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Response of biomass, hydrology and biogeochemistry to alternative approaches of cutting a northern forest: model comparisons

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Abstract The biogeochemical model, PnET-BGC, was modified and parameterized using field data from an experimental whole-tree harvest of watershed (W5) in 1983–1984 at the Hubbard Brook Experimental Forest (HBEF), New Hampshire, USA. The model simulated the hydrology, biomass accumulation, and soil solution and stream water chemistry responses to forest cutting. The parameterized model was then applied to other experimentally cut watersheds at the HBEF; including a devegetation experiment (W2; devegetation and herbicide treatment) and a commercial strip-cut (W4) to evaluate the ability of the model to depict ecosystem responses to a range of cutting regimes. Revisions of algorithms of PnET-BGC improved model performance in predicting short- and long-term dynamics of major elements following various approaches to forest cutting. Despite some initial differences in species composition and biomass accumulation rates among the cut watersheds, simulations of total forest biomass for all three treated watersheds (W2, W4 and W5) were consistent with expectations based on the growth trajectory of a second-growth, reference watershed (W6) at the HBEF. The modified two-soil-layer PnET-BGC captured the immediate increase in stream concentrations

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of NO_3^-, Ca^{2+}, Mg^{2+} and Na^+ as well as enhanced adsorption of SO_4^{2-} following cuttings and indicated a greater response for the devegetation and the whole-tree harvest treatments than the sequential strip-cut of W4. Simulations indicated intense NO_3^- leaching with the devegetation and herbicide treatment and consequent accelerated decline in soil base saturation and a slower recovery pattern during forest regrowth by the end of the simulation period (2100) compared to the other treatments.

Keywords Forest cutting · PnET-BGC · Hubbard Brook Experimental Forest · Biomass accumulation · Biogeochemistry · Stream hydrology

Introduction

Forest biomass harvesting in the northeastern US may intensify over the coming decades in response to increased demand for renewable energy. However, harvesting results in nutrient removal from the forest ecosystem. Sustainable forestry requires that regenerating forests from successive harvests not deplete plant-available pools of nutrients. The long-term effects of repeated forest cutting are uncertain due to limited information on land use history and long-term time series observations. A wide range of positive and negative impacts of intensive forest harvesting on forest production and environmental impacts have been reported, the most notable being the potential for long-term depletion of soil nutrients (Kreutzweiser et al. 2008; Walmsley et al. 2009). Concern remains over the ability of intensively harvested forests to maintain productivity, sequester carbon and nitrogen, and provide other ecosystem services. Repeated clear-cuts could diminish nutrient availability, particularly for carbon, nitrogen and calcium, and ultimately limit plant uptake and forest productivity (Federer et al. 1989; Kreutzweiser et al. 2008; Walmsley et al. 2009; Cleavitt et al. 2017). With the acceleration of land development and demand for forest products, understanding both short- and especially long-term impacts of harvesting practices (e.g., cutting rotation length, intensity) on forest dynamics is a key factor in developing criteria and guidelines for sustainable forest management practices (Peng et al. 2002; Mina et al. 2017).

Biogeochemical models allow for the extrapolation of short-term observations of the response of hydrology, forest biomass and nutrient dynamics to ecosystem stressors to longer-time scales (decades to centuries) and to probe how various disturbances influence forest ecosystems. Therefore, models are important tools to help gain a better understanding of the complex, interacting effects of disturbance on ecological processes. Various logging practices that differ in harvesting intensity and frequency, influence the extent and duration of nutrient losses by biomass removal and drainage (Valipour et al. 2021). Predicting the responses of forest ecosystems to a variety of harvesting techniques over the short and long term is a key factor for understanding the sustainable management of forests.

Unfortunately, few modeling studies have compared simulation results with field measurements to test model performance on short and long-term effects of harvesting (Wei et al. 2003; Bu et al. 2008; Mina et al. 2017; Shifley et al. 2017). We previously modified and tested the biogeochemical watershed model, PnET-BGC, using field observations from a northern hardwood forest watershed that was subjected to a whole-tree harvest (W5) at the Hubbard Brook Experimental Forest (HBEF), New Hampshire, USA (Valipour et al. 2018). The overarching goal of the current study was to apply the parametrized and tested model to other experimentally cut watersheds at the HBEF; including a devegetation experiment (W2; devegetation and herbicide treatment) and a commercial strip-cut (W4). These applications allowed for evaluation of ecosystem effects under a range of cutting approaches, including assessing the depletion of soil nutrient capital (Federer et al. 1989) and effects on carbon sequestration. The specific objectives of this study are: (i) to apply the modified, parametrized model to different cutting experiments at the HBEF (W2, W4) as a test of model performance; (ii) to use the modified model to compare and gain insight on how the forest ecosystem respond to different forest cutting techniques over both the short- and long-term; and (iii) to project short- and long-term patterns of biomass accumulation and changes of nutrient pools and fluxes in soil and streamwater in response to different harvesting strategies.

Methodology

Site description and treatments

The HBEF is located in southern White Mountains of New Hampshire, USA (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 forest management through monitoring and large-scale field experiments. The HBEF includes gauged experimental watersheds with long-term and comprehensive measurements of vegetation, soils, meteorology, hydrology and biogeochemistry, the earliest of which began in 1956 (http://www.hubbardbrook.org). Streamflow is gauged at all the watersheds using v-notch weirs. Some watersheds have been experimentally manipulated by forest cutting (Fig. 1, Bormann and Likens 1979; Likens et al. 1970; http://www.hubbardbrook.org). In this paper, we studied the three experimentally cut watersheds at the HBEF (W5, W2 and W4) and compared the results with the reference watershed (W6). Watershed 6 (W6), with an area of 13.2 ha and an elevation range of 549–792 m, serves as the biogeochemical reference watershed and has not been experimentally manipulated. W5 is adjacent to W6 with an area of 21.9 ha and elevation range of 488–762 m. W5 was subjected to a whole-tree harvest during the fall of 1983 through the winter of 1984. Watershed 2 (W2) with an area of 15.6 ha and elevation range of 503–716 m was devegetated in 1965. All cut vegetation was left on the site and regrowth was prevented for 3 years by herbicide application. Watershed 4 (W4) with an area of 36.1 ha and elevation range 443–747 m was commercially clear-cut in 25-m-wide strips along the elevational contour. The first set of strips was harvested in 1970, and the remaining two sets of strips were harvested in 1972 and 1974; thus, regrowth began in 1971, 1973, and 1975, following the cutting of each strip. An uncut buffer strip was retained along the stream channel in the lower watershed (Table 1).

The climate of the HBEF is humid-continental, with short, cool summers and long, cold winters. Mean monthly air temperature varies approximately between − 9 and 18 °C from January to July, respectively. Average annual precipitation is about 140 cm, of which 25–36% falls as snow (Federer et al. 1990). Soils are predominantly well-drained Spodosols, with an average depth of 0.6–1 m. Vegetation in the study area is dominated by the northern hardwood forest, including American beech (*Fagus grandifolia*), sugar maple (*Acer saccharum*), and yellow birch (*Betula alleghaniensis*). At higher elevation, vegetation includes red spruce (*Picea rubens*), balsam fir (*Abies balsamea*) and paper birch (*Betula papyrifera*). After forest cutting, the pioneer species, pin cherry (*Prunus pennsylvanica*), typically dominates the forest vegetation for two decades (Marks 1974).

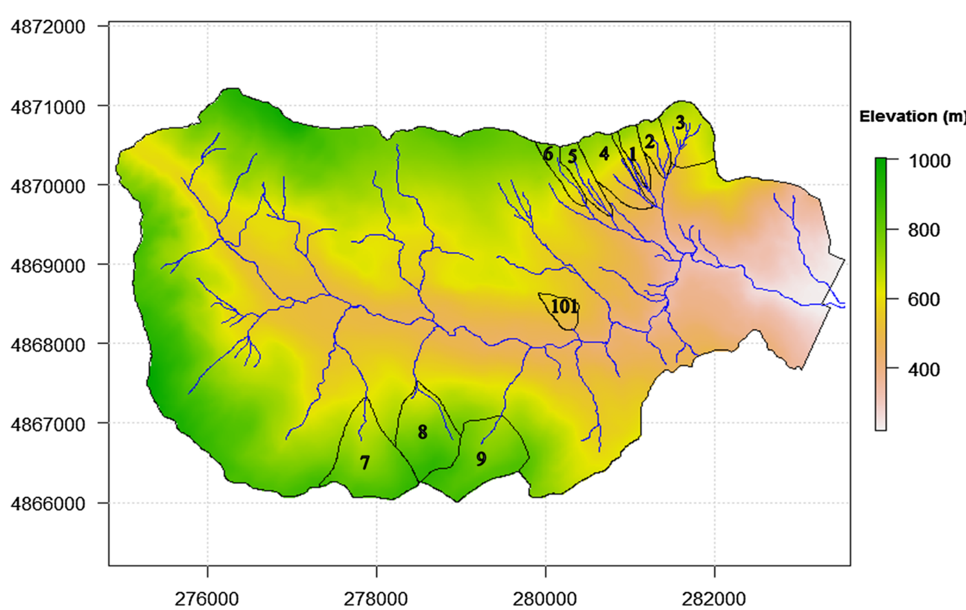

Fig. 1 Elevational map and location of experimental watersheds at the Hubbard Brook Experimental Forest, NH

Table 1 A summary of disturbance history of watershed 6, watershed 5, watershed 4 and watershed 2 at Hubbard Brook Experimental Forest

Watersheds	Disturbance year	Type of disturbance	Mortality	Biomass removal	Elevation (m)
Watershed 6 (reference)	1904	Logging	0.2	0.8	549–792
	1919	Logging	0.59	0.8	
	1938	Hurricane	0.2	0.4	
Watershed[a] 5	1983	Commercially whole-tree harvesting	0.92	0.87	488–762
Watershed[a] 2	1965	Whole-tree harvesting	0.92	0	503–716
	1966	Herbicide application	0.92	0	
	1967	Herbicide application	0.92	0	
	1968	Herbicide application	0.92	0	
Watershed[a] 4	1970	Strip cut	0.35	0.9	442–747
	1972	Strip cut	0.35	0.9	
	1974	Strip cut	0.92	0.9	

[a]Watersheds 5, 4 and 2 include the assumed disturbance history for Watershed 6 in addition to the experimental manipulation indicated

Model description

PnET-BGC is an integrated biogeochemical model, developed to assess the effects of atmospheric deposition, land disturbance and climatic change on vegetation, soils and surface waters, primarily in forest ecosystems (Gbondo-Tugbawa et al. 2001). PnET-BGC was developed by linking two submodels, PnET-CN (Aber and Federer 1992; Aber and Driscoll 1997; Aber et al. 1997) and BGC (Gbondo-Tugbawa et al. 2001) to model the dynamics of major elements (i.e., Ca, Mg, K, Na, Al, C, N, S, P, Cl, Si) in forest ecosystems. PnET-BGC depicts ecosystem processes of photosynthesis, canopy interactions, plant nutrient uptake, accumulation and loss of soil organic matter, soil cation exchange and anion adsorption, nutrient mineralization and nitrification, as well as hydrology, mineral weathering and solution chemical reactions to simulate the fluxes of energy and water and the cycling of nutrients in forest ecosystems (Gbondo-Tugbawa et al. 2001) (Fig. A1).

PnET-BGC was run on a monthly time-step with a spin-up period from year 1000 to 1850 under constant climate, pre-industrial atmospheric deposition and no land disturbance, which allows the model to come to steady-state. Hindcast simulations were then run from 1850 to present by considering historical climate, atmospheric deposition and land disturbance (i.e., forest harvest, blowdown, ice storm). The model can be used to project future conditions under given input scenarios. Model inputs include meteorological data, atmospheric deposition, geochemical properties of soil, vegetation type, element stoichiometry and land disturbance history (Table 1). We used a version of PnET-BGC that considers two layers of soil, 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 utilizes different weathering rates and soil properties for each layer. A detailed sensitivity analysis of model response to variations in model inputs and parameters was conducted in Valipour et al. (2018), Valipour (2019) and Fakhraei et al. (2017).

Meteorological and atmospheric deposition data

The same methodology was used to prepare input data for all watersheds simulated at the HBEF. 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 inputs are limited to the period for which monitoring data are available (meteorology 1955; wet deposition 1963; dry deposition 1990) (http://www.

hubbardbrook.org/). A detailed description of the reconstruction of meteorological and atmospheric deposition data was provided in the previous literature (Chen et al. 2004; Fakhraei et al. 2014, 2016; Valipour et al. 2018).

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). Chemistry of freely-draining soil solutions for W6 were measured during 1984–2017 (https://hubbardbrook.org). For W5, 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) (Johnson et al. 1991, 1997; Dahlgren and Driscoll 1994). Soil chemical data for W5 are available for pre-treatment (1983) and 3 post-treatment years (1986, 1991, 1997) (Johnson et al. 1991, 1997). Note, there are no field measurements of soil characteristics for W2 and W4 following their harvest to compare with the model simulations.

For W5, weathering rates for the upper and lower soil layers were estimated through calibration using soil solution for the Bs2 horizon and stream water chemistry (Dahlgren and Driscoll 1994; Nezat et al. 2004; Chen and Driscoll 2005). To estimate the weathering rates for W2 and W4, we used weathering rates estimated for W5 as the initial values and then adjusted values through calibration with observed stream water chemistry. Parameters and variables used in the model calibration for the study watersheds are summarized in Tables A1–A3 in the supporting information. These parameters were assumed to be constant over the simulation period. A detailed description of the model and its parameters can be found in Aber et al. (1997) and Gbondo-Tugbawa et al. (2001). Model simulations for stream water hydrology and chemistry of study watersheds were compared with the measured data during 1963–2014 (Likens 2017).

Vegetation parameters

PnET-BGC uses site-specific vegetation parameters (variables related to photosynthesis, foliar growth, wood and root turn-over rates and water use efficiency, Tables A1–A3). Values of these parameters for northern hardwood tree species have been obtained from direct field measurements, values reported in the literature or model calibration (Aber and Federer 1992; Aber and Driscoll 1997; Aber et al. 1997). Vegetation parameters are assumed to be constant over the simulation period. Foliar nitrogen concentration is used to predict the rate of photosynthesis and the pattern of biomass accumulation through the simulation (Aber et al. 1997). In order to capture greater total aboveground biomass for W5 than for W4 and W2 in model simulations, we calibrated the model with slightly higher minimum nitrogen concentration in foliar litter for W5 (0.96%) than W4 (0.915%) and W2 (0.911%). This parameter is used as an input to simulate the amount of N allocated to the plant bud for foliage production of the following year.

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, estimated historical tree mortality and percent removal were taken from previous simulations (Aber and Driscoll 1997; Gbondo-Tugbawa et al. 2001). For W5, we used detailed information about the whole-tree harvest (Ryan et al. 1992) to estimate the percent of forest biomass mortality and removal (Ryan et al. 1992; Valipour et al. 2018).

Some assumptions were needed to adjust biomass harvest and regrowth estimates. First, some edge and stream corridor areas were left uncut W4 and W5; we assumed that 8% of the trees on both watersheds were left uncut. Second, in December 1965, all trees and shrubs on watershed 2 were felled and left in place, and during the growing seasons of 1966, 1967 and 1968, herbicide was applied to the watershed to prevent vegetation regrowth. To simulate the W2 disturbance, we assumed mortality of 92% of forest biomass for each year during 1965–1968. Third, because PnET-BGC is limited in its spatial depiction, it was difficult to represent the strip-cut that occurred in W4. We assumed 35% of forest biomass removal for the years 1970 and 1972, and a cumulative total of 92% forest biomass removal by 1974 (Table 1). These assumptions allowed for a better match of both aboveground biomass and stream water chemistry simulations with field observations.

Biomass studies and calculations

A total forest inventory was conducted in W5 during mid-summer of 1982 to quantify biomass prior to the cut. Post-harvest, the sampling approach for forest biomass was adjusted through time to accommodate the greatly changing sizes and density of trees in the watershed (Cleavitt et al. 2017). More detailed information of W5 biomass calculations and analysis can be found in the existing literature (Siccama et al. 1994; Johnson et al. 1995; Fahey et al. 2005; Valipour et al. 2018).

For W2 there was no forest inventory data collected prior to the cut. Following the cut, seventy 10 × 10 m permanent quadrats were established in an evenly distributed, stratified-random manner within a surveyed system of 25 m grid units. Vegetation was sampled within these quadrats by a system of nested plots according to classes of sizes and growth form of plants. Plants were enumerated in the quadrats in mid- to late-July of 1969 1970, 1971, 1973, 1979, and 1988, the 1st, 2nd, 3rd, 5th, 11th, and 20th year of recovery following the last herbicide application (Reiners 1992), and forest biomass was estimated using allometric regression equations for the site (Fahey et al. 2005).

Regeneration of the strip-cut W4 has been surveyed on permanent plots at 1–4-year intervals since the harvest. Fifty-seven 25 × 25 m plots were established for monitoring vegetation. Nineteen plots were chosen randomly for each year of cut. These plots were later classified by elevation with 19 plots at low elevation (440–550 m), 18 at mid-elevation (550–650 m), and 20 at high elevation (650–730 m). Individual stems by species for trees, shrubs, and herbs were measured in these plots (Martin and Hornbeck 1989). Total aboveground biomass values for W4 (1969–2011) were estimated using allometric regression equations (unpublished data from John Battles).

Results

Vegetation simulations

In this study, the parametrized/modified model was applied to different harvesting experiments at the HBEF (W2, W4, W5). The model was tested to assess performance and then used to project ecosystem pools and fluxes including aboveground biomass, soil base saturation, stream water chemistry and element budgets. Both short- and long-term forest ecosystem responses to different harvesting techniques were evaluated. The model generally performed well in the simulation of aboveground biomass in clear-cut watersheds (W4, W2) for both pre-harvest and post-harvest conditions (Fig. 2). Modeled aboveground biomass (154 t ha^{-1}) approximately matched the observed value (169 t ha^{-1}) for the pre-cut year (1969) for the strip-cut W4. Re-growing vegetation consisted mainly of herbs, shrubs, and tree seedlings and sprouts for the first few years after the cuts for both W4 and W2 (Fahey et al. 2005). For the commercial strip-cut W4, aboveground biomass was simulated to be 31 t ha^{-1}, compared with the observed value of 26 t ha^{-1} 6 years after the final third of the watershed was cut (1980), representing around 20% of the pre-cut forest biomass. At 11, 21 and 41 years after the strip cut, simulated aboveground biomass increased to 54, 90 and 125 t ha^{-1}, respectively, corresponding with observed measurements of 48, 85 and 124 t ha^{-1} respectively, approximately 35%, 58% and 81% of aboveground biomass prior to the cut. Model simulations indicated that W4 would reach the aboveground biomass occurring prior to the cut after about 48 years of regrowth (153 t ha^{-1}) (Fig. 2).

For the devegetated and herbicide treated W2, modeled aboveground biomass of 21 t ha^{-1}, compared well with the observed value of 20 t ha^{-1} 5 years after the clear-cut (1972), which represented around 15% of the pre-cut forest biomass. At 11 and 20 years after the clear-cut of W2, simulated aboveground biomass of 58 and 88 t ha^{-1} overestimated the observed values of 40 and 69 t ha^{-1}, respectively, which are approximately 43% and 65% of aboveground biomass prior to the cut. Based on observations in reference watershed 6, the model predicted that W2 approached the value of aboveground biomass that was on the watershed prior to the cut after about 46 years of regrowth (138 t ha^{-1}) (Fig. 2). Comparison with the reference watershed

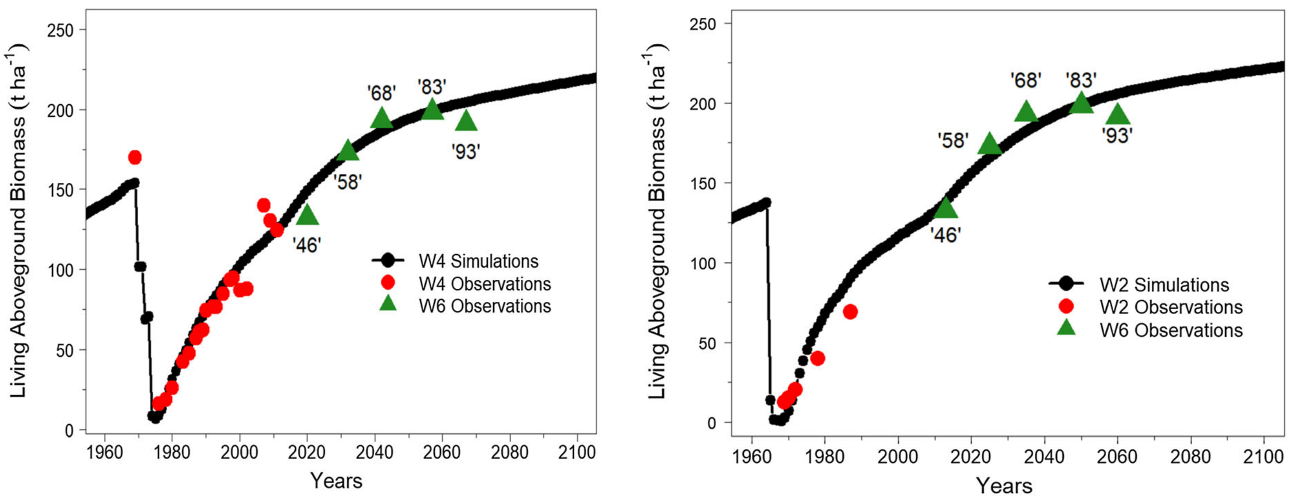

Fig. 2 Simulations of living aboveground biomass accumulation for watershed 4 (W4) and watershed 2 (W2), for the period before and after forest cuts. Model simulations are compared with measured values for W4 and W2 and age-based predictions based on observations from reference watershed 6 (W6) after adjustment for years after cutting

(W6), which is a second-growth forest, suggests that projections of W4 and W2 biomass accumulation are consistent with the expected growth trajectory. By the end of simulation period (2100), the model projected total living aboveground biomass of 218 t ha^{-1} for strip-cut W4 and 221 t ha^{-1} for devegetation/herbicide treated W2, compared with the whole tree-harvested W5 (223 t ha^{-1}) and reference watershed W6 (224 t ha^{-1}).

Stream hydrology

Three statistical metrics were used to evaluate model performance in simulating hydrology for each watershed before and after the harvest: normalized mean error (NME) normalized mean absolute error (NMAE) and normalized root mean squared error (NRMSE) (Janssen and Heuberger 1995; Alewell and Manderscheid 1998). The modeled annual stream flow for W4 adequately captured observed values over the study period 1964–2014, with slight underprediction for the pre-harvest period ($Mean_{obs}$ = 89.72 cm, $Mean_{pred}$ = 84.6 cm, NME_b = − 0.06, Table 2, Fig. 3) and the post-harvest period ($Mean_{obs}$ = 93.87 cm, $Mean_{pred}$ = 91.46 cm, NME_a = − 0.03, Table 2, Fig. 3). Results of simulated hydrologic response showed stream discharge increased in cut watersheds immediately after harvesting the vegetation, and then decreased with forest regrowth. Simulated average annual stream flow for W4 during the multiple years of the strip-cuts until the 1st year of regrowth (1971–1975) indicated a 49% increase from the pre-cut year (1969) compared with a 40% increase for the measured values. The model also effectively simulated annual stream flow for W2 during the study period 1964–2014, with slight underprediction for the post-harvest period ($Mean_{obs}$ = 96.7 cm, $Mean_{pred}$ = 89.3 cm, NME_a = − 0.08, Table 2, Fig. 3). Modeled average annual stream flow for W2 during the devegetation/herbicide treatment years until the first of regrowth (1966–1969) underestimated the percentage increase in stream discharge (27%) from the pre-cut year (1964) compared to the observed value (65%).

Streamwater chemistry simulations

Statistical metrics indicate that the modified model satisfactorily reproduced the long-term patterns (1964–2014) of concentrations of major anions and cations in stream water for both cut watersheds (W4, W2). For the strip-cut W4, the simulated annual volume-weighted concentration of stream water NO_3^- matched measured values for pre-cut and post-cut periods, with some slight underprediction ($Mean_{obs\_b}$ = 21.5 μmol L^{-1}, NME_b = − 0.53, $Mean_{obs\_a}$ = 15 μmol L^{-1}, NME_a = − 0.43, Table 2, Fig. 3). The model was also able to satisfactorily depict stream Ca^{2+} concentrations (pre-treatment: $Mean_{obs\_b}$ = 45.8 μmol L^{-1}, NME_b = 0.1; post-treatment: $Mean_{obs\_a}$ = 35.1 μmol L^{-1}, NME_a = − 0.06,

Table 2 Comparison of modeled and observed values of stream constituents and model performance for the periods prior and after strip-cut W4

Stream constituents	Mean		STD		NME	NMAE	NRMSE
	Observed	Simulated	Observed	Simulated			
Pre-harvest (1966–69)_W4							
Flow	89.72	13.47	84.6	8.96	− 0.06	0.07	0.09
pH	5.82	0.06	6	0.12	0.03	0.03	0.04
Na^{+}	48.17	1.96	54.11	4.2	0.12	0.12	0.15
Mg^{2+}	16.86	0.29	18.75	1.32	0.11	0.12	0.14
Ca^{2+}	45.84	3.02	50.58	3.51	0.1	0.14	0.16
NO_3^{-}	21.56	7.31	10.17	1.13	− 0.53	0.5	0.58
SO_4^{2-}	64.33	3.35	64.78	1	0.01	0.06	0.06
Post-harvest (1980–2014)_W4							
Flow	93.87	24.45	91.46	18.61	− 0.03	0.09	0.14
pH	5.9	0.14	5.9	0.15	0	0.02	0.03
Na^{+}	39.56	2.87	39.63	2.99	0	0.04	0.05
Mg^{2+}	11.82	3.06	11.78	1.76	0	0.14	0.16
Ca^{2+}	35.1	8.74	32.83	4.81	− 0.06	0.14	0.18
NO_3^{-}	15.05	17.86	8.6	2.8	− 0.43	0.7	1.23
SO_4^{2-}	45.52	9.61	44.82	7.19	− 0.02	0.06	0.07

Values represent mean and standard deviation of annual volume-weighted concentrations for the pre-harvest (1966–1969) post-harvest (1980–2014 year) periods. Units for stream constituents are µmol L^{-1} (Flow; cm)

NEM normalized mean error; *NMAE* normalized mean absolute error; *NRMSE* normalized root mean squared error; *STD* standard deviation

Table 2, Fig. 3). Stream Mg^{2+} and Na^{+} concentrations were slightly overpredicted during pre-treatment period (Mg^{2+}: $Mean_{obs\_b}$ = 16.86 µmol L^{-1}, NME_{b} = 0.11; Na^{+}: $Mean_{obs\_b}$ = 48.17 µmol L^{-1}, NME_{b} = 0.12, Table 2, Fig. 3), but close to measured values during the post-treatment period (Mg^{2+}: $Mean_{obs\_a}$ = 11.82 µmol L^{-1}, NME_{a} = 0; Na^{+}: $Mean_{obs\_a}$ = 39.56 µmol L^{-1}, NME_{a} = 0, Table 2, Fig. 3). The modified two soil layer version of PnET-BGC was capable of depicting the immediate increase in stream concentrations of NO_3^{-}, Ca^{2+}, Mg^{2+} and Na^{+} following the progressive strip-cut of W4 (1971–1975), simulating mean values of 44.5, 50.8, 18.9 and 48.7 µmol L^{-1} compared with measured values of 72, 56.2, 17.8 and 46.6 µmol L^{-1} for NO_3, Ca^{2+}, Mg^{2+} and Na^{+}, respectively. The model-simulated peak stream leaching of these elements during the first strip-cut in 1972 was consistent with measurements. The mean annual volume-weighted concentrations of NO_3^{-}, Ca^{2+}, Mg^{2+} and Na^{+} in the stream water from the strip-cut watershed (W4) exceeded those of the reference watershed (W6) by factors of 2, 1.6, 1.3 and 1.5, respectively over the 1971–1975 period.

Simulated stream water pH was within the range of W4 observations, with a slight overprediction before the cut ($Mean_{obs\_b}$ = 5.82, NME_{b} = 0.03) and closely captured the measured values after the treatment ($Mean_{obs\_a}$ = 5.9, NME_{b} = 0). The model simulated a decline in pH values with removal of vegetation during the treatment, with the greatest decline occurring during the second strip cut (1973) and values increasing with the regrowth of vegetation following the third strip cut. The model also performed well in capturing the long-term decreases in stream SO_4^{2-} concentrations with a slight overprediction ($Mean_{obs\_b}$ = 64.3, NME_{b} = 0.01) during the pre-cut period and a slight underprediction during post-cut ($Mean_{obs\_a}$ = 44.8, NME_{a} = -0.02). The model depicted a modest increase in adsorption of SO_4^{2-} in soil under acidic conditions during devegetation processes in W4, indicated by a decline in stream

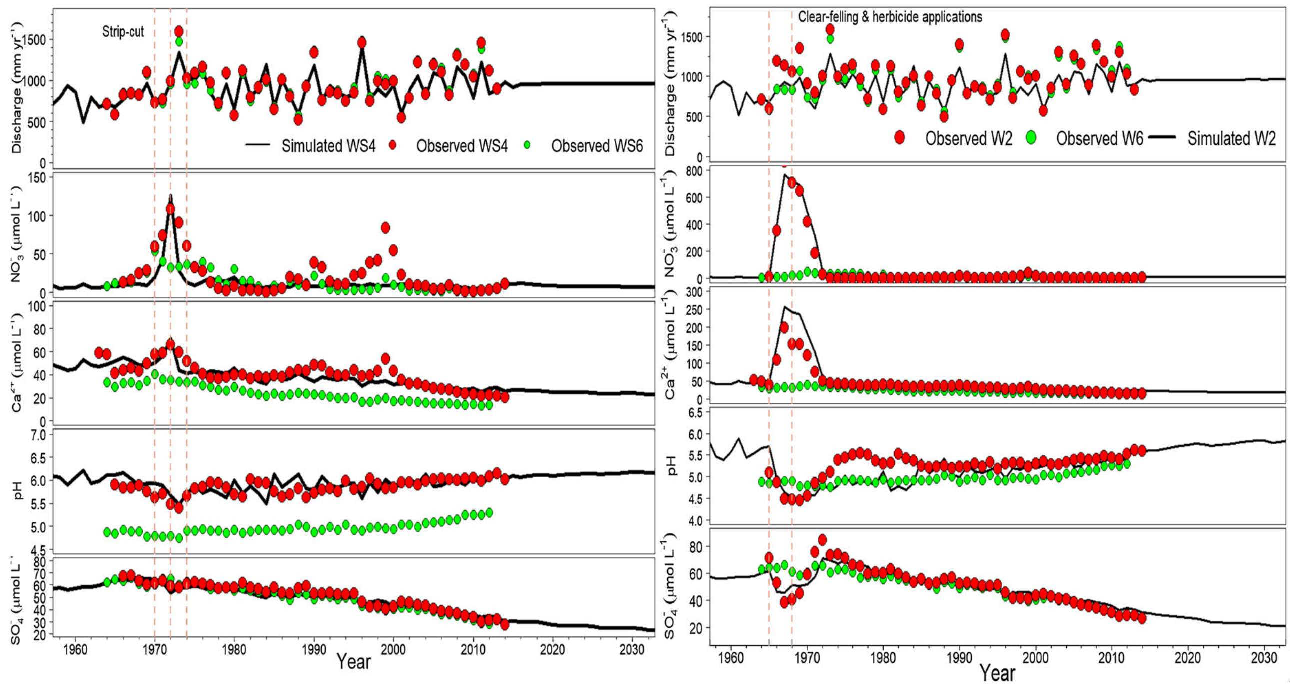

Fig. 3 Comparison between annual streamflow and volume-weighted stream water chemistry from PnET-BGC simulations and observations for W4 and W2, HBEF. Measured values are also shown for the reference watershed (W6). The timing of the cut is indicated by the vertical line

SO_4^{2-} in 1973. Stream SO_4^{2-} then began to increase as soil pH increased associated with forest regrowth; increased NO_3^- retention coincided with desorption of the SO_4^{2-} previously retained by soil.

Watershed 2 (W2) showed a similar pattern of nutrient losses following the 3-year period of devegetation/herbicide treatment, but with a greater degree of response than the other cut watersheds. The modified model reproduced peak values of the concentration and loss of elements in the stream water following the devegetation of W2 (1966–1968), and the rapid recovery of nutrient leaching to pre-cut values with new vegetation growth after 1969. Simulated peak annual volume-weighted concentrations of stream NO_3^-, Ca^{2+}, Mg^{2+} and Na^+ were 564, 198, 58 and 61 μmol L^{-1} which were comparable to the observed values of 529, 137, 44 and 51 μmol L^{-1}, respectively, during the treatment period. For most major elements, annual volume-weighted stream concentrations peaked in the 2nd year after the cutting (1967) and declined during the 3rd year. Modeled annual stream concentrations of NO_3^-, Ca^{2+}, Mg^{2+} and Na^+ also captured the observed patterns for the post-cut recovery period (1980–2014), with some overprediction for NO_3^- ($Mean_{obs\_a} = 5.5$, $NME_a = 0.8$) and Ca^{2+} ($Mean_{obs\_a} = 29$, $NME_a = 0.03$) and slight underprediction for Mg^{2+} ($Mean_{obs\_a} = 8.5$, $NME_a = -0.09$) and Na^+ ($Mean_{obs\_a} = 30$, $NME_a = -0.02$). During the treatment effect period (1966–1971), average annual stream water concentrations exceeded those of the reference watershed (W6) by a factor of 19 for NO_3^-, 4 for Ca^{2+}, 3 for Mg^{2+} and 1.5 for Na^+.

The simulated stream pH of W2 compared well with observations ($Mean_{obs\_a} = 5.3$, $NME_a = -0.04$). A decline in pH was observed the 1st year after the harvest (1966), and pH remained low during the devegetation/herbicide period until the 1st year after this treatment (1969) and then began to increase above pre-cut values. Low performance criteria values indicated that the model performed well in capturing stream SO_4^{2-} concentrations ($Mean_{obs\_a} = 47.2$, $NME_a = 0.01$, 1980–2014). The model depicted the enhanced adsorption of SO_4^{2-} following the W2 cut with the lowest stream SO_4^{2-} concentration in the 2nd year after the clear-cut (1967), followed by subsequent desorption of SO_4^{2-} from soil and increases in stream concentrations (Table 3).

Table 3 Comparison of modeled and observed values of stream constituents and model performance for the period after clear-felling W2 with follow-up herbicide application

Stream constituents	Post-harvest (1980–2014) _W2						
	Mean		STD		NME	NMAE	NRMSE
	Observed	Simulated	Observed	Simulated			
Flow	96.76	23.84	89.33	16.5	− 0.08	0.1	0.13
pH	5.35	0.11	5.15	0.24	− 0.04	0.04	0.06
Na^{+}	29.99	2.15	29.54	2.18	− 0.02	0.06	0.07
Mg^{2+}	8.5	1.99	7.71	1.09	− 0.09	0.16	0.19
Ca^{2+}	28.97	8.52	29.95	6.31	0.03	0.13	0.15
NO_3^{-}	5.46	7.37	9.89	1.99	0.81	1.42	1.61
SO_4^{2-}	47.19	11.63	47.75	9.78	0.01	0.06	0.07

Values represent mean and standard deviation of annual volume-weighted concentrations for the post-harvest (1980–2014 year) periods. Units for stream constituents are μmol L^{-1} (Flow; cm)

NEM normalized mean error; *NMAE* normalized mean absolute error; *NRMSE* normalized root mean squared error; *STD* standard deviation

Seasonal variations in streamwater chemistry

The modified model effectively depicted monthly variations in stream water chemistry for both cut watersheds (W2, W4). Simulated monthly variation of major elements followed similar patterns for both watersheds during the post-cut period 1980–2014 (Fig. 4). Lower monthly concentrations of stream NO_3^{-} were simulated during the fall and winter dormant season than during the growing season. The model slightly overestimated monthly concentrations of stream SO_4^{2-} during the growing season for W2

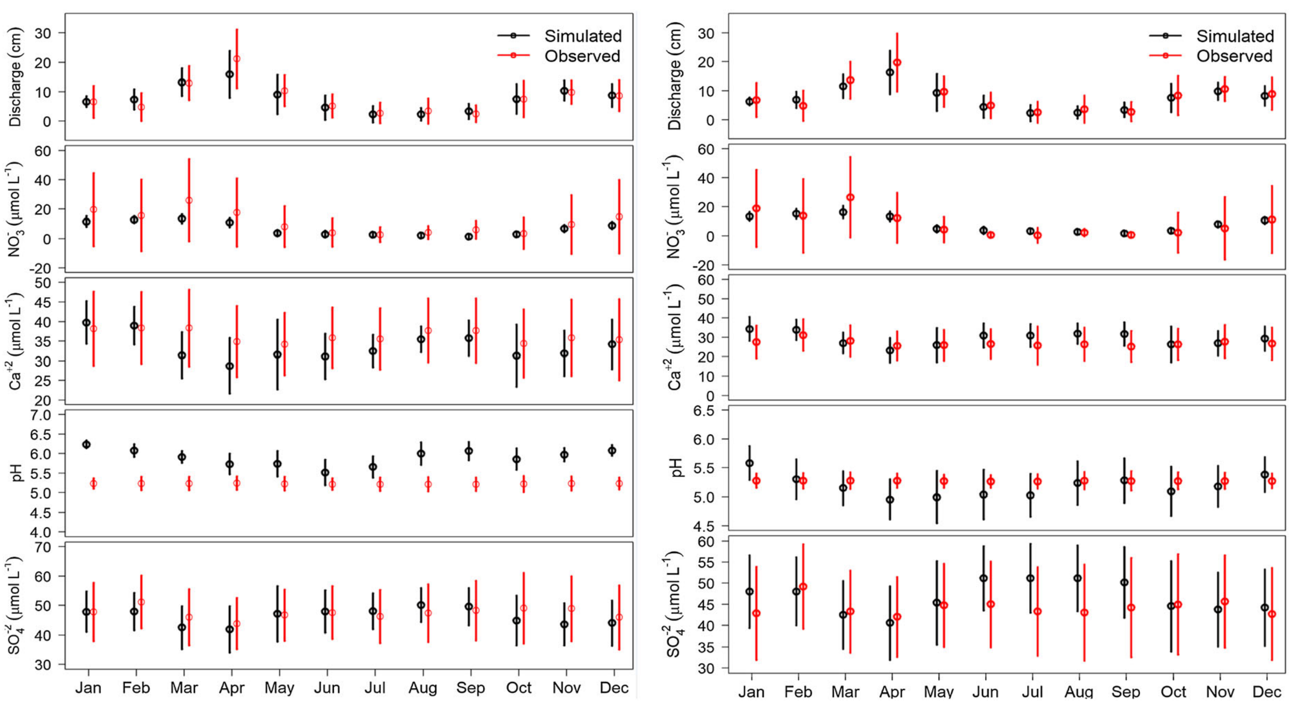

Fig. 4 Comparison between monthly patterns of stream water chemistry from PnET-BGC simulations with observations for W4 (left) and W2 (right) for the post-cut period (1980–2014). Error bars indicate standard deviation and monthly average values

and W4, while underestimating values during the fall and winter for W4. Simulated monthly stream Ca^{2+} concentrations were overestimated for W2 during the growing season, but generally underestimated for W4 except in early winter. Modeled monthly variations of pH for W2 compared well with the measured values but values were generally overestimated for W4.

Nutrient budget simulations

The modified PnET-BGC model was applied to evaluate changes in the source/sink behavior of major elements in the northern hardwood forest in response to cutting disturbances. For all three experimentally treated watersheds at the HBEF (W2, W4, W5), we summarize patterns of nutrient budgets for three different periods: pre-treatment (1960–1964); the treatment effect period which is characterized by a marked response in stream water NO_3^- to cutting (for W4:1971–1975; for W2:1966–1971; for W5:1984–1987; (Fakhraei et al. 2020)); and long-term post-treatment (2046–2050). The values were also compared with the reference watershed (W6).

For all watersheds, soil N mineralization and plant uptake were closely coupled in model simulations, with average annual rates of 100–111 kg N ha^{-1} year^{-1} and 104–114 kg N ha^{-1} year^{-1}, respectively, during 1960–1964 pre-cut period, and slightly higher rates for W5 and W6 followed by W4 and W2 (Fig. 5). Soil N mineralization and plant uptake were greatly influenced by cutting disturbances, and rates were reduced during the treatment effect periods. The greater mineralization rates were estimated for W2 followed by W4 and W5, while the lower rates of plant uptake were simulated for W2 and W5 compared to W4 for the treatment effect periods. Following the clear-cuts, simulated soil N mineralization and plant uptake increased, eventually reaching to pre-cut levels during 2046–2050.

Nitrification rates and stream N leaching followed similar patterns as N soil mineralization. W5 and W6 had similar rates of nitrification and stream N leaching of 3 and 1 kg N ha^{-1} year^{-1}, respectively, exceeding values of W4 and W2 (0.6 and 0.7 kg N ha^{-1} year^{-1} for nitrification rates and N stream leaching, respectively) during the 1960–1964 period. During the treatment periods, the greatest nitrification rates and stream N leaching were simulated for the devegetation/herbicide treatment of W2, followed by W5 and W4. During 2046–2050, the highest rates of nitrification and stream N leaching occurred for W2 and followed by W4, W6 and W5.

Net N release was calculated to estimate the discrepancy between major sources (mineralization/nitrification and atmospheric deposition) and sinks (plant uptake and drainage) of N in the watersheds for the pre-cut, treatment and post-cut simulation periods. This flux represents mobilization of legacy N that accumulated in the ecosystem (largely soil) from historical elevated atmospheric N deposition. Results showed that net N release remained nearly constant for all watersheds during both pre- and post-cut periods at a rate of 0.2 kg ha^{-1}. During the treatment interval, net N release of W2 increased to 1.3 kg N ha^{-1} year^{-1}, and for the whole tree harvest of W5 at the rate of 0.6 kg N ha^{-1} year^{-1}, and the strip cut of W4 with the value of 0.3 kg N ha^{-1} year^{-1} (Fig. 5).

Similar to N, soil Ca^{2+} mineralization and plant uptake were closely coupled over the simulations, with average annual rates of 42–65 kg ha^{-1} year^{-1} and 37–59 kg N ha^{-1} year^{-1}, respectively, during 1960–1964 pre-cut period. The greatest rates were evident for W4 followed by W5, W2 and W6 (Fig. 6). Soil Ca^{2+} mineralization and plant uptake were reduced during the clear-cut experiments, with the greatest decline occurring for W2, followed by W5 and W4. For all cut watersheds with regrowth of new vegetation during the 2046–2050 period, soil Ca^{2+} mineralization and plant uptake increased, though to lower rates than pre-cut values but maintaining the same order of rates as the pre-cut period. Similarly, stream Ca^{2+} flux showed a decreasing pattern from the pre-cut period (1960–1964) to the post-cut period (2046–2050), though during the treatment periods simulations showed increases in stream exports for all cut watersheds. Note that the Ca^{2+} weathering rate was assumed to be a constant value during the simulation period, with the highest rate for W4 (8.7 kg ha^{-1} year^{-1}), then followed by W2 (8 kg ha^{-1} year^{-1}), W5 (6.6 kg ha^{-1} year^{-1}) and W6 (4.8 kg ha^{-1} year^{-1}).

Soil SO_4^{2-} mineralization and plant uptake showed a pattern similar to Ca^{2+}, declining from the range of 20–16 to 16–14 kg S ha^{-1} year^{-1} for soil SO_4^{2-} mineralization, and 22–17 and 15–13 kg S ha^{-1} year^{-1} for SO_4^{2-} plant uptake, respectively, from the 1960–1964 period to 2046–2050 period (Fig. 7). Cutting decreased soil SO_4^{2-} mineralization and plant

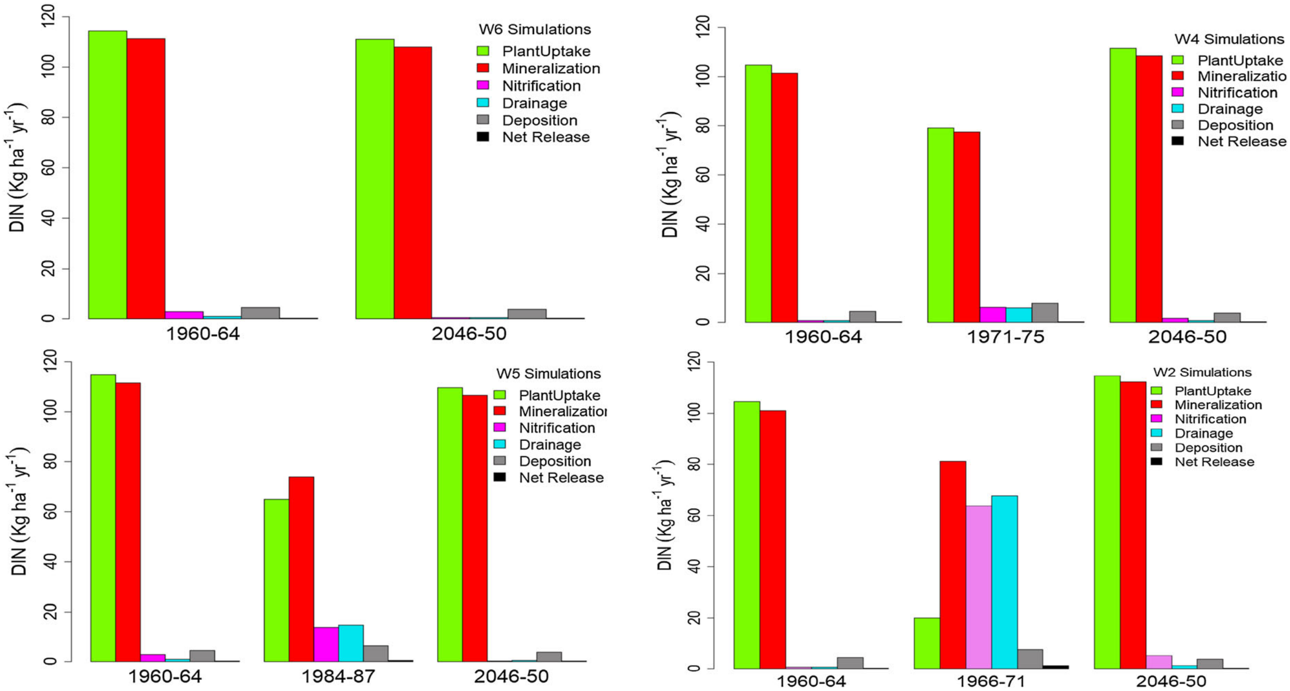

Fig. 5 Comparison of simulated annual average nitrogen budgets for the cut watersheds at HBEF including W5, W4 and W2 during the pre-cut period (1960–1964), the treatment effect period (W5:1984–1987; W2:1966–1971; W4:1971–1975) and post-cut period (2046–2050). Simulations are compared with simulated nitrogen dynamics for the reference watershed (W6) during 1960–1964 and 2046–2050 periods

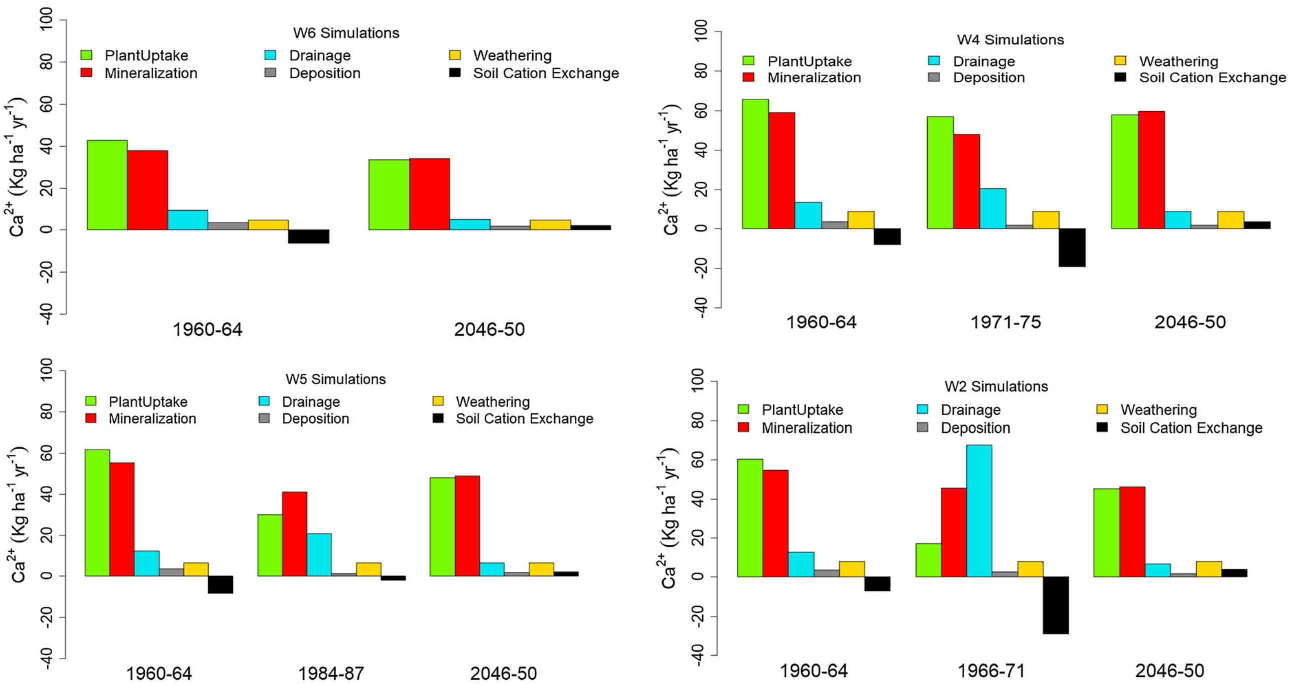

Fig. 6 Comparison of simulated annual average Ca^{2+} budgets for the cut watersheds at the HBEF including W5, W4 and W2 during the pre-cut period (1960–1964), treatment effect period (W5:1984–1987; W2:1966–1971; W4:1971–1975) and post-cut period (2046–2050). Simulations are compared with simulated Ca^{2+} budgets for the reference watershed (W6) during 1960–1964 and 2046–2050 periods

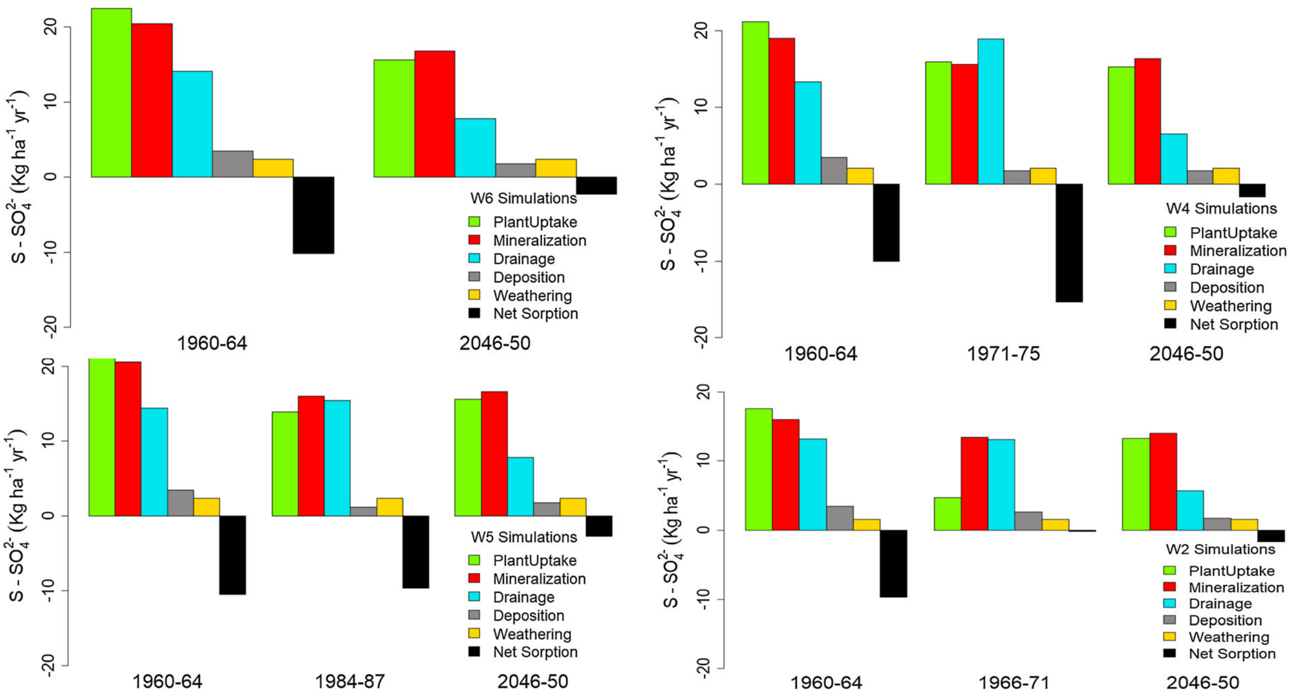

Fig. 7 Simulated SO_4^{2-} budgets for the clear-cut watersheds at HBEF including W5, W4 and W2 during the pre-cut period (1960–1964), treatment effect period (W5:184–187; W2:1966–1971; W4:1971–1975) and post-cut period (2046–2050). Simulated SO_4^{2-} budgets for the reference watershed (W6) during the 1960–1964 and 2046–2050 periods are shown for comparison

uptake in the watersheds during the treatments. Note, SO_4^{2-} weathering rates were assumed to be constant among the watersheds; however, to calibrate stream SO_4^{2-} we assumed a higher monthly S weathering rate for W5 (0.2 g m^{-2-} month^{-1}), than W4 (0.17 g m^{-2} month^{-1}) and W2 (0.13 g m^{-2} month^{-1}).

Simulated historical (1850) soil base saturation was estimated to be around 29% for W4 and 28% for W2, slightly exceeding the values for W5 (25%) and W6 (22%) based on PnET-BGC hindcasts. Long-term acid deposition resulted in declines in soil base saturation in all watersheds at the HBEF until around 2000 (Fig. 8). However, harvesting practices accelerated this loss of soil base saturation, particularly for W2. With regrowth of vegetation and reduction in acid deposition, soil base saturation began to increase gradually for all watersheds but at a slower rate for W2 than W4 and W5.

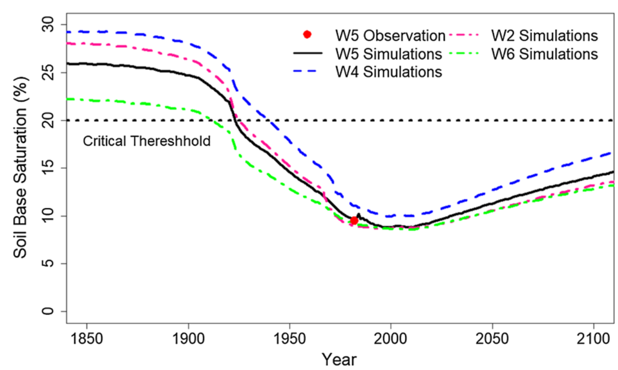

Fig. 8 Simulation of long-term changes of soil base saturation for the reference watershed (W6) and clear-cut watersheds W5, W4 and W2. Measured soil base saturation in 1982 is shown for W5. The horizontal dashed line shows a base saturation of 20% which is thought to be a critical threshold for sugar maple health (Sullivan et al. 2013)

Discussion

The modified PnET-BGC model was effectively able to capture both short- and long-term patterns of aboveground biomass accumulation for different experimentally cut watersheds at HBEF, though with better agreement with observed values for W5 and W4 than W2. The overestimation of biomass accumulation on W2 probably resulted primarily from the effects of the herbicide treatment in depleting the population of fast-growing pin cherry (Reiners 1992). Discrepancies between biomass measurements and simulations for the cut watersheds might be explained by the inability of PnET-BGC to depict shifts in tree species composition following the disturbance as PnET-BGC assumes constant vegetation parameters for the periods before and after forest cutting.

Measurements of regrowing vegetation on the cut watersheds at HBEF show that for the first 5 years after harvest the vegetation is mainly dominated by herbs and shrubs, and biomass accumulation rates closely follow similar patterns for the different cutting treatments (Bormann and Likens 1979; Martin and Hornbeck 1989; Fahey et al. 2005). Around 15 years after the cut, W5 biomass accumulated at a faster rate than for W4 and W2. In order to capture this greater total aboveground biomass for W5 than for W4 and W2 in model simulations, it was necessary to calibrate the model with a slightly higher minimum nitrogen concentration in foliar litter for W5 (0.96%) than W4 (0.915%) and W2 (0.911%). The approach of calibrating the model with higher canopy average foliar N concentration for W5 is consistent with studies reporting greater abundance of pin cherry, a species with exceptionally high foliar N (Mou et al. 1993) in the whole-tree harvested site compared with the strip-cut watershed (W4) or the devegetated watershed (W2) (Hornbeck, et al. 1986; Titus et al. 1998). Hornbeck et al. (1986) compared 10-year regeneration of the northern forest following the progressive strip-cut of W4 and a block clear-cut of W101 at the HBEF, concluding that total aboveground biomass accumulated at a much faster rate on the block clear-cut than on the strip cut, due to higher density of pin cherry in response to a higher initial nutrient release. However, strip-cut harvesting may result in a more desirable mix of commercial species in the regrowing stand.

Despite some initial differences in species composition and biomass accumulation rates among the cut watersheds at the HBEF, simulations of total biomass for all three treated watersheds (W2, W4 and W5) are consistent with the expected growth trajectory of the second- growth reference watershed (W6). These results suggest that though the different harvesting practices influence initial forest composition and growth, the overall impact on total aboveground biomass accumulation is minimal over the long-term at the HBEF. Longer-term model projections of aboveground biomass accumulation patterns and magnitudes (200–300 t ha^{-1}) are generally consistent with simulations using other models such as CENTURY 4.0 (Jiang et al. 2002) and NITMOD (Rolff and Ågren 1999).

The comparison of field measurements from different experimentally cut watersheds with the model simulations confirmed the ability of the modified PnET-BGC model to depict both short- and long-term hydrologic and biogeochemical responses to a range harvesting regimes. However, it is worth exploring some of the reasons for discrepancies between monthly/yearly stream discharge and stream water chemistry measurements and simulations. For example, the model performed better in simulating increases in annual stream flow following the treatment of W5 and W4 than for W2. Model underestimation of stream discharge for W2 might be attributed to overestimation of evapotranspiration during the period of herbicide application. Underprediction of major elements including annual volume-weighted concentrations of NO_3^-, Ca^{2+}, Mg^{2+} and SO_4^{2-} in stream water for W4 during the multiple years of the harvest and the period after the treatment might be explained by natural disturbances such as soil freezing and insect defoliation in the early 1970s and 1980s (Gbondo-Tugbawa et al. 2001; Fitzhugh et al. 2003), and an ice storm in 1998 at the HBEF that caused damage to vegetation and affected the stream water chemistry (Houlton et al. 2003). Another contributing factor may be that PnET-BGC is not spatially structured to depict the physical sequence of the actual strip-cut of W4. Discrepancies between seasonal patterns of the concentrations of major elements in streamwater can also be influenced by the minor disturbances discussed above. Other factors contributing to these model discrepancies include overestimation/underestimation in monthly values of stream flow, mineralization, and plant nutrient uptake.

Our simulations indicate that for all cut watersheds at HBEF NO_3^- concentrations in streams fell below the levels in the reference watershed (W6) for 10–15 years during the initial regrowth, a pattern consistent with the observations. There are several mechanisms that could explain this long-term pattern, including a corresponding decline in nitrification rates, greater uptake of N by the rapidly growing forest, increases in the immobilization of N by soil microbes, and/or an increase in denitrification (Hornbeck et al. 1986). Nutrient budget simulations suggest that this decline in stream NO_3^- could also be associated with decreases in N mineralization during the earlier years of regrowth in cut watersheds possibly because of reduction in litter inputs.

The simulated Ca^{2+} budgets indicated an overall depletion in pools of soil exchangeable Ca^{2+} for all watersheds at the HBEF, including the reference watershed (W6), consistent with long-term elevated SO_4^{2-} and NO_3^- deposition and leaching of soil available cations (Likens et al. 1996). Controls on emissions of SO_2 and NO_x and subsequent decreases in atmospheric S and N deposition and watershed SO_4^{2-} and NO_3^- leaching following the Clean Air Act and subsequent rules (Driscoll et al. 1998; Stoddard et al. 1999; Likens et al. 2002) have curtailed this depletion (Likens et al. 1996). The greatest depletion of soil exchangeable Ca^{2+} occurred on W2 due to the cutting treatment (1966–1971) and related leaching of NO_3^- from the delay of regrowth due to the herbicide treatment coupled with high acid deposition at this time. With regrowing vegetation and controls on emissions of SO_2 and NO_x, Ca^{2+} began to be retained by the soil exchanger eventually approaching steady state conditions (a net retention of 2–3.8 kg ha^{-1} -year^{-1}) for the simulated years 2046–2050.

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; Cleavitt et al. 2017). Simulation results are consistent with previous studies indicating that harvesting regimes with higher intensity can lead to greater depletion of exchangeable Ca^{2+} and consequent reductions in base saturation at the site (Hornbeck, et al. 1986; Aherne et al. 2012; Cleavitt et al. 2017). Simulation results also showed historically ($\sim$ 1850) greater soil base saturation percent for W4 and W2 due to their higher inherent weathering rates making these watersheds able to better withstand soil Ca^{2+} depletion, compared to W5 and W6 which are characterized by lower weathering rates, estimated from model calibration. These differences are probably associated with physiographic factors, especially topography, the thickness of glacial till and mineral content of soils. However, W2 with more intense NO_3^- leaching associated with the herbicide treatment showed the greatest decline in soil base saturation and a slower recovery pattern during forest regrowth, with values decreasing below the base saturation of W5 by the end of the simulation period. The results showed that the amount of stored Ca^{2+} in dead biomass that was left on the site after the treatment in W2 (152 kg ha^{-1}) could not offset the elevated of Ca^{2+} leaching (276 kg ha^{-1}) following the treatment, leading to a decline in soil percent base saturation. The results indicate that the amount Ca^{2+} leaching on W4 which was subject to a more moderate harvesting strategy of strip cutting was projected to recover soil base saturation at a faster rate following the regrowth of vegetation and controls on acid deposition compared to values for W6, W5 and W2.

Studies have shown that forest harvesting can impact site quality by removing essential nutrients (Federer et al. 1989). Moreover in acid sensitive regions impacted by acid deposition, forest harvesting can exacerbate the effects of chronic soil Ca^{2+} depletion on forest health (Juice et al. 2006; Cleavitt et al. 2017). Forest ecosystems characterized by low base saturation and exchangeable Ca^{2+} may experience limited regeneration and increased mortality of sugar maple over the long-term (Schaberg et al. 2006; Sullivan et al. 2013; Cleavitt et al. 2017). Some studies report a critical threshold of 20% soil base saturation for successful regeneration of sugar maple (Sullivan et al. 2013; Cleavitt et al. 2017). Repeated harvesting and intensive tree removal would be expected to aggravate chronic soil available Ca^{2+} depletion by acid deposition (Weetman and Webber 1972; Hornbeck, et al. 1986; Cleavitt et al. 2017). For example, Cleavitt et al. (2017) compared the recovery of forest vegetation of the whole-tree harvest treatment of W5 with the strip-cut on adjacent W4 at the HBEF to evaluate the effects of harvest intensity on soil fertility and species composition over 30 years following the treatments. They concluded that the whole-tree harvest of W5 resulted in greater removal of nutrient cations from the site both as timber products and in stream water and consequent regeneration failure for sugar maple. Our simulations amplify their study. We found that the higher Ca^{2+} weathering rate on W4 may have resulted in a slightly higher historical base saturation than W5 and likely facilitated the more rapid recovery (Fig. 8).

One weakness of the PnET-BGC model is that it does not depict changes in tree species composition following forest cutting, which is an important consideration in understanding watershed recovery. Simulating the dynamics of vegetation composition following disturbance would be a major undertaking;

model development to simultaneously depict the competition among tree species to determine composition would add a level of complexity to the multi-element soil-layer model that would be needed to comprehensively simulate the interactions of major elements with various tree species and these effects on forest growth.

Conclusions

The modified multi-element soil-layer model PnET-BGC was able to depict differences in stream water, soil chemistry and element budgets resulting from different forest cutting experiments. The model also captured the ability of all cut watersheds to limit stream nutrient losses by rapid regrowth of new vegetation. Biomass accumulation in all the cut watersheds was found to approach similar levels by the end of the simulation period (2100). Thus, despite some differences among treatments in effects on soil fertility and base saturation, the model would suggest a high degree of resilience in northern hardwood forests. However, note that we did not investigate the long-term effects of continuing these experimental cuttings on biomass production and soil fertility as forest management practices, though Valipour et al. (2021) conducted hypothetical simulations of the effects of cutting interval and intensity for W5. Increasing demand for bioenergy has necessitated forest managers to develop guidelines to satisfy multiple criteria for forest use, that include timber production and to maintain long-term forest sustainability (Seely et al. 2002; Wang et al. 2014; Creutzburg et al. 2016). Further investigation is needed to contrast different cutting strategies to assure that forest management can satisfy both long-term soil fertility and desired merchantable species composition. Moreover, further experimentation and modeling efforts will be necessary to improve understanding of the effects of forest harvesting approaches coincident with climate change (Valipour et al. 2021). The verified multi-element soil-layer model, PnET-BGC could be used as a diagnostic tool to gain a better understanding of complex interactions of ecological process and their response to multiple ecosystem stressors.

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Data availability The datasets generated during the current study are available from the corresponding author on reasonable request.

Declarations

Conflict of interest The authors declare that they have no conflict of interest.

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