14

Effects of Watershed Experiments on Water Chemistry at the Marcell Experimental Forest

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CONTENTS

Introduction

Watershed studies reveal important information on the effects of forest harvesting and ecosystem disturbances on water and solute yield. Since the first study by Bormann and Likens (1967), studies of nutrient cycling in steep mountainous watersheds have shown that changes to stream yields of nutrients and mineral weathering products vary with vegetation cover, climate, and forest management practices. Most studies show increases in concentrations of nitrate after clearcutting (Bormann et al. 1968; Swank 1988; McHale et al. 2008). Concentrations of other solutes such as sulfate and cations have increased at some watersheds after clearcutting (Bormann et al. 1968; Neal et al. 1992) while other watersheds have shown little or no response (Swank 1988; Martin and Harr 1989).

Although studies in lowland watersheds with large wetlands are less common, results from short-duration studies on peatland ecosystems have reported effects of forest management on concentrations of stream solutes (Verry 1972; Knighton and Stiegler 1981; Ahtiainen 1992; Rosén et al. 1996; Sørensen et al. 2009). The Marcell Experimental Forest (MEF) was established during the 1960s (Chapters 1 and 2) to study the hydrology and ecology of lowland watersheds where upland mineral soils drain to central peatlands (Boelter and Verry 1977). The effects of seven large-scale manipulations on water chemistry have been studied on the MEF watersheds and the data now span up to four decades. In this chapter, we review effects of watershed experiments on solute concentrations and yields in streams and present some findings for the first time.

Watershed Studies at the MEF

Each of six research watersheds at the MEF has a bog or a fen that drains to one or more streams. The ecology, hydrology, and chemistry of bogs differ from fens (Bay 1967; Boelter and Verry 1977). Fens are embedded in the regional aquifer and groundwater flow into fens is a source of water in addition to precipitation and upland runoff. The MEF bog watersheds are perched 5–10 m above the regional aquifer and groundwater from the aquifer does not flow into the bogs. Therefore, the only sources of water are precipitation to bog surfaces and inputs of water via lateral flowpaths through upland soils (Boelter and Verry 1977; Timmons et al. 1977; Chapter 7). Responses to forest harvesting on both types of peatlands have been studied at the MEF: the black spruce at the S1 bog were clearcut with a set of stripcuts in 1969 and the remaining mature trees were cut in 1974; and the forest on the S3 fen was clearcut during 1972.

Even when peatland forests are not harvested, stream-water yield and chemical responses may be influenced by cutting on adjacent uplands. Therefore, clearcutting of upland aspen forests was studied at two watersheds including an experiment in which the uplands regenerated in aspen cover at the S4 watershed and another experiment in which the forest was converted from aspen to conifer cover after clearcutting at the S6 watershed. Vegetation and hydrological responses to harvesting of bog, fen, or upland forests at the MEF are described in Chapters 12 and 13. Water chemistry responses are described in this chapter.

In conjunction with the forest harvest treatments, three additional large-scale experiments included watershed manipulations. The regenerating aspen forest at the S4 watershed was fertilized with ammonium nitrate during 1978 (Berguson and Perala 1988). Aspen sprouting on the S6 uplands was suppressed by cattle grazing during the first three growing seasons after harvest. In addition, ambient sulfate wet deposition was quadrupled on the S6 bog from 2001 to 2008 to determine effects on methylmercury production in a peatland (Chapter 11, Jeremiason et al. 2006).

Long-term measurements in the research watersheds are briefly described here with more details in Chapter 2. Water table elevations in peatlands are measured in each research watershed. Stream stage was measured and streamflow was calculated from stage-discharge relationships at each watershed for at least several years. Stream stage has been measured at permanent stream gages at the outlets of the S2, S4, S5, and S6 watersheds for the duration of the studies. Regressions between water levels at peatland wells and streamflow have been used to calculate monthly and annual water yields for the S1 and S3 watersheds during periods when stream stage was not measured.

Samples of stream water have been collected and analyzed for a variety of solute concentrations in unfiltered water samples. During the 1960s and 1970s, sampling frequency and analyses varied among studies and study sites. Routine biweekly sampling of stream or peatland waters began during 1975 (Chapter 2). Although some data have been reported for samples collected during the 1960s and 1970s (Verry 1972, 1975; Timmons et al. 1977; Knighton and Stiegler 1981; Verry and Timmons 1982), many values from the 1960s, 1970s, 1980s, and 1990s have not yet been entered as electronic data due to the lengthy process of discovering, entering, and checking values from hand-written records and computer printouts. In coming years, as we devote renewed attention to these legacy datasets, we hope that filling gaps as electronic data will enable us to more rigorously evaluate concentrations and yields of solutes in streams.

Nitrate (nitrate + nitrite), ammonium, total organic nitrogen (TON), total phosphorus (TP), and calcium concentrations are the longest and most complete data series. There are sufficient data for those solutes to calculate annual volume-weighted concentrations and solute yields for many years before, during, and after watershed experiments. For some studies, concentrations of other solutes were measured for shorter periods. Stream TON concentrations were calculated by subtracting the ammonium concentration from total Kjeldahl nitrogen, which was measured frequently between 1978 and 1997. Sample concentrations were interpolated linearly to estimate daily concentrations during the intervals between successive samples. Daily concentrations were multiplied by the daily stream-water yield to calculate stream loadings. Loadings were integrated to estimate annual solute yields for each water year. A water year begins on March 1 of the calendar year and ends on February 28 or 29 of the following year. This water year corresponds to the hydrology of perched bog watersheds of north-central Minnesota where streamflow begins during spring snowmelt and stops during the winter (Chapter 2). Annual volume-weighted concentrations were calculated by dividing annual solute yields by annual stream-water yields.

Water and solute yields at the harvested sites were compared to unmanipulated control watersheds (S2 or S5). Temporal patterns of annual volume-weighted concentrations and yields may vary between the control watersheds. To distinguish responses due to experiments from natural variation among different watersheds, we compare the time series of ratios of a particular manipulated watershed to a control watershed (e.g., S6:S2 for the S6 uplands clearcut study) to the ratios of the control watersheds (e.g., S5:S2). We refer to "treatment" ratios when the numerator is the annual concentration or yield data from a manipulated watershed divided by that of a control watershed and to "control" ratios when data from the second control watershed are divided by the control of the manipulated watershed.

Effects of Harvest and Regeneration of Upland Aspen Forests on Stream Chemistry

Aspen (*Populus sp.*), an early succession species, is managed for pulp production throughout northern Minnesota on a short-rotation basis (Bates et al. 1989). The S4 study was designed to determine the effects of upland aspen clearcutting on stream-water and solute yields as well as management strategies for aspen regeneration (Chapter 12). The 25.9 ha upland aspen forest at S4 was harvested with a whole-tree commercial clearcut during the winters of 1970–1971 and 1971–1972. No vegetation on the 8.1 ha central peatland was harvested. Half of the upland area or 34% of the total watershed area was clearcut before snowmelt in 1971. Seventy-one percent of total watershed area was clearcut when the uplands harvesting was completed during January 1972. The upland forest began to regenerate naturally from root sprouting of aspen during the first growing seasons after the two clearcutting periods. In 1978, as a study of forest management in aspen regrowth, ammonium nitrate fertilizer was applied by helicopter at a rate of 340 kg ha^{-1} to the S4 upland forest (Berguson and Perala 1988).

Stream-water yield after harvesting was compared to water yield that was predicted from the pretreatment calibration period during water years from 1962 to 1969 to evaluate the effects of uplands clearcutting on water yield (Chapter 13). Beginning in 1967, water samples were collected from the S5 control watershed and one of the two outlet stream gages (S4N, which is the north outlet) of the S4 watershed (Verry 1975). No samples were collected at S4N and S5 from 1983 to 1988.

Verry (1972) reported values for pH, specific conductivity, apparent color, organic nitrogen, nitrate, nitrite, total N, aluminum, chloride, iron, calcium, copper, sodium, magnesium, manganese, potassium, total phosphate, and zinc before the harvest and the first year after the harvest. After the partial clearcutting, streamflow was 35% lower during peak snowmelt (Verry 1972), and the annual water yield increased by 25% relative to the preharvest period (Chapter 13). Despite the hydrological changes, Verry (1972) showed that stream water chemistry at the partially clearcut S4 watershed did not change relative to the S5 control.

Until now, stream chemistry responses to the S4 clearcutting have not been updated since Verry (1972) published first-year results. From 1972 (second postharvest year) to 1979, stream-water yield increased by 9%–38% relative to the predicted S4 water yield during water years from 1962 to 2005 (Verry 1987; Chapter 13; Figure 14.1a). Unlike increased water yields that persisted for a decade after the harvest, stream nitrogen responses were of short duration. Annual volume-weighted concentration and yield ratios of nitrate + nitrite (Figure 14.1) and ammonium (Figure 14.2) were similar between the control and clearcut watershed until the upland fertilization during 1978. The control (S2:S5) concentration and yield ratios were higher and more variable than the treatment (S4:S5) ratios for nitrate + nitrate during water years before and after 1978. By contrast, the relationships were much different during 1978 when the treatment ratio of nitrate + nitrite concentrations was 96.7 times larger than the control ratio and the yield ratio was 245 times larger (Figure 14.1). The treatment ratio of ammonium concentrations was 6.0 times larger than the control ratio during the 1978 fertilization, which contrasted with the most other years when the ratios were similar (Figure 14.2). The treatment ratio of ammonium yields was 12.0 times larger than the control ratio during the 1978 fertilization. Although the control ratio was 6.6 times larger than the treatment ratio during 1990, yield ratios were similar that year because the water yield that had remained elevated during the first decade after the uplands clearcutting returned to preharvest levels during the 1980s and 1990s. For TON, no data before 1978 were available. We calculated annual volume-weighted concentrations and yields of TON for water years from 1978 to 1997 (Figure 14.3). The treatment ratios were the highest during 1978 and decreased over the next 2 years but remained higher than the levels of the controls through the 1990s.

The magnitudes and fluctuation patterns of calcium concentration ratios were similar during water years before and after clearcutting (Figure 14.4). After fertilization during 1978 through the 2000s, the S4:S5 (treatment) yield ratios were larger than the control ratios. The yield ratios have converged during the last 6 years of record corresponding to water years during the 2000s when the water yield had decreased by as much as 21% relative to the preharvest period (Chapter 13).

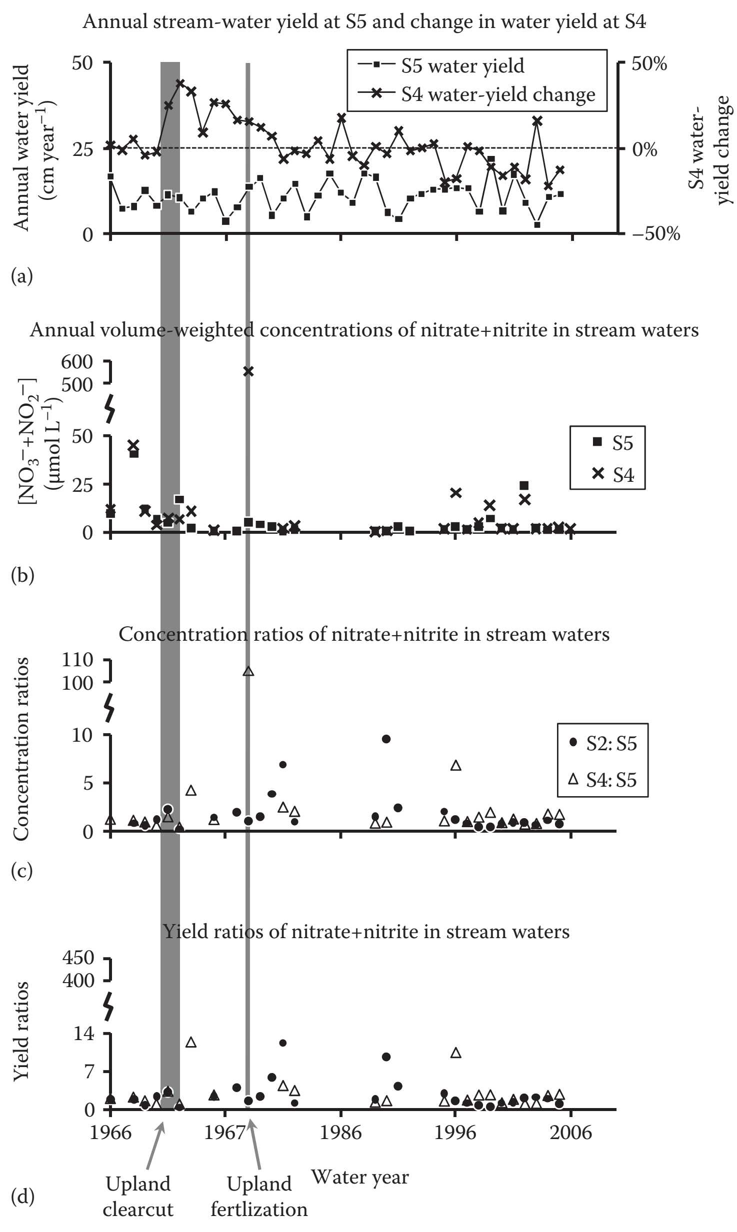

FIGURE 14.1
Annual water yields at the S5 control watershed and changes in stream-water yield at the S4 watershed (a); annual volume-weighted concentrations of nitrate + nitrite for the S5 and S4 watersheds (b); annual stream nitrate + nitrite concentration ratios at S2 relative to S5 (S2:S5) for the control watersheds and S4 relative to S5 (S4:S5) for the experiment (c); and annual stream nitrate + nitrite yield ratios (d).

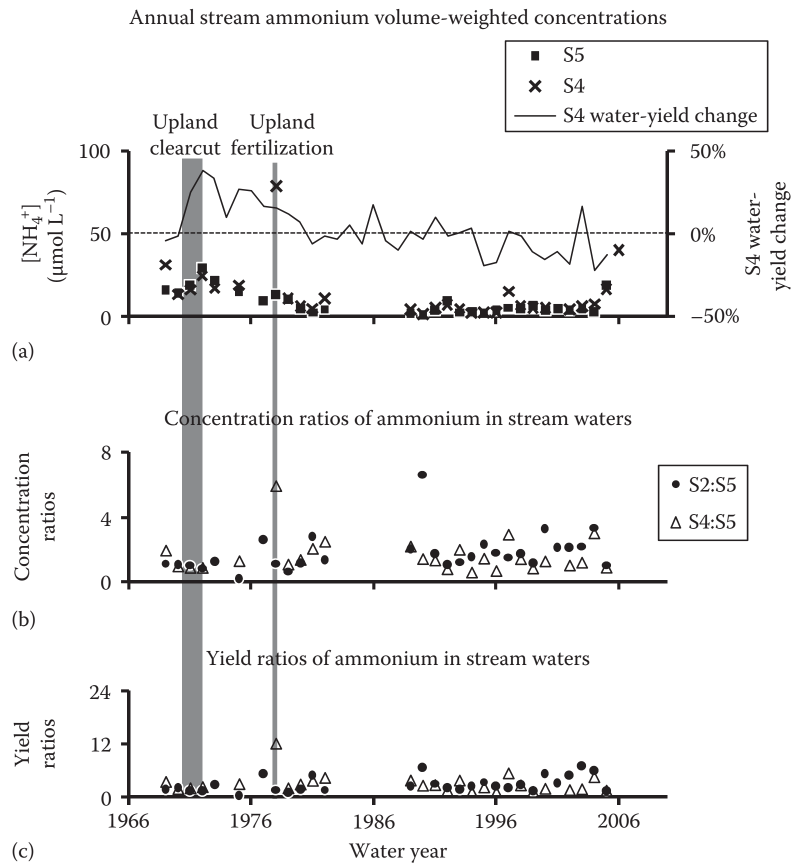

FIGURE 14.2
Changes in annual water yield at the S4 watershed and annual volume-weighted concentrations of ammonium for the S5 control and S4 watersheds (a); annual stream ammonium concentration ratios at S2 relative to S5 (S2:S5) and S4 relative to S5 (S4:S5) (b); and annual stream ammonium yield ratios (c).

Conversion of an Upland Forest from Hardwoods to Conifers

The 6.9 ha upland aspen forest at the S6 watershed was harvested with a whole-tree commercial clearcut from December 1980–June 1981. Most streamflow occurs during spring snowmelt; 22% of the uplands were clearcut before snowmelt and 78% clearcut after most of the annual streamwater yield had occurred. When completed in June, the clearcut was 78% of the watershed area because no vegetation on the 2.0 ha central peatland was harvested. The watershed was fenced during 1981 and grazed with cattle during the first three growing seasons to prepare the site for

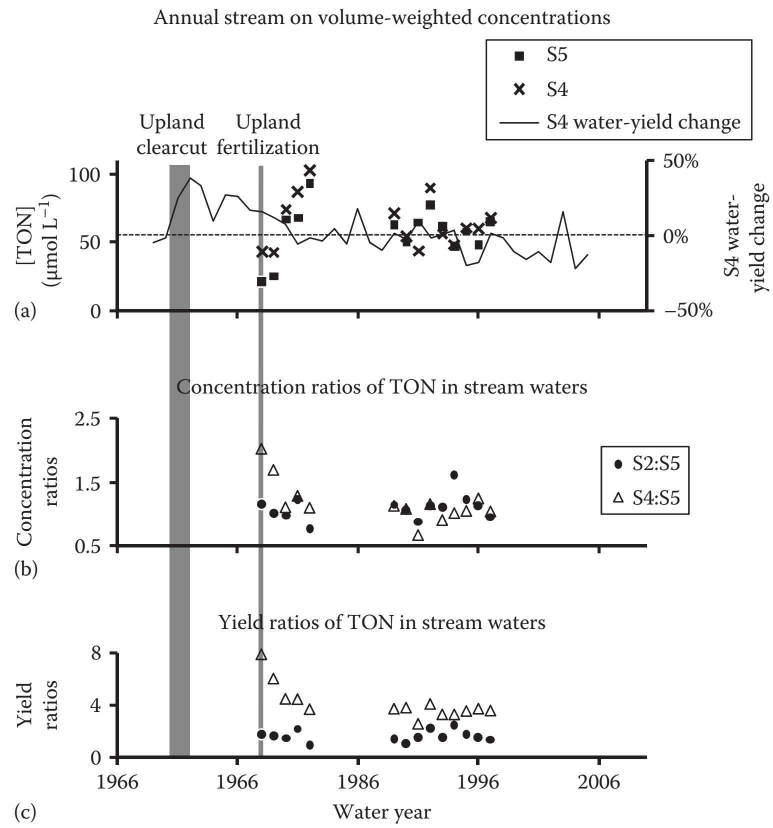

FIGURE 14.3
Changes in annual stream-water yield at the S4 watershed and annual volume-weighted concentrations of TON for the S5 control and S4 watersheds (a); annual stream TON concentration ratios at S2 relative to S5 (S2:S5) and S4 relative to S5 (S4:S5) (b); and annual stream TON yield ratios (c).

planting of red pine and white spruce seedlings during 1983. Grazing was evaluated to assess the cost effectiveness relative to the more common use of herbicides to control vegetation regrowth (Chapter 12). Grazing suppressed the natural regeneration of aspen sprouts, shrubs, and grasses until conifers were planted. During grazing, the uplands were relatively devoid of woody vegetation for 3 years until the conifers were planted during 1983. The canopy of the growing conifer forest closed during the late 1990s. About the time of canopy closure, an experiment was initiated to determine the effects of sulfate deposition on methylmercury production in peatlands. Sulfate was added to the downstream half of the peatland (11% of the total watershed area) as a sodium sulfate solution to increase the ambient wet deposition by a factor of four between the autumn of 2001 and autumn of 2008.

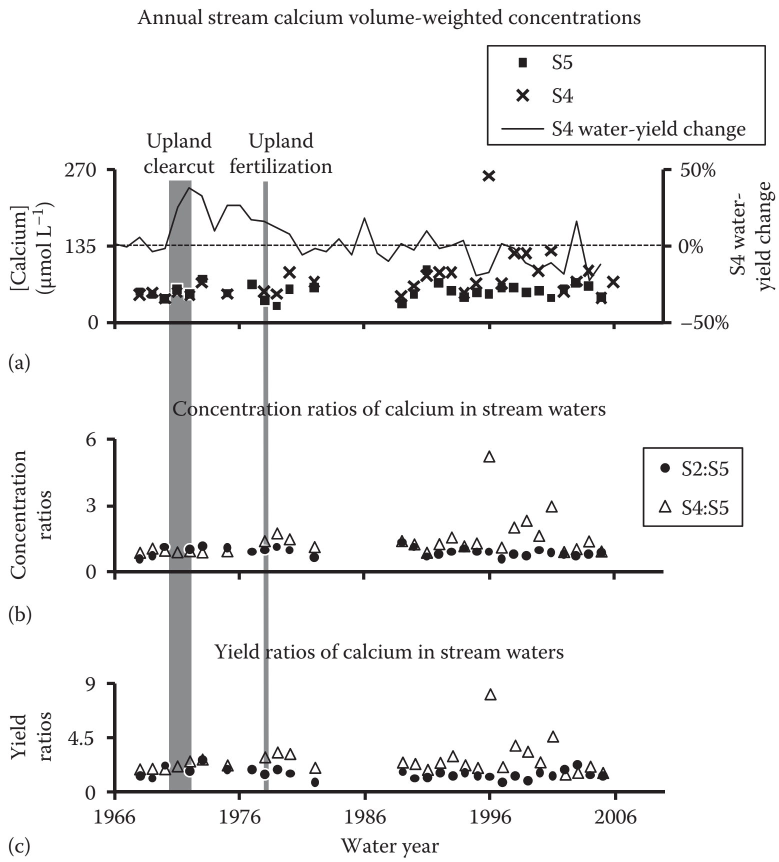

FIGURE 14.4
Changes in annual water yield at the S4 watersheds and annual volume-weighted concentrations of calcium for the S5 control and S4 watersheds (a); annual stream calcium concentration ratios at S2 relative to S5 (S2:S5) and S4 relative to S5 (S4:S5) (b); and annual stream calcium yield ratios (c).

Effects of Uplands Clearcutting and Forest Conversion on Stream Solutes

Stream-water yield after harvesting was compared to water yield predicted from the pretreatment relationship during the calibration period during water years from 1976 to 1979 (Chapter 13). Increases in water yield during the first decade after clearcutting averaged 23% ± 16% (±1 standard deviation) and the largest increase of 63% occurred during 1982 (Figure 14.5a). The average water yield (+1% ± 28%) during the 1990s was similar to preharvest levels. Water yield during the 2000s decreased by an average of 39% ± 19% relative to preharvest levels.

We report the stream-chemistry responses to the clearcutting on the S6 uplands and the forest conversion for the first time. In contrast to the

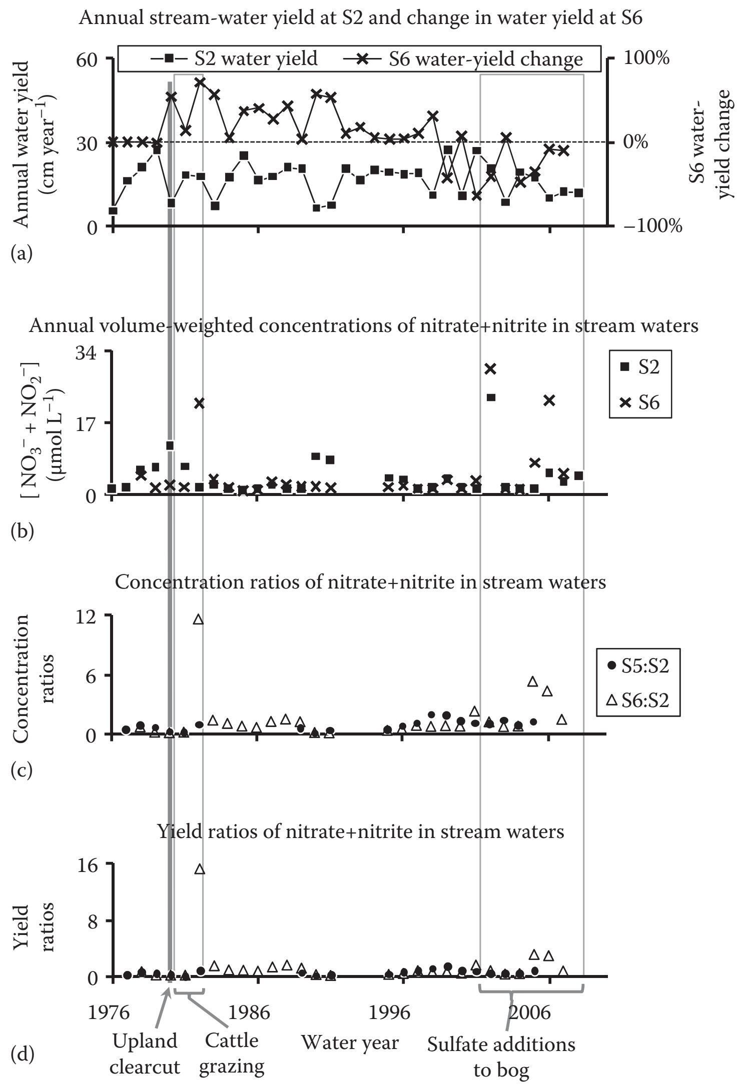

FIGURE 14.5
Annual water yields at the S2 control watershed and changes in stream-water yield at the S6 watershed (a); annual volume-weighted concentrations of nitrate + nitrite for the S2 control and S6 watersheds (b); annual stream calcium concentration ratios at S5 relative to S2 (S5:S2) and S6 relative to S2 (S6:S2) (c); annual stream nitrate + nitrite yield ratios (d).

immediate increase of the water yield that persisted for a decade after harvesting, responses of stream nitrogen species were offset and of short duration (Figure 14.5a). During and 1 year after the upland clearcutting when cattle grazed the watershed, the range and variation of concentrations for nitrate + nitrate (Figure 14.5b), ammonium (Figure 14.6a), and TON

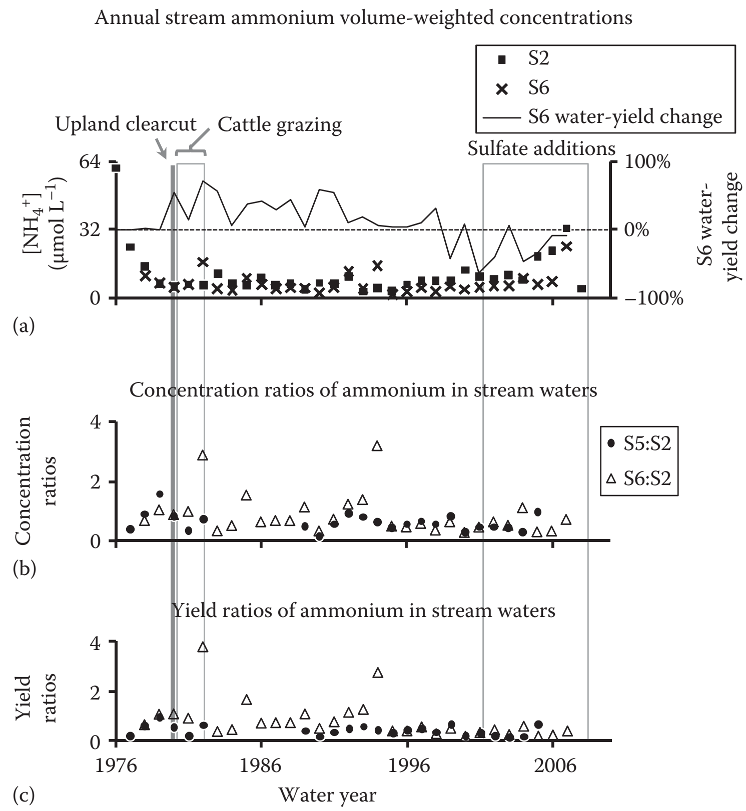

FIGURE 14.6
Changes in annual stream-water yield at the S6 watershed and annual volume-weighted concentrations of ammonium for the S2 control and S6 watersheds (a); annual stream ammonium concentration ratios at S5 relative to S2 (S5:S2) and S6 relative to S2 (S6:S2) (b); and annual stream ammonium yield ratios (c).

(Figure 14.7a) were similar to those for the control watershed. Nitrate + nitrite concentration and yield ratios increased by an order of magnitude during the second water year after the harvest (1982). Annual stream nitrate + nitrite ratios during other years were similar between the controls (S5:S2) and treatment (S6:S2). Stream nitrate + nitrite ratios also were higher after the sulfate addition during water years from 2005 to 2007.

The annual volume-weighted concentration of ammonium was much higher during 1976 than other years at S2 (data for S6 start during 1978). There was a severe summer drought and the lowest stream-water yields on record during 1976. Annual concentration and yield ratios for ammonium were similar before the clearcutting and again from the mid-1990s through 2004. The treatment ratios for annual concentrations and yields were markedly higher during 1982 and 1994 than controls ratios (2 and 14 years after the

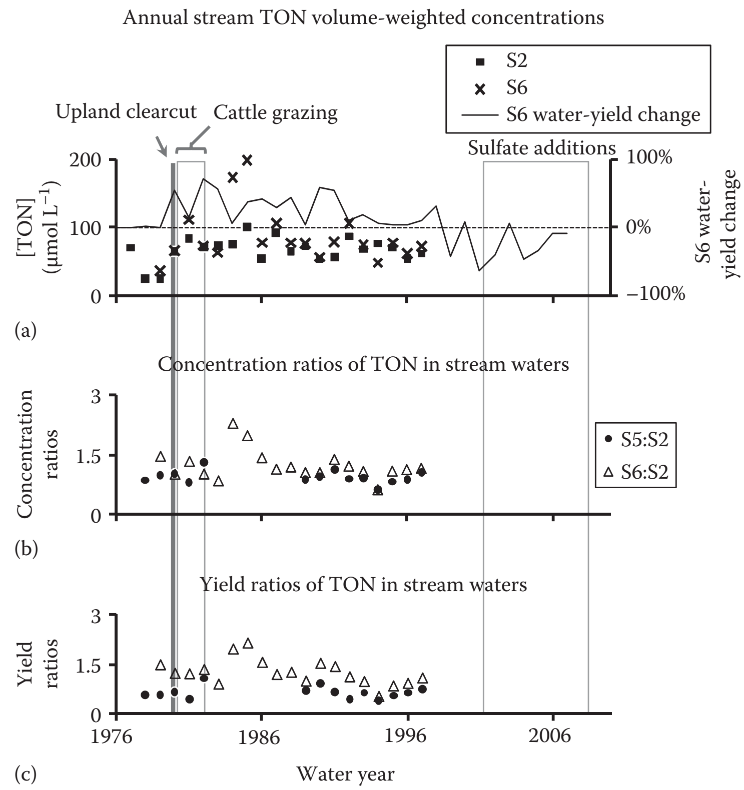

FIGURE 14.7
Changes in annual stream-water yield at the S6 watershed and annual volume-weighted concentrations of TON for the S2 control and S6 watersheds (a); annual stream TON concentration ratios at S5 relative to S2 (S5:S2) and S6 relative to S2 (S6:S2) (b); and annual stream TON yield ratios (c).

clearcutting, respectively). For example, the concentrations were 2.9 times larger and the yield ratios were 3.8 times higher during 1982.

During the fourth and fifth water years after the clearcutting (i.e., 1984 and 1985), the annual volume-weighted concentrations of TON at S6 increased relative to other years and were nearly double those of S2 (Figure 14.7a). From 1989 until 1997, concentrations at the control and experimental watersheds had similar magnitudes and interannual fluctuation patterns (Figure 14.7b).

Responses of TP (Figure 14.8) and calcium (Figure 14.9) did not show brief, large increases like concentrations and yields of nitrogen species. Annual volume-weighted concentrations of TP in stream waters were similar in magnitude and patterns of interannual variation at the S2 and S6 watersheds except the first 7 years after the harvest and several years during the sulfate-addition study. Annual concentration and yield ratios for TP were more variable and larger in magnitude for the treatment than the control watersheds (Figure 14.8). For both TP and calcium, the paucity

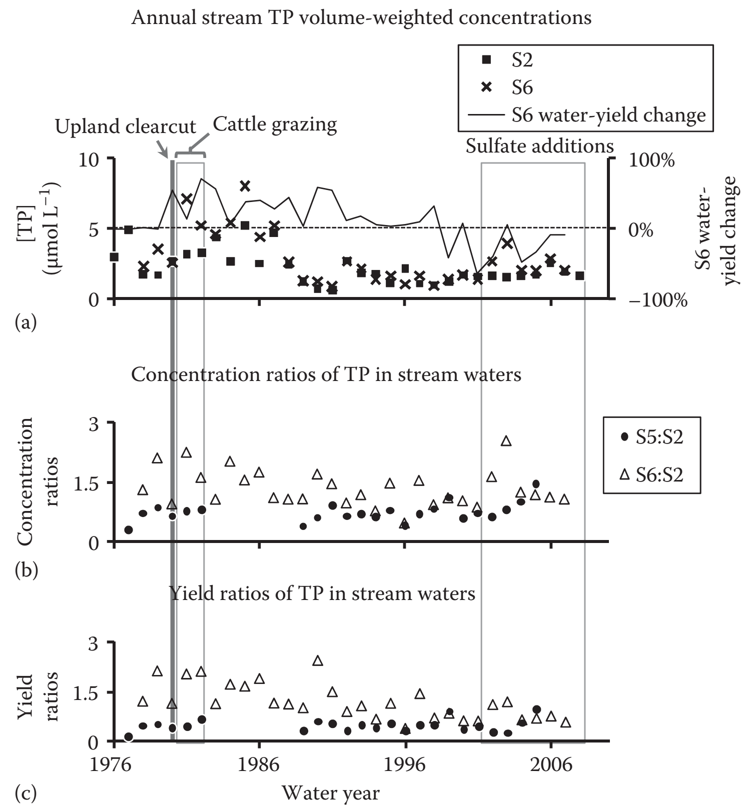

FIGURE 14.8
Changes in annual water yield at the S6 watershed and annual volume-weighted concentrations of TP for the S2 control and S6 watersheds (a); annual stream TP concentration ratios at S5 relative to S2 (S5:S2) and S6 relative to S2 (S6:S2) (b); and annual stream TP yield ratios (c).

of preharvest data prevents a meaningful comparison to the preharvest period. Nonetheless, a pattern does emerge for TP yields after clearcutting. Annual yield ratios have decreased since the upland clearcutting (linear regression having a $p=0.0004$ and R^2 of 0.49) and not changed for the controls ($p=0.34$, $R^2=0.05$). This convergence pattern reflects increases in stream-water yield at S6 during the 1980s that diminished as stream-water yields returned to preharvest levels during the 1990s and decreased further during the 2000s (Figure 14.8). Annual volume-weighted concentrations of calcium in stream waters at the S6 watershed increased relative to the S2 control and were higher throughout the 1980s (Figure 14.9a).

As noted earlier, the uplands were only partially clearcut before snowmelt and most runoff during water year 1980 occurred with snowmelt before the entire upland area was clearcut. This sequence of harvesting may have been a factor related to the similarity of stream solute concentrations and yields between the S6 and control watersheds during the year when clearcutting occurred.

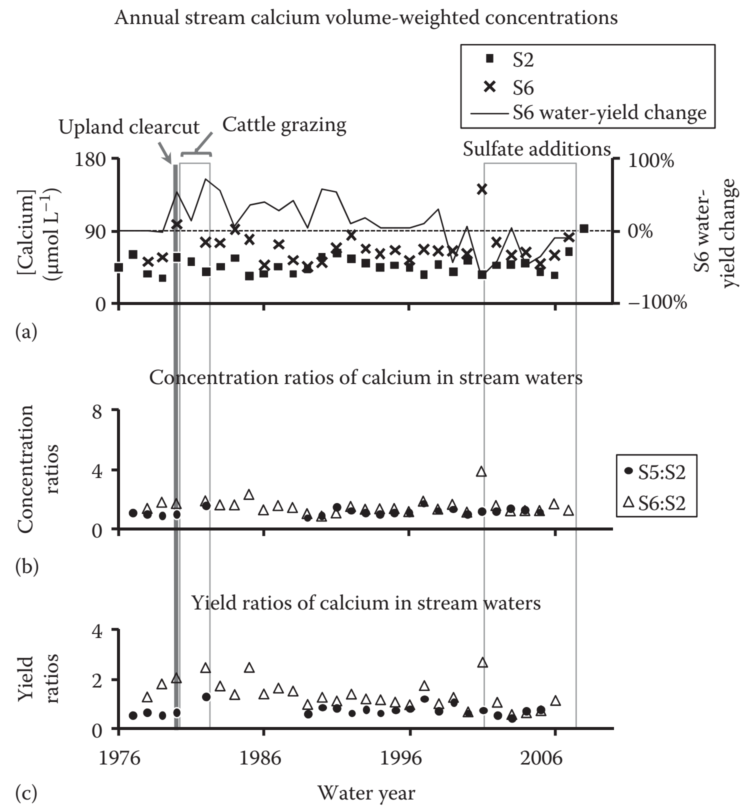

FIGURE 14.9
Annual water yield changes at S6 and annual volume-weighted concentrations of calcium for the S2 control and S6 watersheds (a); annual stream calcium concentration ratios at S5 relative to S2 (S5:S2) and S6 relative to S2 (S6:S2) (b); and annual stream calcium yield ratios (c).

Effect of Cattle Grazing on Stream Fecal Coliform Counts in Streams

The grazing study provided an opportunity to monitor the effects of cattle on stream-water quality during forest conversion. The uplands were grazed for 5–12 weeks each year from 1980 to 1982 to consume aspen regrowth. Cattle were not excluded but did not venture onto the soft organic soils of the peatland. The presence of more than one positive fecal coliform count (per 100 mL sample) per month exceeds the maximum contaminant level for drinking water (U.S. Environmental Protection Agency 2009). To determine whether manure that accumulated on bare upland soils was transported to streams during rainfall and snowmelt runoff events, fecal coliform colonies were counted starting September 1980 (after clearcutting but before the addition of cattle) and ending in 1986. Samples were collected at the S6 weir, about 60 m downstream of the S6 weir where the stream entered a road culvert at the fence boundary, and the S2 weir.

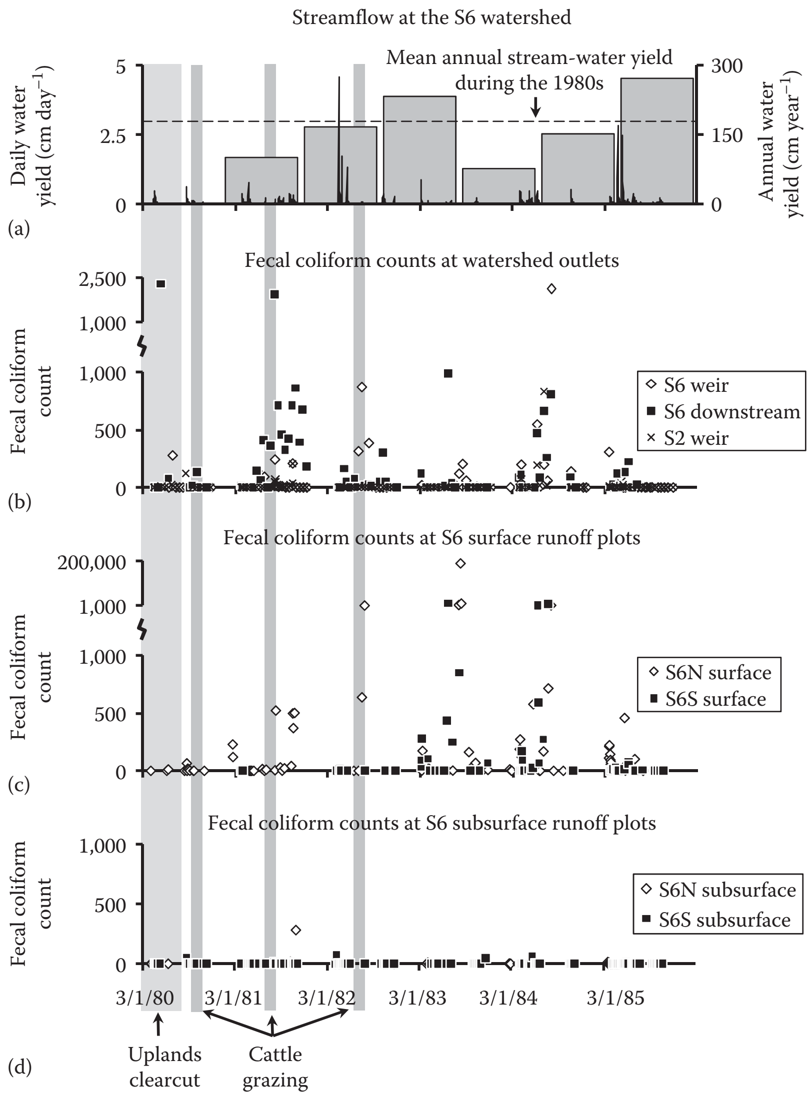

FIGURE 14.10
Streamflow (a) and fecal coliform counts (per 100 mL aliquot) in stream waters (b), upland surface runoff waters (c), and upland subsurface runoff waters (d) from the S2 and S6 watersheds.

Fecal coliform counts in stream water ranged from 0 to 834 at the S2 control watershed and from 0 to 2150 at the S6 weir from 1980 to 1986 (Figure 14.10a). Positive counts at S2 ranged from 5% to 53% of samples among years and the percentages at the S6 weir equaled or exceeded those at the S2 control during every year (Figure 14.11). The mean fecal coliform count of 13 ± 73

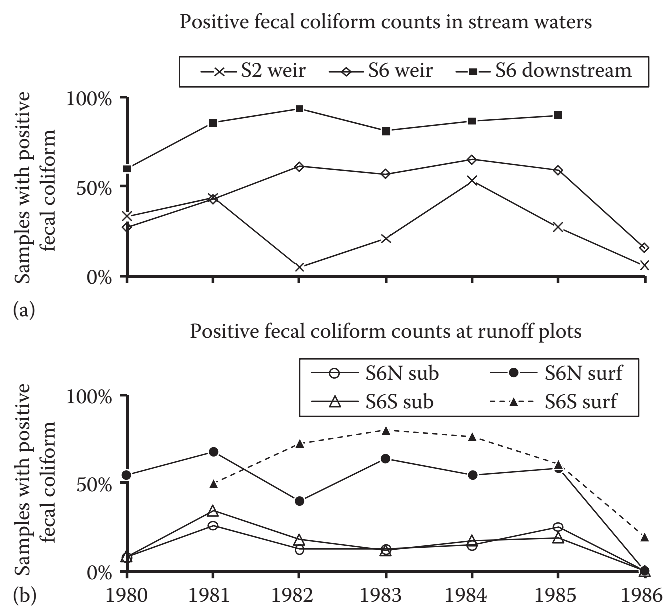

FIGURE 14.11
Positive fecal coliform counts in stream (a) and runoff plot (b) waters from the S2 and S6 watersheds.

(±1 standard deviation) at the S2 watershed was less than the mean of 48 ± 142 at the S6 weir (t-test p = .028).

Fecal coliform counts frequently were higher downstream because cattle entered the channel downstream of the weir. The mean of downstream samples from September 1980 to November 1982 of 212 ± 367 (±1 standard deviation) counts was four times the upstream mean of 53 ± 153 (Figure 14.10b). In addition to cattle sources, unknown natural sources introduced fecal coliform to stream waters as shown by positive counts at the S2 control watershed and positive counts before cattle grazed the S6 uplands. One sample collected at the S6 downstream site before grazing had the highest stream-water count of 2300. Despite natural sources, the frequency of large counts increased after grazing. Larger counts persisted at the S6 weir and downstream for 2 years when levels and the frequency of positive counts returned to levels at the S2 control during 1986. Above-average streamflow during 1985 may have hydrologically flushed residual fecal coliform colonies from the S6 watershed after 2 years of below-average water yield during 1983 and 1984 that coincided with the first 2 years after cattle were removed.

Fecal coliform counts also were measured at surface and subsurface runoff plots on north-facing (site S6S) and south-facing (site S6N) hillslopes. Waters from these samplers represent two flowpaths that transport solutes from upland soils to the lagg zones that surround the bogs (Chapters 2 and 7). Lagg zones preferentially route water and solutes to the watershed outlet during snowmelt and rainfall runoff events when peatland water levels are the highest. Therefore, lagg zones that receive upland waters may route fecal coliform to the watershed outlet. After cattle first grazed the uplands during September 1980, the autumn was relatively dry and fecal coliform counts in surface (Figure 14.10c) and subsurface (Figure 14.10d) runoff were zero. In contrast to consistently low counts at the subsurface runoff plots with means of 2 ± 9 (S6S) and 2 ± 20 (S6N) from 1980 to 1986, counts in water samples from the surface runoff plots increased during grazing at S6N and after grazing at S6S. The highest count of 191,000 was measured in surface runoff from the S6S plot.

Data from the surface runoff plot, S6 weir, and downstream sites suggest that the downstream fecal coliform originated from manure in the stream channel during 1981 and 1982. When counts in surface runoff and S6 weir samples were high during 1982, 1983, and 1984, we hypothesize that some fecal coliform was transported from uplands via the lagg to the outlet stream.

Effect of Sulfate Additions on Stream Methylmercury

Sulfate is an atmospherically deposited pollutant that affects aquatic and peatland ecosystems (Gorham et al. 1984; Hedin et al. 1987) and is linked to the production of highly toxic methylmercury in anoxic organic soils (St. Louis et al. 1996; Branfireun et al. 1999). The sulfate addition at the S6 watershed was designed to assess: (1) the watershed-scale effects of sulfate deposition on methylmercury production in peatlands, and (2) the effects of decadal scale recovery from peak sulfate deposition during the 1970s on methylmercury production and outflow in peatland watersheds. Two important responses of stream chemistry were measured in direct response to the S6 sulfate additions between November 2001 and November 2008. During the first year after the sulfate addition, merthylmercury yields in stream water doubled after sulfate additions quadrupled the wet deposition of sulfate to half the bog surface area (Jeremiason et al. 2006). Sulfate yields in stream waters draining the S6 watershed increased relative to the S2 watershed during water years in the 2000s in contrast to the similar relationships between sulfate yields at the watersheds during the 1980s and 1990s (Figure 14.12).

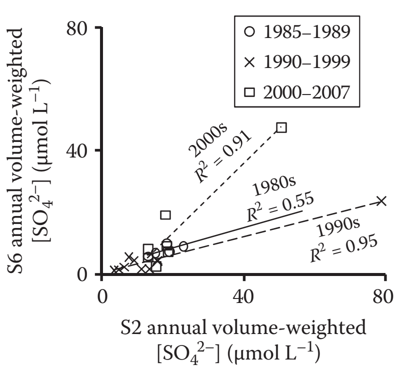

FIGURE 14.12
Annual volume-weighted sulfate concentrations at S6 versus S2.

Additional findings reveal rapid responses of net methylmercury production to sulfate additions: net methylmercury production increased when water levels were the highest, particularly after snowmelt; and net methylmercury production was elevated relative to controls during the first year of the recovery period after sulfate additions ended (Coleman-Wasik 2008). Importantly, these dramatic changes in the production of methylmercury occurred when only 11% of the entire watershed area was affected by the elevated inputs of sulfate.

Chemical Responses to Upland Forest Harvesting in Lowland Peatland Watersheds in Relation to Other Studies

Nitrate and ammonium are redox-sensitive dissolved species of nutrients that play a major role in regulating ecosystem productivity. At the MEF where peatlands are nutrient-poor, little inorganic nitrogen is transported through the peatlands to streams (Urban and Eisenreich 1988; Urban et al. 1988; Chapter 8). The responses of stream nitrogen chemistry to commercial harvests of upland forests at the MEF were of short duration and delayed relative to other upland watershed studies that include commercial clearcuts (Swank 1988; Martin and Harr 1989; Neal et al. 1992; Martin et al. 2000; McHale et al. 2008; Stednick 2008), experimental strip and block cuts (Hornbeck et al. 1987), and multiyear devegetation studies (Bormann et al. 1968; Kochenderfer and Aubertin 1975). However, these patterns at the MEF may be related to the partial clearcutting of the watersheds before snowmelts with completion of the upland clearcutting after most of the annual stream water-yield had already occurred. Climatic variation over the decade that separated the S6 and S4 studies at the MEF may explain some differences in stream nutrient responses to treatments. Nonetheless, results of the experiments show stream nutrient responses to the particular forest-management practices (i.e., 3 year deforestation and cattle grazing at S6 versus the immediate resprouting of aspen at S4).

Nitrate concentrations in stream waters typically increase from twofold to several orders of magnitude within 1 year of clearcutting and higher concentrations may persist from 3 to 20 years (Bormann et al. 1968; Swank 1988; Martin and Harr 1989; Neal et al. 1992; Martin et al. 2000; McHale et al. 2008). First-year increases in stream nitrate concentrations and yields were not detected during 1971 after 34% of the S4 watershed area was clearcut or during 1972 after 71% was clearcut. Annual stream nitrate concentrations and yields changed only during the seventh year after clearcutting when the regenerating forest on the S4 uplands was fertilized. The greater than 500-fold increase in stream nitrate concentrations with fertilization is among the largest reported responses to a forest-management practice. The S4 study results differed from the S6 study in that nitrate concentrations in stream waters increased by an order of magnitude 2 years after the clearcutting and in the third year of cattle grazing.

Comparisons with steep upland watersheds reveal some striking differences from lowland watersheds where oxidized species such as nitrate are chemically reduced in saturated and anoxic organic soils to produce ammonium. In addition, TON concentrations are high due to leaching of soluble organic matter and production of organic acids in peatlands. Nitrogen limitation also leads to near-complete biological assimilation of inorganic nitrogen in peatlands (Urban and Eisenreich 1988; Urban et al. 1988; Chapter 8). Although different from upland watershed studies, the timing response of stream nitrate increases at the MEF S6 clearcutting study was similar to another lowland watershed with wetlands and boreal tree species. Stream nitrate concentrations increased incrementally during the first three growing seasons after uplands clearcutting at a lowland Swedish watershed, 277 Balsjö (Löfgren et al. 2009). However, increases in concentration were at least an order of magnitude larger at 277 Balsjö. One pertinent difference may be drainage ditching of the 277 Balsjö peatland and the consequences of lowered water tables and the altered oxygen status of peatland soils relative to the MEF peatlands that have not been drained.

Concentrations and yields of ammonium in stream waters increased after upland clearcutting at S6 and at 277 Balsjö (Löfgren et al. 2009). The timing of increases in ammonium concentrations and yields at the S6 watershed was similar to nitrate increases but much smaller in magnitude. Ammonium concentrations increased threefold and yields increased fourfold during the third water year after the S6 clearcutting. Ammonium concentrations increased 6-fold and yields increased 23-fold after the S4 fertilization. The increased ammonium concentrations and yields from lowland watersheds with peatlands differed from studies at steep upland watersheds where little to no ammonium was present both before and after clearcuts (Bormann et al. 1968; Hornbeck et al. 1987; Swank 1988).

TP concentrations usually are driven by particulate transport in steep catchments. By contrast, the hummock and hollow microtopography of the MEF peatlands effectively retains most organic and inorganic particulates. Stream TP concentrations at the S6 watershed were higher than at the S2 control during most years from the second through seventh water years after harvest. At two other lowland watersheds with peatlands, TP concentrations showed little or no response during or after clearcutting until peatlands drainage (Ahtiainen 1992; Löfgren et al. 2009). Water-yield deficits drive the long-term declines in TP and calcium yields at watersheds on the MEF. Due to the forest conversion at the S6 watershed, water-yield deficits (relative to the preharvest aspen forests) were larger in magnitude than deficits at the S4 watershed where the upland aspen forest was clearcut a decade earlier and the forest was subsequently fertilized.

Hydrologic Responses to Peatland Harvests

The spatial variation of solute sources, sinks, and transformations between well-aerated upland soils and saturated anaerobic peatland soils affects how water and solutes are delivered to peatland laggs and streams. Solute sources and nutrient availability also differ between ombrotrophic bogs and minerotrophic fens that are embedded in regional groundwater aquifers. Bogs at the MEF are isolated from groundwater inputs due to hydrogeologic settings. The raised domes of ombrotrophic bogs maintain hydraulic gradients that drive water and solute fluxes to lagg zones that surround bogs in perched watersheds (Chapter 7). In contrast to bogs, groundwater flow transports chemically distinct groundwater into fens from regional aquifers (Bay 1967). Fen groundwater tables and streamflow from fens vary in response to interannual variation in groundwater inflow from aquifers. For these reasons, stream solute concentrations and yields may differ when forests on peatlands are clearcut rather than the uplands that drain to those peatlands. Two peatland forests were harvested at the MEF to: (1) determine effects of peatland clearcutting when the upland forests were left intact on water yields and solutes, and (2) determine whether effects on stream solutes differed between bog and fen watersheds. Peatland hydrology was not modified before or after the clearcutting by drainage, a common practice for increasing forest productivity on peatlands in northern Europe and Siberia (Vompersky 1976; Mitsch and Gosselink 1993).

Although solute concentrations have been measured at the S1 and S3 watersheds since the 1960s, there are insufficient data entered in electronic formats for rigorous evaluation of long-term responses to the two peatland harvest studies at the MEF. These analyses will be forthcoming. Here, we summarize and update findings of Knighton and Stiegler (1981). To quantify solute responses of bogs and fens to peatland forest harvesting, Knighton and Stiegler collected water samples from a small pool in each peatland near the stream outlet. They compared stream phosphorus yields to peatland clearcutting at the S3 fen and S1 bog watersheds from 1968 to 1978, compared chemical responses at the treatment watersheds to the S2 bog watershed, and speculated on processes that controlled differences in response between the two wetland types.

Bog Stripcut and Clearcut

Stripcutting retains a seed source to regenerate black spruce stands and is a best management practice to commercially harvest black spruce on peatlands in the Lake States region (Grigal and Brooks 1997). The black spruce forest on the 8.1 ha S1 bog was harvested during staggered whole-tree clearcutting in strips. Eight 30.5 m-wide strips were clearcut during the winter of 1969. The clearcut strips alternated with 45.7 m-wide strips of black spruce that served as the seed source to regenerate black spruce on the cut strips (Verry and Elling 1978). The mature trees that remained in the previously uncut strips were harvested during the winter from 1973 to 1974 after seedlings established on the previously cut strips (Verry and Elling 1978). The aspen and birch forest on the uplands was not harvested except for a small area that was used as a log yarding area.

Changes in water yields and bog water table levels were not detected after the 1969 or 1974 stripcuttings (Chapter 13). TP concentrations in bog waters showed no response to the 1969 stripcutting and then tripled during the year after the 1974 clearcutting (Figure 14.13). Elevated TP concentrations may have lasted longer but no samples were collected during 1975 and concentrations were similar to those at the S2 control by 1976. Clearcutting of the black spruce strip next to the S1 outlet where the samples were collected during 1974 also may be a factor that explains the brief concentration responses at the S1 watershed.

Knighton and Stiegler (1981) measured a large soluble phosphorus pool when surficial peat samples were incubated in a laboratory study (Figure 14.14a). They concluded that more phosphorus may have been transported from the S1 bog if the flow of water through the peat was greater. This finding implied that TP yields to streams may be elevated when wet years and higher flows follow bog clearcuttings.

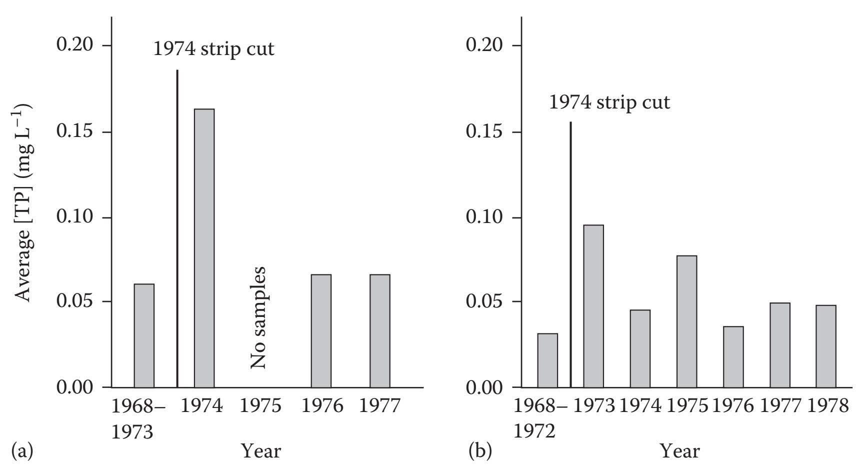

FIGURE 14.13
Average TP concentration of waters from the S1 bog (a) and S3 fen (b). (Modified from Knighton, M.D. and Stiegler, J.H., Phosphorus release following clearcutting of a black spruce fen and a black spruce bog, in *Proceedings of the Sixth International Peat Congress*, August 17–23, 1980, Duluth, MN, International Peat Society, Eveleth, MN, pp. 577–583, 1981.)

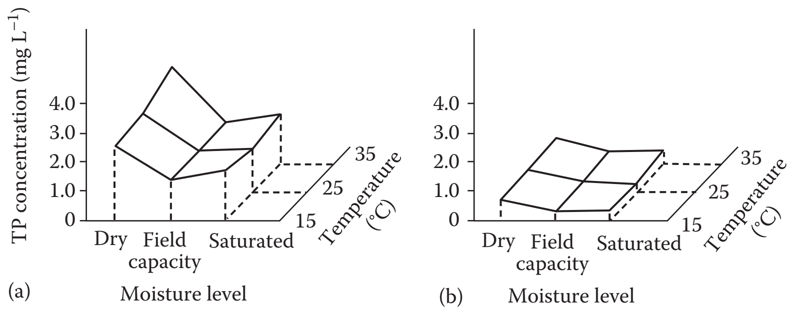

FIGURE 14.14
Concentration of TP in leachates from S1 bog (a) and S3 fen (b) peats. The leachates were extracted from peats that were incubated at different moisture and temperature levels. (Modified from Knighton, M.D. and Stiegler, J.H., Phosphorus release following clearcutting of a black spruce fen and a black spruce bog, in: *Proceedings of the Sixth International Peat Congress*, August 17–23, 1980, Duluth, MN, International Peat Society, Eveleth, MN, pp. 577–583.)

Knighton and Stiegler (1981) discussed how the increased magnitude of water table fluctuations, which Verry (1981) reported for S1 after the 1974 clearcutting, may increase phosphorus mineralization due to transient saturation and anoxia of surficial peats. We reject their hypothesis about controls on phosphorus mineralization after reassessing the entire 1961–2007 record of water table fluctuations. When the original data presented by Verry (1981) were reevaluated, we found that the amplitude of water table fluctuations did not change during or after clearcutting (Chapter 13) and therefore, could not have been a factor affecting phosphorus mineralization.

Fen Clearcut

The 18.6 ha forest on the S3 fen was commercially clearcut during the winter of 1972. The study was designed as site preparation for cedar restoration and to assess effects on stream chemistry. Black spruce and northern white cedar were seeded on the fen after clearcutting and slash burning. Black spruce, alder, and willow currently are the most abundant overstory species because survivorship of northern white cedar was poor in areas that were not protected from browsing by white-tailed deer (*Odocoileus virginianus*). The 34.0 ha upland aspen and birch forest was not harvested though a small portion northwest of the fen had previously been clearcut and replanted in red pine during the 1960s (Chapter 2).

Water yield via streamflow did not change after the peatland harvest. This finding is similar to that in the S1 study (Chapter 13). TP concentrations tripled relative to the pretreatment period from 1968 to 1972. Concentrations remained higher through 1978 when data collection ended. Knighton and Stiegler (1981) speculated that slash burning after clearcutting-induced increases in concentrations and yields after mineralization of phosphorus to soluble forms. The soluble phosphorus pool in surficial peats was less at S3 than at S1 (Figure 14.14b). In contrast to S1, phosphorus release to the stream may have been source (not transport) limited at the S3 fen. This finding suggests that stream TP concentrations are more sensitive to the biogeochemical processes that control phosphorus solubility in fen organic soils than interannual variation in water yields.

Summary

- The physical, chemical, and hydrological settings of lowland watersheds affect water and solute responses to forest harvesting. Because uplands route water through central peatlands that drain to streams, management strategies that affect forest cover and composition on uplands are coupled with the physical and biogeochemical processes that affect water movement and solute availability in peatlands.
- The increases in nitrate and ammonium concentrations in response to upland harvests at the MEF lasted no more than 1 year, whereas the water-yield increases lasted about a decade. The short duration nitrogen responses at the MEF contrast with changes of longer duration after experimental or natural vegetation changes at steep upland watersheds.
- The differences between nitrogen, TP, and calcium reflect fundamentally different biogeochemical processes that affect individual nutrient species when water yields changed after upland clearcuts in the watersheds of the MEF.
- Annual water yields did not change and TP concentrations increased after peatland harvesting on a bog and a fen at the MEF. The duration of increases in TP concentrations at the fen watershed exceeded that at the bog watershed.
- Internal peatland processes regulated the immobilization of upland inputs to laggs and the mobilization of solutes from organic soils after forest management or other forms of ecosystem change such as deposition of atmospheric pollutants. The sulfate addition study at S6 showed that peatlands have a disproportionate effect on the production and release of methylmercury, and that hotspots of biogeochemical transformations are important controls on solute fluxes from lowland watersheds.

References

Ahtiainen, M. 1992. The effects of forest clear-cutting and scarification on the water quality of small brooks. *Hydrobiologia* 243–244(1):465–473.

Bates, P.C., C.R. Blinn, A.A. Alm, and D.A. Perala. 1989. Aspen stand development following harvest in the Lake States Region. *Northern Journal of Applied Forestry* 6:178–183.

Bay, R.R. 1967. Ground water and vegetation in two peat bogs in northern Minnesota. *Ecology* 48(2):308–310.

Berguson, W.E. and D.A. Perala. 1988. Aspen fertilization and thinning research results and future potential. In: *Minnesota's Timber Supply: Perspectives and Analysis*, ed. A.R. Ek and H.M. Hoganson. Grand Rapids, MN: University of Minnesota, pp. 176–183.

Boelter, D.H. and E.S. Verry. 1977. Peatland and water in the northern lake States. General Technical Report NC-31. St. Paul, MN: USDA Forest Service.

Bormann, F.H. and G.E. Likens. 1967. Nutrient cycling. *Science* 155(3761):424–429.

Bormann, F.H., G.E. Likens, D.W. Fisher, and R.S. Pierce. 1968. Nutrient loss accelerated by clear-cutting of a forest ecosystem. *Science* 159(3817):882–884.

Branfireun, B.A., N.T. Roulet, C.A. Kelly, and J.W.M. Rudd. 1999. In situ sulphate stimulation of mercury methylation in a boreal peatland: Toward a link between acid rain and methylmercury contamination in remote environments. *Global Biogeochemical Cycles* 13(3):743–750.

Coleman-Wasik, J.K. 2008. Chronic effects of atmospheric sulfate deposition on mercury methylation in a boreal wetland: Replication of a global experiment. MS thesis. St. Paul, MN: University of Minnesota.

Gorham, E., S.E. Bayley, and D.W. Schindler. 1984. Ecological effects of acid deposition upon peatlands: A neglected field in "acid-rain" research. *Canadian Journal of Fisheries and Aquatic Sciences* 41(8):1256–1268.

Grigal, D.F. and K.N. Brooks. 1997. Forest management impacts on undrained peatlands in North America. In: *Northern Forested Wetlands: Ecology and Management*, ed. C.C. Trettin, M.F. Jurgensen, D.F. Grigal, and J.K. Jeglum. Boca Raton, FL: CRC/Lewis Publishers, pp. 379–396.

Hedin, L.O., G.E. Likens, and F.H. Bormann. 1987. Decrease in precipitation acidity resulting from decreased SO_4^{2-} concentration. *Nature* 325(6101):244–246.

Hornbeck, J.W., C.W. Martin, R.S. Pierce, F.H. Bormann, G.E. Likens, and J.S. Eaton. 1987. The northern hardwood forest ecosystem: Ten years of recovery from clearcutting. RP-NE-596. Broomall, PA: USDA Forest Service.

Jeremiason, J.D., D.R. Engstrom, E.B. Swain, E.A. Nater, B.M. Johnson, J.E. Almendinger, B.A. Monson, and R.K. Kolka. 2006. Sulfate addition increases methylmercury production in an experimental wetland. *Environmental Science and Technology* 40(12):3800–3806.

Knighton, M.D. and J.H. Stiegler. 1981. Phosphorus release following clearcutting of a black spruce fen and a black spruce bog. In: *Proceedings of the Sixth International Peat Congress*, August 17–23, 1980, Duluth, MN. Eveleth, MN: International Peat Society, pp. 577–583.

Kochenderfer, J.N. and G.M. Aubertin. 1975. Effects of management practices on water quality and quantity: Fernow Experimental Forest, W. Virginia. In: *Municipal*

Watershed Management Symposium Proceedings. Upper Darby, PA: USDA Forest Service, pp. 14–24.

Löfgren, S., R. Ring, C. von Brömssen, R. Sørensen, and L. Högbom. 2009. Short-term effects of clear-cutting on the water chemistry of two boreal streams in northern Sweden: A paired catchment study. *Ambio* 38(7):347–356.

Martin, C.W. and R.D. Harr. 1989. Logging of mature Douglas-fir in western Oregon has little effect on nutrient output budgets. *Canadian Journal of Forest Research* 19(1):35–43.

Martin, C.W., J.W. Hornbeck, G.E. Likens, and D.C. Buso. 2000. Impacts of intensive harvesting on hydrology and nutrient dynamics of northern hardwood forests. *Canadian Journal of Fisheries and Aquatic Sciences* 57(S2):19–29.

McHale, M.R., P.S. Murdoch, D.A. Burns, and B.P. Baldigo. 2008. Effects of forest harvesting on ecosystem health in the headwaters of the New York City water supply, Catskill Mountains. New York Scientific Investigations Report 2008–5057: US Geological Survey.

Mitsch, W.J. and J.G. Gosselink. 1993. *Wetlands*. New York: Van Nostrand Reinhold.

Neal, C., R. Fisher, C.J. Smith, S. Hill, M. Neal, T. Conway, G.P. Ryland, and J.A. Jeffrey. 1992. The effects of tree harvesting on stream-water quality at an acidic and acid-sensitive spruce forested area: Plynlimon, mid-Wales. *Journal of Hydrology* 135(1–4):305–319.

Rosén, K., J.-A. Aronson, and H.M. Eriksson. 1996. Effects of clear-cutting on streamwater quality in forest catchments in central Sweden. *Forest Ecology and Management* 83(3):237–244.

Sørensen, R., M. Meili, L. Lambertsson, C. von Brömssen, and K.H. Bishop. 2009. The effects of forest harvest operations on mercury and methylmercury in two boreal streams: Relatively small changes in the first two years prior to site preparation. *Ambio* 38(7):364–372.

St. Louis, V.L., J.W.M. Rudd, C.A. Kelly, K.G. Beaty, R.J. Flett, and N. Roulet. 1996. Production and loss of methylmercury and loss of total mercury from boreal forest catchments containing different types of wetlands. *Environmental Science and Technology* 30(9):2719–2729.

Stednick, J.D. 2008. Long-term water quality changes following timber harvesting. In: *Hydrological and Biological Responses to Forest Practices*, ed., J.D. Stednick. New York: Springer, Vol. 199, pp. 157–170.

Swank, W.T. 1988. Stream chemistry responses to disturbance. In: *Forest Hydrology and Ecology at Coweeta*, eds., W.T. Swank and D.A. Crossley. New York: Springer-Verlag, Vol. 66, pp. 339–357.

Timmons, D.R., E.S. Verry, R.E. Burwell, and R.F. Holt. 1977. Nutrient transport in surface runoff and interflow from an aspen-birch forest. *Journal of Environmental Quality* 6(2):188–192.

Urban, N.R. and S.J. Eisenreich. 1988. Nitrogen cycling in a forested Minnesota bog. *Canadian Journal of Botany* 66(3):435–449.

Urban, N.R., S.J. Eisenreich, and S.E. Bayley. 1988. The relative importance of denitrification and nitrate assimilation in midcontinental bogs. *Limnology and Oceanography* 33(6 part 2):1611–1617.

U.S. Environmental Protection Agency. 2009. National primary drinking water regulations. 816-F-09–004. Washington, DC: U.S. Environmental Protection Agency.

Verry, E.S. 1972. Effect of an aspen clearcutting on water yield and quality in northern Minnesota. In: *Proceedings of a symposium on Watersheds in Transition*, eds., S.C. Csallany, T.G. McLaughlin, and W.D. Striffler. Fort Collins, CO, June 19–22, 1972. Urbana, IL: American Water Resources Association, pp. 276–284.

Verry, E.S. 1975. Streamflow chemistry and nutrient yields from upland-peatland watersheds in Minnesota. *Ecology* 56(5):1149–1157.

Verry, E.S. 1981. Water table and streamflow changes after stripcutting and clearcutting an undrained black spruce bog. In: *Proceedings of the Sixth International Peat Congress*, August 17–23, 1980, Duluth, Minnesota. Eveleth, Minnesota: International Peat Society, pp. 493–498.

Verry, E.S. 1987. The effect of aspen harvest and growth on water yield in Minnesota. In: *Forest Hydrology and Watershed Management—Hydrologie Forestière et Aménagement des Bassins Hydrologiques*. Wallingford, UK: International Society of Hydrological Sciences, Vol. 167, pp. 553–562.

Verry, E.S. and A.E. Elling. 1978. Two years necessary for successful natural seeding in nonbrushy black spruce bogs. Research Note NC-229. St. Paul, MN: USDA Forest Service.

Verry, E.S. and D.R. Timmons. 1982. Waterborne nutrient flow through an upland-peatland watershed in Minnesota. *Ecology* 63(5):1456–1467.

Vompersky, S.E. 1976. Biological foundations of forest drainage efficiency. Washington, DC: USDA Soil Conservation Service and the National Science Foundation.