

Twenty-Three-Year Changes in Upland and Bottomland Forest Soils of Central Illinois

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Abstract: Soil formation is a complex process that varies with climate, relief, parent material, organisms, time, and in response to anthropogenic inputs. Time is often difficult to study. Our objective was to determine if the chemistry of two forest soils changed during a 23-year period in response to decreased acidic deposition (upland and bottomland soils) or by periodic sediment inputs (bottomland soils). These sites were sampled by depth in 1986 and 2009 using the same protocols. During an 8-year period in the 1990s, sediment deposition to the floodplain was measured. Soils at all depths and both sites had trends of greater exchangeable Ca, Mg, and K, increased pH, and decreased exchangeable Al and H. However, because of soil variability, increases were only significant in the base rich bottomland soils and were not thought to be biogeochemically important. Soil organic matter was relatively unchanged at both sites, with greater concentrations of C, N, and S in bottomland soils because of sediment deposition. Flooding was estimated to add 37 Mg ha⁻¹ year⁻¹ of organic-rich sediment to the floodplain soils. Sulfate deposition decreased by a total of 3.1 keq ha⁻¹ during the 24 years between sampling, whereas base cation pools (0–80 cm) in the bottomland and upland soils increased by 458 and 540 keq ha⁻¹ (14 and 85% increases), respectively. Therefore, decreased acidic deposition likely had little effect on base cation pools, and the changes observed were thought to be caused by soil variability and possibly through canopy species variation affecting litterfall chemistry.

Key Words: Soil nutrients, cation chemistry, sediments, silver maple, oak-hickory

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Forest soils provide nutrients for forest growth and health, and understanding their multidecadal changes is important for management purposes. In addition, further knowledge about how forest soils respond to time and anthropogenic factors could help better understand topics such as C storage (Oulehle et al., 2011). Both natural processes such as weathering and sediment inputs and anthropogenic impacts such as acidic deposition are likely to lead to changes in soils (Lawrence et al., 2012). Understanding these changes during long periods can have important land management implications relating to forest health. Retrospective studies, providing direct evidence of chemical changes in forest soils, are relatively rare because of a lack of appropriate samples, documentation, and archiving of samples over decadal or longer periods (Bailey et al., 2005). Richter et al. (2007)

stressed that long-term soil experiments are especially valuable if their time scale data are accompanied by a sample archive that can be analyzed long after collection. Moreover, assessment of soil quality and health and direction of change with time is a primary indicator of sustainable management (Doran, 2002). One example of a multidecadal soil experiment is the work by Johnson et al. (2008), during three decades at Walker Branch Watershed, who studied changes associated with tree uptake, atmospheric deposition, and leaching effects on a forest soil using soil samples taken from 1972, 1982, 1993, and 2004. Lapenis et al. (2004) and Lawrence et al. (2005) used soil samples from 1926, 1964, and 2001 near St. Petersburg, Russia, to demonstrate long-term impacts of acidic deposition on forest soils. Lawrence et al. (2012) documented the recovery of red spruce forest soils in the northeastern United States to acidic deposition. Researchers in these studies were able to reach their conclusions because of the availability of the archived soil samples and the possibility to recreate the original sampling protocols.

A recent review by Lawrence et al. (2013) on measuring environmental change in forest soils summarizes the need for repeated sampling and illustrates the type of results and interpretations that can be made. However, they also discuss the many factors that can affect conclusions from these types of studies, such as changes from storage and soil variability.

Soil acidification is a naturally occurring process; trees naturally acidify soils by donating protons in exchange for base cations needed for growth. This process occurs when they take up cations such as Ca and Mg from the exchange sites on the soil colloids. However, acidic deposition can accelerate this natural process (Yanai et al., 1999) and is now recognized as a key factor in the loss of base cations in the northern hemisphere (Rice and Herman, 2011; Lawrence et al., 2012). A 30-year forest hydrogen (H⁺) budget (Calhoun Experimental Forest, SC) estimated that 38% of soil acidification (Ultisols) was caused by acidic deposition, whereas the other 62% of soil acidification was attributed to the internal functioning of the ecosystem (Markewitz et al., 1998).

Soils that form on alluvium along rivers receive episodic inputs of new sediments eroded from the upstream watershed. Effects of sediment inputs on soil genesis and nutrient cycling are not well documented in the literature. These soils typically have no or little development of pedogenic horizons and are typically classified as Fluvents. However, in much of Illinois, floodplain soils are often classified as Aquolls because the soils have properties and diagnostic features that meet Mollisol requirements, including a mollic epipedon (Buol et al., 2003; Soil Survey Staff, 2013). Sediments enrich these soils and add nutrients that the forest can use (Peterson and Rolfe, 1982; Bartel-Ortiz and David, 1988; David, 1989). Floodplain forests are unique in that they could help regulate and improve downstream water quality (Wright et al., 2001), but most floodplains in North America and Europe have been cultivated (Tockner and Stanford, 2002). Landscape position of a floodplain forest can determine if the flow is from upstream and/or upslope (Brinson, 1993; Lockaby and Walbridge, 1998; Wright et al., 2001).

We took advantage of archived soil samples to evaluate forest soil response to acidic deposition in an upland soil and sediment

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inputs in a bottomland soil during a 23-year period. The overall purpose of this study was to better understand anthropogenic effects on soil processes that operate on a decadal time scale. The specific objectives of this study were to determine if forest soil chemistry changed in (i) an upland forest soil in central Illinois in response to reduced acidic deposition inputs and (ii) in a bottomland floodplain soil in central Illinois that receives periodic sediment inputs as well as reduced acidic deposition inputs.

MATERIALS AND METHODS

Study Sites

Robert Allerton Park, located in Monticello along the Sangamon River in central Illinois, was selected by Bartel-Ortiz and David (1988) as the foundation for a multidecadal soil experiment. Their study focused on the effects of acidic deposition and floodplain sediment deposits on the S constituents of the forest soils. They established baseline data and archived all soil samples that made it possible for further studies on effects of anthropogenic activities.

Silver maple (*Acer saccharinum* L.) was the dominant species in the floodplain at Allerton Park. Eastern cottonwood (*Populus deltoides* Bartram ex Marsh.), American sycamore (*Platanus occidentalis* L.), and American elm (*Ulmus americana* L.) were the subdominant species. Northern red oak (*Quercus rubra* L.) and black oak (*Quercus velutina* Lam.) were the dominant species in the upland. Bitternut hickory (*Carya cordiformis* (Wangenh.) K. Koch), black walnut (*Juglans nigra* L.), shagbark hickory (*Carya ovata* Mill.), white oak (*Quercus alba* L.), hackberry (*Celtis occidentalis* L.), and white ash (*Fraxinus americana* L.) are the subdominant species. Slippery elm (*Ulmus rubra* L.), eastern redbud (*Cercis canadensis* L.), and paw paw (*Asimina triloba* (L.) Dunal) made up the dense woody understory in the upland (Bartel-Ortiz and David, 1988; David, 1989). After a full inventory, few changes in species composition were found between 1986 and 2009 at either site. Silver maple comprised 81 and 73% of the trees greater than 10 cm in 1986 and 2009, respectively, as well as about 50% of the basal area. Oaks and hickories were 73 and 78% of the basal area in 1986 and 2009, respectively. The soil series in the upland site was a Russell silt loam (fine-silty, mixed, superactive, mesic Typic Hapludalfs) on a 4% slope, whereas the bottomland site was on a nearly level floodplain with Sawmill soil series (fine-silty, mixed, superactive, mesic Cumulic Endoaquolls); this was verified during the 2009 field sampling. Russell soils are developed in 50 to 100 cm of Peoria loess over loamy Wisconsinian till with carbonates at greater than 100 cm; Sawmill soils are developed in silty clay loam alluvium, with a dark epipedon greater than 60 cm thick (Fehrenbacher et al., 1984). A 0.1-ha circular plot was established in both upland and bottomland sites in 1986, with a 20 × 20-m square grid placed in the center. The square was further divided into 2 × 2-m subplots, each with unique numbering.

The Sangamon River has a drainage area of 1,484 km² at Allerton Park, and it generally floods annually (Brown and Peterson, 1983; Oldenburg 1983; Polit 1992). Although flooding may occur any time in the year when precipitation is high, most of the events follow spring thaw and occur during the months of March and April, with flood waters generally receding by the end of May (Brown and Peterson 1983; Oldenburg 1983). Brown and Peterson (1983) indicated that average flooding depth in the Sangamon River at Allerton Park was 70 cm, with a maximum depth of 160 cm measured. Polit (1992) estimated that the floodplain soils were typically flooded about 10% of the year. To assess sediment inputs, we used a network of permanent plots

established in 1985 for forest tree measurements. These plots were grouped into three units (Fig. 1), and we made use of a subset of plots in each unit. In February 1991, five cement blocks (0.037 m²) were placed at a random distance and direction from a central stake in 18 plots. The blocks were placed flush with the soil surface for retention of sediments on the rough surface. Steiger et al. (2003) discussed various methods for sediment deposition studies and suggest that flat sediment traps are appropriate for quantifying sedimentation rates and composition. Sediment sampling from the blocks was conducted during September to November of 1998. A total of 74 blocks were located and sampled 8 years after placement (three to five blocks per plot). A stainless steel sampling template (15 × 15 cm) was pushed through the sediment to the top of the block. Surface debris was cleared, and all sediments inside the template were removed. A depth measurement was taken for each block by measurement of sediment thickness on all four sides after the template was removed.

Sample Collection and Analysis

Both upland and bottomland sites were resampled following the protocol of Bartel-Ortiz and David (1988) during the summer of 2009—23 years after the initial sampling. A single 0.1-ha plot was established at each site in 1986 and resampled in 2009. In addition to collecting soil samples, tree canopies were inventoried at each site. Diameter at breast height was recorded for all trees that fell within the 0.1-ha plot. A list of random numbers to 100 was generated, and the first five numbers that appeared in the list were used to choose the soil pits. At the time of sampling, two types of soil conditions were excluded: (i) soil pits initially sampled by Bartel-Ortiz and David (1988) in 1986 and (ii) soils that showed signs of disturbance such as bioturbation and presence of roots. The dimensions of the soil pits dug in the summer of 2009 were approximately 0.5 × 0.5 × 1 m deep. Ten locations were chosen, and soil pits were dug by hand; five in the upland site and five in the bottomland site. Various limiting conditions such as the presence of tree roots and coarse fragments contributed to the differences in the sampling depths chosen in these soil pits. The upland pits were sampled during the first week of July 2009, and the bottomland pits were sampled in early August of 2009. Bulk density samples were obtained using a 7.6-cm-diameter ring collector from 0 to 10 cm, and at 20, 45, and 70 cm within the soil profile, similar to the bulk density sampling in 1986. Bulk soil samples were collected at depths of 0 to 10, 10 to 30, 30 to 60, and 60 to 80 cm in all 10 pits. Approximately 1 to 2 kg of bulk soil was collected from each sampling depth using a trowel. All soil samples were placed into plastic bags in the field and stored in the cooler in the field. Two soil profiles, one in the upland and one in the bottomland, were described at the time of sampling.

Soil samples were air-dried and crushed by hand, and then a 2-mm sieve was used to separate coarse fragments and large organic materials. Sieved samples were used for soil pH, total Kjeldahl N, and exchangeable cation analyses. A small portion of the samples was taken from the 2-mm sieved samples to be finely ground by hand using a mortar and pestle for total C, N, and S analyses. Soil pH was measured with a glass electrode (model 250A; Orion) using a 1:2 ratio of soil to solution with 0.01 M CaCl₂ (Robarge and Fernandez, 1986). Total C and N were determined using an elemental analyzer (Elemental Analyzer, EAS 4010; Costech). Total S was determined using a Leco sulfur determinator (model SC-132) (David et al., 1989). Exchangeable cations were extracted using a vacuum extractor. Different carrier solutions were used: 1 M unbuffered ammonium chloride (NH₄Cl) for exchangeable Ca, Mg, K, and Na and 1 M unbuffered potassium chloride (KCl) for exchangeable Al and

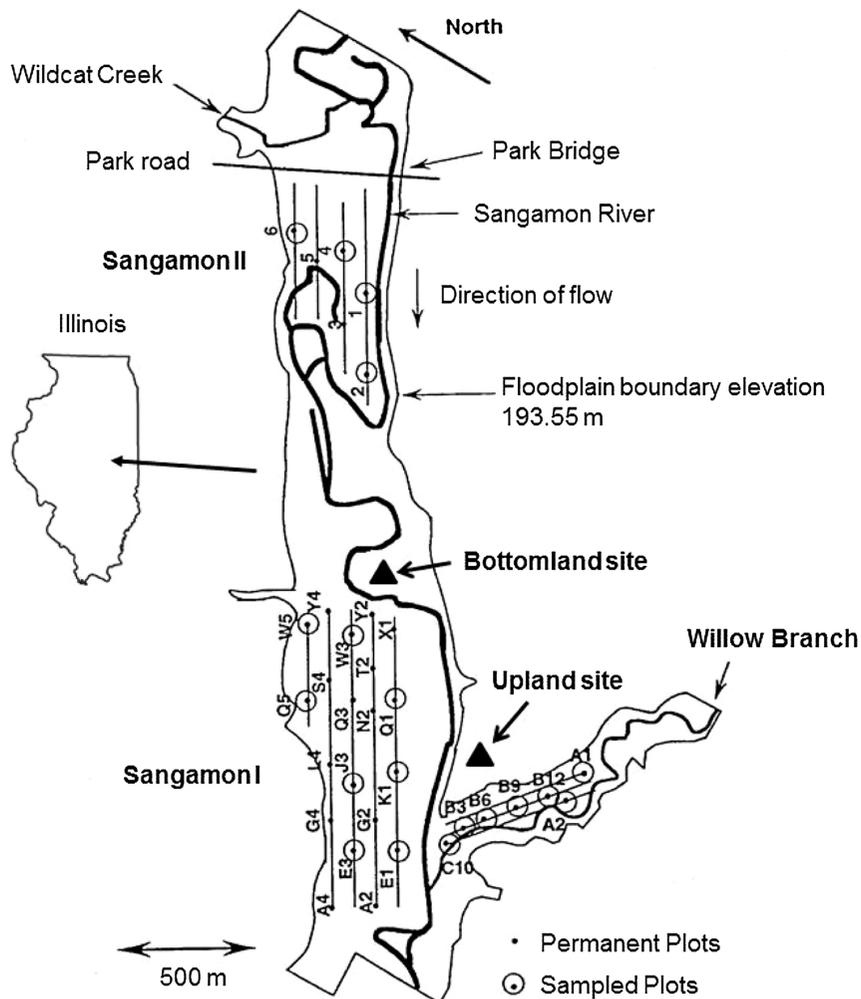


FIG. 1. Map of Allerton Park, IL, showing the locations of permanent plots and plots chosen for this study. Thirty-three permanent plots were established in 1985, and 19 were used in this study for sedimentation deposition estimates. Also shown are the upland and bottomland sites with archived samples (shown as solid triangles), first used by Peterson and Rolfe (1982).

H. Calcium, Mg, K, and Na were measured using atomic absorption, with lanthanum added for Ca and Mg by adsorption and cesium for K by adsorption (Robarge and Fernandez, 1986). Exchangeable acidity was determined by titration after the KCl extraction (Robarge and Fernandez, 1986), with Al determined by inductively coupled plasma emission spectrometry.

Sediment samples collected from the cement blocks were air-dried and sieved (<2 mm) before analysis. Organic matter content was determined by loss-on-ignition (David, 1988). Soil texture was determined by hydrometer for clay and silt, with sand determined by wet sieving through a standard USDA sieve set (Gee and Bauder, 1986).

Archived Soil Samples and Statistics

Archived samples from 1986 were reanalyzed to ensure that differences observed through time were not caused by changes in analytical methods. The archived samples had been stored in closed plastic bags that had been placed in ice cream cardboard containers at room temperature. Both the 1986 and 2009 samples were analyzed in the same laboratory at the University of Illinois under the supervision of Mark David. For exchangeable base cations, 10 samples from the upland and 10 samples from the

bottomland soils from the 1986 sampling were used for comparison. However, for total C, total N, pH, exchangeable Al, and exchangeable H, all 1986 archived soil samples were reanalyzed along with the 2009 samples. Markewitz et al. (1998) used a similar comparison to examine the differences found using archived samples from the Calhoun Experimental Forest. For exchangeable Ca and Mg, the r^2 calculated between the analysis performed in 1987 and during this study was greater than 0.98 and was close to the 1:1 line. The r^2 for exchangeable K was less (0.65), but that could be attributed to the low concentration of exchangeable K in these soils that increased analytical variability. The 1986 cation values were corrected based on this reanalysis using the regression equations developed (linear regression coefficients: Ca b0:-0.21, b1:1.10; Mg b0:-0.09, b1:1.02; K b0:0.01, b1:1.12). For pH, a quadratic regression model using SAS was developed for the upland soils with all depths in the model, with original 1986 analysis, reanalysis of 1986 samples in 2010, and 2009 sample analysis, as terms in the model to determine if a significant change in overall soil profile pH occurred in each set of measurements. We found that all terms were significant ($P < 0.05$) and that soil pH had increased in the acidic upland soils with storage. Therefore, we made our comparison with the original pH values measured in 1986. Two archived samples were used as part of the quality

control protocol when measuring total S. Reanalysis of the data indicated that the analysis performed during this study was 6 and 26% lower for these samples than the original measurement (277 and 100 mg kg⁻¹ in 2009 compared with 261 and 79 mg kg⁻¹ in 1987, respectively). Statistical differences between measurements of the two sampling periods by site and soil depth were determined by *t* test using SAS version 9.3 (SAS Institute, 2011). We also conducted a power analysis using SAS assuming $\alpha = 0.05$ and power of 0.9.

RESULTS

Sedimentation in the floodplain varied from 0.6 to 5.6 cm deposited during the 8-year sampling period, with averages of 2.8, 2.6, and 4.2 cm in the three sampling units, respectively (Table 1). The greatest sediment deposition was measured in the Willow Branch plots, which were located along a stream that entered into the river. Water likely backed into this area, and perhaps, slower currents increased sediment deposition. Sediments were enriched with organic matter, ranging from 7 to 17% across all plots. Sediment deposition was 27.4, 26, and 52.4 Mg ha⁻¹ year⁻¹ when averaged by sampling unit. Soil particle size distribution was mostly silt and clay, with little sand (Table 2). Elevation of plots ranged from 192.3 to 193.4 m, and no relationship was found between sediment depth, mass, or particle size with distance from the river channel or with elevation.

TABLE 1. Depth of Deposited Sediments Along With Organic Matter and Sediment Deposition by Plot in the Bottomland Forest During an 8-Year Period

Unit and Plot	n	Depth cm	Organic Matter %	Sediment Deposition Mg ha ⁻¹ year ⁻¹
Sangamon I				
W5	4	0.6 ± 0.3	17 ± 2.5	19.9
Q5	4	1.4 ± 0.7	13 ± 0.4	19.0
E3	4	2.6 ± 0.9	17 ± 0.7	19.9
W3	4	2.8 ± 1.0	12 ± 0.6	31.9
E1	4	2.0 ± 0.4	12 ± 1.2	18.5
K1	4	2.7 ± 0.2	12 ± 0.5	25.2
Q1	4	4.9 ± 0.6	12 ± 0.4	39.6
J3	4	4.0 ± 0.7	12 ± 0.9	45.4
Mean		2.8	13.5	27.4
Sangamon II				
6	3	1.6 ± 0.9	16 ± 3.1	18.2
1	4	2.7 ± 0.8	11 ± 0.8	27.9
2	5	3.4 ± 0.9	10 ± 0.7	31.9
Mean		2.6	12.3	26.0
Willow Branch				
A1	4	4.3 ± 0.2	7 ± 0.8	65.1
A2	5	4.8 ± 0.8	8 ± 0.2	65.2
B12	5	4.6 ± 0.3	10 ± 0.3	52.8
B9	5	4.1 ± 0.4	9 ± 0.3	52.3
B6	3	5.6 ± 1.0	9 ± 1.0	56.3
B3	4	5.0 ± 0.4	12 ± 0.6	58.4
C10	5	1.2 ± 0.3	10 ± 0.3	16.4
Mean		4.2	9.1	52.4

Values are mean ± 1 SE.

TABLE 2. Mean Soil Texture and Textural Class of Deposited Sediments by Sampling Unit in the Bottomland Forest

Unit	Sand	Silt	Clay	USDA Class
	----- %-----			
Sangamon I	4.8	45.3	49.9	Silty clay
Sangamon II	4.8	51.0	44.2	Silty clay
Willow Branch	3.6	60.8	35.7	Silty clay loam

Bottomland soil pH data measured in 1986 were directly compared with data collected from the 2009 sampling, with a significant increase found at all depths above 60 cm, although the increases were small (Fig. 2). Compared with 1986, increases of less than 0.15 pH units were observed in 2009 from 0- to 60-cm depths, with no change in the 60- to 80-cm soil depth. For the upland soils, we found that storage for 23 years had increased pH. When we compared original 1986 soil pH values with our samples from 2009, we found a significant increase in pH at every soil depth. Soil pH in 2009 increased by 0.2 units in the 0- to 30-cm and 60- to 80-cm depths and by 0.3 units in the 30- to 60-cm depth, respectively, compared with those in 1986.

The differences observed during the past 23 years in soil total C, N, and S concentrations were relatively small at both sites (Fig. 3). Although at certain depths total elemental concentration differences showed statistical significance, they were not large enough changes to be meaningful pedologically. Exchangeable Ca, Mg, and K increased significantly in nearly all soil depths in the bottomland site, with only a few significant increases in upland soils despite relatively large differences (Table 3). Exchangeable Na was negligible and unchanged (<0.15 cmol_c kg⁻¹; data not shown). Exchangeable K concentrations were less than 0.5 cmol_c kg⁻¹ in both soils but did show significant increases in most depths at both sites. Acid cations were only measured in the upland site because the concentrations of exchangeable Al and H in the bottomland site were negligible. Exchangeable Al had no significant changes, with exchangeable H decreasing only in the 0- to 10-cm and 10- to 30-cm depths. Cation exchange capacity increased significantly in all depths at both the upland bottomland sites, but base saturation generally was not significantly different. On a mass basis, the top 80 cm of soil in the upland had increased base cations (Ca, Mg, K) from 632 to 1,172 keq ha⁻¹ from 1986 to 2009, a difference of 540 keq ha⁻¹. Corresponding values for the bottomland soils were 3,187 and 3,644 keq ha⁻¹, an increase of 458 keq ha⁻¹.

DISCUSSION

Flooding and Sediment Effects

Soils in the bottomland site received episodic inputs of sediments during the study period by flooding from the Sangamon River (Table 1). Sediments deposited from the agricultural watershed (>90% corn and soybeans) were likely the contributing factor that maintained the relatively high total elemental concentrations observed in these soils through time. The particle size distribution of sediments was mostly silts and clays, in either the silty clay or silty clay loam textural class (Table 2). These are the dominant surface soil textural classes in the Sangamon River watershed upstream of Allerton Park (Martin, 1991). Peterson and Rolfe (1982) speculated on the source of sediment inputs at this same location and suggested that, although some of the sediments may have been from scouring of the Sangamon River

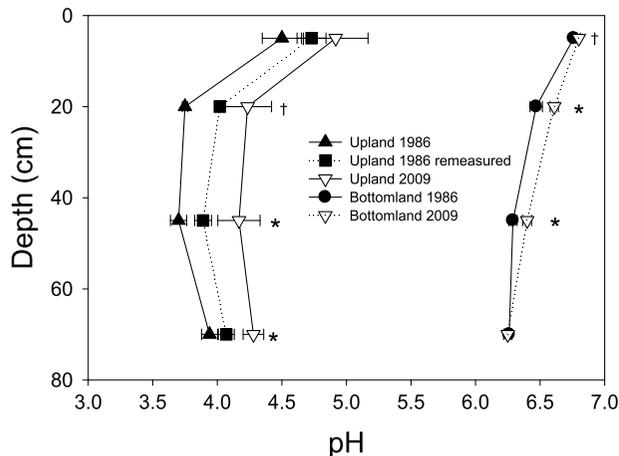


FIG. 2. Mean soil pH (with SE) by depth and site for samples collected and analyzed in 1986, reanalysis of archived 1986 samples for the upland site, and newly obtained samples from 2009. †Significant difference between sampling years at $P < 0.1$ and $P < 0.05$, respectively.

stream channel, most of it was probably derived from the upland areas. Although total C, N, and S all decreased with depth in the floodplain soils, all concentrations were greater than all depths in the adjacent upland soil, demonstrating how enriched these floodplain soils were throughout the entire soil profile.

Average sediment depth measured on the sample blocks was 3.2 cm, a 0.4-cm year⁻¹ sediment deposition rate (Table 1). During the 23 years of our study, sedimentation would have totaled 9.2 cm, and the 0- to 10-cm depth sampled in 1986 would now be in the 10- to 30-cm layer, complicating detecting changes through time. If we had sampled by 10-cm depth intervals through 30 cm, we could have more effectively compared the new 0- to 10-cm depth with the current 10- to 20-cm depth. However, given that there isn't that much variation with depth in base cations in the bottomland soils, sedimentation is clearly adding soil rich in base cations and similar to the underlying profile.

Benedetti (2003) estimated an accretion rate of 1 cm year⁻¹ since 1954 for low-lying floodplain surfaces in Iowa and Wisconsin. He suggested that these rates were less than earlier in the 20th century, when agricultural land disturbance was greater, but were an order of magnitude greater than the long-term average of 0.14 cm year⁻¹ during the Holocene (Benedetti, 2003). Jolley et al. (2010) measured sediment deposition of 0.05 to 2.89 cm year⁻¹ for riparian forests with various disturbance percentages in the watersheds at Ft. Benning, GA, with most of the sediment as sand. The average annual sediment input to the bottomland forest was 37 Mg ha⁻¹ year⁻¹, with an average percent organic matter of 11.6%. Surface soils in the bottomland had about 40 g C kg⁻¹ (Fig. 3). Although this is a large percentage of organic matter for a mineral horizon, the concentration was similar to the deposited sediment. Peterson and Rolfe (1982) measured a sediment input of 38.6 Mg ha⁻¹ year⁻¹ to the Allerton Park floodplain after a single large flood during the spring of 1979 using plastic plates, similar to our average 8-year measurement. Based on discharge measurements at Monticello, IL, at USGS station 05572000, just a few kilometers upstream from the floodplain of Allerton Park, flow records showed 49 days greater than 100 m³ sec⁻¹ (the flow rate that leads to overbank conditions) during 12 events, ranging from 1 to 11 days in length during the 8-year period the blocks were in place. Bhowmik et al. (1986) estimated the annual sediment load in the Sangamon River at Monticello during

the early 1980s at 67,273 Mg year⁻¹. Therefore, the 150-ha floodplain at Allerton Park is estimated to remove about 8% of the annual load of sediment in the Sangamon River, showing the importance of these forests at removing sediment from riverine systems that have moderate erosion rates because of intensive agriculture.

Soil pH

Soil pH is a characteristic that may experience substantial changes during air-dried storage. Raveh and Avnimelech (1978) found that organic matter decomposed or dissolved, causing pH to decrease (Blake et al., 2000). Falkengren-Grerup (1995) found that more acidic soils experience less change. Falkengren-Grerup (1995) found that, during a 6-year period, the archived samples with the greatest acidity changed less than 0.2 units, whereas the least acidic soils decreased by more than 0.6 units. Another study by Prodromou and Pavlatou-Ve (1998) showed pH decreased by 0.3 units on average with acid to neutral soils after 20 years of storage (Blake et al., 2000). Lawrence et al. (2012) also found small

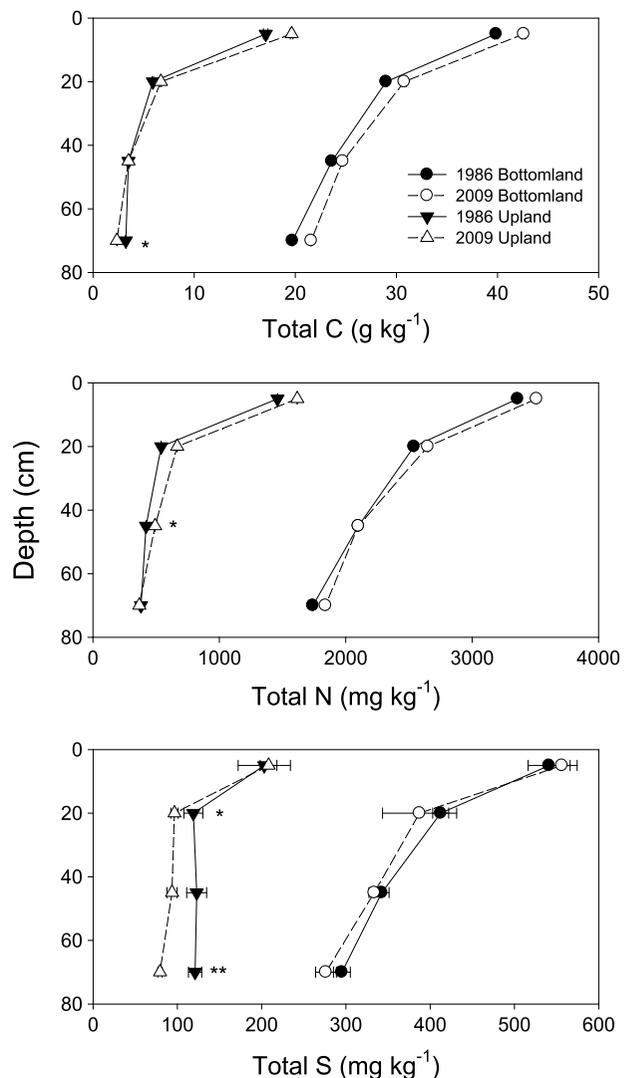


FIG. 3. Mean total C, N, and S concentrations (with SE) of samples collected in 1986 and 2009 by site and depth. *Significant difference between sampling years at $P < 0.05$.

TABLE 3. Mean Values of Soil Chemistry Measurements of Samples Collected From 1989 and 2009 by Site and Depth

Depth, cm	Ca		Mg		K		Al		H		Effective CEC		Base Saturation	
	1986	2009	1986	2009	1986	2009	1986	2009	1986	2009	1986	2009	1986	2009
	-----cmol _c kg ⁻¹ -----													
Bottomland														
0–10	27.9	31.4*	9.8	12.5**	0.3	0.5**	NA	NA	NA	NA	38.0	44.4**	100	100
10–30	25.3	27.8**	9.2	11.2**	0.3	0.4*	NA	NA	NA	NA	34.7	39.4**	100	100
30–60	21.8	24.5*	8.2	10.6**	0.2	0.2	NA	NA	NA	NA	30.2	35.3**	100	100
60–80	20.4	21.6	7.7	9.6**	0.1	0.2	NA	NA	NA	NA	28.2	31.4*	100	100
Upland														
0–10	4.0	6.5	1.0	1.7**	0.2	0.4**	0.6	0.6	0.2	0.1**	6.0	8.6	82.9	92.5
10–30	1.0	2.9	0.3	1.2	0.1	0.3*	2.6	2.1	0.5	0.1*	4.4	6.7**	31.0	64.4
30–60	2.4	6.5	2.3	4.3	0.2	0.3*	5.0	3.7	0.6	0.6	10.6	15.4*	45.9	70.1
60–80	5.5	8.8*	4.2	6.1	0.3	0.3	3.2	2.2	0.4	0.3	13.6	17.6	73.6	85.9*

*,**Within a site and soil depth significant difference between sampling years at $P < 0.05$ and 0.01 , respectively.

but significant decreases in pH in B horizon soils after 12 to 14 years of storage, which is consistent with the results of Blake et al. (2000) and Bailey et al. (2005). Inconsistent with previous studies, the Allerton Park archived upland soil samples were acidic when originally sampled and, during the 23 years of storage, increased by less than 0.3 pH units when reanalyzed. A sample of bottomland archived soil was used repeatedly in pH analysis as part of the quality control protocol, and the reanalysis average of those samples was 0.36 pH units lower than when it was originally analyzed. Based on these results, we used the original soil pH values measured shortly after sampling in 1986.

In the upland soil, pH increased between 0.3 and 0.5 units in each sampling depth, with a significant increase found for all depths except 0 to 10 cm. However, the data were quite variable. For example, in the 10- to 30-cm sampling depth, soil pH measured from samples collected from pits 15 and 33 in 2009 were much closer to the pH measurements from the same sampling depth obtained in 1986 (average, 3.74 and 3.75, respectively). However, samples collected in 2009 from the other three pits in the same sampling depth had an average of soil pH of 4.49, a total of 0.75 units higher. Our only explanation for this variability across the small plot is the species level effects in our mixed species forest, as is discussed later.

Reduced Acidic Deposition Response and Soil Variability

Since the passage of the Clean Air Act amendments in 1990, there has been a decrease in atmospheric deposition of S (NADP, 2013). An NADP site (Bondville, Illinois) is located about 32 km northeast of Allerton Park, and from 1979 to 2009, precipitation at this location averaged 90 cm year⁻¹, with a slight increasing trend. From 1979 to 2009, wet deposition decreased by 0.65 and 5.0 kg ha⁻¹ in nitrate-N and sulfate-S, respectively (Fig. 4). Changes in nitrate-N were minimal throughout the years. A reduction in sulfate-S deposition for the past 30 years was clearly evident. Annual sulfate-S wet deposition averaged 8.4 kg ha⁻¹ from 1979 to 1988 and 7.4 kg ha⁻¹ from 1989 to 1998. The deposition then decreased substantially and started to stabilize for the next 11 years (5.1 kg ha⁻¹ from 1999 to 2003 and 5.0 kg ha⁻¹ from 2004 to 2009).

There was an increasing trend in exchangeable base cations in both soils at each sampled depth, with the larger increase as a

percent of 1986 values in the upland soil, which had much lower exchangeable Ca and Mg concentrations in 1986 than the bottomland soil. However, most of the differences were not significant in the upland soil because of a large amount of variability. A power analysis with an $\alpha = 0.05$ and power of 0.9 indicated that, for the upland soil-exchangeable cations, four to 38 samples would be needed to detect a significant difference, given the mean differences and S.D. measured. However, if S.D. decreased with increasing sample size, the number of samples needed would decrease. Power analysis was also used to determine that, if upland soil-exchangeable cation variability remained the same (coefficients of variability were typically 25 to 60% for Ca, Mg, K, and Al), we would need means to be several times different from each other to detect a significant difference.

For the entire profile (0–80 cm), there was a 458-keq ha⁻¹ increase in base cations for the bottomland soils (a 14% increase from 1986 to 2009), and a 540-keq ha⁻¹ in the upland soils (an 85% increase from 1986). For the upland site, there was also a reduction in exchangeable Al and H, with a corresponding large increase in base saturation, although again, because of soil variability, the differences were not significant. Sulfate deposition decreased by a total of 3.1 keq ha⁻¹ during the 23 years, so it likely had little effect on base cation pools. Given the relatively small

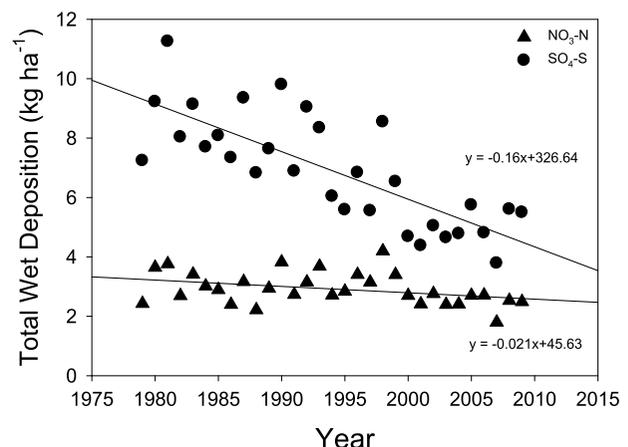


FIG. 4. Atmospheric wet deposition of sulfate and nitrate at the Bondville NADP site from 1979 to 2009.

increase in base cations (as a percent of the total mass) in the bottomland soils and high variability in the upland soils, our interpretation is that the responses measured were likely caused by soil variability. The small reduction in sulfate deposition, little change in stand composition, and lack of other site changes suggest, as the organic matter data showed, that no real change has occurred in forest soil chemistry in these soils.

Soils in east-central Illinois formed in loess over till parent materials, and the till typically has carbonates at depth that explains the high base status. In 1983, the Russell soil profile was described as part of the official USDA soil survey about 200 m north of our site, and characterization data showed 3% carbonates at 1.5 m (USDA, 2013). Therefore, for the upland site, tree roots growing deep into the profile (particularly oaks, which can accumulate Ca) can access this Ca. Pathways such as Ca uptake by trees and allocation to biomass pools have been shown to be the main causes for changes in Ca concentrations in surface soils (Alban, 1982; Johnson and Todd, 1987, 1990; Eriksson and Rosen, 1994; Dijkstra and Smits, 2002). In addition, oaks are known to take up large amounts of Ca that is then stored in above-ground biomass (Weaver and Jones, 1987), some of which is returned to the soil through litterfall. It is possible that canopy level species variation in the upland site led to the variation in soil chemistry that we observed. Qiao et al. (2014) demonstrated that variation in litter inputs affects soil nutrients, and Finzi et al. (1998) concluded that individual tree species in a mixed species stand alters surface mineral soil cation concentrations. Cross and Perakis (2011) concluded that, for organically cycled elements (i.e., C, N, S, P), tree species–soil relationships were observed in old-growth forests of Oregon, but that for weatherable cations (i.e., Ca, Mg, and K), there were no species-level differences. Therefore, the literature supports the concept of species-level effects on soil properties through litterfall, but it seems to be quite site dependent. During soil sampling in 1986 and 2009, we did not note the canopy directly above the sample pits, so we cannot relate the chemistry to individual species. For the bottomland site, exchangeable Ca and Mg in eroded sediments was likely the major source of these base cations, with periodic sediment inputs keeping the base status high and relatively unchanged with depth.

Mean concentrations of total C, N, and S measured in soil samples collected in 2009 were found to be significantly different in some sampling depths than soil samples collected in 1986 at both sites. However, these differences observed during the past 23 years were relatively minimal, which indicated that the total organic matter in the upland and bottomland soils had not substantially changed through time.

CONCLUSIONS

The floodplain soil received substantial inputs of sediment from the agricultural watershed that were rich in organic matter. Therefore, this led to the entire soil profile having large concentrations of C, N, and S that were relatively unchanged during the study period. There were significant increases in base cations in the bottomland soils, although they were small on a percentage basis. In the upland soils, there was little change in soil organic matter. Base cation concentrations in the upland soils were much larger in 2009 compared with 1986, although because of soil variability, most differences were not significant. The trees at the upland site can access base cations from carbonates in the till (upland soil), whereas trees at the bottomland site can access base cations from inputs of eroded sediments and flood waters. At both sites, these processes likely maintained base cation status that led to few changes through time. Variability in upland soil chemistry may have been caused by canopy-level species variation

that affected soils through differences in litterfall chemistry, although further study is needed to investigate this.

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