Attachment 38

Comments on the Draft Plan submitted by Objectors on June 29, 2020, along with all attachments

Attachment 54

Additional Literature Cited

Patterns of Disturbance in Some Old-Growth Mesic Forests of Eastern North America

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PATTERNS OF DISTURBANCE IN SOME OLD-GROWTH MESIC FORESTS OF EASTERN NORTH AMERICA[1]

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Abstract. To characterize the disturbance regime of one type of vegetation, study areas in which relatively small-scale disturbance predominates were chosen in several old-growth mesic forests in the eastern United States. Canopy openings covered 9.5% of total land area. New gaps were formed at an average rate of 1% of total land area per year; old gap area closed at a similar rate primarily by sapling height growth.

With increased gap size, vegetation within gaps increased in woody species diversity, total basal area, and total number of stems. Stems also showed accelerated growth into larger size classes. As gaps aged, stems grew into larger size classes and basal area increased.

Species responses to canopy gaps varied. Some species survived and became established in fairly small gaps (50–100 m^2). Although in large gaps (up to 2009 m^2 in the present study) these species usually increased in total number of stems and basal area, they declined in importance relative to species which rarely survived in small gaps but grew rapidly in large gaps. The disturbance regimes in the forests studied favored tolerant species but allowed opportunists to persist at low densities.

Key words: climax; disturbance; forest regeneration; gaps; Hueston Woods; mixed mesophytic forest; patch dynamics; southern Appalachians; succession; Tionesta; windfalls.

INTRODUCTION

Communities change constantly as individuals die and are replaced. How deaths and replacements occur in time and space has an effect on many aspects of community structure and species composition. The relationship between community properties and the pattern of individual deaths (disturbance regime) has been examined recently (e.g., Jones 1945, Watt 1947, Loucks 1970, Wright 1974, Whitmore 1975, 1978, Connell 1978, Bormann and Likens 1979, White 1979). Detailed descriptions of natural disturbance regimes for various community types are necessary to evaluate recent theories, to understand community properties, and to provide information useful for landscape managers (Pickett and Thompson 1978). The natural disturbance regimes of several communities have been examined in some detail (e.g., Brunig 1973, Heinselman 1973, Henry and Swan 1974, Lorimer 1977, 1980, Zackrisson 1977, Hartshorn 1978, Reiners and Lang 1979, Sprugel and Bormann 1981). However, few studies compare several different communities or have been done in those temperate-zone areas where disturbances are usually small.

The goal of the present paper is to describe disturbance regimes characterized by small gaps created following the death of a single canopy tree, part of a canopy tree, or a very few individuals. A complete description of the disturbance regime involves two parts (Levin and Paine 1974): (1) the size and age distributions and birth and death rates of gaps, and (2) the response of species to the regeneration opportunities existing in gaps of different sizes and ages.

STUDY AREAS

In order to limit consideration to the formation and filling in of small gaps, criteria for choosing a suitable forest stand were that it be (a) without any obvious large-scale human or natural disturbances, as determined from historical records and the presence of very large individuals, and (b) without evidence of extensive chestnut (*Castanea dentata*) mortality (which would greatly affect estimates of more normal rates of gap formation and more normal gap sizes). To decrease variability within and among samples, stands were required to possess reasonably homogeneous canopy species composition for an area of at least several hectares and dominance by some combination of such mesic tree species as hemlock (*Tsuga canadensis*), beech (*Fagus grandifolia*), sugar maple (*Acer saccharum*), yellow birch (*Betula lutea*), yellow buckeye (*Aesculus octandra*), mountain silverbell (*Halesia carolina*), and white basswood (*Tilia heterophylla*).

Some stands within each of the following areas were sampled: Great Smoky Mountains National Park of North Carolina and Tennessee; Joyce Kilmer Wilderness Area of western North Carolina; Walker's Cove Research Natural Area near Asheville, North Carolina; Hueston Woods State Park near Oxford, Ohio; Tionesta Scenic and Natural Areas in northwestern Pennsylvania; Woodbourne Forest and Wildlife Sanctuary in northeastern Pennsylvania; and the Edmund Niles Huyck Preserve near Albany, New York. Species composing >10% of trees ≥ 25 cm dbh for

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each area are, for the Great Smoky Mountains: sugar maple, yellow buckeye, beech, silverbell, white basswood, and hemlock; for Joyce Kilmer: sugar maple, beech, silverbell, basswood, and hemlock; for Walker's Cove: sugar maple, buckeye, beech, and basswood; for Hueston Woods: sugar maple and beech; for Tionesta: beech and hemlock; for Woodbourne: sugar maple, white ash, and hemlock; and for Huyck: beech and hemlock (more details are given in Runkle 1979, 1981).

Although in general the stands studied seemed to fit the criteria concerning disturbance history and species composition listed above, large disturbances may have occurred in some stands and may have been important in determining some of the present species composition. In the Great Smoky Mountains National Park, some selective cutting may have occurred within some of the areas studied. Also, tornadoes that destroyed several hectares of forest have been noted and so may have affected the stands studied at some time in the past. In Joyce Kilmer, windstorms affecting several canopy trees occur periodically (Lorimer 1980) and probably are important in influencing canopy composition, though gaps created by single trees also are important. The present study of Joyce Kilmer included gaps created by single trees and by as many as nine canopy trees, and therefore should cover most of the range of gap sizes which normally occur. In northwestern Pennsylvania as a whole, large-scale disturbances have occurred frequently enough to have generated stands of white pine (*Pinus strobus*), such as those at Heart's Content and Cook's Forest (Morey 1936*b*). Bjorkbom and Larson (1977) state that although mature white pine has not been recorded at Tionesta, windstorms in 1808 and 1870 damaged two large areas within the Tionesta Scenic and Research Natural Areas, causing increases in relatively shade-intolerant hardwood species. In the areas sampled, however, no such disturbances are recorded in the literature. Another influence in Tionesta was heavy browsing by deer, which has seriously affected the regeneration of many hardwoods (Bjorkbom and Larson 1977). Hueston Woods has remained relatively undisturbed since its original purchase in 1797, serving primarily as a source of maple sugar. However, selective logging for desirable species probably occurred, and the undergrowth in some places has received heavy trampling. Some areas within Woodbourne were affected by a hurricane in 1950 (J. Stone, *personal communication*), and by a beech fungal disease (*Nectria coccinea* var. *faginata*); such areas were avoided in my samples. The Huyck Preserve also was affected by the beech fungal disease.

FIELD METHODS

Transects beginning at randomly chosen points were set up along compass lines parallel to the long axis of each suitable study area. At random distances along these transects the point-centered quarter method (Cottam and Curtis 1956) was used to characterize the canopy composition. The first point fell 0–25 paces from the beginning of the transect and subsequent points 25–75 paces ($\approx$17–50 m) apart. At each point, whether or not the point was in a gap, distances to and diameters of nearest trees $\geq$25 cm dbh in each quarter were measured; 25 cm dbh was generally the smallest size at which individuals were capable of creating overstory gaps.

Two types of gaps were defined. The canopy gap was the land surface area directly under the canopy opening. The expanded gap consisted of the canopy gap plus the adjacent area extending to the bases of canopy trees surrounding the canopy gap. The concept of expanded gap was useful for two reasons. First, it included areas directly and indirectly affected by the canopy opening; the effects of light often were offset from the gap center. Therefore simply measuring the canopy gap underestimated the true importance of the gap in the community. Second, at least some of the forestry literature (e.g., Tryon and Trimble 1969) defines "opening" in this way, although a precise definition of "opening" often is not given. For the purposes of this study, gaps were considered indistinguishable from the background vegetation when regeneration within the gap was $\approx$10–20 m tall.

The length of each transect was recorded as the total number of paces. In addition, the number of paces walked along the transect in each canopy gap and each expanded gap was recorded. When the transect intersected an expanded gap, the following additional measurements were taken. The area A for both expanded gaps and canopy gaps was estimated by fitting their length L (largest distance from gap edge to gap edge) and width W (largest distance perpendicular to the length) to the formula for an ellipse (most gaps were shaped at least roughly like an ellipse; $A = \pi LW/4$). Note that two distinctly different types of gap size measurements were taken: first, the fraction of transect in gaps, a quantity used to determine the fraction of total land surface area in gaps, and second, actual gap area, a quantity used to determine gap size distributions. The number and species of all woody stems $\geq$1 m high and the dbh, number, and species of all woody stems $\geq$2 m high were recorded. Where several stems were clearly from the same individual, the few largest were included. This report will refer to individuals $\geq$1 m high within gaps as saplings. The number, species, and type of injury sustained by trees creating the gaps ("gapmakers") also were noted.

Gap age (time since formation) was measured in several ways. Surrounding trees or smaller individuals within the gap were cored, and cores were sanded and examined under a microscope to look for release dates (noticeable and consistent increases in annual ring width). Sprouts that apparently originated when a tree was injured but not killed during gap formation were aged by taking cores, collecting cross sections of the sprout near its junction with the main stem for later laboratory analysis, or counting annual bud scars to determine sprout age. Changes in the rate of height or branch growth for saplings or shrubs within the gap were also noted by counting annual bud scars. Although for some gaps none of these methods provided clear results, in most cases the values probably were accurate to within a few years. For many gaps, several years after initial formation a canopy tree bordering the gap died or was broken off, adding to the gap area. In such cases gap age was dated from the initial disturbance. By convention the age of a gap was the maximum number of winters since formation; for example, for the 1976 data, a gap aged 1 occurred sometime after late summer 1975.

Details for individual gaps are given in Runkle (1979). Species nomenclature follows Radford et al. (1968).

RESULTS

Fraction of land area in gaps

Values for the fraction of land area in canopy gaps ranged from 3.2 to 24.2% for the different study areas (Table 1). Values for the fraction of land surface area in expanded gaps ranged from 6.7 to 47.0%. In general, relative gap area increased from the Pennsylvania and New York beech-hemlock stands to the Ohio beech-sugar maple stand to the southern Appalachians. Within the southern Appalachians trends were less clear.

Gap size distribution

Size distributions for both canopy gaps and expanded gaps were computed in three ways. First, areas for all gaps studied were averaged directly. Although this statistic was a useful description of the gaps analyzed, it did not accurately indicate the size distribution of gaps in the field, since a transect was more likely to intersect a large gap than a small one. Therefore, the second technique used was to divide each gap's area by the square root of its area, a term which should be proportional to its radius. The probability of a gap's being intersected is proportional to its radius. Although this technique accurately described the size distribution of gaps it is also meaningful to ask what was the average gap size associated with each pace or unit gap area. The third technique, therefore, was to weight each gap area by the number of paces (along the transect) which were in the gap. Data were fit to lognormal distributions (Table 2). This distribution is reasonable because it assumes that gap size is a result of many essentially random processes whose effects are multiplicative. Each distribution was checked for lognormality using the Kolmogorov-Smirnov test of goodness of fit (Ostle and Mensing 1975). In no case was the null hypothesis (that the distribution is lognormal) rejected at the .05 level.

TABLE 1. Percent of total land area in gaps, where EG = expanded gap, CG = canopy gap, and stands are as follows: GSM = Great Smoky Mountains National Park (stand numbers are as in Runkle 1979); JK = Joyce Kilmer Wilderness Area; WC = Walker's Cove Research Natural Area; HW = Hueston Woods State Park; TA = Tionesta Scenic and Research Natural Areas; WB = Woodbourne Forest and Wildlife Sanctuary; and HK = Huyck Preserve.

Stand	EG	CG	Total number of paces
GSM1	30.3	16.3	3182
GSM3	22.1	11.1	2568
GSM4	47.0	24.2	1283
GSM5	29.4	13.3	1972
GSM6	30.4	11.2	2036
GSM7	27.6	10.5	1822
GSM9	27.4	8.9	1346
GSM10	30.2	15.8	660
JK	29.7	17.3	1418
WC	20.6	8.2	3409
HW	14.1	7.0	5084
TA	12.0	5.0	10143
WB	6.7	3.2	1327
HK	13.8	4.8	457
All	21.0	9.5	36707

Gap sizes in the southern Appalachians and in Hueston Woods had similar mean values (t test; $P \leq .05$) but the variance in the southern Appalachians was significantly greater (F test; $P \leq .05$). On the other hand, gaps in the southern Appalachians were significantly larger ($P \leq .001$) and more variable in size ($P \leq .01$) than in Tionesta.

TABLE 2. Gap size: lognormal distributional parameters (mean ± SD, $\log_e$) for gap size in square metres (EG = area of expanded gap; CG = area of canopy gap) and sizes of largest gaps sampled. See text for discussion of different types of distributions. Stand symbols are explained in Table 1.

Stand	EG	CG
Size distribution of gaps with sampling bias		
GSM1–10,JK	5.47 ± 0.65	4.18 ± 1.13
WC	5.45 ± 0.69	4.28 ± 1.00
HW	5.43 ± 0.63	3.90 ± 1.09
TA	5.19 ± 0.55	3.85 ± 0.89
Unbiased size distribution of gaps (Gap area weighted by number of paces in gap)		
GSM1–10,JK	5.26 ± 0.63	3.44 ± 1.32
WC	5.20 ± 0.73	3.78 ± 1.00
HW	5.24 ± 0.60	3.33 ± 1.03
TA	5.02 ± 0.61	3.45 ± 0.89
Unbiased size distribution of gap area		
GSM1–10,JK	5.61 ± 0.70	4.73 ± 1.11
WC	5.64 ± 0.68	4.82 ± 0.99
HW	5.64 ± 0.65	4.63 ± 0.92
TA	5.30 ± 0.49	4.23 ± 0.81
Area (m^2) of largest gap sampled		
GSM1–10,JK	2009	1490
WC	804	707
HW	1039	507
TA	506	379

TABLE 3. Canopy gap age distribution by stand (total paces within gaps of each age as percentage of total paces). Gaps which were new in 1977 from study areas originally sampled in 1976 are not included. Stand symbols are explained in Table 1.

	Gap age (yr)															
Stand	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	>15
GSM1	4.0	0.8	0.7	0	0.1	1.9	4.6	0.2	0.6	0.2	1.3	0.3	0	0	0.2	1.2
GSM3	1.4	4.4	1.9	0.8	0	0	0.3	0	0.3	0	0.5	0	0.4	0	0	0.8
GSM4	6.5	0	3.0	1.2	0.3	0.9	1.4	0	1.7	3.7	1.4	0.8	0	0.9	1.2	1.4
GSM5	0.6	0	0.3	0.1	0.4	2.9	1.2	1.2	0	0.4	0.2	0.4	1.9	0	0	2.3
GSM6	2.0	0.2	0.9	0.6	0.7	0.2	0	0	0	1.5	0.2	1.0	0.6	0.8	2.1	0.4
GSM7	0	0.6	0	0.5	0.7	0	1.5	1.9	0.4	0.3	2.8	0	0.3	0	1.0	0.4
GSM9	0.4	1.3	3.0	0.2	1.5	0	0.8	0	0.9	0	0.3	0	0	0	0	0.6
GSM10	2.1	2.4	1.4	0	2.4	5.6	1.2	0	0	0	0.6	0	0	0	0	0
JK	0	7.4	2.5	1.8	0	0	1.0	2.0	0	0.9	0	0	0	0	1.7	0
WC	0.5	0	0.2	1.2	2.7	0.6	0.2	2.0	0	0	0	0	0	0	0	0.7
HW	0.2	1.3	1.2	0.8	0.4	1.0	0.2	0	0.1	0.9	0	0.3	0	0	0.1	0.4
TA	0.2	0.7	0.3	0.6	0.7	0.2	0.8	0.3	0.1	0.1	0.1	0.4	0.2	0	0	0.1
WB	0.5	1.4	0	1.4	0	0	0	0	0	0	0	0	0	0	0	0
HK	1.3	0	0	0	0.9	0	0	0	0	0	2.6	0	0	0	0	0
GSM,JK	2.0	1.8	1.4	0.6	0.5	1.0	1.6	0.4	0.6	0.8	0.8	0.3	0.2	0.2	0.9	0.8

Gap age distribution

To understand gap regeneration it is necessary to know the rates at which gaps are formed (gap birth rates), and the rates at which they close (gap death rates) (Paine and Levin 1981). Gaps die (become indistinguishable from the background vegetation) as a result of (1) lateral extension (branch) growth of canopy trees surrounding the gap, and (2) height growth of individuals either formerly suppressed or newly germinating from seeds. (In some cases stump sprouts of the former canopy individual also are present.) The relative importances of the sources of saplings vary. However, in the mesic forests studied, suppressed individuals were probably the most important, since almost all the major species are at least somewhat tolerant of understory conditions when small. Whether branch growth of large trees or height growth of saplings is more important in gap closure determines whether factors influencing sapling growth within the gap are apt to determine forest composition.

The observed age distribution of gap area, based on the fraction of land in gaps of each age, will be used to determine rates of gap birth and death; the concern here is with total land area in gaps of a certain age, not with amount of area per gap. For these analyses, only the canopy gap, the area directly under the canopy opening, was used. Table 3 gives the age distribution of land area in gaps for each study area individually. It is apparent that no area was in perfect equilibrium with respect to rates of disturbance (gap birth). Also, peak years of gap formation showed little regional synchrony; Great Smoky Mountains (GSM) 6 and GSM9, separated only by a large stream, had quite different distributions of gap age and area.

Lateral extension growth.—During the 1976 field season 384 trees bordering gaps were selected and their dbh measured. A vertical projection of the total lateral extent from the bole to the furthest extension of the crown into the gap was measured for each tree. The data were fit to the regression equation developed by Trimble and Tryon (1966):

TABLE 4. Average rates of lateral extension growth from the literature and from the regression equation: lateral extent (m) = $A + B^*$ Gap Age (yr) + C^* dbh (cm). Numbers missing from the table are not available from the references cited.

Species	A	B	C	r^2	P	Lateral extension growth (cm/yr)	Reference
All	2.42	.035	.017	6	.0001	4.1	Present study
Acer saccharum	0.84	.073	.041	25	.0002	8.3	Present study
Tsuga canadensis	1.06	.063	.021	25	.0001	7.0	Present study
Liriodendron tulipifera	0.90	.044	.066	22		9.4	Trimble and Tryon (1966)
Quercus rubra	0.14	.082	.109	48		16.5	Trimble and Tryon (1966)
Juglans nigra:							
Undisturbed						2.0	Phares and Williams (1971)
Partly released						5.5	Phares and Williams (1971)
Completely released						7.5	Phares and Williams (1971)
Betula lutea:							
16-yr-old stand						18–25	Erdmann et al. (1975)

$$\text{Lateral extent (m)} = A + B \times \text{gap age (yr)} + C \times \text{dbh (cm)}.$$

The annual increase in tree stem diameter was estimated using tree cores selected from canopy individuals sampled to calculate gap age. Overall values (mean ± SE) for stem diameter growth for years following release were 0.36 ± 0.026 cm/yr for a mix of species, 0.32 ± 0.024 cm/yr for hemlock, 0.24 ± 0.026 cm/yr for sugar maple, and 0.86 ± 0.058 cm/yr for tulip tree. Incorporating these results into the regression equations gave average rates of lateral extension growth per year (Table 4). Overall, a growth rate of 4 cm/yr was obtained, although hemlock and sugar maple grew about twice as rapidly. The values obtained were similar to other values in the literature (Table 4).

To determine the effect of lateral extension growth on each study area, gap dimensions were reduced by 4 cm on each side and paces in each gap were reduced by the fraction of gap area that had disappeared. For the 14 study areas, lateral extension growth filled in from 1.4 to 2.7% (average 1.9%) of total gap area each year.

Regeneration height growth and gap closure rates.—The rate at which gaps closed by the growth in height of new or formerly suppressed individuals was estimated in two ways. First, literature estimates of sapling height growth rates following cutting of the overstory were used to derive values for maximum expected time until disappearance of a gap. Second, the observed age distribution of gap area was used to approximate a survivorship function, from which an average rate of disappearance of gap area was computed.

For this study, a maximum value for gap longevity was the time required for new saplings to reach a height of 10–20 m. Many studies on natural growth rates for many species from different areas in the eastern deciduous forest show average growth rates of 0.5–1.0 m/yr following cutting or in naturally created openings (e.g., Kramer 1943, Downs 1946, Kozlowski and Ward 1957, Tryon and Trimble 1969, Marks 1975). Gaps should close even faster because they contain some advance regeneration when formed and because taller individuals grow faster than the rates given above (Laufersweiler 1955, Burton et al. 1969, Tubbs 1977*b*). In general, sprouting was rare or absent in most of the gaps observed. Using a minimum rate of growth in height of 0.5 to 1.0 m/yr and the regeneration height limit of 10–20 m mentioned earlier results in a range of maximum possible gap ages of 10–40 yr.

A more exact method of estimating the rate of gap closure used the observed age distributions of gap area (Table 3). For any group of gaps created during the same year, relatively little gap area will fill in the first few years because the regeneration for the most part will be small. However, on occasion large, formerly suppressed, individuals can eliminate some area even in young gaps. For the next few years, the regeneration in most gaps reaches a height at which gap area is converted rapidly into the background vegetation. Finally, although the annual survival rate of gap area may continue to decrease, the fraction of total land area converting from gaps to the background will decrease due to the relatively small fraction of land area that consists of old gaps.

TABLE 5. Parameters for the logistic model of gap area, $N(t,a) = (49Ke^{r} - 50K)/(49e^{r} - 50 + e^{ra})$, where $N(t,a)$ is the percentage of land area in canopy gaps of age a at time t, and K and r are fitted constants. Stand symbols are explained in Table 1.

Stand	K	r	Inflection point (yr)	$N(t,1)$
GSM1	1.84	0.306	9.2	1.70
GSM3	2.97	1.487	3.4	2.89
GSM4	2.50	0.256	10.1	2.28
GSM5	1.20	0.121	13.8	0.84
GSM6	0.94	0.160	12.6	0.81
GSM7	0.87	0.172	12.2	0.76
GSM9	1.37	0.558	6.4	1.31
GSM10	2.43	0.465	7.2	2.30
JK	3.67	1.263	3.8	3.57
WC	1.01	0.358	8.4	0.94
HW	0.91	0.420	7.6	0.86
TA	0.60	0.325	8.9	0.56
WB	0.88	1.010	4.4	0.85
HK	0.47	0.213	11.1	0.42
GSM1–10,JK	1.49	0.291	9.4	1.37

Of several possible approaches to this process the logistic equation was examined in detail. Assume that the fraction of gap area surviving from age a to age $a + 1$ is independent of a:

$$\frac{dN(t,a)}{da} = -r_c N(t,a)$$

where $N(t,a)$ is the fraction of total land area in gaps of age a at time t, and r_c is a constant rate of gap closure. Next, let the rate of gap closure increase as the fraction of total land area in gaps decreases; when total gap area is small, gaps tend to be older and so should be closing more rapidly due to sapling height growth. A linear relationship will be used as a first-order approximation. Making r_c a linear function of $N(t,a)$ results in

$$\frac{dN(t,a)}{da} = -\left[r - \frac{r}{K}N(t,a)\right]N(t,a)$$

where r and K are constants. From a standard table of integrals, this equation has the following solution:

$$N(t,a) = \frac{K}{1 + e^{ra+b}}$$

where b is a constant. The assumption that at first, from age $a = 0$ to age $a = 1$, lateral extension growth is the only process of importance, in accordance with the pattern of change hypothesized above, gives

TABLE 6. Estimated birth rates (percent of total land area per year) for canopy gaps for each stand. Canopy gap dimensions were increased to compensate for lateral extension growth and then the total revised gap areas in each study area for the most recent 1-, 5-, and 10-yr periods were averaged together. Stand symbols are explained in Table 1.

Stand	Averages over		
	1 yr	5 yr	10 yr
GSM1	4.0	1.1	1.4
GSM3	1.4	1.7	0.9
GSM4	6.6	2.2	2.0
GSM5	0.7	0.3	0.9
GSM6	2.0	0.9	0.7
GSM7	0	0.4	0.6
GSM9	0.4	1.3	0.9
GSM10	2.1	1.8	1.7
JK	0	2.4	1.6
WC	0.5	1.0	0.8
HW	0.2	0.8	0.6
TA	0.2	0.5	0.4
WB	0.5	0.7	0.3
HK	1.3	1.2	1.1
GSM1–10,JK	2.0	1.2	1.1

$$.98 = \frac{N(t,1)}{N(t,0)}.$$

Solving for e^b and substituting back into the equation for $N(t,a)$ results in

$$N(t,a) = \frac{49Ke^r - 50K}{49e^r - 50 + e^{ra}}.$$

This equation was tested for goodness of fit to each study area using the least squares nonlinear procedure of the SAS statistical computer package (Barr et al. 1976), which also computed best fit estimates for r and K. F tests showed all the regressions but one (the Huyck Preserve, with its small sample size) to be highly significant ($P \leq .01$).

The inflection point or age at which gap area was being converted most rapidly into the background vegetation (defined here as 10–20 m tall) was found by solving for the second derivative of the preceding equation, resulting in

$$a = \frac{1}{r}\ln(49e^r - 50).$$

The value of a, the inflection point, was computed for each study area (Table 5); the average value of the 14 study areas was 8.5 yr, a reasonable result given the average rates of sapling height growth discussed previously.

A gap aged a is a fraction $N(t,a)/N(t,1)$ of its original size. An average annual survivorship rate may be computed by assuming the gap loses a constant fraction of its area each year. The fraction of gap area which survives each year (for $a - 1$ years) can be determined from the following equation:

$$\bar{S}_a = \left(\frac{N(t,a)}{N(t,1)}\right)^{1/(a-1)}.$$

This term was decomposed into a survivorship rate from lateral extension growth, $S_L = .98$ (which value should remain roughly constant), and a survivorship rate from sapling height growth, $\bar{S}_{H,a} = \bar{S}_a/S_L$. As an example of the relative importance of these two processes for gaps of different ages, the following results for the southern Appalachians (Great Smoky Mountains and Joyce Kilmer) were obtained:

a	$\bar{S}_a$	$\bar{S}_{H,a}$
2	.98	1.00
5	.96	.98
10	.93	.94
15	.88	.90
20	.85	.87
30	.82	.83

Therefore, after the first few years $\bar{S}_{H,a} < S_L$; i.e., sapling height growth is the more important means of gap closure, implying that sapling growth within even fairly small gaps may be important in determining forest composition.

In using the observed age distribution as a survivorship function it is assumed that the age distribution was approximately stable and stationary, having no major directional changes in gap birth rate. Several factors supported this assumption. First, predictions of the model agreed well with literature values concerning average rates of sapling height growth. Second, results from different study areas were similar, implying that a biological process more basic than random fluctuations was operating. Third, all areas but the one least sampled showed highly significant ($P \leq .01$) fits to the distribution, implying that it was related to a real biological phenomenon.

In addition, homeostatic mechanisms tend to keep the gap age distribution from fluctuating too greatly. The total fraction of land in gaps at time t, $M(t)$, should vary with rates of gap birth and death as follows:

$$dM(t)/dt = B'(t)[1 - M(t)] - D(t)M(t)$$

where $B'(t)$ is the fraction of area not in gaps which is converted into gaps at time t and $D(t)$ is the fraction of gap area which is converted into the background vegetation at time t. Thus, after several years of excessively high disturbance rates, $M(t)$ should be high, $B'(t)[1 - M(t)]$ should be relatively low, and $D(t)M(t)$ should be relatively high, resulting in a gradual decrease in $M(t)$ until more normal values are obtained. Also, as those trees most susceptible to disturbance are eliminated, the remaining individuals should be more resistant.

Birth rate.—The rate at which gaps were formed was estimated in several different ways. The most direct measure was the fraction of total land area covered by gaps ≤ 1 yr old. However, gap birth rates varied from year to year and so some sort of time averaging was necessary. A problem with time averaging was that original gap areas were not known exactly but had to be estimated from the rate of closure discussed previously.

One approach was to increase gap dimensions (length and width) by 8 cm (Table 4) for each year the gap existed. Thus a gap aged 10 yr was assumed to have been 80 cm longer and wider when formed and the original gap area was calculated using these new dimensions. The number of paces in each gap was increased in proportion to this increase in size. All these paces within one study area were summed to result in a new gap age distribution, based on estimated original gap sizes. These estimates of original gap area were averaged for the most recent 5- and 10-yr periods (Table 6). Averages for 5 yr are probably the best available estimates of gap birth rate. Averages for 10 yr are less accurate due to an increase in gap closure by sapling height growth.

A second approach was to use the model described previously (Table 5), letting $a = 1$. In general all methods gave similar estimates, both in actual value and in the relative magnitude of disturbance rates in the different areas. Gap birth rate values from study areas in the southern Appalachians ranged from 0.3 to 3.6%, using different methods, with an average of about 1.2 to 1.7%. Hueston Woods averaged about 0.7 to 0.8% per year of new gaps; Tionesta, 0.5 to 0.7%.

As a check on these values, 54% of the 1976 transect distance was repaced in 1977, resulting in 10 new gaps, for which the canopy gaps made up 1.2% of the ground surface area.

Species responses to gaps

How did different species respond to the variations in gap size and age described above? To help answer this question, Gaussian curves for species were fit using either gap size or age as the abscissa. For gap size, expanded gap area was used in order to include more completely the direct and indirect effects of the gap on forest regeneration. Four measures of species importance were used: total basal area (sum of basal areas of all individuals of the given species within the gap), total number of stems (total number of stems of the given species ≥ 1 m high within the gap), relative basal area (total basal area of the given species divided by the sum of total basal areas of all species), and relative number of stems. The data were further divided into gaps from three major geographic regions: Tionesta, Hueston Woods, and the southern Appalachians (Great Smoky Mountains, Joyce Kilmer, and Walker's Cove). Gaussian curves also were fit for several gap community properties. In all cases Gaussian curves were fit using Cornell Ecology Program 12 (Gauch and Chase 1974). This program computed the percentage of variance accounted for by each fitted curve and for all curves together as a measure of goodness of fit. F values were computed as follows:

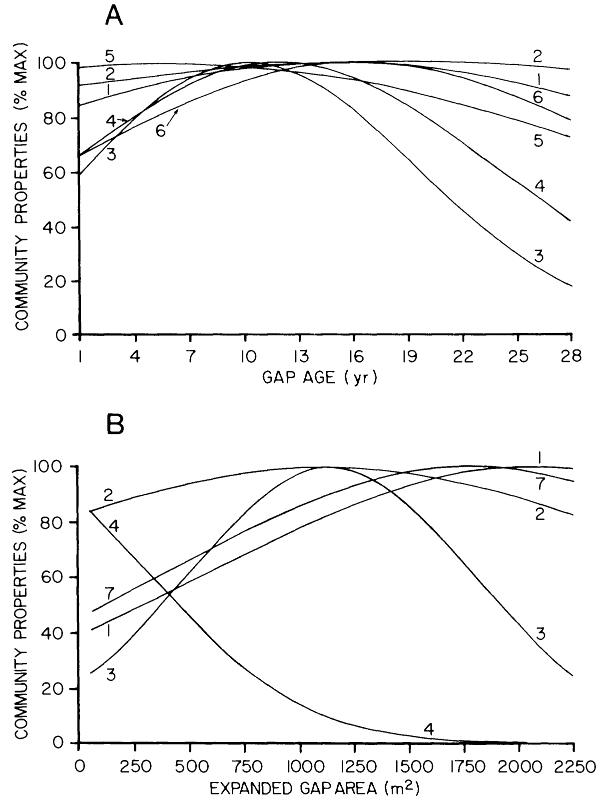

FIG. 1. Fitted Gaussian curves for properties of woody stems within each gap as a function of gap age (A) and expanded gap area (B) for the southern Appalachians as a whole. Community properties included are (1) number of woody species with individuals at least 1 m high; (2) complemented Simpson index, $DS = 1 - \sum_i P_i^2$, where P_i is the average of relative number of stems and relative basal area for sapling species i; (3) total number of stems ≥ 1 m high; (4) density, i.e., number of stems ≥ 1 m high divided by expanded gap area; (5) fraction of stems ≥ 1 m high but < 1.0 cm dbh (this property is plotted only for part A); (6) fraction of stems 1.0–2.5 cm dbh (this property is plotted only for part A); and (7) total basal area of all stems (this property is plotted only for part B).

$$F_{2,n-3} = \left(\frac{PV}{1 - PV}\right)\left(\frac{n - 3}{2}\right),$$

where n was the number of points (gaps) in the sample and PV was the fraction of total variance (corrected for the mean) accounted for by the Gaussian model. This term underestimated the significance of the results because the error mean square probably was inflated by unaccounted for but real factors such as differences in elevation, soils, topography, or geography.

Curves of several community properties vs. gap size and age were fitted for the southern Appalachians, in

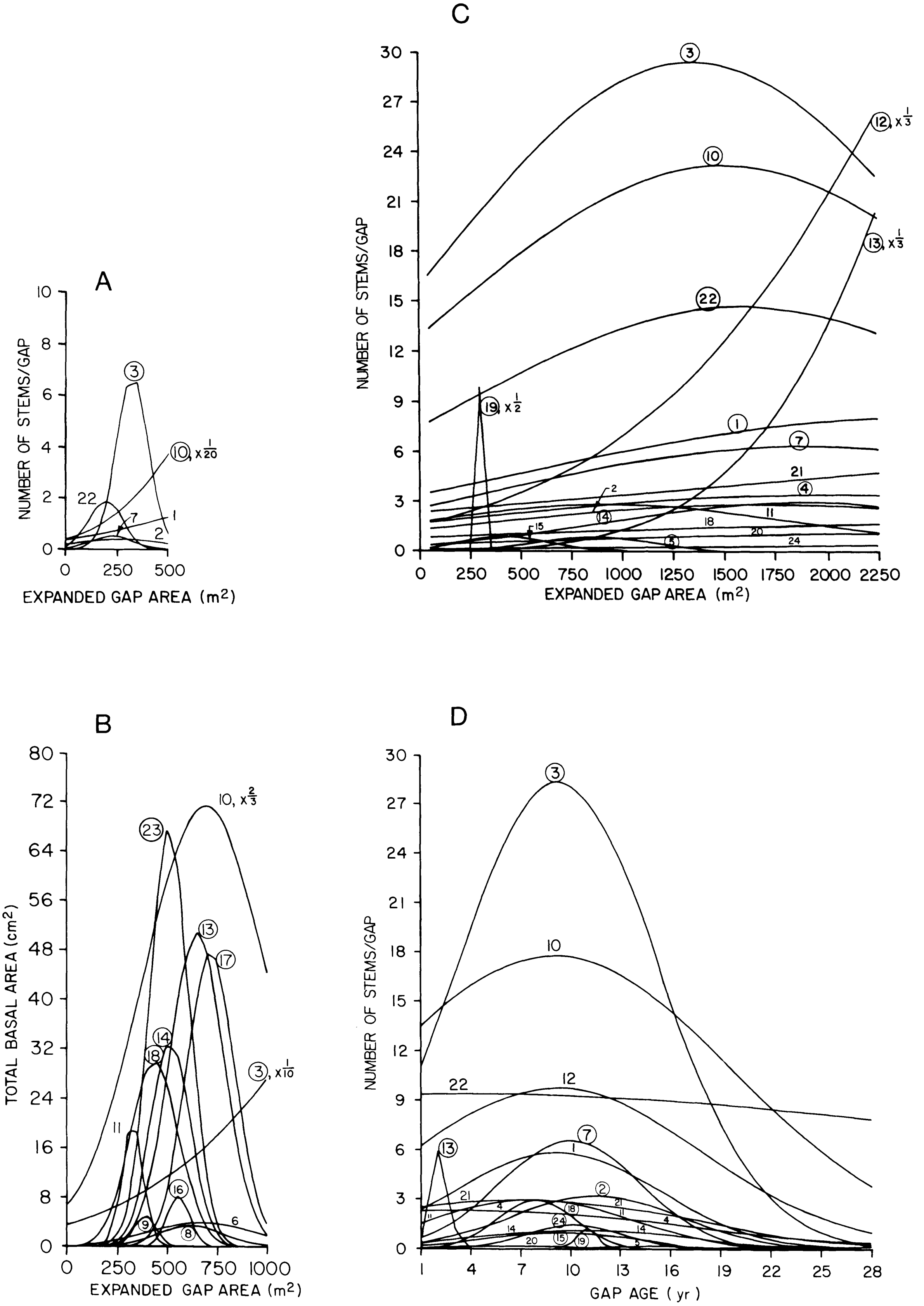
A
NUMBER OF STEMS/GAP
EXPANDED GAP AREA (m^2)
③
⑩, x$\frac{1}{20}$
22
7
1
2
B
TOTAL BASAL AREA (cm^2)
EXPANDED GAP AREA (m^2)
10, x$\frac{2}{3}$
㉓
⑬
⑰
⑭
⑱
11
⑯
③, x$\frac{1}{10}$
⑨
⑧
6
C
NUMBER OF STEMS/GAP
EXPANDED GAP AREA (m^2)
③
⑫, x$\frac{1}{3}$
⑩
⑬, x$\frac{1}{3}$
㉒
⑲, x$\frac{1}{2}$
①
⑦
21
④
2
⑭
15
11
18
20
24
⑤
D
NUMBER OF STEMS/GAP
GAP AGE (yr)
③
10
22
12
⑦
1
⑬
21
4
②
⑱
11
㉔
14
20
⑮
⑲
5

TABLE 7. Variance accounted for (%) by fitting Gaussian curves to sets of species importance values. EG = Expanded gap area. Significance values are symbolized by * for $.01 < P \leq .05$, ** for $P \leq .01$.

	Percent variance accounted for					
	Hueston Woods		Tionesta		Southern Appalachians	
Measure of species importance	Age	EG	Age	EG	Age	EG
Relative density	2	5	1	3	1	1
Relative basal area	4	5	2	2	1	1
Total number of stems	4	20*	0	49**	3*	7**
Total basal area	8	37**	2	17**	1	6**

which the sample size was sufficient to detect significant relationships accounting for only 2–30% of the total variance (Fig. 1). Gap size and age varied more or less independently; correlations between them were very low. As gap size increased, the number of species increased and the concentration of dominance decreased. Total basal area and number of stems increased for most of the range of gap sizes encountered. The decrease in these terms for very large gaps may have been an artifact since few large gaps were sampled. Sapling density (number per square metre) decreased as gap size increased, however, probably because an increasing fraction of the ground surface was covered by fallen boles, branches, and other debris.

Gaps of different ages were interpreted generally to form a single chronological sequence. However, different gaps filled in at different rates, and gaps that were detectable but relatively old ($>$15 yr, say) were in some sense peculiar or else they would already have disappeared. Therefore the response of community properties and species for very old gaps was interpreted with caution. New species and new individuals were added to the gap for 10–15 yr after gap formation (Fig. 1). Although only individuals $\geq$1 m high were measured, these results imply that gaps were open to invasion by new individuals for several years. Whether such new individuals could outcompete those already established is questionable, however. Finally, the fraction of stems $\geq$1.0 m high but $<$1.0 cm dbh increased for 5 yr, after which it declined and the fraction 1.0–2.5 cm dbh increased.

The responses of individual species to differences in gap size and age also were examined, choosing those relationships shown to be most significant in Table 7 (Fig. 2). In Tionesta the number of stems of beech, the dominant species, increased with increased gap size for the range of values recorded. Most other species peaked in number of stems at intermediate values of gap size, perhaps because the higher overall number of stems in larger gaps attracted more deer, favoring the relatively unpalatable beech (Bjorkbom and Larson 1977). Hueston Woods also showed a general direct relationship between number of stems and gap size for most species. The final increase in sugar maple and decrease in most other species may have occurred because the two largest gaps both were created at most 3 yr before my sampling, and relatively few species (other than sugar maple) were abundant. In the southern Appalachians, also, larger gaps contained more individuals of most species; however, few very large gaps were sampled. Densities (number of stems per square metre) for most species were greater in small gaps than in large gaps. Although large gaps probably had more favorable growth conditions and so might be expected to have had higher sapling densities than small gaps, large gaps also had relatively greater area unavailable to sapling growth due to fallen boles, branches, and leaves. In the southern Appalachians most species reached their maximum densities at gap ages of 7–12 yr, in good agreement with the rates of gap closure estimated earlier.

To what degree did species respond individualistically to differences in gap age and size? No two species had identical curves (Fig. 2). However, the variance in the curves was large and much overlap among species existed. Also, in no case was the overall pattern of variation in relative number of stems or relative basal areas significant (Table 7). The dominant species were found in gaps of all ages and sizes.

To examine different species patterns further, weighted average ordinations were run using Cornell Ecology Program 25B (Gauch 1977). Only species occurring in at least 10% (41) of the total number of gaps sampled were used. Their importance (measured as the average of relative number of stems and relative

←

FIG. 2. Fitted Gaussian curves for species importance values: (A) Tionesta, total number of stems vs. expanded gap area; (B) Hueston Woods, total basal area (cm^2) of stems vs. expanded gap area; (C) southern Appalachians as a whole, total number of stems vs. expanded gap area; and (D) southern Appalachians as a whole, total number of stems vs. gap age. Species are, by number, (1) *Acer pensylvanicum*, (2) *A. rubrum*, (3) *A. saccharum*, (4) *Aesculus octandra*, (5) *Aralia spinosa*, (6) *Asimina triloba*, (7) *Betula* spp., (8) *Carya cordiformis*, (9) *Celtis occidentalis*, (10) *Fagus grandifolia*, (11) *Fraxinus americana*, (12) *Halesia carolina*, (13) *Lindera benzoin*, (14) *Liriodendron tulipifera*, (15) *Magnolia acuminata*, (16) *Morus rubra*, (17) *Ostrya virginiana*, (18) *Prunus* spp., (19) *Pyrularia pubera*, (20) *Sambucus pubens*, (21) *Tilia heterophylla*, (22) *Tsuga canadensis*, (23) *Ulmus rubra*, and (24) *Viburnum alnifolium*. Curves which are significant ($P \leq .05$) have the species number circles. Fractions ($\times{}^1/_{20}$, $\times{}^2/_3$, etc.) indicate extent to which the amplitude of a curve has been reduced from its original value to fit on the same scale as the other curves.

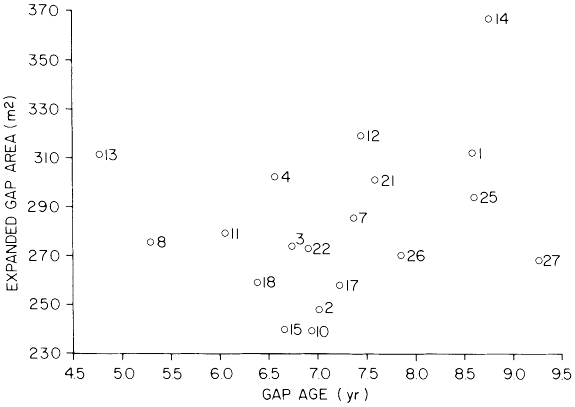

FIG. 3. Distribution of species present in ≥10% of all gaps in relation to gap age and expanded gap area, based on averages of species importance values weighted by gap size and age. Species are numbered as in Fig. 2, with additional species (25) *Amelanchier arborea*, (26) *Magnolia fraseri*, and (27) *Rhododendron maximum*.

basal area) in each gap was weighted by the age of the gap for the first ordination axis and by the expanded gap area for the second ordination axis. Results (Fig. 3) show the scattering of species along one primary gradient, from species reaching maximum importance in small young gaps (understory tolerants, e.g., beech) to those doing best in large old gaps (opportunists, e.g., tulip tree). The correlation coefficient for the two axes is $r = .262$, resulting in $F(1,17) = 4.45$, significant at $P \leq .05$. Two shrubs had somewhat anomalous response patterns. *Lindera benzoin* grew rapidly in large gaps but was overtopped by tree saplings and so did relatively better in young gaps. *Rhododendron maximum*, a shade-tolerant species able to expand vegetatively, did well in fairly small gaps but reached maximum importance in relatively old gaps. It may have grown fairly rapidly even in small gaps; an alternate explanation, however, is that its presence inhibited the growth of other species, so that gaps which were relatively old but still recognizable tended to be those in which site conditions were favorable for rhododendron.

In addition to having somewhat different responses to gap size and age, species also showed differences in the types of injuries they received when they created gaps (Table 8). Only 19% of the gapmakers were uprooted. More commonly trees broke off at some height above ground, about evenly divided between breaks >2.5 m high (28%) and ≤2.5 m high (30%). Finally, about equal numbers died standing (10%) or contributed to a gap by losing large branches though remaining alive (13%). Several species showed significant ($P \leq .05$) propensities for certain types of injuries. Beech was partly uprooted 26% of the time vs. 19% for all species, perhaps due to its shallow spreading root system (Fowells 1965). Living red (*A. rubrum*) and sugar maples and white ash contributed to gaps relatively more than did living trees of other species. Hemlock was less likely to be totally uprooted (7 vs. 14% for all species) but more likely to break at ≤2.5 m (37 vs. 30% for all species). The existence of many snags has important implications for wildlife, a topic of much current interest (Hardin and Evans 1977, Scott et al. 1977, Evans and Conner 1979).

DISCUSSION

The observed gap birth rates of ≈1%/yr (ranging from ≈0.5 to ≈2%/yr for large samples) were similar to disturbance rates for northern conifer forests (Heinselman 1973, Zackrisson 1977), an old-growth beech-maple forest in Indiana (Abrell and Jackson 1977), and tropical rainforests (Leigh 1975, Hartshorn 1978). Inverting the figure for gap birth rate resulted in a natural rotation time, that is, a measure of the average number of years required in nature to regenerate an area equal to the total area under consideration (cf. Heinselman 1973). Both the present study and those studies cited above gave a natural rotation of ≈100 yr, varying from ≈50 to ≈200 yr.

Two questions emerged from these results. First,

TABLE 8. Gapmakers: frequencies (%) for species-injury classes for species represented by more than four trees. Significance levels are marked † for $.05 < P \leq .10$, * for $.01 < P \leq .05$, and ** for $P \leq .01$.

Species	*N*	Alive but injured	Standing dead	Snag >2.5 m high	Snag ≤2.5 m	Partly uprooted	Uprooted
Acer rubrum	9	44*	11	0	22	0	22
A. saccharum	64	20†	12	27	27	3	11
Aesculus octandra	17	18	6	41	12	6	18
Betula lutea	19	16	21	26	26	5	5
Castanea dentata	7	0	13	12	25	0	50*
Fagus grandifolia	237	11	10	25	29	7	19**
Fraxinus americana	8	62**	12	0	12	0	12
Halesia carolina	71	13	7	34	24	6	17
Magnolia fraseri	13	8	0	38	46	8	0
Tilia heterophylla	47	4	2†	40†	40	6	6
Tsuga canadensis	163	10	12	28	37*	6	7*
All	674	13	10	28	30	6	14

TABLE 9. Longevities at key sizes for canopy trees. Minimum, average, and maximum sizes of gapmakers are taken from the present study. Relationships between tree size and age are taken from the literature cited below. Stand symbols are explained in Table 1.

Source of size-age relationships	Stands to which relationship applies	Species	Average age (yr) at 25 cm dbh	Years from 25 cm dbh to average dbh of gapmakers	Age of largest gapmaker (yr)
R. H. Whittaker (personal communication)	GSM	All	91	127	441
		Acer saccharum	79	64	197
		Aesculus octandra	95	141	431
		Fagus grandifolia	101	54	220
		Halesia carolina	78	51	201
		Liriodendron tulipifera	55	153	226
		Tilia heterophylla	54	49	198
		Tsuga canadensis	106	211	525
Morey (1936*a*)	TA	*Tsuga canadensis* (two sites)	185, 119	181, 84	607, 315
		Fagus grandifolia (two sites)	179, 122	93, 85	412, 334
		Betula lutea	100	110	251
Tubbs (1977*a*)	HW	*Acer saccharum* (virgin stand)	177	78	372
		Acer saccharum (managed stand)	93	67	260
Gates and Nichols (1930)	HW	*Acer saccharum*	129	40	229
	TA	*Tsuga canadensis*	165	107	415

how could one reconcile a 100-yr rotation with the fact that dominant forest trees are known to live for much longer periods? Second, were the observed similarities in yearly disturbance rates among the several different communities solely a matter of coincidence or was some underlying mechanism responsible?

The first question was answered partially as follows. The rotation time was not equivalent to the total longevity of a canopy tree but to the average time a tree was canopy size and capable of creating a gap. Therefore, rotation time was approximately equal to the difference in tree age between the time the average tree entered the canopy and the time it died. Setting 25 cm dbh as the approximate lower limit for canopy trees worked fairly well. Of 2921 trees recorded using the point-centered quarter method (nearest four trees either alive and ≥25 cm dbh, or dead and contributing to a gap), only seven individuals were ≥25 cm dbh and dead without creating a gap, and only two individuals <25 cm dbh created gaps. The data from this study on 666 gapmaking individuals ≥25 cm dbh provided an estimate of the size at which trees reaching the canopy die. These sizes were converted to ages using relationships given in the literature and then into the time it took an individual to grow from 25 cm dbh to the average diameter at death for the given species and region (Table 9). The values obtained agree well with the previously estimated natural rotation period of 50–200 yr.

Two factors produce maximum ages greater than these values. First, most of the important species can persist for many years under a closed canopy, growing very slowly (Table 9). Scattered pole-sized survivors were found in many of the gaps studied and undoubtedly were important in gap closure. Second, gaps can occur on one site several times before they occur on a second site. Therefore, individuals on some locations can live longer than the average (Table 9).

Reasons for similarities among different forest types were investigated using a simple model. If an area were subjected to an average rate of disturbance x (fraction of land area per year), a fraction y of the total area would not be affected by disturbances of age $\leq a$. These parameters were related as follows:

$$(1 - x)^a = y$$

This model assumed that the probability of any point undergoing disturbance was independent of the time since last disturbance, with some points likely to undergo a disturbance many times while others remained undisturbed. Table 10 gives the minimum age (a) of five fractions of the stand, $y = 50\%$ through 0.01%, for several rates of disturbance (x). For instance a birth rate of 1%/yr would result in 50% of the stand being over 70 yr old, 10% over 229 yr old, and 1% over 458 yr old. The age at which only 0.01–1% of the stand had not undergone disturbance would approximate the normal maximum life span of the forest dominants. Much literature, for both temperate (Jones 1945, Fowells 1965) and tropical regions (Budowski 1965, Ashton 1969), has suggested that forest dominants usually have life spans of 100–1000 yr. These values correspond to disturbance rates of about 0.5–2.5%/yr, similar to those values given above.

It is unclear whether internal (physiological or structural) constraints or external forces were more important as causes of mortality, although both probably were involved (Bormann and Likens 1979, White 1979). For a forest to maintain itself, disturbance rates need to be low enough so that trees can reach maturity

TABLE 10. Hypothetical gap birth rates (percent of total land area) with expected age distribution for land area.

Birth rate (%)	% of stand at least given age (yr) 50%	10%	1%	.1%	.01%
	Age (yr)				
0.1	693	2301	4603	6904	9206
0.3	231	766	1533	2299	3066
0.5	138	459	919	1378	1837
1.0	70	229	438	687	916
1.5	46	152	305	457	609
2.0	34	114	228	342	456
2.5	27	91	182	273	364
5.0	14	45	90	135	180
10.0	7	22	44	66	87
20.0	3	10	21	31	41

TABLE 12. Observed size distribution for all gaps taken together.

Size class maximum value (m^2)	Canopy gaps Number	Canopy gaps % land area	Expanded gaps Number	Expanded gaps % land area
25	84	0.89	0	0
50	72	1.04	2	0.04
75	57	1.11	13	0.49
100	67	1.92	30	1.12
150	52	1.40	59	2.47
200	28	0.82	70	3.19
400	32	1.28	171	8.98
700	11	0.70	44	3.08
1000	2	0.13	12	0.97
1500	1	0.20	4	0.38
2500	0	0	1	0.26
Sum	406	9.50	406	20.98

and reproduce. On the other extreme, as trees increase in size (age), they decrease in the efficiency of transporting water, nutrients, and photosynthate (Spurr and Barnes 1973, Oldeman 1978), in the favorability of the root/shoot ratio (Borchert 1976), and in the ratio of photosynthetic to nonphotosynthetic tissue (Harper 1977). The net result is a decreased ability to withstand climatic extremes and an increased susceptibility to attack by insects and fungi. Therefore, even in the absence of any severe disturbance, canopy trees in general have only a restricted range of possible longevities, and forests in approximate equilibrium have only a narrow range of possible disturbance rates, falling near 1%/yr.

The preceding analysis implies that a forest's response to disturbance depends not so much on the average rates of disturbance as on the distribution of disturbance in time and space.

To compare openings of different sizes, the most meaningful measure of gap size is the ratio of the diameter D of the gap to the mean height H of the surrounding stand. Several studies have shown that both light and soil moisture in the center of the gap increase as this ratio increases, leveling off when D/H reaches ≈ 2 (Geiger 1965, Minckler and Woerheide 1965, Minckler et al. 1973). For the present study, average stand heights were estimated to be 32 m in the southern Appalachians, 27 m in Hueston Woods, and 25 m for the other northern sites. These estimates are based on occasional direct measurements using an optical rangefinder, lengths of fallen trees, and some literature values (Whittaker 1966). The average of gap width and length was used for the gap diameter. For most gaps $D/H \leq 0.5$, although 18% of the gaps had higher values, one with $D/H = 1.6$ (Table 11).

TABLE 11. Frequency distribution for canopy gap diameter (D)/canopy height (H). Canopy gap segments distinct enough to warrant individual dimensions were treated as separate gaps.

D/H class maximum value	Number of gaps
0.2	89
0.4	189
0.6	99
0.8	30
1.0	8
1.1	2
1.6	1

A great many forestry studies and general reviews state that the selective cutting of individual trees will favor shade-tolerant species such as beech, hemlock, and sugar maple, at the expense of light-demanding species such as black cherry, white ash, tulip tree, and yellow birch (e.g., United States Department of Agriculture 1973, McCauley and Trimble 1975, Leak and Filip 1977, Tubbs 1977*b*). However, openings as small as 400 m^2 have been found sufficient for tulip tree and yellow birch to maintain themselves in a forest (Merz and Boyce 1958, Tubbs 1969, Trimble 1970, Schlesinger 1976).

If "opening," as defined by foresters, is equivalent to "canopy gap," then 1.03% of the land area was in gaps greater than the 400-m^2 limit given above (Table 12). If "opening" is equivalent to "expanded gap," then 4.69% of the total land area was in gaps of the appropriate size. In either case the observed size distribution seemed sufficient to allow some light-demanding species to persist in these forests.

CONCLUSIONS

In areas of deciduous forest protected from large-scale disturbances of wind or fire, disturbances on the scale of a single dead tree made up a significant fraction of the total land area. Gaps in the forest canopy closed primarily due to the height growth of sapling or subcanopy individuals, not to the lateral spread of other canopy trees. Therefore, even small disturbances provided regeneration possibilities for forest species.

Species responses to the regeneration opportunities varied. Tolerant species were present as suppressed saplings before the gap was formed and dominated small, young gaps. Although these species also were abundant in larger, older gaps, their relative importance was lower because of the increased success of opportunists (species unable to survive under a closed canopy or in small gaps but able to grow rapidly in larger gaps), which became more important with time. The observed disturbance regime strongly favored tolerant species but allowed opportunists to persist in low densities.

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LETTER

The Affordable Clean Energy rule and the impact of emissions rebound on carbon dioxide and criteria air pollutant emissions

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Abstract

The Affordable Clean Energy (ACE) rule, the US Environmental Protection Agency's (EPA) proposed replacement of the Clean Power Plan (CPP), targets heat rate improvements (HRIs) at individual coal plants in the US. Due to greater plant efficiency, such HRIs could lead to increased generation and emissions, known as an emissions rebound effect. The EPA Regulatory Impact Analysis for the ACE and other analyses to date have not quantified the magnitude and extent of an emissions rebound. We analyze the estimated emissions rebound of carbon dioxide (CO_2) and criteria pollutants sulfur dioxide (SO_2) and nitrogen oxides (NO_X), using results from the EPA's power sector model, under the ACE in 2030 at model coal plants and at the state and national levels compared to both no policy and the CPP. We decompose emissions changes under a central illustrative ACE scenario and find evidence of a state-level rebound effect. Although the ACE reduces the emissions intensity of coal plants, it is expected to increase the number of operating coal plants and amount of coal-fired electricity generation, with 28% of model plants showing higher CO_2 emissions in 2030 compared to no policy. As a result, the ACE only modestly reduces national power sector CO_2 emissions and increases CO_2 emissions by up to 8.7% in 18 states plus the District of Columbia in 2030 compared to no policy. We also find that the ACE increases SO_2 and NO_X emissions in 19 states and 20 states plus DC, respectively, in 2030 compared to no policy, with implications for air quality and public health. We compare our findings to other model years, additional EPA ACE scenarios, and other modeling results for similar policies, finding similar outcomes. Our results demonstrate the importance of considering the emissions rebound effect and its effect on sub-national emissions outcomes in evaluating the ACE and similar policies targeting HRIs.

1. Introduction

The United States Environmental Protection Agency (EPA) in August 2018 released its proposed Affordable Clean Energy (ACE) rule. The ACE is the proposed replacement to the existing EPA Clean Power Plan (CPP), the carbon dioxide (CO_2) emissions standard for existing power plants. EPA has a legal obligation to regulate greenhouse gas emissions from existing power plants, which was affirmed by the Supreme Court's 2007 decision in *Massachusetts v Environmental Protection Agency* and triggered by the EPA's formal finding in 2009 that greenhouse gas emissions endanger public health and welfare (Massachusetts v Environmental Protection Agency (EPA) 2007, US Environmental Protection Agency (EPA) 2009).

The CPP was finalized in 2015 and established state-based CO_2 emissions goals for affected fossil fuel-fired power plants. The CPP identifies a number of flexible compliance options as part of the 'best system of emissions reductions' (BSER) that the EPA is charged with identifying under section 111(d) of the Clean Air Act. It allows emissions reductions to come from carbon intensity reductions at individual plants—including heat rate improvements (HRIs) or fuel cofiring at the source—or from the substitution of generation towards less carbon-intensive and zero-carbon energy sources (US Environmental Protection Agency (EPA) 2015a). Averaging across electricity generating units (EGUs) and intra- and inter-state trading among units are also allowed. Given the flexible compliance structure, the CPP can be termed a 'systems-based' standard. At the time it was finalized, it was estimated that the CPP would decrease CO_2 emissions by 415 million tons, or 19%, below a business as usual base case level, or 32% below 2005 levels, by 2030 (US Environmental Protection Agency (EPA) 2015b).

The proposed ACE instead employs a narrow 'source-based' regulation, which defines and limits the legally relevant BSER as HRI opportunities at individual coal plants (US Environmental Protection Agency (EPA) 2018a). Heat rate is the amount of fuel input (Btu) used to produce a kWh of electricity; a lower heat rate indicates a more efficient unit, which emits less CO_2 per kWh. As a general rule of thumb, a reduction of 10 million Btu equals roughly a one-ton reduction in CO_2 for coal EGUs. There is considerable heterogeneity in the heat rate of US coal plants and substantial opportunity to make coal plants more efficient Massachusetts Institute of Technology (MIT) 2009, Sargent and Lundy 2009, SFA 2009, DiPietro and Krulla 2010, Campbell 2013, Linn *et al* 2014, Staudt and Macedonia 2014). The ACE sets standards for emissions rate improvements at facilities, but because these standards are based solely on estimated potential for HRIs, we refer to this type of source-based option as a HRI standard. The ACE does not include fuel cofiring among its described emission reduction options. States would be required to submit plans to EPA to implement the rule, taking into account criteria such as remaining useful life, and it is possible states would propose to allow cofiring to achieve comparable emissions reductions. The ACE also allows for the possibility that states determine that no emissions reduction options are feasible.

With the issuance of the proposed replacement regulation, the EPA released a Regulatory Impact Analysis (RIA) that models emissions under the ACE compared to a reference scenario with the CPP and a scenario with no power plant carbon standard (US Environmental Protection Agency (EPA) 2018b). The RIA includes projections of national power sector emissions outcomes, but does not examine or quantify the role that a potential emissions rebound effect may play in driving the emissions outcomes. The rebound effect is a phenomenon in which facilities with high baseline emissions rates are made more efficient through investments to reduce their heat rates, and consequently operate more frequently and remain in operation for a longer period. This phenomenon is well documented in the environmental economics literature, though the majority of evidence focuses on energy efficiency (Greening *et al* 2000, Sorrell *et al* 2009). Previous studies have found evidence that an emissions rebound effect can diminish emissions reductions or even lead to emissions increases following HRIs at high-emissions facilities (Linn *et al* 2014, Keyes *et al* 2018), but no other studies have specifically examined the role of an emissions rebound in the ACE.

We analyze the model plant level results published by EPA to better understand the predicted impact of the ACE on CO_2 emissions from coal plants and the potential impact on total CO_2 emissions at national and state levels (US Environmental Protection Agency (EPA) 2018b). We also analyze the changes in emissions of co-pollutants including sulfur dioxide (SO_2) and nitrogen oxides (NO_X), which affect local air quality and human health.

We conduct a formal decomposition analysis of the estimated national changes in generation and CO_2 emissions between the ACE and a no-policy scenario to examine the underlying drivers of the emissions changes and to estimate the contribution of a potential emissions rebound effect. We provide decomposition results for states that are estimated to experience emissions increases under the source-based ACE rule.

Our analysis largely evaluates the impacts of the ACE based on 2030 projections for a central case we selected from EPA's three illustrative ACE modeling scenarios. In addition, we compare these results to emissions results for 2021–2050 and for the EPA's two other illustrative ACE cases.

This analysis builds upon a study by the same authors that independently models potential national and state-level CO_2 emissions impacts in 2030 for a source-based scenario compared to a scenario with no power plant carbon standard and to a flexible systems-based scenario similar to the CPP (Keyes *et al* 2018). Our findings on the emissions rebound effect are compared to the results of Keyes *et al* (2018).

2. Methods

2.1. Data

We conduct our analysis using results from the EPA's policy scenario modeling for the ACE RIA. EPA used the Integrated Planning Model (IPM) to estimate power sector outcomes from 2021 to 2050. IPM is a dynamic linear programming engineering-economic model of the US power sector. It maps almost 13 000 existing and planned EGUs into about 1700 model plants. The model differentiates power sector outcomes into demand and supply regions and accounts for interstate electricity trade. IPM is solved with fixed electricity demand. EPA uses IPM to project emissions of CO_2 and co-pollutants and a number of other outcomes under various policy scenarios.[7]

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[7] See US Environmental Protection Agency (EPA) (2018b) for a detailed description of modeling assumptions and inputs.

Five scenarios were modeled using IPM: a scenario with no power plant carbon standard, an illustrative scenario with the CPP, and three illustrative ACE scenarios that represent potential state determinations of performance standards and compliance with those standards (US Environmental Protection Agency (EPA) 2018b). The CPP scenario assumes a rate-based implementation applied only to existing fossil-fired EGUs, one of multiple options available to states. Each ACE scenario assumes uniform HRI potential at all coal plants and uniform cost per kW of HRI investment. The ACE scenarios differ in their assumptions about the status of the New Source Review (NSR) provision of the US Clean Air Act. NSR currently requires permitting for major generation sources that make major modifications. The ACE introduces a change in NSR to allow major sources to avoid triggering NSR if modifications do not affect their hourly rate of emissions. The first ACE scenario, 2% HRI at \$50 kW^{-1} at coal plants, assumes that the EPA's proposed revisions to the NSR requirements are not implemented and therefore identifies relatively modest opportunities for HRIs; the second scenario, 4.5% HRI at \$50 kW^{-1}, assumes NSR revisions are implemented and identifies greater opportunities for HRIs; and the third scenario, 4.5% HRI at \$100 kW^{-1}, also assumes NSR revisions are implemented but assumes HRIs have a higher cost, which is more appropriate for plants with relatively low capacity or limited remaining useful life.

Our analysis uses the published output from EPA's IPM model runs. We use the IPM State Emissions datasets to examine total emissions of CO_2 and co-pollutants SO_2 and NO_X at the state and national level. Additionally, we use the IPM RPE datasets, which provide projections of fuel generation and emissions (CO_2, SO_2 and NO_X) for each model plant to evaluate outcomes. Our analysis focuses on emissions outcomes in 2030 for the 4.5% HRI at \$50 kW^{-1} scenario compared to the CPP and no-policy scenarios. We choose this scenario as our ACE central case because it incorporates the implementation of EPA's proposed NSR reform and a lower cost of HRI investment. We also compare these results with the other two ACE scenarios and to results for 2021–2050.

2.2. Decomposition analysis

To analyze estimated changes in EGU generation and associated emissions, we use a logarithmic mean decomposition index approach, based on Ang (2015). We implement Model 1 in table 1 of Ang (2015) and substitute CO_2 emissions for energy consumption (E) and electricity generation for industrial output (Q). This method follows from that used in Palmer *et al* (2018) to decompose modeled emissions changes under a carbon tax. We estimate the contribution of three factors to the change in emissions under the ACE compared to the no-policy scenario: activity, structure, and intensity. The activity factor is emissions changes associated with changes in total electricity generation; the structure factor is emissions changes associated with shifts in generation among fuel types; and the intensity factor is emissions changes associated with changes in emission intensity within fuel types.

The emission intensity of fuel types (the intensity factor) is the factor targeted by a HRI standard and it can change when a policy causes various fossil fuel plants to improve their efficiency. Under a HRI standard, the intensity factor contributes to emissions reductions if the standard successfully reduces the emission intensity of coal plants.

The rebound effect is embodied in changes in the generation mix (the structure factor), which changes when a policy affects the relative competitiveness of generation sources. This can occur under a HRI standard if the standard improves the efficiency of coal plants and thus causes substitution towards coal away from other, lower-emitting generation sources. Our estimate of the rebound effect is likely conservative because the EPA's model holds total demand constant. If demand were allowed to change, the rebound effect would include both the structure factor and the activity factor. Change in demand can occur if the increased efficiency of coal lowers the cost of electricity generation and thus increases total electricity demand, as would be expected in organized wholesale power markets. In regulated markets, these investments could increase or decrease total costs, depending on the reason such investments are previously unrealized. Reasons could include inconsistent pass-through clauses, avoidance of triggering NSR, access to capital, and uncertainty about greenhouse gas regulations (Richardson *et al* 2011, Campbell 2013, Linn *et al* 2014). However, under constant demand, at the national level the activity factor in our analysis is not directly associated with the rebound effect. At the state-level, a change in the activity factor can be associated with the rebound effect because changes in trade flows across states can lead to a net change in generation in some states. This effect is absorbed into the structure factor at the national level. Although electricity demand is held constant, total electricity generation (the activity factor) can still differ on the national level across model scenarios for several reasons: policies may cause changes in trade flows between the US and Canada, or changes in state or regional generation within the US. These changes may affect the total amount of electricity transferred between regions, thus affecting total losses and generation.

3. Results

3.1. National and state-level CO_2 emissions changes

National CO_2 emissions are projected to be slightly lower under the ACE compared to no policy, and higher compared to the CPP, in all modeled years but 2050 (table 1). In 2050, two of the three ACE scenarios have higher CO_2 emissions compared to no policy. Cumulative CO_2 emissions from 2021 to 2050 are

Table 1. National power sector CO_2 emissions (million short tons).

	No policy	CPP	4.5% HRI at \$50 kW^{-1} (ACE central case)	2% HRI at \$50 kW^{-1}	4.5% HRI at \$100 kW^{-1}
2021	1710	1701	1709	1709	1707
2023	1801	1754	1814	1801	1802
2025	1829	1780	1812	1816	1799
2030	1811	1737	1797	1798	1785
2035	1794	1728	1787	1783	1772
2040	1849	1782	1841	1840	1829
2045	1843	1782	1832	1833	1821
2050	1804	1753	1815	1801	1808
2021–2050 cumulative (interpolated)	54 469	52 694	54 261	54 195	53 920

slightly lower under all three ACE scenarios compared to no policy and slightly higher compared to the CPP. In 2030, compared to no policy, CO_2 emissions are projected to be 0.8% lower under the 4.5% HRI at \$50 kW^{-1} scenario, 0.7% lower under the 2% at \$50 kW^{-1} scenario, and 1.5% lower under the 4.5% at \$100 kW^{-1} scenario.

There is substantial variation in state-level outcomes under the ACE. For the 4.5% HRI at \$50 kW^{-1} scenario, 18 states plus the District of Columbia are projected to experience at least small increases in CO_2 emissions in 2030 compared to no policy (figure 1). The numbers are similar for the other two ACE scenarios: 16 states plus Washington, DC for the 2% at \$50 kW^{-1} scenario and 14 states plus Washington, DC for the 4.5% at \$100 kW^{-1} scenario. Compared to the CPP, 22 states and Washington, DC are projected to have emissions increases under the 4.5% HRI at \$50 kW^{-1} ACE scenario (figure 2).[8]

3.2. Coal-fired power plant CO_2 emissions changes

We examine the impact of the ACE on model coal-fired power plants to illustrate the main drivers of emissions changes by focusing on 2030 emissions for the 4.5% HRI at \$50 kW^{-1} scenario, which is our ACE central case. IPM's model coal plants are aggregated representations of constituent coal plants within states, 381 of which were operating in the US in 2016 (US Energy Information Administration (EIA) 2017a). Under EPA's projections of ACE, CO_2 emissions from coal plants are projected to be only slightly lower (0.6%) in 2030 compared to no policy (table 2). While the emissions intensity of coal plants declines by 4.5%, the number of coal plants in operation and total coal-powered electricity generation increase. This shift offsets the benefits of emissions intensity improvements and causes the total emissions reduction to be small compared to the emissions intensity improvements.

Under the EPA's interpretation of section 111(d) of the Clean Air Act as constraining regulations to measures that can be taken at a source (power plant), total CO_2 emissions are actually projected to increase at a number of the affected plants. Of the 333 model coal plants that would be in operation in 2030 under no policy, 93 of those (or 28%) are projected to have higher total CO_2 emissions under the ACE. Additionally, under the ACE five additional model coal plants are projected to be operating in 2030 that would have been idled or retired under no policy.

3.3. Decomposition of CO_2 emissions changes

The decomposition shows the extent to which the rebound effect is projected to offset emissions reductions under the ACE. Total national emissions under the ACE are estimated to decrease by 14.3 million short tons (0.8%) compared to the no-policy scenario in 2030. Our decomposition analysis breaks down the three primary factors driving that change in emissions (figure 3(a)). We find that reductions in emissions intensity within fuel types reduce emissions by 47.4 million tons, mainly due to the lower emissions intensity of coal generation. However, the rebound effect associated primarily with greater utilization of coal plants increases emissions by 32.4 million tons, partially offsetting the reductions from improvements in emissions intensity and resulting in smaller estimated total reductions. Note that the rebound effect is greater on a fleet basis, due to substitution to more efficient units, than researchers have estimated for an individual facility (e.g. Linn *et al* 2014). A slight increase in total electricity generation drives emissions up by an additional 0.6 million tons.

For the 18 states plus DC projected to experience higher CO_2 emissions in 2030 under the ACE compared to no policy (figure 1), total CO_2 emissions are expected to increase by 8.5 million tons. Decomposition reveals that emissions intensity improvements drive down emissions by 14.3 million tons, but these reductions are more than offset by generation mix shifts that drive up emissions by 21.4 million tons and

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[8] Conversely, 25 states are projected to have lower emissions under the the 4.5% at \$100 kW^{-1} scenario compared to the CPP. This is because the CPP creates performance standards for fossil generation sources, and emissions at EGUs can increase under the CPP if their level of generation increases. The CPP is a flexible standard aimed at achieving system-wide emissions reductions.

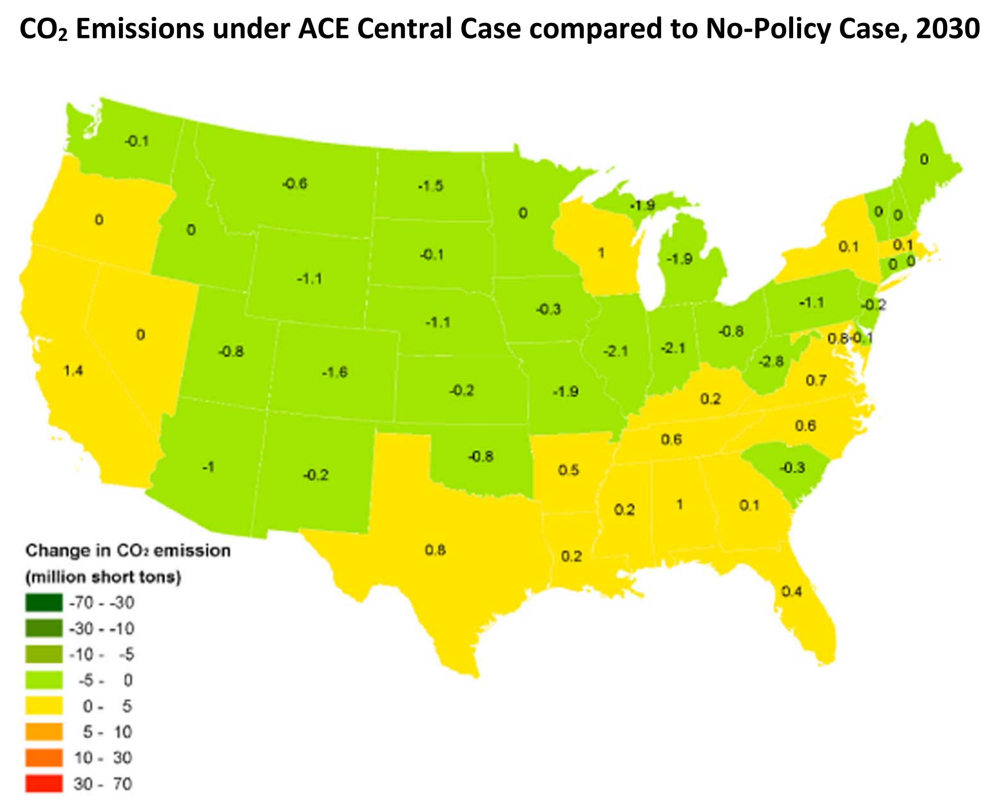

Figure 1. CO_2 emissions under ACE central case compared to No-Policy case, 2030.

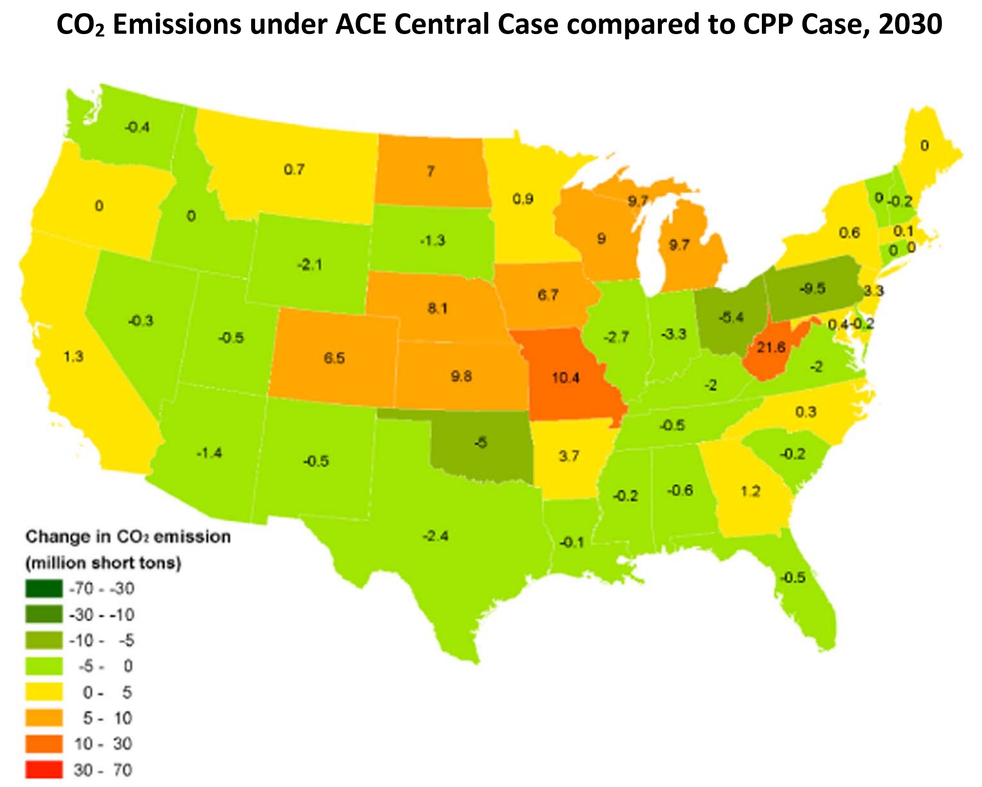

Figure 2. CO_2 emissions under ACE central case compared to CPP case, 2030.

greater total generation that drives up emissions by 1.4 million tons (figure 3(b)). This rebound effect is caused mostly by shifts towards increased coal generation. Of the 18 states that experience total increases in CO_2 emissions, 14 states experience an emissions increase from coal-fired power plants in

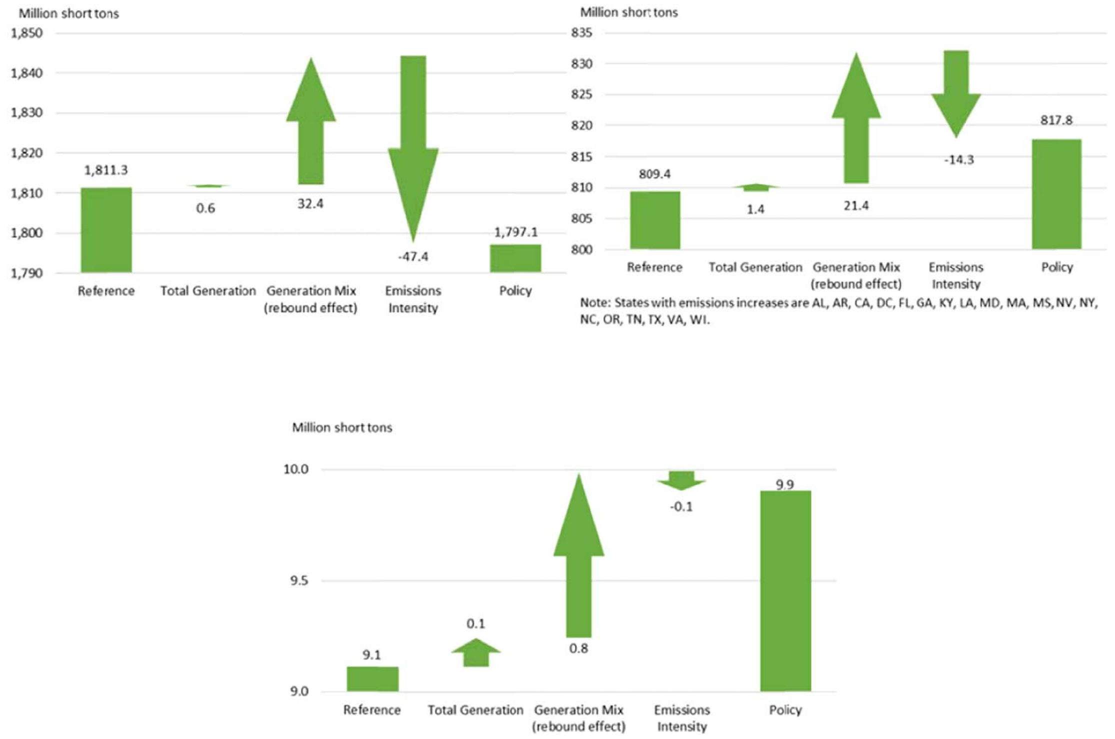

Figure 3. Decomposition of CO_2 emissions change under ACE central case compared to No-Policy case. (a) National, 2030. (b) States with emissions increases, 2030. (c) Maryland, 2030.

Table 2. Comparison of model coal plants between ACE central case and No-Policy case, 2030.

	No policy	ACE central case	Change (level)	Change (%)
Number of model coal plants in operation	333	338	5	1.5%
Total generation (GWh)	937 757	975 633	37 877	4.0%
Total Emissions (Thousand short tons)	1027 456	1020 897	−6559	−0.6%
Emissions intensity (kg kWh^{-1})	0.99	0.95	−0.04	−4.5%
Heat rate (Btu kWh^{-1})	10 395	9930	−465	−4.5%

their state. In the other four states (California, Georgia, Massachusetts, and Oregon) plus DC, the emissions increases are mainly due to increased emissions from natural gas. Increases in state-level natural gas emissions could occur for several reasons that are specific to state and regional electricity markets. This pattern exposes another unintended consequence of the ACE that could diminish emissions reductions in some states.

Maryland has the greatest percent increase in emissions under the ACE compared to no policy in 2030 (8.7%) and provides an informative illustration of the emissions rebound effect. Maryland has two model coal plants in operation under the ACE, neither of which would be in operation with no policy in place. Thus, the shift in the generation mix towards coal drives up emissions by 0.8 million tons and causes an overall increase in emissions in the state (figure 3(c)).

Interstate trade in electricity can exacerbate the emissions rebound in some states, because coal EGUs that become more efficient may compete not only with EGUs in their state but also others in their power market region. For example, the emissions intensity of coal in a net electricity exporting states like Alabama improves in 2030 under the ACE compared to no policy. However, coal generation and total generation increase in the state, suggesting that electricity exports increase. The increase in fossil generation drives up emissions by 2.2 million tons, offsetting the emissions intensity improvements and resulting in a net increase in emissions by 1 million tons.

3.4. Criteria air pollutant emissions changes

National SO_2 emissions in 2030 are projected by EPA to decrease by 0.7% under the ACE compared to no policy, with 19 states showing SO_2 emissions increases (figure 4). National NO_X emissions are projected by EPA to decrease by 1.0%, with 20 states plus DC showing emissions increases (figure 5). Compared to the CPP, national SO_2 emissions are projected by EPA to be 5.9% higher under the ACE and NO_X emissions are projected to be 5.0% higher.

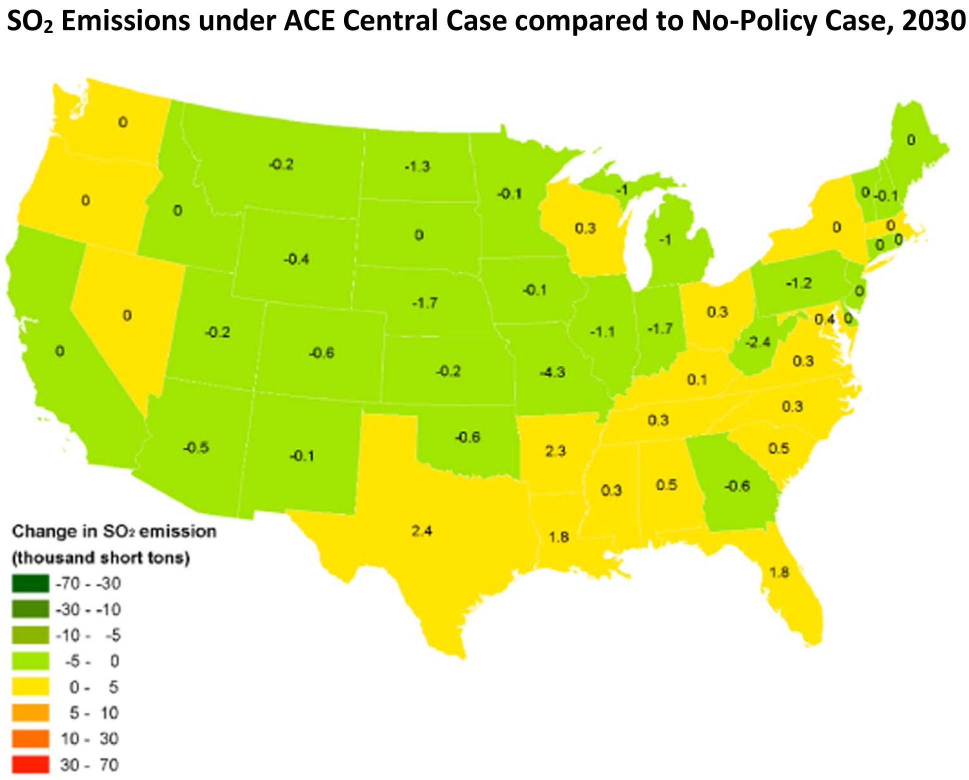

Figure 4. SO_2 emissions under ACE central case compared to No-Policy case, 2030.

4. Discussion

4.1. Comparison of results

Our analysis of ACE impacts using EPA's RIA demonstrates the potential for a rebound effect to occur and limit decrease emissions reductions. Previous studies have found evidence that a rebound effect is associated with HRIs at high-emissions rate facilities, and changes in the operation of these facilities diminishes the reduction in emissions that would otherwise occur (Linn *et al* 2014). Moreover, because these facilities have lower operating costs after the HRIs are made, they are likely to delay their ultimate retirement and may remain in service longer into the future (Burtraw *et al* 2011). Our analysis suggests this is the case, because by 2050 CO_2 emissions under the ACE exceed emissions under no policy. This consideration is important since CO_2 is a stock pollutant that accumulates in the atmosphere each year.

We compare the results of this analysis to another study by the same authors (Keyes *et al* 2018), in which the spatially explicit effects of scenarios constructed independently but similar to the ACE are modeled, including a source-based HRI standard. Keyes *et al* (2018) uses results from IPM to compare their source-based scenario to a no-policy scenario and a systems-based scenario similar to the CPP. Because the modeling conducted for Keyes *et al* (2018) is independent from that used by EPA in its ACE RIA, it provides an alternative estimate of emissions outcomes. Importantly, the results based on EPA's modeling can be compared only qualitatively to the Keyes *et al* modeling results because baseline economic conditions differ between the two sets of model runs. Keyes *et al* (2018) uses power sector modeling based on the electricity industry as it was configured in 2014, and the industry has since undergone substantial changes including retirement of many fossil units. Coal generation declined from 40% of total power generation in 2013 to 31% of total generation in 2017, and overall fossil fuels supplied 62% of total generation in 2017 compared to 67% in 2013 (US Energy Information Administration (EIA) 2018). The analyses also employ different assumptions about policy design and implementation. For example, the source-based standard used in Keyes *et al* (2018) includes cofiring up to 15% with natural gas or biomass as a compliance option, while the ACE does not consider cofiring as a candidate technology for BSER. Therefore, emissions projections in the EPA modeling results are lower for the No-Policy case and the estimated emissions impacts of the source-based policy are smaller compared to Keyes *et al* (2018) (table 3). However, Keyes *et al* (2018) affirm the finding that a rebound effect could lead to emissions increases at individual plants and in some states based on the EPA's modeling.

A notable result from EPA's RIA modeling is that the impact of the CPP on CO_2 emissions compared to no policy is small (4% reduction in 2030) compared to Keyes *et al* (2018), EPA's 2015 RIA for the CPP final rule and the Energy Information Administration's

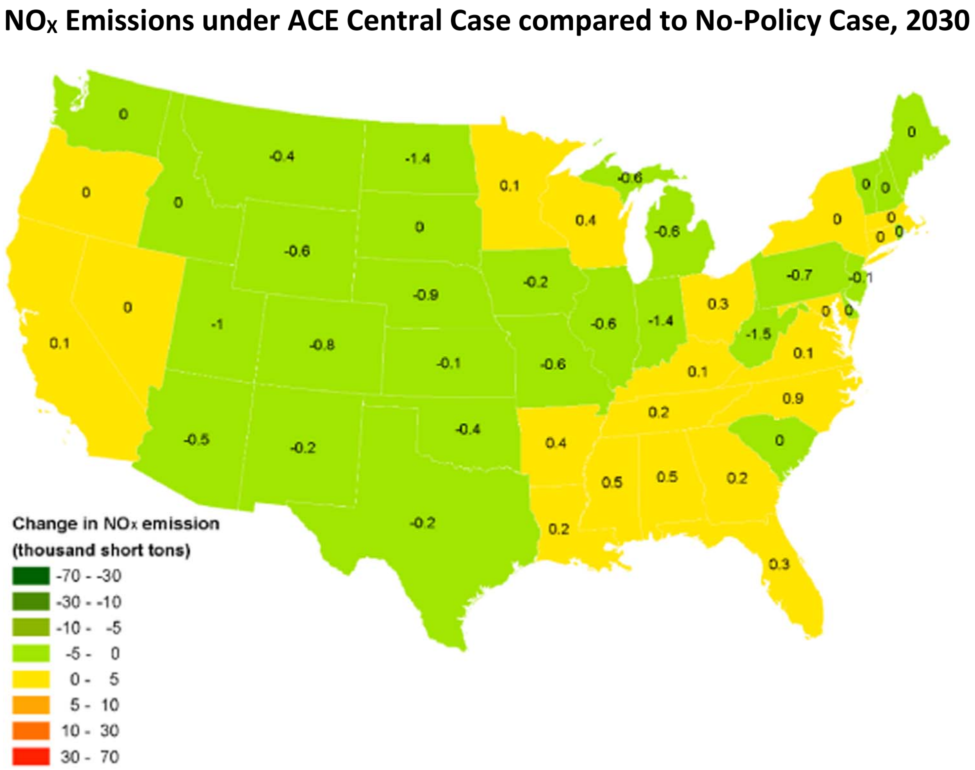

Figure 5. NO$_X$ emissions under ACE central case compared to No-Policy case, 2030.

Table 3. Comparison of source-based scenario modeling results for 2030.

	Current analysis based on EPA's ACE RIA	Keyes *et al* (2018)
CO$_2$ emissions under source-based scenario, million short tons	1797	2386
CO$_2$ emissions under no policy scenario, million short tons	1811	2451
Difference	−0.8%	−2.6%
CO$_2$ emissions under systems-based scenario, million short tons	1737	1466
Difference	3.5%	63%
Number of states with emissions increase compared to no policy scenario	18 states plus DC	8 states
Number of states with emissions increase compared to systems-based scenario	22 states plus DC	46 states

2017 Annual Energy Outlook (US Environmental Protection Agency (EPA) 2015b, US Energy Information Administration (EIA) 2017b). One reason for the relatively small impact of CPP in the ACE re-analysis is that EPA's ACE No-Policy case includes less fossil fuel generation than previous RIAs. Another reason is the set of assumptions that EPA uses for CPP implementation in the ACE RIA, which assumes coverage only for existing generation sources rather than existing and new sources and no incremental energy efficiency investments. These assumptions reduce the projected emissions benefits under the CPP.

The proposed ACE rule, in addition to suggesting changes to power plant carbon standards, also would reform the NSR program for new and significantly modified facilities. As discussed above, the reform to NSR would allow power plants to avoid NSR review as long as their hourly rate of emissions do not increase. This reform may create a loophole for some plants to adopt HRI measures and potentially increase emissions. EPA's projections for the scenario incorporating NSR reform (4.5% HRI at \$50 kW^{-1}) and a scenario without NSR reform (2% HRI at \$50 kW^{-1}) shows minor impacts of NSR reform on CO$_2$ emissions.

4.2. Policy Implications

The CO$_2$ emissions impacts of the ACE have implications for the 20 states that have adopted greenhouse gas emissions targets (Center for Climate and Energy Solutions (C2ES) 2018). Twenty-two states plus DC are projected to have higher emissions under the ACE compared to the CPP, and 11 of these states plus DC currently have greenhouse gas emissions targets in place. These states can be expected to face more difficulty achieving their targets due to the replacement of the CPP. Further, of the 18 states and DC projected to experience higher CO_2 emissions compared to no policy, seven—California, DC, Florida, Maryland, Massachusetts, New York and Oregon—have greenhouse gas emissions targets. For these states, achieving their emissions targets may be more difficult under the ACE compared to having no federal power plant carbon standard in place.

The possibility for the rebound effect to lead to emissions increases at individual plants and for entire states raises the question whether the HRI standard proposed under the ACE qualifies as the 'BSER' that EPA is charged with identifying in its development of a power plant carbon standard under section 111(d) of the Clean Air Act. The projected impact of the rebound effect on CO_2 emissions under the ACE should be taken into consideration in determining whether the BSER requirement has been satisfied.

The change in emissions of co-pollutants under the ACE also has implications for regional air quality and public health. SO_2 and NO_X are precursors to ambient $PM_{2.5}$ and NO_x emissions contribute to ambient ozone, both of which have effects on premature mortality and morbidity. States with increased emissions may experience greater difficulty achieving or maintaining the US National Ambient Air Quality Standards established under the Clean Air Act. EPA estimates that, nationally, the ACE will lead to a slightly lower number of $PM_{2.5}$- and ozone-related premature deaths compared to no policy in 2030, but it estimates that the ACE will substantially increase premature deaths compared to the CPP.

5. Conclusions

Our analysis finds that the projected emissions rebound effect in EPA's ACE RIA undermines emissions reductions from the ACE rule compared to both the CPP and to no power plant carbon standard. Although the emissions intensity of modeled coal plants decreases, the number of operating coal plants and the amount of coal-powered electricity generation increases. Under the ACE central case, the rebound effect causes emissions to increase at 28% of coal plants in 2030. As a result, total CO_2 emissions increase in 18 states plus DC and national CO_2 emissions decrease by only 0.8% in 2030. Further, emissions of SO_2 decline by only 0.7% with increases in 19 states, and emissions of NO_X decline by 1.0% with increases in 20 states plus DC. The other ACE scenarios evaluated show similar outcomes driven by a rebound effect.

Our finding that under a source-based power plant standard the rebound effect can undermine pollutant emissions decreases at the national level and lead to increased emissions at individual coal plants and in a number of states is substantiated by similar findings based on independent power sector modeling (Keyes *et al* 2018). This result, which was not examined in the RIA for the ACE proposed rule, has implications for the defensibility of the ACE as the BSER, for the ability of some states to achieve their greenhouse gas emissions reduction targets, and for jurisdictions that experience poor air quality to protect public health.

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Hellbender

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Sometimes known as the “devil dog” or “snot otter,” the eastern hellbender is North America’s largest salamander.

A fully aquatic species, hellbenders have inhabited the rivers of the eastern United States for the last 65 million years and are indicators of healthy streams. But without direct intervention, they could be headed for extinction.

Hellbenders require very specific habitat conditions to survive. In addition to clean, oxygen-rich water, they need access to prey, the ability to find mates, and large slab rocks with accessible crevices underneath to make homes and protect maturing eggs. Humans have dramatically changed and polluted the landscape, leading to declines in habitat and water quality that now threaten the hellbender’s survival.

Defenders' Impact

Restoring habitat and recovering hellbender populations is a complex conservation challenge with no one simple solution, so Defenders of Wildlife launched the Southeastern Hellbender Conservation Initiative (SEHCI) in 2017.

SEHCI’s mission is to bring together science, education, community outreach and on-the-ground habitat restoration to advance hellbender population recovery on private lands in the Southeast. Through SEHCI, we focus our efforts on identifying priority

Threats & Status → What You Can Do Facts Range & Population Behavior & Reproduction

Threats

Unlike most species, hellbenders breathe through their skin. This makes them particularly vulnerable to sediment, pollutants and reduced oxygen levels in their environment. Nutrients and pesticides, runoff and erosion from agriculture and urban development degrade water quality; mining activities deposit heavy metals and toxins into nearby waterways; and dams and road crossings isolate populations.

Protection Status

Endangered Species Act	IUCN Red List	CITES
! **Endangered**	! **Near Threatened**	! **Appendix III**
The Missouri Distinct Population Segment of the eastern hellbender is listed as endangered; all other populations not listed.		

News

CHARLESTON, S.C. — APRIL 29, 2020

Conservation Groups Challenge EPA's Gutting of Clean Water Protections in Federal Court EPA Rule Paves Way for Wetlands and Small Streams Destruction ›

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THE ROLE OF HEADWATER STREAMS IN DOWNSTREAM WATER QUALITY[1]

Richard B. Alexander, Elizabeth W. Boyer, Richard A. Smith, Gregory E. Schwarz, and Richard B. Moore[2]

ABSTRACT: Knowledge of headwater influences on the water-quality and flow conditions of downstream waters is essential to water-resource management at all governmental levels; this includes recent court decisions on the jurisdiction of the Federal Clean Water Act (CWA) over upland areas that contribute to larger downstream water bodies. We review current watershed research and use a water-quality model to investigate headwater influences on downstream receiving waters. Our evaluations demonstrate the intrinsic connections of headwaters to landscape processes and downstream waters through their influence on the supply, transport, and fate of water and solutes in watersheds. Hydrological processes in headwater catchments control the recharge of subsurface water stores, flow paths, and residence times of water throughout landscapes. The dynamic coupling of hydrological and biogeochemical processes in upland streams further controls the chemical form, timing, and longitudinal distances of solute transport to downstream waters. We apply the spatially explicit, mass-balance watershed model SPARROW to consider transport and transformations of water and nutrients throughout stream networks in the northeastern United States. We simulate fluxes of nitrogen, a primary nutrient that is a water-quality concern for acidification of streams and lakes and eutrophication of coastal waters, and refine the model structure to include literature observations of nitrogen removal in streams and lakes. We quantify nitrogen transport from headwaters to downstream navigable waters, where headwaters are defined within the model as first-order, perennial streams that include flow and nitrogen contributions from smaller, intermittent and ephemeral streams. We find that first-order headwaters contribute approximately 70% of the mean-annual water volume and 65% of the nitrogen flux in second-order streams. Their contributions to mean water volume and nitrogen flux decline only marginally to about 55% and 40% in fourth- and higher-order rivers that include navigable waters and their tributaries. These results underscore the profound influence that headwater areas have on shaping downstream water quantity and water quality. The results have relevance to water-resource management and regulatory decisions and potentially broaden understanding of the spatial extent of Federal CWA jurisdiction in U.S. waters.

(KEY TERMS: rivers/streams; nitrogen; transport and fate; streamflow; headwaters; *SWANCC*; Rapanos.)

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INTRODUCTION

Recent U.S. Supreme Court rulings, related to Clean Water Act (CWA) decisions by federal regulatory agencies (U.S. Army Corps of Engineers and U.S. Environmental Protection Agency), underscore the need for an improved scientific understanding of the influence of headwater areas and upland (low-order) streams on the physical, chemical, and biological integrity of downstream waters, especially those legally classified as "navigable." An important 2001 U.S. Supreme Court ruling (*Solid Waste Agency of Northern Cook County v. U.S. Army Corps of Engineers*; *SWANCC*) and subsequent court decisions interpreting the meaning of *SWANCC* focused on the scope of the CWA permit program as it applies to land development, and have raised questions about the jurisdiction of federal regulatory agencies over various U.S. waterways. The *SWANCC* case narrowed federal authority to protect many upstream and wetland areas, stated as isolated, non-navigable, intrastate waters that are not tributary or adjacent to navigable waters or their tributaries. In subsequent appellate circuit decisions, many questions have been raised about how to interpret the *SWANCC* decision (e.g., the definition of "adjacent") and about what parts of the tributary system are considered jurisdictional under the CWA. These decisions include several recent cases (2006: *Rapanos v. United States*, 04-1034, *Carabell v. Army Corps of Engineers*, 04-1384, and S.D. Warren Co. *v.* ME Board of Environmental Protection, 04-1527) that have not resolved questions about which wetland areas are protected by the CWA.

An improved scientific understanding of the influence of headwater streams on the integrity of downstream navigable waters (especially those that may have less obvious relationships to navigable-in-fact waters; see Federal Register, 2003) is viewed as a central need to assist policy makers, regulatory authorities, and the courts. Of particular interest in determining CWA jurisdiction is whether a "significant nexus" exists between upstream waters and navigable-in-fact waters. Such a connection could be based on evidence that the use, degradation, or destruction of non-navigable headwaters demonstrably affects downstream navigable waters and their tributaries. However, legal ambiguities currently exist as to what constitutes "navigable streams and their tributaries" – i.e., how far upstream does CWA jurisdiction actually extend into tributary reaches. A recent 2006 U.S. Supreme Court decision on the consolidated cases of *Rapanos v. United States & Carabell v. Army Corps of Engineers* failed to explicitly resolve these questions. The ruling specified that Federal CWA jurisdiction requires evidence of a "significant nexus" between upstream waters and navigable waters, based on a technical and scientific judgment by Federal regulators. The cases were remanded to the lower courts for re-evaluation under these guidelines.

Our study provides scientific insight into the coupled hydrological, chemical, and biological influences of headwater systems on downstream navigable waters and their tributaries. An earlier synthesis effort (Nadeau and Leibowitz, 2003) summarized current scientific knowledge of the hydrological and biologic connections between "isolated" wetlands and downgradient surface-water systems. Although a broad range of types of material fluxes and concentrations in headwater and larger streams is ultimately of interest in discussions of headwater connectivity, we focus in this study exclusively on a discussion of nitrogen fluxes in surface waters.

Nitrogen is an essential nutrient that regulates primary production in terrestrial and aquatic ecosystems. Nitrogen inputs to landscapes have increased markedly over the past 50 years across the globe in response to increased food and energy production, which has created an abundant supply of highly reactive forms of nitrogen in air, land, and water (Galloway *et al.*, 2004). Excess nitrogen has been linked to many environmental concerns, including the disruption of forest ecosystem processes (Aber *et al.*, 2003), acidification of lakes and streams (Driscoll *et al.*, 2001), and degradation of coastal waters including high profile water quality issues such as eutrophication, hypoxia, and harmful algal blooms (NRC, 2000). Nitrogen is also the focus of recent USEPA efforts to establish nutrient criteria in U.S. streams, lakes, and estuaries (USEPA, 2000). Moreover, because nitrogen is highly reactive and mobile in terrestrial and aquatic ecosystems, it also serves as a relatively suitable surrogate for many contaminants and potentially toxic substances in water where understanding of the linkages between headwaters and downstream receiving waters is important. Although the complexities of nitrogen cycling in terrestrial and aquatic ecosystems are notable, a considerable body of experimental research and large-scale budgeting and modeling analyses has emerged to support reliable descriptions of the sources and transport of nitrogen over broad spatial scales within streams and rivers.

Our study is organized in two major sections. The first section provides an overview of the principal conceptual frameworks and current watershed research relevant to evaluating the role of headwater streams in controlling nitrogen conditions in downstream waters. This synthesis illustrates current understanding of the coupling of land use, pollutant sources, and hydrological and biogeochemical processes on the landscape and how these activities and processes control the supply and delivery of water and nitrogen flux to headwater streams. We further examine the function that stream channels play in controlling water routing and instream processing and their effects on nitrogen transport from headwaters to downstream waters.

In the second section of the article, we use the water-quality model SPARROW (SPAtially Referenced Regression On Watershed attributes; Smith *et al.*, 1997) to investigate and quantify headwater influences in streams of the northeastern United States. SPARROW is a hybrid statistical/mechanistic watershed model with mass-balance constraints. The model descriptions of landscape and aquatic processes are sufficiently detailed to support an assessment of the effects of headwater processes and pollutant sources on water-quality conditions throughout large river networks. Although progress has been made in empirically modeling the transport of nitrogen in streams (e.g., Seitzinger *et al.*, 2002), most empirical watershed models lack mass-balance constraints and do not separate land and water processes. These features are necessary to accurately quantify nutrient transport in streams of varying sizes in river networks (e.g., Smith *et al.*, 1997; Alexander *et al.*, 2002a,b). Moreover, dynamic mechanistic watershed models (e.g., HSPF; Bicknell *et al.*, 2001), although providing detailed predictions of nitrogen flux over time in response to short-term changes in climate, hydrology, and nutrient cycling dynamics, are frequently applied only in small catchments and lack the spatial detail and observational data needed to quantify the fate of headwater nitrogen sources and cycled nitrogen in large river networks. To enhance our model-based descriptions of nitrogen transport from headwaters to downstream navigable waters and their tributaries, we modify the structure of a previous SPARROW model (Moore *et al.*, 2004) to incorporate observations of nitrogen removal in streams and lakes from the primary literature. We use the refined model to assess the effects of streamflow and nitrogen supply and removal processes in headwaters on the flow and nitrogen conditions in downstream waters.

THE COMPLEX INTERACTIONS OF NITROGEN IN WATERSHEDS

Landscape and Water Interactions

Although nutrients are associated with healthy watersheds and the provision of ecosystem services, they also can act as pollutants. Commonly described as "too much of a good thing," it is the overabundance of nitrogen loadings that leads to negative environmental effects. Nitrogen in the environment has vastly increased in recent decades, largely associated with growing populations and associated land use, from: (1) creation of reactive nitrogen, via the Haber-Bosch process, for fertilizers and other industrial applications; (2) cultivation of vast land areas of crops that host nitrogen-fixing bacteria; and (3) fossil fuel burning and the associated emissions and nitrogen deposition (Smil, 2001). Worldwide, human activities have more than doubled the amount of reactive N entering the environment (Vitousek *et al.*, 1997; Galloway *et al.*, 2004). In an individual watershed, the distribution of human and animal populations, land use, and characteristics of the vegetation and soils set the stage for the types, magnitudes, and geography of nitrogen inputs (Boyer *et al.*, 2002).

Stemming from nitrogen inputs to landscapes, nitrogen fluxes in many surface waters have increased in recent decades, and two-thirds of the nation's estuaries are degraded from nitrogen pollution (Bricker *et al.*, 1999). Nitrogen flux in streams and rivers of any size is the cumulative result of processes that control the supply and transport of nitrogen in terrestrial and aquatic ecosystems. These occur throughout the watershed system from the headwater source areas to the downstream receiving waters (Howarth *et al.*, 1996; Seitzinger *et al.*, 2002; Van Breemen *et al.*, 2002; McClain *et al.*, 2003). As a result, nitrogen pollution and other nutrient problems are increasingly being addressed by researchers and management agencies by considering the intrinsic linkages between terrestrial upland landscapes and the aquatic systems to which they drain (Driscoll *et al.*, 2003; Grimm *et al.*, 2003).

Nitrogen fluxes in surface waters are controlled to a large degree by heterogeneous distributions of nitrogen inputs (Howarth *et al.*, 1996; Boyer *et al.*, 2002). The environmental setting – e.g., climate, topography, vegetation, and soil properties – also shapes both land use (and the types of nitrogen sources) and how nitrogen inputs are mediated. Nitrogen is highly reactive, ensuring biogeochemical processing and transformations in landscapes, including nutrient production mechanisms, assimilation and uptake in plant material, and permanent removal via denitrification (Davidson and Schimel, 1995; Van Breemen *et al.*, 2002; Boyer *et al.*, 2006b). Denitrification is a process whereby the reactive forms of nitrogen are transformed into dinitrogen (N_2) gas, which is highly inert and does not have any adverse environmental consequences (and, in fact, is the dominant component of the earth's atmosphere). Further, nitrogen is highly soluble and is transported easily in water, influenced by hydrological processes including flow paths and residence times of water throughout the watershed (Cirmo and McDonnell, 1997; Band *et al.*, 2001). Collectively, nitrogen sources to landscapes along with coupled hydrological and biogeochemical processes occurring throughout the watershed strongly affect the timing and form of nitrogen delivery to surface waters and the areas of the landscape that contribute nitrogen to streams. In temperate regions, the hydrologically connected soils and land areas that drain to streams expand and contract both laterally and vertically during periods of wetting and drying. During wet periods, this causes saturated areas of the landscape to expand, especially riparian areas, which facilitates both the delivery of nitrogen to streams and its loss via denitrification. Considering such factors, environmental scientists have been successful in simulating nitrogen delivery to surface waters at many spatial and temporal scales (Creed and Band, 1998; Alexander *et al.*, 2000, 2002a; Band *et al.*, 2001; McIsaac *et al.*, 2001; Howarth *et al.*, 2002; Boyer *et al.*, 2006a).

Once nitrogen is delivered to streams or rivers, the aquatic ecosystem itself plays a critical role in modifying the nitrogen (and other material) fluxes, via channel routing and instream processing. Stream channels have a natural dendritic design that plays an intrinsic role in transporting nitrogen and other pollutants from widely dispersed upstream sources and concentrating these materials in downstream waters. Hyporheic zones of streams also play a key role in nitrogen transformations (uptake and cycling) and permanent removal (i.e., denitrification) as nitrogen is exposed to reactive benthic surfaces during transport. The hyporheic zone, literally meaning under the flow, is the zone of sediments beneath and beside the stream where surface water (from the stream) and subsurface water are exchanged, hydrologically linking this zone of sediments to the stream channel. Strong gradients in the oxygen status and nutrient content of streambed sediments occur due to hyporheic exchange, that is, the mixing of the aerated and thus well-oxygenated streamwater with deeper and anoxic subsurface flows (Bencala, 1993). Such redox gradients found in hyporheic regions create metabolically active zones that facilitate transformations of many elements of water quality. Exchange of surface water with the streambed sediments provides opportunities for denitrification to occur (Duff and Triska, 2000). Large fractions of nitrogen inputs to streams are lost via denitrification in hyporheic sediments at all scales from headwater streams to large rivers (Peterson *et al.*, 2001; Thomas *et al.*, 2001; Seitzinger *et al.*, 2002; Böhlke *et al.*, 2004; Mulholland *et al.*, 2004; Boyer *et al.*, 2006b; Triska *et al.*, this issue).

Detailed studies of individual watersheds, where hydrological and biogeochemical processes are measured and observed over space and time, provide a scientific basis to understand the dominant factors controlling water quality and nitrogen and provide insight into how to quantify such responses at watershed and regional scales with modeling approaches. For example, the U.S. Geological Survey's Water, Energy, and Biogeochemical Budgets (WEBB) program was designed to understand processes occurring in small watersheds located in geographically diverse environments that represent a range of hydrological, ecological, and climatic conditions. Controls on nitrogen transport and transformation over a variety of scales are being examined in nested catchments from 3 ha to 110 km^2 (J. Shanley and S. Sebestyen, 2005, personal communication) at the Sleeper's River WEBB site, located in the Green Mountains of northeastern Vermont. Results from this site provide a window into the importance of coupled hydrological and biogeochemical processes that affect water quality. The supply of nitrogen from this forested, headwater catchment to its receiving waters is controlled to a large degree by soil biogeochemical processes that provide sources of nitrogen from organic matter, and hydrological processes that connect the landscape to streamflow. Flow paths and residence times of water in the landscape strongly influence streamwater nitrogen concentrations. The temporal variation of nitrogen in the stream (Figure 1) is tightly linked to cycles of water (e.g., influence of spring snowmelt and associated runoff) and carbon (e.g., in dissolved organic forms, DOC), and reflects contributions of flow and solutes from both upland hillslopes and near stream riparian zones of the landscape (McGlynn *et al.*, 1999; Shanley, 2000).

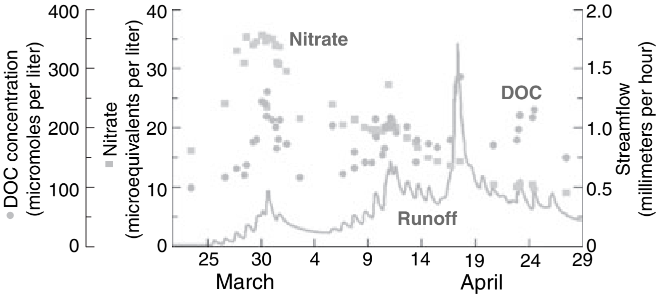

FIGURE 1. Flow Paths and Residence Times of Water in the Landscape Strongly Influence the Magnitude and Variation of Nitrate Concentrations in Headwater Streams. Reprinted from Shanley (2000).

Such results are not limited only to small catchments, but are observed at all watershed scales. For example, nitrogen sources and fate have been studied for over 30 years in the large Fall Creek watershed

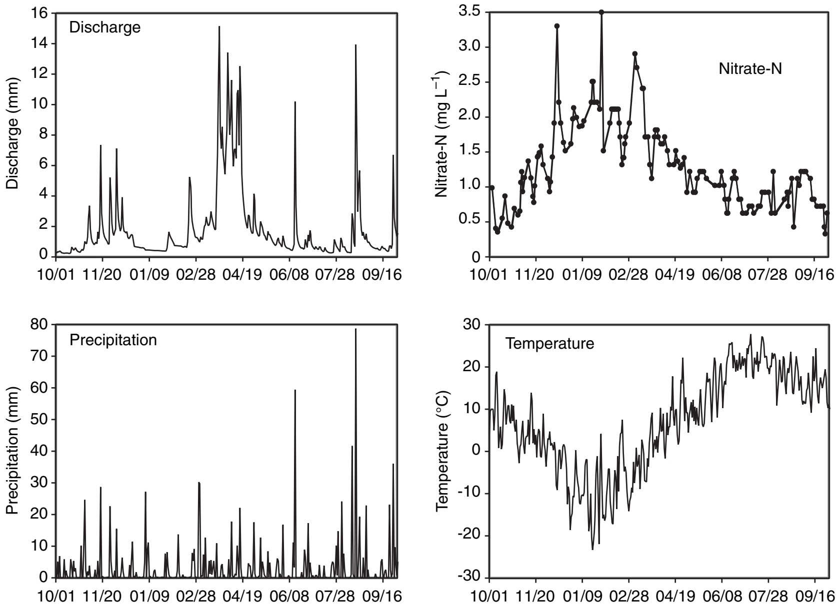

FIGURE 2. Records of Discharge, Precipitation, Nitrate-N, and Temperature at Fall Creek, NY, During 2003-04.

in central New York, a mixed-land-use basin containing large amounts of forest (53%) and agricultural (42%) land that drain an area of 327 km^2. Nitrogen primarily from atmospheric deposition, fertilizers, and manure, is delivered to the stream during rain and snowmelt events, with a large degree of direct connectivity of the upland landscape to the stream. Precipitation and streamflow are well distributed throughout the year (Figure 2). Despite this, instream nitrogen concentrations are notably influenced by seasonal variability, as indicated by air temperature (Figure 2). During the growing season (high temperatures), plants are able to utilize much of the nitrogen inputs to support their growth and productivity. Denitrification, a temperature-dependent process, is also important in consuming nitrogen during these periods. These results are consistent throughout the entire 30-year period of record at the site, and further illustrate the importance of coupled hydrological and biogeochemical controls affecting water quality.

Nitrogen Transport From Headwaters to Higher-Order Streams

Mathematical models of the instream routing and biogeochemical processes that control the transport of nutrients and other solutes provide insight into the influence of headwater catchments and streams on the quality of downstream waters. The dynamics of solute transport in streams can be modeled (e.g., Stream Solute Workshop, 1990; Runkel, 1998) according to the processes of advection, dispersion, ground-water inputs, transient storage (e.g., in hyporheic zones), and nonconservative transport (e.g., uptake, denitrification). One-dimensional, steady state forms of these models provide a simplified description of nutrient transport according to a first-order exponential-decay process (e.g., Newbold *et al.*, 1981; Stream Solute Workshop, 1990; Chapra, 1997; Donner *et al.*, 2004). Nutrient transport is mediated in these models by a reaction-rate coefficient (in units of reciprocal time) and the water time of travel over a given length of stream channel (determined as the product of channel length and the reciprocal of water velocity). The steady-state reaction-rate expression reflects the aggregate, net effects of the physical, hydrological, and biochemical properties of the channel and hyporheic zone on nutrient removal. These model expressions have been advanced as part of *nutrient spiraling* concepts (Newbold *et al.*, 1981); these concepts describe the downstream transport of nutrients as a series of repeated cyclical transformations that entail nitrogen migration to the benthos via biological uptake and organic nitrogen storage and a return to the water column via mineralization and nitrification. Nutrient decay processes in these models may also include the permanent removal of nitrogen from streams via denitrification.

First-order exponential decay functions have been developed to predict nitrogen transport and losses in streams of widely varying sizes, based on empirical observations from the literature of the effects on nitrogen transport of various hydrologic and geometric properties, such as water depth, flow, velocity, and channel slope (Kelly *et al.*, 1987; Molot and Dillon, 1993; Howarth *et al.*, 1996; Alexander *et al.*, 2000, 2002a, 2004; Seitzinger *et al.*, 2002). Studies (Howarth *et al.*, 1996; Peterson *et al.*, 2001; Seitzinger *et al.*, 2002; Boyer *et al.*, 2006b) also indicate that the rates of nitrogen uptake and permanent loss via denitrification in streams generally decline in a downstream direction with increases in stream size (i.e., with increases in mean water velocity, streamflow, and depth). Headwaters and other low-order streams are important locations for nitrogen loss in river networks given that their large benthic surface area relative to the overlying water volume generally leads to greater contact and exchange of water and nitrogen with the hyporheic zone (Alexander *et al.*, 2000; Peterson *et al.*, 2001). Small streams also generally have greater benthic frictional resistance and hyporheic storage (relative to the channel water volume) than large streams and rivers (Harvey and Wagner, 2000; Harvey *et al.*, 2003), which may contribute to their higher observed rates of nitrogen loss.

Based on current understanding of these processes, land-use changes or modifications to stream channels that increase the rates of flow in headwater streams may heighten their influence on the chemical quality of downstream receiving waters. For example, increases in the peak discharge and flashiness of flows that are often associated with urbanization would be likely to reduce the natural processing of nitrogen in low-order streams, increasing the distance over which nitrogen is transported downstream. In addition, stream channelization projects that straighten channels and remove natural pools and riffles are likely to shorten the water travel time in stream reaches; this would also be likely to reduce nitrogen losses and increase downstream transport.

Some exceptions to these general patterns in nutrient transport are of note. One is the importance of floodplains and the riparian areas of large rivers, including, for example, the Mississippi and southeastern U.S. rivers, as sites for nitrogen loss via denitrification during floods. The increase in water depth during floods on these rivers actually increases the contact of nitrogen with microbially reactive floodplain sediments and promotes denitrification (NRC, 2002; Richardson *et al.*, 2004; Scott *et al.*, 2004). Another is the potential for the first-order properties of nitrogen reaction rates to break down in nutrient-enriched waters where denitrification (Garcia-Ruiz *et al.*, 1998) or uptake processes (Dodds *et al.*, 2002) become concentration saturated. Under these conditions, a lower reaction rate would be expected and nitrogen could be transported for longer distances in streams than would occur under nonsaturated conditions. Therefore, headwater catchments with high stream nitrogen concentrations, such as those found in highly urbanized or cultivated catchments, could have an even more far-reaching downstream influence than headwater streams draining relatively undeveloped catchments with low nitrogen concentrations.

Despite the extensive cycling of nitrogen and generally high rates of nitrogen loss in small streams and the terrestrial ecosystems of watersheds (e.g., Howarth *et al.*, 1996; Boyer *et al.*, 2002), there is mounting evidence that the nitrogen in downstream receiving waters is strongly connected to distant landscape sources and responds relatively rapidly to changes in these sources. These connections are observed in watershed studies at small spatial scales, such as those cited earlier, as well as in large-scale studies. One example of the latter is the Mississippi River Basin, where most of the nitrogen loadings at the Mississippi outlet to the northern Gulf of Mexico are transported from distant, inland agricultural watersheds (Alexander *et al.*, 2000). Annual changes in nitrogen load at the outlet correspond closely to contemporaneous annual changes in runoff and nitrogen inputs from agricultural fertilizers and other sources in the basin as well as changes in nitrogen inputs during the preceding 5 years (Goolsby *et al.*, 1999; McIsaac *et al.*, 2001). European studies (e.g., Stalnacke *et al.*, 2003) suggest that improvements in oxygen conditions on the northwestern shelf of the Black Sea in the early and mid-1990s near the outlet of the 800,000 km^2 Danube River Basin occurred in response to upstream reductions in farm subsidies and the use of fertilizers in several eastern European countries following the dissolution of the former Soviet Union in 1991. The nitrogen response to fertilizer reductions has been less rapid (>10 years) in streams draining certain other eastern European watersheds (Stalnacke *et al.*, 2003).

These regional-scale studies suggest that headwater and other low-order streams may play an important role in the observed linkages between landscape pollutant sources, such as agricultural fertilizers and livestock wastes, and the long-distance transport and delivery of nitrogen to higher-order streams and coastal receiving waters. The downstream influences of landscape sources are likely facilitated by the high density of first-order (headwater) streams and their high frequency of tributary connections with all higher-order streams – properties that are intrinsic to dendritic river networks (e.g., see discussion of Tokunaga's Law in Dodds and Rothman, 2000). These characteristics suggest that changes in the physical or chemical condition of headwaters or their catchments could potentially influence both nitrogen and flow conditions in downstream waters. In the following section, we investigate the nature of headwater connections to pollutant sources and higher-order streams and their influence on flow and nitrogen conditions in downstream waters by applying the SPARROW model to a spatially detailed network of streams and rivers.

ASSESSING THE DOWNSTREAM EFFECTS OF HEADWATERS

Model Specification

The steady-state SPARROW model describes nutrient source inputs and one-dimensional transport in terrestrial and aquatic ecosystems, including first-order decay in streams and reservoirs. Model parameters are statistically estimated from a calibration to mean-annual nitrogen loads (mass per unit time) that are computed from periodically measured nutrient concentrations and daily flow measurements at multiple stream monitoring stations. The use of mean-annual loads in the model adjusts for temporal variability related to long-term trends and short-term changes in flow and instream nitrogen cycling and transformation processes. As a consequence, the model estimates the hydrological and biogeochemical processes that affect the long-term supply, loss, and transport of nitrogen in watersheds (Alexander *et al.*, 2000; Schwarz *et al.*, 2006). This mass-balance specification of the model is well suited for assessing the natural and human-related properties of headwaters that govern the long-term generation and transport of nitrogen and its fate in higher-order streams and downstream receiving waters. Notably, mass-balance approaches have generated considerable interest in recent years to further understanding of the long-term effects of nitrogen supply and transport on inland and coastal eutrophication (e.g., Howarth *et al.*, 1996; Vitousek *et al.*, 1997; Carpenter *et al.*, 1998; NRC, 2000; Boyer *et al.*, 2002).

The model structure, supporting equations, and details of the model estimation are given in Schwarz *et al.* (2006). Conceptually, the model is applied to individual stream reaches through a mathematical equation in which F_j' is the model-estimated mean-annual total nitrogen flux leaving reach i. This flux is related to the flux leaving adjacent reaches upstream of reach i, denoted by N_j' where j indexes the set $J(i)$ of adjacent reaches upstream of reach i, plus additional flux that is generated within the incremental reach segment i. In most cases, the set of adjacent upstream reaches $J(i)$ will consist of either two reaches, if reach i is the result of a confluence, or no reaches if reach i is a headwater reach. The functional relationships determining reach i flux are given by

$$F_i^* = \left[\left(\sum_{j \in J(i)} F_j' \right) A\left(\mathbf{Z}_i^{\mathrm{S}}, \mathbf{Z}_i^{\mathrm{R}}; \boldsymbol{\theta}_{\mathrm{S}}, \boldsymbol{\theta}_{\mathrm{R}}\right) + \left(\sum_{n=1}^{N_{\mathrm{S}}} S_{n,i} \alpha_n D_n\left(\mathbf{Z}_i^{\mathrm{D}}; \boldsymbol{\theta}_{\mathrm{D}}\right) \right) A'\left(\mathbf{Z}_i^{\mathrm{S}}, \mathbf{Z}_i^{\mathrm{R}}; \boldsymbol{\theta}_{\mathrm{S}}, \boldsymbol{\theta}_{\mathrm{R}}\right) \right] \varepsilon_i \quad (1)$$

The first summation term represents the amount of flux that leaves upstream reaches and is delivered downstream to reach i, where F_j' equals measured flux, F_j^{M}, if upstream reach j is monitored or, if it is not, is given by the model-estimated flux F_j^*. $A(\cdot)$ is the stream delivery function representing loss processes acting on flux as it travels along the reach pathway. This function defines the fraction of flux entering reach i at the upstream node that is delivered to the reach's downstream node. The factor is a function of measured stream and reservoir characteristics, denoted by the vectors $\mathbf{Z}^{\mathrm{S}}$ and $\mathbf{Z}^{\mathrm{R}}$, with corresponding coefficient vectors $\boldsymbol{\theta}_{\mathrm{S}}$ and $\boldsymbol{\theta}_{\mathrm{R}}$. If reach i is a stream, then only the $\mathbf{Z}^{\mathrm{S}}$ and $\boldsymbol{\theta}_{\mathrm{S}}$ terms determine the value of $A(\cdot)$; conversely, if reach i is a reservoir then the terms that determine $A(\cdot)$ consist of $\mathbf{Z}^{\mathrm{R}}$ and $\boldsymbol{\theta}_{\mathrm{R}}$.

The second summation term represents the amount of flux introduced to the stream network at reach i. This term is composed of the flux originating in specific sources, indexed by $n = 1, \ldots, N_{\mathrm{S}}$. Associated with each source is a source variable, denoted by S_n, and its associated source-specific coefficient, α_n. This coefficient retains the units that convert the source variable units to flux units. The function $D_n(\cdot)$ represents the land-to-water delivery factor. For sources associated with the landscape, this function, along with the source-specific coefficient, represents the rate at which the source variable is converted to nitrogen mass that is delivered to streams. The land-to-water delivery factor is a source-specific function of a vector of delivery variables, denoted by $\mathbf{Z}_i^{\mathrm{D}}$, and an associated vector of coefficients $\boldsymbol{\theta}_{\mathrm{D}}$. For point sources that are described by a measured discharge of mass directly to the stream channel (e.g., municipal wastewater effluent), the delivery factor takes on a value of 1, with no underlying factors acting as determinants, and the estimated source-specific coefficient should be close to 1. The last term in the equation, the function $A'(\cdot)$, represents the fraction of flux originating in and delivered to reach i that is transported to the reach's downstream node and is similar in form to the stream delivery factor defined in the first summation term of the equation. If reach i is classified as a stream (as opposed to a reservoir reach), the nitrogen introduced to the reach from its incremental drainage area receives the square root of the reach's full instream delivery. This assumption is consistent with the notion that contaminants are introduced to the reach network at the midpoint of reach i and thus are subjected to only half of the reach's time of travel. Alternatively, for reaches classified as reservoirs, we assume that the nitrogen receives the full attenuation defined for the reach.

The multiplicative error term, ϵ_i, is applicable in cases where reach i is a monitored reach; the error is assumed to be independent and identically distributed across independent sub-basins in the intervening drainage between stream monitoring sites. Coefficient estimation is performed on the log transforms of the summed quantities in Equation (1) using nonlinear least-squares estimation (Schwarz *et al.*, 2006).

Nitrogen loss in streams is modeled according to a first-order decay process (Chapra, 1997) in which the fraction of the nitrogen mass originating from the upstream node and transported along reach i to its downstream node is estimated as a continuous function of the mean water time of travel (T_i^{S}; units of time) in reach i and a first-order reaction rate that is expressed as a power function of the mean water depth, D_i, such that

$$A\left(\mathbf{Z}_i^{\mathrm{S}}, \mathbf{Z}_i^{\mathrm{R}}; \boldsymbol{\theta}_{\mathrm{S}}, \boldsymbol{\theta}_{\mathrm{R}}\right) = \exp\left(-\theta_{\mathrm{S1}} D_i^{\theta_{\mathrm{S2}}} T_i^{\mathrm{S}}\right) \tag{2}$$

where θ_{S1} (a coefficient in units of length^{-1} time^{-1}) and θ_{S2} are estimated coefficients. A similar power function has been previously evaluated in SPARROW for streamflow (Alexander *et al.*, 2002a; Elliott *et al.*, 2005; Schwarz *et al.*, 2006). The nitrogen loss-rate coefficient (in units of reciprocal time), which is calculated as the product of the estimated coefficients and mean water depth, is dependent on properties of the water column that are proportional to water volume, such as streamflow and depth (Stream Solute Workshop, 1990).

Nitrogen loss in lakes and reservoirs is modeled according to a first-order process (e.g., Kelly *et al.*, 1987) in which the fraction of the nitrogen mass originating from the upstream reach node and transported through the reservoir segment of reach i to its downstream node is estimated as a function of the reciprocal of the areal hydraulic load $\left(q_i^{\mathrm{R}}\right)^{-1}$ (units of length time^{-1}) for the reservoir associated with reach i and an apparent settling velocity coefficient (θ_{R0}; units of length time^{-1}), such that

$$A\left(\mathbf{Z}_i^{\mathrm{S}}, \mathbf{Z}_i^{\mathrm{R}}; \boldsymbol{\theta}_{\mathrm{S}}, \boldsymbol{\theta}_{\mathrm{R}}\right) = \frac{1}{1 + \theta_{\mathrm{R0}}\left(q_i^{\mathrm{R}}\right)^{-1}} \tag{3}$$

Additional details on this formulation are given in Alexander *et al.* (2002a) and Schwarz *et al.* (2006). The areal hydraulic load is estimated in this study as the quotient of the outflow discharge to the surface area of the impoundment, but may also be determined from the ratio of the mean depth to the solute residence time of the impoundment.

Model Estimation

Our application of the model to catchments and streams in the northeastern United States is based on a previous SPARROW application (Moore *et al.*, 2004) to the 1:100,000 scale National Hydrography Dataset (NHD; USGS, 1999). The water-quality and geographic data for the nutrient sources and watershed properties are described in detail in this earlier study (Moore *et al.*, 2004). The parameters of Equations (1)-(3) are estimated using the mean-annual total nitrogen loads at 65 stream monitoring stations. The mean-annual loads were computed by applying flux-estimation procedures to daily records of flow and periodic measurements of total nitrogen concentration; total nitrogen is determined as the sum of dissolved nitrate-nitrite and total organic plus ammonia nitrogen concentration measurements (Moore *et al.*, 2004). The explanatory variables in the model include four nitrogen sources (municipal wastewater discharges, atmospheric deposition, and runoff from cultivated and developed urban and suburban lands), one terrestrial land-to-water attenuation factor (soil permeability) that is applied with equal proportional effect to all sources except municipal wastewater discharges, and a total of three nitrogen-decay coefficients for streams and reservoirs as specified in Equations (2) and (3).

The modeled region contains approximately 42,000 stream reaches having a mean catchment size of 4.4 km^2, based on watershed boundary delineations from 30-m digital elevation data. The mean-annual streamflow for each stream reach was calculated as the sum of the mean-annual runoff for the incremental drainage area of each stream catchment and that from all upstream catchments. For 211 available gaged stream stations, most (53%) had estimated streamflows within 5% of the gaged flow; 83% had estimated flows within 10%, and 93% had estimated flows within 15% of the gaged flow. Time-of-travel estimates for Equation (2) were computed from published regression equations (Jobson, 1996) that estimate mean water velocity as a function of mean streamflow, reach slope, and the total drainage area of each stream reach. Selected properties of the approximately 23,000 headwater NHD reaches are presented in Table 1.

We estimate two additional aquatic transport functions in the model to assist in quantifying the rates of nitrogen removal in northeastern streams and lakes as a continuous function of the size and hydraulic properties of these water bodies. The parameters of these functions are estimated using current literature rates of nitrogen removal reported for streams and lakes in North America, Europe, and New

TABLE 1. Geometric and Hydraulic Properties of NHD Headwater Reaches for Northeastern U.S. Streams.

Metric	Percentiles (Number Reaches = 23,253)				
	10th	25th	50th	75th	90th
Drainage area (km^2)	0.8	1.8	3.7	7.3	12.9
Mean-annual streamflow (m^3/s)	0.02	0.04	0.08	0.15	0.28
Mean water depth* (m)	0.06	0.07	0.10	0.12	0.16
Mean water travel time (days)	0.02	0.05	0.09	0.14	0.19

*Depth = $0.2612Q^{0.3966}$, where Q is the mean-annual streamflow (Alexander *et al.*, 2000).

Zealand (Seitzinger *et al.*, 2002; Böhlke *et al.*, 2004; Mulholland *et al.*, 2004). This information provides a generally comprehensive description of what is currently known about nitrogen transport across large spatial scales, and thus, gives a more refined method for assessing the influence of headwater sources and processes on downstream nutrient conditions.

The stream transport function describes the fraction of nitrogen mass that is transported along the experimentally studied reaches, denoted by TR_i^{S} for reach i, expressed as a function of the stream characteristics according to

$$TR_i^{\mathrm{S}} = \exp\left(-\theta_{\mathrm{S1}} D_i^{\theta_{\mathrm{S2}}} T_i^{\mathrm{S}}\right)\varepsilon_i^{\mathrm{S}} \tag{4}$$

where the variables and coefficients in the exponential function are identical to those in Equation (2), and $\varepsilon_i^{\mathrm{S}}$ is an error term, independent across measurements, having a variance that may differ from the error term appearing in Equation (1). Literature estimates of the nitrogen transport fraction, TR_i^{S}, are based on denitrification and mass-balance measurements of nitrogen loss for 12 streams (see Seitzinger *et al.*, 2002; Böhlke *et al.*, 2004; Mulholland *et al.*, 2004; we use the reported estimates of the mean depth and water time of travel for the studied reaches). Many of the measurements of denitrification are based on summer, low-flow conditions and are assumed to be representative of the rates during other periods of the year.

The reservoir transport function describes the fraction of the nitrogen mass that is transported in experimentally studied lakes, denoted by TR_i^{L} for lake i, expressed according to

$$TR_i^{\mathrm{L}} = \frac{1}{1+\theta_{\mathrm{R0}}\left(q_i^{\mathrm{R}}\right)^{-1}}\varepsilon_i^{\mathrm{L}} \tag{5}$$

where the coefficient and variable in the denominator of the expression are the same as those defined in Equation (3), and $\varepsilon_i^{\mathrm{L}}$ represents an independent and identically distributed error term having a variance that potentially differs from ε_i and $\varepsilon_i^{\mathrm{S}}$ in Equations (1) and (4). The literature estimates of the nitrogen transport fraction, TR_i^{L}, are based on denitrification and mass-balance measurements of nitrogen loss for 36 lakes (see Seitzinger *et al.*, 2002; we use the reported estimates of the mean depth and water residence time for the studied lakes to calculate the areal hydraulic load).

The three components comprising the SPARROW model consist of Equation (1) [with instream delivery fraction given by Equation (2) and reservoir delivery fraction given by Equation (3)] estimated using the instream load observations for 65 stream monitoring stations, Equation (4) estimated using the 12 literature estimates of stream delivery fraction, and Equation (5) estimated using the 36 literature estimates of lake delivery fraction. A two-step procedure was used to simultaneously estimate the coefficients of the three equations. In the first step, the model is estimated using all observations, both those associated with the monitoring station data and those associated with the literature measurements, with each observation given equal weight. The error estimates from this initial model are consistent estimates of the true errors and are used to estimate the relative variances of the three model components. The model was then re-estimated in a second step using weighted nonlinear least squares, weighting each observation according to the respective reciprocal variance (i.e., 1/RMSE^2; RMSE = root mean square error) of the model error (weighting factors: lakes = 1/0.2925; streams = 1/0.0099; monitoring loads = 1/0.16). The weights are used to account for the level of uncertainty associated with the different types of measurements used in the model.

Model Predictions and Simulation Methods

We use the estimated model to investigate the supply and transport of nitrogen and water in streams of varying sizes within the northeastern river network, ranging from small headwater streams to large rivers. Stream size is defined according to the Horton-Strahler stream-order number (Horton, 1945; Strahler, 1957; see Figure 3). We assigned stream-order numbers to NHD reaches using a previously developed algorithm (K. Lanfear, USGS, 2005, written communication). The Strahler ordering system produces a dendritic, hierarchical classification in which headwater streams (i.e., streams with no tributaries) are classified as order 1 with all subsequent streams of the nth order being located downstream of the confluence of two $(n-1)$th order streams. The number of reaches and sum of the incremental drainage area for the NHD streams both decline at a similar rate with increasing stream order (see Figure 3b)

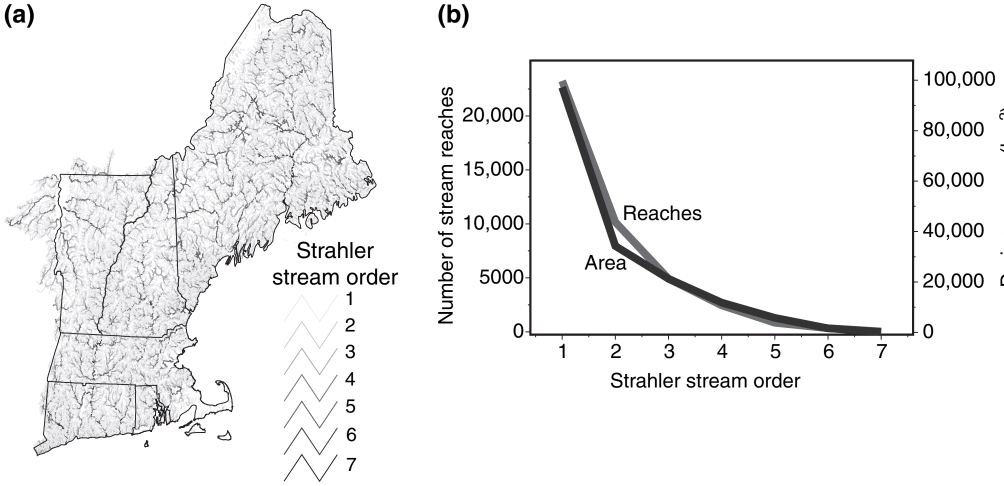

FIGURE 3. Stream Reaches From the National Hydrography Dataset for the Northeastern United States: (a) Strahler Stream-Order Number by Reach; (b) Number of Reaches and Total Drainage Area for Stream Reaches Classified by Strahler Stream-Order Number.

that is generally consistent with Horton's geometric-scaling laws. These scale-invariant laws correspond to the fractal structure of drainage networks (Peckham and Gupta, 1999) and describe fundamental mathematical properties that relate to the similar spatial organization of various topographic and geometric properties, including stream number, drainage area, and stream length, throughout the hierarchy of stream network systems (Rodriguez-Iturbe and Rinaldo, 1997; Peckham and Gupta, 1999).

We use the Horton-Strahler stream classification with the model predictions to quantify the pollutant sources and rates of nitrogen delivery within streams of varying sizes in the northeastern NHD river network. We track nitrogen delivery to NHD reaches from the four pollutant sources within the *incremental* drainage area of each reach. The *incremental* area of a stream reach is defined as the catchment drainage area from which water and nitrogen directly enter the reach, independent of the drainage area of upstream reaches that hydrologically connect to the reach. We summed the mass of nitrogen delivered from all incremental drainage areas of NHD stream reaches within each Strahler stream-order class and for each pollutant source. Similarly, we also use the network data on streamflow to quantify the flow contributions from the incremental drainage areas of different sized NHD reaches by summing the incremental reach flows separately among reaches with similar Strahler stream-order numbers.

We use several model simulations to investigate the influence of nitrogen sources, streamflow, and instream processing in headwater catchments on the mean-annual nitrogen and flow conditions in downstream waters. First, to quantify the downstream contributions of headwater nitrogen loads, we set the total inputs from all nitrogen sources in headwater streams to zero in the model and track the resulting change in nitrogen loads in all higher-order streams (orders 2-7). The results quantify the percentage of the downstream loads in each Strahler stream-order class that originates collectively from the 23,253 headwater catchments. Similar evaluations for mean-annual flow quantify the percentage of the flow in each stream-order class that originates from headwater catchments.

Second, we refine the model simulations to investigate the downstream effects on nitrogen loads from changes in pollutant sources in various collections of randomly selected headwater catchments. These simulations, which randomly select from 10% (2,325 reaches) to 90% of the reaches (20,928), give useful information about the sensitivity of the downstream changes in loads when significant changes occur in the pollutant sources in a subset of headwater reaches.

Finally, to quantify the downstream effects of loss processes (e.g., denitrification) in headwater streams and reservoirs, we set the decay rate to zero in headwater streams and reservoirs and track the change in the nitrogen loads in first- and all higher-order streams. For each stream-order class, we compute the mean of the percentage changes and the standard deviation among all reaches, with the latter metric indicating the spatial variability among streams of the same order. The adjustment to the decay rate in these simulations is identical to setting the water travel time (or areal hydraulic load for reservoirs) to zero because both impart identical effects in the decay functions given in Equations (2) and (3).

Results of the Model Estimation

The parameter coefficients and model performance statistics are given in Table 2. The model explains 95% of the spatial variability in log-transformed mean-annual total nitrogen loads (i.e., $R^2 = 0.95$). All model coefficients are statistically significant for $\alpha = 0.10$. The prediction accuracy is ±44% for individual reaches, based on the RMSE of the model for one standard deviation variability. Model predictions of nitrogen yields from predominantly forested, cultivated, and developed urban and suburban catchments compare favorably with those reported in the literature for similar land uses (e.g., Beaulac and Reckhow, 1982). For example, predicted yields from forested catchments (median = 2.7 kg/ha/year; interquartile range from 1.8 to 3.4 kg/ha/year) are 20-25% of the predicted yields for cultivated and developed catchments.

The inclusion of literature nitrogen loss rates in the model estimation provides sufficient statistical power to quantify nitrogen loss as a continuous function of the hydraulic conditions in streams and reservoirs in the northeastern United States (Table 2; Figure 4). We find that the continuous stream loss function gives first-order nitrogen loss rates (Figure 4a) that decline with increases in mean water depth (also mean streamflow). This inverse relation is consistent with that reported for other SPARROW nitrogen models (Alexander *et al.*, 2002a; Schwarz *et al.*, 2006) and is also consistent with the widely held scientific notion that water-column nitrogen loss rates generally decline with increasing water depth (e.g., Stream Solute Workshop, 1990; Peterson *et al.*, 2001; Thomas *et al.*, 2001). The rates estimated here for small streams (depths < 0.39 m) are generally consistent with the single loss rate (0.82 day^{-1}) that was estimated according to a discrete loss function in the previous northeastern SPARROW model (Moore *et al.*, 2004). The first-order rates from the continuous loss function (Figure 4a) are centered on the previously estimated constant rate and provide a reasonable description of the dimensions of the inverse relation over these smaller stream sizes. Although the literature data include relatively few observations of nitrogen loss in larger streams (those with depths greater then 0.39 m; Figure 4a), these observations provide important complementary information for estimating nitrogen losses in streams of the Northeast. Attempts to estimate the model with a continuous instream loss function (i.e., Equation (2)) using only the load data from the 65 monitoring sites were unsuccessful as the model failed to converge.

The estimated nitrogen loss coefficient (i.e., mass-transfer rate) for reservoirs (Table 2) is similar to that estimated for the lake data alone (Figure 4b) – i.e., 9.9 m/year compared with 10.4 m/year, respectively – and is about five times larger than that estimated in the previous northeastern SPARROW model (Moore *et al.*, 2004; i.e., 9.9 m/year compared with 1.9 m/year, respectively). Based on a re-estimation of the coefficients in this previous model using a fixed reservoir mass-transfer coefficient value of 9.9 m/year, we find that a difference in the reservoir loss rate coefficient of this magnitude has relatively little effect on the estimates of the other coefficients in the earlier model. The general insensitivity of the model coefficients to such changes is consistent with suggestions by Moore *et al.* (2004) that the monitoring sites may be poorly located in relation to the

TABLE 2. Estimated Coefficients for the SPARROW Total Nitrogen Models for Northeastern U.S. NHD Streams.

	Estimated model*		
Predictor Variables	**Coefficient**	**Units**	**Standard Eror**
Sources			
Municipal wastewater	1.42	Dimensionless	0.39
Atmospheric deposition	0.412	Dimensionless	0.058
Cultivated agricultural land	678	$\text{kg/km}^2\text{/year}$	260
Developed urban and suburban land	726	$\text{kg/km}^2\text{/year}$	232
Land-to-water delivery			
Soil permeability	0.387	Dimensionless	0.154
Instream loss			
θ_{S1}	0.0513	$\text{m}^{-1}\ \text{day}^{-1}$	0.0084
θ_{S2}	-1.319	dimensionless	0.076
Reservoir/lake loss	9.9	m/year	1.6
Number of observations	113		
R^2	0.95		
RMSE (root mean square error in %)	44.2		

*The model as defined by Equations (1)-(5) is estimated using load data for the 65 stream monitoring sites and additional literature measurements of the nitrogen loss rate in streams ($N = 12$) and lakes ($N = 36$) in New Zealand, North America, and Europe (data are from Seitzinger *et al.*, 2002; Böhlke *et al.*, 2004; Mulholland *et al.*, 2004).

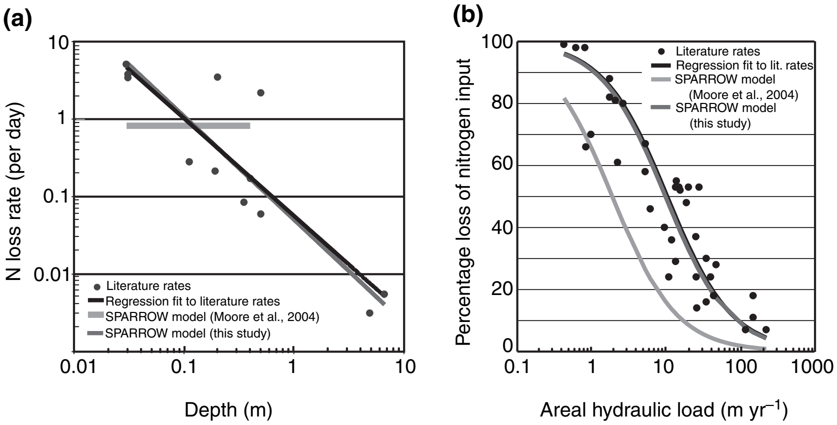

FIGURE 4. Nitrogen Loss in Streams, Lakes, and Reservoirs: (a) Streams in Relation to Mean Water Depth and (b) Reservoirs and Lakes in Relation to the Areal Hydraulic Load. The literature rates are for streams and lakes in North America, Europe, and New Zealand (Seitzinger *et al.*, 2002; Böhlke *et al.*, 2004; Mulholland *et al.*, 2004). The literature rates in (a) were originally reported as a percentage of nitrogen inputs in Seitzinger *et al.* (2002) and are converted to first-order rates here using the corresponding measurements of the water time-of-travel. The regression fit for the literature rates in (a) is obtained from a log-linear regression: $k = 0.0573d^{-1.246}$, where k is the first-order rate coefficient and d is the mean water depth; $R^2 = 0.770$. The regression fit for the literature rates for lakes in (b) is obtained from a nonlinear regression: $N = 1 - [1/(1 + 10.4q^{-1})]$, where N is the fractional nitrogen loss and q is the areal hydraulic load; $R^2 = 0.757$; the estimates are virtually identical to those estimated in the SPARROW model in this study.

reservoirs in the northeastern catchments. However, relatively small rates of nitrogen loss in reservoirs are generally consistent with previous SPARROW models applied in the United States (Smith *et al.*, 1997) and New Zealand (Alexander *et al.*, 2002a).

Other comparisons with the previous northeastern model (Moore *et al.*, 2004) indicate that the model estimated here gives an equally plausible description of nitrogen sources and transport in the northeastern catchments and streams. Although the estimated model yields a slightly higher model error (RMSE = 44.2%) as compared with that for the previous model (RMSE = 40.4%), the changes in the mean estimates of the model coefficients are within the measures of uncertainty as expressed by the standard errors of the coefficients. Differences in the quantities of nitrogen delivered to streams from the various sources are relatively small; the model reported here (Table 2) indicates that the contributions from municipal wastewater sources are about 25% higher than estimated in the previous model, whereas the nitrogen contributions from cultivated and developed urban/suburban lands are about 25% lower. Predictions of nitrogen yield for about 6,600 catchments with predominantly cultivated, developed urban/suburban, or forested land uses differ by less than 25% from the model predictions generated by the previous model.

The Supply and Delivery of Nitrogen and Water to Streams

Based on comparisons of model predictions of flow and the nitrogen loads for the incremental drainages of NHD streams of varying sizes (as defined by Horton-Strahler class; Figure 5), headwaters catchments, in aggregate, account for nearly one-half of the total nitrogen mass supplied to all streams – i.e., headwaters account for 45% of the total nitrogen mass or load that is delivered to all stream reaches from the incremental drainage areas of reaches in the northeastern NHD river network (Figure 5a). By comparison, second- and higher-order streams account for less than 20% of the total nitrogen load that is delivered to all streams. This percentage declines progressively (as does the drainage area; Figure 5b) with increases in stream order.

The nitrogen yields (i.e., loads per unit drainage area) from the incremental drainages (Figure 5b) of headwater streams (mean = 5.5 kg/ha/year) are among the smallest among all stream orders. Atmospheric deposition is the largest source of nitrogen in headwater catchments, accounting for nearly 70% of the total incremental load delivered to headwater streams, with cultivated land and urban/suburban sources accounting for about 27% of the incremental load (see Figure 5c). Most headwater catchments where atmospheric deposition is high are predominantly forested; more than 50% of the headwater catchments have more than 85% forested land area. Cultivated and urban/suburban lands account for more than 10% of the land area in about 75% of the headwater streams. The nitrogen yields increase progressively with stream order (Figure 5b), reflecting the increase in municipal wastewater discharges associated with increases in population in the vicinity of the higher-order streams (see Figure 5c). The large increase in yield in stream order 6 (Figure 5b)

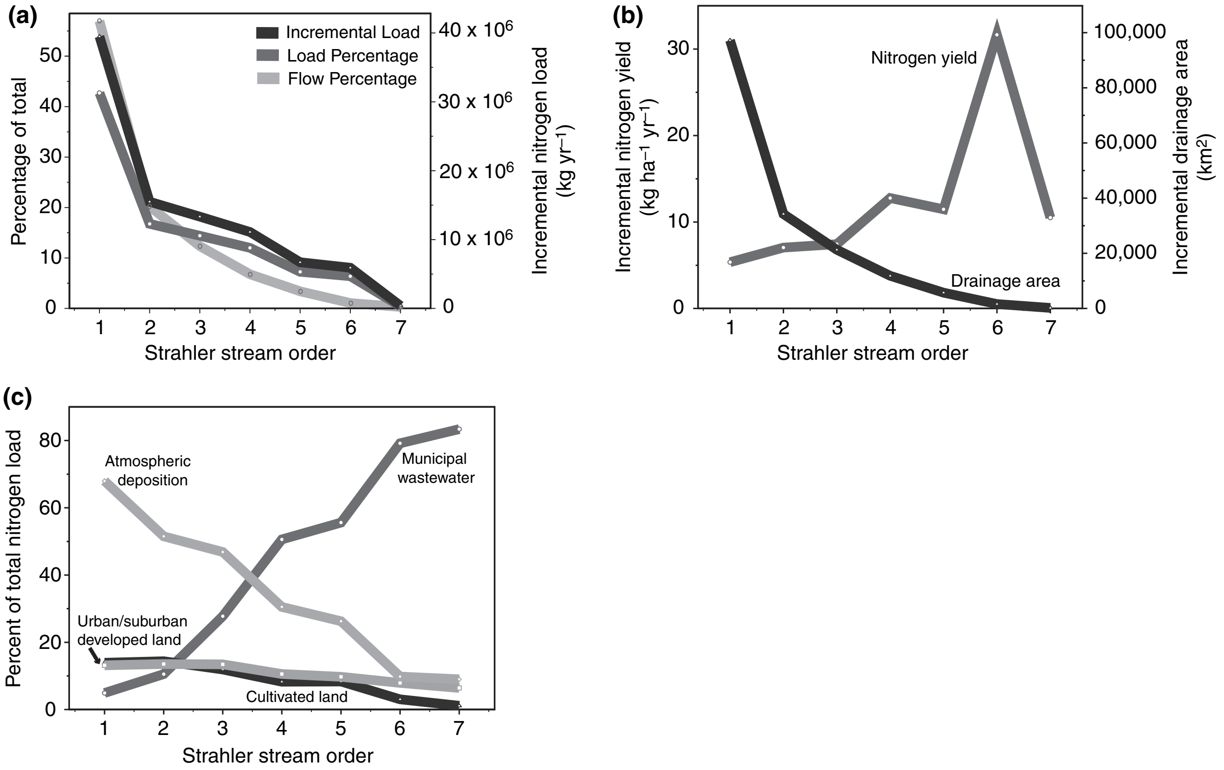

FIGURE 5. Mean-Annual Streamflow and Total Nitrogen Loads, Yields, and Sources for Streams of the Northeastern United States in Relation to Strahler Stream-Order Class: (a) Load From the Incremental Reach Watersheds and the Load and Flow, Expressed as a Percentage of the Sum of the Incremental Load and Flow in Streams of All Orders; (b) Yield and Drainage Area of the Incremental Reach Watersheds; (c) Sources of the Nitrogen Loads, Expressed as a Percentage of the Sum of the Incremental Load in Streams of the Same Order.

includes incoming loads to the lower Connecticut River, where major municipal wastewater discharges occur; note that the percentage of the total incremental load attributable to wastewater discharges increases from 50% in stream order 5 to nearly 80% in stream order 6. Overall, these results indicate that, although the nitrogen yields in headwater streams are generally the smallest among all stream orders (Figure 5b), collectively, the total loads of nitrogen leaving headwater reaches are similar in size to the sum of all loads that originate in the incremental watersheds of higher-order streams.

The mean-annual flow contributions from the incremental drainage areas of NHD reaches (Figure 5a) indicate that first-order streams account for approximately 60% of the total volume of mean-annual flow that is contributed to all northeastern streams. Similar to that observed for other stream properties (e.g., nitrogen load, drainage area), the flow contributions that originate in the incremental watersheds of higher-order streams, expressed as a percentage of the total flow volume in all streams, are relatively small and decline monotonically with increases in stream order, from about 20% for second-order streams to less than 1% for sixth- and seventh-order streams.

Downstream Influences of Headwaters

The results of the model simulations (Figures 6 and 7) indicate a demonstrable effect of the nitrogen sources and flow in headwater catchments on the mean-annual nitrogen and flow conditions in downstream reaches. The percentage of the mean-annual nitrogen load in reaches that is contributed from headwater streams steadily declines with increases in stream order through the sixth-order streams (Figure 6a). We found that second-order streams receive approximately 65% of their nitrogen loads from headwater streams. This percentage contribution of headwater streams ranges from 43% to 87% of the nitrogen loads in second-order streams, based on the two-thirds of the streams that lie within a one standard deviation range in this stream-size class. The lowest contribution of headwater streams to nitrogen loads is about 40% as observed in sixth-order streams. The higher fraction of headwater nitrogen contributions in streams of order 7 as compared with order 6 reflect differences in the load response and potentially the network structure of two independent river basins, the Connecticut and Penobscot (we executed separate simulations for these drai-

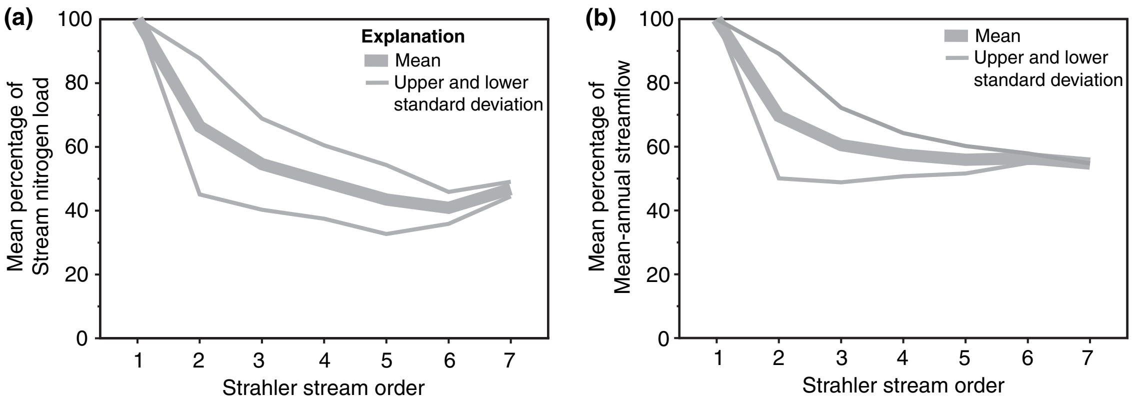

FIGURE 6. The Percentage of the Mean-Annual Nitrogen Load and Streamflow in Streams of the Northeastern United States That Originates in Headwater Catchments: (a) Nitrogen; (b) Streamflow. The estimates are obtained in model simulations by setting the total nitrogen source loadings or streamflow to zero in 23,253 headwater (Strahler order 1) catchments and quantifying the resulting percentage change in the downstream nitrogen loads or flow. The upper and lower standard deviation lines reflect the range of variability (associated with one standard deviation) observed in reaches in the estimated percentage reduction in nitrogen load.

nages and found monotonically decreasing headwater contributions with increasing stream order in each basin that are similar to those shown in Figure 6a for stream orders 1-6).

We find that the percentage of the mean-annual flow in network streams that originates from headwater catchments exhibits a monotonic decline from headwaters to high-order streams similar to that found for nitrogen loads, but is somewhat larger in magnitude than observed for the nitrogen loads (Figure 6b). Headwater catchments contribute approximately 70% of the water volume in second-order streams. Moreover, the flow contributions of headwater catchments to the mean water volume in downstream reaches decline only marginally to about 55% in fourth- and higher-order streams.

The large contributions of headwater nitrogen sources and flow volumes to mean-annual nitrogen loads and flow in streams of all sizes are generally consistent with the high density of headwater streams and the high frequency of their connections to the channels of all higher-order streams; these are intrinsic properties of dendritic river networks. The proportion of all lower-order streams that are tributary to streams of a given Strahler order conforms to fundamental scaling properties defined according to Tokunaga's Law (e.g., see discussion in Dodds and Rothman, 2000). According to this law for commonly observed values of network scaling parameters (Tokunaga, 2003), first-order streams represent the single, most prevalent Horton-Strahler stream-order class with high frequencies of tributary connections to all higher-order streams within river networks. Considering all of the lower-order tributaries to higher-order streams in a network, the percentage of lower-order streams that are theoretically classified as first-order declines with an increase in stream order, but levels off to about 50% (see Table 3). These percentages of first-order tributary connections to higher-order streams are generally similar for the northeastern NHD river network. Therefore, first-order streams are the most frequently occurring tributary to all higher-order streams and represent the origin of a major fraction of the water and nitrogen loadings in streams of all sizes within the northeastern United States.

Refinements to the model simulations to assess the downstream effects of changes in nitrogen sources in a subset of the headwater catchments (Figure 7) provide insight into the magnitude of the water-quality effects in cases where pollutant sources and land use undergo significant changes in a subset of headwater streams. We find that the mean percentage of the stream nitrogen load that originates in headwater catchments declines monotonically with increases in Strahler stream order through the sixth-order streams; the mean percentage shows an approximate leveling in magnitude in fourth- and higher-order streams. The rate of decline is generally similar for simulations involving changes in sources in 50% or more of the headwater reaches; a slightly smaller rate of decline is noted in the mean percentage for simulations involving fewer headwater reaches. The results indicate that nitrogen sources in as few as 50% of the headwater catchments account for 20-25% of the nitrogen loadings in fourth- and higher-order streams; sources in as few as 25% of the headwater catchments account for 10-12% of the nitrogen loadings in fourth- and higher-order streams.

A simulation of the downstream effects of nitrogen loss processes in headwater streams and reservoirs (related to denitrification and long-term storage) indi-

TABLE 3. Headwater Tributary Connections to Higher-Order Streams in River Networks.

	Headwater (First-Order) Streams		
	Percentage of All Lower-Order Tributary Reaches Classified as First-Order Streams		
Strahler Stream-Order Class	**Theoretical***	**New England NHD**	**Number of NHD Stream Reaches**
2	100.0	100.0	11,775
3	66.7	46.5	5,019
4	57.1	54.3	2,527
5	53.3	57.7	1,181
6	51.6	53.5	497
7	50.8	51.1	45

*The estimates are based on Tokunaga's law for describing the average number of streams of a given order that are tributaries to higher-order streams (Dodds and Rothman, 2000). For common values of the network scaling parameters (Tokunaga, 2003), the average number of first-order tributaries to higher-order streams of order v is computed as 2^{v-1}. In the table, the average number of first-order tributaries to a specified stream order is expressed as a percentage of the total number of all lower-order connecting tributaries for that stream order.

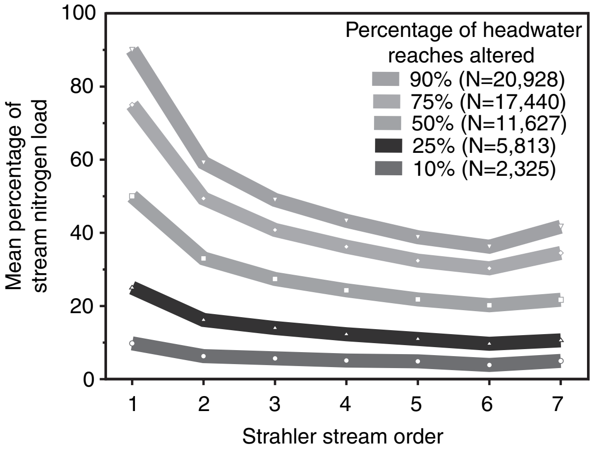

FIGURE 7. The Mean Percentage of the Stream Nitrogen Load in National Hydrography Dataset Reaches Originating From Randomly Selected Subsets of the Headwater Catchments in Relation to Strahler Stream-Order Class. The estimates are obtained in model simulations by setting the total nitrogen source loadings to zero in a randomly selected set of headwater catchments, ranging in number from 2,325 (10% of the total reaches of 23,253) to 20,928 reaches (90% of the total), and quantifying the resulting percentage change in the downstream nitrogen loads.

cates that nitrogen losses in headwaters reduce the nitrogen loads by about 8% in first-order (headwater) streams (standard deviation = ~0-30%), 5% in second-order streams (standard deviation = <1-12%), and about 3-4% in fourth- and higher-order streams. These estimates are calculated as the change in simulated load expressed as a percentage of the original decayed load. The reported changes in load reflect the integrated effects of instream biochemical processing (e.g., denitrification) and water travel times within stream reaches (see Table 1) on the rates of stream nitrogen loss (note that the nitrogen delivered to headwater stream channels from point- or land-based sources is assumed to enter, on average, the midpoint of total channel length of the headwater reach and is therefore subjected to only half of the water time of travel). The large variability in nitrogen loss in headwater streams (i.e., ~0-30%) reflects differences among first-order reaches in the mean water depth and water travel time. Although nitrogen losses in headwaters streams cause relatively small changes in the nitrogen loads in higher-order streams on average, the downstream change in nitrogen loads is actually large relative to the change in headwater loads – i.e., the downstream relative changes in load range from 40% to 60% of the relative change observed in the headwater nitrogen loads.

Uncertainties and Research Needs

Headwater streams are operationally defined in our assessment as Horton-Strahler first-order perennial streams, based on the 1:100,000-scale NHD river network. The Horton-Strahler classification of NHD streams gives a reasonable approximation of headwater locations in relation to those of higher-order streams within the larger drainage network. This definition is based on fundamental principles that describe the hierarchy of the spatial organization of various topographic, hydrologic, and geometric properties of river networks. Comparisons of the Horton-Strahler classification of NHD streams with classifications for more finely resolved 1:24,000-scale streams (Andrews *et al.*, 2002) suggest that NHD headwater channels may be generally classified as second-order streams at this finer scale. Thus, the first-order headwater streams in our study reflect the flow and nitrogen contributions from many smaller streams, including those from intermittent ephemeral streams.

The use of the Horton-Strahler classification to define headwaters has received some criticism (e.g., Gomi *et al.*, 2002; Whiting and Bradley, 1993) because it does not explicitly include hydrological and biological process-related definitions of transitional upland headwater reaches; these are reach locations where the influence of hillslope processes on water and material flux tends to give way to the fluvial routing processes that dominate in higher-order streams. There are, however, intrinsic ambiguities in defining headwater streams that arise from the dynamic spatial and temporal nature of hydrological and biological processes in low-order streams; this contributes, for example, to the lack of consistent definitions of intermittent and ephemeral headwater streams (Meyer and Wallace, 2001).

Additional studies are needed to investigate the effects on our interpretations of alternative definitions of headwater streams in relation to various hydrological- and biogeochemical-process characteristics. This research will demand the use of more spatially detailed digital topography (e.g., 1:24,000 or finer scales) as well as equally refined watershed data, including data on climatic conditions, point and diffuse contaminant sources, and instream nutrient concentrations, for use as input to regional-scale source-transport models.

Our model analyses assume that mean-annual, instream nitrogen losses can be described as a first-order process, mediated by a loss-rate coefficient, the mean-annual solute travel time within stream channels, and mean water depth (or mean-annual streamflow). The first-order assumption of the loss process is potentially subject to some uncertainties, related to the limiting effects of saturation kinetics on denitrification rates (e.g., Garcia-Ruiz *et al.*, 1998), especially in highly developed watersheds where high nitrate concentrations can occur. Under such conditions, for example, highly developed headwater catchments could have more far reaching downstream effects than under the assumed first-order kinetics of the model. The first-order loss function also reflects the aggregate, net time-averaged effect of the hydraulic and biogeochemical properties of streams of varying size; this function does not isolate the effects of specific properties of the benthic sediment, such as organic carbon and oxygen content.

Although our modeling analysis is well suited to examine the natural and human-related processes that control the downstream transport and fate of the nitrogen over annual or longer time periods, it does not include any explicit assessment of the effects of seasonal or other temporal variability in nitrogen loss and streamflow (e.g., heterotrophic and autotrophic production and respiration) on the transport and downstream fate of nitrogen. These short-term processes are included in dynamic mechanistic models (e.g., HSPF; Bicknell *et al.*, 2001), but these models are rarely used to track the geography of nitrogen losses and the downstream transport and fate of nutrients in large watersheds (e.g., Filoso *et al.*, 2004). One difficulty is that the influence of short-term uptake and cycling processes on the downstream fate of various nitrogen forms is not currently well understood, based on available experimental research (Peterson *et al.*, 2001; Grimm *et al.*, 2003). Considerable progress has been made in measuring nitrogen cycling at the reach and catchment scales in small streams (e.g., Peterson *et al.*, 2001; Hall and Tank, 2003; Mulholland *et al.*, 2004; Royer *et al.*, 2004), but longitudinal studies are needed to quantify the effects of autotrophic and heterotrophic uptake and cycling of nutrients in low-order streams on nutrient conditions in higher-order systems. This includes an improved tracking of the separate fate of organic and inorganic nitrogen in models to enhance understanding of the headwater origins of bio-available nitrogen in downstream waters. Observational data and model improvements are also needed to account for the effects of long ground-water residence times that can delay the delivery of nitrogen from land-based sources to downstream waters (e.g., Böhlke and Denver, 1995; McIsaac *et al.*, 2001).

CONCLUSIONS

Our synthesis of existing watershed research and the modeling assessment of northeastern U.S. streams demonstrate the important role that headwaters play in the supply, transport, and fate of water and nitrogen in river networks. This provides important information for the water-resource community regarding decisions on the regulation and management of headwater streams. The results also provide scientific information that potentially broadens understanding of the extent of Federal CWA jurisdiction in waters of the United States, a topic of continuing importance as indicated by recent U.S. Supreme Court cases. The procedures for establishing Federal jurisdiction that have emerged from these cases stress the need for technical and scientific information about whether a “significant nexus” exists between upland waters and downstream navigable waters and their tributaries. Such a connection could be based on evidence that the use, degradation, or destruction of non-navigable headwaters demonstrably influences the waters covered by the CWA.

The results reported here are consistent with the notion that pollutant sources and hydrological and biogeochemical processes in headwaters are physically and bio-chemically connected to the water-quality conditions in downstream waters of widely varying sizes, including navigable waters and their tributaries. Experimental studies of nitrogen transport in streams and rivers indicate that hydrological processes in headwater catchments influence stream nitrogen conditions by controlling the recharge of subsurface water stores and the flow paths and residence times of water through landscapes. The dynamic coupling of hydrological and biogeochemical processes in upland streams further controls the chemical form, timing, and longitudinal distances of nitrogen and other solute transport to downstream waters. Headwater influences on water-quality conditions in downstream waters are likely facilitated by the high density of headwater streams and their high frequency of tributary linkages to the channels of higher-order streams in river networks. These natural dendritic properties of stream networks play an intrinsic role in the delivery of nitrogen and other pollutants to downstream receiving waters from headwater locations throughout watersheds.

Our application of a refined version of the source-transport model SPARROW illustrates many of these concepts. The results demonstrate the prominent influence of headwaters on the mean-annual flow and nitrogen conditions in streams of all sizes in the northeastern United States. We estimate that headwater catchments contribute a majority (~65%) of the nitrogen mass and water volume (~70%) in second-order streams; these contributions decline only marginally to about 40% and 55%, respectively, in fourth- and higher-order streams. We also find that the downstream effects of headwater pollutant sources of nitrogen are generally very large in absolute terms in comparison to the effects of instream processing and long-term nitrogen storage in headwater streams. Nevertheless, the downstream effects of nitrogen processing and storage within headwater streams are still quite large in relative terms, ranging from about 40% to 60% of the magnitude of the relative effects observed in the headwater reaches. Moreover, because of the larger magnitude of nitrogen loads in downstream waters, the magnitude of the change in loadings related to headwater processes is actually quite large in absolute units of nitrogen mass. Our assessment of the potential downstream effects on nitrogen loads related to significant changes in land use or flows in headwater catchments indicates that the downstream nutrient loads change by approximately 50% of magnitude of the percentage of headwater reaches in which these changes occur. Thus, for example, major changes in nitrogen loads in a subset of 25% of the headwater catchments would be expected to change nitrogen loads by about 10-12% in the waters downstream of these headwaters. In view of the comparatively larger headwater flow contributions to downstream waters, we would anticipate generally larger downstream effects on mean-annual streamflow in response to major changes in the land use (e.g., pervious cover) or channel properties (e.g., channelization, water velocity) in headwater catchments and streams.

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VARIATION IN STREAM WATER QUALITY IN AN URBAN HEADWATER STREAM IN THE SOUTHERN APPALACHIANS

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Abstract. We examined the influence of a forested landscape on the quality of water in a stream originating on an urban landscape and flowing through National Forest lands. Sample sites included an urban stream (URB), a site on the same stream but within a National Forest (FOR) and 2 km downstream from the URB site, and a small, undisturbed, forested reference tributary of the main stream (REF). We monitored stream water quality from March 2002 through June 2003. Average base flows for the three stream sites were URB = $184\,\mathrm{L\,s^{-1}}$, FOR = $420\,\mathrm{L\,s^{-1}}$, and REF = $17\,\mathrm{L\,s^{-1}}$. We analyzed weekly stream water samples for NO_3^-, NH_4^+, PO_4^+, Cl^-, K, Ca, Mg, SO_4, SiO_2, pH, conductivity, total suspended solids (TSS), and bacteria on a monthly basis. Most solutes were higher in concentration at the URB site, as were conductivity, TSS, and bacteria counts. Reductions in NO_3^-, NH_4^+, and PO_4^+ concentrations between the URB and FOR sites were inferred from changes in nutrient:chloride ratios. Bacteria populations were greater and more responsive to stream temperature at the URB site. Water quality responses to changes in stream discharge varied among sites but were greater at the URB site. By all measures, water quality was consistently higher at the FOR site than at the URB site.

Keywords: water quality, surface water, urbanization, forest, sediment, bacteria

1. Introduction

Land use is one of the most important factors determining water quality (Allan and Flecker, 1993). As human populations increase and land use patterns change, resource managers, planners, and regulators need to understand the impacts of urbanization along the wildland-urban interface on water quality and aquatic resources. Paul and Meyer (2001) found that the most consistent effect of urbanization on stream ecosystems was an increase in impervious surface areas within urbanized catchments. Runoff from these urbanized surfaces and municipal discharges result in increased loading of nutrients (Tufford *et al.*, 2003) and other contaminants to streams (Davis *et al.*, 2003). Lenat and Crawford (1994) found that suspended sediment yield was greater for an urban catchment than for a forested catchment in the North Carolina piedmont. Swank and Bolstad (1994) found that the percentages of land use in non-forest cover and the surface area of paved roads per unit of land area were among the most important influences on

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baseline water quality in a southern Appalachian watershed. Similarly, Hunsaker and Levine (1995) found that the percentage of land in forest and other uses were the best predictors of overall water quality in river basins of Illinois and Texas. Assessing the potential impacts of urbanization on resource conditions is increasingly important in determining the management of forested lands because such lands are often juxtaposed with urban areas, especially in the eastern U.S.

Regulation of stream water nutrient concentrations by external and internal processes has received much attention. Studies of in-stream processes have focused primarily on nitrate and phosphorus depletion or retention in headwater streams. Swank and Caskey (1982) determined nitrate depletion in a typical southern Appalachian headwater stream following clear-cutting. In their study, denitrifying enzymes in sediments caused the loss of an estimated $1.7\,\mathrm{kg\,N\,yr^{-1}}$ from the watershed by converting nitrate to nitrite. Similarly, Mulholland (1992) found that in an eastern Tennessee stream, in-stream immobilization of inorganic N as a result of microbial and algal uptake resulted in declines in concentration of that element with distance downstream. However, Mulholland *et al.* (1995) demonstrated experimentally that increased in-stream nutrient cycling may offset some longitudinal changes in nutrient concentrations downstream. Peterson *et al.* (2001) demonstrated that despite low ammonium concentrations in stream water, nitrification rates were high and ammonium removal took place along shorter stream distances than did nitrate removal across a variety of biomes. They report that some of the ammonium and nitrate becomes temporarily sorbed onto biofilms and other submerged surfaces, but that release of inorganic nitrogen from the stream bottom can offset effects of N removal to some degree. Terrestrial controls on stream water nutrient concentrations have also been examined. For example, nitrogen uptake and denitrification in riparian zones of forests can reduce NO_3^- concentrations in drainage water entering streams (Groffman *et al.*, 1996; Hill, 1996). In addition, research that has often focused on upland sources of nutrients in agricultural landscapes has shown that soils and riparian vegetation serve as nutrient sinks, thereby buffering streams from upland perturbation (Lowrance *et al.*, 1984; Peterjohn and Correll, 1984).

In the southern Appalachians, the headwaters of major streams and rivers are often occupied by National Forest lands. Indeed, the protection of headwaters of navigable waterways was a basic premise for the establishment of the National Forest system. In this setting, streams drain minimally disturbed watersheds and enter more developed landscapes where water quality can be reduced as a cumulative result of both point and non-point inputs from sedimentation, agricultural runoff, and urban development. In many cases the opposite occurs; streams originate in urban or suburban settings and flow into undisturbed forested landscapes. The objective of this study was to examine (1) variation in water quality among land use types, and (2) the influence of a forested landscape on the quality of water in a stream originating on an urban landscape.

2. Methods

2.1. SITE DESCRIPTION

The study was conducted in the Blue Ridge Physiographic Province of western North Carolina (35°6′ N, 83°6′ W). The region receives approximately 2000 mm of precipitation annually, with less than 10% of this falling as snow or ice. Mean annual temperature is 13 °C. Elevation ranges from approximately 880 m at the lower sampling site to 1050 m at the upper site.

The study area is located approximately 40 km south of the city of Sylva, North Carolina along HWY 107 in Jackson County and lies within the upper Chattooga River watershed. This portion of the upper Chattooga River watershed (East Fork of the Chattooga River) is made up of a mixture of urban, rural, and forested landscapes and is approximately 1500 ha. The largest population center within the watershed is the town of Cashiers, North Carolina, which occupies its extreme northern portion. A municipal sewage treatment facility, which utilizes an aerobic biological treatment method, chlorination (trichloro-s-triazinetrione), and de-chlorination (sodium sulfite), treats sewage from the town. Treated effluent is discharged into Cashiers Creek, a major headwater tributary of the East Fork of the Chattooga River. Within 1 km of the treatment facility Cashiers Creek enters the Nantahala National Forest, from which it exits as the East Fork of the Chattooga River. Several small tributaries flow into the river. Our approach was to assess the condition of (1) Cashiers Creek (Urban:URB) near where it enters National Forest, (2) the Chattooga River near where it exits the National Forest (Forest:FOR), and (3) a small tributary, which lies entirely within National Forests and drains an area of the watershed below Devil's Courthouse and Whiteside Mountain. This tributary is regarded as an undisturbed stream (Reference:REF). Baseflow was taken to be 25% or less of maximum discharge and storm flow 75% or greater than maximum discharge. Streams draining the two sub-watersheds were not sampled for chemistry, bacteria, or TSS; however, because they were undisturbed, we assumed that water quality parameters were comparable to the reference stream. The URB stream reflected the cumulative influences of housing developments, water impoundments, stormwater runoff, roads, and the waste-water treatment facility. The FOR stream travels approximately 1.6 km within the Nantahala National Forest before reaching the downstream sampling site.

2.2. WATER QUALITY PARAMETERS AND METHODS

Stream water samples were collected from March 2002 through June 2003. Automated stream water pumping samplers ([1]American Sigma, Norwalk, CN) were

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[1] The use of trade or firm names in this publication is for reader information and does not imply endorsement by the U.S. Department of Agriculture of any product or service.

installed at each site to provide periodic stream water samples. Stream depth was measured weekly and was combined with data obtained from periodic surveys of channel cross-sections and steam velocity to calculate discharge. The stream water samplers were visited weekly for collection of water samples and to download stream depth data. The sampler can collect a maximum of 24 1-L samples over a 1-wk period. Samples taken during baseflow were composited, and those taken during storms remained discrete. Stream water samples were analyzed for NH_4^+, PO_4^+, and SiO_2 on a Perstorp Model 3590 Autoanalyzer (Wilsonville, OR), K, Na, Ca, and Mg on a Perkin Elmer Model 300 Atomic Absorption Spectrophotometer (Shelton, CN), and SO_4, NO_3^-, and Cl^- on a Dionex Model 4500i Ion Chromatograph (Sunnydale, CA). We also calculated nutrient to Cl^- ratios to characterize biological uptake and retention potentials between the URB and FOR sites for NO_3^-, NH_4^+, and PO_4^+. Because Cl^- is a conservative tracer, these ratios may correct for the effects of dilution from surface and subsurface water sources between the URB and FOR sampling sites. Conductivity and pH were measured using digital conductivity and pH meters (Orion Models 122 and 611, respectively). Total suspended solids (TSS) was determined using a vacuum filtration system with 1.5 micron glass microfiber filters (Whatman, Clifton, NJ). All analyses were conducted at the Coweeta Hydrologic Laboratory.

Monthly grab samples were taken at each site for determination of fecal and total coliform, and fecal Streptococcus population densities. A few grab samples were taken during high stream flows, as well; however, because the sites were remote from our headquarters only two storms were sampled. Standard filtration methods (Millipore 1986) were used in the analysis of stream bacteria. Pre-sterilized HA-type (0.45 um pore size) membrane filters to collect Streptococcus and total coliform, and HC-type filters (0.7 um pore size) were used to collect fecal coliform. Pre-prepared commercial media were used for growth media. Dilutions were conducted using 99 ml commercially pre-loaded dilution bottles.

Fecal coliform:fecal Streptococcus ratios (FC:FS) have been used to differentiate among contamination from human (>4.0), domestic animal (0.1–0.6), and wild animal (<0.1) sources (Geldreich, 1976; Howell *et al.*, 1995). Several criteria need to be met for accurate source identification using this ratio: stream travel time of <24 h, >100 FS counts per 100 ml, and sample pH between 4 and 9 (Geldreich, 1976). During the course of the study, FS fluctuated around 100 for the FOR site and was consistently <100 at the REF site. In contrast, FS was consistently >100 at the URB site. Stream travel time between the URB and FOR sites was estimated to be less than 24 h, and pH was consistently within the range specified.

2.3. DATA ANALYSIS

Data were expressed on a monthly basis for purposes of comparing base and storm flow, and on a weekly basis for examining seasonal differences in water chemistry and TSS among the URB, FOR, and REF stream sites. PROC GLM (SAS Inst.,

1994) was used in multiple comparisons of season and site, and Duncan's multiple range test (SAS Inst., 1994) was used to separate means. Values measured near peak stream flow were used for analysis of stormflows. Simple linear regression (PROC REG, SAS Inst., 1987) was used to examine the relationship between solute concentration and stream discharge for each site. Differences among the slopes of the regression lines were determined using PROC GLM (SAS Inst., 1987) with the appropriate interaction terms. Differences among slopes were interpreted as differences among rates of solute concentration response to changes in discharge. Statistical significance was evaluated at the $\alpha = 0.05$ level.

3. Results and Discussion

3.1. Silicates, conductivity, and pH

Over the study period, average concentrations of solutes other than SiO_2 were consistently higher at the URB stream site than at the FOR and REF sites (Figure 1), and patterns of site-to-site variation in solute concentrations were consistent from season to season (Figure 2). SiO_2 concentration is influenced by a combination of factors. These include use of SiO_2 by diatoms in the construction of frustules; groundwater residence time, which when long can permit SiO_2 to accumulate before entering the stream; and substrate mineral content within the catchment. Conductivity was 3 to 4 times greater at the URB site as at the REF site during all seasons and was near double that at the FOR site (Figure 2). Conductivity was intermediate at the FOR site because of reductions in ionic concentrations in the stream water due to dilution and in-stream processing between the URB and FOR sites. Chloride concentrations ranged from <1.0 ppm at the REF site to near 7 ppm at the URB site at baseflow (Table I). The higher concentration in the URB and FOR streams may be due to the use of chlorine in the treatment of municipal sewage and its subsequent release into the stream above the URB sample site; however, a sulfur compound (Na_2SO_3) is used as a de-chlorinator and oxidizes chlorine to form chloride in the treated effluent before release. The pH level was consistently higher at the URB site than at the other two sites, both seasonally and for the study period (Table I; Figure 3).

3.2. Total suspended solids

TSS at the REF site during stormflow was roughly equivalent to baseflow TSS at the URB site (Table I). There was an approximately 3-fold increase in TSS from base- to stormflow at the REF site, a 4-fold increase at the FOR site, and a 5-fold increase at the URB site. The magnitude of the increase in stormflow sediment at the URB site helps explain the differences among the slopes of the rising limbs of the hydrographs for the three sites (Figure 4). At the URB site, sediment transport and

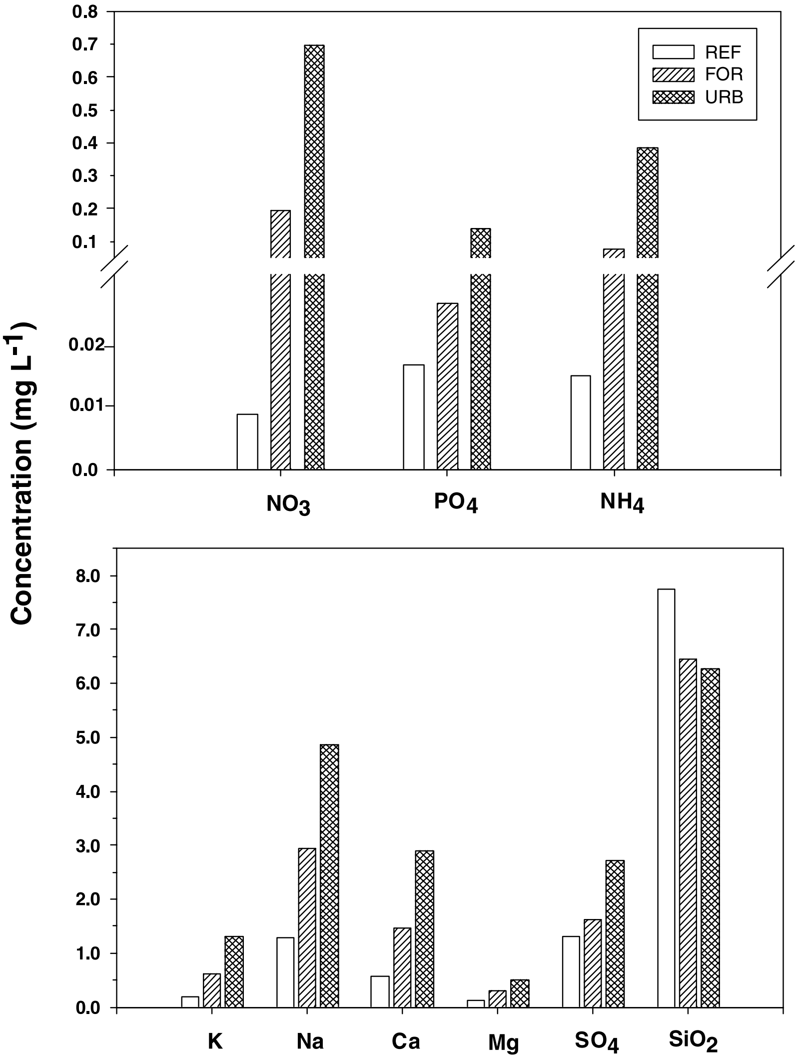

Figure 1. Means of solute concentrations over the study period by site.

delivery were more sensitive to increases in the magnitude of the storm. This was most likely related to the presence of a higher percentage of impervious surfaces that increase stormflow, and to land disturbances that increase sediment loading.

3.3. STREAM DISCHARGE AND WATER QUALITY

Average base flows for the three stream sites, calculated as twenty-five percent or less of maximum flow, were URB = 184 L s^{-1}, FOR = 420 L s^{-1}, and

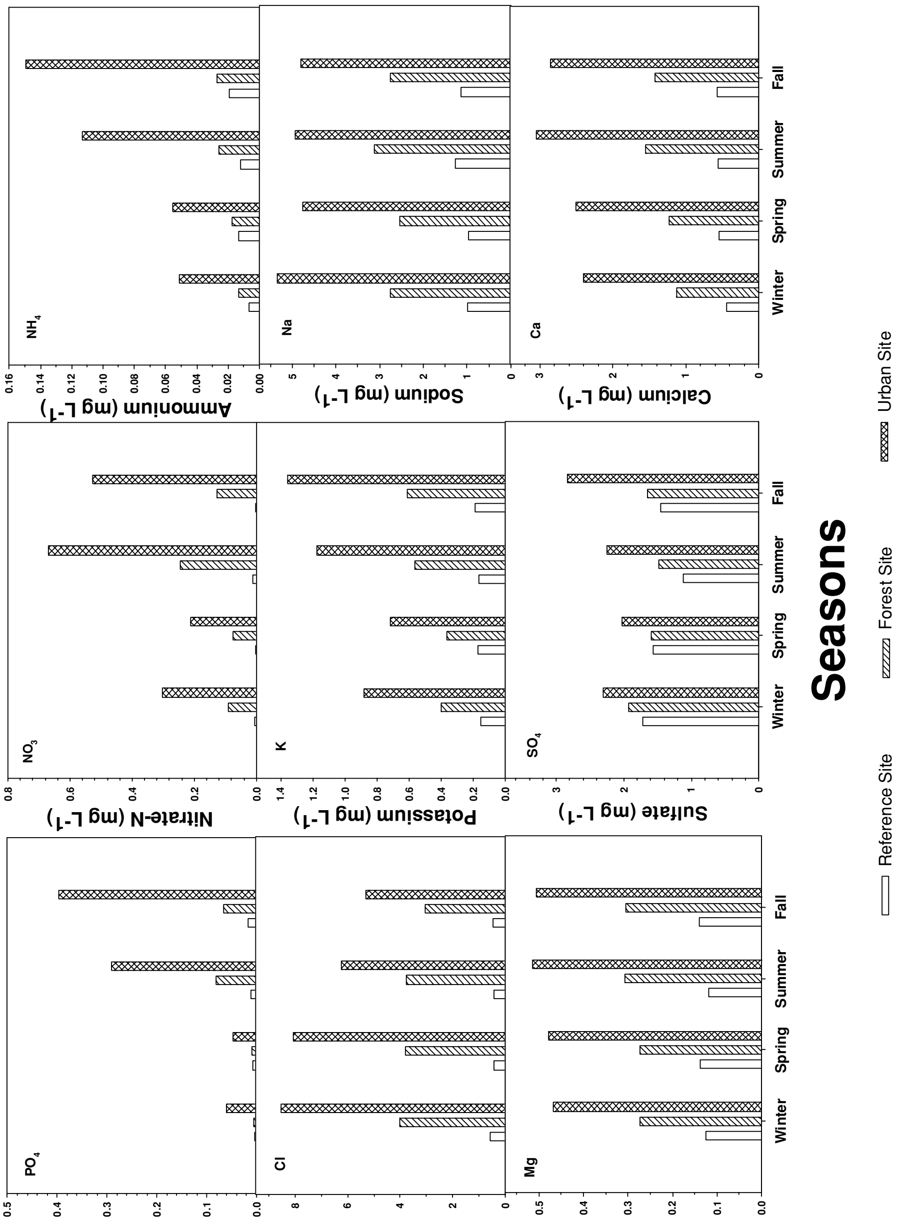

Figure 2. Mean concentration by season and site for major cations and anions in stream water.

TABLE I

Means for stream discharge, pH, solute concentrations, and TSS for base and storm flow by site

Parameter		REF	URB	FOR
Discharge L s^{-1}	Base	17 (<1–48)	182 (<1–523)	418 (52–622)
	Storm	121 (27–161)	1088 (658–1885)	1721 (1383–2352)
pH	Base	6.3 (5.8–6.8)	6.8 (6.5–7.2)	6.6 (6.3–6.9)
	Storm	5.69 (5.2–5.9)	6.68 (6.5–6.8)	6.35 (6.1–6.6)
No$_3^-$ mg L^{-1}	Base	0.007 (0–0.03)	0.60 (0.12–2.18)	0.20 (0.02–0.74)
	Storm	0.001 (0–0.004)	0.10 (0–0.17)	0.059 (0.03–0.12)
NH$_4^+$ mg L^{-1}	Base	0.01 (0–0.03)	0.109 (0.02–0.62)	0.018 (0–0.04)
	Storm	0.06 (0–0.03)	0.046 (0.03–0.08)	0.024 (0–0.04)
PO$_4^+$ mg L^{-1}	Base	0.008 (0–0.03)	0.284 (0.01–1.39)	0.054 (0–0.30)
	Storm	0.003 (0–0.01)	0.022 (0–0.05)	0.010 (0–0.06)
Cl$^-$ mg L^{-1}	Base	0.558 (0.4–2.5)	7.01 (3.5–16.3)	3.98 (2.4–7.5)
	Storm	0.415 (0.3–0.5)	4.91 (2.6–9.1)	3.48 (1.8–7.5)
K mg L^{-1}	Base	0.17 (0.07–0.45)	1.12 (0.33–2.88)	0.67 (0.16–1.06)
	Storm	0.16 (0.07–0.25)	0.92 (0.37–1.31)	0.45 (0.30–0.76)
Na mg L^{-1}	Base	1.21 (0.53–2.65)	5.21 (2.10–10.77)	3.36 (1.71–5.00)
	Storm	0.78 (0.54–1.16)	3.38 (1.85–4.65)	2.57 (1.65–4.98)
Ca mg L^{-1}	Base	0.54 (0.31–1.26)	2.79 (1.82–4.70)	1.57 (1.03–2.24)
	Storm	0.51 (0.38–1.45)	2.49 (1.21–2.83)	1.29 (0.88–2.51)
Mg mg L^{-1}	Base	0.12 (0.08–0.29)	0.50 (0.34–0.83)	0.31 (0.23–0.38)
	Storm	0.14 (0.10–0.23)	0.46 (0.27–0.50)	0.29 (0.23–0.46)
TSS ppm	Base	2.84 (<0.1–31.4)	11.85 (1.5–52.7)	6.42 (1.6–21.6)
	Storm	11.66 (3.2–30.2)	47.8 (12.0–82.9)	41.73 (8.3–205.5)

Values in parentheses represent the observed range.

REF = 17 L s^{-1}. Perennial streams from two undisturbed subwatersheds located between the URB and REF sites contributed an additional 206 L s^{-1} of baseflow to the FOR stream. Summing baseflow quantities from all sources (e.g., URB + REF + the two other streams) accounted for nearly all the flow at the FOR site. Stream discharge response to rainfall varied among sample sites. The hydrograph in Figure 4 illustrates differences in stream discharge response time among sites for a typical storm in June 2002. Peak discharge occurred very near the same time at all sites; however, the slopes of the rising limbs of the hydrographs show that flow increased at a different rate at each site. The falling limbs of the REF and FOR stream hydrographs indicate a typical pattern of a post-storm decrease in stream discharge. In contrast, the response pattern observed on the URB stream appears to be the result of an altered hydrologic regime caused in part by the influence of a large reservoir within 2 km upstream of the sample site. Following storms, the steady release of storm water from the reservoir resulted in a slow decrease

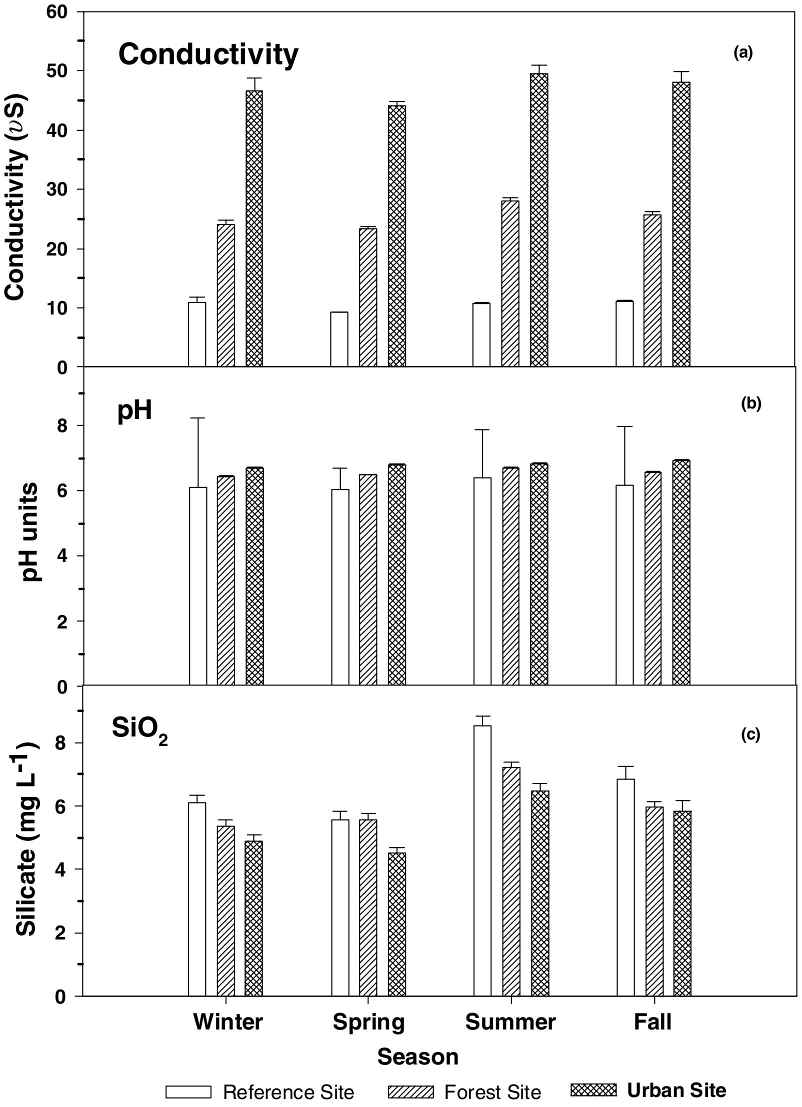

Figure 3. Means for conductivity (a), pH (b), and silicates (c) by season and site. Season was defined as: winter ($n = 18$), Nov. 15–Mar. 15; spring ($n = 18$), Mar. 16–May. 31; summer ($n = 11$), June 1–Aug. 31; fall ($n = 11$), Sept. 1–Nov. 14.

in discharge and may have masked effects of direct storm runoff on water volume and TSS. Similarly, TSS remained higher than pre-storm values for a longer period at the URB site than at the other sites. In contrast, fluvial processes (in-channel translocation of sediments) may be the primary drivers of TSS at the FOR site, rather than near-stream sediment inputs from land disturbance. It is also likely that

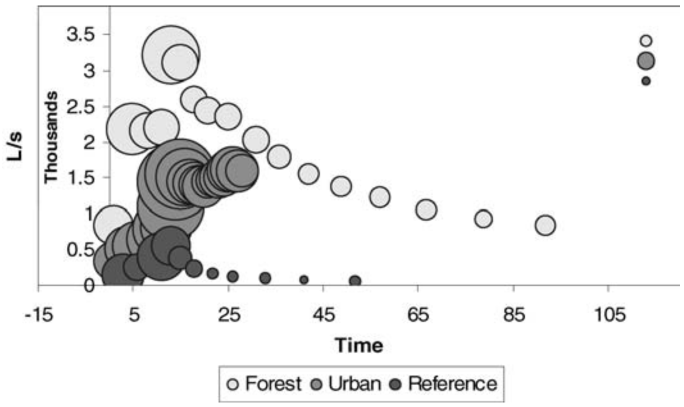

Figure 4. Hydrograph representing changes in stream discharge (Liters sec^{-1}) over time for a selected June 2002 storm for the three study sites. Also known as a sedigraph where the circles represent relative amounts of total suspended sediment (TSS). The three circles in the top right-hand corner represent relative TSS at baseflow for each site.

inputs of finer sediments originating farther up in the watershed contributed some TSS to the FOR site.

In many unimpaired streams, TSS typically begins to decrease before peak discharge and continues to decrease as the storm recedes (Glysson, 1987; Webster *et al.*, 1990; Burke and MacDonald, 1999; Riedel *et al.*, 2004). The REF stream responded in this manner but the URB and FOR streams did not. At these two sites, TSS reached a maximum at peak discharge, but TSS in the FOR stream decreased along the falling limb of the hydrograph. TSS in the URB stream decreased soon after peak discharge but remained high as discharge leveled off.

Single-variable regression models for NO_3^-, NH_4^+, PO_4^+, and Cl^- vs. stream discharge indicate that stream chemistry responses to variation in stream discharge were greatest at the URB site (Table II). For NO_3^-, NH_4^+, PO_4^+, and Cl^-, slopes of the regressions were significantly different from the FOR site (Table II). This suggests that although baseflow and stormflow concentrations of most solutes were greatest at the URB site, the dilution effects of increased discharge on chemical constituents were greater at the URB site than at the FOR site, where there was a steeper rate of decrease in solute concentrations per unit increase in discharge. TSS increased with increasing stream flow at all sites and showed the sharpest increase at the REF site.

3.4. BASE CATIONS AND OTHER CONSTITUENTS

Concentrations of base cations (K, Na, Ca, Mg) were highest at the URB site both seasonally (Figure 2) and for the study period (Figure 1), and this contributed to higher conductivity at that site. At the URB site, Mg concentrations were on the

TABLE II

Parameter estimates for single-variable regressions of responses to stream discharge for the URB, FOR, and REF sites

Variable	Site	Intercept	Slope	Adj R-sq	*F*	*P*
TSS	FOR	−1.931	0.617^{a}	0.16	12.23	0.0009
	REF	1.935	1.581^{ab}	0.13	8.13	0.0061
	URB	5.879	1.012^{b}	0.48	39.88	<0.0001
NO_3^-	FOR	0.248	-0.004^{a}	0.22	18.72	<0.0001
	REF	0.0079	-0.002^{ab}	0.16	10.97	0.0016
	URB	0.743	-0.019^{b}	0.26	15.30	0.0003
NH_4^+	FOR			n.s.		
	REF			n.s.		
	URB	0.128	-0.003^{b}	0.12	5.76	0.021
PO_4^+	FOR	0.0727	-0.00014^{a}	0.18	14.82	0.0003
	REF	0.0096	-0.0022^{ab}	0.07	6.33	0.0421
	URB	0.357	-0.010^{b}	0.19	10.37	0.0024
Cl^-	FOR			n.s.		
	REF			n.s.		
	URB	7.353	-0.059^{b}	0.11	5.5	0.0237

Values for the slope parameter with the same superscript are not statistically different within measured parameter. *F*- and *P*-statistics correspond to the individual site regressions. n.s. represents non-significance at $\alpha = 0.05$.

average more than double the North Carolina Department of Environment and Natural Resources (NCDENR) standard for freshwater. At the REF site, concentrations of cations other than Na were lower than those reported by Swank (1988) for small undisturbed headwater streams in the southern Appalachian region. Swank reported Mg, K, and Ca concentrations 2 to 3 times higher than those we observed. Sodium concentrations at the REF site were 10 percent higher than those reported by Swank. Base cation concentrations at the FOR site were higher than those at the REF site but lower than those at the URB site. Sodium was notably higher at both the URB and FOR sites than at the REF site, probably as a result of the addition of sodium sulfite (Na_2SO_3) to the treated effluent as a de-chlorinator before that effluent was released into Cashiers Creek.

3.5. NITROGEN AND PHOSPHORUS

Stream water nitrate concentration is often used as an index of water quality because of its sensitivity to disturbance. It is highly mobile, and regulated by a variety of biological controls (Swank, 1988). In this study, NO_3^- concentrations were higher and more variable at the URB site during the growing season than at the FOR and REF sites (Figure 5). In addition, concentration increases toward the end of the

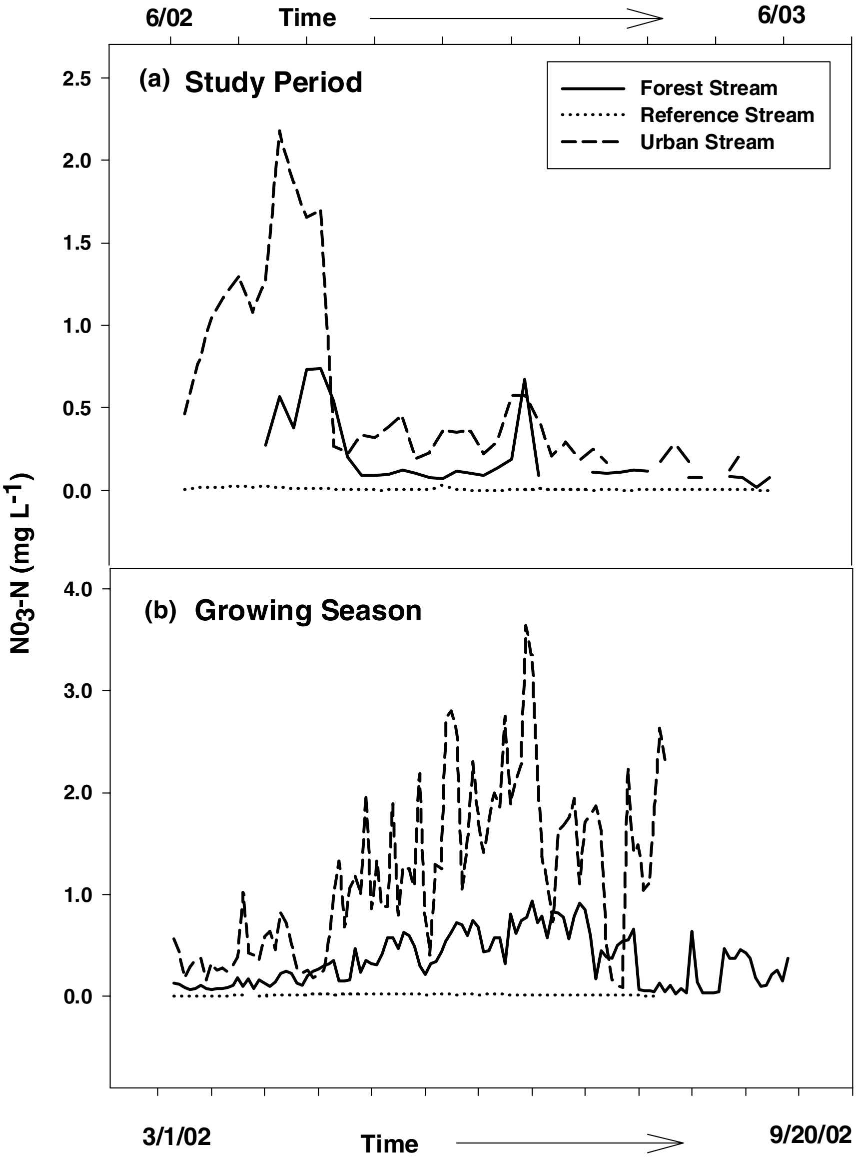

Figure 5. Annual (a) and seasonal (b) variation in stream water nitrate nitrogen concentration for each site. Points along graph in (a) are monthly means by site. Points along graph in (b) are means of weekly composited samples for both base and storm flow.

growing season at the URB, but not at the FOR site. The increase in concentration at the URB site may be a result of inputs from non-point sources such as septic drain-fields in the urban and non-urban portions of the watershed, and possibly fluctuating discharge from the local waste water treatment plant. The lack of an increase at the FOR site is likely due to retention or depletion along the stream reach. Ammonium was higher at the URB site, as well, and likely for the same reasons. Ammonium is produced during the decomposition of organic matter. Where dissolved oxygen is not limiting, ammonium is quickly nitrified to form nitrate, which undergoes denitrification to form nitrite and nitrous oxide gas. There is evidence in this study for substantial reductions in nitrate, ammonium, and phosphorus concentrations between the URB and FOR sites.

Chloride is a biologically inert solute, and therefore useful for assessing in-stream nutrient cycling. When nitrogen or phosphorus is expressed as a ratio of chloride, dilution due to increased flows is taken into account; hence, the change in the ratio reflects real change in the solute concentration and suggests the presence of various mechanisms for nutrient retention. For the study period, ratios of nitrate, ammonium, and phosphorus to chloride decreased from the URB site to the FOR site during base flow by 43, 71, and 66%, respectively, suggesting that there was substantial retention or removal of these solutes between the two sites over the study period (Table III). However, seasonal values varied substantially. Reductions in nitrate occurred during the summer months (−52%) and in the fall (−51%) and in the winter (−25%) but no measurable change was found during the spring (Table III). Ammonium uptake remained similar throughout the year but began to decline in the fall. Although ammonium uptake has been shown to be high in the fall due to fresh organic inputs from litter fall (Tank *et al.*, 2000; Webster *et al.*, 2003), declining stream water temperatures as fall progressed may have reduced uptake (Tank *et al.*, 2000). Phosphorous retention was high during the winter and spring, and remained high relative to nitrate retention during the summer and fall (Table III). Webster *et al.* (2003) also reported greater retention or removal of phosphorous and ammonium than of nitrate in a wide variety of stream ecosystems; however, most streams in their study had greater surface:volume ratios than did our FOR site. The effect of low surface:volume ratios is to mask the apparent significance of detrital dynamics that serve as the source of much of the ammonium in stream water. Swank and Caskey (1982) measured denitrification in sediments of a stream

TABLE III

Percent change in the ratios of nitrate, ammonium, and phosphate to chloride from the URB site to the FOR site by season and study period

	Study period	Winter	Spring	Summer	Fall
		Change in ratios			
Nitrate	−43	−25	∼0	−52	−51
Ammonium	−71	−63	−63	−75	−72
Phosphate	−66	−88	−88	−65	−63

Season was defined as; winter ($n = 18$), Nov. 15–Mar. 15; spring ($n = 18$), Mar. 16–May 31; summer ($n = 11$), June 1–Aug. 31; fall ($n = 11$), Sept. 1–Nov. 14.

draining a 4-yr-old clearcut. They attributed nitrogen loss to denitrification, but suggested that N removal by algae and heterotrophic bacteria likely had occurred as well. Similarly, Mulholland and Hill (1997) found that in-stream processes were important determinants of stream water nitrate and ammonium concentrations and explained much of the strong seasonality they observed.

Nitrate, NH_4^+, and PO_4^+ concentrations were all at least twice as great at the URB site as they were at the FOR site during all seasons. NH_4^+ and PO_4^+ concentrations were highest during the fall, but these fall concentrations were significantly higher than those during other seasons only at the REF site. Tank *et al.* (2000) found that NH_4^+ uptake was greatest in the fall, but our data suggest that in this disturbed stream ecosystem uptake peaked in the spring and was lowest in the fall. Nevertheless, combined NO_3^- removal and retention was greatest in the fall (Table III). Nitrogen retention and depletion in streams are the result of heterotrophic and autotrophic activity, particularly in sediments and during the fall when organic matter inputs and biological activity and demand for nutrients are the greatest (Tank *et al.*, 2000).

3.6. BACTERIA AND STREAM TEMPERATURE

Average stream temperature was generally lower at the REF site than at the URB or FOR sites, and during the summer was lower by more than 2 °C (Figure 6). There were no site-to-site differences in stream water temperature during the winter months. Bacteria population response to seasonal variation in stream water temperature varied considerably among sites (Figure 7). These responses were greatest at the URB site and lowest at the REF site, but populations generally began to increase substantially at approximately 15 °C. McSwain (1977) reported significant declines in total coliform (TC) in the fall when stream water temperature fell below 11 °C in a southern Appalachian headwater stream. At the REF site, fecal coliform (FC) and fecal Streptococcus (FS) showed very little response to increasing stream water temperature, and TC showed only a slight increase. All bacteria types were equally responsive to temperature at the FOR site, but they were considerably less responsive than the URB site in terms of both response to temperature and overall population densities (Figure 7). McSwain (1977) found increases in FC, FS, and TC during the late summer and early fall. These increases coincided with leaf fall and higher stream temperatures. In his study, seasonal variation in bacterial counts were slightly less than those observed at the REF site and considerably less than those at the FOR or URB sites. In our study populations densities varied considerably by season (Figure 6); moreover, FS was undetectable at the REF site during both winter and spring. Population densities of FC and FS at the URB site were significantly greater than those at the REF site but were not significantly different from those at the FOR site. TC had significantly higher population densities at the URB site than at either of the other two sites (Figure 8). These observed differences in stream water bacterial populations between the URB and FOR sites are partially explained by dilution effects; however, declines in populations can also be the result

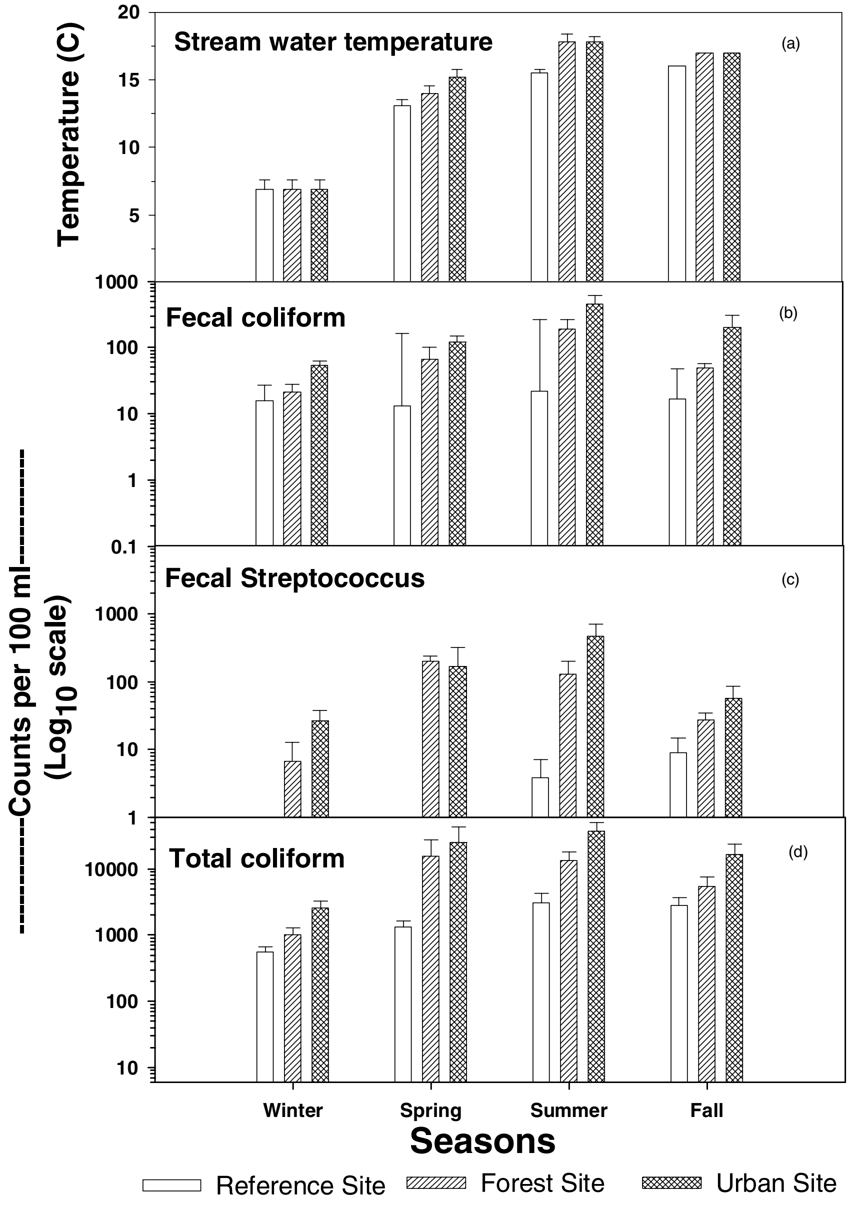

Figure 6. Log transformed fecal coliform, fecal Streptococcus, and total coliform, expressed as colony counts per 100 ml, and stream water temperature (°C) by season of year. Season was defined as: winter ($n = 18$), Nov. 15–Mar. 15; spring ($n = 18$), Mar. 16–May 31; summer ($n = 11$), June 1–Aug. 31; fall ($n = 11$), Sept. 1–Nov. 14. Stream temperature values are based on monthly measurements taken during bacteria sampling.

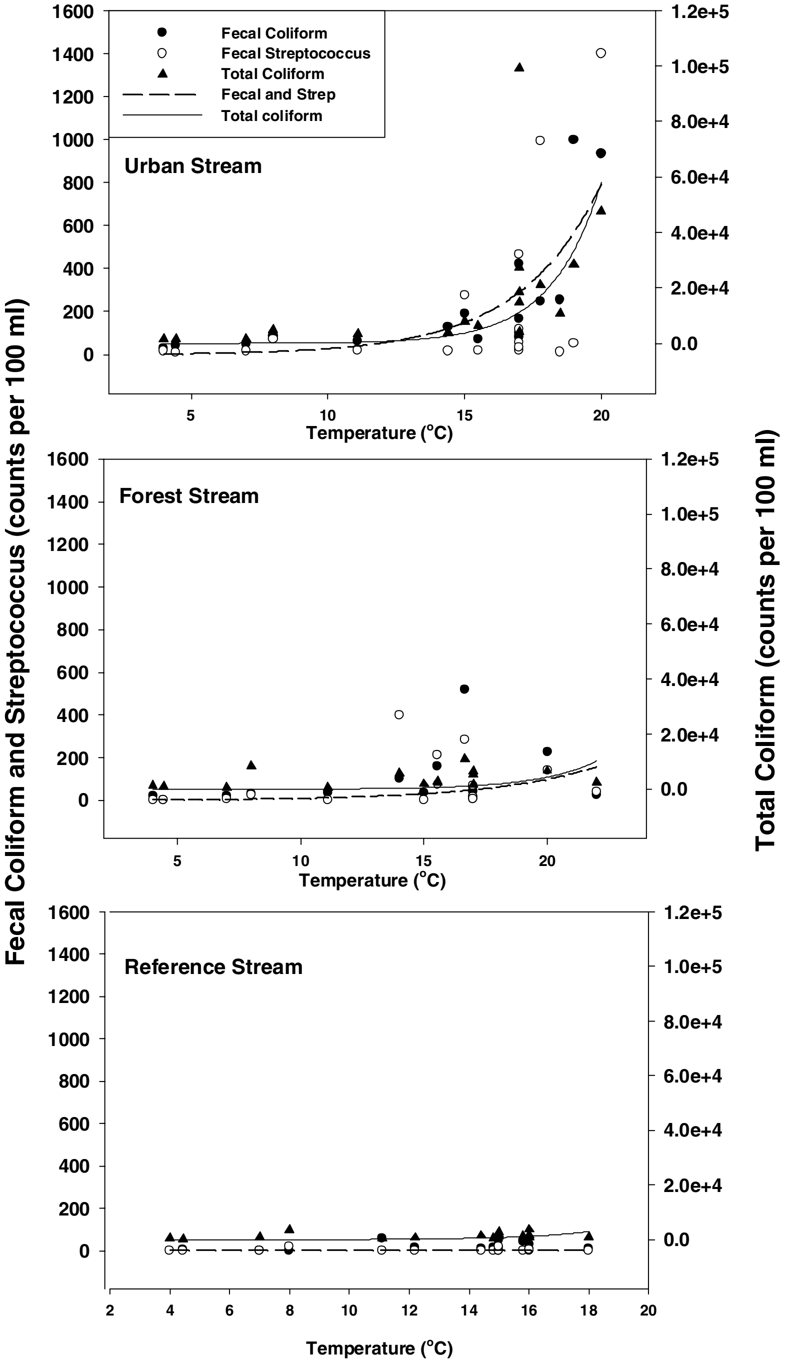

Figure 7. Bacteria responses to increasing stream water temperature by site. Solid line is the response curve ($y = e^{ax}$) for total coliform. The dashed line is the response curve ($y = e^{ax}$) for fecal coliform and fecal Streptococcus, combined.

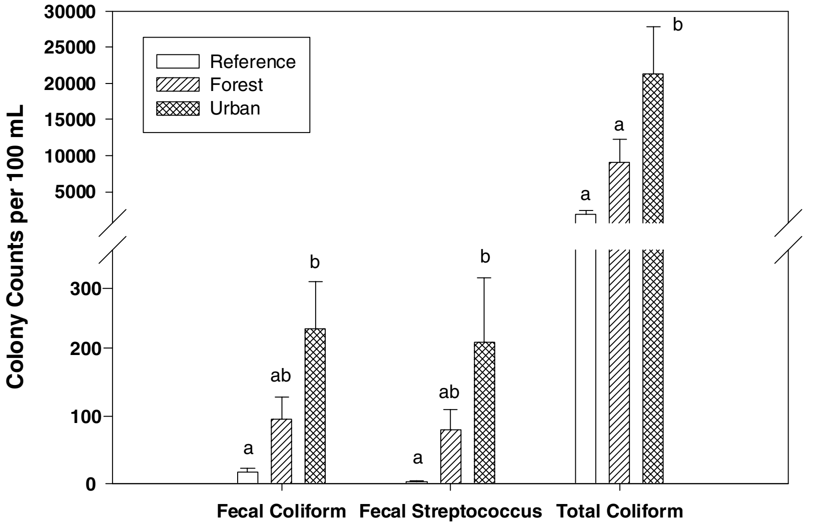

Figure 8. Means of bacteria colony counts (per 100 mL) by site. Bars with the same letter within bacteria type are not significantly different. Error bars represent one standard error of the mean. Means and standard errors are based on sample sizes ranging from 14–18 for each bacteria type.

of in-stream competition, predation, and resource limitation (Janakiraman and Leff, 1999).

The US Environmental Protection Agency National Watershed Database 305(b) report (US Environmental Protection Agency, 1999) ranks FC bacteria pollution as the most widespread pollution problem in the nation's rivers and streams. Non-point sources of fecal contamination that contribute to pollution are often difficult to identify, but human health risks are greater when FC is principally from human sources (Sinton *et al.*, 1993). Fecal coliform:fecal streptococcus ratios have been used to differentiate between contamination from human (>4.0), domestic animal (0.1–0.6), and wild animal (<0.1) sources (Geldreich, 1976; Howell *et al.*, 1995). Doran and Linn (1979) indicate that the FC:FS ratio is useful in distinguishing between domestic animal and wild animal sources, but the usefulness of FC:FS in differentiating between human and nonhuman sources is questionable. Nonetheless, a ratio of 5.28 at the URB site during baseflow suggests the presence of human sources of contamination. During stormflow, however, that ratio decreased to 0.47, which is well below the human contamination threshold value. Bolstad and Swank (1997), in a study of cumulative effects of land use with varying distance downstream, reported FC:FS values of 0.65 and 0.49 for baseflow and stormflow, respectively, at the sampling station furthest downstream in a southern Appalachian stream. The stormflow value for FC:FS at the URB site is similar to the value reported by Bolstad and Swank, but baseflow FC:FS values at the URB site were 8 times higher than those reported by Bolstad and Swank (1997). This could be explained by the proximity of the URB sampling site to the city of Cashiers and the influence of septic systems. Where stream water originating from storm runoff is relatively low in bacteria, point and non-point sources of bacteria, such as septic drain fields in the urban center, have a greater influence on the concentrations of bacterial populations during baseflow than during storms. The FOR site had an FC:FS value of 3.94 during baseflow, and that ratio was reduced to 0.68 during storms. The REF site had the lowest baseflow FC:FS value (3.0) but the highest value for storms (0.83). It is important to note that during the study period only 2 storms were sampled for bacteria due to the remoteness of the study sites. More samples would be required to accurately characterize bacteria populations. Therefore, we suggest caution in interpreting the data beyond a relative comparison among sites.

4. Comparisons with Published Standards

To put our results in context, we compared our data to a compilation of published standards and guidelines (Table IV). Chloride concentrations at baseflow ranged from <1.0 ppm at the REF site to near 8 ppm at the URB site. The addition of sodium sulfite to the treated effluent as a de-chlorinator oxidizes chlorine to form chloride, resulting in higher concentrations of that element in the stream water both at the URB and FOR sites. Still, those concentrations were well below the 230 ppm NCDENR published allowable maximum for aquatic life (Table IV). Standards for cation concentrations do exist, but with the exception of Mg at the URB site, observed concentrations are well below published allowable maximums for freshwater. Maximum Mg concentrations were 4 times greater the NCDENR allowable maximum for freshwater. The source of Mg is uncertain, but is probably a combination of point and non-point sources. Mean values ranged from 0.46 mg L^{-1} during stormflow to 0.50 mg L^{-1} during baseflow (Table I), considerably lower than the NCDENR allowable maximum.

The United States Environmental Protection Agency (US EPA, 1995) standard for drinking water is frequently considered the threshold for desirable versus undesirable water quality. Nitrate concentrations are typically well below this threshold maximum (10 mg L^{-1}; Table IV) in southern Appalachian streams. However, this published EPA standard is useful as a standard reference when one is comparing water quality at different points along streams or in different watersheds for the purpose of assessing the effects of upland disturbance or urbanization. It is important to note that although NO_3^- concentrations were highest at the URB site, these values were well below the EPA allowable maximum for drinking water (10 mg L^{-1}) (Table IV). Moreover, neither of the stream sites in this study are direct sources of drinking water.

The USEPA standard for fecal coliform in stream water, including all surface water, is applicable to "primary contact waters." Primary contact waters are defined as all surface freshwater where human contact during recreation or other uses

TABLE IV

Published standards for parameters measured in the study for which published standards exist

Parameters	Current study maximum	Freshwater CCC[†]	Aquatic life	Human health[****]	Water supply[***]	High Quality Waters (HQW)[c]	Source of standard
Chloride mg L^{-1}	16.3		230		250		NC[*]
Nitrate mg L^{-1}	6.8				10 MCL[‡]		NC
pH	5.8–7.2		6.0–9.0				NC
TSS mg L^{-1}	Baseflow – 53					10 (Tr & PNA)[b]	NC
	Stormflow – 83						NC
Calcium mg L^{-1}	4.7	7.3					ECOTOX[**]
Magnesium mg L^{-1}	0.8	0.2					NC
Potassium mg L^{-1}	2.9	30					NC
Sodium mg L^{-1}	10.8	400					NC 0208
Sulfates mg L^{-1}	6.3				250		NC
Fecal coliform counts/100 ml	1400			200 (N)[a]			USEPA
Total coliform counts/100 ml	99200				50[£] (N)		NC

Included are the observed maxima for the current study, all of which were observed at the URB site. The standards, criteria, or toxic concentrations are either from 15A NCAC 2B or are National Criteria as per USEPA. For a more complete listing, go to http://h2o.enr.state.nc.us/csu/.

[*] North Carolina 2B Standard.

[**] USEPA ECOTOXicology Database System.

[***] Water supply standards are applicable to all Water Supply Classifications. Standard is based on the consumption of fish and water.

[****] Standard for Primary Contact Waters.

[†] Chronic Criterion Concentration.

[‡] Maximum Contaminant Level used in drinking water and groundwater.

[a] Narrative description of limits or additional narrative language (See NC 2B).

[£] Applies only to unfiltered Water Supplies.

[b] Tr = Trout Waters as defined by 2B.0101 and.0301; PNA = Primary Nursery Areas.

[c] HQW = High Quality Waters – see 02B.0101 and 02B.0201.

could occur. The allowable maximum of 200 colony counts per 100 ml (Table III) is based on the mean of a minimum of 5 samples over a 30-day period. Fecal coliform counts often exceeded 200 at the URB site, but those counts were based on monthly samples. Although values for fecal coliform at the URB site were high throughout the study period, and the absolute maximum value observed well exceeds the standard (Table IV), it is unknown whether those values would have exceeded the standard had we applied the sampling criteria stated above.

5. Role of Undisturbed Stream Reaches

There have been numerous studies of the role of undisturbed headwater streams or stream reaches in improving stream water quality. Many have demonstrated the role of near-stream or riparian vegetation in mitigating upland sources of nutrients. For example, forested watersheds conserve nutrients through biological and geochemical processes that retain N and P in upper soil horizons (Wood *et al.*, 1984; Qualls *et al.*, 1991). Riparian vegetation also plays an important role in the regulation of nutrient fluxes through incorporation and storage and through the filtering of sediment and other material released during upslope disturbances, particularly from agricultural activities (Lowrance *et al.*, 1984; Peterjohn and Correll, 1984). However, in-stream processes have been shown to further reduce transport of nutrients, particularly inorganic forms of nitrogen. Net transformations of nutrients from inorganic to organic or particulate form are key mechanisms for nutrient retention in streams (Meyer and Likens, 1979). It has also been shown that prolonged periods of in-stream nutrient retention in undisturbed headwater streams or stream reaches are often punctuated by nutrient losses during storms (Meyer and Likens, 1979; Grimm, 1987). Transient storage, the routing of water along flow paths moving much more slowly than the average in-channel stream velocity (e.g., in pools and low-gradient stream sections), creates zones in which stream metabolism and storage within the channel bed (Grimm and Fisher, 1984; Fellows *et al.*, 2001) increase a variety of in-stream biogeochemical processes (Baker *et al.*, 2000). During high flows these pools of transformed and stored nutrients are flushed out to be taken up or stored in locations farther downstream.

In this study, concentrations of most stream solutes were higher at the URB site than at the FOR and REF sites during base and stormflow. Lower concentrations at the FOR and REF sites probably resulted from a combination of dilution from stream water draining less disturbed upland areas and in-stream processing. This study suggests that undisturbed stream reaches are effective at improving water quality in streams where headwater reaches are heavily affected by urbanization or other land uses. We recognize that our ability to extrapolate these results to other streams with mixed land uses is limited by a lack of replication. However, it is extremely difficult to truly replicate large-scale studies of this nature without introducing numerous confounding factors that make inferences equally limited. Understanding the cumulative effects of mixed land uses will require approaches that combine large-scale monitoring, replicated mid or large-scale studies where possible, and detailed small scale studies and experiments. Our approach is strengthened by the fact that the patterns we observed can be explained by processes determined from small-scale experimental approaches (e.g., Mulholland *et al.* (1995)); however, monitoring of other sites will be needed to increase confidence in the generality of the patterns we observed.

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Eastern Brook Trout: Status and Threats

Produced by Trout Unlimited for the Eastern Brook Trout Joint Venture

DAVID ANDERSON

Eastern Brook Trout: Status and Threats

Background: Brook trout (*Salvelinus fontinalis*) are the only trout native to much of the eastern United States. They have inhabited the East's coldwater streams and lakes ever since the retreat of the continental glaciers across New York and New England, and they have thrived in the ancient valleys of the Appalachians for the last several million years. Arguably the most beautiful freshwater fish, brook trout survive in only the coldest and cleanest water. In fact, brook trout serve as indicators of the health of the watersheds they inhabit. Strong wild brook trout populations demonstrate that a stream or river ecosystem is healthy and that water quality is excellent. A decline in brook trout populations can serve as an early warning that the health of an entire system is at risk.

In pre-Colonial times, brook trout were present in nearly every coldwater stream and river in the eastern United States. Sensitive to changes in water quality, wild brook trout began to disappear as early agriculture, timber and textiles economies transformed the eastern landscape by stripping the region's protective forests and filling the streams with sediment and pollution. As streams gained value as highways for log drives, water sources for farming, and prime locations for factories and mills, the resulting loss in brook trout populations mirrored the broader decline in the health of the region's lands and waters.

Many of these threats to water quality and wild brook trout persist today, as our population and resource needs increasingly expand. New challenges associated with urbanization place additional stresses on the eastern landscape and its remaining brook trout habitat.

A Partnership to Conserve Brook Trout

For many years, the solution to declining brook trout populations was stocking more fish to ensure that fishing opportunities did not suffer. In recent decades, however, state and federal fisheries managers and organizations such as Trout Unlimited have focused on restoring the habitat that brook trout require for their survival. In 2004, in recognition of the need to address regional and range-wide threats to brook trout, a group of public and private entities formed the Eastern Brook Trout Joint Venture (EBTJV) to halt the decline of brook trout and restore fishable populations.

The Eastern Brook Trout Joint Venture is comprised of:

- Fish and wildlife agencies from 17 states
- Federal support from U.S. Geological Survey, U.S. Forest Service, U.S. Fish & Wildlife Service, National Park Service and Office of Surface Mining
- Conservation organizations including Association of Fish & Wildlife Agencies, Trout Unlimited, Izaak Walton League of America, Trust for Public Land and The Nature Conservancy
- Academic institutions including Conservation Management Institute at Virginia Tech and James Madison University

Members of the Eastern Brook Trout Joint Venture are deeply committed to maintaining and restoring brook trout and the watersheds upon which they depend. This summary report describes the first stage of the Joint Venture's efforts to spearhead a collaborative process to improve brook trout habitat and return one of our most beautiful gamefish to its native range.

The maps and data in this publication are based on "Distribution, Status and Perturbations to Brook Trout within the Eastern United States," a technical report by the Joint Venture's assessment team that will be published later in 2006. This first-of-its-kind assessment paints a comprehensive picture of the condition of brook trout populations across their native range from Ohio to Maine to Georgia. The technical report categorizes a variety of threats to brook trout and their habitat and helps to identify restoration and protection priorities. Using satellite imagery and statistical analysis, the report predicts the status of brook trout in areas that lack population data and identifies different levels of environmental stress that brook trout are able to tolerate before they are likely to disappear.

The technical report identifies where wild brook trout populations remain strong, where they are struggling and where they have vanished. Most importantly, it provides state and federal agencies, anglers and community leaders with the tools to identify local rivers and streams that are priorities for protection and restoration. Partners in the Eastern Brook Trout Joint Venture are using the technical report and ongoing analyses to develop a comprehensive strategy for state and the federal agencies to protect and restore brook trout on regional and range-wide scales. This will involve advancing data collection, promoting policies necessary for success, and establishing on-the-ground projects to protect and restore brook trout habitat and populations. The data included in the technical report also will serve as a baseline for tracking and measuring the success of protection and restoration efforts over time. This summary report provides an overview of the data and findings included in the full technical report.

Brook Trout Assessment - Key Findings

The following points summarize the key findings of the technical report:

- Intact stream populations of brook trout (where wild brook trout occupy 90-100% of their historical habitat) exist in only 5% of subwatersheds.
- Wild stream populations of brook trout have vanished or are greatly reduced in nearly half of subwatersheds.
- The vast majority of historically occupied large rivers no longer support self-reproducing populations of brook trout.
- Brook trout survive almost exclusively as fragmented populations relegated to the extreme headwaters of streams.
- Poor land management associated with agriculture ranks as the most widely distributed impact to brook trout across the eastern range.
- Non-native fish rank as the largest biological threat to brook trout.
- Intact subwatersheds of wild brook trout in lakes and ponds are almost exclusively located in Maine, but self-reproducing populations remain in some lakes and ponds in New York, New Hampshire and Vermont.
- More data collection is needed to determine the status of brook trout in various parts of the eastern range, particularly in Maine, New Hampshire, New York, Massachusetts and Pennsylvania.

Brook Trout Status and Distribution

This summary report presents information on the status of brook trout populations in 17 states in the Appalachian region, an area that represents 70% of the historical range of brook trout in the United States. This report also identifies the principal threats identified by regional experts to the continued viability of brook trout populations on a state-by-state basis.

Assessment Methodology:

The assessment team collected existing electronic data on brook trout populations from state and federal agencies in 17 states. The team then traveled to each state and met personally with fisheries biologists to review and classify each individual subwatershed. The team used a consistent classification method based on the percentage of historically occupied habitat still maintaining self-reproducing populations of brook trout. Fisheries biologists then used their expert knowledge to list the greatest local threats to wild, self-reproducing brook trout and their habitat.

In total, the assessment team evaluated 11,400 subwatersheds to determine the strength of brook trout populations. While subwatersheds vary in size, they typically contain 25 to 75 miles of streams. Approximately half (5,563) of those subwatersheds historically supported brook trout. The following table presents the current status of brook trout populations in those subwatersheds where brook trout historically thrived.

Brook Trout Subwatershed Status in the Eastern Range
(See following page and pages 18-19 for full map)

Color	Classification	Description	%
🟩	Intact	90-100% historical habitat occupied by self-reproducing brook trout	5%
🟨	Reduced	50-90% historical habitat occupied by self-reproducing brook trout	9%
🟥	Greatly Reduced	1-50% historical habitat occupied by self-reproducing brook trout	27%
🟪	Present, Qualitative Data	Present, but no quantitative data available on populations	19%
⬜	Extirpated	Brook trout have vanished from this subwatershed	21%
▨	Absent, Unclear History	No brook trout currently present, historical presence unknown	6%
▒	Unknown, No Data	No quantitative or qualitative data exists	13%

The assessment data tells a somber story of brook trout decline across their range, but the data also offers hope for restoration and recovery in many areas. Strong, healthy subwatersheds do exist, but they are rare. The majority of these intact subwatersheds are located in Maine, New Hampshire, New York, Vermont and Virginia. Pennsylvania, Maryland, West Virginia and the other New England states each possess only a handful of these intact subwatersheds. Brook trout are extirpated from over 20% of the subwatersheds across the Eastern range and have vanished from all streams and rivers within those areas.

Based on scientific, on-the-ground information gathered within the last ten years, the following table shows the states with the greatest percentage of intact and extirpated subwatersheds.

States with Highest Percentage of Intact and Extirpated Subwatersheds

State	Number of Intact Subwatersheds	Percentage of Total Subwatersheds
Maine	147	14%
Vermont	33	14%
Virginia	36	9%
New Hampshire	21	8%
New York	62*	5%

State	Number of Extirpated Subwatersheds	Percentage of Total Subwatersheds
Georgia	53	58%
Maryland	83	57%
South Carolina	12	44%
North Carolina	95	40%
New Jersey	94	38%

** New York figure was calculated by multiplying the number of watersheds (5th level hydrologic unit) x 2.5, since subwatershed (6th level hydrologic unit) data is not yet available for the state. On average, there are 2.5 subwatersheds within any given watershed in New York.*

Threats to Brook Trout and Their Habitat

Eastern brook trout reside in the most heavily populated and intensely industrialized region of the United States. Land use decisions made over the past several hundred years have severely impacted the quality of brook trout streams and rivers--largely by removing streamside trees and increasing sedimentation and nutrient runoff. While some sections of the East have regained forest cover and are healing from the widespread clearing of the eastern forests, other areas are undergoing rapid change as our population, road network and water needs continue to grow.

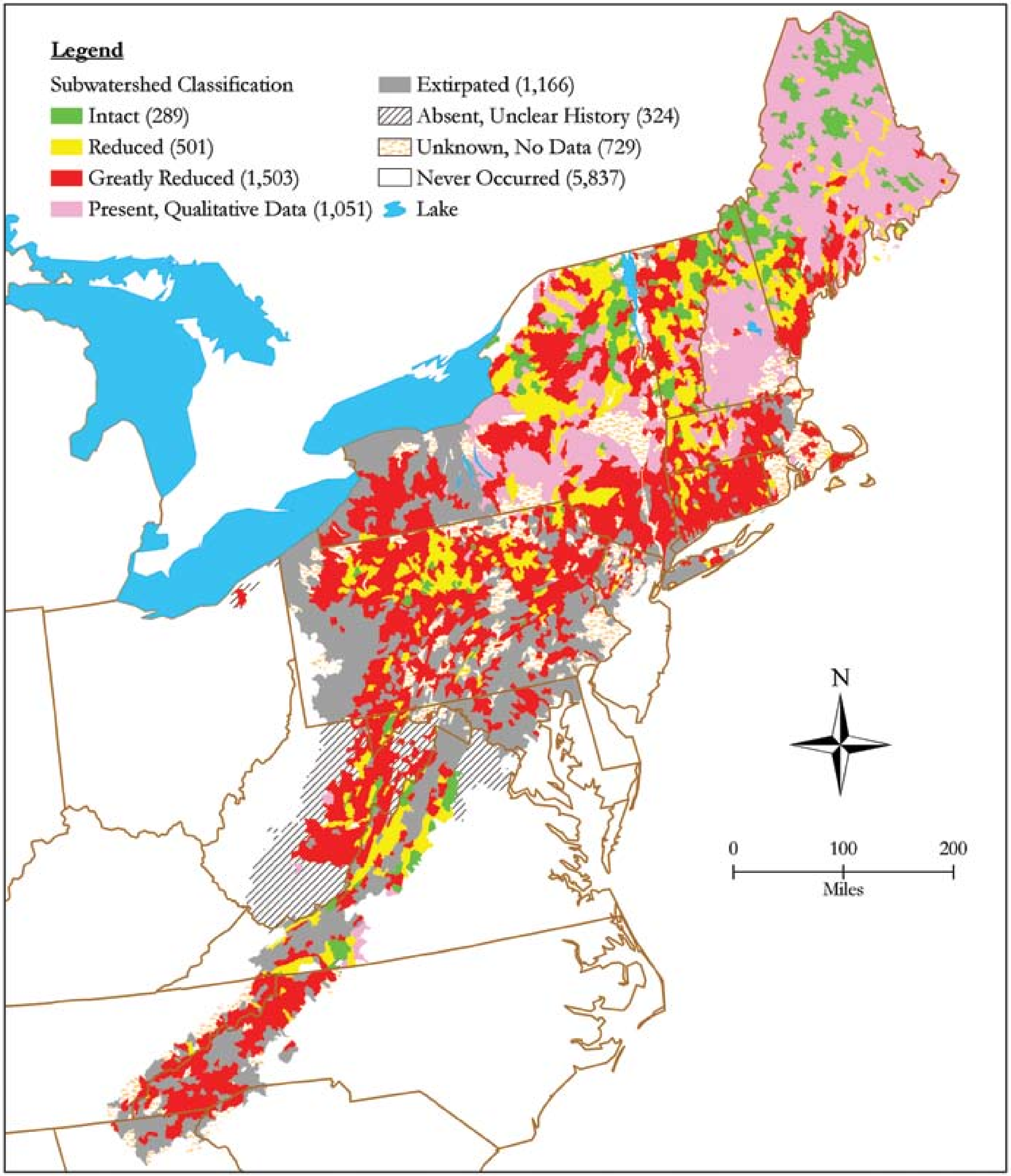
Primary Threats to Brook Trout

Rank	Disturbances (High or Medium)	Number of Subwatersheds	Percentage of Subwatersheds
1	Poor Land Management	1647	37%
2	High Water Temperature	1629	36%
3	Sedimentation (Roads)	1225	27%
4	One or More Non-Native Fish Species	1189	26%
5	Urbanization	1141	25%
6	Riparian Habitat	1029	23%
7	Brown Trout	853	19%
8	Stream Fragmentation (Roads)	767	17%
9	Dam Inundation/Fragmentation	705	16%
10	Forestry	642	14%

Threats information based on professional opinion of regional experts. Figures do not add to 100% because zero, one, or multiple disturbances may occur in each subwatershed.

Brook Trout Population Status in the Eastern U.S. Range by Subwatershed

(See pages 18-19 for a larger map)



Map data derived from state and federal data and compiled in EBTJV assessment results titled, *Distribution, status, and perturbations to brook trout within the Eastern United States, 2006.* Authored by Mark Hudy, US Forest Service; Teresa Thieling, James Madison University; Nathaniel Gillespie, Trout Unlimited; Eric Smith, Virginia Tech. Map created on 2/24/06 by Nathaniel Gillespie, Trout Unlimited.

Regional experts listed poor land management associated with agriculture as the most widespread impact on brook trout habitat in the Eastern United States. Poor land management can involve clearing streamside vegetation, over-grazing sensitive areas, ineffectively managing nutrients and ditching small streams. While these practices cause direct damage to water quality, they also contribute to higher water temperatures and degraded streamside areas – the second and sixth greatest disturbances across the study area.

Roads can have a variety of damaging effects on streams. Sedimentation is listed as the third largest impact to brook trout. Runoff of sand and silt from poorly designed or maintained roads can smother brook trout eggs and the aquatic insects that fish eat. In addition, dams and poorly designed culverts or bridges can act as barriers to fish movement. Streams can quickly become fragmented into sections, isolating brook trout populations from each other and limiting their ability to move and find areas of clean gravel to spawn or colder waters in the summer. Dams also increase water temperatures by slowing down flowing water and exposing it to the air and sun.

Non-native species (such as smallmouth bass, rainbow trout and brown trout) are the only disturbance not related to habitat in the top ten regional impacts to brook trout. These fish can out-compete brook trout in high quality habitat by eating them and forcing them out of the more favorable parts of a stream or lake. Non-native fish also can thrive in lower quality waters that once supported brook trout.

Impacts on water quality and stream health are often complex and interrelated. For example, actions such as removing trees from stream banks, allowing livestock in streams or poorly planning urban development can all cause higher water temperatures, increased sediment and impaired habitat. All of these factors make it more difficult for brook trout to reproduce and survive. In most cases, a combination of negative changes to the surrounding land and stream banks--rather than a single disturbance--causes brook trout to decline or vanish from a particular subwatershed.

Conservation and Restoration Opportunities

Despite their sensitivity to declines in water quality and the introduction of non-native fish, brook trout have managed to persist in countless headwater streams across the eastern United States. Many opportunities currently exist for the restoration of brook trout habitat. For example, working with farmers and other landowners to replant streamside shrubs and trees and fence livestock away from streams can dramatically improve water temperatures and water quality in a relatively short period of time. Many private landowners are currently partnering with federal and local agencies and non-governmental organizations to protect streams on private land. Because farmers and ranchers own so much land throughout the historical range of eastern brook trout, they have a unique opportunity to be at the forefront of the effort to safeguard water quality and restore brook trout populations through cooperative, incentive-based programs.

A host of other opportunities exist for improving brook trout habitat and restoring populations. Liming and other acid abatement techniques can neutralize acid deposition and abandoned mine drainage and make thousands of miles of streams fishable. Protecting forested watersheds can ensure healthy populations and water quality far into the future. Selective removal of non-native fish where appropriate to protect brook trout is an effective management tool that is gaining increasing popularity among biologists. Replacing poorly designed culverts and removing old dams that block fish movement can reconnect fragmented habitat and strengthen or extend brook trout populations downstream.

CLINT FERGUSON

People value brook trout not only for their beauty, their delicious taste, and their sportfish qualities, but also as indicators of the broader health of the watersheds where they live. A sentinel of superior water quality, the brook trout will always mirror the health of the Appalachians and the waters that drain from these landscapes. The assessment information summarized in this report provides new perspectives on the status of brook trout and water quality across the East, allowing analysis at range-wide, regional, state and local scales. This assessment sets a benchmark for fisheries managers, policy makers and citizens to track and assess progress in protecting and restoring eastern waters and their native trout. Collective efforts to restore the brook trout will enable us to protect human health, assure clean and sustainable water supplies and preserve our quality of life for generations to come.

Georgia & South Carolina:

In Georgia and South Carolina, the genetically distinctive Southern Appalachian brook trout has largely retreated into isolated headwater streams on public lands managed by the U.S. Forest Service. These states contain no remaining intact or reduced subwatersheds. While poor land management, roads, and urbanization impact over 75% of brook trout subwatersheds in Georgia, rainbow trout are the most pervasive impact.

Brook Trout Classifications	Number of Subwatersheds	Percentage of Subwatersheds
Intact (>90% habitat occupied)	0	0%
Reduced (50-90% habitat occupied)	0	0%
Greatly Reduced (<50% occupied)	29	25%
Present, Qualitative Data Only	0	0%
Extirpated	65	55%
Absent, Unknown History	0	0%
Unknown, No Data	24	20%
Total	118	100%

Population Status: Southern Appalachian brook trout are present at greatly reduced levels in only 25% of their native range in Georgia and South Carolina. Over 55% of subwatersheds are confirmed or suspected to be extirpated, and population status is unknown in approximately 20%. All remaining brook trout populations exist in isolated headwater streams. In Georgia, all but one of the remaining brook trout subwatersheds are located on national forest lands. Similarly, in South Carolina, three of the seven subwatersheds that support brook trout are located in national forests.

Threats: The Southeast mountains endured harsh land use practices over the past several centuries. Timbering, instream log-drives and poor land use practices increased stream erosion and opened the shaded streams to the sun, degrading waters with silt and raising water temperatures. In response, brook trout retreated to small, headwater streams. To fill the void, many streams were stocked with non-native rainbow trout and brown trout. As the landscape healed and water quality improved, these non-native trout expanded their range and now compete with brook trout for food and space. Rainbow trout now thrive in 96% of Georgia's subwatersheds that have brook trout data.

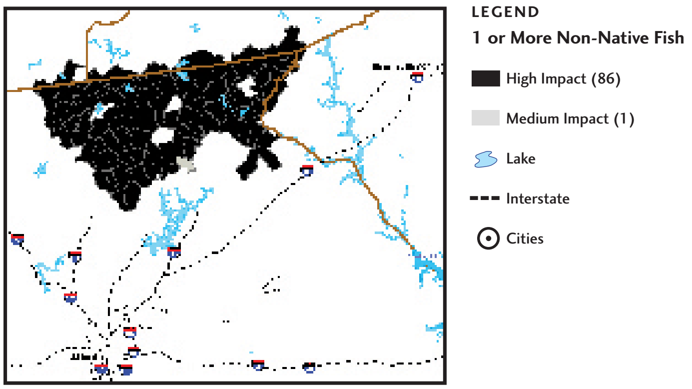
Regional experts identified poor land management, road sediment and urbanization as current impacts to brook trout habitat. These disturbances are widespread throughout the brook trout's southern range, and they result in increased sedimentation that suffocates trout eggs and aquatic insects. Because Georgia and South Carolina represent the extreme southern limit of brook trout, local habitat is particularly sensitive to land use changes that raise water temperatures.

The majority of remaining brook trout streams in Georgia and South Carolina are located in the Chattahoochee and Sumter national forests, where they are protected from future land use changes. The protection and connection of these small, fragmented populations to lower elevation rivers will ensure their long term survival in the face of droughts and floods. The restoration of streamside areas, improvement of stream habitat and the selective removal of non-native fish can strengthen existing populations and restore others to other portions of their original range.

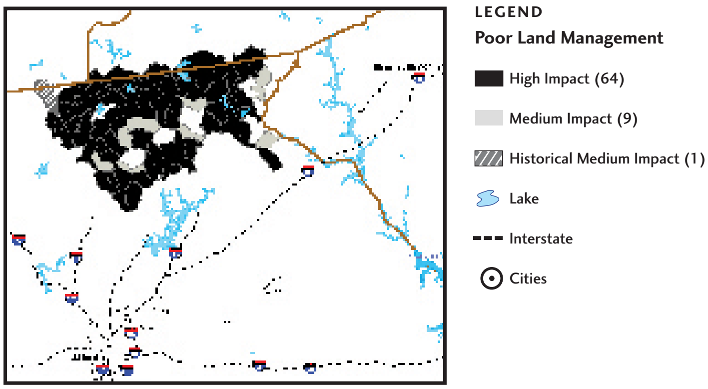
Disturbances (High or Medium)	Number of Subwatersheds	Percentage of Subwatersheds
Rainbow Trout	87	96%
Historical Forestry	83	91%
Poor Land Management	74	81%
Road Sediment	79	87%
Urbanization	76	84%

Threats information based on professional opinion of regional experts. Figures do not add to 100% because zero, one, or multiple disturbances may occur in each subwatershed.

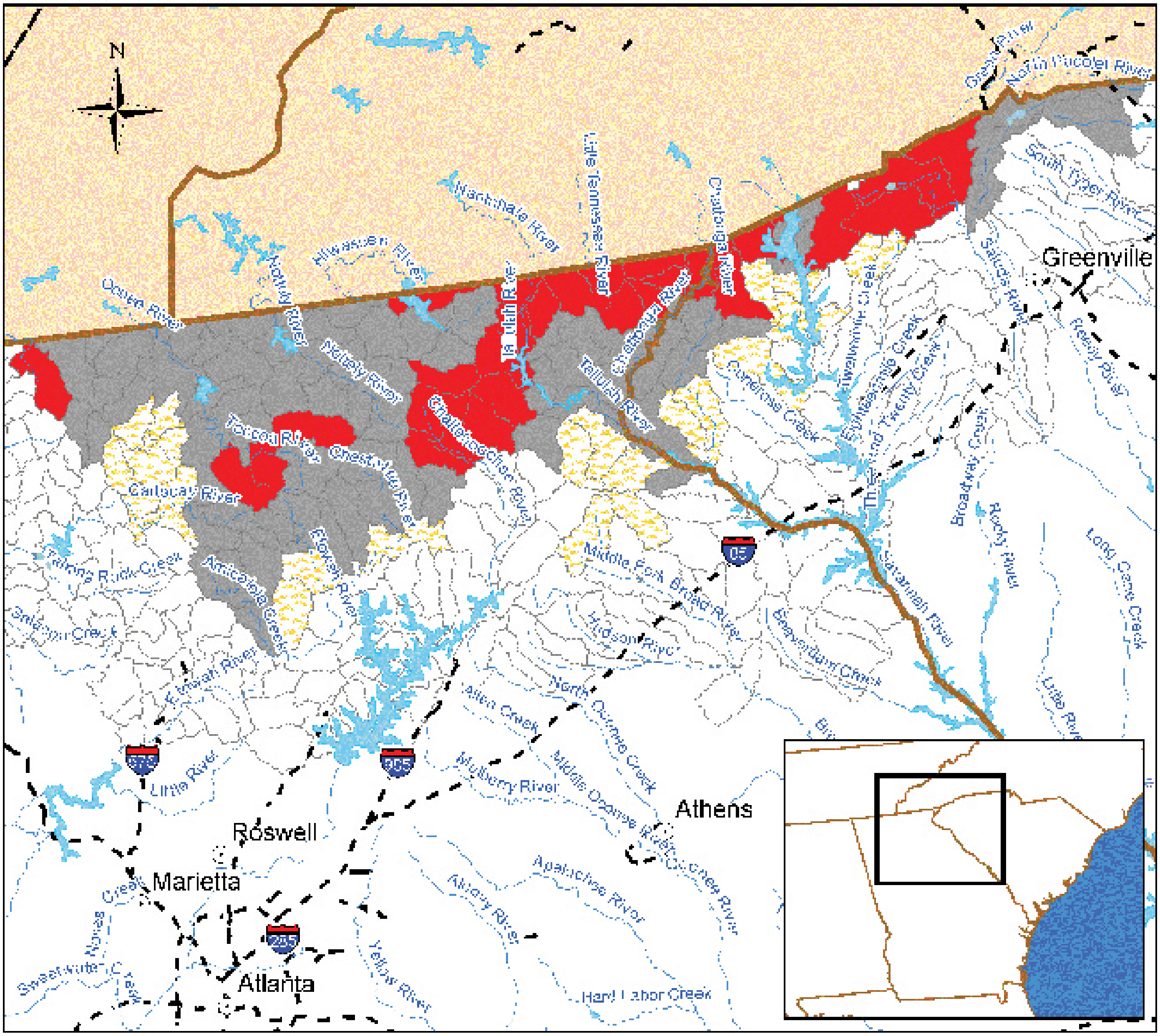
Non-Native Fish Impacts to Brook Trout in Georgia by Subwatershed



Poor Land Management Impacts to Brook Trout in Georgia by Subwatershed



Georgia & South Carolina Brook Trout Population Status by Subwatershed



Legend

Subwatershed Status

🟥 Greatly Reduced (29)

▨ Extirpated (65)

▨ Unknown, No Data (24)

☐ Never Occurred (201)

Rivers

Lake

Interstate

Cities

0 10 20

Miles

Map data derived from state and federal data and compiled in EBTJV assessment results titled, *Distribution, status, and perturbations to brook trout within the eastern United States, 2006.* Authored by Mark Hudy, US Forest Service; Teresa Thieling, James Madison University; Nathaniel Gillespie, Trout Unlimited; Eric Smith, Virginia Tech. Map created on 2/24/06 by Nathaniel Gillespie, Trout Unlimited.

Tennessee & North Carolina:

Brook trout native to the Southern Appalachians are genetically distinctive. Only a handful of subwatersheds in Tennessee and North Carolina still support 50% or more of the brook trout they once did. Many of the largest remaining populations occur on federal lands in headwater streams that escaped previous habitat destruction. Competition with non-native rainbow and brown trout threatens many existing brook trout populations.

Brook Trout Classifications	Number of Subwatersheds	Percentage of Subwatersheds
Intact (>90% habitat occupied)	1	<1%
Reduced (50-90% habitat occupied)	5	2%
Greatly Reduced (<50% occupied)	149	47%
Present, Qualitative Data Only	0	0%
Extirpated	113	36%
Absent, Unknown History	0	0%
Unknown, No Data	49	15%
Total	317	100%

Population Status: Tennessee and North Carolina boast the only remaining intact and reduced subwatersheds in the Southeast, representing less than 3% of the historical subwatersheds where the genetically distinct Southern Appalachian brook trout historically thrived. Almost half of the subwatersheds in these two states are greatly reduced. Where brook trout do persist, populations within greatly reduced subwatersheds often contain considerably less than 50% of historical populations. Brook trout are extirpated in 36% of subwatersheds, and 95 of these 113 extirpated subwatersheds occur in North Carolina. Brook trout data currently is not available for 15% of the total historical subwatersheds in these states.

Threats: The Southeast mountains have suffered from poor land use practices over the past several centuries. Large-scale logging, instream log-drives and poor land management associated with agriculture increased erosion and opened the shaded streams to the sun. As water quality declined and brook trout disappeared, rainbow trout and brown trout were introduced. As forests returned and aquatic habitat improved, these non-native fish expanded their range and now compete with brook trout for food and space. Most remaining high-quality trout habitat is occupied by non-native fish. Rainbow trout are specifically recognized as a threat to brook trout in over 70% of the subwatersheds with brook trout data in these states.

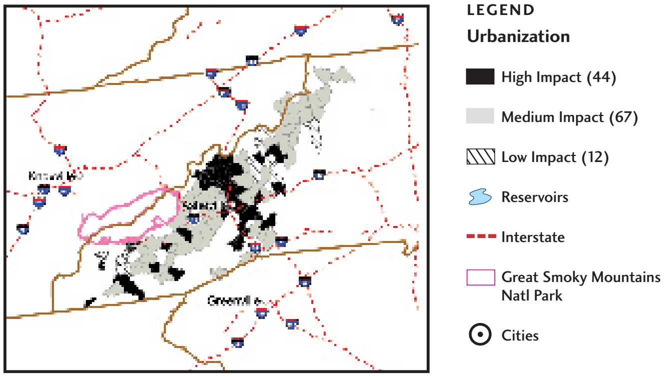
Poor land management continues to contribute to increased water temperatures, sedimentation and nutrient pollution. Regional experts specifically identified urbanization, poor land management and degraded streamside habitat as major threats to brook trout habitat.

Great Smoky Mountains National Park and Cherokee and Nantahala national forests host some of the highest quality trout habitat remaining in the Southeast. Protection and connection of these small, fragmented brook trout populations to lower elevation rivers will ensure their long-term survival in the face of droughts and floods. Continued protection of forested land, cooperative restoration of streamside areas on private land and selective removal of non-native fish can restore healthy populations of brook trout.

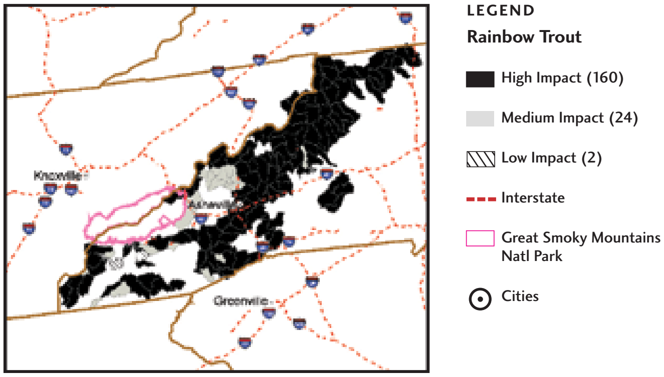
Disturbances (High or Medium)	Number of Subwatersheds	Percentage of Subwatersheds
1 or more Non-Native Fish	185	69%
Rainbow Trout	184	69%
Urbanization	111	41%
Brown Trout	101	38%
Poor Land Management	99	37%
Riparian Habitat	99	37%

Threats information based on professional opinion of regional experts. Figures do not add to 100% because zero, one, or multiple disturbances may occur in each subwatershed.

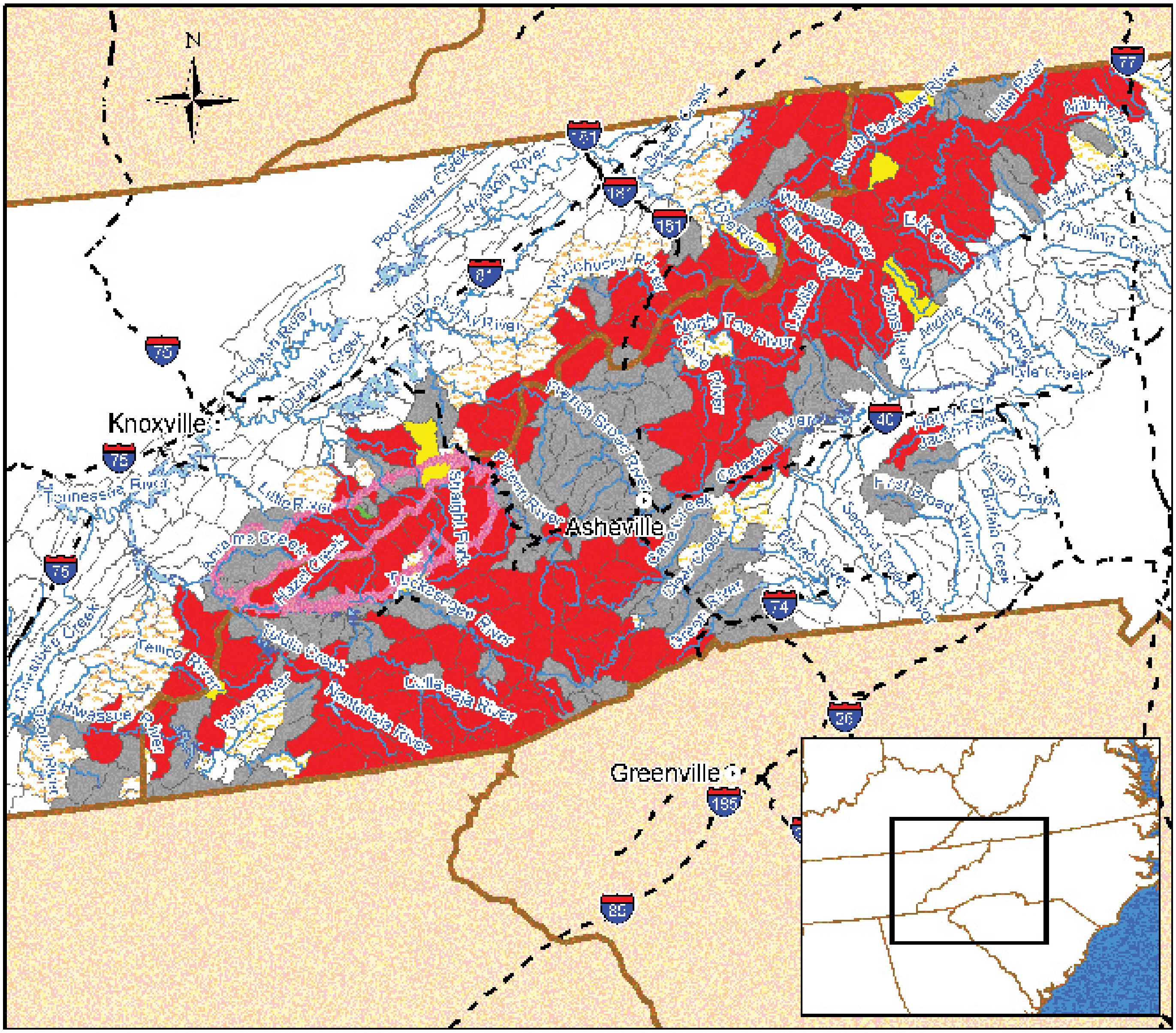
Urbanization Impacts to Brook Trout in Tennessee and North Carolina by Subwatershed



Rainbow Trout Impacts to Brook Trout in Tennessee and North Carolina by Subwatershed



Tennessee & North Carolina Brook Trout Population Status by Subwatershed



Legend

- Rivers & Streams
- Reservoirs
- Interstate
- 🟩 Intact (1)
- 🟨 Reduced (5)
- 🟥 Greatly Reduced (149)
- ▨ Extirpated (113)
- Unknown, No Data (49)
- Never Occurred (228)
- Great Smoky Mountains Nat'l Park
- Cities

0 15 30
Miles

Map data derived from state and federal data and compiled in EBTJV assessment results titled, *Distribution, status, and perturbations to brook trout within the eastern United States, 2006.* Authored by Mark Hudy, US Forest Service; Teresa Thieling, James Madison University; Nathaniel Gillespie, Trout Unlimited; Eric Smith, Virginia Tech. Map created on 2/24/06 by Nathaniel Gillespie, Trout Unlimited.

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Virginia:

The mountains of Virginia, many of them protected under federal ownership, provide the largest stronghold for wild brook trout south of the Mason-Dixon line. Brook trout, however, have been largely extirpated from lower elevations of the state where poor land management, outdated grazing practices, roads and other human changes have degraded water quality and streamside conditions.

Brook Trout Classifications	Number of Subwatersheds	Percentage of Subwatersheds
Intact (>90% habitat occupied)	36	9%
Reduced (50-90% habitat occupied)	80	20%
Greatly Reduced (<50% occupied)	56	14%
Present, Qualitative Data Only	8	2%
Extirpated	148	38%
Unknown, No Data	0	0%
Absent, Unclear History	64	16%
Total	392	100%

Population Status: The Commonwealth of Virginia retains the strongest brook trout populations south of the Mason-Dixon Line, and supports subwatersheds with intact populations (9%) and reduced populations (20%). Brook trout are concentrated in steep mountain streams, where the Shenandoah National Park and the George Washington and Jefferson National Forests protect a large number of these healthy populations. Virginia has excellent data available for brook trout populations; only 2% of subwatersheds lack quantitative data on brook trout.

Despite this relatively strong reservoir, Virginia has lost all populations in 38% of its historical brook trout subwatersheds, an area nearly the size of Connecticut. Brook trout no longer live along a contiguous swath of land stretching from Winchester south through the Shenandoah Valley and continuing south of Roanoke into southwestern Virginia. Brook trout across this area once inhabited valley-bottom spring creeks, which over time have become degraded by farming, timber harvest, and other land use practices that alter water quality and stream habitat.

In Northern Virginia, the majority of subwatersheds were identified as absent, unclear history. This classification indicates that experts are uncertain whether brook trout ever occupied these streams and/or whether they vanished years ago. Elevation and hydrology data suggest that brook trout most likely never occurred in the majority of these absent subwatersheds.

Threats: Regional experts listed high water temperature as the greatest disturbance to brook trout populations across the state. The next three impacts all contribute to high water temperatures as well as to increased sedimentation: poor land management, degraded streamside (riparian) habitat and grazing in sensitive areas. Partnership efforts are underway to restore streamside vegetation and reduce water temperatures and decrease sediment and nutrient inputs in a number of creeks that formerly held brook trout.

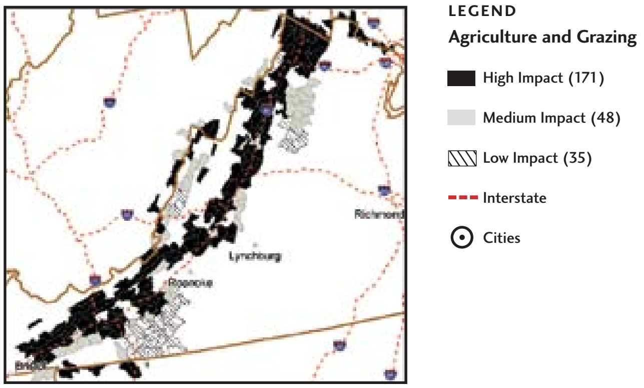
In addition, road and culvert fragmentation of streams are problematic statewide. These threats are attributable to both suburban development and poor road planning for forestry.

The low buffering geology of much of the Appalachian mountains confines acid deposition impacts largely to higher elevations. Regional biologists identified acid deposition as affecting 76 subwatersheds, located largely on federal lands. While not as widespread as other disturbances, acid deposition threatens a large portion of remaining brook trout streams.

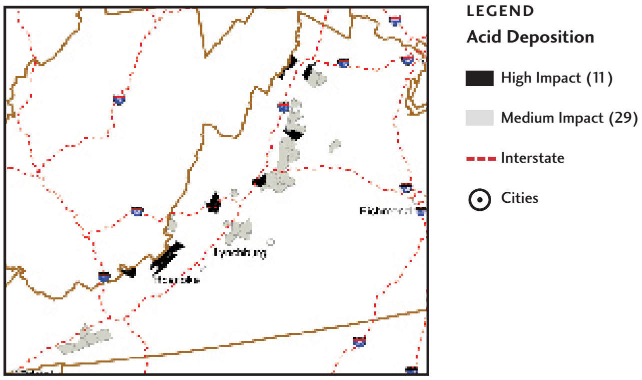
Disturbances (High or Medium)	Number of Subwatersheds	Percentage of Subwatersheds
High Water Temperature	253	77%
Poor Land Management	214	65%
Riparian Habitat	209	64%
Grazing	205	63%
Stream fragmentation (Roads)	198	60%

Threats information based on professional opinion of regional experts. Figures do not add to 100% because zero, one, or multiple disturbances may occur in each subwatershed.

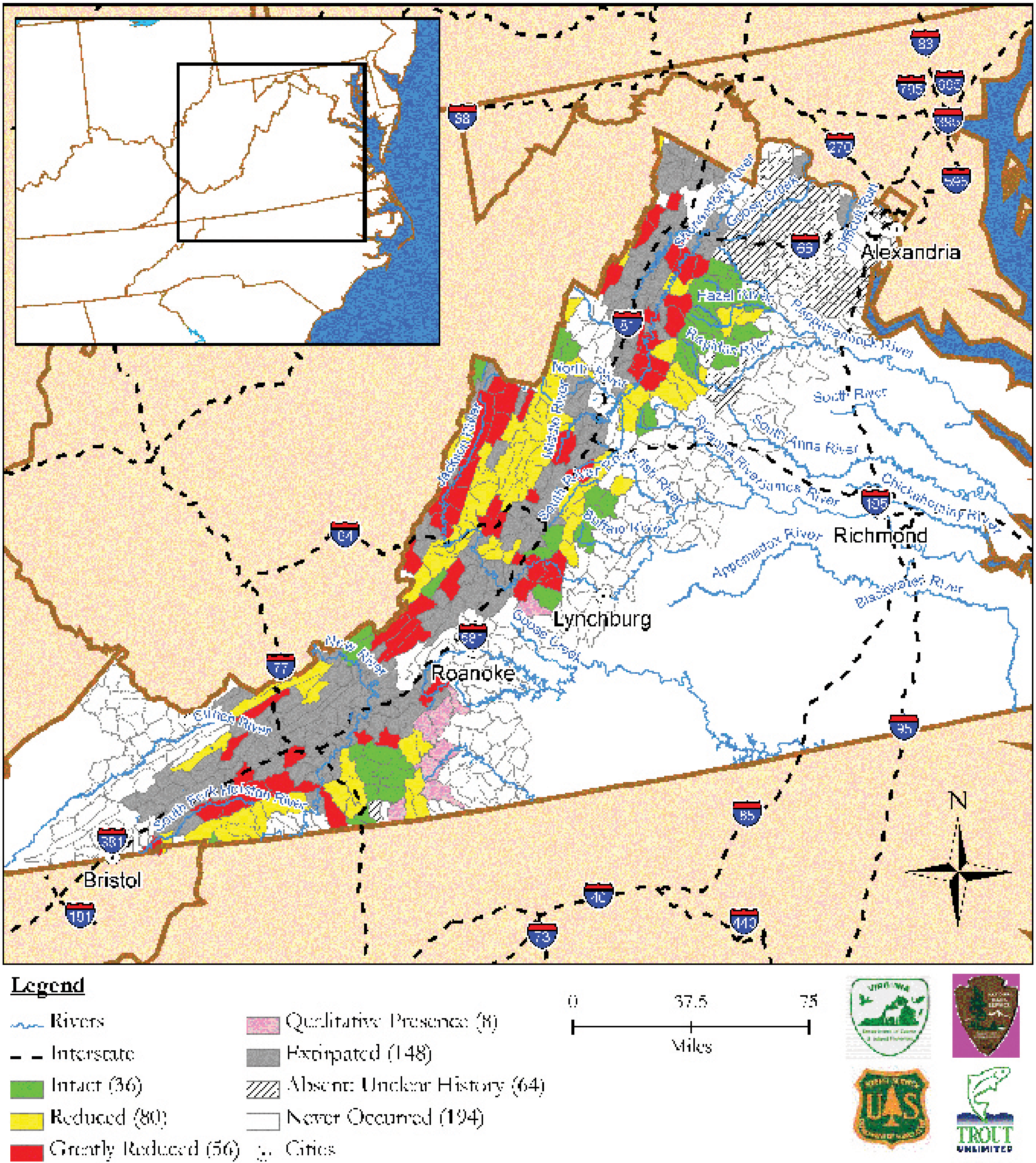
Poor Land Management & Grazing Impacts to Brook Trout in Virginia by Subwatershed



Acid Deposition Impacts to Brook Trout in Virginia by Subwatershed



Virginia Brook Trout Population Status by Subwatershed



Map data derived from state and federal data and compiled in EBTJV assessment results titled, *Distribution, status, and perturbations to brook trout within the eastern United States, 2006.* Authored by Mark Hudy, US Forest Service; Teresa Thieling, James Madison University; Nathaniel Gillespie, Trout Unlimited; Eric Smith, Virginia Tech. Map created on 2/24/06 by Nathaniel Gillespie, Trout Unlimited.

West Virginia:

The majority of West Virginia's remaining brook trout subwatersheds are greatly reduced largely due to poor water quality associated with a long history of poor land management, forestry and mining. In addition, acid deposition and abandoned mine drainage each impair approximately 25% of available brook trout habitat. Further information is required in over half of the state's subwatersheds to determine whether brook trout historically thrived in areas where they are currently absent.

Brook Trout Classifications	Number of Subwatersheds	Percentage of Subwatersheds
Intact (>90% habitat occupied)	4	1%
Reduced (50-90% habitat occupied)	16	4%
Greatly Reduced (<50% occupied)	130	30%
Present, Qualitative Data Only	4	1%
Extirpated	24	6%
Absent, Unclear History	249	57%
Unknown, No Data	7	2%
Total	434	100%

Population Status: West Virginia has very few healthy brook trout subwatersheds -- 1% remain intact and 4% are reduced. The majority of existing brook trout subwatersheds are greatly reduced (30%). While only 6% of the subwatersheds are extirpated, brook trout were documented to be absent from 57% of subwatersheds within the study area. Experts know that brook trout are not living in those subwatersheds, but they are not able to determine whether brook trout historically occurred there or disappeared some time ago. Further investigation is needed to determine if the native range of brook trout is much smaller than previous research indicated, or if the extent of brook trout losses are much more severe than biologists can measure at this point in time.

Threats: West Virginia fisheries experts determined that poor land management and forestry ranked as the two most widespread disturbances to brook trout populations across the state. Both of these land uses can degrade riparian habitat (5th ranked impact) by removing streamside vegetation. Increased nutrients, sediment and higher water temperatures are generally the result of poor land management. Forestry practices can reduce water quality and raise water temperatures, typically due to poorly designed and maintained dirt roads and skid trails, and loss of streamside trees.

Acid deposition was identified as currently affecting 30% of subwatersheds with documented brook trout habitat. Acid deposition often exerts a greater proportional impact on higher elevation areas, precisely where brook trout have retreated in the face of water quality declines in the valleys.

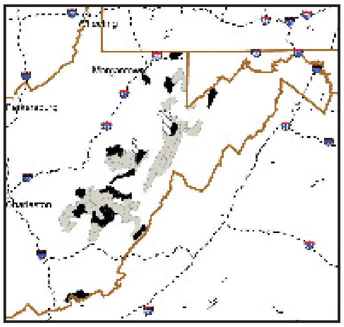
Abandoned mine drainage (AMD) affects almost one quarter of West Virginia's subwatersheds that have brook trout data. AMD often renders entire sections of stream lifeless from toxic, acidic water leaching from mines.

Expanding the state's successful lime dosing program to neutralize acid streams and revive aquatic life can make hundreds of miles of rivers habitable for brook trout. Increasing voluntary programs to protect and replant streamside trees holds great potential for brook trout restoration on private lands throughout the state.

Disturbances (High or Medium)	Number of Subwatersheds	Percentage of Subwatersheds
Poor Land Management	104	59%
Forestry	96	54%
Acid Deposition	53	30%
Abandoned Mine Drainage	43	24%
Riparian Habitat	37	21%
Mining	30	17%

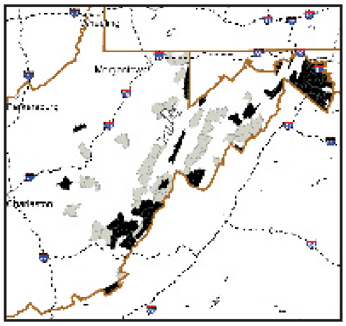
Threats information based on professional opinion of regional experts. Figures do not add to 100% because zero, one, or multiple disturbances may occur in each subwatershed.

Acid Deposition Impacts to Brook Trout in West Virginia by Subwatershed



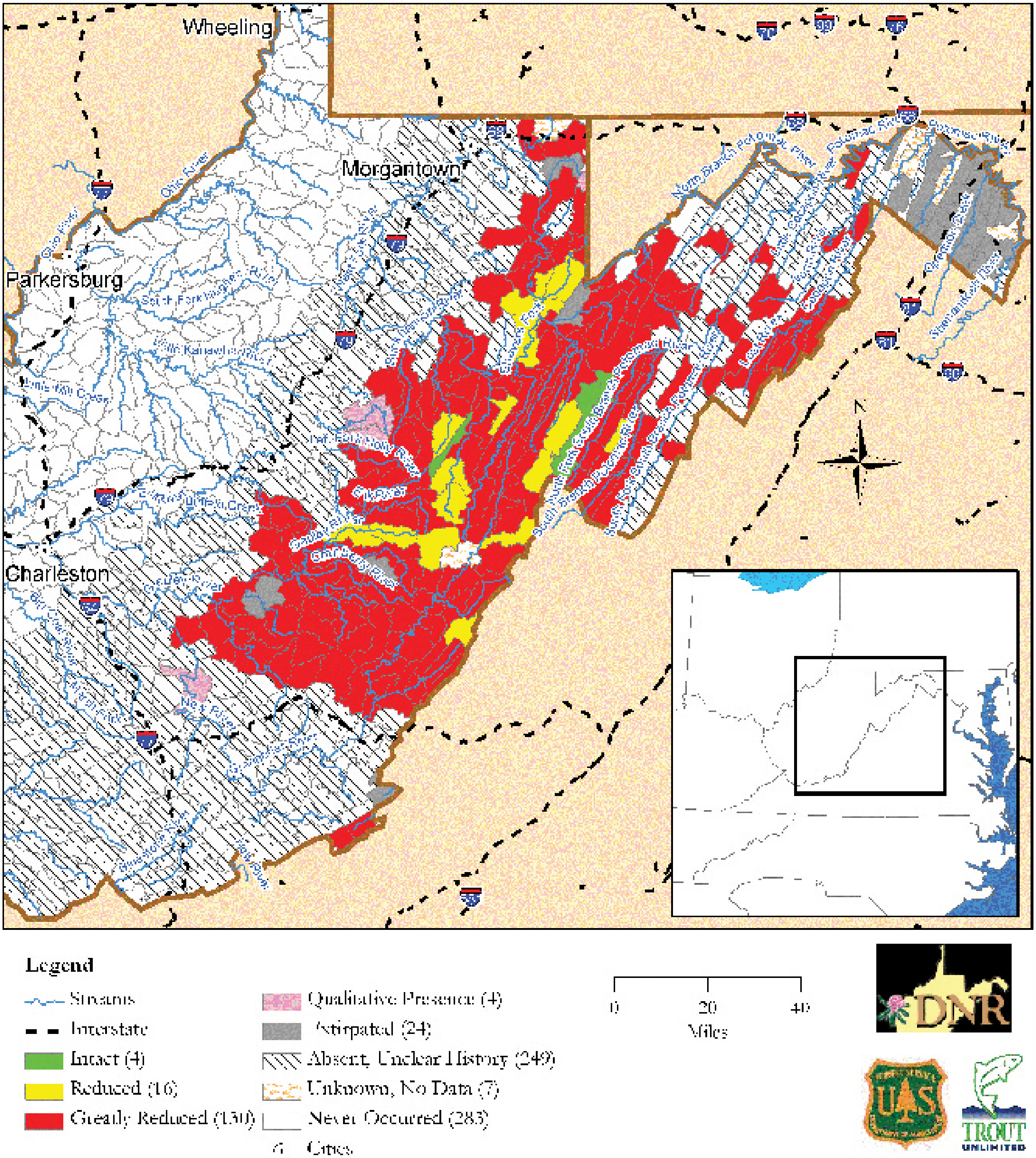
LEGEND
Acid Deposition
■ High Impact (18)
□ Medium Impact (37)
▨ Low Impact (3)
--- Interstate
⊙ Cities

Poor Land Management Impacts to Brook Trout in West Virginia by Subwatershed



LEGEND
Poor Land Management
■ High Impact (54)
□ Medium Impact (50)
▨ Low Impact (4)
--- Interstate
⊙ Cities

West Virginia Brook Trout Population Status by Subwatershed



Map data derived from state and federal data and compiled in EBTJV assessment results titled, *Distribution, status, and perturbations to brook trout within the eastern United States, 2006.* Authored by Mark Hudy, US Forest Service; Teresa Thieling, James Madison University; Nathaniel Gillespie, Trout Unlimited; Eric Smith, Virginia Tech. Map created on 2/24/06 by Nathaniel Gillespie, Trout Unlimited.

Maryland:

Maryland's brook trout populations are greatly diminished from their historical range, with only three intact subwatersheds remaining in the western panhandle. Over 55% of subwatersheds across the state have lost brook trout entirely, and almost 30% contain only small, headwater populations. High water temperature, poor land management, urbanization and water withdrawals exert the greatest impact on brook trout in the state.

Brook Trout Classifications	Number of Subwatersheds	Percentage of Subwatersheds
Present, Intact	3	2%
Present, Reduced	5	3%
Present, Greatly Reduced	42	30%
Present, Qualitative Data	0	0%
Extirpated	83	57%
Absent, Unclear History	0	0%
Unknown, No Data	12	8%
Total	145	100%

Population Status: The state of Maryland has very few healthy brook trout subwatersheds, concentrated in the western panhandle in the more mountainous Appalachian terrain. The Savage River headwaters and a handful of other subwatersheds along the Pennsylvania and West Virginia border represent the remaining strong populations of brook trout in the state. Only 2% of the subwatersheds are intact, and 3% of the subwatersheds are reduced. 29% of the state's subwatersheds are greatly reduced. Brook trout populations are extirpated from 57% of the state's subwatersheds.

Maryland possesses relatively complete data on brook trout. Only 8% of the 145 subwatersheds within the historical range have no brook trout information available, located primarily north of the West Virginia boundary between Hagerstown and Cumberland.

Threats: Regional experts identified high water temperature as a disturbance in a staggering 79% of Maryland subwatersheds with brook trout data. High water temperatures result primarily from urbanization, poor land management and groundwater withdrawals. In the rolling piedmont and coastal plain, historical clearing of forests, insufficient streamside vegetation and ineffective nutrient management have helped usher the brook trout from many of its native waters. Groundwater withdrawals for irrigation and residential use are particularly damaging to stream ecosystems, since groundwater plays a key role in moderating stream temperatures and maintaining flows during droughts.

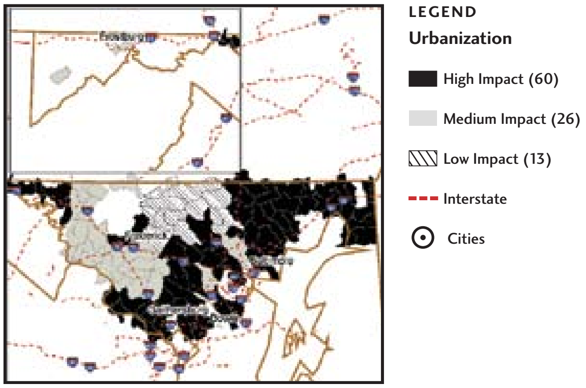
The state's western panhandle is comprised of rugged mountains where poor land management also impacts stream habitat. Other localized impacts threaten the survival of brook trout populations, most notably abandoned mine drainage, acid deposition and road sedimentation. Regional experts cited historical forestry practices as a factor in the loss of brook trout across much of this region.

Protecting the few intact subwatersheds and expanding brook trout populations in the western panhandle and in the Catoctin Mountains will help ensure the long-term health of brook trout in Maryland. A surprising number of brook trout streams survive near Baltimore. Maintaining these populations will be an extraordinary challenge. As forests regrow and the state pursues more creative ways to reduce stormwater runoff, reestablish streamside forests and improve water quality in the Chesapeake Bay watershed, the potential for restoring brook trout in Maryland is strong.

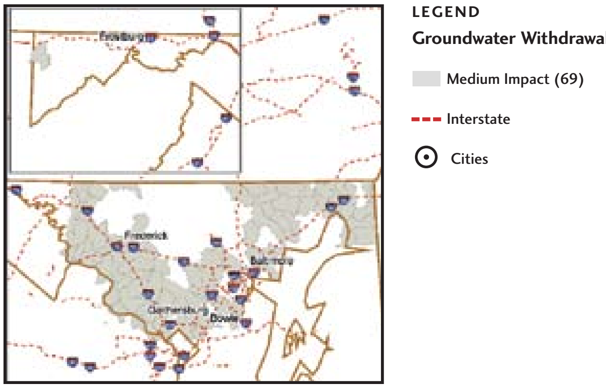
Disturbances (High or Medium)	Number of Subwatersheds	Percentage of Subwatersheds
High Water Temperature	106	79%
Urbanization	100	75%
Poor Land Management	91	68%
Groundwater Withdrawals	75	56%
Surface Water Withdrawals	53	40 %

Threats information based on professional opinion of regional experts. Figures do not add to 100% because zero, one, or multiple disturbances may occur in each subwatershed.

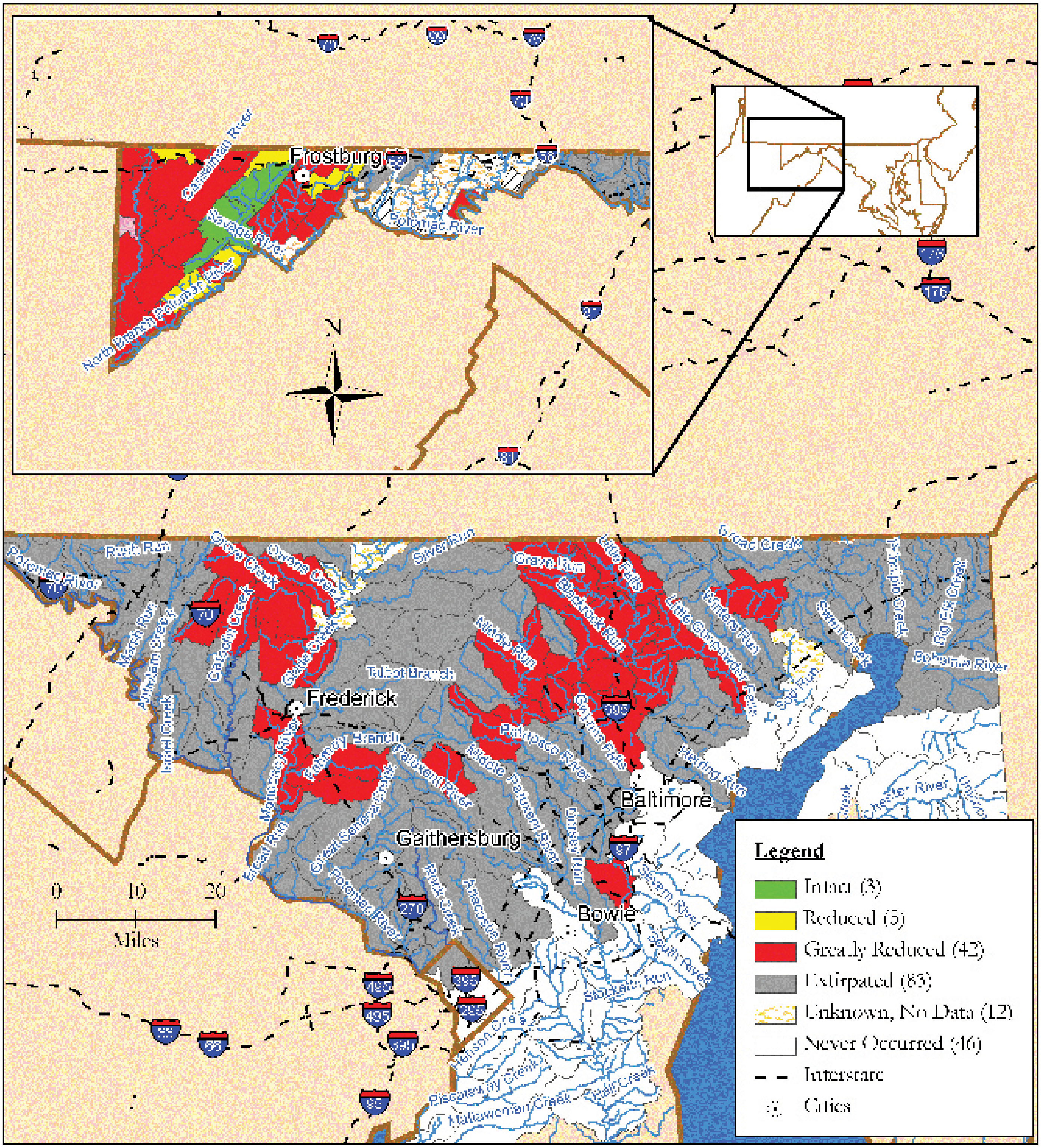
Urbanization Impacts to Brook Trout in Maryland by Subwatershed



Groundwater Withdrawal Impacts to Brook Trout in Maryland by Subwatershed



Maryland Brook Trout Population Status by Subwatershed



Map data derived from state and federal data and compiled in EBTJV assessment results titled, *Distribution, status, and perturbations to brook trout within the eastern United States, 2006.* Authored by Mark Hudy, US Forest Service; Teresa Thieling, James Madison University; Nathaniel Gillespie, Trout Unlimited; Eric Smith, Virginia Tech. Map created on 2/24/06 by Nathaniel Gillespie, Trout Unlimited.

Pennsylvania & Ohio:

Brook trout populations remain intact in very few subwatersheds in Pennsylvania, located primarily in the Allegheny Mountains, Potter and Clinton counties, and the northeastern corner of the state. Brook trout survive mostly in isolated, headwater populations. High water temperatures and sedimentation from poor land management, roads and urbanization impact the most subwatersheds. A few small brook trout populations still survive in Ohio.

Brook Trout Classifications	Number of Subwatersheds	Percentage of Subwatersheds
Intact (>90% habitat occupied)	16	1%
Reduced (50-90% habitat occupied)	118	9%
Greatly Reduced (<50% occupied)	507	39%
Present, Qualitative Data Only	5	<1%
Extirpated	449	34%
Absent, Unclear History	0	0%
Unknown, No Data	218	17%
Total	1313	100%

Population Status: Brook trout historically thrived across Pennsylvania, with the exception of areas in the extreme western and southwestern portion of the state. Today, 1% of the state's historical subwatersheds remain intact, while 9% are reduced. Most of these relatively healthy brook trout subwatersheds are located in the west-central portion of Allegheny National Forest, in the God's Country region including the Genessee River headwaters, Kettle Creek and other tributaries to the West Branch Susquehanna River, and in the state's northeast corner between the Delaware and North Branch Susquehanna Rivers. In 39% of subwatersheds, brook trout are greatly reduced and typically occupy only small, headwater streams. Brook trout have vanished from 34% of historical brook trout subwatersheds. A significant portion of the state (17%) lacks any data on the presence of brook trout.

Until a recent discovery of several remnant populations, brook trout were believed to be extirpated from Ohio. Due to conservation and management efforts, however, brook trout now survive at greatly reduced levels in three subwatersheds. Seven other surrounding subwatersheds have suitable habitat but lack brook trout populations, and no data exists to determine their historical presence. Brook trout are confirmed to be extirpated from one subwatershed in Ohio.

Threats: Regional experts ranked poor land management associated with agriculture as the most widespread disturbance to brook trout habitat across Pennsylvania, impacting almost 50% of subwatersheds with brook trout data. Traditional land uses that remove streamside trees directly contribute to high water temperature, the second most widespread disturbance. Increased partnerships on private lands to reduce water temperature, nutrient runoff and sedimentation could greatly benefit Pennsylvania's water quality and brook trout populations.

Regional experts cited competition and predation from brown trout as the third highest ranked impact across the state. Urbanization and associated road sedimentation ranked among the top five disturbances statewide.

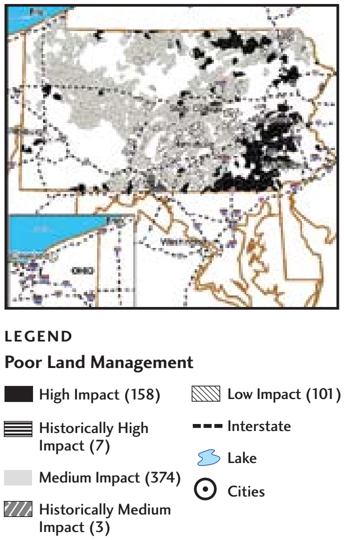
While not as widespread as the top five disturbances, acid deposition impairs clusters of subwatersheds (123 total) with poor buffering geology. Abandoned mine drainage impacts are localized and severe, affecting a group of subwatersheds larger than all of Connecticut. Expanding ongoing efforts to mitigate these water quality impacts could restore many miles of brook trout habitat that currently support little or no aquatic life.

Ohio's few brook trout populations are disturbed by urbanization and poor instream habitat, which lead to higher water temperatures. Dams and impassable culverts contribute to fragmented streams in these subwatersheds.

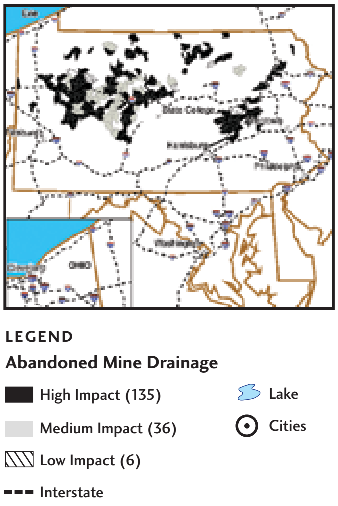
Disturbances (High or Medium)	Number of Subwatersheds	Percentage of Subwatersheds
Poor Land Management	532	49%
High Water Temperature	463	42%
Brown Trout	296	27%
Sedimentation (Roads)	248	23%
Urbanization	233	21%

Threats information based on professional opinion of regional experts. Figures do not add to 100% because zero, one, or multiple disturbances may occur in each subwatershed.

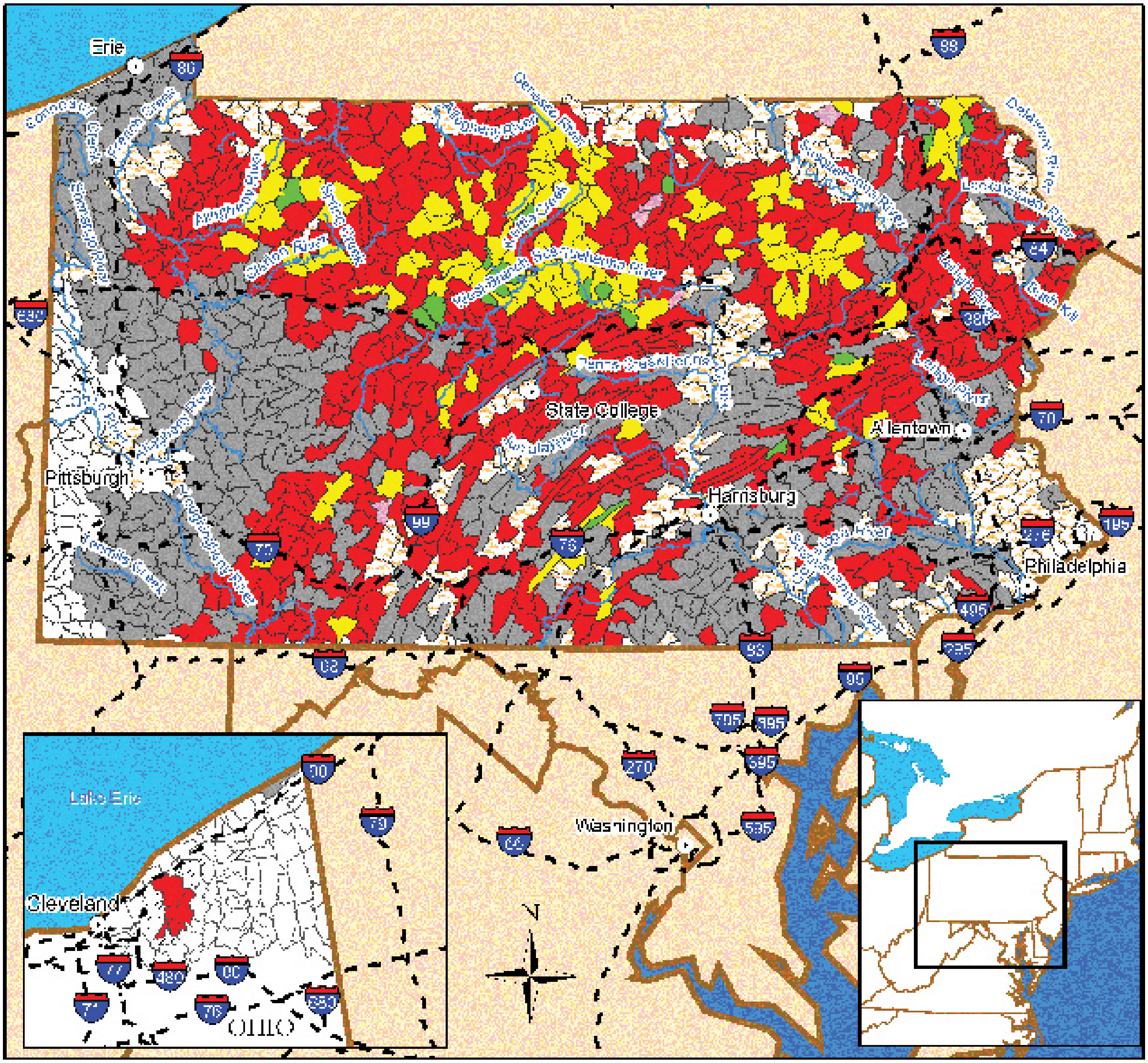
Poor Land Management Impacts to Brook Trout in In Pennsylvania and Ohio by Subwatershed



Abandoned Mine Drainage Impacts to Brook Trout in Pennsyvania and Ohio by Subwatershed



Pennsylvania & Ohio Brook Trout Population Status by Subwatershed



Legend

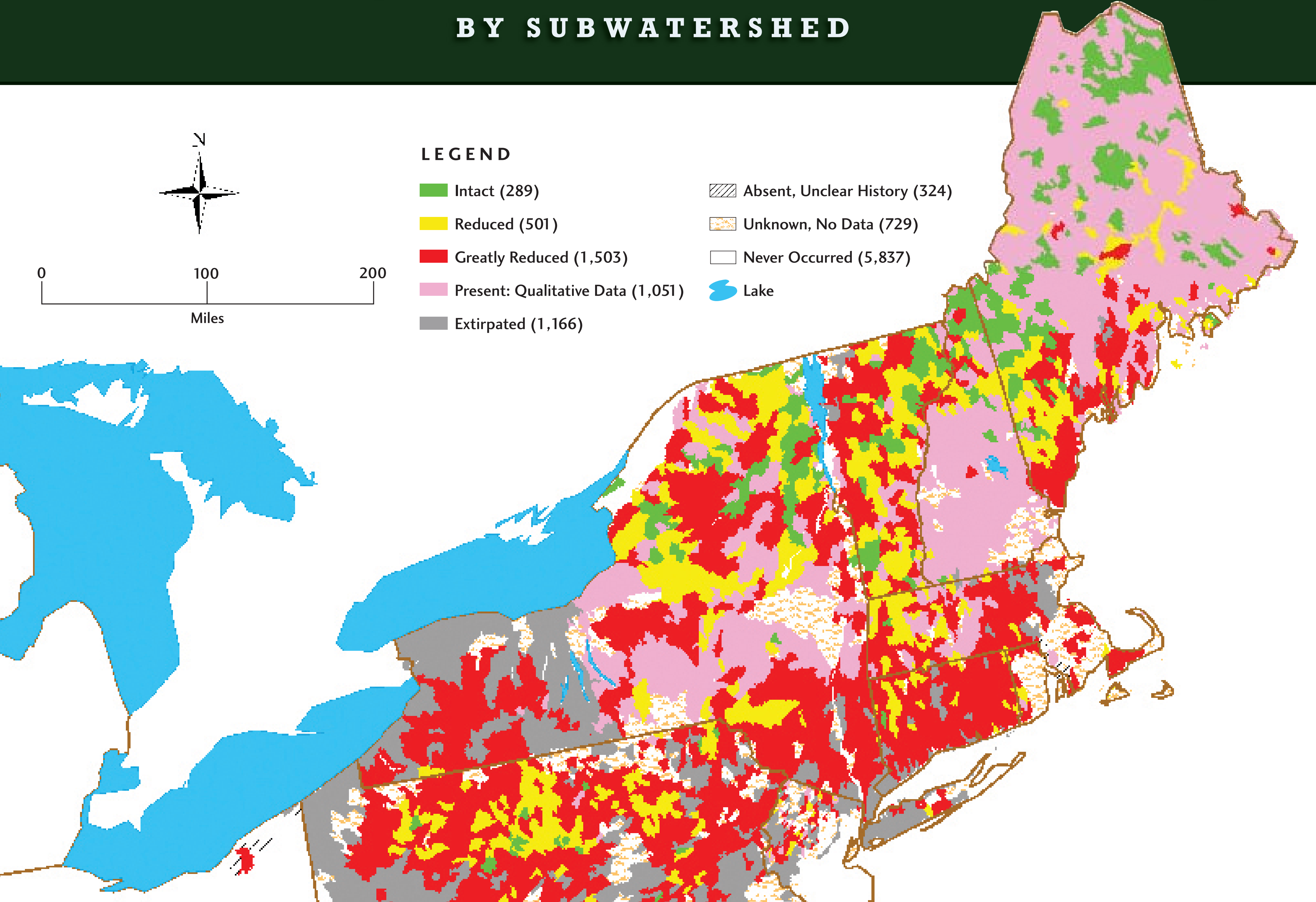
🟩 Intact (16)
🟨 Reduced (118)
🟥 Greatly Reduced (507)
▨ Qualitative Presence (5)
▨ Extirpated (449)
▨ Unkown, No Data (218)
☐ Never Occurred (72)
Rivers
Lake
Interstate
Cities

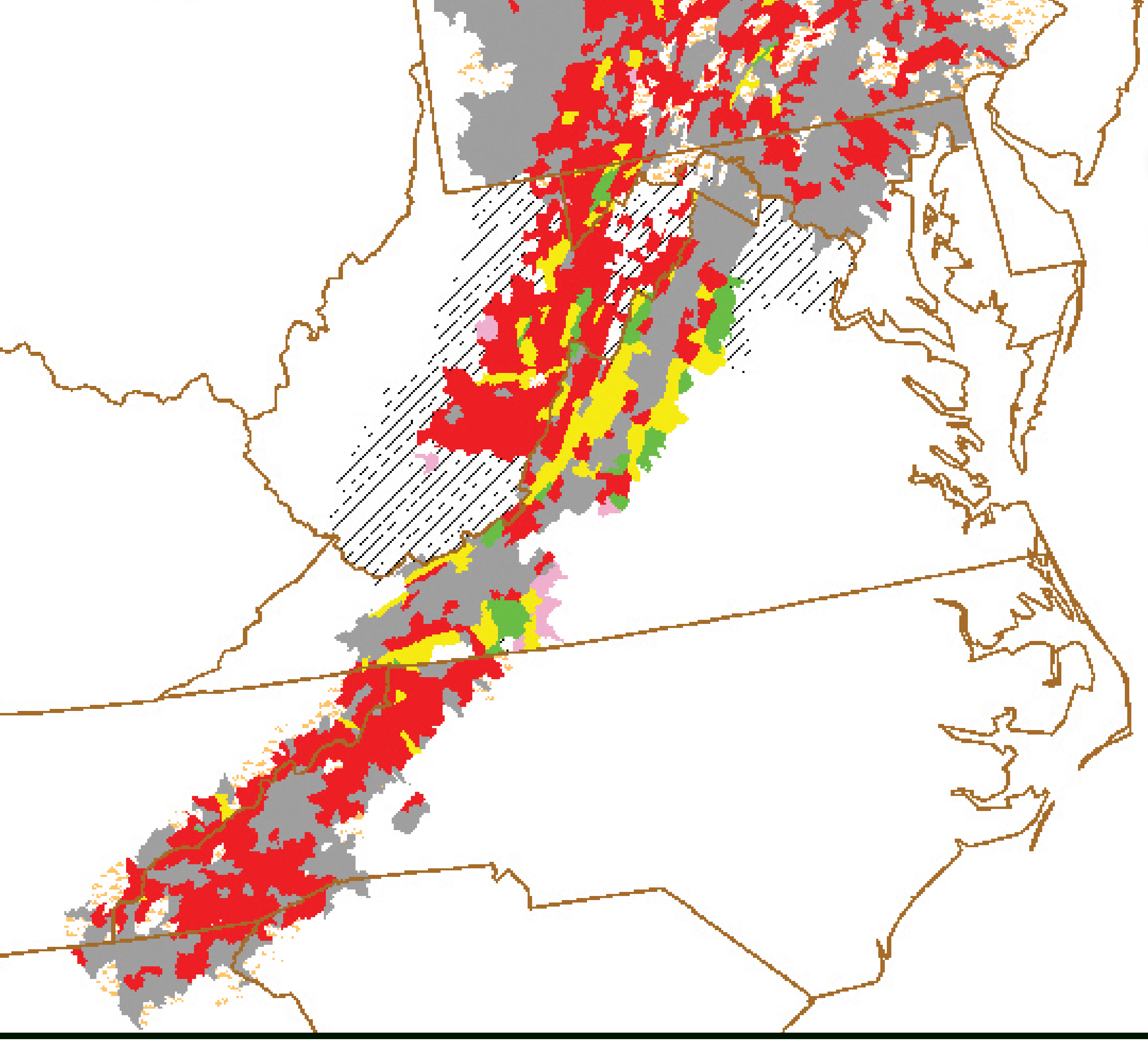
0 25 50
Miles

Map data derived from state and federal data and compiled in EBTJV assessment results titled, *Distribution, status, and perturbations to brook trout within the eastern United States, 2006.* Authored by Mark Hudy, US Forest Service; Teresa Thieling, James Madison University; Nathaniel Gillespie, Trout Unlimited; Eric Smith, Virginia Tech. Map created on 2/24/06 by Nathaniel Gillespie, Trout Unlimited.

Brook Trout Population Status in the Eastern U.S. Range

BY SUBWATERSHED





Strong wild brook trout populations demonstrate that a stream or river ecosystem is healthy and that water quality is excellent. A decline in brook trout populations can serve as an early warning that the health of an entire system is at risk.

Map data derived from state and federal data and compiled in EBTJV assessment results titled, *Distribution, status, and perturbations to brook trout within the eastern United States,* 2006. Authored by Mark Hudy, US Forest Service; Teresa Thieling, James Madison University; Nathaniel Gillespie, Trout Unlimited; Eric Smith, Virginia Tech. Map created on 2/24/06 by Nathaniel Gillespie, Trout Unlimited. For more information, go to **www.brookie.org**.

New Jersey:

Brook trout survive in less than half of their original range in New Jersey, although reduced populations remain in the New Jersey Highlands and the Delaware Water Gap. Urbanization and a variety of environmental impacts associated with industry and roads have played a major role in the loss of brook trout populations. Further assessment data is needed in a third of the state.

Brook Trout Classifications	Number of Subwatersheds	Percentage of Subwatersheds
Intact (>90% habitat occupied)	1	<1%
Reduced (50-90% habitat occupied)	14	6%
Greatly Reduced (<50% occupied)	44	18%
Present, Qualitative Data Only	19	9%
Extirpated	94	38%
Absent, Unclear History	0	0%
Unknown, No Data	76	30%
Total	248	100%

Population Status: Brook trout survive in less than half of their original range in New Jersey. Less than 1% of subwatersheds remain intact, and 6% are reduced. Brook trout populations are greatly reduced in another 18% of the state's subwatersheds. Brook trout are present in another 9% of subwatersheds, but no quantitative data is available. These areas where brook trout persist total less than the number of subwatersheds (38%) where brook trout have been extirpated. Population status is unknown for 30% of the historical brook trout range in New Jersey.

While the New Jersey Highlands are home to most of the healthier brook trout subwatersheds, populations also exist in tributaries to the Delaware River within the Delaware Water Gap National Recreation Area.

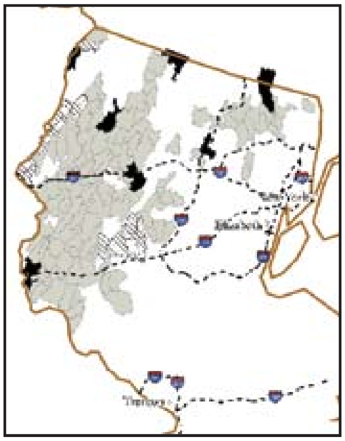
Disturbances (High or Medium)	Number of Subwatersheds	Percentage of Subwatersheds
Sedimentation (Roads)	114	66%
Urbanization	111	65%
Dam Inundation/Fragmentation	100	58%
High Water Temperature	96	56%
Stream Fragmentation (Roads)	95	55%
1 or more Non-Native Fish	65	38%

Threats information based on professional opinion of regional experts. Figures do not add to 100% because zero, one, or multiple disturbances may occur in each subwatershed.

Threats: New Jersey is the most densely populated state in the country. Its industrial legacy has left a heavy footprint on brook trout habitat, with the greatest impacts in the most populated and urbanized regions of the state. State fisheries experts listed sedimentation from roads and urbanization as the top two most widely distributed impacts to native brook trout subwatersheds. A dense road network across much of the state contributes to declines in water quality from sedimentation, warmer water, and non-point source pollutants. In addition, regional experts identified fragmentation by dams--many built more than a century ago for mills and local power--as the third greatest impact across the state. Dams physically dissect a stream and isolate fish populations, and they also increase water temperatures by slowing down water and exposing it to the sun. Road culverts can exert similar ecological impacts by preventing fish from moving past these barriers. Poorly designed culverts may also contribute to sediment pollution.

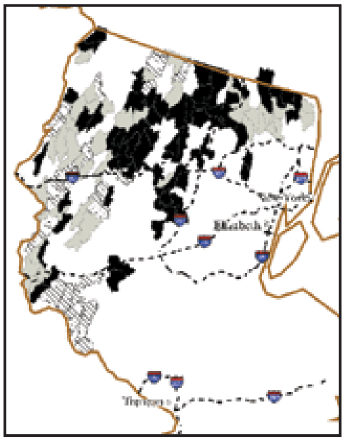
Considering that New Jersey is the most densely populated state in the United States, the presence of brook trout in 59 subwatersheds is encouraging, yet it is tempered by the documentation that more than a third of the historical subwatersheds have lost their native trout.

Sedimentation (Road) Impacts to Brook Trout in New Jersey by Subwatershed



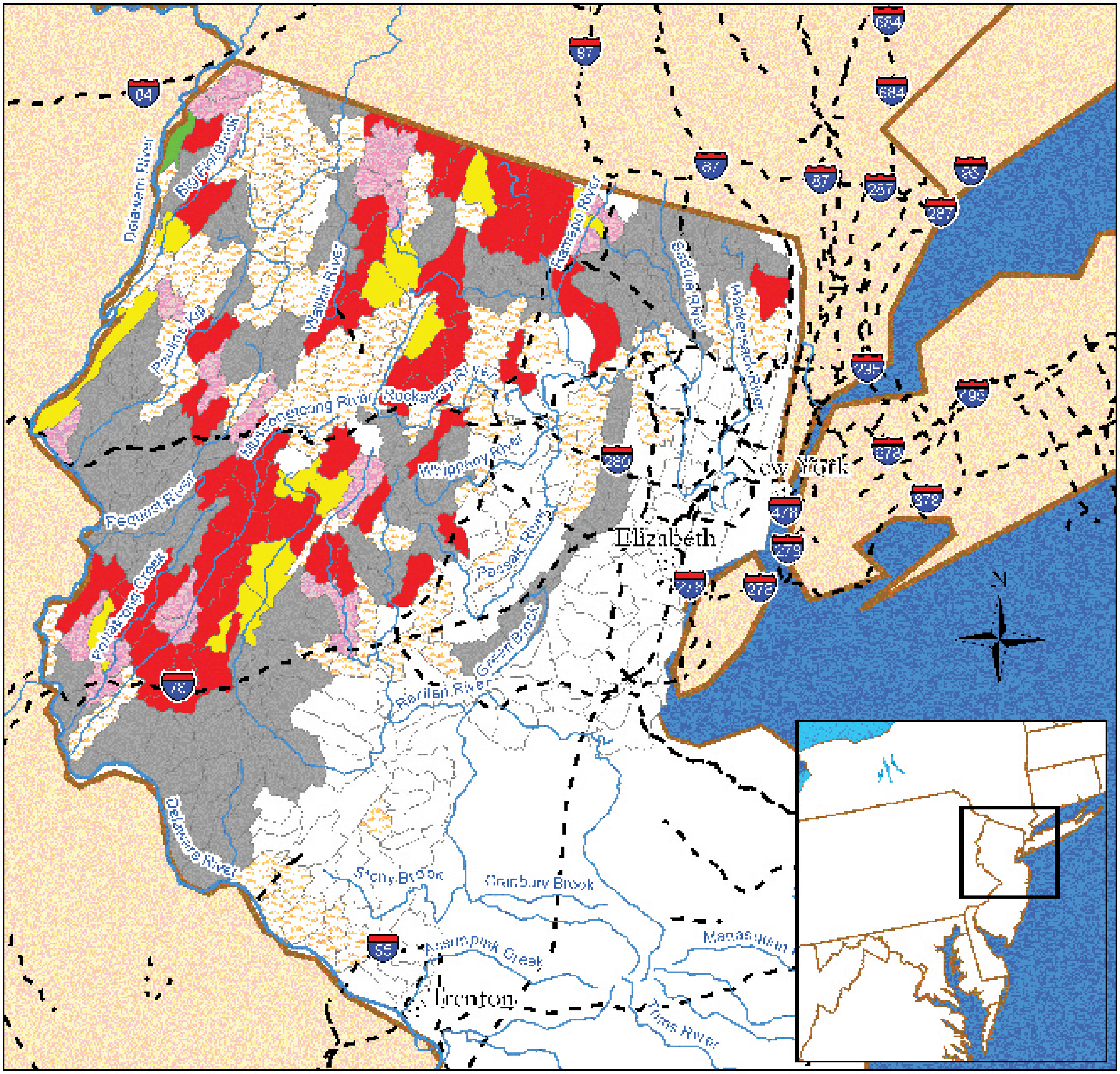
LEGEND
Sedimentation (Roads)
■ High Impact (8)
░ Medium Impact (106)
▨ Low Impact (11)
--- Interstate
⊙ Cities

Dam Inundation/Fragmentation Impacts to Brook Trout in New Jersey by Subwatershed



LEGEND
Inundation/Fragmentation (Dams)
■ High Impact (58)
░ Medium Impact (42)
▨ Low Impact (24)
--- Interstate
⊙ Cities

New Jersey Brook Trout Population Status by Subwatershed



Legend

- 🟩 Intact (1)
- 🟨 Reduced (14)
- 🟥 Greatly Reduced (44)
- ▨ Qualitative Presence (19)
- ▨ Extirpated (94)
- ▨ Unknown, No Data (76)
- ☐ Never Occurred (122)
- Rivers
- Interstate
- Cities

0 10 20
Miles

Map data derived from state and federal data and compiled in EBTJV assessment results titled, *Distribution, status, and perturbations to brook trout within the eastern United States, 2006.* Authored by Mark Hudy, US Forest Service; Teresa Thieling, James Madison University; Nathaniel Gillespie, Trout Unlimited; Eric Smith, Virginia Tech. Map created on 2/24/06 by Nathaniel Gillespie, Trout Unlimited.

New York:

In New York, 5% of watersheds that historically contained brook trout in streams and rivers remain intact, located primarily in portions of the Adirondacks and the Tug Hill Plateau. Western and south central New York have suffered the greatest losses of brook trout. Data gaps remain in the central part of the state from Albany to Syracuse. While many lakes and ponds still contain brook trout, losses have been substantial due to competition with non-native fish and acid rain.

Brook Trout Classifications	Number of Watersheds	Percentage of Watersheds
Intact (>90% habitat occupied)	25	5%
Reduced (50-90% habitat occupied)	63	11%
Greatly Reduced (<50% occupied)	149	27%
Present, Qualitative Data Only	106	19%
Extirpated	129	23%
Absent, Unclear History	0	0%
Unknown, No Data	89	16%
Total	561	100%

Population Status: New York's intact watersheds occur mostly in the Adirondack Park and along the Tug Hill Plateau. The Catskill Mountains support a block of reduced watersheds. The Finger Lakes, Southern Tier and Western New York have suffered the greatest losses, where brook trout live in small, fragmented populations in headwater streams. A large portion of central New York stretching from Albany to Syracuse and south to the Catskills lacks brook trout information. Although New York once boasted vast and famed lake fisheries for brook trout, only two of the 136 watersheds that historically supported lake populations remain intact today.

New York is the only state where subwatersheds are not delineated, and therefore this analysis was conducted on a watershed basis. Data was collected and analyzed separately for streams and rivers and for lakes and ponds.

Threats: High water temperature was listed as the top disturbance to stream populations of brook trout. High water temperatures are a common symptom of various land uses that remove streamside vegetation, particularly poor land management associated with agriculture (ranked fourth). Degraded riparian (streamside) habitat also contributes to increased water temperatures. In addition, regional experts noted the resurgence of beavers--now thriving without natural predators along waterways altered by forestry--as a source of warmer water.

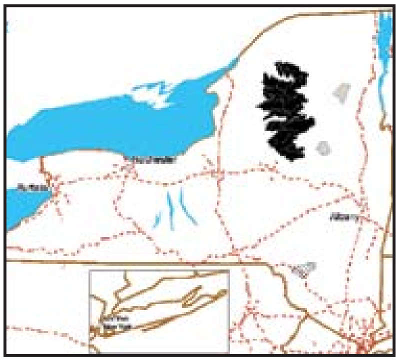
Non-native fish (specifically brown trout) were identified as the second largest stream disturbance. While non-native fish can out-compete brook trout, they are also more tolerant of warmer, more polluted waters that formerly supported brook trout.

The most widespread disturbance to New York's lake and pond populations is non-native fish--specifically smallmouth bass, largemouth bass, and other warm-water fish such as yellow perch and golden shiners. Acid deposition has a severe but regionalized impact on lakes, eliminating or reducing aquatic life in 23 Adirondack watersheds.

Many approaches exist to protect and restore New York's lake populations of brook trout. These include restricting illegal fish introductions into brook trout waters, increasing current monitoring programs, reducing acid deposition, liming acidified ponds and reintroducing native brook trout. Stream populations can benefit from building more partnerships among landowners, agencies and non-profit organizations to restore streamside trees and improve habitat for New York's state fish.

Disturbances (High or Medium)	Number of Watersheds	Percentage of Watersheds
High Water Temperature	282	60%
1 or More Non-Native Fish	245	52%
Brown Trout	218	46%
Poor Land Management	215	46%
Beavers	197	42%
Riparian Habitat	190	40%

Threats information based on professional opinion of regional experts. Figures do not add to 100% because zero, one, or multiple disturbances may occur in each watershed.



Acid Deposition Impacts to Brook Trout in New York by Watershed

LEGEND

Acid Deposition

■ High Impact (20)

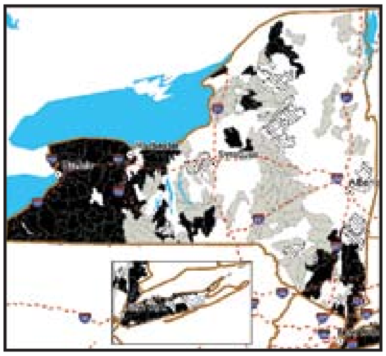
□ Medium Impact (2)

▧ Low Impact (1)

╍ Interstate

Lake

⊙ Cities



High Water Temperature Impacts to Brook Trout in New York by Watershed

LEGEND

High Water Temperatures

■ High Impact (164)

□ Medium Impact (118)

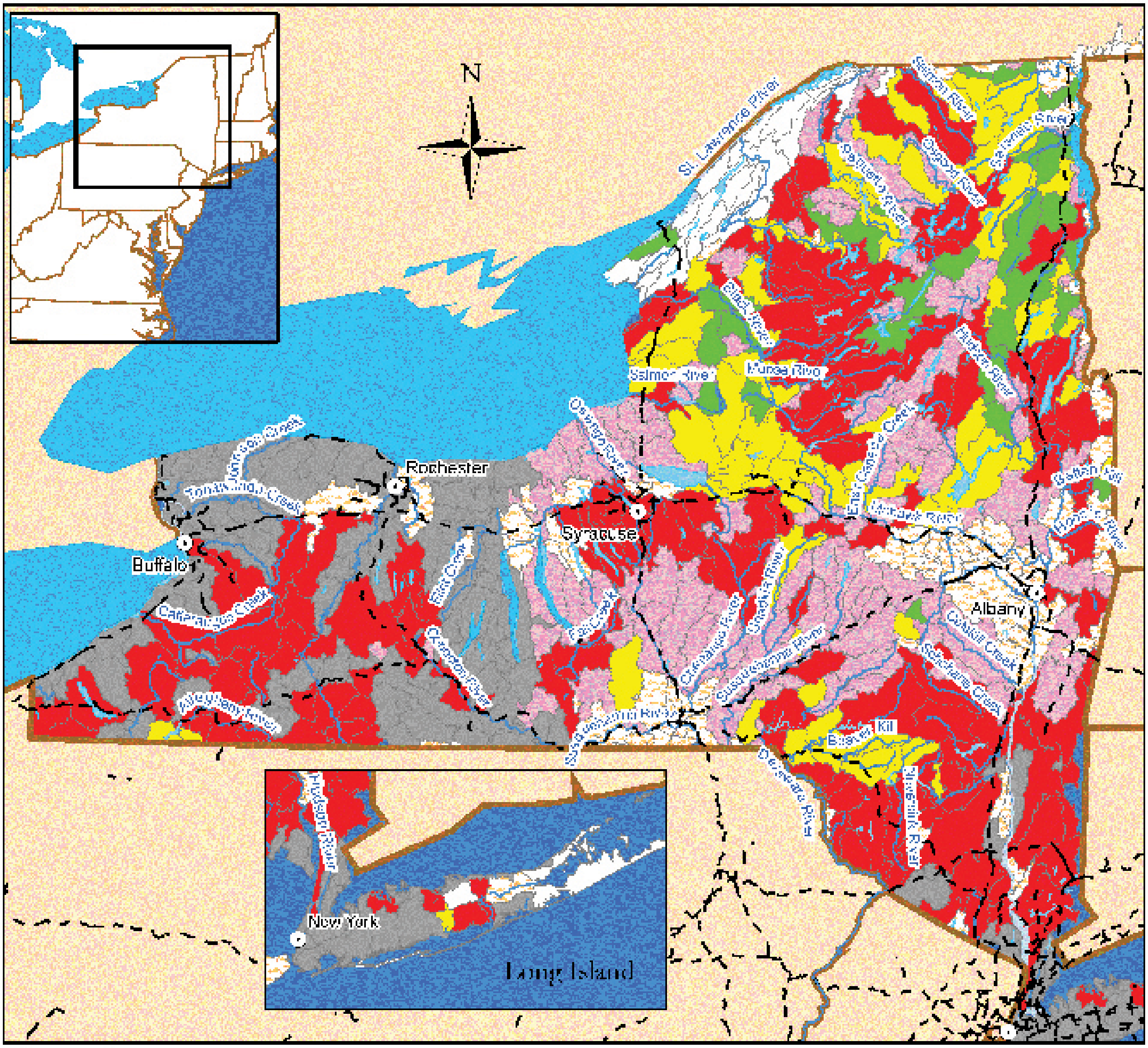
▧ Low Impact (31)

╍ Interstate

Lake

⊙ Cities

New York Brook Trout Population Status by Watershed



Legend

🟩 Intact (25)

🟨 Reduced (63)

🟥 Greatly Reduced (149)

▨ Qualitative Presence (106)

▩ Extirpated (129)

▤ Unknown, No Data (89)

— Never Occurred (36)

〰 Rivers

● Lake

- - - Interstate

● Cities

0 25 50
Miles

Map data derived from state and federal data and compiled in EBTJV assessment results titled, *Distribution, status, and perturbations to brook trout within the eastern United States, 2006.* Authored by Mark Hudy, US Forest Service; Teresa Thieling, James Madison University; Nathaniel Gillespie, Trout Unlimited; Eric Smith, Virginia Tech. Map created on 2/24/06 by Nathaniel Gillespie, Trout Unlimited.

Connecticut & Rhode Island:

While brook trout are still present within most Connecticut and Rhode Island subwatersheds, remaining brook trout populations are small and fragmented. Only one subwatershed remains intact for brook trout in these states. Fairly healthy subwatersheds are scattered within the Housatonic, Connecticut, Thames and Wood River drainages. Extirpated areas are concentrated in the southwest and near Hartford.

Brook Trout Classifications	Number of Subwatersheds	Percentage of Subwatersheds
Intact (>90% habitat occupied)	1	<1%
Reduced (50-90% habitat occupied)	19	10%
Greatly Reduced (<50% occupied)	134	69%
Present, Qualitative Data Only	3	2%
Extirpated	29	15%
Absent, Unclear History	1	<1%
Unknown, No Data	6	3%
Total	193	100%

Population Status: Brook trout remain distributed across much of Connecticut and Rhode Island, although in relatively depleted numbers. In Connecticut, brook trout currently occupy 80% of historical subwatersheds, but the vast majority of these support populations that are greatly reduced. Roughly 10% of the state's subwatersheds support reduced populations, and only one intact subwatershed remains. For the most part, Connecticut's brook trout populations are small and fragmented. These populations are located in the uppermost headwaters of stream systems and have disappeared from the larger river segments they once inhabited due to declines in water quality, increased water temperature, and displacement by non-native fish.

Rhode Island data is only partially available, and further data collection is needed to document the condition of its only native trout. The Wood River contains the healthiest known wild brook trout populations in the state.

Threats: Regional experts noted that nearly every subwatershed in Connecticut suffers from increased water temperatures. High water temperature is a symptom of a variety of human activities that alter streamside vegetation and change the pathways that water takes as it flows across the land and into streams. Urbanization, roads and dam fragmentation represent physical disturbances to brook trout habitat that lead to increased water temperature and decreased water quality.

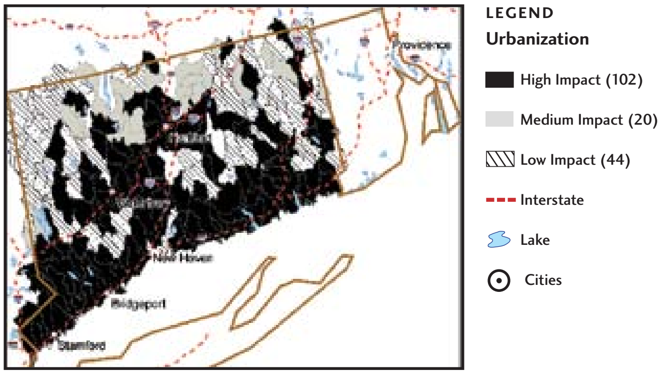
Like much of southern New England, past land uses have left a heavy footprint on Connecticut's landscape. Regional experts identified historical clearing of forests as a profound disturbance that continues to impact streams and water quality today. Urbanization impacts reflect the state's industrial past and current population and development issues. In addition, experts identified dams--many built more than a century ago--as a disturbance to brook trout habitat in almost half of the state's subwatersheds. No impact data was available for Rhode Island.

Water quality has improved across New England following the decline of historical agriculture, the enforcement of the Clean Water Act and the regrowth of forests. By protecting streams from poorly designed development, removing dangerous or nonfunctional dams and improving stormwater management, policy makers and resource managers can improve water quality and help rebuild intact brook trout populations over time.

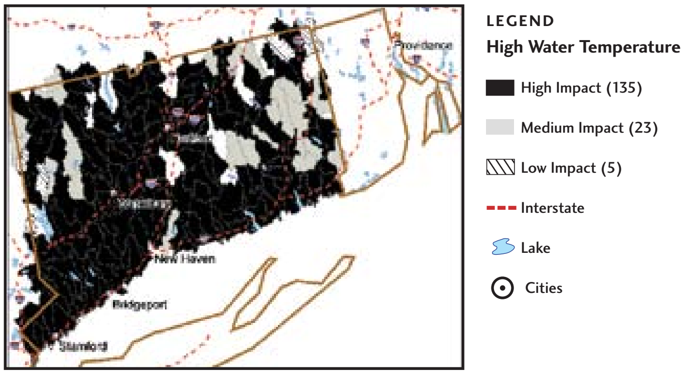
Disturbances (High or Medium)	Number of Subwatersheds	Percentage of Subwatersheds
High Water Temperature	165	94%
Historical Forestry	139	79%
Urbanization	122	69%
Road Sediment	122	69%
Dam Fragmentation	85	48%

Threats information based on professional opinion of regional experts. Figures do not add to 100% because zero, one, or multiple disturbances may occur in each subwatershed.

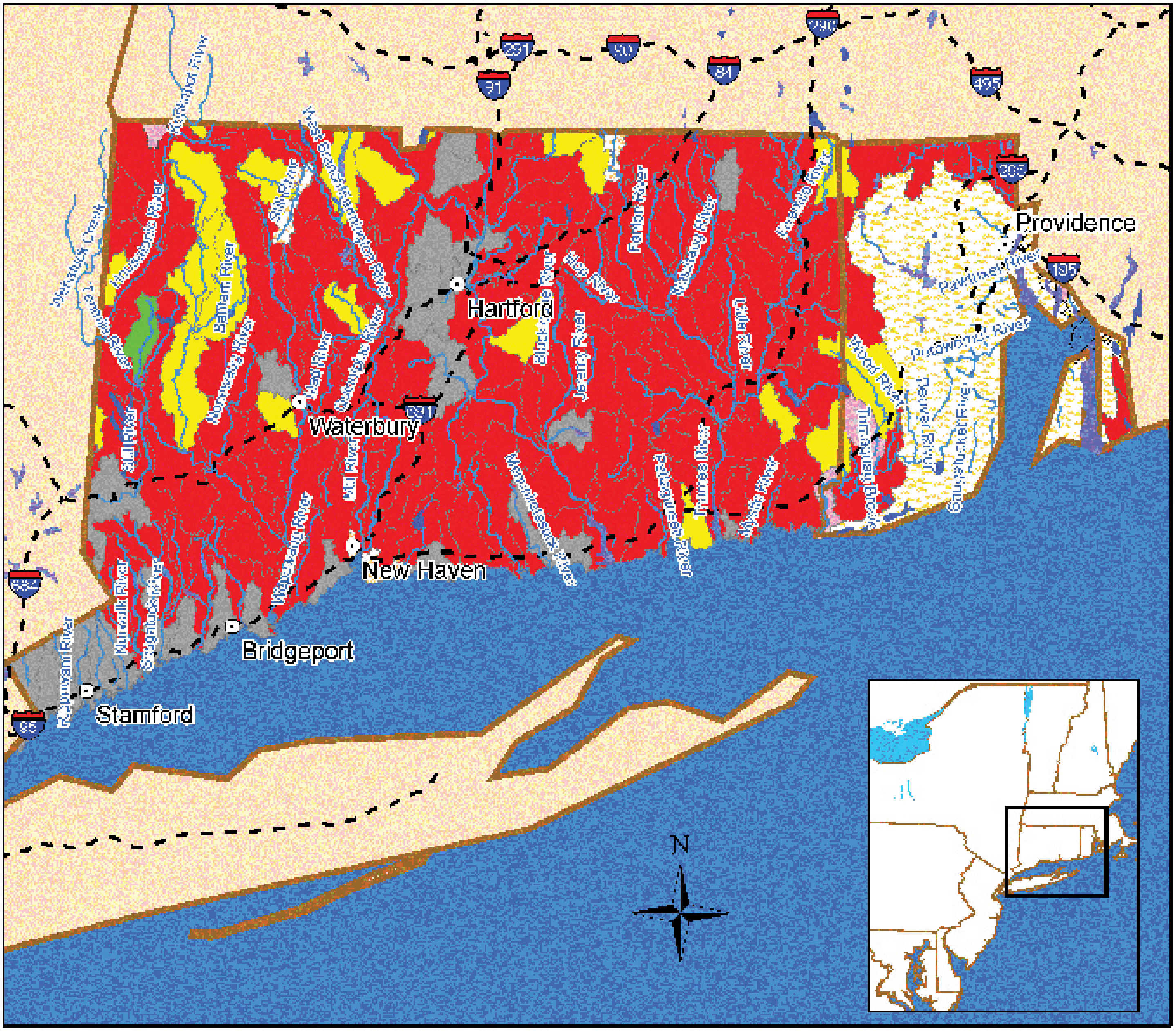
Urbanization Impacts to Brook Trout in Connecticut by Subwatershed



High Water Temperature Impacts to Brook Trout in Connecticut by Subwatershed



Connecticut & Rhode Island Brook Trout Population Status by Subwatershed



Legend

- 〰 Rivers
- 🟩 Intact (1)
- 🟨 Reduced (19)
- 🟥 Greatly Reduced (134)
- ▨ Present, Qualitative Data (3)
- ⬛ Extirpated (29)
- ▨ Absent, Unclear History (1)
- ▨ Unknown, No Data (6)
- ● Lake
- ‒ ‒ Interstate
- • Cities

0 12.5 25
Miles

Map data derived from state and federal data and compiled in EBTJV assessment results titled, *Distribution, status, and perturbations to brook trout within the eastern United States, 2006.* Authored by Mark Hudy, US Forest Service; Teresa Thieling, James Madison University; Nathaniel Gillespie, Trout Unlimited; Eric Smith, Virginia Tech. Map created on 2/24/06 by Nathaniel Gillespie, Trout Unlimited.

Massachusetts:

Western Massachusetts possesses the state's best remaining brook trout habitat, with some comparatively strong brook trout populations in the western Taconics and in parts of the Connecticut River watershed. Coastal "salter" brook trout survive in several subwatersheds along shores including Cape Cod and Martha's Vineyard. Brook trout have vanished from the greater Boston area. Data gaps exist in the central part of the state, while large portions of Eastern Massachusetts lack any population data.

Brook Trout Classifications	Number of Subwatersheds	Percentage of Subwatersheds
Intact (>90% habitat occupied)	1	<1%
Reduced (50-90% habitat occupied)	29	10%
Greatly Reduced (<50% occupied)	80	28%
Present, Qualitative Data Only	34	12%
Extirpated	20	7%
Absent, Unclear History	4	1%
Unknown, No Data	119	42%
Total	287	100%

Population Status: Less than 11% of the subwatersheds in Massachusetts support intact or reduced brook trout populations. These relatively healthy populations are located primarily in the Berkshire and Taconic mountains in the western part of the state, and within portions of the Hoosic, Deerfield and Westfield subwatersheds and several tributaries to the Connecticut River. In 28% of subwatersheds, brook trout are greatly reduced, occupying only isolated headwater stream sections. The sprawling Boston area has lost the greatest amount of brook trout habitat in the state.

Very little data is available for the eastern portion of the state south of Boston to Cape Cod. In addition, 12% of Massachusetts subwatersheds - largely in the central part of the state- have only qualitative data to document the presence of brook trout.

Threats: Massachusetts rivers and streams are heavily burdened by dams and roads. Regional experts identified dam fragmentation as a high or medium disturbance in 65% of all subwatersheds where brook trout status is known. Dams inundate habitat and increase water temperatures by slowing down flowing water and exposing it to the sun.

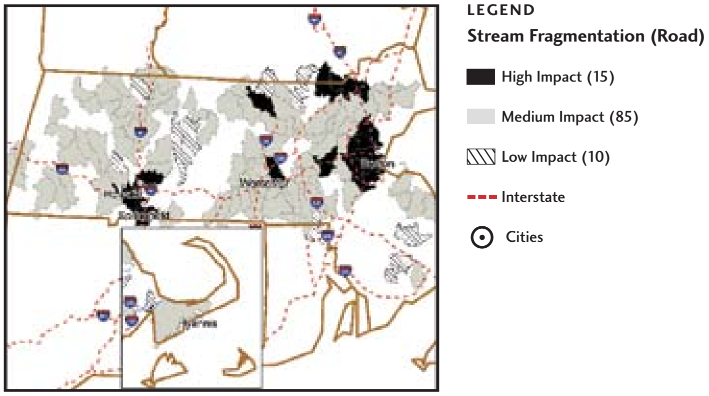
Dams and culverts often form barriers to fish movement, effectively cutting streams into biological fragments. Small, isolated populations of brook trout without connection to a larger population run the risk of vanishing over time as they succumb to natural flood and drought cycles. Because these fragmented populations are isolated from one another, if a population disappears, it cannot be reestablished by other fish from downstream. Removing or breaching unnecessary dams can restore a biological connection between isolated populations, reduce summer water temperatures and re-establish lost stream habitat.

Regional experts ranked stream fragmentation and sedimentation from roads as the second and third most common disturbances to brook trout habitat. In addition, streamside (riparian) and instream habitat degradation were listed as factors in over 50% of the state's brook trout subwatersheds where data is available. Instream habitat losses often result from gravel mining, flood control manipulation and loss of trees.

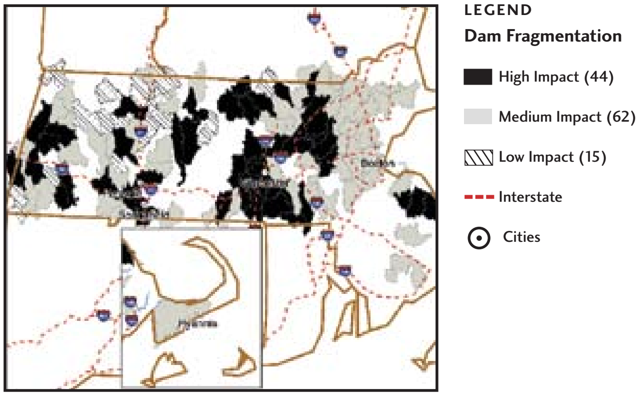
Disturbances (High or Medium)	Number of Subwatersheds	Percentage of Subwatersheds
Dam Inundation/Fragmentation	106	65%
Stream Fragmentation (Roads)	100	61%
Sedimentation (Roads)	96	59%
Riparian Habitat	93	57%
Instream Habitat	91	56%

Threats information based on professional opinion of regional experts. Figures do not add to 100% because zero, one, or multiple disturbances may occur in each subwatershed.

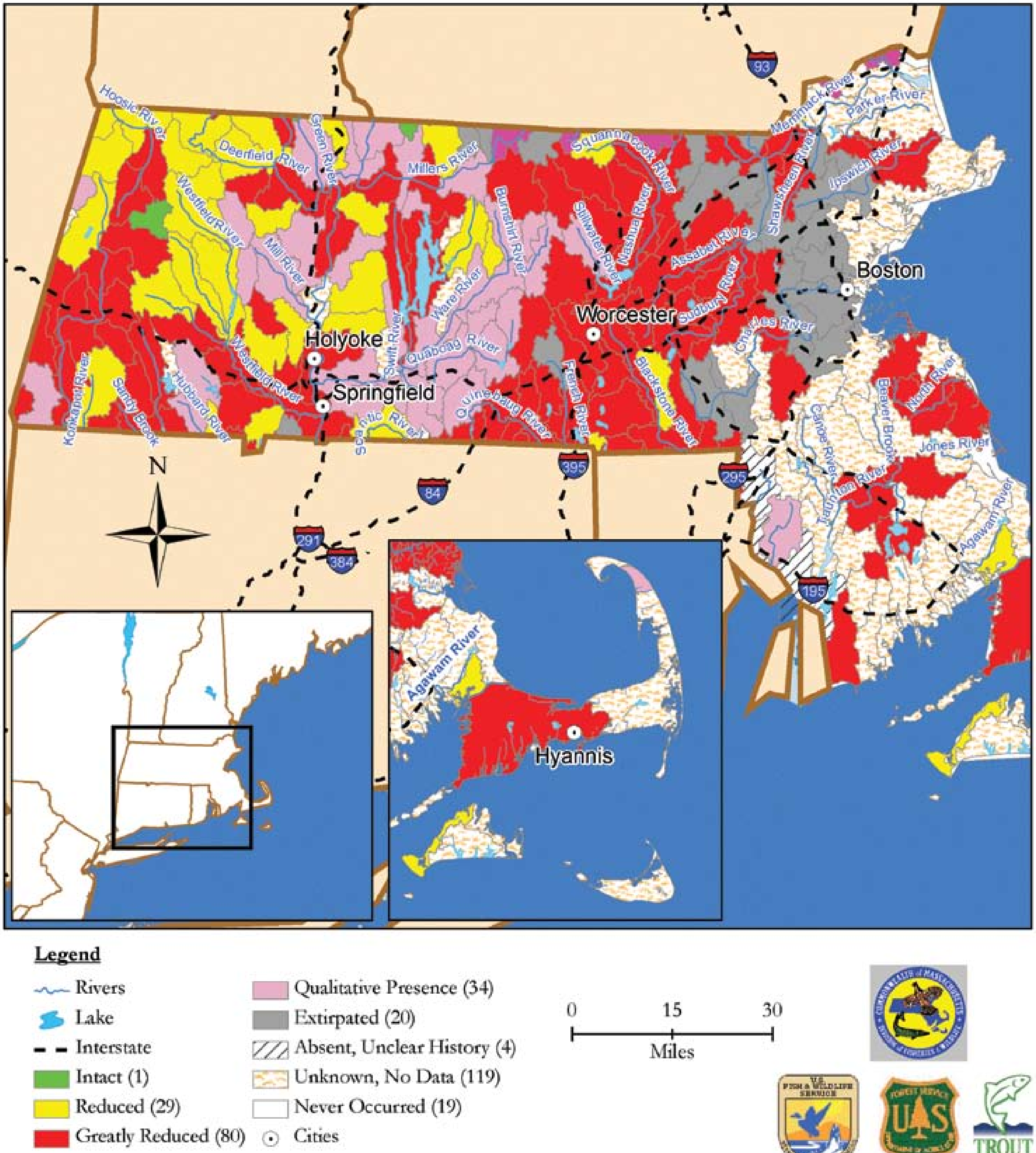
Stream Fragmentation (Road) Impacts to Brook Trout in Massachusetts by Subwatershed



Dam Fragmentation Impacts to Brook Trout in Massachusetts by Subwatershed



Massachusetts Brook Trout Population Status by Subwatershed



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Vermont:

Vermont boasts some of the largest concentrations of intact subwatersheds outside of Maine for wild, self-reproducing populations of brook trout. Most of these subwatersheds are located in the headwaters of the Batten Kill, White and East Branches of the Nulhegan and Passumpsic Rivers. While sedimentation and high water temperatures from roads and poor land management have degraded aquatic habitat in over half of the state, non-native fish have displaced brook trout from many of Vermont's streams and lakes.

Brook Trout Classifications	Number of Subwatersheds	Percentage of Subwatersheds
Intact (>90% habitat occupied)	33	14%
Reduced (50-90% habitat occupied)	64	27%
Greatly Reduced (<50% occupied)	86	36%
Present, Qualitative Data Only	20	8%
Extirpated	6	3%
Absent, Unclear History	0	0%
Unknown, No Data	31	13%
Total	**240**	**100%**

Population Status: Vermont's wild brook trout status is among the best in the East. Nearly 14% of the state supports intact habitat, and another 27% of the subwatersheds are reduced. These intact and reduced subwatersheds are centered in the Batten Kill and White River headwater regions, in several tributaries to Otter Creek, within and adjacent to the Green Mountain National Forest and within much of the sparsely populated Northeast Kingdom. The Green Mountain National Forest protects portions of five intact and 18 reduced subwatersheds. The remainder of Vermont's strong brook trout habitat is located on private land.

Over 35% of brook trout habitat is greatly reduced, concentrated in lower elevation areas on the west side of the state and within the White and Black River watersheds. Extirpated areas are concentrated in the lower reaches of the Winooski and Missiquoi rivers. Brook trout population data is lacking for 21% of the state, mostly located east of Rutland and along the Lake Champlain and New Hampshire borders.

Of the 45 subwatersheds that historically supported lake and pond populations of brook trout, only 2% of these subwatersheds remain intact, 31% are greatly reduced, 31% are extirpated, and in 29% status is unknown.

Threats: The impacts to brook trout in Vermont's streams relate to poor land management as well as to historical timbering and roadbuilding. Six of the top seven disturbances listed by regional experts relate specifically to increased sedimentation and water temperature. Loss of riparian (streamside) habitat and poor land management associated with agriculture are direct causes of higher water temperatures.

Often, a combination of several disturbances determines brook trout's ability to thrive or persist. Replanting trees and restricting livestock from sensitive streamside areas are examples of best management practices that may be used to reverse some of these impacts. Numerous state and federal agencies currently provide incentives to protect or restore streamside vegetation.

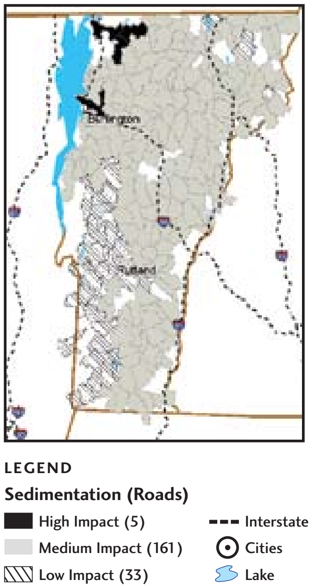
Non-native fish represent an additional threat to wild brook trout in Vermont. Following clearcutting in the 19th century, non-native fish were introduced to degraded waters where brook trout could no longer survive. These non-native fish are more tolerant of high water temperatures and competing species. As stream habitat has recovered, these non-native fish have spread into areas of high water quality where they frequently out-compete wild brook trout.

Regional experts documented that Vermont's lake populations of brook trout have suffered primarily from introductions of smallmouth bass and warmwater fish such as sunfish and yellow perch.

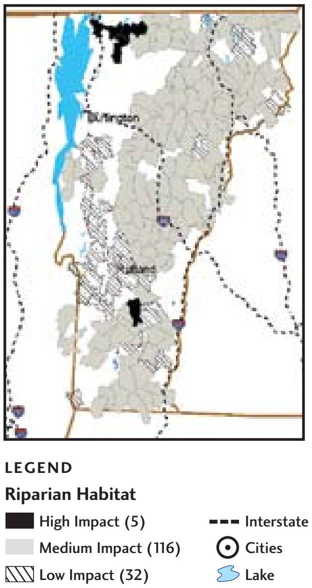
Disturbances (High or Medium)	Number of Subwatersheds	Percentage of Subwatersheds
Sedimentation (Roads)	166	79%
Historical Forestry	163	78%
Riparian Habitat	121	58%
Poor Land Management	121	58%
High Water Temperature	116	56%
1 or More Non-Native Fish	106	51%

Threats information based on professional opinion of regional experts. Figures do not add to 100% because zero, one, or multiple disturbances may occur in each subwatershed.

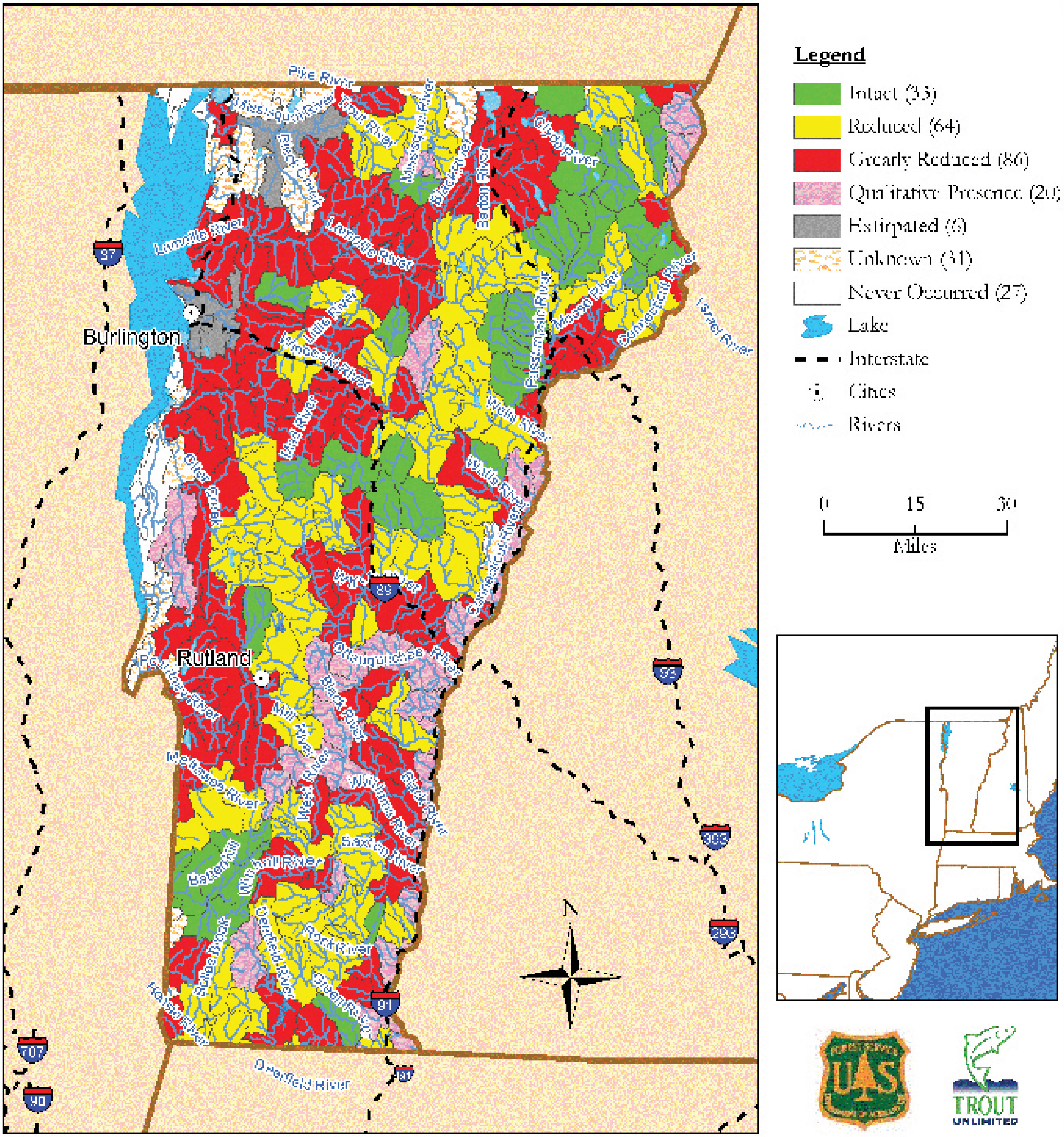
Road Sediment Impacts to Brook Trout in Vermont by Subwatershed



Riparian Habitat Impacts to Brook Trout in Vermont by Subwatershed



Vermont Brook Trout Population Status by Subwatershed



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New Hampshire:

The majority of New Hampshire lacks quantitative brook trout population data for streams, and brook trout status is unknown in the vicinity of Concord and Manchester. Much of northern New Hampshire maintains intact brook trout habitat, including portions of the White Mountains. Only 1% of the state's lake subwatersheds are known to be intact, while 90% of subwatersheds have no data on lake populations of brook trout.

Brook Trout Classifications	Number of Subwatersheds	Percentage of Subwatersheds
Intact (>90% habitat occupied)	21	7%
Reduced (50-90% habitat occupied)	13	5%
Greatly Reduced (<50% occupied)	13	5%
Present, Qualitative Data Only	195	70%
Extirpated	0	0%
Absent, Unclear History	0	0%
Unknown, No Data	37	13%
Total	279	100%

Population Status: In New Hampshire, 7% of subwatersheds are known to support intact, self-reproducing populations of brook trout. These subwatersheds (including the Upper Connecticut River system and the Magalloway, Dead Diamond and Swift Diamond Rivers) represent most of the intact brook trout habitat remaining outside of Maine. Portions of the White Mountain National Forest also support intact subwatersheds, although other areas are reduced or only quantitative data is available. Throughout the majority of the state (70% of subwatersheds), brook trout are known to be present, but insufficient scientific documentation prevents experts from classifying the status of the populations.

New Hampshire boasts over 279 subwatersheds that historically held lake populations of brook trout. For the majority of these subwatersheds (88%), brook trout population status is unknown. Only 1% of subwatersheds are documented as intact - where more than 90% of historical lake and pond habitat is currently occupied by wild, self-reproducing brook trout.

Threats: Like most of New England, New Hampshire suffers from a legacy of intensive timber cutting. Deforestation, associated sedimentation and channelization for log drives degraded stream habitat and depleted many brook trout populations. Regional biologists ranked road sedimentation as the number one threat to brook trout in New Hampshire. Road construction and poorly maintained roads can increase sedimentation and impair water quality. Non-native fish, particularly rainbow trout, were ranked as the second and third most widespread disturbances to brook trout statewide. Smallmouth bass pose a specific threat to lake and large river populations of native brook trout in this state.

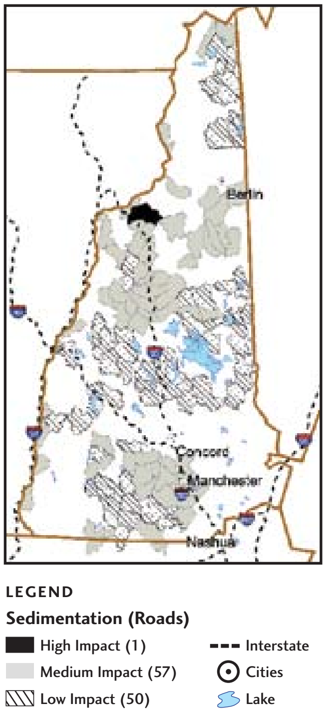
Acid deposition impacts are highest in the southern portion of the White Mountain National Forest and west of Concord and Manchester. In addition, poorly designed road culverts and dams fragment brook trout habitat and restrict fish movement.

State agencies have been working closely with fish habitat biologists to ensure that best design practices minimize the impact of road culverts on brook trout. In addition, habitat restoration work is ongoing in impacted areas to restore vegetation and instream habitat damaged by historical logging and log drives.

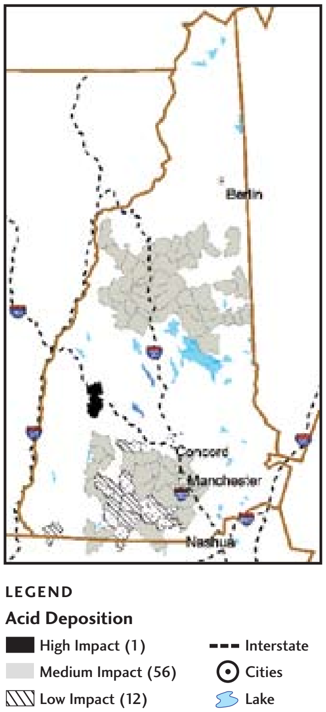
Disturbances (High, Medium or Low)	Number of Subwatersheds	Percentage of Subwatersheds
Sedimentation (Roads)	108	45%
1 or more Non-native Fish	95	39%
Rainbow Trout	74	30%
Acid Deposition	69	28%
Stream Fragmentation (Roads)	66	27%
Dam Indundation/Fragmentation	57	24%

Threats information based on professional opinion of regional experts. Figures do not add to 100% because zero, one, or multiple disturbances may occur in each subwatershed.

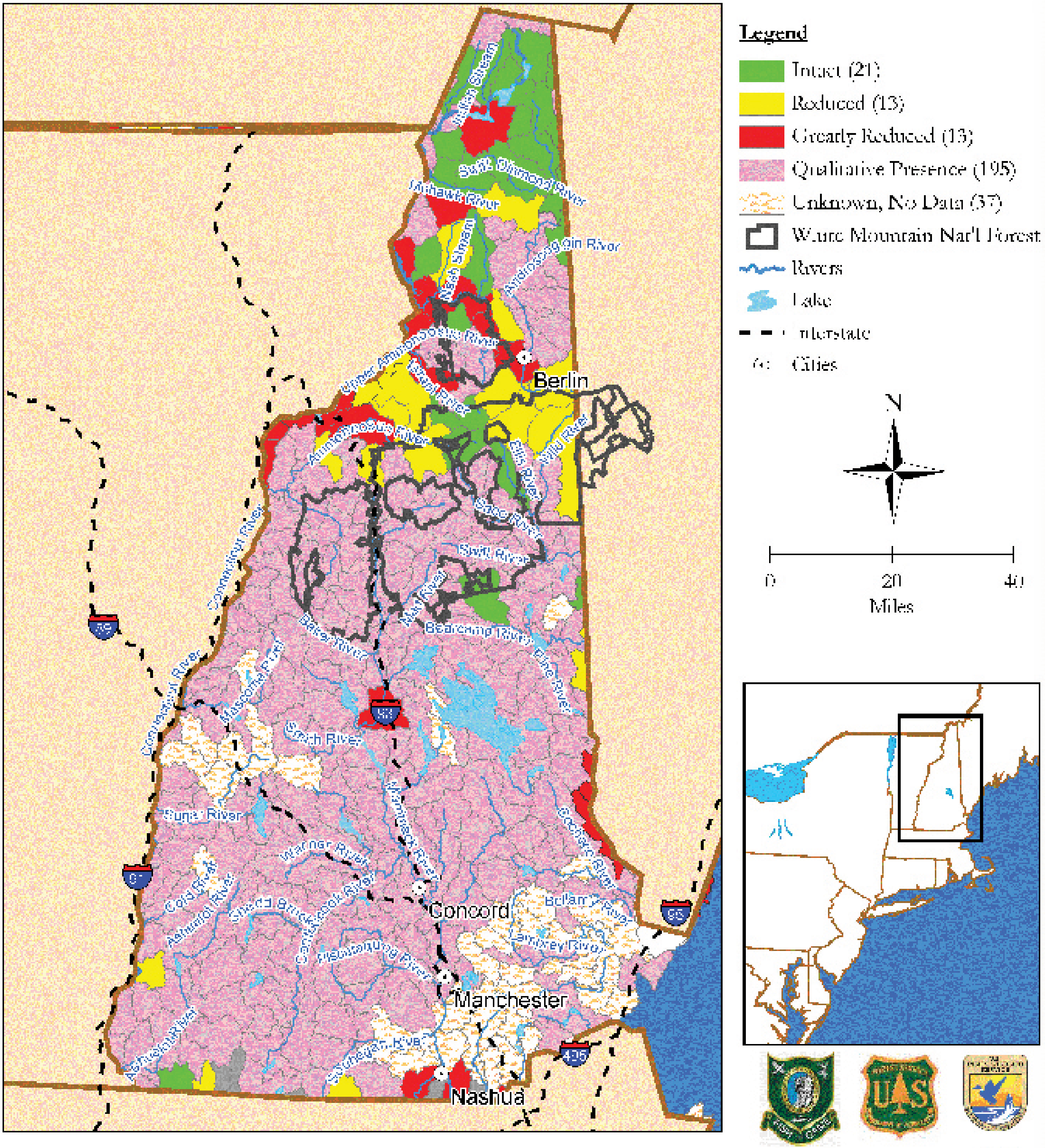
Road Sediment Impacts to Brook Trout in New Hampshire by Subwatershed



Acid Deposition Impacts to Brook Trout in New Hampshire by Subwatershed



New Hampshire Brook Trout Population Status by Subwatershed



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Maine Rivers & Streams:

Maine is the last true stronghold for brook trout in the eastern United States, with as many intact subwatersheds as all other states in the eastern range combined. Over 60% of Maine lacks stream population data for brook trout, although the majority is presumed to be intact. Southern Maine has experienced the greatest reduction in populations, mainly from dams, poor land management and fragmentation of stream habitat by roads and culverts. Sedimentation from certain forestry practices and poorly maintained roads impact brook trout populations in most of the northern half of the state.

Brook Trout Classifications	Number of Subwatersheds	Percentage of Subwatersheds
Intact (>90% habitat occupied)	147	14%
Reduced (50-90% habitat occupied)	76	7%
Greatly Reduced (<50% occupied)	88	8%
Present, Qualitative Data Only	658	64%
Extirpated	5	<1%
Absent, Unclear History	0	0%
Unknown, No Data	61	6%
Total	1035	100%

Population Status: Maine boasts more than twice the number of intact subwatersheds for brook trout populations as the other 16 states in the eastern range combined, yet almost 65% of the state has no quantitative data on brook trout status. Greatly reduced and extirpated subwatersheds are concentrated in the lower Kennebec and Androscoggin drainages, and in the Portland area south to the New Hampshire border.

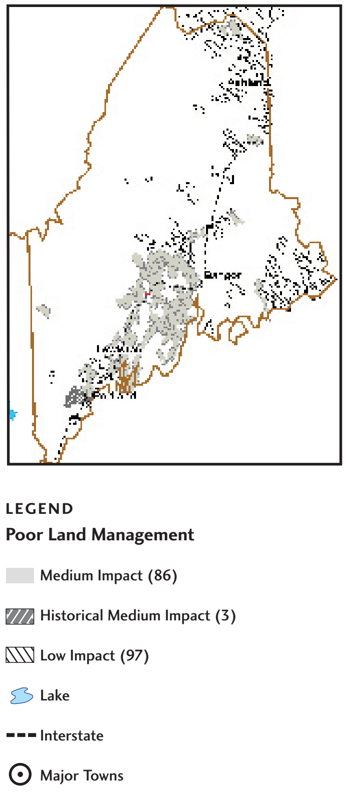
Disturbances (High or Medium)	Number of Subwatersheds	Percentage of Subwatersheds
Beavers	117	12%
Dam Inundation/Fragmentation	105	11%
Poor Land Management	86	9%
Forestry	82	8%
Stream Fragmentation (Roads)	73	7%

Threats information based on professional opinion of regional experts. Figures do not add to 100% because zero, one, or multiple disturbances may occur in each subwatershed.

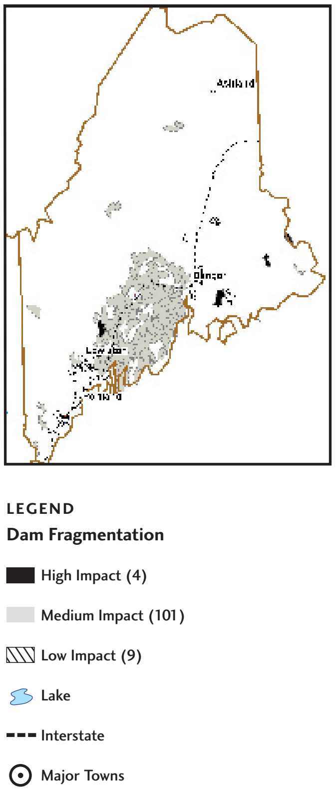
Threats: Generally, Maine's disturbances are relatively less severe than those in the rest of the eastern United States. Southern and coastal areas of Maine increasingly are experiencing urbanization pressures and associated water temperature and sedimentation impacts. Brook trout habitat between Portland and Bangor has been degraded by poor land management and dams. Poor land management practices also impact Down East and northern potato country. Regional experts noted that sedimentation and culvert fragmentation associated with forestry roads exert widespread but less severe impacts north and east of Bangor. Threats from non-native fish appear to be less common than in many other states, with impacts focused in the St. John's, Kennebec, Rapid and Penobscot River drainages.

While Maine's brook trout resources are superior to any other state in the eastern range, stream assessment and monitoring is needed to gauge the extent and status of brook trout populations and to benchmark conditions as Maine undergoes imminent land ownership changes. Excellent water quality, a high percentage of forest cover, and rivers unaltered by dams and development have allowed Maine's native brook trout to thrive in many subwatersheds. However, increasing residential development (particularly along the coast and in southern Maine) and the illegal introduction of non-native fish pose a threat to the best remaining brook trout habitat in the eastern United States.

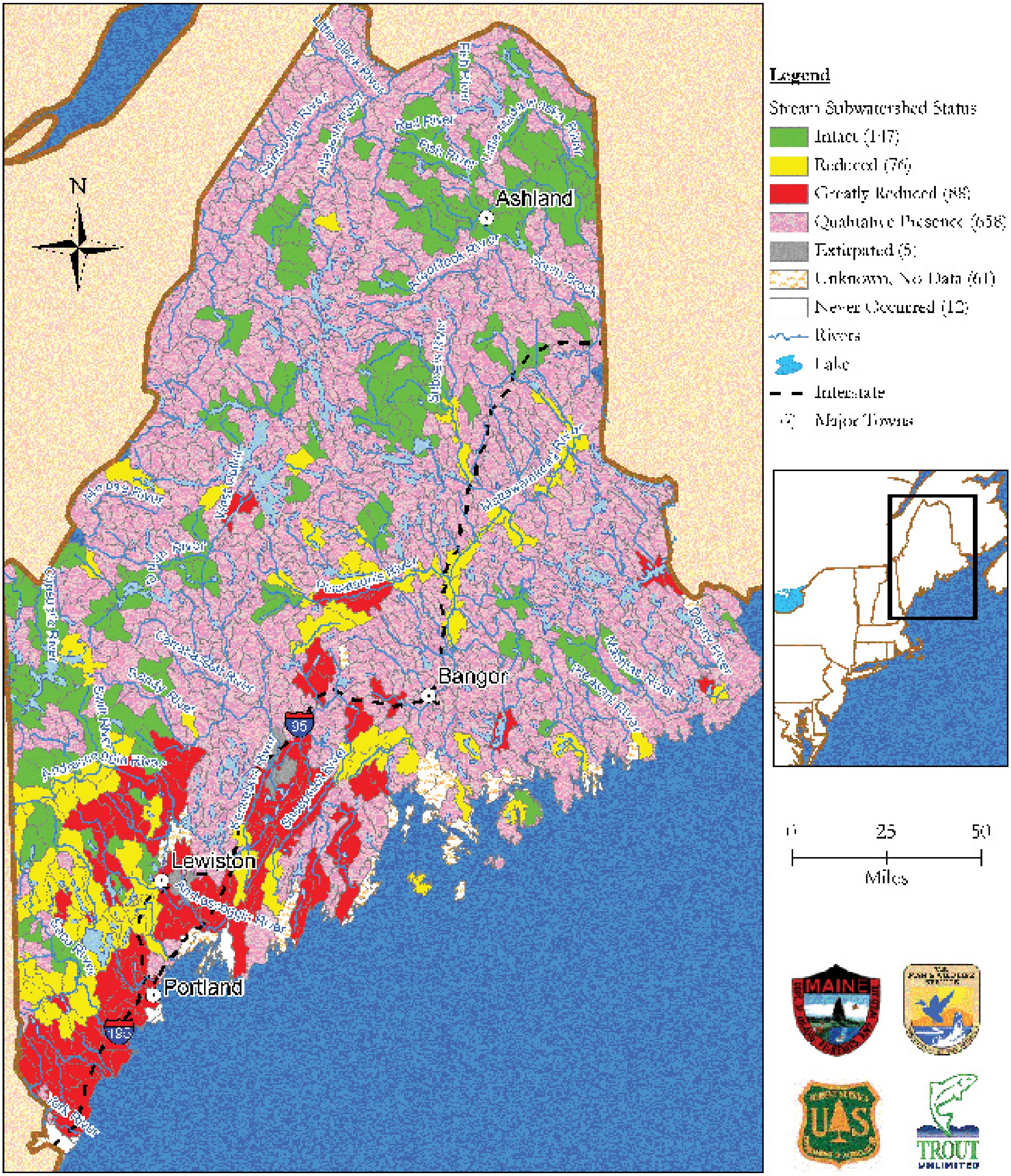
Poor Land Management Impacts to Brook Trout in Maine Streams & Rivers by Subwatershed



Dam Fragmentation/Inundation Impacts to Brook Trout in Maine Streams & Rivers by Subwatershed



Maine Stream and River Brook Trout Population Status by Subwatershed



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Maine Lakes & Ponds:

With 185 intact subwatersheds and many other healthy wild brook trout lakes and ponds, Maine represents the last stronghold for lake and pond brook trout populations. However, these fish populations are extremely vulnerable to introductions of non-native fish. Over 30% of Maine's subwatersheds are greatly reduced, primarily from smallmouth bass and other non-native fish.

Brook Trout Classifications	Number of Subwatersheds	Percentage of Subwatersheds
Intact (>90% habitat occupied)	185	21%
Reduced (50-90% habitat occupied)	35	4%
Greatly Reduced (<50% occupied)	323	37%
Present, Qualitative Data	89	10%
Extirpated	7	1%
Absent, Unclear History	0	0%
Unknown, No Data	235	27%
Total	874	100%

Population Status: Maine is the only state with extensive intact populations of wild, self-reproducing brook trout in lakes and ponds, including some lakes over 5,000 acres in size. Maine's lake and pond brook trout resources are the jewel of the eastern range: lake populations are intact in 185 subwatersheds (18% of the historical range), in comparison to only six intact subwatersheds among the 16 other states. Although brook trout historically thrived in most of Maine's ponds and lakes, over 30% of lake subwatersheds are greatly reduced. Data is not available for another 22% of the state's subwatersheds. Less than 1% of Maine's lake subwatersheds are extirpated.

Disturbances (High or Medium)	Number of Subwatersheds	Percentage of Subwatersheds
1 or More Non-Native Fish	222	25%
Smallmouth Bass	126	14%
Other Cool/Warmwater Fish (Perch, Sunfish, Muskellunge)	121	14%
Largemouth Bass	109	13%
Dissolved Oxygen	43	5%

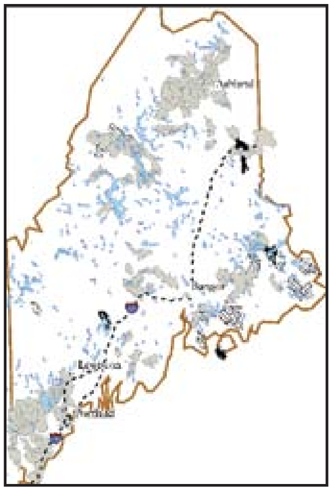
Threats information based on professional opinion of regional experts. Figures do not add to 100% because zero, one, or multiple disturbances may occur in each subwatershed.

Threats: Non-native fish are the dominant threat to Maine's lake and pond populations of brook trout. Many of these fish are illegally introduced by sportsmen who want to catch species of fish other than brook trout. The results are disasterous, as these fish outcompete brook trout within several years. Approximately 25% of Maine's lake subwatersheds that have brook trout data are known to suffer from impacts from non-native fish. Regional experts identified smallmouth bass, largemouth bass, and other cool/warm water fish such as yellow perch, sunfish and muskellunge as the most common introduced species limiting brook trout populations.

Habitat degradation plays only a minimal role in the overall status of Maine's brook trout lakes and ponds. Regional experts identified dissolved oxygen as an impact in less than 5% of Maine's subwatersheds. Dissolved oxygen levels typically are related to pollution of waters due to septic, agriculture and stormwater runoff pollution.

Maine is the only state in the eastern range with large lakes and large populations of self-reproducing brook trout, as well as hundreds of smaller lakes and ponds with self-reproducing brook trout populations. The deliberate spread of non-native fish threatens to eliminate these vulnerable brook trout lakes and ponds. Increased awareness and a more effective means of community self-policing to discourage people from intoducing non-native fish will prevent Maine's brook trout from suffering the same fate as other lake populations in the eastern United States.

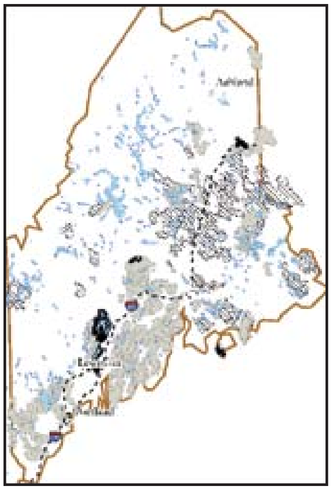
Other Coolwater/Warmwater Fish Impacts to Brook Trout in Maine Lakes & Ponds by Subwatershed



LEGEND
Perch, Sunfish, Muskellunge

■ High Impact (6)
□ Medium Impact (115)
▨ Low Impact (18)
Lake
--- Interstate
⊙ Major Towns

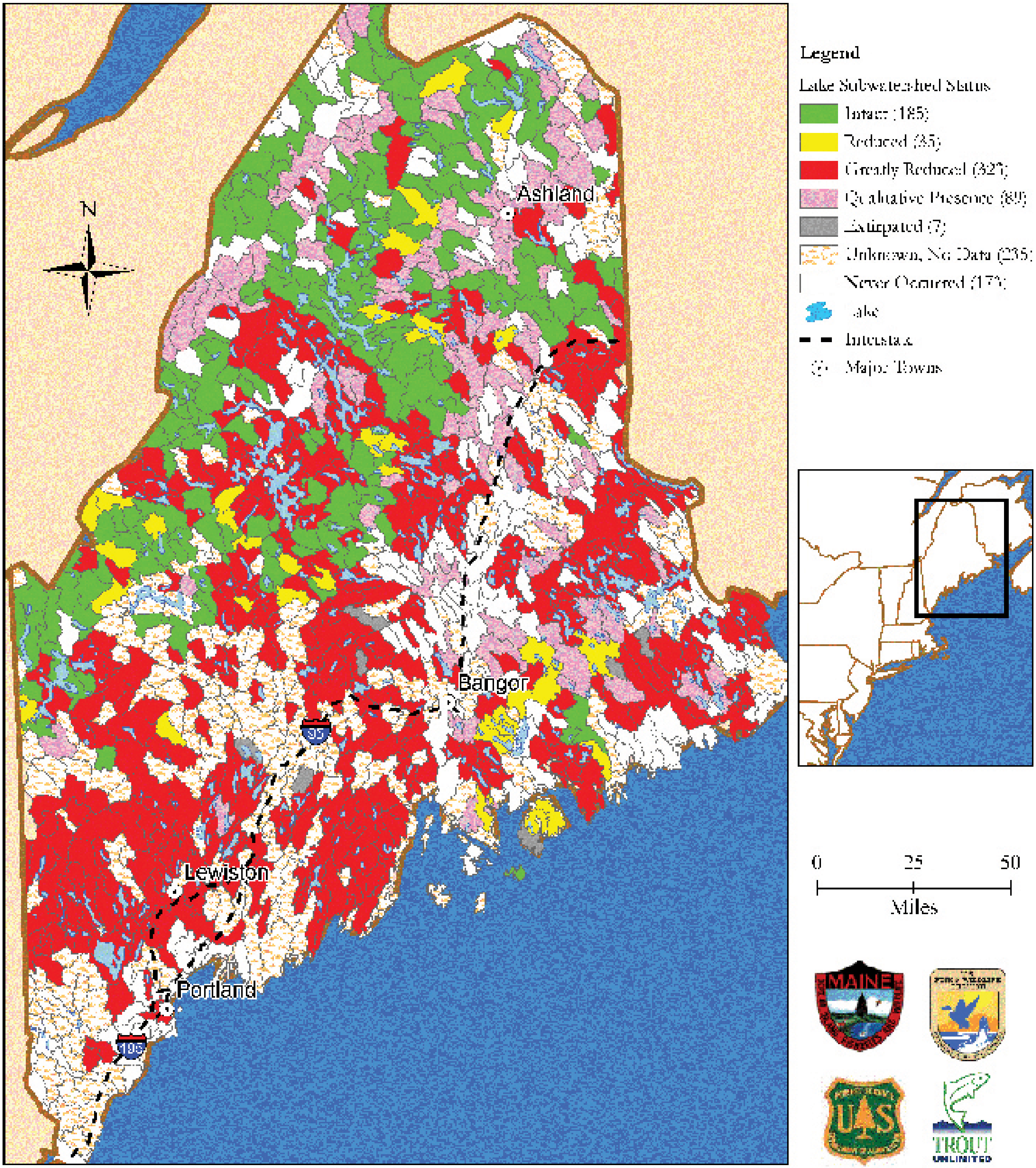
Smallmouth Bass Impacts to Brook Trout in Maine Lakes & Ponds by Subwatershed



LEGEND
Smallmouth Bass

■ High Impact (11)
□ Medium Impact (115)
▨ Low Impact (80)
Lake
--- Interstate
⊙ Major Towns

Maine Lake and Pond Brook Trout Population Status by Subwatershed



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This report is a product of the Eastern Brook Trout Joint Venture, which is a cooperative effort to develop and implement a conservation strategy for brook trout in the East. The following organizations have participated significantly in the EBTJV and the work that produced this report:

Federal Agencies:

U.S. Geological Survey
U.S. Fish and Wildlife Service
National Park Service
U.S. Forest Service
National Resources Conservation Service
Office of Surface Mining

State Agency Participants:

Maryland Department of Natural Resources
Virginia Department of Game and Inland Fisheries
North Carolina Wildlife Resources Commission
Tennessee Wildlife Resources Agency
South Carolina Department of Natural Resources
Georgia Department of Natural Resources
Maine Department of Inland Fisheries and Wildlife
New Hampshire Fish and Game Department
Vermont Department of Fish and Wildlife
Massachusetts Department of Fisheries, Wildlife and Environmental Law Enforcement
Rhode Island Department of Environmental Management
Connecticut Department of Environmental Protection
New York Department of Environmental Conservation
New Jersey Division of Fish and Wildlife
Pennsylvania Fish and Boat Commission
West Virginia Division of Natural Resources
Ohio Department of Natural Resources

University Participants:

Virginia Tech, College of Natural Resources
James Madison University, Department of Biology

Conservation Organizations:

Association of Fish and Wildlife Agencies
Trout Unlimited
Izaak Walton League
Trust for Public Land
The Nature Conservancy
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Chapter 14
Early Successional Forest Habitats and Water Resources

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Abstract Tree harvests that create early successional habitats have direct and indirect impacts on water resources in forests of the Central Hardwood Region. Streamflow increases substantially immediately after timber harvest, but increases decline as leaf area recovers and biomass aggrades. Post-harvest increases in stormflow of 10–20%, generally do not contribute to downstream flooding. Sediment from roads and skid trails can compromise water quality after cutting. With implementation of Best Management Practices (BMPs), timber harvests are unlikely to have detrimental impacts on water resources, but forest conversion from hardwood to pines, or poorly designed road networks may have long lasting impacts. Changing climate suggests the need for close monitoring of BMP effectiveness and the development of new BMPs applicable to more extreme climatic conditions.

14.1 Introduction

Watershed management requires understanding the tight linkages among vegetation, soils, and water quantity and quality. Because of these linkages, forest management activities that alter vegetation, such as creation of early successional habitats, have the potential to impact water resources. From a hydrologic standpoint, we define early successional habitats by the structural and functional attributes that are created by disturbance and influence hydrologic processes. Early successional habitats can be created by either natural disturbances (e.g., hurricanes, tornados, severe wildfires), or human-mediated intentional (e.g., forest cutting)

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and unintentional (e.g., invasive insects and disease introductions) disturbances (White et al., Chap. 3). Defining structural attributes of early successional forests include low leaf, stemwood and sapwood areas, high forest floor mass and coarse woody debris, and a high proportion of fast-growing, shade intolerant species (Keyser, Chap. 15). Defining functional attributes include high leaf-level C gain and low water use efficiencies, rapid organic matter decomposition, and accelerated nutrient cycling and accumulation (Keyser, Chap. 15). Although early successional forest attributes can be maintained with repeated disturbances, these attributes more often are transitional and recovery to pre-disturbance conditions occurs quickly (e.g., leaf area) or over several decades (e.g., species composition). Where disturbances are particularly severe, such as road building or loss of a dominant overstory species, structural and functional attributes may never recover to pre-disturbance conditions (Ellison et al. 2005). Combined, these changes in structural and functional attributes can impact water resources, and land managers need to consider those impacts when managing forests for multiple benefits. In particular, forest harvesting (with and without species conversion) and associated forest operations have the potential to substantially alter both water quantity and quality; in some cases, these changes persist long-term. In short, good land management is good watershed management.

Our understanding of the changes in water resources associated with creating early successional habitats is largely derived from a long history of paired watershed studies that have examined long-term streamflow and water quality responses to forest cutting (Calder 1993; Stednick 1996; Jones and Post 2004; Brown et al. 2005). Paired catchment studies have been critical to understanding how land management and other disturbances affect streamflow and quality. Accurate measurement of streamflow is at the core of paired watershed studies and this typically requires installation of a weir at the watershed outlet (Reinhart and Pierce 1964). Streamwater quality can be measured directly for some parameters (e.g., turbidity, pH, temperature, conductivity) using automated sensors, or water samples can be analyzed in a laboratory for these and other parameters such as nutrient concentration. The primary goal of the paired catchment method is to isolate streamwater response to cutting by accounting for the influences of climate or other factors. Using a paired untreated watershed that serves as a reference, streamflow response to cutting can be determined by examining the difference between expected streamflow (e.g., what would be expected if the watershed had not been treated) from observed streamflow. When measured streamflow differs from expected, the inference is that the treatment alone resulted in the streamflow response. Catchment scale manipulations at experimental watersheds such as the Coweeta Hydrologic Laboratory in the Southern Appalachians of North Carolina, the Fernow Experimental Forest in the Central Appalachians of West Virginia, and Hubbard Brook Experimental Forest in New Hampshire involve various intensities and types of management activities, as well as variation in watershed characteristics such as aspect, elevation, and size (Adams et al. 2008). These long-term watershed studies provide a powerful database from which we can examine the effects of managing for early successional habitats on streamflow amount, timing, and quality.

Annual streamflow generally increases for the first few years after forest canopy removal,, but the magnitude, timing, and duration of the response varies considerably among ecosystems. Using data from water yield studies across the globe, a general model suggests that for each percent of the forest removed streamflow increases 2.5–3.3 mm (Calder 1993; Stednick 1996); however, general models typically explain less than 50% of the variation of the streamflow increase (Stednick 1996) due to high variability in stand structure, pre- and post-harvest species composition, and the interaction between vegetation and climate. In some cases, streamflow returns to pre-harvest levels within 10–20 years. In others, streamflow remains higher, or can even be lower than pre-harvest flow, for several decades after cutting. This wide variation in temporal response patterns is attributable to the complex interactions between climate and vegetation, which can vary considerably from dry to wet to snow-dominated climatic regimes, and with differences in vegetation structure and phenology (coniferous vs. deciduous forest) (McNab, Chap. 2).

While gauged watershed studies provide the foundation for quantifying streamflow responses to forest disturbances, process-level studies are required to fully understand the structural and functional attributes that regulate the magnitude and duration of responses. For example, timber harvest simultaneously alters forest structure by reducing leaf area index, interception surface area, and vegetation height. Harvesting also alters forest function by changing the relative abundance of plant species (Loftis et al., Chap. 5; Elliott, Chap. 7), and the physical environment by changing the energy balance, wind environment, hydrologic flowpaths, and soil temperature and moisture. The topographic/edaphic complexity and high vegetation diversity of forest ecosystems in the Central Hardwood Region is likely to result in a wide range of streamflow response patterns. A more in depth understanding of the factors regulating these response patterns can help managers create and maintain early successional habitats and protect or enhance water resources.

Water quality can also be substantially affected by management activities that create early successional habitats and can have detrimental impacts on aquatic habitats and organisms (Moorman et al., Chap. 11). Research indicates that the harvest of forest biomass in itself has little or no measureable impact on sediment yield. Instead, the primary factors that determine sediment yield are the forest operations required to remove logs, such as roads and skid trails, and the implementation and effectiveness of Best Management Practices (BMPs) that either minimize erosion or prevent sediment from reaching the stream. Stream nutrients can also be impacted by creating and maintaining early successional habitats; however, response magnitude and duration vary considerably among chemical constituents, post-disturbance successional dynamics, and other silvicultural practices such as the use of herbicides or fertilizers.

In this chapter we focus on the first several years after harvesting to assess potential impacts of using forest harvests to create early successional habitats on water resources. To provide examples and illustrate concepts, we use data primarily from long-term studies in the Southern Appalachians, but also include and integrate results of studies from watershed experiments in other areas of the Central Hardwood Region. In addition, we include a discussion of the potential implications of climate change and how associated changes in precipitation regimes might interact with early successional habitats.

14.2 The Hydrologic Budget of Forested Watersheds

The three main components of the hydrologic budget of forested watersheds are **inputs** in the form of rain, snow, and ice (P); **outputs** in the form of transpiration, canopy interception, and soil and forest floor evaporation (evapotranspiration, ET), and groundwater recharge and streamflow (RO or runoff); and change in **soil water storage** (S). Thus, the hydrologic budget can be expressed in terms of a simple mass balance equation: $RO = P - ET \pm S$. Over the long-term, changes in soil water storage (S) are assumed to be negligible so that the storage component of the budget is usually ignored.

Understanding components of the water budget is useful for interpreting and predicting potential impacts of creating and maintaining early successional habitats. ET is the primary component influenced by forest cutting. However, significant alterations to hydrologic flowpaths due to compaction, roads, and other physical changes can influence runoff processes as well, especially stormflow. Timber harvesting alters ET by changing forest structure and function, and the micrometeorological factors that drive transpiration and evaporation. Structural changes include less leaf and stem surface area, and change in the distribution and arrangement of branch surface area. A major functional change that ensues when shifting from mature trees to seedlings, sprouts, and herbaceous vegetation is a decrease in abundance of plant species with conservative water use, resulting in increased transpiration per unit leaf area (Wallace 1988). The vegetation layer can also be more coupled to the atmosphere after forest harvest, thus changing energy balances and wind profiles (Swift 1976; Swank and Vose 1988). For example, Sun et al. (2010) found that net radiation of an 18-year old loblolly pine plantation was 20% higher than a younger stand (4–6 year old) in on the Coastal Plain of North Carolina, resulting in a 25% higher ET in the former.

14.3 Streamflow Responses to Forest Removal

14.3.1 *Amount and Timing*

Forest harvesting increases annual streamflow in almost all cases in the Central Hardwood Region (Jackson et al. 2004). For example, average increases (% increase relative to that expected based on flow in a reference watershed) in water yield for the first 2 years after cutting ranges from 9.1% at Hubbard Brook in New Hampshire, 14.3% at the Fernow in West Virginia, and 23.0% at the Coweeta Hydrologic

Table 14.1 Post-treatment streamflow response expressed as a percentage increase relative to expected streamflow (adapted from Vose et al. 2010)

Experimental forest	Average annual response (first 2 years post-cut)	Minimum	Maximum
Coweeta, NC ($n=6$)	23.0	10.3	44.1
Fernow, WV ($n=3$)	14.3	10.8	18.2
Hubbard Brook, NH ($n=3$)	9.1	1.7	18.9

Laboratory in North Carolina (Table 14.1). Comparing clearcut harvests with and without BMPs in hardwood forest in eastern Kentucky, Arthur et al. (1998) found a 138% (without BMPs) and a 123% (with BMPs) increase in streamflow during the initial 17 month post-cutting period. Water yield was still 15 to 12% greater 8 years after cutting for the BMP and without BMP watersheds, respectively (Arthur et al. 1998). Differences among regions are likely the result of a complex array of factors, but syntheses of worldwide data from watershed experiments suggest that absolute increases after cutting are greatest in high rainfall areas (Bosch and Hewlett 1982; Swank and Johnson 1994). Other factors include soil depth, the proportion of the annual water budget accounted for by ET, and annual snow fall. The amount of steamflow response is greatest during the first few years following treatment and can be estimated for upland hardwood forests using a model (Douglass and Swank 1975) where first year streamflow increase (water yield) is predicted as a function of the amount of basal area removed and an index of solar radiation inputs:

$$\text{Yield} = 0.00224 * (\text{BA} / \text{PI})^{1.4462},$$

where

Yield = first year increase in streamflow (cm),
BA = amount of basal area removed (%), and
PI = solar insolation index.

Highest yields are observed when 100% of the forest is harvested on north facing slopes. On south or west facing slopes where solar radiation inputs are greater, first year responses are lower because ET on harvested south facing slopes is not as responsive to the increased energy load as ET on harvested north facing slopes. The model also includes an equation to predict the exponential decline in streamflow response as the forest re-grows and LAI recovers (Swank and Douglass 1975). Applications of the model indicate good performance in the Southern Appalachians (Swank and Johnson 1994; Swank et al. 2001) and other eastern deciduous and coniferous forests (Douglass and Swank 1975; Douglass 1983).

Forest cutting can also impact streamflow timing throughout the year and alter storm hydrographs. For example, in areas with high snowfall and shallow soils, cutting increases the proportion of annual streamflow in the spring and summer months due to faster snowmelt and reduced transpiration. In areas with deeper soils and higher precipitation, typical of the Southern Appalachians, flow increases are greatest in the late summer and fall, and may extend into the winter months (Swank and Johnson 1994). For example, on a south facing clearcut watershed in the Southern Appalachians, streamflow increased by approximately 48% during August through October, a time when flows from mature forests are typically lowest (Swank et al. 2001). Storm hydrographs (i.e., a graphical analysis of stream flow vs. time during and after storm events) can also be impacted by cutting and the effects of timber harvesting on flooding have been a focus of intense debate and research for the past several decades (Lull and Reinhart 1972; Andreassian 2004; Eisenbies et al. 2007). Flooding is defined by hydrologic events that exceed bankfull. The linkage between timber harvesting, storm hydrographs, and flooding is complex, and can be better understood by examining the components of stormflow, and then dissecting how forest harvesting influences these components. Streamflow is comprised of baseflow and stormflow, with the latter being described by both the magnitude (peakflow) and duration (stormflow volume). Flooding occurrence and severity is determined largely by peakflow (essentially analogous to stage or the height of the stream) and stormflow volume (the amount of flow contributed by the storm). In forests of the Central Hardwood Region, peakflow and stormflow volume are primarily affected by forest operations that create soil disturbances that alter stormflow pathways; chief among these operations is the road network. For example, in the Southern Appalachians, stormflow volume was nearly double on a watershed logged with a high road density (Douglass and Swank 1976) compared to a watershed logged with a low road density (Swank et al. 2001). However, increases were still relatively minor (10% increase for the low road density watershed versus 17% increase for the high road density watershed). Peak discharges increased on the low road density watershed by up to 15% (Swank et al. 2001). In other sites where trees were felled, but no material removed and no roads were built, peakflow rates increased very little over all (<7%) although stormflow volume increased by 11% (Hewlett and Helvey 1970). In West Virginia, peak discharges after logging were up to four times greater during the growing season (Patric and Reinhart 1971) and they were up to 30% greater after cutting in New Hampshire (Hornbeck 1973).

If BMPs are implemented, most of the physical impacts related to harvest soil disturbances (e.g., skid trails, landing decks, etc.) are short-lived and have little impact on flood risk over the long-term. In contrast, construction of roads and associated engineering related to road surfacing, drainage, culvert design and location are much longer lasting. Depending on the design and surface area impacted, these can permanently alter hydrologic flow paths and storm hydrographs. In short, road design needs to focus on "disconnecting" the surface water draining from the road network to the stream network. Analyses of the impacts of cutting on downstream flooding suggests that many extreme flood events are unrelated to forest cutting and associated road networks and skid trails. Instead, they are primarily determined by storm size and intensity (Perry and Combs 1998; Kochendorfer et al. 2007) and occur regardless of forest management activities.

14.3.2 Duration of Streamflow Response

Among the biological and physical process changes that occur with timber harvest, the duration of streamflow response primarily depends on how quickly leaf and sapwood area recover, and the physiological and structural characteristics of the tree species that occupy the site after the cutting. Long-term streamflow responses for six watersheds in the Southern Appalachians illustrate the temporally variable nature of the response. The response depends on both the forest management objective (e.g., thinning, species conversion, clear cut, etc.,) and how the resulting vegetation responds to climate (Fig. 14.1). Few watershed treatments show no effect (e.g., zero line represents no difference between observed and expected flow based on flow from the reference watershed); and more importantly, few of the watersheds have returned to expected levels after 20 years. For example, where timber harvesting was followed by a species conversion (in this case, from deciduous hardwood to conifer, Fig. 14.1a–b), annual streamflow returned to reference levels after approximately 10 years, marking the point in time when canopy closure was complete. Thereafter, streamflow has been about 25% lower on the conifer dominated watershed (relative to the hardwood reference watershed) due to higher interception and year round transpiration by conifers (Swank and Douglass 1974; Ford et al. 2011).

Variation in sapwood area and species composition among hardwood species during succession can also play an important role in determining the magnitude and timing of streamflow responses after cutting (Ford et al. 2011). For example, transpiration rates for a given diameter yellow-poplar (*Liriodendron tulipifera*) are nearly twofold greater than hickory (*Carya* spp.) and fourfold greater than oaks (*Quercus* spp.). Yellow-poplar transpiration and stomatal conductance rates are also much more responsive to climatic variation compared to oaks and hickories (Ford et al. 2011) (Table 14.2). Xylem anatomy and resulting sapwood area are important determinants of stand transpiration (Wullschleger et al. 2001). For example, transpiration of trees with diffuse-porous, ring-porous, semi-ring-porous, and tracheid xylem anatomies vary more among these three xylem types than they do within a type by species (Fig. 14.2). Diffuse ring porous species have greater sapwood area than ring- or semi-ring porous species and as sapwood area increases, potential water transport increases (Enquist et al. 1998; Meinzer et al. 2005). Hence, if the early successional stand is dominated by diffuse porous species such as yellow-poplar, black birch (*Betula lenta*), or red maple (*Acer rubrum*), we would expect that growing season transpiration in an average year to be much greater (and hence, lower streamflow) than stands dominated by ring-porous species such as oaks or hickories, and likewise be more responsive to climatic variation. In most cases, post-harvest or post-disturbance vegetation succession in the Appalachians is a complex mix of species in both space and time (Elliott and Vose 2011) which makes simple extrapolations difficult. For example, as eastern hemlock (*Tsuga canadensis*) declines and its basal area is reduced by attack from an invasive exotic insect, black birch, a diffuse-porous sapwood species, is dominating early successional trajectories of leaf and sapwood area response (Orwig et al. 2002). This shift in species composition has the potential to increase transpiration by 30% (and thus correspondingly decrease streamflow) (Daley et al. 2007).

To fully understand and predict how post-harvest shifts in the relative abundance of tree species regulate streamflow response (e.g., to explain the variation shown in

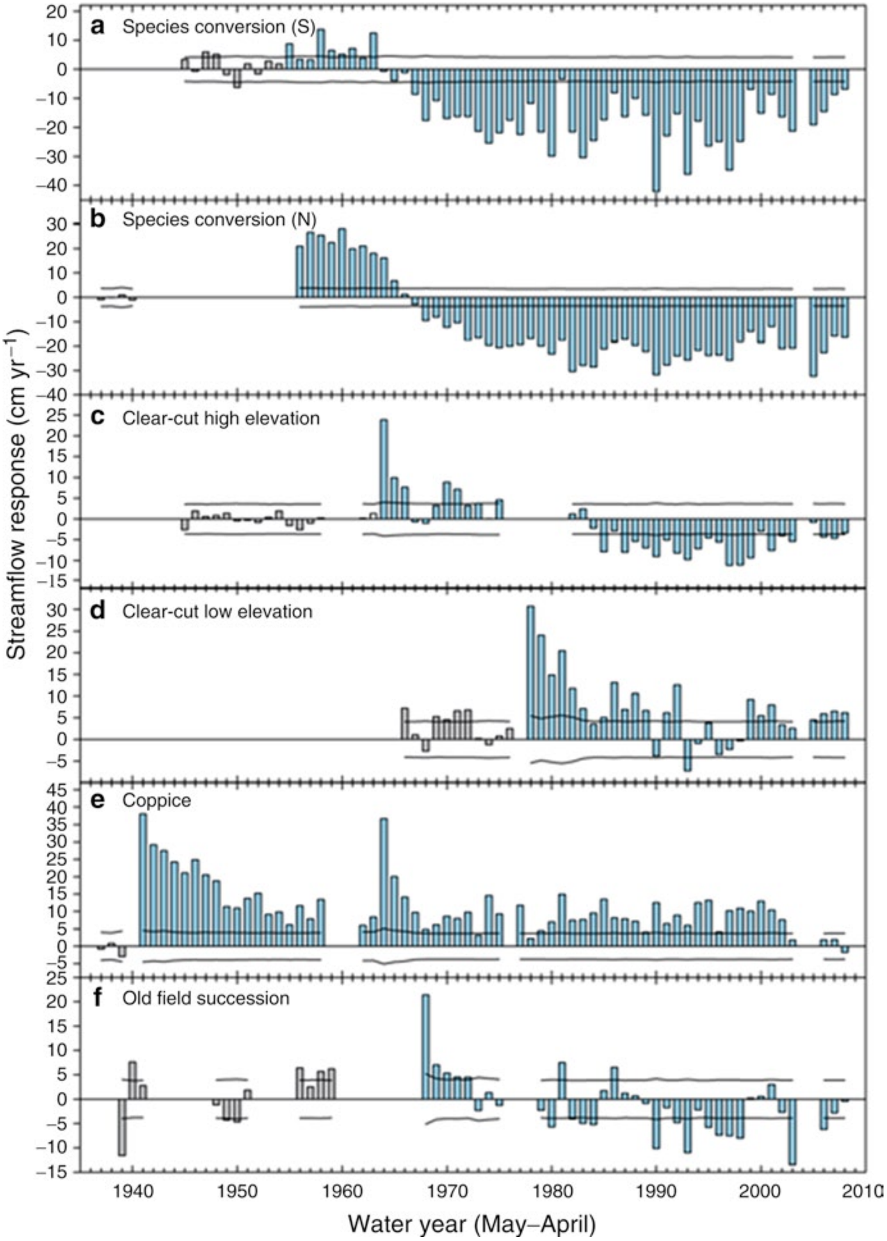

Fig. 14.1 Streamflow response (D, cm yr^{-1}) to forest cutting in the Southern Appalachians (see Swank and Crossley (1988) for site and treatment descriptions). Grey bars depict the calibration period and cyan bars depict streamflow response after treatments. Solid lines on either side of the zero line are 95% confidence intervals; data within the confidence intervals do not differ from zero. Species conversion treatments involved cutting hardwood species and planting *Pinus strobus* on north (N) and (S) facing watersheds (from Ford et al. 2011)

Table 14.2 Mean (standard error) growing season daily transpiration per unit leaf area (E_L, mm) for four hardwood species (Adapted from Ford et al. 2011). Within columns, species not sharing the same lowercase letters denote significant differences among species for that year. Within rows, years not sharing the same uppercase letters denote significant differences among years for that species

	Year		
Species	2004	2005	2006
Carya spp.	0.20 (0.03) b, A	0.19 (0.02) b, A	0.18 (0.02) c, A
Liriodendron tulipifera L.	0.45 (0.05) a, AB	0.39 (0.07) a, B	0.46 (0.03) a, A
Quercus prinus L.	0.21 (0.03) b, A	0.07 (0.01) b, B	0.10 (0.02) cd, AB
Quercus rubra L.	0.10 (0.02) b, A	0.07 (0.02) b, A	0.07 (0.01) c, A

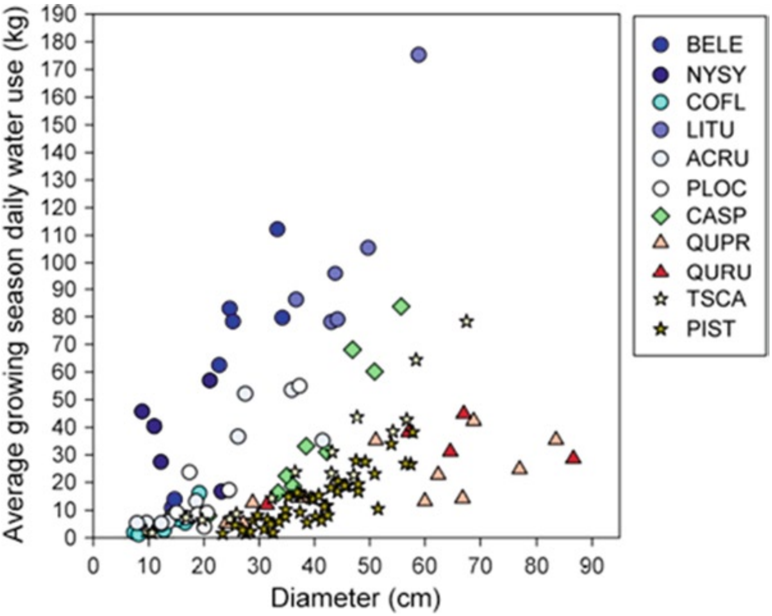

Fig. 14.2 Observed daily water use (DWU) estimated from sap flux density in trees of varying species (legend text denotes first two letters of Latin binomial: BELE *Betula lenta*, NYSY *Nyssa sylvatica*, COFL *Cornus florida*, LITU *Liriodendron tulipifera*, ACRU *Acer rubrum*, PLOC *Platanus occidentalis*, CASP *Carya* spp., QUPR *Quercus prinus*, QURU *Q. rubra*, TSCA *Tsuga canadensis*, PIST *Pinus strobus*) in reference watersheds at Coweeta (except PIST). Symbols represent the mean DWU of replicate trees in each species during the growing season for deciduous species, days of year 128–280 in 2006. Mean DWU during the entire annual period is shown for coniferous species (TSCA is during 2004, PIST is during 2006). LITU, QURU, QUPR, CASP, and PIST data are from (Ford et al. 2011). TSCA data are from Ford and Vose (2007). BELE, NYSY, COFL, ACRU, and PLOC are from (C. Ford and J. Vose, unpublished) but follow the methods in (Ford et al. 2011). Symbols: circles are species with diffuse porous xylem anatomy, diamonds are species with semi-ring-porous xylem anatomy, triangles are species with ring-porous xylem anatomy, stars are for species with tracheid xylem anatomy (from Vose et al. 2011)

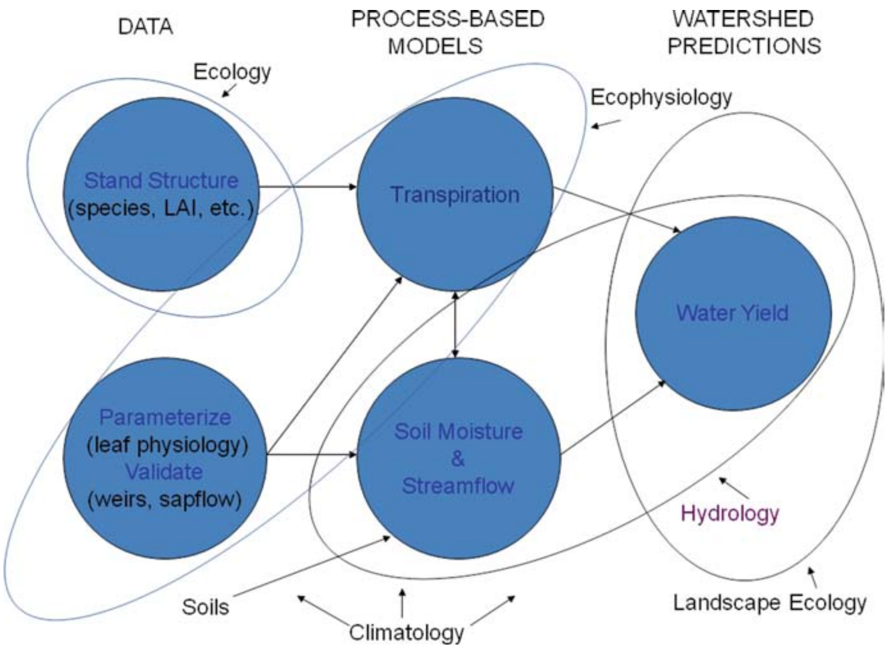

Fig. 14.3 Interdisciplinary approaches to understanding impacts of forest management and other disturbances on water yield requires linking species dynamics and physiology, soil moisture dynamics, and climate across scales ranging from leaves to landscapes (from Vose et al. 2011)

the empirical data shown in Fig. 14.1), we need to be able to link spatially explicit (i.e., cove, midslope, ridge, etc.) predictions of species composition and structure with: (1) species-specific physiology, (2) soil moisture and subsurface flow dynamics, and (3) microclimate. This is a significant departure from traditional hydrologic sciences and requires a multidisciplinary, multi-scale approach (Fig. 14.3).

14.4 Water Quality Responses

Considerable research has been conducted on the effects of forest harvesting on water quality in upland hardwood forests, as well as the development of BMPs to minimize impacts (Kochendorfer and Hornbeck 1999; Jackson et al. 2004; Sun et al. 2004). The most impacted water quality parameter is sediment load, although water temperature and dissolved nutrient concentrations can also be affected. The impact of all of these parameters can be reduced or eliminated with proper planning and BMP implementation. Thus, water quality from streams draining early successional forests can be as high from streams draining undisturbed forested catchments.

Sediment delivery to streams occurs primarily as a result of erosion from roads and skid trails associated with logging (Anderson et al. 1976; Swift 1988; Swank et al. 2001). For example, logging without BMPs resulted in annual sediment losses on the order of 3.1 MT ha^{-1} in the Central Appalachians compared to 0.04 MT ha^{-1} in uncut reference watersheds (Jackson et al. 2004). Careful layout and construction of roads and skid trails minimizes impacts (Swift 1988). However, roads and skid trails are particularly vulnerable to erosion during and shortly after construction, and stream crossings are the most likely locations for sediment delivery to streams. In a study in the Southern Appalachians examining the effectiveness of road construction BMPs, the majority of sediment was generated in two large storms that occurred shortly after new road construction and declined to pre-cut levels after road stabilization and reduced use after logging (Swank et al. 2001). Thus, it is critical to implement BMPs to ensure that newly constructed roads are quickly stabilized and that water and sediment moving from the forest roads and associated components such as ditches and cut banks is dispersed into areas that are disconnected from the streams to ensure infiltration and sediment trapping (Swift and Burns 1999). For example, in eastern Kentucky, BMPs such as streamside buffer strips and proper road construction and rehabilitation reduced suspended sediment considerably compared to a watershed clearcut without BMPs (Arthur et al. 1998). By contrast, other management activities that can be used to create early successional habitats without roads and skid trails (e.g., high intensity prescribed burning) are much less likely to cause a decline in water quality. For example, felling and burning low quality pine-hardwood stands in the Southern Appalachians resulted in no off-site movement of sediment (Swift et al. 1993).

Stream temperature, which affects dissolved oxygen concentration, may also be impacted by timber harvesting and the creation of early successional habitat. However, the magnitude and duration of the increase depends on the width of riparian buffers and the size of the harvested area. In the Central Hardwood Region, removal of forest canopy adjacent to forest streams increases maximum summer stream water temperatures by as much as 6°C (Swift and Messer 1971; Hornbeck and Federer 1975; Swift 1983; Clinton et al. 2010; Clinton 2011). However, maintaining a riparian forest buffer reduces or eliminates this effect (Hornbeck et al. 1986; Moore et al. 2005; Clinton 2011). For example, Clinton (2011) found that a buffer width as narrow as 10 m was adequate to prevent an increase in stream temperature after cutting. In addition, when only small areas of riparian forest canopy are removed, stream temperature responses are often dampened or eliminated within relatively short distances (e.g., 150 m) downstream (Clinton et al. 2010).

Disruption of terrestrial nutrient cycling processes through both alteration of soil abiotic conditions and reduced vegetation nutrient uptake can lead to nutrient transport into streams. Forest ecosystems are characterized by conservative nutrient cycling; most chemical constituents are limiting and tightly cycled by biogeochemical processes. Creating early successional habitats results in a considerable disruption to nutrient cycling processes and alters the environmental characteristics that regulate them. Opening the forest canopy increases soil temperature, and reduced transpiration rates increase soil moisture (Swank and Vose 1988). Both soil temperature and moisture influence nutrient cycling. For example, warmer and wetter soils result in increased nitrogen (N) mineralization and nitrification (Knoepp and Swank 2002; Knoepp and Vose 2007). Hence, these systems can transform N held tightly in organic matter to more mobile inorganic forms such as nitrate-N (NO_3^-).

In undisturbed forests, N typically limits productivity; most available N is used by the vegetation or immobilized by microbes. When nutrient uptake is disrupted by forest harvesting, combined with accelerated mineralization and nitrification, excess nutrients can be transported to streams. Studies examining changes in streamwater chemistry after timber harvesting have found that increases in nutrient concentrations can occur (especially for NO_3^-), losses are generally small relative to overall site nutrient pools and have little or no impact on water quality (Arthur et al. 1998; Martin et al. 2000; Swank et al. 2001). Nutrient responses tend to be greater in higher latitudes where nutrient cycling processes are more limited by temperature compared to responses at lower latitudes and elevations (Hornbeck et al. 1986). However rapid re-establishment of vegetation (both woody and herbaceous) plays a major in sequestering nutrients and re-establishing nutrient cycling processes. Indeed, major losses of nutrients (especially N, but also calcium and potassium) have been observed when vegetation regrowth is precluded by herbicides (Likens et al. 1970). Hence, one of the key BMPs to keep nutrients on site is to ensure rapid re-establishment of vegetation.

14.5 Potential Interactions with Climate Change

Because of the combination of biological and physical controls on hydrologic processes, climate change will both directly and indirectly impact the nation's water resources (Brian et al. 2004; Sun et al. 2008). The direct impacts of climate change on water resources will depend on how climate change alters the amount, type (e.g., snow vs. rain), and timing of precipitation; how this influences baseflow, stormflow, groundwater recharge, and flooding; and how these new hydrologic regimes interact with land use types (see Wear, Chap. 16). Long-term USGS streamflow data suggest that average annual streamflow has increased and this increase has been linked to greater precipitation in the eastern continental USA over the past 100 years (Lins and Slack 1999; Karl et al. 1995; IPCC 2007). However, fewer than 66% of all Global Circulation Models (GCMs) can agree on the predicted change in direction of future precipitation, e.g., wetter vs. drier (IPCC 2007). Inter- and intra-annual precipitation variability in the continental USA is a natural phenomenon related to large-scale global climate teleconnections (e.g., El Niño Southern Oscillation, Pacific Decadal Oscillation, North Atlantic Oscillation). Many regions of the USA have experienced an increased frequency of precipitation extremes over the last 50 years (Easterling et al. 2000a; Huntington 2006; IPCC 2007). As the climate warms in most GCMs, the frequency of extreme precipitation events increases across the globe (O'Gorman and Schneider 2009). However, the timing and spatial distribution of extreme events are among the most uncertain aspects of future climate scenarios (Karl and Knight 1998; Allen and Ingram 2002). Despite this uncertainty, recent experience with droughts and low flows in many areas of the USA indicate that even small changes in drought severity and frequency will have a major impact on society, including drinking water supplies (Easterling et al. 2000b; Luce and Holden 2009).

Most of the world's knowledge of the interactions among management, climate, vegetation, soils, and streamflow has been derived from long-term experiments on paired catchments. A key question is whether this knowledge, built primarily on empirical relationships under historical climate regimes, will allow robust predictions of responses under future climatic regimes. Creating early successional habitats has the potential to alter the hydrological responses to climate change again by influencing biological factors that determine evapotranspiration and physical factors that create soil disturbances or alter hydrologic flow paths. Management activities that favor or replace one species (or several species) over another can alter ET through direct and indirect changes in transpiration or interception (Ford et al. 2011, Stoy et al. 2006). For example, land management practices that favor high transpiration and interception may create conditions that mitigate the impacts of higher rainfall, but worsen the impacts of drought. As a result, streamflow responses (amount and timing) and recovery rates may be different under future climates. In general, hydrologic responses to climate change are larger in the humid Central Hardwood Region (McNab, Chap. 2). than in drier regions, and most climate models suggest the eastern USA will become more water-stressed (Sun et al. 2008). Thus, understanding the role of vegetation in hydrologic processes becomes increasingly important in the Central Hardwood Region as the climate gets warmer and more variable.

14.6 Summary

Because of the tight linkage between vegetation, soils, and water quantity and quality, creating early successional habitats has both direct and indirect impacts on water resources in the Central Hardwood Region. Decades of research using paired catchments in upland hardwood forests has shown:

1. Streamflow increases substantially in the first few years after cutting, but increases decline as sites revegetate and leaf area recovers. Streamflow increases are greater where precipitation is highest and where evapotranspiration represents a large portion of the overall site water budget.
2. The magnitude and rate of recovery to pre-disturbance streamflow depends on species composition and how species vary in transpiration and leaf and sapwood areas. Diffuse-porous species such as blackgum (*Nyssa sylvatica*), red maple, black birch, and yellow-poplar have the highest transpiration rates, while species with ring- or semi-ring porous sapwood, such as oaks and hickories, generally have the lowest transpiration rates for a given diameter. As such, watersheds dominated by the former would be expected to return to pre-cut streamflow levels faster than watersheds dominated by the latter; but depending on how the post-treatment vegetation differs from the pre-treatment vegetation, streamflow responses may be permanently higher or lower than reference conditions.

3. Stormflow increases by 10–20% following cutting and is directly proportional to the density and design of forest roads. However, these increases have not been shown to contribute to downstream flooding.
4. Sediment is the primary concern in terms of water quality responses to cutting and the primary sediment sources are roads and skid trails. BMPs have proven to be effective in reducing sediment.
5. Land managers will need to consider the potential interactions among future climate, changing vegetation structure and function, and physical impacts of forest operations on water resources.

As long as BMPs are properly implemented and maintained, creating early successional habitats in upland hardwood forests by harvesting trees is not likely to have a significant negative impact on either water quantity or water quality. However, it is also clear that forest operations associated with forest cutting (such as roads, stream crossings, culverts, etc.) can create permanent changes to hydrologic flow paths and serve as long-term sources of concern for water quantity and quality. In short, ensuring that BMPs are properly implesmented and functional requires a long-term commitment by land managers. Finally, much of what we know about the effects of disturbances on water resources (and the BMPs required to minimize those effects) has been developed from empirical data under historical climate regimes. Climatic conditions predicted for the eastern USA under climate change scenarios suggests the need for close monitoring of BMP effectiveness and the development of new BMPs applicable to more extreme climatic conditions in the future.

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THE CONTRIBUTION OF HEADWATER STREAMS TO BIODIVERSITY IN RIVER NETWORKS[1]

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ABSTRACT: The diversity of life in headwater streams (intermittent, first and second order) contributes to the biodiversity of a river system and its riparian network. Small streams differ widely in physical, chemical, and biotic attributes, thus providing habitats for a range of unique species. Headwater species include permanent residents as well as migrants that travel to headwaters at particular seasons or life stages. Movement by migrants links headwaters with downstream and terrestrial ecosystems, as do exports such as emerging and drifting insects. We review the diversity of taxa dependent on headwaters. Exemplifying this diversity are three unmapped headwaters that support over 290 taxa. Even intermittent streams may support rich and distinctive biological communities, in part because of the predictability of dry periods. The influence of headwaters on downstream systems emerges from their attributes that meet unique habitat requirements of residents and migrants by: offering a refuge from temperature and flow extremes, competitors, predators, and introduced species; serving as a source of colonists; providing spawning sites and rearing areas; being a rich source of food; and creating migration corridors throughout the landscape. Degradation and loss of headwaters and their connectivity to ecosystems downstream threaten the biological integrity of entire river networks.

(KEY TERMS: biotic integrity; intermittent; first-order streams; small streams; invertebrates; fish.)

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INTRODUCTION

Headwaters (i.e., springs and intermittent, first- and second-order streams) are abundant and unique components of a river network. They are found throughout the network, flowing into other first-order streams or into ones that are much larger. Small streams and springs occur across the range of climatic, geologic, riparian, and biogeographic settings of the United States This diversity produces differences in temperature, light, and hydrologic regimes, water chemistry, substrate type, food resources, and species pools, all of which affect the abundance and diversity of the biota. Because their catchments are not large and are easily influenced by small-scale differences in local conditions, headwater streams are arguably the most varied of all running-water habitats. They offer an enormous array of habitats for microbial, plant, and animal life, but their small size

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also makes them especially sensitive to disruption. Despite their abundance on the landscape and importance as habitat and as the origin of water resources, they are ignored in commonly used cartographic depictions. Small streams are neither named nor adequately indicated on standard topographic maps (1:24,000, USGS 7.5 min quads) (Meyer and Wallace, 2001).

The biota of headwater streams can be placed in five broad groups: (1) species that are unique to these small ecosystems; (2) species that are found in these and larger streams, although their abundance may vary with stream size; (3) species that move into headwaters seasonally as the stream network expands and contracts or as downstream conditions grow less favorable; (4) species that spend most of their lives in downstream ecosystems, but require headwaters at particular life-history stages (e.g., for spawning or nursery areas); and (5) species that live around but not in headwater streams, requiring the moist habitat they provide or feeding on the products of headwaters (e.g., benthic, emerging or drifting insects).

Headwaters are important for all of these groups and therefore are integral to the maintenance of biological diversity in the river network. In the following sections, we (1) provide an overview of the diversity of organisms that depend on small streams, (2) discuss the ecological factors that make these habitats favorable for so many species, (3) illustrate the ecological connectivity that exists between headwater and downstream ecosystems, and (4) discuss the ways in which downstream biota depend upon headwater ecosystems.

BIOLOGICAL DIVERSITY IN SMALL STREAMS

Primary Producers

The algal communities of headwaters are dominated by diatoms (e.g., *Cymbella*, *Gomphoneis*, *Fragilaria*), cyanobacteria (e.g., *Schizothrix*, *Phormidium*), red algae (e.g., *Batrachospermum*), and green algae (e.g., *Stigeoclonium*) (Biggs, 1996). In systems where the headwaters are shaded and low in nutrients, 30-60 algal species are commonly encountered, some of which are not found elsewhere in the river network (Rex Lowe, personal communication). For example, the algal community of a rivulet flowing from an Ontario spring consisted of 34 taxa, 32 of which were diatoms (Sherwood *et al.*, 2000). Although algal taxa richness increased downstream, eight of the taxa found in the first 20 m of the stream were not found at stations further downstream (Sherwood *et al.*, 2000). Rocks and bryophytes in a shaded headwater stream in the southern Appalachians supported 40 algal taxa, 30 of which were diatoms (Greenwood, 2004; Greenwood and Rosemond, 2005). Only a few taxa were abundant; two taxa each represented >20% of the biovolume, whereas each of 29 other taxa represented <1% (Greenwood and Rosemond, 2005). Recent research in continuously flowing Alaskan springs has revealed a diverse algal assemblage that serves as a source of propagules for the downstream flora once those larger streams begin to thaw (Huryn *et al.*, 2005).

Bryophytes (mosses and liverworts) commonly dominate the biomass of primary producers in small streams. Mosses can use only carbon dioxide in photosynthesis and are most diverse and abundant in headwater streams and seeps where water is rich in carbon dioxide (Stream Bryophyte Group, 1999). Bryophyte species richness ranged from 0 to 14 species in small boreal streams (Heino *et al.*, 2005). Four species dominate the bryophyte flora of small, high-gradient Appalachian streams; *Fontinalis dalecarlica* and *Hygroamblystegietum fluviatile* are most abundant in first through third-order streams (Glime, 1968). Mosses and liverworts attach to hard substrates and provide habitat that supports many invertebrate species (Stream Bryophyte Group, 1999).

The types of primary producers found in headwater streams vary greatly as a function of light and hydrologic regime. In well-lit, hydrologically stable springs, a diversity of vascular plants can be found including species endemic to springs such as *Zizania texana* (Texas wild rice) (Hubbs, 1995). A survey of macrophyte diversity in 79 small (mean width 1.9 m), unshaded, lowland streams found 11-24 species per stream (mean = 18.5 species) and a total of 131 species (Baattrup-Pedersen *et al.*, 2003). The headwaters of the Upper Mississippi River flow through bogs and swamps with high vascular plant diversity (Delong, 2005). In headwater streams flowing through steeper and forested catchments, angiosperm diversity is often low and increases as stream width increases. For example, the first 20 m of an Ontario spring-fed stream housed only three vascular plant species, whereas 9-14 species occurred at sites further downstream (Sherwood *et al.*, 2000). In addition to being primary producers in small streams, vascular plants can act as sieves, trapping particles of organic matter (Horvath, 2004). This increases the organic matter availability to consumers in the headwaters, but decreases organic matter transport downstream.

Decomposers

From a taxonomic perspective, bacteria are the least known organisms in headwater streams; however, we know much about their functional role in stream biogeochemical cycles and food webs (e.g., Hall and Meyer, 1998). They are critical to processing of organic matter, which alters nutrient and organic matter exports from small streams to downstream ecosystems as described elsewhere in this series of papers (Wipfli *et al.*, this issue). Dissolved organic carbon (DOC) provides the C source supporting bacterial metabolism and is the most abundant form of organic matter exported from headwaters to downstream ecosystems (Allan, 1995). Leaching of leaf litter is one source of DOC in headwaters that generates a diversity of compounds that differ in their availability to bacteria. Highly labile DOC supports local bacterial metabolism, whereas DOC of intermediate lability is exported and supports bacterial metabolism downstream (Wiegner *et al.*, 2005). Bacteria from headwater sites were able to grow on DOC leached from a nearby riparian species, whereas bacteria collected further downstream were able to use DOC leached from a wider array of species (Koetsier *et al.*, 1997). Genetic diversity of bacteria did not vary significantly with distance downstream in a blackwater stream (McArthur *et al.*, 1992). However, genetic similarity between sites decreased with increasing distance downstream, suggesting genetic differences among headwater and downstream populations of a species (McArthur *et al.*, 1992). Methods for assessing bacterial diversity are recent and still developing, and have not been applied to the entire bacterial assemblages in headwater streams. On the basis of what has been discovered in soils (Tiedje *et al.*, 1999), we would expect the sediments and biofilm of headwater streams to contain at least hundreds to thousands of types of bacteria.

Fungi are also crucial to organic matter dynamics and food webs in headwater streams, and we know considerably more about their diversity than about bacterial diversity. Fungi in headwater streams are primarily hyphomycetes, ascomycetes, and oomycetes. Species composition changes markedly along the course of a stream (Tsui *et al.*, 2001), but is high even in very small streams (Suberkropp and Wallace, 1992; Gulis and Suberkropp, 2004). Over 51 taxa of aquatic hyphomycete fungi have been found in two tiny streams in the southern Appalachians, where inputs of leaf litter from the surrounding forest are high (Gulis and Suberkropp, 2004). When leaf litter inputs to a headwater stream were experimentally eliminated, fungal taxa richness declined from 43 to 36 taxa (Gulis and Suberkropp, 2003). Fungal species composition and richness in headwater streams are strongly influenced by the species composition of riparian vegetation and water chemistry (Bärlocher and Graca, 2002; Gulis and Suberkropp, 2004).

Insects

As water first emerges from the ground in a spring or seep, it provides habitat for an array of insect species. Thirteen species of caddisfly were found within 20 m of the source of an Appalachian springbrook (McCabe and Sykora, 2000). As many as 18 caddisfly species were found in individual California springs (Erman and Erman, 1995), and from 5 to 38 chironomid taxa were identified from individual springs in the High Plains (Blackwood *et al.*, 1995). Unique faunal assemblages have been linked to characteristic water chemistries of springs, reflecting different levels of contamination of their ground-water sources (Williams *et al.*, 1997).

The springs and small seeps that provide habitats at the beginnings of a river network are inadequately mapped. A study in headwater streams of West Virginia and Kentucky illustrates this point. From February through April 2000, Stout and Wallace (2003) sampled from the first continuous flowing water downstream to either a confluence or the point on a topographic map where a solid blue line stream began; i.e., they sampled 34 flowing streams that were unmapped or indicated as intermittent. Their samples included over 86 insect genera in 47 families. Ephemeroptera, Plecoptera, and Trichoptera (EPT) taxa richness, commonly used as an indicator of water quality, increased with distance from the source in these unnamed streams (Figure 1). The seeps where water first emerged from the ground had

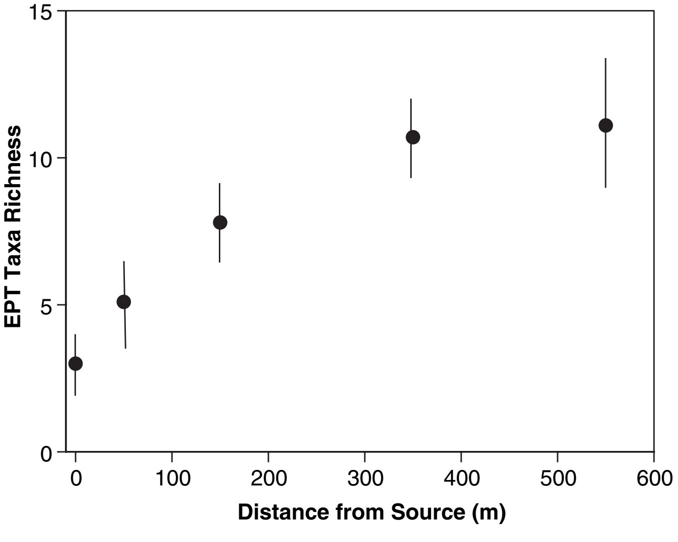

FIGURE 1. Taxa Richness (Mean ± 95% CI) for Insects in the Orders Ephemeroptera, Plecoptera, and Trichoptera Collected From 34 Unmapped Headwater Streams in Kentucky and West Virginia. Richness is plotted *vs.* distance from the point at which water emerged from the ground. Data are from Stout and Wallace (2003).

an average of three EPT taxa; sites within only 150 m of the source had an average of eight EPT taxa; and EPT taxa richness was similar (11 taxa) at 350 and 500 m from the source. Although these streams were either unmapped or designated as intermittent, EPT and other insect taxa with multiyear aquatic life cycles were found in these streams, some with catchments as small as 4 ha (Stout and Wallace, 2003).

Long-term stream research at Coweeta Hydrologic Laboratory in North Carolina provides further evidence of the diversity of aquatic insects in very small, unmapped streams (Table 1). At least 51 families and 145 genera of aquatic insects have been collected over three decades of sampling in eight headwater streams with catchments ranging in size from 5 to 61 ha. None of these streams is shown on standard topographic maps. Putting this taxonomic diversity into some perspective, there are only 33 families and 80 genera of freshwater fishes in the entire state of North Carolina (Menhinick, 1991).

TABLE 1. Diversity in Aquatic Insects Found in Headwater Streams of Coweeta Hydrologic Laboratory in the Southern Appalachian Mountains of Western North Carolina (Courtney, 1994, 2000; Gurtz, 1981; Huryn, 1990; Huryn and Wallace, 1985, 1987a,b, 1988; Lugthart and Wallace, 1992; Wallace *et al.*, 1991, 1999).

Order	**Number of Families**	**Number of Genera**
Ephemeroptera (mayflies)	5	10
Odonata (dragonflies, damselflies)	2	2
Plecoptera (stoneflies)	8	15
Megaloptera (alderflies, dobsonflies)	1	1
Coleoptera (beetles)	3	4
Trichoptera (caddisflies)	13	22
Diptera (true flies)	19	91*
Total	51	145

*Includes Chironomidae.

Small streams contain unique as well as widely distributed insect species. A list of eastern North American stoneflies that occur only in first- and second-order streams includes 60 species in 24 genera and 8 families (R.F. Kirchner and B.C. Kondratieff in Stout and Wallace, 2003). Thirty-six of the 78 caddisfly species in a Sierra Nevada stream network were found only in springs; eight of these were restricted to constant temperature springs (Erman and Erman, 1995). Species composition differed greatly among individual streams; on average, only 23% of species were similar among streams (Erman and Erman, 1995). Insect samples from seven central Oregon springs and seeps included 106 species; 92% of those were found only in the springs and seeps and not in the main creek (Anderson and Anderson, 1995). Most of the uniquely spring species were dipterans.

Aquatic insect diversity is high in the southeastern United States; 40% of the North American aquatic insect fauna can be found in the Southeast (Morse *et al.*, 1997). Much of this richness is in small springs and streams (Morse *et al.*, 1997). For example, extensive sampling in a Louisiana spring complex captured 43 caddisfly species including 5 endemics (Morse and Barr, 1990). Over 650 insect species have been found in Upper Three Runs Creek, a fourth-order stream on South Carolina's Coastal Plain; 180 species are found in its second-order tributaries, and many are found only in the headwaters (Morse *et al.*, 1980, 1983; John Morse personal communication). The spring-fed ravine ecosystems of northern Florida harbor 138 caddisfly and 23 stonefly species, which represent 70% and 55%, respectively, of all Florida species in these orders (Rasmussen, 2004). The high-gradient streams of the Appalachians are also rich in insect species, with collector-gatherers and shredders as the largest contributors to secondary production in the headwaters (Wallace *et al.*, 1992).

Even small streams that do not flow continuously may contain a rich and sometimes unique insect fauna. An intensive study of seven "summer-dry" (i.e., intermittent) streams in western Oregon < 12 km apart found 202 aquatic or semi-aquatic insect species, at least 13 of which were new to science (Dieterich and Anderson, 2000). The two intermittent streams that were in forest settings had more insect species (125-126 species) than a permanent headwater stream (100 species) in the same setting. Considering the entire species pool, 8% were found only in permanent headwaters, 25% were restricted to intermittent streams, and 67% were found in both. Over half of the species found were dipterans, and EPT taxa comprised about 30% of the insect fauna (Dieterich and Anderson, 2000). Somewhat higher taxonomic richness was observed in permanently flowing streams (71-92 taxa) than in intermittent streams (54-93 taxa) in another group of western Oregon streams, although the peak emergence biomass was three times higher in the intermittent streams (Progar and Moldenke, 2002). This emerging biomass provides a food resource for riparian consumers. In these streams, only two EPT genera were unique to the intermittent channels, and most taxa were common to both stream types. In the southeastern United States, 171 taxa were found in six small Alabama streams that varied in their permanence (Feminella, 1996). Only 7% of taxa were found exclusively in intermittent streams, whereas 75% of taxa were found in both perennial and intermittent streams. In the Southwestern United States, 10 species of winter-emerging stoneflies were found in New

Mexico streams that are dry for long periods in spring and autumn (Jacobi and Cary, 1996). Adaptations for life under these conditions include small size, rapid development, and a period of diapause during egg or larval stages. Subarctic Alaskan streams do not flow in winter because they are frozen. Although some dipteran species have adaptations that allow them to survive freezing, most aquatic invertebrates die when streambeds freeze; these species survive by migrating away from a freezing front or remaining in habitats such as headwater springs that do not freeze (Irons *et al.*, 1993; Huryn *et al.*, 2005). These refugia serve as sources of colonists when streams begin to thaw (Huryn *et al.*, 2005).

Mollusks, Crustaceans, and Other Invertebrates

The invertebrate fauna of hardwater springs is dominated by crustaceans, triclads, and mollusks (Glazier, 1991). Although mollusk diversity is generally the greatest in larger rivers, mollusks can also be conspicuous and abundant in headwaters. Many species are headwater specialists with small geographic ranges. For example, members of the prosobranch family Hydrobiidae frequent springs and spring-fed streams throughout the USA. About 200 rare headwater hydrobiid species occur in the USA. (listed by NatureServe 2005 as imperiled or critically imperiled [G2 or G1]), with dozens of narrowly endemic species from the Southeast, the Great Basin and the Northwest (Herschler, 1994; Frest and Johannes, 1999). Nineteen headwater species are either protected by the Endangered Species Act or are rare enough to be considered for listing. Hydrobiids, physids, and lymnaeids are the most abundant mollusks in hardwater springs in the temperate zone (Glazier, 1991). The pleurocerid snails such as *Goniobasis* and *Juga* are often dominant grazers in headwaters of the Southeast and Northwest (Lamberti, 1996; Steinman, 1996). Their absence from intermittent streams has been suggested as one of the factors responsible for high diversity of insect grazers in those systems (Dieterich and Anderson, 2000). Pearl mussels (*Margaritifera* spp.) can also be extremely abundant (>100 m^{-2}) in small streams (Johnson and Brown, 2000).

Crustaceans such as amphipods, isopods and crayfish are conspicuously abundant in headwaters. Microcrustaceans such as cladocerans, ostracods, and copepods also live in headwaters, where they can reach very high densities (>10,000 m^{-2}, Galassi *et al.*, 2002). Although fewer than 10 species of macrocrustaceans inhabit a typical headwater site, species composition varies greatly across headwaters; North America supports 600-700 species of large freshwater crustaceans, many of them in headwater streams. The NatureServe database lists 31 amphipod, 4 isopod, and 11 crayfish species as found in springs and springbrooks; of these, 30 amphipod, 3 isopod, and 5 crayfish species are considered imperiled or critically imperiled (G1 or G2, Larry Master, personal communication). Amphipods and isopods are most common in relatively constant, cool waters, where they can reach high densities (Covich and Thorp, 1991). The southeastern United States has the highest number of crayfish species (Taylor *et al.*, 1996). Crayfish comprise a large portion of the biomass in many headwater streams; e.g., they comprise >90% of macroinvertebrate biomass in perennial headwaters of coastal Washington (Haggerty *et al.*, 2002). Macrocrustaceans are not confined to perennial streams. In fact, total crayfish densities were higher in intermittent than in perennial streams in the south-central United States; two species (*Orconectes puntimanus* and *O. marchandi*) had significantly greater numbers in intermittent streams, whereas abundance of the other two species did not differ with stream type (Flinders and Magoulick, 2003).

Small streams support many invertebrate taxa other than insects, mollusks and crustaceans (Table 2), although they have not been as extensively studied. A typical headwater stream might contain 30-300 species and 20,000-2,000,000 m^{-2} of these other taxa, such as turbellarians, gastrotrichs, and nematodes (Table 2). Species richness in these groups may be as high in headwaters as in larger streams (e.g., Kolasa, 1983), and many can be found in intermittent streams. Many are unique to headwaters; e.g., most of the endemic lumbriculid oligochaetes recently discovered in the Pacific Northwest live in seeps, springs, and small streams (McKey-Fender and Fender, 1988; Fend and Brinkhurst, 2000; Fend and Gustafson, 2001).

Fishes

Stream fish diversity generally increases with increasing stream size along a gradient of increasing habitat heterogeneity, pool development, and habitat volume (Schlosser, 1987). The extent to which species richness changes with stream size varies considerably. From 3 to 11 species were found in a second-order Kentucky stream *vs.* 12-25 in a fourth-order stream (Kuehne, 1962); a Texas headwater stream contained 22 species, whereas downstream sections had 33 species (Evans and Noble, 1979). In some cases, the increase in fish species with increasing stream size occurs as a result of species additions, so that headwater assemblages represent a nested

TABLE 2. Invertebrates Other Than Mollusks, Crustaceans, and Insects That Are Common in Headwaters.

Group	Typical Species Richness in Headwaters	Typical Density in Headwaters (no./m^2)	Key References
Turbellaria	3–30	1,000–10,000	Kolasa (1983, 2002)
Gastrotricha	3–30 (?)	10,000–300,000 (?)	Strayer and Hummon (2001), Balsamo and Todaro (2002)
Rotifera	20–200	10,000–1,000,000	Schmid-Araya (1998), Wallace and Ricci (2002)
Nematoda	10–100	5,000–500,000	Traunsperger (2002)
Tardigrada	1–10	1,000–10,000 (?)	Nelson and McInnes (2002)
Oligochaeta	3–30	1,000–50,000	Schwank (1981a,b)
Acari	5–50	100–10,000	Di Sabatino *et al.* (2002, 2003)
Total	40–450	28,000–1,880,000	

Question marks indicate substantial uncertainty in poorly studied groups.

subset of species found throughout the network (e.g., Taylor and Warren, 2001). In other cases, diversity increases but the species are different from those found in the headwaters. For example, small insectivorous fishes numerically dominate first- and second-order streams in the southeastern Coastal Plain; the same species are rare in larger streams (Paller, 1994). Because headwater streams may contain a unique species assemblage, they can make a significant contribution to regional fish diversity (e.g., Paller, 1994).

The location of a small stream in the network also affects its richness (Matthews, 1998). The fish assemblages in second-order Texas streams flowing into other second order streams had a Shannon diversity index of 0.94, whereas second-order streams flowing into third and fourth-order streams had diversity indices of 1.13 and 1.84, respectively (Whiteside and McNatt, 1972). The higher diversity in the streams that flow into larger streams is a consequence of species from the larger stream moving into the tributaries.

Small streams are characterized by small-bodied species such as small minnows, madtom catfishes, darters, and sculpins (Schlosser, 1987). For example, small-bodied insectivorous fishes are numerically dominant in first-order streams in Mississippi, with species richness ranging from 2 to 36 species (Smiley *et al.*, 2005). Samples from only 14 first-order streams in managed pine forests included 18% of Mississippi's native fish species (Smiley *et al.*, 2005). The fish fauna in cold eastern and western North American headwater streams usually consists of a salmonid species, a sculpin, and 1-3 species of cyprinids or catastomids (Moyle and Herbold, 1987). In high-gradient Southern Appalachian streams brook trout (*Salvelinus fontinalis*) are found furthest upstream, with sculpin (e.g., *Cottus bairdi*), dace (e.g., *Rhinichthys atratulus*), and darters (e.g., *Etheostoma flabellare*) slightly further downstream (Wallace *et al.*, 1992). Throughout the southeastern United States, darters in the genera *Etheostoma* and *Percina* contribute to fish diversity in headwaters with 73 species whose habitat descriptions in the NatureServe database include the terms springs, small streams, headwaters, or small creeks. That database lists 180 fish species whose distributions include springs and springbrooks (L. Master, personal communication).

Springs and spring runs often contain unique fish faunas, including endemics found in only one or two springs (Hubbs, 1995). The NatureServe database identifies 49 fish species as exclusive to springs and springbrooks; 30 of these species are ranked as critically imperiled, imperiled, or extinct (NatureServe ranks of G1, G2, or GX; L. Master, personal communication). Many extirpated and threatened southwestern fishes are spring inhabitants. For example, 13 species of pupfishes (*Cyprinodon* spp.) are found in springs in the southwestern United States, 12 of which have NatureServe ranks of G1, G2, or GX. Six endemic *Gambusia* species occur in stenothermal Texas springs, and those species are replaced by the widespread mosquitofish *Gambusia affinis* in downstream reaches (Hubbs, 1995). Unique spring species are also found in more mesic regions. For example, *Etheostoma nuchale* is a darter endemic to two springs in Alabama (Hubbs, 1995); the coldwater darter, *E. ditrema*, has a similar limited distribution.

Fish also occur in intermittent stream habitats. Ten intermittent tributaries of a river in Colorado contained 11 native fish species. Five of those species penetrated 7-9 km upstream in tributaries that were dry except for isolated pools, which were maintained by an extensive ground-water aquifer (Fausch and Bramblett, 1991). Rogue River tributaries that were dry in summer supported large spawning populations of steelhead salmon (*Oncorhynchus mykiss*) in winter (Everest, 1973 in Erman and Hawthorne, 1976). A striking 39-47% of adult rainbow trout (*O. mykiss*) in Sagehen Creek, California, spawned in one intermittent tributary and only 10-15% spawned in the perennial main channel (Erman and Hawthorne, 1976). Intermittent streams and ephemeral swamps contributed 15% and 23% of coho salmon (*Oncorhynchus kisutch*) smolts, respectively, during 2 years in the 10 km^2 Carnation Creek catchment (Brown and Hartman, 1988). The proportion of smolts from intermittent tributaries was higher during 1 year because extensive flows washed out smolts in the main channel and lower during the other year because low spring flows decreased the connectivity between the main stem and intermittent habitats. A recent study in coastal Oregon streams found 11-21% of adult coho salmon populations spawning in intermittent streams (Wigington *et al.*, 2006). Furthermore, juvenile coho tagged in the main channel entered intermittent tributaries during high autumn flows, and smolts that used intermittent tributaries were larger than those using permanent tributaries (Wigington *et al.*, in review).

Many fish species that spend most of their lives in larger streams, rivers, or lakes use small streams for spawning and nursery areas. In addition to the coho salmon, steelhead, and rainbow trout just described, cutthroat trout (*Oncorhynchus clarki*) and chum salmon (*Oncorhynchus keta*) migrate into very small tributary streams to spawn, navigating riffles with half of their bodies out of the water. During their first summer of life, 81% of brook trout spawned in a Canadian lake moved into tiny tributary streams to take advantage of favorable flows and temperatures (Curry *et al.*, 1997). Fishes other than salmonids also use small tributaries for spawning and nursery areas. For example, the trispot darter (*Etheostoma trisella*) is an imperiled southeastern species that lives along the edge of a small river but spawns in a seepage stream (<1 m wide) flowing through a marshy pasture (Ryon, 1986); the slackwater darter, *Etheostoma boschungi*, spawns in similar habitats.

Hence, we can identify three broad classes of fishes that use headwater streams and springs. Headwater specialists use small streams throughout the year. This group includes species of minnows (*Phoxinus, Rhinichthys, Hemitremia*), pupfish (*Cyprinodon*), topminnows (*Fundulus*), sculpins (*Cottus*), and darters (*Etheostoma* and *Percina*). A second class includes generalists that use headwaters as one of many habitats. Many trout, minnows such as creek chub, madtom catfish (*Noturus*), and small sunfishes (e.g., pygmy sunfishes, *Elassoma*) are in this group. These species may maintain permanent populations in headwaters or move into and out of them as the stream network expands and contracts. Some can be found in water barely deep enough for them to swim, such as the pygmy sunfishes that occur in inflow regions of southeastern swamps. The third group lives in larger systems but uses small streams for spawning and nursery areas as described above.

Headwater fish species are vulnerable to extirpation. In the southeastern United States 25% of the 16 headwater species and 70% of the 10 spring species are considered to be jeopardized (Table 3 and Etnier, 1997). Small-bodied fishes that spawn, feed or seek shelter on the stream bottom are particularly vulnerable (Burkhead *et al.*, 1997; Burkhead and Jelks, 2000). Highland endemic species, many that occupy headwater habitats, are being replaced by more cosmopolitan species as southern Appalachian streams are degraded (Scott and Helfman, 2001). Threats to headwater fishes are not unique to the southeastern United States. Headwater species account for 29% of

TABLE 3. Southeastern Fish Species Whose Preferred Habitat Is Headwaters or Springs According to Etnier (1997).

Headwater Species	**Spring Species**
Notropis chrosomus (rainbow shiner)	*Hemitremia flammea* (flame chub)
N. signipinnis (flagfin shiner)	*Notropis harperi* (redeye chub)
Phoxinus cumberlandensis (blackside dace)*	*Forbesichthys agassizi* (spring cavefish)
P. erythrogaster (southern redbelly dace)	*Fundulus albolineatus* (whiteline topminnow)*
P. tennesseensis (Tennessee dace)*	*F. julisia* (Barrens topminnow)*
P. sp.cf. *erythrogaster* *	*Cottus pygmaeus* (pygmy sculpin)*
Rhinichthys atratulus (blacknose dace)	*Elassoma alabamae* (spring pygmy sunfish)*
Semotilus atromaculatus (creek chub)	*Etheostoma ditrema* (coldwater darter)*
S. lumbee (sandhills chub)*	*E. nuchale* (watercress darter)*
S. thoreauianus (Dixie chub)	*E. tuscumbia* (Tuscumbia darter)*
Catostomus commersoni (white sucker)	
Salvelinus fontinalis (brook trout)	
Etheostoma parvipinne (goldstripe darter)	
E. sagitta (arrow darter)	
E. spectabile (orangethroat darter)	
E. whipplei (redfin darter)	

*Indicates species that Etnier (1997) identified as jeopardized or extinct. This list does not include species that use headwaters for breeding.

all fish species in the Maumee (98 total species) and Illinois (135 species) rivers, and headwater specialists have been particularly vulnerable to extirpation (Karr *et al.*, 1985). From 50% to 64% of headwater species are either declining or extirpated from those rivers (Karr *et al.*, 1985). A tabulation of headwater and spring fish species that are presumed Extinct or listed as Threatened, Endangered, or Candidate species under the Endangered Species Act includes at least 13 species dependent on small or intermittent streams and 23 spring-dwelling species (Table 4). This is an extremely conservative estimate; many more headwater- and spring-dwelling fishes are recognized as imperiled by the American Fisheries Society (Warren *et al.*, 2000).

In contrast to this pattern of threatened species in headwaters, protected headwater streams can serve as a refuge for species extirpated from other parts of the network. For example, the smallest known parasitic lamprey species (*Lampetra minima*) was thought to be extinct after the endemic population in Miller Lake was eliminated via poisoning in 1958. Later collections in small tributaries revealed previously unknown populations of the species (Lorion *et al.*, 2000).

Amphibians and Reptiles

Stream-dwelling amphibians can be found in streams as both larvae and adults (Petranka, 1998). Many spend their entire life history within streams, whereas others use streams while larvae, venture into terrestrial habitats as adults, and return to streams only to reproduce. In Appalachian streams, amphibians are primarily found in habitats that lack fish, but the *Dicamptodon* of the western United States and *Necturus* of the Southeast share their habitats with fishes. The tadpoles of some *Rana* and *Bufo* also survive where fishes are present.

Salamanders (larvae and adults) and frogs (adults) can be the dominant vertebrate predators in systems where they occur (Burton and Likens, 1975; Werner and McCune, 1979), and tadpoles exert significant grazing pressure on algae (Stebbins and Cohen, 1995). The presence of amphibians in headwater streams increases the biodiversity by acting as keystone predators (e.g., Fauth and Resetarits, 1991).

North American amphibian databases list 84 salamander species in 18 genera whose habitats include small streams, seeps, springs, or headwater streams (Table 5). In high-gradient Appalachian streams, 3-5

TABLE 4. Fish Species Associated with Small Streams and Springs That Are Presumed Extinct (*) or Are Listed as Threatened, Endangered, or Candidate Species under the Endangered Species Act.

Small Stream Species	**Spring Species**
Phoxinus cumberlandensis (blackside dace)	*Eremichthys acros* (desert dace)
Catostomus santaanae (Santa Ana sucker)	*Gila intermedia* (Gila chub)
Oncorhynchus clarkii seleniris (Paiute cutthroat trout)	*Lepidomeda albivallis* (White River spinedace)
O. mykiss pop. 10 (steelhead – southern California)	*L. altivelis* * (Pahranagat spinedace)
O. mykiss whitei (Little Kern golden trout)	*Rhinichthys osculus nevadensis* & other subspp. (Ash Meadows speckled dace)
Gasterosteus aculeatus williamsoni (unarmored threespine stickleback)	*Fundulus albolineatus** (whiteline topminnow)
Etheostoma chienense (relict darter)	*Gambusia gaigei* (Big Bend gambusia)
E. cragini (Arkansas darter)	*G. georgei* (San Marcos gambusia)
E. fonticola (fountain darter)	*G. heterochir* (Clear Creek gambusia)
E. okaloosae (Okaloosa darter)	*G. nobilis* (Pecos gambusia)
E. phytophilum (rush darter)	*Cottus paulus* (pygmy sculpin)
E. scotti (Cherokee darter)	*Crenichthys baileyi* (White River springfish)
E. susanae (Cumberland Johnny darter)	*C. nevadae* (Railroad Valley springfish)
	Cyprinodon arcuatus * (Santa Cruz pupfish)
	C. bovines (Leon Springs pupfish)
	C. *diabolis* (Devil's Hole pupfish)
	C. elegans (Comanche Springs pupfish)
	C. macularius (desert pupfish)
	C. nevadensis (Amargosa pupfish) (2 subspp. extinct)
	C. radiosus (Owens River pupfish)
	Empetrichthys latos (Pahrump poolfish)
	E. merriami * (Ash Meadows poolfish)
	Etheostoma nuchale (watercress darter)

Note: This is a very conservative listing of species considered imperiled by experts; e.g., of the 11 species identified as jeopardized by Etnier (1997) (see Table 3), only four are listed here, and one of those is extinct.

TABLE 5. Reptile and Amphibian Genera with Species Whose Habitats Include Small Streams, Seeps, Springs, or Headwater Streams.

	Genus	No. of Species
Salamanders	*Ambystoma*	6
	Amphiuma	3
	Dicamptodon	4
	Desmognathus	17
	Eurycea	25
	Gyrinophilus	4
	Haideotriton	1
	Hemidactylium	1
	Hydromantes	3
	Necturus	5
	Phaeognathus	1
	Plethodon	2
	Pseudotriton	2
	Rhyacotriton	4
	Pseudobranchus	2
	Siren	2
	Stereochilus	1
	Typhlotriton	1
Frogs	*Acris*	2
	Ascaphus	2
	Hyla	2
	Pseudacris	2
	Rana	18
	Smilisca	1
	*Xenopus**	1
Toads	*Spea*	2
	Bufo	6
Turtles	*Chelydra*	1
	Kinosternon	6
	Sternotherus	4
	Apalone	3
Snakes	*Nerodia*	7
	Regina	4
	Seminatrix	1
	Agkistrodon	1
	Farancia	2

Note: Data are from NatureServe (accessed July and October 2005), AmphibiaWeb (http://www.amphibiaweb.org), Global Amphibian Assessment (http://www.globalamphibians.org), IUCN Red List (http://www.redlist.org), and Center for North American Herpetology (http://www.naherpetology.org, accessed October 2005).

*Introduced into North America.

species of salamanders in the genera *Desmognathus, Eurycea*, *Gyrinophilus*, and *Leurognathus* occur and are the dominant vertebrate predators in the smallest headwaters; their secondary production is higher in first-order streams than in third-order streams (Wallace *et al.*, 1992). Salamander larvae feed almost exclusively on aquatic invertebrates (Johnson and Wallace, 2005). In the northeastern United States, stream amphibian diversity is concentrated in headwater streams (reports cited in Lowe and Bolger, 2002). Population size of the spring salamander, *Gyrinophilus porphyriticus*, was the highest in small streams without brook trout and lower where connectivity with downstream ecosystems was compromised (Lowe and Bolger, 2002).

Several frog and toad species also occur in small streams: 28 species of frogs in seven genera and eight species of toads in two genera occur in small streams and springs (Table 5). At least two of these species are considered rare (G1 or G2 in NatureServe, 2003), and one (*Rana fisheri*) is presumed extinct. Also listed is *Xenopus laevis*, a species native to Africa and introduced to novel habitats in North America; introductions of this exotic species may be responsible for the introduction of Chytrid fungi to the USA (Weldon *et al.*, 2004). This is an example of a headwater species with an impact far beyond the headwaters.

Reptiles (chiefly turtles and snakes) may also be found in headwater habitats including intermittent streams (e.g., Stone, 2001). Fourteen species of turtles in 4 genera and 15 species of snakes in five genera are found associated with small streams (Table 5). Although reptiles are not usually restricted to or most abundant in these habitats (Buhlmann and Gibbons, 1997), species in several genera (e.g., *Nerodia*, *Farancia*, and *Regina*) specialize on aquatic prey items. The genera listed in Table 5 represent taxa with the strongest ties to headwater habitats and do not include several species that are only loosely associated with streams (e.g., certain species of *Carolina*, *Elaphe*, *Thamnophis*, and *Nerodia*).

Birds and Mammals

Only a few species of birds (e.g., dippers, *Cinclus mexicanus*) actually live in small streams, but many depend on headwaters for food, water, habitat, or movement corridors. The preferred habitat of Louisiana and northern water thrushes (*Seiurus noveboracensis* and *S. motacilla*) is small headwater streams (Prosser and Brooks, 1998). The Virginia rail (*Rallus limicola*) is listed as a species exclusive to springs and springbrooks in the NatureServe database (L. Master, personal communication). Many other species are attracted to the large hatches of aquatic insects that emerge from headwater streams. Birds like flycatchers can be especially abundant around streams (Murray and Stauffer, 1995), and overall bird abundance may be elevated near headwater streams (Wiebe and Martin, 1998). Bird species richness and evenness were higher in the riparian zone of the first and second-order Michigan streams than in the uplands, and 12 species were found only in the riparian zone (Bub *et al.*, 2004). Abundance of several bird species was closely correlated with aquatic insect emergence in small prairie streams (Gray, 1993). Birds such as herons and kingfishers feed on fish and aquatic invertebrates in pools of intermittent streams (e.g., Tramer, 1977).

TABLE 6. A Minimum Estimate of Taxa Associated with Three Small, Shaded Streams (Average Discharge <2.5 L/s) on Catchments 53, 54, 55 (5–7.5 ha) at Coweeta Hydrologic Laboratory, NC.

Taxon	Estimated Number of Taxa	Reference
Algae	30 diatom species 10 other algal taxa	Greenwood (2004), Greenwood and Rosemond (2005)
Bryophyta	7 moss and 4 liverwort taxa	Greenwood (personal communication)
Fungi	51 taxa	Suberkropp and Wallace (1992), Gulis and Suberkropp (2004, 2003)
Protista	>7 taxa	Vila (1996), Vila (personal communication)
Nematoda	>10 taxa	Vila (1996), Vila (personal communication)
Copepoda	5 species	Vila (1996), Vila (personal communication)
Cladocera	1 species	Vila (1996), Vila (personal communication)
Decapoda	1 species	Wallace *et al.* (personal observations)
Ostracoda	1 species	Vila (1996), Vila (personal communication)
Gastrotrichia	>5 taxa	Vila (1996), Vila (personal communication)
Oligochaeta	>4 taxa	Vila (1996), Vila (personal communication)
Branchiobdellida	1 species	Wallace *et al.* (personal observations)
Rotifera	>10 taxa	Vila (1996), Vila (personal communication)
Turbellaria	>4 taxa	Vila (1996), Vila (personal communication)
Tardigrada	2 taxa	Vila (1996), Vila (personal communication)
Acarina	>3 taxa	Vila (1996), Vila (personal communication)
Bivalvia	1 species	Wallace *et al.* (personal observations)
Ephemeroptera	4 families; 7 genera; >7 species	Wallace *et al.* (personal observations), Wallace *et al.* (1991), Lugthart and Wallace (1992), Wallace *et al.* (1999)
Odonata	2 families; 2 genera; >2 species	Wallace *et al.* (personal observations), Wallace *et al.* (1991), Lugthart and Wallace (1992), Wallace *et al.* (1999)
Plecoptera	6 families; 8 genera; >8 species	Wallace *et al.* (personal observations), Wallace *et al.* (1991), Lugthart and Wallace (1992), Wallace *et al.* (1999)
Coleoptera	3 families; 4 genera; >4 species	Wallace *et al.* (personal observations), Wallace *et al.* (1991), Lugthart and Wallace (1992), Wallace *et al.* (1999)
Trichoptera	14 families; 19 genera; > 20 species	Wallace *et al.* (personal observations), Wallace *et al.*(1991), Lugthart and Wallace (1992), Wallace *et al.* (1999)
Diptera (incl. chironomids)	15 families; 55 genera; >59 species	Wallace *et al.* (personal observations), Wallace *et al.* (1991), Lugthart and Wallace (1992), Wallace *et al.* (1999), Courtney (1994, 2000)
Hemiptera	2 genera; 2 species	Wallace *et al.* (personal observations)
Collembola	1 family; 1 genus; >1 species	Wallace *et al.* (personal observations)
Arachnida	19 genera*	Sanzone (2001)
Amphibia	2 genera; 5 species	Johnson (2001)
Reptilia	> 3 species	Wallace *et al.* (personal observations)
Aves	2 species	Wallace *et al.* (personal observations)
Mammals	4 species	Wallace *et al.* (personal observations)
TOTAL	> 293 taxa	

*Estimated from data on a site ~2 km downstream.

Several bat species forage along streams for emerging insects and drink from the stream (Seidman and Zabel, 2001). Seven bat species in the genera *Myotus*, *Corynorhinus, Lasionycteris* and *Eptesicus* were observed feeding along intermittent streams in California (Seidman and Zabel, 2001). Bat activity was the greatest along the widest intermittent streams, but higher at all stream sizes than at upland sites.

Small mammals found in headwater stream habitats include shrews, voles, and moles. NatureServe (accessed July 2005) lists 5 species of shrews in the genus *Sorex* that are found in and on the banks of headwater streams in the USA. The star-nosed mole (*Condylura cristata*) digs tunnels that lead to small streams and is considered imperiled in the southeastern United States (Harvey and Clark, 1997). Mammals characteristic of small streams in the Pacific Northwest include *Sorex bendirii*, *S. palustris*, *S. pacificus*, *Microtus richardsoni* and *M. longicaudus*; some are obligate headwater species whereas others are widespread but more abundant in headwaters (Richardson *et al.*, 2005). Headwaters are also frequented by species such as raccoon, mink, beaver and otter, which may use them out of proportion to their areal extent on the landscape (Kruuk *et al.*, 1998).

Estimating Biological Diversity in a Headwater Stream

A complete species list does not exist for any headwater stream in the USA. However, based on the studies discussed here, a complete list would likely number in the hundreds to thousands of species. The invertebrate fauna of a first-order German stream (Breitenbach) has been investigated for many years. This 1-m-wide stream is home to 1004 invertebrate taxa (Allan, 1995). Many of these species are small invertebrates living in the hyporheic zone with connections to the ground water. Similar invertebrate diversity is likely to be found in the USA headwater streams. As an example, we consider three first-order, fishless streams (catchments 5-7.5 ha in area and mean discharge < 2.5 l/s) in the southern Appalachian Mountains of North Carolina, which are sites of ongoing long-term ecological research. These heavily shaded streams are in forested catchments and have a dense rhododendron riparian canopy. A list of known diversity in the taxonomic groups found associated with these three small streams is presented in Table 6. The groups about which we know the least (noninsect invertebrates) in these Appalachian streams are very diverse (400 taxa) in the Breitenbach, a small stream where they have been intensely studied (Allan, 1995). It is therefore likely that noninsect invertebrate diversity in the Appalachian streams is considerably higher than what we report here. Birds and reptiles associated with these streams have not been studied, so their diversity is unknown. Thus, we know that at least 293 taxa are associated with these three first-order streams, but their true diversity is likely at least twice that.

Headwater diversity is underestimated not only because of limited sampling, but also because so many headwater species remain undescribed. For example, half of the stonefly species associated with headwaters were described only in the last two to three decades (Stout and Wallace, 2003); new species of hydrobiid snails are continually being described; and a recent survey of ravine streams in the Florida panhandle found a dozen caddisfly species new to science (Rasmussen, 2004). Thorough surveys of small streams routinely discover new species, genera, and even families of invertebrates (Strayer, 2000). This is especially true for the hyporheic fauna living within the streambed, a habitat that is rarely sampled systematically.

THE BIOLOGICAL IMPORTANCE OF SMALL STREAMS IN RIVER NETWORKS

Headwater streams and springs may be small in size, but they provide habitats for a rich array of species, which enhances the biological diversity of the entire river system. Furthermore, the strong biological linkages between these upstream habitats and downstream ecosystems enhance and maintain species diversity downstream. The attributes of headwaters that make them essential habitats and that lead to linkages with other ecosystems are diagrammed in Figure 2 and discussed in this section.

Headwaters Support Many Species That Occur Nowhere Else in the River System

The previous sections provided numerous examples of species found only in headwaters. These species enhance diversity in the entire system (e.g., Paller, 1994). There are many reasons why headwater streams have a unique complement of species; we describe several here.

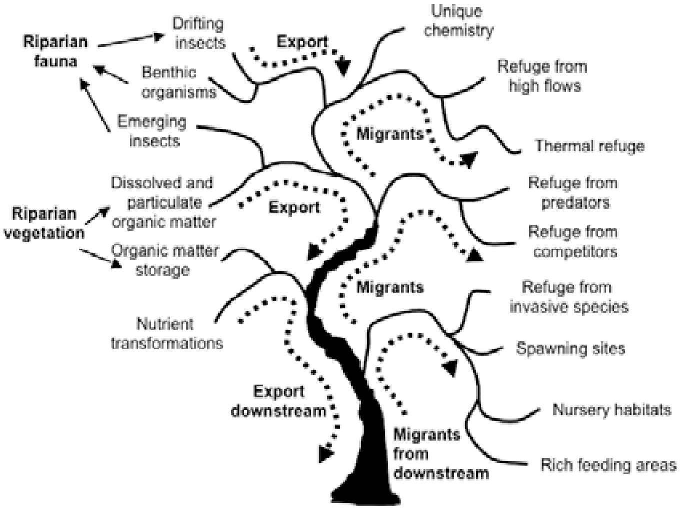

FIGURE 2. Factors That Contribute to the Biological Importance of Headwater Streams in River Networks. Attributes on the right benefit species unique to headwaters and also make headwaters essential seasonal habitats for migrants from downstream. On the left are biological contributions of headwater ecosystems to riparian and downstream ecosystems.

Headwaters Provide Unique and Highly Diverse Physico-chemical Habitats. Headwater streams contribute to species richness in river systems in many ways, chief among which is the diverse array of unique habitats that they provide. As noted by a fish ecologist, "overall, there are probably more environmental, biological and ichthyological differences among different kinds of first-order streams than among stream reaches in higher orders" (Matthews, 1998; p. 311). Headwaters range from steep, swift, and cold mountain streams to warm, low-gradient, swampy tributaries. The light regime in small streams ranges from well lit to heavily shaded. Their chemistry reflects the catchment's soil, geology, and human disturbance regime (e.g., Williams *et al.*, 1997). Their biology reflects the complement of species (both native and introduced) in the region and the presence or absence of barriers to exchange with neighboring ecosystems (e.g., downstream, riparian, or in adjacent valleys). The flow regime in small streams can be fairly constant in ground-water-fed springs, predictably variable from seasonal snow melt, intermittent with isolated pools sustained by ground-water connections, perennial with a flashy hydrograph after rainstorms, or one of many other variations. Small streams can serve as a refuge for species that are vulnerable to being swept downstream. With lower discharge and proximity to refuges from the current, small streams and springs offer a more benign habitat for species unable to maintain position in a strong current (e.g., Glazier, 1991; Dieterich and Anderson, 2000).

Headwaters Provide a Refuge from Predators. The high vulnerability of amphipods to fish predation are considered to be one reason why amphipods reach such high abundance in small fishless springs (Glazier, 1991). Low numbers of predators in intermittent streams is considered to contribute to the high diversity of aquatic insects (Dieterich and Anderson, 2000), the high biomass of emerging insects (Progar and Moldenke, 2002), and crayfish abundance patterns (Flinders and Magoulick, 2003) in those streams. The absence of fish predators in high-elevation Colorado streams results in emerging female mayflies that are larger and more fecund (Peckarsky *et al.*, 2002). The flight of adult stoneflies prior to oviposition is predominantly upstream for distances up to 730 m in a New Hampshire stream network; researchers speculate that this is because of the lower interspecific competition, lower predation risk, and higher food resources in the headwater tributaries (Macneale *et al.*, 2005). The absence of fish predation is considered a factor responsible for the prevalence of salamanders and other amphibians in small streams (Petranka, 1998). The significance of predator-free environments for amphibians is apparent from the lower populations observed in stream networks where trout have been introduced into high mountain lakes (Pilliod and Peterson, 2001).

Headwaters Provide a Refuge from Competitors. Low abundance of competitively dominant species is another explanatory factor for the abundance and diversity of headwater species. The absence of dominant competitors such as the snail *Juga silicula* was considered a factor contributing to the diversity of grazing insects in western intermittent streams (Dieterich and Anderson, 2000). Interannual variation in abundance of native rainbow trout in an intermittent California stream was correlated with the intensity of winter floods, which destroy the eggs of introduced brook trout (*Salvelinus fontinalis*). This leaves fewer brook trout to compete with rainbow trout fry that hatch during the following spring (Erman and Hawthorne, 1976). The brook trout fry are competitive dominants in this stream because they are larger and more aggressive than rainbow trout fry (Erman and Hawthorne, 1976).

Headwaters Provide a Refuge from Alien Species. In the southern Appalachians, populations of native brook trout have been greatly reduced or displaced by the introduced rainbow trout throughout much of the stream network; brook trout persist in small, high-gradient headwater streams (Larson and Moore, 1985; Larson *et al.*, 1995). Headwater pools in a Colorado stream provided habitats for Arkansas darters that were otherwise subjected to predation by an introduced pike (Labbe and Fausch, 2000). Headwater streams are recognized as the refuges for species that have been extirpated downstream and have been identified as the priority targets for freshwater conservation efforts (Saunders *et al.*, 2002).

Headwaters Are Essential for Species Living in Larger Streams

Genetic Linkages. Populations in headwaters are genetically connected to populations living in larger streams, and the genetic structure of stream populations provides a measure of this linkage. Little genetic differentiation from headwaters to downstream reaches was observed for distances up to 2.5 km in a stonefly population (Schultheis *et al.*, 2002), up to 10 km in populations of a mayfly (Monaghan *et al.*, 2001), and up to 20 km in a caddisfly population (Wilcock *et al.*, 2003). This mixing of up- and downstream populations is a result of both larval and adult dispersal and illustrates the scale of biological linkages in river networks.

Species Migrate to Headwaters for Spawning and Nursery Habitats. Small streams serve as vital spawning habitats for species that live in larger streams during most of the year. In addition to the many salmonids that spawn in small streams as discussed earlier, several darters (e.g., *Etheostoma boschungi, E. trisella*) migrate to small streams (<1 m wide) for breeding (Ryon, 1986; Boschung and Mayden, 2004). Many lake-dwelling fish species also migrate to small tributaries for spawning. Examples include kokanee salmon (*Oncorhynchus nerka*, nonanadromous sockeye) and several species of California sucker, including the federally endangered shortnose and Lost River suckers (*Chamistes brevirostris* and *Deltistes luxatus*) (Moyle, 2002). Headwater streams provide a vital rearing habitat for the young of the many species that spawn there. Many of these species support important fisheries and are likely to suffer declines without access to intact headwaters even if the downstream habitats remain intact. Headwaters serve as spawning and nursery grounds for many of the reasons detailed above, namely that they offer a refuge from high flow, competitors, and predators.

Headwaters Provide Rich Feeding Grounds. Small streams are often areas of concentrated food resources for both permanent residents and migrants. Large inputs of leaves from forested riparian zones, the high retention capacity of small streams, and the high rates of primary productivity in unshaded headwaters mean that these streams are rich in food for primary consumers such as crustaceans and insects. Those organisms are eaten by resident and migrant invertebrate and vertebrate predators, and the large hatches of aquatic insects are important to aerial and terrestrial predators. Small streams also receive considerable input of terrestrial insects; e.g., terrestrial invertebrates were a more important food resource for fishes in a first and second-order stream than a third-order stream (Lotrich, 1973).

Headwaters Provide Thermal Refuges. Small streams offer a thermal refuge for species that spend most of their lives in larger systems. They provide warm refuges from freezing for stream fishes during winter (e.g., Power *et al.*, 1999) and cool refuges during summer (e.g., Curry *et al.*, 1997). The Arkansas darter, *Etheostoma cragini*, uses small first-order streams as a summer refuge from heat and drought in the Ozarks (Radwell, 2001). Arkansas darter populations are also found in intermittent streams in Colorado, where their persistence in temporarily isolated pools depends upon a supply of cool groundwater (Labbe and Fausch, 2000). Brook trout in the Ford River, Michigan, retreat to cooler headwaters in summer (Hayes *et al.*, 1998). The success of quillback and introduced carp in midwestern streams has been attributed to the warming of small streams because of human disturbance of the landscape; native species in decline in this region require cooler tributaries (Karr *et al.*, 1985). If headwater streams are thermally degraded, or if barriers to movement are established, downstream species lose access to these thermal refuges.

Headwaters Provide a Source of Colonists and a Network of Movement Corridors. Biological connectivity between headwater and downstream ecosystems is considerable and essential for the maintenance of species diversity in downstream ecosystems (e.g., Labbe and Fausch, 2000). One way in which small streams maintain diversity in the river network is by providing a source of colonists for recovery of downstream systems following disturbance (Lorion *et al.*, 2000; Progar and Moldenke, 2002; Huryn *et al.*, 2005). Small streams also provide movement corridors for plants and animals across the landscape. Their riparian zones provide cooler and more mesic conditions than those found in the uplands (e.g., Richardson *et al.*, 2005). The flight paths of adult aquatic insects are concentrated along streams and riparian zones, which serve as dispersal corridors (e.g., Petersen *et al.*, 2004).

Headwater Biodiversity Affects the Character and Function of Terrestrial and Downstream Ecosystems

Headwaters Supply Food to Neighboring Ecosystems. The diversity of organisms in headwaters creates food resources for other ecosystems and thus provides another ecological linkage between headwater and neighboring ecosystems: "headwater streams are the vertex of a network of trophic arteries flowing from the forest upland to the oceans" (Progar and Moldenke, 2002). Leaf-shredding insects commonly dominate the aquatic insect fauna in forested headwaters, and the fine particles of organic matter that shredders generate are exported as seston to support foodwebs of ecosystems downstream (Vannote *et al.*, 1980). Elimination of aquatic insects from a headwater stream resulted in a 67% reduction in seston export to downstream ecosystems, which was a greater reduction than was caused by a severe drought (Cuffney *et al.*, 1990). Sufficient numbers of drifting aquatic insects and detritus are exported from fishless headwater tributaries to support 100–2,000 young-of-the-year salmonids per kilometer of larger salmon-bearing streams in Alaska (Wipfli and Gregovich, 2002). Emerging insects and transforming amphibians supply food for terrestrial organisms such as spiders, birds, and bats that forage in the riparian zone of small streams (e.g., Richardson *et al.*, 2005). These nutrient and organic matter linkages support riparian and downstream ecosystems. Their significance has been discussed in greater detail elsewhere (Meyer and Wallace, 2001; Freeman *et al.*, this issue; Wipfli *et al.*, this issue).

Biological Activity in Headwaters Affects Connections to Neighboring Ecosystems. Small streams are sites of intense biological activity, whose consequences influence ecosystems downstream. For example, uptake of DOC in headwaters alters the quality and quantity of DOC exported to downstream ecosystems (Wiegner *et al.*, 2005). Uptake of nutrients in headwaters alters nitrogen and phosphorus loading to ecosystems downstream (Meyer and Wallace, 2001; Alexander *et al.*, this issue; Triska *et al.*, this issue).

THREATS TO SMALL STREAMS

Despite their unique contributions to and importance in maintaining the diversity and functional integrity of entire river systems, small streams are continually under threat by human activity (Meyer and Wallace, 2001). The literature describing the biota of headwaters is replete with examples of species threatened by any number of human activities. Threats include ground-water extraction which, in addition to threatening species associated with small springs (e.g., Hubbs, 1995), has caused tributaries of Kansas streams to go dry, resulting in the extirpation of 16 species from the river system (Cross and Moss, 1987). Land-disturbing activities such as agriculture, logging, mining, and urbanization degrade and eliminate headwater habitats (Meyer and Wallace, 2001). These inconspicuous, unnamed, unmapped, and undocumented ecosystems, many of which are on private property, are thus extremely vulnerable to human impacts. The cumulative impact of degraded headwaters contributes to the loss of ecological integrity in ecosystems downstream.

Small streams are thus a vital part of the biological integrity of our nation's waterways. Degradation of headwater habitats and loss of their connections to larger streams have negative consequences not only for inhabitants of small streams but also for the diversity of downstream and riparian ecosystems. In many respects and locales, the biological integrity of entire river networks may be greatly dependent on the individual and cumulative impacts occurring in the many small streams that constitute their headwaters.

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Quantitative review of riparian buffer width guidelines from Canada and the United States

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Abstract

This paper reviewed the provincial, territorial, and state guidelines for the retention of treed riparian buffers after timber harvest in Canada and the United States. Comparisons amongst jurisdictions were facilitated through the use of a standardized template for the classification of waterbodies. Mean buffer widths varied from 15.1 to 29.0 m for different waterbody types when both countries were combined. However, Canadian jurisdictions had wider buffers (except for intermittent streams). In part, this was due to the high percentage of Boreal jurisdictions in Canada and Southeast jurisdictions in the United States. The Boreal region had the widest buffers while Southeastern jurisdictions had the narrowest buffers. Just under half (~44%) of the jurisdictions investigated had three or more modifying factors in the guidelines. Of these, waterbody type, shoreline slope, waterbody size, and presence of fish were the most common. Boreal and Pacific jurisdictions tended to have a more diverse set of waterbody size classes, waterbody types, and other modifying factors. Jurisdictions from the Midwest, Northeast, and Southeast maintained relatively simple 'one-size-fits-all' guidelines. Jurisdictions without modifying factors for slope or presence of fish applied wider baseline buffers than jurisdictions with these factors. A large percentage of jurisdictions (~80%) allowed some selective harvest in buffers. However, these were often accompanied by relatively restrictive prescriptions. In comparison to the ecological recommendations, buffer widths for most jurisdictions were adequate to protect the aquatic biota and habitats but were, generally, less than recommended widths for terrestrial communities. In the future, two management trends are likely to continue, the shift towards more complicated guidelines and the expansion to larger-scale, watershed planning of riparian areas.

Keywords: Riparian; Management; Buffers; Forestry; Streamside; Guidelines; Canada; United States

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1. Introduction

The diversity of biota in riparian areas reflects a spatially and temporally heterogeneous environment created by the varied processes affecting the riparian ecosystem. These include fluvial disturbances (flooding, erosion, sedimentation, geomorphic channel processes), non-fluvial disturbances (fire, insects, wind), variable light environment, variable soils, variable topography, and other upland influences (Gregory et al., 1991; Naiman et al., 1993; Sagers and Lyon, 1997). Understanding the spatial extent of these processes is a critical component of riparian management. The riparian zone can be examined along three spatial axes. These include: longitudinal, vertical, and transverse (after Malanson, 1993; United States Fish and Wildlife Service, 1997). Most of the past and present research and management efforts focus on the transverse properties of riparian areas, particularly its translation into buffers left after harvest. The retention of buffers has been recommended for controlling erosion and sedimentation (Haupt and Kid, 1965; Patric, 1978; Moring, 1982; but see Steedman and France, 2000), moderating stream temperature and light (Brown, 1969; Helvey, 1972; Aubertin and Patric, 1974; Beschta and Weatherred, 1984; Kochenderfer et al., 1997; Johnson and Jones, 2000), inputting fine and large organic debris (Murphy and Koski, 1989; McDade et al., 1990; Robinson and Beschta, 1990; Van Sickle and Gregory, 1990; Bilby and Bisson, 1992; Duncan and Brusven, 1985; France et al., 1996;

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Reid and Hilton, 1998; Hauer et al., 1999), and maintaining invertebrate communities (Newbold et al., 1980; Noel et al., 1986; Carlson et al., 1990; Collier and Smith, 1998; Rask et al., 1998; Whitaker et al., 2000), fish communities (Wesche et al., 1987; Young et al., 1999), nearshore vegetation (Johnson and Brown, 1990; Darveau et al., 1998; Harper and MacDonald, 2001), bird communities (Gilmer et al., 1978; Johnson and Brown, 1990; Darveau et al., 1995; LaRue et al., 1995; Spackman and Hughes, 1995; Van der Haegen and Degraaf, 1996; Ewins, 1997; Kinley and Newhouse, 1997; Whitaker and Montevecchi, 1997; Meiklejohn and Hughes, 1999; Whitaker and Montevecchi, 1999; Whitaker et al., 2000), and mammals (Brusnyk and Gilbert, 1983; Servheen, 1983; Unsworth et al., 1989; Leach and Edge, 1994; Van der Haegen and Degraaf, 1996; Collins and Helm, 1997; Darveau et al., 1998; Darveau et al., 2001; Forsey and Baggs, 2001; but see De Groot, 2002). One underlying objective in riparian management has been to translate the spatial extent of riparian processes and patterns into management practice particularly buffer widths.

The use of riparian buffers has a long history in forestry. Implementation of treed corridors along waterbodies dates back to the 1700s in European forest management (Porter, 1887). The practice of leaving buffers was first applied in United States in late 1960s (Calhoun, 1988 reference in Brosofske et al., 1997). The primary reasons for the use of buffers today are similar to their historical use. For many jurisdictions, the underlying objective is the isolation of upland activities from terrestrial nearshore and aquatic areas. Despite the similarity of purpose, jurisdictions vary widely in the guidelines used in applying buffers. Variances in buffer widths could reflect differences in the integration of ecological, economic, and social factors. As an example, mountainous regions could be more likely to emphasize slope and drainage area in guidelines. Jurisdictions also face differing degrees and types of public and stakeholder scrutiny and economic incentives. Regions vary in the levels of competing interests in the forested lands such as aboriginal, recreation, rural home development, or fisheries. The complexity of guidelines could also reflect the interests of these groups as well as the response of mangers to demonstrate due diligence through greater guideline complexity. For most jurisdictions, the resultant riparian guidelines are a process of weighing all these factors and devising a compromise amongst often conflicting values for riparian areas.

Our primary objective is to review and analyze the structure and underlying riparian values embodied in forest management guidelines throughout jurisdictions in Canada and the United States. It is not an examination of the how effective these guidelines are in maintaining riparian values; this would require an examination of empirical data on water quality and aquatic and terrestrial habitat, and biota. Instead, this paper focuses on buffer width guidelines as one manifestation of resource management used to maintain riparian values. Specific objectives include: (1) comparison of national and regional differences in buffer widths, (2) comparison of modifying factors in structuring guidelines amongst regions, and (3) comparison of guidelines associated with harvest within buffers.

2. Data and analytical methods

A database of riparian management guidelines and regulations was obtained by contacting provinces, territories, and states in Canada and the United States (see Appendix A). We focused on jurisdictions which were able to provide a published record of riparian guidelines. Arizona, District of Columbia, Kansas, New Mexico, and Nunavit were not able to provide these and were not included in this paper. A total of 60 jurisdictions were analyzed. To examine the effect of broad regional geography on riparian guidelines, we categorized jurisdictions into six regions (Table 1). A number of different references were used to create the classification (Bailey and Cushwa, 1981; Environment, 2001; United States Department of Agriculture, 2002). Alaska, Alberta, British Columbia, and Washington were represented in two or more geographic regions. Washington state has separate guidelines for the eastern and western areas of the state. Statistical analysis required at least three jurisdictions in

Table 1
Classification of provinces, territories, and states from Canada and the United States into broad ecological regions

Country	Regions	Jurisdictions
Canada	Boreal	Alberta, Manitoba, Newfoundland, Northwest Territories, Ontario, Quebec, Saskatchewan, Yukon, British Columbia
	Northeast	New Brunswick, Nova Scotia, Prince Edward Island
	Rocky /Intermountain	Alberta, British Columbia
	Pacific	British Columbia
United States	Boreal	Michigan, Minnesota, Wisconsin, Alaska
	Rocky/Intermountain	Colorado, Montana, Utah, Wyoming, Idaho, Nevada, Washington east
	Midwest	Illinois, Indiana, Iowa, Missouri, Nebraska, North Dakota, Oklahoma, South Dakota, Texas
	Northeast	Connecticut, Delaware, Maine, Maryland, Massachusetts, New Hampshire, New Jersey, New York, Ohio, Pennsylvania, Rhode Island, Vermont, West Virginia
	Pacific	Alaska, California, Hawaii, Oregon, Washington west
	Southeast	Alabama, Arkansas, Florida, Georgia, Kentucky, Louisiana, Mississippi, North Carolina, South Carolina, Tennessee, Virginia

Table 2
Standardized template of waterbody types used to facilitate comparisons of guidelines amongst provinces, territories, and states

Waterbody types	Basic description	Size	Slope (%)	Fish-bearing	Drainage basin size
Large permanent stream	Permanent watercourse with defined bank, year-round flows	>5 m width	2.5	No	>50 km^2
Small permanent stream	Permanent watercourse with defined bank, year-round flows	≤ 5 m width	2.5	No	<50 km^2
Intermittent stream	Permanent watercourse with defined bank, no year-round flows	Any width	2.5	No	Not applicable
Small lake	Standing waterbodies	<4 ha	2.5	No	Not applicable
Large lake	Standing waterbodies	>4 ha	2.5	Yes	Not applicable

each category. Within the southwestern region (New Mexico, Arizona, and Nevada) only Nevada had published riparian guidelines, therefore this region was not included in analyses and Nevada was placed in with the Rocky/ Intermountain region.

In order to compare waterbody classifications from different jurisdictions, we applied a standardized template of waterbody types (Table 2) to each jurisdiction. The criteria for the template were based on a preliminary review of all guidelines. Buffer widths and other management prescriptions from each jurisdiction were applied to each waterbody type. In the case of jurisdictions with relatively simple buffer guidelines, one or more classes often had the same buffer width. Other jurisdictions with more complex guidelines would often have to be re-interpreted. As an example, Wisconsin classifies streams as ‘navigable’, we interpreted this as large permanent streams. All subsequent analyses and descriptions of buffer widths were based on the template waterbody types. All buffer widths were reported in metric values.

Guidelines often included factors that modify the baseline buffer width assigned to a waterbody (e.g. presence of fish). The diversity and relative frequencies of different modifying factors were described for all jurisdictions. The use of complementary (i.e. two or more) modifying factors was examined by recording the paired frequency of factors and comparing this to the expected frequency based on independent selection. We also explored the changes to buffer widths associated with the application of five different modifying factors; waterbody type, waterbody size, slope, presence of fish, and selective harvest. Analysis of slope and presence of fish was limited to medium or large streams. Our purpose was to demonstrate general patterns of change to buffer widths rather than to exhaustively catalogue all combinations of factors to all types of waterbodies in all jurisdictions.

Tests of normality distributions on buffer widths indicated that datasets were slightly right skewed, however, all were within range of a normal distribution. As such, we applied a single factor analysis of variance (ANOVA). If significant differences were found, then multiple comparisons were detected using a Tukey Kramer HSD test. Of interest to managers is whether the combinations of factors are selected independently or whether modifying factors are selected by specific combinations. To test for biases in combinations of modifying factors, the probability of independent selection of factors was based on multiplying their individual occurrence within different jurisdictions to determine the random co-occurrence. This was compared to their actual co-occurrence amongst jurisdictions. A chi-squared test was used to compare expected (i.e. random) and actual occurrence. Throughout all statistical tests, a 5% probability was used as a criterion for significance. All analyses were executed on the JMP Statistical Program ver. 4.0.2 (SAS, 2000).

3. Results

3.1. Comparison of Canadian and American jurisdictions

Mean buffer widths varied from 15.1 to 29.0 m for each waterbody type for all jurisdictions from Canada and the United States combined (Table 3). In general, the pooled values from jurisdictions in the United States exhibited narrower buffer widths than in Canada for similarly classified waterbodies (Table 3). For most waterbodies, the mean buffer widths were 33–58% larger across

Table 3
The mean (S.E.) buffer widths summarized for jurisdictions from Canada and the United States combined, and separately for each country. Letters denote significant differences for waterbody types between Canada and the United States (ANOVA, d$f = 1$, post hoc Tukey Kramer HSD test, d$f = 1$, $P < 0.05$)

Waterbody classes	Combined ($n = 60$)	Canada ($n = 12$)	United States ($n = 48$)
Large permanent streams	28.1 (2.7)	43.8 (9.1) a	24.2 (2.3) b
Small permanent streams	21.8 (1.7)	29.6 (4.9) a	19.9 (1.7) b
Intermittent streams	15.1 (1.7)	13.8 (3.2) a	15.5 (2) a
Small lakes	27.6 (3.0)	47.1 (10.9) b	22.9 (2.1) b
Large lakes	29.0 (3.2)	54.6 (11.4) a	22.7 (2.1) b

Table 4
Comparison of regional differences among mean (S.E.) buffer widths for different waterbody types. Letters denote significant differences (ANOVA, $df = 1$, post hoc Tukey Kramer HSD test, $df = 1$, $P < 0.05$)

Waterbody types	Boreal ($n = 13$)	Rocky/Intermountain ($n = 9$)	Pacific ($n = 6$)	Northeast ($n = 16$)	Midwest ($n = 9$)	Southeast ($n = 11$)
Large permanent streams	39.1 (5.6) a	24.4 (7.2) ab	24.3 (8.0) ab	29.7 (7.2) ab	25.7 (5.9) ab	19.4 (3.0) b
Small permanent streams	26.3 (2.6) a	24.2 (7.2) ab	22.7 (7.9) ab	23.7 (4.1) ab	14.4 (1.2) b	17.5 (2.7) b
Intermittent streams	13.9 (3.0) ab	24.2 (7.2) a	21.7 (8.0) ab	13.1 (3.1) ab	11.5 (1.9) b	12.1 (3.4) ab
Small lakes	45.8 (8.2) a	23.0 (6.8) ab	22.7 (3.5) ab	30.6 (7.2) ab	21.7 (5.6) b	17.4 (2.8) b
Large lakes	52.2 (8.8) a	23.0 (6.8) ab	22.7 (3.5) b	30.2 (7.2) ab	21.7 (5.6) b	17.4 (2.8) b

Canadian jurisdictions. The exception was intermittent streams where there was no significant difference (Table 3).

3.2. Regional patterns

There were significant differences in the width of buffers across the different ecological regions. In general, the Boreal region had the widest buffers for all waterbody types except intermittent streams (Table 4). Mean Boreal buffer widths ranged from 13.9 m for intermittent streams to 52.2 m for large lakes. In contrast, the Southeast region had the narrowest mean widths ranging from 12.1 m for intermittent streams to 19.4 m for large streams. Rocky/Intermountain and Pacific regions had relatively little variance amongst waterbody types (±3 m). Both these regions had the widest buffers on intermittent streams. Northeast and Midwest also had relatively little variance amongst types except for intermittent streams which had buffers about half the width of other waterbody types.

3.3. Modifying factors

Across many jurisdictions, a number of modifying factors were commonly used in guideline formulation (Table 5). Just under half (44%) in North America had three or more modifying factors in the guidelines. Thirteen jurisdictions surveyed (22%) used only a single factor. Across all jurisdictions, waterbody type, slope, waterbody size, and presence of fish were the most common modifying factors (Table 5). Less common factors included: drinking water/aesthetics, drainage basin area, forest management practices adjacent to waterbodies, presence of saltwater flow, types of shoreline vegetation, upstream of fishbearing waterbodies, threat of downstream sediment transport, and flow rates.

Both Boreal and Pacific regions had the most diverse set of modifying factors (Table 5). Of the twelve most common factors, Pacific jurisdictions utilized a total of 11 with a mean of 4.8 factors per jurisdiction, while Boreal jurisdictions utilized nine with a mean of 3.5 factors per jurisdiction. Overall, Northeast and Rocky/Intermountain jurisdictions utilized a similar number of factors (8 and 9, respectively) as the Boreal, but jurisdictions within each of these regions featured means of 2.0 and 2.4 factors, respectively. Lastly, Midwest and Southeast guidelines had a relatively low number of factors, 5 and 6, respectively. Mean numbers of factors in these regions were 2.1 and 2.5 factors per jurisdiction, respectively.

Of the jurisdictions that used more than a single modifying factor, the most common combinations of two

Table 5
Mean number of modifying factors and the percentages of jurisdictions using different modifying factors assessed across all jurisdictions and each region

Modifying factor	All	Boreal ($n = 13$)	Rocky/Intermountain ($n = 9$)	Pacific ($n = 6$)	Northeast ($n = 16$)	Midwest ($n = 9$)	Southeast ($n = 11$)
Mean no. per jurisdiction	2.7	3.5	2.4	4.8	2.0	2.1	2.5
Waterbody type	78.7	91.7	77.8	66.7	68.8	88.9	100.0
Slope	49.2	25.0	44.4	50.0	43.8	66.7	63.6
Waterbody size	32.8	58.3	33.3	50.0	25.0	33.3	27.3
Fishbearing	32.8	58.3	33.3	83.3	18.8	11.1	36.4
Drinking water/aesthetics	14.8	16.7	22.2	33.3	18.8	11.1	9.1
Drainage basin area	6.6	25.0	0.0	0.0	6.3	0.0	0.0
Shoreline forest management	8.2	33.0	11.1	16.7	6.3	0.0	0.0
Saltwater flow	9.8	8.3	0.0	33.3	6.3	0.0	18.2
Shoreline vegetation	4.9	25.0	11.1	16.7	0.0	0.0	0.0
Upstream of fishbearing	4.9	0.0	9.1	16.7	6.3	0.0	0.0
Downstream sediment threat	3.3	0.0	0.0	33.3	0.0	0.0	0.0
Flow rates	3.3	0.0	0.0	33.3	0.0	0.0	0.0

Table 6
Most frequent combinations (pairs) of modifying factors used by jurisdictions. *N* represents the number of jurisdictions utilizing a particular combination. The expected percentage is based on independently selecting combinations of modifiers from the frequencies in Table 5

Combination of modifiers	*N*	Actual percentage	Expected percentage
Waterbody type—slope	23	39.0	38.7
Waterbody type—waterbody size	18	30.5	25.8
Waterbody type—fishbearing	18	30.5	25.8
Slope—fishbearing	11	18.6	16.1
Waterbody size—fishbearing	9	15.3	10.8
Fishbearing—drinking water/aesthetics	7	11.9	4.9
Waterbody type—drinking water/aesthetics	7	11.9	11.6

factors were waterbody type with either slope, waterbody size, or presence of fish (Table 6). Combinations of waterbody type with these modifiers were found in $>30\%$ of the guidelines examined. With the exception of waterbody type and slope, and waterbody type and drinking water/aesthetics, all other common combinations were utilized more frequently than would be expected based on independent selection of modifying factors (Table 6). That is, managers appear to be selecting combinations of modifiers to complement each other.

3.4. Waterbody types

Not surprisingly, streams were the most commonly recognized waterbody types across all guidelines (Table 7). Under half of jurisdictions ($\sim 39\%$) further recognized differences between intermittent and permanent flow streams. Lakes and wetlands were the third and fourth most recognized classifications, respectively. Less frequently used types included: waterbodies with exceptional aesthetics or heritage value, ponds, natural springs, saltwater/brackish estuaries, coldwater/warmwater bodies, braided streams, and manmade impoundments and canals.

Across all provinces, territories, and states, a mean number of 2.5 waterbody types were recognized per jurisdiction. Pacific, Boreal, and Southeast jurisdictions had the greatest diversity of waterbody types (8 or 9) and the recognition of 3.0–3.7 waterbody types per jurisdiction (Table 7). Rocky/Intermountain guidelines recognized a mean of 2.7 waterbody types per jurisdiction, with a total diversity of six waterbodies types. Both Northeast and Midwest jurisdictions recognized the least number of waterbody types, 1.8 and 1.9 per jurisdiction, respectively, and a total diversity of 5 and 4 waterbody types, respectively.

3.5. Slope

Results suggested that jurisdictions that do not incorporate shoreline slope as a modifying factor had wider baseline buffers to account for the potential presence of a sloped shoreline. On the other hand, jurisdictions that had specific guidelines for slope had narrower baseline buffers when there was no slope than jurisdictions without slope guidelines. The mean (S.E.) buffer width at 0% slope for jurisdictions with slope modifiers (16.8 m (2.9)) was significantly narrower than jurisdictions without slope guidelines (33.1 m (3.0); ANOVA, $P < 0.05$). Furthermore, in jurisdictions with slope guidelines, the mean (S.E.) addition to the baseline buffer width was 0.79 m (0.08) for each 1% increase in slope. Based on this relationship, we can crudely estimate the degree to which jurisdictions without slope guidelines extend their buffer width. A mean additional buffer of 16.3 m (33.1 − 16.8 m) could potentially account for 21% of slope change in jurisdictions without slope guidelines (Table 8). Northeast, Rocky/Intermountain, Pacific, and Boreal jurisdictions with no slope guidelines had the widest baseline buffers (Table 8). Across these regions, differences between baseline buffers of jurisdictions with and without slope guidelines varied from 18.6 to 29.7 m and accounted for 26–35% of slope. In

Table 7
Mean number of delineated waterbody types per jurisdiction and percentages of waterbody types assessed across all jurisdictions and by regions

Waterbody types	All	Boreal ($n = 13$)	Rocky/Intermountain ($n = 9$)	Pacific ($n = 6$)	Northeast ($n = 16$)	Midwest ($n = 9$)	Southeast ($n = 11$)
Mean no. per jurisdiction	2.5	3.0	2.7	3.7	1.8	1.9	3.2
Streams	69.7	84.6	77.8	83.3	43.8	66.7	90.9
Permanent/intermittent	39.4	38.5	33.3	50.0	18.8	22.2	90.9
Lakes	39.4	69.2	55.6	83.3	25.0	22.2	9.1
Marshes/bogs/wetlands	18.2	30.8	33.3	33.3	0.0	11.1	18.2
Aesthetics/heritage	10.6	15.4	11.1	16.7	12.5	0.0	9.1
Ponds	6.1	15.4	0.0	0.0	6.3	0.0	9.1
Estuaries	6.1	7.7	0.0	33.3	0.0	0.0	9.1
Natural springs	4.5	7.7	11.1	16.7	0.0	0.0	0.0
Cold/warmwater flows	3.0	7.7	0.0	0.0	0.0	0.0	9.1
Braided streams	1.5	0.0	0.0	0.0	0.0	0.0	9.1
Manmade waterbodies	1.5	0.0	0.0	16.7	0.0	0.0	0.0

Table 8
Buffer widths (m) from large streams when shoreline slope is used as a modifier for determining width. Baseline widths are the mean (S.E.) values at 0% slope for jurisdictions with slope guidelines. The No Guideline column is the mean (S.E.) buffer widths for jurisdictions without slope as a modifying factor. The rate column is the change in the mean additional buffer width with each percent increase in slope for jurisdictions with slope guidelines. Slope Accounted (%) represents the amount of slope that could be accounted by the wider baseline buffers in jurisdictions without slope guidelines

Region	Baseline width with guidelines (m)	Rate (m/%)	No. guideline width (m)	Difference (m)	Slope accounted (%)
All	16.8 (2.9)	0.79	33.1 (3.0)	16.3	21
Boreal	30.2 (4.4)	0.62	48.8 (6.9)	18.6	30
Rocky/Intermountain	17.7 (2.7)	0.73	43.2 (9.5)	25.5	35
Pacific	18.0 (3.6)	1.12	47.7 (11.8)	29.7	26
Northeast	15.4 (1.3)	0.79	39.6 (9.0)	24.3	31
Midwest	14.0 (2.1)	0.77	16.5 (2.3)	2.5	3
Southeast	10.1(1.0)	0.56	19.8 (4.2)	9.6	17

contrast, Southeast and Midwest regions exhibited a much lower difference between jurisdictions with and without slope guidelines. Buffers from Southeast and Midwest jurisdictions without slope could potentially account for slopes of 17 and 3%, respectively.

3.6. Waterbody size

Table 9 summarizes the mean ranges used to delineate size classes within waterbody types. Relatively few jurisdictions further classified waterbody types into different size categories beyond defining minimum size criteria. Only 30, 10, and 10% of jurisdictions established further size classes for streams, lakes, and wetlands, respectively. In pooling all jurisdictions, we found three size categories of streams, two size categories of lakes, and two size categories of wetlands. In applying mean values to the breakpoints of these size classes, we estimated that small streams were <5.0 m in width, medium streams were between 5.0–9.3 m while large streams were >9.3 m. Small lakes were >0.9–4.3 ha, while large lakes were >4.3 ha. Small wetlands were 0.3–2.3 ha, while large wetlands were >2.3 ha. Boreal, Rocky/Intermountain, and Pacific region jurisdictions featured the greatest number of divisions by size class within waterbody types. Amongst this group there were relatively few differences between size class boundaries except the Boreal region, which classified large wetlands as >5.0 ha. Both the Midwest and Southeast regions featured three classes for streams, a single class for lakes, and no delineation for wetlands (Table 9). Lastly, the pooled data for the Northeast region indicated two stream classes, and a single class each for lakes and wetlands (Table 9).

It is worth noting that most jurisdictions have some categories reserved for areas of exceptional value, usually historic or natural sites. These areas usually have much wider buffers. Furthermore, a number of jurisdictions ($n = 7$) do not use direct measurements of channel width or surface area as criteria for separating waterbody types. In these jurisdictions, size was part of the formulation but other criteria were also considered. West Virginia and Saskatchewan base their classifications on stream order rather than a direct metric of channel width, while Wisconsin utilizes channel navigability. The Northwest Territories emphasizes the terrestrial and riparian interface as well as floodplain width to classify streams. A number of maritime jurisdictions, (i.e. New Brunswick, Newfoundland, and Nova Scotia), base their classifications on mapping units such as delineation on 1:50,000 maps.

3.7. Presence of fish

Like those for slope guidelines, jurisdictions with fish guidelines have narrower baseline buffers than those without fish guidelines (Fig. 1). Across all jurisdictions, the mean (S.E.) baseline buffer widths for large non-fishbearing streams in jurisdictions that utilize fishbearing modifiers was 18.5 m (4.9), however, there was a significant increase to 45.7 m (6.4) for large streams with fish (ANOVA, $P < 0.05$). In comparison, jurisdictions without

Table 9
Mean waterbody size classes for streams (m), lakes (ha), and wetlands (ha) summarized across all jurisdictions and for each region

Waterbody type	All	Boreal	Rocky/Intermountain	Pacific	Northeast	Midwest	Southeast
Small streams	0.1–4.9	0.4–3.0	0.2–3.7	0.3–3.8	0.0–5.3	0.0–6.1	0.0–6.3
Medium streams	5.0–9.3	3.1–4.3	3.8–5.0	3.9–5.0	>5.3	6.2–12.2	6.4–12.2
Large streams	>9.3	>4.3	>5.0	>5.0		>12.2	>12.2
Small lakes	0.9–4.3	2.1–.5	2.8–5.0	0.4–5.0	>1.1	>0.0	>0.0
Large lakes	>4.3	>7.5	>5.0	>5.0			
Small wetlands	0.3–2.3	0.3–5.0	0.4–3.5	0.6–3.5	>0.0		
Large wetlands	>2.3	>5.0	>3.5	>3.5			

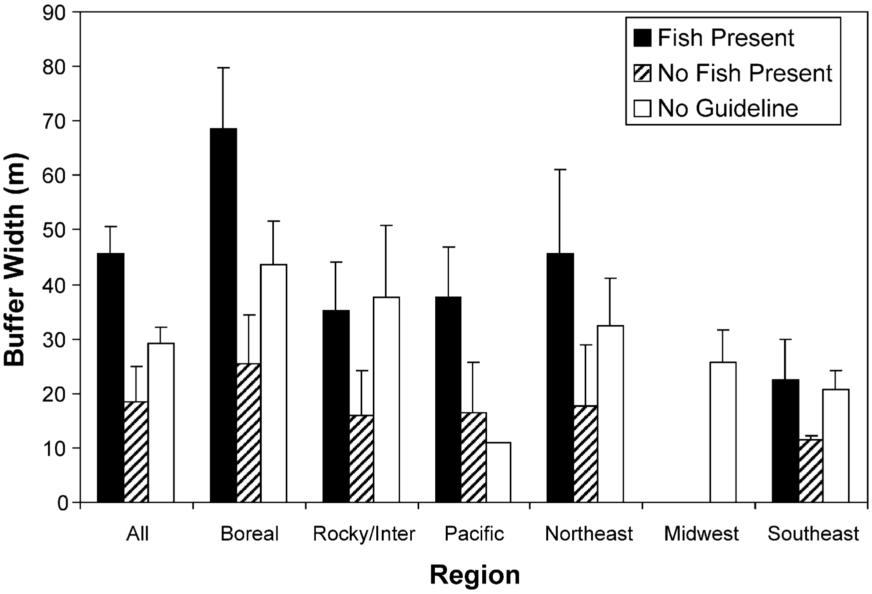

Fig. 1. Mean buffer widths of large streams with fish (first bar) and without fish (second bar) for jurisdictions with fish guidelines, and jurisdictions without fish guidelines (third bar). Error bars represent standard error.

fish guidelines had an intermediate mean baseline (S.E.) buffer of 29.1 m (3.1). There appears to be some compensation for not having fish guidelines by maintaining wider baseline buffers in jurisdictions without fish guidelines. This pattern was present in all regions except the Midwest, where there were no jurisdictions with fish guidelines and for the Pacific where all jurisdictions except Hawaii had fish guidelines (Fig. 1).

3.8. Patterns of selective harvest

About 80% of all jurisdictions allowed some harvest within buffers. Unlike slope or fish guidelines, most jurisdictions added no additional buffer width to areas that permitted harvest within buffers. The mean (S.E.) width of buffers amongst jurisdictions permitting harvesting was 27.4 m (2.9). Surprisingly, jurisdictions that did not allow harvesting had slightly wider buffers, 34.3 m (5.5). Amongst regions, all jurisdictions in the Midwest allowed for selective harvest within buffers, while 62% of Boreal jurisdictions allowed harvest. The remainder of regions were ordered Pacific (83%), Northeast (75%), Southeast (73%), and Rocky Mountain/Intermountain (67%). Like the pooled dataset, there were no clear patterns amongst regions in terms of buffer widths for jurisdictions with and without selective harvest. Jurisdictions with selective harvest in Boreal, Northeast, and Pacific regions had mean buffers wider than jurisdictions without selective harvest, while in Rocky/Intermountain and Southeast jurisdictions the reverse was true (Fig. 2). In neither case were any of the differences statistically significant (ANOVA, $P > 0.05$).

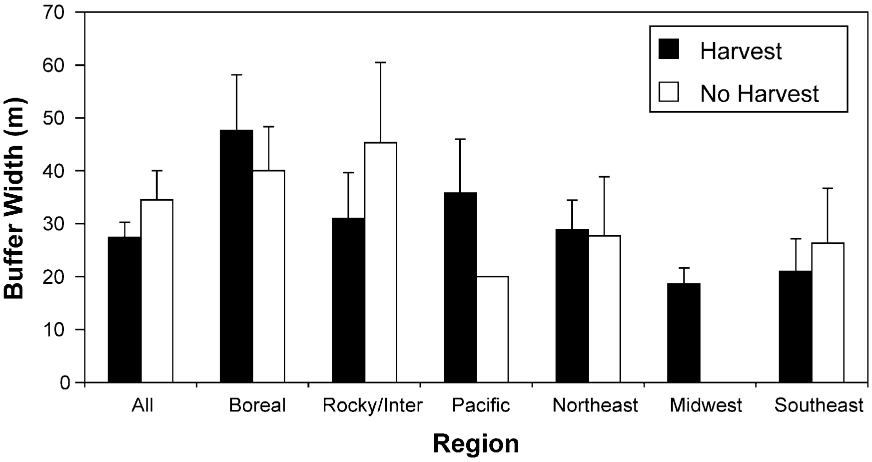

Fig. 2. Mean buffer widths on large streams for jurisdictions with selective harvest (first bar) and jurisdictions without selective harvest (second bar). Error bars represent standard error.

4. Discussion

From an ecological perspective, the primary goal for managers is to match or exceed the width of guidelines to those recommended in the ecological literature. Fortunately, a large body of research literature on riparian function, structure, and biota is available with many studies putting forward recommendations on buffer width. Both theoretical and empirical data are available for sedimentation and erosion control, and stream temperature regulation. As an example, Wong and McCuen (1982) modeled the impacts of substrate characteristics, slope, vegetation roughness, and overland flow patterns and suggested that under most circumstances, buffers less than 60 m were sufficient to control sedimentation. Applying a different model, Cook College Department of Environmental Resources (1989) found that buffer widths from 15 m for slopes less than 1% and 61 m for slopes under 15% would be adequate for sediment reduction. Results from field studies suggest more varied results. Moring (1982) reported that 30 m buffers were unable to prevent increases in stream sedimentation after a partial clearcutting of small watersheds in Oregon. In contrast, Haupt and Kidd (1965) found that 9 m strips were sufficient to remove sediments from cutblock features in Rocky Mountain areas of central Idaho. Low relief boreal systems are unlikely to yield large amounts of sediment even without buffer strips. Steedman and France (2000) found no significant sediment deposition after shoreline harvesting to small coldwater lakes in the Canadian Shield.

In a review of forest practices, Binkley and Brown (1993) noted that almost universal use of intact or partially harvested buffers had significantly reduced increases in stream temperature after harvesting. A number of models exist for prescribing buffer widths to control solar radiation (Beschta and Weatherred, 1984). The important variables include: stream width and volume, buffer forest height and density, amount of watershed cut, solar inputs, and groundwater temperatures. These models suggest that buffers >30 m wide are sufficient to prevent steam temperatures from rising. Aubertin and Patric (1974) found that partially cut (~50% retention) buffers (10–20 m wide) were still able to sufficiently shade streams and prevent temperature increases. Removal of buffers from patch cut and clearcut/burned basins in the H.J. Andrews Experimental Forest in the western Cascades, Oregon, increased the maximum summer temperature earlier in the season (Johnson and Jones, 2000).

Other empirical studies indicate that the removal of buffers from stream systems causes increases in water temperature and light, and subsequent changes in aquatic biota. In a comparison of undisturbed and partially harvested (26–54% tree removal, 6–17 years previously) stream segments in northeast Oregon, Carlson et al. (1990) reported macroinvertebrate densities 20–113 times greater in logged sites, although diversity was the same between logged and undisturbed sites. The increased densities were particularly notable in lower elevation streams and those less shaded by vegetation. Similarly, greater amounts of light after logging led to increases in the density of both invertebrate and periphyton communities in small, buffered (8–9 m) streams in northern New England (Noel et al., 1986). Newbold et al. (1980) studied the impact of varying stream buffer widths on invertebrate communities in northern California streams. Streams with buffers ≥30 m exhibited no impact of harvest on invertebrates, while streams with buffers <30 m experienced changes in species diversity. For the narrower buffers, the changes in diversity were positively correlated to buffer width. Young et al. (1999) found that non-anadromous cutthroat trout streams harvested to streamside margins reached a maximum summer temperature of 30 °C, which was correlated with a four-fold decline in fish density. Fish populations recovered after stream temperatures decreased following revegetation by shrubs and trees. Wesche et al. (1987) found that overhead bank cover provided by riparian vegetation explained the greatest amount of variation in trout population size in Wyoming streams.

In stream and possibly small lake systems, most of fine and large organic debris is derived from treed riparian buffers. France et al. (1996) found that harvesting of the riparian canopy around Boreal Shield lakes reduced the allochthonous inputs of small woody debris by 90%. Similar declines in allochthonous materials have been noted in a number of forested, small stream systems in which treed riparian buffers have been removed (Bilby and Bisson, 1992; Duncan and Brusven, 1985). Treed riparian areas are the source for large woody debris in stream ecosystems (reviewed in Sedell et al., 1990). A number of studies have demonstrated that most (>90%) of the coarse woody inputs originated within a single tree length of the stream bank (Robinson and Beschta, 1990; Van Sickle and Gregory, 1990). In Washington and Oregon, a 30 m strip on one side of a stream provided 85% of the natural uncut input, but a 10 m strip provided less than half the natural input (McDade et al., 1990).

In general, buffer widths recommended for protection of terrestrial riparian components were wider than those recommended for aquatic components. Harper and MacDonald (2001) demonstrated that a distinct lakeshore forest edge community extended for about 40 m around boreal lakes in central Alberta. Johnson and Brown (1990) compared the forest composition of buffers strips (~80 m wide) left after timber harvest to the composition of undisturbed lakeshore forests in Maine. They reported that shrub densities were greater in the buffer strips, but tree and snag densities were greater in undisturbed lakeshore forests. A comparison of different widths and disturbance levels of riparian buffers in mature balsam fir forests in Quebec indicated that narrower buffers (20 m intact and 20 thinned) exhibited greater stem densities of conifer and deciduous shrubs than wider buffers (>40 m) and uncut controls (Darveau et al., 1998).

With respect to maintenance of riparian bird assemblages, Spackman and Hughes (1995) argued that buffer widths of 150 and 175 m would be required to maintain 90 and 95% of preharvest species along mid-order streams in Vermont. Kinley and Newhouse (1997) argued for riparian buffers wider than 50 m in order to maintain bird densities and prevent changes to community structure in hybrid white × Engelmann spruce-lodgepole pine forests of southeastern British Columbia. A comparison of buffer strips (~80 m) left after timber harvest and undisturbed lakeshore areas in Maine indicated that density and species richness were lower in the buffer strip (Johnson and Brown, 1990). In contrast, Whitaker and Montevecchi (1999) found that buffer strips contained a higher total avian abundance than forested streamside controls. This was attributed to the presence of edge and clearcut tolerant species. Riparian buffers were able to maintain many riparian and woodland species, however, interior forest species required wider buffers and some were not present in even the widest buffers (~60 m). Darveau et al. (1995) noted an increased bird density in streamside buffers the year after cutting balsam fir forests of Quebec. These differences declined in following years. Density increases were greatest and subsequent declines were faster in narrower (20 m) buffers. However, thinning (33% tree removal) of 20 m strips did not appear to have as much of an impact on bird species as reducing buffer width did. They suggested that buffer widths of 60 m could support forest dwelling species, whereas buffer widths of 20 m were more useful to ubiquitous species.

For most large mammals, buffers left after harvest are not wide enough to provide source habitat, however, they may provide sufficient cover for foraging and travel. In west-central Idaho, Unsworth et al. (1989) recommended that forested buffers along streams, roads, and dense stands on north-facing slopes be retained for bear cover and bedding. Van der Haegen and Degraaf (1996) found that black bears used riparian buffers as travel corridors in harvested stands in Maine. Brusnyk and Gilbert (1983) found that moose densities were greater in riparian buffer strips (60 m) left after harvesting than in blocks that did not retain buffers.

In contrast, the impact of riparian buffers on small mammals appears variable. Forsey and Baggs (2001) found that track counts were greater for Newfoundland marten, snowshoe hare, and red squirrel in interior, uncut forests than riparian areas, whereas track counts were greater in riparian strips (20 m) after cutting. The authors concluded that buffer strips left after harvest were valuable to these species. Darveau et al. (2001) studied small mammals in balsam fir forests along streams in Quebec and found no difference in the abundance of the two most common small mammal species among buffers of varying width (20, 40 and 60 m). They also reported that meadow vole, which was absent prior to harvest, invaded clearcuts and was a limiting factor to the occurrence of red back vole and deer mouse in buffer strips. They suggested that 20 m buffer strips may work as refuges for small mammals, but that wider strips would provide a more natural habitat for edge-avoiding species. Contrary to these results, a number of studies found little bias in the distribution of mammals in buffers left after harvest. In comparing red squirrel, northern flying squirrel, and eastern chipmunk population parameters within upland and riparian strips and forested blocks, Cote and Ferron (2001) found no differences among treatments and controls. De Groot (2002) found similar results for small mammals within mixedwood boreal forests in north-central Alberta. Abundances and demographics of red-backed voles, deer mice, and meadow voles estimated through trapping did not differ in riparian forest strips (20–200 m) and controls adjacent to small lakes up to four years after their creation. In balsam fir forests of Quebec, Darveau et al. (1998) reported that snowshoe hares made only minimal use of riparian buffer strips regardless of width (widths of 20, 40 and 60 m were tested).

While it is beyond the scope of this paper to thoroughly review all literature (e.g. Wenger, 1999), the wider recommendations associated with some terrestrial species may reflect a direct loss of habitat with reductions in buffer width. In contrast, habitat loss may only occur with very narrow buffers for some aquatic organisms, e.g. amphibians. Viewed in this light, jurisdictions with wider buffers would tend to capture a greater extent of terrestrial riparian functions, structure, and biota. Twenty-nine (48%) jurisdictions make explicit statements about the protection of aquatic and terrestrial habitats and biota in either their preamble or objectives for riparian management. The remaining jurisdictions focus on the protection of aquatic habitats and biota. Based on the ecological literature, the former group should have wider buffers. However, buffer widths were not statistically different between the two groups for any of the waterbody types (ANOVA; P range 0.22–0.90). This suggests that application of wider buffer widths does not necessarily follow from a desire to expand protection to the terrestrial components of the riparian.

One of the more striking patterns in this study was the variance amongst jurisdictions in the complexity of guidelines. In general, jurisdictions seemed to select between management paradigms that either apply relatively simple, guidelines with few factors or more complex guidelines that utilize a large number of factors. Jurisdictions such as those in the Pacific region have been more proactive in the development of complex guidelines. In contrast, most of the jurisdictions in the Midwest retained relatively simple guidelines. It would be tempting to argue that the greater intricacy in guidelines reflects the greater complexity in the ecological setting of Pacific jurisdictions. In part this maybe true, however, many of modifying factors, waterbody types, and size categories found within Pacific jurisdictions are general enough that they could be applied to other jurisdictions. Viewed in this light, the added complexity may not necessarily stem from an inherently more complex underlying riparian ecology.

One possible explanation may be that jurisdictions in the Pacific region, particularly those of continental North America, support significant forest-based economies both in terms of timber harvest and other non-extractive uses such as recreation. All these jurisdictions have been the focus of intense public and regulatory scrutiny over the past few decades. In general, the political response of government and industry has been to produce more complex guidelines. Modifying factors can be interpreted as a priority list for riparian protection. Slope, presence of fish, drinking water, and aesthetics were relatively frequent across all regions. For some jurisdictions in the Boreal, drainage basin area, shoreline vegetation, and types of forestry activity were also utilized. Pacific regions included these and upstream and downstream effects on fish and sediment and saltwater flows. In designing complex guidelines, a common pattern was to use the major classifiers (waterbody size and type) to develop baseline widths and then increase buffer widths when special factors were present. Our analysis suggested that jurisdictions increase guideline complexity by adding specific combinations of modifiers usually by the addition of slope, fishbearing, or drinking water to other more common modifiers (Table 6). The question of whether this approach has resulted in improvements to the management of riparian habitat and biota remains a point of contention and requires empirical data based on large-scale experimentation.

In the case of slope and fishbearing streams, buffer widths used by jurisdictions without these modifying factors were intermediate to those with modifying factors. In essence, these jurisdictions treated all waterbodies as potentially being bordered by slope or bearing fish. In the case of slope, application of a safety margin through additional width maybe warranted. Relationships between slope, buffer width, and sediment transport are monotonic and relatively continuous (Wong and McCuen, 1982). Additional buffer width produces a relatively predictable result in terms of a safety margin for sediment transport. In particular, waterbodies may be better protected by wider buffers from periodic disturbances such as unusually wet years, catastrophic weather, or catastrophic disturbances to upland areas. During these years there is the potential for greater amounts of run-off and erosion potential. Wider buffers may reduce the risk of sediment transport into waterbodies. A number of stream classification systems utilize the occurrence of high flows as a basis of categorization (e.g. Rosgen, 1996).

The relationship between buffer width and presence of fish is less straightforward. Changes in buffer width may cross multiple thresholds such as those for the input of coarse woody debris (McDade et al., 1990; Robinson and Beschta, 1990; Van Sickle and Gregory, 1990) or regulation of stream temperature (Brazier and Brown, 1973). It is unclear whether those jurisdictions without fish guidelines had established relationships between local fish populations/communities and buffer widths. In this case, the application of a wider baseline buffer may not provide an incremental increase in protection unless the additional width crosses a threshold value.

The finding that Canada has wider buffers than the United States was somewhat surprising. In part, it can be explained by the high proportion of Boreal jurisdictions in Canada and Southeast jurisdictions in the United States. Boreal jurisdictions had some of the widest buffers while Southeastern jurisdictions had some of the narrowest. A number of possibilities exist for differences amongst these regions. Generally, more populated regions have a longer history of development in riparian areas. If much of the riparian has historically been allocated to development, it is much more difficult to apply larger buffers that may take away from an already established user group. Also, areas with longer histories of development may have already significantly altered riparian habitats. Hence, the range of riparian values may be significantly changed or lost.

Most jurisdictions (~80%) allowed for the option of harvest within riparian buffers. Types of harvest included single tree selection, group selection, and zoned harvest. Though jurisdictions differed in prescriptions, the general restrictions were similar throughout. These included: (1) retaining at least half the cover, volume, or basal area, (2) minimizing or eliminating machinery traffic, and other ground disturbance, (3) protecting understory and regeneration, (4) preventing direct shoreline erosion or removal of trees with roots that stabilized shorelines, (5) spatially dispersed cutting (single tree or small group selection), and (6) preventing ‘hi-grading’ of large or exceptional timber value trees.

Harvest within buffers attempts to extract some direct, short-term economic benefit from riparian areas and re-introduce or maintain tree-replacing disturbances. Although this can be viewed as contrary to the longstanding riparian management paradigm of protection through preservation, partial harvest has been argued as a management analogue for natural single tree or small group replacement. Ilhardt et al. (2000) and Palik et al. (2000) argue that partial harvest would fit into a probabilistic model for defining transverse riparian values. They suggest that riparian structure, function, and biota are more likely to be found closer to the water’s edge. Palik et al. (2000) further suggests a gradient of decreasing harvest intensity with distance to water’s edge follows this definition. Their model features a continuous gradient from no harvest areas to single tree selection, small group selection, and large group selection with retention as one moves from water’s edge to the upland. From an ecological impact standpoint, partial harvest within buffers, if carefully executed, seems to have relatively little effect on potential short-term impacts such as stream temperature, however, long-term effects such as the potential reduction of large organic inputs have not been evaluated. In practice, a number of jurisdictions have multiple-management zones along some waterbodies (e.g. Washington, British

Columbia). These feature zones of differential harvest rather than a continuous change but still retain the same underlying principles. A no harvest zone closest to water edge, then a series of zones with increasing degrees of harvest as one moves to the upland.

4.1. Management implications

There are a number of trends in buffer guidelines that have management implications. The first is a shift away from 'one or few-sizes-fits-all' buffers. In part, this has been driven by a combination of economic incentives, improvements in best management practices for timber extraction primarily in skidding and road construction, increase in knowledge base, increase in public scrutiny, and a desire to protect the unique ecology of riparian areas. Resource managers potentially find themselves in an ever-shortening cycle of revising and implementing riparian management guidelines. Our results suggest that for most factors such as fishbearing and slope, one or few-sizes rules may potentially apply buffers that are wider than would be warranted by local site conditions. Hence, some would argue this applies inconsistent criteria to the delineation and protection of riparian values in the field.

The current trend has been towards more 'tailor-made' buffers that vary amongst broadly similar harvest areas to within a single harvest area. The primary benefit in using tailor-made buffers is the application of clear criteria to define the riparian. These criteria are specifically defined by the modifying factors selected by jurisdictions, and buffers are applied in a predictable response to these criteria. The caveat to tailor-made buffers is the greater complexity in guidelines. From the dataset of guidelines in this study, we found a total of 14 broad modifying factors and individual jurisdictions applied from 1 to 6 of these factors. For each factor, there can exist two to many classes. Hence, the addition of a single factor can exponentially increase the number of potential classes applied in the field. The upper limit to the number of classes is often set by the practicality of training personnel, costs with planning and prescribing modifiers in the field, and the compliance requirements and monitoring of buffers. In particular, compliance monitoring is generally considered much easier with simple guidelines. In discussions with many resource managers from both regulatory and industry sides, these issues greatly favor the application of simpler guidelines.

A second trend in the application of riparian buffers is their use within a broader watershed framework. The naturally integrative nature of watersheds and their natural segregation within the landscape makes them attractive tools for land management. The application of buffers classified at a stand level and repeated throughout a watershed may not meet broader objectives. A number of issues require a broader perspective for riparian management. These include: aquatic and terrestrial travel corridors for biodiversity, habitat fragmentation, cumulative effects, and downstream water quality and human consumption. At their earliest inception, buffers were largely used to protect aquatic resources. Increasingly, they are viewed as an important component for the maintenance and dispersal of upland species and other ecological values such as old growth or dispersal corridors. Aquatic components have always had a strong research foundation using the watershed framework (e.g. Naiman et al., 1987). We would argue that the current overall emphasis on stand-level prescriptions is due to the underlying principle of aquatic protection for buffers. If aquatic components are isolated from upland activities through buffers then there is no need to adjust guidelines for watershed effects at least as modifications to buffer guidelines. Other impacts such as water yield and peak flow impact are driven by the percentage of watershed harvested rather than buffer area (Keenan and Kimmins, 1993). Extension of riparian management to terrestrial components requires integrating riparian and upland ecological processes and biota. Research on the watershed implications for terrestrial components has accumulated over the last decade (e.g. Knopf and Samson, 1994; Naiman and Rogers, 1997; Lock and Naiman, 1998) but has not yet been translated into comprehensive guidelines that integrate buffer widths at the watershed scale.

5. Conclusions

The overall goals of riparian protection through the use of buffers meets the ecological recommendations for most aquatic and some terrestrial components of the riparian. Most notably core habitat for medium and large mammals and birds were wider than most current guidelines. In these cases, more research would be required to determine how changes in buffer width alter the overall habitat quality for these biota. It could be argued that the variance amongst jurisdictions in the width of buffers suggests emphasis on differing riparian components. In part, these reflect broad differences in ecoregions. However, other correlates such as the history of land use, degree of public scrutiny, and framework for the guidelines themselves contributes to the overall variance. With the last point, jurisdictions choose between having simple 'one or few sizes fits all' or relatively complex guidelines that considers modifying factors such as the presence of fish, slope, and other factors. The number of potential classes for riparian buffers greatly increases with the addition of even a few modifying factors. In the future, two management trends are likely to continue, the shift towards more complicated guidelines and the expansion to larger-scale, watershed planning of riparian areas.

Table A1
Reference list of guidelines from different jurisdictions in Canada and the United States used in this paper

Jurisdiction	Reference
Alabama	Alabama Forestry Commission (Undated). *Alabama's Best Management Practices for Forestry*. Montgomery, Alabama. 28 p. Alabama Forestry Commission. http://www.forestry.state.al.us/publication/BMPs/BMPs.pdf (accessed 2002)
Alaska	Division of Forestry, Department of Natural Resources. (2000). *Alaska Forest Resources and Practices*. Anchorage, Alaska. 22 p. Division of Forestry, Department of Natural Resources. http:/www.dnr.state.ak.us/forestry/pdfs/forprac.pdf (accessed 2002)
Arkansas	Arkansas Forestry Commission. (Undated). *2.0 Streamside Management Zones*. Little Rock, Arkansas. 3 p. Arkansas Forestry Commission. http:/www.forestry.state.ar.us/bmp/smz.html (accessed 2002)
California	California Department of Forestry and Fire Protection. (2000). *California Forest Practice Rules 2000*. Sacramento, California. 230 p. Resource Management, Forest Practice Program, California Department of Forestry and Fire Protection. http://fire.ca.gov/forest_practice.html (accessed 2002)
Colorado	Colorado State Forest Service. (1998). *Colorado Forest Stewardship Guidelines to Protect Water Quality: Best Management Practices (BMPs) for Colorado*. Fort Collins, Colorado. 32 p. Colorado State Forest Service, Colorado State University
Connecticut	Connecticut Resource Conservation and Development Forestry Committee. (1998). *A Practical Guide for Protecting Water Quality While Harvesting Forest Products*. Hartford, Connecticut. 36 p. Connecticut Resource Conservation and Development Forestry Committee, Department of Environmental Protection, State of Connecticut
Delaware	Delaware Department of Agriculture, Forest Service. (1996). *Delaware's Forestry Best Management Practices Field Manual*. Dover, Delaware. 71 p. Delaware Department of Agriculture, Forestry Department
Florida	School of Forest Resources and Conservation. (Undated). *Special Management Zones*. Gainesville, Florida. 13 p. Florida Forestry Information, School of Forest Resources and Conservation, University of Florida. http://www.sfrc.ufl.edu/Extension/ffws/smz.htm (accessed 2002)
Georgia	Georgia Forestry Commission. (1999). *Georgia's Best Management Practices for Forestry*. Dry Branch, Georgia. 71 p. Georgia Forestry Commission
Hawaii	Division Forestry and Wildlife. (2001). *Water Protection and Management Program*. Honolulu, Hawaii. 23 p. Division Forestry and Wildlife, Department of Land and Natural Resources, State of Hawaii
Idaho	Idaho Department of Lands. (1996). *State of Forestry for Idaho-Best Management Practices: Forest Stewardship Guidelines for Water Quality*. Coeur d'Alene, Idaho. 33 p. Idaho Department of Lands, Bureau of Forestry Assistance
Illinois	Illinois Department of Natural Resources. (2000). *Forestry Best Management Practices for Illinois*. Springfield, Illinois. 63 p. Division of Forest Resources, Department of Natural Resources
Indiana	Indiana Department of Natural Resources. (1999). *Indiana Logging and Forestry Best Management Practices, BMP Field Guide*. Indianapolis, Indiana. 85 p. Department of Natural Resources, Division of Forestry
Iowa	Iowa Department of Natural Resources. (1998). *Iowa Forestry: Best Management Practices*. Des Moines, Iowa. 65 p. Iowa Department of Resources. http://www.state.ia.us/government/dnr/organiza/forest/bmps3.htm (accessed 2002)
Kentucky	Division of Forestry. (1997). *Kentucky Best Zones. Management Practice No. 3-Streamside Management*. Frankfort, Kentucky. 47-55 pp. Division of Forestry, Department of Natural Resources. http://www.ca.uky.edu/agc/pubs/for/for67/bmp_03.pdf (accessed 2002)
Louisiana	Louisiana Department of Agriculture and Forestry. (1999). *Recommended Forestry Best Management Practices for Louisiana*. Baton Rouge, Louisiana. 83 p. Louisiana Department of Agriculture and Forestry. http://www.ldaf.state.la.us/divisions/forestry/publications.asp (accessed 2002)
Maine	Maine Department of Environment Protection. (1998). *A Field Guide to Laws Pertaining to Timber Harvesting in Organized Areas of Maine*. Augusta, Maine. Publication DEPL W39-B98. 35 p. Maine Forest Service, Department of Conservation Maine Forest Service. (1994). *Erosion and Sediment Control Handbook for Main Timber Harvesting Operations. Best Management Practices*. Augusta, Maine. Publication SHS#22. 48 p. Forest Information Centre, Maine Forest Service, Maine Department of Conservation Maine Department of Environment Protection. (1999). *Maine Shoreland Zoning: A Handbook for Shoreland Owners*. Augusta, Maine. Publication DEPL W1999-2., 34 p. Maine Department of Environmental Protection
Maryland	Maryland Department of Natural Resources—Forest Service. (2000). *A Guide To Maryland Regulation of Forestry and Related Practices*. Annapolis, Maryland. 81 p. Maryland Department of Natural Resources. http://www.dnrweb.dnr.state.md.us/download/forests/frg.pdf (accessed 2002)
Massachusetts	Kittredge, Jr., D.B. and Parker, M. (1996). *Massachusetts Forestry Best Practices Manual*. Pittsfield, Massachusetts.56 p. Bureau of Forestry, Division of Forests and Parks, Department of Environmental Management, Commonwealth of Massachusetts
Michigan	Michigan Department of Natural Resources. (1994). *Water Quality Management Practices on Forest Land*. Lansing Michigan. 9 p. Forest Management Division, Michigan Department of Natural Resources
Minnesota	Minnesota Forest Resources Council. (1999). *Sustaining Minnesota Forest Resources-Voluntary Site-level Management Guidelines for Landowners, Loggers, and Resource Managers. Part 3. Integrated Guidelines*. St Paul, Minnesota. 78 p. Minnesota Forest Resources Council
Mississippi	Mississippi Forestry Commission. (2000). *Mississippi's BMP's: Best Management Practices for Forestry in Mississippi*. Jackson, Mississippi. 15 p. Publication # 107 (Internet Version). Mississippi Forestry Commission, http://www.mfc.state.ms.us/pdf/bmp2000.pdf">http://http://www.mfc.state.ms.us/pdf/bmp2000.pdf (accessed 2002)
Missouri	Missouri Department of Conservation. (1997). *Missouri Watershed Protection Practice*. Jefferson City, Missouri. 29 p. Missouri Department of Conservation

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Jurisdiction	Reference
Montana	Department of Natural Resources and Conservation. (1993). *Montana Guide to the Streamside Management Zone Law and Rules*. Missoula, Montana. 35 p. Department of Natural Resources and Conservation, Department of Natural Resources and Conservation
Nebraska	Nebraska Forest Service. (Undated). *Forestry: Best Management Practices for Nebraska*. Lincoln, Nebraska. 6 p. School of Natural Resource Sciences, University of Nebraska. http://www.ianr.unl.edu/pubs/forestry/nfs/nfs-1.htm (accessed 2002)
Nevada	State of Nevada. (1997). *Nevada Forest Practice Regulations (Statutes) for Forestry. Chapter 528 Forest Practice and Reforestation NRS 528.053. Certain activities prohibited near bodies of water; Nevada Revised Statutes*. Carson City. Nevada. 528-8 pp. Nevada State Legislature. http://www.leg.state.nv.us/lawl.cfm (accessed 2002)
New Hampshire	New Hampshire Division of Forests and Lands. (Undated). *Forest Operations Manual*. Concord, New Hampshire. 31 p. New Hampshire Division of Forests and Lands. http://www.nhdfl.com/for_mgt_bureau/manual/Forest%20Operations%20Manual.pdf (accessed 2002)
New Jersey	New Jersey Forest Service. (Undated). *New Jersey Forestry and Wetlands Best Management Practices Manual*. Jackson, New Jersey. 31 p. Forest Resource Education Center
New York	Division of Lands and Forests. (Undated). *Timber Harvesting Guidelines*. Albany, New York. 4 p. Division of Lands and Forests, New York State Department of Environmental Conservation. http://www.dec.state.ny.us/website/dif/privland/privassist/bmp.html (accessed 2002)
North Carolina	North Carolina Division of Forest Resources. (1989). *Forestry Best Management Practices Manual*. Raleigh, North Carolina. 67 p. North Carolina Division of Forest Resources, Department of Environment, Health, and Natural Resources Department of Environment, Health, and Natural Resources. (1990). *Best Management Practices for Forestry in the Wetlands of North Carolina*. Raleigh, North Carolina. 28 p. Department of Environment, Health, and Natural Resources
North Dakota	North Dakota State Forest Service. (1999). *North Dakota Forestry Best Management Practices*. Bottineau, North Dakota. 29 p. North Dakota State Forest Service
Ohio	Ohio Division of Forestry. (Undated). *Fact Sheet: Best Management Practices for Logging Operations*. Columbus, Ohio. 4 p. Division of Forestry Publications, Ohio Division of Forestry. http://www.hcs.ohio-state.edu/ODNR/Education/pdf/logging.pdf (accessed 2002)
Oklahoma	Oklahoma Cooperative Extension Service. (1998). *Riparian Area Management Handbook*. Stillwater, Oklahoma. Publication E-952. 96 p. Oklahoma Cooperative Extension Service, Oklahoma State University
Oregon	Oregon Department of Forestry. (2002). *Division 635 Water Protection Rules: Purpose, Goals, Classification and Riparian Management Areas. Oregon Administrative Rules 629-635-0000 to 629-635-0310*. Salem, Oregon. 10 p. Oregon State Archives, Oregon Secretary of State. http://www.arcweb.sos.state.or.us/rules/Rules/fpa-635.htm (accessed 2002)
Pennsylvania	Division of Forest Advisory Services. (1999). *Inventory Manual of Procedure for the Fourth State Forest Management Plan*. Harrisburg, Pennsylvania. 49 p. Bureau of Forestry, Department of Conservation and Natural Resources, Commonwealth of Pennsylvania
Rhode Island	Rhode Island Forest Conservators Organization. (Undated). *Best Management Practices for Rhode Island. Water Quality Protection and Forest Management Guidelines*. North Scituate, Rhode Island. Rhode Island Forest Conservators Organization. (accessed 2002)
South Carolina	South Carolina Forestry Commission. (1994). *Best Management Practices: Streamside Management Zones*. Columbia, South Carolina. 4 p. South Carolina Forestry Commission. http://www.state.sc.us/forest/rbsmz.htm (accessed 2002) South Carolina Forestry Commission. (Undated). *Best Management Practices for Braided Systems: A Supplement to the 1994 BMP Manual*. Columbia, South Carolina. 5 p. South Carolina Forestry Commission. http://www.state.sc.us/forest/braid.htm (accessed 2002)
South Dakota	South Dakota Department of Agriculture. (Undated). *South Dakota Forestry Best Management Practices-Forest Stewardship Guidelines for Water Quality*. Rapid City, South Dakota. 32 pp. Resource Conservation and Forestry, South Dakota Department of Agriculture
Tennessee	Division of Forestry. (1993). *Guide to Forestry Best Management Practices*. Nashville, Tennessee. 41 p. Division of Forestry, Tennessee Department of Agriculture
Texas	Texas Forest Service. (2000). *Texas Forestry Best Management Practices*. College Station, Texas. 108 p. Texas Forest Service
Utah	State of Utah, Non-Point Source Task Force. (1998). *Nonpoint Source Management Plan-Silvicultural Activities*. Salt Lake City, Utah. 92 p. Division of Forestry, Fire, and State Lands, Department of Natural Resources
Vermont	Vermont Agency of Natural Resources. (1987). *Acceptable Management Practices for Maintaining Water Quality on Logging Jobs in Vermont*. Waterbury, Vermont. 51 p. Vermont Agency of Natural Resources, Department of Forests, Parks and Recreation
Virginia	Virginia Department of Forestry. (Undated). *Forestry BMP guide for Virginia*. Charlottesville, Virginia. 31 p. Virginia Department of Forestry. http://state.vipnet.org/dof/wq/bmpguide.htm (accessed 2002)
Washington	Washington Forest Practices Board. (2000). *Washington Forest Practices Board Manual: Section 7 Guidelines for Riparian Management Zones*. Olympia, Washington. 44 p. Washington State Department of Natural Resources. http://www.wa.gov/dnr/htdocs/fp/fpb/fpbmanual/se07.html (accessed 2002)
West Virginia	Center for Agricultural and Natural Resources Development. (Undated). *Best Management Practices-Soil and Water Conservation*. Morgantown, West Virginia. 3 p. Center for Agricultural and Natural Resources Development. West Virginia University Extension Service. http://www.wvu.edu/~agexten/forestry/bestprac.htm (accessed 2002)
Wisconsin	Wisconsin Department of Natural Resources. (1997). *Wisconsin's Forestry: Best Management Practices for Water Quality-Field Manual*. Madison, Wisconsin. 76 p. Bureau of Forestry, Wisconsin Department of Natural Resources. http://www.dnr.state.wi.us/org/land/forestry (accessed 2002)

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Table A1 (*continued*)

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Alberta	Alberta Sustainable Resource Development. 1994. *Alberta Timber Harvest Planning and Operating Ground Rules*. Edmonton, Alberta. 57 p. Alberta Sustainable Resource Development. http://www3.gov.ab.ca/srd/forests/fmd/manuals/ProvGR94.doc (accessed 2002)
British Columbia	British Columbia Ministry of Forests. (1995). *Forest Practices Code-Riparian Management Area Guidebook. British Columbia Ministry of Forests*. Victoria, British Columbia. 80 p. British Columbia Ministry of Forests. http://www.for.gov.bc.ca/tasb/legsregs/fpc/fpcguide/riparian/rip-toc.htm (accessed 2002)
Manitoba	Manitoba Conservation. (1990). *Recommended Buffer Zones for Protecting Fish Resources in Lakes and Streams in Forest Cutting Areas*. Winnipeg, Manitoba. 3 p. Forest Planning and Practices, Manitoba Conservation Manitoba Conservation. (1996). *Consolidated Buffer Management Guidelines*. Winnipeg, Manitoba. 4 p. Forest Planning and Practices, Manitoba Conservation
New Brunswick	New Brunswick Department of Natural Resources and Energy. (2000). *A Vision for New Brunswick Forests, Goals and Objectives for Crown Land Management*. Fredericton, New Brunswick. 47 p. New Brunswick Department of Natural Resources and Energy, http://www.gov.nb.ca/dnre/vision.htm (accessed 2002) New Brunswick Department of Natural Resources and Energy. (1999). *Watercourse Buffer Zone Guidelines for Crown Land Forestry Activities*. Fredericton, New Brunswick. New Brunswick Department of Natural Resources and Energy
Newfoundland	Forest Resources and Agrifoods. (Undated). *Schedule IV, Environmental Protection Guidelines for Ecologically Based Forest Resource Management (Stand Level Operations)*. Cornerbrook, Newfoundland. Forest Resources, Forest Resources and Agrifoods
Nova Scotia	Nova Scotia Natural Resources. (1986). *Forest/Wildlife Guidelines and Standards for Nova Scotia*. Truro, Nova Scotia. 19 p. Nova Scotia Natural Resources
Northwest Territories	Forest Management Division, Department of Resources, Wildlife and Economic Development. (2000). *Northwest Territories Timber Harvest Planning and Operating Ground Rules*. Yellowknife, Northwest Territories. 50 p. Forest Management Division, Department of Resources, Wildlife and Economic
Ontario	Ontario Ministry of Natural Resources. (1998). *Timber Management Guidelines for the Protection of Fish Habitat*. Toronto, Ontario. Ontario Ministry of Natural Resources Ontario Ministry of Natural Resources. (1998). *Code of Practice for Timber Management Operations in Riparian Areas*. Toronto, Ontario. Ontario Ministry of Natural Resources. http://www.mnr.gov.on.ca/MNR/forests/forestdoc/guidelines/pdfs/code_prac.pdf (accessed 2002)
Prince Edward Island	Prince Edward Island Department of Agriculture and Forestry. (2001). *Watercourse Buffer Zones: Forestry Operations*. Charlottetown, Prince Edward Island. 2 p. Prince Edward Island Agriculture and Forestry Information Centre, http://www.gov.pe.ca/af/agweb/library/documents/bufferzones/forestry.php3 (accessed 2002)
Quebec	Government du Québec. (1966). *Forest Act, Standards of forest management for forests in the public domain. Division II Section 2*. Gazette Officielle du Québec 128 (19): 2169
Saskatchewan	Saskatchewan Environment and Resources Management. (1985). *Guidelines For The Protection of Fish Habitat During Forest Operations*. Regina, Saskatchewan, Saskatchewan Environment and Resources Management Environment and Resources Management
Yukon	British Columbia Ministry of Forests. (1995). *Forest Practices Code-Riparian Management Area Guidebook.*Victoria, British Columbia. 80 p. British Columbia Ministry of Forests. http://www.for.gov.bc.ca/tasb/legsregs/fpc/fpcguide/riparian/rip-toc.htm (accessed 2001 at the time of reporting Yukon was using British Columbia's guidelines as an interim measure)

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Appendix A

See Table A1.

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Soil physical property changes at the North American Long-Term Soil Productivity study sites: 1 and 5 years after compaction[1]

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Abstract: The impact of forest management operations on soil physical properties is important to understand, since management can significantly change site productivity by altering root growth potential, water infiltration and soil erosion, and water and nutrient availability. We studied soil bulk density and strength changes as indicators of soil compaction before harvesting and 1 and 5 years after harvest and site treatment on 12 of the North American Long-Term Soil Productivity sites. Severe soil compaction treatments approached root-limiting bulk densities for each soil texture, while moderate compaction levels were between severe and preharvest values. Immediately after harvesting, soil bulk density on the severely compacted plots ranged from 1% less than to 58% higher than preharvest levels across all sites. Soil compaction increases were noticeable to a depth of 30 cm. After 5 years, bulk density recovery on coarse-textured soils was evident in the surface (0–10 cm) soil, but recovery was less in the subsoil (10–30 cm depth); fine-textured soils exhibited little recovery. When measured as a percentage, initial bulk density increases were greater on fine-textured soils than on coarser-textured soils and were mainly due to higher initial bulk density values in coarse-textured soils. Development of soil monitoring methods applicable to all soil types may not be appropriate, and more site-specific techniques may be needed for soil monitoring after disturbance.

Résumé : Il est important de comprendre l'impact des interventions dictées par l'aménagement forestier sur les propriétés physiques du sol étant donné qu'elles peuvent modifier de façon significative la productivité d'une station en altérant le potentiel de croissance des racines, l'infiltration d'eau, l'érosion du sol et la disponibilité de l'eau et des nutriments. Les auteurs ont étudié les changements dans la résistance et la densité apparente du sol en tant qu'indicateurs de la compaction du sol avant la récolte ainsi qu'un et 5 ans après la récolte et la préparation du terrain dans 12 stations du projet nord-américain de productivité des sols à long terme. Les traitements de compaction sévère du sol s'approchaient de la densité apparente inappropriée pour les racines pour chaque texture de sol alors que les degrés modérés de compaction du sol se situaient entre des valeurs allant de sévères à celles obtenues avant la récolte. Immédiatement après la récolte dans les parcelles où la compaction du sol était sévère, la densité apparente du sol variait de 1 % moins élevée à 58 % plus élevée qu'avant la récolte pour l'ensemble des sites. L'augmentation de la compaction du sol était observable jusqu'à une profondeur de 30 cm. Après 5 ans, le rétablissement de la densité apparente dans les sols à texture grossière était évident en surface (0–10 cm) mais pas aussi évident en profondeur (10–30 cm); presque aucun rétablissement n'était apparent dans les sols à texture fine. Mesurée en pourcentage, l'augmentation de la densité apparente initiale était plus forte dans les sols à texture fine comparativement aux sols à texture grossière

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surtout à cause d'une densité apparente initiale plus élevée dans les sols à texture grossière. La mise au point de méthodes de suivi des sols applicables à tous les types de sol pourrait ne pas être appropriée et des techniques mieux adaptées à chaque site pourraient être nécessaires pour le suivi du sol après une perturbation.

[Traduit par la Rédaction]

Introduction

Increased forest management and concern over changes in soil productivity are among the topics debated by forest managers and the public. A key element in this debate is the use of mechanized equipment to extract timber products and the subsequent soil compaction and recovery times (Greacen and Sands 1980; Cullen et al. 1991; Froehlich and McNabb 1984; Jansson and Johansson 1998; Landsberg et al. 2003; Miller et al. 2004). A potential consequence of severe soil compaction is the significant loss of site productivity (Powers 1991; Morris and Miller 1994). Where soil compaction occurs, total porosity decreases and soil strength and volumetric water content increase, resulting in increased water runoff and soil erosion, less rooting volume, and poor aeration (Greacen and Sands 1980; Elliot et al. 1998; Williamson and Neilsen 2000). Ultimately, the degree of compaction caused by harvesting or site preparation is affected by soil properties (e.g., texture, organic matter, and water content) at the time of disturbance (Bock and VanRees 2002).

Changes in soil water content from compaction affect temperature flux; which results in altered microclimatic conditions (Fleming et al. 1998), leading to reduced root growth and stand productivity (Greacen and Sands 1980; Gerard et al. 1982). Direct correlations of compaction impacts on forest plant growth are frequently unclear because compaction is often associated with other detrimental disturbances, such as soil displacement, mixing, and rutting. In addition, plant growth on compacted areas (skid trails, landings etc.) has sometimes been found to be greater than on nonimpacted soil because of reduced weed competition (Miller et al. 1989; Miller and Anderson 2002).

Various studies have shown that once compacted, forest soils often recover slowly (many decades) to undisturbed levels of bulk density or soil strength (Sands et al. 1979; Froehlich et al. 1985; Tiarks and Haywood 1996). Recovery rates are dependent on many factors, but chief among them are number of repeated harvest cycles, soil moisture conditions during harvest, soil texture, and rock-fragment content (Miller et al. 1996; Williamson and Neilsen 2000; Liechty et al. 2002). The extent of compaction, initial bulk density, depth of impact, and subsequent soil recovery are all factors that determine the consequences of timber harvesting or site preparation on productivity. In addition, duration and variability of compaction can be significant from site to site or at depth in the soil profile (Beckett and Webster 1971; Blythe and Macleod 1978; Courtin et al. 1983). For instance, variability within soil textural groups, forest stands, or on skid trails can be as great as or greater than the variability between them (Courtin et al. 1983).

Few studies have assessed the long-term effects of compaction on soil productivity or forest sustainability on large, relatively uniform study plots. However, many studies have assessed the effects of harvesting operations or skid trail construction on changes in soil compaction level (Table 1). Often data are not collected over a long time period, are confounded by other site disturbances, do not directly assess compaction impacts on subsequent vegetation growth, or do not have a base-line comparison. The impetus for initiating the North American Long-Term Soil Productivity (LTSP) study was to test the linkage between soil impacts and tree growth (Powers et al. 1990; Fleming et al. 2006; Powers 2006). In this paper we evaluate (1) the effectiveness and variability of compaction treatments on the LTSP sites across a variety of soil textural classes and (2) the recovery of soil bulk density and soil strength 1 and 5 years after harvesting and site preparation.

Materials and methods

The North American LTSP study sites were established to conform to the National Study Plan described by Powers (2006). A series of plots (0.4 ha in size) with common treatment protocols were installed in major timber types and on different soil groups throughout the United States and Canada (Table 2). All data used in this paper came from 12 LTSP sites that were at least 5 years old. Additional descriptions of each installation can be found in Powers (2006) and Fleming et al. (2006). Main soil treatments (3 × 3 factorial design) were three levels of organic matter removal (bole-only removal, whole-tree removal, and whole-tree plus forest-floor removal) and three levels of compaction applied to the soil surface (none, moderate, and severe). At most study locations, main treatments were split in half to provide a weed versus no-weed (herbicide) comparison. All study sites had three replications of each treatment.

The aspen stands at the Huron-Manistee, Ottawa, and Chippewa sites were winter logged to protect suckering roots. Other sites were harvested during the summer, but all plots receiving the no-compaction treatment were not driven on during either harvesting or site preparation. The desired compaction level was achieved by driving over plots with heavy equipment (e.g., bulldozer, grappler, asphalt roller) or compressing with high ground pressure equipment. Logging debris and forest-floor material were removed before compaction so that mineral and organic components would not be mixed. At each of the 12 sites, compaction was deliberately scheduled when the soil was near field capacity to ensure maximum macropore reduction. Severe compaction was intended to approach, but not meet, growth-limiting bulk densities or soil strength for each particular soil texture (Daddow and Warrington 1983), and we attempted to reach bulk density levels within 20% of the approximate growth-limiting bulk density in the surface 0–10 cm of soil. Moderate compaction levels were designed to come close to the midpoint between no and severe compaction. After mineral soil compaction was complete, forest floor and slash were

Table 1. Average change in surface (0–15 cm) bulk density immediately after harvest operations for various small-scale skid-trail studies.

Soil texture	Bulk density Initial ($Mg{\cdot}m^{-3}$)	Final ($Mg{\cdot}m^{-3}$)	Increase (%)	Reference
Sand	1.35	1.60	16	Sands and Bowen 1978
Volcanic ash over limestone till	0.53	0.93	41	Cullen et al. 1991
Volcanic ash over quartzite till	0.76	0.92	18	Cullen et al. 1991
Tertiary volcanic ash	1.67	1.81	8	Cullen et al. 1991
Silt loam over glacial till	0.95	1.4	33	Jansson and Johansson 1998
Sandy loam	0.92	1.15	20	Allbrook 1986
Loam	0.72	0.96	25	Aust et al. 1993
Loamy volcanic ash	0.93	1.07	15	Froehlich et al. 1986
Volcanic ash	0.84	1.08	28	Froehlich et al. 1986
Silty clay	1.19	1.32	11	Corns 1988
Clay over till	1.05	1.29	20	Corns 1988
Loam (eolian)	0.67	0.70	12	Corns 1988

Table 2. Sample size, site characteristics, and soil properties for 5-year-old Long-Term Soil Productivity installations.

Textural class	Installation name*	$n^{\dagger}$	Clay content (%)	Rock-fragment content (%)	Preharvest bulk density ($Mg{\cdot}m^{-3}$)
Sand	Huron-Manistee	8	2	1	0.96
Loamy sand	Nemagos Lake	10	3	11	1.1
Sandy loam	Goldsboro	4	12	0	1.33
Sandy loam	Rogers	5	15	22	0.91
Skeletal-loam	Topley	9	15	35	1.45
Fine sandy loam	Malbis	10	12	0	1.36
Very fine sandy loam	Chippewa	8	10	1	1.02
Silt loam	Freest (1–3)	10	6	0	1.32
Silt loam (volcanic)	Council	16	17	3	0.67
Cherty silt loam	Carr Creek	4	26	44	1.48
Clay loam	Challenge	5	21	30	0.94
Clay	Ottawa	8	60	0	1.03

Note: Texture, clay content, rock-fragment content, and average preharvest bulk density are from the surface (0–10 cm). British Columbia soil depth is 0–20 cm.

*For more information on each installation, see Powers et al. (2006).

†Sample size for each plot and depth interval.

returned, as needed, to achieve each plot treatment combination. Methods of compaction, measurement of compaction, and organic matter removal varied for each LTSP installation; however, main and split-plot treatments were consistently maintained. Each plot was regenerated with tree species indicative of surrounding native forest types.

Pre-and post-harvest (at 1 and 5 years) collection of soil strength and bulk density were conducted in a manner that conformed to established published protocols (i.e., Blake and Hartge 1986; Muller and Hamilton 1992; Lichter and Costello 1994; Page-Dumroese et al. 1999), but were necessarily different at each installation because of differences in rock-fragment amounts and size, sampling equipment, or timing (Table 2). Bulk density samples were collected from the 0–10, 10–20, and 20–30 cm depths on the Malbis, Freest (all three sites from Powers (2006)), Missouri, Goldsboro, Council, Ottawa, Huron-Manistee, and Chippewa sites; at 0–10 and 10–20 cm depths on the Nemagos Lake site; and at 0–20 cm depth on the Topley site. Soil strength was measured at Council and Freest using a recording penetrometer at 1.5 cm increments adjacent to the bulk density sampling sites. Three penetrometer measurements (replicates) at each sampling point were taken to a depth of 60 cm. Soil strength measurements were taken at approximately the same time each year to minimize seasonal soil moisture differences. Rock-fragment content was measured by either field estimates or gravimetric laboratory mass. Total bulk density was corrected for rock-fragment content as necessary (Andraski 1991; Page-Dumroese et al. 1999). Soil texture was determined using established published protocols (i.e., Gee and Bauder 1986). Several study sites had clay content >20%, but none of the sites had an appreciable component of shrink–swell clays. At the Council and Challenge sites, pore volume was estimated using undisturbed cores and a pressure chamber (Lenhard and Bloomsburg 1979). At the Topley, Missouri, and Council sites, average fifth-year soil moisture and temperature were recorded on two subplots (no herbicide applied) at the 10 cm soil depth using moisture and temperature wafers (ELE International/SoilTest, Inc., Loveland, Colorado) and an analog output sensor.

On four sites (Malbis, Freest, Goldsboro, and Council), we estimated the number of samples necessary to be within

Table 3. Average bulk density and change from undisturbed values for select Long-Term Soil Productivity installations (installation name followed by textural class in parentheses).

		1 year after site treatment		5 years after site treatment	
Depth (cm)	Compaction level	Avg. bulk density ($Mg \cdot m^{-3}$)*	Change from preharvest (%)[†]	Avg. bulk density ($Mg \cdot m^{-3}$)*	Change from preharvest (%)[†]
Huron-Manistee (sand)					
0–10	None	1.12 (11)a	14	1.03 (22)a	6
	Moderate	1.28 (8)b	24	1.16 (18)b	17
	Severe	1.34 (8)c	28	1.20 (8)b	19
	p value	<0.001		<0.0001	
10–20	None	1.31 (8)a	10	1.28 (7)a	8
	Moderate	1.41 (7)b	16	1.38 (6)b	14
	Severe	1.45 (8)c	18	1.43 (7)c	17
	p value	<0.0001		<0.0001	
20–30	None	1.39 (10)a	5	1.37 (7)a	3
	Moderate	1.46 (7)b	9	1.44 (11)b	8
	Severe	1.51 (6)c	12	1.49 (9)c	11
	p value	<0.001		<0.001	
Nemagos Lake (loamy sand)					
0–10	None	1.04 (10)a	–3	1.00 (15)a	–9
	Severe	1.25 (11)b	15	1.06 (15)a	–3
	p value	<0.0001		0.7132	
10–20	None	1.14 (13)a	4	1.17 (14) a	7
	Severe	1.44 (9)b	19	1.27 (12)b	2
	p value	<0.0001		<0.0001	
Goldsboro (sandy loam)					
0–10	None	1.22 (16)a	9	1.13 (22)a	14
	Moderate	1.46 (13)b	40	1.25 (23)a	24
	Severe	1.45 (13)b	23	1.17 (20)a	13
	p value	0.0462		0.9637	
10–20	None	1.41 (12)a	23	1.44 (21)a	7
	Moderate	1.54 (10)b	28	1.48 (18)a	11
	Severe	1.52 (10)b	27	1.41 (21)a	9
	p value	0.0195		0.6799	
20–30	None	1.47 (11)a	24	1.45 (20)a	8
	Moderate	1.60 (11)b	30	1.41 (19)a	11
	Severe	1.57 (16)b	29	1.31 (25)a	–18
	p value	0.0305		0.7013	
Topley (skeletal-loam)					
0–20	None	1.56 (18)a	8	1.66 (15)a	14
	Moderate	1.74 (18)b	14	1.79 (16)b	16
	Severe	1.75 (18)b	17	1.74 (13)b	16
	p value	<0.0001		0.0016	
Malbis (fine sandy loam)					
0–10	None	1.26 (7)a	–6	1.22 (10)a	–5
	Moderate	1.33 (12)b	–1	1.27 (11)b	–4
	Severe	1.35 (15)b	–1	1.28 (9)b	–5
	p value	<0.0001		<0.0001	
10–20	None	1.43 (6)a	0	1.34 (11)a	–7
	Moderate	1.50 (9)b	6	1.45 (10)b	3
	Severe	1.51 (8)b	5	1.45 (11)b	1
	p value	<0.0001		<0.0001	
20–30	None	1.49 (6)a	3	1.39 (10)a	–3
	Moderate	1.51 (7)b	6	1.46 (8)b	–1
	Severe	1.51 (7)b	3	1.43 (10)c	–3
	p value	0.0584		<0.0001	
Chippewa (very fine sandy loam)					
0–10	None	1.17 (14)a	14	1.14 (15)a	12
	Moderate	1.34 (10)b	25	1.21 (14)b	17
	Severe	1.39 (10)c	28	1.24 (12)b	19
	p value	<0.0001		0.001	
10–20	None	1.45 (12)a	12	1.42 (10)a	10

Table 3 (*concluded*).

Depth (cm)	Compaction level	1 year after site treatment: Avg. bulk density ($Mg \cdot m^{-3}$)*	1 year after site treatment: Change from preharvest (%)†	5 years after site treatment: Avg. bulk density ($Mg \cdot m^{-3}$)*	5 years after site treatment: Change from preharvest (%)†
	Moderate	1.57 (10)b	18	1.45 (10)ab	12
	Severe	1.62 (9)b	31	1.49 (11)b	14
	p value	<0.0001		0.0141	
20–30	None	1.56 (12)a	10	1.39 (11)a	–1
	Moderate	1.63 (9)b	14	1.39 (35)a	–1
	Severe	1.65 (7)b	15	1.53 (37)a	8
	p value	0.0050		0.099	
Freest (1–3) (silt loam)					
0–10	None	1.31 (8)a	–3	1.31 (9)a	–3
	Moderate	1.41 (8)b	7	1.36 (8)b	4
	Severe	1.42 (8)b	8	1.35 (10)b	2
	p value	<0.0001		0.0050	
10–20	None	1.54 (7)a	2	1.53 (8)a	2
	Moderate	1.62 (5)b	4	1.61 (8)b	3
	Severe	1.60 (6)b	3	1.60 (7)b	2
	p value	<0.0001		<0.0001	
20–30	None	1.54 (4)a	2	1.60 (8)a	–4
	Moderate	1.59 (5)b	2	1.52 (7)b	0
	Severe	1.56 (6)a	1	1.57 (7)a	–1
	p value	0.0021		0.0434	
Council (silt loam; volcanic)					
0–10	None	0.60 (4)a	–2	0.61 (4)a	0
	Moderate	0.72 (2)b	6	0.70 (3)b	3
	Severe	0.83 (3)b	18	0.83 (3)b	18
	p value	<0.0001		<0.0001	
10–20	None	0.68 (5)a	6	0.65 (5)a	0
	Moderate	0.76 (4)ab	8	0.74 (5)a	5
	Severe	0.92 (6)b	22	0.91 (6)b	21
	p value	0.0543		<0.0001	
20–30	None	0.71 (4)a	30	0.70 (5)a	28
	Moderate	0.81 (5)b	47	0.78 (4)a	43
	Severe	0.95 (3)b	58	0.93 (5)b	50
	p value	<0.0001		<0.0001	
Ottawa (clay)					
0–10	None	1.10 (11)a	7	1.16 (13)a	12
	Moderate	1.18 (12)b	13	1.20 (10)a	15
	Severe	1.27 (10)c	19	1.29 (12)b	21
	p value	<0.0001		0.0002	
10–20	None	1.21 (8)a	1	1.29 (8)a	7
	Moderate	1.25 (8)b	4	1.34 (7)b	11
	Severe	1.29 (8)c	7	1.38 (7)b	14
	p value	<0.0001		<0.0001	
20–30	None	1.21 (7)a	–1	1.29 (18)a	5
	Moderate	1.24 (7)b	1	1.28 (10)a	4
	Severe	1.28 (7)c	4	1.37 (6)b	11
	p value	<0.0001		0.0042	

Note: Values in parentheses are the coefficients of variation (%).

*In each column, within each location and depth, values with the same letter are not significantly different.

†Negative values as percent change from preharvest indicate a lower bulk density than was originally sampled before harvest.

15% of the mean both preharvest and 5 years after harvest and site preparation activities based on preharvest sampling. We show 0.4 ha preharvest data and 0.2 ha (split-plot) postharvest samples, since half of each plot was treated with herbicide. For each plot, the number of samples necessary to estimate the mean value and degree of confidence was determined using the following equation:

$$n = \frac{t_{n-1}^2 S_y^2}{E^2}$$

where n is the number of samples necessary, t_{n-1}^2 is the value of the Student's t distribution with $n - 1$ degrees of freedom, S_y^2 is the variance of the population (assumed to be the same as the sample population), and E^2 is the allowable error (Freese 1962).

Statistical analyses

Data from each site were subjected to an analysis of variance (ANOVA). Mean separation was tested using Dunn's multiple comparison test. Significant differences are noted between each compaction level (none, moderate, and severe), soil depth (0–10, 10–20, and 20–30 cm), location (Malbis, Freest, etc.), and sample period (1 and 5 years after harvest). Because organic matter was removed prior to compaction then returned to each plot, 1- and 5-year bulk density results were unaffected by the organic matter removal treatments. Therefore, results from the different organic matter removal treatment plots were combined for each level of compaction.

Results and discussion

Compaction efficiency

Moisture content is one of the most important factors influencing the compactiblity of soils (Soane 1990); hence all study sites were compacted when soil moisture was near field capacity. For all sites after 1 year, moderate compaction in the 10–20 cm depth resulted in an increase in bulk density ranging from 4% to 28%, while severe compaction plots at this same depth resulted in an increase ~~in~~ bulk density ranging from 3% to 31% (Table 3). Our ability to compact soil deeper in the soil profile (20–30 cm) was just as variable as at the 10–20 cm depth. The relatively fine-textured volcanic ash-cap soils in Idaho were extremely susceptible to deep (20–30 cm depth) soil compaction (47% increase in bulk density in moderate compaction plots; 58% increase in bulk density in severe compaction plots), while one of the other fine-textured sites (Ottawa) did not exhibit such increases in bulk density at this depth.

In a field study of this magnitude, it is very difficult to establish and accurately measure soil bulk density values within narrowly defined treatment specifications across different soil types and using different equipment. This is clearly shown in the establishment of compacted plots at the Malbis site, where after 1 year bulk density values were less at the 0–10 cm depth after compaction than prior to treatment. Deeper in the soil profile (10–20 and 20–30 cm), bulk density increases of 3%–6% were measured 1 year after compaction. The measurement techniques used may not have been precise enough to differentiate the changes in bulk densities between years.

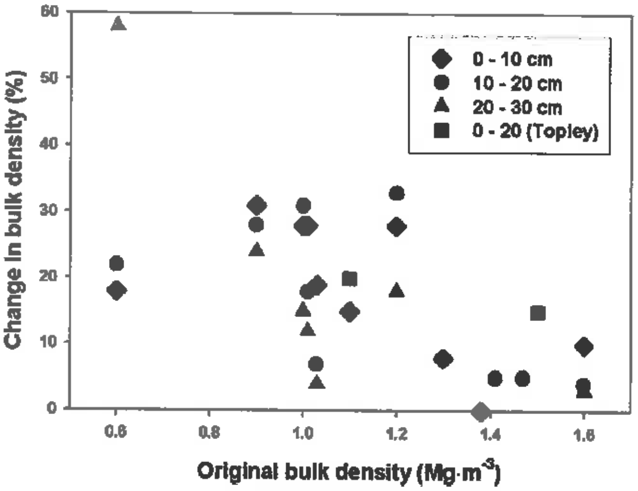
Generally, differences between the moderate and severe compaction levels were small, if detectable at all. Before harvesting, soil bulk density values in the 10–20 and 20–30 cm depths were not significantly different for most sites (Table 3). On cherty silt loam plots (Carr Creek, Missouri), surface bulk densities (0–10 cm depth) in the moderate and severe treatments were 8% (ending bulk density: 1.65 $\mathrm{Mg \cdot m^{-3}}$) and 15% (ending bulk density: 1.78 $\mathrm{Mg \cdot m^{-3}}$) higher than bulk density in the uncompacted control (F. Ponder, personal communication, 2005). On a clay-loam soil (Challenge), bulk density increased 18% in both the moderate and severe compaction treatments. However, on the sandy-loam Rogers site, moderate compaction plots increased 13% above preharvest conditions and the severe compaction plots increased 19%. Both the Challenge and Rogers sites had increased bulk density with increasing depth (R. Powers, personal communication, 2005). The Nemagos Lake site (with no mid level of compaction) also showed a significant increase in bulk density in the severe compaction treatment as compared to bulk density at preharvest levels. Soil organic matter in the mineral soil is important for reducing the impacts of machine traffic on soil bulk density changes (Soane 1990). Because organic matter on the LTSP sites was removed prior to compaction, we saw no significant impact of organic matter level on bulk density. However, as roots and organic material decay within the soil profile, the importance of organic matter for maintaining soil structure may become more evident.

Levels of soil compaction (as measured by bulk density) achieved with these large-scale field plots were often of a similar magnitude to those reported for a variety of skid-trail studies implemented with ground-based equipment (Table 1). This indicates that, in most cases, we were able to mimic small-scale changes on large-scale plots. As vegetation develops on these treatments, we will be able to determine how applicable skid-trail studies are to larger areas and whether recovery time is affected (see also Fleming et al. 2006).

Daddow and Warrington (1983) summarized numerous studies and delineated 1.75 $\mathrm{Mg \cdot m^{-3}}$ as the growth-limiting bulk density for sandy loams and loamy sands. In addition, they defined 1.4 $\mathrm{Mg \cdot m^{-3}}$ as being limiting to root growth in fine-textured soils. Lousier (1990) indicated that soil bulk densities near 1.2–1.4 $\mathrm{Mg \cdot m^{-3}}$ were sufficient to stop root growth in most forest ecosystems. Our data indicate that bulk densities higher than these root-limiting levels already existed on the Malbis, Freest, Goldsboro, Missouri, and Topley sites. Since these sites all supported highly productive stands, setting broad rules of root-limiting bulk densities may not be feasible. However, the high initial soil densities strongly influenced the degree of compaction attained and may indicate that these sites could be susceptible to productivity losses with small increases in bulk density.

Overall, initial soil bulk density determined the degree of severe compaction (Fig. 1). As initial bulk density increased, the level of change decreased. Fine-textured soils often had the lowest initial bulk density, but the largest increase after treatment, with a majority of compaction occurring after a single equipment pass. This pattern of a larger percent increase in bulk density on fine-textured soils has been measured elsewhere (Williamson and Neilsen 2000). Percent increase in bulk density has been suggested as a method for determining change in soil productivity after trafficking; however, this may limit activities on soils with low initial bulk densities. In addition, sites with a high initial bulk density may exhibit a detrimental change in macroporosity due to subsequent trafficking that may go undetected with a percent increase standard. The percent increase criteria for soil compaction of varying soil types also may not reflect changes to biological properties or plant growth response (Williamson and Nielsen 2000). Landres et al. (1999) pro-

Fig. 1. Percent change in soil bulk density after severe compaction relative to initial soil bulk density.



pose a soil evaluation system based on the natural range of variation where soil properties after harvesting are compared to preharvest conditions. In this case, preharvest data would be collected to evaluate the natural range of variation for similar landscapes and then used to determine significant changes due to management (Landsberg et al. 2003). Assessment of preharvest conditions has also been recommended for evaluation of detrimental changes in soil nutrients due to displacement and burning (Page-Dumroese et al. 2000). In the USDA Forest Service, soil quality standards and guidelines set a 15% increase in bulk density for determining a detrimental disturbance (Powers et al. 1998). This guideline requires some survey of undisturbed soil conditions for a postharvest comparison. However, the British Columbia Ministry of Forests uses a postharvest visual assessment of disturbance relative to adjacent undisturbed soil as a proxy for regulating long-term effects (British Columbia Ministry of Forests 1997; Curran 1999), and along with some preharvest work determines the appropriate silvicultural prescription and possible restoration needs.

Pore-size distribution

Compaction affects pore-size distribution and therefore available water, mainly because soil volume decreases during compression of pore space (Startsev and McNabb 2001). Changes in soil porosity were assessed at three LTSP installations (Council, Challenge, and Rogers). On the Council site after severe compaction of the silt loam volcanic ash surface soil, total porosity declined 25% in the 10–20 cm depth (data not shown). At this same depth, macropore volume declined 34%, while micropore volume remained relatively unchanged (<5%). On the Challenge and Rogers sites, severe compaction of soils with varying textures also resulted in overall decline of total porosity (9%, loam; 20%, clay; and 13%, sandy loam) throughout the soil profile (to a depth of 45 cm) (Gomez et al. 2002). On both sites, 5-year growth responses to compaction treatment were inconclusive (Gomez et al. 2002; D. Page-Dumroese, unpublished data). On the Challenge and Rogers sites, ponderosa pine (*Pinus ponderosa* Dougl. ex P. & C. Laws.) growth differences were related to soil texture, water, and air regimes, not to specific soil physical property changes (Gomez et al. 2002). Compaction-caused reductions in total porosity may result in little change in moisture retention, and therefore plant growth proceeds relatively unaffected until root growth is inhibited (Sands et al. 1979). However, soil texture is important for determining the impact of increased micropores. For example, on a soil with high clay content, 10% air-filled porosity (*v*/*v*) may be adequate for plant growth (Håkansson 1990), while on a sandy soil (with a low content of fine material) air-filled porosity may need to be near 30% for air permeability to be adequate (Håkansson and Lipiec 2000).

Soil temperature and moisture content during the growing season for three sites is shown in Figs. 2*a* and 2*b*. Although generally not statistically significant, severe compaction often resulted in a slight increase in average soil temperature at 20 cm throughout the growing season. On the Topley and Carr Creek sites, severe compaction generally resulted in increased moisture content at 20 cm regardless of organic matter treatment. However, on the Council site, severe compaction did not increase soil moisture during the growing season. Additionally, on the Council site, soil water declined slightly as more organic matter was removed from the soil surface. Compaction has been shown to have a variable effect on soil moisture content of forest soils, and a significant increase in soil bulk density may not affect soil water (Froehlich and McNabb 1984). During compaction, micropores may be unaffected and soil porosity changes could be confined to the mesopore space (Startsev and McNabb 2001), resulting in little change in soil moisture content. Changes in pore-size distribution are highly dependent on soil texture and soil water regime, and the use of soil porosity as a monitoring tool for managers will require site-specific data (Gomez et al. 2002).

Five-year recovery

After 5 years, every site except Topley exhibited some level of bulk density recovery as compared to the 1-year postharvest measurement (Table 3). In general, for both compaction treatments (moderate and severe), the Malbis site had fully recovered to predisturbance levels (compaction levels less than or not significantly greater than preharvest levels). On the Freest sites after 5 years, plots with the greatest amount of residual compaction were the surface 0–10 cm in the moderate compaction plot (within 4% of the preharvest level). The Chippewa plots (very fine sandy loam) showed full recovery in the 20–30 cm depth 5 years after treatment, while the other two depths had an average recovery of 26% (0–10 cm depth) and 35% (10–20 cm depth). On the Council plots, which showed the greatest initial change in bulk density, there was only a slight amount of recovery in the surface soil after 5 years. Surprisingly, the clay soil (Ottawa) at the 20–30 cm depth showed an increase in bulk density after 5 years in all three compaction treatments. This increase may only be a reflection of site variability, but other factors, such as organic matter loss after the canopy was removed or raindrop impact on the exposed soil, may contribute to this increase. Fine-textured soils appear to be the slowest to recover after site treatment. In fact, the clay-loam soil (Challenge) has not recovered to preharvest

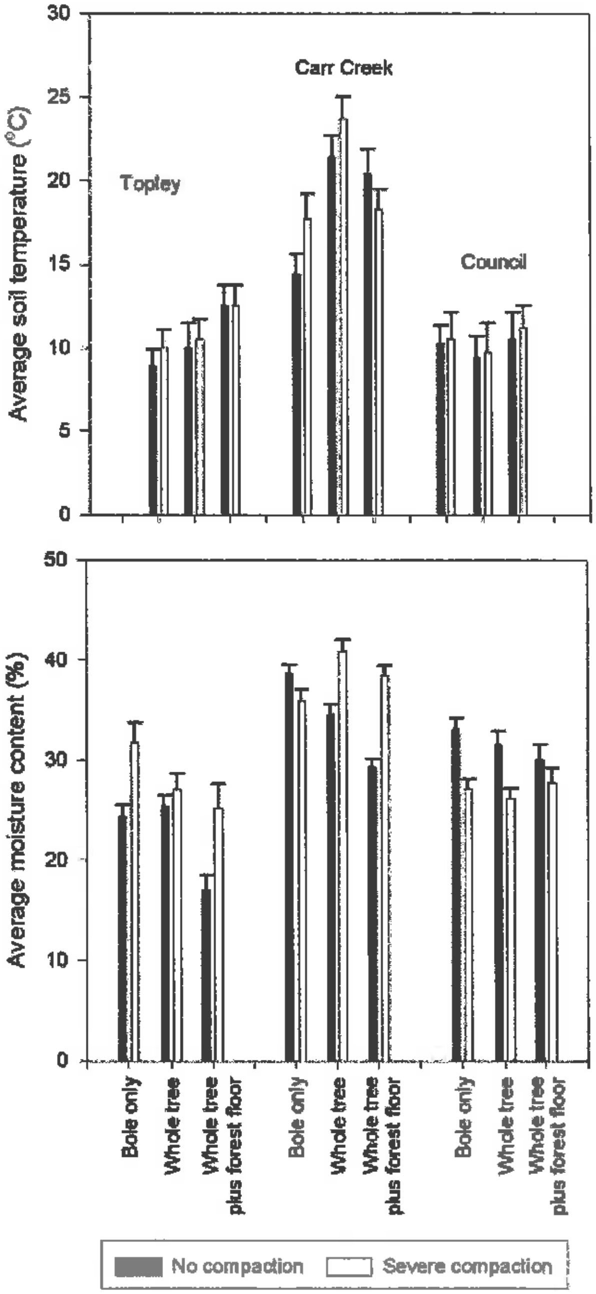


Fig. 2. Average fifth growing season (May–September) (*a*) temperature and (*b*) moisture at 20 cm soil depth on three soil textures as affected by compaction and organic matter level. Error bars represent the standard error of the mean.

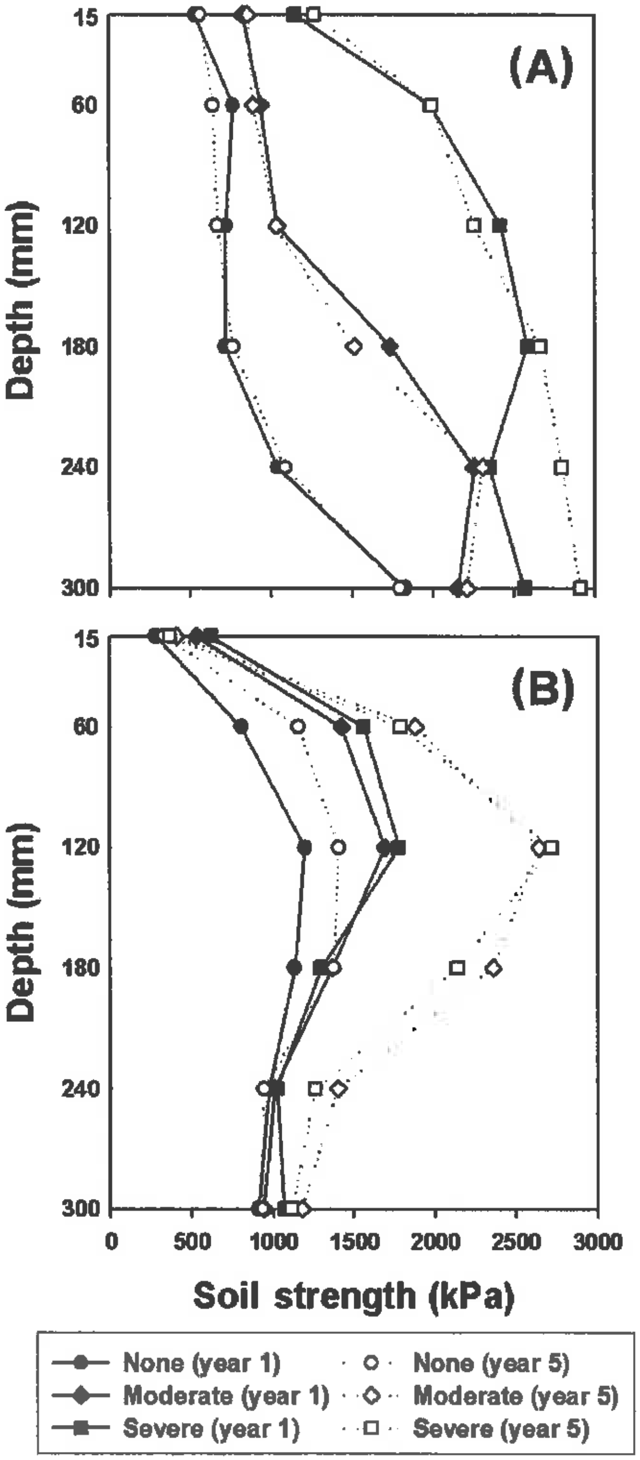


Fig. 3. Average soil strength in the soil profile as affected by treatment compaction level for (*a*) Council and (*b*) Freest 1–3.

conditions after 10 years (R. Powers, personal communication, 2005; data not shown).

Examination of herbicide impacts show that only plots on the Malbis and Freest sites had a significant reduction in surface (0–10 cm) bulk density with understory reestablishment (data not shown). Vegetation regrowth on the Malbis sites may have accelerated a bulk density decrease in the 0–10 cm depth of the moderate compaction plots. These plots had returned to the original preharvest bulk density after 5 years (p = 0.0056). On the surface (0–10 cm) of the Freest plots soil had recovered to near predisturbance levels in the herbicide-treated, severely compacted plots (p = 0.0482; 2% higher than preharvest). Although there was no significant herbicide effect at the other sites, the level of recovery at these study sites after 5 years was unexpected, since many authors report a return to the initial, uncompacted state is

Table 4. Average number of soil bulk density samples necessary in each 0.4 ha plot (preharvest) and 0.2 ha plot (postharvest) to be within 15% of the population mean ($\alpha = 0.1$) for select Long-Term Soil Productivity study sites (installation name followed by textural class in parentheses).

		Postharvest (5 years after treatment)					
		No compaction		Moderate compaction		Severe compaction	
Depth (cm)	Preharvest (undisturbed)	No herbicide	Herbicide	No herbicide	Herbicide	No herbicide	Herbicide
Goldsboro (sandy loam)							
0–10	15	19	15	8	6	10	12
10–20	9	7	6	8	10	10	9
20–30	12	8	10	5	8	8	7
Malbis (fine sandy loam)							
0–10	9	6	3	31	10	7	5
10–20	11	6	6	16	7	13	6
20–30	14	3	4	8	13	11	4
Freest (1–3) (silt loam)							
0–10	15	10	17	12	8	13	14
10–20	25	8	23	5	8	18	11
20–30	10	7	17	8	10	11	9
Council (silt loam; volcanic)							
0–10	11	11	8	15	10	14	13
10–20	10	8	5	9	9	10	10
20–30	8	9	4	8	9	9	7

Note: Sample numbers are based on sites with three replicates. Sample sizes (*n*) are shown in Table 2.

often slow or nonexistent (Hatchell et al. 1970; Froehlich and McNabb 1984; Corns and Maynard 1998; Stone and Elioff 1998). Recovery to preharvest levels on these LTSP sites can be attributed to a host of environmental factors such as high rock-fragment content, a fluctuating water table, or freeze–thaw cycles (Fleming et al. 1998; Stone and Kabzems 2002). Slower recovery on the Idaho sites may be due to compression of the glass shards of the volcanic surface soil (Shoji et al. 1993). Although soil bulk density increases are fairly easy to quantify, the direct effects on vegetation regeneration and growth are not always immediately apparent (Miller et al. 1989; Powers and Fiddler 1997; Kozlowski 1999; Gomez et al. 2002; Miller and Anderson 2002; Landsberg et al. 2003). Sites with a high initial bulk density, but with small bulk density increases after treatment, exhibit the fastest recovery, since incremental increases are small.

Soil strength measurements

Compacted soils resist penetration by plant roots because of either small or rigid pores that prevent roots from growing through the soil. Penetrometer values represent a measure of mechanical resistance of the soil to root penetration (Sands et al. 1979). An example of soil strength change after compaction treatment is shown in Figs. 3*a* and 3*b*. Both the Freest and Council sites show an increase in soil strength with depth and compaction intensity. Although bulk density measurements at the Freest sites indicate recovery after 5 years, soil strength measurements do not reflect this same recovery. Contrary to results on the Freest sites, on the Council plots both soil bulk density and soil strength measurements indicate little recovery after 5 years.

Reduced root penetration at high soil strength has been demonstrated in a variety of field studies, including loblolly pine (*Pinus taeda* L.) (Hatchell et al. 1970), Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) (Youngberg 1959), and radiata pine (*Pinus radiata* D. Don) (Sands et al. 1979) stands. When soil strength exceeded 3000 kPa in radiata pine plantations, root growth was restricted (Sands et al. 1979). However, root-limiting soil strength, on a variety of soils (loamy fine sand, fine sandy loam, very fine sandy loam, and loam) in the United States, was found to be closer to 2500 kPa (Taylor et al. 1966). Both the Freest and Council sites approached these two assessments for root-limiting soil strength values. Since soil strength values decrease as soil water content rises, root growth may be proceeding on these sites at high water contents, but may be restricted as soils dry (Gomez et al. 2002). The impacts of increasing soil strength are mixed, and data can be found supporting tree growth reductions, increases, both increases and decreases, or no effect (Sands and Bowen 1978; Greacen and Sands 1980; Miller et al. 2004). On the sandy-loam soil (Rogers), compaction increased the number of days that water was available for plant uptake from 45 days (no compaction) to 131 days (severe compaction). However, on the clay-loam soil (Challenge), days of available water decreased with increasing compaction (Gomez et al. 2002). In the southern United States, soil strength, not bulk density, was found to be the critical impedance factor controlling root penetration into the soil profile (Taylor and Burnett 1964). In addition, clay and volumetric water content have been highly correlated with resistance to root penetration (Gerard et al. 1982). Differences in total organic matter and soil surface traffic load will also affect the degree to which soil strength changes during harvest or site preparation activities (Sands et al. 1979; Williamson and Neilsen 2000; Liechty et al. 2002).

Soil strength measurements are relatively easy to collect on sites once initial instrument setup is complete (Atwell 1993) and could be a method for evaluating a site before actual soil quality monitoring is conducted. Areas can be initially defined as similar or different and a sampling system devised for a given site. This may be more time consuming, but may offer a more productive way to gather soil strength data for interpretation of short- and long-term harvest and site preparation effects. In addition, collecting a gravimetric soil moisture sample concurrently with soil strength is necessary to adjust for the possible influence of soil moisture between sample dates (Landsberg et al. 2003).

Variability within plots

For several LTSP sites we estimated the sample size that was necessary to be within 15% of the mean 0.4 ha plot bulk density value. Estimates for 0.4 ha ranged from 8 to 25 samples before harvesting and from 3 to 23 samples postharvest (Table 4). The smaller range in sample size is similar to the samples sizes we selected (Table 2). After 5 years, the herbicide-treated plots generally required a smaller sample size than the untreated plots. This is likely due to fewer roots in the surface soil horizons, which can contribute to higher variability. In addition, we calculated the number of sample points necessary to be within 15% of the mean for soil strength on the Freest and Council 0.4 ha plots. Optimum sample size on the Freest plots was calculated to be approximately 38 sample points and for Council it was 20 sample points in each 0.2 ha plot (data not shown). Usually, selection of sample size for each site was dictated by field crew availability, time constraints, and budgets.

Lateral variability is often a problem in forest ecosystems, even within small areas (Courtin et al. 1983), and most forest studies are limited to the forest floor and surface mineral soil and do not include the deeper mineral soil physical properties (e.g., Grier and McColl 1971). Beckett and Webster (1971) have reported that up to half the plot variability can be present within 1 m^2 and within-plot variability changes little with the size of the plot. In British Columbia, on both high- and low-productivity ecosystems, bulk density was one of the least variable measurements, and the estimate of the required sample size (±20% with 90% confidence) was 4–6 samples on 0.8–3.3 ha plots. Sample sizes of 14–28 were needed if a 95% confidence and ±10% error were used (Courtin et al. 1983). These values are similar to our calculations of the number of bulk density samples needed to accurately approximate mean conditions on each 0.4 ha plot. Because of problems associated with large numbers of samples (cost, analysis, transport, etc.), several authors have noted that when studying other highly variable soil properties (i.e., some forest elemental concentrations, rock-fragment content), compositing is often necessary to reduce within-plot variability (Mader 1963; Courtin et al. 1983). Assessment of the importance of soil physical changes in the broader context of the range of natural variability is important when considering the impacts of timber harvesting or mechanical site preparation and their subsequent consequences on vegetation response (Block et al. 2002; Bock and VanRees 2002).

Fig. 4. Soil strength contour lines of a severely compacted plot on the Council plots for (*a*) the entire plot (65 m on a side) and for (*b*) a 5 m^2 subplot within the larger treated area.

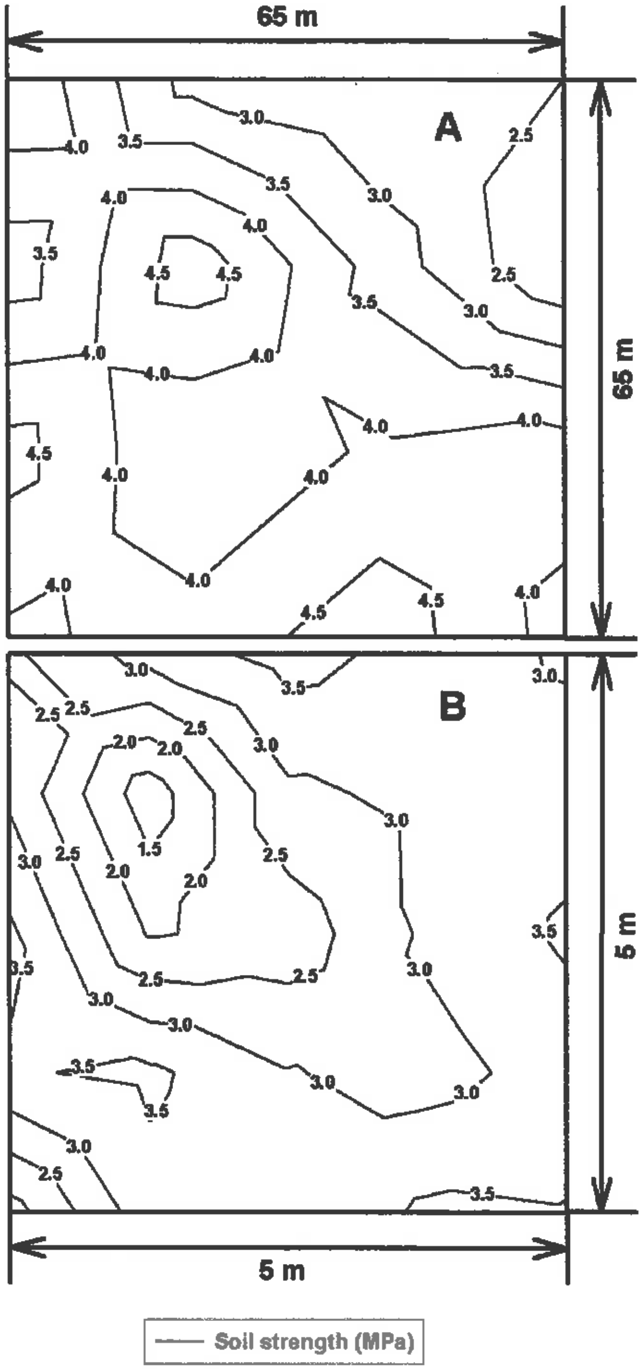
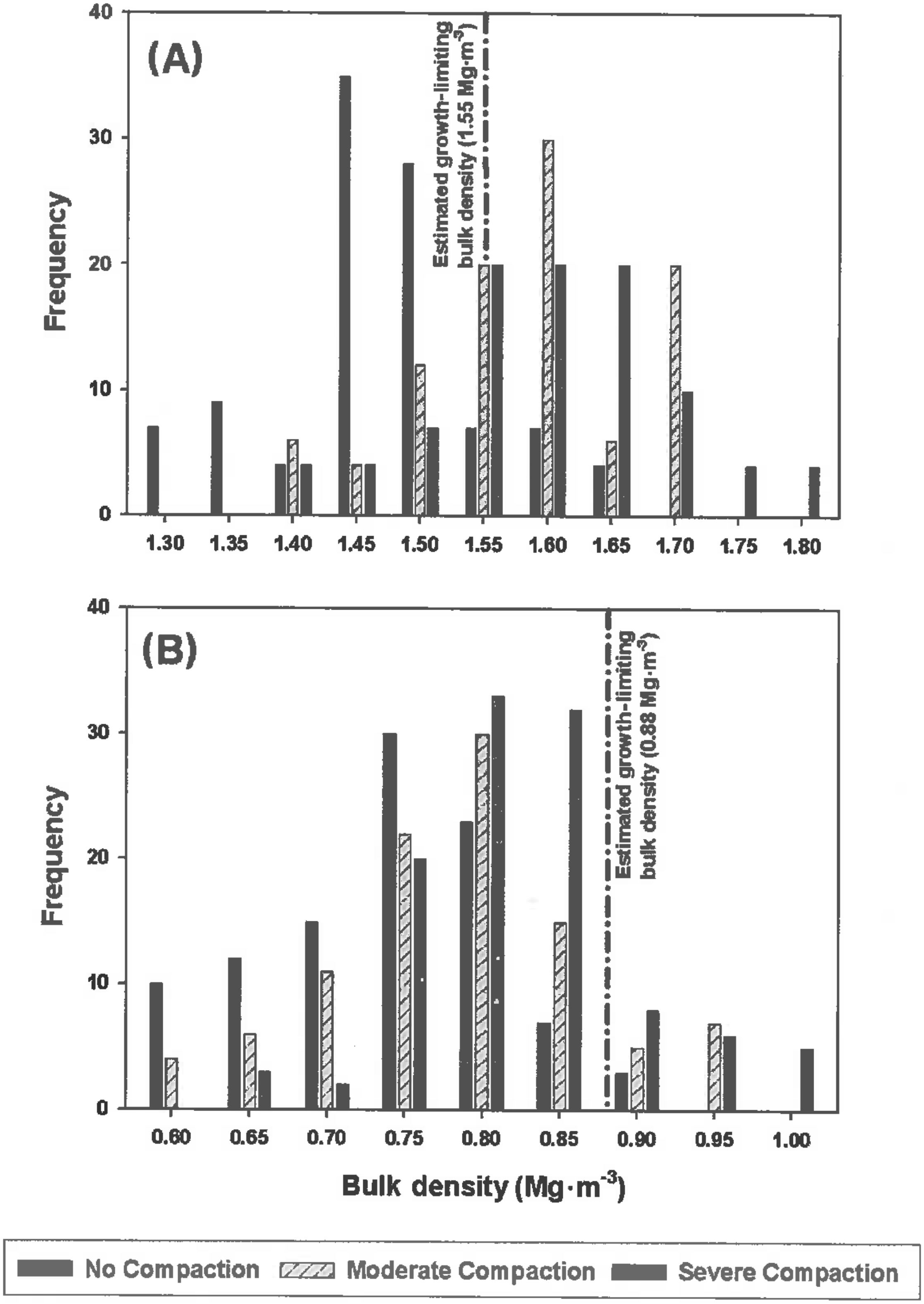


Fig. 5. Frequency distribution of soil bulk density values 1 year after compaction from (*a*) Malbis (fine sandy loam) and (*b*) Council (silt loam, volcanic). Note initial differences in soil bulk density range.



Although one aim of this study was to achieve uniform compaction, we were not always successful. On a fairly large scale (0.4 ha) we evaluated a severe compaction plot (0–15 cm depth) in Idaho for soil strength at 48 sample points (16 points with 3 replicate samples at each point) using the standard sampling protocol (Fig. 4*a*). Soil strength values range from 2500 to 4800 kPa, and although some differences occur, soil strength changes appear fairly uniform, except near large tree stumps. At a smaller scale (5 m^2), the same plot was intensively sampled (225 sample points: 75 points with 3 replicate samples at each point) (Fig. 4*b*). This smaller scale and intensive sampling scheme shows some of the soil strength variation that is possible across any given plot. At this scale, soil strength measurements range from 1200 to 4800 kPa. The small-scale plot did not have any tree stumps, but several were present near the edges. Spatial variability associated with these soil strength measurements is not uncommon. Small-scale plot data are highly influenced by traffic variability, while large-scale plot variability can be attributed to larger landscape features such as stumps or microtopography (Carter et al. 2000). Spatial dependence of soil strength measurements often occurs at more than one scale (O'Sullivan et al. 1987; Carter et al. 2000); however, spatial variability of the subsurface horizons is likely to be less pronounced than surface variability (Carter et al. 1999). These variability results are not surprising considering the factors that affect compaction of uneven ground surfaces. Since we had removed logging debris and applied a relatively uniform traffic pattern, the variability of compaction during operational logging is likely much higher.

We also evaluated the uniformity of bulk density values from two locations. The frequency distributions of bulk density values 1 year after harvesting for each compaction treatment on the Malbis (Fig. 5*a*) and Council (Fig. 5*b*) sites show that compaction does not occur uniformly (all one value). Rather, bulk density values shifted from low to high bulk density as traffic intensity increased. For both sites, plots with no postharvest compaction showed a range from many very low bulk densities to a few high values. After compaction, there were few low bulk density values and many higher values. For example, on the Malbis no-compaction treatments, a majority (~75%) of the bulk density values were less than the growth-limiting bulk density (Daddow and Warrington 1983). After treatment (severe compaction), nearly 60% of the samples had values greater than reported growth-limiting bulk densities. On the Council site no-compaction plots, approximately 90% of each 0.4 ha plot was below the estimated growth-limiting bulk density. Severe compaction did not substantially increase the amount of each plot below the estimated root-limiting bulk density, but more values were closer to this value than before compaction. This same bulk density shift was noted on the Challenge and Rogers plots and also occurred on the other LTSP study sites as well (i.e., bulk density was not uniform across the entire plot).

Knowing the spatial distribution of soil compaction in relation to root distribution is critical to understanding the effects of compaction on forest productivity. Roots use soil resources far from the main stem and are able to adjust their distribution to maximize available resources (Sands and Bowen 1978). In the example shown in Fig. 3*b*, after canopy closure, root distribution of each tree will cover a larger area than the spatial pattern of soil strength. Using the growth of young trees to measure impacts of soil compaction must be done with caution under these circumstances. Tree growth in areas of high soil strength may improve as roots expand into areas of lower strength. Compaction effects on productivity will not be clear until all trees in a given stand have had the opportunity to spread out into the entire available soil volume.

Spatial variability also affects the way in which soil quality standards are used to establish limits of allowable impact. For example, the Malbis compaction treatments increased bulk density by only 3%–6%. However, this small change in absolute values increased the frequency of densities above the critical level from 25% to 60% (Fig. 5*a*). Absolute densities at Council increased by 6%–58% relative to the original densities, while the frequency of densities greater than the growth-limiting value changed very little. Soil characteristics, including differences between the initial density and the growth-limiting density, as well as the spatial variability, need to be considered during establishment of standards.

Conclusion

In all instances we were able to significantly increase soil bulk density above the undisturbed level. Attaining a severe compaction level was difficult and not always a significant increase over the moderate compaction level. One major component in determining our compaction "success" was plot variability. Plot layout, rocks, soil texture, stumps, and initial bulk density all influenced our ability to achieve two levels of compaction on all sites. Plot variability was not always captured with our relatively small sample size, but predicted plot sample numbers were fairly uniform for most study sites (from 9 to 14). Within-plot variability on these fairly large plots may necessitate altering sampling protocols in the future. Soil penetrometers may be practical tools for detecting within-plot differences and could be used as a method for prescreening sites before intensive sampling begins on any management area.

Soil bulk density and soil strength showed a range of recovery; from none to full recovery after 5 years. However, the change in pore-size distribution on two of the study sites may indicate that while bulk density has decreased over time, macroporosity may not have recovered. In addition, on the Freest sites, although bulk density showed recovery, soil strength readings did not. Similar responses on other sites (i.e., bulk density decrease and no change in soil strength) may also be occurring.

The LTSP installations offer an opportunity to assess soil changes over a long time period than is normally evaluated for smaller scale field studies. By the time trees on these sites reach crown closure, compaction effects on growth should be evident. Maintaining soil productivity is critical following any harvesting or site preparation activities. However, the decisions about how many passes logging equipment can take, where logging equipment should be allowed to travel, and how much soil impact is acceptable are all dependent on the soil texture, forest type, available slash to buffer equipment, initial condition of the soil, and preharvest conditions. Loss of macropores on fine-textured soil may prove to be more deleterious to plant growth than a percent change in bulk density. Site-specific sampling schemes will be needed to predict the longevity and extent of compaction, especially on areas where compaction is not uniform. The protocols for soil compaction sampling may require premanagement assessments to establish base-line levels as a comparison. This base-line information will likely provide more meaningful information about the impacts of harvesting or site preparation on long-term productivity and site sustainability. Any consequences of soil compaction must also be measured against tree performance over a long period of time.

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Universal Soil Loss Equation (USLE)-Predicted Soil Loss for Harvesting Regimes in Appalachian Hardwoods

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ABSTRACT: *Soil erosion from forest harvesting is a major environmental concern. While there has been research comparing soil erosion on clearcut regeneration harvests with that on uncut forests, there has been little focus on the differences among common silvicultural harvests. Forest certification standards that are currently being evaluated for adoption across the country often encourage uneven-aged systems over even-aged or two-aged systems. We estimated soil loss using the Universal Soil Loss Equation (USLE) for forest land on five harvested treatments in the southern Appalachians. Treatments included a clearcut, leave-tree harvest, shelterwood, group selection, and uncut control. Results predicted that the group selection would have approximately 10 tons/ac more soil loss over a 100 yr rotation than the other harvested treatments. The higher rate was primarily from skid trails when the treatment was reentered for harvesting. These results should be considered when weighing the benefits of uneven-aged silviculture over even-aged or two-aged silviculture. North. J. Appl. For. 19(2):53–58.*

Key Words: Universal Soil Loss Equation (USLE), soil erosion, timber harvests, silvicultural system.

Forest harvesting is often blamed for causing excessive amounts of nonpoint source pollution (McNulty 1995, Binkley and Brown 1993, Sopper 1975). Concern for maintaining water quality and site productivity led to the passage of water quality legislation and the subsequent development of Best Management Practices (BMPs) to minimize the effects of harvesting (Park et al. 1994). However, harvesting on steep slopes in the Appalachians is still viewed as a potential cause of erosion and decreases in water quality (Kochenderfer et al. 1997).

Forest management uses several harvesting methods to achieve a variety of goals. Methods are designed primarily to enhance the value of an existing stand or to manipulate the regeneration for the next stand, but they can also be used for aesthetics, water yield, or wildlife management (Smith 1997). Foresters are now evaluating the effectiveness of two-aged (leave-tree) and uneven-aged (group selection, single-tree selection) silvicultural regeneration methods as an alternative to even-aged methods (clearcut, shelterwood, seed tree). The Forestry Stewardship Council (FSC) forest certification standards call for minimizing clearcuts (FSC 2001), and the proposed guidelines for the southeastern United States state that "uneven-aged management systems should be used when feasible" (FSC 1998). The Monongahela National Forest in West Virginia decreased the amount of even-aged silviculture during the past 5 yr in response to criticisms of clearcutting (Myers 1999). Each silvicultural regeneration method serves a purpose, but the assumption that uneven-aged or two-aged silvicultural methods are patently less damaging to the environment than even-aged systems oversimplifies the issue and may lead to future stand composition and structure that fail to meet the stated management objectives.

There is little research comparing the effects of commonly used silvicultural treatments in the Appalachian hardwoods on soil erosion (Bormann and Likens 1979, Patric 1980). Most studies focus on nutrient loadings from harvesting (Bolstad and Swank 1997, Hornbeck et al. 1993) but do not actually compare silvicultural treatments. Several studies have compared the erosion associated with different land uses and harvesting techniques. Using the USLE, Gianessi et al. (1986) found that forestry affects soil erosion less than any other nonurban land use (Table 1). Similar results were found by measuring sediment (Yoho 1980, Grayson et al. 1993). Yoho (1980) also reviewed differences in forest harvesting quality and the resulting erosion rates. Using BMPs while harvesting reduced sediment concentrations by 20% in Virginia (Park et al. 1994). Natural disturbances common to the Appalachians such as

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Table 1. Average annual sheet and rill erosion by land use. Adapted from Gianessi et al. (1986).

State	Cropland	Pasture	Forest	Other*
	(tons/ac/yr)			
Virginia	6.18	3.51	0.82	5.32
West Virginia	2.57	4.16	2.07	48.58
U.S. average	4.37	1.29	0.91	8.46

* Includes farmsteads, mines, quarries, pits, and other rural lands.

fire, ice, or windthrow can also cause high rates of erosion, depending on the severity of the disturbance (Scott and Van Wyk 1990, Shahlaee et al. 1991).

This study compares soil loss estimates from four silvicultural harvest treatments and nonharvested controls in the Southern Appalachians. These treatments are then evaluated for the long-term implications of different forest regeneration techniques. The silvicultural treatments included even-aged, two-aged, and uneven-aged methods. Because erosion is a natural process that occurs on undisturbed forestland, the erosion rates from harvested areas must first be put into context with rates from presently undisturbed forests. By comparing our harvested treatments with a non-harvested control, we can set baselines and tolerable limits of erosion that are comparable to those of undisturbed forests (Fowler and Heady 1981).

The universal soil loss equation (USLE), as modified for forestland by Dissmeyer and Foster (1984), is the most widely used method of predicting soil loss in forestry (Lane et al. 1992). This updated version of the original USLE was designed for forestland, whereas the original equation was specific to agriculture (Wischmeier and Smith 1978). The equation was developed using 35 watersheds in the South that were also sampled for sediment to compare accuracy. While there have been some criticisms of the USLE, and there exist more updated computer models such as the Revised Universal Soil Loss Equation (RUSLE) and the Water Erosion Prediction Project (WEPP) (Larson et al. 1997, Montgomery et al. 1997, Lane et al. 1992), the USLE remains the best known because of its sound scientific basis, ease of use, low cost, and direct application in forest systems. The updated USLE for forestland allows foresters to quickly and easily compare potential soil loss following different harvesting methods and evaluate which method is likely to have the least influence on soil erosion.

It is important to recognize that soil loss predicted from the USLE is site soil movement, which is not synonymous with delivery of sediment to stream channels. The USLE predicts sheet and rill erosion. The Dissmeyer and Foster guide (1984) defines erosion as "the amount of soil delivered to the toe of the slope where either deposition begins or where runoff becomes concentrated." The equation does not estimate gully, landslide, soil creep, stream channel erosion, or erosion from a single storm. The soil that is deposited downslope does not necessarily have an impact on water quality. It is only when the eroded soil enters a stream that it actually affects water quality (Yoho 1980). Gianessi et al. (1986) estimated that the ratio of gross erosion to sediment for forestland is 0.50 and 0.52 for Virginia and West Virginia, respectively.

Methods

Site Selection and Description

The USLE data collection and analysis is part of a larger, long-term research project designed to examine the effects of different levels of silvicultural disturbance on the tree, shrub, and herbaceous strata in Southern Appalachian forests (Wender et al. 1999, Wender 2000). Four sites were selected to represent common upland hardwood forest types in the southern Appalachians; specifically, sites that covered at least 35 ac with minimal silvicultural disturbance in the last 15 yr and were relatively uniform in stand composition, age, structure, and geophysical characteristics. Specific criteria included were: mid-elevation (2000–4000 ft) stands dominated by red and white oaks (*Quercus* spp.), hickories (*Carya* spp.), and maples (*Acer* spp.); a maturing overstory between 50–150 yr; moderate slopes (10–4%); average site index between 60–70 ft (base age 50 for upland oaks); and predominantly south-facing aspects.

Two of the sites are located in southwestern Virginia in the Clinch Ranger District of the Jefferson National Forest. The sites are located in Wise and Scott Counties in the Cumberland Plateau physiographic region. The remaining two sites are located in the Allegheny Plateau physiographic region in Randolph County, West Virginia. The sites are hilly, with an average slope of 2%.

The Ridge and Valley and Allegheny Plateau physiographic provinces are characterized by a moderately moist, temperate, mesothermal climate. Precipitation is distributed throughout the year, without a distinct dry season, although spring is consistently the wettest season. Temperature and precipitation for both regions can exhibit considerable local variation because of differences in relief, aspect, and vegetation patterns.

The study sites are contained within Braun's (1950) Oak-Chestnut Forest region of the eastern deciduous forest. Hammond (1998) characterized the pretreatment vascular plant community of the sites. *Quercus* species were the dominant overstory component of all sites. *Q. rubra, Q. prinus, Q. alba,* and *Acer rubrum* were the major components of the Virginia sites and one West Virginia site. The Virginia sites had an average preharvest basal area of 127 ft^2/ac, while the West Virginia site had 154 ft^2/ac of basal area. The second West Virginia site was dominated by *A. rubrum, Magnolia fraseri, Liriodendron tulipifera, A. saccharum,* and *Q. rubra,* and had a preharvest basal area of 142 ft^2/ac.

The soils of all sites are derived from sandstone and shale residuum and colluvium. As is typical of most Appalachian forests, the soils are rocky, well drained, and acidic, and are without exceptional moisture-holding capacity. All study sites fall within the mesic soil temperature class, meaning the winter-to-summer range of soil temperature at 50 cm is 8–15°C (Daniels et al. 1973). Soils were identified using soil surveys of Scott and Wise Counties, VA, and Randolph County, WV (Jurney 1951, Perry et al. 1954, Pyle et al. 1982). The Virginia site soils are predominantly classified as Muskingum series, a fine-loamy, mixed, mesic Typic Dystrochrept. The West Virginia sites are predominantly from the Gilpin and Dekalb series. The Gilpin series is classified as a fine-loamy, mixed, mesic Typic Hapludult, and the Dekalb series is classified as a loamy-skeletal, mixed, mesic, Typic Dystrochrept. These soils are often found together and form the Gilpin-Dekalb-Buchanan association. While other soil types may be found on the treatments, the series described are the dominant types across the sites.

Treatments were implemented in 1999 and included (1) clearcut (even-aged), (2) shelterwood (even-aged), (3) leave-tree harvest (two-aged), (4) group selection (uneven-aged), and (5) uncut control. Each treatment plot was 5 ac, with no buffers between treatments. Because there were no buffers between treatments, data were not collected within 1.5 chains of each treatment boundary. An on-site project forester designed skid trail placement in accordance with applicable BMPs, and all skid trails were located along the slope contour. Skid trails were designed to be temporary and were closed to vehicle access after harvesting was completed. There are no permanent roads in the treatments. Conventional harvesting methods using chainsaws and cable skidders were employed for all treatments. In the clearcut treatment, all stems greater than 2 in. dbh were felled. Nonmerchantable trees were felled and left on the site. Mast, snag, or cull trees could be left for wildlife purposes, but could not exceed 4 stems/ac. The shelterwood treatment was designed to leave 50-60 ft^2/ac evenly distributed over the treatment area, with removal of residual trees in 5–10 yr once adequate advanced regeneration is present. Residual trees were dominant or codominant stems. In the two-aged leave-tree harvest, trees in the dominant or codominant crown classes were retained such that the residual stand consisted of no more than 20 trees/ac (20 ft^2/ac). The group selection treatment typically had three small openings (0.25–0.5 ac in size) with improvement cutting between the group cuts. This silvicultural treatment will be repeated every 20 yr in the group selection, with 100% of the treatment area cut after 100 yr. No silvicultural activity occurred in the control treatment.

Data Collection and Analysis

To examine the effects of harvesting on soil erosion, we collected USLE data at each site during the growing season 1 and 2 yr following harvest. Collecting data 1 yr after harvest and a second time the following year allowed us to compare erosion rates between years to estimate the recovery time to predisturbance erosion levels. In each treatment, we collected USLE data for each factor in the equation as suggested by Dissmeyer and Foster (1984) on a grid with 6 plots spaced 2 × 4 chains apart with a 1.5 chain offset into the 5 ac treatment plot. Plots were located by pacing from the northwest corner of each treatment. If a plot fell on a skid trail, it was moved an additional 33 ft so that it was located away from any skid trails. In addition, we collected USLE data for each skid trail in the treatment, including length and width to determine area. In this manner, it is possible to separate treatment and skid trail effects. The temporary nature of the plots caused variation in plot locations between years.

USLE Factors

Data collection followed the standards set in "A Guide for Predicting Sheet and Rill Erosion on Forest Land" (Dissmeyer and Foster 1984). The USLE uses a number of factors and subfactors to estimate soil loss [Equation (1)].

$$A = RKLSCP \tag{1}$$

where

A = soil loss (tons/acre/year)

R = rainfall and runoff factor (EI)

K = soil erodibility factor

LS = slope length and the slope steepness factor

CP = cover management practice factor

The rainfall and runoff index (R) is the effect of raindrop impact on runoff (Risse et al. 1993). Areas of high annual precipitation and intense thunderstorms generally have higher R values. We determined R values for each site using Wischmeier and Smith's map of isoerodent lines of erosion index units (EI) for the United States (Dissmeyer and Foster 1984). The R factor for all sites is 150 EI units.

The soil erodibility factor (K) reflects each soil type's inherent susceptibility to soil erosion. The Natural Resources Conservation Service developed K factors for most soil types. The K factor for the Muskingum and Gilpin soil series is 0.28, while that for the Dekalb is 0.24. Because soil surveys were used to identify the soils, and it is not known on a plot basis what the exact series is, we used a K factor of 0.28 for all plots.

The LS factor represents the slope length (L) and the slope steepness (S). These two measurements are combined using the formula $LS = (\lambda / 72.6)^m\,(65.41 \sin^2\theta + 4.65 \sin\theta + 0.065)$, where λ = angle of slope in degrees; and $m = 0.2$ for gradients less than 1%, 0.3 for 1 to 3% slopes, 0.4 for 3.5 to 4.5% slopes, and 0.5 for slopes of 5% and greater. The slope length was either measured by pacing or noted as 467 ft if the entire slope was uniform. A clinometer was used to determine slope gradient.

The CP factor is the product of several subfactors and represents the management practice factor for untilled and tilled forestland. We treated all treatment plots as untilled and all skid trail plots as tilled. The major subfactors for the treatment plots are (1) bare soil, (2) fine roots, (3) canopy height, (4) steps, (5) onsite storage, and (6) organic matter. For the skid trail plots the subfactors are (1) bare soil, (2) canopy height, (3) steps, (4) onsite storage, and (5) invading vegetation. All subfactors were estimated visually for the area surrounding the plot center. The plots were not a fixed size.

We calculated soil erosion estimates (A) for each measurement plot 1 and 2 yr following harvest. Differences between treatment erosion estimates were tested using PROC GLM in SAS V8 using a random complete block design ($n = 4$) with subsampling ($n = 6$) (SAS Institute 1999). To test for differences between blocks, we first calculated weighted averages based on the area in skid trails and treatments. These averages were then analyzed in SAS using a random complete block design. Percent area in skid trails was also compared by this method. All differences among means were tested using Duncan's multiple range test at $\alpha = 0.05$. Statistical analysis was performed for years 1 and 2 postharvest.

After obtaining erosion estimates for years 1 and 2, we extrapolated the data to a 100 yr rotation. We made several assumptions when calculating these estimates. Recovery rates after year 2 were based on the ratio of recovery between years 1 and 2. Once a treatment's erosion rate dropped to the control rate at year 2, we used the control rate for the rest of the rotation for that treatment. We cut one-fifth of the group selection every 20 yr and removed the shelterwood overstory at year 7. For these treatments, we assumed that erosion rates would again increase to the year 1 and 2 rates after the additional entries with the same recovery times.

Results

Estimated erosion in the clearcut and leave-tree treatments, excluding skid trails, was significantly higher than in the control 1 yr after harvesting (Table 2). The group selection and shelterwood treatments were not significantly different from the control, clearcut, or shelterwood. By year 2 there were no significant differences among the treatments. On a rotation-length basis, the group selection harvest had the highest projected rates of erosion, 24.4 tons over the 100 yr rotation, due to the multiple entries into the block. This was over 5 tons higher than the second highest treatment, the clearcut.

When the treatment and skid trail plots were combined, the results followed the same trends in year 1 as the treatment plot-only analysis (Table 3). Only the predicted soil loss from the leave-tree treatment was significantly different from the control at age 2, though it was not statistically different from that of the other harvested treatments. The group selection treatment again had the highest projected erosion rates for the rotation, 29.4 tons. The clearcut, leave-tree, and shelterwood treatments had very similar soil loss rates of approximately 19.5 tons. The control had slightly more erosion in the first year after harvest than in the second year. We feel this is likely due to using different sample points between years and a possible edge effect from adjacent harvested treatments.

Area in skid trails was highest in the group selection treatment (Table 3). All harvested treatments had a significantly larger percent area in skid trails than the control. There were no statistical differences among harvested treatments.

Table 2. Estimated erosion (tons/ac/yr) in treatments excluding skid trails during a 100 yr rotation with harvest at year 1 in Appalachian hardwood stands in Scott and Wise Counties, VA, and Randolph County, WV. Means with the same letter within years are not significantly different at $\alpha = 0.05$ using Duncan's Multiple Range Test.

	Years since harvest		Average erosion for 100 yr rotation*	Sum erosion for 100 yr rotation*
Treatment	1	2		
			·········(tons/ac/yr)··········	
Clearcut	4.41 a	0.74 a	0.19	19.0
Leave tree	4.01 a	0.90 a	0.19	18.8
Group selection†	2.17 ab	0.16 a	0.24	24.4
Shelterwood†	1.75 ab	0.32 a	0.18	17.7
Control	0.21 b	0.14 a	0.14	14.2

[2] Average and sum estimates were calculated based on the ratio of recovery between years 1 and 2; e.g., the recovery ratio for the clearcut between years 1 and 2 was 0.74/4.41 = 0.17. Year 3 estimated soil loss equaled 0.13 tons/ac/yr (0.74*0.17 = 0.13). However, the value of 0.14 tons/ac/yr was used because soil loss was not allowed to go below the year 2 control amounts.

† One-fifth of the group selection will be harvested every 20 yr, and the shelterwood

Discussion

These data clearly show that silvicultural regeneration methods requiring multiple entries into a stand over a rotation period (group selections and shelterwoods) have the potential to cause as much or more erosion than one-entry harvests (clearcuts and leave-tree harvests). While the shelterwood and group selection treatments initially had lower soil movement rates than the clearcut and leave-tree harvests, after factoring in additional harvests, the group selection had 15.3 more tons of erosion than the control for the 100 yr rotation. The shelterwood had approximately the same soil movement as the clearcut and leave-tree, with about 5.5 more tons than the control. This is a critical point when deciding what type of harvest to implement. Multiple entries redisturb recovering skid trails, and it had been documented that up to 90% of the sediment from logging is due to permanent and temporary roads (Yoho 1980). While the skid trails in this study are temporary (there are no permanent roads in the treatments), repeated entries are expected to create the same disturbance intensities as in the initial harvests.

The soil movement rate of 0.14 tons/ac/yr found in the control was similar to the erosion rate of 0.1 tons/ac/yr suggested for responsibly managed forestland in the Appalachian region by Patric (1980). This rate is based on erosion

Table 3. Estimated erosion in treatments and skid trails (tons/ac/yr) during a 100 yr rotation with harvest at year 1 in Appalachian hardwood stands in Scott and Wise Counties, Virginia, and Randolph County, West Virginia. Treatment means with the same letter within years are not significantly different at $\alpha = 0.05$ using Duncan's Multiple Range Test.

	Years since harvest		Average erosion for 100 yr rotation*	Sum erosion for 100 yr rotation*	Percent of 5 ac treatment plot in skid trails
Treatment	1	2			
			···················(tons/ac/yr)·················		
Clearcut	4.76 a	0.84 ab	0.19	19.5	6.1 a
Leave tree	5.29 a	1.12 a	0.20	19.7	5.4 a
Group selection†	2.47 ab	0.47 ab	0.29	29.4	9.4 a
Shelterwood†	2.38 ab	0.47 ab	0.17	19.7	7.1 a
Control	0.21 b	0.14 b	0.14	14.1	0 b

* Average and sum estimates were calculated based on the ratio of recovery between years 1 and 2. The recovery ratio for the clearcut between years 1 and 2 was 0.84/4.76 = 0.18. Year 3 estimated soil loss equaled 0.15 tons/ac/year (0.84*0.18 = 0.15). Year 4 estimated soil loss equaled 0.03 (0.15*0.18 = 0.03). However, the value of 0.14 tons/ac/yr was used because soil loss was not allowed to go below the year 2 control amounts.

† One-fifth of the group selection is harvested every 20 yr, and the shelterwood overstory removed at year 7, with years 1–5 erosion estimates assumed after each harvest period.

caused by normal geologic processes, regardless of anthropogenic effects. The slightly higher erosion rates in the control than the rate cited by Patric (1980) could be due to using the soil survey to identify the dominant soil series instead of identifying each soil type for every plot. There may be some soil types in the treatments that have lower *K* factors, such as the Dekalb series, that would lower the estimated erosion rates.

The rate of soil erosion peaked 1 yr after harvesting and then greatly decreased by year 2. Kochenderfer et al. (1997) reported that sediment export returned to preharvest levels by the third year after selectively harvesting a gauged West Virginia watershed. The results from our study also indicate that erosion will return to preharvest levels by year 3 or soon thereafter. Estimated soil movement decreased by approximately 80% between years 1 and 2 postharvest in all harvested treatments. According to Borman and Likens (1979), sediment yields from a careless clearcut in West Virginia were 1.35 tons/ac. This mass is roughly one-fifth of the USLE erosion estimates for clearcutting in this study. Patric (1976) and Smith and Stanley (1965) estimate sediment losses for carefully clearcut forests at 0.06 to 0.17 tons/yr.

Because the data were extrapolated over a 100 yr rotation based on the calculations made 1 and 2 yr after harvest, the rotation-length estimates will not be as accurate as actual measurements. The average and total rotation erosion estimates presented here point out the potential long-term management implications of different harvesting methods. We feel that the rotation estimates are reasonable and even if the actual numbers do change, the relative proportions among the treatments probably will not.

Management Implications

Emphasis should be placed on the value of careful design and placement of skid trails by professional foresters. Depending on how the road and skid trail system is implemented, sediment yields can vary by 25 times (Yoho 1980). Swift (1986) reported downslope sediment movement ranging from 2 to 314 ft, depending on site and road conditions. The skid trails in the harvested treatments in this study covered less than 10% of the total area. Careless logging without BMPs can disturb up to 40% of the area (Yoho 1980). The skid trails in this study were also laid out on the contour. Hornbeck and Reinhart (1964) reported that skid trails placed perpendicular to the contour on severe slopes in the Appalachians resulted in 40 tons/ac of sediment from the skid trails in the first year after harvesting (cited from Yoho 1980). Compare this with the highest rate of erosion from the skid trails in this study of 5.3 tons/ac in the leave-tree treatment, and the importance of properly planned skid trails is made clear.

The rate of soil erosion after harvesting in upland hardwoods rapidly approaches undisturbed forest erosion levels. The skid trails account for the greatest proportion of soil erosion. Therefore, a harvest treatment requiring multiple entries into a stand may cause more erosion on a longer time scale. Every time the stand is entered, the skid trails are redisturbed and subjected to rates of erosion similar to the initial harvest. The results presented here identify the relative impacts of multiple entries. The use of group selection as a harvest method may not be the best choice from the standpoint of site impact, and other methods should be considered, unless it is the only regeneration method that will result in achieving the desired future stand condition. In addition, the results suggest that erosion rates from clearcut, shelterwood, and leave-tree harvests return to baseline mature forest levels within the first few years after harvesting. The results from this study indicate that one- and two-entry silvicultural harvest treatments that follow BMP guidelines cause minimal soil movement and can quickly recover to pre-harvest erosion levels.

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Article

Suitability of Soil Erosion Models for the Evaluation of Bladed Skid Trail BMPs in the Southern Appalachians

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Abstract: This project measured soil erosion rates from bladed skid trails in the mountains of Virginia following a timber harvest, and compared measured erosion to four erosion model predictions produced by Universal Soil Loss Equation—Forest (USLE-Forest), Revised Universal Soil Loss Equation, v.2 (RUSLE2), Water Erosion Prediction Project—Road (WEPP-Road) using default files, and WEPP-Road using modified files in order to assess the utility of the models for these conditions. Skid trails were segregated into six blocks where each block had similar trail slopes and soils. Each block contained four skid trail closure treatments: (1) bare soil (Control); (2) residual limbs and tops (Slash); (3) grass seed (Seed); and (4) fertilizer, seed, and straw mulch (Mulch). All treatments had waterbars, the minimum trail closure best management practice (BMP), to provide upslope and downslope borders of experimental units. Site cover characteristics on each experimental unit were collected quarterly as input parameters for erosion models. The suitability of soil erosion models were evaluated based upon statistical summaries, linear relationships with measured erosion rates, Nash-Sutcliffe Model Efficiency, and a nonparametric analysis. Treatments were measured to have erosion rates of 15.2 tonnes ha^{-1} year^{-1} (Control), 5.9 tonnes ha^{-1} year^{-1} (Seed), 1.1 tonnes ha^{-1} year^{-1} (Mulch), and 0.8 tonnes ha^{-1} year^{-1} (Slash). It was determined that WEPP-Road: Modified (p-value = 0.643) and USLE-Forest (p-value = 0.307) were the most suitable models given their accuracy; however USLE-Forest may be better for making management decisions given its practicality.

Keywords: bladed skid trails; forest operations; forest harvesting; soil erosion modeling; best management practices

1. Introduction

The United States Environmental Protection Agency (USEPA) has identified sediment as the most damaging nonpoint-source pollutant in the U.S. [1]. Forest operations have the potential to produce substantial amounts of soil erosion that may be delivered as sediment in streams [2], thus a variety of forestry best management practices have been developed to either reduce soil erosion or interrupt delivery of eroded material to streams. In the southern Appalachians of the U.S., primary sources of soil erosion associated with forest operations are forest roads [3], overland [4] and bladed skid trails [5], and stream crossings [6]. Roads and skid trails are potentially highly erosive due to exposure of bare soil, terrain slope steepness, low road drainage standards [7,8], and traffic during poor weather conditions. The combination of these factors are known to increase erosion; therefore increasing the possibility of stream sedimentation and degradation [9,10]. Skid trails are potentially of more concern than haul roads because skid trails typically have lower standards than roads and skid trails may comprise a larger percentage of the harvest area [11]. Bladed skid trails are often used in the steep terrain of the region to facilitate skidder operator safety and operational efficiency. They differ from overland skid trails in that a bulldozer is used to construct the road, as opposed to having equipment simply drive on the surface of the soil [3,4]. Kochenderfer [12] estimated that up to 84% of exposed mineral soil in a harvest area was due to skid trails. More recently Worrell et al. [13] reported that bladed skid trails comprised approximately 8% of harvest area in the Appalachian Mountains. Wade et al. [5] measured erosion produced by bladed skid trails in the Piedmont region and determined that sediment production was strongly influenced by the application of forestry best management practices (BMPs). Trails with only waterbars produced 1.1 tonnes ha^{-1} year^{-1} of erosion whereas trails using slash or mulch cover produced <4 tonnes ha^{-1} year^{-1}.

Best management practices for skid trails have been developed to reduce the impacts of forest operations on water quality [14]. Skid trail BMPs include pre-harvest planning (e.g., layout of bladed skid trails), water control structures (e.g., water bars), and the use of ground cover on skid trails [15]. Commonly suggested methods of ground cover for bladed skid trails include grass seed, straw mulch, and residual limbs and tops from the forest harvest (slash) [16–19]. These methods of ground cover have been found to be both effective and economical in the past [4,5,20].

Soil erosion has the potential to reduce site productivity [21,22] and negatively impact water quality [2], thus quantification of the effects of forest best management practices on soil erosion are clearly important. However, on-site measurement of erosion is often costly and time consuming, thus models are commonly used to estimate erosion potentials [23,24]. Several models were developed to allow agricultural land managers to estimate and prioritize erosion issues and have been adapted to forest use over time [25]. Erosion models can be used by forest managers to make silvicultural, management, or even forest engineering decisions [26]. They are frequently modified to maintain and increase their accuracy and dependability [23]. One of the oldest and most widely applied soil erosion models is the Universal Soil Loss Equation (USLE) that was originally developed by the USDA in 1954 to estimate potential sheet and rill erosion from agricultural lands. The USLE is an empirical model that has been adapted to predict erosion from rangelands, minelands, watershed, and forest lands [27]. Dissmeyer and Foster [28] modified the USLE for use on forestlands (USLE-Forest). The USLE-Forest is relatively simple to use and has been widely used successfully on skid trails in the Piedmont physiographic region [4,5,20]. The USLE-Forest equation components are:

$$A = RKLSCP \tag{1}$$

where A is the annual soil loss per unit area, R is the rainfall and runoff factor, K is the soil erodibility factor, L represents the slope-length factor, S is the slope-steepness factor, C is the cover and management factor, and P represents the support practices factor [28]. R is determined based upon the average weather conditions at the location of interest. K is a function of multiple soil characteristics: soil texture, organic matter content, structure, and permeability. K values can be found in soil surveys or soil descriptions [29]. More accurate K-value estimates can be obtained by completing a nomograph included in the USLE-Forest manual. The L value is "the ratio of soil loss from the field slope length to that from a 72.6-foot (22.13 m) length under identical conditions" [28]. Likewise, slope-steepness factor (S) is defined as "the ratio of soil loss from the field slope gradient to that from a 9-percent slope under otherwise identical conditions" [28]. These two variables can be determined from a table found in A Guide to Predicting Sheet and Rill Erosion on Forest Land, written by Dissmeyer and Foster [28]. Cover and management (CP) factors are based upon the amount of bare soil, presence of canopy, soil reconsolidation, high organic matter content, fine roots, residual binding effects, onsite storage, and natural sediment trapping resulting in steps, and can be derived from tables published by Dissmeyer and Foster [28].

The USLE was later revised and converted to a computerized format, labeled the RUSLE or Revised Universal Soil Loss Equation. This model was first produced in the early 1990's, and RUSLE1.06 and RUSLE2 were both released in 2003. Although the original empirical algorithm from the USLE was kept, it was modified for improved accuracy by deriving soil loss factors in new ways. This revision included changes to make the model more suited for use with forest lands. Other improvements included updated rainfall coefficients, after changing some of the R factors in the eastern US based on weather data collected from more than 1200 weather stations. Soil erodibility (K) is varied seasonally for increased accuracy. The LS factor is improved in that it takes into account the "susceptibility of the soil to rill erosion relative to interrill erosion" and the cover factor uses a new algorithm for determining cover based on prior land use, canopy cover, soil cover, and soil surface roughness [30]. RUSLE2 has no specific data files for forest roads, however there are "highly disturbed land" files that can be modified to suit different forest road treatments [24].

The Water Erosion Prediction Project (WEPP) is a physically-based model produced by the U.S. Department of Agriculture Natural Resource Conservation Service (NRCS) and U.S. Forest Service (USFS) to replace the USLE formula. WEPP "models soil erosion as a process of rill and interrill detachment and transport" [31] as opposed to empirically modeling the ground conditions [32,33]. The WEPP model had additional potential utility because it estimates daily conditions that affect erosion, over the course of a year. In this, senescence, plant growth, residue accumulation and decomposition, as well as daily temperatures and soil water availability are taken into account to provide a very detailed estimate of soil loss over time. An additional benefit is the ability to model complex slopes and forest road profiles, with features such as cutslopes and fillslopes, ditches, and road surfaces [23]. Four types of data files are required to run WEPP: (1) a climate file, to include data on daily precipitation and temperature; (2) a hillslope file, which can contain multiple points to describe a slope's shape; (3) a soils file, which can include multiple soil types across the hillslope; and (4) a management file containing information on soil disturbances and vegetative conditions present [26]. Weather data are obtained through Cligen, the USDA's weather resource. This weather file models weather data on a daily basis for more than 1000 climates [34]. Using the hillslope file, WEPP determines the erosion or deposition rates for at least 100 points of the hillslope if there is any runoff predicted that day [35]. Because WEPP, like other models, was originally intended for cropland or rangelands, there have been many efforts to adapt it for forest uses [36–41]. One of these interfaces is the WEPP-Road model interface. This program allows the user to determine the amount of sediment delivered to the stream through the forest buffer and amount of sediment eroded from each portion of the road, as well as determine the presence of a sediment plume in the forest [42]. At this time, the selections for cover and land use scenarios appear to limit WEPPs utility for estimation of erosion for many eastern forest management regimes [43].

There have been several attempts to assess the performance of these three models. Wade et al. [24], compared sediment trap data to predictions by all three models. Erosion rates were estimated from different sections of bladed skid trail in the Piedmont of Virginia using sediment traps, and were then compared to erosion rates predicted by USLE, RUSLE2, and WEPP models. It was found that overall, all three models performed well enough for identifying erosion hazards and making management decisions. When comparing the modeled data, it was determined that USLE-Forest ranged from 0.9× to 2.2× the actual erosion rates from data collected from the sediment traps. RUSLE2 ranged from 0.4× to 2× the actual erosion, and WEPP-Road ranged from 2.3× to 7.5× [24]. These data indicated that the USLE-Forest and RUSLE2 can be useful at approximating erosion rates, but WEPP-Road values should only be used for ranking purposes on bladed skid trails. Foster, Toy, and Renard [44] found similar results when comparing USLE, RUSLE1.06, and RUSLE2. WEPP modeling efforts can be improved with laborious programming, but is time consuming and requires many measurements to modify the working files [45]. Lang et al. [45] found that soil erosion models worked best when estimating erosion rates less than 11.2 Mg ha^{-1} $year^{-1}$; however when erosion rates surpassed this amount model estimates varied widely. Croke and Netherly [25] compared the USLE and WEPP on skid trails in Australia and concluded that the USLE was more user friendly while the WEPP model was a better predictor of erosion on skid trails. However, their investigation indicated that neither method was wholly satisfactory for estimation of erosion. One important distinction to note is the difference between empirical models, which are simpler to use but are based on observations and measurements; and physically-based models which replicate erosion processes as equations [32,33]. Both types of models have their own advantages and applications [32].

Overall, the literature clearly indicates that erosion from skid trails can be a significant source of nonpoint source pollution from forestry operations [2,46–48] and that rates of erosion for different types of skid trail BMPs are warranted in order to evaluate BMP efficacy. This aspect of the problem is addressed by a companion paper [49]. Furthermore, the literature indicates that erosion models have been used with varying success to estimate erosion from skid trails, but modeled erosion rates from bladed skid trails in mountainous terrain have not been compared to direct erosion measures.

The primary objective of this study was to compare measured erosion rates from four bladed skid trail closure methods in mountainous terrain with those produced by the Universal Soil Loss Equation (USLE-Forest), the Revised Universal Soil Loss Equation (RUSLE2), the Water Erosion Prediction Project (WEPP-Road: Default), and a more modified version of WEPP (WEPP-Road: Modified).

2. Materials and Methods

2.1. Research Site

The study site is located in the Ridge and Valley physiographic province, on Virginia Tech's Fishburn Forest located in Montgomery County, Virginia (Figure 1). This physiographic province is characterized by long mountain ridges with constant linear valleys in between them. The average yearly precipitation is 103.86 cm [50]. The average high and low temperatures for this location in January are 5.3 °C and −5.9 °C. The average high and low temperatures in July are 27.9 °C and 15.6 °C [50]. Rainfall data were collected from a nearby weather station for the duration of the study period (Figure 2) [51] and were used to compare the effects of rainfall on erosion rates [49].

The soils are very shallow, well drained silt loams, being derived mostly from shale, siltstone, and sandstone residuum. Berks, Weikert, Berks-Weikert and Clymer soil series (*Lithic Dystrudepts*) dominate the site [29]. The harvested stands were primarily mixed upland hardwood-pine stands, composed of white pine (*Pinus strobus* L.), chestnut oak (*Quercus montana* Willd.), and white oak (*Quercus alba* L.). Slopes in this region range from 0% to 100%.

The site was harvested in late 2014-early 2015 using a shelterwood overstory removal of upland hardwoods and pine. Skid trails were laid out in a "logger's choice" arrangement. Skid trail slopes ranged from 0–35%, with an average slope of 16%. Skid trail sideslopes ranged from 5–45%. The skid trails were divided into 6 blocks based on slope class. Two blocks were arranged in each slope class: Gentle (0–10%), Moderate (11–20%), and Steep (>20%). Each block of treatments contained the four closure methods that were compared in this experiment.

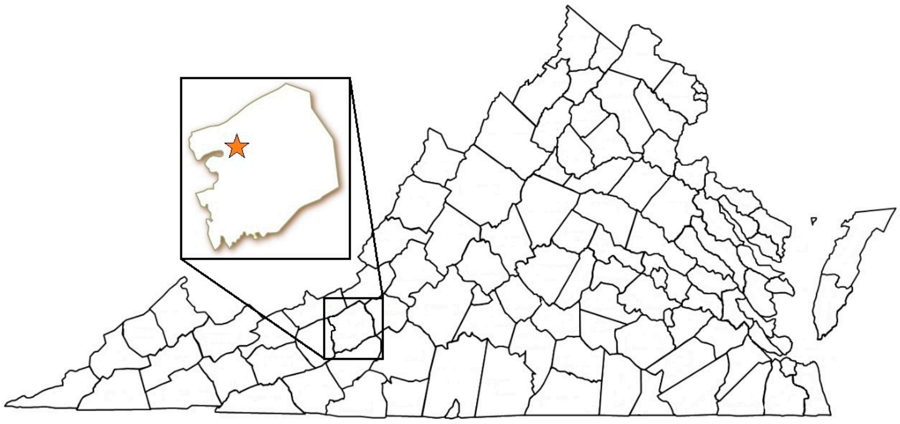


Figure 1. Timber harvest was located in Montgomery County of southwestern Virginia, located on the southeastern coast of the United States. Map is not to scale.

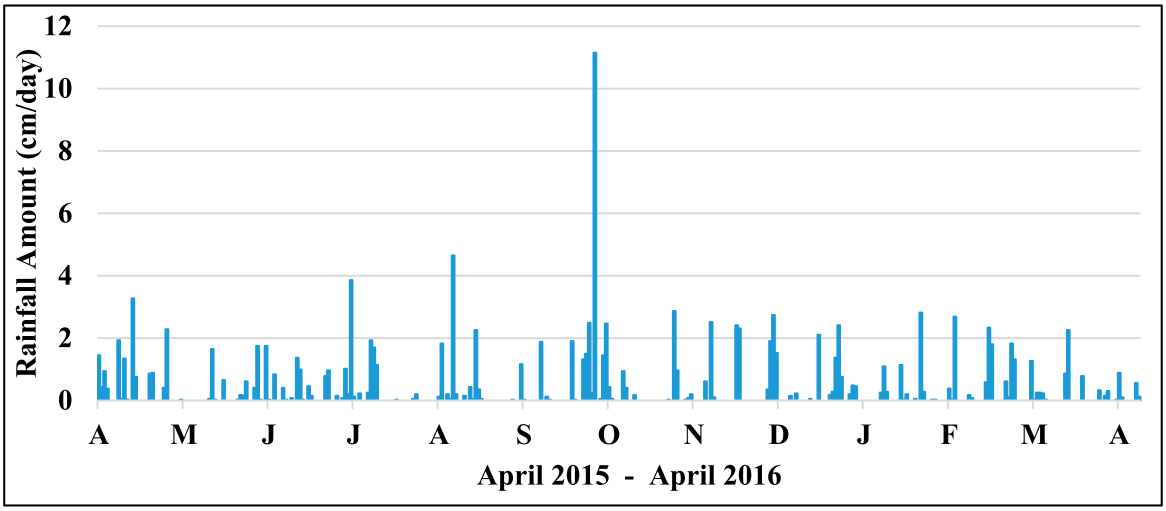


Figure 2. Daily rainfall amounts over the course of 1 year of data collection.

Treatments consisted of 15.2 m segments of skid trail, approximately 2 m wide. On steep slopes (>20%) the treatments were shortened to 12.2 m in order to comply with state BMP guidelines [14]. Closure methods were randomly assigned to each of the 24 experimental units. Units were separated by waterbars. Skid trail treatments were closed to vehicle traffic over the course of the study period in order to avoid any effects of heavy trafficking on soil erosion rates [52].

2.2. BMP Treatments

Four types of treatments were used in this study: (1) waterbars only (Control), (2) waterbars with grass seed (Seed), (3) waterbars with grass seed, fertilizer, and straw mulch (Mulch), and (4) waterbars with slash (Slash) (Figure 3). The Control treatment consists of waterbars with no ground cover treatments and represents the minimum acceptable BMPs as a control reference to which the other treatments were compared. For the Seed treatment, grass seed was applied at the time of skid trail closeout (April 2015) using a mix of 50% perennial ryegrass (*Lolium perenne* L.) seed and 50% K-31 fescue (*Festuca arundinacea* Schreb.), based on suggestions from the VDOF BMP manual [14]. Seeds were spread with a hand operated seeder to ensure adequate coverage, at a rate of approximately 168 kg/ha. For the Mulch treatment, the same grass seed mixture was applied, followed by fertilizer and straw mulch. Mulch was spread by hand to ensure near total coverage, at a depth of 3–6 cm across each experimental unit [14]. Fertilizer [5-10-10 (N, P_2O_5, K_2O)] was added at a rate of 336 kg/ha to provide sufficient nutrient availability for the grass. Slash treatments utilized residual slash from on-site logging operations, and was primarily composed of yellow-poplar (*Liriodendron tulipifera* L.), hickory (*Carya spp.*), scarlet oak (*Quercus coccinea* Münch.), chestnut oak (*Quercus montana* Willd.), white oak (*Quercus alba* L.), white pine (*Pinus strobus* L.), and Virginia pine (*Pinus virginiana* Mill.). Slash was hand applied onto skid trails to ensure similar coverage and then compacted with a bulldozer to make contact with the ground. After being tracked in by the bulldozer, slash was at a depth of 0.6–0.9 m.

Figure 3. A comparison of the four skid trail closeout best management practices (BMPs) used in the study.

2.3. Sediment Trap Installation and Measurement

A full description of the collection of field data and the effectiveness of skid trail closure methods is available from Vinson et al. [49]. Sediment traps were used to measure erosion rates in the field over the course of the year. These sediment traps consisted of silt fences that were joined to the downslope waterbars so that they collected all runoff from the skid trail treatment (Figure 4). Berms were constructed on either side of the skid trail to limit overland flow and to ensure runoff from the treatment made it into the sediment trap. Within each sediment trap, metal pins were driven into the ground at regular intervals in a grid pattern. The depth of the sediment was measured at each sediment pin on a monthly basis, as was the area of the sediment collected. From this a volumetric accumulation of sediment was determined over time. Bulk density samples were taken from the accumulated sediment, and this was used to convert the volume of collected sediment to a gravimetric amount.

Figure 4. An example of the silt fence sediment traps used in the study. Sediment pins were driven into trap area on a grid pattern and measured at regular intervals.

2.4. Erosion Model Parameters

For modeling, each experimental unit was divided into three sections. The first section being the downhill side of the upslope waterbar, the second section being the actual skid trail surface, and the third being the uphill side of the downslope waterbar (Figure 5a,b). Sections 1 and 2 were modeled together. Since the two waterbars have sides that are contributing to the area, they needed to be accounted for in the modeling as well. The slope and length of every section was measured using a total station. The USLE was used to estimate erosion from each section of each treatment, and estimates were combined in a weighted average total erosion estimate for each treatment. Grass treatments had model estimates determined both before and after seed germination for a comparison, as ground cover values were measured in the field every 3 months to account for variations in seasons, the establishment of grass, and decomposition of slash and mulch. Slope, climate data, soil characteristics, and cover practices were determined for each experimental unit and input into all three models to estimate soil loss. Actual erosion rates were converted to tonnes ha^{-1} year^{-1} in order to compare estimates provided by all three models. For each treatment area the following data were collected: ground cover, slope gradient and slope length, percent of soil in clay, sand, and silt, soil rock content, and rainfall data.

2.5. USLE-Forest Parameters

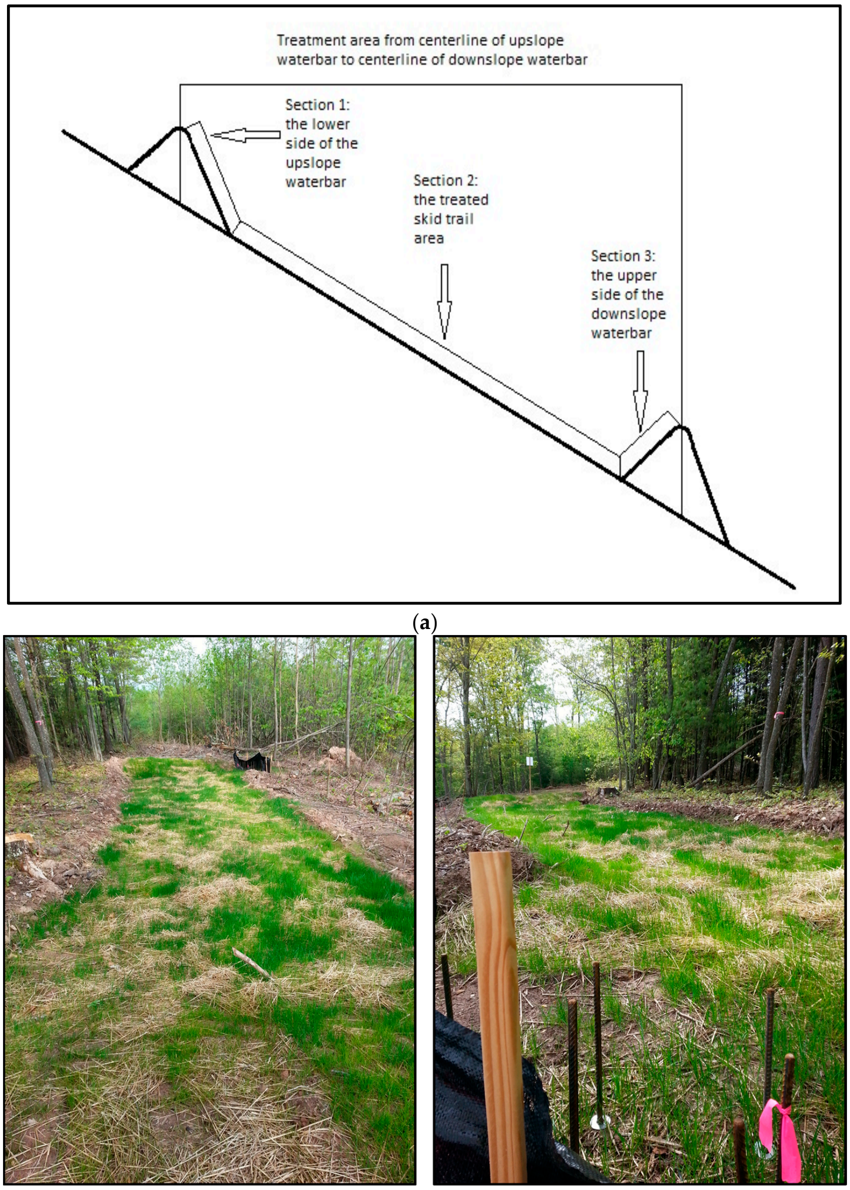
A rainfall runoff factor of 150 was used as it was derived from a rainfall contour map provided by the USLE-Forest manual [28]. A soil erodibility factor of 0.43 was obtained from the Montgomery County, VA Soil Survey [29]. A total station was used to measure the slope length and gradient for the upper and lower waterbars, and the section of bladed skid trail located between the two. Slope lengths were often too small to be found in the USLE-Forest manual, and therefore were obtained using the equation:

$$LS = (\lambda/72.6)^{m}(65.41\sin\theta^{2} + 4.65\sin\theta + 0.065) \quad (2)$$

where λ is the slope length in feet, θ is the slope angle in degrees, and m is 0.2 for <1% slopes, 0.3 for 1% to 3% slopes, 0.4 for 3.5% to 4.5% slopes, and 0.5 for ≥5% slopes [28]. The bladed skid trails were considered to be tilled soil, therefore having CP factors to include bare soil, residual binding, and soil reconsolidation; canopy effect; steps; onsite storage; invading vegetation; and contour tillage. Bare soil percentages were calculated by creating transects across the treatment, with evenly spaced points. At each point, ground cover was determined to be either bare or covered, and ground cover percentage was calculated. Ground cover included vegetation, straw mulch, woody residues, rock fragments, and leaf litter. These measurements were collected quarterly over the course of a year to cover the span of four seasons. A weighted average of the four periods was used to determine a final annual erosion rate for each treatment.

2.6. RUSLE Parameters

Erosion estimates were also predicted using RUSLE2. Montgomery county weather and soil files were imported into the program to more accurately estimate soil loss. Climatic data were accessed from the NRCS database [29] for Montgomery County, Virginia. Daily and monthly average rainfall rates were included in these data. Montgomery county soil survey indicated the Berks-Weikert complex as the soil series for the site [29], the soil file for which was then imported into the program. The soil file contains information on the erodibility of the soil, the soil texture, and acceptable loss rates. For every treatment, a slope profile was created based on the measured slope and length of each section of the treatment area. Management files had to be created for each BMP treatment, as there were no pre-made files to represent forest roads or skid trails. All operations were set to occur in late April to coincide with the initial site installation. The “highly disturbed land/blade cut” option was selected to represent the Control treatments. Seed treatments used this file, but with the modification of “broadcast seed operation” also used. “Fescue” and “Ryegrass” were used as the species of seed applied, and the "live surface cover" was modified to represent the percentage of ground cover contributed by the germination of the grass seed as time increased. Mulch treatments used this file; however it was modified to include the "add mulch" operation in the form of "bale straw or residue." The type of mulch chosen in this instance was "wheat straw." The option "specify cover directly" was chosen and modified for each treatment to correspond with cover percentages measured in the field. Slash treatments were best represented by the "highly disturbed/blade cut" option, followed by the "add mulch" operation, with "prunings, orchard and vineyard, flail shredded" chosen as the material. The cover was again manipulated by modifying the "specify cover directly" parameter, and by modifying the decomposition half-life of the material to 1800 days, as based on rates used by Wade et al. [24] to represent the decomposition rate of woody debris from southern Appalachian hardwood forests.



(**a**)

(**b**)

Figure 5. (**a**) Profile of skid trail treatment. Sections 1 and 2 (lower side of upslope waterbar and skid trail surface) were modeled together; Section 3 (upper side of downslope waterbar) was modeled separately; (**b**) Photographs of a Mulch treatment from the upslope waterbar looking down toward treatment, downslope waterbar, and sediment trap (first photograph), and from the sediment trap looking upslope to waterbar at top of treatment (second photograph).

2.7. WEPP-Road: Default Parameters

WEPP-Road is dependent upon four different types of files to predict soil erosion rates. The software features a database that contains basic files for each of these that can be easily modified to best represent the site. The four types of files are: (1) climate; (2) soil characteristics; (3) slope length and gradient; and (4) land management operations. A climate file for Blacksburg, Virginia is embedded into the software and was therefore chosen as the best representative of the site conditions, as the weather station is less than 8 km (5 miles) away from the study site. Within the WEPP-Road soils database, the file most similar to a Berks-Weikert complex was the "Disturbed Skid Clay Loam," which was chosen for modeling on this site. Soil rock content for each treatment varied from 10–36%, and was directly correlated with slope steepness. Therefore, it was determined that rock content of the soil would be a parameter which needed modification for each treatment, as well as factored into the ground cover in the initial conditions and management files. Slope length and gradient values were modified for each treatment as they were measured with the total station. The "Forest Bladed Road" management file was used for the Control treatments, and was modified for the others. Initial conditions were modified by their initial rill and interrill ground cover percentage, as measured in the field. Seed treatments used this base of "Forest Bladed Road" as the initial conditions and then were modified with the "fescue" and "annual ryegrass at a low fertilization rate." Mulch treatment management files used this file as a base, however "fescue residue" was added as a mulch at a rate of 0.788 kg m^{-2}. Similar to RUSLE2, there are no management files in WEPP-Road that represent Slash treatments. Since there are no woody residue mulch treatments, the same "fescue residue" mulch was chosen. The actual application rate (by weight) that was used to apply slash in the field was used to model this treatment, similar to methods used by Wade et al. [24]. All treatments were modeled for one year.

2.8. WEPP-Road: Modified Parameters

WEPP-Road was then used to model these treatments a second time, using files that were modified to more accurately represent the soil and treatment conditions throughout the year. The primary reason for this being that WEPP-Road has a large number of parameters that can be manipulated when using the model. We wished to determine if collecting more data and making use of more of the model parameters would provide a significantly more accurate soil erosion estimate, and how much more labor would be needed to accomplish this. A soils file was created for each experimental unit based on the "Disturbed Skid Clay-Loam" file used earlier, but modified with the soil rock content and soil particle size present in each of the experimental units. The model was used to calculate interrill erodibility, rill erodibility, critical shear, and effective hydrologic conductivity instead of using the default preset values in place for that particular file. The weather file remained the same, as the Blacksburg, Virginia climate file was determined to be accurately representative of the study site based on its geographic proximity. Slope gradient and length were created once again based on measurements taken in the field with a total station. For the Control treatment, the same "Forest Bladed Road" management file was used for the control treatments, and was modified for the others. The Forest Bladed Road file was modified with initial rill and interrill ground cover percentage, as well as bulk density of the experimental units. This time, the "Initial Plant" field in the "Initial Conditions" file was changed to "Skid Trail-Disturbed," and the "Days Since Last Tillage" field was modified to reflect that the disturbance had just occurred (0 days). Seed treatments used this base of "Forest Bladed Road" as the initial conditions and then were modified with the "fescue" and "annual ryegrass at a low fertilization rate." Mulch treatment management files used this file as a base, however "annual ryegrass at a high fertilization rate" was used instead of "annual ryegrass at a low fertilization rate;" and "fescue residue" was added as a mulch at a rate of 0.788 kg m^{-2}. Once again, there are no files in WEPP-Road that represent woody material for Slash treatments. In this instance, the "Rock" file was chosen in the "Residue Added" field, and was modified to represent the decomposition rate of woody material. This file was chosen because it is the closest available file that could be modified to represent a slash treatment. The actual application rate (by weight) that was used to apply slash in the field was used to model this treatment. All treatments were modeled for one year.

2.9. Data Analysis

Treatment effects for each erosion model were analyzed using JMP statistical software [53]. A variety of methods were used to compare the trapped and modeled estimates including: (1) summary statistics; (2) linear relationships; (3) Nash-Sutcliffe Model Efficiency (NSE) [54]; and (4) a nonparametric analysis. Summary statistics were analyzed to examine means and standard deviations for each treatment using each erosion model. Linear relationships, and NSE were evaluated to determine the accuracy of the models when compared to the actual trapped erosion rates, and a nonparametric comparison for each pair using the Wilcoxon method was conducted to compare these models to each other. Similar comparisons have been conducted by Wade et al. [24] and Croke and Netherly [25].

3. Results

3.1. BMP Treatment Effectiveness

The sediment collected in traps clearly indicates an overall effectiveness of the BMPs compared. Control treatments with waterbars only were measured to have an erosion rate of 15.2 tonnes ha^{-1} $year^{-1}$. Seed treatments were measured to have an erosion rate of 5.9 tonnes ha^{-1} $year^{-1}$, and Mulch treatments eroded at a rate of 1.1 tonnes ha^{-1} $year^{-1}$. Slash treatments eroded at a rate of 0.8 tonnes ha^{-1} $year^{-1}$. Each model ranked the BMP treatments as having the Control as the most erosive, and the Mulch treatment as least erosive. All models tended to over-estimate the erosion rates of Slash treatments. The Control treatments represent the minimum level of BMPs that are acceptable for skid trail closeout, and was measured to have eroded at rates 2.8× to 8× that of seeded treatments, the next most erodible treatment [49]. Mulch and Slash treatments both reduced average sediment rates to minimal amounts. Adding mulch and fertilizer provided the trail with immediate ground cover, which was not attained by the Seed treatments due to the time necessary for germination. Mulch also aided in the retention of soil nutrients and moisture, as well as reduced predation of the grass seeds from wildlife. Slash provided immediate ground cover, and offers the additional benefits of reducing traffic on the trail, in the form of four-wheelers and pedestrians. After cost analysis, Slash was also shown to provide the greatest benefit in soil erosion reduction per dollar spent in installation [49]. This is due to the fact that no materials are needed to be purchased to install a slash treatment, since slash is already present following the harvest. For all treatments, as ground cover increased, soil erosion decreased. Slope and length did have effects upon the erosion rates, as did rock content of the soil. Steeper slopes in this soil series tended to feature higher rock fragment contents, which acted to increase soil cover over time as the soil around them was eroded away. More information on BMP effectiveness and erosion rates over time may be found in Vinson et al. [49].

3.2. Model Suitability

Models were evaluated using the four different techniques outlined earlier. Each of the model predictions was compared to the trapped sediment data after one year (Table 1). WEPP-Road: Modified had the closest overall mean erosion rate estimate for the Control treatments, while RUSLE2 had the closest overall mean erosion rate for the Seed and Mulch treatments and USLE-Forest provided the closest overall mean erosion rate for the Slash treatments. It is to be noted that for the Control and Slash treatments, the estimates provided by RUSLE2 were more than double the next closest estimate, indicating that its results may be very inconsistent based on the conditions being modeled.

Linear relationships were also used to determine model accuracy. Each of the sets of modeled data were compared to the data collected by sediment traps. Accurate models are expected to have a linear relationship to the collected data [24]. In this study, RUSLE2 was shown to have the highest

R^2 value amongst the three, at 0.6069 (Figure 6). This indicates that RUSLE2 has the best estimated linear relationship with the trapped data. The linear relationship of WEPP-Road: Default to measured data has the second highest R^2 value of 0.5855 (Figure 7), followed by USLE-Forest with an R^2 value of 0.4652 (Figure 8). When compared to the 1:1 line, the inclination of the trend lines of these models indicate that they both tend to overestimate erosion rates. Lastly, the relationship of WEPP-Road: Modified to the measured erosion data has the lowest R^2 value (0.0977) which is indicative of a poor model accuracy (Figure 9). This could have occurred due to inadequacies in modeling just one specific treatment.

Table 1. Summary statistics for each of the models analyzed. For each treatment, there is an asterisk (*) next to the model prediction that is closest to the measured amount.

Treatment	**Method**	**Erosion Rate (tonnes ha^{-1} year^{-1})**	**Std Dev** [a]	**Std Error Mean** [b]	**Lower 95%**	**Upper 95%**
Control	Measured	15.2	12.1	5.0	2.4	27.9
	USLE-Forest [c]	24.1	11.3	4.6	12.3	36.0
	RUSLE2 [d]	66.4	29.2	11.9	35.8	97.1
	WEPP-Road [e]: Default	27.1	6.9	2.8	19.8	34.3
	WEPP-Road [e]: Modified	* 10.8	9.5	3.9	0.8	20.7
Seed	Measured	5.9	5.4	2.2	0.2	11.6
	USLE-Forest [c]	16.5	12.5	5.1	3.4	29.7
	RUSLE2 [d]	* 6.4	3.6	1.5	2.6	10.2
	WEPP-Road [e]: Default	12.7	6.2	2.5	6.2	19.2
	WEPP-Road [e]: Modified	2.8	2.3	0.9	0.4	5.2
Mulch	Measured	1.1	1.0	0.4	0.1	2.1
	USLE-Forest[c]	0.3	0.3	0.1	0.0	0.7
	RUSLE2 [d]	* 0.6	0.4	0.2	0.2	1.1
	WEPP-Road [e]: Default	1.6	0.7	0.3	0.9	2.4
	WEPP-Road [e]: Modified	0.5	0.2	0.1	0.2	0.7
Slash	Measured	0.8	0.6	0.2	0.2	1.4
	USLE-Forest [c]	* 2.3	1.9	0.8	0.3	4.3
	RUSLE2 [d]	21.8	11.0	4.5	10.3	33.3
	WEPP-Road [e]: Default	7.3	3.6	1.5	3.5	11.0
	WEPP-Road [e]: Modified	2.4	1.8	0.7	0.5	4.2

[a] Standard Deviation; [b] Standard Error of the Mean; [c] Univsersal Soil Loss Equation—Forest; [d] Revised Universal Soil Loss Equation, v.2; [e] Water Erosion Prediction Project—Road.

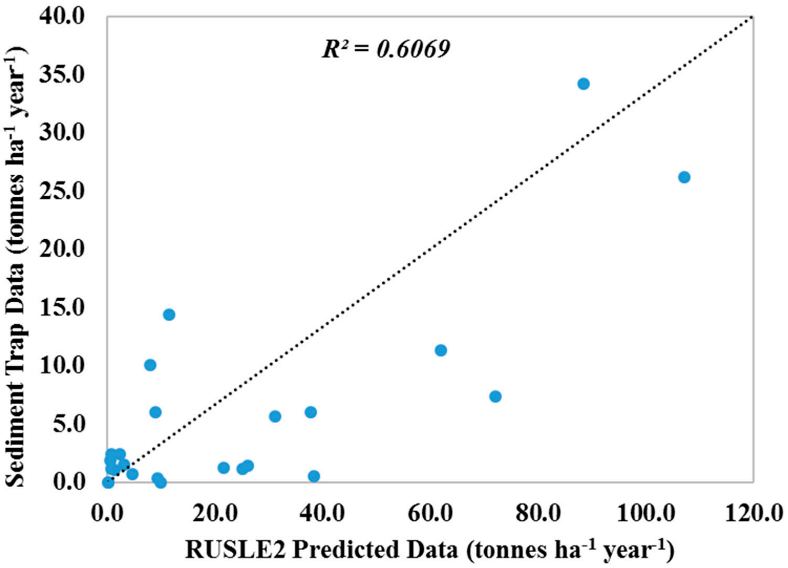


Figure 6. Linear relationship of RUSLE2 modeled data and actual measured erosion data. Line on graph represents a 1:1 relationship.

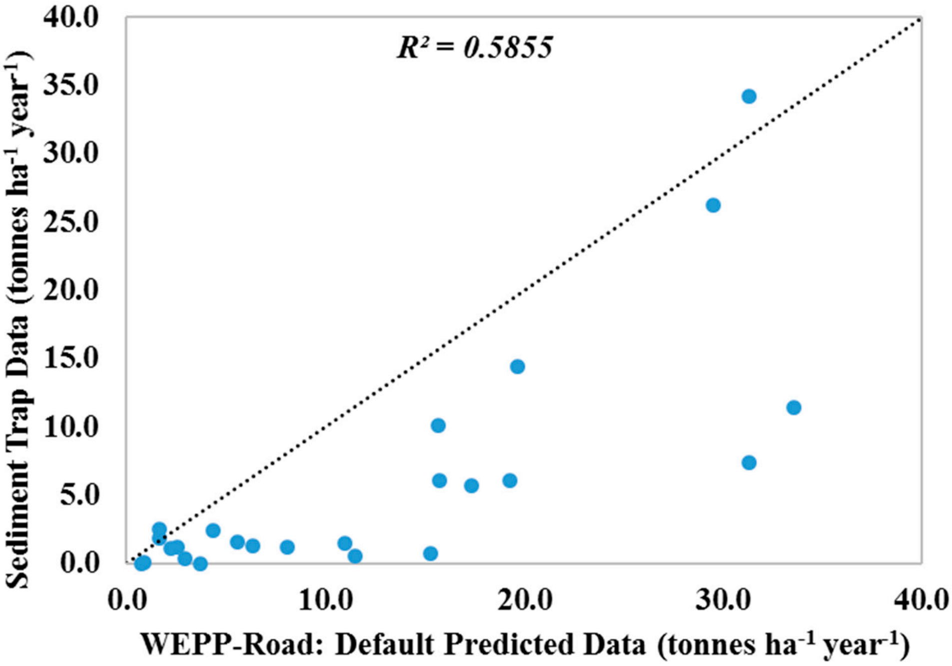


Figure 7. Linear relationship of WEPP-Road modeled data and actual measured erosion data. Line on graph represents a 1:1 relationship.

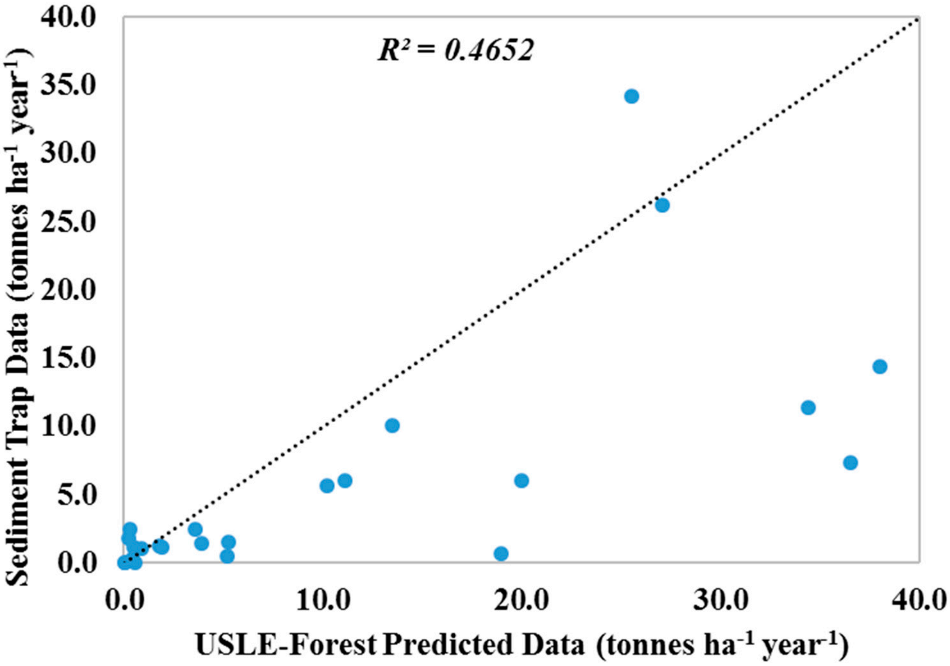


Figure 8. Linear relationship of USLE-Forest modeled data and actual measured erosion data. Line on graph represents a 1:1 relationship.

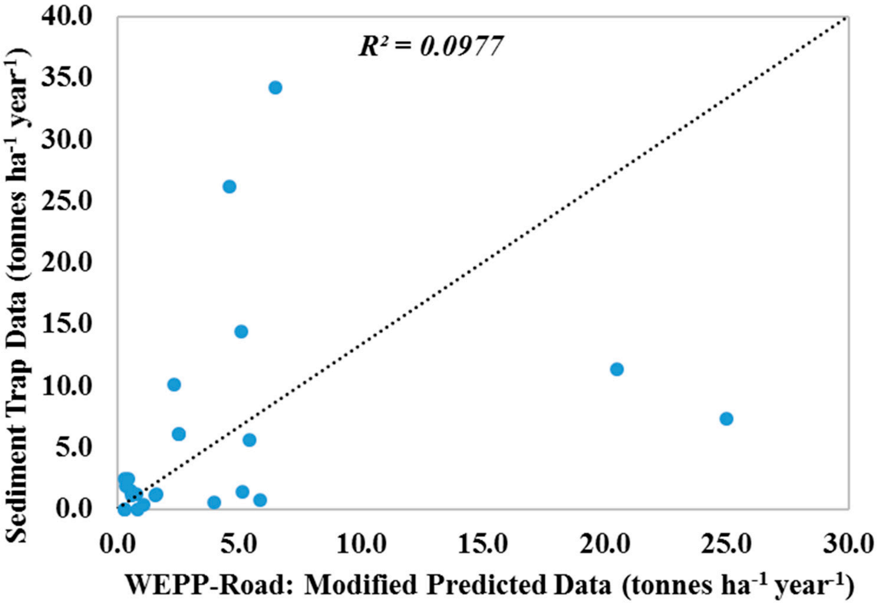


Figure 9. Linear relationship of WEPP-Road: Modified modeled data and actual erosion data. Line on graph represents a 1:1 relationship.

The Nash-Sutcliffe Model Efficiency (NSE) is commonly used to evaluate hydrologic models. The range of efficiency is from $-\infty$ to 1, with values from 0–1 indicating that the model is a good predictor of the measured values. As values approach 1, the model is a more accurate representation of true values. Negative values indicate that the mean of the measured values are a better predictor than the model itself [54], with lower values representing less suitable models. NSE was calculated for each of the treatments and each of the models as a whole (Table 2). All values were negative with the exception of the WEPP-Road: Default (0.15) and RUSLE2 (0.23) models at predicting Mulch treatment erosion rates. The NSE values for the other two models are negative for this treatment (−0.29, and −0.24), however they are substantially greater in value than most other treatment categories. This is evidence that the models did reasonably well at predicting erosion from Mulch treatments. Control treatments were found to have the lowest values for each model, indicating that all models were insufficient at predicting soil loss from bare soil treatments. When evaluating the entire model over all types of treatments, RUSLE2 has a much lower NSE value (−1174.15) than USLE-Forest (−146.35), WEPP-Road: Default (−115.01), and WEPP-Road: Modified (−102.72) indicating that it is the least suitable of the three models. Using this "whole model evaluation," WEPP-Road: Modified has the highest NSE score and would be ranked the most accurate of the four.

Table 2. A comparison of predicted erosion rates and their Nash-Sutcliffe Model Efficiency (NSE) values for the whole model and for each treatment type.

		Control	Seed	Mulch	Slash	Whole Model NSE [a]
Trapped Sediment	(tonnes ha^{-1} year^{-1})	15.15	5.87	1.10	0.78	–
USLE-Forest [b]	(tonnes ha^{-1} year^{-1})	24.14	16.54	0.33	2.30	–
	NSE [a]	−476.81	−276.44	−0.29	−4.87	−146.35
RUSLE2 [c]	(tonnes ha^{-1} year^{-1})	66.44	6.36	0.62	21.77	–
	NSE[a]	−5681.25	−8.90	0.23	−651.77	−1174.15
WEPP-Road [d]: Default	(tonnes ha^{-1} year^{-1})	27.06	12.70	1.62	7.25	–
	NSE [a]	−442.32	−94.39	0.15	−60.95	−115.01
WEPP-Road [d]: Modified	(tonnes ha^{-1} year^{-1})	10.75	2.05	0.45	2.37	–
	NSE [a]	−520.42	−46.55	−0.24	−4.49	−102.72

[a] Nash-Sutcliffe Model Efficiency; [b] Univsersal Soil Loss Equation—Forest; [c] Revised Universal Soil Loss Equation, v.2; [d] Water Erosion Prediction Project—Road.

Lastly, the models were analyzed using a nonparametric comparison for each pair using the Wilcoxon method (Table 3). For this method, each model was individually compared to the measured data to find significance. In this instance, WEPP-Road: Default and RUSLE2 were considered to be significantly different to the measured data (p-value = 0.0046, p-value = 0.0154).

Table 3. Nonparametric analysis comparing each model to the measured data using Wilcoxon method. $\alpha = 0.05$.

Model	Score Mean Difference	Standard Error Difference	p-Value
USLE-Forest [a]	4.13	4.04	0.3074
RUSLE2 [b]	9.79	4.04	* 0.0154
WEPP-Road [c]: Default	11.45	4.04	* 0.0046
WEPP-Road [c]: Modified	−1.88	4.04	0.6427

[a] Univsersal Soil Loss Equation—Forest; [b] Revised Universal Soil Loss Equation, v.2; [c] Water Erosion Prediction Project—Road.

4. Discussion

Results indicate that the BMPs effectively provided ground cover necessary to reduce erosion. Generally, as ground cover increased, erosion rates decreased. It was seen in the field that rock fragments had a major impact on ground cover and therefore erosion rates [49], which may have been difficult for the models to assess. Slash, seed, and straw mulch have been shown to reduce erosion from skid trails and temporary roads. Slash and straw mulch both provide immediate cover, especially during the initial months at which one can expect erosion rates to be the highest. Slash is readily available, lasts longer than straw mulch, and is more effective at reducing trail traffic. Both slash and straw mulch may also improve the chemical and physical properties of the soil through decomposition. This study indicates that additional road closure BMPs can be used to enhance the minimal effects of waterbars only. Erosion models were shown to have varying degrees of accuracy and suitability based upon their use. Similar conclusions were also reached by Wade et al. [24], Brown et al. [43], and Lang et al. [45].

USLE-Forest was slightly different than the other models in terms of management and cover practices. Whereas RUSLE2 and WEPP-Road model a specific operation and make assumptions based upon its effects, USLE-Forest models these effects directly. This has a noticeable impact on the accuracy of the model. However, many field measurements are required to produce a feasible value from this model. Soils, ground cover, and canopy cover must all be measured in the field. However, this does allow for a more "field available" prediction, whereas RUSLE2 and WEPP-Road both require the use of computer software. The USLE was shown to be the most user-friendly of the three models, in that it can easily be performed with a manual in the field with relatively minimal training and still provide an acceptable estimate of soil loss. Of all the models compared, USLE-Forest provided a consistently, reasonably reliable prediction with minimal difficulty.

RUSLE2 was determined to be the least suitable of the four models assessed, in that its NSE values and nonparametric p-values are all the least favorable of the models. One of the factors affecting the model accuracy is the aforementioned soil rock content. While soils files were accurate enough for the model, it did not take into account the increased soil ground cover from the high soil rock content over time. Other factors include the fact that operations are modeled as such instead of the effects that those operations had upon the ground surface [44]. The primary reason for poor performance of this model would be the fact that there are no management files available for bladed roads or slash treatments. However, RUSLE2 was able to model Seed and Mulch treatments exceptionally well. This shows that RUSLE2 can sufficiently model soil loss for certain ground conditions, but overall may be too inconsistent to be trusted.

WEPP-Road (both Default and Modified) was shown to be the most accurate of the four models based on NSE. This can be attributed to a number of factors. This is the only model that takes into account soil rock content in its analysis, which could have helped to make predictions more accurately. In addition to this, there are forest road and skid trail treatment files available, which gives WEPP-Road an advantage over RUSLE2. One major disadvantage to WEPP is that it does not feature any wood or wood-fiber based mulches to represent slash treatments. Both WEPP-Road and RUSLE2 are at a disadvantage, in that when compared to USLE-Forest, they are difficult to learn initially. They also require significant computer use, which is not always practical for field management decisions.

WEPP-Road: Modified outperformed WEPP-Road: Default. Of the predictions that the modified WEPP model produced, 71% were closer to the measured value than the default WEPP predictions. Lang et al. [45] found similar results when comparing soil erosion models to trapped data from forest haul roads. However, there were some treatments that WEPP-Road: Default modeled better than WEPP-Road: Modified. Inaccuracies in the modified version may have arisen from issues with certain parameters, resulting in the low correlation of modeled points in a linear relationship. It was noted that when using the WEPP model, as the rock content of the experimental unit was increased, the predicted erosion rate dramatically increased. This is not reflective of what was measured or observed in the field measurements, and is also contrary to what other studies have found regarding the effects of soil rock content on the erosion of soils [55,56]. For this reason, we perceive WEPP to be limited in its uses of producing accurate soil erosion predictions on steep, rocky slopes.

5. Conclusions

The primary objective of this study was to evaluate models based on the similarity of their predictions to erosion data collected in the field. After having modeled 24 experimental units over the course of a year using all four models, they were analyzed to determine accuracy. Four BMP treatments were compared to show that adding grass seed; fertilizer, grass seed, mulch; or slash were able to significantly reduce the amount of soil erosion from a bladed skid trail. Mulch and Slash treatments were both the most effective at reducing soil erosion, as they provide immediate ground cover. Based on the Nash-Sutcliffe Model Evaluation and a nonparametric analysis, USLE-Forest and WEPP-Road (both Default and Modified) were significantly better than the other models applied to this site. RUSLE2 was shown to be insufficient for use in modeling bladed skid trails, having over-predicted almost every value. However, of all the soil erosion models, RUSLE2 featured the best linear relationship with the measured erosion data. Each model was able to rank the BMP treatments as having the Control as the most erosive, and the Mulch treatment as least erosive. All models overestimated the erosion rates for Slash treatments, with RUSLE2 placing it at the second-highest erosion rate.

USLE-Forest and WEPP-Road (both Default and Modified) were shown to have been the best suited for this site. With improvements in management and soil files for RUSLE2 and the Default WEPP-Road models, we can expect model accuracy to significantly increase, therefore broadening their applicability to more varied sites. However, as can be seen with results from our Modified WEPP-Road model, as more files are modified and accuracy is further increased the labor involved and time required to complete the modeling drastically increases. This challenge could lead to other models like USLE-Forest being better suited for making forestland management decisions due to their ease of use and ability to provide an acceptable erosion estimate with fewer field measurements and less time required. There are additional research opportunities for comparing these models under different conditions globally.

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How Much Soil Disturbance Can Be Expected as a Result of Southern Pine Beetle Suppression Activities?

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Forest Service

Rocky Mountain Research Station

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Abstract

Land managers have long recognized the importance of maintaining soil productivity in the context of sustainable forest management. Soil disturbance that results in impaired hydrologic function and changes in certain soil properties (e.g., structure, organic matter) may be detrimental to soil productivity. Little is known about the degree of soil disturbance that results from salvage logging implemented in response to large-scale disturbances in the Southern Region of the Forest Service, U.S. Department of Agriculture. We assess the degree of soil disturbance following harvest operations after an outbreak of southern pine beetle on the Bienville and Homochitto National Forests in Mississippi. Post-implementation monitoring was carried out on 37 cutting units using methods consistent with the Forest Soils Disturbance Monitoring Protocol. Soil disturbance was detected on approximately 52 percent of the cutting units; the majority of soil disturbance was class 1 and 2. High levels of soil disturbance were avoided in part by effective communication between soil scientists, timber sale administrators, and equipment operators. High levels of detrimental soil disturbance were from excessive rutting when logging operations occurred during high soil moisture conditions, which are suboptimal for soil strength. These data provide a baseline for evaluating soil disturbance recovery in the Southern Region and indicate the magnitude of soil disturbance to be expected during salvage logging activities.

Keywords: Southern pine beetle, salvage logging, soil monitoring, soil disturbance, FSDMP

Cover: Top, Woody residues resulting from harvest operations (photo: R.M. Bergstrom, USDA Forest Service); center, Skid trail through a harvest unit (photo: R.M. Bergstrom, USDA Forest Service); bottom, Log landing during harvest operations to remove southern pine beetle-killed trees (photo: R.M. Bergstrom, USDA Forest Service).

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Contents

Introduction

Soil quality is both a driver and an indicator of forest ecosystem health. The health of forest ecosystems is largely dependent on hydrologic function and soil organic matter dynamics. Hydrologic function has long been recognized to be influenced by various soil properties such as bulk density, structure, and porosity (Amacher et al. 2007; Schoenholtz et al. 2000). Management-induced changes in soil properties have the potential to impact soil quality, and in turn, impair long-term soil productivity (Burger et al. 2010). The North American Long-Term Soil Productivity (LTSP) study began in 1989 in an effort to understand the connection between soil disturbance and forest productivity. This research was undertaken at least partially in response to the mandate in the National Forest Management Act of 1976 that forest management not permanently impair land productivity (Powers 2006). However, because of the diversity of sites, harvest methods, soil, and disturbance regime, additional data are needed to quantify the types of soil disturbance. Understanding the role of disturbance in maintaining or improving long-term soil productivity (and associated hydrologic function) remains a management goal within the Forest Service, U.S. Department of Agriculture (USDA FS 2010).

The awareness that forest management operations have the potential to impair long-term soil productivity has led private and public landholders to recognize the need for both maintaining soil productivity and developing soil monitoring guidelines and soil quality standards (Neary et al. 2010). Without standardized monitoring protocols, the potential exists for the evolution of soil monitoring methods that are self-suiting, but produce data not necessarily comparable with data gathered by other methods. Sharing questions about soil monitoring and coordinating across sites with common protocols allow for cross-site analysis and increase the capacity to understand key drivers of long-term soil productivity.

The Forest Soil Disturbance Monitoring Protocol (FSDMP) was developed to provide a common protocol for the monitoring of soil quality, using visual soil disturbance indicators that indicate potential decreases in soil productivity or hydrologic function (Page-Dumroese et al. 2009). It is also an integral part of the adaptive management process (Curran et al. 2005) and develops a common language for reporting soil disturbance (Curran et al. 2007). The FSDMP is used extensively by forest specialists conducting soil monitoring and assessment surveys before and after land management activities. The protocol provides a rapid tool for estimating the amount and types of disturbances caused by land management activities. Disturbances include compaction, rutting, displacement, severe burning, and loss of soil cover.

Moderate amounts of soil disturbance on the Coastal Plain of South Carolina have been associated with high site productivity 5 years after stand replacement (Eisenbies et al. 2005), and the authors provide evidence that trees on disturbed sites performed as well or better than trees on minimally disturbed sites. Furthermore, severely compacted sites on the DeSoto National Forest in Mississippi had significantly greater loblolly pine (*Pinus taeda* L.) volume as compared to sites with lower amounts of soil compaction after 5 years of growth (Scott et al. 2004). Stagg and Scott (2006) indicate that planted pine biomass was unaffected by soil compaction 5 and 10 years post-treatment. However, data from a long-term study site in a loblolly pine plantation in North Carolina showed that organic matter removal reduced species diversity more than soil compaction did 14 years post-treatment (Vierra and Blank 2010).

Although the relationship between soil compaction and productivity loss is not always evident, this kind of soil disturbance should not be dismissed as inconsequential. Severe and moderate soil compaction have been shown to hinder succession and reduce woody understory biomass (Stagg and Scott 2006). A study from 2010 on the Croatan National Forest in North Carolina found that trees which would normally be measured for site index curve development had significantly reduced height growth 14 years after organic matter removal and soil compaction (Eaton et al. 2010). However, the effects of soil compaction cannot be broadly applied to other soil types without considering their inherent physical properties. For example, seeding survival is more affected by soil compaction in fine-textured soils than in sandy soils (Miwa et al. 2004). Furthermore, soil compaction or soil erosion or both are often the consequence of excessive soil rutting. Generally, soil rutting is most evident on sites with high soil moisture content during logging operations. High soil moisture content makes soil more susceptible to puddling (smearing of surface pores), decreased resistance to mechanical stresses, soil mixing, and displacement. Vidrine et al. (1999) noted that the average rut depth was 13.0 inches (33 cm) in the most severely disturbed areas on a sandy loam soil in Louisiana.

Generally, soil recovery (or formation) in the Forest Service's Southern Region (Region 8: the 13 southeastern States, as well as Puerto Rico) occurs more quickly than in colder and drier regions of the United States following disturbance, due to the pace of weathering induced by the climate in the Southeast. Even in areas that exhibit high areal extent and severity of soil disturbance, soils are expected to naturally recover from disturbance impacts relatively quickly. Dickerson (1976) estimated that complete recovery of soil bulk density may occur in 8 to 12 years on the northern Mississippi Coastal Plain. Similarly, soil bulk density increases induced by compacting the soil were shown to have recovered by 5 percent between 1 and 5 years post-treatment (Scott et al. 2004). Even after severe compaction on southern LTSP sites with a fine sandy loam soil, recovery to predisturbance bulk density occurred in less than 5 years (Page-Dumroese et al. 2006). Deeply rutted soils in the South Carolina Coastal Plain had partial recovery within 2 years of disturbance due to the positive effects of clays, which have high shrink-swell properties (Miwa et al. 1999).

Monitoring soil quality is a major task associated with watershed management programs and restoration activities within the Forest Service. As such, it is important that soil quality responses to land management activities are measured and understood within the context of maintaining long-term soil health and site productivity. The majority of widely available data regarding the effects of land management activities on soil productivity or quality, using soil disturbance as a proxy, is concentrated in the Forest Service's Northern Region (Region 1: northern Idaho, Montana, North Dakota, and part of South Dakota) and Pacific Northwest Region (Region 6: Oregon and Washington). However, additional data are needed from other Forest Service regions to expand the applicability of the FSDMP. With the potential for forest disturbances related to forest health and wildfire, salvage logging operations may temporarily take priority over a national forest's normal plan of work related to vegetation management. Studies have not begun to link soil disturbance indicators to soil quality in the South, especially as related to salvage logging operations.

The objective of this study was to assess the degree of soil disturbance associated with salvage logging operations in two national forests in the South, in part to establish a baseline for expected impacts of this type of management activity, and to evaluate the long-term recovery of soil disturbance in these ecosystems.

Methods

Managers treated almost 6,300 acres (2,600 ha) on the Bienville and Homochitto National Forests in Mississippi (fig. 1) during the southern pine beetle (SPB; *Dendroctonus frontalis*) suppression effort in 2017. Approximately 5,200 acres (2,100 ha) of cut and leave were accomplished between March and November 2017; 1,100 acres (445 ha) of cut and remove were accomplished between March 2017 and January 2018. Approximately 400 to 500 trees/acre (990 to 1,200 trees/ha) were cut during this effort, depending on the age class of treated timber stands. Residual material from cut and leave operations included "jackstraw" trees with logging slash distributed across the cutting units. In addition, logging slash was distributed across cutting units in cut and remove operations. Both types of operations left standing dead timber in the center of the cutting units. Mechanical equipment and hand operations (chainsaw) were used to implement treatments in varying ground and weather conditions. The soil monitoring

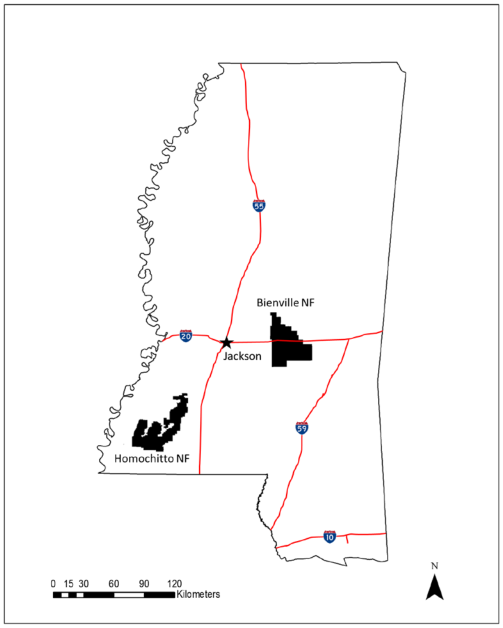

Figure 1—Location of the Bienville and Homochitto National Forests in Mississippi.

data in this paper include only harvest units that were completed using mechanical equipment.

The soil texture of the dominant soil map component for 62 percent of cutting units was either silty clay loam or silt loam. Thirty-two percent of dominant soil map units within cutting units were sandy loam. The remainder of cutting units were mapped as loamy soils. The majority of slopes ranged between 2 and 8 percent (Soil Survey Staff 2017).

During the 2017 SPB suppression efforts, operators used ground-based feller-buncher cutting machines, rubber-tired skidders, and knuckle boom loaders. The feller buncher was used to cut and group stems so that the skidder could move bundles of wood to the knuckle boom loader. Bundles were placed so that the skidder stayed on or close to the main skid trail in order to decrease the amount of soil disturbance throughout a cutting unit. In some instances slash was placed on the main skid trail to mitigate soil rutting or soil displacement. Soil moisture conditions, at the surface or at depth (or both), were assessed daily to safeguard against excessive ground disturbance. Soil interpretations derived from Web Soil Survey (Soil Survey Staff 2017) were used to establish a strategy for timing logging operations during wet ground conditions.

Thirty-seven cutting units were evaluated using the soil disturbance classification scheme described in the FSDMP (Page-Dumroese et al. 2009). Random transects were oriented within each cutting unit and monitoring points were established every 50 feet (15 m). Soil disturbance was assessed at each monitoring point for degree of compaction, rut depth, displacement, burn severity, and soil cover. After data on these attributes were collected, each monitoring point was assigned a soil disturbance class that also evaluated the point within the context of the landscape. Because there are few data on the relationship of disturbance class to the potential loss in long-term productivity (validation data), assignment of “detrimental soil disturbance” (DSD) based on visual observations was made after discussions with other Forest Service soil scientists. For these cutting units and soil types detrimental disturbance was noted when the soil disturbance class was class 3. Cutting units were evaluated 1 to 3 months after logging was completed.

Results and Discussion

Over 2,400 soil disturbance monitoring points were gathered in 37 harvest units, each with an average of 66 monitoring points. Monitored units had an average of 45 percent (areal extent) of land with class 0 soil disturbance (fig. 2). Approximately 52 percent of the cutting units had soil disturbance in some combination of soil disturbance classes 1, 2, and 3. Approximately 5 percent of the cutting units showed class 3 soil disturbance (fig. 2). The amount of undisturbed land in these cutting units was higher than reported by two studies in central and southern Alabama using similar harvest methods. These studies found that only 25 percent of land area within cutting units was left in an undisturbed condition (Carter and Grace 2012; McDonald et al. 1998). Our results suggest that current ground-based harvest methods (e.g., feller buncher) when used during a similar time of year can limit the amount of soil disturbance that may

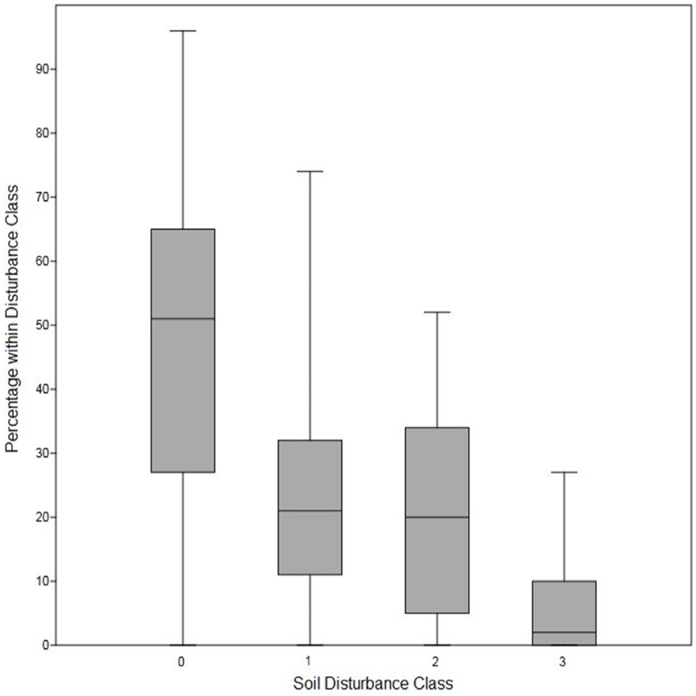

Figure 2—Overall soil disturbance in each Forest Soil Disturbance Monitoring Protocol class for 2017 southern pine beetle suppression treatment units (n = 37). Box plots show median soil disturbance class (horizontal line), middle 50 percent (box), and upper and lower 25 percent (bars).

impact site productivity and soil health. The lack of disturbance in surface organic matter and mineral soil also indicates that these sites are likely to be more resilient to future stressors (e.g., climate change, drought, and insect or disease outbreaks).

Management Implications

National forest units (forests and regions) in the western and northwestern United States have linked current soil quality standards and guidelines for long-term productivity on some soils (Page-Dumroese et al. 2000). The concepts of soil or site productivity, as they relate to sustainable forest management, can be linked to thresholds derived from FSDMP data. For instance, in the Northern Region (Region 1) adherence to soil quality guidelines means that DSD is limited to 15 percent of an activity area. Another example of DSD in Region 1 is the removal of 1 or more inches (2.5 cm) of any surface soil horizon from a continuous area greater than 100 ft^2 (9 m^2). It has been difficult to demonstrate the link between sustainable forest management and soil property changes observed using the FSDMP in the Southern Region (Region 8). In general, soils in the Southeast are more resilient and recover relatively quickly after management activities. However, findings from the LTSP installations across North America have pointed out that loss of the forest floor leads to declines in soil carbon content, which can also lead to reduced nutrient availability (Powers et al. 2005).

Although there are no common soil standards that are correlated to soil disturbance data across Region 8, there is utility in monitoring activity units using the FSDMP as it provides a consistent language and approach to assessing soil disturbance. This monitoring will also prove useful in assessing the degree of disturbance expected with different land management, restoration activities, and long-term productivity.

While acknowledging the difficulty of drawing generalizations between management-induced changes in soil properties and soil or site productivity in the context of sustainable forest management in the Southeast, we can present these data in the context of the Regional Soil Standards from Region 1. These standards state that no more than 15 percent of an activity area can exhibit DSD. As mentioned earlier, we assumed classification of a sample location as disturbance class 3 indicated that the disturbance was detrimental to long-term soil hydrologic function or site productivity. When using the Region 1 guidelines about the areal extent of DSD to compare to the SPB suppression effort, we found that DSD was exceeded in 4 of the 53 harvest units. One of these units had an unusually high proportion of log landings and skid trails for the size of the unit. Excessive rutting in the other three units contributed to exceeding the DSD threshold. Soil gilgai (small depressions holding water), microknolls, and microdepressions were observed on one of these units, making it difficult to determine a baseline for accurate rutting depth measurements. Generally, the severely rutted areas were observed in lower slopes and drainages. Logging under drier soil moisture conditions will reduce the likelihood of severe rutting. It will be beneficial to revisit the units with the highest disturbance and evaluate their rates of natural recovery or the need for restoration before the next harvest cycle. Soil property changes evaluated with the FSDMP that are most likely to affect soil or site productivity include the loss of porosity and an increase in bulk density associated with compaction, rutting, and organic matter removal. However, these changes in soil properties do not always correspond to losses in soil productivity.

Rutting is visually distinct and usually occurs on wet sites or sites with a seasonal high water table that coincides with harvesting (Aust et al. 1995; Burger et al. 1989). Rutting can also occur on uplands under certain soil conditions and after numerous passes. Rutting increases bulk density and reduces macropore space and saturated hydraulic conductivity. In the SPB suppression harvest units, the rutting hazard for most of the soils is listed as "severe" (fig. 3), meaning that ruts have the potential to form readily, based on soil characteristics (i.e., depth to water table, rock fragments, soil classification). However, the overall low percentage of class 3 (rut depth >4 inches [10 cm]) soil disturbance indicates that the degree of rutting observed in the 2017 SPB treatment units was lower than in the harvest operations described by Vidrine et al. (1999) and Aust et al. (1995). The difference between these older studies and our current monitoring effort could be attributed to a change in logging equipment. The use of rubber-tired skidders operating on larger and higher pressure tires results in less

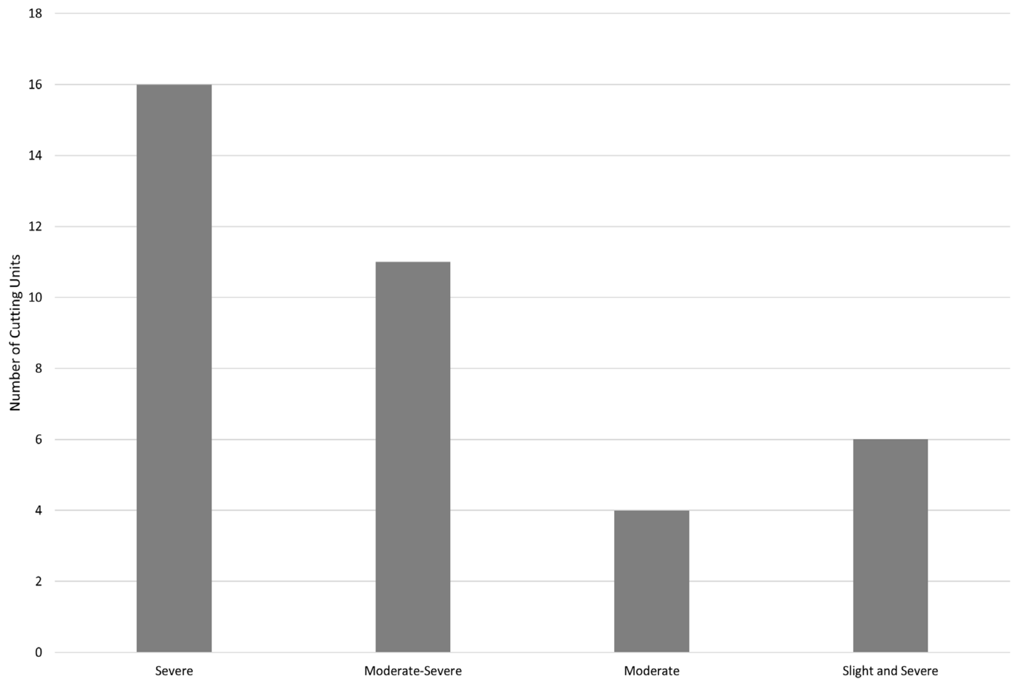

Figure 3—Soil rutting hazard rating for cutting units. Ratings indicate the hazard of surface rut formation through the operation of harvest equipment. A rating of "slight" indicates that the soil is susceptible to little or no rutting, "moderate" indicates that rutting is likely, and "severe" indicates that ruts form easily. Ratings are based on soil characteristics; interpretations were obtained from Web Soil Survey (Soil Survey Staff 2017).

change in soil bulk density and porosity and shallower ruts (Carter 2011). A broad-scale look at the soil disturbance monitoring data suggests that ground-disturbing activities from the 2017 efforts did not result in soil disturbance outside the range of what is currently known. This is a tribute to the local timber sale staff and operators, who were experienced and worked closely together throughout the logging season. Reduced soil disturbance is also linked to effective communications among loggers, timber sale administrators, and soil scientists.

There are no Region 8 Forest Plan or Regional Soil Standards to compare the efficacy of these operations with respect to soil disturbance. Nor was it possible to determine which soil property changes justified a particular soil disturbance class. These data do provide a baseline for estimating the degree of soil disturbance to anticipate when similar harvest unit operation methods are used under similar soil conditions. A missing component from the overall Region 8 soil quality strategy is to monitor the effectiveness of best management practices applied to salvage logging and disturbance classes to validate changes in particular soil properties (e.g., soil organic matter, respiration, porosity) that are key to sustainable forest management and site productivity. Monitoring efforts must move past considering soil disturbance data in a vacuum. Data gathered in Region 8 should be linked to ecosystem health indicators such as forest understory characteristics or soil microbial diversity. This information, combined with specific soil property data gathered using the FSDMP, will prove valuable for estimating the effects of land management activities on the soil resource and its recovery.

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Technical Support Document: -
Technical Update of the Social Cost of Carbon for Regulatory Impact Analysis - Under Executive Order 12866 -

Interagency Working Group on Social Cost of Greenhouse Gases, United States Government

With participation by

Council of Economic Advisers
Council on Environmental Quality
Department of Agriculture
Department of Commerce
Department of Energy
Department of the Interior
Department of Transportation
Department of the Treasury
Environmental Protection Agency
National Economic Council
Office of Management and Budget
Office of Science and Technology Policy

August 2016

See Appendix B for Details on Revisions since May 2013

Preface

The Interagency Working Group on the Social Cost of Greenhouse Gases (formerly the Interagency Working Group on the Social Cost of Carbon) has a longstanding commitment to ensure that the social cost of carbon estimates continue to reflect the best available science and methodologies. Given this commitment and public comments on issues of a deeply technical nature received by the Office of Management and Budget and federal agencies, the Interagency Working Group is seeking independent expert advice on technical opportunities to update the social cost of carbon estimates. The Interagency Working Group asked the National Academies of Sciences, Engineering, and Medicine in 2015 to review the latest research on modeling the economic aspects of climate change to inform future revisions to the social cost of carbon estimates presented in this technical support document. In January 2016, the Academies' Committee on the Social Cost of Carbon issued an interim report that recommended against a near-term update to the social cost of carbon estimates, but included recommendations for enhancing the presentation and discussion of uncertainty around the current estimates. This revision to the TSD responds to these recommendations in the presentation of the current estimates. It does not revisit the interagency group's 2010 methodological decisions or update the schedule of social cost of carbon estimates presented in the July 2015 revision. The Academies' final report (expected in early 2017) will provide longer term recommendations for a more comprehensive update.

Executive Summary

Executive Order 12866 requires agencies, to the extent permitted by law, "to assess both the costs and the benefits of the intended regulation and, recognizing that some costs and benefits are difficult to quantify, propose or adopt a regulation only upon a reasoned determination that the benefits of the intended regulation justify its costs." The purpose of the social cost of carbon ($SC\text{-}CO_2$)[1] estimates presented here is to allow agencies to incorporate the social benefits of reducing carbon dioxide (CO_2) emissions into cost-benefit analyses of regulatory actions. The $SC\text{-}CO_2$ is the monetized damages associated with an incremental increase in carbon emissions in a given year. It is intended to include (but is not limited to) changes in net agricultural productivity, human health, property damages from increased flood risk, and the value of ecosystem services due to climate change.

The interagency process that developed the original U.S. government $SC\text{-}CO_2$ estimates is described in the 2010 Technical Support Document on the Social Cost of Carbon (TSD) (Interagency Working Group on Social Cost of Carbon 2010). Through that process the Interagency Working Group (IWG) selected $SC\text{-}CO_2$ values for use in regulatory analyses. For each emissions year, four values are recommended. Three of these values are based on the average $SC\text{-}CO_2$ from three integrated assessment models (IAMs), at discount rates of 2.5, 3, and 5 percent. In addition, as discussed in the 2010 TSD, there is extensive evidence in the scientific and economic literature on the potential for lower-probability, but higher-impact outcomes from climate change, which would be particularly harmful to society and thus relevant to the public and policymakers. The fourth value is thus included to represent the marginal damages associated with these lower-probability, higher-impact outcomes. Accordingly, this fourth value is selected from further out in the tail of the distribution of $SC\text{-}CO_2$ estimates; specifically, the fourth value corresponds to the 95th percentile of the frequency distribution of $SC\text{-}CO_2$ estimates based on a 3 percent discount rate. Because the present value of economic damages associated with CO_2 emissions change over time, a separate set of estimates is presented for each emissions year through 2050, which is sufficient to cover the time frame addressed in most current regulatory impact analyses.

In May of 2013, the IWG provided an update of the $SC\text{-}CO_2$ estimates based on new versions of each IAM (DICE, PAGE, and FUND). The 2013 update did not revisit other IWG modeling decisions (e.g., the discount rate, reference case socioeconomic and emission scenarios, or equilibrium climate sensitivity). Improvements in the way damages are modeled were confined to those that had been incorporated into the latest versions of the models by the developers themselves in the peer-reviewed literature. The IWG subsequently provided additional minor technical revisions in November of 2013 and July of 2015, as described in Appendix B.

The purpose of this 2016 revision to the TSD is to enhance the presentation and discussion of quantified uncertainty around the current $SC\text{-}CO_2$ estimates, as a response to recommendations in the interim report by the National Academies of Sciences, Engineering, and Medicine. Included herein are an expanded

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[1] Throughout this Technical Support Document (TSD) we refer to the estimates as "$SC\text{-}CO_2$ estimates" rather than the more simplified "SCC" abbreviation used in previous versions of the TSD.

graphical presentation of the SC-CO_2 estimates highlighting a symmetric range of uncertainty around estimates for each discount rate, new sections that provide a unified discussion of the methodology used to incorporate sources of uncertainty, and a detailed explanation of the uncertain parameters in both the FUND and PAGE models.

The distributions of SC-CO_2 estimates reflect uncertainty in key model parameters chosen by the IWG such as the sensitivity of the climate to increases in carbon dioxide concentrations, as well as uncertainty in default parameters set by the original model developers. This TSD maintains the same approach to estimating the SC-CO_2 and selecting four values for each emissions year that was used in earlier versions of the TSD. Table ES-1 summarizes the SC-CO_2 estimates for the years 2010 through 2050. These estimates are identical to those reported in the previous version of the TSD, released in July 2015. As explained in previous TSDs, the central value is the average of SC-CO_2 estimates based on the 3 percent discount rate. For purposes of capturing uncertainty around the SC-CO_2 estimates in regulatory impact analysis, the IWG emphasizes the importance of considering all four SC-CO_2 values.

Table ES-1: Social Cost of CO_2, 2010 – 2050 (in 2007 dollars per metric ton of CO_2)

Year	5% Average	**3% Average**	2.5% Average	High Impact (95th Pct at 3%)
2010	10	**31**	50	86
2015	11	**36**	56	105
2020	12	**42**	62	123
2025	14	**46**	68	138
2030	16	**50**	73	152
2035	18	**55**	78	168
2040	21	**60**	84	183
2045	23	**64**	89	197
2050	26	**69**	95	212

While point estimates are important for providing analysts with a tractable approach for regulatory analysis, they do not fully quantify uncertainty associated with the SC-CO_2 estimates. Figure ES-1 presents the quantified sources of uncertainty in the form of frequency distributions for the SC-CO_2 estimates for emissions in 2020. To highlight the difference between the impact of the discount rate on the SC-CO_2 and other quantified sources of uncertainty, the bars below the frequency distributions provide a symmetric representation of quantified variability in the SC-CO_2 estimates for each discount rate. When an agency determines that it is appropriate to conduct additional quantitative uncertainty analysis, it should follow best practices for probabilistic analysis.[2] The full set of information that underlies the frequency distributions in Figure ES-1, which have previously been available upon request, are now available on Office of Management and Budget's (OMB) website for easy public access.

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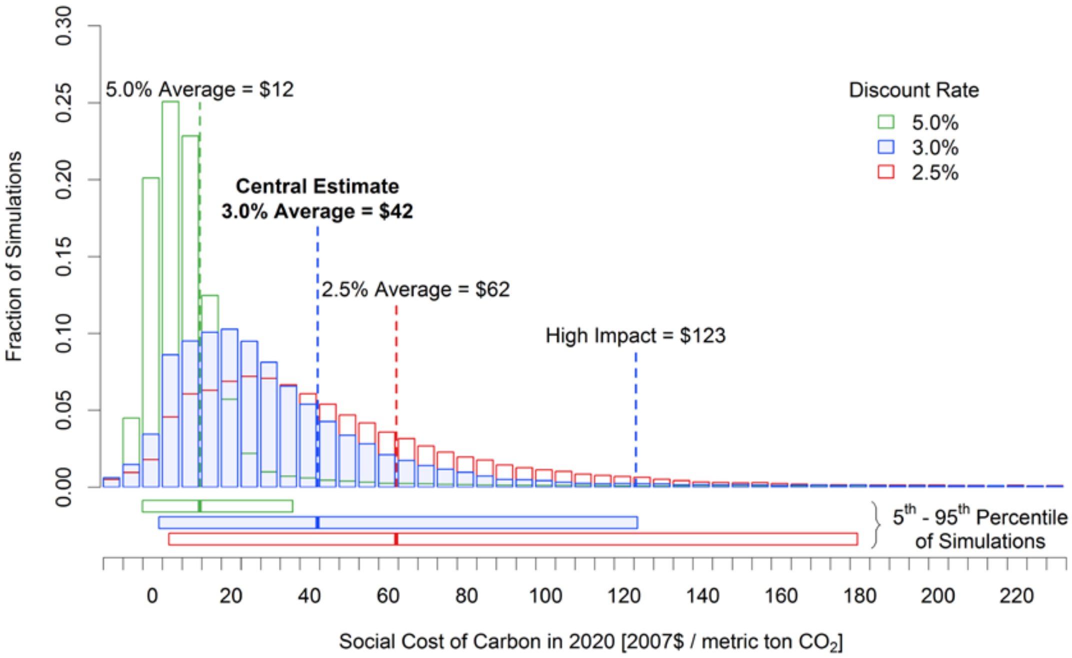
[2] See e.g. OMB Circular A-4, section on *Treatment of Uncertainty*. Available at: https://www.whitehouse.gov/omb/circulars_a004_a-4/#e.

Figure ES-1: Frequency Distribution of SC-CO₂ Estimates for 2020[3]



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[3] Although the distributions in Figure ES-1 are based on the full set of model results (150,000 estimates for each discount rate), for display purposes the horizontal axis is truncated with 0.1 to 0.6 percent of the estimates lying below the lowest bin displayed and 0.2 to 3.7 percent of the estimates lying above the highest bin displayed, depending on the discount rate.

I. Purpose

The purpose of this document is to present the current schedule of social cost of carbon ($SC\text{-}CO_2$) estimates, along with an enhanced presentation and discussion of quantified sources of uncertainty around the estimates to respond to recommendations in the interim report of the National Academies of Sciences, Engineering, and Medicine (National Academies 2016).[4] Because the last substantive update to the $SC\text{-}CO_2$ estimates occurred in May 2013, this document maintains much of the earlier technical discussion from the May 2013 TSD. The $SC\text{-}CO_2$ estimates themselves remain unchanged since the July 2015 revision.

E.O. 13563 commits the Administration to regulatory decision making "based on the best available science."[5] Additionally, the IWG recommended in 2010 that the $SC\text{-}CO_2$ estimates be revisited on a regular basis or as model updates that reflect the growing body of scientific and economic knowledge become available.[6] By early 2013, new versions of the three integrated assessment models (IAMs) used by the U.S. government to estimate the $SC\text{-}CO_2$ (DICE, FUND, and PAGE) were available and had been published in the peer-reviewed literature. While acknowledging the continued limitations of the approach taken by the IWG in 2010 (documented in the original 2010 TSD), the May 2013 TSD provided an update of the $SC\text{-}CO_2$ estimates based on the latest peer-reviewed version of the models, replacing model versions that were developed up to ten years earlier in a rapidly evolving field. It did not revisit other assumptions with regard to the discount rate, reference case socioeconomic and emission scenarios, or equilibrium climate sensitivity. Improvements in the way damages are modeled were confined to those that had been incorporated into the latest versions of the models by the developers themselves in the peer-reviewed literature. The agencies participating in the IWG continue to investigate potential improvements to the way in which economic damages associated with changes in CO_2 emissions are quantified.

Section II summarizes the major features of the IAMs used in this TSD that were updated in 2013 relative to the versions of the models used in the 2010 TSD. Section III presents the $SC\text{-}CO_2$ estimates for 2010 – 2050 based on these versions of the models. Section IV discusses the treatment of uncertainty in the analysis. Section V provides a discussion of other model limitations and research gaps.

II. Summary of Model Updates

This section briefly reviews the features of the three IAMs used in this TSD (DICE 2010, FUND 3.8, and PAGE 2009) that were updated by the model developers relative to the versions of the models used by the IWG in 2010 (DICE 2007, FUND 3.5, and PAGE 2002). The focus here is on describing those model updates that are relevant to estimating the social cost of carbon, as summarized in Table 1. For example, both the DICE and PAGE models now include an explicit representation of sea level rise damages. Other

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[4] In this document, we present all social cost estimates per metric ton of CO_2 emissions. Alternatively, one could report the social cost per metric ton of carbon emissions. The multiplier for translating between mass of CO_2 and the mass of carbon is 3.67 (the molecular weight of CO_2 divided by the molecular weight of carbon = 44/12 = 3.67).

[5] http://www.whitehouse.gov/sites/default/files/omb/inforeg/eo12866/eo13563_01182011.pdf

[6] See p. 1, 3, 4, 29, and 33 (Interagency Working Group on Social Cost of Carbon 2010).

revisions to PAGE include: updated adaptation assumptions, revisions to ensure damages are constrained by GDP, updated regional scaling of damages, and a revised treatment of potentially abrupt shifts in climate damages. The DICE model's simple carbon cycle has been updated to be more consistent with a more complex climate model. The FUND model includes updated damage functions for sea level rise impacts, the agricultural sector, and reduced space heating requirements, as well as changes to the transient response of temperature to the buildup of GHG concentrations and the inclusion of indirect effects of methane emissions. Changes made to parts of the models that are superseded by the IWG's modeling assumptions—regarding equilibrium climate sensitivity, discounting, and socioeconomic variables—are not discussed here but can be found in the references provided in each section below.

Table 1: Summary of Key Model Revisions Relevant to the IWG SC-CO_2 Estimates

IAM	Version used in 2010 IWG Analysis	Version Used since May 2013	Key changes relevant to IWG SC-CO_2
DICE	2007	2010	Updated calibration of the carbon cycle model and explicit representation of sea level rise (SLR) and associated damages.
FUND	3.5 (2009)	3.8 (2012)	Updated damage functions for space heating, SLR, agricultural impacts, changes to transient response of temperature to buildup of GHG concentrations, and inclusion of indirect climate effects of methane.
PAGE	2002	2009	Explicit representation of SLR damages, revisions to damage function to ensure damages do not exceed 100% of GDP, change in regional scaling of damages, revised treatment of potential abrupt damages, and updated adaptation assumptions.

A. DICE

DICE 2010 includes a number of changes over the previous 2007 version used in the 2010 TSD. The model changes that are relevant for the SC-CO_2 estimates developed by the IWG include: 1) updated parameter values for the carbon cycle model, 2) an explicit representation of sea level dynamics, and 3) a re-calibrated damage function that includes an explicit representation of economic damages from sea level rise. Changes were also made to other parts of the DICE model—including the equilibrium climate sensitivity parameter, the rate of change of total factor productivity, and the elasticity of the marginal utility of consumption—but these components of DICE are superseded by the IWG's assumptions and so will not be discussed here. More details on DICE2007 can be found in Nordhaus (2008) and on DICE2010 in Nordhaus (2010). The DICE2010 model and documentation is also available for download from the homepage of William Nordhaus.

Carbon Cycle Parameters

DICE uses a three-box model of carbon stocks and flows to represent the accumulation and transfer of carbon among the atmosphere, the shallow ocean and terrestrial biosphere, and the deep ocean. These parameters are “calibrated to match the carbon cycle in the Model for the Assessment of Greenhouse Gas Induced Climate Change (MAGICC)” (Nordhaus 2008, p. 44).[7] Carbon cycle transfer coefficient values in DICE2010 are based on re-calibration of the model to match the newer 2009 version of MAGICC (Nordhaus 2010, p. 2). For example, in DICE2010, in each decade 12 percent of the carbon in the atmosphere is transferred to the shallow ocean, 4.7 percent of the carbon in the shallow ocean is transferred to the atmosphere, 94.8 percent remains in the shallow ocean, and 0.5 percent is transferred to the deep ocean. For comparison, in DICE 2007, 18.9 percent of the carbon in the atmosphere is transferred to the shallow ocean each decade, 9.7 percent of the carbon in the shallow ocean is transferred to the atmosphere, 85.3 percent remains in the shallow ocean, and 5 percent is transferred to the deep ocean.

The implication of these changes for DICE2010 is in general a weakening of the ocean as a carbon sink and therefore a higher concentration of carbon in the atmosphere than in DICE2007 for a given path of emissions. All else equal, these changes will generally increase the level of warming and therefore the SC-CO_2 estimates in DICE2010 relative to those from DICE2007.

Sea Level Dynamics

A new feature of DICE2010 is an explicit representation of the dynamics of the global average sea level anomaly to be used in the updated damage function (discussed below). This section contains a brief description of the sea level rise (SLR) module; a more detailed description can be found on the model developer’s website.[8] The average global sea level anomaly is modeled as the sum of four terms that represent contributions from: 1) thermal expansion of the oceans, 2) melting of glaciers and small ice caps, 3) melting of the Greenland ice sheet, and 4) melting of the Antarctic ice sheet.

The parameters of the four components of the SLR module are calibrated to match consensus results from the IPCC’s Fourth Assessment Report (AR4).[9] The rise in sea level from thermal expansion in each time period (decade) is 2 percent of the difference between the sea level in the previous period and the long run equilibrium sea level, which is 0.5 meters per degree Celsius (°C) above the average global temperature in 1900. The rise in sea level from the melting of glaciers and small ice caps occurs at a rate of 0.008 meters per decade per °C above the average global temperature in 1900.

The contribution to sea level rise from melting of the Greenland ice sheet is more complex. The equilibrium contribution to SLR is 0 meters for temperature anomalies less than 1 ^{o}C and increases linearly from 0 meters to a maximum of 7.3 meters for temperature anomalies between 1 ^{o}C and 3.5 °C. The contribution to SLR in each period is proportional to the difference between the previous period’s sea

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[7] MAGICC is a simple climate model initially developed by the U.S. National Center for Atmospheric Research that has been used heavily by the Intergovernmental Panel on Climate Change (IPCC) to emulate projections from more sophisticated state of the art earth system simulation models (Randall et al. 2007).

[8] Documentation on the new sea level rise module of DICE is available on William Nordhaus’ website at: http://nordhaus.econ.yale.edu/documents/SLR_021910.pdf.

[9] For a review of post-IPCC AR4 research on sea level rise, see Nicholls et al. (2011) and NAS (2011).

level anomaly and the equilibrium sea level anomaly, where the constant of proportionality increases with the temperature anomaly in the current period.

The contribution to SLR from the melting of the Antarctic ice sheet is -0.001 meters per decade when the temperature anomaly is below 3 °C and increases linearly between 3 °C and 6 °C to a maximum rate of 0.025 meters per decade at a temperature anomaly of 6 °C.

Re-calibrated Damage Function

Economic damages from climate change in the DICE model are represented by a fractional loss of gross economic output in each period. A portion of the remaining economic output in each period (net of climate change damages) is consumed and the remainder is invested in the physical capital stock to support future economic production, so each period's climate damages will reduce consumption in that period and in all future periods due to the lost investment. The fraction of output in each period that is lost due to climate change impacts is represented as a sigmoid, or "S"-shaped, function of the temperature anomaly in the period.[10] The loss function in DICE2010 has been expanded by including a quadratic sub-function of SLR. In DICE2010 the temperature anomaly coefficients have been recalibrated to avoid double-counting damages from sea level rise that were implicitly included in these parameters in DICE2007.

The aggregate damages in DICE2010 are illustrated by Nordhaus (2010, p. 3), who notes that "…damages in the uncontrolled (baseline) [i.e., reference] case … in 2095 are $12 trillion, or 2.8 percent of global output, for a global temperature increase of 3.4 °C above 1900 levels." This compares to a loss of 3.2 percent of global output at 3.4 °C in DICE2007. However, in DICE2010, annual damages are lower in most of the early periods of the modeling horizon but higher in later periods than would be calculated using the DICE2007 damage function. Specifically, the percent difference between damages in the base run of DICE2010 and those that would be calculated using the DICE2007 damage function starts at +7 percent in 2005, decreases to a low of -14 percent in 2065, then continuously increases to +20 percent by 2300 (the end of the IWG analysis time horizon), and to +160 percent by the end of the model time horizon in 2595. The large increases in the far future years of the time horizon are due to the permanence associated with damages from sea level rise, along with the assumption that the sea level is projected to continue to rise long after the global average temperature begins to decrease. The changes to the loss function generally decrease the IWG SC-CO_2 estimates slightly given that relative increases in damages in later periods are discounted more heavily, all else equal.

B. FUND

FUND version 3.8 includes a number of changes over the previous version 3.5 (Narita et al. 2010) used in the 2010 TSD. Documentation supporting FUND and the model's source code for all versions of the model

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[10] The model and documentation, including formulas, are available on the author's webpage at http://www.econ.yale.edu/~nordhaus/homepage/RICEmodels.htm.

is available from the model authors.[11] Notable changes, due to their impact on the SC-CO_2 estimates, are adjustments to the space heating, agriculture, and sea level rise damage functions in addition to changes to the temperature response function and the inclusion of indirect effects from methane emissions.[12] Each of these is discussed in turn.

Space Heating

In FUND, the damages associated with the change in energy needs for space heating are based on the estimated impact due to one degree of warming. These baseline damages are scaled based on the forecasted temperature anomaly's deviation from the one degree benchmark and adjusted for changes in vulnerability due to economic and energy efficiency growth. In FUND 3.5, the function that scales the base year damages adjusted for vulnerability allows for the possibility that in some simulations the benefits associated with reduced heating needs may be an unbounded convex function of the temperature anomaly. In FUND 3.8, the form of the scaling has been modified to ensure that the function is everywhere concave and that there will exist an upper bound on the benefits a region may receive from reduced space heating needs. The new formulation approaches a value of two in the limit of large temperature anomalies, or in other words, assuming no decrease in vulnerability, the reduced expenditures on space heating at any level of warming will not exceed two times the reductions experienced at one degree of warming. Since the reduced need for space heating represents a benefit of climate change in the model, or a negative damage, this change will increase the estimated SC-CO_2. This update accounts for a significant portion of the difference in the expected SC-CO_2 estimates reported by the two versions of the model when run probabilistically.

Sea Level Rise and Land Loss

The FUND model explicitly includes damages associated with the inundation of dry land due to sea level rise. The amount of land lost within a region depends on the proportion of the coastline being protected by adequate sea walls and the amount of sea level rise. In FUND 3.5 the function defining the potential land lost in a given year due to sea level rise is linear in the rate of sea level rise for that year. This assumption implicitly assumes that all regions are well represented by a homogeneous coastline in length and a constant uniform slope moving inland. In FUND 3.8 the function defining the potential land lost has been changed to be a convex function of sea level rise, thereby assuming that the slope of the shore line

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[11] http://www.fund-model.org/. This report uses version 3.8 of the FUND model, which represents a modest update to the most recent version of the model to appear in the literature (version 3.7) (Anthoff and Tol, 2013a, 2013b). For the purpose of computing the SC-CO_2, the relevant changes (between 3.7 to 3.8) are associated with improving consistency with IPCC AR4 by adjusting the atmospheric lifetimes of CH_4 and N_2O and incorporating the indirect forcing effects of CH_4, along with making minor stability improvements in the sea wall construction algorithm.

[12] The other damage sectors (water resources, space cooling, land loss, migration, ecosystems, human health, and extreme weather) were not significantly updated.

increases moving inland. The effect of this change is to typically reduce the vulnerability of some regions to sea level rise based land loss, thereby lowering the expected SC-CO_2 estimate. [13]

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[13] For stability purposes this report also uses an update to the model which assumes that regional coastal protection measures will be built to protect the most valuable land first, such that the marginal benefits of coastal protection is decreasing in the level of protection following Fankhauser (1995).

Agriculture

In FUND, the damages associated with the agricultural sector are measured as proportional to the sector's value. The fraction is bounded from above by one and is made up of three additive components that represent the effects from carbon fertilization, the rate of temperature change, and the level of the temperature anomaly. In both FUND 3.5 and FUND 3.8, the fraction of the sector's value lost due to the level of the temperature anomaly is modeled as a quadratic function with an intercept of zero. In FUND 3.5, the coefficients of this loss function are modeled as the ratio of two random normal variables. This specification had the potential for unintended extreme behavior as draws from the parameter in the denominator approached zero or went negative. In FUND 3.8, the coefficients are drawn directly from truncated normal distributions so that they remain in the range $[0,\infty)$ and $(-\infty,0]$, respectively, ensuring the correct sign and eliminating the potential for divide-by-zero errors. The means for the new distributions are set equal to the ratio of the means from the normal distributions used in the previous version. In general the impact of this change has been to decrease the range of the distribution while spreading out the distributions' mass over the remaining range relative to the previous version. The net effect of this change on the SC-CO_2 estimates is difficult to predict.

Transient Temperature Response

The temperature response model translates changes in global levels of radiative forcing into the current expected temperature anomaly. In FUND, a given year's increase in the temperature anomaly is based on a mean reverting function where the mean equals the equilibrium temperature anomaly that would eventually be reached if that year's level of radiative forcing were sustained. The rate of mean reversion defines the rate at which the transient temperature approaches the equilibrium. In FUND 3.5, the rate of temperature response is defined as a decreasing linear function of equilibrium climate sensitivity to capture the fact that the progressive heat uptake of the deep ocean causes the rate to slow at higher values of the equilibrium climate sensitivity. In FUND 3.8, the rate of temperature response has been updated to a quadratic function of the equilibrium climate sensitivity. This change reduces the sensitivity of the rate of temperature response to the level of the equilibrium climate sensitivity, a relationship first noted by Hansen et al. (1985) based on the heat uptake of the deep ocean. Therefore in FUND 3.8, the temperature response will typically be faster than in the previous version. The overall effect of this change is likely to increase estimates of the SC-CO_2 as higher temperatures are reached during the timeframe analyzed and as the same damages experienced in the previous version of the model are now experienced earlier and therefore discounted less.

Methane

The IPCC AR4 notes a series of indirect effects of methane emissions, and has developed methods for proxying such effects when computing the global warming potential of methane (Forster et al. 2007). FUND 3.8 now includes the same methods for incorporating the indirect effects of methane emissions. Specifically, the average atmospheric lifetime of methane has been set to 12 years to account for the feedback of methane emissions on its own lifetime. The radiative forcing associated with atmospheric methane has also been increased by 40% to account for its net impact on ozone production and stratospheric water vapor. This update to the model is relevant for the SC-CO_2 because most of the damage functions are non-linear functions of the temperature anomaly, which represents the fact that as the climate system becomes more stressed an additional unit of warming will have a greater impact on damages. Accounting for the indirect effects of CH_4 emissions on temperature will therefore move the model further up the damage curves in the baseline, making a marginal change in emissions of CO_2 more impactful. All else equal, the effect of this increased radiative forcing will be to increase the estimated SC-CO_2 values, due to greater projected temperature anomaly.

C. PAGE

PAGE09 (Hope 2013) includes a number of changes from PAGE2002, the version used in the 2010 TSD. The changes that most directly affect the SC-CO_2 estimates include: explicitly modeling the impacts from sea level rise, revisions to the damage function to ensure damages are constrained by GDP, a change in the regional scaling of damages, a revised treatment for the probability of a discontinuity within the damage function, and revised assumptions on adaptation. The model also includes revisions to the carbon cycle feedback and the calculation of regional temperatures.[14] More details on PAGE09 can be found in Hope (2011a, 2011b, 2011c). A description of PAGE2002 can be found in Hope (2006).

Sea Level Rise

While PAGE2002 aggregates all damages into two categories—economic and non-economic impacts—PAGE09 adds a third explicit category: damages from sea level rise. In the previous version of the model, damages from sea level rise were subsumed by the other damage categories. In PAGE09 sea level damages increase less than linearly with sea level under the assumption that land, people, and GDP are more concentrated in low-lying shoreline areas. Damages from the economic and non-economic sectors were adjusted to account for the introduction of this new category.

Revised Damage Function to Account for Saturation

In PAGE09, small initial economic and non-economic benefits (negative damages) are modeled for small temperature increases, but all regions eventually experience economic damages from climate change, where damages are the sum of additively separable polynomial functions of temperature and sea level rise. Damages transition from this polynomial function to a logistic path once they exceed a certain proportion of remaining Gross Domestic Product (GDP) to ensure that damages do not exceed 100 percent of GDP. This differs from PAGE2002, which allowed Eastern Europe to potentially experience large benefits from temperature increases, and which also did not bound the possible damages that could be experienced.

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[14] Because several changes in the PAGE model are structural (e.g., the addition of sea level rise and treatment of discontinuity), it is not possible to assess the direct impact of each change on the SC-CO_2 in isolation as done for the other two models above.

Regional Scaling Factors

As in the previous version of PAGE, the PAGE09 model calculates the damages for the European Union (EU) and then, assumes that damages for other regions are proportional based on a given scaling factor. The scaling factors in PAGE09 are based on the length of each region's coastline relative to the EU (Hope 2011b). Because of the long coastline in the EU, other regions are, on average, less vulnerable than the EU for the same sea level and temperature increase, but all regions have a positive scaling factor. PAGE2002 based its scaling factors on four studies reported in the IPCC's third assessment report, and allowed for benefits from temperature increases in Eastern Europe, smaller impacts in developed countries, and higher damages in developing countries.

Probability of a Discontinuity

In PAGE2002, the damages associated with a "discontinuity" (nonlinear extreme event) were modeled as an expected value. Specifically, a stochastic probability of a discontinuity was multiplied by the damages associated with a discontinuity to obtain an expected value, and this was added to the economic and non-economic impacts. That is, additional damages from an extreme event, such as extreme melting of the Greenland ice sheet, were multiplied by the probability of the event occurring and added to the damage estimate. In PAGE09, the probability of discontinuity is treated as a discrete event for each year in the model. The damages for each model run are estimated either with or without a discontinuity occurring, rather than as an expected value. A large-scale discontinuity becomes possible when the temperature rises beyond some threshold value between 2 and 4°C. The probability that a discontinuity will occur beyond this threshold then increases by between 10 and 30 percent for every 1°C rise in temperature beyond the threshold. If a discontinuity occurs, the EU loses an additional 5 to 25 percent of its GDP (drawn from a triangular distribution with a mean of 15 percent) in addition to other damages, and other regions lose an amount determined by their regional scaling factor. The threshold value for a possible discontinuity is lower than in PAGE2002, while the rate at which the probability of a discontinuity increases with the temperature anomaly and the damages that result from a discontinuity are both higher than in PAGE2002. The model assumes that only one discontinuity can occur and that the impact is phased in over a period of time, but once it occurs, its effect is permanent.

Adaptation

As in PAGE2002, adaptation is available to help mitigate any climate change impacts that occur. In PAGE this adaptation is the same regardless of the temperature change or sea level rise and is therefore akin to what is more commonly considered a reduction in vulnerability. It is modeled by reducing the damages by some percentage. PAGE09 assumes a smaller decrease in vulnerability than the previous version of the model and assumes that it will take longer for this change in vulnerability to be realized. In the aggregated economic sector, at the time of full implementation, this adaptation will mitigate all damages up to a temperature increase of 1°C, and for temperature anomalies between 1°C and 2°C, it will reduce damages by 15-30 percent (depending on the region). However, it takes 20 years to fully implement this adaptation. In PAGE2002, adaptation was assumed to reduce economic sector damages up to 2°C by 50-90 percent after 20 years. Beyond 2°C, no adaptation is assumed to be available to mitigate the impacts of climate change. For the non-economic sector, in PAGE09 adaptation is available to reduce 15 percent of the damages due to a temperature increase between 0°C and 2°C and is assumed to take 40 years to fully implement, instead of 25 percent of the damages over 20 years assumed in PAGE2002. Similarly, adaptation is assumed to alleviate 25-50 percent of the damages from the first 0.20 to 0.25 meters of sea level rise but is assumed to be ineffective thereafter. Hope (2011c) estimates that the less optimistic assumptions regarding the ability to offset impacts of temperature and sea level rise via adaptation increase the SC-CO_2 by approximately 30 percent.

Other Noteworthy Changes

Two other changes in the model are worth noting. There is a change in the way the model accounts for decreased CO_2 absorption on land and in the ocean as temperature rises. PAGE09 introduces a linear feedback from global mean temperature to the percentage gain in the excess concentration of CO_2, capped at a maximum level. In PAGE2002, an additional amount was added to the CO_2 emissions each period to account for a decrease in ocean absorption and a loss of soil carbon. Also updated is the method by which the average global and annual temperature anomaly is downscaled to determine annual average regional temperature anomalies to be used in the regional damage functions. In PAGE2002, the scaling was determined solely based on regional difference in emissions of sulfate aerosols. In PAGE09, this regional temperature anomaly is further adjusted using an additive factor that is based on the average absolute latitude of a region relative to the area weighted average absolute latitude of the Earth's landmass, to capture relatively greater changes in temperature forecast to be experienced at higher latitudes.

III. SC-CO_2 Estimates

The three IAMs were run using the same methodology detailed in the 2010 TSD (Interagency Working Group on Social Cost of Carbon 2010). The approach, along with the inputs for the socioeconomic emissions scenarios, equilibrium climate sensitivity distribution, and discount rate remains the same. This includes the five reference scenarios based on the EMF-22 modeling exercise, the Roe and Baker equilibrium climate sensitivity distribution calibrated to the IPCC AR4, and three constant discount rates of 2.5, 3, and 5 percent.

As was previously the case, use of three models, three discount rates, and five scenarios produces 45 separate frequency distributions of SC-CO_2 estimates in a given year. The approach laid out in the 2010 TSD applied equal weight to each model and socioeconomic scenario in order to reduce the dimensionality down to three separate distributions, one for each of the three discount rates. The IWG selected four values from these distributions for use in regulatory analysis. Three values are based on the average SC-CO_2 across models and socioeconomic and emissions scenarios at the 2.5, 3, and 5 percent discount rates, respectively. The fourth value is included to provide information on the marginal damages associated with lower-probability, higher-impact outcomes that would be particularly harmful to society. As discussed in the 2010 TSD, there is extensive evidence in the scientific and economic literature of the potential for lower-probability, higher-impact outcomes from climate change, which would be particularly harmful to society and thus relevant to the public and policymakers. This points to the relevance of values above the mean in right skewed distributions. Accordingly, this fourth value is selected from further out in the tails of the frequency distribution of SC-CO_2 estimates, and, in particular, is set to the 95th percentile of the frequency distribution of SC-CO_2 estimates based on a 3 percent discount rate. (A detailed set of percentiles by model and scenario combination and additional summary statistics for the 2020 values is available in Appendix A.) As noted in the 2010 TSD, "the 3 percent discount rate is the central value, and so the central value that emerges is the average SC-CO_2 across models at the 3 percent discount rate" (Interagency Working Group on Social Cost of Carbon 2010, p. 25). However, for purposes of capturing the uncertainties involved in regulatory impact analysis, the IWG emphasizes the importance and value of including all four SC-CO_2 values.

Table 2 shows the four selected SC-CO_2 estimates in five year increments from 2010 to 2050. Values for 2010, 2020, 2030, 2040, and 2050 are calculated by first combining all outputs (10,000 estimates per model run) from all scenarios and models for a given discount rate. Values for the years in between are calculated using linear interpolation. The full set of revised annual SC-CO_2 estimates between 2010 and 2050 is reported in the Appendix and the full set of model results are available on the OMB website.[15]

Table 2: Social Cost of CO_2, 2010 – 2050 (in 2007 dollars per metric ton of CO_2)

Year	5% Average	**3% Average**	2.5% Average	High Impact (95th Pct at 3%)
2010	10	**31**	50	86
2015	11	**36**	56	105
2020	12	**42**	62	123
2025	14	**46**	68	138
2030	16	**50**	73	152
2035	18	**55**	78	168
2040	21	**60**	84	183
2045	23	**64**	89	197
2050	26	**69**	95	212

As was the case in the 2010 TSD, the SC-CO_2 increases over time because future emissions are expected to produce larger incremental damages as physical and economic systems become more stressed in response to greater climatic change, and because GDP is growing over time and many damage categories are modeled as proportional to gross GDP. The approach taken by the IWG is to compute the cost of a marginal ton emitted in the future by running the models for a set of perturbation years out to 2050. Table 3 illustrates how the growth rate for these four SC-CO_2 estimates varies over time.

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[15] https://www.whitehouse.gov/omb/oira/social-cost-of-carbon.

Table 3: Average Annual Growth Rates of SC-CO_2 Estimates between 2010 and 2050

Average Annual Growth Rate (%)	5.0% Avg	3.0% Avg	2.5% Avg	3.0% 95th
2010-2020	1.2%	3.2%	2.4%	4.4%
2020-2030	3.4%	2.1%	1.7%	2.3%
2030-2040	3.0%	1.9%	1.5%	2.0%
2040-2050	2.6%	1.6%	1.3%	1.6%

The future monetized value of emission reductions in each year (the SC-CO_2 in year *t* multiplied by the change in emissions in year *t*) must be discounted to the present to determine its total net present value for use in regulatory analysis. As previously discussed in the 2010 TSD, damages from future emissions should be discounted at the same rate as that used to calculate the SC-CO_2 estimates themselves to ensure internal consistency—i.e., future damages from climate change, whether they result from emissions today or emissions in a later year, should be discounted to the base year of the analysis using the same rate.

Current guidance contained in OMB Circular A-4 indicates that analysis of economically significant proposed and final regulations from the domestic perspective is required, while analysis from the international perspective is optional. However, the IWG (including OMB) determined that a modified approach is more appropriate in this case because the climate change problem is highly unusual in a number of respects. First, it involves a global externality: emissions of most greenhouse gases contribute to damages around the world even when they are emitted in the United States—and conversely, greenhouse gases emitted elsewhere contribute to damages in the United States. Consequently, to address the global nature of the problem, the SC-CO_2 must incorporate the full (global) damages caused by GHG emissions. Second, climate change presents a problem that the United States alone cannot solve. Other countries will also need to take action to reduce emissions if significant changes in the global climate are to be avoided. Emphasizing the need for a global solution to a global problem, the United States has been actively involved in seeking international agreements to reduce emissions. For example, the United States joined over 170 other nations and signed the Paris Agreement on April 22, 2016, signaling worldwide commitment to reduce GHG emissions. The United States has been active in encouraging other nations, including emerging major economies, to take significant steps to reduce emissions. Using a global estimate of damages in U.S. regulatory analyses sends a strong signal to other nations that they too should base their emissions reductions strategies on a global perspective, thus supporting a cooperative and mutually beneficial approach to achieving needed reduction. Thirteen prominent academics noted that these "are compelling reasons to focus on a global [SC-CO_2]" in a recent article on the SC-CO_2 (Pizer et al. 2014). In addition, adverse impacts on other countries can have spillover effects on the United States, particularly in the areas of national security, international trade, public health, and humanitarian concerns. When these considerations are taken as a whole, the IWG concluded that a global measure of the benefits from reducing U.S. emissions is appropriate. For additional discussion, see the 2010 TSD.

IV. Treatment of Uncertainty

Uncertainty about the value of the SC-CO_2 is in part inherent, as with any analysis that looks into the future, but it is also driven by current data gaps associated with the complex physical, economic, and behavioral processes that link GHG emissions to human health and well-being. Some sources of uncertainty pertain to aspects of the natural world, such as quantifying the physical effects of greenhouse gas emissions on Earth systems. Other sources of uncertainty are associated with current and future human behavior and well-being, such as population and economic growth, GHG emissions, the translation of Earth system changes to economic damages, and the role of adaptation. It is important to note that even in the presence of uncertainty, scientific and economic analysis can provide valuable information to the public and decision makers, though the uncertainty should be acknowledged and when possible taken into account in the analysis. This section summarizes the sources of uncertainty that the IWG was able to consider in a quantitative manner in estimating the SC-CO_2. Further discussion on sources of uncertainty that are active areas of research and have not yet been fully quantified in the SC-CO_2 estimates is provided in Section V and in the 2010 TSD.

In developing the SC-CO_2 estimates, the IWG considered various sources of uncertainty through a combination of a multi-model ensemble, probabilistic analysis, and scenario analysis. For example, the three IAMs used collectively span a wide range of Earth system and economic outcomes to help reflect the uncertainty in the literature and in the underlying dynamics being modeled. The use of an ensemble of three different models is also intended to, at least partially, address the fact that no single model includes all of the quantified economic damages. It also helps to reflect structural uncertainty across the models, which is uncertainty in the underlying relationships between GHG emissions, Earth systems, and economic damages that are included in the models. Bearing in mind the different limitations of each model (discussed in the 2010 TSD) and lacking an objective basis upon which to differentially weight the models, the three IAMs are given equal weight in the analysis.

The IWG used Monte Carlo techniques to run the IAMs a large number of times. In each simulation the uncertain parameters are represented by random draws from their defined probability distributions. In all three models the equilibrium climate sensitivity is treated probabilistically based on the probability distribution described in the 2010 TSD. The equilibrium climate sensitivity is a key parameter in this analysis because it helps define the strength of the climate response to increasing GHG concentrations in the atmosphere. In addition, the FUND and PAGE models define many of their parameters with probability distributions instead of point estimates. For these two models, the model developers' default probability distributions are maintained for all parameters other than those superseded by the IWG's harmonized inputs (i.e., equilibrium climate sensitivity, socioeconomic and emissions scenarios, and discount rates). More information on the uncertain parameters in PAGE and FUND is presented in Appendix C.

For the socioeconomic and emissions scenarios, uncertainty is included in the analysis by considering a range of scenarios, which are described in detail in the 2010 SC-CO_2 TSD. As noted in the 2010 TSD, while the IWG considered formally assigning probability weights to the different socioeconomic scenarios selected, it came to the conclusion that this could not be accomplished in an analytically rigorous way given the dearth of information on the likelihood of a full range of future socioeconomic pathways. Thus, the IWG determined that, because no basis for assigning differential weights was available, the most transparent way to present a range of uncertainty was simply to weight each of the five scenarios equally for the consolidated estimates. To provide additional information as to how the results vary with the scenarios, summarized results for each scenario are presented separately in Appendix A. The results of each model run are available on the OMB website.

Finally, based on the review of the literature, the IWG chose discount rates that reflect reasonable judgements under both prescriptive and descriptive approaches to intergenerational discounting. As discussed in the 2010 TSD, in light of disagreement in the literature on the appropriate discount rate to use in this context and uncertainty about how rates may change over time, the IWG selected three certainty-equivalent constant discount rates to span a plausible range: 2.5, 3, and 5 percent per year. However, unlike the approach taken for consolidating results across models and socioeconomic and emissions scenarios, the SC-CO_2 estimates are not pooled across different discount rates because the range of discount rates reflects both uncertainty and, at least in part, different policy or value judgements.

The outcome of accounting for various sources of uncertainty using the approaches described above is a frequency distribution of the SC-CO_2 estimates for emissions occurring in a given year for each of the three discount rates. These frequency distributions reflect the uncertainty around the input parameters for which probability distributions were defined, as well as from the multi-model ensemble and socioeconomic and emissions scenarios where probabilities were implied by the equal weighting assumption. It is important to note that the set of SC-CO_2 estimates obtained from this analysis does not yield a probability distribution that fully characterizes uncertainty about the SC-CO_2 due to impact categories omitted from the models and sources of uncertainty that have not been fully characterized due to data limitations.

Figure 1 presents the frequency distribution of the SC-CO_2 estimates for emissions in 2020 for each of the three discount rates. Each of these distributions represents 150,000 estimates based on 10,000 simulations for each combination of the three models and five socioeconomic and emissions scenarios.[16] In general, the distributions are skewed to the right and have long right tails, which tend to be even longer for lower discount rates. To highlight the difference between the impact of the discount rate on the SC-CO_2 and other quantified sources of uncertainty, the bars below the frequency distributions provide a symmetric representation of quantified variability in the SC-CO_2 estimates conditioned on each discount rate. The full set of SC-CO_2 results through 2050 is available on OMB's website. This may be useful to analysts in situations that warrant additional quantitative uncertainty analysis (e.g., as recommended by OMB for rules that exceed $1 billion in annual benefits or costs). See OMB Circular A-4 for guidance and discussion of best practices in conducting uncertainty analysis in RIAs.

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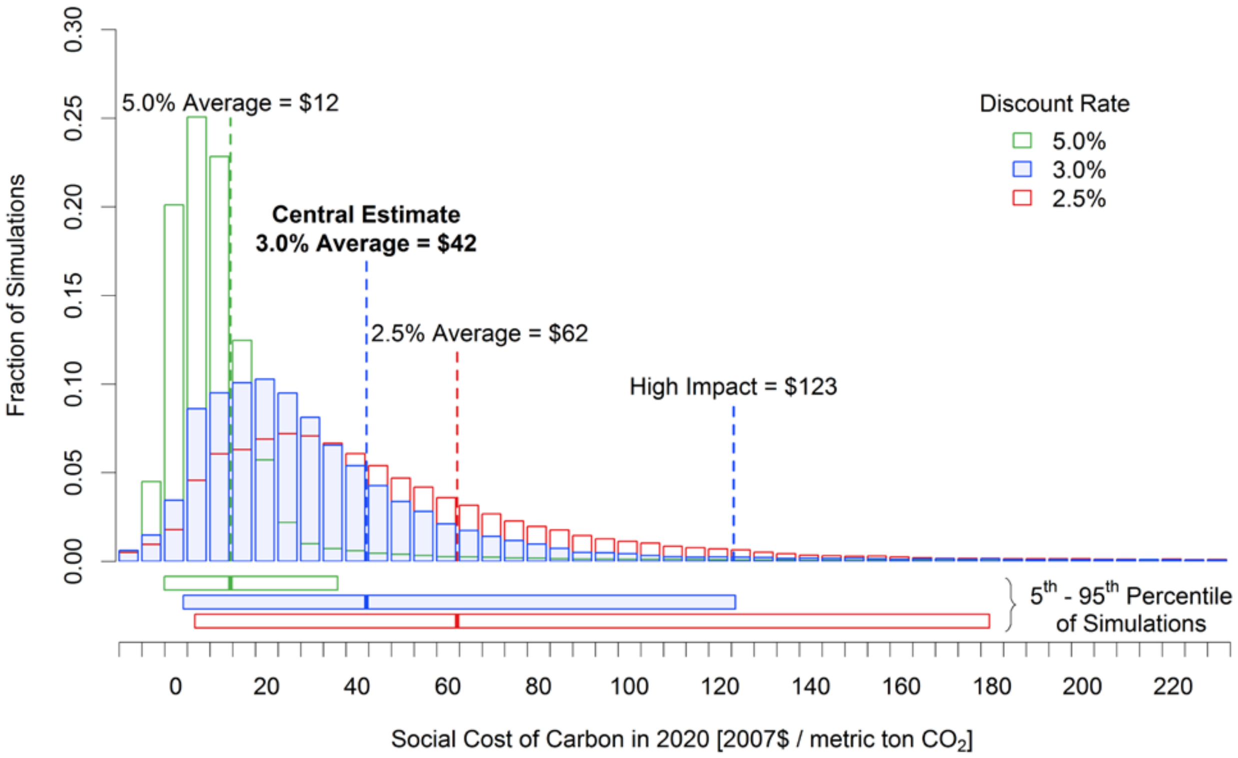
[16] Although the distributions in Figure 1 are based on the full set of model results (150,000 estimates for each discount rate), for display purposes the horizontal axis is truncated with 0.1 to 0.6 percent of the estimates lying below the lowest bin displayed and 0.2 to 3.7 percent of the estimates lying above the highest bin displayed, depending on the discount rate.

Figure 1: Frequency Distribution of SC-CO_2 Estimates for 2020 (in 2007$ per metric ton CO_2)



As previously described, the SC-CO_2 estimates produced by the IWG are based on a rigorous approach to accounting for quantifiable uncertainty using multiple analytical techniques. In addition, the scientific and economics literature has further explored known sources of uncertainty related to estimates of the SC-CO_2. For example, researchers have published papers that explore the sensitivity of IAMs and the resulting SC-CO_2 estimates to different assumptions embedded in the models (see, e.g., Hope (2013), Anthoff and Tol (2013a), and Nordhaus (2014)). However, there remain additional sources of uncertainty that have not been fully characterized and explored due to remaining data limitations. Additional research is needed in order to expand the quantification of various sources of uncertainty in estimates of the SC-CO_2 (e.g., developing explicit probability distributions for more inputs pertaining to climate impacts and their valuation). The IWG is actively following advances in the scientific and economic literature that could provide guidance on, or methodologies for, a more robust incorporation of uncertainty.

V. Other Model Limitations and Research Gaps

The 2010 SC-CO_2 TSD discusses a number of important limitations for which additional research is needed. In particular, the document highlights the need to improve the quantification of both non-catastrophic and catastrophic damages, the treatment of adaptation and technological change, and the way in which inter-regional and inter-sectoral linkages are modeled. While the more recent versions of the models discussed above offer some improvements in these areas, further research is still needed. Currently, IAMs do not include all of the important physical, ecological, and economic impacts of climate change recognized in the climate change literature due to a lack of precise information on the nature of damages and because the science incorporated into these models understandably lags behind the most recent research.[17] These individual limitations do not all work in the same direction in terms of their influence on the SC-CO_2 estimates; however, it is the IWG's judgment that, taken together, these limitations suggest that the SC-CO_2 estimates are likely conservative. In particular, the IPCC Fourth Assessment Report (Meehl et al. 2007), which was the most current IPCC assessment available at the time of the IWG's 2009-2010 review, concluded that SC-CO_2 estimates "very likely…underestimate the damage costs" due to omitted impacts. Since then, the peer-reviewed literature has continued to support this conclusion, as noted in the IPCC Fifth Assessment report (Oppenheimer et al. 2014).

Another area of active research relates to intergenerational discounting, including the application of discount rates to regulations in which some costs and benefits accrue intra-generationally while others accrue inter-generationally. Some experts have argued that a declining discount rate would be appropriate to analyze impacts that occur far into the future (Arrow et al. 2013). However, additional research and analysis is still needed to develop a methodology for implementing a declining discount rate and to understand the implications of applying these theoretical lessons in practice.

The 2010 TSD also discusses the need to more carefully assess the implications of risk aversion for SC-CO_2 estimation as well as the substitution possibilities between climate and non-climate goods at higher temperature increases, both of which have implications for the discount rate used. EPA, DOE, and other agencies continue to engage in research on modeling and valuation of climate impacts that can potentially improve SC-CO_2 estimation in the future. See the 2010 SC-CO_2 TSD for the full discussion.

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[17] See, for example, Howard (2014) and EPRI (2014) for recent discussions.

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Appendix A

Table A1: Annual SC-CO_2 Values: 2010-2050 (2007$/metric ton CO_2)

Year	5% Average	**3% Average**	2.5% Average	High Impact (95th Pct at 3%)
2010	10	**31**	50	86
2011	11	**32**	51	90
2012	11	**33**	53	93
2013	11	**34**	54	97
2014	11	**35**	55	101
2015	11	**36**	56	105
2016	11	**38**	57	108
2017	11	**39**	59	112
2018	12	**40**	60	116
2019	12	**41**	61	120
2020	12	**42**	62	123
2021	12	**42**	63	126
2022	13	**43**	64	129
2023	13	**44**	65	132
2024	13	**45**	66	135
2025	14	**46**	68	138
2026	14	**47**	69	141
2027	15	**48**	70	143
2028	15	**49**	71	146
2029	15	**49**	72	149
2030	16	**50**	73	152
2031	16	**51**	74	155
2032	17	**52**	75	158
2033	17	**53**	76	161
2034	18	**54**	77	164
2035	18	**55**	78	168
2036	19	**56**	79	171
2037	19	**57**	81	174
2038	20	**58**	82	177
2039	20	**59**	83	180
2040	21	**60**	84	183
2041	21	**61**	85	186
2042	22	**61**	86	189
2043	22	**62**	87	192
2044	23	**63**	88	194
2045	23	**64**	89	197
2046	24	**65**	90	200
2047	24	**66**	92	203
2048	25	**67**	93	206
2049	25	**68**	94	209
2050	26	**69**	95	212

Table A2: 2020 Global SC-CO_2 Estimates at 2.5 Percent Discount Rate (2007$/metric ton CO_2)

Percentile	1st	5th	10th	25th	50th	Avg	75th	90th	95th	99th
Scenario[18]	PAGE									
IMAGE	6	10	15	26	55	123	133	313	493	949
MERGE Optimistic	4	6	8	15	32	75	79	188	304	621
MESSAGE	4	7	10	19	41	104	103	266	463	879
MiniCAM Base	5	8	12	21	45	102	108	255	412	835
5th Scenario	2	4	6	11	24	81	66	192	371	915

Scenario	DICE									
IMAGE	25	31	37	47	64	72	92	123	139	161
MERGE Optimistic	14	18	20	26	36	40	50	65	74	85
MESSAGE	20	24	28	37	51	58	71	95	109	221
MiniCAM Base	20	25	29	38	53	61	76	102	117	135
5th Scenario	17	22	25	33	45	52	65	91	106	126

Scenario	FUND									
IMAGE	-14	-2	4	15	31	39	55	86	107	157
MERGE Optimistic	-6	1	6	14	27	35	46	70	87	141
MESSAGE	-16	-5	1	11	24	31	43	67	83	126
MiniCAM Base	-7	2	7	16	32	39	55	83	103	158
5th Scenario	-29	-13	-6	4	16	21	32	53	69	103

Table A3: 2020 Global SC-CO_2 Estimates at 3 Percent Discount Rate (2007$/metric ton CO_2)

Percentile	1st	5th	10th	25th	50th	Avg	75th	90th	95th	99th
Scenario	PAGE									
IMAGE	4	7	9	17	36	87	91	228	369	696
MERGE Optimistic	2	4	6	10	22	54	55	136	222	461
MESSAGE	3	5	7	13	28	72	71	188	316	614
MiniCAM Base	3	5	7	13	29	70	72	177	288	597
5th Scenario	1	3	4	7	16	55	46	130	252	632

Scenario	DICE									
IMAGE	16	21	24	32	43	48	60	79	90	102
MERGE Optimistic	10	13	15	19	25	28	35	44	50	58
MESSAGE	14	18	20	26	35	40	49	64	73	83
MiniCAM Base	13	17	20	26	35	39	49	65	73	85
5th Scenario	12	15	17	22	30	34	43	58	67	79

Scenario	FUND									
IMAGE	-13	-4	0	8	18	23	33	51	65	99
MERGE Optimistic	-7	-1	2	8	17	21	29	45	57	95
MESSAGE	-14	-6	-2	5	14	18	26	41	52	82
MiniCAM Base	-7	-1	3	9	19	23	33	50	63	101
5th Scenario	-22	-11	-6	1	8	11	18	31	40	62

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[18] See 2010 TSD for a description of these scenarios.

Table A4: 2020 Global SC-CO_2 Estimates at 5 Percent Discount Rate (2007$/metric ton CO_2)

Percentile	1st	5th	10th	25th	50th	Avg	75th	90th	95th	99th
Scenario	PAGE									
IMAGE	1	2	2	4	10	27	26	68	118	234
MERGE Optimistic	1	1	2	3	6	17	17	43	72	146
MESSAGE	1	1	2	4	8	23	22	58	102	207
MiniCAM Base	1	1	2	3	8	20	20	52	90	182
5th Scenario	0	1	1	2	5	17	14	39	75	199

Scenario	DICE									
IMAGE	6	8	9	11	14	15	18	22	25	27
MERGE Optimistic	4	5	6	7	9	10	12	15	16	18
MESSAGE	6	7	8	10	12	13	16	20	22	25
MiniCAM Base	5	6	7	8	11	12	14	18	20	22
5th Scenario	5	6	6	8	10	11	14	17	19	21

Scenario	FUND									
IMAGE	-9	-5	-4	-1	2	3	6	10	14	24
MERGE Optimistic	-6	-4	-2	0	3	4	6	11	15	26
MESSAGE	-10	-6	-4	-1	1	2	5	9	12	21
MiniCAM Base	-7	-4	-2	0	3	4	6	11	14	25
5th Scenario	-11	-7	-5	-3	0	0	3	5	7	13

Table A5: Additional Summary Statistics of 2020 Global SC-CO$_2$ Estimates

Discount rate:	5.0%				3.0%				2.5%			
Statistic:	Mean	Variance	Skewness	Kurtosis	Mean	Variance	Skewness	Kurtosis	Mean	Variance	Skewness	Kurtosis
DICE	12	26	2	15	38	409	3	24	57	1097	3	30
PAGE	21	1481	5	32	68	13712	4	22	97	26878	4	23
FUND	3	41	5	179	19	1452	-42	8727	33	6154	-73	14931

Appendix B

The November 2013 revision of this TSD is based on two corrections to the runs based on the FUND model. First, the potential dry land loss in the algorithm that estimates regional coastal protections was misspecified in the model's computer code. This correction is covered in an erratum to Anthoff and Tol (2013a) published in the same journal (*Climatic Change*) in October 2013 (Anthoff and Tol (2013b)). Second, the equilibrium climate sensitivity distribution was inadvertently specified as a truncated Gamma distribution (the default in FUND) as opposed to the truncated Roe and Baker distribution as was intended. The truncated Gamma distribution used in the FUND runs had approximately the same mean and upper truncation point, but lower variance and faster decay of the upper tail, as compared to the intended specification based on the Roe and Baker distribution. The difference between the original estimates reported in the May 2013 version of this TSD and this revision are generally one dollar or less.

The July 2015 revision of this TSD is based on two corrections. First, the DICE model had been run up to 2300 rather than through 2300, as was intended, thereby leaving out the marginal damages in the last year of the time horizon. Second, due to an indexing error, the results from the PAGE model were in 2008 U.S. dollars rather than 2007 U.S. dollars, as was intended. In the current revision, all models have been run through 2300, and all estimates are in 2007 U.S. dollars. On average the revised SC-CO_2 estimates are one dollar less than the mean SC-CO_2 estimates reported in the November 2013 version of this TSD. The difference between the 95th percentile estimates with a 3% discount rate is slightly larger, as those estimates are heavily influenced by results from the PAGE model.

The July 2016 revision provides additional discussion of uncertainty in response to recommendations from the National Academy of Sciences, Engineering, and Medicine. It does not revisit the IWG's 2010 methodological decisions or update the schedule of SC-CO_2 estimates presented in the July 2015 revision. The IWG is currently seeking external expert advice from the National Academies on the technical merits and challenges of potential approaches to future updates of the SC-CO_2 estimates presented in this TSD. To date, the Academies' committee has issued an interim report that recommended against a near-term update to the SC-CO_2 estimates, but included recommendations for enhancing the presentation and discussion of uncertainty around the current estimates. This revision includes additional information that the IWG determined was appropriate to respond to these recommendations. Specifically, the executive summary presents more information about the range of quantified uncertainty in the SC-CO_2 estimates (including a graphical representation of symmetric high and low values from the frequency distribution of SC-CO_2 estimates conditional on each discount rate), and a new section has also been added that provides a unified discussion of the various sources of uncertainty and how they were handled in estimating the SC-CO_2. Efforts to make the sources of uncertainty clear have also been enhanced with the addition of a new appendix that describes in more detail the uncertain parameters in both the FUND and PAGE models (Appendix C). Furthermore, the full set of SC-CO_2 modeling results, which have previously been available upon request, are now provided on the OMB website for easy access. The Academies' final report (expected in early 2017) will provide longer term recommendations for a more comprehensive update. For more information on the status of the Academies' process, see: http://sites.nationalacademies.org/DBASSE/BECS/CurrentProjects/DBASSE_167526.

Appendix C

This appendix provides a general overview of the parameters that are treated probabilistically in each of the three integrated assessment models the IWG used to estimate the SC-CO_2. In the DICE model the only uncertain parameter considered was the equilibrium climate sensitivity as defined by the probability distribution harmonized across the three models. By default, all of the other parameters in the model are defined by point estimates and these definitions were maintained by the IWG. In the FUND and PAGE models many of the parameters, beyond the equilibrium climate sensitivity, are defined by probability distributions in the default versions of the models. The IWG maintained these default assumptions and allowed these parameters to vary in the Monte Carlo simulations conducted with the FUND and PAGE models.

Default Uncertainty Assumptions in FUND

In the version of the FUND model used by the IWG (version 3.8.1) over 90 of the over 150 parameters in the model are defined by probability distributions instead of point estimates, and for 30 of those parameters the values vary across the model's 16 regions. This includes parameters related to the physical and economic components of the model. The default assumptions in the model include parameters whose probability distributions are based on the normal, Gamma, and triangular distributions. In most cases the distributions are truncated from above or below. The choice of distributions and parameterizations are based on the model developers' assessment of the scientific and economic literature. Complete information on the exact probability distributions specified for each uncertain parameter is provided through the model's documentation, input data, and source code, available at: http://www.fund-model.org/home.

The physical components of the model map emissions to atmospheric concentrations, then map those concentrations to radiative forcing, which is then mapped to changes in global mean temperature. Changes in temperature are then used to estimate sea level rise. The parameters treated probabilistically in these relationships may be grouped into three main categories: atmospheric lifetimes, speed of temperature response, and sea level rise. First, atmospheric concentrations are determined by one box models, that capture a single representative sink, for each of the three non-CO_2 GHGs and a five box model for CO_2, that represents the multiple sinks in the carbon cycle that operate on different time frames. In each of these boxes, the lifetime of additions to the atmospheric concentration in the box are treated as uncertain. Second, parameters associated with speed at which the climate responds to changes in radiative forcing are treated as uncertain. In the FUND model radiative forcing, R_t, is mapped to changes in global mean temperature, T_t, through

$$T_t = T_{t-1} + \frac{1}{\theta_1 + \theta_2 ECS + \theta_3 ECS^2}\left(\frac{\psi ECS}{\ln(2)} R_t - T_{1-t}\right),$$

where the probability distribution for the equilibrium climate sensitivity, ECS, was harmonized across the models as discussed in the 2010 TSD. The parameters θ_i define the speed at which the temperature anomaly responds to changes in radiative forcing and are treated as uncertain in the model. Third, sea level rise is treated as a mean reverting function, where the mean is determined as proportional to the current global mean temperature anomaly. Both this proportionality parameter and the rate of mean reversion in this relationship are treated as uncertain in the model.

The economic components of the model map changes in the physical components to monetized damages. To place the uncertain parameters of the model associated with mapping physical endpoints to damages in context, it is useful to consider the general form of the damage functions in the model. Many of the damage functions in the model have forms that are roughly comparable to

$$D_{r,t} = \alpha_r Y_{r,t} \beta_{r,t} \left(\frac{y_{r,t}}{y_{r,b}} \right)^{\gamma} \left(\frac{N_{r,t}}{N_{r,b}} \right)^{\phi} T_t^{\delta}, \tag{1}$$

where α_r is the damage at a 1 °C global mean temperature increase as a fraction of regional GDP, $Y_{r,t}$. The model considers numerous changes that may reduce a region's benchmark vulnerability to climate change. For example, γ represents the elasticity of damages with respect to changes in the region's GDP per capita, $y_{r,t}$, relative to a benchmark value, $y_{r,b}$; ϕ represents the elasticity of damages with respect to changes in the region's population, $N_{r,t}$, relative to a benchmark value, $N_{r,b}$; and the projection $\beta_{r,t}$ provides for an exogenous reduction in vulnerability (e.g., forecast energy efficiency improvements the affect space cooling costs). Once the benchmark damages have been scaled due to changes in vulnerability they are adjusted based on a non-linear scaling of the level of climate change forecast, using a power function with the exponent, δ.

Some damage categories have damage function specifications that differ from the example in (1). For example, agriculture and forestry damages take atmospheric concentrations of CO_2 and the rate of climate change into account in different forms, though the method by which they calculate the monetized impact in these cases is similar with respect to accounting for GDP growth and changes in vulnerability. In other cases the process by which damages are estimated is more complex. For example, in estimating damages from sea level rise the model considers explicit regional decision makers that choose levels of coastal protection in a given year based on a benefit-cost test. In estimating the damages from changes in cardiovascular mortality risk the model considers forecast changes in the proportion of the population over the age of 65 and deemed most vulnerable by the model developers. Other damage categories may also have functional forms that differ slightly from (1), but in general this form provides a useful framework for discussing the parameters for which the model developers have defined probability distributions as opposed to point estimates.

In many damage categories (e.g., sea level rise, water resources, biodiversity loss, agriculture and forestry, and space conditioning) the benchmark damages, α_r, are treated as uncertain parameters in the model and in most case they are assumed to vary by region. The elasticity of damages with respect to changes in regional GDP per capita, γ, and the elasticity with respect to changes in regional population, ϕ, are also treated as uncertain parameters in most damage functions in the model, though they are not assumed to vary across regions. In most cases the exponent, δ, on the power function that scales damages based on the forecast level of climate change are also treated as uncertain parameters, though they are not assumed to vary across regions in most cases.

Figure C1 presents results of an analysis from the developers of the FUND model that examines the uncertain parameters that have the greatest influence on estimates of the SC-CO_2 based on the default version of the model. While some of the modeling inputs are different for the SC-CO_2 estimates calculated by the IWG these parameters are likely to remain highly influential in the FUND modeling results.

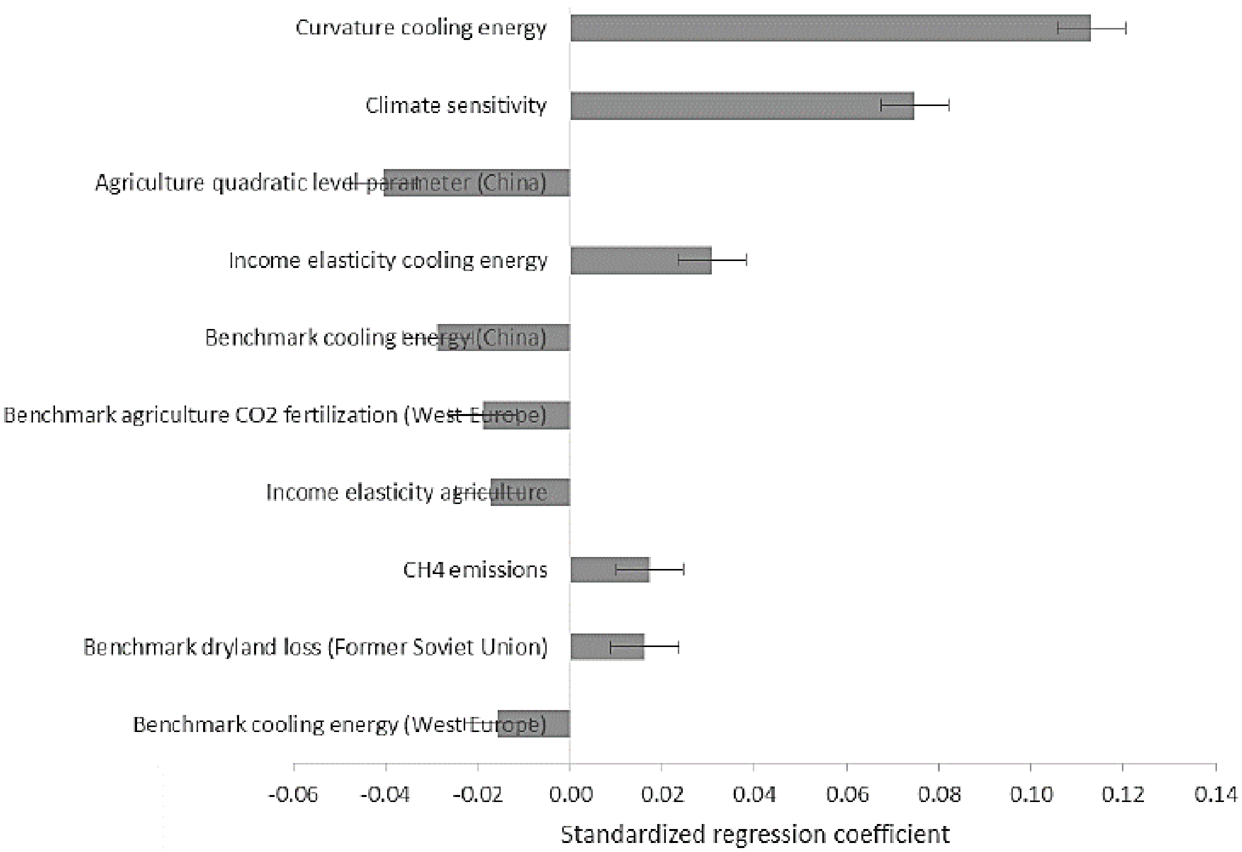

Figure C1: Influence of Key Uncertain Parameters in Default FUND Model (Anthoff and Tol 2013a)[19]

Default Uncertainty Assumptions in PAGE

In the version of the PAGE model used by the IWG (version PAGE09) there are over 40 parameters defined by probability distributions instead of point estimates.[20] The parameters can broadly be classified as related to climate science, damages, discontinuities, and adaptive and preventive costs. In the default version of the model, all of the parameters are modeled as triangular distributions except for the one variable related to the probability of a discontinuity occurring, with is represented by a uniform distribution. More detail on the model equations can be found in Hope (2006, 2011a) and the default minimum, mode, and maximum values for the parameters are provided in Appendix 2 of Hope (2011a). The calibration of these distributions is based on the developer's assessment of the IPCC's Fourth Assessment report and scientific articles referenced in Hope (2011a, 2011b, 2011c). The IWG added an uncertain parameter to the default model, specifically the equilibrium climate sensitivity parameter, which was harmonized across the models as discussed in the 2010 TSD.

In the climate component of the PAGE model, atmospheric CO_2 concentration is assumed to follow an initial rapid decay followed by an exponential decline to an equilibrium level. The parameters treated probabilistically in this decay are the proportion of the anthropogenic CO_2 emissions that enter the atmosphere, the half-life of the CO_2's atmospheric residence, and the fraction of cumulative emissions that ultimately remains in the atmosphere. A carbon cycle feedback is included to represent the impact of increasing temperatures on the role of the terrestrial biosphere and oceans in the carbon cycle. This feedback is modeled with probabilistic parameters representing the percentage increase in the CO_2 concentration anomaly and with an uncertain upper bound on this percentage.

The negative radiative forcing effect from sulfates is modeled with probabilistic parameters for the direct linear effect due to backscattering and the indirect logarithmic effect assumed for cloud interactions. The radiative forcing from CO_2, all other greenhouse gases, and sulfates are combined in a one box model to estimate the global mean temperature. Uncertainty in the global mean temperature response to change in radiative forcing is based on the uncertain equilibrium climate sensitivity parameter and uncertainty in the half-life of the global response to an increase in radiative forcing, which defines the inertia of the climate system in the model. Temperature anomalies in the model vary geographically, with larger increases over land and the poles. Probabilistic parameters are used for the ratios of the temperature anomaly over land relative to the ocean and the ratio of the temperature anomaly over the poles relative to the equator. The PAGE model also includes an explicit sea level component, modelled as a lagged function of the global mean temperature anomaly. The elements of this component that are treated

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[19] Based on a coefficients of standardized regression of parameter draws on the SC-CO_2 using FUND 3.8.1 under Ramsey discounting with a pure rate of time preference of one percent and rate of relative risk aversion of 1.5. The 90 percent confidence intervals around the regression coefficients are presented as error bars.

[20] This appendix focuses on the parameters in the PAGE model related to estimating the climate impacts and principle calculation of the monetized damages. There are over 60 additional parameters in the model related to abatement and adaptation, which may be highly relevant for purposes other than estimating the SC-CO_2, but are not discussed here.

probabilistically include: sea level rise from preindustrial levels to levels in the year 2000, the asymptotic sea level rise expected with no temperature change, the predicted sea level rise experience with a temperature change, and the half-life of the sea level rise.

In the economic impacts module, damages are estimated for four categories: sea level rise, economic damages, non-economic damages, and damages from a discontinuity. Each damage category is calculated as a loss proportional to GDP. The model first calculates damages for a "focus region" (set to the European Union) assuming the region's base year GDP per capita. Damages for other regions are assumed to be proportional to the focus region's damage, represented by a regional weighting factor.

Economic damages, non-economic damages, and damages from sea level rise are modeled as polynomial functions of the temperature or sea level impact, which are defined as the regional temperature or sea level rise above a regional tolerable level. These functions are calibrated to damages at some reference level (e.g., damages at 3°C or damages for a ½ meter sea level rise). The specification allows for the possibility of "initial benefits" from small increases in regional temperature. The variables represented by a probability distributions in this specification are: the regional weighting factors; the initial benefits; the calibration point; the damages at the calibration point; and the exponent on the damage functions.

The damages from a discontinuity are treated differently from other damages in PAGE because the event either occurs or it does not in a given model simulation. In the PAGE model, the probability of a discontinuity is treated as a discrete event, where if it occurs, additional damages would be borne and therefore added to the other estimates of climate damages. Uncertain parameters related to this discontinuity include the threshold global mean temperature beyond which a discontinuity becomes possible and the increase in the probability of a discontinuity as the temperature anomaly continues to increase beyond this threshold. If the global mean temperature has exceeded the threshold for any time period in a model run, then the probability of a discontinuity occurring is assigned, otherwise the probability is set to zero. For each time period a uniform random variable is drawn and compared to this probability to determine if a discontinuity event has occurred in that simulation. The additional loss if a discontinuity does occur in a simulation is represented by an uncertain parameter and is multiplied by the uncertain regional weighting factor to obtain the regional effects.

Damages for each category in each region are adjusted to account for the region's forecast GDP in a given model year to reflect differences in vulnerability based on the relative level of economic development. Specifically, the damage estimates are multiplied by a factor equal to the ratio of a region's actual GDP per capita to the base year GDP per capita, where the ratio exponentiated with a value less than or equal to zero. The exponents vary across damage categories and in each case are treated as uncertain parameters.

Finally, in each region damages for each category are calculated sequentially (sea level rise, economic, non-economic, and discontinuity, in that order) and are assessed to ensure that they do not create total damages that exceed 100 percent of GDP for that region. Damages transition from a polynomial function to a logistic path once they exceed a certain proportion of remaining GDP, and the proportion where this transition begins is treated as uncertain. An additional parameter labeled the "statistical value of civilization," also treated as uncertain, caps total damages (including abatement and adaptation costs described below) at some maximum level.

Figure C2 presents results of an analysis from the developers of the PAGE model that examines the uncertain parameters that have the greatest influence on estimates of the SC-CO_2 based on the default version of the model. Although some of the modeling inputs are different for the SC-CO_2 estimates calculated by the IWG, these parameters are likely to remain highly influential in the PAGE modeling results.

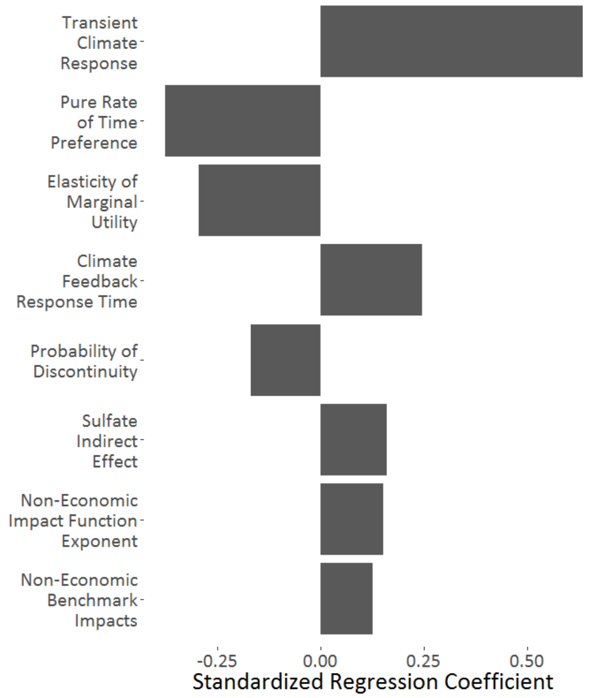

Figure C2: Influence of Key Uncertain Parameters in Default PAGE Model (Hope 2013)[21]

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[21] Based on a standardized regression of the parameters. The values give the predicted increase in the SC-CO_2 in 2010 based on a one standard deviation increase in the coefficient, using the default parameters for PAGE09 under Ramsey discounting with an uncertain pure rate of time preference and rate of relative risk aversion.