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Site-specific Critical Acid Load Estimates for Forest Soils in the Osborn Creek Watershed, Michigan

Trevor Hobbs, Jason Lynch, and Randy Kolka



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Abstract

Anthropogenic acid deposition has the potential to accelerate leaching of soil cations, and in turn, deplete nutrients essential to forest vegetation. The critical load concept, employing a simple mass balance (SMB) approach, is often used to model this process. In an evaluation under the U.S. Forest Service Watershed Condition Framework program, soils in all 6th level watersheds on the Huron-Manistee National Forests (HMNF) in Michigan were assigned the lowest score of “3—Impaired Function” due to exceedance of the critical load of acidity as determined by national-scale estimates. The impetus for this research was to test the relevance of national-scale critical acid load estimates at the 6th level watershed scale by using site-specific field data in the SMB model where possible.

The Osborn Creek watershed on the HMNF served as a case study. Field data were collected to estimate soil mineral weathering rates, nutrient uptake rates, and forest growth characteristics at five sites containing sandy, nutrient-poor soils. Critical acid loads and exceedances were developed under “best” and “worst” case scenarios given the uncertainty in the SMB model. Despite the high likelihood of actual exceedance and some evidence for soil acidification across the watershed, base saturation remains excessively high (>100 percent) at most sites. Other field data suggest that these soils receive significant external inputs of base cations that may outweigh what is produced through weathering onsite within the rooting zone. Trees show no visible signs of decline.

Overall, the SMB approach may not adequately capture the complexity of nutrient cycling at all of the sample sites. The variability of soils, weathering estimates, and nutrient uptake rates between and within sites makes extrapolation of these results to other HMNF watersheds difficult to justify. Management programs aimed at improving our understanding of base cycling in complex glacial terrain, as well as mitigating the risks associated with nutrient depletion from frequent timber harvests and fuels reduction practices, are suggested.

Cover Art

A stylized fisheye photo looking up at the canopy of a sugar maple stand in the Osborn Creek watershed.
Photo by Trevor Hobbs, Huron-Manistee National Forests.

Quality Assurance

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The Authors

TREVOR HOBBS is a soil scientist with the Huron-Manistee National Forests in Cadillac, MI.

JASON LYNCH is an ecologist with the U.S. Environmental Protection Agency, Office of Air and Radiation, Office of Atmospheric Programs in Washington, DC.

RANDY KOLKA is a research soil scientist with the U.S. Forest Service, Northern Research Station in Grand Rapids, MN.

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INTRODUCTION

Soils play a crucial role in maintaining healthy forest watersheds by providing substrate for forest growth and acting as a filter for precipitation entering lakes, streams, and groundwater aquifers. Acid deposition, primarily derived from nitrogen and sulfur compounds released into the atmosphere via combustion (automobile and industrial exhaust), and deposited via precipitation and dry deposition (gases, particles, and aerosols), has the potential to disrupt normal soil function. Acid deposition can leach soil cations, and in turn, deplete essential nutrients for healthy vegetation growth, affect the chemistry of nearby lakes and streams, and create toxic conditions for aquatic biota.

Watershed Condition Framework

The U.S. Forest Service classifies the health of watersheds according to the Watershed Condition Framework (WCF). The WCF is a nationwide U.S. Forest Service management program intended to establish a consistent, easy-to-compare, and credible process for evaluating and improving the health of watersheds on national forests and grasslands (Potyondy and Geier 2011). Under the WCF, each 6th level HUC (hydrologic unit code) watershed is assigned a score indicating its overall health. The score is based on 12 environmental attributes, such as water quality, soils, ozone, and riparian vegetation. The WCF score is used to prioritize watershed restoration efforts and track the progress of watershed improvements.

All watersheds in the Huron-Manistee National Forests (HMNF; located in the northern Lower Peninsula of Michigan) received the lowest WCF score of “3—Impaired Function” for the soil contamination attribute (Table 1). This attribute was given a score of 3 because of supposed soil contamination due to excess acid deposition. This determination was based on exceedance of critical loads of acidity that were modeled at the national scale (McNulty and Boggs 2010; McNulty et al. 2007, 2013). As applied to soils, the critical load of acidity, hereafter called critical acid load (CAL), may be defined as the highest depositional load that will not cause chemical changes in soil leading to long-term harmful effects on ecosystem structure and function (Nilsson and Grennfelt 1988). To date, national-scale CAL estimates represent the only available CAL estimates for forest managers in Michigan. However, most CAL estimates modeled at the regional or national scale rely on broad assumptions and aggregation of coarse-resolution datasets, which may or may not accurately reflect actual soil

Table 1.—Excerpt from Condition Rating Rule Set for soil condition indicator in USDA Watershed Condition Framework Technical Guide (adapted from Potyondy and Geier 2011: 29)^a

Rating	Description of soil contamination
Good (1)— Functioning Properly	No substantial areas of soil contamination in the watershed exist. When atmospheric deposition is a source of contamination, sulfur and/or nitrogen deposition is more than 10 percent below the terrestrial critical load.
Fair (2)— Functioning at Risk	Limited areas of soil contamination may be present, but they do not have a substantial effect on overall soil quality. When atmospheric deposition is a source of contamination, sulfur and/or nitrogen deposition is 0–10 percent below the terrestrial critical load.
Poor (3)— Impaired Function	Extensive areas of soil contamination may be present. When atmospheric deposition is a source of contamination, sulfur and/or nitrogen deposition is above the terrestrial critical load.

^aOther soil attributes considered in the overall soil score include soil productivity and soil erosion (not reproduced here).

conditions. As such, the impetus for this research was to develop site-specific CAL estimates using as many empirical data as possible. A 6th level HUC watershed on the HMNF, the Osborn Creek watershed, was used as a case study.

Research Questions

The following questions drive this research:

- What is the critical acid load for forest soils in the Osborn Creek watershed according to site-specific data analysis?
- Has the CAL been exceeded? If so, by how much?
- What management strategies can the U.S. Forest Service implement to monitor and mitigate forest soil contamination due to excess acid deposition?

Objectives

With these research questions in mind, the following objectives were developed to guide this project:

- 1) Deconstruct the process of deriving CAL estimates by describing key assumptions and sources of input data for each model parameter.
- 2) Develop a range of CAL estimates for forest soils in the Osborn Creek watershed using site-specific field data to the extent possible.
- 3) Compare the resulting site-specific estimates to national-scale modeled estimates for this watershed.
- 4) Determine if the WCF score of “3—Impaired Function” for the soil contamination attribute is appropriate for the Osborn Creek watershed considering actual soil conditions.
- 5) Discuss the relevance of results in light of model uncertainties and suggest management considerations for maintaining soil nutrients.

BACKGROUND AND MODELING APPROACH

The topic of critical loads has a long history and wide literature base. For brevity, this report includes a modest amount of background information on the topic, with emphasis placed on new literature relevant to the ecosystems and soils found in the study area and literature published by the U.S. Forest Service. Most of the additional guidance comes from the UBA Critical Load Mapping Manual (UBA [Federal Environmental Agency] 2004).

Modeling Critical Acid Loads for Forest Soils

The critical load concept is a well-established method for determining environmental thresholds of anthropogenic pollutants, such as nitrogen deposition, acid rain, lead, and mercury, for various ecosystem compartments (e.g., lakes, streams, forests, and soils) (UBA 2004). Federal land management agencies in the United States use critical loads as a tool to identify vulnerable ecosystem compartments and protect them from deleterious effects of anthropogenic pollutants (Burns 2011, Fenn et al. 2011, Porter et al. 2005). In the Midwest, nitrogen (N) and sulfur (S) compounds released into the atmosphere via combustion, as well

as from agricultural fertilizer production and use and from agricultural waste, are the main contributors to anthropogenic soil acidification from deposition (Alewell 2003, Driscoll et al. 2001). Soil base cations (calcium, magnesium, potassium, and sodium), mainly derived via soil mineral weathering along with base cation deposition, are the primary buffering agent protecting soils against excessive acidic atmospheric deposition (Sverdrup and Warfvinge 1993). When long-term acid deposition exceeds soil buffering capacity, the critical acid load (CAL) is exceeded. Critical load exceedance indicates an increased risk of damage to the ecosystem. Some consequences of increased soil acidity are leaching of essential soil nutrients (Schaberg et al. 2010), and aluminum toxicity (Cronan and Grigal 1995), both of which reduce forest growth (Ouimet et al. 2001) and acidify surface waters (McDonnell et al. 2012, Siemion et al. 2014).

Simple Mass Balance Method

Soil CALs are difficult to determine because their components are often hard to measure directly, causing values to vary depending on the approach and availability of environmental data (Pardo 2010, Rodríguez-Lado et al. 2007). The most common approach is the simple mass balance (SMB) model (also referred to as the “steady state method”) (Sverdrup and De Vries 1993). The SMB model considers inputs from atmospheric deposition and base cation weathering and outputs from leaching loss and biomass removal. If outputs exceed inputs, there will be a net loss of base cations from the soil, leading to increased acidification. This approach is often used for setting CALs on the national to regional level because it can be a first step to understanding impacts of soil acidification from deposition and because data are often not available at the site-specific scale (Nasr et al. 2010, Ouimet et al. 2006, Whitfield et al. 2010). By substituting as many site-specific data in the SMB equation as possible, the SMB model can potentially be made more accurate and reliable (Bosman et al. 2001, Freer-Smith and Kennedy 2003, Whitfield et al. 2011).

Simple Mass Balance Model

The simple mass balance equation is:

$$CAL(S+N) = BC_{dep} - Cl_{dep} + BC_w - BC_u + N_i + N_u + N_{de} - ANC_{le,crit}, \quad (1)$$

Where

$CAL(S+N)$ = the forest soil critical acid load for sulfur (S) and nitrogen (N),

BC_{dep} = deposition of base cations: calcium (Ca^{2+}), magnesium (Mg^{2+}), potassium (K^+), and sodium (Na^+),

Cl_{dep} = chloride deposition,

BC_w = base cation weathering,

BC_u = uptake of base cations (Ca^{2+} , Mg^{2+} , K^+) by trees,

N_i = nitrogen immobilization,

N_u = uptake of nitrogen by trees,

N_{de} = denitrification, and

$ANC_{le,crit}$ = the acceptable forest soil acid-neutralizing capacity leaching rate.

The critical acid load exceedance value is:

$$Ex(S+N)_{dep} = (S+N)_{dep} - CAL(S+N), \quad (2)$$

Where

$(S+N)_{dep}$ = total sulfur (S) and nitrogen (N) deposition.

In the SMB equation (eqn. 1) and equation 2, all input parameters are expressed in equivalents per hectare per year (eq/ha/yr). (Please refer to the conversion factors on p. 40 to convert units from metric to English.)

N-Nutrient. In this study, a threshold for N-nutrient will not be considered when determining the total CAL. Easily applicable indicators that demonstrate ecosystem shifts caused by excess N-nutrient have not been developed for Michigan forests, nor has a consistent set of data on N dynamics emerged in the local forest ecology literature that could be reasonably adopted for this study. Furthermore, McNulty et al. (2007, 2013) and McNulty and Boggs (2010) did not assess the N-nutrient critical load in their national-scale estimates. Although data on N deposition will be obtained to represent certain parameters of the SMB model, the focus of this study is on the capacity for forest soils in the study area to neutralize anthropogenic acidic inputs, rather than on defining limits that prevent impacts of eutrophication.

Explanation and Comparison of Data Inputs Used in the Simple Mass Balance Equation

The following sections provide a synopsis of each input parameter used in the SMB equation. The purpose of this review is to deconstruct the model inputs, and identify sources of input data, assumptions, and inherent limitations of the SMB model. At the end of each section, the methods and data sources used to derive an input value for that parameter are described. It should be emphasized that the SMB model is a simplification of reality. A complete understanding of the complexity of acidification status in forest soils is almost never practical. Some input data will inevitably need to be modeled or assumed from the literature. Data obtained in this study potentially could lead to a different (and more accurate) CAL determination than national-scale estimates for soils in the Osborn Creek watershed.

Atmospheric Deposition of Sulfur+Nitrogen, Base Cations, and Chlorine. Atmospheric deposition of ions to soils occurs via wet deposition (i.e., rain and snow), dry deposition (i.e., particulates and gases), and cloud/fog inputs. Acidifying compounds such as nitrate (NO_3^-), nitric acid (HNO_3), ammonium (NH_4^+), sulfur dioxide (SO_2), and sulfate (SO_4^{2-}), as well as the base cations calcium (Ca^{2+}), sodium (Na^+), magnesium (Mg^{2+}), and potassium (K^+), and chlorine (Cl), are some of the typical analytes measured in existing monitoring networks. Dry deposition data are more difficult to measure and model than wet deposition data because of the variety of local factors that influence deposition (i.e., elevation, slope, aspect, and canopy structure). Cloud and fog deposition exhibits significant spatial and temporal variation, and is generally excluded as a source of atmospheric deposition in SMB models for low-elevation regions. Acid deposition data are collected regularly at monitoring stations across the United States. Typically, data describing both wet and dry (particulate) forms are available. The two major sources of atmospheric deposition data in the United States are the National Atmospheric Deposition Program (NADP) and the Clean Air Status and Trends Network (CASTNET).

This study uses a new method for estimating total (S+N) deposition developed by the Total Deposition (TDEP) Science Committee and NADP that combines measured data from various monitoring stations with modeled data from the Community Multi-scale Air Quality (CMAQ) model. Monitoring data for the CMAQ model come from several sources: CASTNET, Ammonia Monitoring Network (AMoN), Southeastern Aerosol Research and Characterization (SEARCH), and National Trends Network (NTN). Data from each source span different years and undergo different transformations before being used as input to the CMAQ model. TDEP data combine monitoring data and CMAQ output by using inverse distance weighting interpolation methods, giving higher weight to measured data. The output contains annual total deposition data (S+N, wet and dry) at 4-km² resolution. Further details of the methods used to derive deposition estimates can be found in Schwede and Lear (2014).

Base Cation Weathering. Data accurately describing mineral weathering are necessary if the derived CALs are to be confidently relied upon for answering management questions at the local scale (McDonnell et al. 2010). Basic soil characteristics that influence mineral weathering rates include mineralogy, surface area (related to overall soil texture and the inherent surface roughness of different minerals), moisture content, temperature, carbon dioxide concentration, strength of organic acids, and pH (Sverdrup and Warfvinge 1993). Several methods for calculating mineral weathering have been developed. In general, simpler methods rely on more assumptions than complex methods, and as such, often result in greater uncertainty (Hodson and Langan 1999, Kolka et al. 1996). Common examples used in CAL studies include the soil texture approximation (STA) method (UBA 2004) and the PROFILE model (Warfvinge and Sverdrup 1992).

The STA method has the fewest data requirements, and therefore is the least expensive to implement. Weathering rates are assigned to soils based on a combination of soil texture and mineralogical “classes” (UBA 2004). In contrast, PROFILE is a kinetic soil chemical weathering model that combines transition-state theory, experimentally derived dissolution reaction data, and an independent database of soil geophysical data to develop base cation weathering rates under current deposition (Warfvinge and Sverdrup 1992). PROFILE requires more-detailed soils input data than the STA method. Although the underlying assumptions and accuracy of the PROFILE model have been questioned (Hodson et al. 1997), over time it has become a standard to which other methods are compared. Some studies have evaluated other approaches to estimating mineral weathering and their impacts on critical load estimates (Whitfield et al. 2006, 2011). More recent studies have taken hybrid approaches by combining the STA and PROFILE methods, developing regression models that explain their relationship, and extrapolating those regressions to unmeasured soils (Whitfield and Watmough 2012).

Additional weathering rate calculation methods, such as the zirconium depletion method (Kirkwood and Nesbitt 1991) and the pedological mass balance (PMB) method (Elgi and Fitze 2000), use the relative concentrations of immobile (nonweatherable) elements and labile elements (base cations) in the rooting zone (E- through B-horizons) versus that of the soil parent materials (C-horizon). The PMB method, as revised by Elgi and Fitze (2000), adds another level of accuracy by incorporating net changes (positive and negative) in horizon volume due to weathering. Both of these methods rely on the underlying assumptions that 1) the original soil parent materials were mineralogically homogenous throughout the profile, and 2) the C-horizon represents unweathered soil (i.e., conditions at time zero). Researchers have not yet reached a consensus on the best method to determine mineral weathering.

Table 2.—Soil texture classes as a function of clay and sand content (UBA 2004)

Texture class	Name	Definition
1	Coarse	Clay <18% and sand ≥65%
2	Medium	Clay <35% and sand >15%, but clay ≥18% if sand ≥65%
3	Medium fine	Clay <35% and sand <15%
4	Fine	35% ≤ clay < 60%
5	Very fine	Clay ≥60%

Table 3.—Weathering rate classes, ranging from low to high (1 to 6), as a function of parent material and texture classes (UBA 2004)

Parent material	Texture class				
	1	2	3	4	5
Acidic	1	3	3	6	6
Intermediate	2	4	4	6	6
Basic	2	5	5	6	6
Organic	Class 6 for Oe and class 1 for other organic soils				

Each method presents advantages and disadvantages depending on how well underlying assumptions fit with actual soil conditions in the region under study.

The national-scale estimates to which site-specific CALs derived in this study are compared, use the STA method. The derivation of base cation weathering rate from the STA method involves first categorizing the soil into a basic texture class (Table 2) and mineral weathering rate class (Table 3) (UBA 2004). After soils are categorized, the weathering rate is derived as follows:

$$BC_w = z \times 500 \times (WR_c - 0.5) \times \exp\left(\frac{A}{281} - \frac{A}{273 + T}\right), \quad (3)$$

Where

z = soil depth (m),

WR_c = weathering rate class (Table 3),

T = average annual soil temperature ($^{\circ}\text{C}$), and

A = 3600 (K).

Equation 3 determines a weathering rate for Ca+Mg+K+Na in eq/ha/yr.

This study used the PMB method, as revised by Elgi and Fitze (2000). The PMB method has been successfully applied in long-term weathering studies of glacially derived silicate parent materials (Lichter 1998, Olsson and Melkerud 2000, Schroth et al. 2007). Input data for the PMB method include bulk density, elemental geochemistry, and horizon thickness throughout the soil profile. Long-term weathering rates are calculated as follows:

$$m_{j,flux}(z_w) = \sum_{a=1}^n C_{j,p} \rho_p \left(\frac{1}{\epsilon_{i,w} + 1} \right) \tau_{j,w} \Delta z_w \quad (4)$$

$$\epsilon_{i,w} = \left(\frac{\rho_p C_{i,p}}{\rho_w C_{i,w}} \right) - 1 \quad (5)$$

$$\tau_{j,w} = \left(\frac{C_{j,w} C_{i,p}}{C_{j,p} C_{i,w}} \right) - 1, \quad (6)$$

Where

$m_{j,flux}$ = change in mass of element j (in this case, each of the major base cations) (kg/m²),

z_w = weathered horizon,

$C_{j,p}$ = concentration of labile element j in the parent material (kg/tonne),

$C_{i,p}$ = concentration of immobile element i in the parent material (kg/tonne),

$C_{j,w}$ = concentration of labile element j in the weathered horizon,

$C_{i,w}$ = concentration of immobile element i in the weathered horizon,

ρ_p = bulk density of the parent material (tonne/m³),

ρ_w = bulk density of the weathered horizon (tonne/m³),

$\epsilon_{i,w}$ = strain (volumetric changes due to pedogenic weathering with respect to the volume of the initial material),

$\tau_{j,w}$ = the open-system mass transport function for element j , and

Δz_w = the depth of the weathered horizon (m).

In this study, titanium (Ti) is used as the immobile element i , and Ca, K, Mg, and Na are used as the labile elements j . The open-system mass transport function ($\tau_{j,w}$) (eqn. 6) for each of the major base cations (Ca^{2+} , K^+ , Mg^{2+} , and Na^+) is calculated based upon their concentrations relative to the immobile element in the weathered horizon versus the parent materials. The strain calculation (eqn. 5) is incorporated into the mass flux equation to determine net gains or losses of each labile element j in the weathered horizons. Mass flux calculations for each horizon are then summed throughout the profile. See Theoretical Background section in Elgi and Fitze (2000) for further explanation of terms.

N Immobilization. In the SMB model, N immobilization (N_i) represents the long-term net retention of N in the soil root zone (under ambient N deposition) (UBA 2004). Nitrogen immobilization accounts for the amount of inorganic nitrogen converted to organic nitrogen and retained in soil N pools, primarily by microbes (aerobes, anaerobes, fungi, and bacteria). Nitrogen immobilization is counterbalanced by N mineralization, which is the release of inorganic forms of nitrogen as a by-product of detritus consumption by soil microbes (Paul 2007). In N-limited environments (such as upland forest ecosystems), the amount of N immobilized depends on the amount and type of leaf litter, the productivity and efficiency of soil microbes, and, ultimately, soil temperature and soil moisture (with maximum N immobilization occurring at moderate levels of both moisture and temperature). Higher values for N_i would result in higher critical load estimates (the ecosystem is able to retain more N), whereas lower N_i values would result in lower critical load estimates (the ecosystem is able to retain less N). There is little consensus regarding acceptable N_i values for critical load assessments of forest soils, primarily because they are difficult to determine with certainty.

Table 4.—Examples of commonly used values for nitrogen (N) immobilization in critical load studies that employed the simple mass balance approach

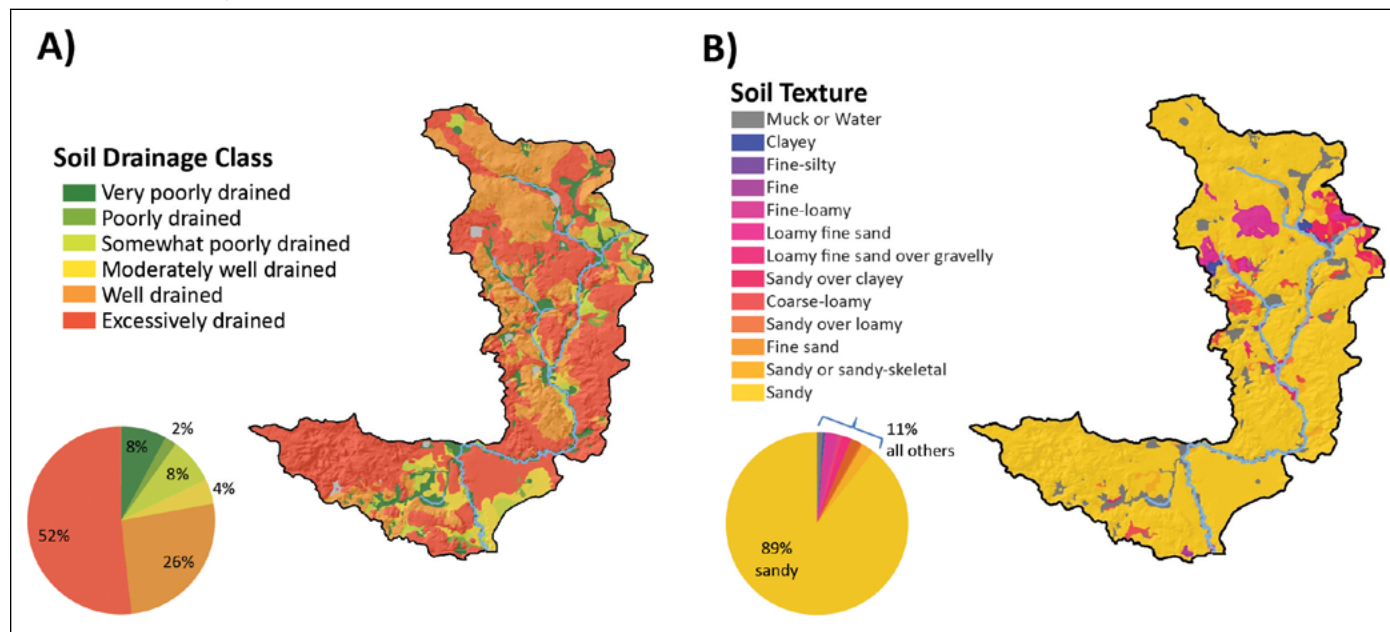
Value for N (kg/ha/yr)	Justification	Reference
0.2–0.5	Chronosequence study of Norwegian soils; recognized that values may be an underestimation.	Rosen et al. (1992)
0.0	Lack of site-specific data; assumption that production of N in humus is compensated for by N fixation.	Arp et al. (1996)
2.0	Lower end of the range suggested by Downing and others (1993).	Posch (1997)
0.0	Lack of data to suggest otherwise; study area characterized by disturbance processes (e.g., tree throw, fire) that would eventually export N from soils.	Oiomet et al. (2001)
2.0	Assumed, though reference was made to much higher N immobilization rates used in Europe (Posch et al. 2001) as justification as a precautionary value.	Watmough and Dillon (2003)
0.0 and 8.8	To represent the worst- and best-case scenarios: 0.0 = all N leached, 8.8 = all N retained in ecosystem, respectively.	Whitfield et al. (2010)

Most forest ecosystems where anthropogenic acidification is a concern are currently accommodating and adjusting to elevated levels of N deposition, making it impossible to measure in situ N retention under “long-term” (i.e., natural) conditions. Many studies rely on previously published or recommended values, most of which originate from Rosen et al. (1992), with minor adjustments and supporting justifications or assumptions (Table 4). Rosen et al. (1992) measured stable carbon-nitrogen compounds in a variety of soil profiles in Norway, and derived average N immobilization rates by dividing the N content in soils by landform age (i.e., the chronosequence approach). This approach is possible for landscapes in which the deglacial chronology is reasonably well constrained by absolute dates. Values for N ranging from 0.2 to 0.5 kg/ha/yr (14.3 to 35.7 eq/ha/yr) were reported (Rosen et al. 1992). These values have been adopted for use in other critical load studies of formerly glaciated environments dominated by Spodosols, and are suggested as the range of N immobilization for northern climates in the UBA mapping manual (UBA 2004). Other studies have used different values for N_i with a variety of supporting justifications (Table 4).

In this study, an approach similar to Whitfield et al. (2010) was used, primarily because there are no locally derived data that can be used as a meaningful substitute for long-term sustainable N immobilization in the study area. This approach also allowed the sensitivity of the N_i parameter to be tested under different deposition scenarios, and therefore allowed for more meaningful comparison of overall results to national-scale estimates.

Denitrification. In the SMB model, denitrification (N_{de}) represents the long-term loss of N from the soil to the atmosphere via reduction of nitrate (conversion to N gases) (UBA 2004). Generally, denitrification caused by denitrifying bacteria occurs in soils when pore space saturation is greater than 60 percent (Paul 2007). Consequently, denitrification is

Figure 1.—Distribution, by location and proportion of the total, of A) soil drainage class and B) surface soil texture in the Osborn Creek watershed (Soil Survey Staff 1996).



more significant in wet soils than dry soils. In areas where soil acidification is a concern, denitrification can be an important mitigating process as it prevents excess NO_3^- from entering groundwater (Wexler et al. 2014). However, in many such areas (e.g., well-drained, sandy upland forest soils), denitrification is often negligible, if not absent. Only in isolated pockets of poorly drained soils, in wetland soils, and along riparian areas would denitrification occur year-round (McLaughlin et al. 2011). Denitrification may play a minor role on other sites that experience periodic pulses of increased saturation throughout the year (i.e., during spring snowmelt, or because of a fluctuating water table). Higher values for N_{de} would result in higher critical load estimates (the ecosystem is able to process and release more N back to the atmosphere), whereas lower N_{de} values would result in lower critical load estimates (the ecosystem is less able to process and release N back to the atmosphere).

Most SMB critical load studies on well-drained upland forest sites assume that denitrification is insignificant, and therefore N_{de} values are set to zero (Aherne 2008, Pardo and Driscoll 1996, Warfvinge et al. 1993, Watmough et al. 2006). Zero values for N_{de} are a precautionary approach to setting critical loads where no local data exist. In an SMB study of British Columbia soils, the availability of local data permitted Mongeon et al. (2010) to set N_{de} values to 0.035 kg/ha/yr. A meta-analysis of reported N_{de} values (from 25 studies) in forest soils by Barton et al. (1999) found that more than half of studies reported denitrification rates of <1 kg N/ha/yr.

For this study, N_{de} values are assumed to be zero because the majority (78 percent) of soils in the watershed are excessively or well drained (Fig. 1). Furthermore, most of the U.S. Forest Service land in the watershed is occupied by red pine (*Pinus resinosa*; i.e., conifers). Therefore, considering the absence of site-specific measurements, and the fact that reported denitrification values for coniferous stands are generally <0.1 kg/ha/yr, a zero value for N_{de} is conservative and reasonable.

Nitrogen Uptake and Base Cation Uptake. In the SMB model, N_u accounts for the amount of nitrogen that trees extract from the soil for growth. Similarly, BC_u accounts for the quantity of base cations extracted from the soil for tree growth. When trees are harvested, both N_u and BC_u represent long-term average removals of N and base cations from the ecosystem (UBA 2004). The values are highly dependent on stand rotation period (Vadeboncoeur et al. 2014), stocking density, and harvesting practices (i.e., proportion of trees removed vs. left behind onsite) (Hazlett et al. 2014). Higher values for N_u effectively increase the CAL estimate; higher values for BC_u effectively decrease the CAL estimate. In other words, N contributes to soil acidity, whereas base cations neutralize soil acidity. Values for N_u and BC_u can be derived via site-specific measurements or published values.

The following equations were used in this study to calculate N_u and BC_u (collectively termed “uptake”):

$$Uptake_{bark} = Bark\ Volume \times NC_{bark} \times SG \quad (7)$$

$$Uptake_{bole} = Bole\ Volume \times NC_{bole} \times SG, \quad (8)$$

Where

NC = nutrient concentration: nitrogen and base cation concentration in bark or bole, and

SG = specific gravity (density) of bark or bole wood (g/cm^3).

The following sources of data were used to derive values for input into equation 7:

- 1) Species-specific volumetric equations for trees in the Lake States (Miles and Hill 2010) were used to calculate total tree volume and volume of bark on a tree-by-tree basis, assuming stem-only harvest.
- 2) Base cation and nitrogen concentrations for bark and bole of individual species were calculated based on samples collected from the field.
- 3) Specific gravity values for bark and bole of individual species were calculated based on samples collected from the field.

In the SMB model, BC and N uptake rates represent nutrient loss via harvest over a typical rotation period. In this study, the BC and N uptake rates assume that trees are harvested at present, rather than specifying the length of a rotation period. In addition, species-specific formulas used to estimate bark and bole volume include terms that compensate for volume left behind according to typical harvest methods for that species. This approach was considered more appropriate than extrapolating BC and N concentration data forward to a typical rotation period for two reasons: 1) At best, rotation periods can only be estimated—often with a 20-year buffer—because, in practice, forest management activities are executed within a complex framework of overlapping priorities, socio-economics, and policy, which collectively determine when stands are actually harvested; and 2) many of the stands in this study are close to or at their intended age of harvest.

Critical Leaching Rate of Forest Soil Acid Neutralizing Capacity. In the SMB equation (eqn. 1), $ANC_{le,crit}$ represents the highest acceptable leaching rate of soil acid neutralizing capacity. This parameter of the SMB model determines the critical chemical criterion for adverse impacts to a receptor in the environmental compartment of concern, that is, damage to fine roots

Table 5.—Ranges of the gibbsite equilibrium constant (K_{gibb}) as a function of soil organic matter content (UBA 2004)

Soil type	Organic matter (%)	K_{gibb} (m ⁶ /eq ²)
Mineral soils; C-horizon soils with low organic matter	<5	950–9,500
Soils with low organic matter; B- and C-horizons	5–15	300–3,000
Soils with some organic matter; A- and E-horizons	15–30	100
Peaty and organic soils; organic layers	>70	9.5

(receptor) in the soil (compartment). This parameter can be modeled in several different ways, so care must be taken in setting chemical criteria that are appropriate for the environment and receptor of concern. Choosing the right input values depends on whether the receptor is more sensitive to acidity (pH) or the toxic effects of high aluminum (Al) concentrations (UBA 2004). In upland forest ecosystems with low soil organic matter content, the relationship between available base cations in soil (soil nutrients) and levels of monomeric Al (which can be toxic to fine roots) is considered the most appropriate criterion to model. With these terms in mind, the following equation is used:

$$ANC_{(le,crit)} = -Q^{\frac{2}{3}} \times \left(1.5 \times \frac{BC_{dep} + BC_w - BC_u}{K_{gibb} \times \left(\frac{BC}{Al}\right)_{crit}} \right)^{\frac{1}{3}} - 1.5 \times \frac{BC_{dep} + BC_w - BC_u}{\left(\frac{BC}{Al}\right)_{crit}}, \quad (9)$$

Where

Q = precipitation surplus (in m³/ha/yr),

BC_{dep} = base cation deposition,

BC_w = base cation weathering,

BC_u = base cation uptake,

K_{gibb} = the gibbsite equilibrium constant, and

$\left(\frac{BC}{Al}\right)_{crit}$ = critical base cation-to-aluminum ratio.

The gibbsite equilibrium constant describes the relationship between hydrogen (H) and Al ions in solution. Common input values for K_{gibb} (Table 5) are based primarily on data from an intensive forest monitoring study in Europe (De Vries et al. 2003), and are widely used in regional and national-scale CAL estimates. The critical base cation-to-aluminum ratio ($BC:Al_{crit}$) is commonly set to 1 for coniferous forests and 10 for deciduous forests (UBA 2004). These BC:Al values are the international standard considered to acceptably protect tree roots from Al toxicity.

Instead of using the default K_{gibb} constant in this study, we use a version of the $ANC_{le,crit}$ equation that incorporates local soils data to describe the relationship between H ions and

Table 6.—Summary of simple mass balance input data used in this study

Model parameter ^a			This study ^b	Data source	Scale of input data ^c	
Deposition data		$(S+N)_{dep}$ (wet and dry)	MO	Wet+dry: CMAQ (U.S. EPA 2014) Years: 2000–2012	4 km ²	
		BC_{dep}				
		Cl_{dep}				
Mineral weathering		BC_w	T	ME	Field	Δ
			M	ME	Lab	Δ
Base cation uptake		BC_u	C	ME	Field/Lab	Δ
			SG	ME	Field/Lab	Δ
Nitrogen immobilization		N_i	A/MO	Wet+dry: CMAQ (U.S. EPA 2014) Years: 2000–2012	4 km ²	
Nitrogen uptake		N_u	C	ME	Field/Lab	Δ
			SG	ME	Field/Lab	Δ
Denitrification		N_{de}	A	--	NA	
Leaching of acid neutralizing capacity, $ANC_{le,crit}$	Discharge	Q	MO	(Michigan State University Extension 2014) Years: 1981–2010	Δ	
	pH/pAl equilibrium constant	K_{alox}	MO	(NCSS, n.d.)	NA	
	Critical base cation-to-aluminum ratio	$\left(\frac{BC}{Al}\right)_{crit}$	A	(UBA 2004)	NA	

^a Abbreviations: S = sulfur, N = nitrogen, BC = base cation, Cl = chlorine, T= soil texture, M= soil mineralogy, C = content, SG = specific gravity, Al = aluminum.

^b Abbreviations: MO = modeled, A = assumed, ME = measured.

^c Abbreviations: Δ = site-specific, NA = not applicable.

exchangeable Al in soil solution. This option more adequately reflects site-specific leaching processes in the soils under investigation (UBA 2004). The following equation was used:

$$[Al] = K_{Alox} \times [H]^a, \quad (10)$$

Where

Al = exchangeable aluminum (in eq/m³)

H = hydrogen ions in solution (in eq/m³).

K_{alox} (the equilibrium constant) and exponent a were derived by plotting the logarithmic forms of these data and inserting a power trendline (in the form of $Y = K \times X^a$). These data were used instead of the K_{gibb} equilibrium constant in equation 9.

Input Data for the Simple Mass Balance Equation

Table 6 summarizes the SMB input data used in this study. Two input parameters were modeled based on actual data, four were assumed based on values reported in the literature, and six—including three of the most influential input parameters (BC_w , BC_u , and N_u)—were measured at the plot level (see the Methods section for more details). State-of-the-science



Figure 2.—Location of Osborn Creek watershed (black outline) within the greater White River watershed (light gray shading). Approximate centroid (latitude/longitude) of Osborn Creek watershed: 43.6252° N/–86.2083° W.

analysis tools were used to model input data wherever possible. Because we used the same modeling approach as McNulty and others (2007, 2013) and McNulty and Boggs (2010), the results of this work can be directly compared to results from national-scale estimates.

STUDY AREA

Geography

The Osborn Creek watershed is located in the southwestern portion of the Huron-Manistee National Forests, in central Oceana County, MI (Fig. 2). Elevations across the watershed range between 200 and 300 m above sea level. The area of the watershed is about 5,880 ha. The watershed falls within Forest Service Management Area 4.4—Rural (U.S. Forest Service 2012). Rural areas are primarily managed for recreation, firewood, and moderate-to-high volumes of softwood timber products. Only 8 percent of land within the watershed is federally owned. Private land in the watershed forms a patchwork of cover types including cultivated crops, deciduous and evergreen forests, herbaceous cover, and wetlands.

Geology

The surrounding region is characterized by gently rolling glacial hills of moderate relief and broad lowland outwash valleys (Schaetzl et al. 2013). Present landforms and drainage patterns in the study area owe their origin to former glacial ice margin positions and associated morphosequence deposits of the Lake Michigan Lobe (Farrand and Eschman 1974). In general, upland areas are primarily ground moraine or end moraine composed of a range of coarse-textured sediments. Lowland areas are primarily deep glacial outwash deposits composed of well-sorted medium-to-coarse sand (Farrand and Bell 1982, and personal

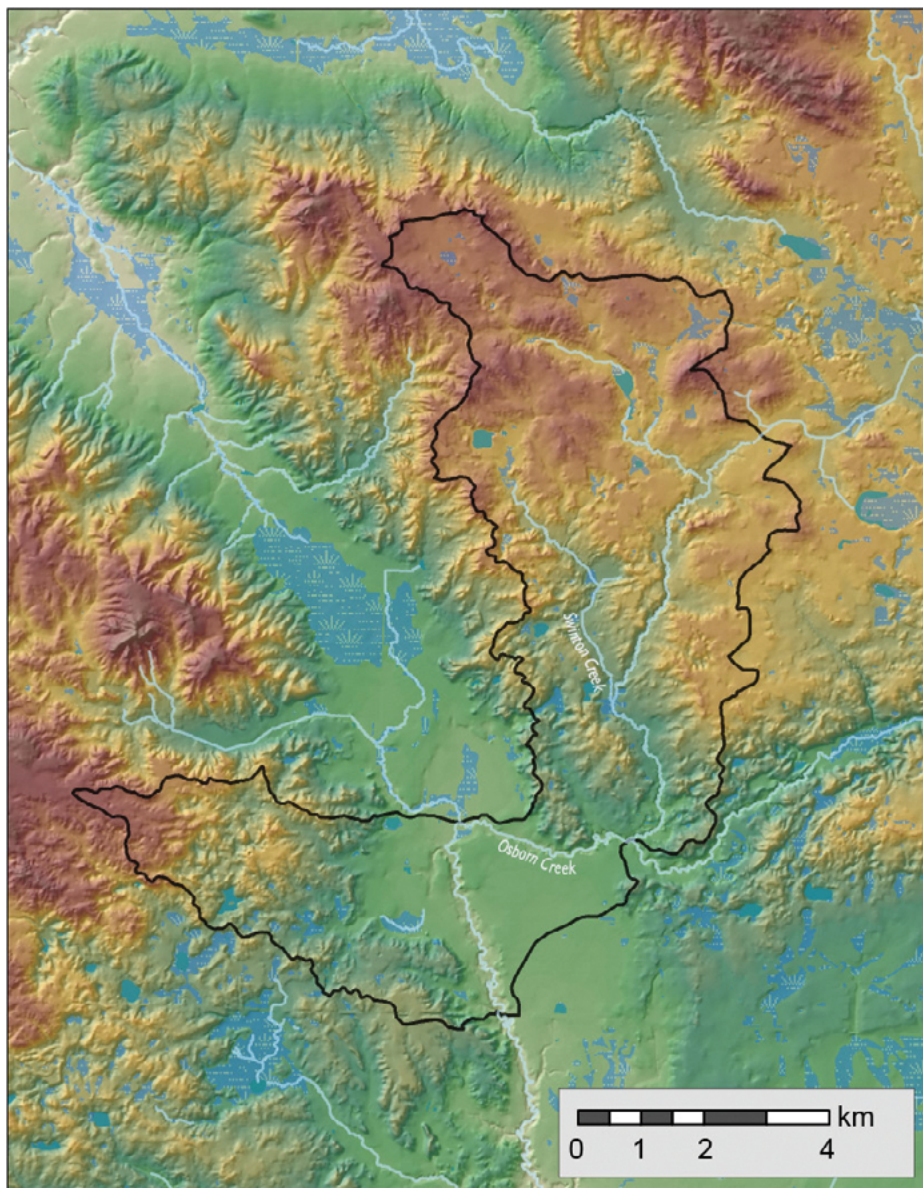


Figure 3.—Digital elevation model (10-m resolution) and transparent hillshade of Osborn Creek watershed (black outline) and surrounding area. Yellow and brown areas represent higher elevations. Blue and pale green areas represent lower elevations. Streams are light blue; wetlands and lakes are dark blue.

observations 2014). Drift thickness ranges between 90 and 150 m in the study area. Road cuts and exposures of the underlying glacial stratigraphy are uncommon. Most geologic interpretation is therefore based on surface morphology and shallow soil boring.

The irregular topography of the watershed is typical of glaciated landscapes in the Great Lakes region (Fig. 3). A variety of ice-contact environments are evident among the landforms present in the watershed including kames, kettles, subglacial drainage networks (discontinuous eskers), and ice-contact outwash. The northern and northeastern parts of the watershed are locally referred to as “overwashed moraine” because clean sand of varying thickness and extent overlies sandy loam diamict at shallow depths; topographic roughness in such areas decreases markedly compared to the surrounding terrain (Fig. 3). Sand composes nearly all of the surficial deposits in the study area, indicating widespread meltwater processes during deglaciation. The lithology of surficial sediments in the study area reflects a wide-ranging mixture of subcropping bedrock units in the Michigan Basin, with greater concentrations of minerals that occur in locally subcropping bedrock. Underlying

bedrock units in the area include the Michigan Formation (primarily composed of shale, with interbedded sandstone, dolostone, limestone, and evaporate) and the Marshall Formation (primarily composed of sandstone, with interbedded siltstone) (Milstein 1987). In general, more-resistant granitic minerals make up a greater proportion of the surficial deposits than less resistant (more weatherable) minerals. Indeed, recent mineralogical investigations by the U.S. Geological Survey (Smith et al. 2013) indicate that upland soils in nearby areas are primarily composed of quartz (~84 percent) with lesser amounts of potassium feldspar (~12 percent) and plagioclase (~4 percent).

Soils and Vegetation

About 78 percent of the watershed is composed of excessively drained and well-drained soils; minor pockets (10 percent) of poorly drained and very poorly drained soils occur in swamps and riparian areas. Surface textures in the watershed are mostly sand (Fig. 1). On forested land within the watershed, cation exchange capacity (at pH 7.0) is estimated to be low, typically ranging between 1 and 4 meq/100 g; pH ranges from 5.0 (strongly acid) to 6.5 (slightly acid) (Soil Survey Staff 1996). The most common soils in the watershed are classified as Typic Udipsamments and Entic Haplorthods. Some of the soils classified as Entic Haplorthods are occasionally found overlying loamy till. Other soils contain loamy sand to sandy loam bands (lamellae or other nonpedogenic layers, or a combination) with total thickness of 5 to 15 cm within 1.56 to 4.5 m of the surface (Host et al. 1993). These substratum layers have been shown to significantly increase available moisture and nutrients for trees (Hannah and Zahner 1970, Schaetzl 1992).

Before European settlement, most of the vegetation in the study area was beech/sugar maple/hemlock (*Fagus* spp./*Acer saccharum*/*Tsuga* spp.) forest (Comer et al. 1995). The southwestern portion of the watershed supported white pine (*Pinus strobus*)/mixed hardwood forest. All of the land in the watershed was most likely cleared of timber at some point in the mid-to-late 1800s. Areas of more fertile soil were put into agricultural production and remain as such to the present day. Most areas of nutrient-poor soils were cleared for agriculture and subsequently abandoned in the early-to-mid 1900s due to insufficient productivity or low profitability. These areas reverted to mixed hardwood regrowth or were planted with red pine throughout the early 20th century. Today, U.S. Forest Service land within the watershed primarily supports red pine, mixed oaks (white oak [*Quercus alba*], black oak [*Q. velutina*], and northern red oak [*Q. rubra*]), bigtooth aspen (*Populus grandidentata*), and minor areas of sugar maple.

Surface Water and Groundwater

Two creeks (Osborn Creek and Swinton Creek) flow through the Osborn Creek watershed (Fig. 3). Osborn Creek and Swinton Creek join the North Branch White River south of the watershed divide. These streams are all considered high-quality cold water systems supporting brook trout (*Salvelinus fontinalis*), Iowa darter (*Ettheostoma exile*), and other species adapted to clean, cool water and a narrow pH range. Wetland communities compose about 10 percent (6.12 km²) of the total watershed area, and occur mostly in riparian zones. Within the watershed are five small lakes, of which four are internally drained seepage lakes (Fig. 3). Groundwater makes up a large portion of total stream flow in the Osborn Creek watershed. Regional models indicate that as much as 80 percent of total stream discharge in the surrounding White River watershed is attributed to groundwater, perhaps representing

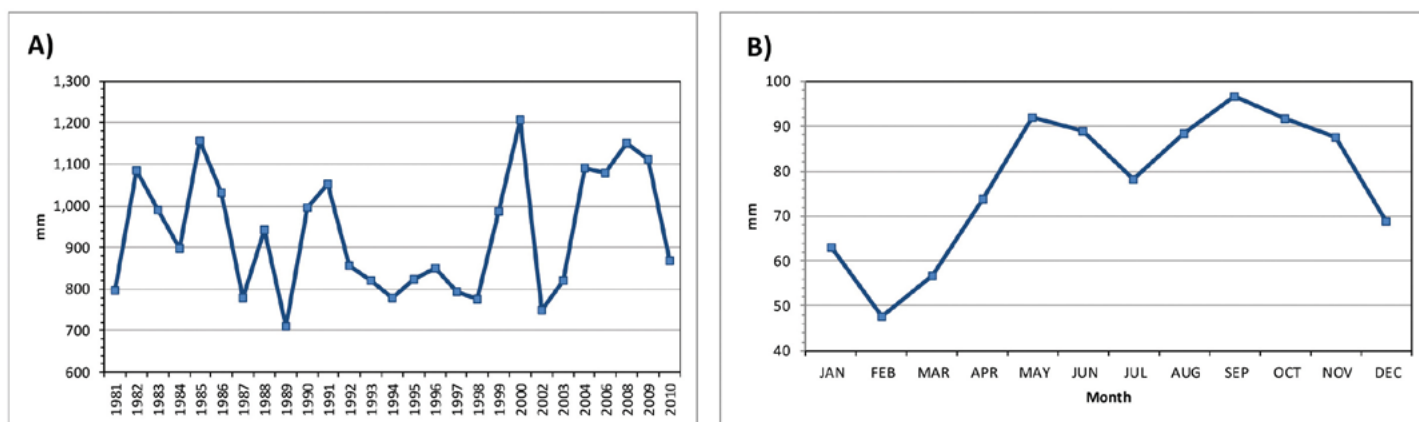


Figure 4.—A) Annual precipitation totals and B) average monthly precipitation at the Hart, MI weather station (located about 18 km from the center of the Osborn Creek watershed), 1981–2010.

the most groundwater-dependent (base flow) fluvial system in the contiguous United States (Wolock 2003). Water table depths vary throughout the watershed, generally mimicking the topography, with typical values of 0 to 3 m in low areas and 24 to 30 m in higher areas. Factors that contribute to consistent groundwater flow include thick coarse-textured glacial drift, highly permeable soils, significant annual snowpack, and precipitation exceeding evapotranspiration in surrounding upland regions.

Climate

The region surrounding the Osborn Creek watershed has a humid warm continental climate with cold winters and warm summers. Proximity to Lake Michigan (~20 km to the west) provides a moderating effect on temperature throughout the year, and increases cloudiness and snowfall in the fall and winter. For this study climate statistics for 1981–2010 were derived from the weather station nearest Osborn Creek (Hart weather station, ID #3632, latitude 43.7366° N/longitude -86.3594° W) (Michigan State University Extension 2014). Average annual precipitation is 765 to 1079 mm (± 1 standard deviation). Precipitation is well distributed throughout the year (Fig. 4). Average annual snowfall is 1128 to 3020 mm (± 1 standard deviation). Mean July temperature is 20.6 °C. Mean January temperature is -5.1 °C.

METHODS

Field Methods

Forest Sampling Sites

Five study sites on U.S. Forest Service land on the Huron-Manistee National Forests (HMNF) were chosen for detailed sampling of vegetation and soils (Fig. 5). Sandy soil types were preferentially chosen for sampling as they are generally considered more susceptible to acidification than finer-textured soils. A secondary consideration was to capture a range of typical forest cover types found on these soils across the watershed. Prior to the fieldwork, the approximate center of each forest stand was estimated on maps and marked with a point. The latitude/longitude coordinates of these points were calculated in ArcMap (Esri, Redlands, CA) and loaded onto handheld global positioning system (GPS) devices (Garmin Oregon® 550t; Garmin International, Olathe, KS) for field navigation. After navigating to the site in

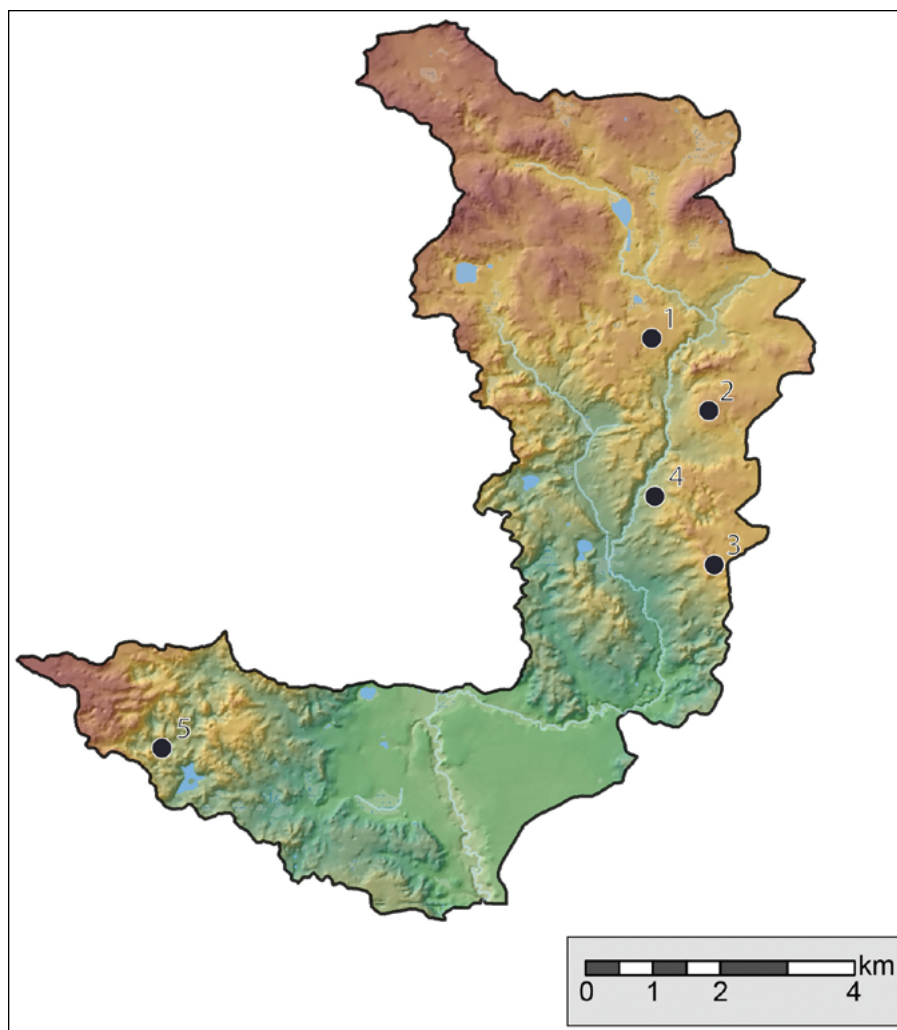


Figure 5.—Location of the five forest soil sample sites in the Osborn Creek watershed.

the field, we adjusted the locations (if necessary) to avoid disturbed ground or stands having more than one dominant overstory species. When the final site location was established, a new GPS point was marked. The center of each site was flagged and a 20 m × 20 m square plot surrounding the center was measured and staked. The plot edges were aligned with the four major cardinal directions. Each plot was further subdivided into four quadrants (northeast, northwest, southeast, and southwest). All plots were at least 90 m from a road.

Vegetative Sampling

The dominant overstory and understory vegetative characteristics for each site were recorded. The number of dominant tree species was counted within each quadrant and summed for the entire plot. Five individual trees—one from each of the four quadrants, and one tree closest to the center of the plot—were selected within each plot for tissue sampling. Mature trees were preferentially chosen for sampling. At each sample tree, diameter at breast height (d.b.h., 1.37 m) and tree height (length of bole and total height) were measured. A bole sample was taken from each of the five trees with a 6.4-mm-diameter increment borer. Trees with rotten cull were not sampled. Bark samples were removed from the same five trees with a 5-cm-diameter hole saw, which was inserted into each tree at breast height.

Soil Sampling

Soil pits were located in the center of each 20 m × 20 m plot. Soil pits were excavated by shovel to a depth of about 120 to 150 cm. Soil profiles were described and samples collected from each horizon according to Natural Resources Conservation Service methods (Schoeneberger et al. 2012). The depth, character, and distribution of roots were recorded. Four bulk density samples were taken at each pit. Bulk density samples were gathered with a sharpened polyvinyl chloride pipe core (diameter: 7.6 cm; length: 9.7 cm) and sledge hammer, according to methods in Blake and Hartge (1986). The surface bulk density sample (A-horizon) was taken vertically (from above) next to the soil pit. The remaining three bulk density samples were carefully extracted horizontally from the pit face at different depths according to the horizon sequence: one sample each in the eluvial (E-) horizon, illuvial (B-) horizon, and C-horizon. All horizon and bulk density samples were transported from the field in waterproof sample bags.

Laboratory Analysis

Vegetation

Bark and bole samples were air dried, measured, and weighed in the geomorphology laboratory at the HMNF. Tree age was determined by visual inspection after taking high-resolution photographs of bole samples with a mounted EF 200mm zoom lens (Canon, Tokyo, Japan) and counting rings between the inner bark and pith. Volume measurements of bark and bole were used along with sample weight to compute specific gravity of the samples. Bole sample volume was calculated using the standard equation for a cylinder. Bark sample volume was calculated by taking the average of 12 evenly spaced thickness measurements (mm) around the vertical edge of the sample and multiplying by the area of the bark sample. This method assumed that bark surface roughness was reasonably accounted for by the average of the 12 thickness measurements. Bark and bole samples were then packaged and shipped to the Cornell Nutrient Analysis Laboratory for geochemical analysis using inductively coupled plasma-atomic emission spectroscopy (ICP-AES), following nitric acid-perchloric acid-hydrofluoric (HNO_3 - HClO_4 -HF) acid digestion. Prior to ICP-AES, remaining bole wood was carefully removed from bark samples as necessary.

Soils

Soil horizon and bulk density samples were weighed immediately upon return to the HMNF geomorphology laboratory. Samples were then air dried for several days and weighed again to calculate field moisture content. The bulk density of horizons that fell between bulk density sample depths was estimated by extrapolating the bulk density results throughout the profile and assigning a value corresponding to the depth of the midpoint of each horizon. This procedure assumes that bulk density changes linearly with depth between sample points. Horizon samples were lightly ground with a mortar and pestle and passed through a 2-mm sieve to remove and weigh coarse fragments. The remaining fine-earth fraction was then homogenized by running it through a sample splitter four times, ensuring a representative subsample (~50 g) for particle-size analysis. Samples were next shaken for 2 hours in a water-based solution with sodium hexametaphosphate as the dispersant. Then particle size was analyzed by using the Bouyoucos hydrometer method (Bouyoucos 1962). The remaining portion of horizon samples were repackaged and shipped to the Cornell Nutrient Analysis Laboratory for ICP-AES, following HNO_3 - HClO_4 -HF digestion. In addition, the extractable bases calcium (Ca^{2+}), potassium (K^+), and magnesium (Mg^{2+}), and effective cation

exchange capacity (ECEC), were measured in an unbuffered ammonium chloride (NH_4Cl) solution. Base saturation was estimated as the sum of NH_4Cl -extractable bases/ECEC $\times$ 100. Soil pH (1:1 ratio with deionized water) and organic matter content (Loss on Ignition method) were also measured.

Data Calculation Methods

Deposition

For nitrogen (N) and sulfur (S) deposition data, raster grids for total N and total S deposition were downloaded from TDEP v. 2014.01 for each year from 2000 through 2012 (U.S. EPA 2014). For base cation (BC) and chloride (Cl^-) deposition data, raster grids of each BC analyte (Ca, Mg, K, and sodium [Na]) and Cl were downloaded for 2000 through 2012 (NADP 2014). Both datasets were loaded into a new ArcMap document. Raster calculator was used to compute the average for each analyte over the period of analysis. Then, site-specific values were obtained by extracting the averages at each study site. Data were converted from kg/yr to eq/ha/yr if necessary.

Mineral Weathering

Most studies that use the pedological mass balance (PMB) method for weathering rate calculations limit analysis to the rooting zone (E- through deepest B-horizons). In this study, the entire profile was analyzed (including the A-horizon down to the depth of sampling in the unweathered parent material). To account for base cations that are associated with organic matter, the concentration of exchangeable base cations (Ca, K, and Mg) at field pH is subtracted from the concentration of bases obtained through complete acid digestion of minerals, similar to the approach in Schroth et al. (2007). This correction was not done for Na (Na is assumed to be nonbiocycled). This correction procedure effectively limits the derivation of PMB mass flux calculations to nonexchangeable (solid) mineral phases. Base cations adsorbed to organic matter can accumulate at the soil surface due to preferential plant uptake and atmospheric deposition (Lichter 1998). For the purpose of obtaining a long-term weathering rate with the PMB method, these labile base cation enrichments to the surface and lower horizons are considered products of—or external inputs to—the soil weathering system and could drastically offset weathering losses from horizon flux calculations. Failing to make this adjustment could result in an underestimation of mass loss due to weathering.

To determine a weathering rate, the results of the PMB method were then divided by the age of the landscape. Absolute ages have not yet been determined for specific geologic units in the study area. However, the morainic system surrounding the watershed very likely correlates to either the Inner Port Huron or Outer Port Huron Moraine. Blewett et al. (1993) report a minimum limiting radiocarbon age of the Inner Port Huron Moraine of 12,960 (± 350), based on paleo-lake sediment samples taken next to the Inner Port Huron Moraine in Antrim County, MI. These radiocarbon dates calibrate to 15,077 (± 449) calendar years before present. Therefore, as a conservative estimate, a value of 15,100 years is used for the age of soils in the Osborn Creek watershed.

Nitrogen and Base Cation Uptake

Gross volume for each tree was calculated by using field collected data for d.b.h. and tree height. Gross volume is the total volume of wood in the central stem of trees 12.7 cm or larger in diameter, from a 30-cm stump to a minimum 10.2-cm top diameter outside bark (d.o.b.), or where the central stem breaks into limbs, all of which are less than 10 cm in d.o.b.

Gross volume equations do not account for rotten or missing form cull (Miles and Hill 2010). Diameter at breast height was then converted to diameter inside bark (d.i.b.) using the species-specific bark ratio equations in Dixon and Keyser (2014). Bark volume was obtained by subtracting gross volume using d.b.h. from gross volume using d.i.b. This calculation assumes uniform bark thickness throughout the tree bole, and also assumes a stem-only harvest (leaves and branches are left onsite).

The concentrations of BC and N derived from ICP-AES analyses in bark and bole were multiplied by their respective volumes and specific gravity values to derive total BC and N content per sampled tree using equations 7 and 8. Total BC and N content was divided by tree age to derive an annual uptake rate per tree. BC and N tree uptake rates were averaged across data for sampled trees at the site (n=5). Then, uptake averages were multiplied by the number of trees within the 20 m × 20 m plot and converted to eq/ha/yr. The uptake rates for bark and bole were added to derive species-specific total uptake for each site.

Precipitation Surplus

Precipitation surplus (Q) was estimated from daily precipitation and reference evapotranspiration data at the Hart weather station (Michigan State University Extension 2014). The period of analysis was 1997 through 2013 as reference evapotranspiration data are not available before this time. Reference evapotranspiration data were subtracted from precipitation data on a daily basis. Precipitation exceeding evapotranspiration is considered to represent surplus soil moisture (unused by plants) that passes completely through the soil profile (i.e., infiltration). Daily precipitation surplus results were summed for each month, and monthly precipitation surplus was summed for each year. Finally, an annual average was calculated for the period of analysis (17-year average).

Critical Leaching of Acid Neutralizing Capacity

For hydrogen (H⁺) and aluminum (Al) input into equation 10, lab data for Typic Udipsamments and Entic Haplorthods within Major Land Resource Area 94 (Michigan and Wisconsin Sandy Drift) were downloaded from the National Cooperative Soil Survey (NCSS) soils characterization database (NCSS, n.d.). Exchangeable Al (all unbuffered methods) and pH data for horizons with a pH of less than 5.5 were used (Al generally becomes insoluble at pH >5.5). Exchangeable Al was converted to equivalents, $\log_{10}$ transformed (defining $pAl = 3 - \log_{10}([Al]/3)$), and plotted against pH for each group of sample horizons separately (A-, E-, B-, and C-horizons). In order for these data to be incorporated into equation 9, the formulas for $ANC_{le,crit}$ have to be adapted by replacing the exponent 3 by a (a modified version of the K_{gibb} constant, as presented in equation 10) and 1/3 by 1/ a (UBA 2004). The K_{Alox} and a values were inserted into equation 9 for each horizon group to determine which had the lowest $ANC_{le,crit}$ value. The lowest value was then adopted for the whole soil profile to protect the entire pedon from unacceptable leaching. This method assumes that the exchange of base cations between soil and soil solution are in steady-state.

RESULTS AND DISCUSSION

Site-specific Tree Uptake

On average, the base cation (BC) and nitrogen (N) uptake rates derived in this study (Table 7) were higher than values used by McNulty et al. (2007) in national-scale critical acid load (CAL) estimates by 76.1 and 167.9 eq/ha/yr, respectively (Table 8). White oak (site 3) exhibited the greatest difference in BC and N uptake rates between the two studies. Red pine (site 1) exhibited the least difference in BC uptake between the studies. Northern red oak (site 5) exhibited the least difference in N uptake between the studies. The BC and N uptake rates for sugar maple and white oak (sites 2 and 3) derived here were more than twice as high as those in McNulty et al. (2007).

The differences in uptake rates between the studies are very likely related to the method used to derive annual volume increment and nutrient concentration estimates. In McNulty et al. (2007), the volume of growth allocated to bark was assumed to be 15 percent for deciduous trees and 11 percent for coniferous trees. In this study, the average ratio of bark to bole volume was 16.2 percent for deciduous trees (n=15 samples) and 17.7 percent for coniferous trees (only red pine in this study, n=10) (calculated from data in Table 7).

If bark volume had been assumed in this study to be similar to that in McNulty et al. (2007), uptake rates for red pine bark would have been underestimated. This may partially explain the positive difference in uptake rates for red pine between the two studies (Table 8). The method used in this study to estimate nutrient uptake assumes that tissue concentrations in sampled trees adequately represent the entire population of trees across the stand (i.e., average data from sampled trees were extrapolated to unsampled trees). This assumption is based on the premise that nutrients and water in the soil are consistent across the site and not limiting, and therefore tissue concentrations are largely controlled by genetics (Alban 1988). If this premise were true, then BC and N concentrations should exhibit a narrow range at the site level. Overall, samples from hardwood sites exhibited considerable variation in calcium (Ca), potassium (K), and magnesium (Mg) concentrations in bark and bole of sampled trees (Fig. 6). Sodium was omitted, as mentioned earlier, because it is not considered biologically important (Bailey et al. 2003). Sugar maple (site 2) exhibited the greatest range of bole wood BC concentrations, whereas red pine (site 4) exhibited the least. White oak (site 3) exhibited the greatest range of bark BC concentrations, and red pine (site 1) the least. Hardwoods had a greater range of tissue nutrient concentrations than conifers (red pine only) in this watershed (Fig. 6). This observation may be related to the growth characteristics of each stand. Stands with homogenous forest structure and closed canopy (such as red pine plantations: sites 1 and 4) may have more consistent tissue concentrations because light and space are consistently limited throughout the stand.

Significant intraspecies differences in nutrient concentrations occurred in red pine (sites 1 and 4) in this watershed, even on sites with very similar soils and stand development (Table 9). To better understand intraspecies variability, we tested sites 1 and 4 for significant differences in Ca, K, Mg, and N concentrations in both bark and bole. A preliminary F-test for equality of variances indicated that the variances of sites 1 and 4 were not significantly different (data not shown). Therefore, an independent two-sample *t*-test (equal variance) was run (Table 9). The mean concentrations of bole Mg and bark N were significantly higher in site 1 than site 4. Mean concentrations of all other nutrients were not significantly different between the two sites.

Table 7.—Physical characteristics of the five trees sampled at each site (with number of trees in that species on the plot shown in parentheses)

Site and forest cover	Tree age	Specific gravity of bole	Specific gravity of bark	Diameter at breast height	Tree height	Bole volume	Bark volume
	<i>yr</i>	----- <i>g/cm³</i> -----		<i>cm</i>	<i>m</i>	----- <i>m³</i> -----	
Site 1 Red pine (38)	46	0.28	0.33	25.4	20	2.39	0.42
	45	0.33	0.35	24.6	18	2.12	0.37
	42	0.29	0.30	24.4	18	2.05	0.36
	47	0.27	0.39	25.4	19	2.34	0.41
	47	0.31	0.27	27.7	18	2.59	0.46
Site 2 Sugar maple (9)	91	0.46	0.72	47.5	30	9.44	1.16
	55	0.50	0.69	64.3	34	19.35	2.41
	86	0.46	0.60	35.8	26	5.00	0.60
	96	0.48	0.56	54.1	29	11.78	1.45
	88	0.46	0.74	51.1	27	9.86	1.21
Site 3 White oak (16)	82	0.47	0.50	47.2	31	11.37	2.11
	95	0.52	0.64	45.5	26	8.90	1.63
	79	0.48	0.48	40.1	26	7.20	1.31
	79	0.50	0.58	44.5	24	8.18	1.49
	99	0.49	0.43	43.7	25	8.05	1.47
Site 4 Red pine (49)	48	0.28	0.27	32.3	20	3.55	0.64
	46	0.28	0.35	23.4	21	2.12	0.37
	50	0.28	0.24	29.0	19	2.87	0.51
	46	0.28	0.27	27.2	16	2.27	0.40
	42	0.28	0.36	26.9	18	2.41	0.43
Site 5 Northern red oak (14)	111	0.45	0.49	47.2	31	8.16	1.49
	114	0.49	0.71	45.5	26	6.40	1.15
	109	0.47	0.67	40.1	26	5.14	0.91
	106	0.43	0.37	44.5	24	5.97	1.07
	117	0.40	0.50	43.7	25	5.92	1.06

Table 8.—Tree uptake values for each site compared to uptake values used for corresponding species in McNulty et al. (2007)

Site and forest cover	Nitrogen uptake <i>eq/ha/yr</i>			Base cation uptake <i>eq/ha/yr</i>		
	From			From		
	This study	McNulty et al. (2007)	Difference	This study	McNulty et al. (2007)	Difference
Site 1 Red pine	103.0	59.1	43.9	86.5	77.1	9.4
Site 2 Sugar maple	205.0	101.8	103.2	560.4	190.5	369.9
Site 3 White oak	273.4	129.7	143.7	599.9	599.9	386.1
Site 4 Red pine	148.9	59.1	89.8	108.3	108.3	31.2
Site 5 Northern red oak	129.3	129.7	-0.4	256.6	256.6	42.8
Average difference	—	—	76.1	—	—	167.9

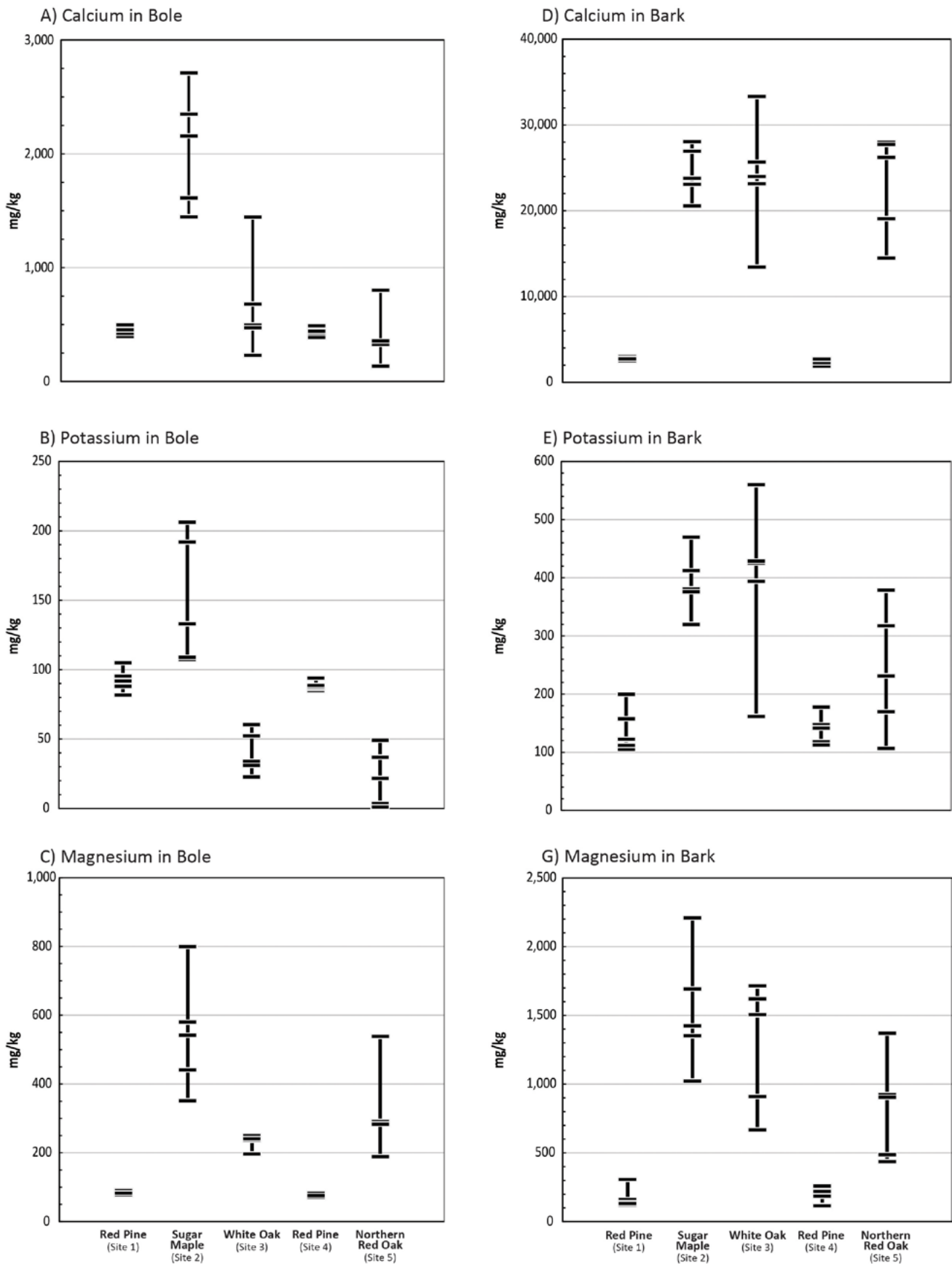


Figure 6.—Concentrations of base cations in A–C) bole samples and D–F) bark samples by site, Osborn Creek watershed. Note: The y-axis scale is different on each graph. The value for an individual tree sample is indicated by a dash.

Table 9.—Average concentrations of exchangeable base cations and nitrogen by component for sites 1 and 4 (red pine plantations) and results of paired *t*-test between variables (n=5)

Tree component and base cation	Site 1 <i>mg/kg</i>	Site 4 <i>mg/kg</i>	<i>t</i> -test	<i>p</i> -value
Bole				
Calcium	437.76	425.96	0.46	0.65
Potassium	92.38	88.32	0.97	0.35
Magnesium	85.83	75.75	3.10	0.01
Nitrogen	610.70	774.05	-1.01	0.34
Bark				
Calcium	2,713.06	2,422.16	1.62	0.14
Potassium	139.26	139.41	-0.01	0.99
Magnesium	171.79	180.07	-0.18	0.85
Nitrogen	2092.32	1588.99	3.11	0.01

Although the number of samples in each population (n=5) is small, these data suggest that tree nutrient demand may vary depending on factors other than genetics. When multiplied by growth characteristics from each stand (Table 7), these small differences in nutrient concentrations were compounded such that the final calculated uptake rates differed considerably between sites 1 and 4 (Table 8). On average, sites 2, 3, and 5 exhibited greater BC uptake than N uptake (Fig. 7). Nitrogen uptake exceeded BC uptake for sites 1 and 4. Sugar maple, white oak, and northern red oak (sites 2, 3, and 5) exhibited much wider ranges of uptake rates at the site level for both BC and N than red pine (sites 1 and 4). Sites showing large differences in uptake rates between this study and the study by McNulty et al.

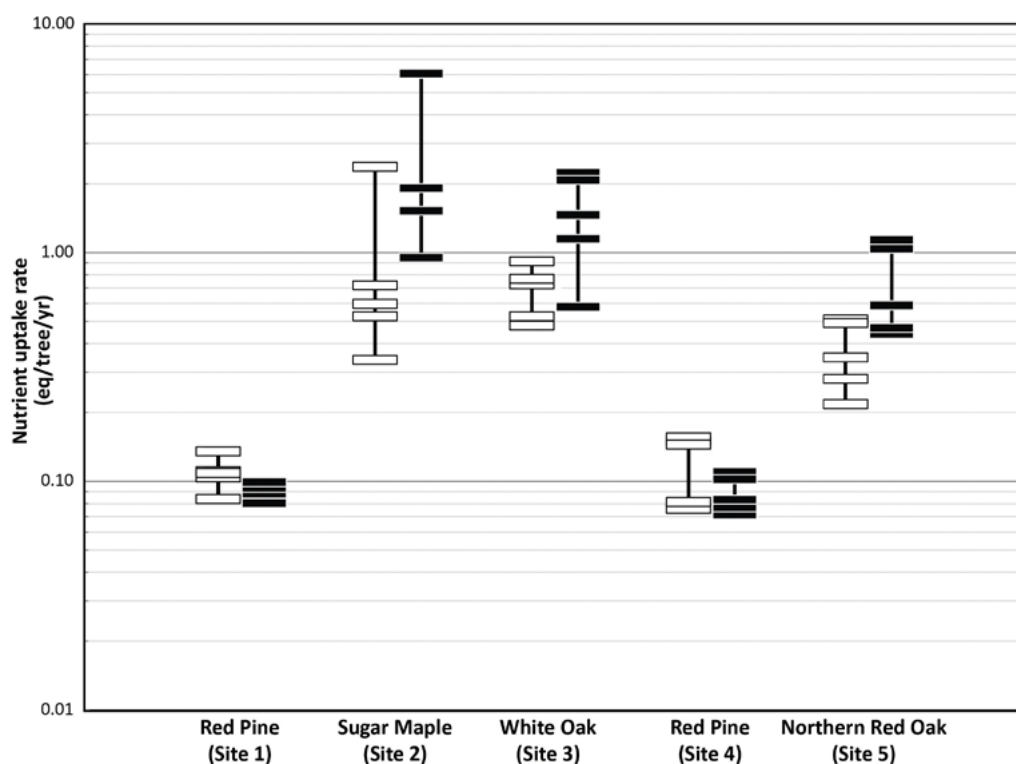


Figure 7.—Whole-tree base cation (BC) and nitrogen (N) uptake rates for sampled trees, Osborn Creek watershed. Hollow white dashes represent N uptake rates. Solid black dashes represent total uptake rates for Calcium+Potassium+Magnesium. Note: Y-axis values are logarithmic to highlight smaller ranges in red pine values (sites 1 and 4).

(2007) (sites 2 and 3; Table 7) also exhibited greater variation in d.b.h., tree height, and tissue concentrations (Table 7, Fig. 6). Data from Figure 7 suggest that uptake rates assigned to hardwoods have greater uncertainty than uptake rates assigned to red pine in this study. More sample data would be needed to verify the relevance of uptake rates derived in this study for species exhibiting considerable BC and N variation at the site level. The final site-specific CAL estimate for hardwood sites should be interpreted in light of these observations where forest harvesting occurs.

The uptake rates used in McNulty et al. (2007) are based on species-group averages from the tree chemistry database (Pardo et al. 2005). For example, red pine was grouped with eastern white pine (*P. strobus*) and jack pine (*P. banksiana*) in that study, so the values used for comparison in Table 8 were actually the average for the three species. Although the dataset in Pardo et al. (2005) is larger than the dataset collected in this study, the inclusion of multiple species in uptake terms in McNulty et al. (2007) may dilute the precision and applicability of those estimates at the watershed scale. Nutrient concentration data in the tree chemistry database were compared to data in this study on an individual species basis (rather than species group) to better understand how data derived here fit with published values. With the exception of N concentrations in red pine, data derived here for all species were less than or within the range of values published in the tree chemistry database (Fig. 8). Given the combination of 1) lower nutrient concentrations derived here compared to data in the tree chemistry database and 2) higher uptake values derived here compared to McNulty et al. (2007), the most likely explanation for the differences in uptake between the studies (Table 8) involves differences in methods used to calculate volume. Overall, the considerable site-level variation in BC and N concentrations and uptake values in this study suggests that stand characteristics and soil factors have an inseparable connection to growth and uptake, and that assigning a single value to a species across a watershed (let alone a species-group across an entire continent) may not be appropriate. Nonetheless, site-specific data are always preferred over regional data when calculating critical acid loads (Pardo 2010). Future research aimed at characterizing the influence of edaphic factors, climate, and stand factors on intraspecies variability in nutrient uptake at multiple scales would be beneficial for critical load modelers.

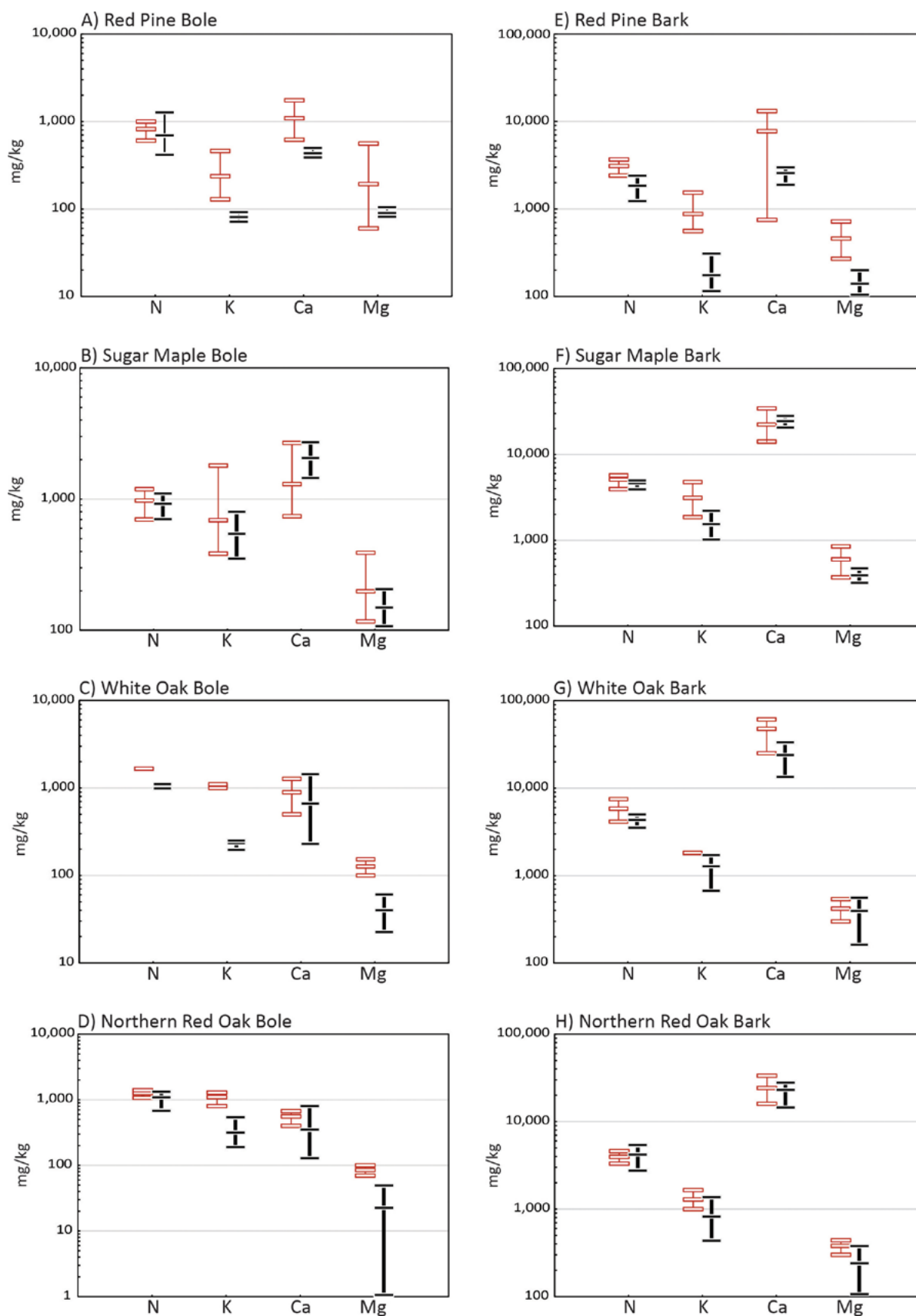


Figure 8.—Comparison of concentration data for nitrogen (N), potassium (K), calcium (Ca), and magnesium (Mg) from the tree chemistry database (Pardo et al. 2005) (hollow red dashes) and this study (solid black dashes). The three data points for each nutrient represent maximum, mean, and minimum values from each dataset. Note: The y-axis scale is different on each graph.

Table 10.—Physical and chemical characteristics of soil horizons at the five sample sites

Site and forest cover	Horizon	Depth	Sand	Silt	Clay	Loss on ignition	pH	Effective cation exchange capacity	Calcium	Potassium	Magnesium	Base saturation	Inorganic carbon ^a
		cm	Percent						cmol/kg			Percent	
Site 1 Red pine	A/E	0–25	97.4	0.5	2.1	9.26	4.46	2.82	2.53	0.14	0.38	108	<bdl
	Apb/E	25–45	92.8	7.0	0.2	2.47	4.49	0.19	0.11	0.04	0.02	94	<bdl
	BC	45–90	96.6	3.4	0.0	1.50	4.61	0.25	0.14	0.04	0.03	88	<bdl
	C/E&Bt	90–140	96.1	1.0	2.9	0.14	4.84	0.31	0.17	0.04	0.03	77	<bdl
Site 2 Sugar maple	A	0–5	86.1	11.4	2.5	11.04	3.97	0.85	4.52	0.11	0.90	654	<bdl
	E	5–10	88.7	10.3	1.1	0.89	4.11	0.34	0.46	0.02	0.10	172	<bdl
	Bw	10–20	93.4	5.9	0.7	1.24	5.16	0.52	1.35	0.09	0.43	357	<bdl
	Btw	20–60	94.4	4.1	1.5	0.73	5.01	0.32	0.82	0.07	0.25	357	<bdl
	C	60–140	97.3	1.2	1.5	0.13	5.28	0.08	0.20	0.02	0.02	302	0.56
Site 3 White oak	A	0–6	86.7	7.6	5.7	25.80	3.25	1.25	1.23	0.28	0.88	192	0.38
	E	6–12	95.3	2.5	2.1	1.35	3.94	0.28	0.09	0.18	0.04	113	<bdl
	Bt	12–17	95.5	3.6	0.9	2.24	4.00	0.25	0.09	0.04	0.02	62	<bdl
	Bw	17–80	96.4	3.6	0.0	1.43	4.24	0.23	0.08	0.03	0.01	48	<bdl
	C	80–150	98.6	1.4	0.0	0.17	4.48	0.09	0.29	0.01	0.01	349	0.38
Site 4 Red pine	A	0–5	95.5	3.8	0.7	1.87	3.78	0.26	0.19	0.04	0.02	100	<bdl
	E	5–20	94.2	3.7	2.1	1.67	4.09	0.28	0.12	0.03	0.01	59	<bdl
	Apb	20–40	90.9	7.7	1.3	1.53	4.09	0.33	0.13	0.03	0.01	52	0.38
	Btw	40–60	92.6	5.4	2.1	1.49	4.24	0.26	0.09	0.02	0.00	43	<bdl
	C	60–100	95.1	4.5	0.4	0.30	4.67	0.10	0.05	0.01	0.02	77	0.38
Site 5 Northern red oak	A	0–10	94.4	3.9	1.8	2.60	3.43	0.53	0.22	0.10	0.25	105	<bdl
	E	10–24	92.1	4.0	4.0	1.00	3.67	0.36	0.06	0.03	0.03	31	<bdl
	Btw	24–45	94.5	3.3	2.2	1.89	4.27	0.34	0.07	0.03	0.00	29	<bdl
	Bw	45–110	94.3	4.3	1.4	0.70	4.53	0.18	0.06	0.02	0.01	49	<bdl
	C	110–150	97.3	2.1	0.6	0.24	4.93	0.27	0.05	0.02	0.01	28	<bdl

^a <bdl = below detectable limit.

Soil Horizon Characteristics

Sampled soils on all sites had textures and horizon sequences typical of sandy, nutrient-poor soils found across the Huron-Manistee National Forests (Table 10). Mean clay content across all horizons was 1.6 percent. With the exception of the A-horizon at sites 1, 2, and 3, organic matter was less than 3 percent in all horizons. The pH of surface horizons ranged from 3.25 (ultra acid) to 4.46 (extremely acid). All soils exhibited a gradual increase in pH with depth. The effective cation exchange capacity (ECEC) was very low across all horizons (average 0.45 cmol/kg, n=24), reflective of a very low pH and clay content, and minimal organic matter content. Exchangeable pools of base cations (Ca, K, and Mg) at all sites (except site 4) were more than twice as great in surface horizons than the sum of all subsurface horizons, reflective of higher organic matter contents in surface horizons. Base saturation ranged from moderate to supersaturated (>100 percent) across all sites, reflective of very low ECEC. Base saturation was highest in surface horizons at all sites except site 3. Base saturation decreased in the rooting zones at sites 2, 3, 4, and 5, then slightly increased with depth. Base saturation was excessively high at site 2 (≥172 percent in all horizons).

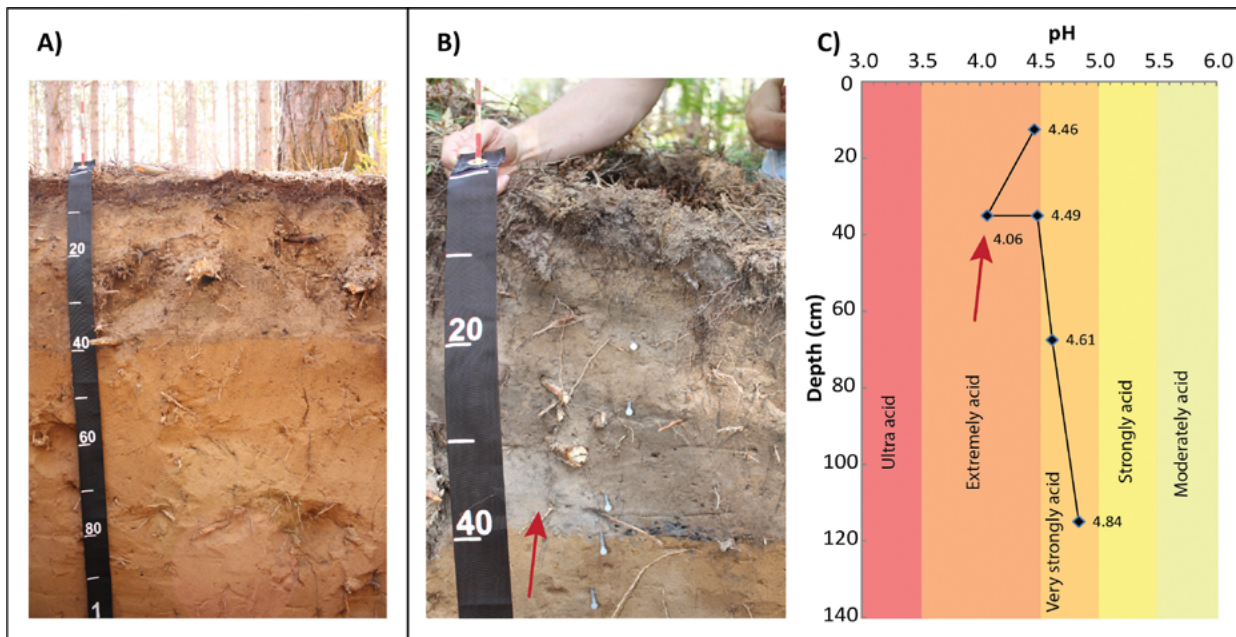


Figure 9.—Soil pit at A) site 4 and B) site 1, Osborn Creek watershed. Note the lighter-colored buried plow (Apb) horizon between 20 and 40 cm compared to the overlying A-horizon in A). Red arrow in B) points to the portion of the Apb horizon that appears to be leached relative to the remainder of the horizon, as shown by C) the sharp decline to an extremely low pH within that horizon. Photos by Trevor Hobbs, Huron-Manistee National Forests.

Soils under red pine plantations (sites 1 and 4) contained a distinct buried plow (Apb) horizon with an abrupt lower boundary (Fig. 9), indicating that these two sites were farm fields before the U.S. Forest Service acquired the land in the 1930s. The original plow horizons were probably buried during site preparation prior to planting red pine in the early 1960s. The Apb horizons at sites 1 and 4 were distinctly lighter colored than their overlying A-horizons (Munsell color 10YR 5/2 vs. 10YR 2/2). At site 1, an additional pH sample (not presented in Table 10) was taken from the middle of the Apb horizon near a large root where there was possible evidence of accelerated leaching (Fig. 9B). The pH near the large root was lower (4.06) than at the middle of the horizon (4.49). Together, these observations suggest that acidic conditions in the subsoil have facilitated the translocation of humus out of Apb horizons in a relatively short time span (~46 years since planting). This interpretation is supported by other studies demonstrating that conifer species produce aggressive weathering agents via root exudation (i.e., low-molecular-weight organic acids) (Augusto et al. 2002, Schroth et al. 2007), and may even facilitate accelerated weathering under nutrient stress in order to obtain necessary nutrients, especially during early stand development prior to canopy closure (Balogh-Brunstad et al. 2008, Shi et al. 2014).

Weathering Results

Bulk density ranged between 0.65 and 1.66 g/cm³ in all horizons of sampled soils, and gradually increased with depth (Table 11). Sites 1 and 4 (red pine plantations) exhibited higher surface bulk densities than sites 2, 3, and 5 (hardwoods), reflective of legacy soil compaction associated with site preparation. Major BC concentrations were lowest in A- and E-horizons at all sites, typical of soils dominated by podzolization. At all sites, Mg concentrations reached a maximum in the lowest B-horizons. At sites 1 and 2, Ca concentrations in surface horizons were about 3.5 times greater than in their respective C-horizons. Titanium (Ti) concentrations exhibited considerable variability with depth and across all sites.

Table 11.—Select physical and chemical characteristics of soil profiles at the five sample sites

Site and forest cover	Horizon	Horizon thickness	Bulk density	Calcium (Ca) ^a	Potassium (K) ^a	Magnesium (Mg) ^a	Sodium (Na) ^a	Titanium (Ti) ^a
		<i>m</i>	<i>g/cm³</i>	<i>mg/kg</i>				
Site 1 Red pine	A/E	0.25	1.36	84.60	63.42	112.48	3.98	23.19
	Apb/E	0.20	1.61	14.84	94.23	180.52	5.27	27.37
	BC	0.45	1.65	35.75	116.04	366.67	6.46	40.16
	C/E&Bt	0.50	1.54	18.92	96.36	314.89	5.54	32.22
Site 2 Sugar maple	A	0.05	0.67	420.66	306.54	221.69	5.39	7.76
	E	0.05	0.76	71.09	260.34	79.88	1.89	6.86
	Bw	0.1	1.02	221.74	288.32	369.76	3.31	6.97
	Btw	0.4	1.55	147.16	242.64	333.70	2.32	10.10
	C	0.8	1.66	121.52	249.45	233.94	3.66	20.22
Site 3 White oak	A	0.06	0.65	14.63	286.89	199.10	10.32	11.91
	E	0.06	0.81	20.83	176.55	85.15	0.75	4.54
	Bt	0.05	0.98	43.49	254.32	165.05	1.47	8.64
	Bw	0.63	1.44	47.57	260.64	203.16	2.01	5.84
	C	0.7	1.62	19.86	250.90	163.03	0.07	18.89
Site 4 Red pine	A	0.05	1.28	25.60	260.35	162.07	0.91	8.01
	E	0.15	1.32	31.35	263.90	177.00	1.35	6.28
	Apb	0.2	1.42	39.79	262.27	179.81	2.97	8.25
	Btw	0.2	1.46	56.23	292.25	334.39	2.90	9.74
	C	0.8	1.72	59.27	261.92	323.68	1.60	13.76
Site 5 Northern red oak	A	0.1	0.88	0.08	212.28	61.44	1.33	5.27
	E	0.14	1.13	20.86	231.12	70.64	0.36	8.34
	Btw	0.21	1.41	42.30	267.62	212.97	2.92	6.65
	Bw	0.65	1.46	27.04	236.51	187.32	1.02	9.12
	C	0.4	1.61	46.42	268.59	256.06	1.12	7.85

^a Base cation concentration data for Ca, K, and Mg presented here represent nonexchangeable quantities, derived by subtracting concentrations in extractable forms (from an ammonium chloride solution) from concentrations after nitric acid-perchloric acid-hydrofluoric acid digestion. Data for Na and Ti are from nitric acid-perchloric acid-hydrofluoric acid digestion.

In results from the strain analyses (Table 12), positive strain values suggest that a horizon has expanded in volume (dilation) since time zero, resulting in a decrease in Ti concentration in the weathered horizon relative to the parent material. Negative strain values suggest that a horizon has decreased in volume (collapse) since time zero, resulting in an increase in Ti concentration in the weathered horizon relative to the parent material. Sites 1 and 5 exhibited slightly positive strain values at the surface with gradually decreasing strain values at depth. Sites 2, 3, and 4 exhibited drastic increases in strain up-profile, generally reaching a maximum at or just below the surface. Because Ti is considered nonweatherable, positive strain values in weathered horizons are attributable to decreases in bulk density, inherent mineralogical heterogeneity of the original soil parent materials, or both. In a soil weathering study of successive beach ridges (dune sand), Lichter (1998) attributed up-profile decreases in bulk density to the accumulation of organic matter or bioturbation (confounding edaphic factors that introduce heterogeneity over time), or a combination thereof, and cited the correlation between Ti concentrations and bulk density as supporting evidence. In this study, soil horizon

Table 12.—Strain analyses and mass flux for each horizon and final weathering rate (WR) for the entire pedon at the five sample sites

Site and forest cover	Horizon	Horizon thickness (m)	Strain ($\epsilon_{i,w}$) ^a	Calcium	Potassium	Magnesium	Sodium	Base cation	WR ^b
-----eq/ha/yr-----									
Site 1 Red pine	A/E	0.25	0.57	8.0	-0.3	-21.1	0.0	-13.4	-36.0
	Apb/E	0.20	0.13	-0.1	0.7	-15.3	0.1	-14.6	
	BC	0.45	-0.25	3.0	-0.5	-10.4	-0.1	-8.0	
	C/E&Bt	0.50	0.00						
Site 2 Sugar maple	A	0.05	5.46	4.3	1.2	2.4	0.0	7.9	130.7
	E	0.05	5.44	0.6	1.1	0.0	0.0	1.7	
	Bw	0.1	3.72	4.0	3.5	16.1	0.1	23.7	
	Btw	0.4	1.14	11.7	12.4	73.2	0.1	97.4	
	C	0.8	0.00						
Site 3 White oak	A	0.06	2.95	0.0	0.9	2.1	0.1	3.1	126.3
	E	0.06	7.32	0.3	1.0	1.2	0.0	2.5	
	Bt	0.05	2.61	0.6	1.2	2.4	0.0	4.2	
	Bw	0.63	2.64	12.4	28.1	75.5	0.5	116.5	
	C	0.7	0.00						
Site 4 Red pine	A	0.05	1.31	-0.2	1.2	-0.9	0.0	0.1	35.6
	E	0.15	1.86	0.3	4.8	3.2	0.0	8.3	
	Apb	0.2	1.02	0.4	5.1	-2.2	0.2	3.5	
	Btw	0.2	0.66	1.4	5.3	16.8	0.2	23.7	
	C	0.8	0.00						
Site 5 Northern red oak	A	0.1	1.73	-0.9	0.5	-5.3	0.0	-5.7	-101.7
	E	0.14	0.34	-1.5	-1.5	-17.4	-0.0	-20.4	
	Btw	0.21	0.35	0.3	2.0	-0.6	0.2	1.9	
	Bw	0.65	-0.05	-8.4	-12.1	-56.9	-0.1	-77.5	
	C	0.4	0.00						

^a Positive strain values suggest that horizon volume has increased (dilation); negative values suggest that horizon volume has decreased (collapse). C-horizon strain values are always 0.000 by definition.

^b Positive WR values indicate net gain of base cations and equate to a “negative” weathering rate.

Ti concentrations exhibited a weak relationship to bulk density (data not shown). Therefore, soil horizons that exhibit drastic positive strain values (dilation) in Table 12 are likely to be more indicative of mineralogical heterogeneity of the parent materials than increases in bulk density up-profile.

This result is surprising given the close proximity and presumed similar genetic origin of these soils (i.e., outwash). Because the PMB method relies on the assumption that the original parent materials are homogenous, a common approach is to “normalize” parent material mineralogy data before calculating weathering losses. Parent material normalization effectively accounts for bioturbation. No such transformations were done with soil geochemical data in this study. We found no apparent significant correlation between Ti concentrations, bulk density, organic matter, or any other soil factors that would warrant transformation of the data in an attempt to produce more-accurate results. Furthermore, the sample size of parent materials across the study area was small (n=5). As such, the final mineral weathering rates for sites 2, 3, and 4 should be interpreted with caution.

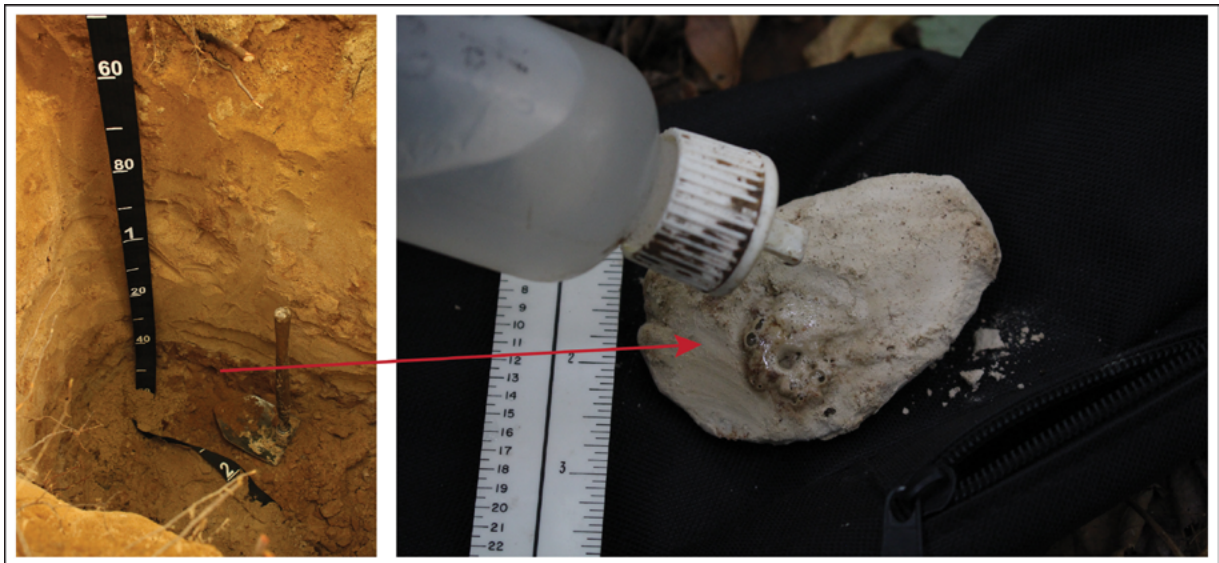


Figure 10.—A) Bottom of soil pit at site 2 showing contact with underlying diamict and B) chalky nodule found in the diamict showing strong effervescence in hydrochloric acid. Red arrow shows where nodule was extracted. Photos by Trevor Hobbs, Huron-Manistee National Forests.

The weathering rates for all soils ranged between -36.2 and $+130.7$ eq/ha/yr (Table 12). Three of the soils (sites 2, 3, and 4) exhibited positive net fluxes of base cations throughout the profile and, therefore, “negative” weathering rates. Negative weathering rates calculated by the PMB method have primarily been attributed to inherent heterogeneity of immobile elements or major oxides throughout the profile (Whitfield et al. 2011). This explanation may be partially representative of conditions at sites 2, 3, and 4, which exhibited significant positive strain not attributable to bulk density decreases. However, the positive BC fluxes at sites 2, 3, and 4 were disproportionately large compared to corresponding strain values (Table 12). Instead, net base cation gains may be indicative of significant external BC input at these sites. Possible external sources include atmospheric deposition of BC, preferential plant uptake from deeper soil layers containing high BC concentrations, or influxes of BC from shallow lateral groundwater flow during periods of heavy recharge. The latter explanation is supported by results for Mg concentrations, which showed the largest increase in the lowest B-horizon at sites 2, 3, and 4 (Table 12). Assuming that Mg held on clay exchange sites was sufficiently accounted for prior to calculating mass flux, BC concentration increases of such magnitude were probably not due to pedogenic translocation of clays, especially considering the low clay contents of these soils (<3 percent). Furthermore, at site 2, a clear lithologic discontinuity at a depth of about 140 cm was evident. The underlying diamict contained highly weathered chalk fragments that effervesced strongly in 10 percent hydrochloric acid (Fig. 10). These chalk fragments probably originated from underlying Paleozoic sedimentary carbonates in the Michigan Basin (many of which are dolomitized or high-Mg limestone) (Catacosinos et al. 2001) and subsequently would have been incorporated into the local till as ice advanced from the west. These nodules may be the source of elevated Mg and inorganic carbon in lower horizons at sites 2, 3, and 4 (Table 10); these elements would periodically mix with soil waters during periods of heavy rain or rapid spring snowmelt. Other soil weathering studies in areas with significant groundwater flow or sand over calcareous substrata, or both, document a transition from silicate- to carbonate-dominated weathering regimes below the primary rooting zone (Graf Pannatier et al. 2004; Jin et al. 2008, 2012). Such a scenario is suggested here at sites 2, 3, and 4.

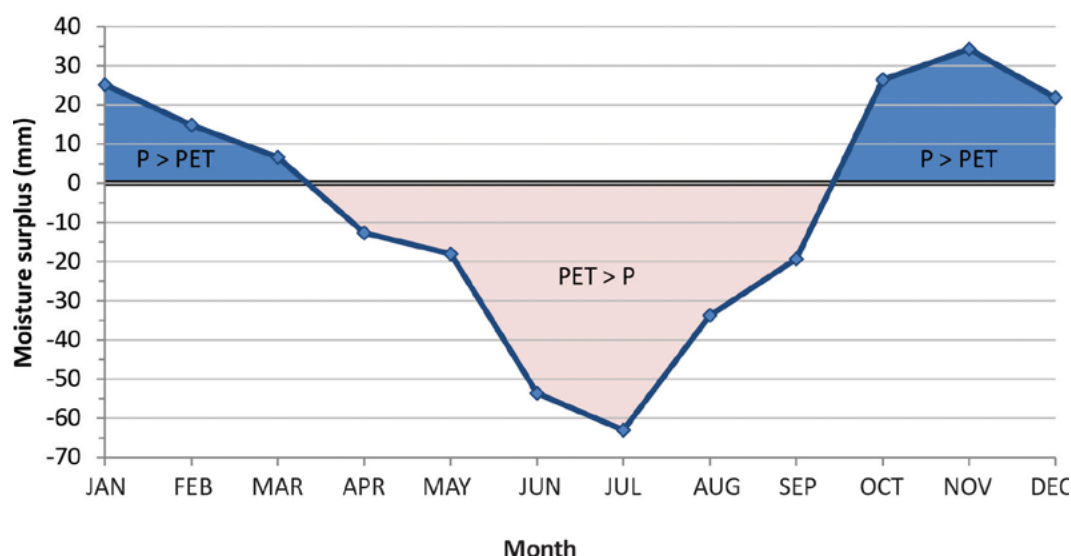


Figure 11.—Average moisture surplus by month at the Hart, MI weather station, 1997–2013. Blue areas represent periods when precipitation (P) exceeded reference potential evapotranspiration (PET). Pink area represents period when reference potential evapotranspiration exceeded precipitation.

Critical Leaching of Acid Neutralizing Capacity

Precipitation Surplus

Average annual precipitation surplus at the Hart weather station for the period of analysis was 189 mm/yr. The highest soil infiltration occurred in November prior to soil freezing and as photosynthesis completely stops. Precipitation lags behind evapotranspiration in June and July, creating droughty conditions in the sandiest of soils (Fig. 11).

Relationship between pH and Exchangeable Aluminum

The relationship between pH and exchangeable aluminum (pAl) was generally weak across all horizon groups (Table 13). However, values derived for K_{alox} were still within the range suggested as default values using the K_{gibb} equilibrium constant for mineral soils with less than 5-percent organic matter (Table 5) (UBA 2004). This finding supports the use of the K_{alox} values derived here for calculating $ANC_{le,crit}$ in this study.

Table 13.—Equilibrium constant K_{alox} and exponent a , derived from the National Cooperative Soil Survey soil characterization database for Typic Udipsamments and Entic Haplorthods in Michigan and Wisconsin Sandy Drift, and results of regression analysis of pH and aluminum concentrations using data from the soil characterization database

Soil horizon group	K_{alox}	a	Number of samples analyzed	r^2
A	6,026	0.31	59	0.28
E	9,550	0.29	44	0.19
B	2,512	0.38	168	0.24
C	2,455	0.41	34	0.40

Table 14.—Site-specific values for acid neutralizing capacity derived from standard gibbsite equilibrium constant (K_{gibb}) and the equilibrium constant K_{alox} for different horizon groups analyzed from National Cooperative Soil Survey soil characterization data

Site	K_{gibb}	K_{alox}			
	(organic matter <5%)	A-horizons	E-horizons	B-horizons	C-horizons
		eq/ha/yr			
1	130.2	147.9	147.9	147.9	147.9
2	26.3	37.5	37.5	37.5	37.5
3	26.3	37.5	37.5	37.5	37.5
4	351.4	375.7	375.7	375.7	375.7
5	357.6	381.9	381.9	381.9	381.9

Site-specific Acid Neutralizing Capacity Rate

Site-specific acid neutralizing capacity rate ($ANC_{le,crit}$) values are presented in Table 14. As a means of comparison, results of the $ANC_{le,crit}$ equation using both the standard K_{gibb} equilibrium constant for sandy soils with less than 5-percent organic matter and the horizon group K_{alox} constants derived here from local soils data are presented. The formula for $ANC_{le,crit}$ requires that the sum of BC_{dep} and BC_w exceed BC_u , or that there is a minimum (nonnegative) amount of BC concentration in soil waters, below which base cations cannot be taken up by vegetation (UBA 2004). Sites 2, 3, 4, and 5 did not meet this requirement, due to a combination of negative weathering rates (sites 2, 3, and 4), and high uptake rates. Thus, for determining site-specific $ANC_{le,crit}$, a minimum soil water BC concentration of 0.1 eq/m³ was used to calculate a default BC leaching term in the form of:

$$BC_{le} = BC_{dep} + BC_w - (0.1 \times [Q]), \quad (11)$$

Where

BC_{dep} = base cation deposition,

BC_w = base cation weathering, and

Q = precipitation surplus (in m³/ha/yr) (UBA 2004).

This formula (eqn. 11) was used in place of the $(BC_{dep} + BC_w - BC_u)$ term in the $ANC_{le,crit}$ equation (eqn. 9) for sites 2, 3, 4, and 5. Furthermore, site 4 was assigned the weathering rate of site 1 (red pine, 36.2 eq/ha/yr) and sites 2 and 3 (hardwoods) were assigned the weathering rate of site 5 (northern red oak, 101.8 eq/ha/yr). The use of the standard K_{gibb} resulted in slightly lower $ANC_{le,crit}$ values than any of the horizon group K_{alox} values (Table 14). Furthermore, within each K_{alox} horizon group, no differences between resulting $ANC_{le,crit}$ values were observed (to two decimal places), with the exception of a minor difference in the C-horizon group as applied to site 5. The minimal difference between each horizon group is very likely due to the small Q used for all sites (the $ANC_{le,crit}$ equation appears sensitive to Q).

Table 15.—Site-specific input parameters to the simple mass balance equation, critical acid loads (CALs), and exceedances for the five sample sites

Site	BC_{dep}	Cl_{dep}	BC_w^a	BC_u	$N_{i\text{worst}}^b$	$N_{i\text{best}}^b$	N_u	$ANC_{le,crit}$	$S+N_{dep}$	CAL 1 ^c	CAL 2 ^c	Ex 1 ^d	Ex 2 ^d
1	148.9	21.7	36.2	86.5	0	638.7	103.0	147.9	1,209.7	32.0	670.7	1,177.7	539.0
2	148.4	21.6	101.8	560.4	0	536.7	205.0	37.5	1,209.7	-164.3	372.4	1,374.0	837.3
3	148.4	21.6	101.8	599.9	0	468.3	273.4	37.5	1,209.7	-135.4	332.9	1,345.1	876.8
4	148.6	21.6	36.2	108.3	0	592.9	148.9	375.7	1,209.7	-171.9	421.0	1,381.6	788.7
5	152.8	22.1	101.8	256.6	0	538.6	129.3	381.9	1,135.2	-276.7	261.9	1,411.9	873.3

^a Weathering results from the pedological mass balance method at sites 2, 3, and 4 were negative; therefore, the base cation weathering rate (BC_w) from site 1 (red pine) was assigned to site 4 (also red pine), and BC_w from site 5 (hardwood) was assigned to sites 2 and 3 (also hardwoods).

^b $N_{i\text{worst}}$ represents the worst-case scenario and $N_{i\text{best}}$ the best-case scenario for long-term nitrogen (N) retention in soils. The $N_{i\text{worst}}$ value of 0 assumes that all N is leached from the soil. The $N_{i\text{best}}$ value is set to $N_{dep} - N_u$, representing a scenario where all deposited N (N_{dep}) that is not taken up by vegetation (N_u) is retained in the soil.

^c CAL 1 = critical acid load estimate using $N_{i\text{worst}}$ term; CAL 2 = critical acid load estimate using $N_{i\text{best}}$ term.

^d Ex 1 = exceedance of critical acid load under CAL 1; Ex 2 = exceedance of critical acid load under CAL 2.

Critical Acid Loads and Exceedances

To calculate CAL estimates and exceedances for sample sites (Table 15), we used two scenarios for describing the fate of N in soils. In the worst-case scenario, $N_i = 0$; in other words, all N_{dep} that is not taken up by vegetation (N_u) is leached from the soil. In the best-case scenario, $N_i = N_{dep} - N_u$ (i.e., all N_{dep} that is not taken up by vegetation is retained in the soil system). Critical acid load estimates were calculated for each scenario using both $N_{i\text{worst}}$ (CAL 1) and $N_{i\text{best}}$ (CAL 2). Under the worst-case scenario, CALs ranged from -276.7 to 32.0 eq/ha/yr. Under the best-case scenario, CALs ranged from 332.9 to 670.7 eq/ha/yr. Negative CAL estimates for sites 2, 3, 4, and 5 under the worst-case scenario resulted from a combination of very low weathering rates and high tree uptake rates. Two scenarios for exceedance (corresponding to CAL 1 and CAL 2) are also presented in Table 15. Under scenario 1, exceedances ranged from 1,177.7 to 1,411.9 eq/ha/yr. Under scenario 2, exceedances ranged from 539.0 to 876.8 eq/ha/yr.

The site-specific CAL estimates under the worst-case and best-case scenarios were lower than national-scale CAL estimates by McNulty et al. (2010) for the same locations (Table 16). A critical acid load was not calculated for site 1 in McNulty et al. (2010), perhaps because only a minor amount of forest land was contained in that cell of their raster analysis, and therefore was excluded. For the other four sites, CALs under the worst-case scenario in this study were 869.2 eq/ha/yr lower on average, and CALs under the best-case scenario in this study were 335.0 eq/ha/yr lower on average, than those reported by McNulty et al. (2010). Exceedances under the worst-case scenario in this study were 975.1 eq/ha/yr higher on average than those reported by McNulty et al. (2010). Exceedances under the best-case scenario were 441.0 eq/ha/yr higher on average than what was determined by McNulty et al. (2010). The CALs and exceedances under each scenario in this study were drastically different, demonstrating the uncertainty associated with long-term N_i . However, the fate of N is probably better represented by the worst-case scenario than the best-case scenario. Data from the Michigan Nitrogen Deposition Gradient Study support this interpretation (Zak et al. 2008, Zogg et al. 2000). Zak et al. (2008) concluded that about 20 percent of anthropogenic N applied to their sites was assimilated into forest soil pools, with the remainder lost to leaching. Regardless of the fate of N at sites in this study, both scenarios suggest that soils are indeed at risk for acidification under current deposition rates, and perhaps more at risk than was predicted by national-scale CAL estimates (McNulty and Boggs 2010, McNulty et al. 2007).

The estimated CALs and exceedances presented here are heavily influenced by BC_u and N_u . Considerable variation in BC_u and N_u was found across and within all sites in this study. To illustrate the influence of BC and N uptake in the simple mass balance model, CALs and exceedances were determined using the lowest and highest BC and N uptake data for individual trees at each site (Table 17). The lowest BC uptake combined with the highest N uptake represents the best-case scenario (highest CAL) for soil acidification status at each site. In contrast, the highest BC uptake combined with the lowest N uptake represents the worst-case scenario (lowest CAL) for soil acidification status at each site. Furthermore, because of the uncertainty associated with the fate of N at these sites, exceedance was calculated using only S_{dep} in Table 17. Even under the best-case uptake scenario while using only S_{dep} (ignoring the potential for acidification from excess N), all sites were in exceedance of CALs.

In this study, we assume that BC_u and N_u are representative of losses due to stem-only harvest. In reality, the BC_u and N_u terms are accurate and applicable only when the sites are actually logged and nutrients are taken offsite. This method was used as a cautious approach to assess the potential for soil acidification assuming all trees will eventually be harvested. Although harvesting has not yet occurred, many of these stands are at or beyond their intended rotation age. The red pine stands in particular (sites 1 and 4) will probably be harvested more intensively than any of the other sites. Sites 2, 3, and 5 (hardwood stands) may or may not be harvested at all. Regardless of the intended rotation period or date of harvest, the CALs and exceedances presented here suggest that soils are at an increased risk for acidification if harvesting continues on these sites.

Although the estimated CALs and exceedances presented here are also influenced by BC_w rates, the difference in weathering estimates calculated by different methods does not significantly influence the main conclusions of this study. National-scale estimates using the soil texture

Table 16.—Critical acid loads (CALs) and exceedances as mapped by McNulty et al. (2010) at locations in this study and the difference (diff) in CALs and exceedances between this study and McNulty et al. (2010) for two nitrogen immobilization (N_i) scenarios^a

Site	Critical acid load		
	From McNulty et al. (2010)	Diff scenario 1 _t	Diff scenario 2 _t
<i>eq/ha/yr</i>			
1	0	0	0
2	764.9	–929.2	–392.5
3	768.0	–903.4	–435.1
4	770.5	–942.4	–349.5
5	424.9	–701.6	–163.0
Average difference		–869.2	–335.0
Site	Exceedance		
1	0	0	0
2	320.9	1,053.1	516.4
3	314.3	1,030.8	562.5
4	313.0	1,068.6	475.7
5	664.1	747.8	209.2
Average difference		975.1	441.0

^a Scenario 1 uses the $N_{i\text{worst}}$ term, which assumes that all N is leached from the soil, whereas scenario 2 uses the $N_{i\text{best}}$ term, which assumes that all N not taken up by vegetation is retained in the soil.

Table 17.—Critical acid loads and exceedances from empirical tree nutrient data in this study for two nitrogen immobilization scenarios^a

Site	Critical acid load	
	Best-case scenario	Worst-case scenario
	$BC_{ulow} + N_{uhigh}$	$BC_{uhigh} + N_{ulow}$
1	68.2	1.7
2	243.7	–1,094.7
3	326.2	–509.4
4	–108.5	–253.4
5	–125.1	–469.9
Site	Exceedance ^b	
1	399.8	466.3
2	224.2	1,562.6
3	141.8	977.4
4	576.4	721.4
5	592.4	937.8

^a Scenarios differ in relative rate of base cation uptake (BC_u) and of nitrogen uptake (N_u).

^b Exceedances calculated with deposited sulfur only.

approximation (STA) method predicted that mineral weathering in this location is 239.7 eq/ha/yr (McNulty and Boggs 2010). Despite the difficulties that the pedological mass balance (PMB) method presented in estimating mineral weathering at these sites, the difference between BC_w using the STA method and using the PMB method here would not result in a shift from exceedance to nonexceedance for any sites under either CAL scenario (calculations not shown).

MANAGEMENT RECOMMENDATIONS

Applicability of Results

The critical load concept is a useful tool for assessing the potential for adverse impacts caused by acid deposition and harvesting. Exceedances of critical acid loads (CALs) presented here may or may not actually indicate a specific ecosystem response, but rather suggest that adverse impacts to trees and soil nutrient pools are more or less likely. Even though typical ecosystem responses associated with CAL exceedance (such as fine root or foliar damage) were not specifically investigated during fieldwork, no gross signs of tree decline were noticed. The exceedances under best- and worst-case scenarios presented here suggest that soils in the Osborn Creek watershed are likely to undergo a reduction in capability of supporting healthy, vigorous stands if current rates of atmospheric deposition and presumed harvest scenarios continue. However, until more is known about N dynamics in these stands, and until specific dose-response relationships relating acid deposition to changes in a soil chemical indicator (and subsequent biological harm) are defined for different forest types in Michigan, quantifying the likelihood of adverse impacts will remain difficult.

Much recent work in Michigan, as part of the Michigan Nitrogen Deposition Gradient Study research program, has focused on understanding nitrogen (N) cycling and retention in forest soils, specifically with respect to impacts on soil fungal and microbial populations and carbon dynamics under elevated anthropogenic N deposition (Fisk et al. 2002; Hassett and Zak 2005; Zak et al. 2004, 2008; Zogg et al. 2000). Among important conclusions about soil acidification, the research program suggests that the influence of acid deposition in Michigan has been waning since passage of the 1990 amendments to the Clean Air Act of 1970, trees do not appear to be in decline, and soil base saturation remains high (Talhelm et al. 2012). These conclusions may be relevant to the Osborn Creek watershed as well. Base saturation remains high at all sites, with the exception of values approaching less than 30 percent in the subsoil at site 5 (Table 10). However, base saturation may not be an appropriate chemical indicator for this watershed. Soils in this region are predominantly sand textured, and as such, small differences in base cation concentration can have a large influence on base saturation calculations. Moreover, the Michigan Nitrogen Deposition Gradient study focuses solely on sugar maple stands, and therefore, conclusions from that research program may not be directly applicable to stands of other hardwoods and conifers in the Osborn Creek watershed.

Watershed Condition Framework Rating

Soils in all watersheds on the Huron-Manistee National Forests (HMNF) were given a score of “3—Impaired Function” in the Watershed Condition Framework (WCF) rating system. If it is important from a management perspective to adhere to the operational definitions associated with each WCF rating (Table 1), then a rating of 3 should be retained for this watershed. Indeed, sulfur (S) and N deposition are above the critical loads as determined in

this study. However, exceedance of CALs has not yet produced observable negative impacts on forests in this watershed. If the observations and results from this study are considered in context, then a rating of “2—Functioning at Risk” is probably a more appropriate description of soil conditions across the Osborn Creek watershed.

Several factors complicate the widespread applicability of the results in this study. These factors include:

- The wide range of base cation and nitrogen uptake estimates within and across sites
- The difficulty in estimating mineral weathering rates using empirical data
- The influence of geologic (nonpedogenic) layers below the soil pedon, which may provide “external” nutrients to varying degrees depending on their lithology and the depth of rooting
- The apparent chemical influence of groundwater in lower pedons at some sites
- The excessively high base saturation in contrast to low pH in soils.

Biogeochemical cycling between soils and trees is highly spatially heterogeneous at the watershed scale. Given the wide range of CALs and exceedances presented here and the high degree of uncertainty associated with some parameters of the simple mass balance model—despite detailed site-specific data—extrapolating the results of this research to assess the acidification status of soils in other watersheds on the HMNF is not recommended. Doing so would not provide any more-reliable insight than what has already been determined by national-scale CAL estimates for this area (McNulty and Boggs 2010, McNulty et al. 2007). In that sense, the national-scale CAL estimates are viewed as an excellent starting point to assess the potential for soil acidification at the regional scale, which can then be used to target specific watersheds for more detailed and site-specific modeling.

Monitoring Soil Nutrients

According to site-specific analysis, all sites in this study are likely to undergo CAL exceedance under presumed harvest scenarios. These results raise questions about the most effective management strategies to maintain soil nutrient pools and protect soils from increased acidification—especially in light of current fuels reduction and short-term rotation harvest practices common in some areas of the HMNF. Some national forests, particularly in the mid-Atlantic region, where acid deposition has historically been high relative to other parts of the country, have developed monitoring programs aimed at understanding and tracking the impacts of acid deposition on soils and forest health (Connolly et al. 2007, Farr et al. 2009). Those forests can easily demonstrate and justify the costs of conducting such work because of continued high acid deposition (especially N), and the observed effects directly related to acid deposition in both terrestrial and aquatic ecosystems (Driscoll et al. 2003, Siemion et al. 2014, Sullivan 2012). In the western Lower Peninsula of Michigan, however, acid deposition has historically been moderate; total $(S+N)_{dep}$ was about 17.9 kg/ha/yr from 2000 to 2012 in the Osborn Creek watershed.

Regardless of the impacts of acid deposition, forests in the region are already subject to a host of stresses from pests such as emerald ash borer (*Agrilus planipennis*) and gypsy moth (*Lymantria dispar*) and diseases such as oak wilt (causal agent: *Ceratocystis fagacearum*) and beech bark disease (a complex that includes the insect *Cryptococcus fagisuga* and the fungi *Neonectria* spp.) (Michigan Department of Natural Resources 2014). Groundwater across the

region is generally considered to be well buffered due to high carbonate content inherited from underlying calcareous bedrock units in the Michigan Basin. Furthermore, the underlying glacial stratigraphy (and hence, surface water/groundwater interactions) is exceedingly complex and not mapped in detail. The sandiest of soils in Michigan, which are logically considered to be at risk of acidification, often contain an unpredictable arrangement of thin geogenic clayey layers (i.e., “textural banding”) that have been demonstrated to attract roots well below typical rooting depths (Hannah and Zahner 1970), and therefore very likely play a role in nutrient cycling and soil acid-base chemistry. In the Lower Peninsula, these features have been formally recognized and incorporated into county soil survey data only where U.S. Forest Service ecological mapping protocols were used (i.e., on U.S. Forest Service lands) (Host et al. 1993). Therefore, significant differences exist between map unit concepts across political boundaries in Michigan. These inconsistencies add more uncertainty to landscape-scale CAL modeling efforts that rely on publicly available soils data.

From a broad management perspective, existing conditions may dampen the urgency for monitoring the impacts of acid deposition on the HMNF, especially in light of the inherent complexity of detecting direct links between acid deposition and ecosystem effects in this area. Moreover, new research suggests that climate change may have an impact on the factors that regulate CAL limits. For example, an increase in BC weathering and N uptake and retention due to warming temperatures and shifting precipitation patterns may benefit ecosystems susceptible to acidification (McNulty et al. 2013). Despite these uncertainties, we recommend an intensified soil nutrient management effort beyond what is already laid out in Forest Service Manual 2550 and the HMNF Land and Resource Management Plan (U.S. Forest Service 2012), specifically for ensuring long-term supply of soil base cations under current harvest and fuels reduction practices. An important first step would be to obtain more-sophisticated modeling tools to calculate mineral weathering (i.e., PROFILE), and plant uptake rates (i.e., the GrowUp model; Coordination Centre for Effects 2017) to reduce uncertainty related to those parameters of the SMB model. Further steps would involve using existing soil geochemical data from the U.S. Geological Survey (Smith et al. 2013) with additional field data to create Forest-wide maps that more accurately predict mineral weathering rates. Similarly, more field data should be collected to better understand the spatial variability and edaphic conditions that contribute to tree base cation (BC) and N uptake variability at the Forest-wide scale. At the site-specific scale, nondestructive methods for accurately and quickly measuring timber volume at monitoring plots (i.e., photogrammetry or LiDAR) could add greater certainty to BC and N uptake estimates.

These additional steps could be used to develop a Forest-wide nutrient sensitivity map, similar to what is in place on the Monongahela National Forest in West Virginia (Connolly et al. 2007), or to develop sustainable harvest interpretation ratings (applied to soil map units), similar to what is in place for Connecticut (Natural Resources Conservation Service 2016). This map could then be used to develop guidelines for long-term, sustainable fuels reduction and harvest practices tailored to the capability of individual sites and soils to withstand repeated nutrient removals. Finally, testing and applying a soil chemical indicator that specifically relates acid deposition levels to a detectable symptom of biological damage for different tree species found across the HMNF would be informative. This effort ideally should be taken on by a government agency (such as the U.S. Forest Service), university, or other research organization that demonstrates a high level of expertise in both critical load modeling and the complexities of Michigan soils. The most commonly applied soil chemical indicators that relate deposition to biological damage (such as BC:Al ratios, Ca:Al ratios,

and maintaining base saturation >20 percent) may or may not be relevant for some tree species in the northern Lower Peninsula, which have adapted over millennia to naturally acidic soils dominated by podzolization. Results of such efforts—combined with a U.S. Forest Service program to continually monitor base saturation before, during, and after harvests and plantings in areas determined to be at high risk—would greatly enhance the environmental analyses and disclosure of impacts to soils and aquatic resources associated with U.S. Forest Service timber management activities.

CONCLUSIONS

Site-specific critical acid loads were modeled for five sites in the Osborn Creek watershed using empirical data as much as possible and the simple mass balance (SMB) approach. On average, the base cation (BC) and nitrogen (N) uptake rates derived in this study were higher than values used in national-scale critical acid load (CAL) estimates by 76.1 and 167.9 eq/ha/yr, respectively (McNulty and Boggs 2010; McNulty et al. 2007, 2013). Considerable variation in BC and N uptake rates was found between and within species in both bark and bole wood. Hardwood species exhibited a greater range of BC and N uptake than conifer species (red pine). Homogenous stands such as red pine plantations have more consistent nutrient values probably because light and space are consistently limited. Species-specific tree nutrient concentration data in this study are within or below the range of concentrations reported in the tree chemistry database (Pardo et al. 2005). Considered in combination, these observations suggest that values for BC_u and N_u are heavily influenced by the method used to calculate tree volume. The method for calculating tree volume in this study is based on stem-only harvest and species-specific volumetric equations (Miles and Hill 2010) using field measurements at five trees per site. Overall, the considerable variation in BC and N concentrations and uptake values demonstrated at the site level in this study suggest that stand characteristics and soil factors have an inseparable connection to growth and uptake, and that assigning a single value to a species across a watershed (let alone a species-group across an entire continent) may not be appropriate. Future research aimed at characterizing the influence of edaphic factors, climate, and stand factors on intraspecies variability in nutrient uptake at multiple scales would be beneficial for larger scale CAL modeling efforts.

Our results point to signs of acidification in the soils studied. Surface horizons, which receive the bulk of atmospheric acid deposition, had pH values of 3.25 to 3.97 at four of the five sites. Buried plow horizons in red pine plantations (sites 1 and 4) exhibited clear signs of humus eluviation (lighter color and lower pH), suggesting that acid deposition—presumably in conjunction with low-molecular-weight organic acid production in the root zone—has facilitated the translocation of humus out of these horizons since the time of planting (~46 years ago). However, despite acidic soil conditions, base saturation remained high to super-saturated in most horizons. The combination of very low weathering rates, droughty conditions during summer months, very low cation exchange capacity, and significant external base cation inputs from a variety of potential offsite sources may be responsible for such high base saturation and exchangeable cation pools at these sites.

Empirical mineral weathering rates using the pedological mass balance method in this study for site 1 (36.2 eq/ha/yr) and site 5 (101.8 eq/ha/yr) were lower than what was determined by national-scale estimates using the soil texture approximation method for the same locations (239.9 eq/ha/yr) (McNulty and Boggs 2010; McNulty et al. 2007, 2013). Although three of the sites (2, 3, and 4) yielded negative weathering estimates, the positive net flux of base

cations at these sites is not considered erroneous, but rather suggests that at present these soils may receive more nutritional support from external BC sources (i.e., atmospheric deposition or groundwater), which could outweigh what is produced through weathering onsite within the rooting zone alone. In such soils, applying the soil texture approximation method may overestimate actual mineral weathering rates. Overall, soils data collected in this study highlight the idea that the SMB model may not fully capture the nuances of nutrient cycling in forest soils with sandy-over-calcareous substrata, periodic base-rich groundwater influence, or a combination of both. CAL modeling in such areas should ideally incorporate into the SMB model additional parameters that estimate the flux of BC through the soil system from upslope sources delivering weathering products to the root zone, shallow groundwater fluxes, and pedogenic or geologic layers tapped by deep-rooted trees.

By using results from national-scale estimates, soils in all watersheds in the Huron-Manistee National Forests were assigned the lowest score of “3—Impaired Function” due to CAL exceedance in the Watershed Condition Framework rating system. This study found that exceedance occurred in soils in the Osborn Creek watershed under worst- and best-case scenarios of N retention, BC and N uptake, and mineral weathering estimates. Despite the high likelihood of exceedance, trees and roots exhibited no clear visual signs of biological damage. Furthermore, base saturation in soils remained high. As such, trees may not be at immediate risk of aluminum toxicity due to excess base cation depletion. However, if current deposition rates and harvesting practices continue in perpetuity, the likelihood of adverse impacts from acidification may increase.

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English Conversion Factors Applicable to Text, Tables, and Figures

When you know:	Multiply by:	Then add:	To find:
centimeters (cm)	0.394		inches
cubic meters	35.3		cubic feet
degrees Celsius (°C)	$1.8 \times ^\circ\text{C}$	32	degrees Fahrenheit
hectares (ha)	2.47		acres
kilometers (km)	0.621		miles
meters (m)	3.28		feet
millimeters (mm)	0.0394		inches
square kilometers (km ²)	0.386		square miles
tonnes	1.102		tons

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Anthropogenic acid deposition has the potential to accelerate leaching of soil cations, and in turn, deplete nutrients essential to forest vegetation. The critical load concept, employing a simple mass balance (SMB) approach, is often used to model this process. In an evaluation under the U.S. Forest Service Watershed Condition Framework program, soils in all 6th level watersheds on the Huron-Manistee National Forests (HMNF) in Michigan were assigned the lowest score of “3—Impaired Function” due to exceedance of the critical load of acidity as determined by national-scale estimates. The impetus for this research was to test the relevance of national-scale critical acid load estimates at the 6th level watershed scale by using site-specific field data in the SMB model where possible.

The Osborn Creek watershed on the HMNF served as a case study. Field data were collected to estimate soil mineral weathering rates, nutrient uptake rates, and forest growth characteristics at five sites containing sandy, nutrient-poor soils. Critical acid loads and exceedances were developed under “best” and “worst” case scenarios given the uncertainty in the SMB model. Despite the high likelihood of actual exceedance and some evidence for soil acidification across the watershed, base saturation remains excessively high (>100 percent) at most sites. Other field data suggest that these soils receive significant external inputs of base cations that may outweigh what is produced through weathering onsite within the rooting zone. Trees show no visible signs of decline.

Overall, the SMB approach may not adequately capture the complexity of nutrient cycling at all of the sample sites. The variability of soils, weathering estimates, and nutrient uptake rates between and within sites makes extrapolation of these results to other HMNF watersheds difficult to justify. Management programs aimed at improving our understanding of base cycling in complex glacial terrain, as well as mitigating the risks associated with nutrient depletion from frequent timber harvests and fuels reduction practices, are suggested.

KEY WORDS: soil acidification, critical load, simple mass balance, weathering rate, nutrient cycling

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