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Soil, water and nutrient loss under conventional and organic vegetable production managed in small farms versus forest system

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Abstract

Agricultural systems with conventional tillage and intensive use of agrochemicals, especially those on high slopes and with shallow soils, have the potential to release pollutants. This study aimed at evaluating the soil, water and nutrient lost via agricultural runoff in large plots (small catchments) under conventional and organic farming of vegetables as well as under forest (control) system in a Cambisol in the Campestre catchment. Samples of runoff were collected biweekly for one year through a Coshocton wheel. The soil and water losses from the conventional farming were 218 and 6 times higher, respectively, than forest. Under organic farming the soil and water losses were 12 and 4 times higher, respectively, than forest. However the soil losses (0.5 to 114 kg ha $^{-1}$ year^{-1}) are considered low in agronomy but environmentally represent a potential source of surface water contamination by runoff associated pollutants. The concentrations and losses of all forms of phosphorus (P) were higher in the conventional system (9.5, 0.9 and 0.3 mg L^{-1} of total P for conventional, organic and forest systems, respectively), while the organic system had the highest concentrations and losses of soluble nitrogen (4.7, 38.6 and 0.4 mg L $^{-1}$ of NO_3-N, respectively). The percentage of bioavailable P was proportionally higher in the organic system (91 % of total P lost was as bioavailable P), indicating greater potential for pollution in the short term.

Keywords: farming systems, catchment, runoff, manure, phosphorus, nitrogen

1 Introduction

Runoff from agricultural fields is the primary source of diffuse pollution. Nutrients in runoff may be carried in soluble fractions or adsorbed onto sediment (Sharpley *et al.*, 2001). Nitrogen (N) and phosphorus (P) are essential nutrients for plants and are therefore, extensively applied in agricultural production. However, they may cause pollution (Hart *et al.*, 2004; Kay *et al.*, 2009). Eutrophication is the main problem associated with these nutrients (Correll, 1998; Daniel *et al.*, 1998; Smith *et al.*, 1999); however, the nitrogen as nitrate (NO_3-N) is also associated with human health problems and nitrogen as ammonium (NH_4-N) is inimical to aquatic environments (Smith *et al.*, 1990).

Conservation tillage systems have proven effective in controlling soil loss mainly by increasing soil cover

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(Derpsch *et al.*, 1991) which reduces particulate nutrient losses. Soluble nutrient losses may however, still occur (Sharpley & Halvorson, 1994; Bertol *et al.*, 2003) via water loss (Cogo *et al.*, 2003). Organic farming, a conservation agricultural system, relies on a number of farming practices based on ecological cycles, and brings benefits towards reducing soil erosion (Arnhold *et al.*, 2014) due to a much better vegetative cover and soil structure which brings more resistance to soil erosion processes (Gomiero *et al.*, 2011).

Manure application is a common practice in organic farming. When properly used, it is an excellent way to recycle nutrients and add organic matter to the soil (Eghball *et al.*, 2002) improving the medium's chemical, physical and biological attributes and consequently improving crop productivity (Haynes & Naidu, 1998; Ndayegamiye & Cote, 1989; Mellek *et al.*, 2010; Gomiero *et al.*, 2011). However, successive applications to soil may result in negative effects on the aquatic system through runoff and or leaching (Hooda *et al.*, 2000; Shigaki *et al.*, 2006).

According to Willer & Lernoud (2013), 37.2 million hectares of agricultural land are organic. In Brazil, about 1.8 % of farms are under organic management. Organic farming is very popular in Parana state and among the main crops under cultivation are the vegetables (Hamerschmidt *et al.*, 2000). The organic farming of vegetables in Parana occupies 1,231 hectares with 1,208 farms, characterized by small family farms (Popia *et al.*, 2000). The metropolitan region of Curitiba accounts for 70 % of the vegetable production of Parana, and Colombo city is the main producer, with 36 % of vegetable production in the metropolitan region of Curitiba (Almeida, 2003).

In the Campestre catchment (Colombo, Parana, Brazil), agriculture is mainly vegetable production characterized by family labor using the conventional system of plowing and harrowing and heavy application of mineral and organic fertilizers (poultry litter). It is also common to use water from streams for irrigation. Some farms have adopted an organic system which is characterized by no chemical pesticides usage, application of organic fertilizers, soil tillage predominantly by animal traction and high diversification of crops. Most of the land occupied by agriculture is not recommended for intensive use due to the shallow soils and steep slopes and should be occupied by pasture or forest (Ribeiro *et al.*, 2014). Moreover, almost 50 % of the riparian zone (30 m each side of the river) is not covered by native vegetation. This fragile environment with intensive production of vegetables mainly under conventional tillage has a large potential of water contamination by soil and water losses with runoff associated pollutants as nutrients and pesticides.

To find technologies that combine economic and environmental sustainability is a challenge, and organic farming can be a good alternative for clean food production. Research that focuses on quality of soil and water are required to achieve this. The aim of this study was to evaluate the water, sediment, phosphorus and nitrogen loss in large plots (small catchments) with intensive vegetable production under organic and conventional systems managed by family farmers using a forest system as a reference. This research may help to design best management practices (including organic production) for ecologically fragile soil environments.

2 Materials and methods

2.1 Characterization of the experimental area

The study was conducted on three large plots (small catchments) in the Campestre catchment, Colombo, Parana state, Brazil, located on S 25º17′ W 49°13′, at an altitude of 1,027 meters. The climate is a subtropical humid mesothermal (Cfb) by Köppen and the original vegetation was Tropical Forest. The average annual rainfall is between 1,400 and 1,600 mm. Agricultural production at the Campestre catchment follows the family farming model (less than 30 hectares and the income is exclusively from the farm), predominantly a conventional system with some farmers producing organic crops. Besides the vegetable production systems (organic and conventional) a third area consisting of secondary forest was used as reference (control). We chose similar sites, regarding size of the area, soil type, slope, texture and shape of the slope, but some differences were unavoidable as this is a field study on a catchment scale. Also, soil management and cropping in this area was done by each farmer following their usual systems, so results of this study represent the farmer common practice.

This study was carried out on two paired small farms and, due to the high diversity of soil and crop management, it was not possible to replicate the system. We therefore studied one large plot for each management system (organic, conventional and forest).

The organic system was 0.32 ha with an average slope of 18 % and generally clay textured (500, 440, 60 g kg^{-1} clay, silt and sand, respectively, at 0.0–0.2 m). This area has been managed by a farmer under the organic system with manure fertilization (poultry litter) for over ten years. The soil management was restricted to ploughing with animal traction in bands of about 6 to 10 meters wide, across the slope. Seedlings were planted manually on small ridges, with planting time and species type alternated between bands in order to have crops at different development stages. The main crops were lettuce (*Lactuca sativa* L.), swiss chard (*Beta vulgaris* L.) and broccoli (*Brassica oleracea* L.). Vegetable crops were growing in the area all year round.

The conventional system had 0.30 ha area with an average slope of 12 % and medium texture (280, 370, 350 g kg^{-1} clay, silt and sand, respectively, at 0.0–0.2 m). In this system, the farmers used machinery for soil tillage (ploughing and harrowing) and applied both mineral and organic (poultry litter) fertilizer. This area had a single summer crop (cauliflower – *Brassica oleracea* L. var. *botrytis* L.), followed by a long fallow period.

The forest system was 0.16 ha with an average slope of 21 % and medium texture (250, 340, 410 g kg^{-1} clay, silt and sand, respectively, at 0.0–0.2 m). Bracatinga (*Mimosa scabrella* Benth.), manduirana (*Senna macranthera* (DC. ex Collad.) H.S. Irwin & Barneby) and guavirova (*Campomanesia xanthocarpa* Mart.) (all native plants) were the predominant species. Eucalyptus (*Eucalyptus* spp.), an exotic plant, were also present.

Based on interviews with farmers, it was estimated that application of poultry litter was 48 Mg ha^{-1} year^{-1}. At 75 % dry matter and 2.2 %, 2.4 % and 1.7 % of N, P_2O_5 and K_2O, respectively, the application rate translated to 1,056, 1,152 and 816 kg ha^{-1} year^{-1} of N, P_2O_5 and K_2O by organic fertilization. In the conventional system, 4 Mg ha^{-1} year^{-1} of mineral fertilizer (10-10-10), resulting on addition of 400 kg ha^{-1} year^{-1} of N, P_2O_5 and K_2O was applied in addition to the organic fertilization.

The soil in all three systems was classified as Cambisol and the chemical and physical attributes are presented in Tables 1 and 2.

2.2 Analysis of soil and soil losses

In the lower part of each plot (small catchment) a drainage channel was constructed in order to allow water flow. The channel had its base and sides compacted to avoid erosion. The Coshocton wheel (Lal, 1994) was connected through a PVC pipe to a 65 L plastic bucket (with a lid), collecting 1 % of the flow (so 1 % of water including suspended solids was retained in the bucket). Runoff was sampled biweekly for a year (September 2007 to August 2008). The amount of runoff was measured using graduated devices (bucket and beaker). After homogenization, runoff samples (500 mL bottles) were taken to analyse sediment and nutrients. A portion of these samples were frozen for chemical analysis with a part being reserved to estimate sediment loss.

Table 1: *Soil chemical attributes (0.0–0.2 m) in the vegetable plots and forest system from Campestre Catchment, Colombo, Brazil.*

Systems	Al	H+Al	Ca	Mg	K	CEC	P	C_{org}
			($cmol_c\ kg^{-1}$)				($mg\ kg^{-1}$)	($g\ kg^{-1}$)
Conventional	1.2	8.7	4.7	1.3	0.9	15.5	151.7	33.9
Organic	0.0	5.1	8.5	2.4	1.5	17.7	40.2	35.2
Forest	4.2	12.5	2.8	1.5	0.2	17.1	3.9	29.9

CEC = cation exchange capacity; P = Mehlich extractable; C_{org} = organic C; K = exchangeable K

Table 2: *Some soil physical attributes (0.0–0.2 m) in the vegetable plots and forest system from Campestre Catchment, Colombo, Brazil.*

Systems	Particle size ($g\ kg^{-1}$)			MWD	ρ_b	Porosity (%)			K_s
	Clay	Silt	Sand	(mm)	($Mg\ m^{-3}$)	Macro	Micro	Total	($mm\ h^{-1}$)
Conventional	280	370	350	3.0	0.9	22	40	62	175
Organic	500	440	60	3.6	0.9	20	45	62	186
Forest	250	340	410	3.6	0.9	30	33	63	215

MWD = mean weight diameter of water-stable aggregates; ρ_b = soil bulk density; K_s = saturated hydraulic conductivity

The concentration of solids obtained from a 50 mL aliquot dried at 105 °C was multiplied by the runoff measured in the bucket and then multiplied by 100 to determine soil loss. This value was added to the amount of sediment trapped (coarser material) in the Coshocton wheel.

Next to the plots, two rain gauges were installed to record the distribution and amount of daily rainfall during the experiment. The annual amount of rainfall during the experimental period (September 2007 to August 2008) was 1,227 mm, with highest rainfall amounts in November (214 mm), December (305 mm) and February (228 mm).

2.3 Analysis of phosphorus and nitrogen

Dissolved reactive P (DRP), nitrogen as nitrate (NO_3 -N) and nitrogen as ammonium (NH_4-N) were analysed on thawed samples passed through a 0.45 μm membrane filter. The DRP was determined by spectrophotometry at 880 nm following the ascorbic acid methodology (APHA, 1995). Ammonium-N was determined by the phenol method using spectrophotometry at 640 nm (APHA, 1995). Nitrate-N was determined by spectrophotometry at 220 nm using metallic zinc to determine the interference (Norman & Stucki, 1981).

Total P (TP) and total N (TN) in the non-filtered sample were determined using the Kjeldahl digestion (APHA, 1995). Total P was determined by spectrophotometry at 880 nm following the ascorbic acid methodology (APHA, 1995). Total N Kjeldahl was determined by spectrophotometry at 640 nm following the same methodology used to determine ammonium-N (APHA, 1995). Total N Kjeldahl refers to the soluble ammonium as well as nitrogen associated with the sediment, either as organic or mineral. So, the particulate N which represents the nitrogen associated with the sediment, either in organic or mineral, was determined by the difference between total N Kjeldahl and ammonium-N. The nitrate was not recovered in the total N Kjeldahl methodology, thus the total N (TN) was obtained by the sum of the Kjeldahl N and NO_3-N (Sharpley & Menzel, 1987).

The particulate P (PP) was obtained by subtracting DRP from total P concentration. Bioavailable P (BP) was determined by the membrane filter impregnated with iron oxide according Myers & Pierzynski (2000) and Sharpley (1993). The particulate bioavailable P (PBP) was obtained by the difference from BP and DRP. The particulate non-bioavailable P (PNBP) was obtained by subtracting the PBP from the PP.

Nutrient losses (g ha^{-1}) were estimated as the product of the concentrations of nutrients and runoff volume.

2.4 Statistical analysis

Losses of soil, water and nutrients were compared using the Wilcoxon signed rank test, utilizing the R version 2.11.0 (R Development Core Team, 2009). The Wilcoxon test was used rather than the paired t-test because the daily loss distributions contained influential outliers. A critical p-value of 0.017 was used to account for the multiple testing issue of carrying out three pairwise comparisons. This maintains a Type I error of 0.05 within each soil, water and nutrient.

3 Results

3.1 Soil and water losses

Annual soil losses in conventional and organic systems were respectively 218 and 12 times greater than in forest system corroborated by the statistically lowest soil losses in the forest system per the Wilcoxon test. Statically however, there was no difference between the two disturbed soil systems (Table 3). The months of December (305 mm of rainfall) and February (228 mm of rainfall) had the greatest soil loss (Figure 1) corresponding with highest water loss (Figure 2).

Water loss in all systems were very low and almost zero in the forest system (Table 3). Less than 1 % of rainfall was lost through runoff (from a total of 1227 mm during the experimental period). The Wilcoxon test does not show significant differences between organic and conventional systems. Statistically significant differences in water loss were, however, observed between agricultural (conventional and organic) and forest system (Table 3).

Table 3: *Effect of management system on annual losses of soil and water in small scale vegetable farming and forest system during Sep. 2007 to Aug. 2008 in Campestre Catchment, Colombo, Brazil.*

Systems	*Soil losses* ($kg\ ha^{-1}\ year^{-1}$)	*Water losses* ($mm\ year^{-1}$)
Conventional	113.58	1.98
Organic	6.03	1.21
Forest	0.52 *	0.32 *

* Significantly different from conventional and organic system according to Wilcox test, p<0.017.

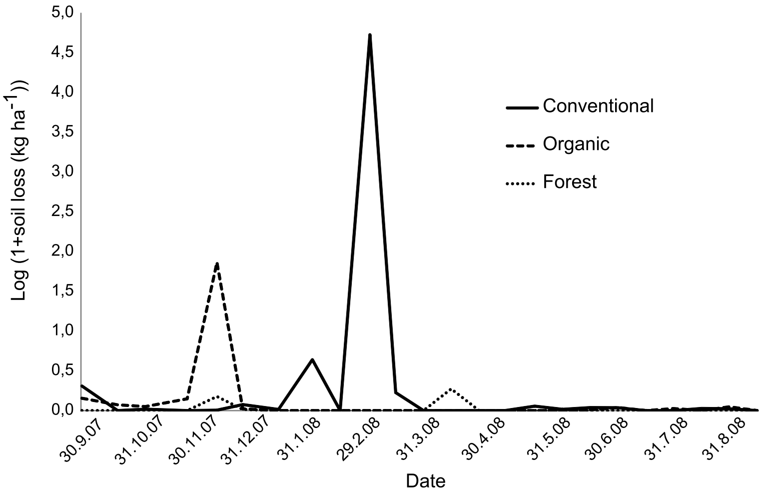

Fig. 1: *Soil loss in vegetable plots and forest system during Sep. 2007 to Aug. 2008 in Campestre Catchment, Colombo, Brazil.*

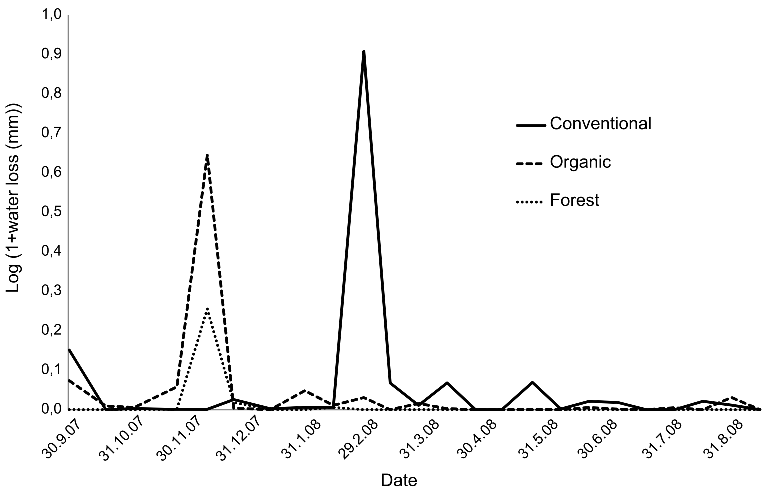

Fig. 2: *Water loss in vegetable plots and forest system during Sep. 2007 to Aug. 2008 in Campestre Catchment, Colombo, Brazil.*

3.2 Concentration and loss of phosphorus

The annual weighted average concentration (Table 4) and the annual loss (Table 5) of total P, particulate P, dissolved reactive P and bioavailable P in runoff were greatest in conventional followed by organic and least in the forest system. The Wilcoxon test showed statistically significant differences in P loss between conventional and forest, and between organic and forest, but not between conventional and organic (Table 5) as was also observed in the annual soil and water loss.

The maximum amount of total P lost was 0.188 kg ha^{-1} year^{-1} in the conventional system (Table 5) with almost 50 % as particulate P (Figure 3).

3.3 Concentration and nitrogen loss

Unlike phosphorus, the highest concentration (Table 4) and loss (Table 5) of soluble nitrogen (NO_3-N and NH_4-N) was found in the organic system. Nitrate-N, which is the most damaging to human health, had an average concentration (Table 4) in the organic system 8 times greater than in the conventional system. The loss of NO_3-N (Table 5) in the organic system was 5 times more than in the conventional system. Also, NO_3 -N loss is approximately 59 % of the total N in the organic system while in the conventional system approximately only 14 % was lost (Figure 4). In our study, 69 % of the total N was lost as particulate N (Figure 4) agreeing with Sharpley & Menzel (1987) who observed that 64 % of the total N was lost in a catchment with conventional tillage as particulate N.

Table 4: *Effect of management system on weighted mean concentration of nutrients in small scale vegetable farming and forest system during Sep. 2007 to Aug. 2008 in Campestre Catchment, Colombo, Brazil.*

Systems	*BP*	*PBP*	*DRP*	*PP*	*TP*	*KjN* ($mg\ L^{-1}$)	*TN*	*PN*	NH_4-N	NO_3-N
Conventional	4.63	0.45	4.18	5.30	9.48	29.73	34.37	23.72	6.01	4.66
Organic	0.82	0.45	0.37	0.54	0.92	27.52	66.06	17.25	10.25	38.62
Forest	0.18	0.0	0.14	0.16	0.30	26.18	26.56	15.30	10.88	0.38

BP = bioavailable P; PBP = particulate bioavailable P; DRP = dissolved reactive P; PP = particulate P; TP = total P; KjN = Kjeldhal N; TN = total N; PN = particulate N; NH_4-N = Ammonium N; NO_3-N = Nitrate N

Table 5: *Effect of management system on losses of nutrients in small scale vegetable farming and forest system during Sep. 2007 to Aug. 2008 in Campestre Catchment, Colombo, Brazil.*

Systems	*BP*	*PBP*	*DRP*	*PP*	*TP* ($g\ ha^{-1}\ year^{-1}$)	*KjN*	*TN*	*PN*	NH_4-N	NO_3-N
Conventional	92	9	83	106	188	590	682	471	120	93
Organic	10	5	5	6	11	333	800	209	124	468
Forest	0.57 *	0.24 *	0.33 *	0.52 *	0.85 *	84 *	85 *	49 *	35 *	1 *

BP = bioavailable P; PBP = particulate bioavailable P; DRP = dissolved reactive P; PP = particulate P; TP = total P; KjN = Kjeldhal N; TN = total N; PN = particulate N; NH4-N = Ammonium N; NO_3-N = Nitrate N
* Significantly different from conventional and organic systems according to Wilcox test, p<0.017.

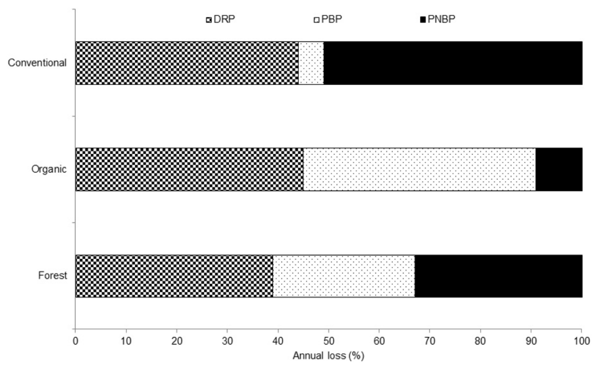

Fig. 3: *Annual loss (%) of dissolved reactive P (DRP), particulate bioavailable P (PBP) and particulate non-bioavailable P (PNBP) in runoff of vegetable plots and forest system during Sep. 2007 to Aug. 2008 in Campestre Catchment, Colombo, Brazil.*

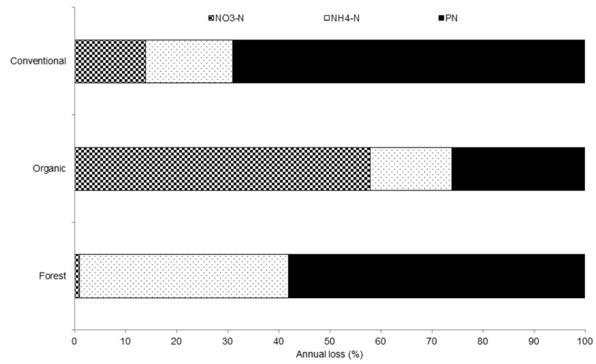

Fig. 4: *Annual loss (%) of nitrate N* (NO_3*-N), ammonium N* (NH_4*-N)) and particulate N (PN) in runoff of vegetable plots and forest system during Sep. 2007 to Aug. 2008 in Campestre Catchment, Colombo, Brazil.*

4 Discussion

Soil losses (0.5 to 114 kg ha^{-1} year^{-1}) are considered low when compared with soil loss tolerance (T value) for Cambissols obtained by Bertol & Almeida (2000) (approximately 7,500 kg ha^{-1} year^{-1}). Soil loss tolerance refers to the maximum rate of annual soil loss that can occur and still permit crop productivity to be sustained economically. The T value is very questionable by soil scientists but it has been considered as a reference value on some private and governmental conservation programs to control soil erosion. However, it is important to point out that the soil and water losses, even if not of agronomical importance, environmentally can cause damage by runoff associated pollutants (Sharpley *et al.*, 2001) if these pollutants reach the hydric system.

Soil loss is strongly affected by water loss, and in our study the water losses (0.1 to 2.0 mm year^{-1}) were very low (less than 1 %) compared to the total precipitation of the studied period (1,227 mm year^{-1}). The lower water losses were possible due to the low volume and intensity of precipitation during the experiment (in that region the average annual rainfall is between 1,400 and 1,600 mm) but also to the high infiltration. The saturated hydraulic conductivity was 175, 186 and 215 mm h^{-1} for conventional, organic and forest system, respectively. Final infiltration rates above 150 mm h^{-1} are classified as high in the soil quality test kit guide by USDA (1999).

The annual soil loss was low but it was not well distributed during the year around (Figure 2). In the conventional system, the soil loss in February was 111 kg ha^{-1} (98 % of the total lost annually) and in the organic system the soil loss in December was 5 kg ha^{-1} (83 % of total lost annually). Despite being in different months, both occurred in summer (season of higher volume and intensity of rainfall), but the high soil loss in these two months was not only a result of the high rainfall. It was also a result of the vegetative cover; otherwise both systems should have high soil and water losses in the same months. In February (228 mm of rainfall), the soil was bared in the conventional system while in the organic system the soil was covered by several crops (lettuce, swiss chard and broccoli) which resulted in high soil loss in the conventional system. On the other hand, in December (305 mm rainfall), half of the area in the organic system was with bared soil and in the conventional system the soil was covered by weeds resulting in higher soil loss in the organic system.

On overall, conservation agriculture systems with high plant diversity, minimizes the time that the soil is bare and exposed to runoff, and consequently has been more efficient on controlling soil erosion than conventional agriculture (Gomiero *et al.*, 2011; Fang *et al.*, 2012; Arnhold *et al.*, 2014; Palm *et al.*, 2014). In our study, the soil loss in the conventional system was 19 times greater than in the organic system, even not statistically different. The least soil loss in the organic system (even with higher slope) was possible due to the higher percentage of soil covered by crops all year around. Also, the aggregate stability in the organic system was better than in the conventional system (mean weight diameter of water-stable aggregates of 3.6 and 3.0 mm, respectively).

On overall, higher the sand content, higher the water infiltration into the soil (USDA, 1999). In our study, even with higher clay content in the organic system, the saturated hydraulic conductivity was greater resulting on less water loss by surface runoff. However, the organic system was not so efficient on decreasing water loss as soil loss. The water loss in the conventional system was only 1.6 greater than the organic system (Table 3). This result demonstrated that vegetative cover and soil structure had a greater influence in soil loss than in water loss.

The highest concentration of runoff P from agricultural systems (9.48 and 0.92 mg L^{-1} of total P for conventional and organic farming, respectively) can cause eutrophication (Correll, 1998; Daniel *et al.*, 1998; Smith *et al.*, 1999). The highest P concentration in the conventional system can be explained by the high P input (organic and mineral fertilizations). It was estimated, based on the interview with the farmers, application of 1,152 and 400 kg ha^{-1} year^{-1} of P_2O_5 by organic and mineral fertilization, respectively. The organic farming applied 1,152 kg ha^{-1} year^{-1} P_2O_5 equivalent of organic fertilizer. The forest (unfertilized system), as expected, had the lowest P concentration (0.30 mg L^{-1} of total P), though, this low P concentration can cause environmental problems (total P of 0.1 mg L^{-1} is related with eutrophication) (Correll, 1998; Daniel *et al.*, 1998). Sharpley *et al.* (1994) associated the transport of P in nonfertilized forest with the natural fertility of the soil and the amount of nutrients retained in the litter, which enriches the water runoff in natural systems.

The higher concentrations of runoff P in the conventional system compared to the organic system can be explained by the higher soil loss as well as the higher soil P content (Pote *et al.*, 1999). Fine sediments like clay and organic matter enriched with P have low densities and can be carried long distances (Smith *et al.*, 1999; McDowell *et al.*, 2001), however, even with higher clay and higher soil organic carbon, the organic system was efficient on reducing runoff P concentrations by reducing soil loss. A direct correlation between soil loss and runoff P concentration is expected, especially for particulate P which represents the P associated to the sediment (Sharpley *et al.*, 2001).

The bioavailable P is the dissolved reactive P in addition to the particulate bioavailable P which is readily available to growing aquatic plants, and represents a short term problem for water quality (Sharpley *et al.*, 1994; Reynolds & Davies, 2001). In the organic system we found lower total P losses, but 91 % of total P lost was as bioavailable P (45 % DRP and 46 % PBP), whereas in the conventional system only 49 % was lost as bioavailable P (44 % DRP and 5 % PBP) (Figure 3). This suggests that the organic system could be more harmful over short time periods. The higher proportion of bioavailable P in the organic system could be due to the higher contribution from plant material as well as from preferential transport of clay and organic particles in runoff (higher clay content and soil organic carbon in the organic system). Sharpley *et al.* (1992) also found lower proportion of bioavailable P in conventional tillage compared to no-till system.

Considering that the application of mineral P to grow vegetables is generally higher than 100 kg ha^{-1} year^{-1} (in this study 175 kg ha^{-1} year^{-1} of mineral P and 500 kg ha^{-1} year^{-1} of organic P) the P losses (in this study lower than 188 g ha^{-1} year^{-1}) represented less than 1 % of the applied P. Studies in general show losses lower than 5 % of the total applied (Sharpley *et al.*, 2001) but despite not representing a huge cost for crop production, this loss can represent a huge challenge to aquatic systems due to P pollution.

Unlike phosphorus, the highest concentrations of soluble nitrogen were found in the organic system 38.62 mg L^{-1} of NO_3-N and 10.25 mg L^{-1} of NH_4-N. The higher concentration of soluble nitrogen, even with lower soil and water losses could be a result of higher clay content; however, considering that the runoff phosphorus concentration was not affected by clay content, the higher soluble nitrogen was possible affected by the better soil biological diversity and activity which increases mineralization and consequently nitrate and ammonium concentration in soil (Eghball *et al.*, 2002; Gomiero *et al.*, 2011). The higher amount of soluble nitrogen (NO_3-N and NH_4-N) lost by runoff in the organic system, even with lower water losses reflects the influence of the nutrient concentration. Water loss by runoff was very low (less than 1 % of the total precipitated), consequently most of water was lost by subsurface flow, and as well known, leaching is the main process of NO_3-N transport from soil to water (Smith *et al.*, 1990). So, considering the greater concentration of NO_3-N present in runoff, reflecting the high soil nitrogen concentration, subsurface water should also be monitored in order to detect possible nitrate contamination problems.

5 Conclusions

Soil, water and nutrient losses were lower in the forest system compared to conventional and organic systems indicating the importance of forest in fragile environment. Considering the vegetables production, the runoff and thus the potential for water contamination by associated pollutants was affected by soil vegetative cover. So, best management practices including cover crops should be adopted, particularly during periods of intense rainfall. The organic system better controls runoff and sediment yield over the conventional system; however a more efficient nutrient management should also be applied to avoid water contamination. Soluble nitrogen concentrations and the percentage of bioavailable phosphorus were higher in the organic system, indicating greater potential for pollution in the short term.

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References

Almeida, L. (2003). *Mudanças técnicas na agricultura: perspectivas da transição agroambiental em Colombo – PR.* Ph.D. thesis Universidade Federal do Parana, Curitiba.

APHA (1995). *Standard Methods for the Examination of Water and Wastewater, 19th Ed.*. American Public Health Association, Washington, USA.

Arnhold, S., Lindner, S., Lee, B., Martin, E., Kettering, J., Nguyen, T. T., Koellner, T., Ok, T. S. & Huwe, B. (2014). Conventional and organic farming: Soil erosion and conservation potential for row crop cultivation. *Geoderma*, 219-220, 89–105.

Bertol, I. & Almeida, J. A. (2000). Tolerância de perda de solo por erosão para os principais solos do estado de Santa Catarina. *Revista Brasileira de Ciência do Solo*, 24, 657–668.

Bertol, I., Mello, E. L., Guadagnin, J. C., Zaparolli, A. L. & Carrafa, M. R. (2003). Nutrients losses by water erosion. *Scientia Agricola*, 3, 581–586.

Cogo, N. P., Levien, R. & Schwarz, R. A. (2003). Perda de solo e água por erosão hídrica influenciadas por métodos de preparo, classes de declive e níveis de fertilidade do solo. *Revista Brasileira de Ciência do Solo*, 27, 746–753.

Correll, D. L. (1998). The role of phosphorus in the eutrophication of receiving waters: A review. *Journal of Environmental Quality*, 27, 261–266.

Daniel, T. C., Sharpley, A. N. & Lemunyon, J. L. (1998). Agricultural phosphorus and Eutrophication: A Symposium Overview. *Journal of Environmental Quality*, 27, 251–257.

Derpsch, R., Roth, C. H., Sidiras, N. & Köpke, U. (1991). *Controle da erosão no Paraná, Brasil: Sistemas de cobertura do solo, plantio direto e preparo conservacionista*. IAPAR, Londrina.

Eghball, B., Wienhold, B. J., Gilley, J. E. & Eigengerg, R. A. (2002). Mineralization of manure nutrients. *Journal of Soil and Water Conservation*, 57, 470–473.

Fang, N. F., Shi, Z. H., Li, L., Guo, Z. L., Liu, Q. J. & Ai, L. (2012). The effects of rainfall regimes and land use changes on runoff and soil loss in a small mountainous watershed. *Catena*, 99, 1–8.

Gomiero, T., Pimentel, D. & Paoletti, M. G. (2011). Environmental impact of different agricultural management practices: conventional vs. organic agriculture. *Critical Reviews in Plant Sciences*, 30, 95–124.

Hamerschmidt, I., Silva, J. C. B. V. & Lizarelli, P. H. (2000). Agricultura orgânica. Boletim Serie Produtor 65, EMATER-PR, Curitiba.

Hart, M. R., Quin, B. F. & Long Nguyen, M. (2004). Phosphorus runoff from agricultural land and direct fertilizer effects: A review. *Journal of Environmental Quality*, 33, 1954–1972.

Haynes, J. R. & Naidu, R. (1998). Influence of lime, fertilizer and manure applications on soil organic matter content and soil physical conditions: a review. *Nutrient Cycling in Agroecosystems*, 5, 123–137.

Hooda, P. S., Edwards, A. C., Anderson, H. A. & Miller, A. (2000). A review of water quality concerns in livestock farming areas. *Science of the Total Environment*, 250, 143–147.

Kay, P., Edwards, A. C. & Foulger, M. (2009). A review of the efficacy of contemporary agricultural stewardship measures for ameliorating water pollution problems of key concern to the UK water industry. *Agricultural Systems*, 99, 67–75.

Lal, R. (1994). *Soil erosion research methods. 2ed.*. Water Conservation Society, Ankeny.

McDowell, R. W., Sharpley, A. N., Condron, L. M., Haygarth, P. M. & Brookes, P. C. (2001). Processes controlling soil phosphorus release to runoff and implications for agricultural management. *Nutrient Cycling in Agroecosystems*, 59, 269–284.

Mellek, J. E., Dieckow, J., Silva, V. L., Favaretto, N., Pauletti, V., Vezzani, F. M. & Souza, J. L. (2010). Dairy liquid manure and no-tillage: physical and hydraulic properties and carbon stocks in a Cambisol of Southern Brazil. *Soil & Tillage Research*, 110, 69–76.

Myers, R. G. & Pierzynski, G. M. (2000). Using the iron method to estimate bioavailable phosphorus in runoff. In G. M. Pierzynski (Ed.), *Methods of phosphorus analysis for soils, sediments, residuals, and waters*. Southern Cooperative Series Bulletin 396. Accessed 01/02/2014.

Ndayegamiye, A. & Cote, D. (1989). Effect of long-term pig slurry and solid cattle manure application on soil chemical and biological properties. *Canadian Journal of Soil Science*, 69, 39–47.

Norman, R. J. & Stucki, J. W. (1981). The determination of nitrate and nitrite in soils extracts by ultraviolet spectrophotometry. *Soil Science Society of America Journal*, 45, 347–353.

Palm, C., Blanco-Canqui, H., DeClerck, F., Gatere, L. & Grace, P. (2014). Conservation agriculture and ecosystem services: An overview. *Agriculture, Ecosystems and Environment*, 187, 87–105.

Popia, A. F., Cidade Junior, H. A. & Almeida, R. (2000). Olericultura orgânica. Boletim Serie Produtor 43, EMATER-PR, Curitiba.

Pote, D. H., Daniel, T. C., Nichols, D. J., Sharpley, A. N., Moore, P. A., Miller, D. M. & Edwards, D. R. (1999). Relationship between phosphorus levels in three ultisols and phosphorus concentrations in runoff. *Journal of Environmental Quality*, 28, 170–175.

R Development Core Team (2009). R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

Reynolds, C. S. & Davies, P. S. (2001). Sources and bioavailability of phosphorus fractions in freshwaters: a British perspective. *Biological Reviews*, 76, 27–64.

Ribeiro, K. H., Favaretto, N., Dieckow, J., de Paula Souza, L. C., Minella, J. P. G., de Almeida, L. & Ramos, M. R. (2014). Quality of surface water related to land use: a case study in a catchment with small farms and intensive vegetables production in southern Brazil. *Revista Brasileira de Ciência do Solo*, 38 (2), 656–668.

Sharpley, A. N. (1993). An innovative approach to estimate bioavailable phosphorus in agricultural runoff using iron oxide-impregnated paper. *Journal of Environmental Quality*, 22, 597–601.

Sharpley, A. N., Chapra, S. C., Wedepohl, R., Sims, J. T., Daniel, T. C. & Reddy, K. R. (1994). Managing agricultural phosphorus for protection of surface waters: Issues and options. *Journal of Environmental Quality*, 23, 437–451.

Sharpley, A. N. & Halvorson, A. D. (1994). The management of soil phosphorus availability and its impact on surface water quality. In R. Lal, & B. A. Stewart (Eds.), *Soil Processes and Water quality*. Lewis Publishers, Boca Raton.

Sharpley, A. N., McDowell, R. W. & Kleinman, J. A. (2001). Phosphorus loss from land to water: Integrating agricultural and environmental management. *Plant & Soil*, 237, 287–307.

Sharpley, A. N. & Menzel, R. G. (1987). The impact of soil and fertilizer phosphorus on the environment. *Advances in Agronomy*, 41, 297–324.

Sharpley, A. N., Smith, S. J., Jones, O. R., Jones, O. R., Berg, W. A. & Coleman, G. A. (1992). The transport of bioavailable phosphorus in agricultural runoff. *Journal of Environmental Quality*, 21, 30–35.

Shigaki, F., Sharpley, A. N. & Prochnow, L. I. (2006). Animal-based agriculture, phosphorus management and water quality in Brazil: Options for the future. *Scientia Agricola*, 63, 194–209.

Smith, S. J., Schepers, J. S. & Porter, L. K. (1990). Assessing and managing agricultural nitrogen losses to the environment. In B. A. Stewart (Ed.), *Advances in Soil Science* (pp. 1–43). Lewis Publishers, Chelsea, USA.

Smith, V. H., Tilman, G. D. & Nekola, J. C. (1999). Eutrofication: impacts of excess nutrient inputs on freschwater, marine, and terrestrial ecosystems. *Environmental Pollution*, 100, 179–196.

USDA (1999). Soil quality test kit guide. Washington, USDA-ARS-Soil Quality Institute. 82p.

Willer, H. & Lernoud, J. (2013). *Organic Agriculture Worldwide: Key results from the FiBL-IFOAM survey on organic agriculture worldwide 2013*. Research Institute of Organic Agriculture (FIBL), Frick, Switzerland.