, which would be applicable to both streams and polders. This value was higher, and less strict, than the corresponding SRP threshold of 9 µg P L^{-1} established for Australia[30] but lower than the MCAS values observed in streams and polders (340 and 230 µg P L^{-1}, respectively), confirming the environmental deterioration of these surface waters.

It should be noted that the existence of high SRP concentrations in streams seems to contradict the conclusions of Correll[27] that SRP always tends to be a small fraction of TP because of the strong adsorption of orthophosphate onto stream sediments, which causes the TP:SRP balance to be strongly shifted towards the TP side. In the intensive farming conditions of this study, however, this balance may have been modified. This change could have been caused, in part, by a larger input of soluble P to these surface water bodies. This P is likely to originate from single-point sources, such as untreated dairy-farm effluent, but also from more diffuse inputs, like P carried by water runoff.

The diffuse P input tends to increase when there is a boost in soil labile P,[31] which is the P fraction most related to P uptake by crops. At these study sites, soil labile P, as measured by the Bray-P1 method at a soil depth of 0 to 20 cm, has increased over many years from a basal value[32] of 4 mg P kg^{-1} to an average of 69 mg P kg^{-1}, with a range of 22 to 113 mg P kg^{-1} (Figure 4). This increase was more than was needed to maximize crop productivity. For example, the optimum Bray-P1 value required for lucerne (*Medicago sativa*), the relevant species with the higher P requirement is 25 mg P kg^{-1}for this soil depth.[33]

Since 2000, these farms have shifted from conventional to no-till farming, and nowadays, almost all farms have made this transition.[34] Under no-till planting, phosphate fertilizer is usually applied to the soil surface without being mixed in, which leads to a high P stratification at the soil surface, further increasing the concentration of soluble P in runoff.[35] Perdomo et al[36] have also observed strong labile P stratification in soils under intensive management in other areas of the Santa Lucia River basin.

A lower rate of soil erosion under no-till farming also reduces the fraction of particulate P carried away by runoff, leading to a decrease in sediment transport; thus, the specific soil surface capable of P adsorption also diminishes, leading to an increase in SRP.[37] According to James and Barko,[38] soluble forms of P comprised between 48% and 57% of the flow-weighted concentration of TP between May and September 2002, at Upper Eau Galle River (Wisconsin, USA), whereas all bioavailable P (including SP plus labile particulate phosphorus) represented 79% of TP. Goyenola et al[39] also reported a high proportion of soluble P (86.4%) in streams located in areas undergoing intensive farming in another zone of the Santa Lucia River basin. In our case, this median-estimated value was 88%.

It should be noted that the contribution of point sources of water pollution from cities and industries located within the Santa Lucia River basin do not affect the water quality of the catchments under study because there is no direct hydrologic connection between these 2 zones. Thus, the most plausible explanation of the observed concentration rise of P, and other nutrients in these water bodies, is that it is mainly related to the intensification of agriculture in the area. The implications of this conspicuous case of surface water contamination are obvious because these catchments are in the area that provides potable water to more than half of the Uruguayan population. Although Uruguay has already implemented farm policies to protect water quality in this area, they are primarily based on preventing soil erosion and restricting P fertilization to fields that are below agronomic soil test phosphorus (STP) thresholds. However, as Baker et al[40] has noted, one of the BMPs that show larger potential to reduce dissolved reactive phosphorus in runoff in similar situations is the reduction of STP stratification by a one-time soil inversion, as well as the application of P fertilizer below the soil surface instead of on top.

Conclusions

1. The MCAS of TP (770 and 350 μg P L^{-1}) and TN (1659 and 1567 μg N L^{-1}) in courses and polders, respectively, were higher than the most permissive thresholds established by USEPA for both water types, even though these water bodies mainly receive discharge from agricultural sources and are not influenced by cities or industries.
2. Soil labile P (Bray 1) at the surveyed sites ranged from 22 to 113 mg P kg^{-1} (0-20 cm), well above the original value of 4 mg P kg^{-1}. This suggests that part of the increase in P concentration in water could be related to P carried away through runoff. Another source of pollution still present at many sites comes from the direct discharge into waterways of dairy and swine effluents. Consequently, BMPs should be enforced to minimize the impact of these 2 contamination sources.
3. Even with the application of these BMPs, water quality improvement would be slow because current labile P levels in both soil and stream sediments are high, and their decrease would be sluggish because P equilibrium is strongly displaced towards the solid phase.

Author Contributions

PB, CP, and SD conceived and designed the experiments; contributed to the writing of the manuscript; agree with manuscript results and conclusions; jointly developed the structure and arguments for the paper; and reviewed and approved of the final manuscript. PB analysed the data and wrote the first draft of the manuscript. CP and SD made critical revisions and approved final version.

Acknowledgements

The authors are grateful to Victoria Mancassola, Sebastian Peluffo, and José Pedro Dieste for their help on collecting water samples from the different sites.

REFERENCES

1. Achkar M, Dominguez A, Pesce F. Cuenca del Río Santa Lucía – Uruguay Aportes para la discusión ciudadana [in Spanish]. http://www.redes.org.uy/2012/12/12/cuenca-del-rio-santa-lucia-uruguay-aportes-para-la-discusion-ciudadana/. Published 2012. Accessed February 3, 2017.
2. MVOTMA. Plan de acción para la protección del agua en la cuenca del Santa Lucía [in Spanish]. http://mvotma.gub.uy/sala-de-prensa/item/10004440-plan-de-acción-para-la-protección-del-agua-en-la-cuenca-del-santa-lucía.html. Published 2013. Accessed February 2, 2017.
3. Barbazán M, del Pino A, Moltini C, Hernández J, Rodríguez J. Caracterización de materiales orgánicos aplicados en sistemas agrícolas intensivos de Uruguay [in Spanish]. *Agrocien Uruguay*. 2011;15:82–92.
4. Dokulil MT, Teubner K. Eutrophication and climate change: present situation and future scenarios. In: Ansari AA, Singh Gill S, Lanza GR, Rast W, eds. *Eutrophication: Causes, Consequences and Control*. Dordrecht, The Netherlands: Springer; 2011:1–16.
5. Daniel T, Sharpley A, Lemunyon J. Agricultural phosphorus and eutrophication: a symposium overview. *J Environ Qual*. 1998;27:251.
6. Dodds WK, Jones RJ, Welch E. Suggested classification of stream trophic state: distributions of temperate stream types by chlorophyll, total nitrogen and phosphorus. *Water Res*. 1998;32:1455–1462.
7. Executive Branch Uruguay. Poder Ejecutivo de Uruguay. Decreto 253/79-Normas para prevenir la contaminación ambiental mediante el control de la contaminación de aguas [in Spanish]. http://www.mvotma.gub.uy/ciudadania/item/10003601-decreto-253-079.html. Published 1979. Accessed February 3, 2017.
8. DINAMA. Evolución de la calidad en la cuenca del Santa Lucía 10 años de información [in Spanish]. http://mvotma.gub.uy/images/santa_lucia/Evoluci%C3%B3n%20de%20la%20calidad%20en%20la%20cuenca%20del%20Santa%20Luc%C3%ADa%20-%2010%20a%C3%B1os%20de%20informaci%C3%B3n%20OK.pdf. Published 2015. Accessed February 4, 2017.
9. DINAMA. Monitoreo y evaluación de calidad de agua Rio santa Lucia. Resultados del rediseño del programa (periodo 2011) [in Spanish]. http://mvotma.gub.uy/ciudadania/biblioteca/documentos-de-ambiente/item/download/5490_ec4478f9b147946d8a9cf162cd02cc4f.html. Published 2011. Accessed February 4, 2017.
10. DINAMA. Monitoreo y evaluación de calidad de agua Río Santa Lucía [in Spanish]. http://mvotma.gub.uy/images/EVOLUCI%C3%93N_DEL_ESTADO_TR%C3%93FICO_DEL_R%C3%8DO_SANTA_LUC%C3%8DA-2005-2011-vweb.pdf. Published 2013. Accessed February 4, 2017.
11. El PAIS. *Algas en el agua: algunas pueden producir toxinas*. Montevideo, Uruguay: Diario EL PAIS [in Spanish]. http://historico.elpais.com.uy/13/03/12/pciuda_701911.asp. Published 2013. Accessed February 3, 2017.
12. Durán A, Califra A, Molfins JH, Linn W. *Keys to Soil Taxonomy for Uruguay*. Washington, DC: USDA, Natural Resources Conservation Service (NRCS). https://scholar.google.com/scholar_lookup?title=+Keys+to+Soil+Taxonomy+for+Uruguay&author=Durán+A&author=Califra+A&author=Molfino+JH&author=Lynn+W&publication_year=2006. Published 2006. Accessed February 2, 2017.
13. García de Souza M, Alliaume F, Mancassola V, Dogliotti S. Carbono orgánico y propiedades físicas del suelo en predios hortícolas del sur de Uruguay. *Agrocien Uruguay*. 2011;15:70–81.
14. Castaño J, Gimenez A, Ceroni M, Furest J, Aunchayna R, Bidegain M. Caracterización agroclimática del uruguay 1980-2009 [in Spanish]. INIA Serie Technique 193. 2011:33. http://www.ainfo.inia.uy/digital/bitstream/item/2538/1/18429021211104157.pdf.
15. Bremmer J, Mulvaney C. Nitrogen-total. In: Page AL, Miller RH, Keeney DR, eds. *Methods of Soil Analysis Part 2: Chemical and Microbiological Properties*. 2nd ed. Madison, WI: Agronomy 9/2, American Society of Agronomy; 1982:595–624.
16. Rhine ED, Sims GK, Mulvaney RL, Pratt EJ. Improving the Berthelot reaction for determining ammonium in soil extracts and water. *Soil Sci Soc Am J*. 1998;62:473–480.
17. Keeney D, Nelson D. Nitrogen-inorganic forms. In: Page AL, Miller RH, Keeney DR, eds. *Methods of Soil Analysis Part 2: Chemical and Microbiological Properties*. 2nd ed. Madison, WI: Agronomy 9/2, American Society of Agronomy; 1982:643–698.
18. Pote DH, Daniel T. Analyzing for total phosphorus and total dissolved phosphorus in water samples. In: Pierzynski GM, ed. *Methods of Phosphorus Analysis for Soils, Sediments, Residuals, and Water*. Citeseer; 2000:94–97.
19. USEPA. Ecoregional nutrient criteria for rivers and streams. https://www.epa.gov/sites/production/files/2014-08/documents/criteria-nutrient-ecoregions-sumtable.pdf. Published 2002. Accessed February 2, 2017.
20. ANZECC & ARMCANZ. *Australian and New Zealand Guidelines for Fresh and Marine Water Quality*. Canberra, ACT: Australian and New Zealand Environment and Conservation Council, Agriculture and Resource Management Council of Australia and New Zealand, 2000;1–103.
21. Crouzet P, Leonard J, Nixon S, et al. *Nutrients in European Ecosystems – European Environment Agency*. http://www.eea.europa.eu/publications/ENVIASSRP04. Published 2000. Accessed February 5, 2017.
22. Di Rienzo JA, Casanoves F, Balzarini MG, Gonzalez L, Tablada M, Robledo CW. InfoStat versión 2011. Grupo InfoStat, FCA, Universidad Nacional de Córdoba, Argentina. 2011;8:195–199. http://www.infostat.com.ar.

23. Yu S, He ZL, Stoffella PJ, et al. Surface runoff phosphorus (P) loss in relation to phosphatase activity and soil P fractions in Florida sandy soils under citrus production. *Soil Biol Biochem*. 2006;38:619–628.
24. He ZL, Zhang MK, Stoffella PJ, Yang XE, Banks DJ. Phosphorus concentrations and loads in Runoff water under crop production. *Soil Sci Soc Am J*. 2006; 70:1807.
25. Bartley R, Speirs WJ, Ellis TW, Waters DK. A review of sediment and nutrient concentration data from Australia for use in catchment water quality models. *Mar Pollut Bull*. 2012;65:101–116.
26. Smith AP, Western AW, Hannah MC. Linking water quality trends with land use intensification in dairy farming catchments. *J Hydrol*. 2013;476:1–12.
27. Correll DL. The role of phosphorus in the eutrophication of receiving waters: a review. *J Environ Qual*. 1998;27:261.
28. Birr A, Mulla D. Evaluation of the phosphorus index in watersheds at the regional scale. *J Environ Qual*. 2001;30:2018–2025.
29. Vidon P, Campbell MA, Gray M. Unrestricted cattle access to streams and water quality in till landscape of the Midwest. *Agric Water Manag*. 2008;95:322–330.
30. LAWA. Dissolved reactive phosphorus. https://www.lawa.org.nz/learn/glossary/d/dissolved-reactive-phosphorus-(drp)/. Published 2012.
31. Sharpley A, Jarvie HP, Buda A, May L, Spears B, Kleinman P. Phosphorus legacy: overcoming the effects of past management practices to mitigate future water quality impairment. *J Environ Qual*. 2014;42:1308–1326.
32. Hernández J, Berger A, Deambrosi E, Lavecchia A. Soil phosphorus tests for flooded rice grown in contrasting soils and cropping history. *Commun Soil Sci Plant Anal*. 2013;44:1193–1210.
33. Ayala W, Bermúdez R, Carámbula M, Morón A, Bermúdez R. Seminario de actualización técnica: fertilización fosfatada de pasturas en la Región Este. INIA Serie Technique 172. 2008. http://www.inia.uy/Publicaciones/Documentos%20compartidos/18429240309123740.pdf.
34. Ernst O, Siri-Prieto G. Impact of perennial pasture and tillage systems on carbon input and soil quality indicators. *Soil Tillage Res*. 2009;105:260–268.
35. Kleinman PJA, Sharpley AN, Saporito LS, Buda AR, Bryant RB. Application of manure to no-till soils: phosphorus losses by sub-surface and surface pathways. *Nutr Cycl Agroecosystems*. 2009;84:215–227.
36. Perdomo C, Barreto P, Piñeiro V. Pérdidas de fósforo desde suelos agrícolas hacia aguas superficiales: resultados preliminares para Uruguay y posibles medidas de manejo para mitigar riesgos [in Spanish]. *IV Simp Nac Agric Paysandú-Uruguay*. 2015;1:77–94.
37. Smith DR, Warnemuende-Pappas EA. Vertical tillage impacts on water quality derived from rainfall simulations. *Soil Tillage Res*. 2015;153:155–160.
38. James WF, Barko JW. Biologically labile and refractory phosphorus loads from the agriculturally-managed upper Eau Galle river watershed, Wisconsin. *Lake Reserv Manag*. 2009;2381:37–41.
39. Goyenola G, Meerhoff M, Teixeira-de Mello F, et al. Phosphorus dynamics in lowland streams as a response to climatic, hydrological and agricultural land use gradients. *Hydrol Earth Syst Sci Discuss*. 2015;12:3349–3390.
40. Baker DB, Johnson LT, Confesor RB, Crumrine JP. Vertical stratification of soil phosphorus as a concern for dissolved phosphorus runoff in the Lake Erie Basin [published online ahead of print January 19, 2017]. *J Environ Qual*. doi:10.2134/jeq2016.09.0337.