Agriculture, Ecosystems and Environment 134 (2009) 19–23

Contents lists available at ScienceDirect

Agriculture, Ecosystems and Environment

journal homepage: www.elsevier.com/locate/agee

Short communication

Atmospheric deposition contributes little nutrient and sediment to stream flow from an agricultural watershed

R.W. McDowell [a,*], A.N. Sharpley [b]

[a] AgResearch, Invermay Agricultural Centre, Private Bag 50034, Mosgiel, New Zealand
[b] Department of Crop, Soil and Environmental Sciences, Division of Agriculture, University of Arkansas, Fayetteville, AR, USA

ARTICLE INFO

Article history:
Received 11 March 2009
Received in revised form 19 June 2009
Accepted 30 June 2009
Available online 30 July 2009

Keywords:
Phosphorus
Nitrogen
Wind erosion
Sediment
Water quality

ABSTRACT

Atmospheric deposition of nutrients within agricultural watersheds has received scant attention and is poorly understood compared to nutrient transport in surface and subsurface water flow pathways. Thus, we determined the deposition of phosphorus (P), nitrogen (N), and sediment in a mixed land use watershed in south-central Pennsylvania (39.5 ha; 50% corn–wheat–soybean rotation, 20% pasture, and 30% woodland), in comparison with stream loads at several locations along its reach between 2004 and 2006. There was a significant difference in deposition rates among land uses ($P < 0.05$) with more P and N deposited on cropland (1.93 kg P and 10.71 kg N ha^{-1} yr^{-1}) than pasture (1.10 kg P and 8.06 kg N ha^{-1} yr^{-1}) and woodland (0.36 and 2.33 kg N ha^{-1} yr^{-1}). Although not significant, sediment showed the same trends among land uses. A significant relationship was found between P in deposition and P in soil <10-m away from the samplers suggesting much of the deposited sample was derived from local soil. Samplers adjacent to the stream channel showed deposition rates (1.64 kg P and 8.83 kg N ha^{-1} yr^{-1}) similar to those on cropland. However, accounting for the surface area of the stream, direct deposition of P, N, and sediment probably accounted for <3% of P and <1% of N and sediment load in stream flow from the watershed (1.41 kg P, 27.09 kg N, and 1343 kg sediment ha^{-1} yr^{-1} at the outlet). This suggests that strategies to mitigate nutrient and sediment loss in this mixed-land use watershed should focus on runoff pathways.

1. Introduction

Among sources contributing to nutrient loss to streams, atmospheric nutrient deposition is probably the least understood. Processes such as wind erosion, burning and tillage can cause particles containing nitrogen (N) and phosphorus (P) to become airborne. While this can alter soil and crop productivity (Larney et al., 1998), later deposition of these particles has been recognised as an important source of nutrients in receiving lakes and marine water bodies (Kocak et al., 2004; Shaw et al., 1989).

Most studies that have examined atmospheric deposition focus on N not P (Asman et al., 1998; Howarth, 2006; NADP, 2004). However, in most streams and rivers, eutrophication is limited by P inputs. Of those studies that have examined P, most examine deposition on a large scale. For example, Tabatabai (1983) estimated that wet deposition (rainfall and snow) contributed from 0.01 to 0.1 kg P ha^{-1} yr^{-1} to land in the US. Others have also established that dry deposition (e.g., wind borne particles) contributes an equal, if not greater, amount of P than wet deposition (Ann and James, 2001; Shaw et al., 1989). In the extremely arid environment of eastern Inner Mongolia, Li et al. (2004) measured dry deposition rates of 0.01–0.7 kg P ha^{-1} d^{-1}. Obviously the rate of deposition will be highly variable dependant on a number of factors like vegetation, topography, climate and location. For lakes, Cole et al. (1990) noted loads varying from 0.08 to 1.93 kg ha^{-1}, generally decreasing, along with those particles eroded by wind, with distance from the shore. These loads are comparable to annual loads measured in some streams (Li et al., 2004; Ryden et al., 1973). However, while a much greater ratio of land to surface water exists in stream and rivers than most lakes, little work has identified if deposition, most likely derived from the surrounding land use, is an important source of nutrients and sediment for streams and rivers.

One of the major variables affecting the deposition of N and P into streams and rivers may be local land use. Past studies have indicated that deposition is related to the degree of agricultural activity (Asman, 2002; Pearson and Stewart, 1993). In the Northeast U.S., cropland, pasture land, and woodland are common within the same watershed (Pionke et al., 2000). Each land use has a different potential for P losses to streams via runoff processes due to different management such as P inputs in fertiliser and manure,

* Corresponding author. Tel.: +64 3 489 9262; fax: +64 3 489 3739.
E-mail address: richard.mcdowell@agresearch.co.nz (R.W. McDowell).

0167-8809/$ – see front matter
doi:10.1016/j.agee.2009.06.016

and tillage practices. Similar to loss by runoff, particles eroded by wind contain much P (Larney et al., 1998), meaning that stream P concentrations could reflect the local land use. The objective of this paper was to determine the relative potential for deposition of P, and secondly N and sediment, among different land uses and into areas of a stream draining a mixed agriculture – forestry watershed in the Northeast U.S.

2. Materials and methods

2.1. Sites and sampling

The study was conducted in a 39.5 ha subwatershed of Mahantango Creek named FD36, a tributary to the Susquehanna River and ultimately the Chesapeake Bay (Fig. 1). FD36 is typical of upland agricultural watersheds within the nonglaciated, folded and faulted, Appalachian Valley and Ridge Physiographic Province. Soils are Dystric Cambisols (Berks and Calvin series), Stagnic Luvisols (Hustontown series) and Haplic Acrisols (Leck Kill and Hartleton series) with slopes ranging from 1 to 20%. Climate is temperate and humid with an average rainfall of 1100 mm yr^{-1} and streamflow of 450 mm yr^{-1}.

Land use is predominantly cropped to a soybean (*Glycine max* [L.] Merr.)–wheat (*Triticum aestivum* L.)–corn [*Zea mays* L.] rotation (50%), with a smaller area in pasture for hay (20%), and the remainder wooded (30%) (Sharpley et al., 2008). Crop tillage between 2004 and 2006 was chisel plowed and disked, with approximately 20% residue cover obtained from visual observation of twenty 1 m^2 quadrants randomly located around the cropped area. In the last 10 years, cropped land north of the stream channel (Fig. 1) received a surface application of about 60 m^3 ha^{-1} yr^{-1} pig slurry in spring (about 100 kg P ha^{-1} yr^{-1}, assuming a slurry P concentration of 1.6 g L^{-1}; Sharpley and Moyer, 2000). South of the stream channel, approximately 5 Mg ha^{-1} yr^{-1} poultry manure was added each spring (about 85 kg P ha^{-1} y^{-1}, assuming a manure P content of 16.9 g kg^{-1}; Sharpley and Moyer, 2000).

The watershed is divided into four segments (named flume 1, at the outlet, to 4) based on topography and drainage patterns derived from a detailed topographic survey and visual reconnaissance (Fig. 1). Since May 1996, streamflow below each segment has been continuously monitored using recording H-flumes, and water samples taken automatically during storms at 5- to 120-min intervals using programmable stage-activated samplers. Baseflow samples are taken at each flume at weekly intervals.

The number of sites for the collection of deposition reflected the proportional area of land use in the watershed: five sites were placed in representative positions (i.e. at least 30 m away from the edge of the landuse) within cropland and three each within pasture and woodland (Fig. 1). It is recognised that this placement strategy would have avoided the collection of debris near the edge of the woodland, but thought necessary for the comparison of landuse. However, the mean wind speed measured near the watershed outlet was low (6 km/h with a maximum gust of 39 km/h for 5 min from 2004 to 2006), and predominantly from the west, blowing woodland debris away from the stream (T. Troutman, *pers. comm.*). Samplers were also located on the stream banks near each of the four flumes just above the maximum recorded water height for 1997 to present. These were placed here to estimate deposition into the stream. Each site had three replicate deposition samplers installed in January 2004. Deposition samplers were modified from Li et al. (2004) and designed to collect bulk samples of both wet and dry deposition to the ground or stream. This included potential returns from vegetation, which Tskuda et al. (2006) noted will contribute input via deposition to waterways and hence should be included in estimates. The samplers consisted of 43-cm long, by 29-cm wide, and 25-cm deep plastic containers, with a 5-cm wide lip at the top to help prevent raindrop splash of soil into the collectors. On the edge of each container was positioned a 1.25-cm steel mesh with bristles from plastic rope glued in place every fourth hole (88 in total) to prevent birds from perching on them. The samplers were dug into the ground or bank to 5 cm depth, leaving 20 cm of sidewall plus rope to help prevent potential

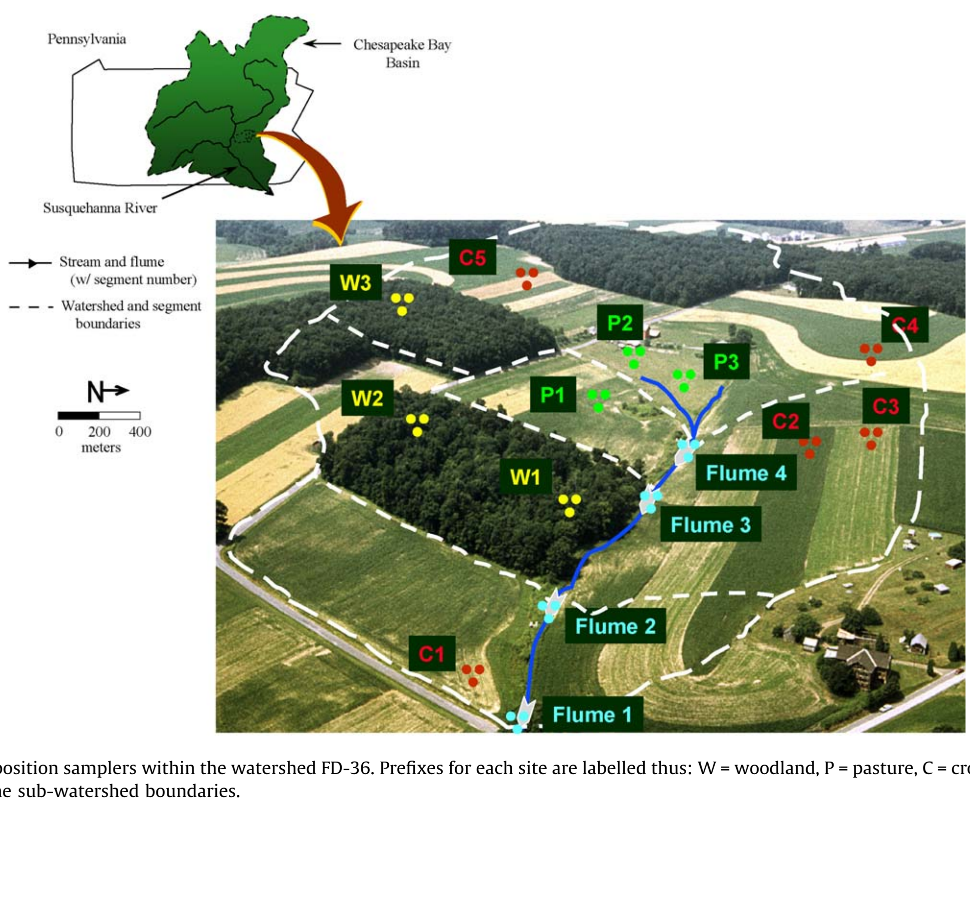

Fig. 1. Location of the deposition samplers within the watershed FD-36. Prefixes for each site are labelled thus: W = woodland, P = pasture, C = cropland, and F = flume. The dashed line represents the sub-watershed boundaries.

transfer of soil via raindrop splash while still collecting as much deposition to near ground level as possible. Samplers were filled with 5 L of deionised water and checked weekly to either top-up water or to remove any insects and pieces of vegetation >3 cm. Although these may be blown into the stream channel there inclusion would cause unduly large variation when comparing land uses. At fortnightly intervals, water was added to increase the sample volume to 10 L and the sample removed for later analysis of N, P and sediment. The sampler was then washed and refilled with 5 L of deionised water. Due to the number of samplers required and the cost involved in employing 35 AeroChem Metrics deposition collectors, preliminary work in New Zealand had calibrated our modified design against that of an adjacent AeroChem Metrics Model 301 deposition collector for 1 year. Data indicated that the total load of sediment and P collected every 3 weeks by the modified collector was within 20% of the total P and sediment deposited within the Aerochem Metrics collector. The use of deionised water in our samples was used in response to the observation that during high winds, material was blown out of the samplers. However, this required the adjustment of nutrient and sediment concentrations according to the volume collected and the precipitation measured near the watershed outlet. The reader should therefore be cautioned that concentrations may have been subject to variation due to different volumes of precipitation in the collectors and measured the watershed outlet rain gauge. Loads were expressed on a kg ha^{-1} basis according to the mass of nutrient or sediment deposited and the area of the collector.

In April 2005, pairs of soil cores (2.5 cm diameter, 0–5 cm depth) were collected on four transects (north, south, east, west) every 2 m, up to 10 m away from each deposition sampler (except near the flumes). The samples were bulked, air-dried, and sieved (2 mm) for later analysis.

2.2. Analyses

An unfiltered stream water sample was digested using low-N potassium persulphate and total P (TP) and total N (TN) determined via standard auto-analyzer techniques (Patton and Kryskalla, 2003). An unfiltered sub-sample was also filtered through a glass fibre filter paper (nominal pore size 0.7 μm), the paper dried, and sediment determined gravimetrically. The same analyses were also conducted for samples taken from the deposition samplers. For soil samples, Mehlich-3 soil P concentration was determined by colorimetric determination of filtered solutions after extraction of 1 g soil with 10 mL of 0.2 M CH_3COOH, 0.25 M NH_4NO_3, 0.015 M NH_4F, 0.013 M HNO_3, and 0.001 M EDTA for 5 min (Mehlich, 1984).

Data for mean concentration and load of N, P and sediment in deposition samplers and in streamflow was subjected to an analysis of variance after testing for normality. Land use and location of samplers (site) were used as treatments for comparison. The interaction between land use and site was also tested. Preliminary analysis indicated that there was no significant temporal effect (seasonally or annually) on concentrations and loads. For woodland, this may have been influenced by the removal of large pieces of vegetation, and for cropland, by only short periods of bare ground. Hence, only the mean annual concentration or load is presented. The least significant difference at the $P < 0.05$ level (LSD_{05}) is given for comparison between treatment means.

3. Results and discussion

Mean concentrations and loads of TP, TN, and sediment within the deposition samplers are given in Tables 1 and 2. There was a significant difference between the mean concentrations and loads of all constituents measured between land uses, sites, and for the interaction between land uses and sites. Among land uses, the greatest mean concentration and load for all constituents was found in cropland, followed by pasture, and then woodland. The concentration and load of constituents in deposition samplers at stream flumes (see Fig. 1), was generally closer to that in cropland than either pasture or woodland.

Table 1
Mean concentrations of total P, total N, and sediment for each land use and site collected by the deposition samplers.

Landuse	Site	TP (mg L^{-1})	TN (mg L^{-1})	Sediment (g L^{-1})
Woodland	1	0.288	2.170	0.169
Woodland	2	0.309	2.191	0.185
Woodland	3	0.365	2.243	0.197
Woodland	Overall mean	0.321	2.210	0.184
Pasture	1	1.744	8.894	0.306
Pasture	2	0.800	7.549	0.214
Pasture	3	0.546	6.834	0.200
Pasture	Overall mean	1.030	7.760	0.240
Cropland	1	1.631	9.355	0.396
Cropland	2	2.578	11.107	0.435
Cropland	3	1.537	10.050	0.416
Cropland	4	1.028	8.874	0.290
Cropland	5	2.316	10.768	0.457
Cropland	Overall mean	1.818	10.030	0.399
Flumes	1	1.955	10.240	0.455
Flumes	2	2.656	9.319	0.522
Flumes	3	1.200	7.894	0.372
Flumes	4	0.946	6.748	0.253
Flumes	Overall mean	1.689	8.550	0.400
$LSD_{05\text{-landuse}}$[a]		0.294	0.919	0.069
$LSD_{05\text{-site}}$		0.309	1.034	0.078
$LSD_{05\text{-landuse} \times \text{site}}$		0.603	2.019	0.152

[a] The least significant difference at the $P < 0.05$ level is given for means of landuse, sites and the interaction of landuse and sites.

Table 2
Mean loads of total P, total N, and sediment for each landuse and site collected by deposition samplers.

Landuse	Site	TP (kg ha^{-1})	TN (kg ha^{-1})	Sediment (Mg ha^{-1})
Woodland	1	1.264	6.933	0.785
Woodland	2	1.143	6.226	0.834
Woodland	3	1.444	7.369	0.971
Woodland	Overall mean	1.284	6.843	0.863
Pasture	1	7.760	30.301	1.447
Pasture	2	3.178	29.256	0.941
Pasture	3	2.207	24.815	0.875
Pasture	Overall mean	4.382	28.124	1.088
Cropland	1	6.828	34.643	1.689
Cropland	2	10.905	46.017	1.886
Cropland	3	6.475	39.950	1.896
Cropland	4	4.232	33.367	1.142
Cropland	5	10.025	44.922	2.110
Cropland	Overall mean	7.693	39.780	1.744
Flumes	1	8.822	39.339	2.062
Flumes	2	6.797	33.153	2.045
Flumes	3	5.148	31.173	1.835
Flumes	4	2.879	26.395	1.067
Flumes	Overall mean	5.911	32.515	1.752
$LSD_{05\text{-landuse}}$[a]		0.460	1.230	0.130
$LSD_{05\text{-site}}$		0.387	1.277	0.124
$LSD_{05\text{-landuse} \times \text{site}}$		0.755	2.493	0.242

[a] The least significant difference at the $P < 0.05$ level is given for means of landuse, sites and the interaction of landuse and sites.

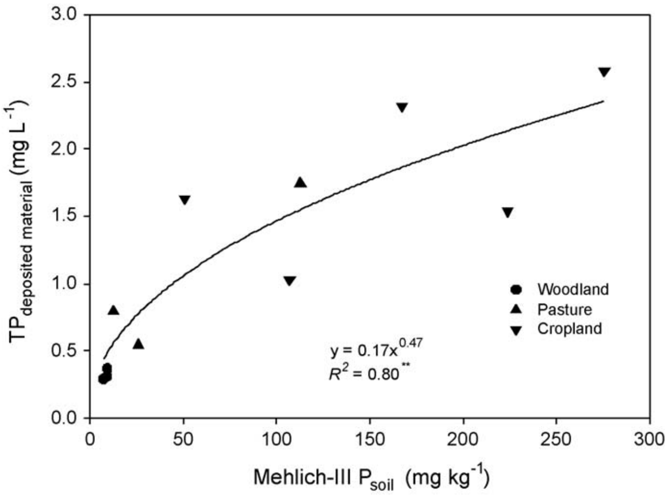

Fig. 2. Relationship between mean total P (TP) from deposition samples during 2004–2006 and Mehlich-III P determined on soil samples taken within a 10-m radius of the deposition samplers in April 2005.

3.1. Influence of local materials

Past studies have shown linkages between wet and dry deposition of P and local land use activities. For instance, Hendry et al. (1981) noted that wet deposition of P in Florida was greatest in areas of intensive agricultural activity and near phosphate mines and least in forested areas and near the coast. Within small watersheds such as FD-36 (39.5 ha), the likelihood of highly variable nutrient and sediment concentrations within wet deposition is minimal, although different rainfall patterns may dictate that volume would vary. Indeed, comparison of P concentrations in wet deposition from two rain gages, one within the watershed and one 3-km away in a similar mixed landuse watershed, indicted no significant difference. Dry deposition, largely from wind erosion, varies closest to the ground and over short distances (Li et al., 2004). The influence of local activities like land use and the application of P fertilisers and manure on deposited material was evident by the close relationship between total P in the deposited samples and Mehlich-3 P in soils within a 10-m radius of the collectors (Fig. 2; $R^2 = 0.80$; $P < 0.01$).

The mean N to P concentration ratio of the material deposited in the samplers was 6.1. Although insects and pieces of vegetation that could be removed from the samplers by hand were done so weekly, remaining biogenic material (e.g., pollen and small seeds) was not. The N to P ratio in the deposited material was low, and according to Downing and McCauley (1992) more characteristic of soil and sediment than wet deposition. This is unsurprising, given the quantity of P in the samples (0.34–2.67 kg P ha^{-1} yr^{-1}, Table 2) compared to the small quantity P returned to the land in wet deposition (0.01–0.1 kg P ha^{-1} yr^{-1}; Tabatabai, 1983). While an exact proportion of wet to dry deposition cannot be determined, if the greatest load measured by Tabatabai (1983) was used (0.1 kg P ha^{-1} yr^{-1}), then a maximum of 4% in cropland to 29% in woodland of the P collected by samplers was as wet deposition. In comparison, Tskuda et al. (2006) measured dry deposition as 54% of total deposition, but losses were small (~0.3 kg P ha^{-1} y^{-1}) and measurements were taken in a sheltered forest. In contrast, Anderson and Downing (2006) measured 2.6 times greater quantities of dry than wet deposition in an agricultural region. Irrespective of the type of deposition, the load of P measured in our study, except cropland, was within the range summarized from 24 studies by Tskuda et al. (2006) of 0.07–1.68 kg P ha^{-1} to land annually.

3.2. Potential contribution to the stream

Concentrations and loads of P in streamflow and TP, TN, and sediment in deposited material tended to increase from flume four to flume one (Table 2). McDowell et al. (2001) noted a similar trend for P measured at each flume during baseflow and attributed this to the influence of stream bed sediments. Given the similar relationship with deposited P concentrations, this suggests that a proportion of P may have arisen from deposited sediments, which are in turn derived from wind erosion of surrounding soil (Fig. 2). Literature data has estimated P deposition rates that at first inspection could be an important source of P in streamflow. For example, during summer, Cole et al. (1990) measured losses to lakes that, if expanded to include an entire year, ranged from 0.33 to 7.73 kg P ha^{-1} yr^{-1}. Newman (1995) estimated that a general range of P deposition rates on land could be from 0.01 to 10 kg P ha^{-1} yr^{-1}. In our study, both deposition rates and stream flow loads of TP at each flume (Tables 2 and 3) were within this range. However, while deposition rates may seem high on a per hectare basis the overall contribution will depend on the surface area of the stream.

In Table 3, contributions to stream load are presented using the deposition rates to the stream, as measured at each flume, and channel dimensions to calculate surface area of the stream. Although Cole et al. (1990) point out that deposition rates decline exponentially with distance from the shore of a lake, due to the deposition of heavy particles like pollen (~20 μm; Sugita, 1993), in a narrow stream (i.e. <5 m width) there is unlikely to be much change. The channel within FD36 is about 2-m wide (±25%). The

Table 3
Mean annual loads of total P (TP), total N (TN), and sediment in streamflow for each flume between 2004 and 2006 and the contributions to stream loads on an area and percentage basis according to channel dimensions.

Flume	Stream loads			Area of stream channel[a] (m^2)	Input to stream[b]			Proportion of stream load		
	TP (kg ha^{-1})	TN (kg ha^{-1})	Sediment (Mg ha^{-1})		TP (kg)	TN (kg)	Sediment (Mg)	TP (%)	TN (%)	Sediment (%)
1	1.410	27.090	1.343	172	0.754	4.562	0.236	1.4	0.4	0.4
2	0.531	11.588	0.545	444	0.602	3.886	0.201	3.0	0.9	1.0
3	0.443	6.479	0.446	212	0.300	2.414	0.110	2.4	1.3	0.9
4	0.418	10.232	0.513	664	0.191	1.753	0.071	1.9	0.7	0.6
Overall mean	0.700	13.847	0.712							
LSD_{05}[c]	0.601	5.323	0.704							

[a] Area calculated assuming 2-m channel width and lengths for each segment of watershed for each flume from McDowell et al. (2001).
[b] Calculated from deposition to flumes in Table 2 and the area of stream channel (e.g., 0.191 kg P = 0.0664 ha × 2.879 kg P ha^{-1} for flume 4). Note that contributions to flume 1 (farthest downstream) include contributions from upstream flumes two to four, whereas flume 2 includes contributions from flumes 3 and 4, and flume 3 includes the contribution from flume 4.
[c] The least significant difference at the $P < 0.05$ level is given for comparison between mean loads of flumes.

corresponding surface area of stream channel exposed to deposition is about 660-m^2 before flume four, 210-m^2 before flume three, 440-m^2 before flume two, and 170-m^2 before flume one (each sub-watershed length of reach is given in Table 3). Assuming that deposition rates at each flume are representative of rates along the length of reach in each sub-watershed, the contribution of atmospheric deposition to mean annual stream loads of TP were generally <3 and $<1\%$ for TN and sediment.

4. Conclusions

Data showed a good relationship between P concentrations in deposition and soil <10 m of the deposition sampler suggesting much of the deposited sample was locally derived. Deposition increased with the degree of land management, such that woodlands received 0.36 kg TP and 2.33 kg TN $ha^{-1}\ yr^{-1}$, while cropped areas received 1.93 kg TP and 10.71 kg TN $ha^{-1}\ yr^{-1}$. Sediment deposition averaged 0.22 Mg $ha^{-1}\ yr^{-1}$ on wooded land and 0.44 Mg $ha^{-1}\ yr^{-1}$ on cropped land. Deposition measured at four points in the stream bank was equivalent to stream loads for P, but calculations shows that when the surface area of the stream was accounted for deposition would account for <3% of stream flow loads. The proportion for N and sediment was generally <1% suggesting that mitigation strategies for nutrients and sediment in this mixed-land use watershed should focus on runoff pathways.

Acknowledgements

The authors acknowledge the assistance of David Otto in construction and sample collection from deposition samplers and of Todd Strohecker, Mike Reiner and Terry Troutman for maintenance of flumes and sampling equipment in FD-36 and for collection of flow data and stream flow samples. We also thank Mary Kay Lupton, Joan Weaver, Charles Montgomery and Paul Spock for the analysis of P forms in water samples.

References

Anderson, K.A., Downing, J.A., 2006. Dry and wet atmospheric deposition of nitrogen, phosphorus and silicon in an agricultural region. Water Air Soil Pollut. 176, 351–374.
Ann, H., James, R.T., 2001. Variability, uncertainty, and sensitivity of phosphorus deposition load estimates in South Florida. Water Air Soil Pollut. 126, 37–51.
Asman, W.A.H., 2002. Ammonia and ammonium. In: Holton, J.R., Pyle, J.A., Curry, J.A. (Eds.), Encyclopaedia of Atmospheric Sciences. Academic Press, London, UK, pp. 2365–2376.
Asman, W.A.H., Sutton, M.A., Schjorring, J.K., 1998. Ammonia: emission, atmospheric transport and deposition. New Phytol. 139, 27–48.
Cole, J.J., Caraco, N.F., Likens, G.E., 1990. Short-term atmospheric transport: a significant source of phosphorus to an oligotrophic lake. Limnol. Oceanogr. 35, 1230–1237.
Downing, J.A., McCauley, E., 1992. The nitrogen:phosphorus relationship in lakes. Limnol. Oceanogr. 37, 936–945.
Hendry, C.D., Brezonik, P.L., Edgerton, E.S., 1981. Atmospheric deposition of nitrogen and phosphorus in Florida. In: Eisenreich, S.J. (Ed.), Atmospheric Pollutants in Natural Waters. Ann Arbor Sci. Publ., Ann Arbor, MI, pp. 199–215.
Howarth, R.W., 2006. Atmospheric deposition and nitrogen pollution in coastal marine ecosystems. In: Visgilio, G.R., Whitelaw, D.M. (Eds.), Acid in the Environment—Lessons Learned and Future Prospects. Springer, New York, NY, pp. 97–116.
Kocak, M., Kubilay, N., Mihalopoulos, N., 2004. Ionic composition of lower tropospheric aerosols at a Northeastern Mediterranean site: implications regarding sources and long-range transport. Atmos. Environ. 38, 2067–2077.
Larney, F.J., Bullock, M.S., Janzen, H.H., Ellert, B.H., Olson, E.C.S., 1998. Wind erosion effects on nutrient redistribution and soil productivity. J. Soil Water Conserv. 53, 133–140.
Li, F.-R., Zhao, L.-Y., Zhang, H., Zhang, T.-H., Shirato, Y., 2004. Wind erosion and airborne dust deposition in farmland during spring in the Horqin Sandy Land of eastern Inner Mongolia, China. Soil Till. Res. 75, 121–130.
McDowell, R., Sharpley, A., Folmar, G., 2001. Phosphorus export from an agricultural watershed: linking source and transport mechanisms. J. Environ. Qual. 30, 1587–1595.
Mehlich, A., 1984. Mehlich 3 soil test extractant: a modification of Mehlich 2 extractant. Commun. Soil Sci. Plant Anal. 15, 1409–1416.
NADP, 2004. National Research Support Program – 3, in United States National Atmospheric Deposition Program, Program Office.
Newman, E.I., 1995. Phosphorus inputs to terrestrial ecosystems. J. Ecol. 83, 713–726.
Patton, C.J., Kryskalla, J.R., 2003. Methods of analysis by the U.S. Geological Survey National Water Quality Laboratory: evaluation of alkaline persulfate digestion as an alternative to Kjeldahl digestion for the determination of total and dissolved nitrogen and phosphorus in water. U.S. Geological Survey, Water Resources Investigations Report 03-4174. U.S. Geological Survey, Branch of Information Services, Federal Center, Denver, CO, 33 pp.
Pearson, J., Stewart, G.R., 1993. Tansley review No. 56. The deposition of atmospheric ammonia and its effects on plants. New Phytol. 125, 283–305.
Pionke, H.B., Gburek, W.J., Sharpley, A.N., 2000. Critical source area controls on water quality in an agricultural watershed located in the Chesapeake Basin. Ecol. Eng. 14, 325–335.
Ryden, J.C., Syers, J.K., Harris, R.F., 1973. Phosphorus in runoff and streams. Adv. Agron. 25, 1–45.
Sharpley, A.N., Moyer, B., 2000. Forms of phosphorus in manures and composts and their dissolution during rainfall. J. Environ. Qual. 29, 1462–1469.
Sharpley, A.N., Kleinman, P.J.A., Heathwaite, A.L., Gburek, W.J., Folmar, G.J., Schmidt, J.P., 2008. Phosphorus loss from an agricultural watershed as a function of storm size. J. Environ. Qual. 37, 362–368.
Shaw, R.D., Trimbee, A.M., Minty, H., Fricker, H., Prepas, E.E., 1989. Atmospheric deposition of phosphorus and nitrogen in central Alberta with emphasis on Narrow Lake. Water Air Soil Pollut. 43, 19–134.
Sugita, S., 1993. A model of pollen source area for an entire lake surface. Quat. Res. 39, 239–244.
Tabatabai, M.A., 1983. Atmospheric deposition of nutrients and particles. In: Schaller, F.W., Bailey, G.W. (Eds.), Agricultural Management and Water Quality. Iowa State Univ. Press, Ames, IA, pp. 92–108.
Tskuda, S., Sugiyama, M., Harita, Y., Nishimura, K., 2006. Atmospheric phosphorus deposition in Ashiu, Central Japan—source apportionment for the estimation of true input to a terrestrial ecosystem. Biogeochemistry 77, 117–138.