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Modeling the interactions of forest cutting and climate change on the hydrology, biomass and biogeochemistry of a northeastern forest

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Abstract

Global and regional environmental disturbances, including harvesting and climate change, can lead to integrated and interactive effects on forest ecosystems, altering their structure and function, and therefore long-term sustainability. Understanding both short- and 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. Process ecosystem models are useful tools to improve predictive understanding of complex, interacting ecological process and their response to disturbance. 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 biomass, and soil 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 this disturbance regime. Moreover, the dynamics of nutrients to and from major pools, including mineralization and plant uptake, were poorly predicted.

The overall goal of this dissertation was to develop a simulation tool to evaluate short and long-term effects of harvesting on the hydrology and biogeochemistry of the northern forest. In the first phase of dissertation, PnET-BGC was modified and tested using field observations from an experimentally whole-tree harvested northern hardwood watershed (W5) at HBEF. In the second phase of dissertation, the parametrized/modified model was 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 confirm the ability of the model to depict ecosystem

response to a range of harvesting regimes. In the third phase of dissertation, the confirmed model was used as a heuristic tool to investigate long-term changes in aboveground biomass accumulation and nutrient dynamics under three different harvesting intensities (40%, 60%, 80% watershed cutting) for three rotation lengths (30, 60, 90 years) under both constant (current climate) and changing (MIROC5-RCP4.5) future climate through the year 2200.

In this dissertation, 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. Revisions of algorithms of PnET-BGC significantly improved model performance in predicting short- and long-term dynamics of major elements for evaluating effects of various forest cutting strategies at the HBEF. The comparison among cut watersheds showed that around 15 years after the cuts, W5 biomass accumulated at a faster rate than W4 and W2. Despite some initial differences in species composition and biomass accumulation rates among the cut watersheds, simulations of total biomass for all three treated watersheds (W2, W4 and W5) are consistent with the expected growth trajectory of a second-growth watershed (W6) at the HBEF. These results suggest that though the different harvesting practices influence initial forest composition and growth, the overall impact on total aboveground biomass is minimal over the long-term at the HBEF.

The modified two-soil-layer PnET-BGC was capable of capturing the immediate increase in stream concentrations of NO_3^-, Ca^{2+}, Mg^{2+} and Na^+ as well as enhanced adsorption of SO_4^{2-} following the treatments and indicated a greater response for the devegetated W2 and the whole-tree harvested W5 than the strip-cut W4. Modeled soil solution Bs horizon and stream water chemistry successfully captured the rapid recovery of leaching nutrients to pre-cut levels after the treatments. Accurate simulation of vegetation regrowth allowed for improved prediction of the chemical response of soil and streamwater to cutting disturbance, indicating the important role of plant uptake in regulating the recovery of the forest ecosystem. Simulations for W2 showed more intense NO_3^- leaching associated with the herbicide treatment resulting in an accelerated decline in soil base saturation, to values lower than those anticipated from the effects of acid atmospheric deposition alone, and a slower recovery pattern during forest regrowth by the end of the simulation period (2100). A first-order sensitivity analyis showed that simulations by the model to a given level of perturbation of input parameters are more sensitive under mature forest (pre-cut) conditions than for an aggrading forest (post-cut conditions).

Simulations of the interactions between forest harvest practices and future climate change for W5 demonstrated the greater sensitivity of forest ecosystem nutrient pools to logging strategies under climate change which included fertilization effects of atmospheric carbon dioxide, relative to constant climate conditions. These effects are accentuated with a shortening of the length of cutting interval and increasing forest harvesting intensity. Simulations of both constant and varying climate conditions considered showed greater sensitivity to varying the length of cutting period than altering cutting intensities. My simulations suggest that tree harvesting under constant current climate should affect living tree biomass and woody debris more than soil carbon, while under climate change, loss of soil organic matter pools may adversely affect site fertility. Depletion of soil base cations is accelerated under climate change due to increases in soil mineralization, coupled with increased plant uptake and enhanced biomass accumulation. Nitrogen is predicted to be the element which experiences the greatest relative loss over both short- and long- periods under different harvesting strategies, particularly with changing climate. Simulations show that all management options under climate change enhance both timber production and overall carbon storage in comparison to stationary climate, but with greater potential for a reduction in long-term soil fertility.

Modeling the interactions of forest cutting and climate change on the hydrology, biomass and biogeochemistry of a northeastern forest

by

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1. Introduction

Forest ecosystems provide a variety of important ecological services, such as wildlife habitat, clean air and water, sequestration of atmospheric carbon and nitrogen, and production of biomass. Anthropogenic activities, such as harvesting, can change the structure and function of forest ecosystems and the services provided. Ecosystem response to harvesting disturbance varies remarkably as a function of site characteristics, forest species composition, land use history, and the method, intensity and frequency of harvesting (Kreutzweiser et al., 2008). 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 ecosystem services due to limited information on land use history and long-term time series observations. Repeated clear-cuts could diminish nutrient availability particularly for carbon, nitrogen and calcium and ultimately limit plant uptake and forest productivity (Cleavitt et al., 2018; Federer et al., 1989; Kreutzweiser et al., 2008; Walmsley et al., 2009). Projected changes in global climate resulting from increasing greenhouse gas concentrations are also expected to bring a variety of consequences to forested ecosystems over the next century (Creutzburg et al., 2016). Few studies have been conducted to assess the complex interaction of harvesting disturbance with future projected climate change over the long term. It is anticipated that increases in CO_2 concentration, precipitation and temperature could increase forest productivity over the rotation length of harvesting as long as increases in temperature do not cause water stress or limit the availability of N for vegetation growth through increased soil decomposition. Concurrent exposure to multiple environmental change factors complicates predictions further due to interactions that can either amplify or compensate for the effects of individual stressors. With the acceleration of land development and the demand for forest products, understanding both short- and long-term impacts of harvesting practices (e.g., cutting rotation length, intensity) on forest dynamics in the context of climate change is a key factor in developing criteria and guidelines for sustainable forest management practices (Mina et al., 2017; Peng et al., 2002).

Experimental manipulations and historical observations provide some insight into short-term effects of different ecosystem stressors. However, to make informed management decisions and evaluate the best options to maintain forest productivity and function, it will be necessary to conduct a long-term assessment of the effects of harvesting practices coupled with climate change on hydrological and biogeochemical processes of forest watersheds. To assess the potential impacts of harvesting strategies under changing climate, a multi-faceted approach is needed that is capable of resolving multiple stressors. Mathematical models allow for the extrapolation of short-term observations of hydrology and nutrient dynamics (years to decades) to longer-time scales (decades to centuries) and to probe how multiple disturbances simultaneously influence forest ecosystems. Hydrochemical watershed models, therefore, are important tools to help to gain a better understanding of the complex, interacting effects on ecological processes.

1.1. Dissertation Objectives

The overarching goal of this dissertation is to evaluate both short-term and long-term responses of various harvesting strategies (i.e., cutting rotation length, intensity) on hydrology, biomass accumulation and nutrient dynamics in soil and streamwater of different watersheds of a northern hardwood forest at the Hubbard Brook Experimental Forest (HBEF), New Hampshire, USA, using a hydrochemical model, PnET-BGC. PnET-BGC is a forest-soil water model that simulates energy, water, and elements fluxes at the watershed scale (Gbondo-Tugbawa et al. 2001).

The following are the specific objectives of the dissertation:

1. a) To evaluate the model processes responsible for the prediction of nutrient leaching losses, and modify the algorithms as needed to improve model performance; b) to simulate aboveground biomass accumulation and net primary productivity (NPP) following forest cutting of watershed 5 (W5) at the HBEF; c) to project monthly and annual stream water hydrology, soil solution chemistry, stream water chemistry and nutrient budgets; d) to compare model simulations with experimental observations and reference measurements for W5 and W6, respectively; and e) to conduct an input parameter sensitivity analysis for both pre- and post-treatment periods.

2. a) To apply the modified, parametrized model to different cutting experiments at the HBEF (W2, W4) as a test of model performance; and b) to use the modified model to compare and gain insight on how the forest ecosystem responds to different forest cutting techniques over both the short- and long-term.

3. a) To project short- and long-term patterns of biomass accumulation and changes in hydrology and nutrient pools and fluxes in soil and streamwater as a function of harvesting intensity and rotation length; b) to couple hypothetical tree-cutting scenarios with future projections of climate change and carbon dioxide fertilization; and c) to examine the interplay between these two types of disturbances and inform forest management decisions by executing the trade-offs between timber production and ecosystem storage of nutrients.

These goals and these objectives were accomplished by analyses of field observations and model calculations. The following hypotheses were developed based on a review of the literature (provided in Chapter 2):

1.2. **Dissertation Hypothesis**

The following are the specific hypotheses of the dissertation:

1.2.1. Hypothesis 1:

The removal of vegetation by logging disturbance will disrupt biogeochemical process of forest ecosystems resulting in rapid short-term increases in nutrient availability in soil solutions and export of dissolved nutrients to the receiving waters. There are a variety of factors that interact to diminish nutrient losses and regulate ecosystem recovery following forest cutting including regrowth of new vegetation and rapid plant nutrient uptake, and reduction in mineralization and run-off. Depiction of these processes in a forest biogeochemical model to simulate harvesting effects offers an effective comprise among simplicity, biological realism and predictability.

1.2.2. Hypothesis 2:

Various logging practices, including harvesting intensity and frequency of logging influence the extent and duration of nutrient losses by biomass removal and drainage. Predicting the responses of forest ecosystems to a variety of harvesting techniques over short-term and long-term is a key factor for understanding the sustainable management of forests. The comparison of field measurements from different experimentally cut watersheds with simulations using a well-established biogeochemical model will confirm the ability of the model to depict ecosystem response to a range of harvesting regimes.

1.2.3. Hypothesis 3:

Intensive timber harvesting, consisting of short length rotation and high biomass intensity has the potential to reduce forest productivity and nutrient availability over a long period. Anticipated changes in climate in the Northeast (i.e., increases in temperature and precipitation, carbon dioxide fertilization) will interact with forest harvesting to diminish the sustainability of forest stands due to accelerated loss of soil carbon and elevated leaching. Carbon dioxide fertilization will attenuate these effects.

I addressed these goals, objectives and hypotheses through a three-phase research approach. In the first phase, I evaluated the capability of a version of PnET-BGC that considers multiple-layers of soil to simulate the short-term response of a forest watershed to intensive logging treatment (whole-tree harvest, W5). This was 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. This phase consisted of developing input data (meteorology and atmospheric deposition, forest disturbance from cutting and biomass removal), parametrization and model algorithm modifications. Model results were compared against measured hydrology, biomass and soil solution and stream water chemistry for a commercially whole-tree-harvested watershed (W5) at the HBEF. Model simulations considered major element fluxes including mineralization, nitrification and plant uptake in the harvested ecosystem. In this phase of study, I also conducted a detailed sensitivity analysis of model parameters for periods before and after harvesting. This analysis helped to identify the parameters that are most sensitive in model simulations and how this sensitivity changes over for rapidly regrowing and a more mature forest.

The second phase of the dissertation involved application of 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 model simulations under a range of cutting approaches, the assessing depletion of nutrient capital (Federer et al., 1989) and effects on carbon sequestration.

In the third phase of study, I assessed the effects of alternative harvesting practices (harvesting method, intensity and rotation length) on long-term ecosystem productivity, carbon sequestration (biomass, litter and soil carbon) and nutrients reserves through a series of hypothetical simulations. These hypothetical harvesting scenarios were also coupled with a future climate change scenario to evaluate the potential effects of climate change and forest fertilization from increases in atmospheric carbon dioxide on forest recovery from clear-cutting.

2. Literature Review

2.1. Harvesting impacts on forest ecosystems

Forest harvesting causes short- and long-term changes to the internal processes of forest ecosystems, altering energy flux, hydrology, biogeochemical processes, 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 test model performance and 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 the dynamics of 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 nutrient 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).

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 a lack of time series observations (Kreutzweiser et al., 2008; Thiffault et al., 2011).

Process models can be used to determine the response of the structure and function of terrestrial ecosystems to disturbance. Such models are effective tools that can provide insight into transformations of important processes 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., 2003a). 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 Agren (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 et al., 2000; Prescott, 2005; 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 the 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.

2.2. **Assessment of harvesting practices under future climate change**

Climate change is also increasingly recognized as an important regulator of ecosystem structure and function (Bormann et al., 1977; Dib et al., 2014; Huang et al., 2007; Kirilenko and Sedjo, 2007; Pourmokhtarian et al., 2017). Projected changes in future climate including increases in air temperatures and changes in precipitation by atmosphere-ocean general circulation models (GCMs) and also projected increases in atmospheric CO_2 which are expected to influence forest regeneration, growth, mortality and biogeochemical processes (Peng et al., 2002). Such changes can alter ecosystem services of forest ecosystems including biomass production for energy (Creutzburg et al., 2016).

Increasing demand for timber harvesting and forest products, and concern over increasing concentration of greenhouse gases have compelled forest managers to consider the ability of forest ecosystems to sequester carbon and conserve nutrients when developing long-term management strategies (Seely et al., 2002). An important embedded concept in the term "sustainability of timber yield" is: keeping forests economically profitable while maintaining the structure and function of the forest ecosystems (Peng et al., 2002). Understanding both short- and 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 (Mina et al., 2017; Peng et al., 2002). However, climate is changing and this will also disrupt forest ecosystems processes (Boisvenue and Running, 2006; Ollinger et al., 2008). As the time scale of climate change is comparable to that of forest ecosystem recovery and development following harvesting, it will be important to investigate forest harvesting effects in the context of climate change.

Only a few studies have been conducted to examine the long-term ecological impacts of alternative logging events and these have generally relied on modeling approaches (Blanco et al., 2005; Jiang et al., 2002; Rolff and Ågren , 1999; Vadeboncoeur et al., 2014; Wei et al., 2003b). This research has generally shown that repeated clear-cuts can diminish soil nutrient availability, ultimately limiting plant uptake resulting in declines in forest productivity over the long term. None of these studies have considered the effects of climate change when quantifying the response of ecosystem nutrients pools to harvesting.

Climate change is anticipated to have major consequences for forest ecosystems (Creutzburg et al., 2016; Dai et al., 2016; Ollinger et al., 2008). Atmosphere-Ocean General Circulation Models (AOGCMs) from the Intergovernmental Panel on Climate Change Fifth Assessment Report (IPCC AR5) (Flato et al., 2013) estimate potential changes in climate during the 21st century under several greenhouse gas forcing scenarios called, representative concentration pathways (RCPs) (Moss et al., 2008, 2010). For example, atmospheric CO_2 concentrations are anticipated to reach approximately 940 ppm CO_2-equivalent under the RCP8.5 scenario and approximately 540 ppm CO_2-equivalent under the RCP4.5 scenario by the end of the current century (i.e., 2099) (Moss et al., 2008, 2010; Riahi et al., 2011; Thomson et al., 2011). The magnitude of corresponding changes in climate attributes are expected to be highly variable over space and time (Flato et al., 2013; Pourmokhtarian et al., 2017; Wuebbles et al., 2017). Some studies have suggested that regardless of the effects of variations in CO_2, temperature, and precipitation, forest growth rates have been increasing since the middle of the 20th century (Bascietto et al., 2004; Boisvenue and Running, 2006; Kirilenko and Sedjo, 2007). In contrast, other reports indicate varying trends during different periods or overall declines in forest productivity, mainly related to water stress, and increases in insects, disease and wildfire as a result of increases in temperatures (Boisvenue and Running, 2006; Creutzburg et al., 2016; Miller et al., 2004; Schimel et al., 2001).

Climate change will likely interplay with timber harvesting in complex ways (Creutzburg et al., 2016; Wu et al., 2017). Limited studies have examined the interaction of frequent intensive harvesting of forests coincident with changes in climate (Scheller and Mladenoff, 2005; Wu et al., 2017). For example, Aherne et.al (2012) modeled the interactive effects of forest harvesting and climate change on soil and stream chemistry using the hydrochemical model MAGIC (Model of Acidification of Groundwater in Catchments), concluding that accelerated weathering rates under future climate change can compensate loss of nutrients due to harvesting. However, effects of CO_2 fertilization on forest growth and changes in soil mineralization due to increases in temperature and changes in soil moisture were not considered in this analysis. LANDIS-II (a spatial forest landscape model) was applied to examine long-term effects of forest management and climate change on aboveground biomass, concluding that climate change is unlikely to significantly change forest carbon storage (Creutzburg et al., 2016). However, that study also did not consider the effects of CO_2 fertilization on forest growth. Wu et.al (2017) investigated the integrated effects of harvesting and climate change, finding declines in aboveground biomass as a result of long-term cutting (Wu et al., 2017). However, this work did not consider the effects of CO_2 fertilization on aboveground biomass production.

Further experimentation and modeling efforts will be necessary to improve understanding of the interactions between forest cutting approaches and future climate change. Such research should enhance predictive capabilities that could ultimately be used to develop sustainable forest management practices. Process ecosystem models are effective analytical tools to depict complex interactions among ecosystem components and their response to stressors, providing insight on the transformations of key processes that regulate the structure and function of forest ecosystems, and eventually could be used to inform decision making. Few studies have rigorously compared model simulations against experimental harvesting data to test and verify simulations that would improve confidence in the extrapolation of short-term observations of nutrient dynamics to longer time scales. Previous modeling approaches have largely focused only on the dynamics of C and/or N stocks in soil or vegetation to evaluate the impacts of various logging regimes (Mina et al., 2017; Shifley et al., 2017) and most have not considered CO_2 fertilization effects on forest growth (Aherne et al., 2012; Creutzburg et al., 2016; Wu et al., 2017).

3. Methodology

3.1. 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 forest management through monitoring and large scale field experiments. HBEF encompasses ten experimental watersheds with relatively 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 except one (W101), and five watersheds have been experimentally manipulated (Ca was applied to Watershed 1 and Watersheds 2, 4, 5 and 101 were cut) (Figure 3.1, Bormann and Likens, 2012; Likens et al., 1970; http://www.hubbardbrook.org). In this dissertation, I 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, without experimental manipulation. W5 is adjacent to W6 with an area of 21.9 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. 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 three 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. The remaining two sets of strips were harvested in 1972 and 1974. 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 3.1).

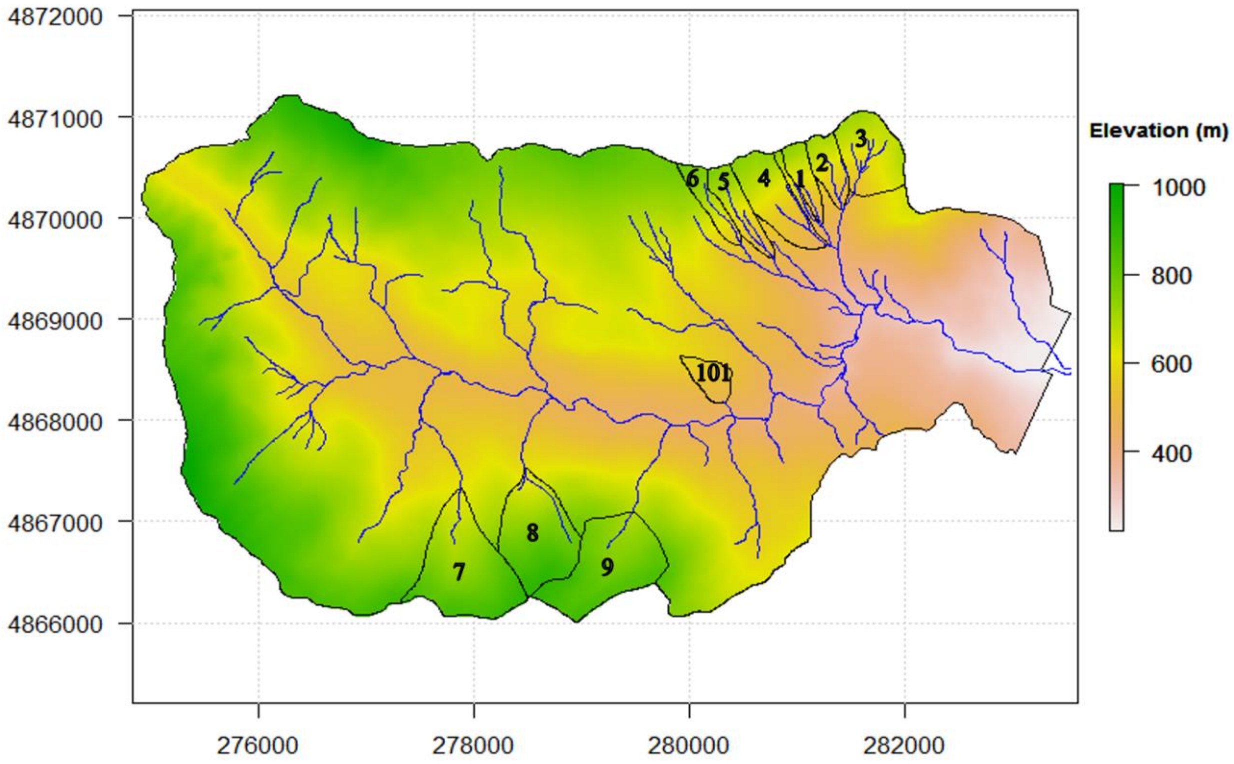


Figure 3.1. Elevational map and location of experimental watersheds at the Hubbard Brook Experimental Forest, NH.

The climate of the HBEF is humid-continental, with short, cool summers and long, cold winters. Mean monthly air temperature varies approximately between -9 °C 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 allegheniensis*). At higher elevation, vegetation includes red spruce (*Picea rubens*), balsam fir (*Abies balsamea*) and paper birch (*Betula papyrifera*).

Table 3.1. A summary of disturbance history of Watersheds 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.

3.2. **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 (Gbondo-Tugbawa et al., 2001). PnET-BGC was developed by linking two submodels, PnET-CN (Aber et al., 1997; Aber and Driscoll, 1997; Aber and Federer, 1992) 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, organic matter 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 (Figure 3.2).

In this dissertation, I 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.

PnET-BGC is typically 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 are 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.

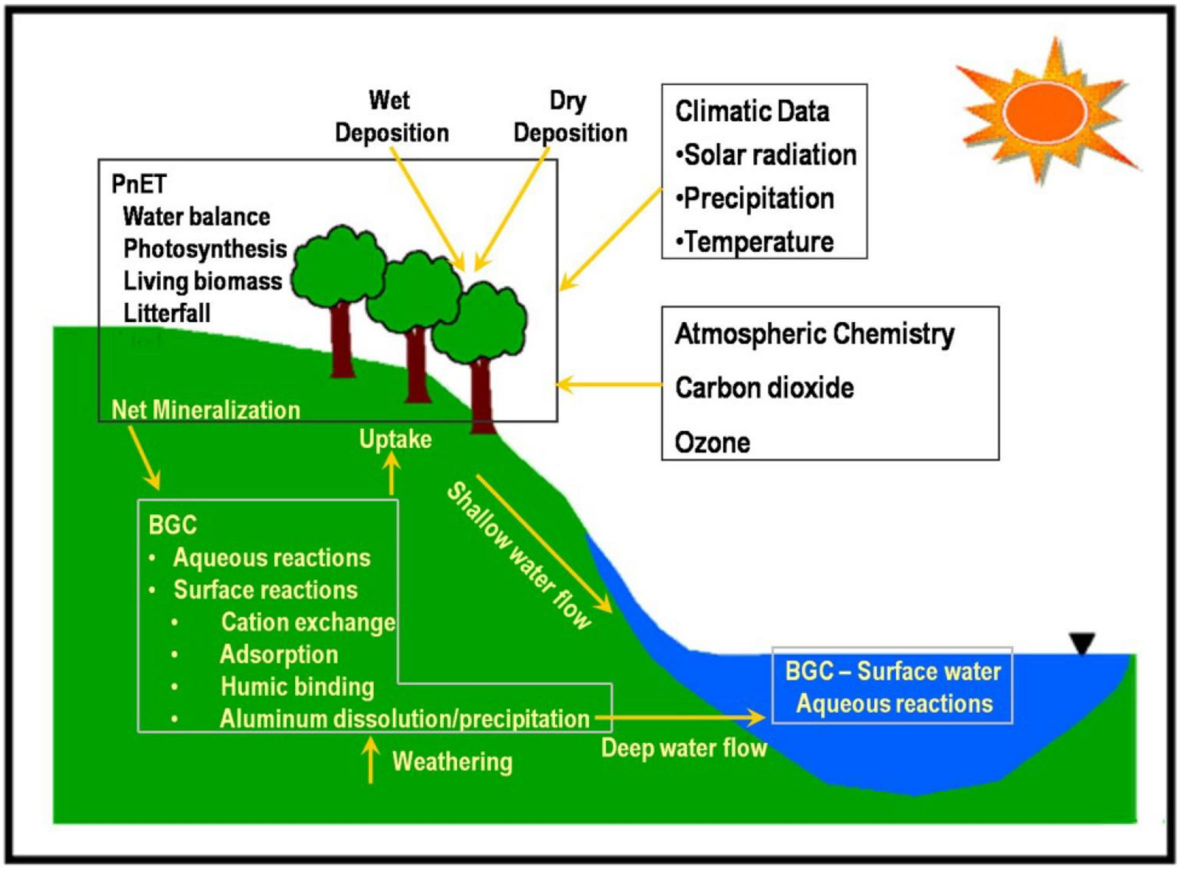


Figure 3.2. Schematic illustration of inputs, processes, interactions and outputs of PnET-BGC.

I added algorithms to the multilayer soil version of PnET-BGC (Chen and Driscoll, 2005) to consider the effects of increasing atmospheric CO_2 on forest ecosystem processes. The effects of atmospheric CO_2 on vegetation, including the response of stomatal conductance and fertilization of biomass, were implemented using a sub-model of photosynthesis and phenology developed by (Ollinger et al., 2008, 2002, 1997).

3.3. Data preparation and model inputs

3.3.1. Meteorological and atmospheric deposition data

I used the same methodology 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 summary of the data sets used in this dissertation was provided in Table 3.2. For the period prior to measurements (1850-1963), reconstructed input data were used (Chen et al., 2004; Fakhraei et al., 2016, 2014). The reconstruction of atmospheric wet deposition is described in Tables 3.3 and 3.4. PnET-BGC uses dry deposition calculated from dry to wet deposition ratios for each element (Table 3.5).

Table 3.2. Major data sets used in this dissertation.

Database	Public Available Source	Major Information Provided
National Atmospheric Deposition Program (NADP)	http://nadp.sws.uiuc.edu/NTN/	Precipitation and wet deposition
Clean Air Status and Trends Network (CASTNET)	https://www.epa.gov/castnet	Dry deposition
Hubbard Brook Experimental Forest, New Hampshire (HBEF)	http://www.hubbardbrook.org/data/dataset_search.php	Meteorological data, climate change, streamwater chemistry and vegetation data
Syracuse University, Civil & Environmental Eng. Department		Soil chemistry data
GIS Map Source	http://data.hubbardbrook.org/gis/	Hubbard Brook GIS Coverages

Table 3.3. Empirical relationships between annual national emissions (Tg/yr) and concentration of solutes in wet deposition (mg/l) measured at HBEF, 1964-2012.

	Concentration in Wet Deposition (mg/l)							
Constituents	Ca^{2+}	Mg^{2+}	K^{+}	Na^{+}	NH_4^{+}	NO_3^{-}	Cl^{-}	SO_4^{2-}
Intercept (mg/l)	0.04	0.01	0.04	0.07	0.06	-0.27	-0.07	-0.02
Slope (mg/l)	0.01	0.002	0.002	0.003	0.005	0.08	0.06	0.09
R^2	0.51	0.41	0.19	0.17	0.15	0.54	0.47	0.89
P-value	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001
National Emission	PM10	PM10	PM10	PM10	NO_x	NO_x	PM10	SO_2

Table 3.4. Wet deposition reconstruction for model simulation over 1000-2200, W5, HBEF.

Time periods	References
1000-1850	Galloway et al., 1982 (Poker Flat, Alaska)
1850-1900	Linear ramp increase
1900-1962	Linear regression model (Table 3.3)
1962-2014	Long-term measurements
2014-2200	Average the last three years of observations

Table 3.5. Constant dry to wet deposition ratios over (1000-2200) were applied for base cations, Cl^-, NH_4^+, NO_3^- and a varied range for SO_4^{2-} in order to balance streamwater SO_4^{2-} chemistry for model simulations.

Chemical constituents	Dry to wet deposition ratio (1000-2200)	References
Na^+	0.3	Gbondo-Tugbawa et.al. 2001
Mg^{2+}		
K^+		
Ca^{2+}		
NH_4^+	0.07	Average current measurements (1964-2012); CASTNET/NADP
Cl^-	0.01	
NO_3^-	0.087	
SO_4^{2-}	0.1 (1000-1900; 1990-2200)	
	0.64 (1900-1970)	Stream calibration (Lovett et al., 1996)
	0.34 (1970-1990)	

3.3.2. 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). 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) (Dahlgren and Driscoll, 1994; Johnson et al., 1991, 1997). Soil chemical data for W5 are available for pre-treatment (1983) and three post-treatment years (1986, 1991, 1997) (Johnson et al., 1997, 1991). Note, there are no field measurements on soil characteristics for W2 and W4 following their harvest to compare with the model simulations. Model simulations for stream water hydrology and chemistry of studied watersheds were compared with the measured data during 1963-2013 (Likens, 2017).

For W5, weathering rates for the upper soil 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). To estimate the weathering rates for W2 and W4, I used weathering rates obtained for W5 as the initial values and then these values were adjusted through calibration with observed stream water chemistry. Parameters and variables used in the model calibration for the study watersheds are summarized in Table A1-A3 in the Appendix. 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., (1995,1997) and Gbondo-Tugbawa et al., (2001).

3.3.3. 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) and assumed to be constant over the simulation period. 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). Note, in order to capture greater total aboveground biomass for W5 than for W4 and W2 in model simulations, I calibrated the model with slightly higher minimum nitrogen concentration in foliar litter for W5 (0.96%) than W4 (0.915%) and W2 (0.911%).

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 A1-A3 in Appendix); 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.

3.3.4. 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. For W5, I 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 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, I 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 3.1).

For W2 and W4 I did not have access to the detailed information indicating percentage of uncut trees along the watershed boundary or stream channel. Hence, to simplify comparison among watershed simulations, I assumed 8% uncut trees for all treated watersheds. In December 1965, all the trees and shrubs on watershed 2 were felled and left in place. During the growing seasons of 1966, 1967 and 1968, herbicides were applied to the watershed to prevent vegetation regrowth. To simulate W2 disturbance, I assumed mortality of 92% of forest biomass for each year during 1965-68 in which all dead trees were left in place (0% removal). Since PnET-BGC is limited in its spatial depiction, it was difficult to represent the strip-cut that occurred in W4. I assumed 35% of forest biomass removal for the years 1970 and 1972, and 92% forest biomass removal for 1974 (Table 3.1). This assumption allowed for a better match of both aboveground biomass and stream water chemistry simulations with field observations.

3.3.5. Biomass studies and calculations

A total forest inventory was conducted in W5 during mid-summer of 1982 to quantify biomass prior to the cut. I 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, I used an annual average biomass increment on W6

(2.98 t ha^{-1} 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, trees were measured ≥ 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, trees were measured ≥ 1.5 cm DBH in random 1m 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 $\leq$ DBH < 7.5 cm) were measured in a nested 30 m^2 transect.

Seventy 10 x 10 m permanent quadrats were established in an evenly distributed, stratified-random manner on W2 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 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). Forest inventory data for W2 were not available on HBEF website to estimate total aboveground biomass after the treatment for W2. Therefore, I used published data from a previous study conducted by Fahey et.al (2005).

Regeneration on the strip-cut W4 has been surveyed on permanent plots at 1- to 4-year intervals since the harvest. Fifty-seven 25 x 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 m to 550 m), 18 at mid-elevation (550 m to 650 m), and 20 at high elevation (650 m to 730). From these plots, individual stems by species for trees, shrubs, and herbs were classified (Martin et al., 1989). Total aboveground biomass values for W4 (1969- 2011) were obtained through personal communication (unpublished data by Dr. John Battles).

3.3.6. Simulation of hypothetical management scenarios

To conduct hypothetical cutting scenarios for W5, I developed multiple management scenarios, varying the rate and intensity of harvesting, combined with scenarios of stationary and hypothetical future climate change. All management scenarios were conducted for a 180-year period (e.g. 2020-2200). To evaluate forest response to various management scenarios, I defined various cutting scenarios (CS) including three cutting rotation lengths (30, 60 and 90 years) with three watershed area cutting levels (cutting intensity 40%, 60% and 80%), in addition to the reference no cutting scenario (NCS). The 30-year cutting scenarios simulates harvest for pulp (e.g. paper, biofuel). The 60- and 90-year cutting scenarios simulate raw timber harvest. I coupled these management scenarios with two climate scenarios including continuing the current climate (stationary climate) and future climate projections from a single AOGCMs model. Hence, a suite of 20 scenarios were developed for this analysis.

AOGCMs from the Intergovernmental Panel on Climate Change Fifth Assessment Report (IPCC AR5) (Flato et al., 2013) estimate potential changes in climate during the 21st century under a suite of greenhouse gas forcing scenarios called, representative concentration pathways (RCPs) (Moss et al., 2008). Downscaled AOGCM climate projections for the emission scenarios of RCP 4.5 and RCP 8.5 for the HBEF show increases in average annual air temperature ranging from 3.02 to 8.22 °C and in average annual precipitation ranging from 15.44 to 29.9 cm for the period 2070–2100, relative to the reference period 1970-2000 for W5.

I selected outputs from the Model for Interdisciplinary Research on Climate version 5 (MIROC5) of the Center for Climate System Research, Japan (Watanabe et al., 2010) under a lower CO_2 emissions scenario (RCP 4.5) which projects moderate changes in climate variables. The results of MIROC5 were statistically downscaled to obtain a higher spatial resolution for the HBEF (Hayhoe et al., 2008; Pourmokhtarian et al., 2017). I used downscaled MIROC5 outputs (RCP 4.5) for the 2018-2100 period. To extrapolate climatic variables for the period beyond of that of MIROC5 outputs (years 2101-2200), monthly averages of the last five years (2096-2100) of the climate projections were used.

3.4. 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 of W5 at HBEF 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, I 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. I then examined whether underprediction of plant nutrient uptake might be responsible for the 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.

In the original simulations there was a prolonged period of elevated nitrification that continued for approximately a decade after the clear-cut. In order to improve model performance for short-term simulations, I hypothesized that when the forest is young with limited litter input, there would be a decrease in the decomposition rate of organic matter. 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. I 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 A1 in Appendix). However, this parametrization was not adequate to capture observations. I also modified the algorithm that calculates nitrification rate by adding a new constant parameter (Table A1) that allows for greater ammonium assimilation by plants and thereby reduces leaching of NO_3^-. I 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 (Figures 4.1c,4.5 and Table A1-A3).

3.5. Model evaluation and sensitivity analysis

Three statistical methods were used to evaluate model performance for each watershed 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 using watershed 5 simulations 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 (1970-72) and 5 years after the cut (1988-90) to compare model sensitivity for the mature and strongly aggrading forest, respectively. For this analysis, I 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:

Equation [1]: $$S^Y_{Xi} = \frac{\partial Y/Y}{\partial X_i/X_i}$$

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).

4. Modification and parametrization of model to simulate short-term and long-term effects of an experimental whole-tree harvest on forest biomass, stream discharge, soil and stream water chemistry.

4.1. Results and discussion

4.1.1. 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 (Figure 4.1a, b). Simulations of annual volume-weighted concentrations of stream NO_3^- were compared with observations (Figure 4.1a) and watershed NO_3^- flux simulations were compared before (Figure 4.1b) and after model modifications (Figure 4.1c). 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 (Figure 4.1c). To avoid repetitive plots, I show modeled stream NO_3^- concentrations after model modifications in Figure 4.5. Similar problems with short-term simulations were evident for other elements, without model modification. Model simulations were compared before and after model modifications based on the values of normalized root mean square error (NRMSE). 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 (Figure A1 in Appendix).

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, the 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., 1995, 1991; 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.

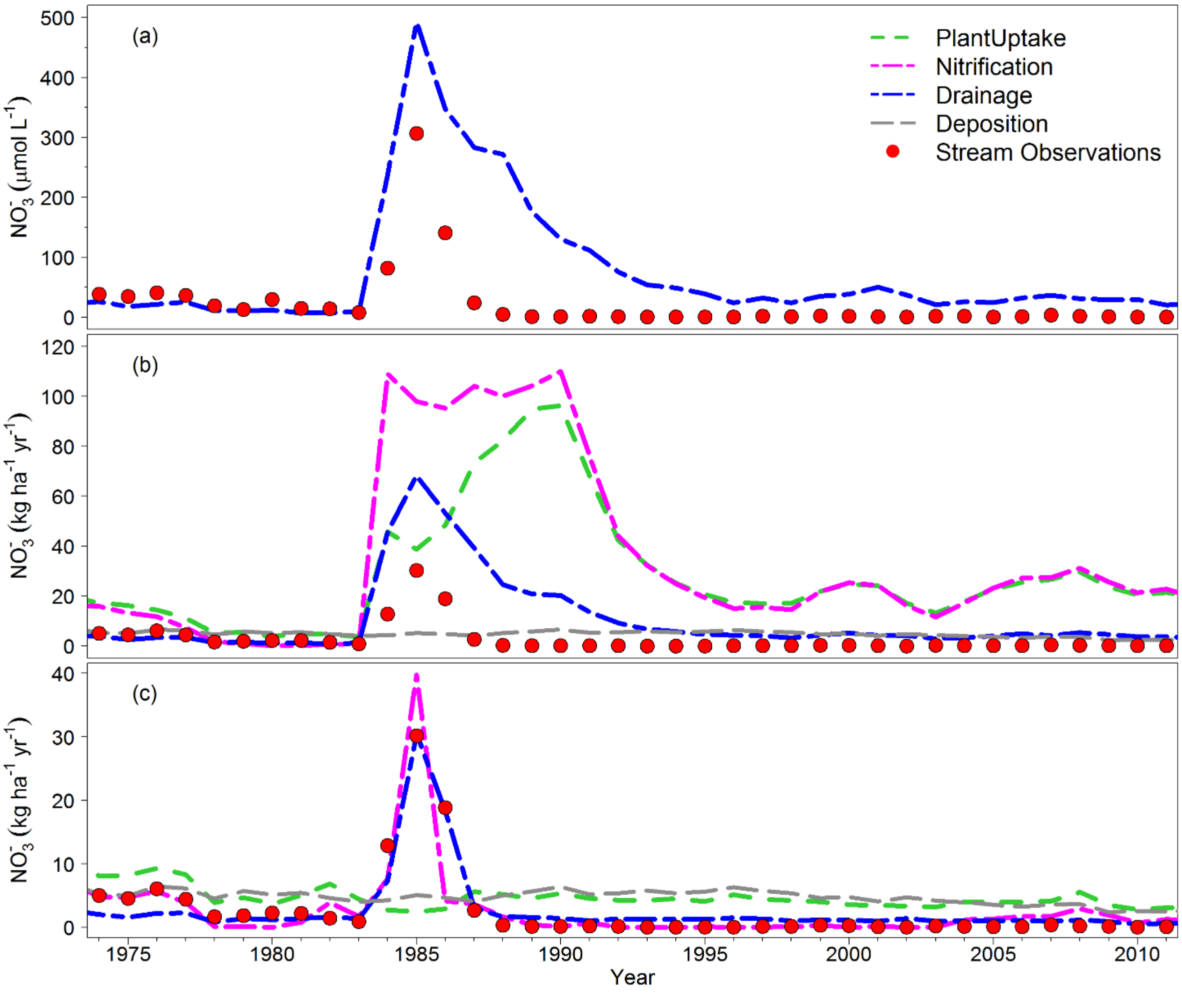


Figure 4.1. Comparison of model simulations with observations for annual volume-weighted NO_3^- concentrations in stream water (a) and modeled NO_3^- budget flux for before (b) and after (c) model algorithm modification and parameterization.

4.1.2. Vegetation simulations

The model generally performed well in the simulation of aboveground biomass for both pre-harvest and post-harvest conditions (Figure 4.2). 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. I project that W5 would reach the aboveground biomass value prior to the cut in about 70 years (190 t ha^{-1}) (Figure 4.2). 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 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.

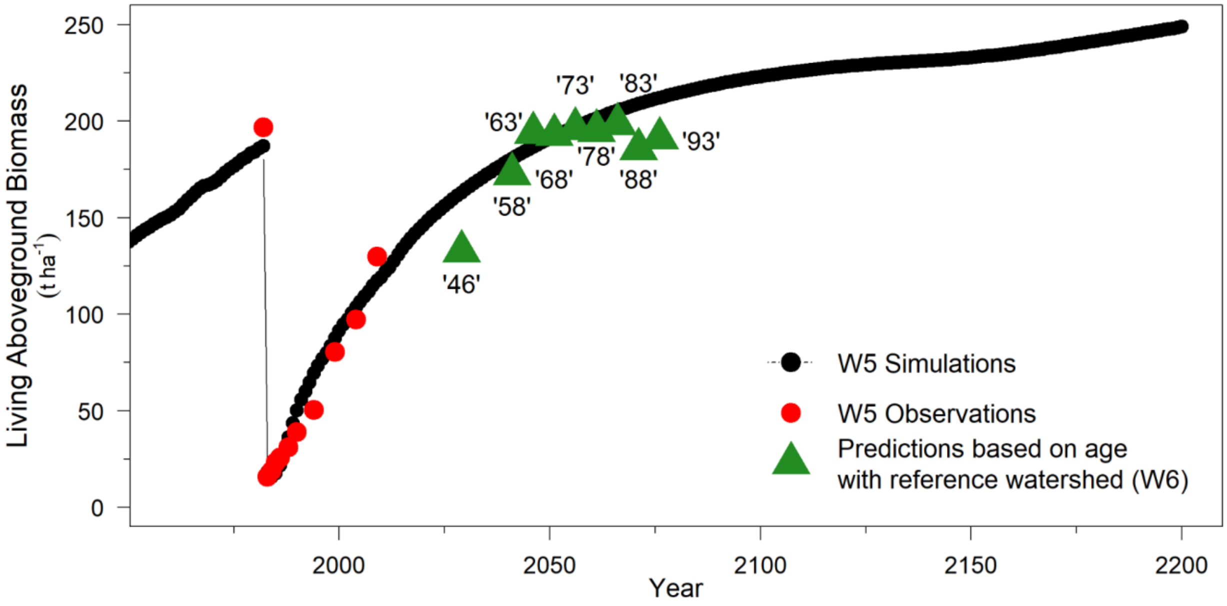


Figure 4.2. Simulations of living aboveground biomass accumulation, Watershed 5, HBEF, for the period before and after the whole-tree harvest. Model simulations are compared with measured values for Watershed 5 and Watershed 6 adjusted to years after cutting.

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 (Figure 4.3), 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 (Figure 4.3). 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.

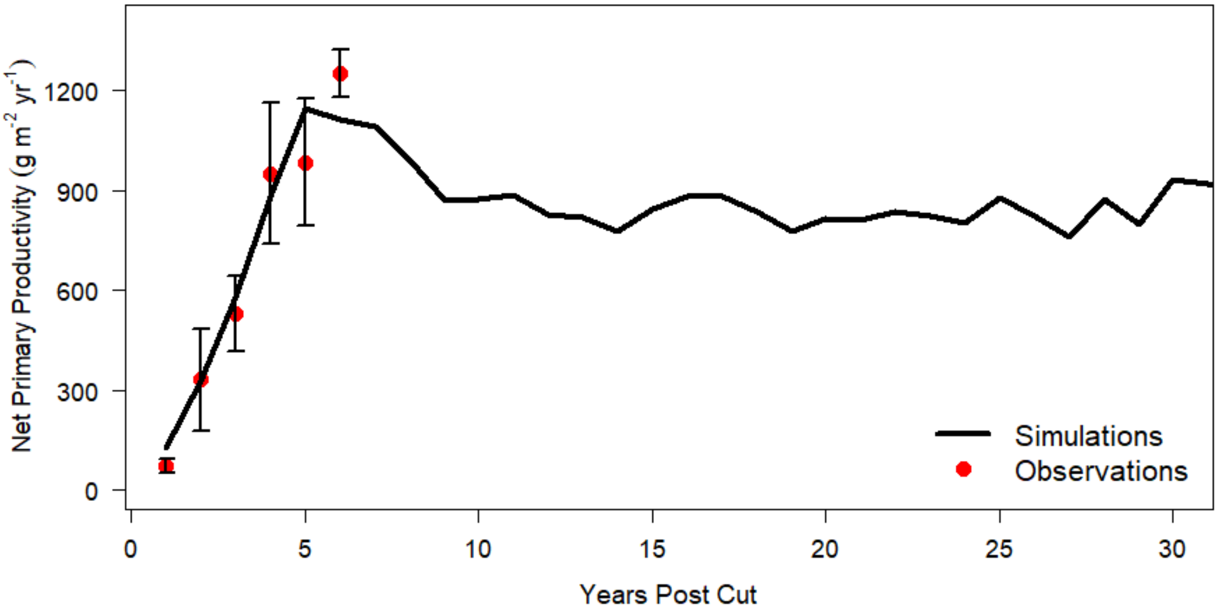


Figure 4.3. Comparison between simulated aboveground net primary productivity of WS5 and measurements from different clear-cut stands in the vicinity of the HBEF. Data shown from (Bormann and Likens, 1979).

4.1.3. Soil solution chemistry simulations.

Simulated solution of the upper soil layer was compared with areal annual average concentrations of Bs horizon soil solution (1984-1997 years) of the three zones in the watershed including SFB (20%, spruce-fir-birch), HH (30%, high-elevation hardwood) and LH (50%, low-elevation hardwood) (Dahlgren and Driscoll, 1994; Johnson et al., 2000). The model effectively captured observed values of major elements in soil solutes (Table 4.1). PnET-BGC simulated the peak in NO_3^- concentration into soil solution in the second year after the clear-cut and reproduced the rapid decrease in NO_3^- with the regrowth of vegetation (NME_a= 0.5, $NMAE_a$= 0.7 and $NRMSE_a$= 0.9, Table 4.1, Figure 4.4). Note that the model performed better in simulating soil solution NO_3^- than streamwater values. Mean annual simulated soil solution Ca^{2+} increased to 50 µmol L^{-1} with biomass removal, closely agreeing with the areal average observation of 40 µmol L^{-1} for 1984-86. The model effectively projected the decreasing trend of Ca^{2+} leaching into soil solution after the treatment (NME_a= 0.06, $NMAE_a$=0.09 and $NRMSE_a$=0.1, Table 4.1, Figure 4.4). The model underestimated the overall increase in average concentration of Al_m and Na^+ in soil solutions at the time of logging, 27 and 36 µmol L^{-1} in comparison with measurements of 53 and 30 µmol L^{-1}, respectively, for 1984-86. PnET-BGC satisfactorily captured the decrease in concentrations of Al_m and Na^+ in soil solutions with the regrowth of trees (NMEa= -0.56, $NMAE_a$= 0.56 and $NRMSE_a$= 0.8 for Al_m, Table 4.1, Figure 4.4) and NME_a= -0.33, $NMAE_a$= 0.33 and $NRMSE_a$= 0.34, for Na^+, Table 4.1, Figure 4.4). The model performed reasonably well in depicting the rapid increase in soil solution concentrations of Mg^{2+} following cutting, with a simulated mean annual value of 21 µmol L^{-1} versus an observed value of 18.6 µmol L^{-1} for 1984-86 and the decrease in concentrations with the regrowth of trees, simulated mean value of 7.4 µmol L^{-1} compared with observed mean value of 7.1 µmol L^{-1} for 1988-1997 (NME_a= 0.04, $NMAE_a$= 0.08 and $NRMSE_a$= 0.09, Table 4.1, Figure 4.4) periods. Generally, the model underestimated soil solution SO_4^{2-} with a mean concentration of 41.1 µmol L^{-1}, compared with measured mean concentration of 51.8 µmol L^{-1} for 1984-86. The model underpredicted soil solution SO_4^{2-} concentrations but captured the overall trend of decreasing concentrations with decreases in atmospheric SO_4^{2-} deposition (NME_a= -0.12, $NMAE_a$= 0.12 and $NRMSE_a$= 0.14, Table 4.1, Figure 4.4). Modeled soil solution DOC was overpredicted compared with measured values (NME_a= 0.2, $NMAE_a$= 0.2 and $NRMSE_a$= 0.25, Table 4.1, Figure 4.4). A small discrepancy between mean annual simulated and observed pH indicated good performance of the model in depiction of this parameter (NME_a= 0.01, $NMAE_a$= 0.01 and $NRMSE_a$= 0.02, Table 4.1, Figure 4.4).

Table 4.1. Summary of simulated and observed soil solution of chemical constituents and model performance results.

Soil Solution Constituents	Post-harvest (1987-1997)						
	Mean		STD		NME	NMAE	NRMSE
	Observed	Simulated	Observed	Simulated			
pH	4.71	4.8	0.09	0.02	0.02	0.02	0.03
[a]DOC	645.62	817.62	101.91	81.39	0.27	0.27	0.30
Na^+	26.87	17.91	2.18	1.08	-0.33	0.33	0.34
Mg^{2+}	7.08	7.68	1.41	0.86	0.04	0.08	0.09
[b]Al_m	21.64	9.42	3.59	0.56	-0.56	0.56	0.58
Ca^{2+}	20.42	21.64	2.91	1.95	0.06	0.09	0.10
NO_3^-	7.31	9.26	2.90	3.04	0.27	0.48	0.57
SO_4^{2-}	47.12	41.32	5.86	2.7	-0.12	0.12	0.14

Units for soil solutions constituents are µmol L^{-1}.

NEM: normalized mean error. NMAE: normalized mean absolute error. NRMSE: normalized root mean squared error. STD: Standard Deviation.

[a] DOC: Dissolved organic carbon.

[b] Al_m: Total monomeric Al.

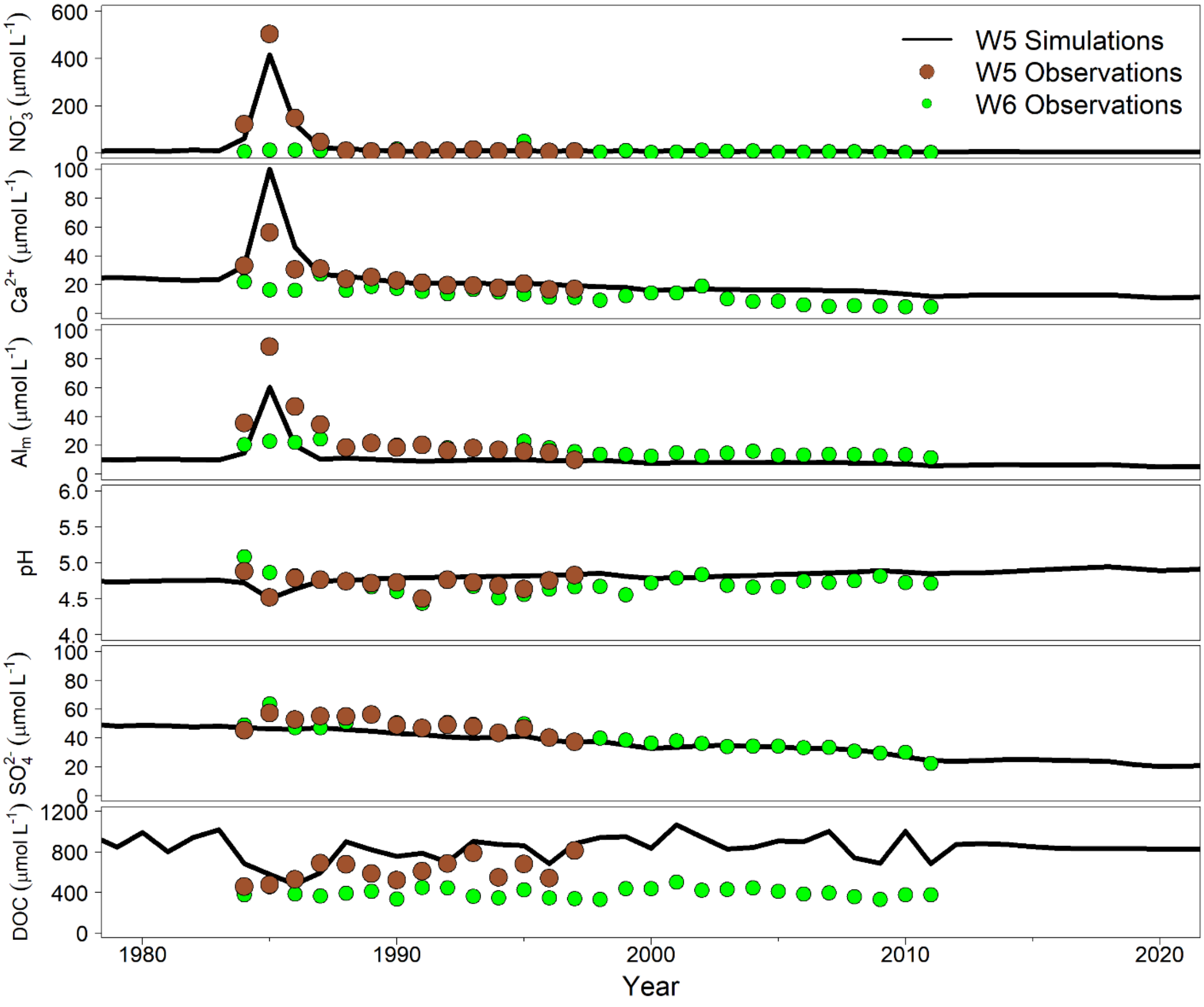


Figure 4.4. Comparison of simulations of soil solution chemistry (Bs soil horizon) by PnET-BGC with measured values in response to the clear-cut of W5, HBEF. Simulation results are compared with measured soil solution chemistry (Bs soil horizon) for reference watershed (W6).

4.1.4. 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 4.2, Figure 4.5) and an underprediction for the post-harvest period

(NME_a= -0.01, $NMAE_a$= 0.06 and $NRMSE_a$= 0.07, Table 4.2, Figure 4.5). 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 (Figure 4.5).

4.1.5. 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 (Figure 4.5). 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 4.2).

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 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 4.2). 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 4.2) 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 Driscoll, 2015). Forest cutting 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 4.2, Figure 4.5). Predicted pH after the treatment closely matched measured values (1988-2012, NME$_a$= 0, NMAE$_a$= 0.01 and NRMSE$_a$= 0.02, Table 4.2, Figure 4.5). 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 4.2).

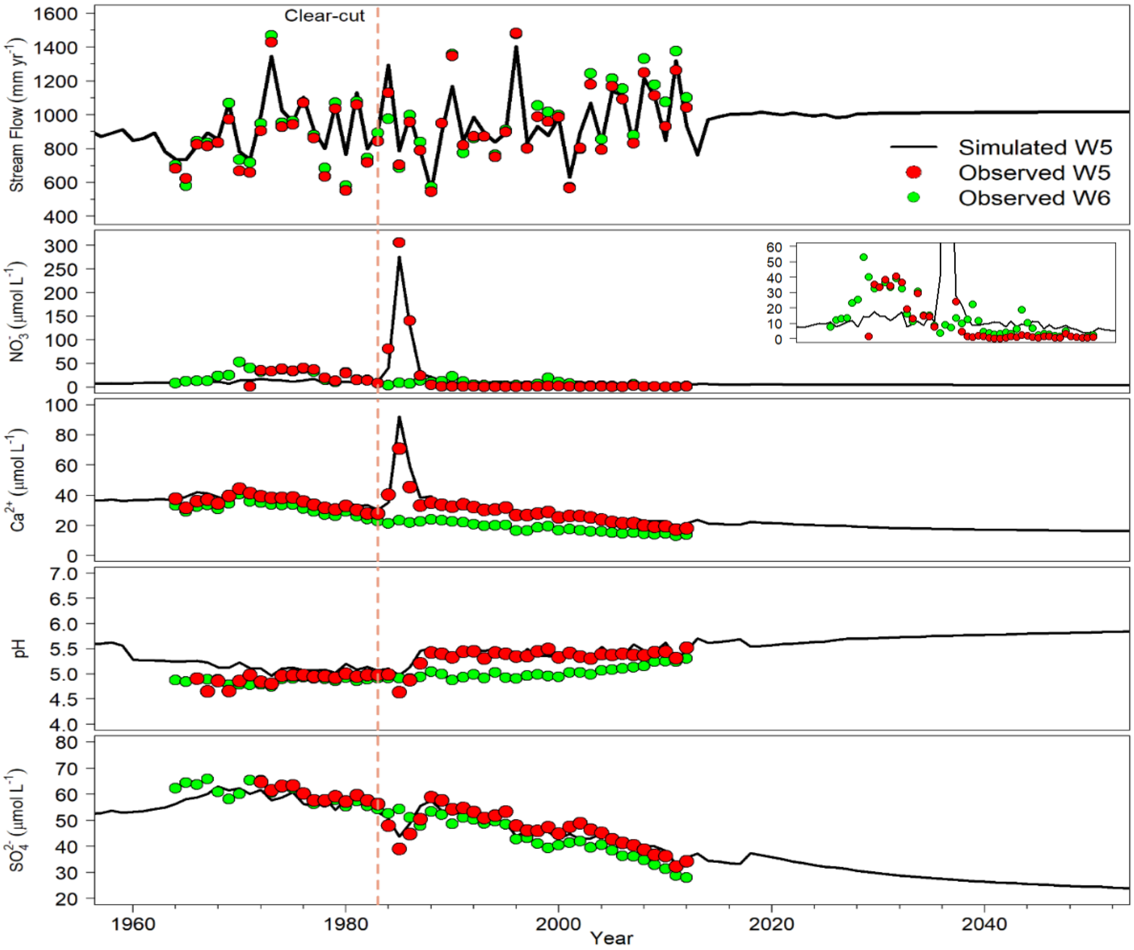


Figure 4.5. Comparison between streamflow and annual volume-weighted stream water chemistry from PnET-BGC simulations and observations for W5, HBEF. The results are also shown with stream water chemistry for the reference watershed (W6).

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 4.2, Figure 4.5) and post-treatment periods (NME_a= - 0.02, $NMAE_a$= 0.04 and $NRMSE_a$= 0.05, Table 4.2, Figure 4.5) 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 (Figure A2).

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 4.2, Figure A2). 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, 2.35, 2.2, 1.3, 2.5 and 3, respectively over the 1984-86 period. I 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.

Table 4.2. 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

4.1.6. 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, I did not consider these disturbances in the 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 (Figure 4.6). 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 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 of snowpack development and snowmelt and canopy development and senescence, can have a considerable effect on simulations of hydrology and nutrients.

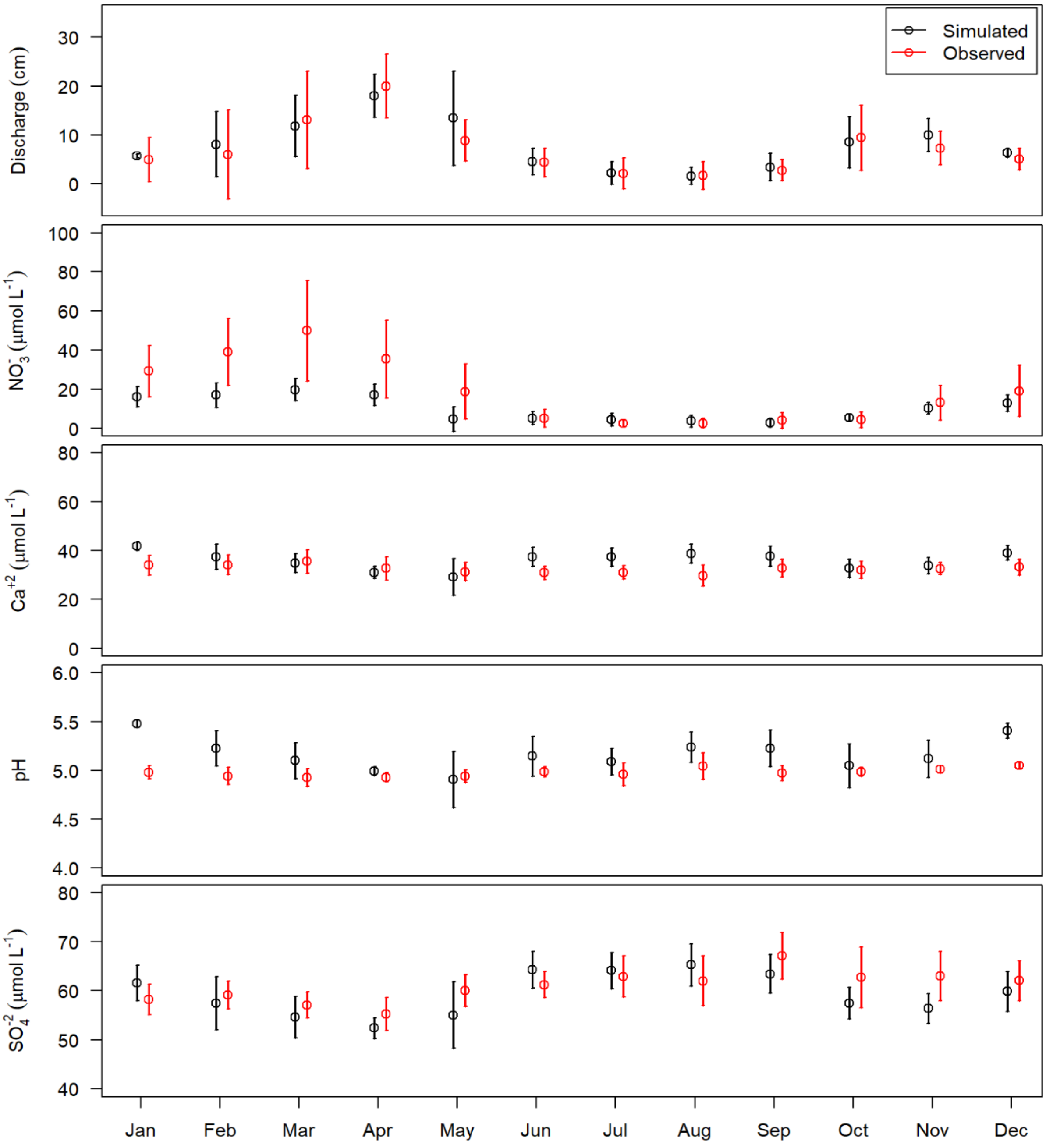


Figure 4.6. Comparison between monthly patterns of stream water chemistry PnET-BGC simulations with observations for the pre-cut period (1974-1982).

4.1.7. 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 (Figure 4.7). 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) (Figure 4.7a). 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) (Figure 4.7a). 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} (Figure 4.7b). 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 the soil exchangeable pool of Ca^{2+} was estimated to be 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 (Figure 4.7b). 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 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 (Figure 4.7c). 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.

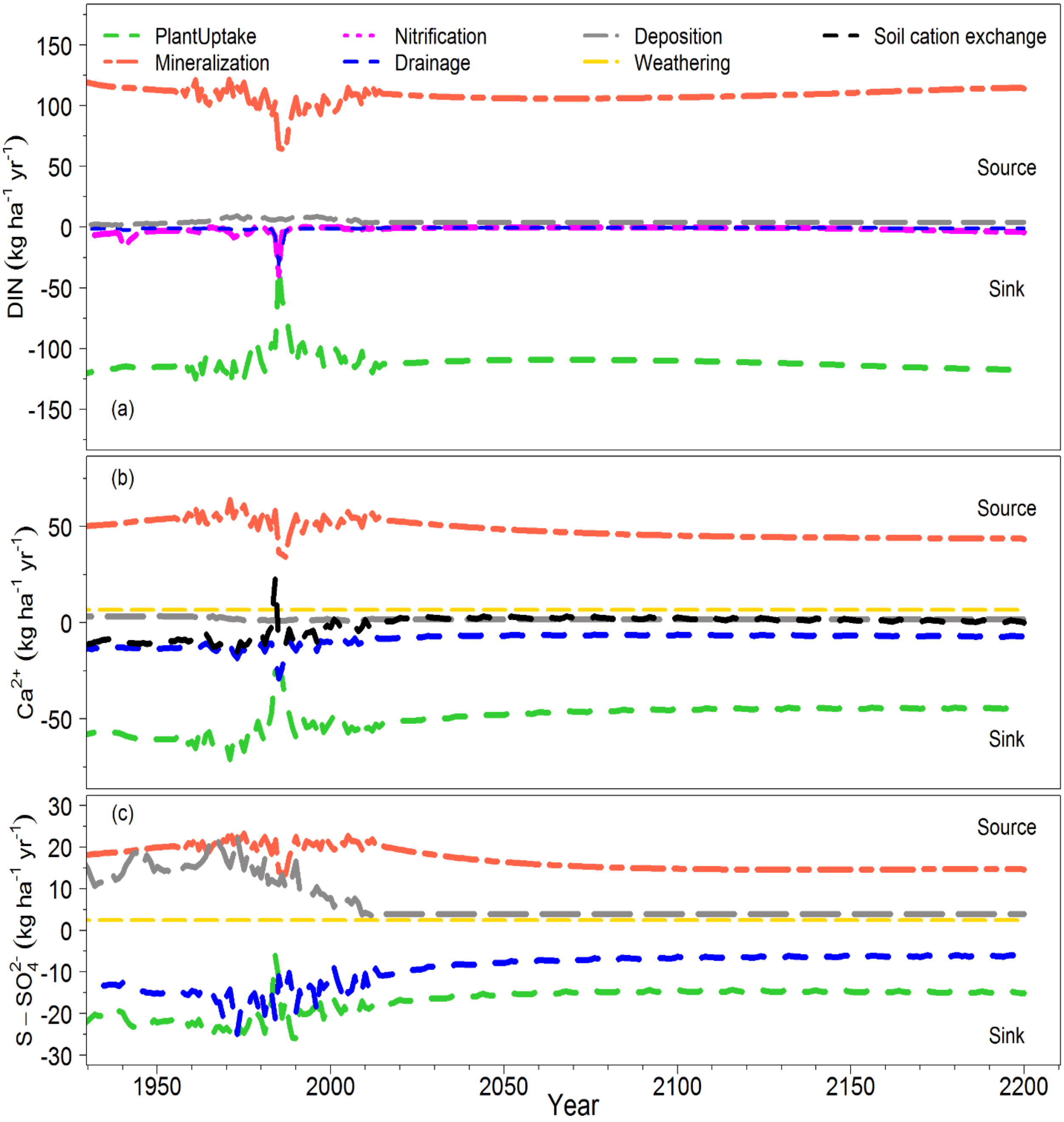


Figure 4.7. Simulation of nutrient budgets for W5, HBEF, including total dissolved inorganic nitrogen (DIN) (a), Ca^{2+} (b) and $S\text{-}SO_4^{2-}$ (c).

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 (Figure 4.8), 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 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.

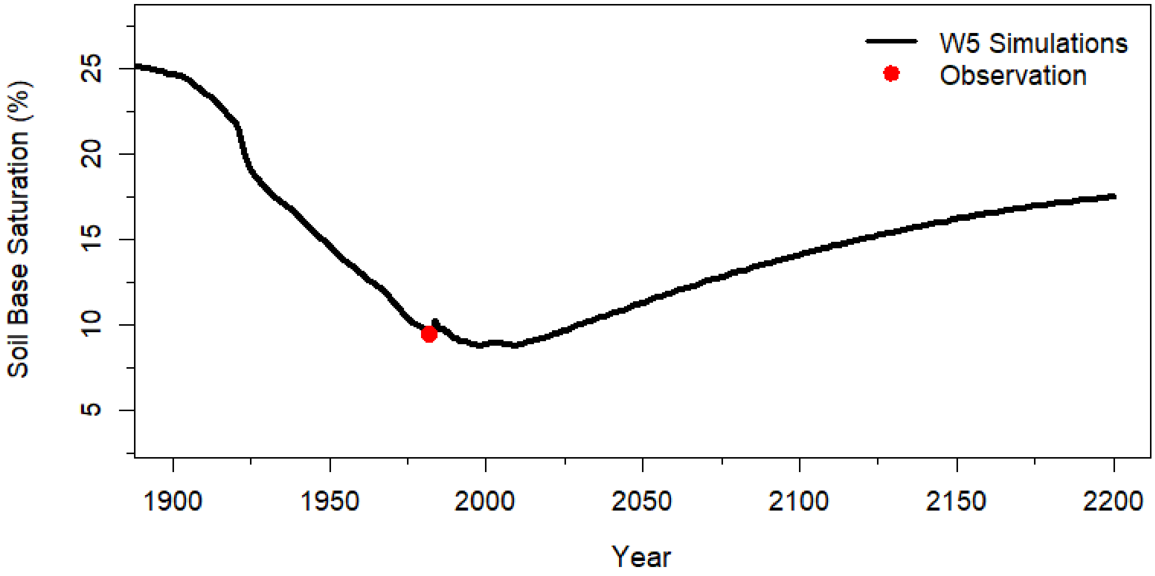


Figure 4.8. Simulation of long-term changes of soil base saturation for W5, HBEF, compared with measured soil base saturation in 1982.

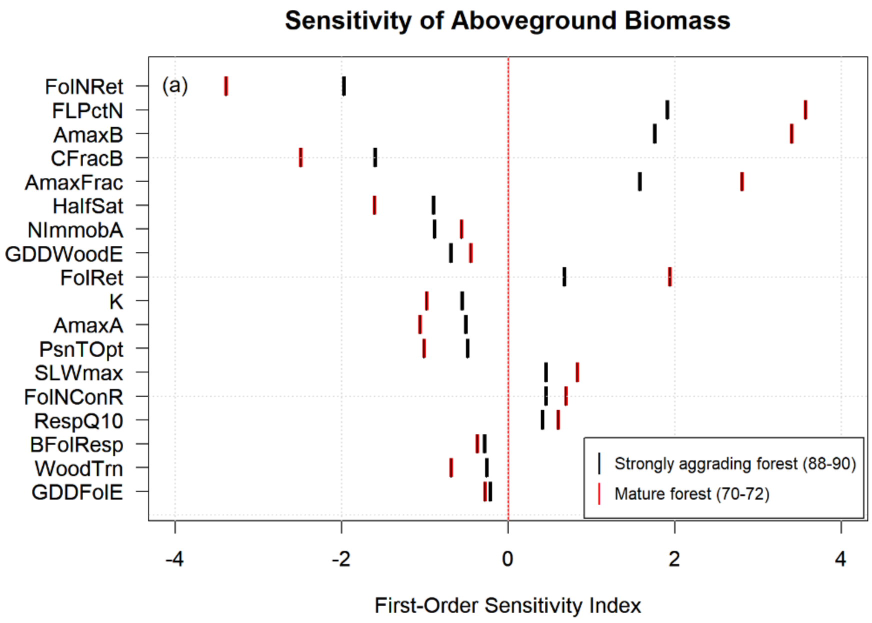
4.1.8. Sensitivity Analysis

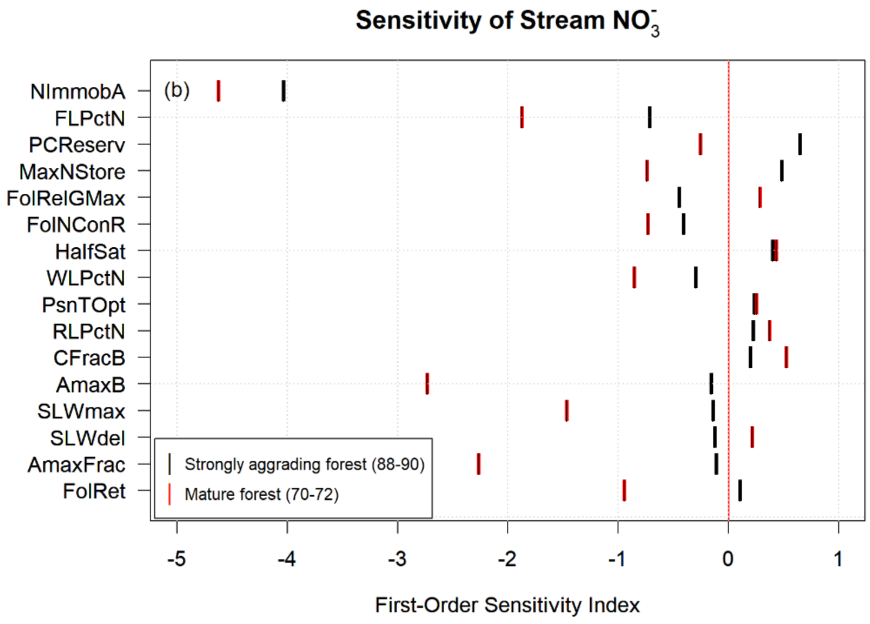
I selected 71 model parameters in PnET-CN such as input variables related to photosynthesis, foliar growth, wood and root turn-over rates and water use efficiency constant, as well as hydrological and chemical process in soil including soil hydrologic conductivities, saturation capacities, weathering rates, cation exchange capicity (CEC), soil mass and SO_4^{2-} adsorption capacity (in BGC) to evaluate their sensitivity in controlling the response of three state variables: discharge, stream NO_3^- concentrations and aboveground biomass, which were selected to represent both biotic and abiotic ecosystem processes. Previous analyses have not evaluated model sensitivity under different conditions of forest growth, but have considered model sensitivity in simulations of mature forest conditions (Aber et al., 1996; Fakhraei et al., 2017; Gbondo-Tugbawa et al., 2001). I did not include climate drivers because the results of this sensitivity were reported in previous studies (Aber et al., 1996; Pourmokhtarian et al., 2012). I conducted the sensitivity analysis of the model parameters 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).

The 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) (Figure 4.9). Biomass simulations are highly sensitive to parameters related to nitrogen retranslocation following senesce (FolNRet), minimum N concentration in foliar litter (flpctn) and the slope of maximum gross photosynthesis rate (AmaxB). With a 10% increase of both minimum N concentration in foliar litter (flpctn) and the slope of maximum gross photosynthesis rate (AmaxB), biomass projections increase by around +35%, while a 10% increase in the parameter associated with N retranslocation following senesce (FolNRet) decreases biomass predictions by 35% over pre-cut conditions (Figure 4.9a). These parameters are also the most sensitive for biomass simulations during the post-treatment conditions but with smaller impact (around 20%) (Figure 4.9a). Parameters related to coefficients AmaxFrac and CFracB, which determine the fraction of maximum photosynthesis and the fraction of carbon allocated to wood, are also relatively sensitive parameters. Wood turnover rate (WoodTrn) and the respiration factor as a fraction of maximum photosynthesis (BFolResp) are among the least sensitive parameters for biomass predictions (Figure 4.9a).

Stream NO_3^- predictions are most sensitive to parameters related to the fraction of mineralized N that is immobilized by microbes (NImmobA) and the slope of the maximum gross photosynthesis rate (AmaxB) with respect of foliar nitrogen concentration variable, particularly for the mature forest. A 10% increase in these parameters resulted in 46% and 27% decreases, respectively in stream NO_3^- concentrations in model simulations (Figure 4.9b). In contrast for an aggrading forest, model sensitivity for stream NO_3^- predictions were much lower (below 10% for all parameters except NImmobA (40%)) (Figure 4.9b). Discharge generally demonstrates limited response to a 10% increment of selected parameters, typically showing less than 5% change, except for the fraction of mineralized N that is immobilized (NImmobA) which increases discharge by 10% and 15% during the pre-cut and post cut conditions, respectively (Figure 4.9c). A reduction in net N mineralization and N assimilation can influence vegetation growth and demand for water plant uptake, eventually increasing stream discharge.

The model uses constant inputs of minimum N concentration in foliar litter (flpctn) and nitrogen retranslocation following senesce (FolNRet) parameters to calculate foliar N concentration that along with the slope coefficient of the photosynthesis equation (AmaxB) affect the rate of 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. The N immobilization factor (NImmobA) has a greater effect on stream NO_3^- than biomass processes. 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 dominate model simulations for 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 regulate ecosystem biotic and abiotic processes, including nutrient uptake, mineralization, nitrification, and transpiration. Results from the sensitivity analysis will be helpful for future model applications at other sites and demonstrate the necessity of field measurements to constrain important parameter values.





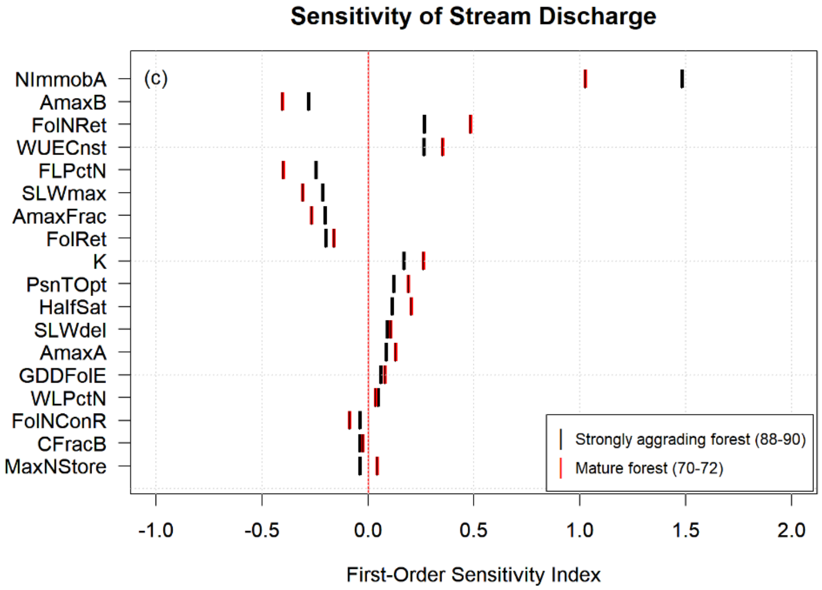


Figure 4.9. First order sensitivity results of PnET-BGC inputs to predictions of aboveground biomass (a), stream NO_3^- concentrations (b) and stream annual discharge (c) for strongly aggrading and mature forests.

4.2. 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.

5. Model comparison of experimental approaches of forest cutting on biomass accumulation, hydrology and biogeochemistry in a northern forest

5.1. Results

5.1.1. Vegetation simulations

In this phase of study, I applied a version of PnET-BGC that was modified and parametrized to different harvesting experiments at the HBEF (W2, W4). Model performance was tested and used to project ecosystem pools and fluxes including aboveground biomass, soil base saturation, stream water chemistry and element budgets to evaluate both short- and long-term forest ecosystem response to different harvesting techniques. The model generally performed well in the simulation of aboveground biomass from clear-cut watersheds (W4, W2) for both pre-harvest and post-harvest conditions (Figure 5.1). 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} six 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 and t ha^{-1} respectively, approximately 35%, 58% and 81% of aboveground biomass prior to the cut. Model simulations project that W4 would reach the aboveground biomass occurring prior to the cut after about 48 years of regrowth (153 t ha^{-1}) (Figure 5.1).

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} following the five 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 and t ha^{-1}, respectively, which are approximately 43% and 65% of aboveground biomass prior to the cut. I anticipate that W2 would approach the value of aboveground biomass that was on the watershed prior to the cut after about 46 years of regrowth (138 t ha-1) (Figure 5.1). Comparison with the reference watershed (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 devegetated/herbicide treated W2, compared with the whole tree-harvested W5 (223 t ha^{-1}) and reference watershed W6 (224 t ha^{-1}).

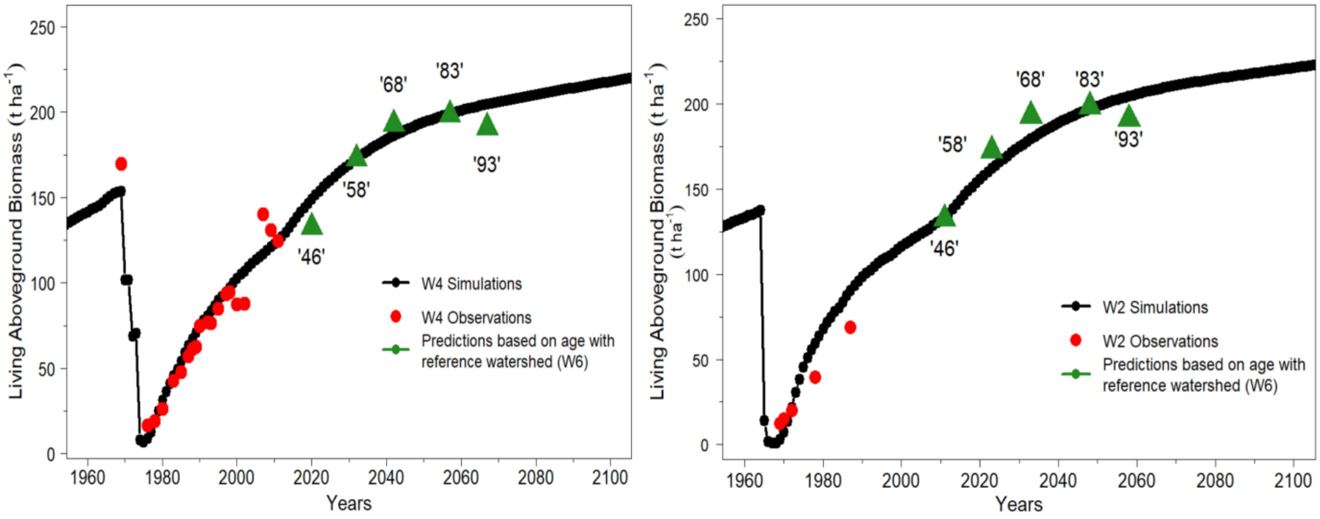


Figure 5.1. Simulations of living aboveground biomass accumulation for watershed 4 (W4) and watershed 2 (W2), HBEF, for the period before and after forest cuts. Model simulations are compared with measured values for W4 and W2 and watershed 6 (W6) after adjustment for years after cutting.

5.1.2. Stream hydrology

The reduction of transpiration associated with forest harvesting coupled with variation in precipitation quantity have a profound effect on ecosystem hydrology. Hydrologic factors also have a profound role in export of nutrients via stream water. 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 5.1, Figure 5.2) and for the post-harvest period ($Mean_{obs}$=93.87 cm, $Mean_{pred}$= 91.46 cm, NME_a=-0.03, Table 5.1, Figure 5.2). 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 stream flow for W4 during the multiple years of the strip-cuts until the first year of regrowth (1971-75) indicated a 49% increase from the pre-cut year (1969) compared with a 40% increase for the measured values. The model also efficiently 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 5.1, Figure 5.2). Modeled average annual stream flow for W2 during the devegetation/herbicide treatment years until the first of regrowth (1966-69) underestimated percentage of increase in stream discharge (27%) from the pre-cut year (1964), compared to the observed value (65%).

5.1.3. 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^- roughly matched measured values for pre-cut and post-cut periods, with some 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 5.1, Figure 5.2). 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 5.1, Figure 5.2). 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 5.1, Figure 5.2), but closely captured 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 5.1, Figure 5.2). The modified two-layer version of PnET-BGC was capable of capturing 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 46.58 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, 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 the factors of 2, 1.6, 1.3 and 1.5, respectively over the 1971-75 period.

Simulated 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 process, with the greatest decline occurring during the second strip-cut (1973), and values increased with the regrowth of vegetation during the third strip cut. The model also performed well showing the long-term decreases in stream SO_4^{2-} concentrations during pre-cut treatment, with slight overprediction (Mean$_{obs\_b}$=64.3, NME$_b$=0.01) and post-cut, with slight underprediction (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 SO_4^{2-}in 1973. Stream SO_4^{2-} then began to increase as soil pH increased during the biomass regrowth, and increased NO_3^- retention coinciding with desorption of the SO_4^{2-} previously retained by soil.

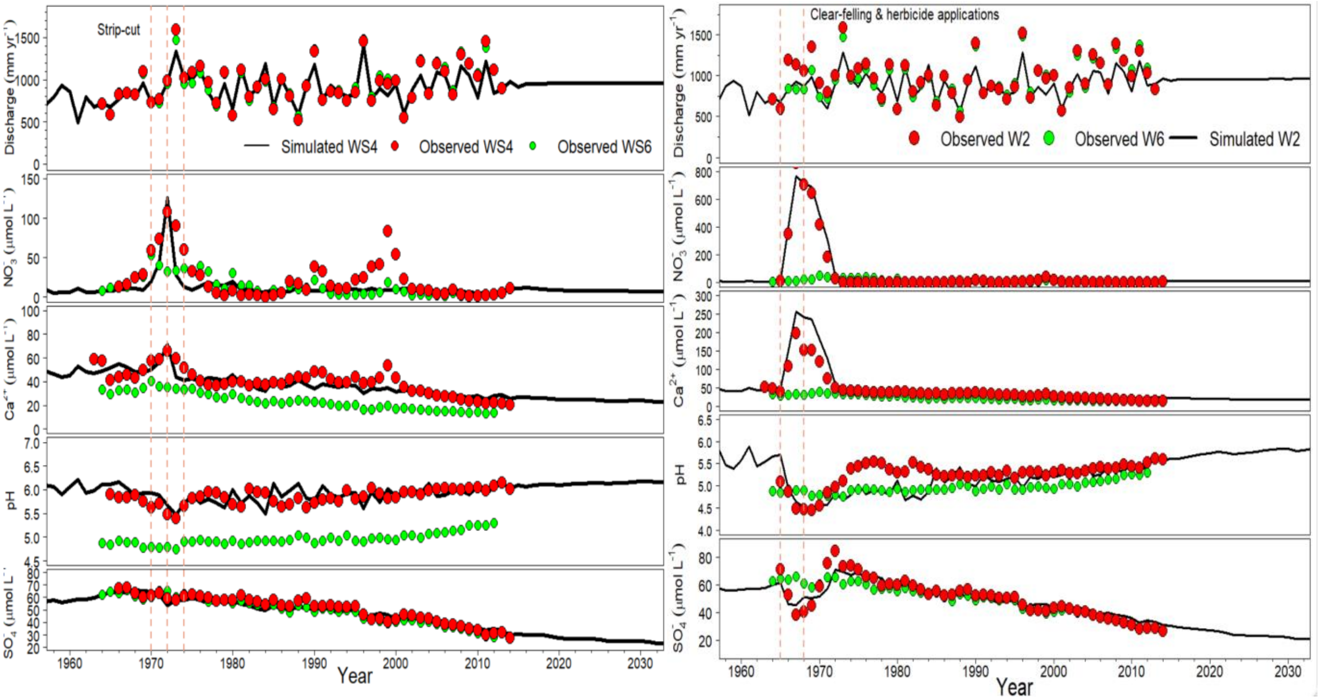


Figure 5.2. Comparison between annual streamflow and volume-weighted stream water chemistry from PnET-BGC simulations and observations for W4 and W2, HBEF. Measured values are also shown for the reference watershed (W6). The timing of the cut is shown by the vertical line.

Watershed 2 (W2) showed a similar pattern of nutrient losses following the 3-year period of devegetation/herbicide treatment, but with greater degree of response. The modified model could nearly reproduce peak values of the concentration and loss of elements in the stream water following the intensive harvest of W2, and the rapid recovery of nutrient leaching to pre-cut values with new vegetation growth. 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 compared with the observed values of 529, 137, 44 and 51 µmol L^{-1}, respectively, during the treatment effect period (1966-71). For most major elements, annual volume-weighted stream concentrations peaked in the second year after the cutting (1967) and declined during the third year. Modeled 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 (1966-71), average annual stream water concentrations exceeded those of 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). Values showed a decline following the first year after the harvest (1966) and remained low during the devegetation/herbicide treatment until the first year after the treatment (1969) and then begin 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 second year after the clear-cut (1967), followed by subsequent desorption of SO_4^{2-} from soil and increases in stream concentrations.

Table 5.1. Comparison of modeled and observed values of stream constituents and model performance for the periods prior and after strip-cut W4 [a].

Stream Constituents	Prior Pre-harvest (1966-69) _W4						
	Mean		STD		NME	NMAE	NRMSE
	Observed	Simulated	Observed	Simulated			
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

Stream Constituents	Post-harvest (1980-2014) _W4						
	Mean		STD		NME	NMAE	NRMSE
	Observed	Simulated	Observed	Simulated			
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

[a] Values represent mean and standard deviation of annual volume-weighted concentrations for the pre-harvest (1966-69) 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 5.2. Comparison of modeled and observed values of stream constituents and model performance for the period after clear-felling W2 with follow-up herbicide application [a].

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

[a]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.

5.1.4. 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 (Figure 5.3). Lower monthly concentrations of stream NO_3^- were projected during the winter dormant season and fall, compared with monthly observations. The model slightly overestimated monthly concentrations of stream SO_4^{2-} during the growing season for W2 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.

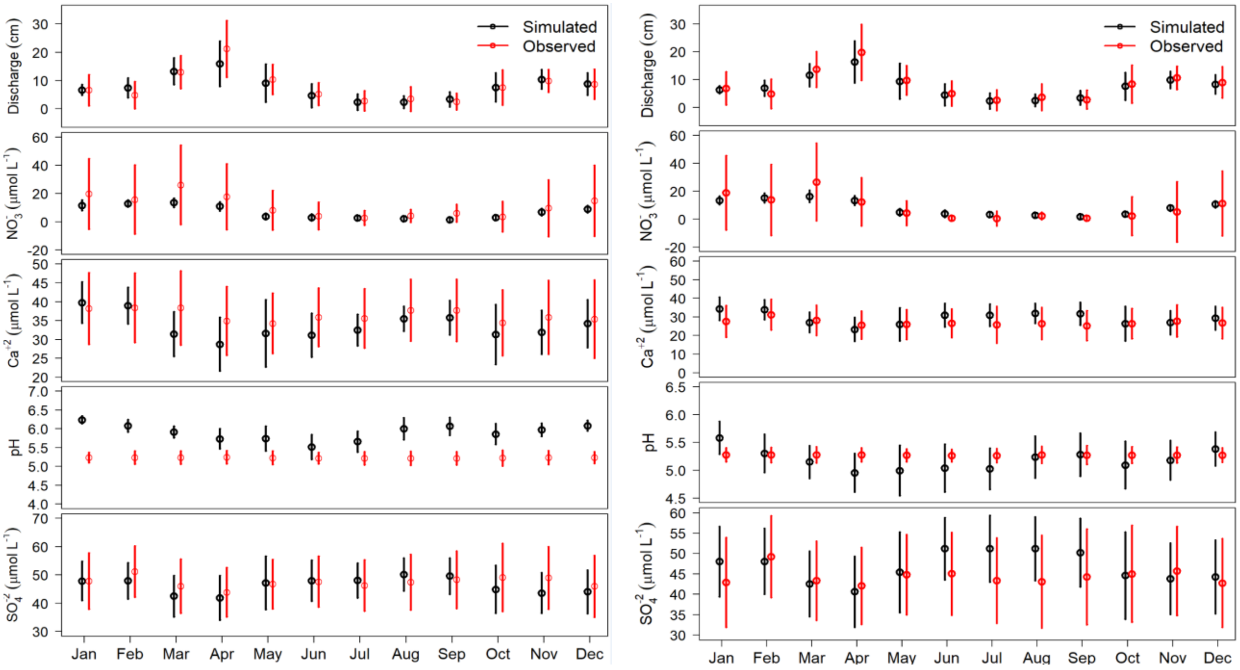


Figure 5.3. Comparison between monthly patterns of stream water chemistry PnET-BGC simulations with observations for W4 (left) and W2 (right), HBEF for the post-cut period (1980-2014). Error bars indicate standard deviation and monthly average values.

5.1.5. Nutrient budget simulations

The modified PnET-BGC model was applied to evaluate changes in the source/sink behavior of the northern hardwood forest for major elements in response to cutting disturbances. Also, element budgets were compared for different watershed cuts over both short- and long-term periods. For all three experimentally treated watersheds at the HBEF, I summarized patterns of nutrient budgets for three different periods: the pre-treatment (1960-64); the treatment effect period which is characterized by a marked response in stream water NO_3^- to the cutting (for W4:1971-75; for W2:1966-71; for W5:1984-87; Fakhraei et al., 2019); and post-treatment (2046-50) and compared values with the reference watershed (W6);

For all watersheds, soil N mineralization and plant uptake were closely coupled, with average annual rates of 100-111 kg N ha^{-1} yr^{-1} and 104-114 kg N ha^{-1} yr^{-1}, respectively during 1960-64 pre-cut period, with slightly higher rates for W5 and W6 followed by W4 and W2 (Figure 5.4). Soil N mineralization and plant uptake were greatly influenced by cutting disturbances of the watersheds and rates were reduced to average values of 77 and 80 kg N ha^{-1} yr^{-1} for W4 (vs 121 and 116 kg N ha^{-1} yr^{-1} for W6 during 1971-75); 81 and 20 kg N ha^{-1} yr^{-1} for W2 (vs 113 and 119 ha^{-1} yr^{-1} for W6 during 1966-71) and; 74 and 65 kg N ha^{-1} yr^{-1} for W5 (vs 111 and 107 ha^{-1} yr^{-1} for W6 during 1984-87), respectively for the treatment effect periods. Following the clear-cuts, simulated soil N mineralization and plant uptake increased in all cut watersheds eventually reaching average annual rates of 106-112 kg N ha^{-1} yr^{-1} and 109-114 kg N ha^{-1} yr$^{-1,}$ respectively, during 2046-50, with the greatest rates for W2 followed by W4, W6 and W5. 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} yr^{-1}, respectively, and exceeded values of W4 and W2 (0.6 and 0.7 kg N ha^{-1} yr^{-1} for nitrification rates and N stream leaching, respectively) during the 1960-64 period. During the treatment periods, the greatest nitrification rates and stream N leaching were simulated for the devegetation/herbicide treatment of W2, with the values of 64 and 68 kg N ha^{-1} yr^{-1}(vs the corresponding values of 2 and 1.3 kg N ha^{-1} yr^{-1} for W6 during 1966-71), followed by W5 (14 and 15 kg N ha^{-1} yr^{-1} vs 1.8 and 1.2 kg N ha^{-1} yr^{-1} for W6 during 1984-87) and W4 (6 and 6 kg N ha^{-1} yr^{-1} vs 6.8 and 2.2 kg N ha^{-1} yr^{-1} for W6 during 1971-75). During 2046-50, the highest rates of nitrification and stream N leaching occurred for W2 (nitrification rate: 5.1 kg N ha^{-1} yr^{-1}, N stream leaching:1.3 kg N ha^{-1} yr^{-1}) and followed by W4), W6 (0.5 and 0.62 kg N ha^{-1} yr^{-1}) and W5 (0.3 and 0.58 kg N ha^{-1} yr^{-1}).

Net N release was calculated to estimate discrepancy between major sources and sinks of N in the watersheds for the pre-cut, treatment and post-cut simulation periods. Results showed that N net release remained nearly constant for all watersheds during both pre- and post-cut periods at the rate of 0.2 kg ha^{-1}. During the treatment interval net N release of W2 of 1.3 kg N ha^{-1} yr^{-1} during the treatment effect period, followed by the whole tree harvest of W5 at the rate of 0.6 kg N ha^{-1} yr-1, and the strip cut of W4 with the value of 0.3 kg N ha^{-1} yr^{-1} (Figure 5.4).

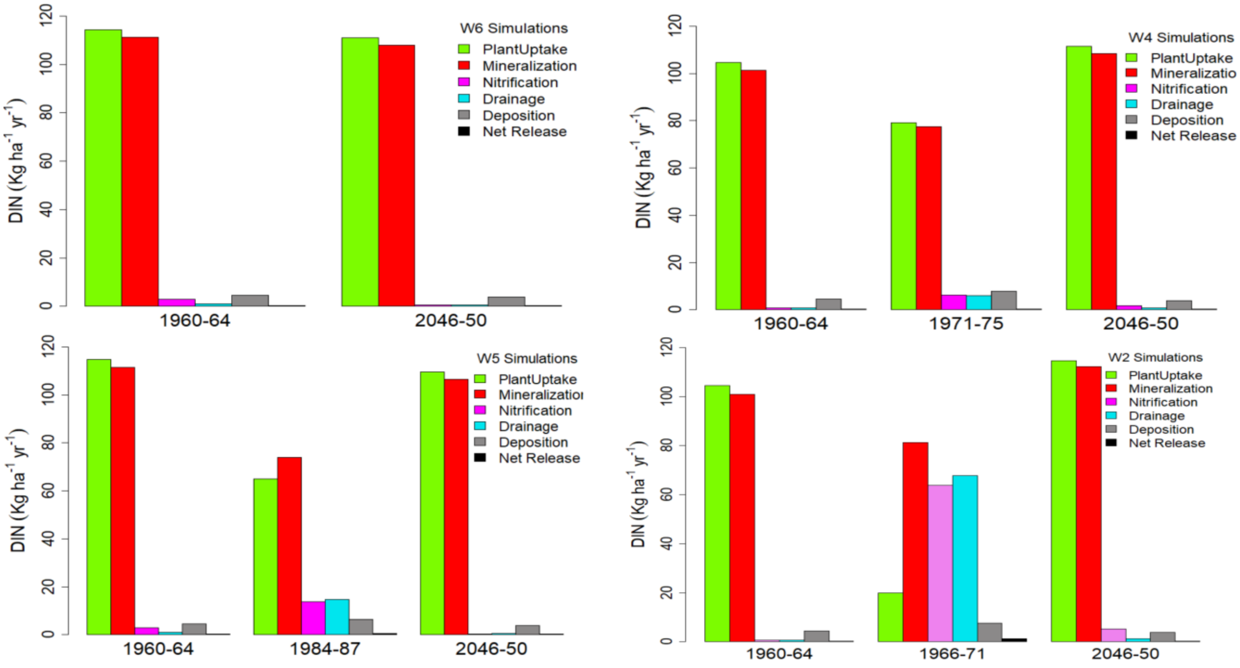


Figure 5.4. Comparison of simulated annual average nitrogen budgets for the cut watersheds at the HBEF including W5, W4 and W2 during the pre-cut period (1960-64), the treatment effect period (W5:1984-87; W2:1966-71; W4:1971-75) and post-cut period (2046-50). Simulations are compared with simulated nitrogen for the reference watershed (W6) during 1960-64 and 2046-50 periods.

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} yr^{-1} and 37-59 kg N ha^{-1} yr^{-1}, respectively, during 1960-64 pre-cut period, with the greatest rate for W4 and followed by W5, W2 and W6 (Figure 5.5). Soil Ca^{2+} mineralization and plant uptake were reduced during the clear-cut experiments, with the greatest decline occurring for W2 (45 and 17 kg ha^{-1} yr^{-1}), followed by W5 (41 and 30 kg ha^{-1} yr^{-1}) and W4 (57 and 48 kg ha^{-1} yr^{-1}). 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 the pre-cut values, ranging between 34-59 kg ha^{-1} yr^{-1} and 33-57 kg ha^{-1} yr^{-1}, respectively, but maintaining the same order of the magnitude of rates as the pre-cut period. Similarly, stream Ca^{2+} flux showed a decreasing pattern from the pre-cut period (1960-64) to the post-cut period (2046-50), though during the treatment periods increases in stream exports were simulated for all cut watersheds. Using a mass balance approach, net loss from the soil pool of exchangeable Ca^{2+} was estimated to vary between 6.1-8.3 kg ha^{-1} yr^{-1} among the watersheds during the pre-cut period 1960-64 and reached the peak values of 29, 19 and 2 kg ha^{-1} yr^{-1} for W2, W4 and W5, respectively during the treatment period (Figure 5.5). Simulated net Ca^{2+} began to show net soil retention during regrowth, reach rates between 2-4 kg ha^{-1} yr^{-1} among the watersheds for the simulated years 2046-50. 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} yr^{-1}), then followed by W2 (8 kg ha^{-1} yr^{-1}), W5 (6.6 kg ha^{-1} yr^{-1}) and W6 (4.8 kg ha^{-1} yr^{-1}).

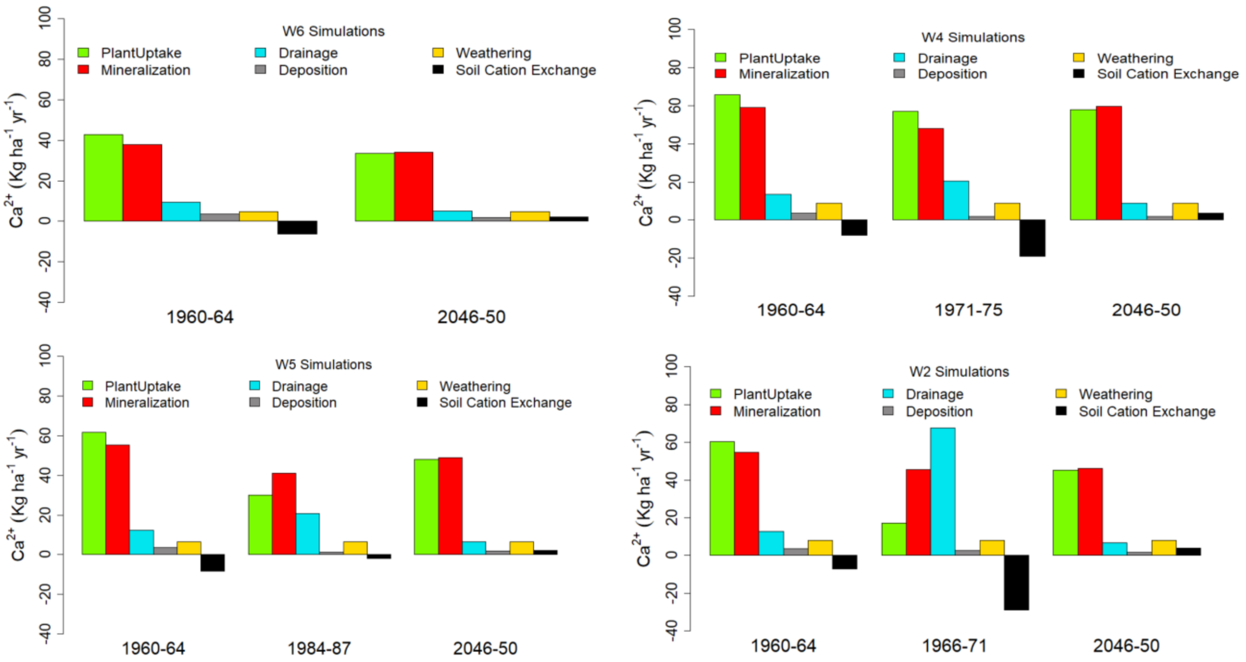


Figure 5.5. Comparison of simulated annual average Ca^{2+} budgets for the cut watersheds at the HBEF including W5, W4 and W2 during the pre-cut period (1960-64), treatment effect period (W5:1984-87; W2:1966-71; W4:1971-75) and post-cut period (2046-50). Simulations are compared with simulated Ca^{2+} budgets for the reference watershed (W6) during 1960-64 and 2046-50 periods.

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} yr^{-1} for soil SO_4^{2-} mineralization, and 22-17 and 15-13 kg S ha^{-1} yr^{-}1 for SO_4^{2-} plant uptake, respectively, from the 1960-64 period to 2046-50 period (Figure 5.6). Cutting decreased soil SO_4^{2-} mineralization and plant uptake in the waterhseds during the treatments. Note, SO_4^{2-} weathering rates were assumed to be constant among the watersheds. In order to calibrate stream SO_4^{2-} I assumed a higher monthly S weathering rate for W5 (0.2 gr m^{-2}), than W4 (0.17 gr m^{-2}) and W2 (0.13 gr m^{-2}).

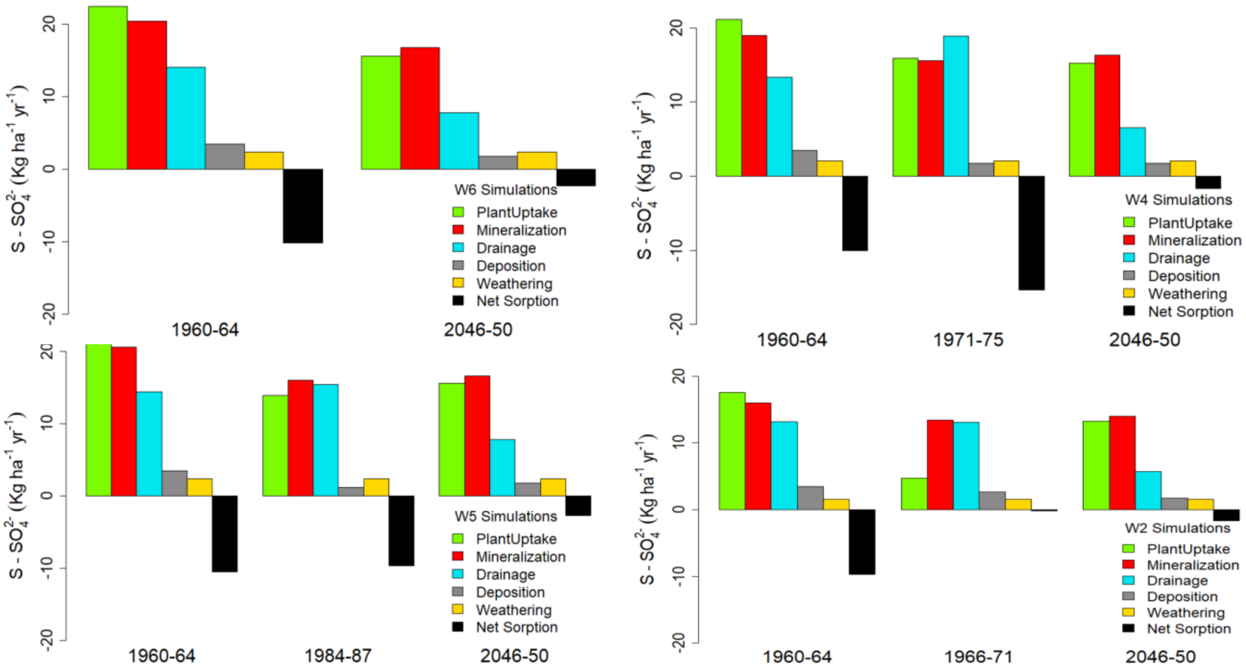


Figure 5.6. Comparison of annual average of simulated SO_4^{2-} budget for the clear-cut watersheds at HBEF including W5, W4 and W2 during the pre-cut period (1960-64), treatment effect period (W5:184-87; W2:1966-71; W4:1971-75) and post-cut period (2046-50). Simulations are also compared with simulated SO_4^{2-} budget for the reference watershed (W6) during 1960-64 and 2046-50 periods.

Simulated soil base saturation was around 29% for W4 and 28% for W2 historically (1850), exceeding the values of W5 (25%) and W6 (22%). Long-term acid deposition caused declines in soil base saturation in all watersheds at the HBEF until around 2000 (Figure 5.7). However, harvesting practices accelerated reduction in 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.

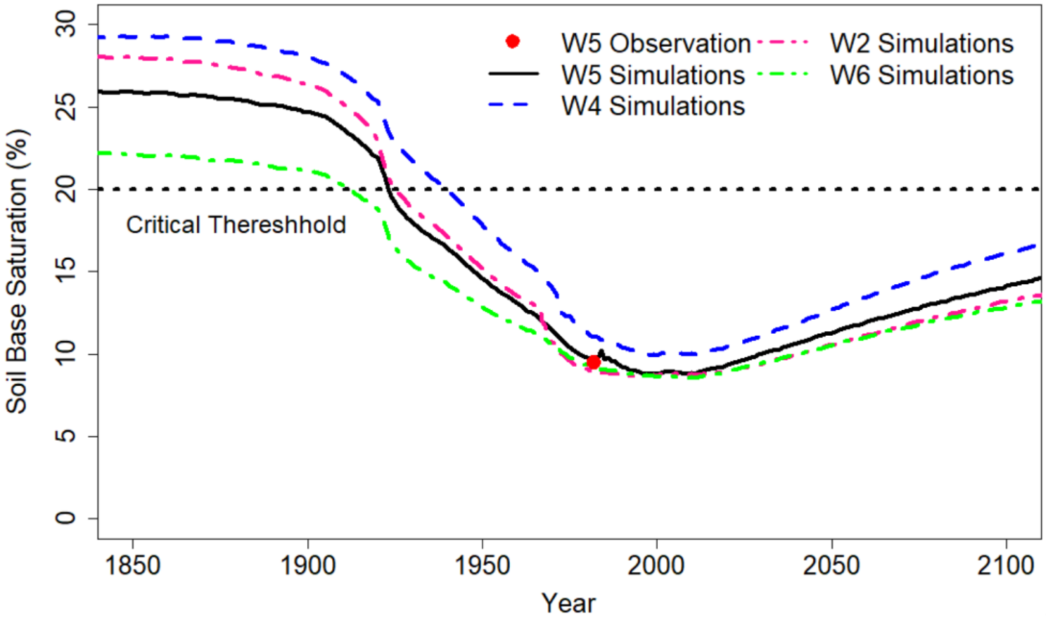


Figure 5.7. Simulation of long-term changes of soil base saturation for the reference watershed (W6) and clear-cut watersheds W5, W4 and W2. HBEF. Measured soil base saturation in 1982 year is shown for W5.

Model simulations were also used to compare cumulative Ca^{2+} loss in streamwater due to chronic leaching of strong acid anions from acid deposition (with the assumption of no cutting treatment) during 1960-2020, with the cumulative stream Ca^{2+} loss caused by the cutting treatments for each watershed. The amount of stored Ca^{2+} in biomass which was removed from the site from cutting and harvesting (for W4 and W5) or cut and was left on the site (for W2) was also estimated using model simulations. Note, to estimate cumulative Ca^{2+} loss in stream waters caused by acid deposition, I subtracted the annual values of stream Ca^{2+} from the simulated preindustrial value (1850) for each watershed. The results show that cumulative Ca^{2+} loss due to chronic leaching associated with inputs of acid deposition was estimated to be highest for W2 (276 kg ha^{-1}), followed by W5 (274 kg ha^{-1}), W4 (264 kg ha^{-1}) and W6 (205 kg ha^{-1}). The amounts of Ca^{2+} in biomass removed from the site due to treatment were 127 and 181 kg ha^{-1} for W5 and W4, respectively, and 176 kg ha^{-1} was cut at W2 but left on the site following the treatment. The results showed the cumulative stream Ca^{2+} loss due to cutting treatments were 252, 26 and 5 kg ha^{-1}, respectively for W2, W5 and W4.

5.2. **Discussion**

My results indicated that the modified model could 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. For W2, lack of biomass observations for the pre-cut period and limited biomass measurements for the period after the clear-cut challenge a rigorous evaluation of the simulation of the biomass accumulation pattern. Discrepancies between biomass measurements and simulations for cut watersheds might be explained by the inability of PnET-BGC to depict shifts in tree species composition following the disturbance. PnET-BGC uses 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 five years after the cut that vegetation is mainly dominated by herbs and shrubs, and biomass accumulation rates closely follow similar levels for the different cutting treatments (Bormann and Likens, 1979; Fahey et al., 2005; Martin et al., 1989). Around 15 years after the cut, W5 biomass accumulated at a faster rate than W4 and W2. Note, in order to capture greater total aboveground biomass for W5 than for W4 and W2 in model simulation, it was necessary to calibrate the model with 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 at the commercial whole-tree harvested site compared with the strip-cut watershed (W4) or the cut watershed with trees and residual slash left on site and herbicide application to suppress the pin cherry seed bank and germination (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 HBEF, concluding that total aboveground biomass accumulated at much faster rate on the block clear-cut than on the strip cut, due to higher density of pin cherry in the block cutting in response to a higher initial nutrient release. However, the strip-cut harvest 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 a second- growth watershed (W6) at the HBEF. These results suggest that though the different harvesting practices influence initial forest composition and growth, the overall impact on total aboveground biomass is minimal over the long-term at the HBEF. 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). Jiang et al. (2002) investigated the influence of different harvesting regimes, including conventional harvest (100% removal of stem but branches, needles and roots left on site) and whole-tree harvesting (100% removal of stem and 90% removal of branches and needles but all roots left on site) on the carbon stocks of a Chinese boreal forest using long-term simulations by CENTURY 4.0. They projected aboveground biomass around 150 Mg C ha^{-1} for conventional harvest and 130 Mg C ha^{-1} for whole-tree harvest, respectively, over 100-year simulations, comparable to values for my projections for the cut watersheds by 2100 year (around 100 t C ha^{-1}) (Jiang et al., 2002a). Rolff and Ågren (1999) used the ecosystem model, NITMOD, to project maximum aboveground biomass of Norway spruce stands of around 250 t ha^{-1} for highly productive stands, 220 t ha^{-1} for medium productivity stands and approximately 100 t ha^{-1} for the least productive sites in 100-year simulations (Rolff and Ågren, 1999). Simulations of PnET-BGC projected around 220 t ha^{-1} of aboveground biomass accumulation by the year 2100 for all cut watersheds.

The comparison of field measurements from different experimentally cut watersheds with the model simulations confirmed the ability of the modified model to depict both short- and long-term hydrologic and biogeochemical responses to a range harvesting regimes. However, there are some discrepancies between monthly/yearly stream discharge and stream water chemistry measurements and simulations. The model performed better in simulations of increases in annual stream flow following the treatment of W5 and W4 than for W2. Underestimation of stream discharge for W2 might be attributed to the modeled overestimation of evapotranspiration during the 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 some minor natural disturbances such as soil freezing and insect defoliation in the early 1970s and 1980s (Fitzhugh et al., 2003; Gbondo-Tugbawa et al., 2001), and an ice storm in 1998 at HBEF that caused some damage to vegetation and affected the stream water chemistry in the ways that were not considered in simulations because the intensity of the disturbances have not been quantified. Another contributing factor may be that PnET-BGC is not spatially structured to adequately depict the physical sequence of actual strip-cut of W4. For W2, overprediction of some major solutes including NO_3^- and Ca^{2+} in stream water during the treatment years and the period after the cut might be explained by different factors including overestimation of nitrification rates during the treatment years and overestimation in mineralization resulting from overestimation of biomass production for the period after the cut. 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.

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 to 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, a greater uptake of N by the rapidly growing new stand, increases in the immobilization of N by soil organisms, and/or an increase in denitrification (Hornbeck et al., 1986). Nutrient budget simulations suggest that it can be associated to decrease in N mineralization during the earlier years of regrowth in cut watersheds possibly because of reduction in litter inputs.

My nutrient budgets developed from simulations showed that the major response of the nutrient cycles in cut watersheds occurred during the treatment period, though there are some differences in nutrient fluxes from the pre- harvest period (1960-64) compared to the long-term recovery values after the cuts (2046-50). For example, N mineralization was higher in W5 than W2 and W4 prior to cut (1960-64), possibly due to higher biomass production, but longer term simulations (2046-50) indicated greater N mineralization in W2 (112 kg ha^{-1}) followed by W4 (108 kg ha^{-1}) and W5 (106 kg ha^{-1}) which had values similar to the reference watershed value (W6, 108 kg ha^{-1}). Mineralization depends on several factors, such as total quantity of biomass production and litter inputs to the soil, temperature, precipitation and soil moisture Bormann and Likens, 1979; Yanai et al., 2003). My results suggest that greater N mineralization of W2 might be related to the greater biomass accumulation by 2050 compared to the other cut watersheds and therefore higher litter inputs to the soil, as well as observed temperature increases caused by climate change from the cut to present and long-term decomposition of organic matter left on the site following the experimental harvest in 1965. Simulations also showed that net N release was highest during the time of watershed cutting and increased with the intensity of the cut. This discrepancy among simulated N sources and sinks might be explained by the model limitations in depicting denitrification processes or organic decay rates of soil layers.

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 emissions of SO_2 and NO_x and elevated SO_4^{2-} and NO_3^- deposition and leaching (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; Likens et al., 2002; Stoddard et al., 1999) have curtailed this depletion (Likens et al. 1996). The greatest depletion of soil exchangeable Ca^{2+} occurred on W2 due to the treatment (1966-71) associated leaching of NO_3^- from the delay of regrowth associated with the herbicide treatment coupled with high acid deposition at this time. With regrowing vegetation and controls on emissions of SO_2, simulated exchangeable 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} yr^{-1}) for the simulated years 2046-50. Soil base saturation is considered a critical indicator of soil acidification stress due to atmospheric acid deposition or forest cutting (Cleavitt et al., 2018; Driscoll et al., 2001; Sullivan et al., 2013). My simulation results are consistent with previous studies indicating that harvesting regimes with higher intensity can lead to greater depletion of exchangeable Ca and reductions in base saturation at the site (Aherne et al., 2012; Cleavitt et al., 2018; Hornbeck et al., 1986). Simulation results also showed historically (~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 W5 and W6 which are characterized by lower weathering rates, estimated from model calibration. 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 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 indicated 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 region impacted by acid deposition forest harvesting can exacerbate chronic soil Ca^{2+} depletion (Cleavitt et al., 2018; Juice et al., 2006). Forest ecosystems characterized by low base saturation and exchangeable Ca^{2+} may experience limited regeneration and health of sugar maple over the long-term with additional depletion by acid deposition or forest harvesting (Cleavitt et al., 2018; Federer et al., 1989; Schaberg et al., 2006; Sullivan et al., 2013). Some studies report a critical threshold of 20% soil base saturation for successful regeneration of sugar maple (Cleavitt et al., 2018; Sullivan et al., 2013). Repeated harvesting and intensive tree removal would be expected to aggravate chronic soil available Ca^{2+} depletion by acid deposition (Hornbeck et al., 1986; Weetman and Webber, 1972). Cleavitt et al. (2018) 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, though in both cut watersheds soil nutrient depletion was supposedly still large enough to limit regeneration of sugar maple, with more a more severe impact on W5. My work serves to amplify their study. I found that a higher Ca^{2+} weathering rate on W4 may have resulted in a slightly higher historical base saturation than W5 and likely allowed for the more rapid recovery (Figure 5.7). Furthermore, some studies highlight that the processes of mineral weathering, atmospheric deposition and detrital recycling on many base-poor soils cannot offset soil available Ca^{2+} depletion due to the combined effects of forest harvesting plus acid deposition (Cleavitt et al., 2018).

As mentioned above, PnET-BGC does not depict changes in tree species composition following forest cutting. The effects of the shift in species composition is an important consideration in understanding watershed recovery from forest cutting. However, it would be a major undertaking to simulate the dynamics of vegetation, hydrology and biogeochemistry following the disturbance. Model development to simultaneously depict the competition among tree species to determine composition would add an additional level of complexity to a multi-element soil-layer model that would be needed to comprehensively simulate the interactions of major elements with various species and these effects on competitive growth. It seems unlikely that there is sufficient information for all tree species to parametrize and calibrate such a model.

5.3. **Conclusions and suggestions for further research**

In this phase of study, 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 tree 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 the cut watersheds was found to approach similar levels by the end of the simulations period (2100). 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 (Creutzburg et al., 2016; Seely et al., 2002; Wang et al., 2014). Therefore, among different cutting approaches considered in this phase of study, it is clear that the treatment applied in W2 (clear-felling with cut trees were left in place and herbicide application) would not be used as a forest management option because the cut biomass is not utilized and the resulting substantial loss of essential nutrients from the site. Compared with whole-tree harvest, progressive strip cutting attenuates effects on hydrologic and nutrient cycles. However, there is concern that both cutting practices result in a less desirable mix of commercial species in the new regrowing stand (Martin et al., 1989). Therefore, further investigation is needed to contrast different cutting strategies to make sure 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. 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.

6. Simulation of the effects of forest harvesting under changing climate to inform long-term sustainable forest management using a biogeochemical model, PnET-BGC

6.1. Results

In this phase of study, I used a version of PnET-BGC that was modified to simulate forest harvesting (Valipour et al. 2018) and projected various ecosystem pools and fluxes including aboveground biomass, woody debris, soil organic matter and element budgets to evaluate long-term site productivity under varied forest management practices. Under stationary climate conditions for the reference (no future cutting) scenario, modeled carbon stock in living aboveground biomass increased to 112 t ha^{-1} by 2200 year, 30% and 70% higher than the values in 1982 (the year before the experimental W5 clear-cutting) and 2019 (the year before the initiation of hypothetical logging regimes), respectively. For all harvesting scenarios, model simulations showed rapid declines in biomass at the time of harvest and then slow increases with the regrowth of the trees (Figure 6.1). Under different harvesting regimes, the carbon stock in aboveground biomass increased to different levels by the end of simulations (2200) that were lower than the no harvesting scenario, with greater declines as the harvest rotation shortened and harvest intensity increased. The most conservative scenario, with a 90-year rotation length and a 40% watershed cutting level, showed only a minor decline of biomass accumulation relative to the reference no harvest scenario (112 t C ha^{-1}), reaching to 101 t C ha^{-1} by the end of the simulation (2200). In contrast, the most aggressive cutting scenario of a 30-year rotation period with an 80% intensity, resulted in aboveground biomass of 64 t C ha^{-1} by 2200.

Under future climate change, an increase in biomass accumulation rate was projected for all forest management scenarios, compared with constant climate conditions. In the reference (no harvest) scenario, modeled carbon stock in aboveground biomass was approximately three times

(330 t ha^{-1}) greater than the value simulated for stationary climate conditions in 2200, four times the value of aboveground biomass of 1982 (84 t ha^{-1}) and five times the value of 2019 (64 t ha^{-1}). Harvesting regimes resulted in lower projections of aboveground biomass accumulation compared to the reference (no logging) scenario, varying between 136 and 306 t ha^{-1} (Figure 6.1). Simulations suggest that aboveground carbon storage increases by extending the logging rotation length and cutting a smaller fraction of the watershed area.

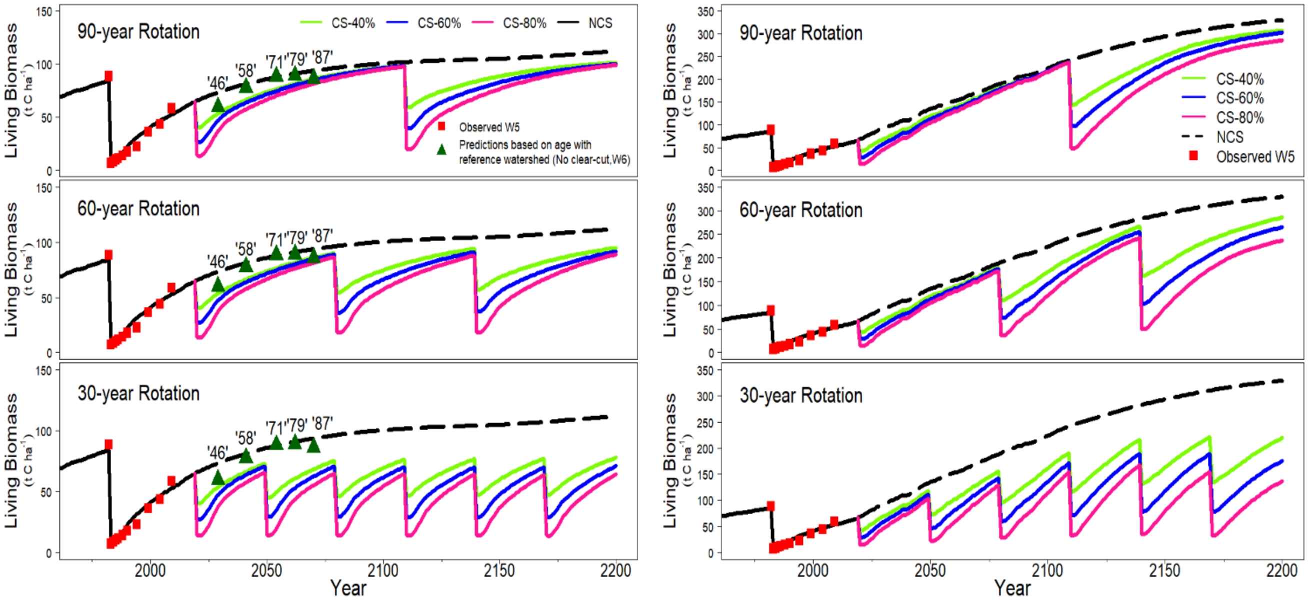


Figure 6.1. Simulation of temporal dynamics of carbon storage in aboveground biomass across ten forest management scenarios under stationary climate (left) and future climate change (right) conditions. Model simulations are compared with measured values for clear-cut W5, HBEF in 1983 and W6 adjusted for years after cutting.

Modeled carbon storage in woody debris followed a similar pattern as aboveground biomass over the simulation period (Figure 6.2). Under continuing current climate, harvesting regimes reduced the amount of woody debris, with pools ranging between 11t ha^{-1} and 21 t ha^{-1}by the year 2200, compared with the no harvest scenario (24 t ha^{-1}).

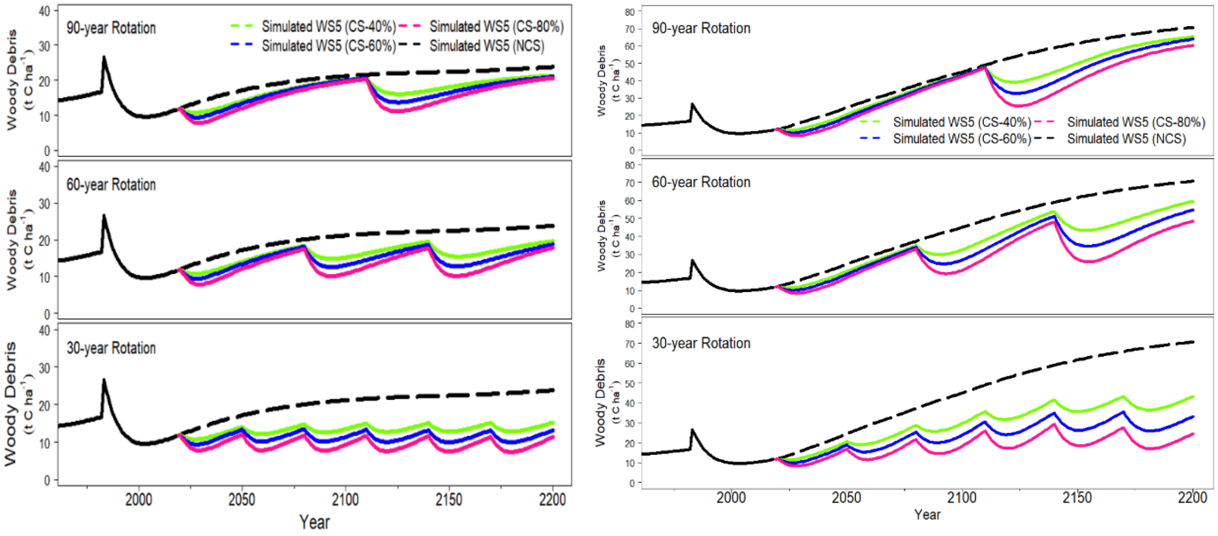


Figure 6.2. Simulation of temporal dynamics of carbon storage in woody debris pool across ten forest management scenarios under stationary climate (left) and future climate change (right) conditions.

The simulated soil organic carbon (SOC) pool approximately captured observations during the early years following the experimental clear-cut in 1983 except for an overprediction of the minimum value 8 years after the harvest (Figure 6.3). For the constant climate period, the soil organic pool slowly accumulated carbon reaching 72 t ha^{-1} by the end of the simulation period, 6% higher than the value in year 2019 (67 t ha^{-1}). The soil organic carbon pool showed little response to various harvesting scenarios, with values ranging between 56 t ha^{-1} and 69 t ha^{-1}by the end of the simulation period. Overall, the pool of soil organic carbon remained relatively stable, with only minor declines from logging regimes. In contrast, under changing climate, the modeled soil organic pool with no harvest lost carbon slowly, decreasing to 52 t ha^{-1} by 2200. In contrast, the interactive effects of climate change with harvesting regimes accelerated the decline in the soil organic carbon pool over the 180-year simulation period, with values between 23 t ha^{-1} and 47 t ha^{-1} in 2200 year, depending on the harvesting approach considered.

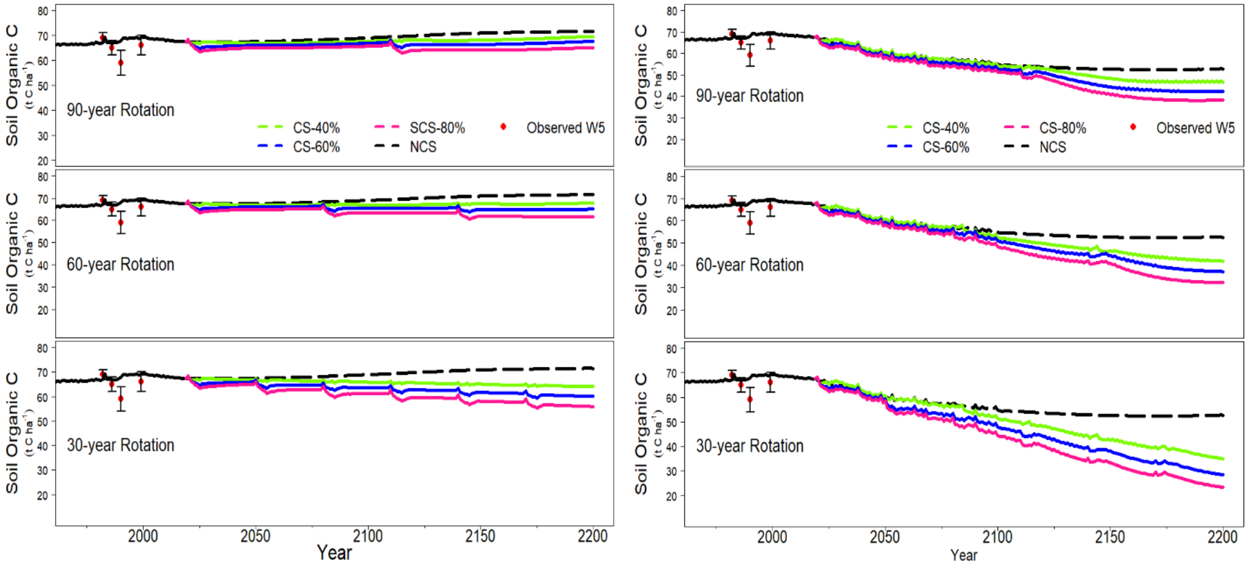


Figure 6.3. Simulation of temporal dynamics of carbon storage in soil organic matter across ten forest management scenarios under stationary climate (left) and future climate change (right) conditions. Model simulations are compared with measured values for clear-cut W5, HBEF in 1983.

Total ecosystem stored carbon (i.e., sum of aboveground biomass, woody debris and soil) reached a maximum value of 207 t C ha^{-}1 for the no cutting scenario in 2200, while harvesting of trees decreased this value in the range of 131-192 t C ha^{-1}, with greater reductions for shorter logging rotation lengths and greater watershed cutting intensities (Figure 6.4, A2). The stored carbon was distributed among the ecosystem pools with highest fraction of carbon occurring as aboveground living biomass (44-60%), followed by soil (35-37%) and woody debris (8-13%) under all management practices (Figure 6.4). Simulations indicated that greater cumulative amounts of wood products would be extracted under the more intense removal scenarios over the 180-year simulation period of management practices, varying between 63-303 t C ha^{-1}. By the end of the simulation period, the amounts of total sequestered carbon (i.e., sum of total ecosystem stored carbon and total removed wood products) under all logging management scenarios exceeded the value for the reference scenario without harvesting (207 t C ha^{-1}), ranging between

255 and 434 t C ha^{-1}. Overall, under constant climate, woody debris and living aboveground biomass experienced the largest decline in carbon sequestration with harvesting regimes compared to the soil organic pool, ranging between 10-52% and 9-42%, respectively by 2200. The lowest decline in carbon sequestration occurred in the soil organic pool (3-22%) with tree harvesting, in comparison to the no harvest scenario.

Climate change increased carbon allocation to the woody debris pool (71 t C ha^{-1}) by 2200 for the no harvest scenario, though with the clear-cutting practices considered this value decreased to 24-65 t C ha^{-1} (Figure 6.2, 6.4). The carbon stock of soil had the lowest fraction of total ecosystem C pool under changing climate (7-12%) by the end of simulation period. Greater amounts of carbon were accumulated in the aboveground biomass (57- 85%) followed by woody debris (13-18%) for all scenarios of management practices by the end of simulation period (Figure 6.4). Total ecosystem carbon stored (aboveground biomass, woody debris and soil) peaked at 452 t C ha^{-1} in 2200 for the no cutting scenario, while harvesting trees decreased this value in the range of 184-418 t ha^{-1} under changing climate (Figure 6.4, A2). Cutting trees resulted in higher total carbon sequestration by the end of simulations, ranging between 536 t ha^{-1} and 794 t ha^{-1} relative to the no cut scenario (452 t ha^{-1}) (Figure 6.4). For all logging practices, except the 30-year cutting periods, soil exhibited the largest decline in carbon accumulation (12-39%), followed by woody debris (7-31%) and aboveground biomass (6-28%), relative to the reference no harvest regime under changing climate. For all intensities of tree removal for the 30-year cutting interval, woody debris experienced a larger decline in carbon (39-65%) than aboveground biomass carbon (33-59%) and soil carbon (33-56%).

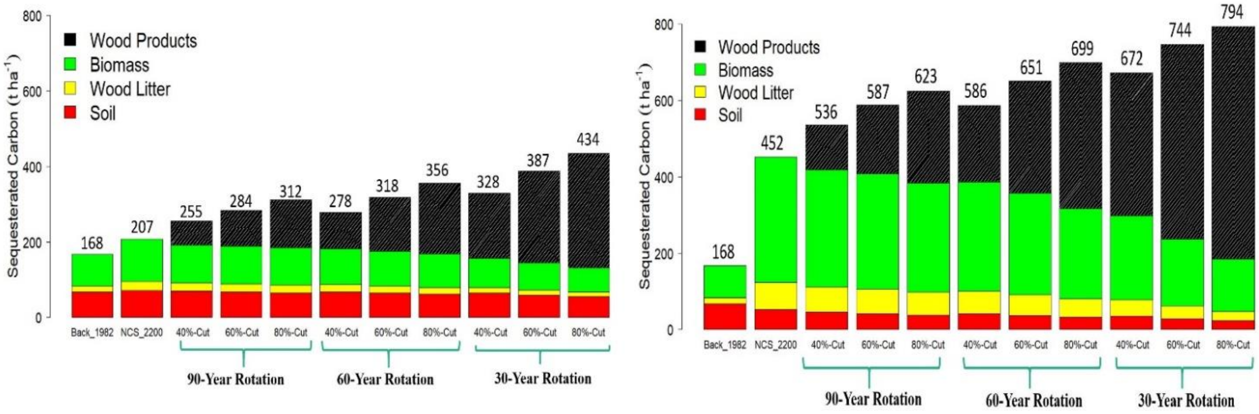


Figure 6.4. Comparison of simulated sequestrated/stored carbon in different ecosystem pools including soil, woody debris, aboveground biomass and cumulative removed wood products at the end of simulation period (2200) for ten scenarios of forest management under stationary (left) and changing (right) climate conditions. The values are also compared with background values, prior to experimental clear-cut W5, HBEF (1982).

A similar analysis was conducted to evaluate the effects of tree harvesting strategies on N and Ca^{2+} stocks over the long-term. For the harvesting simulations, temporal variations of N in plant, woody debris and soil pools were similar to those of carbon pools. Under constant climate, levels of stored N in plant, woody debris and soil were reduced by 10-37%, 11-52% and 3-22% by the end of the simulation period compared to corresponding values for the no harvest scenario (655 kg ha^{-1}, 131 kg ha^{-1} and 4491 kg ha^{-1} respectively). Under changing climate, the response of N stocks in vegetation, woody debris and soil were predicted to be greater than under stationary climate, declining by 7-58%, 8-67% and 11-54%, respectively at the end of the simulation period relative to the corresponding values for the no harvest scenario, 1745 kg ha^{-1}, 369 kg $ha^{-1}$and 3208 kg ha^{-1}, respectively (Figures A4, A5, A6, 6.5). Under all management scenarios with stationary climate, the simulated distribution of N stocks among ecosystem pools varied between 85-88% for soil, 10-12% for total plant biomass and 1-2% for woody debris. With the climate change scenario, the percentage of N stocks in soil, plant matter and woody debris ranged between 56-63%, 31-36% and 5-7%, respectively, for all management practices considered.

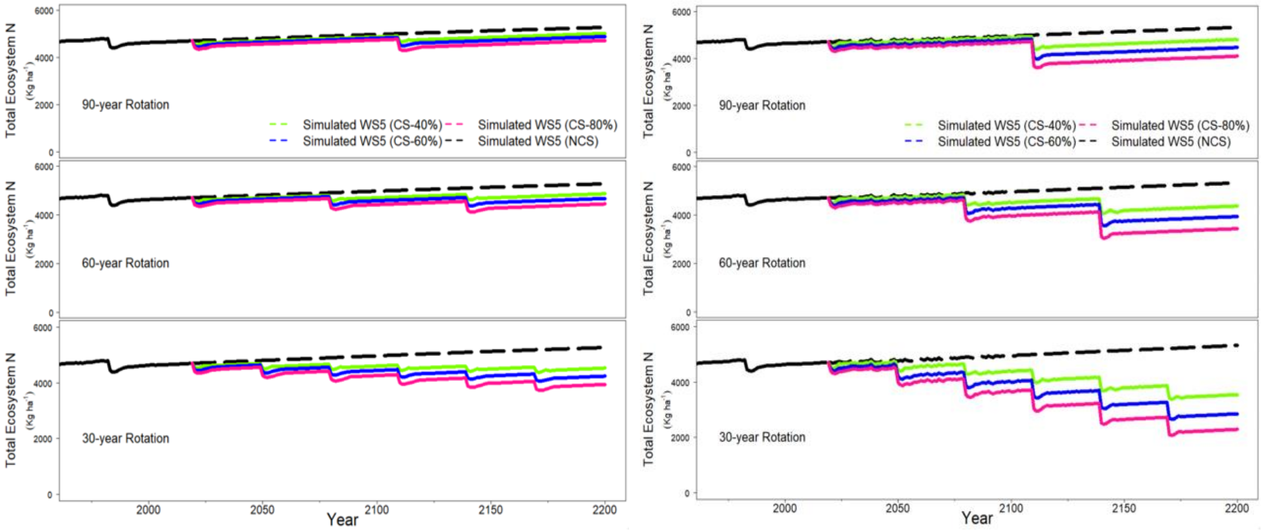


Figure 6.5. Simulation of temporal dynamic of total ecosystem stored N (sum of stored N in plant, woody debris and soil) across ten forest management scenarios under stationary climate (left) and future climate change (right) conditions.

Simulations of exchangeable soil Ca^{2+} stocks showed a declining pattern by the end of the simulations for both stationary and changing climate conditions under the no harvest scenario, reaching to 825 kg ha^{-1} and 627 kg ha^{-1} in 2200, respectively (Figure A7). Logging practices accelerated the decline in exchangeable Ca^{2+} in soil with values ranging from 6-20% below no harvest values for constant and 15-51% for changing climate across the range of harvesting practices considered. Stored Ca^{2+} in plant biomass was also diminished by harvesting scenarios, ranging between 10-33% for constant and 9-53% for climate change scenarios, relative to the reference no harvest scenario levels of 185 kg ha^{-1} and 608 kg ha^{-1}, respectively (Figure A8). A higher fraction of available Ca^{2+} occurred in soil (81-84%) rather than plant biomass (15-18%) under constant climate for all management actions. While an approximately equivalent fraction of available Ca^{2+} was simulated in biomass (48-54%) and soil (45-51%) under climate change for all management approaches.

Under current climate with no future cutting, long-term simulations indicated that the percent soil base saturation that was depleted from historical acid deposition gradually increased from a minimum value of about 7% (in 2000) by the end of the simulation period, but remained depleted below pre-acid deposition values (26% vs 17.4%). The harvesting of trees resulted in a slight increase in percent soil base saturation the first year after the clear-cut, consistent with elevated leaching of nutrient cations in soil and stream water. However, depletion of soil available cations continued to decrease to below pre-cut levels during the early years after the cut, and then increased gradually with vegetation regrowth. Long-term harvesting practices were predicted to amplify the depletion of soil base saturation, reducing values to 17-14% by the end of the simulation period, relative to the reference no-cut scenario (Figure 6.6). For a no harvest scenario under climate change, percent soil base saturation increased from a minimum value around 2000, but at a slower rate than under the constant climate scenario reaching 15%. The interaction of climate change with harvesting regimes increased the decline in percent soil base saturation, with values ranging between 10-13.7% in 2200, relative to the reference no-cut scenario.

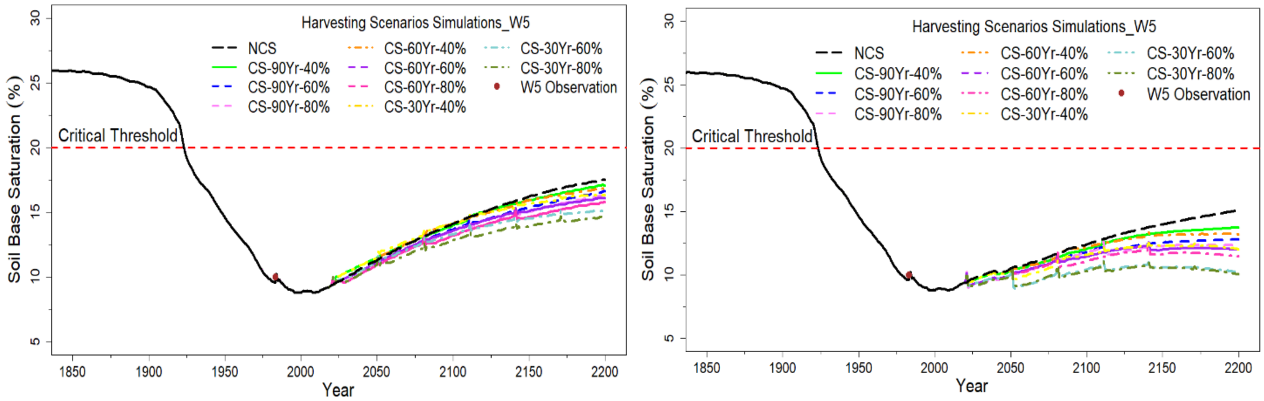


Figure 6.6. Simulation of temporal changes of soil percent base saturation across ten forest management scenarios under stationary climate (left) and future climate change (right) conditions. Model simulations are compared with measured value prior to experimental clear-cut W5, HBEF (1982).

Nutrient budgets were calculated using the model to demonstrate the changes in source/sink behavior of elements associated with intensive timber-harvesting and changing climate. A 60-year cutting period of 80% of the watershed area was selected to illustrate temporal variations in nutrient pools and fluxes under both constant and changing climate (Figure 6.7). Soil mineralization and plant uptake of different elements were closely coupled and remained relatively stable through the simulations under constant climate with no harvesting. With the removal of trees, the model could reproduce the reduction in mineralization and plant uptake rates at the time of the harvest and then values increased with the regrowth of trees, although at lower rates. With harvesting regimes, the rate of plant uptake was reduced by 5-22% for N, 5-17% for Ca^{2+} and 4-17% for SO_4^{2-} at the end of simulations, relative to the reference no harvest values of 117 kg ha^{-1} yr^{-1}, 44 kg ha^{-1} yr^{-1} and 15 kg S ha^{-1} yr^{-1}, respectively (Figure 6.7). The model was also able to capture the elevated rates of nitrification and NO_3^- leaching, and subsequent increases in leaching of nutrient cations (e.g. Ca^{2+}, Mg^{2+}) in soil solutions and stream water at the time of harvest followed by the rapid recovery of leaching nutrients to pre-cut levels during the early years after the cut (Figure 6.7, 6.8). The model also captured soil adsorption of SO_4^{2-} at the time of harvesting and then its subsequent release into soil solution and stream water during the early years after the harvest due to increases in pH (Figure 6.7, 6.8). For an equivalent rotation period, greater leaching of nutrients occurred with increased intensity of harvesting. Modeled peaks in nutrient leaching (i.e., NO_3^-, Ca^{2+} and Mg^{2+}) at time of harvesting declined with sequential cuttings for each logging management scenario (Figure 6.8). For instance, the cutting of trees at a 60-year rotation period with 80% intensity caused elevated leaching of elements in the second year to decline following each successive harvest, by values ranging between 28-39 kg N ha^{-1} for NO^-_3, 23-31 kg ha^{-1} for Ca^{2+} and 7-8 kg ha^{-1} for Mg^{2+} by the end of simulation period (Figure 6.8).

Climate change was predicted to increase annual variabilities in hydrology and nutrient pools by the end of the century (2100). The combination of harvesting with climate change resulted in a similar pattern of nutrient budgets as those under stationary climate. For the logging practices considered, the rate of plant uptake declined by 6-48% for N, 7-42% for Ca^{2+} and 4-43% for S at the end of the simulation period, relative to the values for the reference no harvest scenario of 116 kg N ha^{-1}, 51 kg Ca^{2+} ha^{-1} and 15 kg S ha^{-1}, respectively (Figure 6.7). Simulated peaks in nutrient leaching with logging events showed higher values under climate change than constant climate, but also followed the declining pattern of peak loss (Figure 6.8) through the simulation period (2200). For example, the cutting of trees at a 60-year rotation period with 80% intensity led to an elevated leaching in the second year following each cut, but the peak leaching value decreased through the simulation period by values, ranging between 26- 66 kg N ha^{-1} for NO_3^-, 20-50 kg ha^{-1} for Ca^{2+} and 6-13 kg ha^{-1} for Mg^{2+} by the end of the simulation period (Figure 6.8).

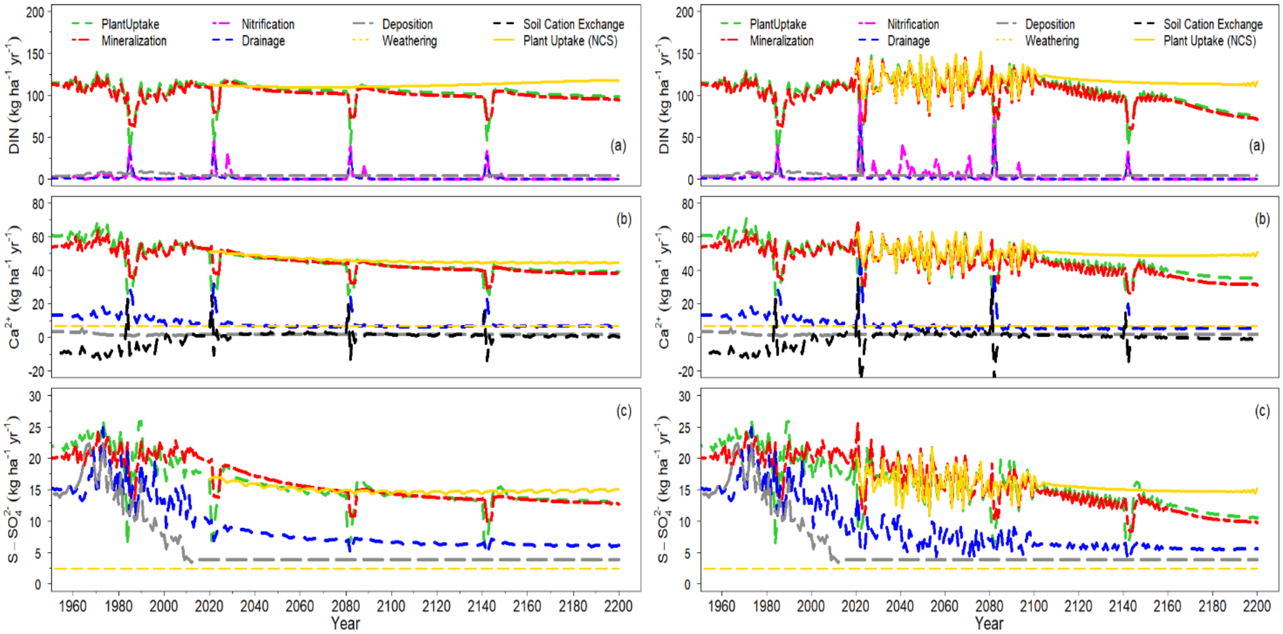


Figure 6.7. Simulations of annual nutrients budgets for W5, HBEF, including total dissolved inorganic nitrogen (DIN) (a), Ca^{2+} (b) and $S\text{-}SO_4^{2-}$ (c) for a 60-year cutting rotation with 80% cutting intensity under stationary climate (left) and future climate change (right).

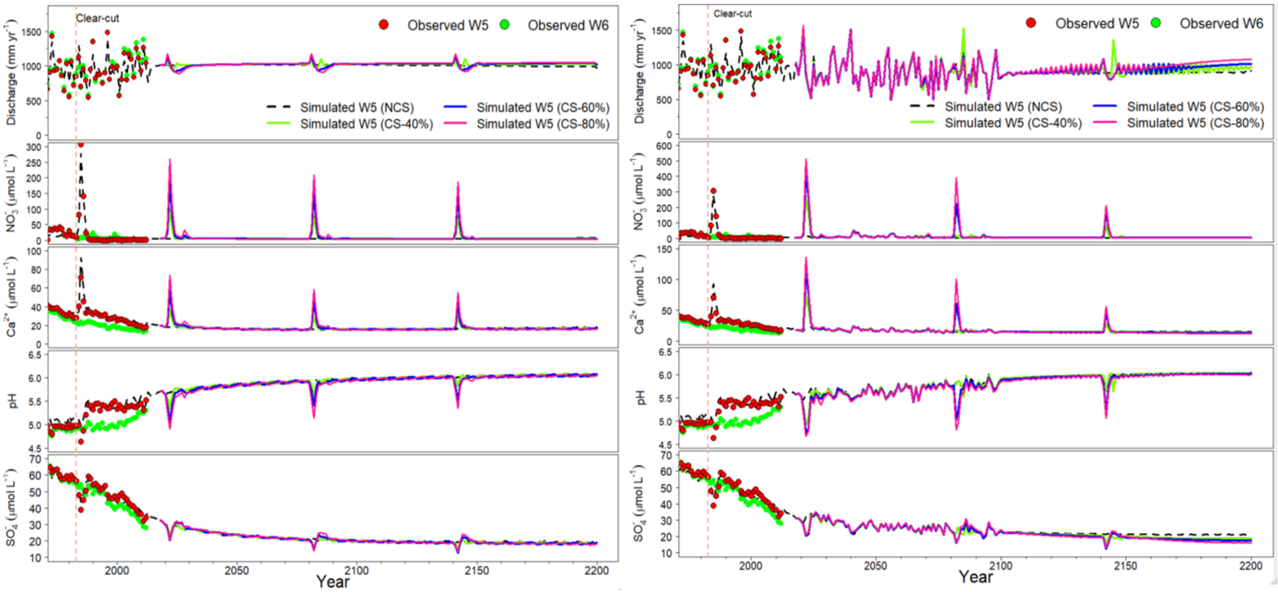


Figure 6.8. Simulation of stream water discharge and chemistry for 60-year cutting period with three intensity levels (40%, 60% and 80%). Simulations are also compared with measurements of W5 (experimental clear-cut watershed in 1883) and W6 (reference watershed).

Cumulative leaching of nutrients over the 180-year simulation period of management practices indicated slightly higher losses under constant climate by 2200 than for changing climate for a no future cutting scenario. Cumulative leaching of NO_3^-, Ca^{2+} and Mg^{2+} reached levels of 125 kg N ha^{-1}, 1211 kg ha^{-1} and 322 kg ha^{-1}, respectively, from 2020-2200 for stationary climate without cutting, compared to the values of 116 kg ha^{-1}, 1045 kg ha^{-1} and 293 kg ha^{-1} for varying climate (Figure 6.9). Cumulative leaching of nutrients from 2020 to 2200 increased substantially with logging scenarios compared to the reference no cutting scenario for both climate conditions. The range of cumulative stream loss of NO_3^- under climate change conditions (139-391 kg N ha^{-1}) exceeded values under constant climate (181-513 kg ha^{-1}) for various harvesting conditions. In contrast, under climate change conditions cumulative leaching of Ca^{2+} and Mg^{2+} ranged from 1077-1247 kg ha^{-1} and 301-339 kg ha^{-1}, respectively, for different harvesting actions by 2200 remaining below the range of 1226-1367 kg ha^{-1} and 344-375 kg ha^{-1}, respectively, under stationary climate.

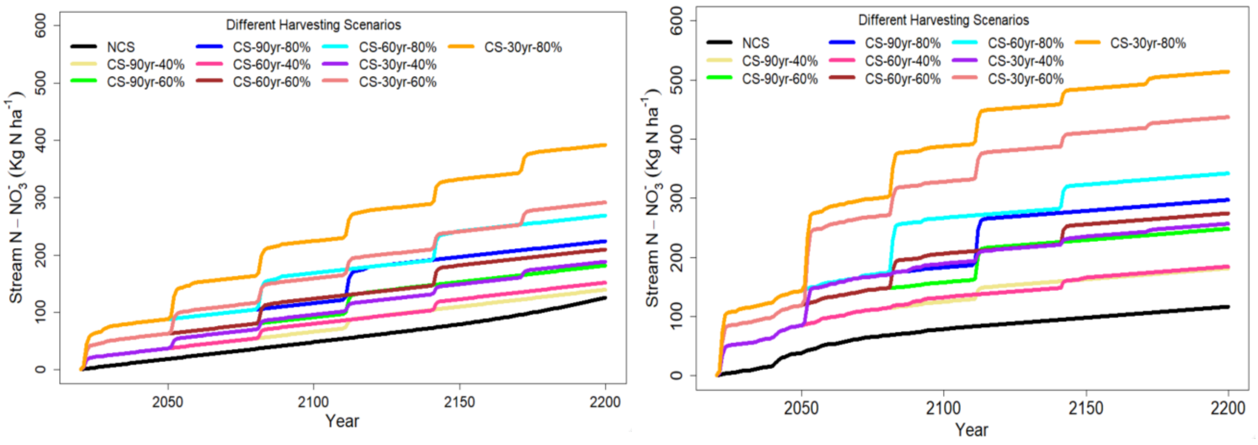


Figure 6.9 Simulations of cumulative stream leaching of NO_3^- for ten forest management scenarios for W5, HBEF under stationary climate (left) and future climate change (right).

6.2. Discussion

In this phase of study, I evaluated multiple forest harvest practices under future climate change to examine the interactions of these disturbances on major nutrient pools as a measure of long-term forest productivity.

My results are consistent with much the literature showing the ability of forests to maintain large stores of carbon and other nutrients in various ecosystem pools (e.g. aboveground living biomass, woody debris, soil pools) under both stationary and climate change conditions without future cutting. Cutting strategies, however, are projected to decrease ecosystem nutrient pools to varying degrees (Aherne et al., 2012; Jiang et al., 2002; Scheller and Mladenoff, 2005; Wu et al., 2017). Under all forest management strategies considered, the rates of carbon sequestration and nutrient assimilation (e.g. N, Ca^{2+}, Mg^{2+}) in aboveground living biomass and woody debris were projected to be significantly higher under changing climate than stationary climate, mainly due to the fertilization effects of CO_2 associated with enhanced water use efficiency and growth rate

(Boisvenue and Running, 2006; Ollinger et al., 2008; Pourmokhtarian et al., 2017; Scheller and Mladenoff, 2005). Conversely for all management scenarios, climate change resulted in a marked loss of soil organic matter leading to elevated leaching of soil nutrients compared with constant climate conditions, primarily due to higher rates of soil mineralization caused by increases in temperature and increases in precipitation and runoff (Boisvenue and Running, 2006; Creutzburg et al., 2016; Pourmokhtarian et al., 2017). Overall, my simulations demonstrated the greater sensitivity of forest ecosystem pools to logging strategies under climate change relative to constant climate conditions. These effects are accentuated with shortening the length of the cutting intervals and increasing watershed harvesting intensity. These simulations highlight the challenges in maintaining the long-term productivity of managed forests with changing climate. Simulations of both climate conditions considered showed greater sensitivity to varying the length of cutting period than altering cutting intensities, consistent with previous reports (Blanco et al., 2005; Jiang et al., 2002; Wei et al., 2003). Under equivalent cutting frequency, the effects of increasing harvesting intensity on reduction of ecosystem stored carbon, including aboveground biomass, woody debris and soil organic matter, was greater for changing climate than constant climate.

The literature is inconsistent on effects of logging on the forest floor, including increases, decreases or no change in soil carbon (Creutzburg et al., 2016; Hartmann et al., 2012; Nave et al., 2010; Palviainen et al., 2004; Yanai et al., 2003). Results from my study confirm previous analyses suggesting that tree harvesting under constant current climate should mostly affect biomass and woody debris rather than soil carbon (Blanco et al., 2005; Creutzburg et al., 2016; Jiang et al., 2002; Wei et al., 2003). In contrast, simulations suggest climate change will have negative impacts on soil fertility. Soil organic matter plays a significant role in supporting soil nutrient availability, which is essential to maintain site productivity, particularly in N-limited forests. Nitrogen is generally the growth-limiting nutrient in temperate forest ecosystems and changes in N availability impact forest productivity (Aherne et al., 2012; Seely et al., 2002). My simulations suggest that the soil carbon to nitrogen ratios fluctuate with harvesting events (in the range of 15-17), and eventually generally exceed values for the reference no cutting scenario by 2200 under both climate conditions (Figure A20). The exception to this pattern occurred for the 30-year cutting period under climate change with soil C:N remaining below the reference values. Results of soil C: N confirm that harvesting would decrease both soil carbon and nitrogen storage, but nitrogen loss would generally exceed that of organic carbon. My results suggest this effect would be heightened under climate change, potentially increasing nitrogen limitation of the northern forest ecosystem. Soil C: N ratio was much lower with a 30-year cutting period relative to the reference no cut scenario under climate change likely due to significant reduction of the aboveground biomass and woody debris pool that results in a rapid loss of the soil organic pool, limiting plant growth. This result confirms previous findings that litter pools represent a small fraction of the total ecosystem carbon pool, but rates of litter production has a significant impact upon carbon stored in the SOM pool (Seely et al., 2002).

My results show that as more wood is extracted from the forest via harvesting the greater the depletion of nutrient cations from the soil exchangeable complex, resulting in a decline in soil base saturation percent (Aherne et al., 2012; Blanco et al., 2005). Depletion of soil base cations is accelerated under climate change due to increases in soil mineralization, plant uptake and enhanced biomass accumulation. Previous findings by Aherne et al. (2012) also show that the increasing harvest intensity by a shift from stem-only harvest to whole-tree harvest approximately doubled the removal of biomass, tripled the removal of base cations and quadrupled the removal of N from the study catchments under current climate. However, their study indicated that climate change would compensate for the depletion of soil base cations caused by harvesting, mainly due to increased weathering rates associated with warmer temperatures. Note that Aherne et al. (2012) did not depict the effects of CO_2 fertilization on enhanced tree growth and nutrient uptake, while I did not consider the effects of enhanced weathering under increases in temperature. The effects of nutrient cation loss are particularly problematic for a site like Hubbard Brook which are characterized by naturally low rates of mineral weathering and soil pools of exchangeable nutrient cations.

Few studies have evaluated the effects of forest harvesting under changing climate conditions. Moreover, those studies have largely focused on the dynamics of carbon. Hence, there is little research to compare with my work. My results were consistent with the study of Wu et al. (2017) who suggested that although climate change can increase production of the forest biomass, land-use change (including town expansion, deforestation and forest conversion) can significantly reduce overall forest biomass production (by 53%-57% until 2050). Their study indicated that land use change can have a greater influence on aboveground forest biomass than climate change and land-use change becomes more prominent as the future simulation period progresses. However, again their work did not consider CO_2 fertilization effects in their predictions, while I did not consider land-use change in my analysis. Creutzburg et al. (2016) found patterns inconsistent with my predictions, suggesting that climate change would decrease carbon sequestration in aboveground biomass and woody debris relative to current climate due to increases in heterotrophic respiration. They indicated that long-term impacts of forest harvesting would be less evident under changing climate than stationary climate. Their analysis simulated a slow increase in soil C accumulation (by 8% until 2100) with harvesting under both climate conditions due to the contribution of decomposition of roots from the harvested trees. The effects of CO_2 fertilization on biomass productivity was not considered in their study.

My study is consistent with previous reports that the implementation of harvesting regimes would impact stream hydrology and water quality over both short- and long- periods (Aherne et al., 2012; Blanco et al., 2005). Simulations show increases in annual stream discharge in the first year after the cut due to a decrease in transpiration under both climate conditions. Over long-term simulations higher stream discharge is also evident associated with intensive tree removal, due to lower plant demand for water with declines in biomass productivity, and particularly under climate change conditions, due to increases in precipitation from changing climate. Cutting trees resulted in elevated export of N and base cations, immediately after harvesting, with higher impacts under climate change mainly due to a greater reduction in the amount of biomass and higher precipitation. However, long-term cumulative leaching of base cations would likely be mitigated under climate change relative to stationary climate, due to the higher nutrient uptake of regrowing vegetation. Nitrogen is predicted to be the element which experiences the greatest relative loss over both short- and long- periods under different harvesting strategies, particularly with changing climate. Previous studies have also showed that overexploitation of biomass leads to greater loss of nutrients particularly nitrogen and phosphorus, compared to the other nutrients (e.g. Ca^{2+}, Mg^{2+}, K^{+}) (Aherne et al., 2012; Blanco et al., 2005).

HBEF streams are highly sensitive to inputs of acidity and have acid neutralizing capacity (ANC) values that are near zero or negative (Likens et al., 1996). My simulations show that forest harvesting can significantly influence stream ANC, with projected reductions of 7% and 20% by 2200 under stationary climate and changing climate, respectively. These estimated reductions in ANC appear to be driven by decreases in soil base saturation, associated with harvesting and elevated leaching from strong acid anions and increases in runoff. This result is in consistent with the findings of Aherne et al. (2012). These patterns suggest that long-term forest sustainability would require ensuring adequate supply of major nutrients over the long-term to maintain forest health and mitigate against stream acidification.

Increased demand for timber production has encouraged forest managers to develop guidelines to satisfy multiple criteria that not only maintain forests as carbon reservoirs but also to consider the ability of forests to sequester carbon (Creutzburg et al., 2016; Seely et al., 2002; Wang et al., 2014). As my results demonstrate, undisturbed mature forests can store significantly greater quantities of carbon compared with those managed for timber production. However there are potential management strategies that can mitigate against this loss of carbon (Creutzburg et al., 2016; Scheller and Mladenoff, 2005; Seely et al., 2002). Much literature reports that the ability of a forest to accumulate carbon declines with increasing age of the forest when ecosystem respiration begins to balance or exceed primary production (Scheller and Mladenoff, 2005; Seely et al., 2002). My results confirm that under both stationary and climate change conditions the rates of carbon sequestration varies with cutting intervals, with a sharp increase from the intervals of 30-60 years, but only a modest increase from 60-90 years (Figure A21).

Management options from my analyses were used to examine the trade-offs between timber production and long-term carbon storage under both stationary and climate change conditions. Total carbon storage at the end of simulation interval (2200) and its relation to cumulative harvested stemwood carbon for the 180-year hypothetical simulation period are depicted using a diagonal line that denotes the 1:1 relationship between the variables (Figure 6.10). Management options falling above the line favor timber production while management scenarios falling below the line favor ecosystem carbon storage and nutrient retention. Simulations show that all management options under climate change enhance both timber production and carbon storage than under stationary climate, but as discussed, with greater potential for a reduction in long-term soil fertility.

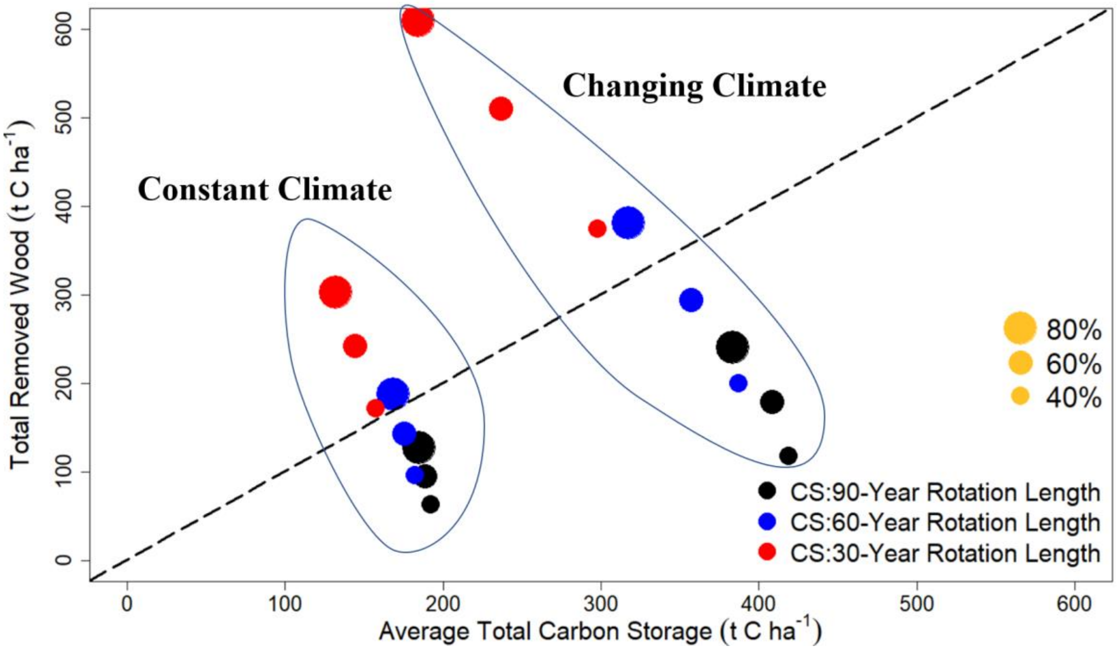


Figure 6.10. Simulation of total stored carbon in the ecosystem at the end of simulations (2200 year) and its relation to cumulative harvested stemwood carbon over the 180-year simulation period for various cutting regimes (from 2020 year until 2200 year) under stationary and changing climate conditions for W5, HBEF. The diagonal line denotes a 1:1 relationship between variables.

6.3. Conclusions and suggestions for further research

Overall, simulations show that all management options under climate change enhance both timber production and carbon storage in comparison to stationary climate, but with greater potential for a reduction in long-term soil fertility. Cumulative stream leaching of NO_3^- under climate change exceeded the values under constant climate for the various cutting regimes. However, greater uptake of Ca^{2+} by regrowing trees in the climate change scenario resulted in decreases in stream Ca^{2+} leaching compared to stationary conditions for different logging practices. Compared with the reference no-cutting scenario, plant nutrient uptake and soil base saturation decreased to different levels by the end of simulations for different logging scenarios, but with greater impact under changing climate. There are sources of uncertainty and limitations in the model results. PnET-BGC does not directly account for the changes in tree species composition following the clear-cut that might significantly influence dynamics of vegetation, hydrology and biogeochemistry. Weathering rates of elements through the simulation period and atmospheric deposition rates for the future years were assumed to be constant, but both may be influenced under changing climate, altering availability of nutrient sources (Aherne et al., 2012; Vadeboncoeur et al., 2014). Moreover, indirect effects of climate change, such as increases in wildfire, insect defoliation and diseases were not considered but are clearly potentially important (Creutzburg et al., 2016; Wei et al., 2003). Few studies have evaluated the long-term effects of CO_2 emissions and its interaction with other global-change drivers (e.g. harvesting) (Pourmokhtarian et al., 2017). Uncertainty in predictions of greenhouse gas emission and climate data in any climate change scenario can affect model simulations. These limiting factors caution against over interpreting my quantitative predictions. In addition, forestry practices are very site dependent. Hence, I suggest caution when extrapolating results from one site to another without proper evaluation of specific impacts. Nevertheless, my work shows how a multi-element soil-layer model can be used as a useful diagnostic tool to gain a better understanding of the complex interactions of ecological processes and their response to multiple ecosystem stressors. The results of different combinations of harvesting and climate scenarios provide important insight into the relative importance of different factors on key ecosystem processes including carbon sequestration, nitrogen cycling, and biomass accumulation.

7. Conclusions

Understanding the dynamics of forest ecosystems, and the factors that influence these dynamics provide a basis for the sustainable use of forest resources and the conservation of their ecosystem services. Ecosystem modeling can be implemented as a diagnostic tool to gain a better understanding of the complex interactions of ecological process and their response to multiple ecosystem stressors. Models of nutrient cycling are of particular importance for evaluation of sustainable forest management practices. The first phase of this dissertation presented the first effort to test and modify the multi-element soil-layer model PnET-BGC against field measurements made for a whole-tree harvest in W5, HBEF in order that the model could be used as a tool to effectively quantify short- and long-term effects of forest cutting. 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 depict short- and long-term changes in hydrology, aboveground forest biomass, and soil solution and stream water chemistry following the cut. The modified model was applied to further evaluate model performance in simulating various cutting practices using other experimentally cut watersheds (the strip-cut W4, the enforced devegetated W2) at the HBEF. The results showed that the modified model was able to depict differences in ecosystem response to different tree cutting experiments. Comparison of biomass accumulation patterns among the cut watersheds at the HBEF indicated some initial differences in their biomass accumulation rates. However, simulations of biomass accumulation for all cut watersheds were consistent with a long-term growth trajectory of a second- growth watershed (W6). The overall impact of these different cutting experiments at the HBEF on total aboveground biomass appears to be minimal over the long-term (2100). The modified model was able to depict changes in hydrology, stream water and soil chemistry and element budgets resulting from different tree cutting experiments. The greatest loss of major nutrients immediately after disturbance occurred in W2 due to the intensive treatment (clear-felling with cut trees were left in place and herbicide application) and followed by the whole-tree harvest of W5. The strip-cut approach used at W4 mitigates nutrient losses compared to the other experiments. These results highlight the important role of plant nutrient uptake in regulating the recovery of the forest ecosystems from biomass disturbance. The model captured the rapid attenuation of nutrient loss to pre-cut levels with regrowth of vegetation for all treated watersheds. Simulations indicate that for all cut watersheds at HBEF NO_3^- concentrations in streams fell below the levels in undisturbed forest (W6) within 10 to 15 years during the initial regrowth, associated to decrease in litter inputs and N mineralization during the earlier years of regrowth. Long-term simulations of soil percent base saturation showed greater depletion and slower recovery for W2 than W4 and W5 following the treatment possibly due to intensive experimental method used and elevated loss of NO_3^- which enhanced base cation leaching Among different cutting approaches at the HBEF, it is obvious that the treatment applied in W2 would not be used as a forest management option because the cut biomass was not extracted from the site. Moreover, the extended herbicide treatment delayed regrowth, resulting in substantial loss of essential nutrients via stream leaching. Compared with the whole-tree harvest, the progressive strip cutting attenuated effects on hydrologic and nutrient cycles. However, there is concern that both cutting practices resulted in a less desirable mix of commercial species in the new regrowing stand (Martin et al., 1989).

A sensitivity analyis was conducted in the first phase using W5 simulations, showing that model calculations are generally more responsive to vegetation parameters than abiotic parameters and that the sensitivity of the model under more mature forest (pre-cut) conditions is greater than for simulations of the aggrading forest (post-cut conditions). The sensitivity analyis results of W5 were also helpful in the calibration of the model to other watersheds and demonstrate the necessity of field measurements to constrain important parameter values.

Interactive simulations of forest harvest practices and future climate change and carbon dioxide fertilization of W5 showed that the sensitivity of forest ecosystem pools to logging strategies was greater under climate change compared to constant climate conditions. These effects are accentuated with shortening the length of the cutting intervals and increasing forest harvesting intensity. Simulations of both constant and varying climate conditions considered showed greater sensitivity to varying the length of the cutting period than altering cutting intensities. My simulations suggest that living tree biomass and woody debris could be more influenced than soil carbon following tree harvesting under constant current climate. In contrast, tree logging under climate change appears to have more negative impacts associated with loss of soil organic matter pools and consequently on-site fertility over the long-term. My results suggest that under both stationary and climate change conditions the rate of carbon sequestration in forest biomass depends upon the cutting interval. For example, simulations suggest a sharp increase in forest biomass accumulation from the cutting intervals of 30-60 years, but only a modest increase when the cutting interval is increased from 60-90 years. Over the long-term, greater depletion of soil exchangeable base cations was evident from harvesting under changing climate due to increases in soil mineralization, plant uptake and enhanced biomass accumulation. Moreover, nitrogen is predicted to be the element which experiences the greatest relative loss over both short- and long- periods under different harvesting strategies, particularly with changing climate. These patterns suggest that the long-term sustainability of forest resources will require management practices that ensure an adequate supply of major nutrients to maintain forest health and mitigate against soil and stream acidification. Simulations show that all management options under climate change enhance both timber production and carbon storage in comparison to stationary climate, but result in greater potential for a reduction in long-term soil fertility due to loss of soil organic matter associated with increases in temperature. The results of different combinations of harvesting and climate scenarios provide important insight into the relative importance of different factors that influence key ecosystem processes, including carbon sequestration, nitrogen cycling, soil base saturation and biomass accumulation.

8. Future research recommendations

Application of the multi-element soil-layer model allowed for a more comprehensive quantification of short- and long-term responses to forest cutting and interactions with climate change and to gain some insights into biogeochemical processes, but led to some questions which should be addressed in future efforts. The following research suggestions could be useful in improving the overall assessments of long-term forest sustainability under future cutting practices:

1. Further field measurements are needed shortly after forest cutting and over the long-term after the treatment to minimize model uncertainty with model parametrization and calibration. For example, measurements of mineralization/nitrification rates for the period immediately after the cut can help to develop model algorithms. This information can result in better simulations of nutrients budget and streamwater chemistry.

2. Spatial modifications to the model to simulate a large watershed through nested small sub-catchments. To accommodate this model configuration various inputs and catchment characteristics for each individual sub-catchment would be needed. For example, this approach would enable user to mimic better cutting practices such as a strip-cut and implementing a buffer strip, and also to select dominant vegetation type of each sub-catchment. This approach would also allow for examination of spatial variation in watershed response to harvesting disturbance.

3. Model modifications to directly account for the changes in tree species composition following the clear-cut that might significantly influence dynamics of vegetation, hydrology and biogeochemistry.

4. Consideration of future changes in acid deposition and weathering rates of elements through the simulation that both may be influenced under changing climate, altering the availability of nutrient sources.

5. Considering indirect effects of climate change through the simulation, such as increases in wildfire, insect defoliation and diseases.

6. Evaluation of long-term impacts of forest logging under a range of potential future climate projections with/without CO_2 emission to account for the effects of variability of climate conditions that might influence hydrology, availability of nutrients and plant growth.

7. Changing the single soil organic matter pool used in PnET-BGC with a simple turn-over rate to multiple soil organic matter pools with different turn-over rates to allow for slower nutrient mineralization from the soil organic pool. This approach would be particularly helpful in simulating the transition of all slash left on the site and dead root biomass from the harvest are added to the soil organic pool. Such an algorithm would help decrease leaching of elements into soil solutions and stream water immediately after cutting.

Appendix

Table 0.1. Input parameters required to run PnET-BGC for Watershed 5 in Hubbard Brook.

Parametrization						
Soil Variables						
Layer	*Name*	*Value*	*Layer*	*Name*	*Value*	*Definition (Unit)*
Upper	Soil Depth	0.6	**Lower**	Soil Depth	0.3	Depth of Soil (m)
	Soil Mass	350		lSoilMass	170	Mass of Soil (kg/m^2)
	CEC	0.0821		lCEC	0.021	Cation Exchange Capacity (mol/kg)
	SO_4Ad	0.01		lSO_4Ad	0.025	SO_4^{2-} Adsorption Capacity (mol/kg)
	SiteDOC	0.02		-	-	Site Density or Moles of Organic Anion Sites per Moles of Organic C (µmol/µmol C)
	DocFrac	0.175		-	-	Fraction of Dissolved Organic carbon
Weathering Rates						
Upper	Ca^{2+}	0.035	**Lower**		0.02	$gr/m^2/mo$
	Mg^{2+}	0.009			0.007	
	Na^+	0.021			0.02	
	Al^{3+}	0.053			0.04	
	K^+	0.005			0.004	
	$S\text{-}SO_4^{2-}$	0.0005			0.02	
	$Si\text{-}SiO_2$	0.1			0.22	
Hydrological Variables						
Upper	USATC	34.3	**Lower**	LSATC	9.9	Saturation Capacity (cm/mo)
	UWHC	14.2		LWHC	4.2	Water Holding Capacity (cm/mo)
	Ksat	20		lksat	4.5	Conductivity at Saturation (cm/mo)
	Kwhc	0		lKwhc	1.2	Conductivity at Water Holding Capacity (cm/mo)
				baseflow	0.11	Minimum Subsurface Flow (cm/mo)
	MaxDownFlow	17.5				Vertical Conductivity (cm/mo)
Canopy Variables						
	WoodTurnover				0.22	Fractional Mortality of Live Wood per Year (yr^{-1})
	RootAllocB				3	Root Allocation Coefficient
	RootAllocD				1	Root Allocation Coefficient
	FLPctN				0.96	Min N Concentration in Foliar Litter, %
	Kho				0.07	Decomposition Constant for SOM, yr^{-1}
	Kho[a]				0.043	Decomposition Constant for SOM, yr^{-1}
	ContrN[b]				3.5	New constant variable limiting nitrification

[a] Used for 2-4 years following the clear-cut.

[b] Used for 3-7 years following the clear-cut.

Table A.2. Input parameters required to run PnET-BGC for Watershed 4 in Hubbard Brook.

Parametrization						
Soil Variables						
Layer	*Name*	*Value*	*Layer*	*Name*	*Value*	*Definition (Unit)*
Upper	Soil Depth	0.6	**Lower**	Soil Depth	0.3	Depth of Soil (m)
	Soil Mass	350		lSoilMass	170	Mass of Soil (kg/m^2)
	CEC	0.0821		lCEC	0.021	Cation Exchange Capacity (mol/kg)
	SO_4Ad	0.01		lSO_4Ad	0.025	SO_4^{2-} Adsorption Capacity (mol/kg)
	SiteDOC	0.02		-	-	Site Density or Moles of Organic Anion Sites per Moles of Organic C (µmol/µmol C)
	DocFrac	0.175		-	-	Fraction of Dissolved Organic carbon
Weathering Rates						
Upper	Ca^{2+}	0.038	**Lower**		0.035	gr/m^2/mo
	Mg^{2+}	0.008			0.008	
	Na^+	0.028			0.03	
	Al^{3+}	0.053			0.04	
	K^+	0.005			0.004	
	S-SO_4^{2-}	0.0001			0.017	
	Si-SiO_2	0.1			0.22	
Hydrological Variables						
Upper	USATC	34.3	**Lower**	LSATC	9.9	Saturation Capacity (cm/mo)
	UWHC	14.2		LWHC	4.2	Water Holding Capacity (cm/mo)
	Ksat	20		lksat	4.5	Conductivity at Saturation (cm/mo)
	Kwhc	0		lKwhc	1.2	Conductivity at Water Holoding Capacity (cm/mo)
				baseflow	0.11	Minimum Subsurface Flow (cm/mo)
	MaxDownFlow	17.5				Vertical Conductivity (cm/mo)
Canopy Variables						
	WoodTurnover				0.22	Fractional Mortality of Live Wood per Year (yr^{-1})
	RootAllocB				3	Root Allocation Coefficient
	RootAllocD				1	Root Allocation Coefficient
	FLPctN				0.915	Min N Concentration in Foliar Litter, %
	Kho				0.07	Decomposition Constant for SOM, yr^{-1}
	Kho[a]				0.043	Decomposition Constant for SOM, yr^{-1}
	ContrN[b]				3.5	New constant variable limiting nitrification

[a] Used for 2-7 years following the first strip cut.

[b] Used for 3-7 years following the first strip cut.

Table A.3. Input parameters required to run PnET-BGC for Watershed 2 in Hubbard Brook.

Parametrization						
Soil Variables						
Layer	*Name*	*Value*	*Layer*	*Name*	*Value*	*Definition (Unit)*
Upper	Soil Depth	0.6	**Lower**	Soil Depth	0.3	Depth of Soil (m)
	Soil Mass	350		lSoilMass	170	Mass of Soil (kg/m^2)
	CEC	0.0821		lCEC	0.021	Cation Exchange Capacity (mol/kg)
	SO_4Ad	0.01		lSO_4Ad	0.025	SO_4^{2-} Adsorption Capacity (mol/kg)
	SiteDOC	0.02		-	-	Site Density or Moles of Organic Anion Sites per Moles of Organic C (µmol/µmol C)
	DocFrac	0.175		-	-	Fraction of Dissolved Organic carbon
Weathering Rates						
Upper	Ca^{2+}	0.035	**Lower**		0.02	gr/m^2/mo
	Mg^{2+}	0.009			0.007	
	Na^+	0.021			0.02	
	Al^{3+}	0.053			0.04	
	K^+	0.005			0.004	
	S-SO_4^{2-}	0.0005			0.02	
	Si-SiO_2	0.1			0.22	
Hydrological Variables						
Upper	USATC	34.3	**Lower**	LSATC	9.9	Saturation Capacity (cm/mo)
	UWHC	14.2		LWHC	4.2	Water Holding Capacity (cm/mo)
	Ksat	20		lksat	4.5	Conductivity at Saturation (cm/mo)
	Kwhc	0		lKwhc	1.2	Conductivity at Water Holoding Capacity (cm/mo)
				baseflow	0.11	Minimum Subsurface Flow (cm/mo)
	MaxDownFlow	17.5				Vertical Conductivity (cm/mo)
Canopy Variables						
	WoodTurnover				0.22	Fractional Mortality of Live Wood per Year (yr^{-1})
	RootAllocB				3	Root Allocation Coefficient
	RootAllocD				1	Root Allocation Coefficient
	FLPctN				0.96	Min N Concentration in Foliar Litter, %
	Kho				0.07	Decomposition Constant for SOM, yr^{-1}
	Kho[a]				0.043	Decomposition Constant for SOM, yr^{-1}
	ContrN[b]				3.5	New constant variable limiting nitrification

[a] Used for 2-7 years following the clear-cut.

[b] Used for 3-7 years following the last herbicide application.

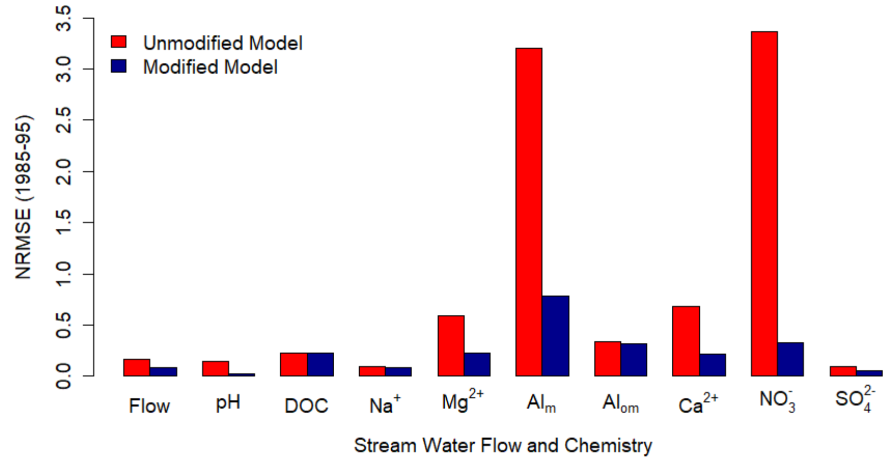


Figure A1. Comparison of model performance before and after algorithm modification in simulating streamwater chemsitry after the clear-cut based on values of NRMSE (1985-95 years).

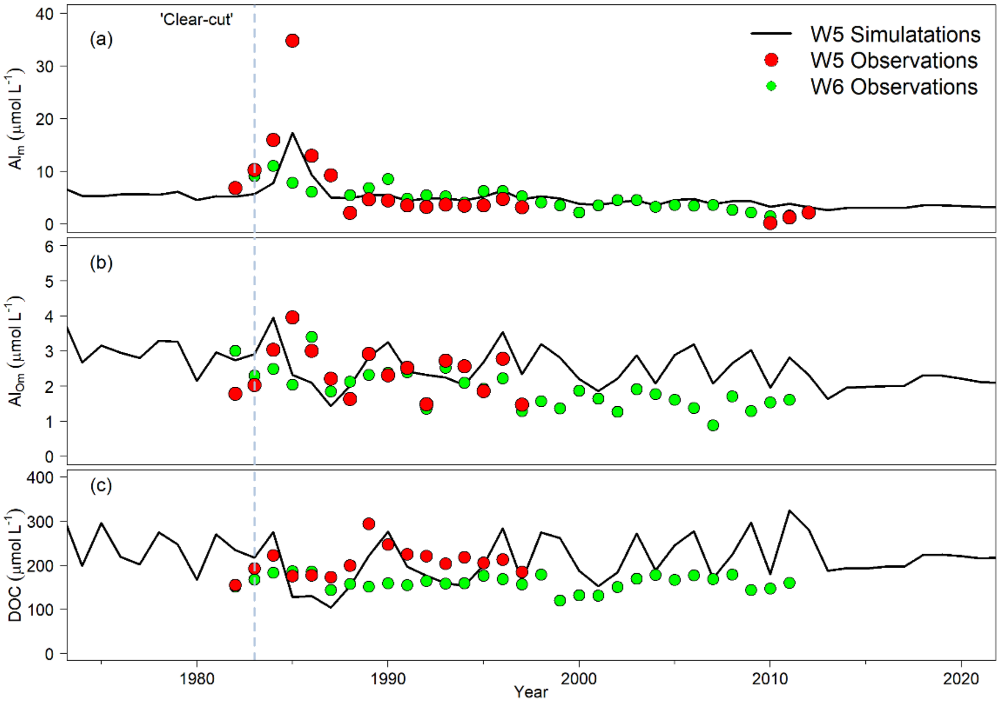


Figure. A2. Comparison between simulations of total monomeric aluminum, Al_m (a), organic monomeric aluminum, Al_{om} (b) and dissolved organic carbon concentrations, DOC (c) in stream water chemistry by PnET-BGC with observations for W5, HBEF. The results are also compared with stream water chemistry for reference watershed (W6).

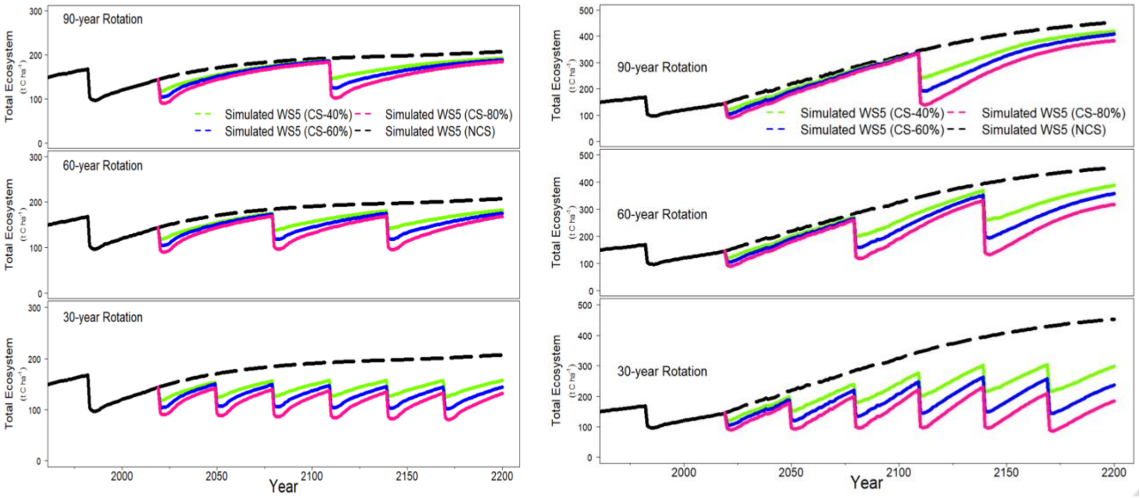


Figure A3. Simulation of temporal dynamics of total ecosystem stored carbon (e.g. sum of aboveground biomass, woody debris and soil) across ten forest management scenarios under stationary climate (left) and future climate change (right) conditions.

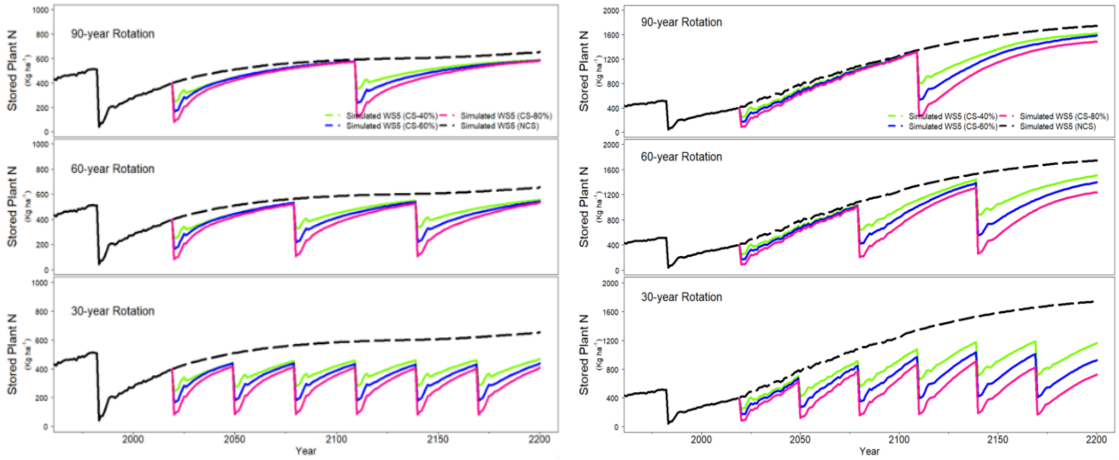


Figure A4. Simulation of temporal dynamic of N stored in plant across ten forest management scenarios under stationary climate (left) and future climate change (right) conditions.

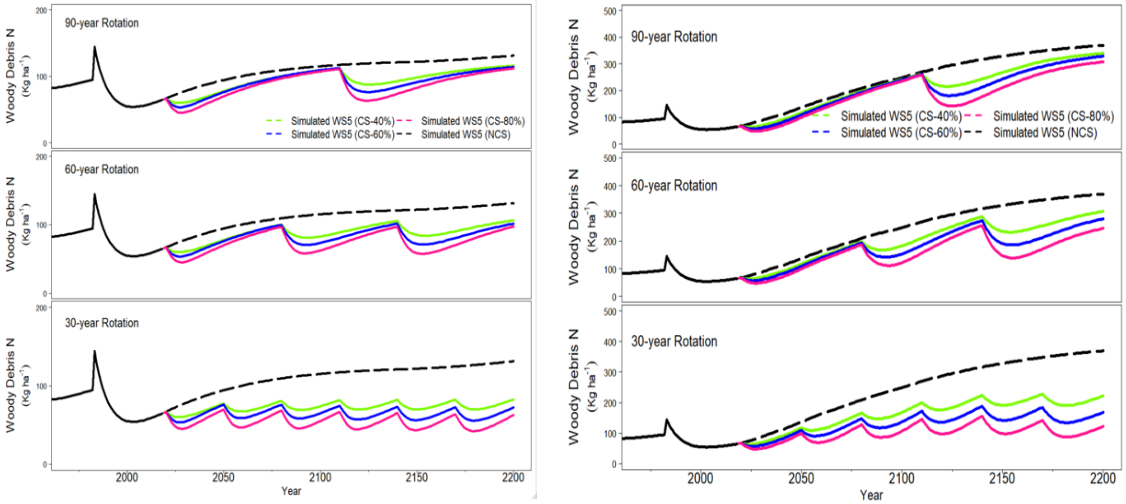


Figure A5. Simulation of temporal dynamic of N stored in woody debris pool across ten forest management scenarios under stationary climate (left) and future climate change (right) conditions.

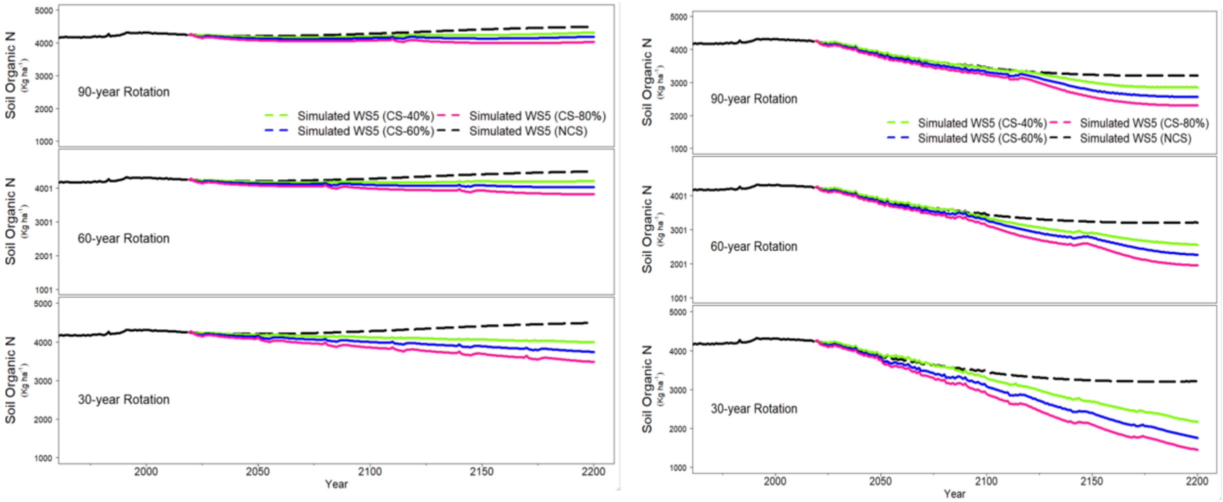


Figure A6. Simulation of temporal dynamic of soil N matter across ten forest management scenarios under stationary climate (left) and future climate change (right) conditions.

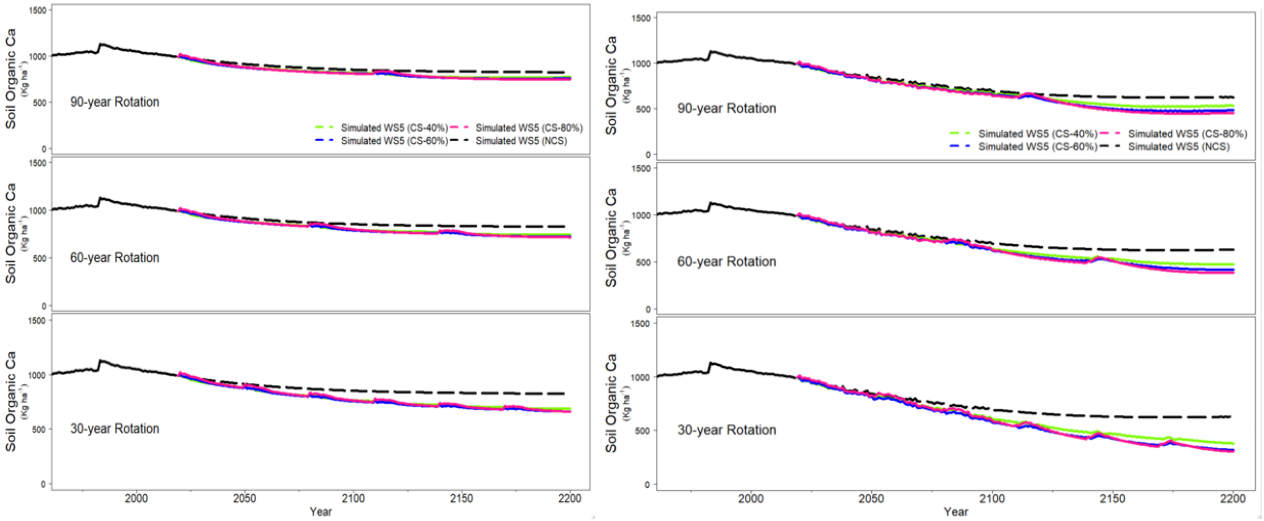


Figure A7. Simulation of temporal dynamic of soil Ca across ten forest management scenarios under stationary climate (left) and future climate change (right) conditions.

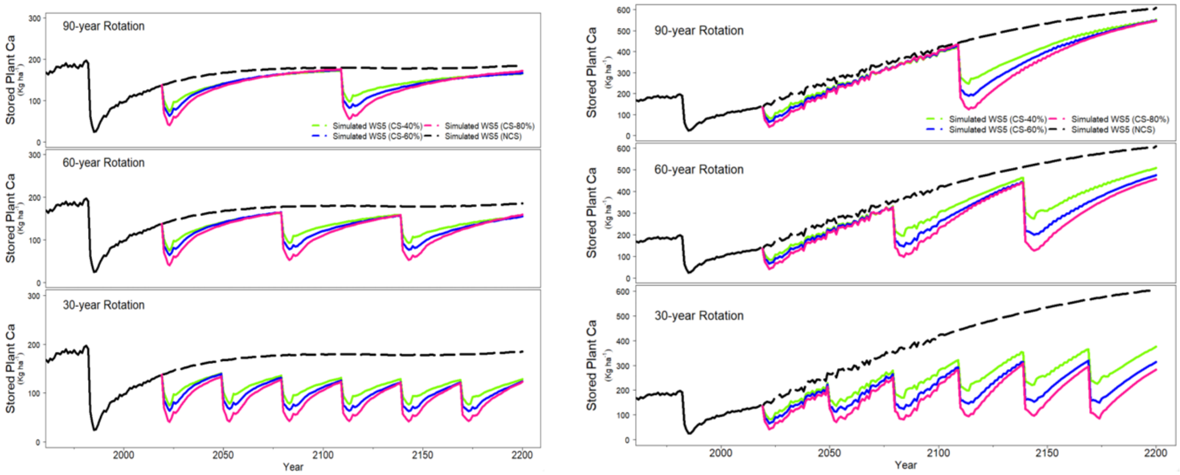


Figure A8. Simulation of temporal dynamic of stored Ca in plant across ten forest management scenarios under stationary climate (left) and future climate change (right) conditions.

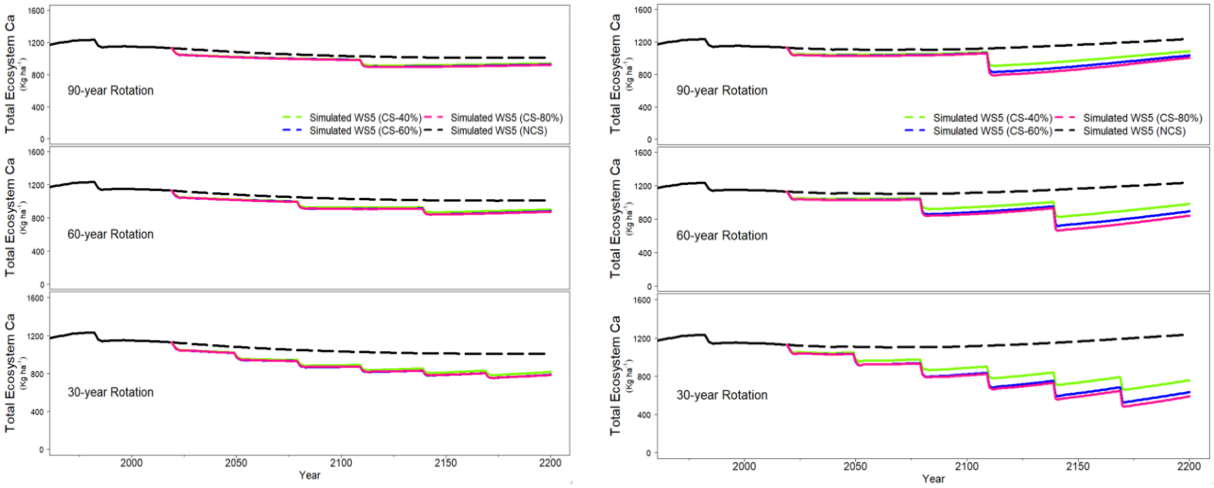


Figure A9. Simulation of temporal dynamic of total ecosystem stored Ca^{2+} across ten forest management scenarios under stationary climate (left) and future climate change (right) conditions.

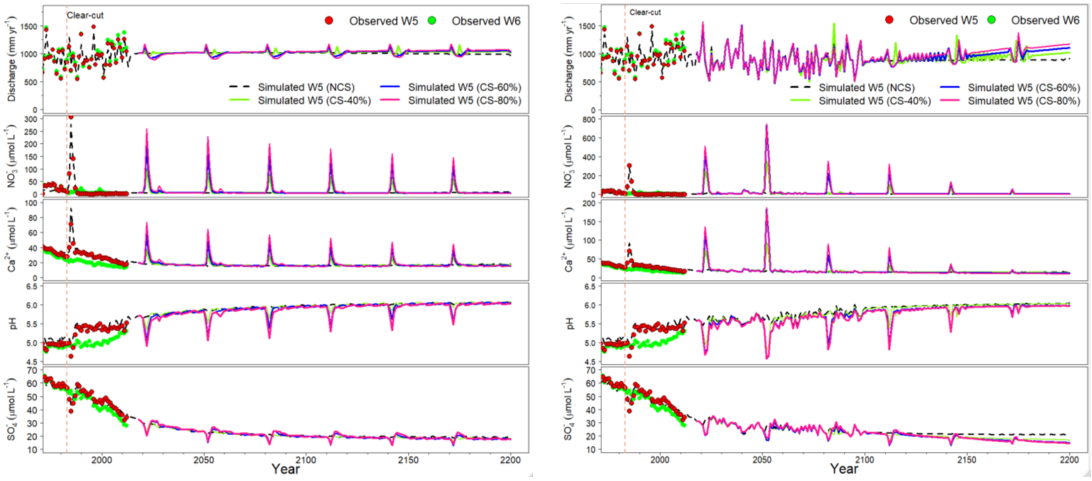


Figure A10. Simulations of stream water discharge and chemistry for 30-year cutting period with three intensity levels (40%, 60% and 80%). Simulations are also compared with measurements of W5 (experimental clear-cut watershed in 1883) and W6 (reference watershed).

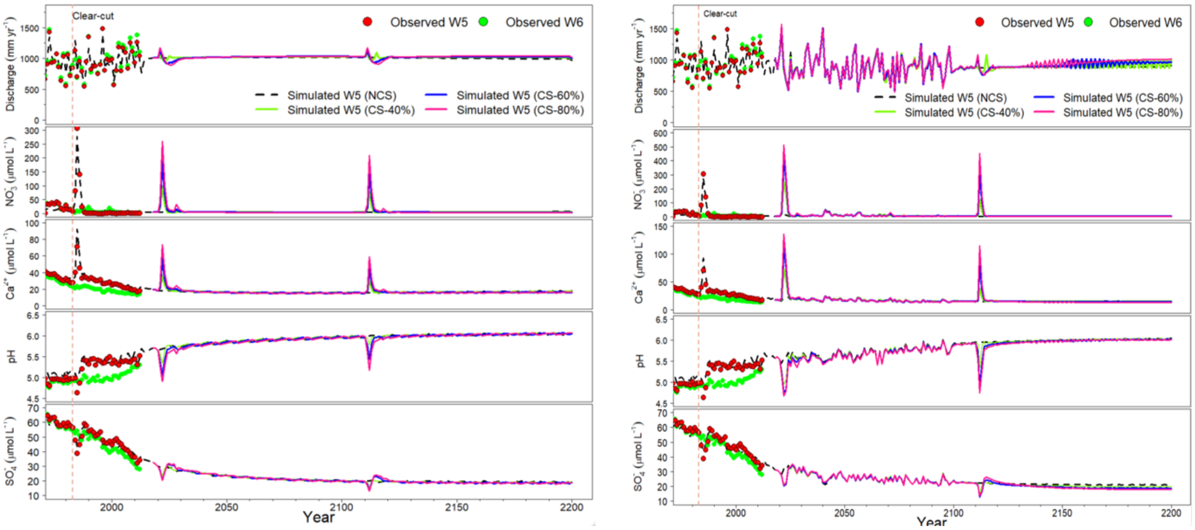


Figure A11. Simulation of stream water discharge and chemistry for 90-year cutting period with three intensity levels (40%, 60% and 80%). Simulations are also compared with measurements of W5 (experimental clear-cut watershed in 1883) and W6 (reference watershed).

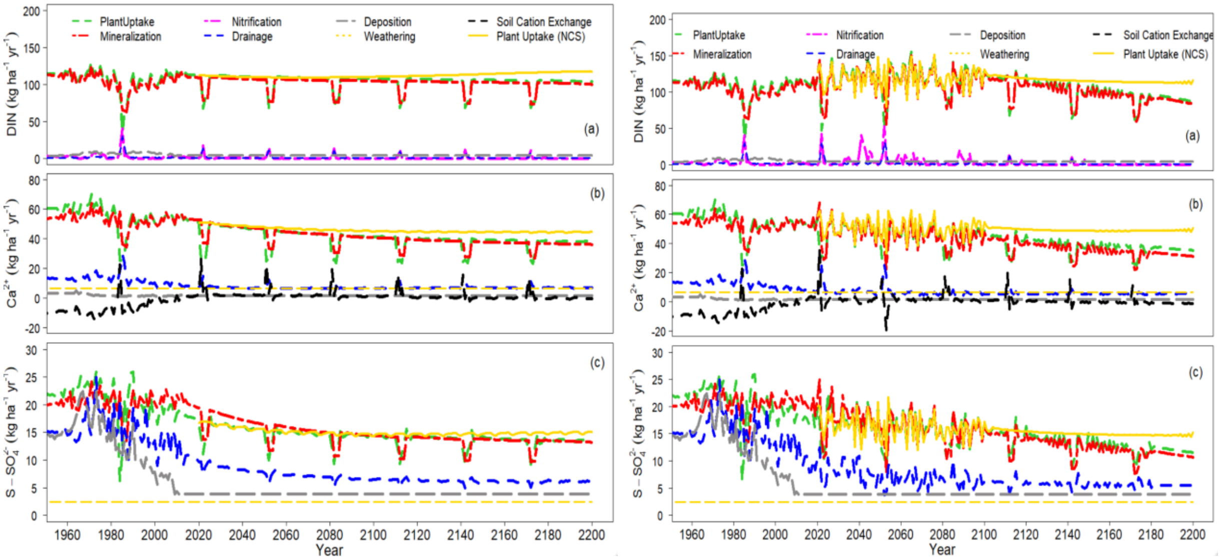


Figure A12. Simulations of annual nutrients budgets for W5, HBEF, including total dissolved inorganic nitrogen (DIN) (a), Ca^{2+} (b) and S-SO$^{2-}{}_{4}$ (c) for a 30-year cutting rotation with 40% cutting intensity under stationary climate (left) and future climate change (right).

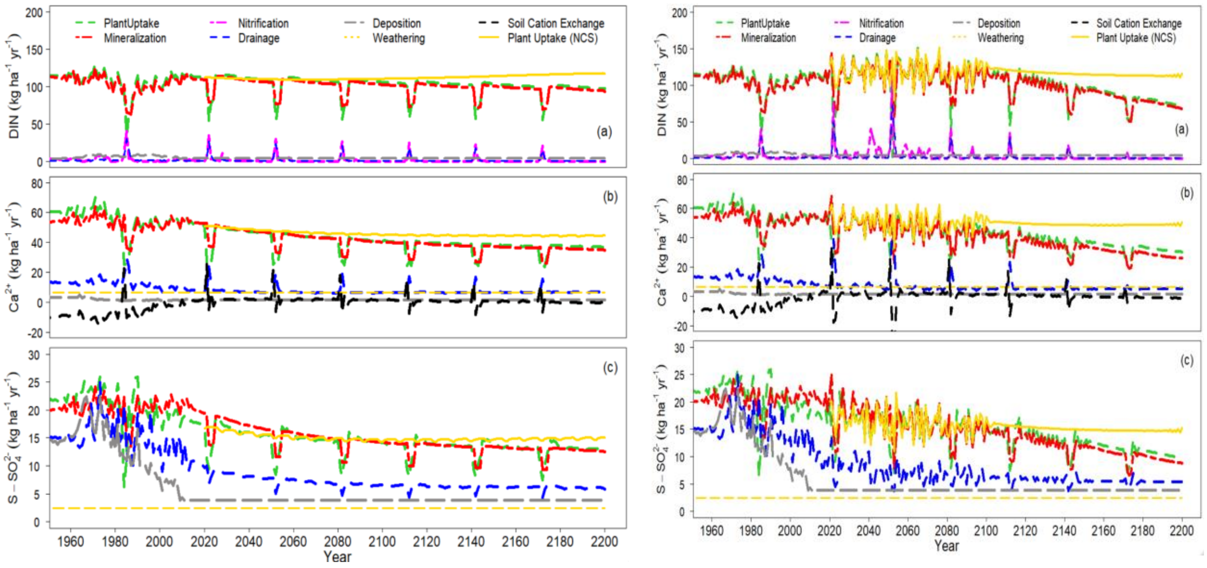


Figure A13. Simulations of annual nutrients budgets for W5, HBEF, including total dissolved inorganic nitrogen (DIN) (a), Ca^{2+} (b) and S-SO$^{2-}_4$ (c) for a 30-year cutting rotation with 60% cutting intensity under stationary climate (left) and future climate change (right).

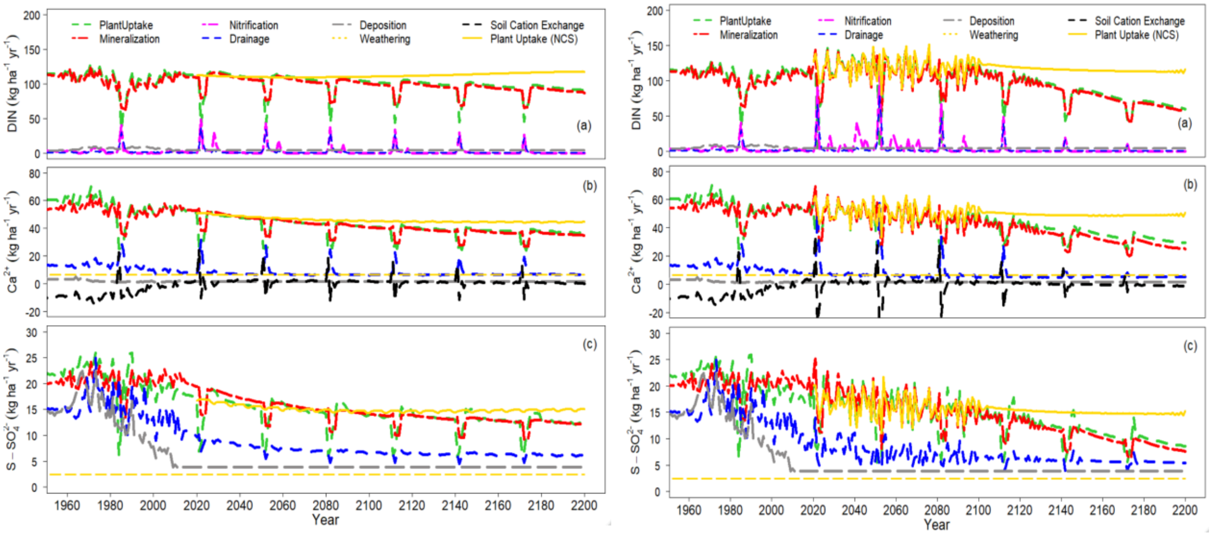


Figure A14. Simulations of annual nutrients budgets for W5, HBEF, including total dissolved inorganic nitrogen (DIN) (a), Ca^{2+} (b) and S-SO$^{2-}_4$ (c) for a 30-year cutting rotation with 80% cutting intensity under stationary climate (left) and future climate change (right).

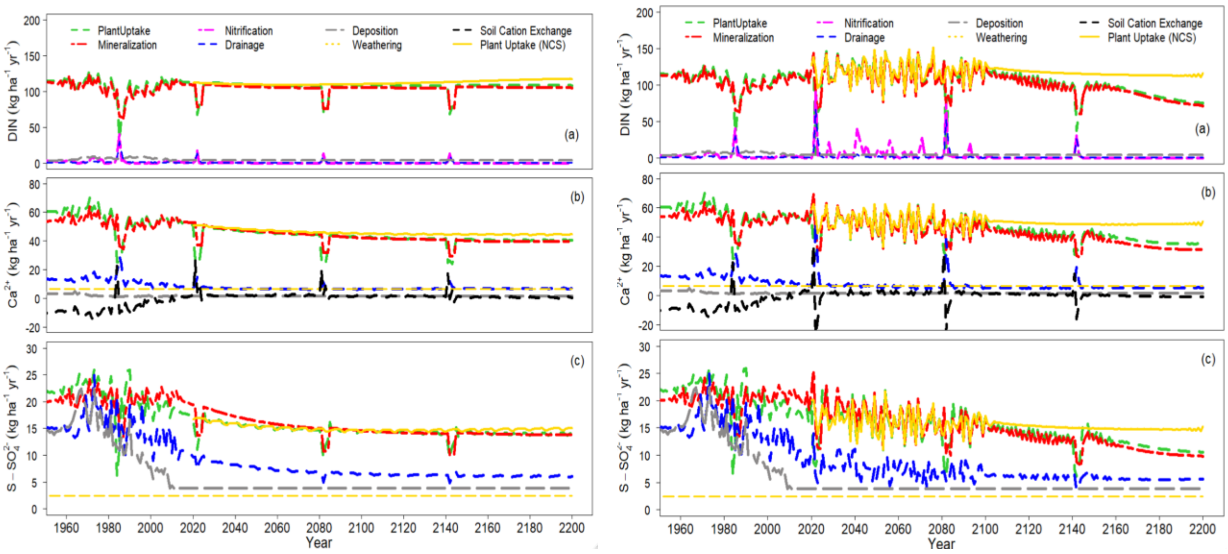


Figure A15. Simulations of annual nutrients budgets for W5, HBEF, including total dissolved inorganic nitrogen (DIN) (a), Ca^{2+} (b) and $S\text{-}SO^{2-}_4$ (c) for a 60-year cutting rotation with 40% cutting intensity under stationary climate (left) and future climate change (right).

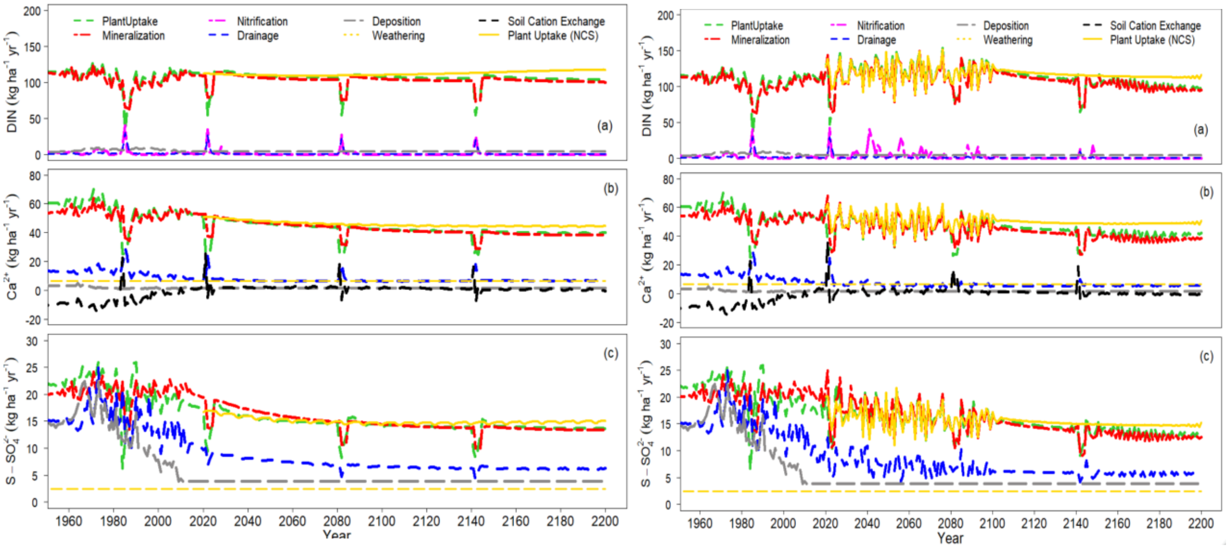


Figure A16. Simulations of annual nutrients budgets for W5, HBEF, including total dissolved inorganic nitrogen (DIN) (a), Ca^{2+} (b) and $S\text{-}SO^{2-}_4$ (c) for a 60-year cutting rotation with 60% cutting intensity under stationary climate (left) and future climate change (right).

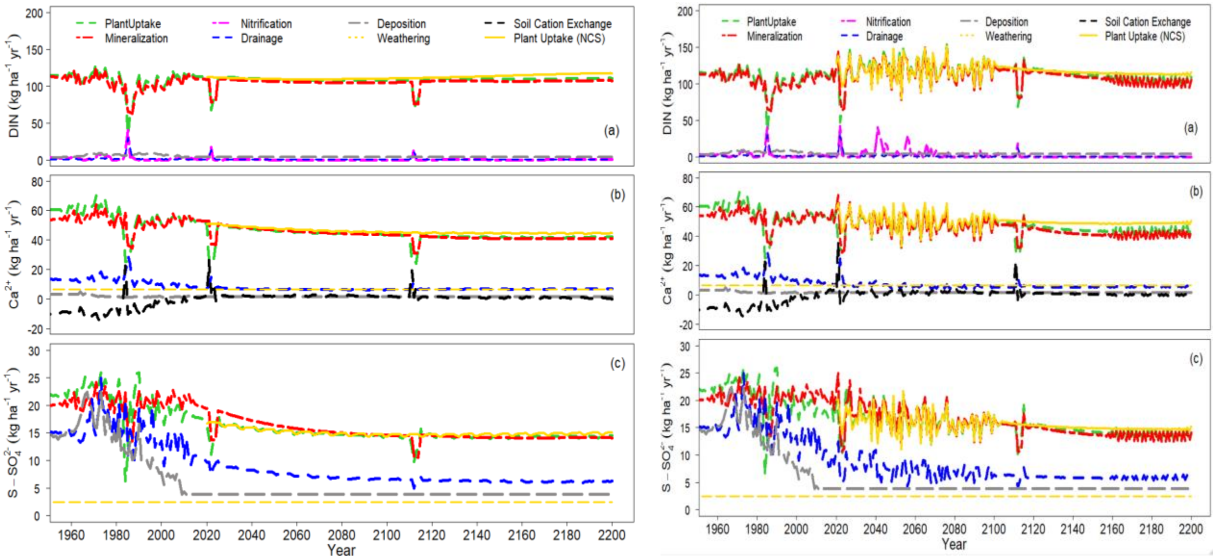


Figure A17. Simulations of annual nutrients budgets for W5, HBEF, including total dissolved inorganic nitrogen (DIN) (a), Ca^{2+} (b) and S-SO$^{2-}{}_{4}$ (c) for a 90-year cutting rotation with 40% cutting intensity under stationary climate (left) and future climate change (right).

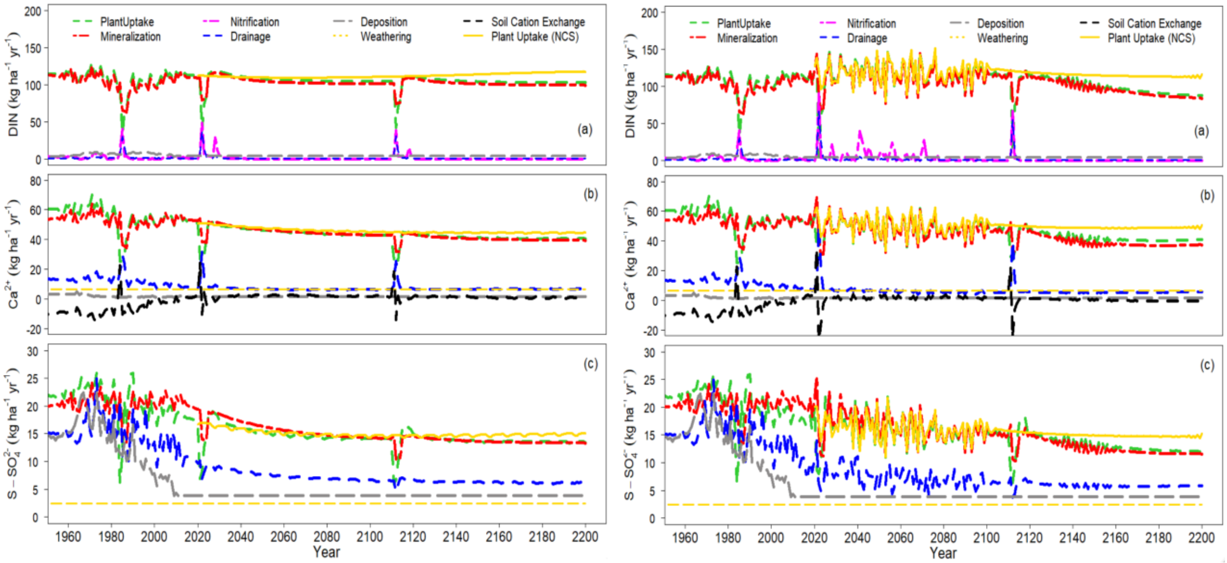


Figure A18. Simulations of annual nutrients budgets for W5, HBEF, including total dissolved inorganic nitrogen (DIN) (a), Ca^{2+} (b) and S-SO$^{2-}{}_{4}$ (c) for a 90-year cutting rotation with 60% cutting intensity under stationary climate (left) and future climate change (right).

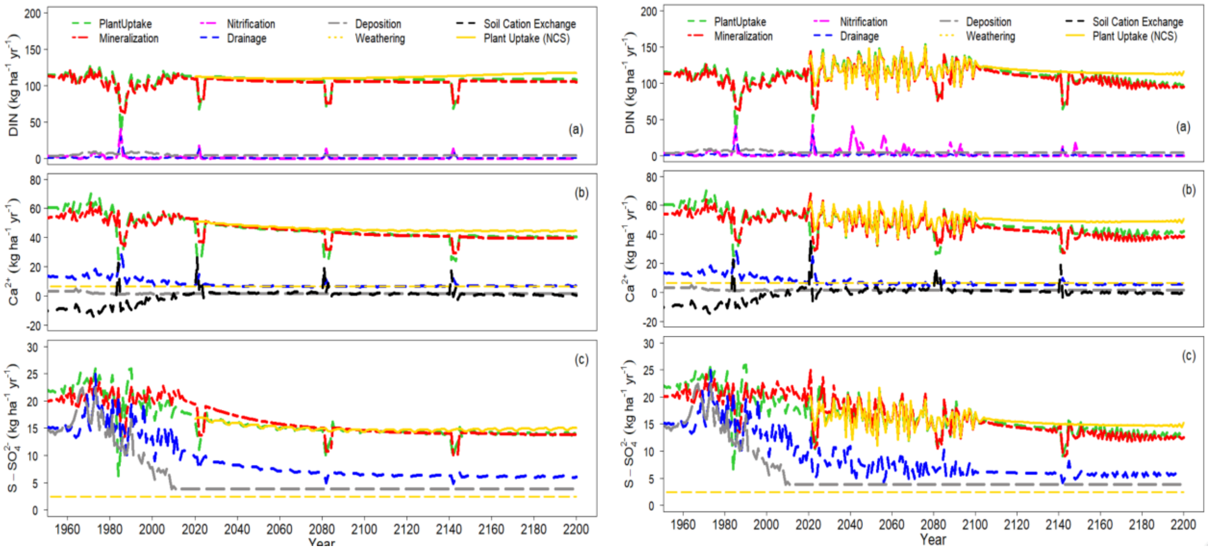


Figure A19. Simulations of annual nutrients budgets for W5, HBEF, including total dissolved inorganic nitrogen (DIN) (a), Ca^{2+} (b) and S-SO$^{2-}{}_4$ (c) for a 90-year cutting rotation with 80% cutting intensity under stationary climate (left) and future climate change (right).

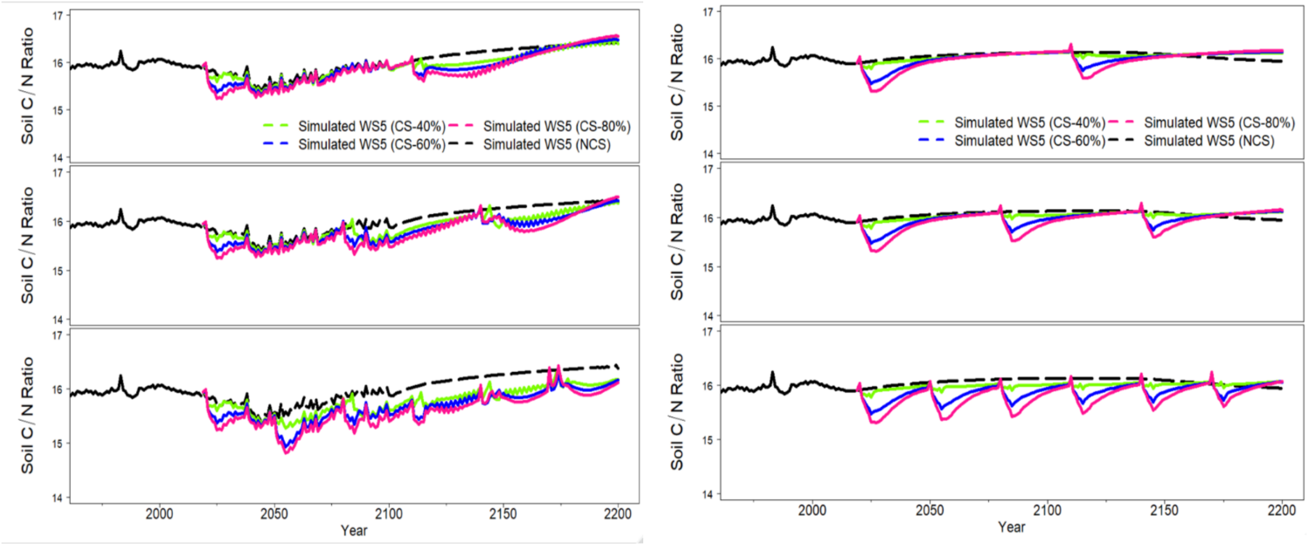


Figure A20. Simulated temporal pattern of soil carbon to nitrogen ratio across ten forest management scenarios under stationary climate (left) and future climate change (right) conditions.

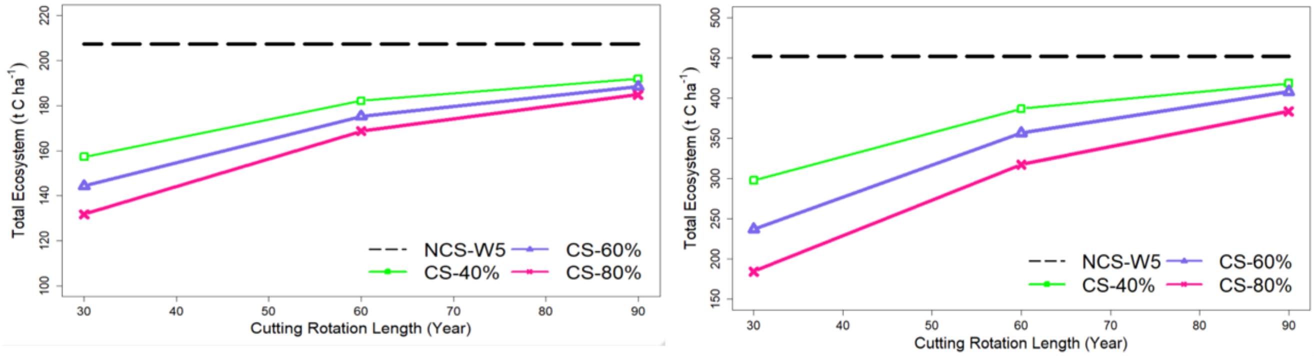


Figure A21. Comparison of effect of various rotation lengths on simulated sequestration of total ecosystem stored carbon across ten forest management scenarios under stationary climate (left) and future climate change (right) conditions.

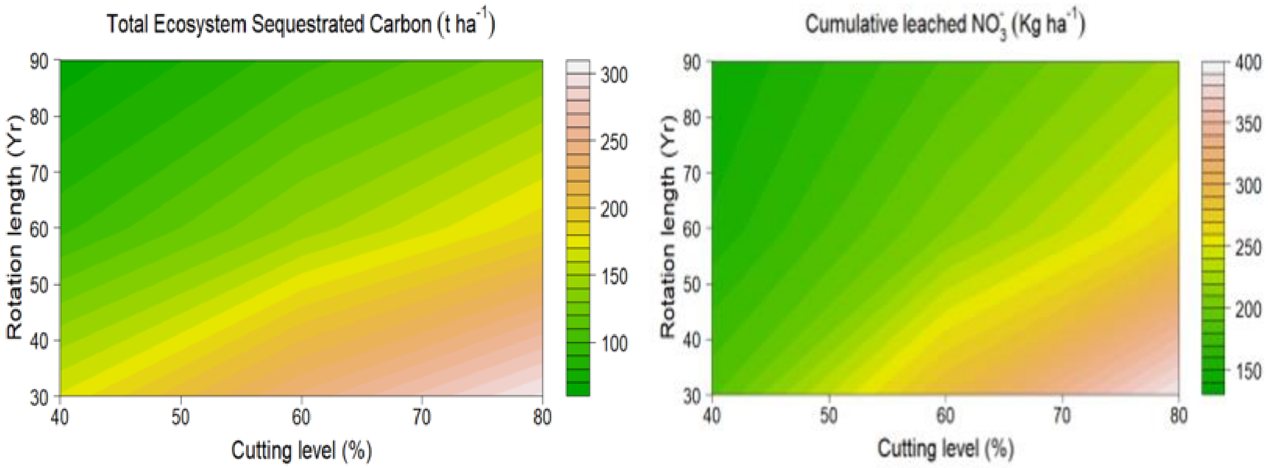


Figure A22. Simulated total ecosystem sequestered carbon and cumulative stream leaching of $NO^-{}_3$ in response to various logging rotation length and intensity levels.

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- **Valipour, M.,** Driscoll, C.T, Johnson, C., Battles, J.J, Campbell J.L, Fahey T., "Interactive simulation of forest harvesting and climate change scenarios to evaluate long-term forest sustainability using a biogeochemical model, PnET-BGC”, In progress to submit Global Change Biology Journal.

- **Valipour, M.,** Driscoll, C.T, Johnson, C., Battles, J.J, Campbell J.L, Fahey T., “Comparison of short- and long-term impacts of various forest logging approaches on biomass accumulation, hydrology and biogeochemistry in a northern forest using PnET-BGC model”, In progress to submit Environmental Science and Technology Journal.
- **Valipour, M.,** Driscoll, C.T, Johnson, C., Battles, J.J, Campbell J.L, Fahey T., 2018. 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. Science of the Total Environment, 645, 244 –256. https://doi.org/10.1016/j.scitotenv.2018.07.066
- **Valipour, M.,** Mirbagheri, S.A., (Oct. 2010) “Quantitative and qualitative evaluation of wastewater in the rural areas of Latian Watershed and zonation of pollutants with GIS” Presented and Published on 4th Conference and Exhibition on Environmental Engineering, Tehran University, Iran.

Selected Presentations and Conferences

- **Valipour, M.,** Driscoll, C.T, "Interactive simulation of forest harvesting and climate change scenarios to evaluate long-term forest sustainability using a biogeochemical model, PnET-BGC", Oral Presentation, American Geophysical Union (AGU) Conference, December 10- 14, 2018, Washington, D.C., USAA.
- **Valipour, M.,** Driscoll, C.T, "Modeling the integrated effects of forest harvesting and climate change scenarios on dynamics of vegetation, hydrology and nutrients using PnET-BGC model", Poster Presentation, Long Term Ecological Research (LTER) All Scientists’ Meeting, October 1-4, Pacific Grove, CA, USA.
- **Valipour, M.,** Driscoll, C.T, "Disturbances and the sustainability of a northern hardwood forest: Evaluation through modeling biomass and cycle of nutrients under different intensive timber harvesting regimes coupled with future climate change projections", Oral Presentation, Hubbard Brook Annual Cooperators Meeting, July 12- 13, 2018, Hubbard Brook Experimental Forest, New Hampshire, USA.

- **Valipour, M.,** Driscoll, C.T, Johnson, C., "Evaluation of forest management practices through application of a biogeochemical model, PnET-BGC", Poster Presentation, American Geophysical Union (AGU) Conference, December 11- 15, 2017, New Orleans, Louisiana, USA.
- **Valipour, M.,** Driscoll, C.T, Johnson, C., "Predicting the dynamics of biomass and nutrients under different harvesting techniques in a northern hardwood forest", Oral Presentation, Ecological Society of America (ESA) Conference, August 6-11, 2017, Portland, Oregon, USA.
- **Valipour, M.,** Driscoll, C.T., "Comparison of impacts of various logging regimes on biomass accumulation, hydrology and biogeochemistry in the northeastern US through modified ecosystem model, PnET-BGC", Oral Presentation, Hubbard Brook Annual Cooperators Meeting, July 12- 13, 2017, Hubbard Brook Experimental Forest, New Hampshire, USA.
- **Valipour, M.,** Driscoll, C.T., "Application of an integrated biogeochemical model to different timber-harvesting", ECS Research Day, Poster & Oral Presentations, April 25, 2017, Department of Engineering and Computer Science, Syracuse University.
- **Valipour, M.,** Driscoll, C.T., Johnson, C., "Modeling short-term and long-term effects of an experimental whole-tree harvest on forest productivity, soil and stream water chemistry", Poster Presentation, Northeast Ecosystem Research Cooperative Conference, March 28-29, 2017, Saratoga Springs, New York, USA.
- **Valipour, M.,** Driscoll, C.T., "Comparison of different harvesting treatments using a Multi-soil layer biogeochemical simulator", Oral Presentation, Environmental Group Seminar, February 20, 2017, L.C. Smith College of Engineering, Syracuse University, Syracuse, NY, USA.
- **Valipour, M.,** Driscoll, C.T., "Modeling of dynamics of vegetation, hydrology and nutrients following an intensive treatment of a northern hardwood forest", Oral Presentation, Hubbard Brook Annual Cooperators Meeting, July 13 - 14, 2016, Hubbard Brook Experimental Forest, New Hampshire, USA.
- **Valipour, M.,** Driscoll, C.T. "Simulation of the response of soil and streamwater Chemistry to the effects of whole –tree clear-cutting using PnET-BGC", Nunan Poster Competition, Department of Engineering and Computer Science, March 2016, Syracuse University, Syracuse, NY, USA.

- **Valipour, M.,** Driscoll, C.T, "Modeling soil and streamwater chemistry to the effects of intensive timber harvesting using a hydro-chemical model", Oral Presentation, Hubbard Brook Annual Cooperators Meeting, July 8 - 9, 2015, Hubbard Brook Experimental Forest, New Hampshire, USA.
- **Valipour, M.,** Driscoll, C.T, "Data reconstruction methods and model calibration results to evaluate ecosystem behavior to an intensive treatment", Oral Presentation, Environmental Group Seminar, November 30, 2015, L.C. Smith College of Engineering, Syracuse University, Syracuse, NY, USA.

Research & Teaching Assistant

Teaching Assistant at Civil and Environmental Engineering Department, Syracuse University:

- Spring 2019: Sustainability of Civil and Environmental Systems
- Fall 2018: Principles of Fluid Mechanics
- Spring 2018: Water Resources Engineering
- Spring 2018: Civil and Environmental Engineering Design (Senior Capstone Design)
- Fall 2016: Introduction to Environmental Engineering
- Fall 2014: Principles of Fluid Mechanics

Research Assistant at Civil and Environmental Engineering Department, Syracuse University:

- Spring 2016-Fall 2017. Project: “Model optimization, parametrization and calibration for different harvesting techniques in an experimental forest, Hubbard Brook, NH, US”. Funded by National Science Foundation and Long-term Ecological Research (NSF, LTER)
- Spring 2015. Project: “Evaluation of hydrologic performance, ecosystem interactions and functional limitations of the varied and widespread green infrastructure systems”
- Spring-Summer 2014. Project: “Evaluation of fate and transport of contaminants in groundwater systems”

Job Experiences

Executive (Isfahan City, Iran, 2008 – 2010)

- Employer: Tarh Afra Eng. Consultancy Company

Academic (Isfahan City, Iran, 2009-2014)

- **Part-Time** Instructor, Isfahan University, 5th Top Iranian University, 2012 – 2014
- **Full-Time** Instructor, Najaf Abad Private University, May. 2010 – Feb.2011
- **Part-Time** Instructor, Isfahan Payame Nour University, Zarin Shahr Payame Nour University, Tiran Payame Nour University, Feb. 2009 - May. 2010

Taught Undergrad Courses: Design of Urban Stormwater Systems, Fluid Mechanics, Hydraulics, Soil and Water Conservation, Statics, Construction Materials and Methods, Concrete Technology, Rural Road Construction and Maintenance.

Computer Skills

Programing languages: R, C++, SQL, Python, SAS.

Software: ArcGIS, AutoCAD, ModFlow.

Big data analytics and machine learning tools: Apache Spark, Hadoop, Weka.

Awards & Memberships

- Outstanding TA Award, Civil and Environmental Engineering Department, Syracuse University, 2019
- Nelson L. Nemerow Memorial Scholarship, 2018
- GSO Travel Grant Award, 2017
- Woman in Science and Engineering and Future Professional Program, 2015-2018
- American Society of Civil Engineers (ASCE) Membership, 2015-Present