

CHANGES IN VILLAGE-SCALE NITROGEN STORAGE IN CHINA'S TAI LAKE REGION

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Abstract. Long-term change in nitrogen storage in densely populated rural landscapes of China's Tai Lake Region was investigated by comparing soil and sediment N storage within an entire village ecosystem under traditional vs. contemporary management. Contemporary data were gathered on site from 1993 to 1996 by field surveying and sampling. Traditional period data, ~1930, were obtained from historical sources, interviews, and back estimation. N storage in the top 40 cm of soil and in low-density sediments (depth to density >1.3 g/cm³) was estimated within 35 village landscape components that were then aggregated into village-scale estimates and compared using Monte Carlo uncertainty analysis and a data quality index. Our results demonstrate with 76% probability that village soil and sediment N storage has increased from 1930 to 1994, most likely by 25% over the 1930s level, or ~1.4 Mg N/ha on average. A 20% increase in agricultural soil N concentrations caused more than half of the increase, potentially improving soil fertility. Sediment accumulation in village canals, ponds, and marshes caused the remaining N storage increase, after sediment use for fertilizer ended in 1982, increasing the risk of flooding and impeding irrigation. Sedimentation at current rates will fill most canals within 25 years, and N concentration in agricultural soils may now be declining. Compounding these problems, village food security is threatened by a 30% decline in agricultural soil N per person since 1930 and a doubling in the proportion of village soil N under buildings and infrastructure, from 5% in 1930 to 11% in 1994. Village landscapes in the Tai Lake Region sequestered 1.7 Tg N and 16 Tg C between 1930 and 1994, forming a significant regional sink that may become a source of atmospheric C and N emissions, if organic N use continues to decline.

Key words: agroecology; anthropic soils; anthropogenic ecosystems; China; ecological history; land use; landscape ecology; sedimentation; soil nitrogen sequestration; soil organic matter; sustainable agriculture; wetlands.

INTRODUCTION

Nitrogen limits primary productivity in most terrestrial ecosystems, including agroecosystems (Richardson 1952, Vitousek and Howarth 1991). To overcome this limitation and maintain food security, densely populated agricultural regions in developing nations now use synthetic N fertilizers to boost yields, resulting in groundwater pollution, N saturation of aquatic ecosystems, and reactive N emissions to the atmosphere (Smil 1991, Matthews 1994, Ma 1997). To reduce these problems, methods must be developed to sustain high yields without overloading agroecosystems with N.

Traditional agriculture in China's Tai Lake Region sustained exceptionally high rice (*Oryza sativa*) yields for centuries under N-limiting conditions without high N inputs, water pollution, or declines in productive

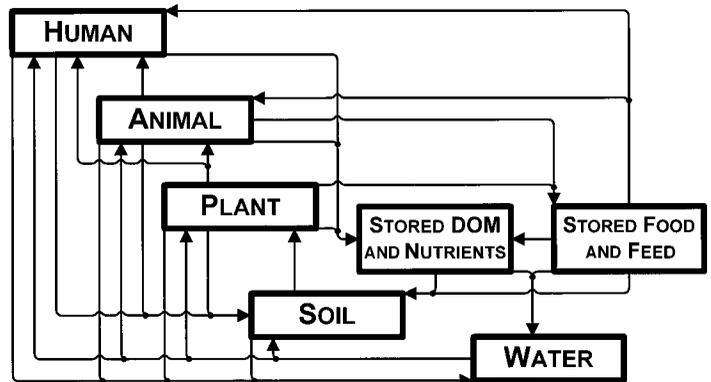
capacity (Ellis and Wang 1997). Though the region's current N-intensive practices have doubled yields and support twice the population, N overapplication and pollution are now endemic in the region (Ellis and Wang 1997). We hypothesize that high traditional rice yields were sustained by efficient recycling of N resources while current N-intensive methods have decreased this recycling, coupling high N inputs with high outputs to the environment.

We can test this hypothesis by comparing current and traditional village-scale N cycling and storage in the Tai Lake Region. Village-scale comparisons are necessary because traditional farmers recycled N-rich materials like manure, biomass, and sediments across village landscapes to increase agroecosystem productivity (Ellis and Wang 1997). We compare village-scale N cycling using a standard seven pool model (Fig. 1) because cycling measures are sensitive to model structure (Finn 1976). Human, animal, and plant pools correspond to basic trophic levels while decomposers are included within the soil, water, and "stored" pools. We differentiate "stored food and feed" and "stored dead

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FIG. 1. Seven-pool compartmental model of village-scale N cycling. DOM = dead organic matter. Only pool-to-pool fluxes are shown (input/output fluxes are not shown).



organic matter (DOM) and nutrients" pools (DOM = manures, plant biomass, compost, and other material) because these pools govern food security and fertilizer availability in traditional systems.

In this article, we begin our comparison of village-scale N cycling by comparing the size of soil N pools within a single village under traditional (~1930) vs. contemporary (1994) management in China's Tai Lake Region (Ellis et al. 2000). Most N within terrestrial ecosystems is stored in soil organic matter pools, so that small changes in soil N storage can transform ecosystems from sinks to sources of N and C, with potentially large impacts on global biogeochemistry (Simpson et al. 1977, Stevenson 1986, Galloway et al. 1995). Soil N is sensitive to changes in fertilization, irrigation, cropping systems, and tillage (Stevenson 1982), and all of these practices have changed in China's transition from traditional to modern management (Luo and Han 1990).

This analysis focuses on the soil N pool measurable using total Kjeldahl nitrogen analysis (TKN; Bremner and Mulvaney 1982). This "soil TKN pool" accounts for the bulk of soil N in most soils, including both organic and inorganic N pools (not including nitrate; Stevenson 1986). The village soil N pool also includes a "belowground biomass pool" (roots and macrofauna typically removed prior to TKN analysis) and a "surface DOM pool" (soil surface DOM and litter usually excluded from soil samples). These pools are not included in the current analysis. Seasonal variation in soil TKN is buffered by large, slow-turnover, organic N pools (Bhuiyan 1949, Stevenson 1986), so we analyze soil TKN at a single annual time point and consider monthly and annual differences to be indistinguishable.

To investigate the sources of long-term anthropogenic change in soil N storage, we stratify village landscapes into biogeochemically distinct "ecotope" landscape components and estimate soil N within each component under traditional and contemporary management (Ellis et al. 2000). By aggregating these components into village-scale estimates, we can determine the contribution of specific management changes to the net change in soil N storage across village landscapes.

METHODS

Study site

This study was conducted in Xiejia Village of Wujin County, a typical alluvial floodplain rice/wheat (*Triticum aestivum*) subsistence village in the Tai Lake Region of Jiangsu Province (location: 31.5° N, 120.1° E; Ellis et al. 2000). The village is constructed on low-lying land (2.5–5 m above sea level) reclaimed from wetlands near Tai Lake many centuries ago. Our estimates for traditional village management describe the average state of Xiejia Village ~1930, defined as the period between 1924 and 1935, and our contemporary estimates are for 1994, defined as the period between January 1993 and December 1995. The landscape boundaries of Xiejia Village were defined in the field by village leaders in 1994, and have changed little since 1930 (Ellis et al. 2000).

Soil classification

Terrestrial soils in Xiejia Village were classified by national soils experts using field profiles in March 1994, and by reference to the Wujin County soil map (Wujin County Soil Survey Office 1986). To aid in diagnosis, soil horizons were sampled within three profiles dug in paddy fields and two in rainfed fields for analysis of percentage organic matter, TKN, and soil texture (Institute of Soil Science, Chinese Academy of Sciences 1978).

Soil and sediment boundaries

We include aerobic soils within village ecosystems from the soil surface to a depth of 40 cm. This depth was selected as a standard because (1) it includes most plant-available nutrients in soils typical of Xiejia Village (Wujin County Soil Survey Office 1986, Wuxi Municipality Agriculture Bureau 1989), (2) it is convenient for sampling, and (3) it corresponds to the rice rooting depth predicted for Xiejia paddy soils (36 cm, based on A horizon depth = 15 cm and the equation: root depth = $2.73 \times A$ horizon depth $- 5.27$, $R^2 = 0.93$, from the Institute of Soil Science, Academia Sinica 1990:681, Fig. 35.2). Submerged soil and sediment

were also included to a 40-cm depth, except when low-density sediments useful for fertilizer (average bulk density $\sim 0.6 \text{ g/cm}^3$) extended below 40 cm. In this case, sediment was considered part of village ecosystems down to the depth of the dense soil layer (bulk density $> 1.3 \text{ g/cm}^3$).

Statistical methods

Overview.—To quantify observational uncertainty in village-scale change estimates, we use probability distributions that describe our “degree of belief,” or betting odds, for the mean of each variable measured or estimated in this study. Details of this change analysis are provided in Ellis et al. (2000), but will be presented in abbreviated form here. We first parameterize probability distribution functions (PDFs) to describe our degree of belief in variables measured directly (measured variables) or by subjective methods (subjective variables), as outlined below and detailed together with the estimation method of each variable. We then use Monte Carlo methods to calculate probability densities for village-scale change estimates and other variables that are calculated as a function of other variables. Because variables range from the well sampled to the subjective to somewhere in between, we use a “data quality pedigree” system to quantify and express data quality across all of our estimates (Costanza et al. 1992, Ellis et al. 2000).

Measured variables.—Whenever Normality is not rejected by fitting tests at $\alpha > 0.05$, we use Normal(sample mean, SE) (SE = standard error) PDFs to describe sampled estimates. (Note: we use an $f(\mu, \sigma)$ formulation instead of the conventional $f(\mu, \sigma^2)$; Ellis et al. 2000.) When Normality is rejected, we select another open-form PDF with statistically significant fit (Lognormal, Gamma, Pearson V, Weibull, Gumbel). When no open-form PDF fit to sample data, we use a Normal or other open-form PDF that best approximates the sample mean, median, variance, and range (maximum–minimum) (Ellis et al. 2000). We use PDFs of the form $f(\text{sample mean, SD})$ (SD = sample standard deviation) to represent estimates from nonrandom samples and samples not identical with the variable to be estimated, as when regional soil samples are used in place of village samples. To represent subsamples from nonrandom samples, we use PDFs in the form: $f(\text{subsample mean, subsample mean} \times \text{sample SD/sample mean})$.

Subjective variables.—For 1930s variables and others that were too costly or infeasible to measure, we describe our degree of belief using methods similar to those for parameterizing subjective prior PDFs in Bayesian statistics (Berger 1980:63, Ellis et al. 2000). We specify PDFs for subjectively estimated variables with mean, median, coefficient of variation (CV), range, and shape that characterize our degree of belief in the possible values of the variable, or we describe a histogram that was then fit to a PDF. PDFs for variables

with well-characterized measurement uncertainty are prepared using PDF families typical for comparable sampled estimates, such as Normal and Lognormal. Though we also use closed-form PDFs, such as truncated Normal and Lognormal, symmetric Beta ($(\text{Beta}(\alpha, \alpha) - 0.5) \times \text{range} + \text{average}$) and Beta-Subjective (Palisades Corporation 1996), all PDFs are prepared in accordance with the principle that they should include all possible values for a variable. We accomplish this by using PDFs with cvs 2 to 50 times larger than cvs for comparable measured variables and by increasing ranges by at least 20% over our original range estimates (Ellis et al. 2000).

Data quality pedigree.—We use a data quality pedigree system to facilitate data quality comparisons between estimates (Ellis et al. 2000). To do this, we score estimates on their statistical (stat), empirical (emp), and methodological (meth) quality using an ordinal scale from 0 to 4 (higher is better) based on a table of standards modified from Costanza et al. (1992, Ellis et al. 2000: Table 6). The resulting pedigree is expressed as a vector of the three scores {stat, emp, meth} or by a grade from zero to one = $(\text{stat} + \text{emp} + \text{meth})/12$. In general, variables with pedigrees greater than {2, 2, 2} (grade > 0.5) are sampled measurements and those below are subjective approximations, with larger values representing higher levels of statistical definition, direct observation, and use of standard methods and lower values demonstrating increased subjectivity.

Monte Carlo and pedigree calculation methods.—To calculate and compare village soil N pools and other estimates derived by calculations from other variables, we use Latin Hypercube Simulation (LHS), an advanced Monte Carlo method (Morgan and Henrion 1990:204, Ellis et al. 2000). We use medians from 10 000 LHS iterations to represent our best estimates for the mean of variables and use boxplots, cvs, and 90% credible intervals (CIN) to describe the range of most likely values for the mean of the variable (Morgan and Henrion 1990:84, Ellis et al. 2000). Multiple estimates on the same variable made by different methods are averaged using normalized inverse variance weights, or using equal weights when all estimates are subjective (grade < 0.5) (Ellis et al. 2000). Pedigrees for calculated variables are computed using a pedigree calculation algorithm (Ellis et al. 2000).

Landscape classification and area measurements

We stratify village landscapes into ecologically homogeneous and biogeochemically distinct landscape components, or ecotopes (Golley 1998:90), using the classification hierarchy: landscape \rightarrow management \rightarrow biota \rightarrow group, as summarized in Table 1 and detailed in Ellis et al. (2000). We use Xiejia Village ecotope area estimates from Ellis et al. (2000) in our current soil N storage calculations. These were estimated for 1994 using aerial photography, field sampling, household surveys, and other field methods, and for 1930

using historical land use data, 1930s aerial photography, village elder interviews, and back-estimation from 1994 areas.

Soil TKN concentration

Sampling.—We sampled aerobic soils in 15 management types grouped into six land use types as described in Table 2. We used these sample types to anticipate variability in soil N caused by fertilization, tillage, and proximity to buildings, so that differences in soil N could be detected using small samples. Each sample consisted of five, 40 cm deep, soil cores extracted using a 3.2-cm inner diameter, T-handle, soil coring tool. Soils were sampled in 19–20 May 1994 and 23–26 May 1995, prior to flooding rice transplanting paddies. Areas were defined for each sample and marked on village maps before going to the field. Paddy soil sample cores were collected from five different fields using a “Z” pattern across 2- to 18-ha sample areas. Cores for other sample types were collected from widely spaced plots or patches within sampling areas. A set of 14 paddy soil A-horizon samples (four 0–20 cm cores/sample) were also taken from a cluster of seven paddy fields in an especially fertile area of the village prior to development of experimental plots. Samples were broken up for air drying on the day of sampling. Anaerobic sediments were sampled from canal bottoms using a bucket on 7 June 1994 and from three horizons (0–5, 5–15, and >15 cm depth) after canal drainage on 19 January 1995. Sediment samples were added to tared plastic bottles and 10–20 mL 0.05 mol/L H_2SO_4 was mixed in immediately, followed by chilling and transport to the laboratory within 24 h.

Chemical analysis.—All samples were transported to the Nanjing Institute of Soil Science for analysis. Prior to chemical analysis, aerobic soils were oven dried for 3–4 h at 60°C followed by 100°C for 1.5–2 h, while sediments were air dried for 3–5 d, crushed and oven dried for 6 h at 80°C followed by 6 h at 105°C. Large roots and macrodebris were removed and samples were ground to pass through 0.25-mm mesh. TKN was determined using the micro Kjeldahl method (Bremner and Mulvaney 1982:610–616), and %C was determined using the Walkley-Black method (Nelson and Sommers 1982:570–571) using 0.133 mol/L $K_2Cr_2O_7$ and a sample size from 0.1 to 0.5 g. Soil percent organic matter (%OM) was calculated by multiplying %C by 1.724 (Commission of Agricultural Chemistry, Soil Science Society of China 1983).

1994 ecotope soil TKN estimates.—Sample year effects were nonsignificant ($P > 0.3$) so samples were pooled across years (Table 2) (tested by General Linear Model (GLM), SPSS 1995). As samples did not differ significantly from Normality, we used Normal PDFs to parameterize soil TKN estimates. For sample types that were identical to ecotopes, we parameterized ecotope soil TKN using Normal(sample mean, SE) PDFs. When ecotope definitions were similar to but not identical

with sample types, or when sample type TKN differed significantly within ecotopes, we used the sample mean for the most similar sample type, or we averaged the sample types within the ecotope, to make Normal(sample mean, sample mean \times cv) PDFs using the maximum cv for the land use type (Table 2). For terrestrial ecotopes that did not overlap with any sample type, we used a Normal PDF with the grand mean of all soil samples and a cv equal to twice the maximum cv of all the sample types (30%). Our anaerobic sediment sample was small and highly skewed, so we used our sample mean in a Lognormal(mean, range/4) PDF with range derived from the minimum and maximum measurements found in Chinese reference sources and increased by 20% of the mean (cv = 40%; Zang et al. 1974, Lu and Shi 1982, Beijing Agricultural University 1985, Luo and Lin 1991). Anaerobic soil was not sampled, so TKN was parameterized using a Beta-Subjective PDF with mean set to the grand mean of soil sample TKN and mode adjusted to give a 50% cv, bounded by zero and a conservative maximum (5 g N/kg).

1930 ecotope soil TKN estimates.—Paddy soil A horizon TKN \sim 1930 was estimated from 11 published data points calculated from 24 samples in the Tai Lake Region of Jiangsu Province before synthetic N use became common in the mid-1960s (Chen 1930, Tschau et al. 1934, Wujin Municipality Agriculture Bureau 1959, Xu 1964, Jiangsu Province Soil Survey Working Group 1965, Wujin County Soil Survey Office 1986). We combined 1930s measurements for Tai Lake Region paddy soil ($n = 3$) with 1950–1959 measurements for the same paddy soil type as Xiejia ($n = 8$), because their difference was nonsignificant (one-tailed t test $P = 0.23$). We then discarded a single outlier from this sample because it was more than two times the interquartile range over the 75th percentile (0.22 g N/kg, Wuxi County \sim 1959; Jiangsu Province Soil Survey Working Group 1965). This point was also a high outlier when combined with a 1994 Xiejia paddy A horizon sample ($n = 17$) that had a much higher mean than the \sim 1930 sample. Because the sample represents regional paddy soils, not those of Xiejia Village, we parameterized our Xiejia 1930 soil A horizon TKN estimate using a Normal PDF with the 10-point sample mean and sd. We also produced a PDF including the outlier to test the effects of this sampling decision on village scale calculations.

We extrapolated 40-cm paddy soil TKN from our 1930s A horizon PDF using slope and intercept parameters derived by linear regression of paired 40-cm and A horizon TKN data from three paddy profiles in Xiejia Village combined with four 1980s profiles for the same soil type in Wujin and Wuxi counties (Wujin County Soil Survey Office 1986, Wuxi Municipality Agriculture Bureau 1989). The regression parameters were parameterized as Normal(mean, SE) PDFs and used for extrapolation even though the regression was not sta-

TABLE 1. Ecotope key and soil N storage per hectare.

Ecotope label†	Land- scape‡	Management§	Biota	Description	Soil N storage (Mg N/ha)	
					1930	1994
Dwellings	T	constructed	human	residential buildings, including outbuildings	7.5 (1.8, 14.5)¶	7.4 (3.9, 11.2)
Chicken house			animals	confined chicken produc- tion, medium scale	...	7.4 (3.9, 11.2)
Public buildings			sealed	industry, government, edu- cation and commerce buildings	...	6.0 (3.0, 9.3)
Roads, paved				paved roads and covered transportation infrastruc- ture	...	6.0 (3.0, 9.3)
Irrigation, sealed				sealed irrigation ditches and agricultural infrastruc- ture	...	6.0 (3.0, 9.3)
Roads, unpaved			bare	unpaved roads, paths and unsealed access areas	6.1 (1.4, 11.9)	6.0 (3.0, 9.3)
Irrigation, unsealed				unsealed earthen irrigation ditches	5.9 (1.4, 11.5)	5.8 (4.2, 7.6)
Bare earth				bare, compacted earthen agricultural infrastruc- ture	5.9 (1.4, 11.4)	5.8 (4.2, 7.6)
Paddy, double crop		paddy	annual	rice paddy, double crop- ping	5.2 (1.1, 10.0)	6.8 (5.6, 7.9)
Paddy, transplant- ing				rice paddy, transplanting area, often single crop- ping	...	5.4 (4.4, 6.4)
Orchard, irrigated		irrigated	perennial	irrigated orchard, <i>Prunus persica</i> dominant	...	5.8 (5.0, 6.5)
Vineyard, irrigated				irrigated vineyard, <i>Vitis</i> sp. dominant	...	5.8 (5.0, 6.5)
Field borders		rainfed	annual	paddy field border crops and weeds	4.7 (1.1, 9.1)	5.8 (4.7, 7.1)
Crops, rainfed				rainfed annual crops, small-scale	4.6 (1.1, 8.7)	5.6 (5.1, 6.2)
Orchard, rainfed			perennial	rainfed orchard, <i>Prunus persica</i> dominant	4.7 (1.1, 8.8)	5.8 (5.1, 6.5)
Vineyard, rainfed				rainfed vineyard, <i>Vitis</i> sp. dominant	...	5.8 (5.0, 6.5)
Mulberry				rainfed mulberry, <i>Morus alba</i> dominant	4.7 (1.1, 8.9)	...
Annual weeds		fallow	annual	annual weeds, larger patches	7.0 (1.6, 13.5)	6.8 (3.6, 10.4)
Tall grasses			perennial	tall grass thicket, <i>Miscan- thus sinensis</i> dominant	6.4 (1.5, 12.3)	6.3 (3.3, 9.5)
Bamboo				Bamboo thicket, <i>Phyllos- tachys nigra</i> dominant	8.0 (1.9, 15.4)	7.9 (6.0, 10.0)
Brush and weeds				Brush and weeds	7.0 (1.6, 13.5)	6.8 (3.6, 10.4)
Trees, medium				Medium trees, saplings and scrub	7.0 (1.6, 13.5)	6.8 (3.6, 10.4)
Trees, mature				Mature trees, closed cano- py, mostly deciduous	7.0 (1.6, 13.5)	6.8 (3.6, 10.3)
Graves				Grave vegetation, perenni- als and annuals	7.0 (1.6, 13.4)	6.8 (3.6, 10.3)
Public trees				Roadside area, government planted <i>Metasequoia glyptostroboides</i> sap- lings, crops and annual weeds	...	3.9 (2.1, 6.0)
Wetland crops	M	rainfed	annual	lentic wetland; nonrice rooted emergent and floating leaf crops	6.3 (2.3, 13.2)	15.6 (6.4, 40.8)
Wetland vegetation		fallow	annual	lentic wetland: rooted, emergent and floating leaf vegetation	...	15.6 (6.4, 40.8)
Pond margins	PM	mixed	mixed	pond margins: variable vegetation and livestock	6.4 (3.0, 11.2)	7.6 (4.4, 13.6)
Fishponds	PA	animal	animals	pond depths: fish, herbivo- rous, small-scale	6.7 (2.3, 13.3)	10.3 (3.3, 32.9)

TABLE 1. Continued.

Ecotope label†	Land- scape‡	Management§	Biota	Description	Soil N storage (Mg N/ha)	
					1930	1994
Pond, crops		rainfed	annual	pond depths: nonrooted, floating crops	...	8.1 (2.7, 38.3)
Pond, fallow		fallow	annual	pond depths: floating and submergent vegetation	...	8.2 (2.8, 24.7)
Canal margins	CM	mixed	mixed	canal margins; variable vegetation and livestock	6.4 (3.0, 11.2)	8.0 (4.5, 15.0)
Canal, crops	CA	rainfed	annual	village canal depths: non- rooted, floating crops	6.6 (2.3, 12.6)	13.7 (4.6, 39.2)
Canal, fallow		fallow	annual	village canal depths: float- ing and submergent veg- etation	6.6 (2.3, 12.6)	13.6 (4.7, 38.7)
County canal, fallow	CB	fallow	annual	county canal depths: float- ing and submergent veg- etation	6.6 (2.3, 12.6)	12.7 (4.3, 37.6)

† Ecotope labels used in charts (Ellis et al. 2000: Table 5).

‡ Landscape class (Ellis et al. 2000: Table 1). T = terrestrial plain, M = marsh, PM = pond margin, PA = pond, CM = canal margin, CA = village canal, CB = county canal.

§ Management class (Ellis et al. 2000: Table 2).

|| Biota class (Ellis et al. 2000: Table 3).

¶ Median and 90% credible intervals (CIN) (in parentheses) estimated for soil and sediment TKN storage per hectare.

tistically significant ($R^2 = 0.27$, $P = 0.19$) because this facilitated use of 1930s sample data in parameter estimation. The 55% cv of the resulting 40-cm TKN estimate was similar in magnitude to the cv of a “worst case” sample from our Xiejia 1994 40-cm soil sample set ($n = 72$; SD/mean for the two lowest and the highest point or vice versa). We therefore used this as the stan-

dard cv for all 1930s soil TKN PDFs, as this was sufficient to cover the full range of 40-cm soil TKN estimates. By assuming that soil TKN in all cropped ecotopes changed in parallel with the 20% paddy soil N increase, we specified soil TKN PDFs for 1930 rainfed crop ecotopes using their 1994 means in Normal(1994 mean \times 0.8, 1994 mean \times 55%) PDFs. We

TABLE 2. Total Kjeldahl N in oven dry soil, sorted by land use and then by management, 1994 and 1995 samples combined.

Land use and management	<i>n</i>	Mean† (g N/kg)	95% confidence limits‡ (g N/kg)	cv§ (%)
Bamboo (a)	3	1.44 ^a	1.11–1.77	12
Fallow area (ab)	6	1.26	1.08–1.43	7
near buildings	3	1.36 ^{ab}	1.23–1.49	5
near canals	3	1.15 ^{bcd}	0.842–1.46	14
Paddy (bc)	23	1.15	1.08–1.21	3
rapeseed	3	1.26 ^{abc}	1.14–1.38	5
wheat	14	1.20 ^{bc}	1.14–1.26	2
transplanting area¶	6	0.965 ^{ef}	0.852–1.08	6
Field border (bc)	3	1.08 ^{def}	0.929–1.24	7
Perennial crops (bc)	10	1.08	1.01–1.15	3
intercropped	5	1.11 ^{cde}	1.04–1.18	3
monocropped	5	1.05 ^{def}	0.927–1.17	6
Rainfed, annual crops (c)	26	1.06	0.985–1.13	3.5
vegetables	10	1.15 ^{bcd}	1.03–1.27	5
fallow	3	1.12 ^{bcd}	0.795–1.44	15
rapeseed + broadbean	4	1.03 ^{cdef}	0.846–1.21	9
broadbean	3	1.01 ^{cdef}	0.931–1.10	4
rapeseed	3	0.973 ^{def}	0.852–1.09	6
wheat	3	0.866 ^f	0.713–1.00	8.5
All samples	71	1.12	1.08–1.17	2

† Means of management types with different letters are statistically significantly different ($P < 0.05$) as tested using least significant differences (LSD) from GLM ($P = 0.001$; SPSS 1995).

‡ Two-tailed Z confidence limits.

§ SE/mean.

|| Land use types with different letters in parentheses are statistically significantly different ($P < 0.05$) as tested using LSDs from GLM ($P = 0.007$).

¶ Paddy transplanting area and winter fallow.

used our 1930 rainfed orchard estimate to represent the mulberry ecotope. For all other ecotopes, we prepared soil TKN PDFs with the same mean as in 1994, but with cvs increased to 55%.

Soil and sediment density

We measured soil horizon thickness and bulk density in all horizons within the top 40 cm of soils in four rainfed crop sites, one paddy site, and two fallow land sites (near buildings and near canals). To maximize differences between samples, we distributed sample sites nonrandomly across the village in areas with divergent vegetation cover and management types. In paddy and rainfed crop sites, we paired three A horizon samples with single samples from each lower horizon, while a single sample was taken from each horizon in fallow sites. Three 100-cm³ soil cores were taken per sample from each horizon using 5 cm deep steel rings hammered vertically into horizons. Cores were trimmed, removed from rings, transported in tared plastic bags, and weighed. Soil core water content was determined by adding 2–5 g soil to tared steel containers, weighing, burning in 95% ethanol three times, and reweighing. Soil bulk density was calculated as core fresh mass \times sample dry mass/sample fresh mass/100 cm³. The 40-cm soil layer density was then calculated by averaging the three core densities per horizon sample and weighting the horizon densities by horizon thickness/40 cm.

Though soil A horizon density differed significantly between management and vegetation types in all sample sites and all horizons differed according to management in paddy sites (GLM, $P > 0.001$), 40-cm soil densities did not differ significantly ($P = 0.13$) due to the small sample size and the homogeneity of lower horizons. Sample cvs (SE/mean) were all $< 1\%$, but samples were small, nonrandom, and not specific to ecotopes, so we parameterized ecotope 40-cm soil densities as Normal PDFs using sample means and 10% cvs for paddy and fallow ecotopes and 5% cvs for rainfed and irrigated ecotopes (the rainfed sample was largest, $n = 12$). To parameterize soil densities for unmeasured terrestrial ecotopes, we specified a BetaSubjective PDF using the mean, maximum, and minimum of all of the 40-cm soil density samples ($cv = 10\%$). A BetaSubjective PDF for aquatic sediment density was prepared from mean, minimum, and maximum density data in Munsiri (1995; $cv = 40\%$, range increased by 20%). A BetaSubjective PDF for anaerobic soil density (Fig. 2) was parameterized using mean and range data from all Xiejia B and W horizon samples (range increased by 20%; $n = 30$; $cv = 10\%$). We used identical soil and sediment density PDFs for 1930 and 1994, assuming that the change over this period was much less than uncertainty in these estimates.

Soil and sediment volume

Soil volume for terrestrial ecotopes was calculated by multiplying ecotope area by a 0.4-m soil depth con-

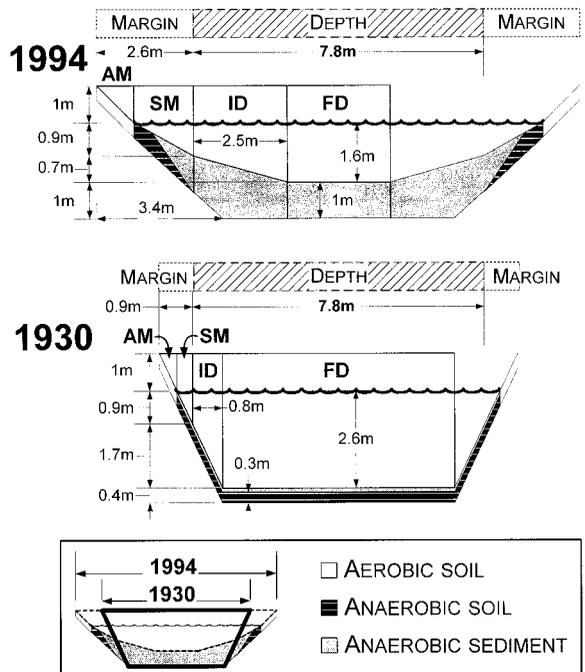


FIG. 2. Typical village canal cross-section geometry at seasonal maximum water level, 1930 and 1994. Dimensions are the average of 13 canal cross-section samples. Margin and depth zones of canals are illustrated, as are soil and sediment geometry. Nested canals inside the box at bottom illustrate the relationship between canal shape in 1930 and in 1994. Canal cross-section components: AM = aerobic margin, SM = submerged margin, ID = inclined depth, FD = flat depth. Soil in aerobic margins is 0.4 m deep in both 1930 and 1994.

stant. To estimate soil and sediment volumes for 1994 aquatic and wetland ecotopes, data from field and indirect methods were combined. We parameterized village canal dimensions using Normal PDFs for sample data generated by polynomial fits to field measurements from 13 village canal cross sections surveyed in 1995 (Fig. 2; Ellis et al. 2000). Village canal sediment thickness in 1994 (depth to density > 1.3 g/cm³) was estimated as ~ 1 m deep from observations on drained canals (BetaSubjective, $cv = 30\%$), and as ~ 0.1 m deep in 1930 by elder village informants who collected sediments at that time (BetaSubjective, $cv = 50\%$). By coupling these estimates with the approximation that traditional sediment harvests in the 1930s gave canals a trapezoidal shape, and that canal margins were one-third the 1994 width in 1930 ($cv = 50\%$), we described village canal geometry as illustrated in Fig. 2.

Marsh and pond cross sections were surveyed in a single location each. County canals were assumed to be 1 m deeper than village canals in both 1930 and 1994 (BetaSubjective, $cv = 50\%$) with the same sediment depth as village canals. Marsh maximum water depth was estimated as 0.8 m (BetaSubjective, $cv = 36\%$), and roadside pond depth was estimated as ~ 2.5 m (BetaSubjective, $cv = 50\%$), with no sediment ac-

cumulation. Sediment depth in managed ponds was estimated as three-fourths canal sediment depth (BetaSubjective, $cv = 37\%$) and marsh sediment thickness was calculated as the difference between 1930 canal depth and 1994 marsh depth (~ 1.8 m thick) because marshes developed from the filling in of canals.

To estimate soil and sediment volumes in canals, we divided them into the four cross section components illustrated in Fig. 2. Similar components were used for county canals, ponds, and marshes, though marshes had no ID component. We derived soil and sediment cross section (XS) areas for the aerobic margin (AM), submerged margin (SM), and inclined depth (ID) components using PDFs for canal geometry variables in trigonometric calculations. Soil and sediment volumes for AM, SM, and ID components were calculated as the product of XS area and perimeter while flat depth (FD) volumes were calculated as sediment thickness \times (depth ecotope area - ID width \times perimeter). The volume of soil eroded from canal banks from 1930 to 1994 was calculated as the difference between the 1994 and 1930 total canal margin XS areas (not including soil and sediment) multiplied by the 1994 canal margin perimeter. We assumed that all AM components contained 100% aerobic soil, that all 1994 ID and FD components were 100% anaerobic sediment, and that 1994 SM components were 1:1 sediment: anaerobic soil (Fig. 2). SM, ID, and FD components from 1930 were assumed to be 1:4 sediment: anaerobic soil (0.1 m sediment depth/0.4 m standard soil depth).

Soil N and C pool size calculations

We calculated ecotope soil TKN pools by multiplying ecotope-level estimates of soil TKN concentration (kilograms N/kilograms dry soil) by bulk density (kilograms dry soil/cubic meters) and volume (cubic meters). Soil TKN per hectare of terrestrial ecotopes was calculated by multiplying soil depth, density, and TKN concentration. Soil TKN per hectare of wetland and aquatic ecotopes was calculated by dividing ecotope soil TKN pool size by ecotope area. Village and landscape class soil TKN pools were calculated by adding ecotope estimates. The change in soil TKN pool size from 1930 to 1994 was calculated by subtracting 1930 values from 1994 values, so that positive differences connote increases from 1930 to 1994. We tested our decision to discard an outlier from the 1930 paddy soil A horizon TKN sample by recalculating the 1930 to 1994 village soil N pool difference using a PDF including the outlier, which had a 9% higher median.

We used C:N ratios to calculate C storage changes for terrestrial soils using a Lognormal PDF with the standard mean for Huangnitu paddy soil (9.7; Office for Soil Survey of China 1993:614), and 15% cv , increased over the 10% cv for the population of all available Huangnitu A horizon C:N samples ($n = 60$). We used our village sediment sample to make a Lognormal

PDF for the C:N ratio of sediments, also with a 15% cv , though the direct population cv estimate was 8%.

RESULTS

Soil characteristics

Xiejia Village soils are formed of alluvial deposits of loess-based yellow-brown earth (Xiong and Li 1987: 221) that classify as Udifluvents in the Entisol order (Foth 1990). Under the Chinese national classification system, all village terrestrial soils classify as "Huangnitu" (yellow clay) periodically submergic (redoxic) paddy soils based on their "A-P-W" horizonation (Xu et al. 1980, Office for Soil Survey of China 1993:614) even though some areas within the constructed, fallow and rainfed crop land-use types had A-B horizons more typical of nonpaddy soils.

Soil TKN concentrations ranged from 0.7 to 1.7 g N/kg across our 40-cm sample of Xiejia terrestrial soils (Table 2), differing significantly between land use types (GLM $P = 0.007$) and between management types (GLM $P = 0.001$). Uncultivated soils generally had the highest TKN concentrations (Bamboo and Fallow management types), though double-cropped paddy soils also had relatively high TKN (~ 1.2 g N/kg; Table 2). Soil organic matter content paralleled TKN, so that soil C:N ratios were tightly constrained across all of our 1994 terrestrial soil samples ($n = 154$, $\bar{X} = 10.2 \pm 0.7$ SD). Village anaerobic sediments had higher TKN concentrations ($n = 9$, $\bar{X} = 2.3 \pm 0.1$ SE g N/kg) and C:N ratios ($\bar{X} = 12.7 \pm 0.3$ SE) than terrestrial soils.

Changes in soil N concentration

Pre-1980 soil and sediment N concentration data were too few in nonagricultural areas of the Tai Lake plain to test for long-term change. However, sufficient data were available to test whether Tai Lake Region paddy soil N concentration has changed since the late 1950s. Paddy soil A horizon N was lower in the 1930s (1929 to 1959 pooled sample, $n = 10$, $\bar{X} = 1.12$ g N/kg) than in 1982 (Huangnitu samples from Wuxi and Wujin counties, $n = 7$, $\bar{X} = 1.45$ g N/kg; Wujin County Soil Survey Office 1986, Wuxi Municipality Agriculture Bureau 1989) and 1994 (Xiejia Village, $n = 17$, $\bar{X} = 1.56$ g N/kg) when tested using Bonferroni-protected least significant differences (LSD $\alpha = 0.05$, GLM $P < 0.001$; SPSS 1995). The 1982 and 1994 samples were indistinguishable.

By converting the 1930s paddy soil A horizon data to a 40-cm estimate, 1930 paddy soil TKN concentration in Xiejia Village was estimated to be 0.95 g N/kg, with a 90% CIN of 0.2–1.8 g N/kg and a pedigree of {2, 2, 1}. When this estimate was subtracted from our 1994 measurement (median = 1.18 g N/kg, CIN = 1.14, 1.22; pedigree = {4, 3, 3}), we observed a concentration increase of $\sim 20\%$ over the 1930 value, or ~ 0.2 g N/kg (CIN = -0.6, 1.0; pedigree = {2, 2, 1}), with a 69% chance that the difference was greater than zero.

Changes in sediment storage

Soil and sediment N storage in village wetland and aquatic ecotopes increased from 7 Mg N/ha in 1930 (CIN = 3, 12; pedigree = {0, 1, 0}) to 11 Mg N/ha in 1994 (CIN = 5, 29; {1, 1, 1}), a total increase of 5 Mg N/ha (CIN = -4, 23; pedigree = {0,0,0}). Bank erosion of canal margins from 1930 to 1994 released about 43 Gg of soil to canals, containing ~47 Mg N (CIN = 21, 84; Fig. 2). This loss nearly equaled the total mass of canal sediments in 1994 (~51 Gg), but could account for only about half of the N stored in canal sediments (110 Mg N; CIN = 40, 280). Based on the assertion by village informants that bank erosion and sediment accumulation began when sediment harvest and canal maintenance ended in 1982, we estimate annual sediment deposition in canals as 0.4 Gg sediment·ha⁻¹·yr⁻¹ (CIN = 0.2, 0.8) with a total N deposition of 0.9 Mg N·ha⁻¹·yr⁻¹ (CIN = 0.4, 2.5) and a depth of 7 cm/yr (CIN = 4, 13). Though bank erosion could account for nearly all of the sediment mass annually deposited in canals, ~0.5 Mg of N·ha⁻¹·yr⁻¹ (CIN = -0.1, 2) was sequestered in canal sediments by some other process, with a 91% chance that this N sequestration was greater than zero.

Village-scale change in soil N storage

The soil TKN pool of Xiejia Village contained 0.9 Gg N in 1930 (CIN = 0.4, 1.4; {1, 1, 1}) and 1.1 Gg N in 1994 (CIN = 0.96, 1.3; {2, 2, 2}; Fig. 3A). By subtracting these estimates using LHS, we deduce a 76% probability that the soil TKN pool of Xiejia Village increased from 1930 to 1994, most likely by ~0.2 Gg N (CIN = -0.3, 0.7; {1, 1, 1}; Fig. 3B). This represents an average increase of 1.4 Mg N per hectare of village area (CIN = -1.9, 4.4), and is equivalent to a 25% increase over the 1930 village soil TKN pool.

In tandem with the village soil N increase, the distribution of soil N among ecotopes changed significantly from 1930 to 1994 (Fig. 4). In 1930, >70% of village soil N was in the two largest ecotopes, paddy and mulberry, and soil N storage per hectare was distributed evenly (Fig. 4, Table 1). By 1994, <70% of village soil N was in the paddy, rainfed crop, and dwelling ecotopes, and the distribution of soil N was uneven, with aquatic and wetland ecotopes storing much more soil N/hectare than terrestrial ecotopes (Fig. 4, Table 1). Construction increased the percentage of village soil N under buildings and infrastructure from 5% in 1930 to 11% in 1994. Patterns of change in soil N storage per hectare and per ecotope are illustrated in Figs. 5 and 6, respectively.

Sources of village-scale change.—Three processes changed village soil N storage from 1930 to 1994: sediment accumulation, increased N concentration in agricultural soils, and altered landscape structure (Figs. 4–6). Village N storage in wetland and aquatic soil and sediment increased by 0.09 Gg (CIN = -0.01, 0.3)

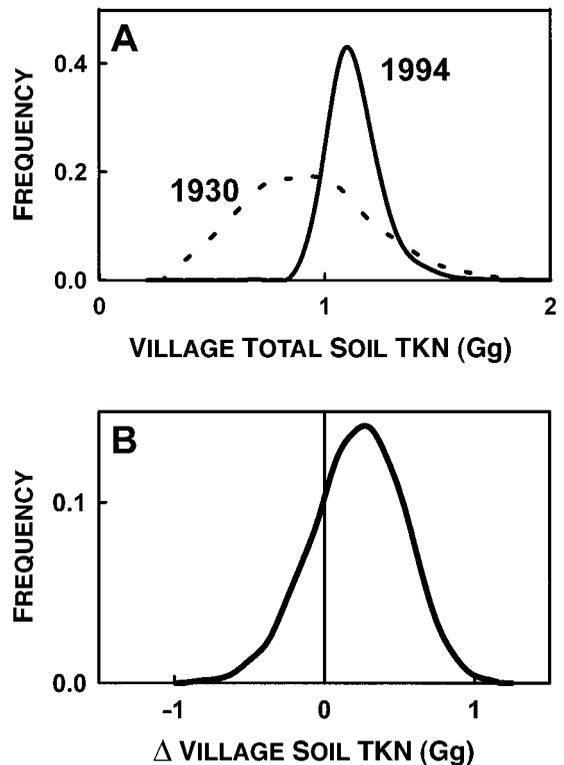


FIG. 3. Changes in Xiejia Village soil total Kjeldahl nitrogen (TKN) pool size from 1930 to 1994. Frequency distributions in (A) represent our degree of belief in specific values for the size of the village soil TKN pool in 1930 and 1994, as marked, while the distribution in (B) represents the probability of a difference between these pools. Positive values in (B) represent an increase since 1930.

from 1930 to 1994, accounting for about half of the village soil N increase over this period. Sediment accumulation was the most important contributor to this increase, as per hectare soil and sediment N storage rose by 180% over 1930s levels. Still, canal bank erosion and new pond construction enlarged the total area of wetland and aquatic ecotopes by 2% of the total village area, boosting wetland and aquatic area by 38% over the 1930 area (Ellis et al. 2000). Area expansion combined with sedimentation increased soil and sediment N storage in canals, ponds, and marshes from ~6% of village soil N in 1930 to ~15% in 1994 (Fig. 4).

Terrestrial soil N storage increased by 14% from 1930 to 1994, adding 0.12 Gg N (CIN = -0.4, 0.6) to the village soil N pool and causing the other half of the village soil N increase (Fig. 6). Driving this increase was a 21% rise in agricultural soil N/ha, a 1.2 Mg N/ha (CIN = -2.8, 4.6) increase over the 1930 level (5.2 Mg N/ha, CIN = 1.8, 9.1). However, the total agricultural soil N storage increase was only ~15%, significantly lower than the soil N/hectare increase, mostly because total agricultural land declined by ~5% from 1930 to 1994 (Ellis et al. 2000).

If village landscape structure had stayed constant

from 1930 to 1994, increased agricultural soil N/hectare would have raised terrestrial soil N storage by 25% more than observed, as tested by substituting 1994 TKN values for 1930s values. About half of this 25% potential increase was eliminated by the 2% decline in terrestrial area as a proportion of total village area. A 12% decline in paddy area and 4% decline in fallow area as a proportion of village area eliminated most of the remaining potential increase (Ellis et al. 2000), because these ecotopes have higher N/hectare than most other terrestrial ecotopes.

Uncertainty in soil N storage estimates

Soil TKN and density PDFs contributed more uncertainty to village soil N pool estimates than did area and volume, because converting the latter to constants had little effect, while converting the former decreased the cvs for N pool and change estimates by more than half and boosted the likelihood of a village soil N increase to >99%. Six variables dominated uncertainty in our 1930 village soil N pool estimate, as indicated by normalized multivariate stepwise regression coefficients >0.1 (overall $R^2 = 0.94$) (Morgan and Henrion 1990:208). Paddy soil variables were most influential (40 cm TKN prediction slope and intercept, A horizon TKN, and density) with lesser influence by mulberry TKN and total village area. In 1994, the seven most influential variables were sediment and paddy soil density, sediment TKN and depth, dwelling soil TKN, village area, and fallow soil density (overall $R^2 = 0.91$). These same thirteen variables dominated uncertainty in our village soil N change estimate, with the 1930s variables contributing much more uncertainty than 1994 variables. Converting the six key 1930s PDFs to constants reduced the cv of our village soil N change estimate from 134% to 58% without changing its median, raising the probability of an increase to 97%. Converting the key 1994 PDFs had almost no effect, while just reducing the cv of our 1930 paddy soil TKN estimate from 55% to 10% reduced the village soil N change cv to 73%, boosting the probability of an increase to 92%. Still, our estimate of village soil N change was not greatly affected by discarding the outlier in our 1930 paddy soil A horizon TKN sample (6% lower median, 3% lower probability of positive change).

DISCUSSION

Long-term change in anthropogenic agricultural soils

Paddy soil N concentration increased by ~20% in Xiejia Village from 1930 to 1994. We did not expect soil N to increase, because organic fertilizer use has declined dramatically in the Tai Lake Region since 1930 (Ellis and Wang 1997) and paddy soil N usually decreases when synthetic N is used without organic N inputs in the region (Li 1991, Ma and He 1993, Wang

et al. 1993). The increase may partly be explained by our methods, which compared a 1994 village sample with a 1930 regional sample, because paddy soil N concentrations near Xiejia are 4–8% higher than average for the Tai Lake Region and Wujin County (Wujin County Soil Survey Office 1986, Institute of Soil Science, Academia Sinica 1990:556). Still, our estimated soil N increase agrees with the 30% increase in Wujin County paddy soil N from 1959 to 1983 (Wujin County Soil Survey Office 1986:88) and similar increases observed near Shanghai during the 1970s (Xi 1981) and across South China since the 1930s (Lindert et al. 1996).

Long-term increases in paddy soil N may have resulted from several factors. First, the use of synthetic N in N-limited traditional agroecosystems probably boosted soil organic N storage by increasing the production of roots, litter, and soil biomass (Paustian et al. 1997). Alternatively, the exceptionally high organic N inputs used in the Tai Lake Region from the 1950s to early 1970s, combined in later years with synthetic N, may have resulted in a legacy of elevated soil N (Ellis and Wang 1997). If organic N inputs were responsible for the increase from 1930 to 1994, then soil N levels may currently be declining because of recent trends away from organic inputs (Ellis and Wang 1997).

Evidence for current trends in Tai Lake Region paddy soil are equivocal. Unpublished reports of the Soil and Fertilizer Station of the Jiangsu Department of Agriculture and Forestry indicate a 20% N increase in Tai Lake Region paddy soils from 1985 to 1996. Conversely, Wujin County paddy soil N declined by 7% from 1973 to 1983 (Wujin County Soil Survey Office 1986:88) and declines have been observed during the late 1980s across the Tai Lake Region (Huang 1994) and in a Zhejiang village (Wang and Shao 1994). Xiejia Village paddy soil N was slightly higher in 1994 than the 1983 averages for Wujin and Wuxi counties, but this difference was nonsignificant (t test $P > 0.15$) and subject to sampling bias.

Soil N and sustainable village management

The transition from traditional to contemporary village management has increased the storage of soil and sediment N with both potentially beneficial and harmful effects. Increased soil organic N may have improved soil fertility and increased crop yields, though rice yields can decline even when soil N increases (Cassman et al. 1997). On the other hand, elevated soil organic N levels may increase the risk of N loss in runoff, nitrate leaching and denitrification, with aerobic soils in rainfed and irrigated agroecosystems posing an especially high risk of polluting village drinking water with nitrate (Meisinger and Randall 1991).

Sediment accumulation.—In 1982, decollectivization of agriculture ended communal canal maintenance and sediment harvest for fertilizer in the Tai Lake Region, primarily because synthetic fertilizers are cheaper

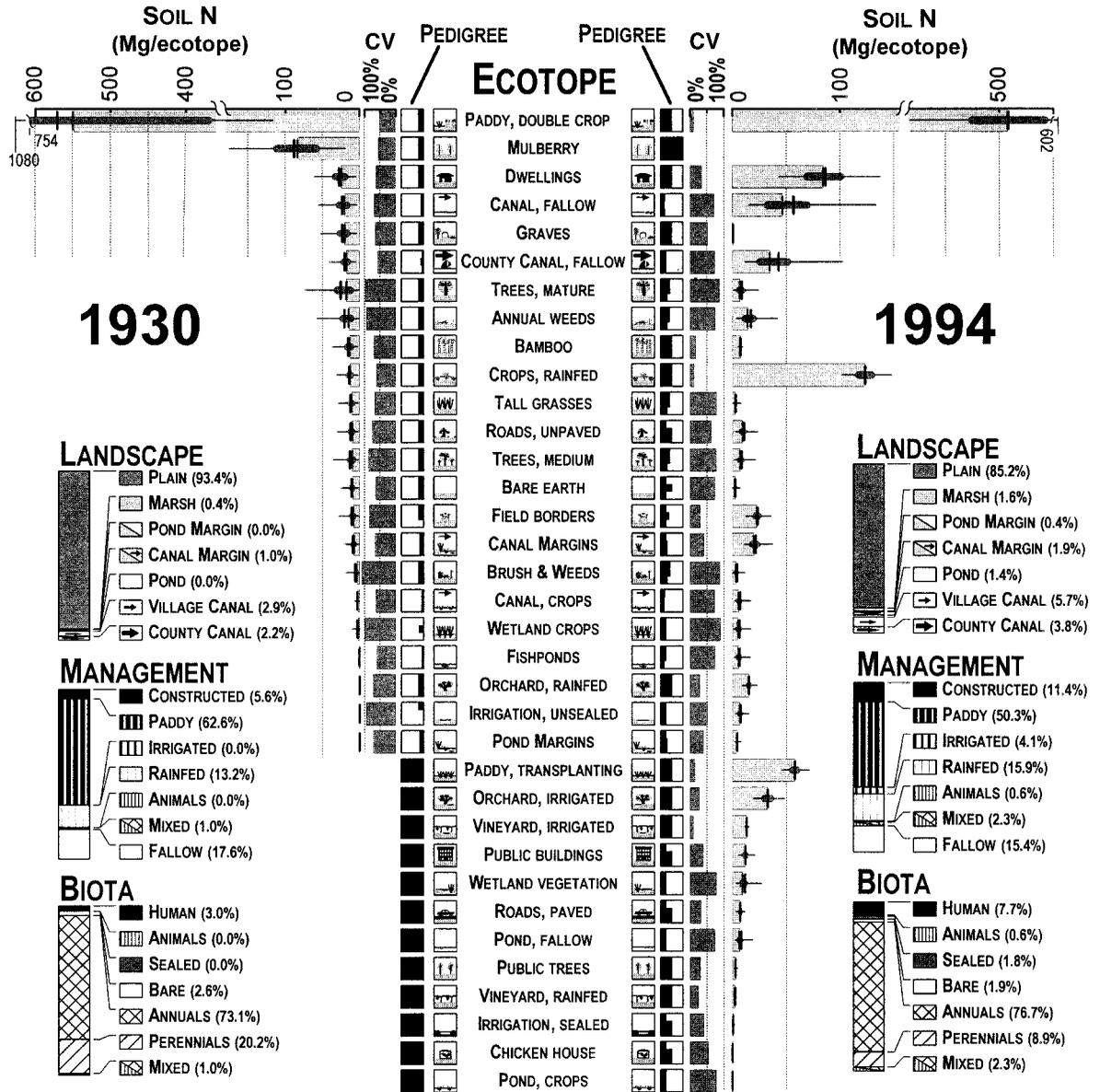


FIG. 4. Soil TKN in ecotopes and landscape classes in Xiejia Village, 1930 and 1994. Ecotopes are sorted by median soil TKN per ecotope in 1930, largest to smallest, top to bottom, and then by 1994 medians for ecotopes not present in 1930 (black pedigree boxes). Ecotopes are represented by icons labeled as in Table 1. Pedigree boxes describe data quality of ecotope soil TKN estimates; cv bars represent their coefficients of variation. In soil TKN charts, light gray horizontal bars describe medians; rounded dark gray boxes cover interquartile ranges; black and dark gray vertical lines depict means and medians, respectively; and whiskers are drawn to 5th and 95th percentiles. When the interquartile ranges of estimates do not overlap, they are different with $\geq 75\%$ probability; this probability is $>95\%$ when whiskers do not overlap. Bar charts at the bottom left and right of the figure show the percentage of village soil TKN in landscape, management, and biota classes (Table 1).

and easier to apply (Ellis and Wang 1997). Since then, sediment has been accumulating in canals at an annual rate of ~ 7 cm/yr, sequestering ~ 0.9 Mg N \cdot ha $^{-1}$ ·yr $^{-1}$ in a total of ~ 400 Mg sediment \cdot ha $^{-1}$ ·yr $^{-1}$. Our estimate is similar to accretion rates in aquaculture ponds (1–30 cm/yr; Boyd 1995:225, Munsiri et al. 1995) and restored wetlands near rice paddies (0.4–0.9 Mg N \cdot ha $^{-1}$ ·yr $^{-1}$; Comín et al. 1997), but is greater than the

maximum for natural wetlands (~ 2.6 cm/yr; Johnston 1991) and bed/ditch cropping systems (0.4 Mg N \cdot ha $^{-1}$ ·yr $^{-1}$; Luo and Lin 1991). Village soil N storage was increased by sediment accretion because we included low-density sediments in our definition of soil depth. Therefore, our village accounting of sediment N will eventually decline due to subsidence, even if canals continue to fill.

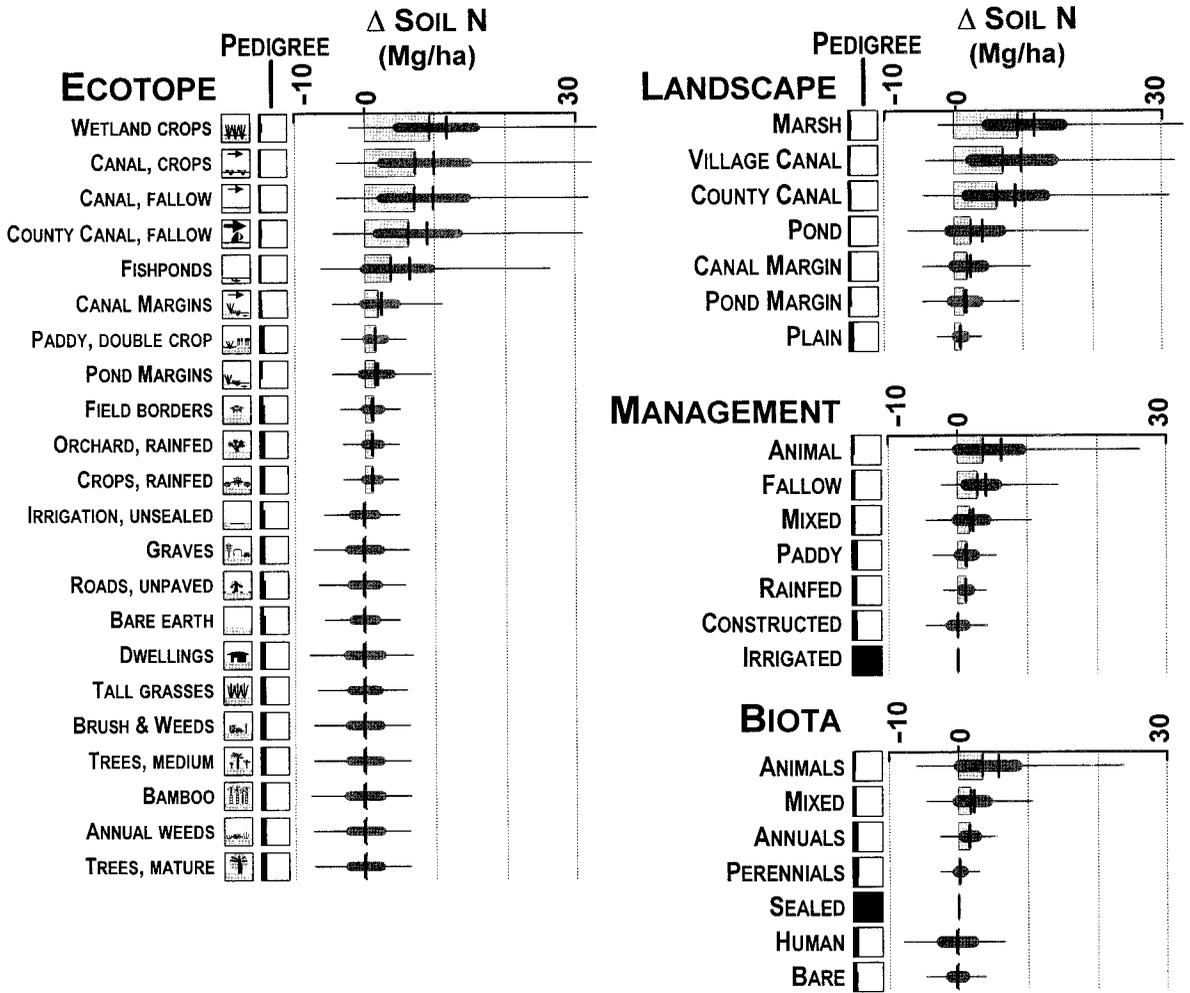


FIG. 5. Change in soil TKN per hectare in ecotopes and landscape classes in Xiejia Village from 1930 to 1994. Ecotopes are sorted by their median change per hectare, largest to smallest, top to bottom. Ecotopes not present in 1930 or 1994 are omitted. Positive changes indicate increases since 1930. Pedigree boxes, cvs, and boxplots are as described in Fig. 4. Bar charts at the right illustrate changes in soil TKN per hectare within landscape, management, and biota classes (Table 1).

Village canal sedimentation is blamed for excessive damage caused by the floods of 1991 (Bu 1994), and sediments blocked irrigation canals during a 1994 regional drought, causing local conflict over responsibility for canal maintenance. Ironically, these same sediments were once so valuable that farmers quarreled over rights to harvest them for fertilizer. Traditional use of sediment fertilizers raised soil levels by as much as 0.5 cm/yr in a process of “reverse erosion” that is thought to contribute to the exceptionally high long-term agricultural productivity of the Tai Lake Region (King 1911, Gong 1981). Elder village informants estimated that 8 cm of sediment was harvested annually for compost in the 1930s. Though canal bank erosion has increased sediment deposition substantially in recent years, an 8 cm/yr accumulation is plausible, given that 1930s sediments were only about half as dense as 1994 sediments (twice as thick per unit mass), according to informants. Based on these rough estimates, an-

nual village sediment use could have been as high as 5 Mg N/yr in the 1930s, providing up to 50 kg N/ha to paddy fields.

Canal sedimentation is an unsustainable process. If canals and marshes are allowed to fill completely, their role in trapping runoff N and P will be lost, furthering the decline of aquatic ecosystems across the region. As canals now average ~1.6 m deep, most canals will fill completely within 25 years if accretion continues at current rates. Removal of sediments would buffer against flooding, improve canal traffic, and improve soil structure, though sediments would release about half their nitrogen to the atmosphere and/or groundwater following exhumation, because terrestrial soils have about half the N content of anaerobic sediment. Ideally, sediments would again be used for fertilizer, though traditional methods are far too labor intensive for current conditions.

Population and N in village landscapes.—The pop-

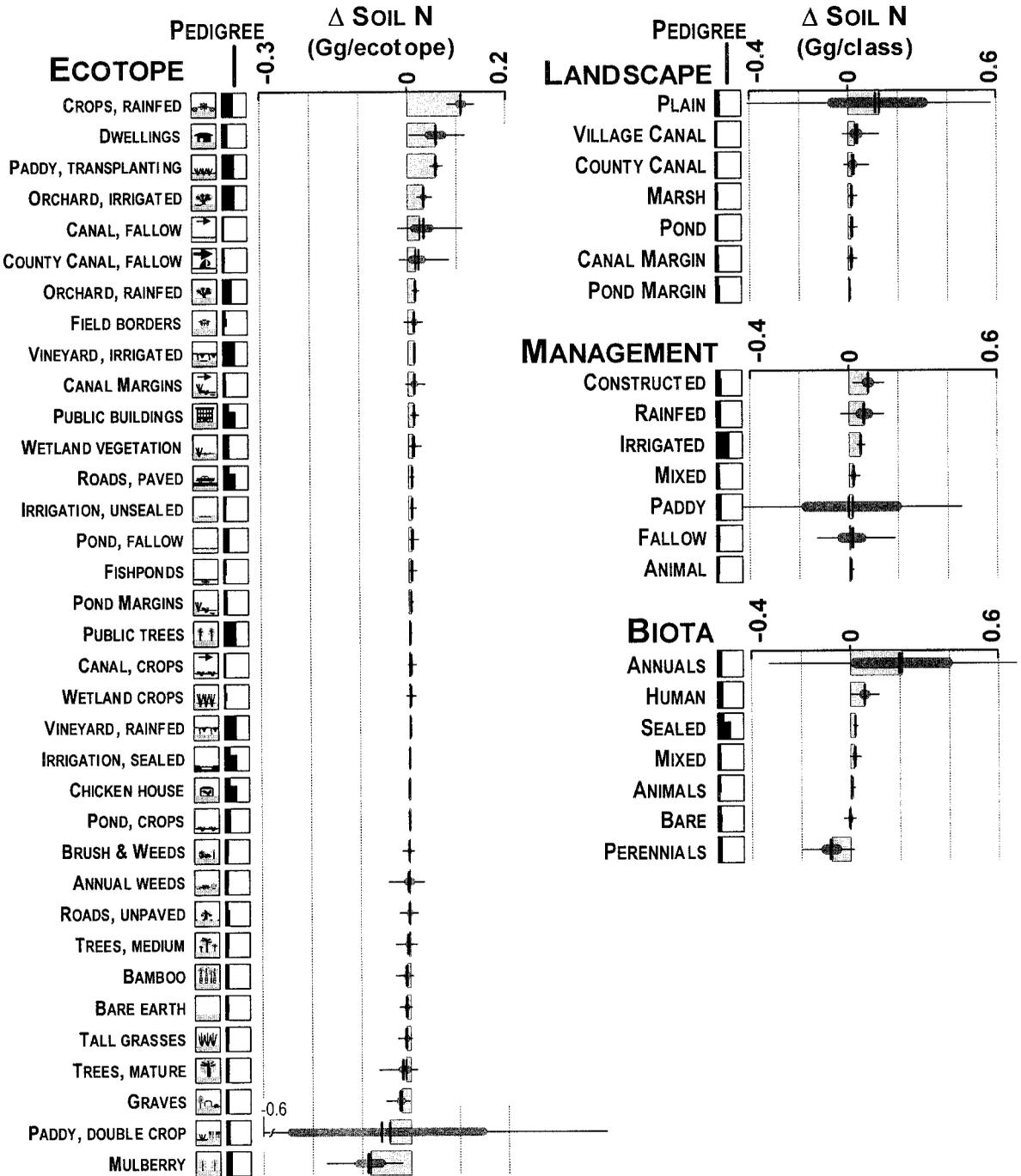


FIG. 6. Change in soil TKN per ecotope and per landscape class in Xiejia Village from 1930 to 1994. Ecotopes are sorted by their median change per hectare, largest to smallest, top to bottom. Positive changes indicate increases since 1930. Graph components are identical to those in Fig. 5, except that total soil TKN change per ecotope and per landscape class is substituted for change in soil TKN per hectare.

ulation of Xiejia Village doubled from 1930 to 1994 while areas of paddy and agricultural land decreased (Ellis et al. 2000). As a result, village, agricultural, and paddy soil N per person declined relative to 1930 by 30%, 34%, and 43%, respectively. Over the same period, new construction increased the percentage of vil-

lage soil N under buildings and infrastructure from 5% to 11%, reducing potential primary productivity and sequestering soil N for the long term (Ellis et al. 2000). As soil N is a form of natural capital (Clark 1992), the 5% decline in village soil N under agriculture makes sense, considering the falling importance of agricul-

tural income and the low price for synthetic N in the Tai Lake Region.

Village paddy soil N storage did not change significantly from 1930 to 1994 because increased soil N concentration compensated for decreased paddy area: village paddy soil N would have been ~20% lower without the concentration increase. As grain from household paddy land supplies Tai Lake Region villagers with most of their dietary protein and energy (Ellis and Wang 1997), and soil organic N is a key factor in rice yield (Watanabe et al. 1981), paddy soil N per capita may be a more robust indicator of village food security than per capita paddy area. Both paddy soil N per capita and rice yield per capita have declined since 1930, suggesting long-term threats to village food security in the Tai Lake Region (Ellis and Wang 1997).

Anthropogenic landscapes as sinks for N and C

Simpson et al. (1977) have described anthropogenic changes in ecosystem N loading and sediment accretion as potential controls on atmospheric N and C, but most studies have focused on biogeochemical change in undisturbed ecosystems. This study describes long-term changes in the biogeochemistry of a highly disturbed anthropogenic landscape undergoing the transition from traditional to modern management. Our evidence for increased N storage in terrestrial soils applies directly to >2000 km² of Huangnitu paddy soils in China, and more generally to paddy soils in China (25 300 km²), and to densely populated rice/wheat subsistence village landscapes in subtropical and temperate regions (~4 × 10⁶ km²; Whittlesey 1936). Village sediment accretion, however, is limited to areas that have recently abandoned sediment use for fertilizer, such as the Tai Lake and Pearl River Delta Regions.

We estimate that the top 40 cm of terrestrial soil in Xiejia Village has accumulated 0.9 Mg N/ha since 1930, with a 90% CIN of (-2.4, 3.9) and a 69% probability of a net increase. Assuming no change in C:N ratio and allowing for uncertainty, village terrestrial soils have accumulated 8 Mg C/ha (CIN = -20, 40) since 1930. This terrestrial sink was driven by even greater per hectare increases in agricultural soil N and C (1.2 Mg N, 11 Mg C) that have overridden potential C emissions caused by conversion of paddy land to rainfed crops and other uses (Cai 1996). Averaged over 60 years, annual per hectare N and C accumulation in village agricultural soils was 0.02 Mg N and 0.2 Mg C, similar to C sequestration in abandoned agricultural soils in temperate zones (0.03–1 Mg C·ha⁻¹·yr⁻¹; Paus-tian et al. 1997). Using a conservative estimate of terrestrial village area within the Tai Lake Plain (uniform, 0.5–1.5 × 26 500 km² area, from: Gong 1988), we estimate a 68% chance that the region's terrestrial soils were a sink for N and C, most likely sequestering 1.7 Tg N (CIN = -5, 8.5) and 16 Tg C (CIN = -46, 82) since 1930. Averaged over 60 years, this is only 28 Gg N and 270 Gg C per year. However, trends in soil N

and C were most likely not constant over this period and paddy soils may have changed from sinks to sources in the past 10 to 20 years because of changes in organic and inorganic fertilizer management. If N storage in fallow areas has increased as a carryover effect of synthetic N subsidy of agroecosystems, our assumption of 1994 fallow levels in 1930 has caused us to underestimate long-term change in soil N and C storage.

Since 1982, village canals, ponds, and marshes have annually accumulated 0.9 Mg N (CIN = 0.4, 2.5) and 12 Mg C (CIN = 5, 33) per hectare, with a >99% chance that these areas are sinks, not sources. Based on estimates of canal bank erosion, only about half of sediment N and C accumulation should be considered a net accumulation. Using conservative estimates of canal, marsh, and pond areas for the Tai Lake Region (uniform, 0.2–2 × Tai Lake Plain area × the average canal, marsh, and pond proportional areas in Wujin and Wuxi Counties: Gong 1988, Zhu and Xu 1988, Tan 1994), annual sediment accumulation in the region's wetland and aquatic ecosystems is 0.04 Tg N (CIN = 0.01, 0.2) and 0.5 Tg C (CIN = 0.1, 2). We assumed that sediment N content in the 1980s was the same as in 1994 (1.8–2.9 g N/kg), because older data were similar (1.3–2.9 g N/kg; Zang et al. 1974), and sediment N is more a function of C availability than N inputs (Johnston 1991). If sediment N concentration has increased, we have underestimated this sink.

Evidence and uncertainty in biogeochemical budgets

Sensitivity analysis demonstrated that our 1930 paddy soil N estimate was the greatest source of uncertainty in estimating long-term change in village soil N pools and that soil TKN and density were greater sources of uncertainty than were area and volume estimates. Short of new evidence from 1930, this accurately represents the state of available knowledge and puts an upper limit on inference about village soil N in this period. Sensitivity analysis also yielded recommendations for measurements on several present-day variables with major influence on uncertainty in village soil N storage (paddy soil density and canal sediment depth, density, and TKN). These variables also had low pedigrees (grades < 0.5), indicating that improving these measurements should not be difficult. Our estimates of the probability of accumulation vs. decline in village soil and sediment N pools were sometimes as low as 68%. This realistically portrays our uncertainty in these trends and presents results in the form of "betting odds" (70:30, 3:1) that are useful in guiding decision-making when absolute certainty is not possible.

Our estimates of long-term change in village soil and sediment N and C sequestration are derived from complex mixtures of direct measurements, calculations, and rough estimates, as are global biogeochemical budgets (Galloway et al. 1996, Vitousek et al. 1997). As results from biogeochemical budgeting are usually reported as

points or ranges, without indicators of probabilistic uncertainty or data quality, our estimates may appear to be of lower quality or greater uncertainty than those in the literature. We believe this demonstrates success in using observational uncertainty analysis to calculate and visualize the uncertainty inherent in estimating long-term change in complex anthropogenic landscapes.

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