Reprinted from the *Journal of Environmental Quality*
Volume 25, no. 2, March–April 1996, Copyright © 1996, ASA, CSSA, SSSA
677 South Segoe Rd., Madison, WI 53711 USA

Estimating the Effects of Changing Land Use Patterns on Connecticut Lakes

Cathryn K. Field, Peter A. Siver,* and Anne-Marie Lott

ABSTRACT

Changes in land use of 30 Connecticut lake watersheds between 1934, 1970, and 1990 were quantified using aerial photographs. Results were used with existing land use models to estimate changes in concentrations of total phosphorus (TP) and total nitrogen (TN) over the 56-yr period. On average, the watersheds have increased in urban-residential land cover from 2% in 1934 to 16% in 1990, and decreased in agricultural land from 20 to 7% during the same time period. The mean percentage of forested land has remained relatively constant. Based on the land use models of Norvell et al. (1979) and Frink (1991), the mean estimated total phosphorus concentration (eTP) increased from 15 μg L^{-1} in 1934 to 25 μg L^{-1} in 1990. The eTP concentrations increased in 26 of the 30 study lakes. In contrast, the mean estimated concentration of total nitrogen (eTN) increased only 20% from 374 to 450 μg L^{-1}. Principal component analysis (PCA) was used to score each study lake according to its current trophic and ionic condition using chemical data from 1991 to 1993, and the results regressed against the 1990 land use types. Trophic scores were most highly correlated with forest cover, while ionic scores were most highly correlated with forest cover and the degree of urban-residential land cover. The effect of water retention time is discussed. Land use models provide useful tools in the management of lakes.

Many Connecticut waterbodies had already undergone significant eutrophication by the late 1930s when Deevey (1940) conducted the first large-scale limnological survey in the state. Since the Deevey study, lakes have become even more eutrophied (Frink and Norvell, 1984; Canavan and Siver, 1994). In addition to an increased nutrient state, the quantity of dissolved salts in Connecticut lakes has also increased (Siver, 1993; Siver et al., 1996). Although there is no evidence as yet of decreasing pH in Connecticut's waterbodies (Marsicano and Siver, 1993), a small group of lakes with decreasing alkalinities has been identified (Siver et al., 1996).

Connecticut was almost entirely forested before European settlement. By the mid- to late-1800s, three-quarters of the state had been converted to agricultural use (Bell, 1985). Shortly thereafter, farmers gradually, but continuously, began turning away from agriculture and toward the manufacturing centers that were developing along the major rivers and the coast (Bell, 1985). Much of the land once used for agriculture reverted back to forest (Stephens, 1976; Bell, 1985). Today, Connecticut is approximately two-thirds forested, one-sixth urban and residential, and one-sixth agriculture (Bell, 1985).

Most researchers agree on the general effects of various land use types on water quality. Agricultural land use has been associated with increased nutrient loading, particularly N (Dillon and Kirchner, 1975; Pionke and Urban, 1985; Frink, 1991; Turner and Rabalais, 1991). Urban land use has also been linked to increased nutrient loading (Gaynor, 1979; Ayers et al., 1980; Osborne and Wiley, 1988), as well as dissolved salt concentrations (Prowse, 1987; Smith et al., 1987; Mattson and Godfrey, 1994). Sewage, lawn fertilizers, phosphate-containing laundry detergents, and pet wastes are important sources of P and N in runoff water from urban and residential areas (Uttormark et al., 1974; Downing and McCauley, 1992). Both forest (Schlosser and Kar, 1981; Lowrance et al., 1984; Peterjohn and Correll, 1984; Detenbeck et al., 1993) and wetland (van der Valk et al., 1979; Johnston et al., 1990; Detenbeck et al., 1993) land covers either export small quantities of nutrients, or may even act as sinks for nutrients.

Around the mid-1930s the use of concentrated man-made N and P fertilizers began to increase significantly. After World War II, fertilizer use increased steadily and peaked around 1980 (Turner and Rabalais, 1991). The composition of fertilizers also changed. In 1929 a typical N fertilizer was composed of 48% ammonium, 19% nitrate, and 33% organic N. By 1949, close to 80% of the N was in the form of ammonium (Turner and Rabalais, 1991). Thus, even though the total acreage of agricultural land has decreased, the levels of nutrients in runoff on a per-acre basis have undoubtedly increased.

Forest, agricultural, and urban land use do not contribute P and N to runoff water in the same proportions. The TN to TP ratio (TN/TP) is highest in forest runoff, less in agricultural runoff, and lowest in urban runoff (Uttormark et al., 1974; Downing and McCauley, 1992). However, in terms of total nutrient load, both TN and TP levels in forested runoff are considerably reduced compared with either agricultural or urban runoff. Phosphorus is usually the limiting factor in fresh water systems (Vollenweider, 1968), although N can also be limiting under certain conditions (Uttormark et al., 1974; Omernik 1976; Downing and McCauley, 1992). Several studies have suggested that a decrease in the TN/TP ratio resulting from an increase in the percentage of urban land in the watershed may cause a shift from P to N limitation (Downing and McCauley, 1992; Canavan and Siver, 1994).

Although many researchers have examined the relationships between nonpoint pollution and water quality (see above), few have developed quantitative models for lakes that involve land cover type (Uttormark et al., 1974; Norvell et al., 1979; Reckhow et al., 1980; Summer et al., 1990; Frink, 1991). Land cover–based export coefficient models have been specifically developed for Connecticut lakes that estimate springtime concentrations of TP (Norvell et al., 1979) and TN (Frink, 1991). The models are based on water retention characteristics and

Botany Dep. Connecticut College, 270 Mohegan Ave., New London, CT 06320. Received 13 Feb. 1995. *Corresponding author (pasiv@conncoll.edu).

Published in J. Environ. Qual. 25:325–333 (1996).

Abbreviations: TP, total phosphorus; TN, total nitrogen; eTP, estimated total phosphorus concentration; eTN, estimated total nitrogen concentration; PCA, principle component analysis; USGS, U.S. Geological Survey; Black-M, Black Pond in Meriden; Black-W, Black Pond in Woodstock; Crystal-E, Crystal Lake in Ellington.

Table 1. Percentages of land use in 30 Connecticut lake watersheds during 1934, 1970, and 1990. See text for descriptions of group designations.†

		1934						1970						1990					
Lake	Group	Lake	U-R	A-O	Forest	Marsh	Water	Lake	U-R	A-O	Forest	Marsh	Water	Lake	U-R	A-O	Forest	Marsh	Water
Alexander	D	0.34	0.02	0.14	0.49	0	0.01	0.34	0.09	0.15	0.41	0	0.01	0.34	0.25	0.04	0.36	0	0.01
Amos	E	0.14	0.03	0.64	0.13	0.06	0.01	0.13	0.05	0.35	0.38	0.09	0.01	0.13	0.13	0.16	0.48	0.08	0.01
Ball	E	0.35	0.04	0.52	0.09	0	0	0.33	0.25	0.18	0.24	0	0	0.33	0.37	0.15	0.15	0	0
Bashan	C	0.22	0.03	0.07	0.67	0.01	0	0.22	0.11	0.05	0.63	0	0	0.21	0.18	0.04	0.57	0	0
Beseck	E	0.09	0.07	0.28	0.56	0	0	0.09	0.10	0.15	0.67	0	0	0.09	0.12	0.12	0.66	0.01	0
Bigelow	A	0.01	0.02	0.06	0.83	0	0.09	0.01	0.03	0.02	0.85	0.01	0.09	0.01	0.04	0.01	0.83	0.02	0.09
Billings	A	0.22	0	0	0.73	0.05	0	0.22	0.02	0	0.72	0.04	0	0.22	0.03	0	0.71	0.04	0
Black-Meriden	C	0.10	0	0.15	0.70	0.05	0	0.11	0.08	0.22	0.55	0.02	0.02	0.09	0.13	0.15	0.58	0.03	0.02
Black-Woodstock	B	0.16	0.01	0.14	0.70	0	0	0.15	0.03	0.07	0.75	0	0	0.15	0.07	0.05	0.73	0	0.01
Breakneck	A	0.02	0	0.02	0.89	0.06	0	0.08	0	0	0.92	0	0	0.08	0	0.01	0.88	0.03	0
Coventry	D	0.19	0.04	0.24	0.51	0.01	0	0.19	0.33	0.10	0.37	0.01	0	0.19	0.33	0.07	0.40	0.01	0
Crystal-Ellington	B	0.11	0.03	0.12	0.73	0.01	0	0.11	0.09	0.06	0.72	0.01	0.01	0.11	0.14	0.03	0.71	0.01	0.01
Gardner	B	0.17	0.06	0.19	0.52	0.06	0	0.18	0.07	0.12	0.59	0.04	0	0.18	0.21	0.05	0.52	0.04	0
Hayward	B	0.12	0.02	0.19	0.67	0	0	0.11	0.08	0.11	0.66	0.04	0	0.10	0.13	0.06	0.68	0.03	0
Kenosia	E	0.02	0.07	0.44	0.42	0.05	0.01	0.02	0.21	0.12	0.61	0.03	0.03	0.02	0.37	0.05	0.50	0.05	0.02
Linsley	D	0.04	0.03	0.31	0.48	0.10	0.04	0.04	0.20	0.22	0.37	0.13	0.05	0.04	0.39	0.11	0.28	0.13	0.04
Long	B	0.04	0.01	0.12	0.79	0.03	0.01	0.03	0.06	0.04	0.86	0	0.01	0.04	0.12	0.05	0.78	0.01	0.01
Mashapaug	A	0.09	0	0.05	0.84	0	0.01	0.10	0	0.03	0.82	0.01	0.01	0.10	0.06	0.02	0.80	0.02	0.01
Mohawk	A	0.14	0	0	0.85	0.02	0	0.14	0.01	0	0.85	0	0	0.14	0.03	0	0.83	0	0
Mt. Tom	E	0.08	0.03	0.34	0.54	0	0	0.08	0.04	0.35	0.52	0.02	0.01	0.08	0.10	0.20	0.60	0.02	0
Pattagansett	D	0.05	0.03	0.12	0.70	0.03	0.07	0.05	0.12	0.08	0.69	0	0.07	0.05	0.23	0.04	0.60	0	0.06
Quassapaug	D	0.25	0.01	0.20	0.51	0.02	0	0.25	0.10	0.17	0.43	0.04	0	0.25	0.16	0.10	0.47	0.03	0
Quonnipaug	B	0.06	0.01	0.16	0.73	0.03	0.01	0.06	0.03	0.11	0.78	0.03	0	0.06	0.07	0.09	0.75	0.02	0.01
Rogers	E	0.06	0.01	0.16	0.73	0.04	0	0.06	0.04	0.04	0.82	0.04	0	0.06	0.06	0.03	0.79	0.05	0.01
Silver	B	0.12	0.02	0.28	0.51	0.04	0.03	0.12	0.15	0.15	0.52	0.04	0.03	0.12	0.21	0.06	0.52	0.07	0.03
Terramuggus	D	0.23	0.05	0.21	0.52	0	0	0.23	0.29	0.07	0.41	0	0	0.24	0.40	0.03	0.33	0	0
Tyler	E	0.04	0.01	0.22	0.68	0.04	0.01	0.05	0.02	0.19	0.70	0.05	0.01	0.05	0.03	0.13	0.72	0.05	0.02
Uncas	A	0.07	0	0.02	0.69	0.20	0.03	0.07	0	0.01	0.78	0.11	0.03	0.07	0.01	0	0.77	0.11	0.03
West Hill	D	0.30	0.03	0.08	0.60	0	0	0.31	0.19	0.01	0.49	0	0	0.32	0.21	0	0.47	0	0
Wonoscopmuc	E	0.22	0.01	0.53	0.24	0	0	0.22	0.16	0.22	0.41	0	0	0.22	0.27	0.16	0.36	0	0
Mean		0.13	0.02	0.20	0.60	0.03	0.01	0.14	0.10	0.11	0.62	0.02	0.01	0.14	0.16	0.07	0.59	0.03	0.01
SD		0.10	0.02	0.16	0.19	0.04	0.02	0.09	0.09	0.09	0.18	0.03	0.02	0.09	0.12	0.06	0.19	0.03	0.02

† U-R = urban-residential; A-O = agriculture and open fields; water = other water bodies.

the percentages of different types of land use within a watershed. Both models are highly significant with r^2 values of 0.65 for P and 0.57 for N.

The objectives of this study were threefold. The first was to characterize land use changes in Connecticut lake watersheds since 1934. Estimates were made for 30 watersheds during three study periods, 1934, 1970, and 1990. Second, we utilized the land use-based models for Connecticut to estimate quantitative changes and identify trends in the TP and TN concentrations of the study lakes. Third, the land use data for 1990 was correlated with water chemistry data collected between 1991 and 1993. Two variables, trophic score and ionic score, were derived using PCA and correlated with land use type to determine if the latter could be used to estimate the former.

METHODS

Land use patterns of the watersheds of 30 lakes (Table 1; Fig. 1) were determined from high resolution aerial photographs with a scale of ca. 1 cm:120 m for the periods 1934, 1970, and 1990 (90 maps). The aerial photographs are archived in the State of Connecticut Library (1934) or the Department of Environmental Protection (1970 and 1990), both located in Hartford, CT. Watershed boundaries were determined using USGS topographical maps in conjunction with Connecticut drainage basin maps, printed to the scale of the aerial photographs, and overlayed onto the photographs. Coverages were digitized and analyzed using PC ARC/INFO.

The names and locations of the 30 study watersheds correspond to those listed in previous publications (Conn. Dep. Environ. Prot., 1982; Frink and Norvell, 1984; Canavan and Siver, 1994). In these publications there are two Black Ponds and two Crystal Lakes. To avoid confusion we have listed the initials of the town in which the waterbody is located after the name of the waterbody: Black-M is Black Pond in Meriden, Black-W is Black Pond in Woodstock, and Crystal-E is Crystal Lake in Ellington.

Land use was categorized as forest, agricultural–open field, urban–residential, marsh, and water (including the lake). These land use categories were selected to conform to the categories utilized by Norvell et al. (1979) and Frink (1991). Although other researchers have suggested distinguishing between the various types of active agriculture and fallow land (Uttormark et al., 1974; Dillon and Kirchner, 1975; Hill, 1981; Beaulac and Reckhow, 1982), it was not practical to do so because the distinction could not easily be made from the aerial photographs. The *marsh* category included all types of wetlands. For all of the watersheds the *urban–residential* category is composed primarily of residential areas.

Norvell et al. (1979) developed a model for estimating spring concentrations of TP based on fractions of land use cover in the watershed and the chemical retention time of P. The model is as follows:

$$\text{eTP} = [(Q + 1.2)/(Q + 12)] \times (170U + 54A + 10W)/D \quad [1]$$

where

eTP = the estimated total P concentration in μg L^{-1} (or ppb),
Q = water load on the lake in m yr^{-1},
D = water export from entire watershed in m yr^{-1}, or
D = Q/watershed to lake area ratio,

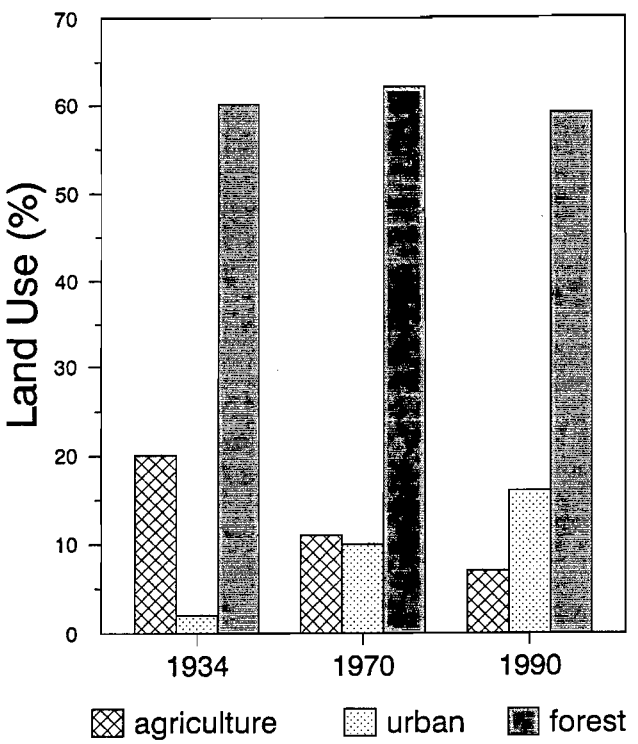

Fig. 1. Mean land use percentages for 30 Connecticut lake watersheds in 1934, 1970, and 1990. The agriculture category also includes open fields and the urban category residential areas.

A = fraction of agricultural land in the watershed,
U = fraction of urban or residential land in the watershed, and
W = fraction of wooded land in the watershed.

Frink (1991) later developed a similar model using the same data set for estimating spring concentrations of TN:

$$\text{eTN} = [(Q + 1)/(Q + 5)] \times (1340U + 760A + 240W)/D \quad [2]$$

where eTN = the estimated total N concentration in $\mu g\ L^{-1}$ (or ppb).

The *wooded* component in both models is a composite of forest, wetlands, and open water with the assumption that the nutrient export of each of the three cover types is the same (see Norvell et al., 1979; Frink, 1991).

Values for Q were obtained from Norvell et al. (1979) for 17 of the lakes. For the remaining study lakes (n = 13), Q was calculated using the following formula:

$$Q = \frac{0.032 \times \text{Rate of drainage} \times \text{Watershed area}}{\text{Lake area}} \quad [3]$$

where Q is in m yr^{-1}. Rate of drainage, in m^3 (sec × $km^2)^{-1}$, is an estimate of the amount of water being exported per unit time per unit of watershed area. The rate of drainage for each watershed was estimated using 30-yr streamflow averages according to the method utilized by the Connecticut Department of Environmental Protection (Thomas, 1966).

The Norvell et al. (1979) and Frink (1991) models were utilized to estimate changes in TP and TN concentrations between the 1934, 1970, and 1990 time periods. The land cover estimates determined from the aerial photographs for each time period were used in each analysis. ANOVA analyses were used to determine if there were any significant differences between either eTP or eTN concentrations of the lakes as a group for the three time periods. Because there was only a single estimate for each lake per time period, the lakes were grouped together for the analyses. The ANOVA analyses were done using SPSSX (Norušis, 1985) on a DEC minicomputer.

In the final part of the study, 1990 land use data obtained from the aerial photographs were correlated with both the trophic condition and the dissolved ionic chemistry of the receiving waterbodies. Correlations were initially made with measured summer TP, TN, and specific conductivity values. To further examine these relationships, 28 of the study lakes were also characterized for trophic and ionic chemistry using PCA according to Canavan and Siver (1994). In PCA analysis new variables (represented by axes) are derived that are linear combinations of the original variables. The first axis is derived such that it explains the maximum amount of variation within the data set. Subsequent axes are derived with the added constraint that they be orthogonal to the previously derived axes.

In this study PCA analyses were run on two sets of measured chemical variables to derive trophic and ionic scores for each study lake. The first set of measured variables included chlorophyll-*a*, Secchi disk depth, TP, and TN, and yielded PCA scores for each lake referred to as trophic scores. The second set of measured variables included Ca^{2+}, Mg^{2+}, Na^{+}, K^{+}, alkalinity, Cl^{-}, and SO_4^{2-}, and yielded scores referred to as ionic scores. Only the first extracted PCA axis was used in each analysis. Each PCA analysis was based on a correlation matrix, and Euclidean Distance plots are presented (ter Braak, 1983, 1990). Chemical data for the lakes were obtained from Canavan and Siver (1994), and the PCA was run using CANOCO version 3.1 (ter Braak, 1990). Correlation and regression analyses using SPSSX (Norušis, 1985) were used to examine the relationships between land use types and trophic and ionic scores. Unless otherwise stated, a significance level of $P < 0.05$ was used.

RESULTS

Changes in Land Use Since 1934

The 30 study watersheds represented a wide range of land use patterns. The mean percentage of agriculture/open field land use for all watersheds was greatest in 1934 at 20%, and decreased to 11% by 1970 and 7% by 1990 (Table 1; Fig. 1). Between 1934 and 1990, nearly every watershed exhibited a decrease in agriculture. In contrast, the mean percentage of urban–residential land use steadily increased from 2% in 1934 to 16% in 1990 (Table 1; Fig. 1). Almost every watershed showed an increase in urban land during the 56-yr period. The mean percentage of forested land in 1970 (62%) was only slightly higher than the mean percentages in both 1934 and 1990 (60 and 59%, respectively) (Table 1; Fig. 1). Thus, the mean decrease in agriculture (13%) was closely coupled with the mean increase in urban land (14%). Despite the minimal change in the mean amount of forested land use over all study sties, significant changes in individual watersheds were observed.

The changes in land use during the 56-yr period in the 30 watersheds were catalogued into five groups (A–E) (Table 1). Category A consisted of watersheds with primarily natural vegetation (forest and wetlands) during the 56 yr. Category A watersheds were composed of at least 75% natural vegetation and approximately

10% or less manipulated lands (agriculture and/or urban land). The change in the amount of natural vegetation between the study periods was <5%. Category B consisted of watersheds with a mean decrease in agriculture of 12%, a mean increase in urban land of 11%, and little change in forested land. Category C contained watersheds that exhibited reductions in forested land of at least 10%, additions in urban land of >10%, and little change in agriculture. Watersheds in Category D experienced similar changes to those in Category C; however, they also had significant decreases in agriculture (mean decline of 13%) and greater amounts of urban development (25 vs. 14%). Watersheds in Category E exhibited mean increases in forest and urban land of 10 and 15%, respectively, at the expense of a large decrease in agriculture of 26%.

The PC ARC/INFO was also used to determine where changes in land use had taken place since 1934. In many of the watersheds, much of the agricultural land in 1934 had reverted back to forests by 1990, and a lesser amount became urban land. In other watersheds, a large percentage of 1934 agricultural land was developed into urban land by 1990. In the majority of the 30 watersheds, many of the forested areas in 1934 remained forested in 1990; a small percentage was developed as urban land, and an even smaller amount was converted to agriculture. In all cases, most of the urban land cover remained urban in subsequent years. The percentages of open water and wetlands remained relatively stable for the 56-yr period in most watersheds. However, a few lakes experienced changes in surface area and shape due primarily to damming.

Estimated Changes in Nutrient Concentrations

The land use–based models for estimating spring TP (Norvell et al., 1979) and TN (Frink, 1991) concentrations were applied to each lake for the three time periods 1934, 1970, and 1990 (Table 2). The mean spring eTP levels increased significantly ($P < 0.01$) from 15 to 25 μg L^{-1} between 1934 and 1990, and concentrations increased in a unidirectional pattern in 26 of the 30 study lakes. In 1990, the eTP levels ranged from a low of 6 μg L^{-1} in Billings to a high of 91 μg L^{-1} in Kenosia (Table 2). The eTP concentrations more than doubled in eight of the lakes since 1934, whereas seven had eTP levels within 20% of their 1934 estimates. Even though only two lakes had eTP levels >25 μg L^{-1} in 1934, 11 were estimated above this value by 1990. A slightly larger percentage of the increase in eTP occurred between 1970 and 1990 (32%) than between 1934 and 1970 (27%), reflecting the greater degree of urban development during the last 20 yr of the study period.

Between 1934 and 1990, the mean spring eTN for the study lakes increased significantly from 374 to 450 μg L^{-1}. Most of the lakes ($n = 23$) experienced increases in eTN levels since 1934, although the percentage increase was less than that for eTP (Table 2). As opposed to eTP, eTN levels did not double in any of the lakes since 1934; on the other hand, 1990 estimates were within 20% of 1934 eTN levels for 13 of the lakes. The eTN levels increased by a larger percentage between 1970 and 1990 than between 1934 and 1970 (12 vs. 8%).

The eTN/eTP ratio ranged from 16 to 41 in 1934,

Table 2. Estimates of total nitrogen (eTN), total phosphorus (eTP), and (eTN)/(eTP). Estimates were calculated for 1934, 1970, and 1990 using Norvell et al. (1979) and Frink's (1991) land use based models.

	eTN (ppb)			eTP (ppb)			eTN/eTP		
Lake	1934	1970	1990	1934	1970	1990	1934	1970	1990
Alexander	197	239	310	6	10	16	31	24	19
Amos	704	601	579	28	24	26	25	25	22
Ball	417	506	600	15	25	32	27	20	18
Bashan	233	283	352	8	12	18	30	24	20
Beseck	498	462	454	24	23	23	21	20	19
Bigelow	427	418	429	21	21	24	20	20	18
Billings	180	191	198	4	5	6	41	28	34
Black-M	319	409	504	11	19	26	30	22	19
Black-W	273	278	307	9	10	12	31	28	25
Breakneck	353	263	260	13	7	7	26	38	35
Coventry	352	557	558	13	32	32	27	17	17
Crystal-E	376	406	471	13	18	23	28	23	20
Gardner	291	264	359	12	11	20	24	24	18
Hayward	367	426	475	14	19	24	27	22	20
Kenosia	812	800	1009	51	62	91	16	31	11
Linsley	457	569	737	24	40	62	19	14	12
Long	365	410	477	16	21	30	23	20	16
Mashapaug	260	250	319	8	7	13	33	36	25
Mohawk	222	237	255	6	7	8	40	34	30
Mt. Tom	485	508	479	21	23	24	23	22	20
Pattagansett	352	461	562	15	27	39	23	17	14
Quassapaug	278	344	374	9	14	17	32	25	22
Quonnipaug	376	397	448	15	16	22	25	25	21
Rogers	381	352	379	15	15	18	26	24	21
Silver	387	471	476	15	24	27	26	20	18
Terramuggus	353	510	597	13	27	35	28	19	17
Tyler	460	430	399	20	19	18	23	23	22
Uncas	265	276	277	8	8	9	33	35	30
West Hill	306	283	293	9	14	15	32	20	20
Wonoscopomuc	483	481	562	18	22	29	28	22	19
Mean	374	403	450	15	19	25	27	24	21

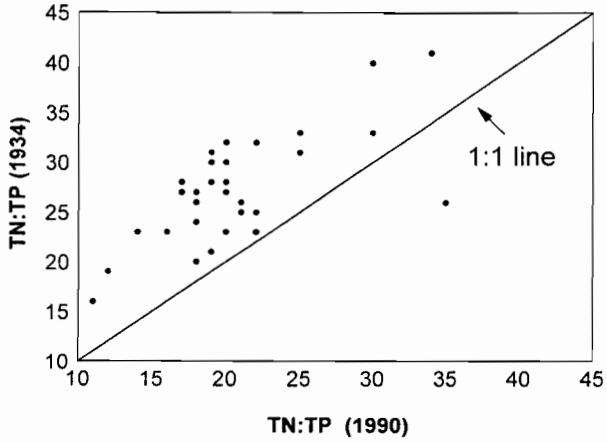

Fig. 2. Comparisons of eTN/eTP ratios between 1934 and 1990 for 30 Connecticut lakes. The line represents a 1:1 line. Values above or below the line indicate that the eTN/eTP was higher or lower, respectively, in 1934.

with a mean of 27 (Table 2). By 1970, the mean eTN/eTP ratio had decreased to 24 and the values ranged from 14 to 38. The mean eTN/eTP ratio continued to decline between 1970 and 1990 to 21, with values ranging from 11 to 35. Based on ANOVA analysis, the set of eTN/eTP ratios for 1990 were significantly lower ($P < 0.05$) than those for 1934 (Fig. 2), and significantly correlated with the amount of urban ($r = -0.73$; $P < 0.01$) and wooded ($r = 0.75$; $P < 0.01$) land.

The measured summer TP concentrations for the study lakes were significantly correlated with spring eTP levels derived from the models ($r = 0.48$; $P < 0.05$). Spring eTN concentrations were also significantly correlated with measured summer TN levels ($r = 0.61$; $P < 0.01$).

Correlations between Current Land Use and Lake Conditions

Based on the PCA analysis for trophic state, Axis I explained 72% of the variance, while Axis II explained an additional 13% (Fig. 3). The position of each lake along PCA Axis I, referred to as the *trophic score*, reflects the overall trophic condition of the lake; the higher the score, the higher the trophic condition. For the ionic score analysis, PCA Axis I, referred to as *ionic score*, explained 62% of the variance, and Axis II an additional 18% (Fig. 4). Higher ionic scores along PCA Axis I reflect higher concentrations of dissolved ions. Canavan and Siver (1994) discuss further the spread of the lakes along Axes I and II for both PCA-derived scores.

The study lakes exhibited a wide diversity of both trophic and ionic scores (Table 3; Fig. 3 and 4). Trophic scores ranged from −1.57 (West Hill) to 0.94 (Linsley), with a mean of −0.32. Ionic scores ranged from −0.79 (Mohawk) to 2.39 (Kenosia), with a mean of 0.1. Of the five watersheds with exceptionally high ionic scores, all had high percentages of urban land in the surrounding watershed, ranging from 21 to 39%. In addition, the watersheds of three of the five waterbodies are located in the Marble Valley, a geologic region in western Connecticut with bedrock composed primarily of marble (Brooks and Deevey, 1963; Bell, 1985).

Significant correlations were found between land use types and PCA scores (Table 4). Trophic scores were positively correlated with agriculture and urban land covers, and negatively correlated with the percentage of forest cover. When all study lakes were included in the analysis, the correlations were only modestly significant. The best predictor of trophic score was forest cover with a Pearson correlation values (r) of −0.66; agriculture and urban land had slightly lower values of 0.55 and 0.53, respectively. The r value for trophic condition and forest cover increased to −0.87 (Fig. 5), and for urban land to 0.81 when lakes with high retention times (i.e., >511 d) were removed. Unlike the forest and urban land use regressions, the r value relating trophic score and agriculture (0.52) did not significantly change when high retention time lakes were omitted.

The division of the lakes above and below retention times of 511 d was based on two lines of reasoning.

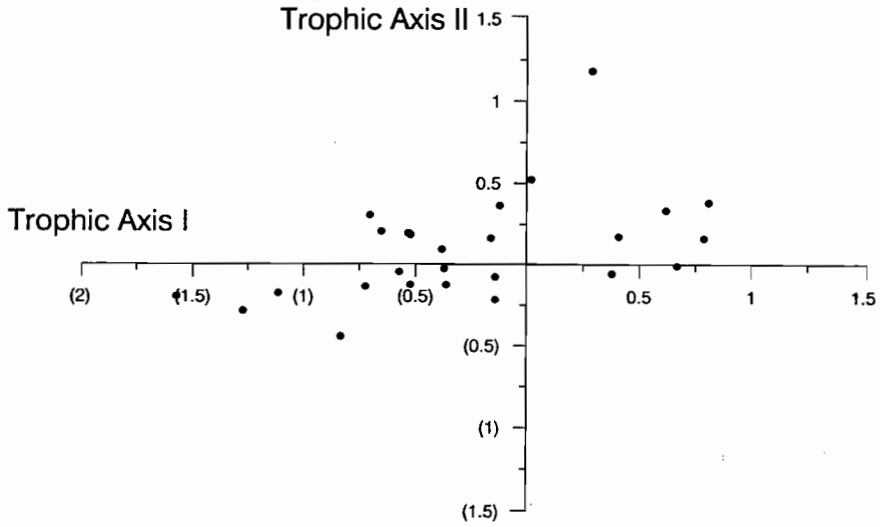

Fig. 3. Euclidean distance plot for Axes I and II of PCA-derived trophic scores of 27 lakes.

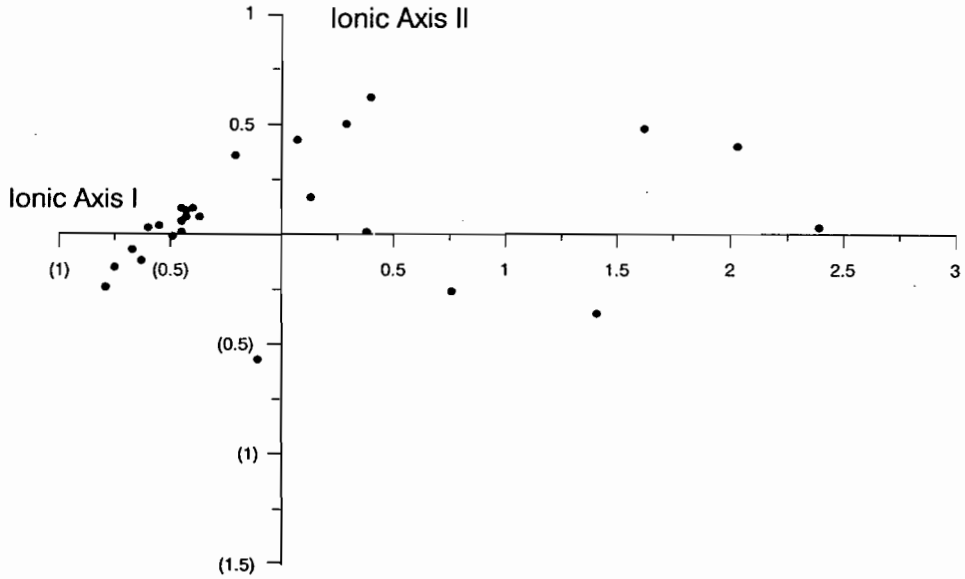

Fig. 4. Euclidean distance plot for Axes I and II of PCA-derived ionic scores of 27 lakes.

First, lakes with the largest retention times were removed one at a time and the regression analysis was redone using the remaining lakes. Based on this analysis, the significance of the resultant relationship continued to increase until the 511-d mark. If additional lakes were removed, the significance began to decline. Second, there was a large gap between 511 d and the lake with the next highest retention time.

Ionic scores were also significantly and positively correlated with the percentage of urban and agricultural land in the watershed, and negatively correlated with the percentage of forest cover (Table 4). Ionic score was most strongly correlated with urban (Fig. 6) and forest land. A regression of the ionic score and percent urban land yielded an r value of 0.72, which increased to 0.85 when lakes with retention times >511 d were omitted. Pearson correlation coefficients for ionic scores and forested land use increased from $r = -0.80$ to -0.90 when lakes with high retention times were removed. The r for agriculture, 0.52, was small compared with the other two land use types, but was a significant relationship nonetheless (Table 4).

Many of the relationships between PCA scores and lake water chemistry variables were statistically significant as well (Table 4). Ionic score was found to be an excellent predictor of specific conductivity ($r = 0.99$). A slightly weaker correlation was found between trophic score and conductivity. Trophic and ionic scores were also significantly related ($P < 0.01$) to estimated values of TN and TP derived from the models, with r values ranging from 0.61 to 0.82 (Table 4).

Table 3. Trophic and ionic scores for 1990 derived from PCA. High trophic scores represent lakes with high trophic states, while high ionic scores represent lakes with high salt content.

Lake	Trophic score	Ionic score
Alexander	−0.70	−0.63
Amos	0.41	0.40
Ball	0.81	2.03
Bashan	−1.27	−0.60
Beseck	0.38	0.38
Bigelow	−1.00	−0.75
Billings	−0.72	−0.67
Black-M	−0.16	0.76
Black-W	−0.52	−0.49
Breakneck	−1.31	−0.74
Coventry	−0.12	0.29
Crystal-E	−0.52	0.07
Gardner	−0.14	−0.37
Hayward	−0.37	−0.45
Kenosia	0.67	2.39
Linsley	0.94	1.62
Long	−0.83	−0.21
Mashapaug	−1.11	−0.43
Mohawk	−0.53	−0.79
Mt. Tom	−0.38	0.13
Pattagansett	−0.14	−0.43
Quassapaug	0.29	−0.45
Quonnipaug	N/A	N/A
Rogers	−0.36	−0.40
Silver	0.79	1.41
Terramuggus	N/A	N/A
Tyler	−0.31	−0.11
Uncas	−0.57	−0.55
West Hill	−1.57	−0.45
Wonoscopomuc	−0.65	1.75
Mean	−0.32	0.10

DISCUSSION

Since the 1930s, percentages of urban land use in the study watersheds have been increasing primarily at the expense of agricultural cover. With the interaction of all factors, some such as urbanization acting to deforest areas, and others such as old field succession acting to reforest areas, it is not surprising that we observed little change in the mean percentage of forested land since 1934. The overall pattern of change exhibited in the 30 study watersheds mirrored trends observed throughout Connecticut (Stephens, 1976; Bell, 1985) and in many areas of the USA (USDA, 1990; Turner and Rabalais, 1991). Despite these generalizations, it is important to remember that on an individual watershed basis, significant changes in each of the land use types may have occurred between 1934 and 1990.

Table 4. Pearson correlation coefficients (*r*) and results of significance tests between 10 land use and chemical variables.†

	W	U	A	eTP	eTN	TP	TN	µS	TS	IS
W	1.00	−0.91**	−0.57**	−0.68**	−0.76**	−0.48*	−0.59**	−0.76**	−0.66**	−0.80**
U		1.00	0.19	0.71**	0.71**	0.31	0.44*	0.65**	0.53**	0.72**
A			1.00	0.22	0.40**	0.43*	0.54**	0.51**	0.55**	0.52**
eTP				1.00	0.96**	0.48*	0.54**	0.73**	0.61**	0.75**
eTN					1.00	0.30	0.61**	0.79**	0.68**	0.82**
TP						1.00	0.69**	0.38*	0.53**	0.42*
TN							1.00	0.76**	0.61**	0.76**
µS								1.00	0.59**	0.99**
TS									1.00	0.61**
IS										1.00

*, ** $P < 0.05$ and $P < 0.01$, respectively.
† W = forests; U = urban-residential; A = agriculture-open field; eTP = estimated total phosphorus; eTN = estimated total nitrogen; TP = measured total phosphorus; TN = measured total nitrogen; µS = measured specific conductivity; TS = trophic score; and IS = ionic score. See text for details.

Trophic Scores

The PCA-derived trophic scores based on current conditions of the lakes were found to be highly correlated with the three primary land use types. Trophic scores were most highly correlated with the percentages of forest cover in the watersheds. The most oligotrophic lakes in our study (i.e., those with the lowest trophic score) had the highest percentages of forest cover. Other researchers have also found that as the amount of forest cover increases, the levels of TN and TP decrease, and the quality of the receiving waterbody increases. Nutrient export coefficient models that use percentages of land use typically assign the lowest export coefficients to forest cover and considerably higher values to both urban and agricultural land (Uttormark et al., 1974; Norvell et al., 1979; Reckhow et al., 1980; Frink, 1991). We conclude that the amount of forest cover in a watershed in Connecticut can accurately be used to estimate the overall trophic condition of a receiving waterbody.

In this study, strong correlations were also found between the trophic scores of the receiving lakes and the percentages of both urban and agricultural land in the watersheds. As the percentages of these land uses increased, the trophic condition of the lake worsened. This is in agreement with many studies that have found both urban and agricultural land contribute significantly greater concentrations of N and P to runoff than does forested land. In the nutrient models utilized in this study, Norvell et al. (1979) and Frink (1991) reported that urban and agricultural land use contributed 17 and 5.4 times more P and 5.6 and 3.2 times more N than forested land, respectively. Although the coefficients differed, Uttormark et al. (1974) and Reckhow et al. (1980) assigned the same rank to each land use type in their nutrient export coefficient models.

Uttormark et al. (1974) and Beaulac and Reckhow (1982) further noted that nutrient loading rates increased with increasing percentages of impervious surface area and residential housing in the watershed. Gaynor (1979) agreed, observing that TP levels were significantly greater downstream from intensive housing areas than from less developed areas. Osborne and Wiley (1988) found that urban land use influenced soluble P levels more than agriculture, and explained most of the variance in NO_3-N levels.

Much work has also been done on the effects of agriculture on water quality. Significantly higher nutrient levels have been found in waters draining active agricultural land compared with those draining forested areas (Dillon and Kirchner, 1975; Omernik, 1976; Peterjohn and Correll, 1984; Pionke and Urban, 1985), especially if fertilizers were used (McColl, 1978; Beaulac and Reckhow, 1982). In our study, the relationship between trophic score and agricultural cover may have been even stronger if we were able to separate open field cover from true agricultural lands. Presumably, open fields did not contribute as much to the nutrient loading as actively fertilized fields.

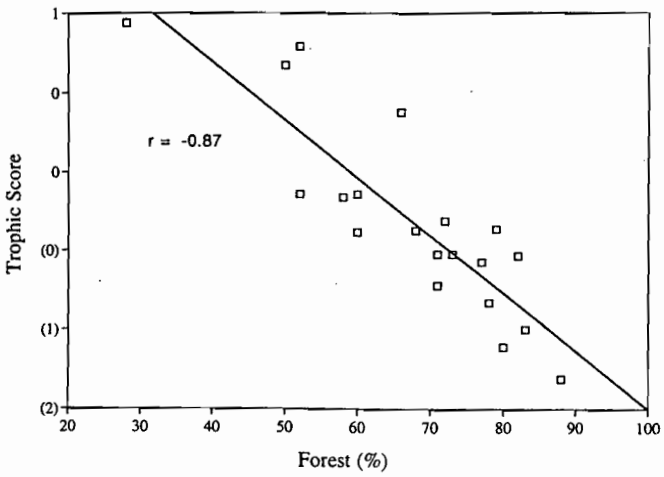

Fig. 5. Relationship between the percentage of forest cover in the watershed and the lake water trophic score for 1990.

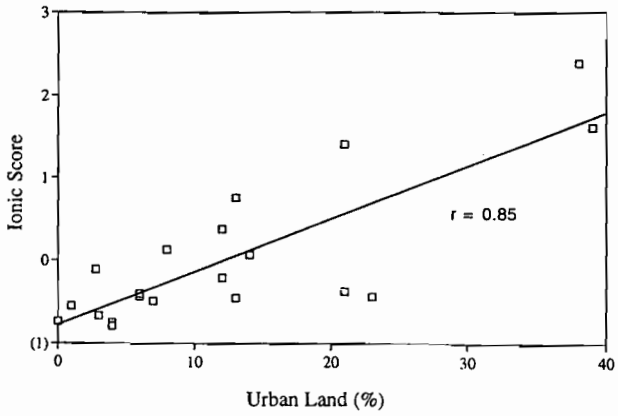

Fig. 6. Relationship between the percentage of urban-residential land in the watershed and the ionic score for 1990.

Ionic Scores

Results from our study support previous studies that have linked elevated dissolved salt concentrations in runoff with decreasing forest cover or increasing urban and agricultural cover. Bormann and Likens (1979) found significantly higher ion concentrations following forest clearcutting and use of pesticides. When McColl (1978) converted scrubland to fertilized pasture by mowing concentrations of dissolved ions (particularly Ca^{2+}, K^+, and SO_4^{2-}) significantly increased. In a comprehensive study of Na^+ and Cl^- levels in Massachusetts streams, Mattson and Godfrey (1994) observed the highest concentrations in urban and residential areas where use of road salt as a deicing agent was most intensive. Smith et al. (1987) also found a significant relationship between Na^+ and Cl^- concentrations in waterbodies and rates of highway road salt use. In addition, researchers have linked high levels of dissolved and suspended materials to urban runoff. Prowse (1987) found that the dissolved load per unit area for an urban subcatchment was 13 times greater than that of a rural subcatchment upstream. Lettenmaier et al. (1991) found that Ca^{2+}, Mg^{2+}, and K^+ levels were positively associated with percent urban cover. It is clear that increased concentrations of dissolved ions in runoff result from urbanization.

On agricultural land, both mechanized tilling techniques and fertilizer use affect the chemistry of runoff water. The erosion caused by plowing contributes significantly to suspended sediments as well as dissolved salt levels in runoff water (Detenbeck et al., 1993). In addition, the use of manure and KCl fertilizer can significantly increase the concentration of Cl^- (Pionke and Urban, 1985). Pionke and Urban (1985) found much higher concentrations of Cl^- in groundwater underlying agricultural land than forested land. Detenbeck et al. (1993) also found higher Cl^- concentrations in lakes that had watersheds with more agricultural cover relative to wetland or forested cover.

Lake Retention Time

The retention time of a lake also strongly influenced the degree of correlation between PCA scores and land use types. Lakes with low retention times have water chemistry that is more similar to incoming water than lakes with high retention times (Frink, 1991). We hypothesize that retention time will have a greater effect on the trophic score than on the ionic score because of the greater rate of removal of N and P by microorganisms relative to the higher concentrations of the more conservative dissolved ions (e.g., Na^+). Thus, lakes with high retention times would have effectively lower trophic scores.

Several aspects of this study support this hypothesis. First, trophic and ionic scores were more highly correlated with land use when lakes with retention times more than 511 d were removed. Second, ionic scores were found to be more conservative indicators of land use than trophic scores. That is, when lakes with high retention times were removed from the analyses, relationships between land use types and trophic scores improved more so than those with ionic scores.

Norvell et al. (1979) and Frink (1991) recognized the importance of retention time and incorporated it as a variable in their land use-based models. Utilizing the Norvell et al. (1979) model, we estimated that the mean spring TP concentration of our study lakes has increased approximately 67% since the 1930s, similar to observations based on actual historical chemical records (Siver et al., this volume). This indicates that the Norvell et al. (1979) model can be used to effectively estimate trends in TP based on changes in land use.

Total Nitrogen/Total Phosphorus Ratios

Because increases in eTN concentrations have been less than those for eTP, the mean eTN/eTP ratio has decreased from 27 to 21 since 1934. The idea that urban land use results in higher export coefficients for P relative to N, and thus a decrease in the TN/TP ratio, is supported by other studies. Uttormark et al. (1974) reported very low TN/TP ratios in runoff from urban land compared to either forest or agricultural cover. Likewise, Downing and McCauley (1992) indicated that increased urbanization in a watershed would result in a decrease in the TN/TP ratio. Downing and McCauley (1992) further hypothesized that waterbodies would have a greater tendency toward N limitation with increased urbanization, especially if TP levels were >30 μg/L. The decrease in the eTN/eTP ratio estimated from the models for Connecticut lakes is consistent with trends observed since the 1970s by Canavan and Siver (1994).

Model Considerations

Norvell et al. (1979) and Frink (1991) related land use to spring nutrient concentrations, the season during which lake water chemistry is most similar to the incoming water. Relationships using summer nutrient levels were poor at best, presumably due to complications from such factors as thermal stratification and internal nutrient loading. Thus, it was not surprising that the correlations between contempory measured summer TP and TN concentrations and model derived estimates for spring were only marginally significant.

There are other inherent deficiencies in land use-based models for estimating nutrient concentrations of receiving waterbodies, especially on an historical level. First, since the models were based on chemical data from the 1970s when fertilizer use was extensive, it is reasonable to believe that their use may overestimate nutrient levels for the 1930s when fertilizers were not widely utilized. Another factor that could improve land use models is consideration of the effects of waterbodies and wetlands positioned upstream of the receiving lake (Norvell et al., 1979; Frink, 1991). An upstream waterbody would essentially increase the retention time of the runoff, thereby altering its chemistry before entering the primary receiving lake. Although land use models often incorporate retention times of the primary waterbody, modifications due to upstream lakes and wetlands, as well as the percentage of the runoff from the watershed that flows through such areas, are not. An upstream lake or wetland would serve as a *filter* for

dissolved nutrients and ions through uptake by macrophytes and other organisms in the wetland (Detenbeck et al., 1993). Using GIS and PCA analysis, Detenbeck et al. (1993) found that the greater the wetland extent, the lower the lake trophic state, TP, TN, and chlorophyll-*a*, and the greater the Secchi disk depth. Furthermore, they found that the closer the wetland was to the lake, and hence, the more of the watershed that drained through it, the lower the trophic state. The findings of Detenbeck et al. (1993) lend great support to this study and suggest improvements that could be incorporated into the Connecticut-based land use models. In summary, although the Norvell et al. (1979) and Frink (1991) models could be improved, especially by using GIS-derived statistics, they still provide a very useful tool in the management of aquatic resources.

ACKNOWLEDGMENTS

This project was funded with a grant from the National Science Foundation (grant DEB-9306587), and the KECK Undergraduate program at Connecticut College. Comments from Paul Bukaveckas and two external reviewers strengthened the manuscript.

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