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Research article

Environmental impacts on water resources from summer crops in rainfed and irrigated systems

Elisa Darré[a], Mónica Cadenazzi[b], Sebastián R. Mazzilli[c], Juan F. Rosas[d], Valentín D. Picasso[e,*]

[a] Departamento de Sistemas Ambientales, Facultad de Agronomía, Universidad de la República, Av. E. Garzón 780, Montevideo, Uruguay
[b] Departamento de Biometría, Estadística y Computación, Facultad de Agronomía, Universidad de la República, Ruta 3 km 363, Paysandú, Uruguay
[c] Departamento de Producción Vegetal, Facultad de Agronomía, Universidad de la República, Ruta 3 km 363, Paysandú, Uruguay
[d] Universidad ORT Uruguay and Centro de Investigaciones Económicas (CINVE), Av. Uruguay 1242, 11100, Montevideo, Uruguay
[e] Department of Agronomy, University of Wisconsin-Madison, 1575 Linden Dr., Madison, WI, 53706, USA

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ABSTRACT

Irrigation is an intensification technology to increase productivity in agricultural systems, but the impacts of irrigation on the environmental performance of crops are not well understood. We evaluated impacts on water use and quality of rainfed and irrigated systems for corn and soybean production in temperate South America using nonparametric ANOVA tests for small sample sizes. We modeled blue water footprint, ecotoxicity, N and P balance, and eutrophication potential for six farms producing corn and soybean in rainfed and irrigated systems in Uruguay. Crop yields were 5948 and 7862 kg ha^{-1} for corn and 2482 and 3423 kg ha^{-1} for soybean, under rainfed and irrigation, respectively. The average blue water footprint for irrigated systems was 264 m^3 ton^{-1} and zero for rainfed systems, with no difference between corn and soybean. The ecotoxicity was greater for soybean than for corn (1679 vs 325 CTUe kg^{-1}) but there were no statistically significant differences in ecotoxicity between rainfed and irrigated systems. Based on Usetox methodology, insecticides had a greater ecotoxic effect (3.2×10^6 CTUe ha^{-1}) than herbicides (7.3×10^4 CTUe ha^{-1}), despite the lower doses applied (insecticides: 0.51 kg ha^{-1}; herbicides: 6.83 kg ha^{-1}). The aquatic eutrophication potential (based on Impact 2002 + methodology) among rainfed and irrigated systems presented no differences (29 vs 24 kgPO$_{4\text{-eq}}$ ha^{-1} for corn and 19 vs 27 kgPO$_{4\text{-eq}}$ ha^{-1} for soybean). The standardized environmental impacts for corn calculated per ha were similar than those per kg of grain when comparing rainfed vs irrigated systems. For soybean, however, standardized environmental impacts per ha were greater in the irrigated than in the rainfed systems, but were similar per kg of grain (except for water footprint). In summary, irrigation resulted in higher productivity and increased blue water footprint than rainfed, but in the set of farms analyzed it did not significantly increase inputs use, so no differences were detected in nutrient balance, eutrophication potential, or ecotoxicity. Soybeans had greater environmental impacts than corn in ecotoxicity and N excess per unit of area, but no statistically significant difference was found in the other indicators. These indicators may be useful as a predictive tool for resource management. Decision makers should consider the trade-offs between productivity, water use, and water quality when using irrigation for intensification of crop production.

1. Introduction

Irrigation has been proposed as an intensification strategy for rainfed crops with significant yield increases (Foley et al., 2011). The world agricultural production has grown between 2.5 and 3 times over the last 50 years, while the cropped area has grown only by 12%. Noticeably, 40% of the increase in food production came from irrigated areas (FAO, 2011). However, an inadequate use of irrigation in intensive agricultural systems can negatively impact water quality due to pesticides and other hazardous substances (Brown and Harris, 2005), N and P from fertilizers (Martínez and Ojeda, 2011), and other chemicals such as sulfate, chloride, or ammonium (Abrahao et al., 2012). The inefficient water use in agricultural production has led to depletion of aquifers and reduction of river flows in some areas (FAO, 2011). The nitrogen (N) and phosphorus (P) excess from cultivated lands have accelerated the eutrophication processes (Kounina et al., 2013). Models

* Corresponding author.
E-mail addresses: e.darre.castell@gmail.com (E. Darré), mcadenazzi2010@gmail.com (M. Cadenazzi), smazzilli@fagro.edu.uy (S.R. Mazzilli), frosas@cinve.org.uy (J.F. Rosas), picassorisso@wisc.edu (V.D. Picasso).

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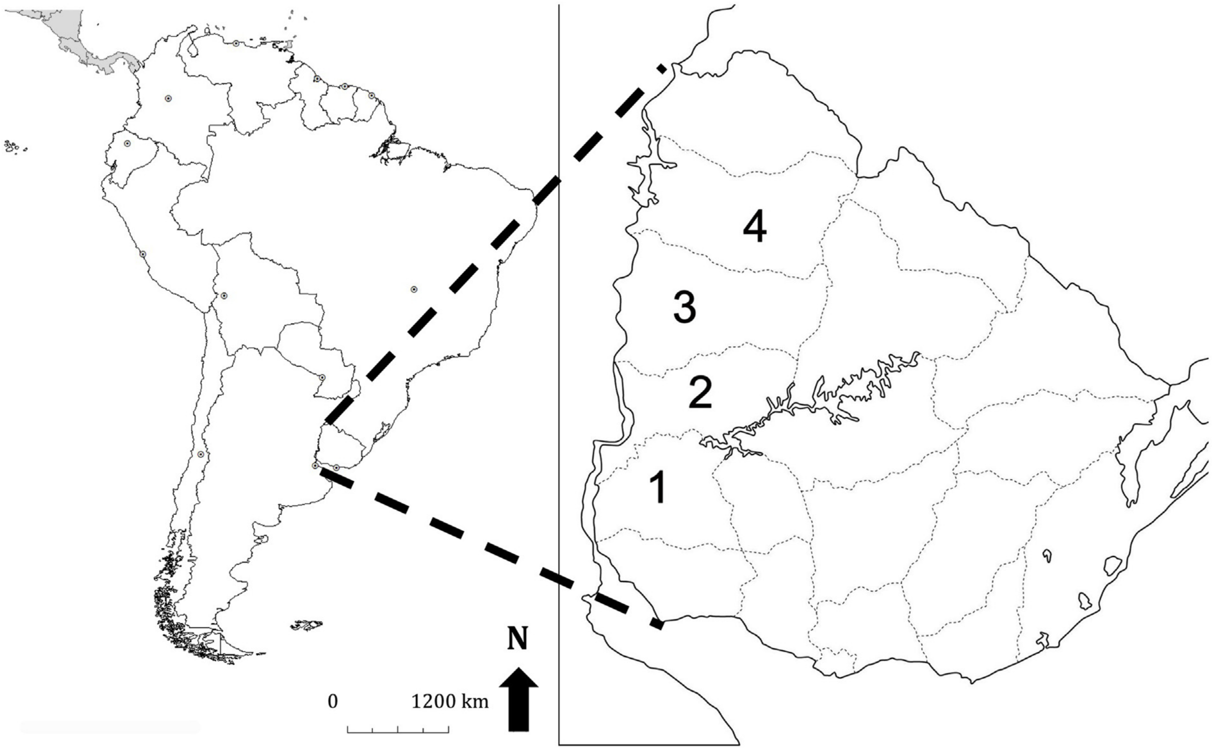

Fig. 1. Maps of South America and Uruguay, showing the location of the western Littoral area were the study cases were located: 1- Soriano, 2- Río Negro, 3- Paysandú and 4- Salto.

forecast a 20% increase of eutrophication of surface waters and coastal areas globally by 2050 (WWDR, 2015). Furthermore, acidification, ecotoxicity, and consequently loss of biodiversity has been documented in irrigated agriculture (Jolliet et al., 2014; Vidal and Kruk, 2008).

Around the world, many agricultural regions use high amounts of water for agriculture (FAO, 2011). For instance, in Uruguay, 86% of the extracted water volume was used for crop irrigation, 9% for drinking water supply, 3% for industries, and 2% for recreation and other uses (De León and Delgado, 2012). The main source is the surface water although groundwater is used for irrigation of fruits and vegetables (77% of the requested permits; De León and Delgado, 2012). Sustainable irrigation practices should be emphasized in regions where water resources are available (Dourado Hernández et al., 2013), as it is Uruguay. Mekonnen et al. (2015) suggest that improvements in rainfed crop efficiency are needed, in part, to reduce the expansion of irrigated agriculture to water scarce regions of Latin America and the Caribbean. Irrigated area in Uruguay has expanded in recent decades, especially supplemented irrigation for extensive summer crops such as corn (maize) and soybean (Failde et al., 2013), but with mixed results depending on the design and efficiency of the irrigation technology (García Petillo, 2012). These summer crops had the greatest growth in the last decade, when significant technological transformations such as irrigation have occurred (Giménez, 2017).

There is little information on the environmental impacts of irrigation in extensive crops in Uruguay. Dogliotti et al. (2004) modeled the productivity and environmental performance of horticulture crop rotations with and without irrigation, focusing on soil erosion and soil organic matter, but their model was not used for extensive crops. Nonetheless, it is known that several water bodies in Uruguay show organic and toxic pollution associated with the accumulation of non-biodegradable solid wastes (Mello de Carvalho, 2013). Acidification and presence of pesticides, herbicides and heavy metals in rivers has been linked to agricultural areas (Conde et al., 2002), and eutrophication has been associated with increased use of fertilizers (Díaz, 2013). However, there are no studies directly assessing the potential water quality impact (i.e., ecotoxicity, nutrient balance, eutrophication) and water use (i.e., water footprint) of supplemented irrigation on traditionally extensive rainfed crops in Uruguay.

Modeling environmental performance of cropping systems can help guiding private management and public policy decisions (Schultink, 2000). The environmental performance can be evaluated per ha of land area or per kg of crop produced depending on the goals of the assessment (Picasso et al., 2014). Irrigation usually increases crop yields, but also it may require greater use of inputs (pesticides and fertilizers). Therefore, environmental impacts per unit of area may increase with irrigation. However, the environmental impacts of irrigation per unit of crop production depends on the crop yield increase relative to input use increase. If irrigation increases crop yields relatively more than the increase in use of inputs, then environmental impact per unit of crop is reduced (Xiao-Tang et al., 2009). However, if the increase in inputs is relatively greater than the increase of yields, the environmental impacts of crops can be increased (Clark and Tilman, 2017). Furthermore, different crops (i.e., corn vs soybean) could have different environmental impacts due to different type and amounts of inputs used.

Therefore, the objective of this study was to estimate potential ecotoxicity, nutrient balance, water eutrophication, and blue water footprint for two summer crops with and without irrigation in Uruguay; specifically, we assessed the differences between the environmental impacts on water resources per hectare and per kg of grain. Our hypothesis was that irrigated systems would result in greater productivity, input use, and environmental impacts on water use and quality than rainfed systems. We also hypothesized that water environmental impacts would be greater for soybean than corn, based on the greater use of pesticides.

2. Materials and methods

2.1. Study cases

In Uruguay, 78% of the agricultural land for row crop production is

Table 1
Crop rotations evaluated for irrigated and rainfed systems in 6 farms in the western Littoral area of Uruguay: farm location, years evaluated, water source (rainfed vs irrigation) and sequence of winter and summer crops grown are shown. Data provided by the farmers.

Farm location and years	Water source	Year 1		Year 2		Year 3		Year 4	
		Winter	Summer	Winter	Summer	Winter	Summer	Winter	Summer
A	Rainfed	Wheat	Soy	CC - R	Soy	Wheat	Sorghum	CC - R	Soy
Soriano 2010–2014	Irrigation	Wheat	Soy	CC - R	Corn	Wheat	Soy	CC - R	Corn
B	Rainfed	Barley	Soy	ChF	Corn	CC - O	Soy		
Rio Negro 2013–2016	Irrigation	Barley	Soy	ChF	Corn	CC - O	Soy		
C	Rainfed	CC - O	Soy	CC - O	Corn				
Rio Negro 2014–2016	Irrigation	CC - O	Soy	CC - O	Corn				
D	Rainfed (r) and Irrigation (i)		Rice	CC - O	Soy (i)	CC - O	Soy (i)	CC - O	Corn (i)
Salto 2012–2016			Rice	CC - O	Soy (i)	CC - O	Soy (i)	CC - O	Soy (i)
			Rice	CC - O	Soy (r)	CC - O	Soy (r)	CC - O	Sorghum (r)
E	Rainfed (r) and Irrigation (i)								
Rio Negro 2011–2014		CC - O	Soy (i)	CC - O	Corn (r)	Wheat (r)	Corn (i)	CC - O	Soy (r)
F	Rainfed (r) and Irrigation (i)								
Rio Negro 2011–2014		ChF	Corn (r)	CC - O	Soy (i)	CC - O	Corn (i)	CC - O	Soy (r)

CC - R: Ryegrass cover crop. CC - O: Oats cover crop. ChF: Chemical Fallow: period between the first herbicide treatment that takes place after the harvest and the direct seeding of the next crop.

located in the western Littoral region (Fig. 1) generating an intensive use of land (Uruguay XXI, 2016). In recent years there has been a 138% increase in land for agricultural use in the country, led by the increase in soybean area (1,140,000 ha), becoming the main export product and turning Uruguay one of the six main exporters in the world. Average yields are 1937 kg ha^{-1} (MGAP-DIEA, 2016; Uruguay XXI, 2016). Similarly, the area under irrigation has expanded from 52,000 ha in the 70s to approximately 222,000 ha in 2012, mainly from extensive cereals, oilseeds, and pastures (Sawchik, 2012). Rice is produced under irrigation with 161,000 ha and average yields of 8094 kg ha^{-1} placing Uruguay as one of the ten largest exporters of rice in the world. Winter crops are wheat (330,000 ha, 3610 kg ha^{-1}) and barley (93,000 ha, 3610 kg ha^{-1}), and other summer crops are corn (83,000 ha, 5867 kg ha^{-1}) and sorghum (67,000 ha, 3606 kg ha^{-1}) (MGAP-DIEA, 2016; Uruguay XXI, 2016).

Data of inputs and yields were obtained from six farms in the western Littoral region of Uruguay over three to six years from 2010 to 2016 (Table 1). In this area the dominant soils are developed on Cretaceous-age sandstones, clay loess and loess. The argiudoles associated with hapluderts predominate. The topography is gently undulating. Natural fertility is high in the southern zone (Soriano and Río Negro) and is average in Paysandú and Salto. Considering the favorable physical conditions, these are considered the most productive agricultural soils in Uruguay. The dominant native vegetation is grasslands and grasslands associated with trees. The average annual temperature in Uruguay is 17.7 °C, ranging from 19.8 °C in January and February to 16.6 °C in June and July. The annual accumulated precipitation range between 1200 and 1600 mm (Castaño et al., 2011). Climate allows for two crops per year: "winter" crops (planted in fall and harvested in spring, like spring wheat and barley) and "summer" crops (planted in spring and harvested in fall, like soybean, corn, sorghum, and rice). Due to soil erosion concerns, there is an increased use of cover crops like oats and ryegrass, which are planted after the summer crops to avoid a winter fallow period and to provide agroecosystem services like soil cover and erosion control, but not harvested for grain. A description of the six study cases is presented in Table 1. The first three farm cases (A, B, and C) provide a side by side comparison of two rotations with and without irrigation. Farm cases D, E, and F, provide information of farms where irrigation was used for some crops in the rotation, but not others. For the analyses we included only the crops that were both irrigated and rainfed in the same farm. These six farms were selected by experts (agronomists, extension agents) because they used crop rotations that are typically for the region, they used irrigation technologies based on best management practices, and they kept good records of inputs used and yields obtained. Therefore, these farms are considered the best-case scenario for yields and environmental performance among commercial systems in Uruguay.

2.2. Environmental impact estimation models

The estimated environmental impacts were: blue water footprint, ecotoxicity, nutrient balance, and eutrophication. Other environmental impacts such as green and grey water footprint, acidification, other chemical pollutants, were not included in this study because the methodologies to compute them require data not available at the farm level. However, we consider that the selected indicators provide enough breadth of environmental impacts for comparing rainfed and irrigated systems. Further research is required in the future to expand the impact assessment.

Each crop within each farm was considered a system with inputs and outputs (Fig. 2), for evaluating each environmental impact. The input accounted for water footprint was irrigated water only (see next section). The inputs accounted for nutrient balance and eutrophication were synthetic fertilizers, biological nitrogen fixation (BNF), and atmospheric deposition, and the outputs were nutrients harvested in grain. The inputs accounted for ecotoxicity were all the pesticides applied. Although there are nutrients and pesticides that remain in the farm, or get lost to air, we did not account for those outputs in the model. Therefore, all our results should be considered potential maximum values, not actual nutrients or pesticide losses.

2.2.1. Blue water footprint

The water footprint is a multidimensional indicator that allows the characterization of the volume of water used for the production of a good or service, considering the volume of fresh water consumed and contaminated in the process (Hoekstra et al., 2011). It is expressed in terms of m^3 ton^{-1}, and can be subdivided in three types: blue water footprint (the consumption of surface and groundwater extracted - when it corresponds); green water footprint (the volume of rainwater that is consumed by the vegetation and does not become runoff); and grey water footprint (the volume of fresh water required to assimilate a load of pollutants). The calculation of the blue water footprint was chosen to specifically compare crops under the irrigation system, according to the following equation: blue water footprint (m^3 ton^{-1}) = CWU/Y; where: CWU is the Crop Water Use - based on the irrigation water applied, expressed in m^3 ha^{-1}, and Y is the crop yield, expressed in ton ha^{-1}. The information on irrigated water applied and yields were obtained from the farm records (Table 2).

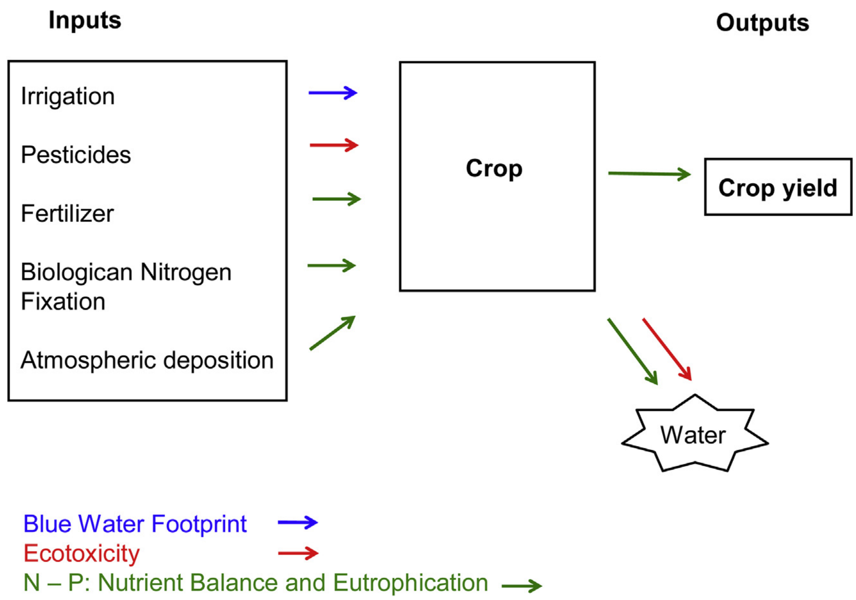

Fig. 2. Model representation of the production system, indicating inputs and outputs used for assessing the environmental impacts of crop production.

2.2.2. Ecotoxicity

The ecotoxicity potential for each crop in water was calculated according to the equation: $EP = \Sigma_i (S_i \times CF_i)$ where: EP is the ecotoxicity potential expressed in Comparative Toxic Units (CTUe), and was calculated for each type of pesticide (fungicides, insecticides, herbicides) per crop; S_i is the rate in kg ha^{-1} of pesticide i used in the crop (dose), multiplied by the amount of active ingredient present in the pesticide (kg); and CF_i is the characterization factor, defined as the potentially affected fraction (PAF) of species per cubic meter of water in a day and per unit mass of the compound or pollutant emitted (PAF m^3 day kg$_{emitted}$$^{-1}$) (Rosenbaum et al., 2008). These characterization factors (Supplemental Table 1) where obtained by the Usetox methodology (www.usetox.org). This indicator depends only on the inputs used per unit of area and not on crop yields. We also calculated the ecotoxicity per kg of grain as EPG (CTUe kg^{-1}) = EP (CTUe ha^{-1})/Yield (kg ha^{-1}).

2.2.3. Nutrient balance (N and P)

The nutrient balance analysis of the systems followed the methodology of Koelsch and Lesoing (1999). We computed the nutrients excess (NE) of the system as the difference between nutrient inputs and outputs per unit of area. Values above 0 indicate nutrient excess suggesting contamination risk. The inputs included are: (i) the proportion of N or P of each fertilizer product used (based on the description of laguiasata.com) multiplied by the fertilizer dose (reported by the farmers) in kg ha^{-1}, (ii) the estimated N biological fixation (for soybean) as the 60% of the harvested N (Collino et al., 2015; Salvagiotti et al., 2008) and (iii) the natural atmospheric N deposition in kg ha^{-1} (Carnelos et al., 2014). In the case of the outputs, we included the N and P harvested in the grain which were estimated based on 88% dry matter, 1.44% and 7.18% N, and 0.30% and 0.64% P for corn and soybean respectively (Fernández Mayer, 2014). We also calculated the nutrient excess per unit of grain as NEG (kg ha^{-1}) = NE (kg ha^{-1})/ Yield (kg ha^{-1}).

The fertilizers used in the crop rotations (and their N-P-K content) were Diammonium phosphate (18-46-0), Super concentrated nitrogen (7-40-5), Potassium phosphate (0-40-20), FanafosK nitrogen (6-30-0), Superphosphate (0–20/22-0), Urea (46-0-0), Sol UAN (32-0-0), Solmix 28 (28-0-0), Solmix 26 (26-0-0), Potassium chloride (0-0-60), and 0–33/34-0 +6S, 25-33-0, 2–25/25-25 +3S.

2.2.4. Eutrophication potential

We calculated the aquatic eutrophication potential (AEP), which considers N and P losses from the system (Brentrup et al., 2004) and is

Table 2
Averages of input (water, pesticides, N and P fertilizers) use and yields of crops (corn and soybean) evaluated for irrigated and rainfed systems from six farms in the western Littoral area of Uruguay. Farm codes correspond to Table 1. Summary of data provided by farmers.

Farm	Water system	Corn					Soy				
		Crop yield kg ha^{-1}	Water use m^3 ha^{-1}	Total Pesticides kg ha^{-1}	Total N fertilizer kg ha^{-1}	Total P fertilizer kg ha^{-1}	Crop yield kg ha^{-1}	Water use m^3 ha^{-1}	Total Pesticides kg ha^{-1}	Total N fertilizer kg ha^{-1}	Total P fertilizer kg ha^{-1}
A	Rainfed	–	–	–	–	–	2150	0	7.0	5	12
	Irrigated	7500	2400	8.5	120	24	2600	1600	4.6	4	11
B	Rainfed	5400	0	11.9	105	18	2500	0	10.2	0	7
	Irrigated	10,400	2300	12.2	171	26	3500	1850	10.2	0	12
C	Rainfed	5000	0	10.7	118	41	2957	0	17.7	30	15
	Irrigated	7000	1580	10.1	150	41	3613	790	17.7	30	15
D	Rainfed	–	–	–	–	–	1300	0	2.3	4	11
	Irrigated	7500	3200	9.7	145	48	3300	1350	8.7	8	21
E	Rainfed	8786	0	3.7	85	24	2872	0	5.6	0	22
	Irrigated	5596	210	5.6	87	11	4037	450	2.9	1	16
F	Rainfed	4605	0	5.3	121	12	3296	0	5.6	0	22
	Irrigated	8452	1680	5.7	83	22	3214	350	2.9	10	27

computed as the amount of a toxic substance (i.e., the nutrient excess calculated in the previous section) multiplied by the characterization factor of the substance (Rosenbaum et al., 2008). The characterization factors of the N and P were obtained from the IMPACT 2002 + methodology (N: 0.10; P: 3.06, Jolliet et al., 2003). These characterization factors are expressed in $kgPO_4$-equivalents kg^{-1} and the AEP in $kgPO_4$-equivalents. We calculated AEP both per unit of ha and per unit of grain. This indicator represents a potential value, whereas the actual eutrophication depends on additional factors such as soil type, rainfall, evapotranspiration, runoff, among others, not included in this study.

2.3. Data analyses

Estimates for each environmental variable were computed for each individual crop in each crop rotation, both in rainfed and irrigated systems. We compared the impacts of the two crops of interest (corn *vs* soybean) and also, we compared the impacts of the two water management systems of interest (irrigated *vs* rainfed) for each crop. ANOVA and t-tests to compare between systems were conducted using two models. The first model considered crop (corn or soybean) and farm (1–6) as fixed effects. The second model considered water management system (irrigated or rainfed) and farm as fixed effects (analyses performed by crop). Because the data did not comply with the corresponding ANOVA assumptions (normality and homogeneity of variance) and sample size was small, an equivalent nonparametric method was applied (Kruskal-Wallis and Mann-Whitney). The InfoStat software was used for the statistical analysis (Di Rienzo et al., 2017).

In order to compare multiple environmental impacts, data were presented as an amoeba plot, showing standardized values for each environmental indicator computed for the crops under the irrigation and rainfed systems. The value of each system was divided by the maximum value obtained from all systems to standardize each variable.

3. Results

3.1. Comparison of irrigated vs rainfed systems

Crop yields were greater for irrigated vs rainfed systems on average (Table 3). As expected, blue water footprint was greater for irrigated systems *vs* rainfed (Table 3). Rainfed systems had a blue water footprint of zero because no water is extracted, while irrigated systems had a blue water footprint of 264 ± 136 m^3 ton^{-1}.

There were no differences between rainfed and irrigated systems for ecotoxicity both per hectare and per kg of grain (Table 3). There were no statistically significant differences in ecotoxicity between rainfed and irrigated systems. Based on Usetox methodology, insecticides had a greater ecotoxic effect (3.2E+06 CTUe ha^{-1}) than herbicides (7.3E+04 CTUe ha^{-1}), despite the lower doses applied (insecticides: 0.51 kg ha^{-1}; herbicides: 6.83 kg ha^{-1}). The highest rates of pesticides applied corresponded to the herbicides, being Glyphosate and 2–4 D Amine the ones that topped the list for both crops. Herbicides comprised 90% of the total pesticides applied, nevertheless, insecticides were the ones that represented the greatest contribution of ecotoxic impact (99%) of the total. The most common active ingredients of the insecticides were: Imidacloprid, Lambdacihalotrina, Beta cyfluthrin, Tiametoxam, Triflumuron, Bifenthrin and Chlorpyrifos.

No differences were found for excess of nutrients or eutrophication potential between rainfed and irrigated systems (Table 3). Excess of N was greater than excess of P, but the eutrophication potential was greater for P.

3.2. Comparison of corn vs soybean systems

No differences for the blue water footprint were found between the corn and soybean (Table 3). Soybeans had more than double the total ecotoxicity than corn expressed per unit of area (4.7 vs 1.9 E+6 CTUe ha^{-1}), and more than five times expressed per kg of grain (1679 vs 325 CTUe kg^{-1}). This was mainly explained by the greater ecotoxicity of insecticides in soybean, despite the fact that herbicide ecotoxicity was greater for corn (Table 3). The N excess per area was 50% greater in soybean than in corn, and no differences were found for P excess; no differences were found for total eutrophication potential for freshwater toxicity between crops (Table 3).

3.3. Environmental impacts per unit of area vs per unit of product

Standardized environmental indicators were calculated dividing each value by the maximum value of the indicator for all combinations of crops and water management systems for indicators per unit of land and per unit of product (Fig. 3). The environmental impacts calculated per ha and kg of grain show a similar pattern for corn systems, where nutrient impacts and ecotoxicity are greater for rainfed systems (although no statistical differences were detected), and blue water footprint is significantly greater for irrigated systems (Fig. 3). In the soybean case, results differ when accounted per unit of land or unit of product. Values per ha show that rainfed systems have less impacts for all indicators than irrigated systems (although no statistical differences were detected). In contrast, values per kg of grain are similar for both water management systems for nutrients and ecotoxicity, but not for water footprint (where irrigated systems have more impact than rainfed, Fig. 3).

4. Discussion

4.1. Blue water footprint

The average blue water footprint of irrigated corn was 171 ± 77 m^3 ton^{-1} (yield: 7.9 ± 1.8 ton ha^{-1}) and for soybeans of 357 ± 196 m^3 ton^{-1} (yield: 3.4 ± 0.4 ton ha^{-1}). This is consistent with the estimations of Mekonnen and Hoekstra (2011) who indicated that the higher the yield, the smaller the water footprint should be. Their study showed the world average blue water footprint (in the period 1996–2005) for corn was 294 m^3 ton^{-1} (yield: 6.0 ton ha^{-1}) and 926 m^3 ton^{-1} (yield: 2.5 ton ha^{-1}) for soybean. Our results of blue water footprint of irrigated systems are lower than the world averages. The difference between these crops has also been observed in Argentina but water footprint was greater for corn (220 m^3 ton^{-1} for corn vs 130 m^3 ton^{-1} for soybean, Zarate et al., 2014). The low yields observed in our study for crops under irrigation could be associated with an inefficient use of water (Lankford, 2012). This could be enhanced by improving the management of the irrigation strategy, which would translate into an improvement of the use of water resources (Álvarez et al., 2016), for example, adjusting the water balance models (Giménez et al., 2016, 2017). Another possible explanation for the low yields observed is soil degradation due to continuous agricultural use that negatively impacts physical and chemical soil properties (Ernst et al., 2018).

4.2. Ecotoxicity

The lack of differences for ecotoxicity between irrigated and rainfed crops might be due to the similarity in pesticides rates used in both systems, but this does not mean a lack of ecotoxicological effect given that the type and combination of pesticides applied influence the ecotoxicity values (Nordborg et al., 2014; Yang and Suh, 2015). Indeed, pesticides and especially insecticides have serious impacts in human and ecosystem health (Juraske and Sanjuán, 2011). In this study, the ecotoxic effect of the insecticides was greater than the effect of herbicides, even though herbicides were applied in larger doses. At the same time, the ecotoxicity for soybean was higher than for corn, which has been documented in the region (Bennet, 2012).

Crops grown in Europe have often lower ecotoxicity effect than

Table 3
Mean (M) and standard deviation (SD) for yields and environmental indicators (blue water footprint, pesticide ecotoxicity, N and P balance, and N and P eutrophication potential) for corn and soybeans from irrigated and rainfed systems in six farms in the western Littoral area of Uruguay, and p-values for two model comparisons: comparing crops (corn vs soybean), and comparing rainfed vs irrigation systems for each crop. P-values < 0.05 are highlighted in bold. EP = Ecotoxicity Potential (per ha); EPG = Ecotoxicity Potential per unit of grain; AEP = Aquatic Eutrophication Potential.

	Corn				Soy				Corn vs Soy	Rainfed vs irrigated	
	Rainfed		Irrigated		Rainfed		Irrigated			Corn	Soy
	M	SD	M	SD	M	SD	M	SD	p-values		
Yield (ton ha^{-1})	5.9	1.7	7.9	1.8	2.5	0.7	3.4	0.4	**< 0.01**	0.20	**0.01**
Irrigation (m^3 ha^{-1})	0	0	1443	763	0	0	1177	600	0.88	**0.03**	**< 0.01**
Blue water footprint (m^3 ton^{-1})	0	0	171	77	0	0	357	196	0.66	**0.03**	**< 0.01**
Fungicides EP (10^3 CTUe ha^{-1})	1.3	2.2	3.8	2.2	5.1	5.1	7.0	11.6	0.58	0.49	0.69
Insecticides EP (10^6 CTUe ha^{-1})	1.8	2.8	1.8	2.8	3.8	3.0	5.4	2.9	**0.02**	0.97	0.51
Herbicides EP (10^6 CTUe ha^{-1})	0.1	0.1	0.1	0.1	0.0	0.1	0.0	0.1	**< 0.01**	0.74	0.44
Total EP (10^6 CTUe ha^{-1})	1.9	2.8	1.9	2.8	3.9	3.0	5.4	2.8	**0.02**	0.89	0.52
Fungicides EPG (CTUe kg^{-1})	0.1	0.2	0.5	0.3	3.0	4.3	2.1	3.3	**< 0.01**	0.26	0.33
Insecticides EPG (CTUe kg^{-1})	358	568	250	405	1778	1370	1560	777	**< 0.01**	0.97	0.81
Herbicides EPG (CTUe kg^{-1})	22	23	19	13	8	19	7	16	**0.04**	> 0.99	0.15
Total EPG (CTUe kg^{-1})	380	563	270	398	1789	1362	1569	763	**< 0.01**	> 0.99	0.81
N Inputs (kg ha^{-1})	112	14	128	38	105	28	143	19	0.68	0.89	**0.03**
N Outputs (kg ha^{-1})	75	21	100	23	157	42	216	28	**< 0.01**	0.20	**0.01**
N Excess (kg ha^{-1})	48	18	38	20	52	18	73	15	**0.04**	0.49	0.10
N Excess (kg kg^{-1})	0.7	0.4	0.4	0.2	0.3	0.1	0.3	0.0	0.54	0.34	0.69
P Inputs (kg ha^{-1})	24	11	25	11	13	7	18	6	0.08	0.97	0.37
P Outputs (kg ha^{-1})	16	4	21	5	14	4	19	3	0.63	0.20	0.01
P Excess (kg ha^{-1})	8	11	7	9	5	3	6	3	0.22	0.97	0.15
P Excess (kg kg^{-1})	0.6	0.9	0.4	0.5	0.3	0.3	0.3	0.1	0.31	0.74	0.23
N AEP ($kgPO_{4\text{-eq}}$ ha^{-1})	4.8	1.8	3.8	1.9	5.2	1.8	7.3	1.5	**0.04**	0.49	0.10
P AEP ($kgPO_{4\text{-eq}}$ ha^{-1})	24.7	34.1	20.6	27.3	13.9	10.1	19.4	7.6	0.22	0.89	0.15
Total AEP ($kgPO_{4\text{-eq}}$ ha^{-1})	29.4	34.8	24.3	28.9	19.1	10.5	26.8	7.8	0.19	0.69	0.15
N AEP ($kgPO_{4\text{-eq}}$ kg^{-1})	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.44	0.40	0.90
P AEP ($kgPO_{4\text{-eq}}$ kg^{-1})	1.8	2.6	1.1	1.5	1.0	0.9	1.1	0.4	0.31	0.63	0.23
Total AEP ($kgPO_{4\text{-eq}}$ kg^{-1})	1.9	2.6	1.2	1.5	1.0	0.9	1.1	0.4	0.36	0.51	0.23
Number of cases	4		4		7		7				

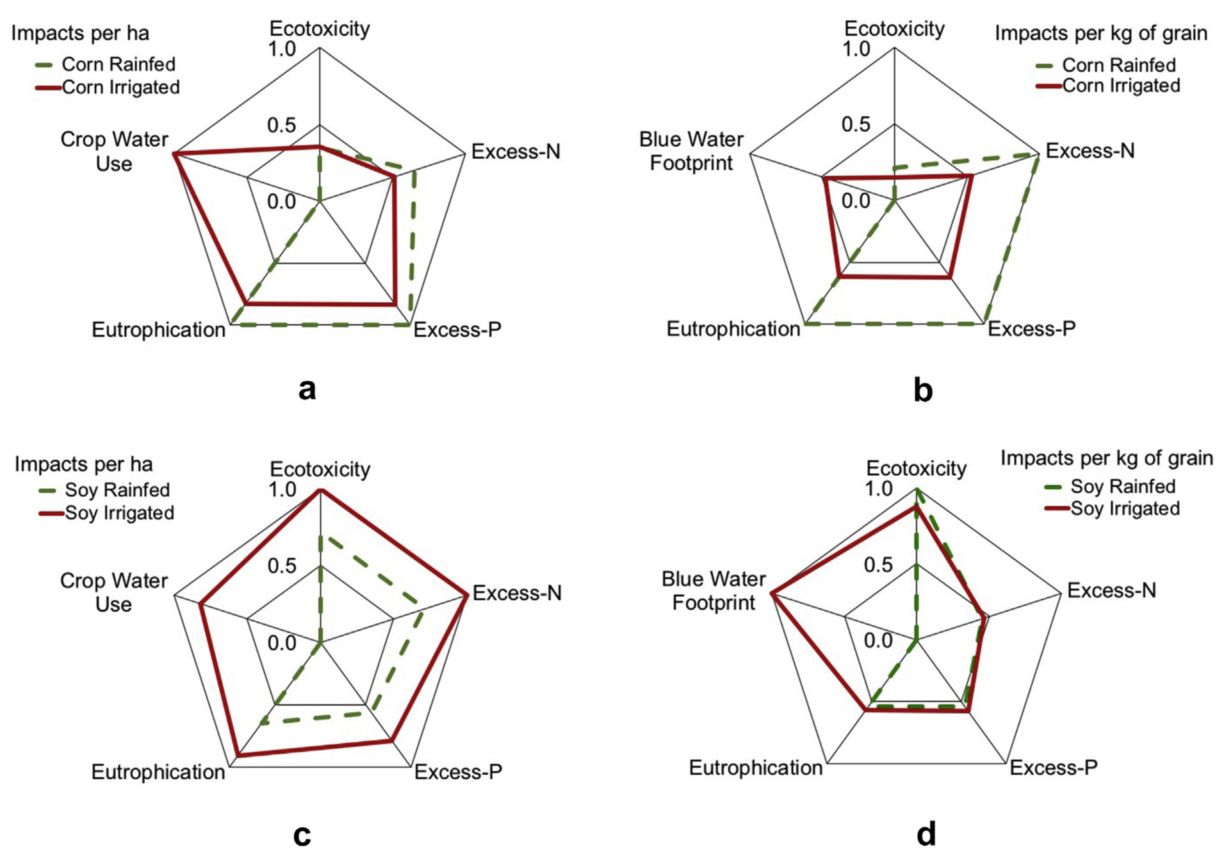

Fig. 3. Standardized environmental indicators for soybean and corn in irrigated and rainfed systems, presented per unit of land (ha) and per unit of grain (kg) analyzed in the western Littoral area of Uruguay. See Table 3 for statistical comparisons.

those grown in North and South America, due to the fact that most toxic pesticides are banned in Europe (Nordborg, 2013). The active compounds present in the most prominent insecticides used in the farms analyzed here (Imidacloprid and Tiametoxam - chemical group: Neonicotinoides; Lambda cyhalothrin, Beta cyfluthrin, Deltamethrin and Bifenthrin: chemical group: Pyrethroids; Chlorpyrifos chemical group: Organophosphates; Triflumuron and Lufenuron - chemical group: Benzoylureas), have toxicity effects at different trophic levels, from algae, crustaceans, fish, bees, birds, to humans. Environmentally, they present different degree in soil's mobility, water solubility, persistence in the water's sediment and different degrees of bioaccumulation (Aparicio et al., 2015), thus, generating a long-term effect. Lambda cyhalothrin, for example, is classified according to the database of pesticides of the European Union as very high toxicity to aquatic organisms and harmful at long-time effects (Nordborg et al., 2017), and small doses generate large negative effects on the environment (Münze et al., 2015; Nordborg et al., 2014). Therefore, insecticides have major ecotoxic impacts (Bunzel et al., 2015; Räsänen et al., 2015). The insecticide thiamethoxam + lambda-cyhalothrin has one of the highest ecotoxic effects, although one of the active ingredients (Thiamethoxam) is not in the Usetox program and its effect could not be assessed in this study.

On the other hand, atrazine was one of the herbicides that presented the highest value of ecotoxic impact and one of the pesticides with the highest application rate. This herbicide was recently banned in Uruguay (Presidencia de la República, 2016). However, it may cause long-term adverse effects in the aquatic environment, because it is very persistent in water sediments, it has medium to high persistence in soil, and it has high to extreme mobility. Atrazine is classified as very toxic to aquatic organisms, given to it is an endocrine disruptor and has reproductive effects (Fleeger et al., 2003; Graymore et al., 2001; Rohr and McCoy, 2010). In another study that compared grey water footprint (an indicator of freshwater pollution) of crops in South America, Uruguay presented one of the highest values compared to other countries in the region, and pesticides had higher grey water footprint than fertilizers (Carmona Vega, 2010).

4.3. Nutrient balance

When N and P from fertilizers exceed crop requirements, their excess becomes a risk to the water quality in terms of eutrophication and changes in the trophic chain (Smith et al., 1999; van Wijnen et al., 2015; Withers and Eunice, 2002). The analyzed irrigation cases seem to have good agronomic management given the relationship between crop yield and nutrient inputs and therefore, the lack of differences in the excess of nutrients. Soybean requires a significant amount of nutrients to achieve adequate growth and yield, presenting superior nutrient requirements per kg produced than corn (García, 2004). Corn is a crop with high water requirement and has a high sensitivity to water stress, since droughts during the vegetative period can affect yield (Aparicio et al., 2002).

In addition, there is a relationship between water availability during the crop cycle and the fertilization's response (Villalobos González et al., 2016). The observed difference between yields (rainfed – irrigated: 1910 kg ha^{-1}) for corn was lower than previously reported yield differences for irrigated vs rainfed corn in Argentina, ranging from 2533 to 4164 kg ha^{-1} (Aquiles et al., 2012; Ehrt et al., 2003; Pedrol et al., 2008). This could indicate that N use efficiency could be improved (Dobermann, 2007). The fertilizer inputs applied to corn under irrigation could be four times the fertilizer inputs in rainfed systems, but if not properly managed, these nutrients could be rapidly transported to water (Monteagudo et al., 2012). Furthermore, intense rainfall events play an important role in the transport of such nutrients (van Geer et al., 2016); heavy storms or intense rain events, can generate considerable losses of particulate P, associated to the soil erosion (Mohamadia and Kavian, 2015). The amount of dissolved P from fertilizers may be relatively large in the runoff generated during the first storm after application (Vadas et al., 2008). Therefore, improvements in N and P use efficiency must be prioritized for conserving water quality (van Wijnen et al., 2015).

4.4. Eutrophication potential

Phosphorus is the limiting nutrient in temperate freshwater lakes (Blomqvist et al., 2004) but nitrogen could be limiting primary production in intermediate latitudes (Fabre et al., 2010), so the excess of both nutrients must be considered to assess eutrophication potential. In Uruguay, P concentration for several rivers (e.g., Santa Lucia, de la Plata, Negro, and Uruguay) were higher than the current standard established in regulation (decree 253/79); and cyanobacteria blooms have also been reported (Kruk et al., 2013). The water quality is deteriorated in all the country, with many rivers being above the eutrophic limit (Chalar et al., 2013; Kruk et al., 2015; Rodríguez-Gallego et al., 2017).

The lack of the observed differences in eutrophication potential between irrigated and rainfed systems in our study could be related to the few cases included. Nevertheless, there was a trend between crops similar to the one reported in other studies in the region, where values of the total AEP for corn ($1.12\,E^{-03}$) were higher than those of soybean ($5.68\,E^{-04}$) (Franzese et al., 2013), as well as higher values for the AEP N for soybean (Austin et al., 2006). This trend was also observed for the grey water footprint in the region, where the corn had the highest values given the fact it is a heavily fertilized crop (Mekonnen et al., 2015; Zarate et al., 2014). On the other hand, Uruguay seem to have lower values for total AEP for soybean, comparing to countries like Brazil and Argentina (Geraldes Castanheira, 2014).

5. Conclusions

Our results support the hypothesis that the use of irrigation in agricultural systems resulted in greater productivity and blue water footprint than rainfed systems. However, here were no differences in nutrient balance, eutrophication potential, or ecotoxicity because in the farms analyzed, the irrigated systems did not increase much the fertilizer or pesticide inputs compared to the rainfed systems.

Ecotoxicity was greater for soybean than for corn, given the greater use of insecticides in soybean. No significant differences were found in blue water footprint and eutrophication between corn and soybean systems in these farms.

Because our methodology was based on case studies from real farms, with data generated over different years, our results should be interpreted considering this limitation: they don't come from an experiment under controlled conditions. Climate variability across years, for instance, is a relevant variable that was not controlled in our study and may affect the results. This is one possible reason why we did not find many significant differences between systems.

According to the World Program for the Evaluation of Water Resources, Uruguay is classified as one of the best countries respect to the water quality and availability in Latin America (WWDR, 2016). Uruguay has been highlighted for its great productive potential due to its richness in water resources, however, the greatest challenge is the proper management of these resources. Given the prospects of increasing the production of food for export markets, and the current policies of promotion of irrigation, policy makers and society in general should carefully consider the trade-offs between productivity, water use, and water quality involved in the intensification of crop production based on irrigation technologies.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at https://doi.org/10.1016/j.jenvman.2018.11.090.

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