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20 Soil Carbon in Northern Forested Wetlands: Impacts of Silvicultural Practices

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Wetlands are typified by a hydrologic regime that induces anoxia in the soil during the growing season, and by vegetation that is adapted to O_2-deficient soils. This combination of hydrology, soil, and vegetation gives rise to site conditions that greatly affect ecosystem processes and landscape functions. Wetlands are now widely recognized for the important functions and values they provide, and concern exists regarding the integrity of the wetland resource and the sustainability of those attributes (Conserv. Found., 1988). Soil C is inextricably linked to many wetland functions including element cycling and storage (Hemond, 1980; Clymo, 1984a; Verhoeven, 1986; Damman, 1990; Verry & Urban, 1992), water quality (Hemond, 1990; Marin et al., 1990; David & Vance, 1991), hydrology (Boelter & Verry, 1977; Ingram, 1983), and vegetation dynamics (Vasander, 1982; Laine et al., 1992). Carbon accumulation and storage in wetland soils also are particularly important ecosystem processes that affect the global C budget. Estimates of C storage in northern peatlands alone range from 257 Pg (Eswaran et al., 1993) to 455 Pg (Post et al., 1982; Gorham, 1991), which comprise 16 and 33% of the respective estimates of the global soil C pool. Eswaran et al. (1993) reported an additional 51 Pg C in mineral soils classified in the aquic suborders, which are characteristically wetland soils. These estimates of soil C in wetlands are constrained by inadequate data on wetland area, organic horizon depth, and soil bulk density (Schlesinger, 1986; Eswaran et al., 1993). However, they demonstrate the importance of wetland soils as a global C sink.

The proportion of northern wetlands which are forested is unknown; however, statistics from five countries demonstrate that forests comprise a significant

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Table 20–1. Area of wetlands and forested wetlands occuring North America and Eurasia, and the proportion of commercial forest land occuring as wetlands.

Country	Total wetland area	Forested wetland area	Proportion of commercial forest land in wetlands	Source†
	$*10^6$ ha	$*10^6$ ha	%	
Sweden	8.4	3.4§	20	1
Finland	10.4‡	?	47	2
USA	42.3	21.1	9	3,4
USSR	?	245.0	22	5
Canada	1110.0‡	?	?	6

†1 = Hanell (1991), 2 = Paivanen (1991), 3 = Frayer (1991), 4 = Abernethy and Turner (1987), 5 = Vompersky (1991), 6 = Haavisto and Jeglum (1991).
‡Estimate is for peatland area only, it does not include mineral soil wetlands.
§Does not include noncommercial forest land.

proportion of the wetland resource, and that commercial wood product operations are an important use of those wetlands (Table 20–1). In the northern USA forests comprise 51 to 66% of the total wetland area (Cubbage & Flather, 1993). The wetland resource in individual states may comprise a larger proportion of the commercial forest land area than the national average; for example in Michigan, Minnesota, and Wisconsin the proportions are 22, 29, and 16%, respectively (Smith & Hahn, 1987, 1989; Hahn & Smith, 1987). Estimates of forested wetlands within the vast Canadian peatlands are not available, however Haavisto and Jeglum's (1991) assessment of commercial forests operations demonstrates that wetlands are important. Similarly, in Scandinavia and the former USSR, intensive management of forested wetlands has been an integral component of commercial forest operations (Hanell, 1991; Paivanen, 1991; Vompersky, 1991).

Management regimes on these northern, forested wetlands vary considerably, ranging from harvesting alone to intensive silvicultural systems designed to maximize wood fiber production. Because of concern about wood utilization efficiency and sustained site productivity, management of forested wetlands typically involves harvesting, site preparation, reforestation, and stand tending. Drainage systems also are a common part of forest management prescriptions, particularly in the southeastern USA, Finland, and the former USSR (Allen & Campbell, 1988; Paivanen, 1991; Vompersky, 1991). Management prescriptions for forested wetlands in the northern USA and Canada do not usually involve drainage, although there is ongoing research and applications using such systems (Jeglum & Overend, 1991).

Common silvicultural practices used in northern forested wetlands include clearcut harvesting (stem-only and whole-tree), site preparation (mounding), replanting, and fertilization. How do these practices affect soil C dynamics in wetlands? What are the associated effects on ecosystem processes? What is the resultant effect on soil C balance at different spatial scales? These questions and others related to the impacts of silvicultural operations on northern forested wetlands have not been adequately studied. Johnson (1992) recently completed a review of forest management impacts on soil C in forests, but only one of the approximately 100 papers reviewed addressed northern forested wetlands. Johnson (1992) reported that harvesting alone generally caused little net change in upland soil C, while more intensive disturbances associated with site preparation usually reduced soil C levels. Organic matter decomposition in wetland soils is sensitive to the same biotic and abiotic factors which affect upland soils. However, the effect of silvicultural disturbances in wetlands may not be the same because of differences in silvicultural practices and site conditions, especially the large C content and anoxic properties of wetland soils.

Forested wetlands are an important component of northern landscapes. Ecological processes which affect wetland functions (e.g., modification of hydrologic regimes, water quality, habitat, or C storage) and societal values (e.g., aesthetics, recreation, forest products) are sensitive to the biotic and abiotic properties of the wetland and its C cycle. Accordingly, an understanding about the effects of forest management practices on C dynamics is necessary if those practices are to be used in a way that does not degrade wetland functions and values. This paper has three objectives: (i) review the distribution and function of C in northern forested wetlands, (ii) review how different silvicultural practices affect soil C levels, and (iii) consider the potential for recovery of soil C following disturbance by silvicultural practices.

WETLAND PROPERTIES AND PROCESSES

There are many classification systems and terms which are used to describe forested wetlands (Mader, 1991). Unfortunately, none of these are ideal for characterizing soil C content or the affect of soil C on wetland functions, such as hydrology, water quality, or nutrient cycling. This is because classification systems either lack specificity with respect to soil and hydrologic properties, or consensus on definitions and nomenclature. Generally, wetlands are classified on the basis of landform, vegetation, depth of peat accumulation, and trophic status. For purposes of this paper we distinguish three categories of forested wetlands: (i) mineral soil, (ii) histic-mineral soil, and (iii) peatlands (Histosols). These categories reflect the degree to which the wetlands have accumulated soil C (Table 20–2).

Mineral soil wetlands commonly occur in landforms which have a fluctuating water table and extended aerobic periods during the growing season. Organic matter does not accumulate in these soils because decomposition is not sufficiently limited by saturation. Accordingly, C flux is approximately balanced between inputs and outputs. Mineral soil wetlands are morphologically similar to upland soils except for the presence of hydric soil indicators (e.g., low chroma mottles within the upper 25 cm, seasonal high water table or flooding). Peatlands (Histosols) are wetlands that have a thick (>40 cm) accumulation of surface organic matter. Peatlands have traditionally been characterized by the source of water which sustains the plant community. Fens are minerotrophic peatlands supplied from groundwater and precipitation. In contrast, bogs are ombrotrophic, and receive water primarily from precipitation (Table 20–2). These water quality and hydrologic characteristics of peatlands have a significant effect on the vegetation and peat type. However, the present vegetation does not necessarily reflect

Table 20–2. Classification and characteristics of northern forested wetlands reflecting accumulated soil C.

Wetland property or attribute	Wetlands soil class			
	Mineral soil	Histic–mineral soil	Peatlands (Histosols)	
			C-accumulating wetlands	
Common wetland nomenclature†	Swamp, palustrine forest	Swamp, mire, palustrine forest	Fen, treed fen, palustrine forest	Bog, treed bog, palustrine forest
Common tree species	*Acer rubrum* L., Ulmus americana, *Fraxinus pennsylvanica* Marshall, *F. nigra* Marshall, *Populus tremula* L.	*Picea mariana* (Mill.) B.S.P., *Larix laricina* (Du Roi) K. Koch., *Pinus banksiana* Lamb., *Acer rubrum* L.	*Thuja occidentalis* L., Acer rubra, *Pinus sylvestris* (L.), *Picea abies* (L.) Karst	*Picea mariana* (Mill.) B.S.P., *Larix laricina* (Du Roi) K. Koch., *Pinnus sylvestris* (L.)
	Landform/hydrology classification‡			
Limnogenous LF: riparian zones WS: flooding, groundwater, precipitation	X	X		
Topogenous LF: topographic depressions or lows WS: groundwater, precipitation	X	X	X	
Ombrogenous LF: raised organic soils WS: precipitation				X

Table 20–2 (continued).

Wetland property or attribute	Forested wetlands			
		Water characteristics		
Trophic status	Minerotrophic	Minerotrophic	Minerotrophic	Ombrotrophic
Soil water pH§	5.0–6.5	4.0–5.5	5.7–7.0	3.8–4.1
		Soil properties		
Organic horizon thickness	0–5 cm	5–40 cm	40+ cm	40+ cm
Primary component of the O horizon	Tree litter	Bryophytes, sedges, tree litter	Bryophytes, sedges, grasses, tree litter	Bryophytes, tree litter

†Nomenclature used to describe forested wetlands [Moore & Bellamy, 1974: Cowardin et al., 1979, Natl. Wetlands Working Group (NWWG), 1988].
X = Common property or characteristic.
‡Categorization of wetlands types by landform (LF) and source of water sustaining the wetland (WS) (adapted from Damman & French, 1987).
§From Boelter and Verry (1977), and Damman and French (1987).

the composition of accumulated organic matter because it is a function the current hydrologic conditions, which may not have been the same during successional development of the wetland (Swanson & Grigal, 1989).

Intermediate between mineral soil wetlands and peatlands are the histic-mineral soil wetlands, which have a histic epipedon 5 to 40 cm thick. These wetlands are nascent peatlands, functioning as a C sink and having hydrologic and biogeochemical processes that are strongly affected by organic matter accumulation. Swamp is a common term used in conjunction with this wetland soil type (Table 20–2). Histic-mineral soil wetlands are not generally considered a peatland (Histosol) until the accumulation of surface organic materials is greater than 40 cm (Soil Survey Staff, 1975; NWWG, 1988).

Carbon Distribution

Soil

Soil C represents the largest C pool in northern forested wetlands, ranging from 50 to 1300 t ha^{-1} (Fig. 20–1). The amount of C stored in a particular wetland depends on the C concentration, thickness, and bulk density of the soil horizons. The rate of C accumulation in peatlands is dependent on wetland productivity, stage of successional development, and organic matter decomposition (Clymo, 1984b). Average C accumulation rates in peatlands range from 0.1 to 0.8 t C ha^{-1} yr^{-1} (Fig. 20–1).

Soil C in forested peatlands and histic-mineral soil wetlands occurs primarily as partially decomposed organic matter. Accumulated organic matter (i.e., peat) in these wetlands has a direct effect on soil hydrologic properties (Boelter & Verry, 1977; Ingram, 1983; Glaser, 1987; Siegel & Glaser, 1987; Verry, 1988). The degree of peat decomposition is the critical parameter affecting water retention, bulk density, and hydraulic conductivity (Boelter, 1969) and C mineralization (Wieder & Yavitt, 1991; Hogg, 1993). Two distinct zones are recognized within a peatland to distinguish areas of biological activity, hydrology, and edaphic properties (Fig. 20–2). The acrotelm, or surface zone, is characterized by a variable soil water content, a relatively high hydraulic conductivity, and active nutrient cycling and decomposition. The thickness of the acrotelm is defined by the lower limit of natural groundwater fluctuations. In contrast, the catotelm is permanently saturated, the peat is generally highly decomposed, biological activity is very low, and water flow is minimal due to a very low hydraulic conductivity (Ingram, 1983). In histic-mineral soils the surface organic horizons would be analogous to the acrotelm. These terms are not used in reference to mineral soil wetlands.

Vegetation

Northern forested wetlands contain diverse plant communities, which have developed in response to climate, hydrology, and substrate conditions (Teskey & Hinkley, 1978; Hofstetter, 1983; Glaser, 1987; NWWG, 1988). The occurrence and productivity of these plant communities vary with respect to water chemistry, water flow, water table depth, and climate (Richardson, 1978; Damman & French, 1987; Damman, 1990). Tree species which are common to northern

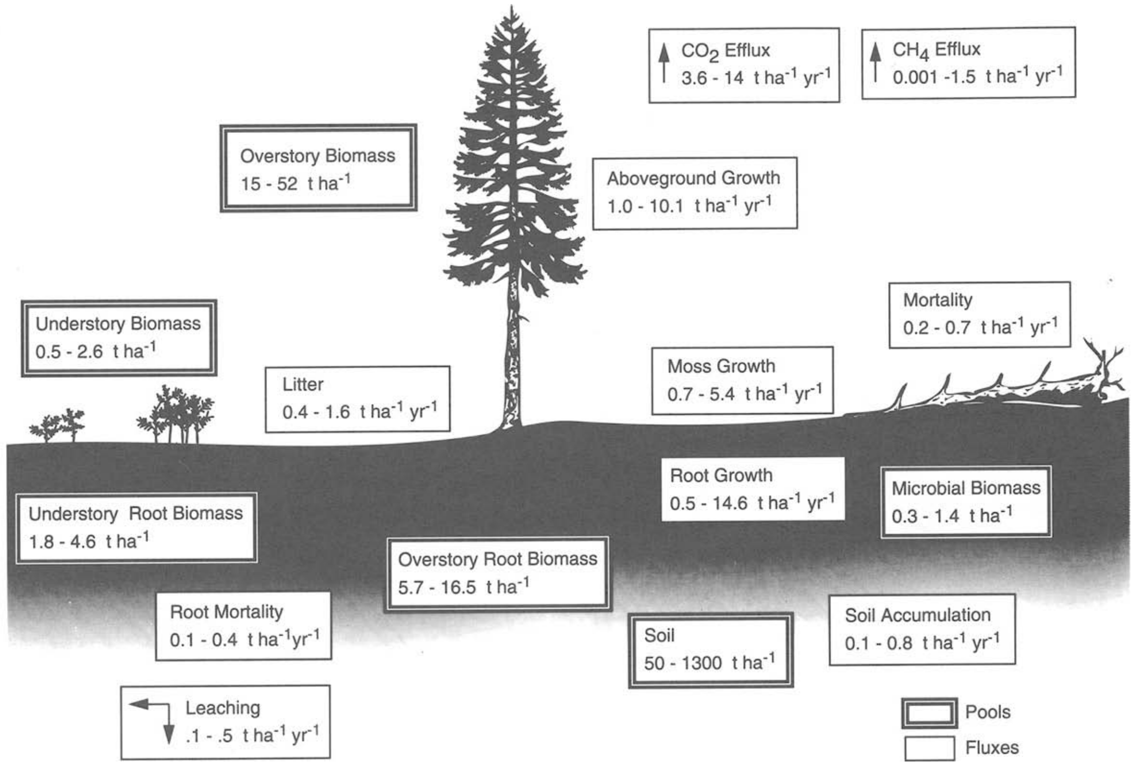

Fig. 20–1. Distribution and fluxes of C in northern forested wetlands. The values encompass the range reported by Reader and Stewart (1972), Reiners (1972), Parker and Schnieder (1975), Reader (1978), Richardson (1978), Brinson et al. (1981), Kenttamies (1981), Vasander (1982), Van Cleve et al. (1983), Grigal (1985), Grigal et al. (1985), Armentano and Menges (1986), Williams and Sparling (1988), Brock and Bregman (1989), Haland and Braekke (1989), Moore (1989), Rochefort et al. (1990), Gorham (1991), Kim and Verma (1992), Verry and Urban (1992), Bartlett and Harriss (1993).

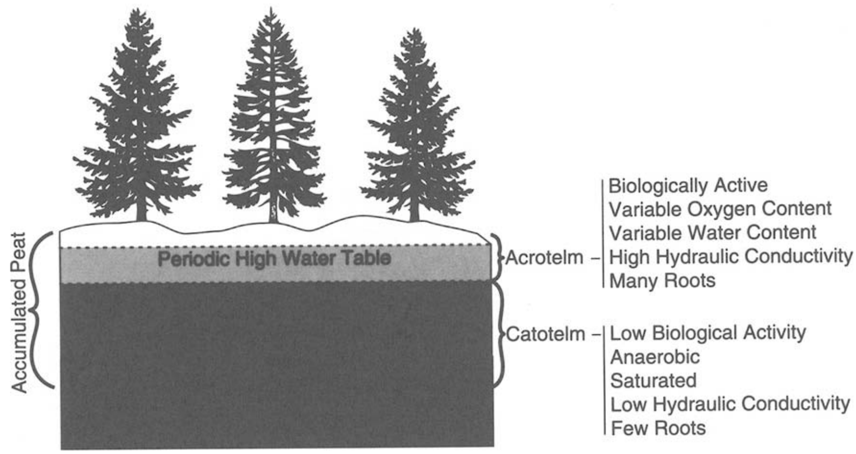

Fig. 20–2. Location and functional properties of the acrotelm and catotelm in peatlands (after Damman & French, 1987).

wetlands are presented in Table 20–2. The forest overstory comprises the largest biomass C pool, which is typically 20 to 100 times greater than C in the understory vegetation (Fig. 20–1). There is little available information on the belowground biomass in forested wetlands (Santantonio et al., 1977). The few studies conducted indicate that roots comprise 10 to 30% of the total C biomass pool. Net aboveground productivity of northern forested wetlands range from 1 to 10 t ha^{-1} yr^{-1}, which is generally lower than other temperate freshwater wetlands and upland forests (Richardson, 1978). Understory vegetation is a significant proportion of total wetland net productivity, comprising 5 to 50% of aboveground growth.

Sphagnum is perhaps the single most important plant affecting northern forested wetlands. While there are several hundred *Sphagnum* species, approximately 20 are important in the development of peatlands (Clymo, 1987). The occurrence of different *Sphagnum* species reflects microsites of varying pH conditions, available nutrients and water, the light (Andrus, 1986). *Sphagnum* affects wetland processes by increasing the water holding capacity of the peat (Boelter & Verry, 1977), acidification (Clymo, 1984a), and peat accumulation (Clymo, 1984b). The slow decomposition rate and high water retention characteristics of *Sphagnum* are important mechanisms of paludification and development of deep peatlands (Moore & Bellamy, 1974). The acidification and exchange capacity of *Sphagnum* affects plant distribution and nutrient cycling within the wetland (Hemond, 1980; Crum, 1988). *Sphagnum* also is an important component of net wetland productivity, often exceeding the productivity of trees (Grigal, 1985; Brock & Bregman, 1989).

Organic Matter Decomposition

The development of histic-mineral soil wetlands and peatlands requires that the rate of organic matter production exceed the rate of decay. Organic matter accumulation occurs until a steady state exists between production and decay within the organic horizons (Clymo, 1992). Soil water content, temperature, acidity, and substrate quality are the primary factors affecting the rate of organic matter decomposition in wetlands (Moore & Bellamy, 1974; Clymo, 1984b; Hogg, 1993). Estimates of long-term C accumulation rates, derived from ^{14}C dated peat cores, range from 8 to 41 g C m^{-2} yr^{-1}, with an average of 21 g C m^{-2} yr^{-1} (Tolonen et al., 1992). However, these accumulation rates do not account for sustained decay throughout the peat column. When this factor is considered, using Clymo's (1984b) decomposition model, the actual long-term C accumulation rate is 33% lower than the estimate derived from ^{14}C peat cores (Tolonen et al., 1992). Peatland surface accretion rates range from 0.1 to 4.3 mm yr^{-1}, with values commonly ranging between 0.5 to 2 mm yr^{-1} (Tolonen et al., 1992). Peat accumulation is not constant with time, the rate of accumulation declines as the thickness of peat increases (Clymo, 1984b).

Organic matter decomposition is most active in the surface soil or acrotelm, which has higher temperature and O_2 content, and a larger supply of labile C than deeper in the soil (Clymo, 1984b; Lieffers, 1988; Hogg, 1993). However, in peatlands having a hummock/hollow relief organic matter decomposition rate in the surface O horizons has been reported to be greater in the hummocks due to increased aeration (Farrish & Grigal, 1985), or less in hummocks due to higher acidity and water table fluctuations (Rochefort et al., 1990). Hogg et al. (1992) demonstrated that frequent wetting and drying cycles increase organic matter decomposition, as compared to extended dry periods alone.

Substrate quality and site conditions also are important factors affecting decomposition in forested wetlands. Hogg (1993) reported that CO_2 released from *Sphagnum* peat from near the surface was greater than older peat from 10- to 12-cm depth, when both peats were incubated under the same abiotic conditions. A primary factor contributing to the lower decay rate of deeper peat was its increased proportion of nonhydrolysable C (Jorgensen & Richter, 1992). Nutrient status of peatlands affects organic matter decomposition, with fens exhibiting higher decomposition rates than bogs (Farrish & Grigal, 1988). Interactions among wetland conditions and substrate quality also affect C mineralization from peat (Wieder & Yavitt, 1991).

Efflux of C from wetlands occurs both as gas and liquid, with gaseous outputs of C (CO_2 and methane [CH_4]) being the dominate pathway (Fig. 20–1). Soil temperature is a primary factor affecting organic matter decomposition. Correspondingly, CO_2 and CH_4 flux from wetlands is positively correlated to soil temperature (Kim & Verma, 1992; Bartlett & Harriss, 1993; Dunfield et al., 1993). Gas fluxes are highest during the summer, corresponding to the period of warmest soil temperatures. Although organic matter decomposition is low during the winter, it can be a significant proportion of the annual decay (Taylor & Jones, 1990; Braekke & Finer, 1990). Dise (1992) reported that CH_4 emissions during the winter can comprise 4 to 21% of the annual flux. Microbial population size and substrate availability also affect soil gas flux (Yavitt et al., 1987).

Soil aeration is another important factor affecting organic matter decomposition, and in peatlands it is strongly influenced by water table depth (Mannerkoski, 1985, p. 1–190). Soil aeration has a significant effect on the proportion of CO_2 and CH_4 in the total gas flux from a wetland (Moore & Knowles, 1989; Freeman et al., 1993). Methane oxidation is very rapid in aerated wetland soils, hence CH_4 emission is low when the water table recedes below the soil surface (Bartlett & Harriss, 1993). In contrast, CO_2 flux is stimulated when the water table is lowered (Kim & Verma, 1992; Freeman et al., 1993).

Carbon output in waters emanating from northern wetlands has long been considered to be high, but this C flux has not been adequately considered in estimating total C losses from wetlands, or in terms of watershed functions. Hemond (1990) hypothesized that riparian wetlands were the primary source of dissolved organic carbon (DOC) in streams in recently glaciated terrain. Other studies have recently established a relationship of DOC in streams with the occurrence of adjacent wetlands (Heikkinen, 1990; Koprivnjak & Moore, 1992). Organic acids comprise a significant proportion of the DOC, and are the principal anions providing the charge balance in waters from northern forested wetlands (Urban et al., 1987; Sallantaus, 1988). However, the transport of organic acids and associated cations are sensitive to the hydrologic flow path, especially when mineral soils are involved (Cronan, 1990; Heikkinen, 1990; David & Vance, 1991).

EFFECTS OF SILVICULTURAL PRACTICES

Wetness is a major constraint on silvicultural operations and vegetation productivity in northern wetlands. Accordingly, silvicultural practices are designed to ameliorate the wetness limitations. Drainage has been used extensively to manage water table levels, and it is most commonly used in conjunction with harvesting, site regeneration, and site improvement prescriptions (Paavilainen & Paivanen, 1988). The following discussion reviews the effects of those practices on soil C levels and the corresponding impact to soil and water processes.

Forest Drainage

Drainage systems are used in many forested wetlands in northern Europe and Asia (Heikurainen et al., 1978; Jeglum & Overend, 1991; Vompersky, 1991). Drainage has been a particularly important practice in Finland and the former USSR, where over 5.5 million hectares have been drained in each country (Paivanen, 1991; Vompersky, 1991). This practice has not been prevalent in northern forested wetlands of North America, although there is a history of drainage dating to the early 1900s (Zon & Averell, 1929) and considerable opportunities exist for expanded use in Canada (Haavisto & Jeglum, 1991). Silvicultural drainage systems typically use open ditches in a variety of configurations that lower the water table 30 to 73 cm below the soil surface (Paivanen & Wells, 1977; Terry & Hughes, 1978; Braekke, 1983).

Until recently the emphasis of studies on drained wetlands focused primarily on forest growth responses and nutrient cycling, with little regard to changes in soil C levels. Significant changes in soil C following drainage are expected due to induced changes in both biotic and abiotic conditions. However, only recently have studies of previously drained peatlands documented reductions in soil C (Table 20–3). These changes in soil C following drainage do not necessarily

Table 20–3. Change in soil C levels following silvicultural practices in northern forested wetlands.

Wetland type	Site characteristics	Silvicultural treatments	Assesment interval	Net change in soil C	Mean annual C loss	Source†
			yr	g C m^{-2}	g C m^{-2} yr^{-1}	
Peatlands						
Fen	Minerotrophic, tall sedge and tall sedge pine fens	Drainage	30	+1300–1800	+40–60	1
	Herb-rich pine fen, peat 1.7 m	Drainage	20	–650	–33	2
	Topogenous poor fen, peat 0.8–0.09 m.	Drainage, fertilization, planted	28	–3400‡	–120	3
Bog						
	Oligotrophic, low-sedge bog and cottongrass pine bog	Drainage	30	–1200 to 4800	–40 to –160	1
	Ombrotrophic bog	Drainage, fertilization	1	—	–700§	4
	Ombrotrophic bog, peat 2.7–3.7 m	Drainage, fertilization, seeded	27	–7850‡	–290	3
Histic-mineral soil						
	Conifer swamp, 15 cm *Sphagnum* epipedon	Whole-tree, harvest	1.5	–1800	–1200	5
	Conifer swamp, 15 cm *Sphagnum* epipedon	Whole-tree, harvest, mound, planted	1.5	–2200	–1500	5

†1 = Laine et al. (1992), 2 = Sakovets and Germanova (1992), 3 = Braekke (1987), 4 = Silvola (1988), 5 = Trettin et al. (1992).
‡A factor of 0.5 was used to convert original data which was presented as organic matter to C.
§Based on CO_2 flux.

reflect total C loss or gain, but represent a net change that includes both inputs and outputs. Bogs generally exhibited greater soil C losses than fens, however, the cause of this difference has not been addressed. Drainage effects on soil C levels in forested, histic-mineral soil wetlands have not been documented, however, a reduction in soil C would be expected. Reductions in soil C from forest drainage is generally less than the 7.8 to 11.2 t ha^{-1} yr^{-1} which Armentano and Menges (1986) reported for agriculturally drained wetlands. Drainage does not always result in net reduction of soil C, although changes in soil C pools are difficult to detect when based on peat depth (Anderson et al., 1992). Increased soil C following drainage was documented in one study (Laine et al., 1992), which was attributed to improved site quality affects on belowground biomass production.

Pathways of C loss following drainage include both gas emissions and leaching. Carbon dioxide flux increases following drainage (Lahde, 1969; Silvola, 1986), while CH_4 flux is usually reduced (Bartlett & Harriss, 1993). Armentano and Menges (1986) reported C loss, as CO_2, from drained forested wetlands in the northern USA as approximately 2.8 t ha^{-1} yr^{-1}. That amount is within the range of C losses reported from forested peatlands which were measured by differences in soil C pools (Table 20–3). Increased DOC export from drainage of northern, forested peatlands also has been widely reported (Table 20–4; Heikurainen et al., 1978; Ahtiainen, 1988; Lundin, 1988; Heikkinen, 1990). However, reductions in DOC also have been found (Kenttamies, 1981; Hovi, 1988). Exports of DOC following drainage are a function of water balance alterations (Sallantaus, 1988). Drainage of fens reduces the normal influx of groundwater thereby reducing the total water flux through the wetland. In contrast, hydrologic inputs to bogs are not as affected by drainage because precipitation is the primary water source. The specific hydrologic response in drained wetlands is dependent on the drainage system design, regional hydrology, geomorphic setting, vegetation type, soil properties, and climate (Starr & Paivanen, 1986).

Reductions in soil C pools after draining peatlands, either as gases or DOC, is a direct result of increased organic matter decomposition due to changes in soil aeration and temperature regimes (Lieffers & Rothwell, 1987; Lieffers, 1988). Drainage systems increase the depth to water table thereby effectively increasing the volume of aerated soil (Munro, 1984; Mannerkoski, 1985, p. 1–190). The effects of drainage on soil temperature are varied, with reports of no change, to increases of 2 to 4°C in the upper 30 cm of peat (Table 20–5). Although either increased aeration or temperature alone would have a stimulatory effect on organic matter decomposition, the combined effect would have the most significant impact (Oades, 1988).

Harvesting

Wood fiber products in northern forested wetlands are usually harvested by clear-cutting, either using whole-tree or stem-only harvest systems. Clearcutting offers efficiencies in harvest operations, and it is an effective method for regenerating wetland forests. Both types of clearcutting systems reduce aboveground biomass, but whole-tree harvesting involves a greater removal of smaller

Table 20–4. Dissolved organic C export and change in water concentrations following silvicultural practices in northern forested wetlands.

Wetland type	Site description	Silvicultural treatments	Assessment interval	DOC export	Change in DOC†	Source‡
			yrs	g C m^{-2}	mg L^{-1}	
Peatlands						
Fen	Tall sedge fen, peat 3.2 m	Drainage	4	—	–0.5	1
	minerotrophic fen	Drainage	30	14	+18.0	2
		Undrained	30	8	—	2
Bog	Herb-rich pine fen	Drainage	20	—	+48.0§	3
	Ombrotrophic bog	Drained	30	16	+6	2
		Undrained	30	12		2
Histic-mineral soil	Conifer swamp, 15 cm *Sphagnum* epipedon	Whole-tree harvest	4	—	–2§	4
	Conifer swamp, 15 cm *Sphagnum* epipedon	Whole-tree harvest, mound, plant	4	—	+3§	4

†Change in concentration as compared to undisturbed reference site.
‡ 1 = Lundin (1988), 2 = Sallantaus (1992), 3 = Sakovets and Germanova (1992), 4 = J.W. McLaughlin (unpublished data).
§Soil water sampled at approximately 1-m depth.

Table 20–5. Change in midsummer soil temperature following silvicultural practices.

Wetland type	Site description	Silvicultural treatments	Assessment interval	Soil depth	Temperature change†	Source
			years	cm	°C	
Peatlands						
Fen	Small sedge mire, peat depth 2.4–3.2 m	Drainage	4	10, 20, 50, 75	0	1
	Treed fen, peat depth = 1.3 m	Drainage	2	10	+3	2
		Drainage	2	30	+2	2
	Conifer swamp, 6 cm histic epipedon	Drained,	11	5	+2.2	3
		Harvested	11	50	+2.5	3
Histic-mineral soil	Conifer swamp, 15 cm *Sphagnum* epipedon	Whole-tree	1	5	+2.6	4
		Harvest	1	25	+2.7	4
	Conifer swamp, 15 cm *Sphagnum* epipedon	Whole-tree harvest, bed, plant	1	5	+5.8	4
			1	25	+4.6	4

†Mean temperature difference, over the assessment interval, between the disturbed and undisturbed reference site.
‡ 1 = Latja and Kurimo (1988), 2 = Lieffers (1988), 3 = Kubin and Kemppainen (1991), 4 = Trettin and Jurgensen (1992).

diameter material, sometimes including foliage (Mann et al., 1988). While many studies have been conducted comparing the effects of these two harvesting systems on biomass pools in upland soils we found no such studies on northern wetlands.

Musselman and Fox (1991) reported that clearcutting on all soils in the USA could reduce soil C pools 25 to 50%. Although a recent review of 13 studies showed that harvesting may either increase or decrease soil C in upland soils, the net effect was usually less than a 10% change in mineral soil C content (Johnson, 1992). Changes in the C content of the forest floor also increased or decreased depending on the quantity of logging residue. Effective maintenance of soil C would most likely occur for stem-only harvesting where a proportion of the overstory biomass remains on site. Although leaving woody residue may have a short-term effect on the size of the soil C pool, it is unlikely that the residue additions would change the soil C equilibrium levels, nor does it represent an effective mechanism for reducing nutrient losses (Fahey et al., 1991; Titus & Malcolm, 1992).

Only one study was found which considered the effect of harvesting alone on soil C pools in northern wetlands, since most silvicultural prescriptions in northern forested wetlands involve drainage, site preparation, or both (Table 20–3). Trettin et al. (1992) reported a 28% reduction in soil C 18 mo after whole-tree harvesting a histic-mineral soil swamp. The annualized C loss following harvesting alone was greater than that reported for peatlands which were drained (Table 20–3). However, these losses reported for peatlands represent a net change in soil C that includes C inputs and outputs over several decades; in contrast, the short-term response reported for the histic-mineral soil wetland involves C loss predominately.

The impact of harvesting on soil C in peatlands and histic-mineral soil wetlands is expected to be different from upland soils. Unlike upland forests, these forested wetlands are characterized by thick surface O horizons, which have developed because of slow organic matter decomposition rates. Changes in soil temperature, aeration, and nutrient availability after harvesting increase organic matter decomposition rates (Trettin & Jurgensen, 1992) and cause reductions in soil C levels. Higher soil temperature following harvesting is a primary factor contributing to increased organic matter decomposition (Table 20–5). This effect occurs rapidly following harvesting and can persist for more than 10 yr (Kubin & Kemppainen, 1991).

Water table depth greatly affects soil aeration, hence changes in the water table following harvesting also would impact organic matter decomposition. The affect of water table depth on aeration is largely controlled by the hydrologic regime, but it also varies temporally (Munro, 1984) and spatially within the wetland (Trettin, unpublished data). Depth to the water table in undrained wetlands, where the water table normally falls to at least 30 cm below the soil surface during the growing season, will typically decrease following harvesting (Verry, 1988). On these sites, anoxia induced by the higher water table would limit the rate of organic matter decomposition. In undrained wetlands where the water table normally remains within 30 cm of the soil surface, the amplitude of water table fluctuations after harvesting will increase, but the average depth will remain the same (Verry, 1986, 1988). In drained peatlands the water table typically rises following harvesting (Paivanen, 1982).

Mineralization of woody residue and soil organic matter following harvesting of forested wetlands increases DOC and mineral nutrient leaching (Ahtiainen, 1988; Hughes et al., 1990). The leached C and nutrients can have a direct effect on the productivity and acid-base chemistry of nearby streams. Huttunen et al. (1988) measured increases in both algal and bacteria primary production following clearcutting of forested wetlands in Finland, but no effect was found when an uncut buffer strip was left adjoining the stream. While DOC exports are an important pathway for C loss in all harvested peatlands, the flux may be dependent on the type of harvest system used. Whole-tree harvest removes more C as biomass, leaving less residue for subsequent decay. Stem-only harvest leaves more residue on the site, which is subject to decay and release as CO_2, CH_4 and DOC. Whether there is a difference in total C loss between these two types of harvesting systems in wetlands has not been studied.

Site Preparation

Site preparation is used on forested wetlands to establish sites for planting seedlings, and for the control of competing vegetation. Numerous site preparation systems are used on northern wetlands, which mostly involve some form of mounding (Sutton, 1993). Mounds are created either by disking or excavation. These mounds provide an elevated planting site and mitigate against increased water table levels following harvesting. Disked mounds (i.e., bedding) mix surface soil horizons into the center of the planting bed, while mounds created by excavation typically do not mix the surface soils (Attiwill et al., 1985; Sutton, 1993).

The degree of soil mixing as a result of site preparation greatly affects organic matter decomposition. Ross and Malcolm (1988) found in a laboratory study that mixing peat with mineral soil under both aerated and saturated conditions increased organic matter decomposition and nutrient mineralization in accordance to the degree of mixing and soil aeration levels. Results from a field study comparing harvesting and site preparation demonstrated that bedding caused greater reductions in soil C pools, as compared to harvesting alone (Table 20–3). These C losses occurred within the first 18 mo following harvest, and were a result of increased organic matter decomposition in the O horizons. Measurements 30 mo later showed no changes in soil C among site preparation treatments (J.W. McLaughlin, unpublished data). The loss of soil C as a result of bedding was greater than annual losses reported for drained peatlands or harvested histic-mineral soil wetlands (Table 20–3).

As was shown earlier for timber harvesting, soil temperature increases as a result of site preparation and is a primary factor affecting organic matter decomposition (Table 20–5). This is especially true for mounding, which increased the rate of soil warming in the spring and increased the maximum midsummer soil temperature in a histic-mineral soil wetland (Trettin & Jurgensen, 1992). Sutton (1993) also reported that mounding increased soil temperature.

The most pronounced effect of site preparation on organic matter decomposition occurs when both soil aeration and temperature are affected (Hogg et al., 1992). Elevated soil temperature increased decomposition of peat under aerated conditions, but temperature did not have an obvious effect on organic matter decomposition when saturated peat was incubated in a laboratory study (Hogg et al., 1992). A similar temperature and water regime interaction on the cellulose decay was reported by Donnelly et al. (1990). Field measurements of organic matter decomposition following site preparation of a histic-mineral soil wetland found increased decomposition in the upper 25 cm of soil (Fig. 20–3) which was attributed to increased temperature and aeration (Trettin & Jurgensen, 1992). Harvesting alone did not increase soil temperature to the same extent as mounding, and the affect was evident in the lower decomposition rate. Increased organic matter decomposition as a result of mounding also have been reported for other northern (Sutton, 1993) and southern (Bridgham et al., 1991) forested wetlands.

RECOVERY OF SOIL CARBON

Increased organic matter decomposition rates following silvicultural practices reduces the ability of forested wetlands to serve as a C sink. In many cases, increased decomposition rates can transform a wetland from a C sink to a C source. Long-term accumulation rates of soil C in northern peatlands range from 8 to 41 g m^{-2} yr^{-1} (Tolonen et al., 1992). The time necessary to replace soil C loss

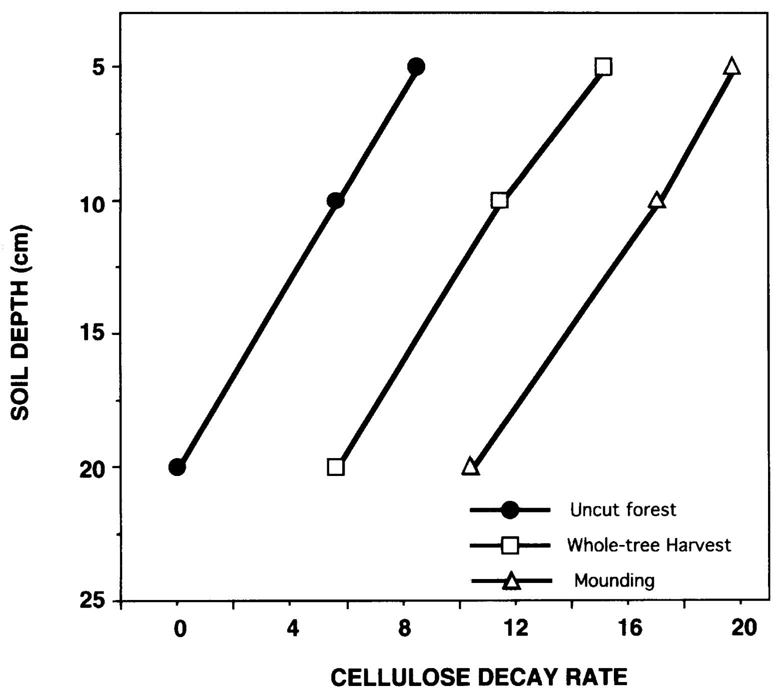

Fig. 20–3. Rate of cellulose decay in a histic-mineral soil swamp in northern Michigan 1 yr following whole-tree harvesting and whole-tree harvesting plus mounding. Cellulose decomposition rate was measured by Cotton Strip Assay (after Trettin & Jurgensen, 1992).

resulting from silvicultural practices could take decades to centuries. Reductions in soil C pools after harvesting range from approximately 1000 to 5000 g m^{-2} (Table 20–3). If postdisturbance accumulation rates were the same as predisturbance, the recovery time to return soil C to preharvest levels would range from 40 to 120 yr, using an accumulation rate of 23 g m^{-2} yr^{-1} (Gorham, 1991). However, it is unlikely that peat accumulation on disturbed wetlands would be the same as under predisturbance conditions because of changes in plant species composition and productivity, hydrology, and nutrient cycling. On drained peatlands the recovery of soil C may never occur because long-term CO_2 emissions (approx. 30 g m^{-2} yr^{-1}) exceeds average C accumulation rates (Gorham, 1991). Shifts in C allocation among the soil and vegetation will affect wetland functions, but there is little empirical data on which to assess this potential response.

The duration of impacts on soil C accumulation resulting from silvicultural practices is dependent on wetland properties and the type forest management practice. Results from several studies in northern peatlands demonstrated that C loss was still evident 20 to 30 yr after the wetland was drained (Table 20–3). Vompersky et al. (1992) suggested that improved plant growth following drainage may mitigate against loss of soil C. However, even if increased wetland productivity effectively reduces net C losses or perhaps even increases C storage on the site, any soil C loss should be expected to change soil hydrologic properties and nutrient cycling within the wetland. Only one study (Laine et al., 1992) has documented a net increase in soil C following drainage as a result of increased stand productivity. Braekke (1992) reported increased root growth following drainage and fertilization, but he did not determine the change in soil C levels. Sakovets and Germanova (1992) reported that total C content on a drained fen increased following drainage as a result of increased aboveground biomass production, although soil C levels actually decreased.

Mitigation of soil C loss by increased biomass production has only been demonstrated for fens. There are currently no studies which suggest that ombrotrophic wetlands have the potential for productivity gains to compensate for soil C loss. Changes in C allocation from soil C to above- and/or belowground biomass represents a functional change in the wetland that may temporarily balance the C flux and storage, but it is doubtful that increased C in biomass will replace the C accumulation function of peatlands and histic-mineral soils because cultivated biomass does not represent a long-term C sink (Vitousek, 1991).

CONCLUSIONS AND PERSPECTIVES

A significant, but yet uncertain, proportion of northern wetlands are forested, accumulate soil C, and are important for wood production in many countries. Carbon accumulation in these wetlands is a result of biomass production exceeding the rate of organic matter decomposition. Biotic and abiotic factors which affect organic matter accumulation include hydrology, geomorphic setting, microclimate, and vegetation composition and production. A unique aspect of C-accumulating wetlands is that the affect of those developmental factors on wetland functions change as organic matter accumulates. Accordingly, wetland functions, including soil water storage, modification of water quality, nutrient cycling, habitat conditions, and others, are sensitive to soil C levels. Carbon storage alone is an important wetland function that affects the global C budget.

The C balance in forested wetlands is sensitive to various silvicultural practices. However, information characterizing the impacts of silvicultural practices in terms of long-term C dynamics and wetland functions are very limited. Harvesting, site preparation, and drainage each cause a reduction in soil C. However, the magnitude of C loss is dependent on wetland type (bog, fen, swamp), management regime imposed, and time since disturbance. The actual loss of soil C caused by these silvicultural practices is difficult to ascertain with existing information. Most available data is from long-term studies that permit only an assessment of the net change in soil C storage. The loss of soil C following silvicultural practices occurs as CO_2, CH_4, and DOC. However, the relative importance of gaseous and aqueous pathways for C flux have not been adequately addressed. Increased soil temperature and aeration are the primary factors causing soil organic matter decomposition to increase following harvesting and site preparation. Carbon losses are greatest in drained and site-prepared wetlands, where aerated soil volume and water table fluctuations are the greatest.

In contrast, biomass production on wetlands sometimes increases following drainage and site preparation. That response has been postulated as a mechanism for mitigating soil C loss following drainage. Currently, only one study has measured an actual increase in soil C following drainage, all others have reported a net C loss. Increased C accumulation in biomass following other silvicultural practices (e.g., harvesting, site preparation) also may be considered to balance the loss of soil C. However, changes in C accumulation among different pools (e.g., biomass, soil) within the wetland will affect ecosystem functions.

Given the diversity of wetland types and the importance of these ecosystems for wood products, additional research is needed to determine the effects of silvicultural practices on soil C and associated ecosystem processes. Studies are needed to characterize the magnitude of soil C loss following silvicultural practices, and to determine the partitioning among the gas and aqueous pathways for C flux. These data are needed to determine the impacts on wetland functions, including, the global C balance, stream water quality, stream biota, hydrology, and element cycling and transport. Other studies are needed to further characterize the mechanisms controlling soil C loss, in particular the interaction between moisture and temperature regimes in field-scale experiments. Finally, long-term research is needed to assess the functional response of different wetland types to altered patterns of C allocation and storage following silvicultural practices. This information is particularly important for designing best management practices to sustain inherent wetland functions.

To address the growing public concern on the management of forested wetlands, regional, interdisciplinary studies are needed to enhance the knowledge-base regarding the effects of silvicultural practices. Ideally, such studies should have a common treatment design that is applied to regionally important wetland types. A good example of a comprehensive, integrated study is SUOSILMU, a Finnish program designed to determine the role of peatlands in the biogeochemical cycling of C (Kanninen & Anttila, 1992). In the USA, opportunities for integrated studies on managed wetlands are provided through research cooperatives (Shepard et al., 1993) and consortia (e.g., Consortium for Research on Southern Forested Wetlands). Expansion of these efforts, particularly in northern regions, is needed in order to provide an informed basis for decisions on the regulation and management of forested wetlands.

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REFERENCES

Abernethy, Y., and R.E. Turner. 1987. U.S. forested wetlands: 1940–1980. BioScience 37:721–727.

Ahtiainen, M. 1988. Effects of clear-cutting and forestry drainage on water quality in the Nurmes-study. p. 206–219. *In* Proc. Int. Symp. Hydrology of Wetlands in Temperate and Cold Regions, Joensuu, Finland. 6–8 June. Vol. 1. No. 4/88. Publ. Acad., Helsinki, Finland.

Allen, H.L., and R.G. Campbell. 1988. Wet site pine management in the southeastern United States. p. 173–184. *In* D.L. Hook (ed.) The ecology and management of wetlands. Vol. 2. Croom Helm, London.

Anderson, A.R., D.G. Pyatt, J.M. Sayers, S.R. Blackhall, and H.P. Robinson. 1992. Volume and mass budgets of blanket peat in the north of Scotland. Suo 43:195–198.

Andrus, R.E. 1986. Some aspects of *Sphagnum* ecology. Can. J. Bot. 64:416–426.

Armentano, T.V., and E.S. Menges. 1986. Patterns of change in the carbon balance of organic soil-wetlands of the temperate zone. J. Ecol. 74:755–774.

Attiwill, P.M., N.D. Turvey, M.A. Adams. 1985. Effects of mound-cultivation (bedding) on concentration and conservation of nutrients in a sandy podzol. For. Ecol. Manage. 11:97–110.

Bartlett, K.B., and R.C. Harriss. 1993. Review and assessment of methane emissions from wetlands. Chromosphere 26:261–320.

Boelter, D.H. 1969. Physical properties of peat related to degree of decomposition. Soil Sci. Soc. Am. Proc. 28:433–435.

Boelter, D.H., and E.S. Verry. 1977. Peatland and water in the northern Lake States. USDA-FS, Gen. Tech. Rep. NC-31. North Central For. Exp. Stn., St. Paul, MN.

Braekke, F.H. 1983. Water table levels at different drainage intensities on deep peat in northern Norway. For. Ecol. Manage. 5:169–192.

Braekke, F.H. 1987. Nutrient relationships in forest stands: Effects of drainage and fertilization on surface peat layers. For. Ecol. Manage. 21:269–284.

Braekke, F.H. 1992. Root biomass changes after drainage and fertilization of a low-shrub pine bog. Plant Soil 143:33–43.

Braekke, F.H., and L. Finer. 1990. Decomposition of cellulose in litter layer and surface peat of low-shrub pine bogs. Scand. J. For. Res. 5:297–310.

Bridgham, S.D., C.J. Richardson, E. Maltby, and S.P. Faulkner. 1991. Cellulose decay in natural and disturbed peatlands in North Carolina. J. Environ. Qual. 20:695–701.

Brinson, M.M., A.E. Lugo, and S. Brown. 1981. Primary productivity, decomposition, and consumer activity in freshwater wetlands. Annu. Rev. Ecol. Syst. 12:123–161.

Brock, T.C.M., and R. Bregman. 1989. Periodicity in growth, productivity, nutrient content and decomposition of *Sphagnum recurvum var. mucronatum* in a fen woodland. Oecologia 80:44–52.

Clymo, R.S. 1984a. *Sphagnum*-dominated peat bog: A naturally acid ecosystem. Philos. Trans. R. Soc. London, B 305:487–499.

Clymo, R.S. 1984b. Limit of peat bog growth. Philos. Trans. R. Soc. London, B 303:605–654.

Clymo, R.S. 1987. Interaction of *Sphagnum* with water and air. p. 513–529. *In* T.C. Hutchinson and K.M. Meema (ed.) Effects of atmospheric pollutants on forests, wetlands, and agricultural ecosystems. Springer-Verlag, Berlin, Germany.

Clymo, R.S. 1992. Models of peat growth. Suo 43:127–136.

The Conservation Foundation. 1988. Protecting America's wetlands: An action agenda. Conserv. Found., Washington, DC.

Cowardin, L.M., V. Carter, F.C. Golet, and E.T. LaRoe. 1979. Classification of wetlands and deepwater habitats of the United States. FWS/OBS-79/31. U.S. Fish Wildlife Serv., Washington DC.

Cronan, C.S. 1990. Patterns of organic acid transport from forested watersheds to aquatic ecosystems. p. 245–260. *In* E.M. Perdue and E.T. Gjessing (ed.) Organic acids in aquatic ecosystems. John Wiley & Sons, Chichester, England.

Crum, H.A. 1988. A focus on peatlands and peat mosses. Univ. Michigan Press, Ann Arbor.

Cubbage, F.W., and C.H. Flather. 1993. Forested wetland area and distribution: A detailed look at the South. J. For. 91:35–40.

Damman, A.W.H. 1990. Nutrient status of ombrotrophic peat bogs. Aquilo Ser. Bot. 28:5–14.

Damman, A.W.H., and T.W. French. 1987. The ecology of peat bogs of the glaciated northeastern United States: A community profile. U.S. Fish Wildlife Serv. Biol. Rep. 85(7.16). U.S. Dep. Interior, Washington, DC.

David, M.B., and G.F. Vance. 1991. Chemical character and origin of organic acids in streams and seepage lakes of central Maine. Biogeochemistry 12:17–41.

Dise, N.B. 1992. Winter fluxes of methane from Minnesota peatlands. Biogeochemistry 17:71–83.

Donnelly, P.K., J.A. Entry, D.L. Crawford, and K. Cromack, Jr. 1990. Cellulose and lignin degradation in forest soils in response to moisture, temperature, and acidity. Microbiol. Ecol. 20:289–295.

Dunfield, P., R. Knowles, R. Dumont, and T.R. Moore. 1993. Methane production and consumption in temperate and subarctic peat soils: Response to temperature and pH. Soil Biol. Biochem. 25:321–326.

Eswaran, H., E. Van Den Berg, and P. Reich. 1993. Organic carbon in soils of the world. Soil Sci. Soc. Am. J. 57:192–194.

Fahey, T.J., P.A. Stevens, M. Hornung, and P. Rowland. 1991. Decomposition and nutrient release from logging residue following conventional harvest of Sitka spruce in north Wales. Forestry 64:289–301.

Farrish, K.W., and D.F. Grigal. 1985. Mass loss in a forested bog: Relation to hummock and hollow microrelief. Can. J. Soil Sci. 65:375–378.

Farrish, K.W., and D.F. Grigal. 1988. Decomposition in an ombrotrophic bog and a minerotrophic fen in Minnesota. Soil Sci. 145:353–358.

Frayer, W.E. 1991. Status and trends of wetlands and deepwater habitats in the conterminous United States, 1970's to 1980's. School Forestry Wood Products, Michigan Technol. Univ., Houghton, MI.

Freeman, C., M.A. Lock, and B. Reynolds. 1993. Fluxes of CO_2, CH_4 and N_2O from a Welsh peatland following simulation of water table draw-down: Potential feedback to climate change. Biogeochemistry 19:51–60.

Glaser, P.H. 1987. The ecology of patterned boreal peatlands of northern Minnesota: A community profile. U.S. Fish Wildlife Serv. Biol. Rep. 85(7.14). U.S. Dep. Interior, Washington, DC.

Gorham, E. 1991. Northern peatlands: Role in the carbon cycle and probable responses to climatic warming. Ecol. Monogr. 1:182–195.

Grigal, D.F. 1985. *Sphagnum* production in forested bogs in northern Minnesota. Can. J. Bot. 63:1204–1207.

Grigal, D.F., C.G. Buttleman, and L.K. Kernik. 1985. Biomass and productivity of the woody strata of forested bogs in northern Minnesota. Can. J. Bot. 63:2416–2424.

Haavisto, V.F., and J.K. Jeglum. 1991. Peatland potentially available for forestry in Canada. p. 30–37. *In* J.K. Jeglum and R.P. Overend (ed.) Peat and peatlands diversification and innovation, Quebec City, Quebec. 6–10 Aug. 1989. Vol. 6. Canadian Soc. Peat and Peatlands, Darmouth, Nova Scotia.

Hahn, J.T., and W.B. Smith. 1987. Minnesota's forest statistics, 1987: An inventory update. Gen. Tech. Rep. NC-118. USDA-FS, North Central For. Exp. Stn., St. Paul, NM.

Haland, B., and F.H. Braekke. 1989. Distribution of root biomass in a low-shrub pine bog. Scan. J. For. Res. 4:307–316.

Hanell, B. 1991. Peatland forestry in Sweden. p. 19–25. *In* J.K. Jeglum and R.P. Overend (ed.) Peat and peatlands diversification and innovation, Quebec City, Quebec. 6–10 Aug. 1989. Vol. 6. Canadian Soc. Peat and Peatlands, Darmouth, Nova Scotia.

Heikkinen, K. 1990. Nature of dissolved organic matter in the drainage basin of a boreal humic river in northern Finland. J. Environ. Qual. 19:649–657.

Heikurainen, L., K. Kenttamies, and J. Laine. 1978. The environmental effects of forest drainage. Suo 29:49–58.

Hemond, H.F. 1980. Biogeochemistry of Thoreau's Bog, Concord, Massachusetts. Ecol. Monogr. 50:507–526.

Hemond, H.F. 1990. Wetlands as the source of dissolved organic carbon to surface waters. p. 301–313. *In* E.M. Perdue and E.T. Gjessing (ed.) Organic acids in aquatic ecosystems. John Wiley & Sons, Chichester, England.

Hofstetter, R.H. 1983. Wetlands in the United States. p. 201–244. *In* A.J.P Gore (ed.) Mires: Swamp, bog, fen, and moor. Vol. 1. Elsevier Sci. Publ. Co., Amsterdam.

Hogg, E.H. 1993. Decay potential of hummock and hollow Sphagnum peats at different depths in a Swedish raised bog. Oikos 66:269–278.

Hogg, E.H., V.J. Lieffers, and R.W. Wein. 1992. Potential carbon losses from peat profiles; effects of temperature, drought cycles, and fire. Ecol. Applic. 2:298–306.

Hovi, A. 1988. Organic carbon dynamics in small brooks before and after forest drainage and clear-cutting. p. 220–231. *In* Proc. Int. Symp. Hydrology Wetlands in Temperate and Cold Regions, Joensuu, Finland. 6–8 June. Vol. 1. No. 4/88. Publ. Acad. Finland, Helsinki, Finland.

Hughes, S., B. Reynolds, and J.D. Roberts. 1990. The influence of land management on concentrations of dissolved organic carbon and its effects on the mobilization of aluminum and iron in podzol soils in mid-Wales. Soil Use Manage. 6:137–144.

Huttunen, P., A.L. Holopainen, and A. Hovi. 1988. Effects of silvicultural measures on primary production in forest brooks. p. 239–248. *In* Proc. Int. Symp. Hydrology Wetlands in Temperate and Cold Regions, Joensuu, Finland. 6–8 June. Vol. 1. No. 4/88. Publ. Acad. Finland, Helsinki, Finland.

Ingram, H.A.P. 1983. Hydrology. p. 67–158. *In* A.J.P Gore (ed.) Mires: Swamp, bog, fen, and moor. Vol. I. Elsevier Sci. Publ. Co., Amsterdam.

Jeglum, J.K., and R.P. Overend (ed.) 1991. Peat and peatlands diversification and innovation, Quebec City, Quebec. 6–10 Aug. 1989. Canadian Soc. Peat Peatlands, Darmouth, Nova Scotia.

Johnson, D.W. 1992. Effects of forest management on soil carbon storage. Water Air Soil Pollut. 64:83–120.

Jorgensen, R.G., and G.M. Richter. 1992. Composition of carbon fractions and potential denitrification in drained peat soils. J. Soil Sci. 43:347–358.

Kanninen, M., and P. Anttila (ed.). 1992. The Finnish research programme on climate change. No. 3/92. Publ. Acad. Finland, Helsinki, Finland.

Kenttamies, K. 1981. The effects of water quality of forest drainage and fertilization in peatlands. Publ. Water Res. Inst., no. 43. Natl. Board Waters, Helsinki, Finland.

Kim, J., and S.B. Verma. 1992. Soil surface CO_2 flux in a Minnesota peatland. Biogeochemistry 18:37–51.

Koprivnjak, J.F., and T.R. Moore. 1992. Sources, sinks, and fluxes of dissolved organic carbon in subarctic fen catchments. Arctic Alpine Res. 24:204–210.

Kubin, E., and L. Kemppainen. 1991. Effect of clearcutting of boreal spruce forest on air and soil temperature conditions. Acta For. Fenn. 225:1–42.

Lahde, E. 1969. Biological activity in some natural and drained peat soils with special reference to oxidation-reduction conditions. Acta For. Fenn. 94:1–69.

Laine, J., H. Vasander, and A. Puhalainen. 1992. Effects of forest drainage on the carbon balance of mire ecosystems. p. 170–181. *In* Proc. 9th Int. Peat Congr., 9th, 22–26 June. Vol. 1. Uppsala, Sweden. Int. Peat Soc., Jyska, Finland.

Latja, A., and H. Kurimo. 1988. Temperature changes in the soil and close to the ground on wetlands drained for forestry. p. 46–51. *In* Proc. Int. Symp. Hydrology Wetlands Temperate and Cold Regions, Joensuu, Finland. 6–8 June. Vol. 1. No. 4/88. Publ. Acad. Finland, Helsinki, Finland.

Lieffers, V.J. 1988. *Sphagnum* and cellulose decomposition in drained and natural areas of an Alberta peatland. Can. J. Soil Sci. 68:755–761.

Lieffers, V.J., and R.L. Rothwell. 1987. Effects of drainage on substrate temperature and phenology of some trees and shrubs in an Alberta peatland. Can. J. For. Res. 17:97–104.

Lundin, L. 1988. Impacts of drainage for forestry on runoff and water chemistry. p. 197–205. *In* Proc. Int. Symp. Hydrology Wetlands Temperate Cold Regions, Joensuu. 6–8 June. Vol. 1. No. 4/88. Publ. Acad. Finland, Helsinki, Finland.

Mader, S.F. 1991. Forested wetland classification and mapping: A literature review. NCASI Tech. Bull. 606. Natl. Council Pap. Ind. Air Stream Improvement, New York.

Mann, L.K., D.W. Johnson, D.C. West, D.W. Cole, J.W. Hornbeck, C.W. Martin, H. Riekerk, C.T. Smith, W.T. Swank, L.M. Tritton, and D.H. Van Lear. 1988. Effects of whole-tree and stem-only clearcutting on postharvest hydrologic losses, nutrient capital, and regrowth. For. Sci. 34:412–428.

Mannerkoski, H. 1985. Effect of water table fluctuation on the ecology of peat soil. Publ. 7. Dep. Peatland For. Univ. Helsinki, Helsinki, Finland.

Marin, L.E., T.K. Kratz, and C.J. Bower. 1990. Spatial and temporal patterns in the hydrochemistry of a poor fen in northern Wisconsin. Biogeochemistry 11:63–76.

Moore, P.D., and D.J. Bellamy. 1974. Peatlands. Springer-Verlag, New York.

Moore, T.R. 1989. Growth and net production of *Sphagnum* at five fen sites, subarctic eastern Canada. Can. J. Bot. 67:1203–1207.

Moore, T.R., and R. Knowles. 1989. The influence of water table levels on methane and carbon dioxide emissions from peatland soils. Can. J. Soil Sci. 69:33–38.

Munro, D.S. 1984. Summer soil moisture content and the water table in a forested wetland peat. Can. J. For. Res. 14:331–335.

Musselman, R.C., and D.G. Fox. 1991. A review of the role of temperate forests in the global CO_2 Balance. J. Air Waste Manage. Assoc. 41:798–807.

National Wetlands Working Group. 1988. Wetlands of Canada. Ecol. Land Classification Ser., no. 24. Sustainable Devel. Branch. Environ. Canada, Ottawa, Ontario, and Polysci. Publ., Montreal, Quebec.

Oades, J.M. 1988. The retention of organic matter in soils. Biogeochemistry 5:35–70.

Paavilainen, E., and J. Paivanen. 1988. Use and management of forested hydric soils. p. 203–212. *In* D.L. Hook (ed.) The ecology and management of wetlands. Vol. 2. Croom Helm, London.

Paivanen, J. 1982. The effect of cutting and fertilization on the hydrology of an old forest drainage area. Folia For. 515:4–19.

Paivanen, J. 1991. Peatland forestry in Finland: Present status and prospects. p. 3–12. *In* J.K. Jeglum and R.P. Overend (ed.) Peat and peatlands diversification and innovation, Quebec City, Quebec. 6–10 Aug. 1989. Canadian Soc. Peat Peatlands, Darmouth, Nova Scotia.

Paivanen, J., and E.D. Wells. 1977. Guidelines for the development of peatland drainage systems for forestry in Newfoundland. Dep. Fish. Environ., Canadian For. Serv. Inf. Rep. N-X-156. St. Johns, Newfoundland.

Parker, G.R., and G. Schnieder. 1975. Biomass and productivity of an alder swamp in northern Michigan. Can. J. For. Res. 5:403–409.

Post, W.M., W.R. Emanuel, P.J. Zinke, and A.G. Stangenberger. 1982. Soil carbon pools and world life zones. Nature (London) 298:156–159.

Reader, R.J. 1978. Primary production in northern bog marshes. p. 53–62. *In* Freshwater wetlands. Acad. Press, New York.

Reader, R.J., and J.M. Stewart. 1972. The relationship between net primary production and accumulation for a peatland in southeastern Manitoba. Ecology 53:1024–1037.

Reiners, W.A. 1972. Structure and energetics of three Minnesota forests. Ecol. Monogr. 42:71–94.

Richardson, C.J. 1978. Primary productivity values in fresh water wetlands. p. 131–145. *In* P.E. Greeson et al. (ed.) Wetland function and values: The state of our understanding. AWRA, Minneapolis, MN.

Rochefort, L., D.H. Vitt, and S.E. Baley. 1990. Growth, production, and decomposition dynamics of *Sphagnum* under natural and experimentally acidified conditions. Ecology 71:1986–2000.

Ross, S.M., and D.C. Malcolm. 1988. Modeling nutrient mobilization in intensively mixed peaty heathland soil. Plant Soil 107:113–121.

Sakovets, V.V., and N.I. Germanova. 1992. Changes in carbon balance of forested mires in Karelia due to drainage. Suo 43:249–252.

Sallantaus, T. 1988. Water quality and man's influence on it. p. 80–98. *In* Proc. Int. Symp. Hydrology Wetlands Temperate Cold Regions, Joensuu, Finland. 6–8 June. Vol. 2. No. 4/88. Publ. Acad. Finland, Helsinki, Finland.

Sallantaus, T. 1992. Leaching in the material balance of peatlands—preliminary results. Suo 43:253–258.

Santantonio, D., R.K. Hermann, and W.S. Overton. 1977. Root biomass studies in forest ecosystems. Pedobiologia 17:1–31.

Schlesinger, W.H. 1986. Changes in soil carbon storage and associated properties with disturbance and recovery. p. 195–219. *In* J.R. Trabalka and D.E. Reichle (ed.) The changing carbon cycle—A global analysis. Springer-Verlag, New York.

Shepard, J.P., A.A. Lucier, and L.W. Haines. 1993. Industry and forest wetlands: Cooperative research initiatives. J. For. 91:29–33.

Siegel, D.I., and P.H. Glaser. 1987. Ground water flow in a bog-fen complex Lost River peatland, northern Minnesota. J. Ecol. 75:743–754.

Silvola, J. 1986. Carbon dioxide dynamics in mires reclaimed for forestry in eastern Finland. Ann. Bot. Fennici 23:59–67.

Silvola, J. 1988. Effect of drainage and fertilization on carbon and nutrient mineralization of peat. Suo 39:27–37.

Smith, W.B., and J.T. Hahn. 1987. Michigan's forest statistics, 1987: An inventory update. USDA-FS Gen. Tech. Rep. NC-112. North Central For. Exp. Stn., St. Paul, MN.

Smith, W.B., and J.T. Hahn. 1989. Wisconsin's forest statistics, 1987: An inventory update. USDA-FS Gen. Tech. Rep. NC-130. North Central For. Exp. Stn., St. Paul, MN.

Soil Survey Staff. 1975. Soil taxonomy: A basic system of soil classification for making and interpreting soil surveys. USDA-SCS Agric. Handb. 436. U.S. Gov. Print. Office, Washington, DC.

Starr, M.R., and J. Paivanen. 1986. Runoff response to peatland forest drainage in Finland: A synthesis. p. 43–50. *In* S.P. Gessel (ed.) Forest site productivity. Martinus Nijoff Publ., Dordrecht, the Netherlands.

Sutton, R.F. 1993. Mounding site preparation: A review of European and North American experience. New For. 7:151–192.

Swanson, D.K., and D.F. Grigal. 1989. Vegetation indicators of organic soil properties in Minnesota. Soil Sci. Soc. Am. J. 53:491–495.

Taylor, B.R., and H.G. Jones. 1990. Litter decomposition under snow cover in a balsam fir forest. Can. J. Bot. 68:112–120.

Terry, T.A., and J.H. Hughes. 1978. Drainage of excess water—why and how? p. 148–166. *In* W.E. Balmer (ed.) Proc. Soil Moisture—Site Productivity Symp., Myrtle Beach, SC. 1–3 Nov. 1977. Southeast For. Exp. Stn., Asheville, NC.

Teskey, R.O., and T.M. Hinkley. 1978. Impact of water level changes on woody riparian and wetland communities. U.S. Fish Wildlife Serv. Biol. Rep. FWS/OBS-78/88. U.S. Dep. Interior, Washington, DC.

Titus, B.D., and D.C. Malcolm. 1992. Nutrient leaching from the litter layer after clearfelling of sitka spruce stands on peaty gley soils. Forestry 65:389–416.

Tolonen, K., H. Vasander, A.W.H. Damman, and R.S. Clymo. 1992. Rate of apparent and true carbon accumulation in boreal peatlands. p. 319–333. *In* Proc. Int. Peat Congr., 9th, 22–26 June. Vol. 1. Uppsala, Sweden, Int. Peat Soc., Jyska, Finland.

Trettin, C.C., and M.F. Jurgensen. 1992. Organic matter decomposition response following disturbance in a forested wetland in northern Michigan, U.S.A. p. 392–399. *In* Proc. Int. Peat Congr., 22–26 June. Uppsala, Sweden. Int. Peat Soc., Jyska, Finland.

Trettin, C.C., M.R. Gale, M.F. Jurgensen, J.W. McLaughlin. 1992. Carbon storage response to harvesting and site preparation in a forested mire in northern Michigan, U.S.A. Suo 43:281–284.

Urban, N.R., S.J. Eisenreich, and E. Gorham. 1987. Proton cycling in bogs: Geographic variation in northeastern North America. p. 577–598. *In* T.C. Hutchinson and K.M. Meema (ed.) Effects of atmospheric pollutants on forests, wetlands, and agricultural ecosystems. NATO ASI Ser. Vol G16. Springer-Verlag, Berlin.

Van Cleve, K., L. Oliver, R. Schlentner, L.A. Viereck, and C.T. Dryness. 1983. Productivity and nutrient cycling in taiga forest ecosystems. Can. J. For. Res. 13:747–766.

Vasander, H. 1982. Plant biomass and production in virgin, drained, and fertilized sites in a raised bog in southern Finland. Ann. Bot. Fennici 19:103–125.

Verhoeven, J.T.A. 1986. Nutrient dynamics in minerotrophic peat mires. Aquat. Bot. 25:117–137.

Verry, E.S. 1986. Forest harvesting and water: the Lake States experience. Water Res. Bull. 22:1039–1047.

Verry, E.S. 1988. The hydrology of wetlands and man's influence on it. p. 41–61. *In* Proc. Int. Symp. Hydrology Wetlands Temperate Cold Regions, Joensuu, Finland. 6–8 June. Vol. 2. No. 4/88. Publ. Acad. Finland, Helsinki. Int. Peat Soc., Jyska, Finland.

Verry, E.S., and N.R. Urban. 1992. Nutrient cycling at Marcell Bog, Minnesota. Suo 43:147–154.

Vitousek, P.M. 1991. Can planted forests counteract increasing atmospheric carbon dioxide? J. Environ. Qual. 20:348–354.

Vompersky, S.E. 1991. Current status of forest drainage in the USSR and problems in research. p. 13–18. *In* J.K. Jeglum and R.P. Overend (ed.) Peat and peatlands diversification and innovation, Quebec City, Quebec. 6–10 Aug. 1989. Canadian Soc. Peat and Peatlands, Darmouth, Nova Scotia.

Vompersky, S.E., M.V. Smagina, A.I. Ivanov, and T.V. Glukhova. 1992. The effect of forest drainage on the balance of organic matter in forest mires. p. 17–22. *In* O.M. Bragg et al. (ed.) Peatland ecosystems and man: An impact assessment. Dep. Biol. Sci., Univ., Dundee, England.

Wieder, R.K., and J.B. Yavitt. 1991. Assessment of site differences in anaerobic carbon mineralization using reciprocal peat transplants. Soil Biol. Biochem. 23:1093–1095.

Williams, B.L., and G.P. Sparling. 1988. Microbial biomass carbon and readily mineralized nitrogen in peat and forest humus. Soil Biol. Biochem. 20:579–581.

Yavitt, J.B., G.E. Lang, and R.K. Wieder. 1987. Control of carbon mineralization to CH_4 and CO_2 in anaerobic, *Sphagnum*-derived peat from Big Run Bog, West Virginia. Biogeochemistry 4:141–157.

Zon, R., and J.L. Averell. 1929. Drainage of swamps and forest growth. Res. Bull. 89. Agric. Exp. Stn., Univ. Wisconsin, Madison.