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The application of an integrated biogeochemical model to simulate dynamics of vegetation, hydrology and nutrients in soil and streamwater following a whole-tree harvest of a northern hardwood forest

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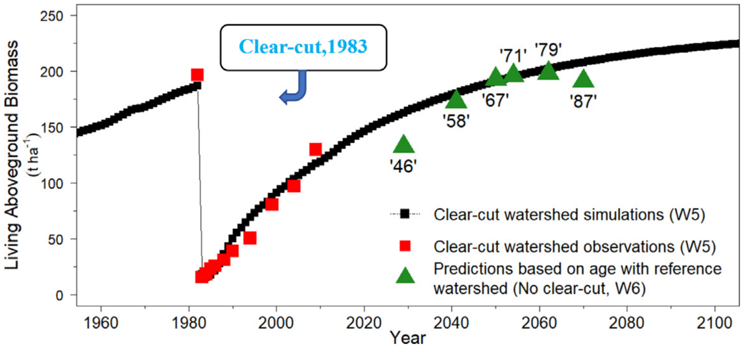
HIGHLIGHTS

- PnET-BGC was used to quantify biogeochemical effects of forest cutting.
- Model revisions improved performance in quantifying harvesting effects.
- Simulations depict coupling of net mineralization with plant uptake regulate recovery.
- Model simulations are more sensitive under mature forest conditions than aggrading growth.
- Findings are compared with other studies and models in the literature.

GRAPHICAL ABSTRACT



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ABSTRACT

Understanding the impacts of clear-cutting is critical to inform sustainable forest management associated with net primary productivity and nutrient availability over the long-term. Few studies have rigorously tested model simulations against field measurements which would provide more confidence in efforts to quantify logging impacts over the long-term. The biogeochemical model, PnET-BGC has been used to simulate forest production and stream chemistry at the Hubbard Brook Experimental Forest (HBEF), New Hampshire, USA. Previous versions of PnET-BGC could accurately simulate the longer-term biogeochemical response to harvesting, but were unable to reproduce the marked changes in stream NO_3^- immediately after clear-cutting which is an important impact of forest harvesting. Moreover, the dynamics of nutrients in major pools including mineralization and plant uptake were poorly predicted. In this study, the model was modified and parametrized allowing for a lower decomposition rate during the earlier years after the clear-cut and increased NH_4^+ plant uptake with the regrowth of new vegetation to adequately reproduce hydrology, aboveground forest biomass, and soil solution and stream water chemistry in response to a whole-tree harvest of a northern hardwood forest watershed (W5) at the HBEF. Modeled soil solution and stream water chemistry successfully captured the rapid recovery of leaching nutrients to pre-cut levels within four years after the treatment. The model simulated a substantial increase in aboveground net primary productivity (NPP) from around 36% to 97% of pre-cut aboveground values within years 2

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to 4 of the cut, which closely reproduced the measured values. The projected accumulation of aboveground biomass 70 years following the harvest was almost 190 t ha^{-1}, which is close to the pre-cut measured value. A first-order sensitivity analysis showed greater sensitivity of projections of the model outputs for the mature forest than the strongly aggrading forest.

1. Introduction

Forest harvesting causes short- and long-term changes to the internal processes of forest ecosystems, altering hydrology, biogeochemical processes, energy flux and species composition (Hume et al., 2018; Parolari and Porporato, 2016; Yanai et al., 2003). Few modeling studies have compared their simulations with the field measurements to examine short and long -term effects of harvesting (Bu et al., 2008; Mina et al., 2017; Shifley et al., 2017; Wei et al., 2003). The biogeochemical model, PnET-BGC has previously been used to simulate northern hardwood watersheds at the Hubbard Brook Experimental Forest (HBEF), New Hampshire, USA for the various applications (Aber et al., 1997; Aber and Federer, 1992; Gbondo-Tugbawa et al., 2001; Pourmokhtarian et al., 2012). However, the previous versions of PnET-BGC have been unable to effectively reproduce the marked changes in nutrients concentrations in soil and stream water immediately following harvesting and the dynamics of nutrients in major pools including mineralization and plant uptake (Aber et al., 1997; Aber and Driscoll, 1997). This paper is the first effort to test and modify the PnET-BGC model to be used as a tool to effectively quantify short- and long-term effects of forest clear-cutting.

Ecosystem response to logging disturbance varies significantly, depending on site conditions, forest species composition, land use history, and the method and frequency of harvesting (Clarke et al., 2015; Hume et al., 2018; Levers et al., 2014). Despite this, the long-term effects of forest management practices on nutrient cycles remain poorly understood (Hume et al., 2018; Martin et al., 2015; Parolari and Porporato, 2016; Shifley et al., 2017). Most studies have shown changes in biogeochemical processes following forest cutting in response to changes in species composition, root uptake demand, soil conditions and microbial activity, leading to a short-term (1–3 year) increase in nutrient concentrations in soil solutions and export to receiving waters (Kreutzweiser et al., 2008; Nave et al., 2010; Thiffault et al., 2011). However, some studies have reported little or no change in soil chemical properties (Keenan and Kimmins, 1993; Kreutzweiser et al., 2008). The long-term implications of harvesting of forest ecosystems are less clear due to limited information on land use history and lack of time series observations (Kreutzweiser et al., 2008; Thiffault et al., 2011).

Process models have been developed to quantify the structure and function of terrestrial ecosystems in response to disturbance. Such models are effective tools that can provide insight into transformations of important elements that regulate the structure and function of forest ecosystems. For example, FORCAST (Forest Canopy Atmosphere Transfer) was used to evaluate long-term site productivity and nutrient loss of timber-harvesting of a lodgepole pine forest in the central interior of British Columbia (Wei et al., 2003). CENTURY 4.0 (Grassland and agroecosystem dynamics model) was applied to examine depletion of biomass carbon stocks, woody litter and soil carbon under long-term, frequent and intensive harvesting of boreal forests in China and central Canada (Jiang et al., 2002). Rolff and Ågren (1999) modeled the dynamics of nitrogen in managed (different harvesting intensities) Norway spruce stands in Scandinavia using NITMOD (Nitrogen Model), concluding that cumulative nitrogen leaching and a reduction in aboveground biomass production were a consequence of long-term, frequent and intensive logging. Sustainable management of forest ecosystems to conserve nutrients, while maintaining productivity and nutrient availability was a major concern in all these studies.

Logging is thought to have a significant effect on forest floor structure (Hartmann et al., 2012; Palviainen et al., 2004; Yanai et al., 2003). The curve of Covington (1981) has been used to infer the loss of soil organic matter and nutrient budgets due to accelerated decomposition (Aber and Federer, 1992; Covington, 1981; Hartmann et al., 2012). However, some studies have shown increases in the mass of soil organic matter within the years after harvesting attributed to the lower decomposition rates (Mattson and Smith, 1993; Prescott, 2005; Prescott et al., 2000; Yanai et al., 2003). Variations in the response of soil organic matter storage with cutting at different sites may be due to variations in the mixing of the forest floor into the mineral soil, soil disturbance with harvesting operation, the dynamic of woody debris, change in litter quality and quantity, alterations in microbial community, increased leaching of dissolved organic carbon or CO_2 release into atmosphere, and climate change (Johnson et al., 1985; Liu, 2014; Ryan et al., 1992; Yanai et al., 2003).

Various experimental and modeling studies have been conducted at Hubbard Brook to investigate the biogeochemical response of forests to harvesting. Dahlgren and Driscoll (1994) examined patterns in the concentrations of major elements in soil solutions and stream water after a whole-tree harvest of Watershed 5 (W5) at the HBEF, documenting acidification of stream water and potentially toxic concentrations of dissolved inorganic aluminum. Johnson et al. (1991) concluded there was no substantial change in the total pool of soil organic matter or the quantity of exchange sites in the solum three years after W5 was cut. Dib et al. (2014) compared the ability of the RothC (Rothamsted Carbon Model) and CENTURY models to simulate changes in soil organic carbon (SOC) pools following harvest of W5; the models predicted minimum SOC after 45 and 14 years, respectively, compared with a minimum measured value after 8 years. Based on published studies, Federer et al. (1989) concluded that depletion of soil calcium and other nutrients with repeated harvesting of various eastern US forests would limit long-term sustainable forestry.

Poor knowledge of how biological, geochemical and hydrological processes change during the recovery from ecosystem disturbance, poor quantitative understanding of land use history and limited time series observations challenge the development and testing of models used to simulate effects of forest harvesting (Mina et al., 2017; Parolari and Porporato, 2016; Shifley et al., 2017). Fortunately, studies of the experimental whole-tree harvest of W5 at the HBEF provide detailed, comprehensive field measurements to use in evaluating and revising the PnET-BGC model to assess both short- and long-term effects of logging regimes and improve future projections.

The primary goal of this study was to evaluate, modify if necessary and apply the biogeochemical model, PnET-BGC, to simulate short- and long-term hydrologic and biogeochemical response of a temperate deciduous forest watershed to a whole-tree harvest disturbance. In future studies, the parameterized/modified model will be applied to the other forest watersheds that have undergone various harvesting regimes to further test the model and gain confidence in performance. The verified model will be ultimately used in the simulation of hypothetical harvesting scenarios to quantify short- and long-term effects on watershed hydrology and nutrient stores to inform practices of sustainable forest management.

The specific objectives of this analysis were to: (1) simulate aboveground biomass accumulation and net primary productivity (NPP) following forest cutting on W5 at the HBEF; (2) evaluate model processes responsible for the prediction of nutrient leaching losses and modify the algorithms as needed to improve model performance; (3) project monthly and annual stream water hydrology, soil solution chemistry, stream water chemistry and nutrient budgets; (4) compare model simulations with experimental observations and reference measurements on W5; and (5) conduct an input parameter sensitivity analysis for both pre- and post-treatment periods.

2. Methodology

2.1. Site description

The HBEF is located in southern White Mountains of New Hampshire (43°56′ N, 71°45′ W). The site was established by the U.S. Forest Service in 1955 to improve understanding of the response of northeastern US temperate forests to management through monitoring and field experiments. The HBEF has ten watersheds with long-term biogeochemical records, the earliest of which began in 1963. More detail about the HBEF can be found in supplemental information of this paper as well as in previous studies (Bormann and Likens, 1979; Likens et al., 1970). In this paper, we present data from two experimental watersheds at HBEF. Watershed 6 (W6), with an area of 13.2 ha and an elevation range of 549–792 m, serves as the biogeochemical reference watershed, without experimental manipulation. W5 is adjacent to W6 with area of 22.5 ha and elevation range of 488–762 m. W5 was subjected to whole-tree harvest during the fall of 1983 through the winter of 1984 (Table A1, Fig. A1).

2.2. Model description

PnET-BGC is an integrated biogeochemical model, developed to assess the effects of atmospheric deposition, land disturbance and climatic conditions on vegetation, soils and surface waters, primarily in forest ecosystems (Chen and Driscoll, 2005; Gbondo-Tugbawa et al., 2001). For the simulation of W5, we used a version of PnET-BGC that considers multiple-layers of soil, depicting surface organic ("O") and mineral B horizons to form the upper layer and the C horizon as the lower layer, in order to better capture seasonal variation in stream discharge and chemistry (Chen and Driscoll, 2005). This version considers hydrological characteristics that determine water exchange between the two layers, and also utilizes different weathering rates and soil properties for each layer. More detail about the model can be found in the supplementary information and previous studies (Aber et al., 1997; Aber and Driscoll, 1997; Aber and Federer, 1992; Gbondo-Tugbawa et al., 2001).

2.3. Data preparation and model inputs

2.3.1. Meteorological and atmospheric deposition data

Meteorological data (photosynthetically active radiation, precipitation, maximum and minimum temperature) and atmospheric deposition (dry and wet) vary monthly over the simulation period. Direct measurements of these parameters are available since the initiation of the HBEF in the mid-1950s and 1960s (Table A2 in the supplemental information provides a summary of the data sets used in this study). For the period prior to measurements (1850–1963), we used reconstructed input data (Chen et al., 2004; Fakhraei et al., 2016, 2014; Fakhraei and Driscoll, 2015). The reconstruction of atmospheric wet deposition is described in Tables A3 and A4. PnET-BGC uses dry deposition calculated from dry to wet deposition ratios for each element (Table A5).

2.3.2. Hydrology, weathering and soil data

Hydrological parameters for upper and lower soil layers were calculated based on an end-member mixing and flow analyses (Chen and Driscoll, 2005). Effects of whole-tree clear-cutting on soil processes were observed for the pre-treatment (1983) and over the post-cut period (1984–1997) using chemistry data from zero-tension lysimeters in Oa (3–6 cm below surface of forest floor), Bhs and Bs horizons (19–26 and 40–49 cm beneath the surface of the mineral soil, respectively) in three elevation zones (low and high elevation deciduous forest and high-elevation coniferous forest) (Dahlgren and Driscoll, 1994; Johnson et al., 1991, 1997). Soil chemical data are available for pre-treatment (1983) and three post-treatment years (1986, 1991, 1997) (Johnson et al., 1991, 1997). Model simulations for stream water hydrology and chemistry were compared with the measured data during 1963–2013 (Likens, 2017).

Weathering rates for the upper layer were estimated through calibration using soil solution for the Bs2 horizon (Chen and Driscoll, 2005; Dahlgren and Driscoll, 1994; Nezat et al., 2004). For the lower soil layer, weathering rates were obtained by calibration using stream water chemistry. Note that the sum of the weathering rates for the two soil layers is equivalent to the soil weathering rate for the entire soil profile used in previous model simulations with a single (lumped) soil layer (Gbondo-Tugbawa et al., 2001). Parameters and variables used in the model calibration are summarized in Table A6 in the supplemental information. A detailed description of the model and its parameters can be found in Aber et al. (1997, 1995) and Gbondo-Tugbawa et al. (2001).

2.3.3. Vegetation parameters

PnET-BGC uses site-specific vegetation parameters. Values of these parameters for northern hardwood tree species are obtained from direct field measurements, values in the literature or model calibration (Aber et al., 1997; Aber and Driscoll, 1997; Aber and Federer, 1992). Model simulations are known to be highly sensitive to the minimum nitrogen concentration in foliar litter. This parameter is used as an input to simulate the amount of N allocated to plant bud for foliage production of the following year. This input parameter assures the simulated foliar nitrogen concentrations above the reported minimum nitrogen concentration in foliar litter of 0.8%–0.9% (Aber et al., 1997; Aber and Driscoll, 1997). With this parametrization, modeled foliar nitrogen concentration were comparable to observations, ranging from below 2% to 2.4% in mixed stands (hardwood-conifer) (Aber et al., 1996). Foliar nitrogen concentration is used to predict the rate of photosynthesis and the pattern of biomass accumulation through the simulation (Aber et al., 1997). Root production contributes significantly to the soil organic matter pool. The default procedure in the model, which allocated twice the monthly foliar carbon production as monthly increments of root carbon, underpredicted the soil organic carbon pool in simulations. To reproduce the soil organic carbon pattern at the HBEF, it was necessary to increase root carbon allocation to three times foliar production (Table A6 in supplemental information); this condition may reflect the apparently high rhizosphere carbon flux (Phillips and Fahey, 2005) and root turnover (Tierney and Fahey, 2002) observed in this forest.

2.3.4. Land use history

The HBEF was selectively logged for red spruce in the 1880s and then logged intensively from 1910 to 1917. The areas comprising W5 and W6 experienced some salvage removal following the hurricane of 1938 and damage from an ice storm in 1998. However, there are limited data on biomass impacts from the historical logging events. As a result, historical tree mortality and removal percent estimated for use in the previous simulations (Aber and Driscoll, 1997; Gbondo-Tugbawa et al., 2001) were also applied for this analysis. We used detailed information on Hubbard Brook website (https://hubbardbrook.org/watersheds/watershed-5) to estimate the percent of forest biomass mortality and removal for the whole-tree harvest of W5 in winter 1983–1984. All living trees above 2 cm diameter at breast height (dbh) were cut and those >10 cm dbh were removed from the watershed. 8% of large trees were not cut, but left in a buffer strip along the watershed boundary. Moreover, dead trees above 10 cm dbh and felled trees on steep slopes were not removed and left as slash on the cut watershed (13%). For simulations, we estimated mortality of 92% of forest biomass and 87%

removal of aboveground biomass from the watershed (https://hubbardbrook.org/watersheds/watershed-5; Johnson et al., 1995) (Table A1).

2.3.5. Biomass studies and calculations

A total forest inventory was conducted in W5 during midsummer of 1982 to quantify biomass prior to the cut. We used the dimensional analysis method of Whittaker et al. (1974) with revised allometric equations (Siccama et al., 1994) based on parabolic volume (PV), which consider tree height in addition to diameter, to estimate aboveground biomass. Calculated biomass values are comparable with other results at the HBEF (Fahey et al., 2005; Johnson et al., 1995; https://hubbardbrook.org/watersheds/watershed-5). In order to calculate total aboveground biomass on W5 at the time of the cut in 1983, we used an annual average biomass increment on W6 (2.98 t $\mathrm{ha}^{-1}\ \mathrm{yr}^{-1}$) over the 1977–1982 period, and applied this value to the 1982 W5 biomass estimate to allow for 1.5 years of growth from the time of inventory to the cut (https://hubbardbrook.org/watersheds/watershed-5).

Post-harvest, the sampling approach for forest biomass was adjusted through the time to accommodate the greatly changing sizes and density of trees in the watershed (Cleavitt et al., 2018). In the first post-harvest tree survey in 1990, we measured trees ≥1.5 cm DBH, and composition and abundance were assessed in 199 random transects, 1 m by 25 m in area. In the 1994, 1999, and 2004 surveys, we measured trees ≥1.5 cm DBH in random 1 m by 25 m transects (38 in 1994; 101 in 1999 and 2004). In 2009, tree surveys were conducted in a total of 101 random circular plots; trees were sampled in two size classes; individuals ≥7.5 cm DBH were measured in a 100 m^2 area and smaller trees (1.5 cm ≤ DBH < 7.5 cm) were measured in a nested 30 m^2 transect.

2.4. Rationale for PnET-BGC modifications to simulate harvesting effects

Clear-cutting can cause substantial changes to the internal processes of forest ecosystems (Bormann and Likens, 1979). Aggrading forest ecosystems can regulate biotic and abiotic processes especially nutrient uptake, organic matter mineralization, nitrification, and transpiration. The original version of PnET-BGC could accurately simulate the longer-term biogeochemical response to harvesting, but, was unable to adequately depict key biotic and abiotic processes immediately after clear-cutting that determine nutrient leaching from soil to stream water. Various processes interact to minimize nutrient losses from the ecosystem after disturbance, including the rapid regrowth of fast-growing, shade-intolerant tree species such as pin cherry (*Prunus pensylvanica*), white birch (*Betula papyrifera*) and trembling aspen (*Populus tremuloides*) (Marks, 1974), increased evapotranspiration (Hornbeck et al., 1997) and a reduction in decomposition rate through alteration of microclimate over the growing season (Bormann and Likens, 1979; Marks and Bormann, 1972; Yanai et al., 2003).

A number of factors were evaluated to reconcile the overprediction of nutrient loss during the period immediately following the whole-tree harvest (Fig. 1a,b) including: 1) inaccurate prediction of nutrient uptake due to inaccurate simulation of regrowing plant biomass; 2) inaccurate prediction of nutrient uptake due to inaccurate stoichiometry of regrowing biomass (Marks, 1974); and 3) inaccurate decomposition rate of soil organic matter when biomass and litter input are low (Ryan et al., 1992; Yanai et al., 2003).

In order to calibrate biomass, we parameterized the model for vegetation parameters which determine canopy photosynthesis rate, to better reproduce aboveground biomass accumulation. However, this modification did not enhance model performance for the short-term simulation of stream water chemistry. We then examined whether underprediction of plant nutrient uptake might be responsible for the

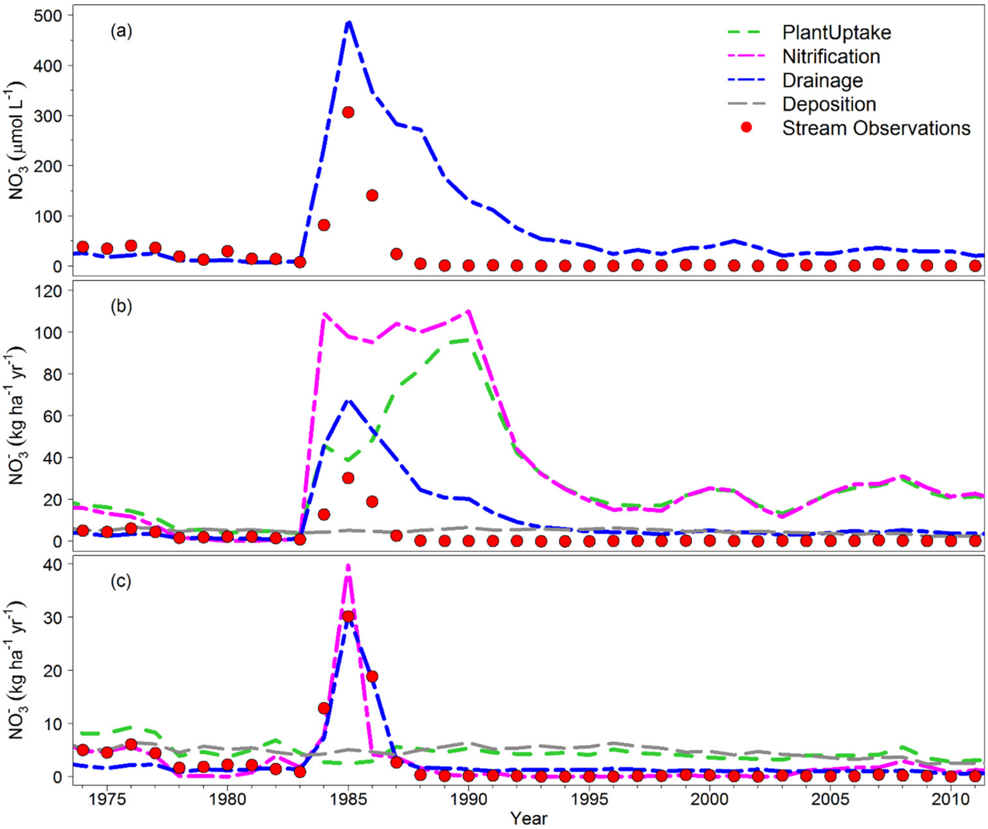

Fig. 1. Comparison of model simulations with observations for annual volume-weighted NO_3^- concentrations in stream water (a) and modeled NO_3^- budget flux for before (b) and after (c) model algorithm modification and parameterization.

overprediction of stream nutrient export, possibly due to errors in vegetation tissue element stoichiometry. However, vegetation stoichiometry values used in model simulations were in the range of measured values (Marks, 1974; Marks and Bormann, 1972; Whittaker et al., 1979), indicating an acceptable accuracy of the stoichiometry of regrowing biomass used in the model.

Note that in the original simulations there was a prolonged period of elevated nitrification that continued for approximately a decade after the clear-cut (Fig. 1b). In order to improve model performance for short-term simulations, we hypothesized that when the forest is young with limited litter input, there would be a decrease in the decomposition rate of organic matter. Note, PnET-BGC assumes a single soil organic matter pool with a constant turnover rate which is equivalent to the slow pool in Century model developed for the mature forests (Aber et al., 1997; Parton et al., 1993). At the time of disturbance, the model assumes all slash left on the site and dead root biomass from the harvest are added to the soil organic pool to be mineralized at a constant turnover rate. A reduction in decomposition rate slows nutrient mineralization from the soil organic pool, decreasing leaching of elements into soil solutions and stream water. This observation of a short-term decrease in soil organic matter decay is consistent with field observations conducted at various sites (Johnson et al., 1995; Prescott et al., 2000; Ryan et al., 1992; Yanai et al., 2003).

Under this reduced decomposition rate, coupled with the rapid regrowth of vegetation, plant demand for ammonium was able to keep pace with soil nitrogen mineralization, resulting in less nitrification in model simulations. We parametrized the model with lower rates of decomposition for the years immediately after the cut (years 2 to 4) to mitigate the over-prediction of nutrient leaching (Table A6 in supplemental information). However, this parametrization was not adequate to capture observations. We also modified the algorithm that calculates nitrification rate by adding a new constant parameter (Table A6) that allows for greater ammonium assimilation by plants and thereby reduces leaching of NO_3^-. We also modified model algorithms to depict plant nutrient uptake as a function of regrowing biomass in order to capture the observed plant nutrient uptake values. These parametrization/modifications of the algorithms improved the model performance so that it was better able to simulate stream chemistry and plant uptake in the years immediately after the clear-cut (Figs. 2c,5 and Table A6).

2.5. Model evaluation and sensitivity analysis

Three statistical methods were used to evaluate model performance before and after the harvest: normalized mean error (NME) normalized mean absolute error (NMAE) and normalized root mean squared error (NRMSE) (Alewell and Manderscheid, 1998; Janssen and Heuberger, 1995). NME reflects an average bias value for model predictions. A negative NME value indicates underprediction and a positive value indicates overprediction. NMAE and NRMSE are good estimators of model performance in capturing observed trends. NRMSE gives a relatively high weight to large errors since the errors are squared before they are averaged.

A first order sensitivity analysis was conducted to identify the input parameters that most affect the response variables of considerable interest in model calculations (Y). This analysis was conducted for periods before and 5 years after the cut to compare model sensitivity for the mature and strongly aggrading forest, respectively. For this analysis, we examined relative change in the state variable of interest (Y_i) divided by the relative change in input factor (X_i) (Jørgensen and Bendoricchio, 2001). Therefore, sensitivity of a parameter ($S^{Y}{}_{Xi}$) is represented as follows:

$$S^{Y}{}_{Xi} = \frac{\partial Y/Y}{\partial X_i/X_i} \tag{1}$$

A higher $S^{Y}{}_{Xi}$ value reflects greater sensitivity of model projections to an input of interest. Previous analyses have not evaluated model sensitivity for a strongly aggrading forest, but have considered model sensitivity in simulations of mature forest conditions (Aber et al., 1996; Fakhraei et al., 2017; Gbondo-Tugbawa et al., 2001).

3. Results and discussion

3.1. Comparison of performance before and after model modifications

The initial version of PnET-BGC was able to accurately predict the long-term biogeochemical response to logging, but was unable to adequately depict the biotic and abiotic processes immediately after the clear-cut which determine the extent of nutrient leaching from the soil to streamwater (Fig. 1a,b). Simulations of annual volume-weighted concentrations of stream NO_3^- were compared with observations (Fig. 1a) and watershed NO_3^- flux simulations were compared before (Fig. 1b) and after model modifications (Fig. 1c). The original model could not adequately reproduce the relatively rapid decline of stream NO_3^- that followed the initial pulse of leaching after clear-cutting (within 4 years); the dynamics of nutrients in major pools including mineralization and plant uptake also were poorly predicted. However, with the modifications to the model, the pattern of NO_3^- fluxes in major pools, including stream output and plant uptake, became consistent with measurements over the simulation period (Fig. 1c). To avoid repetitive plots, we show modeled stream NO_3^- concentrations after model modifications in Fig. 4. Similar problems with short-term

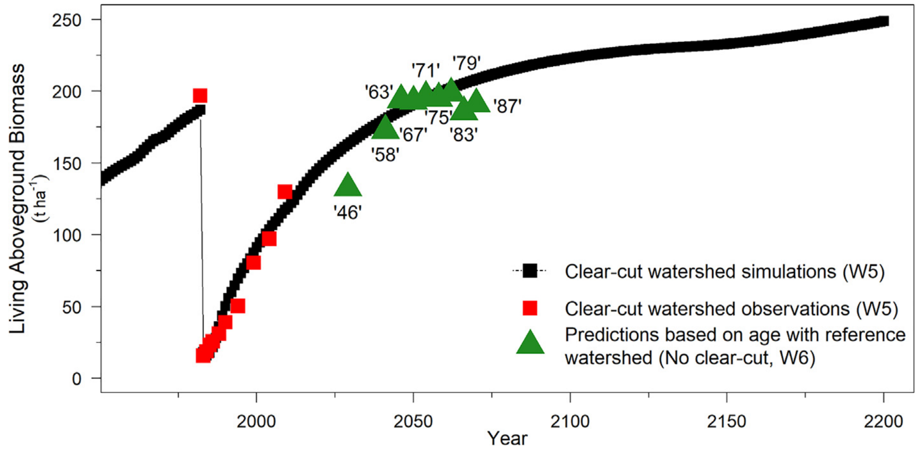

Fig. 2. Simulations of living aboveground biomass accumulation, Watershed 5, HBEF, for the period before and after the whole-tree harvest. Model simulations are compared with measured values for Watershed 5 and Watershed 6 adjusted to years after cutting.

simulations were evident for other elements, without model modification. Model projections were greatly improved after model modifications particularly for NO_3^- (by 90%), SO_4^{2-} (by 40%), Ca^{2+} (by 70%), Mg^{2+} (by 60%) and inorganic monomeric Al (Al_m) (by 75%) over the 1985–95 simulation period (Fig. A2 in supplemental information).

The principal model modifications that improved performance in depicting nutrient dynamics were a reduction in the rate of organic matter decomposition and nutrient mineralization after clear-cutting and an increase in the rate of vegetation assimilation of ammonium, which reduced the soil nitrification rate. Unfortunately, there are no direct field measurements of decomposition/mineralization rates for the period immediately after the cut to compare with the modeled rates. However, our model modifications/parametrizations are consistent with studies which show a reduction in decay rate immediately after logging (Johnson et al., 1995; Liu, 2014; Prescott, 2005; Ryan et al., 1992; Yanai et al., 2003). Although early studies at the HBEF suggested rapid loss of forest floor organic matter after forest harvest (Covington, 1981), subsequent research on W5 showed that this “loss” was in part due to physical mixing of organic matter into mineral soil during harvest (Johnson et al., 1991, 1995; Ryan et al., 1992) or possibly because of increased dissolved organic carbon (DOC) leaching (Johnson et al., 1995). As a result, actual organic matter decay was slower than what might be suggested in Covington (1981) (Yanai et al., 2003).

Moreover, rapid plant N assimilation may be related to the effects of species composition on nutrient uptake. Notably, PnET-BGC does not directly account for the changes in tree species composition. It is possible that the increase in fast-growing, early-successional species like pin cherry, birch and aspen on the cut watershed (Mou et al., 1993) contributes to the higher ammonium assimilation following the harvest. Nevertheless, the modified model can be used effectively as a tool to compare model simulations with available observations, provide insight into unmeasured biotic processes following the cut and make long-term simulations of alternative harvesting strategies. This analysis also demonstrates the necessity for additional measurements in the future to further test model performance.

3.2. Vegetation simulations

The model generally performed well in the simulation of aboveground biomass for both pre-harvest and post-harvest conditions (Fig. 2). Modeled aboveground biomass matched well with the observed value for the pre-cut year (1983). In the first few years after the cut, re-growing vegetation consisted mainly of herbs, shrubs and tree seedlings and sprouts (Johnson et al., 1995). Six years after the harvest, simulated aboveground biomass of 43 t ha^{-1}, compared well with the observed value of 40 t ha^{-1}, which represents approximately 22% of the pre-cut forest biomass. At 16 and 26 years after the clear cut, simulated aboveground biomass increased to 79 and 118 t ha^{-1}, respectively, corresponding with observed measurements of 80 and 130 t ha^{-1} respectively, approximately 46% and 61% of aboveground biomass prior to the cut. We project that W5 would reach the aboveground biomass value prior to the cut in about 70 years (190 t ha^{-1}) (Fig. 2). Differences between W5 modeled biomass and observations might be due to the inability of PnET-BGC to depict shifts in tree species following the harvest.

Comparison with the reference watershed (W6), which is a second-growth forest, reveals that mid-rotation projections of W5 biomass accumulation are consistent with the expected growth trajectory. Discrepancies between W5 and W6 aboveground biomass might be explained by the site differences, species composition, and uncertainty in measurements. Longer-term model projections of the aboveground biomass accumulation patterns are generally consistent with other studies (Bormann and Likens, 1979; Jiang et al., 2002; Rolff and Ågren, 1999; Wei et al., 2003). Long-term simulations using JABOWA (Northeastern forest growth simulator) that depicts four separate phases of regrowth designated as reorganization, aggrading, transition and steady state of the ecosystem, estimated roughly 300–400 t ha^{-1} total biomass by the steady state phase, 500 years after forest clear-cutting (Bormann and Likens, 1979); however, more recent measurements of old-growth forests in the region call into question the steady-state projections of JABOWA (Keeton et al., 2011). Simulations of PnET-BGC projected 250 t ha^{-1} of aboveground biomass accumulation for the shorter simulation period by the year 2200. Rolff and Ågren (1999) used the ecosystem model, NITMOD, to project maximum aboveground biomass of Norway spruce stands at around 250 t ha^{-1} in highly productive stands, 220 t ha^{-1} in medium productive stands and approximately 100 t ha^{-1} for the least productive sites.

The simulated pattern of increasing aboveground net primary productivity (NPP) shortly after cutting agreed with observations across different cut stands in the vicinity of the HBEF (Fig. 3), although there was some deviation from the measured values 5 and 6 years after harvesting (Bormann and Likens, 1979). Within 2 years after the clear-cut, simulated NPP rapidly increased, achieving 36% of the value of a 55-year-old stand, comparable with the observed value (Fig. 3). After around 10 years, aboveground NPP reached a near steady state value of around 836 g m^{-2} yr^{-1} which is close to the average observed value of a 55-year-old stand (Bormann and Likens, 1979) and the pre-cut value.

3.3. Stream hydrology

Hydrology plays an important role in the loss of nutrients. Hydrologic output is largely influenced by variations in precipitation quantity and evapotranspiration during the growing season. The modeled annual stream flow adequately captured observed values over the study period 1964–2012, with slight overprediction for the pre-harvest period ($NME_b = 0.08$, $NMAE_b = 0.09$ and $NRMSE_b = 0.1$, Table 1, Fig. 4) and an underprediction for the post-harvest period ($NME_a = -0.01$, $NMAE_a = 0.06$ and $NRMSE_a = 0.07$, Table 1, Fig. 4). Stream flow at W5 was 15% greater than W6 in the first year after the cut (1984) due to a decrease in transpiration. Moreover, simulated stream flow for W5 indicated a 61% increase from the pre-cut year (1982) to the first year after the clear-cut (1984) compared with an increase of 57% for the measured values. Over the intermediate term (13–30 years) a relatively small decrease in discharge from cutover watersheds at HBEF has been attributed to higher transpiration rates associated with increased stomatal conductance of early successional than mature forest species (Hornbeck et al., 1997); however, as noted (Section 3.1) because the PnET-BGC model does not simulate changes in forest species composition, this pattern was not captured in the simulations (Fig. 4).

3.4. Streamwater chemistry simulations

After revising algorithms and parameters in PnET-BGC, simulations of surface water constituents sufficiently captured observations for the pre-harvest and post-harvest periods. If NO_3^- is not assimilated by vegetation, it acts as a relatively conservative solute and facilitates the mobilization of major cations from soil (Johnson and Cole, 1980). As noted earlier, the model satisfactorily reproduced peak values of the concentrations and losses of NO_3^- to streamwater following the harvest of W5, and the rapid recovery of watershed NO_3^- retention. The simulated increase in the mean volume-weighted stream concentration of NO_3^- (146 μmol L^{-1}) agreed with the mean observation (176 μmol L^{-1}) over the 1984–86 period (Fig. 4). Despite the improvement in simulation of stream NO_3^- after the cut with model modifications, metrics of model performance indicate that NO_3^- remains a challenging solute to simulate (Table 1).

The model satisfactorily reproduced the long-term pattern (1964–2012) of basic cation concentrations in stream water as indicated by low performance criteria values. For example, the simulated annual volume-weighted concentration of stream water Ca^{2+} nearly

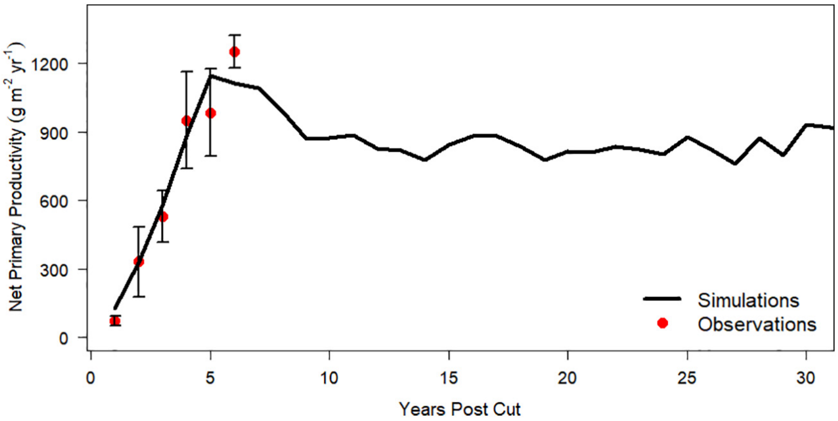

Fig. 3. Comparison between simulated aboveground net primary productivity of WS5 and measurements from different clear-cut stands in the vicinity of the HBEF. Data shown from (Bormann and Likens, 1979).

matched the measured values for post-cut and pre-cut periods, with only slight overprediction ($NME_b = 0.02$ and $NME_a = 0.04$ Table 1). Simulated stream concentrations of Ca^{2+} also satisfactorily captured the pattern in observations before and after the clear-cut ($NRMSE_b = 0.1$ and $NRMSE_a = 0.08$). In the years immediately following the clear-cut (3 years), the model slightly overestimated the measured values, but reproduced the rapid increase in the leaching of stream Ca^{2+} (simulated mean of 59 vs observed mean of 52.3 μmol L^{-1}). The model was also able to sufficiently depict stream Na^+ concentrations (pre-treatment: $NME_b = 0.03$, $NMAE_b = 0.06$, $NRMSE_b = 0.08$; post-treatment: $NME_a = -0.06$, $NMAE_a = 0.06$, $NRMSE_a = 0.08$). Stream Mg^{2+} concentrations were slightly overpredicted during pre-treatment period ($NME_b = 0.13$, $NMAE_b = 0.13$, $NRMSE_b = 0.14$; Table 1) and closely captured during post-treatment period ($NME_a = 0$, $NMAE_a = 0.04$, $NRMSE_a = 0.05$). PnET-BGC closely depicted the immediate increase in stream concentrations of Na^+ and Mg^{2+} following logging (1984–1986), simulating mean values of 40 and 23 μmol L^{-1} compared with measured values of 38.5 and 21.8 μmol L^{-1} for Na^+ and Mg^{2+}, respectively.

In the model, pH is calculated from the difference in charge balance between major cations and anions and equilibrium reactions of dissolved inorganic carbon, aluminum and organic acids (Fakhraei and

Table 1
Comparison of modeled and observed values of stream constituents and model performance for the periods prior to and after whole-tree harvesting[a].

Stream constituents	Pre-harvest (1966–82)						
	Mean		STD		NME	NMAE	NRMSE
	Observed	Simulated	Observed	Simulated			
Flow	85.34	92.10	20.48	16.37	0.08	0.09	0.11
pH	4.90	5.10	0.10	0.07	0.04	0.04	0.05
DOC[b]	–	242.69	–	42.32	–	–	–
Na^+	33.13	33.97	3.07	3.91	0.03	0.06	0.08
Mg^{2+}	13.81	15.61	1.86	1.65	0.13	0.13	0.14
Al_m[c]	–	5.44	–	0.54	–	–	–
Ca^{2+}	35.53	36.14	4.50	3.16	0.02	0.09	0.10
NO_3^-	27.04	12.81	11.73	3.30	−0.53	0.53	0.62
SO_4^{2-}	60.20	57.30	2.75	2.55	−0.05	0.05	0.05
Al_{om}[d]	–	2.97	–	0.41	–	–	–
ANC[e]	–	5.96	–	2.63	–	–	–
Stream constituents	Post-harvest (1988–2012)						
	Mean		STD		NME	NMAE	NRMSE
	Observed	Simulated	Observed	Simulated			
Flow	97.26	96.01	22.74	19.57	−0.01	0.06	0.07
pH	5.39	5.41	0.06	0.10	0.00	0.01	0.02
DOC[b]	223.72	207.61	30.57	48.45	−0.07	0.19	0.21
Na^+	30.10	28.23	2.08	1.65	−0.06	0.06	0.08
Mg^{2+}	10.69	10.65	2.00	1.80	0.00	0.04	0.05
Al_m[c]	3.85	5.00	0.81	0.57	0.30	0.32	0.35
Ca^{2+}	25.86	26.87	5.39	4.57	0.04	0.07	0.08
NO_3^-	1.19	8.44	1.00	3.55	6.09	6.40	6.88
SO_4^{2-}	45.77	44.87	7.20	6.03	−0.02	0.04	0.05
Al_{om}[d]	2.30	2.63	0.57	0.50	0.14	0.25	0.27
ANC[e]	6.07	8.43	4.31	2.30	0.39	0.65	0.86

[a] Values represent mean and standard deviation of annual volume-weighted concentrations for the pre-harvest (1966–82 year) and post-harvest (1988–2012 year) periods. Units for stream constituents are μmol L^{-1} (ANC; μeq L^{-1}, Flow; cm). NEM: normalized mean error. NMAE: normalized mean absolute error. NRMSE: normalized root mean squared error. STD: Standard deviation.

[b] DOC: Dissolved organic carbon.

[c] Al_m: Total monomeric Al.

[d] Al_{om}: Organic monomeric Al.

[e] ANC: Acid neutralizing capacity.

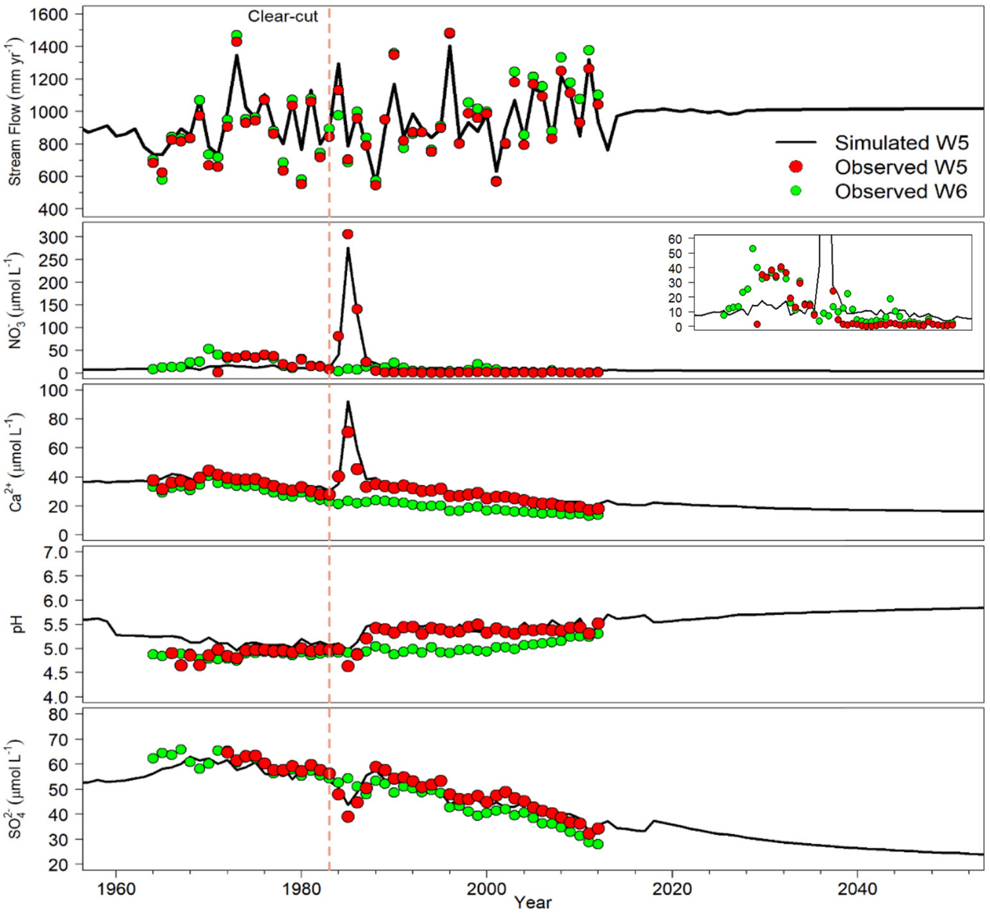

Fig. 4. Comparison between streamflow and annual volume-weighted stream water chemistry from PnET-BGC simulations and observations for W5, HBEF. The results are also shown with stream water chemistry for the reference watershed (W6).

Driscoll, 2015). Devegetation enhances the leaching of nutrients, particularly NO_3^-, causing decreases in pH. With regrowth of vegetation and exhaustion of the supply of readily decomposable substrate, leaching of dissolved solutes declines and pH begins to increase. Underprediction of NO_3^- and SO_4^{2-} in 1985, coupled with overprediction of basic cations, resulted in the slight overprediction of pH in stream water. Simulated pH lies within the range of observations, with a slight overprediction before the clear-cut, which is consistent with the overprediction of Na^+, Mg^{2+} and Ca^{2+} and underprediction of SO_4^{2-} and NO_3^- (1966–83, $NME_b = 0.04$, $NMAE_b = 0.04$ and $NRMSE_b = 0.05$, Table 1, Fig. 4). Predicted pH after the treatment closely matched measured values (1988–2012, $NME_a = 0$, $NMAE_a = 0.01$ and $NRMSE_a = 0.02$, Table 1, Fig. 4). Acid neutralizing capacity (ANC) measurements are available for 1988–2012, and the model approximately captured the observations ($NME_a = 0.39$, $NMAE_a = 0.6$ and $NRMSE_a = 0.85$, Table 1).

The model performed well in capturing long-term stream SO_4^{2-} concentrations (1972–2012). Low performance criteria values for the pre-treatment ($NME_b = -0.05$, $NMAE_b = 0.05$ and $NRMSE_b = 0.05$, Table 1, Fig. 4) and post-treatment periods ($NME_a = -0.02$, $NMAE_a = 0.04$ and $NRMSE_a = 0.05$, Table 1, Fig. 4) are indicative of the close agreement of simulations with measured stream SO_4^{2-} concentrations. The model depicted the enhanced adsorption of SO_4^{2-} in soil under acidic conditions immediately following the clear-cut and the subsequent desorption of SO_4^{2-} as soil pH increases during biomass regrowth and NO_3^- retention (Mitchell et al., 1989). The long-term decreases in stream SO_4^{2-} concentrations are consistent with controls on emission of SO_2 following the Clean Air Act and subsequent rules (Driscoll et al., 1998; Stoddard et al., 1999).

The model generally captured stream water Al measurements over the 1988–1992 period, with $NME_a = 0.32$, $NMAE_a = 0.32$ and $NRMSE_a = 0.35$ for total monomeric Al and $NME_a = 0.14$, $NMAE_a = 0.25$ and $NRMSE_a = 0.27$ for organic monomeric Al. In the years immediately after the clear-cut (1984–86), the model projected leaching of total and organic monomeric Al of 12.1 and 2.7 μmol L^{-1} respectively, in comparison with measured concentrations of 21.2 and 3.3 μmol L^{-1}, respectively. The model performed poorly in simulating the peak stream total monomeric Al, and underpredicted concentrations in the second year after the harvest (Fig. A3).

Volume weighted concentrations of dissolved organic carbon (DOC) simulated by the model were roughly comparable with observations, with slight overprediction in pre-harvest years and underestimation for post-harvest years ($NME_a = -0.07$), and roughly captured the trend in concentrations ($NMAE_a = 0.2$ and $NMSE_a = 0.2$, Table 1, Fig. A3). However, the simulated stream DOC indicated more variability than observed values which may be associated with the effects of temperature and precipitation depicted in the model decomposition algorithm (Aber et al., 1997; McClaugherty et al., 1985). The model partitions metabolized organic matter into DOC and CO_2, and simulates soil sorption of DOC to depict losses of dissolved organic matter. The stream monomeric Al, DOC and the site density of the organic anions are important factors in the simulation of dissolved organic matter, Al speciation, pH and ANC in streamwater. The model depicts organic solutes as a triprotic analog to simulate pH, ANC and the speciation of monomeric Al (Fakhraei and Driscoll, 2015; Gbondo-Tugbawa et al., 2001).

The mean annual volume-weighted concentrations of NO_3^-, Ca^{2+}, Mg^{2+}, Na^+, total and organic monomeric Al in the stream water from the clear-cut watershed (W5) exceeded those of the reference watershed (W6) by the factors of 27, 2.5, 2.2, 1.3, 2.5 and 3, respectively over the 1984–86 period. We summarized PnET-BGC performance in simulations of stream water hydrology and chemistry based on NME values. NME below 10% indicates the model performed well. NME values between 10% and 20% are acceptable, but beyond this range is poor agreement. Based on these criteria, the model performed satisfactorily in depicting stream flow, pH, Ca^{2+}, SO_4^{2-}, Na^+, DOC for observations during pre- and post-treatment periods, and for Mg^{2+} during the post-treatment period. The model simulated Mg^{2+} and Al_{om} to an acceptable degree during post-cut period. NO_3^-, Al_m and ANC were the most challenging solutes to simulate effectively. Part of the challenge in simulating ANC stems from the fact that HBEF streams are highly sensitive to inputs of acidity and have ANC values near zero or negative. Note, PnET-BGC simulation results for the soil solutions chemistry are available in the supplementary information (Fig. A4, Table A7).

3.5. Seasonal variations in streamwater chemistry

The modified version of PnET-BGC with two soil layers adequately captured monthly variations in the chemistry of stream water before and after the clear-cut of W5. The pattern of monthly variation in stream water chemistry was similar before and after the harvest, although much lower concentrations of stream NO_3^- occurred after the cut (1990–2012). For the pre-cut period (1974–1982), the model underpredicted the high leaching of NO_3^- in streamwater during the winter dormant season, possibly because of some minor disturbances of soil freezing events and insect defoliation in early 1960, 1970 and 1980 (Fitzhugh et al., 2003; Gbondo-Tugbawa et al., 2001). Note, we did not consider these disturbances in our simulations because their intensities have not been quantified. During the summer and fall, when stream NO_3^- concentrations were lower, modeled values more closely agreed with observed values (Fig. 5). Monthly stream water Ca^{2+}, Mg^{2+} and Na^+ concentrations exceeded observed values during growing season and early winter, probably due to underprediction in net vegetation uptake.

Monthly variations of SO_4^{2-} indicated an overprediction in the growing season and underprediction in winter, possibly because of underestimation in plant uptake and mineralization, respectively. pH is highly influenced by errors in prediction of major ions. Some discrepancy between simulated and with observed pH was evident in the growing season and early winter season, related to overprediction of Ca^{2+}, Mg^{2+} and Na^+ and underprediction of NO_3^-. Overall, the model performed well in depicting seasonal variation in pH. Furthermore, monthly modeled streamflow was in good agreement with monthly measurements. Note that some factors, including the timing in snowpack development and snowmelt and canopy development and senescence, can have a considerable effect on simulations of hydrology and nutrients.

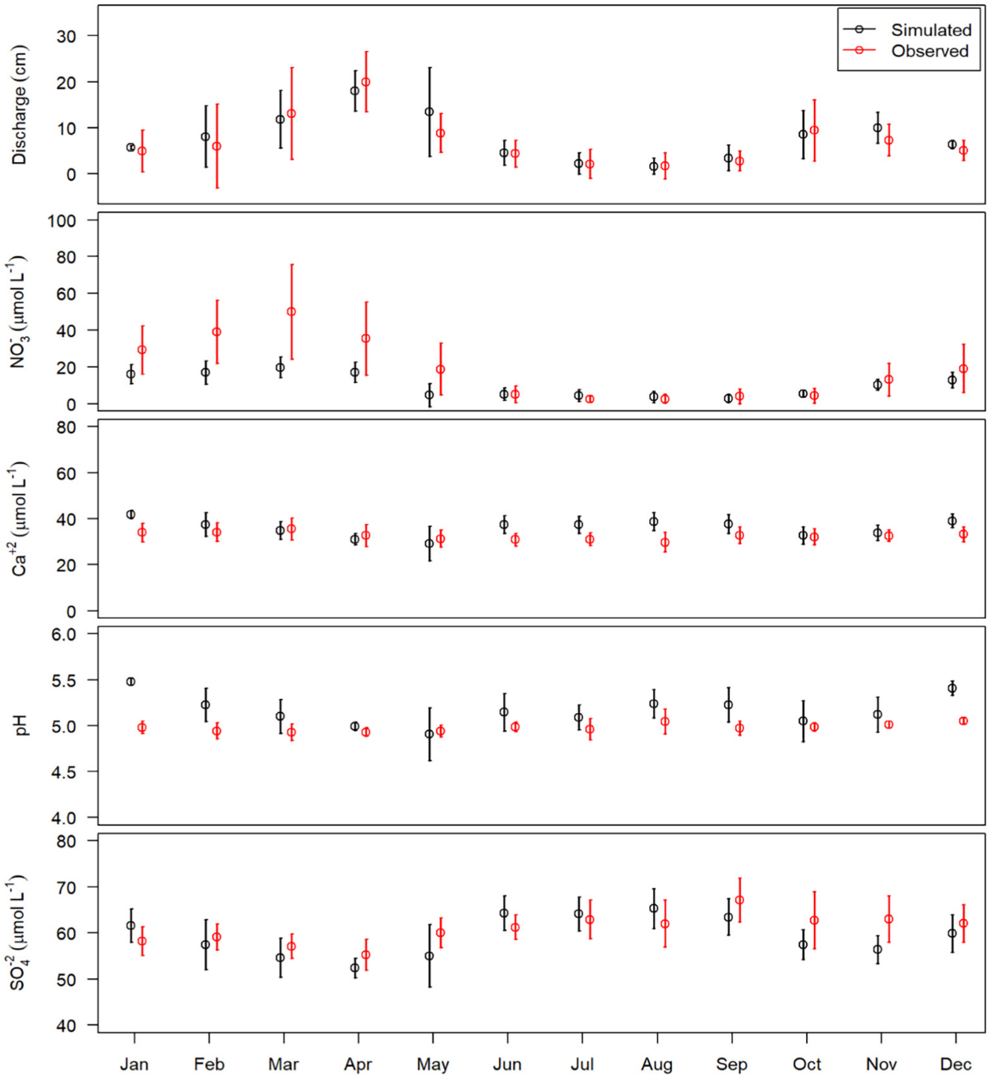

Fig. 5. Comparison between monthly patterns of stream water chemistry PnET-BGC simulations with observations for the pre-cut period (1974–1982).

3.6. Nutrient budget simulations

Nutrient budgets were calculated using the model to illustrate how the major sources and sinks of elements were affected by the intensive timber-harvesting of W5 (Fig. 6). Soil N mineralization and plant uptake were closely coupled, with average annual rates of 107 and 112 kg N ha^{-1} yr^{-1}, respectively in the early 1980s before the clear-cut, compared with measured values of 91–119 kg N ha^{-1} yr^{-1} (Bohlen et al., 2001; Melillo, 1977) for N mineralization and 88.2–114.4 kg N ha^{-1} yr^{-1} (Whittaker et al., 1974) for N plant uptake. Following the clear-cut, model simulated N mineralization increased slightly during the first year (110 kg N ha^{-1} yr^{-1} in 1984 vs 91 kg N ha^{-1} yr^{-1} in 1983) and then decreased to approximately 65 kg N ha^{-1} yr^{-1} in the second year after the cut (1985) (Fig. 6a). With biomass removal and reduced plant demand, plant N uptake reached a minimum value of 33 kg N ha^{-1} yr^{-1} in 1985, coinciding with the decline in N mineralization. With regrowing vegetation, both mineralization and plant uptake rates increased until around 1990, approaching pre-cut levels, but at slightly lower rates. There was a rapid increase in nitrification, peaking at 40 kg N ha^{-1} yr^{-1} in the second year after logging (1985) (Fig. 6a). After this initial period, plant uptake demand for N increased, resulting in decreases in nitrification and less stream NO$_3^-$ leaching.

The model closely captured average Ca^{2+} plant uptake of 58 kg ha^{-1} yr^{-1} in the early 1980s before the treatment, comparable with measurements (50–67 kg ha^{-1} yr^{-1}) (Whittaker et al., 1974). Plant Ca^{2+} uptake declined to 25 kg ha^{-1} yr^{-1} in the second year after the harvest (1985) before increasing to the near the pre-cut level; however, by 2100 the simulated average annual rate of Ca^{2+} uptake was only 44 kg ha^{-1} yr^{-1} (Fig. 6b). After an initial increase in soil Ca^{2+} mineralization in the first year following the clear-cut (58 kg ha^{-1} yr^{-1} in 1984 vs 47 kg ha^{-1} yr^{-1} in 1983), the rate decreased to roughly 35 kg ha^{-1} yr^{-1} in the second year after the cut (1985 year) and then increased but to a lower rate of 45 kg ha^{-1} yr^{-1} than the pre-cut levels of 55 kg ha^{-1} yr^{-1}.

Using a mass balance approach, depletion of soil exchangeable pool of Ca^{2+} was estimated around 9.6 kg ha^{-1} yr^{-1} for the years 1960–1980 due to elevated leaching from acid deposition (Likens et al., 1996). A net Ca^{2+} adsorption of 22 kg ha^{-1} yr^{-1} to the soil exchange complex was calculated in the first year following the clear-cut (1984). However, with regrowing vegetation, simulated exchangeable Ca^{2+} began to desorb from soil at a rate of 6 kg ha^{-1} yr^{-1} before reaching steady state conditions (a net adsorption of 1.3 kg ha^{-1} yr^{-1}) for the simulated years after 2000 (Fig. 6b). Weathering rate is important for the supply of Ca^{2+} to the watershed and in stream Ca^{2+} calibration that was assumed to be a constant value of 6.6 kg ha^{-1} yr^{-1} during the simulation period, higher than the assumed value of 3.5 kg ha^{-1} yr^{-1} for W6 by (Gbondo-Tugbawa et al., 2001) and significantly lower than the value of 21 kg ha^{-1} yr^{-1} for W6 (Likens et al., 1977) estimated using a mass balance approach.

The model effectively simulated SO$_4^{2-}$ uptake in early the 1980s (mean of 20 kg S ha^{-1} yr^{-1}), with values comparable to the observed range (9.46–27.46 kg S ha^{-1} yr^{-1}) (Whittaker et al., 1974). Plant uptake

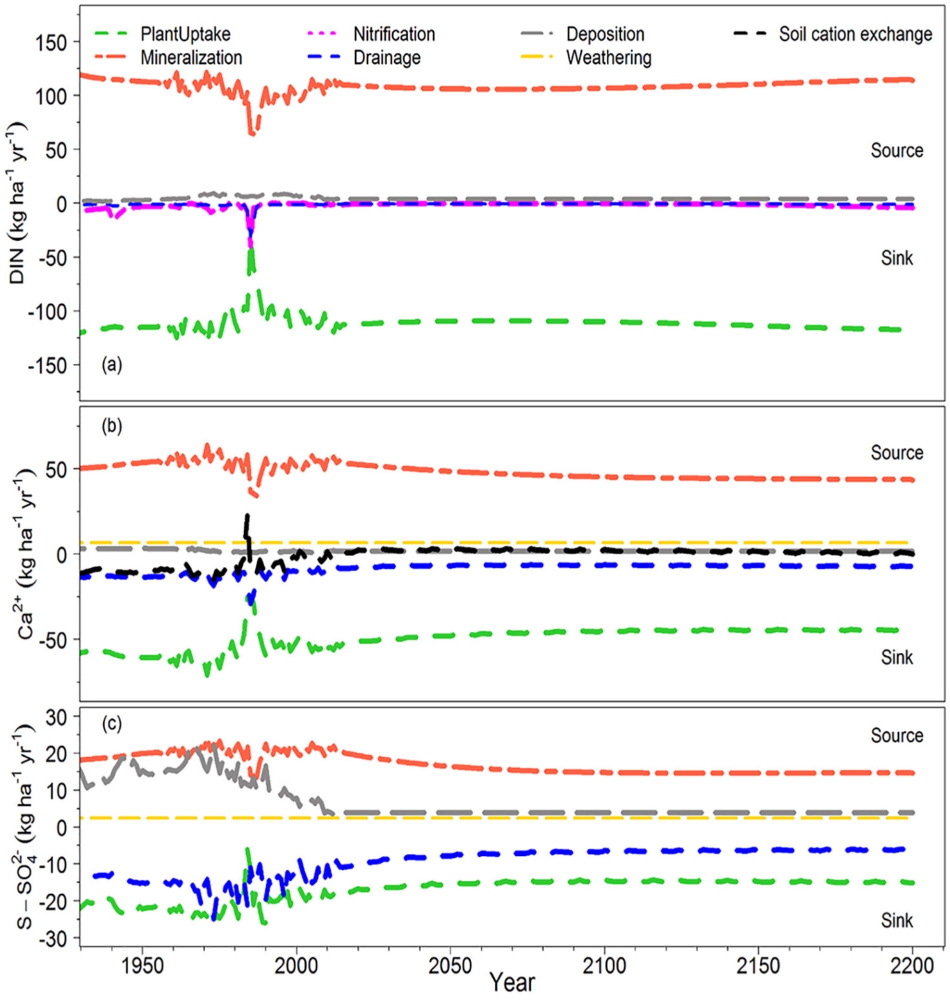

Fig. 6. Simulation of nutrients budgets for W5, HBEF, including total dissolved inorganic nitrogen (DIN) (a), Ca^{2+} (b) and S-SO$_4^{2-}$ (c).

of sulfate decreased to 6 kg S ha^{-1} yr^{-1} in the second year after the cut (1985). Soil SO_4^{2-} mineralization showed a pattern similar to soil N and Ca mineralization following the treatment (Fig. 6c). The overall pattern of SO_4^{2-} dynamics was dominated by the atmospheric deposition of SO_4^{2-} which was relatively high during 1960–1980 (17.4 kg S ha^{-1} yr^{-1}) but declined to 3.8 kg S ha^{-1} yr^{-1} by the end of the simulation.

Soil base saturation is considered a critical indicator of soil acidification stress due to atmospheric acid deposition or forest cutting (Driscoll et al., 2001, Sullivan et al., 2013). PnET-BGC-simulated soil base saturation was around 25% historically (1850), with values decreasing associated with high acid deposition to around 9.6% in 1980s (Fig. A5), similar to the measured value of 9.5% in W5 prior to harvesting (Johnson et al., 1991). Simulated soil base saturation showed a slight increase in the first year after the clear-cut in 1984, consistent with elevated leaching of nutrient cations in soil and stream water and then decreased to below 10% by around 2000 due to depletion of exchangeable basic cations in soil during the earlier years after the clear-cut coupled with high acid deposition. With the competing processes of regrowth of new vegetation and reduction in acid deposition, percent soil base saturation gradually increased to values below historical values by the end of the simulation period (17.4%).

Long-term simulation of soil exchangeable cation pools indicated depletion of soil exchangeable Ca declined over time, eventually approaching steady state over long-term simulations. Soil and stream water acidification can occur due to forest cutting, resulting in depletion of basic cations associated with elevated leaching of strong acid anions (Federer et al., 1989). Soil and stream water in the northern hardwood forest of the HBEF are prone to the acidification due to low soil base saturation (10%, W5), surface water pH (5.5, W5) and ANC (10 μeq l^{-1}, W5), which are below critical chemical thresholds of 20%, 6 and 50 μeq l^{-1}, respectively (Cronan and Grigal, 1995; Gbondo-Tugbawa and Driscoll, 2002; MacAvoy and Bulger, 1995). Long-term simulations indicated that low values of soil base saturation are likely to persist beyond year 2200 (17.4%). Loss of nutrient cations from the soil exchangeable pool impact forest productivity and the health of sensitive tree species such as sugar maple over the long-term (Cleavitt et al., 2018). Future intensive forest cutting in northern hardwood forests has the potential for depletion of available soil nutrients that could limit plant growth of sensitive species.

3.7. Sensitivity analysis

We conducted a model sensitivity analysis for both the pre-treatment and post-treatment periods, representing state variables as 3-year mean values for the mature forest (1970–72) and the post-cut, rapidly-aggrading forest (1988–90). Our sensitivity analyis showed that model simulations are generally more responsive to vegetation parameters than abiotic parameters and that the sensitivity of the model to mature forest (pre-cut) conditions is greater than for simulations of the aggrading forest (post-cut conditions) (Fig. A6). In brief, the model sensitivity showed biomass and stream NO_3^- are highly sensitive to parameters related to nitrogen retranslocation following senescence (FolNRet), minimum N concentration in foliar litter (flpctn), the slope of maximum gross photosynthesis rate (AmaxB) and the fraction of mineralized N that is immobilized by microbes (NImmobA). The model uses these constant inputs to calculate foliar N concentration and gross photosynthesis in broad-leaved deciduous forests. Higher values of flpctn and AmaxB and a lower value of FolNRet result in a higher gross photosynthesis rate, causing more biomass production and less leaching of NO_3^- to stream water. Higher rates of N immobilization in soil limit NO_3^- leaching.

Long-term measurements at the HBEF enabled effective calibration and testing of the model mainly based on field-based data. However, a sensitivity analysis helps to identify the most important parameters that influence model simulations for conditions before and after the harvest. Model sensitivity results were consistent with previous reports, reflecting the importance of vegetation variables (Aber et al., 1996). Lower model sensitivity for the period after the harvest may reflect the ability of the aggrading northern hardwood ecosystem to tightly regulate ecosystem biotic and abiotic processes, including nutrient uptake, mineralization, nitrification, and transpiration. Results from the sensitivity analysis should be helpful for future model applications at other sites and demonstrate the necessity of field measurements to constrain important parameter values.

4. Conclusions and future research

Utilization of forest biomass as an energy source has raised interest in improving the understanding of the short- and long-term effects of intensive harvesting to inform practices for sustainable forest management. Process-based modeling is a useful diagnostic tool that enables extrapolation of short-term observations of nutrient dynamics to longer time scales (e.g., until 2200 year). Revisions of algorithms of PnET-BGC significantly improved model performance in predicting short- and long-term dynamics of major elements for evaluating effects of forest harvesting at the HBEF. Modeled soil Bs horizon and stream water chemistry successfully captured the rapid recovery of leaching nutrients to pre-cut levels after harvest. The model was also used to evaluate nutrient budgets after the harvest and the role of net mineralization and plant uptake in regulating the recovery of the forest ecosystem. A first-order sensitivity analysis was conducted to identify important vegetation parameters, particularly at the time of harvest in model simulation.

This study is the first step in the testing and evaluation of PnET-BGC as a tool to quantify effects of forest clear-cutting. Few studies have rigorously compared model simulations against experimental harvesting data to test and verify simulations and improve confidence in efforts to extrapolate short-term observations of nutrient dynamics to longer time scales. Previous modeling approaches have focused on dynamics of C and N stocks in soil or vegetation and assessed their depletion in response to long-term forest harvesting practices (Mina et al., 2017; Shifley et al., 2017; Sullivan et al., 2013). Typically, forest ecosystem models do not consider other nutrients (e.g., Ca^{2+}, SO_4^{2-}) or soil/stream acid-base chemistry, which can affect tree growth and ecosystem structure and function, and hence the sustainability of different logging practices. Application of the multi-element soil-layer model allows for a more comprehensive quantification of short- and long-term responses to forest disturbance. However, additional testing and application of the model is needed for additional sites and harvesting approaches, which could lead to simulations of short- and long-term impacts of different forest cutting approaches (e.g., cutting rotation length and intensity) and investigation of how changing climate influences the recovery of forest from harvesting.

Acknowledgements

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2018.07.066.

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