ENVIRONMENTAL IMPACTS OF APPLYING MANURE, FERTILIZER, AND SEWAGE BIOSOLIDS ON A DAIRY FARM[1]

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ABSTRACT: Farms that once spread only manures are now also applying sewage biosolids (sludge) and/or other wastes such as those from food processing. The objective of this study was to monitor environmental impacts at a dairy farm applying these materials. Fields were selected representing recent waste applications of manure (M1, M2), sewage biosolids (B1, B2), or fertilizer only control (F1, F2), although most fields had historical biosolids applications. Fields representing each treatment were not experimental replicates because of varying applications and soil characteristics. Septage and food processing wastes were also applied. Soil percolates were collected with wick lysimeters. Runoff was sampled at seven stream sites. Test field soils and alfalfa (*Medicago sativa*) were analyzed for trace elements. Cumulative trace metal loadings were low, at most only 1 percent of USEPA Part 503 limits. Surface soil enrichment was most evident for Mo, P, and S. Alfalfa tissue showed no trends of concern. The B2 site had the greatest percolate concentrations for 6 of 13 elements. Percolate Cu was somewhat elevated at Sites M1, M2, B2, and F1. Percolate sodium was elevated on all M and B fields and sulfur was greatest at M2, B1, and B2. Soluble orthophosphate correlated with stream discharge during intensive monitoring of Stream Sites S1 (fertilizer) and S2 (biosolids). Peaks in S2 streamwater Mo lagged large runoff events by five days. Total streamwater export of Cu, Na, Mo, and soluble P were greater from the S2 biosolids subwatershed than from the S1 fertilizer subwatershed. Percolate concentrations exceeded corresponding streamwater concentrations in most cases.
(KEY TERMS: nonpoint source pollution; sludge; waste/sewage treatment; runoff; percolate; farm management.)

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INTRODUCTION

With increased emphasis on recycling and reuse of organic wastes, farms that once dealt solely with animal manures are also land applying waste streams such as food processing wastes and sewage biosolids (sludge). A concurrent trend in agriculture is the concentration of more animal units per area of cropland. Taken together, the increased organic waste loadings on croplands can result in greater environmental impacts, affecting both on-site soil quality (via accumulation of contaminants) and increased concentrations of nutrients and contaminants in runoff and/or soil percolate. Historic overapplication of P through fertilizers and organic wastes has contributed to the potential for off-site impacts by surface runoff and, more recently recognized, by subsurface transport (Sims *et al,*. 1998).

On a watershed scale, McFarland and Hauck (1999) were able to show a strong correlation between instream storm event N and P concentrations and several indices of agricultural use in the watershed, including percent area in intensive agriculture, percent of area applied with dairy wastes, and dairy cow density. Similarly, Hunter *et al.* (2000) found a correlation between agricultural land uses (sheep stocking densities and application of manures and sewage biosolids) and surface water fecal coliforms. Randall *et al.* (2000) concluded that application of dairy

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manure on a clay loam soil did not lead to greater N and P leaching if it was applied at optimal rates and immediately incorporated.

In terms of sewage biosolids contribution to nutrient cycling and potential losses, the primary focus of research has been on N, with some attention given to P (see reviews of Krogmann *et al.,* 1997, 1998, 1999). Recent studies by Elliott *et al.* (2002) and Penn and Sims (2002) found that P leaching from laboratory soil/amendment mixtures was lower for sewage biosolids than from fertilizer or poultry waste. Siddique *et al.* (2000) likewise found greater P leachability from fertilizer versus sewage biosolids amended columns. Maguire *et al.* (2000) found that mean oxalate-extractable P levels in soils from 11 field sites were doubled by biosolids application, but observed that oxalate-extractable Al and/or Fe often increased, mitigating potential P release.

At the small watershed scale, Grey and Henry (2002) noted that a single 13.5 tons per hectare (tons/ha) application of sewage biosolids to 40 percent of a forested watershed had little observable impact on runoff P, but resulted in an increase in runoff NO_3-N after nine months (with concentrations up to 1 mg/l correlating with runoff events). Hart and Nguyen (1994) reported that application of 10 tons/ha of liquid digested sludge to aspen saplings resulted in significant increases in NO_3-N, with ground water monitoring wells peaking at 4.3 mg/l.

In addition to desirable nutrients and organic matter, sewage biosolids contain potentially toxic trace elements and synthetic organic chemicals (Hale and LaGuardia, 2002; National Research Council, 2002). While substantial progress has been made in reducing these levels (National Research Council, 2002; Stehouwer *et al.,* 2002), trace element concentrations in sewage biosolids remain greater than those in most fertilizers or manures (Raven and Loeppert, 1997; McBride and Spiers, 2001). Even comparable concentrations in sewage biosolids and fertilizers would result in far greater contaminant loadings from sewage biosolids due to greater bulk application rates (tons/ha magnitudes compared to kg/ha for fertilizers). The long term fate of trace elements in sewage biosolids has been the subject of multiple studies (Alloway and Jackson, 1991; Dowdy *et al.,* 1991; Barbarick *et al.,* 1998; Richards *et al.,* 1998; Sloan *et al.,* 1998; Baveye *et al.,* 1999; McBride *et al.,* 1999), with many reporting immobility but others observing loss and/or leachability of at least a portion of applied trace elements.

The objective of the present study was to monitor environmental impacts (percolate and streamwater concentrations, soil enrichment, and crop uptake of trace elements and nutrients) associated with the use of soil amendments including fertilizer, manure, and biosolids on a commercial dairy farm.

SITE HISTORY AND MONITORING APPROACH

Overview

The study was performed on the Leo Dickson and Sons farm (Bath, New York) a large (850 head) commercial dairy farm in the central southern tier of New York State. This farm has a consistent record of land application of sewage biosolids since 1978. The farm has somewhat distinct areas of fields where recent primary waste applications are manure (closest to the main barn), sewage biosolids, or no wastes (fertilizer only) on the fields farthest from the farm center. It should be noted that working farms are economic enterprises and that maintaining equivalent treatments on fields so that they can serve as experimental replicates is not a realistic priority. Most fields have had some history of sewage biosolids applications. Nevertheless, two fields each were selected for which the recent primary management pattern has been manure application, sewage biosolids application, or fertilizer application.

Soil percolate was sampled using passive wick lysimeters (Boll *et al.,* 1992), which are able to passively extract percolate from unsaturated soil. Their use also avoided the potential problem of adsorption of trace metals by ceramic porous cup lysimeters (McGuire *et al.,* 1992; Wenzel *et al.,* 1997): in prior work excellent agreement in percolate trace metal concentrations between in-field wick lysimeters and soil columns from the same site was observed (Richards *et al.,* 1998, 2000). Stream sites were sampled to determine element mobility in runoff. The present distribution of trace metals within the soil was examined in vertical profiles, and alfalfa growing on test fields was sampled.

Site History and Soil Types

All corn (*Zea mays*) grown on the Dickson and Sons farm is used for silage. Alfalfa is grown as well. Farm wide average lime application rates are 0.4 tons/ha per year. Starter fertilizer is used on all corn fields at a rate of 196 kg/ha of 15-15-15, resulting in elemental applications of 29 kg/ha N, 13 kg/ha P, and 24 kg/ha K. Fields not receiving organic wastes are applied with an average of 336 kg/ha of 15-15-15 (50 kg/ha N, 22 kg/ha P, and 42 kg/ha K); actual application rates are based on commercial soil testing.

Two fields from each of the three soil management patterns (primarily manure, primarily sewage biosolids, and primarily fertilizer) were selected and designated M1/M2, B1/B2, and F1/F2 respectively. Note that these two fields cannot be regarded as experimental replicates of each management pattern because of differences in soil characteristics and history. Soil characteristics and taxonomic information are summarized in Table 1. Existing commercial soil testing results (Spectrum Analytic Inc.; samples processed in mid to late fall) for soil characteristics and Mehlich 3-extractable nutrients are summarized in Table 2 (italics indicate calculations). As is typical of long term dairy farms in the Northeast U.S., available P already exceeded maximum recommended levels for all M and B treatments, while K levels were adequate for half the plots.

Land Application Practices

Manure from the dairy barn is scraped frequently during the day and includes sawdust used as freestall bedding. Manure is stored in a lagoon along with milkhouse wastewater, and is subsequently land-applied at 8 to 10 percent total solids at a rate of 65,000 to 75,000 l/ha. Beginning in 1998, cheese processing waste (whey) was added to the lagoon on a daily basis so that it comprised two-thirds of the volume spread (1/3 manure, 2/3 whey; 6 to 8 percent TS).

TABLE 1. Soil Series Information for Sampled Fields (USDA-SCS, 1978) and Taxonomic Classifications (USDA-NRCS, 2001).

Field	Soil Series and Slope	Soil Taxonomy	Depth, inches (cm)	Comments
M1	Lordstown Channery Silt Loam (12 to 20 percent)	Coarse loamy, mixed, active, mesic Typic Dystrudepts	> 36 (91) to bedrock	Glacial till, moderate permeability
M2, B1	Fremont Silt Loam, (2 to 8 percent)	Fine loamy, mixed, semiactive, acid, mesic Aeric Endoaquepts	32 (81) to fragipan	Strongly acid glacial till, somewhat poorly drained
B2, F1	Volusia Channery Silt Loam (8 to 15 percent)	Fine loamy, mixed, active, mesic Aeric Fragiaquepts	15 (38) to fragipan	Strongly acid glacial till, somewhat poorly drained
F2	Mardin Channery Silt Loam (8 to 15 percent)	Coarse loamy, mixed, active, mesic, Typic Fragiudepts	19 (48) to fragipan	Strongly acid glacial till, moderate permeability

TABLE 2. Commercial Soil Testing Results: Soil Characteristics and Mehlich 3-Extractable Nutrients (mean of fall 1995 and 1997 samplings).

	Field					
	M-1	**M-2**	**B-1**	**B-2**	**F-1**	**F-2**
CEC (cmol/kg)	13	18	13	11	11	12
Soil pH	6.3	5.4	7.0	6.6	6.9	6.2
Organic Matter (percent)	4.0	3.8	3.3	3.7	3.4	3.4
Available P (kg/ha)	473	223	380	161	41	66
Ratio	*3.5*	*2.0*	*3.0*	*1.2*	*0.3*	*0.5*
Available K (kg/ha)	897	715	620	227	237	163
K Saturation (percent)	8.2	4.7	5.3	2.4	2.5	1.6
Ratio	*1.6*	*1.2*	*1.1*	*0.5*	*0.5*	*0.3*
SO4-S (mg/kg)	30	40	32	27	24	17
Zn (mg/kg)	9	9	10	5	3	4
B (mg/kg)	0.2	0.2	0.4	0.2	0.2	0.1

Note: "Ratio" (italicized to signify calculated) indicates the ratio of actual available nutrients to the maximum optimum level recommended for the field. A ratio > 1.0 indicates excess nutrients.

The reported 70,000 l/ha land application rate and solids contents resulted in dry matter loading rates of 8 tons/ha/y for manure (relevant to years prior to addition of processing wastes) and 3.5 tons/ha/y for the mixed manure/food processing wastes. While most spreading precedes planting, some manure was spread in winter.

Land application of sewage biosolids, first used in 1978 to improve fallow fields, takes place from approximately April 1 until the end of May on fields being planted for corn. Application then shifts to fields being prepared for forage seeding in August and, following that, to fallow land. Both liquid and dewatered sewage biosolids are collected from various treatment plants throughout the region of the state. Stabilization processes used vary widely by plant, with some lime stabilization of liquid biosolids performed at the farm. Of the wastes land applied on this farm, sewage biosolids are by far the best documented, with annual analysis and application summary reports filed in compliance with state regulations. Mean application rates in 1999 (LaBella Associates, 1999, unpublished report number 20084) were 3.8 tons/ha on 76 ha for dewatered sludge, and 4.6 tons/ha on 107 ha for liquid wastes. These loadings account for 31 percent (dewatered sludge) or 70 percent (liquid biosolids and other wastes) of corn N requirements.

Table 3 summarizes the known cumulative application history through the 2000 cropping season. Test fields were selected on the basis of recent total mass loadings. However, cumulative biosolids metals loadings indicate widely varying application histories, presumably due to application of higher concentration wastes at some point. Note that only the M2 field was entirely free of historical biosolids applications. Overall rates are low, with maximum (B2) levels only on the order of 1 percent of USEPA Part 503 cumulative limits (USEPA, 1993). Actual trace element application rates from manure can be estimated from values shown later. While Raven and Loeppert (1997) found that rock phosphates (along with sewage biosolids) had the greatest trace element concentrations, research at Cornell University (McBride and Spiers, 2001) has indicated that fertilizers used in New York have relatively low levels of trace contaminants. This, in combination with the fact that fertilizer mass loadings are an order of magnitude lower than biosolids or manures, suggests that trace element loadings from fertilizers were negligible.

The most recent sewage biosolids sources and imported masses for the entire farm are shown in Table 4. Liquid biosolids and other liquid wastes applied to fields more distant from the main dairy barn are a mixture of liquid biosolids, liquid food processing waste, liquid manure from a heifer barn and septage. These wastes are mixed and lime-stabilized prior to application. Table 4 shows the recent relative contributions from various sources.

Recent mean sewage biosolids analyses from independent certified laboratories are shown in Table 5. All biosolids applied on this farm are classified under New York state standards as class B biosolids. Concentrations for regulated trace elements were all well below the Part 503 "Exceptional Quality" levels. Also shown are composite grab sample results for dairy manure (January 2001 sampling) and mixed dairy manure/cheese processing waste lagoon (30 percent animal manure and 70 percent processing wastes; mean of 1998 and 2001 samplings). The cheese processing waste was reported to have a pH level between 4.5 and 5.7 and solids content of 6.5 to 7.5 percent TS. Cow manure analysis is included to show the impact of the processing waste, particularly on elevating Na and Cu and diluting Zn.

TABLE 3. Cumulative Historical Test Field Sewage Biosolids Loadings: Number of Annual Loadings (as of 2000) and Cumulative Trace Metal Loadings as of the End of 1999 (kg/ha).

Field	Loadings	Cd	Cr	Cu	Pb	Ni	Zn
		cumulative kg/ha					
B1	15	0.06	0.28	5.52	0.56	0.54	7.45
B2	11	0.35	2.71	20.42	3.79	1.80	28.29
M1	4	0.15	1.00	5.44	1.72	0.86	6.28
M2	0	0	0	0	0	0	0
F1	4	0.07	0.92	5.03	1.15	0.93	10.66
F2	6	0.16	1.58	8.53	1.61	1.24	15.29
USEPA Part 503 Cumulative Limit		39	3,000	1,500	300	420	2,800

Source: LaBella Associates (1999, unpublished Report No. 20084) and USEPA (1993).

TABLE 4. Recent Farm Wide Cumulative Imports and Sources of Sewage Biosolids.

	1997	1998	1999
Dewatered Sewage Biosolids			
Wet Mass (metric tons)	2,289	2,072	2,248
Dry Mass (metric tons)	434	434	550
Liquid Biosolids and Other Wastes*			
Volume (m^3)	12,491	10,977	9,841
Dry Mass (metric tons)	467	394	293
Contributing Sources (percent of volume)			
Municipal Sewage Biosolids	48	60	26
Food Processing Waste	28	9	34
Heifer Manure	13	21	31
Septage	11	10	9

*Liquid wastes also include food processing, heifer barn, and septage inputs (LaBella Associates, 1997, 1998, 1999, unpublished Report Nos. 98052, 99090, and 20084, respectively).

ANALYTICAL MATERIALS AND METHODS

Soil Water Sampling

The locations of the selected test fields (two for each waste application scenario: M1/M2, B1/B2, and F1/F2) are shown in Figure 1. On each test field a 10 m by 10 m sampling plot was delineated for the installation of two passive wick lysimeters (Boll *et al.,* 1992; Richards *et al.,* 1998). Installation was facilitated by backhoe excavation of a 2 m deep trench just downhill from the sampling plot. Samplers consisted of an enclosed box (34 by 34 cm square, 1 m tall), with 25 fiberglass braided wicks (0.9 m long) (Pepperrell Braiding Co., Pepperrell Massachusetts) suspended vertically from the upper surface. Samplers were installed 0.6 to 0.7 m below the soil surface in 0.7 m long horizontal tunnels excavated laterally from the

TABLE 5. Recent Sewage Biosolids Analyses and Composite Grab Sample Results of Dairy Manure and Mixed Lagoon.

Parameter*	EPA Part 503 EQ Limits	Liquid Biosolids/Wastes	Dewatered Biosolids	Dairy Manure	Mixed Lagoon**
pH		6.1	7.1	8.4	4.3
TS Percent		3.6	25.2	12.4	4.8
		mg/kg (dry weight basis)			
As	41	13.1	7.5	0.34	0.50
B		–	–	21.9	11.72
Ca		–	–	18,600	17,800
Cd	39	4.4	2.9	0.23	0.29
Cr	1,200	17.9	23.9	2.95	2.82
Cu	1,500	368	521	92	592
Hg	17	2.1	2.1	–	–
K		6,919	2,408	31,000	31,700
Mg		–	–	6,220	4,550
Mn		–	–	213	74.6
Mo		13.2	13.0	2.68	2.12
Na		–	–	11,360	27,300
Ni	420	17.7	20.1	10.3	11.8
P		11,454	24,665	6,380	9,960
Pb	300	39.8	62.3	1.19	1.56
S		–	–	5,530	3,777
Sr		–	–	44.9	39.6
Zn	2,800	388	816	210	94.7
TAN		7,738	1,748	–	–
TKN		40,036	23,927	–	–
Total PCB		4.0	1.1	–	–

*TAN = total ammoniacal N; TKN = total Kjeldahl N; Total PCB = total polychlorinated biphenyls.
**Dairy manure/cheese processing waste lagoon (30 percent manure and 70 percent processing wastes).
Notes: Sewage biosolids analyses are the means of 1997 to 1999 annual mass weighted mean values from LaBella (1997, 1998, 1999, unpublished Report Nos. 98052, 99090, and 20084, respectively). USEPA Part 503 EQ limits included for comparison.

trench. The samplers were installed approximately 1.5 m apart and were pressed upward against the exposed face of undisturbed subsoil. The bottom of the undisturbed soil profile was leveled and carefully handpicked to reopen flow paths disturbed by excavation.

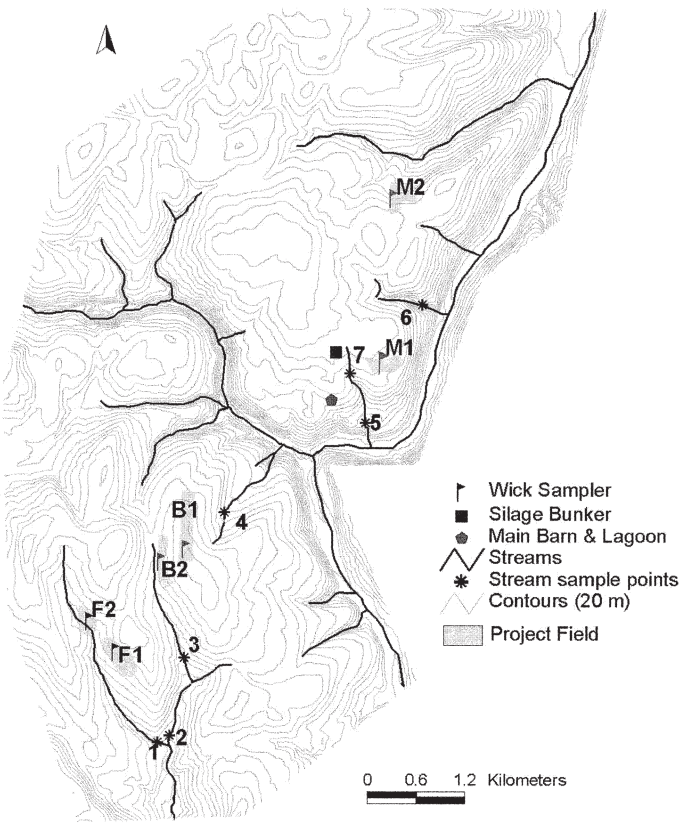

Figure 1. Contour and Location Map of Test Fields (sewage biosolids B1, B2; fertilizer F1, F2; and manure M1, M2), Farm Facilities and Surface Stream Sampling Sites.

Plexiglas plates supported by springs pressed the upper end of the wicks against the soil. Each suspended wick was encased in 13 mm (i.d.) clear PVC tubing. The vertical suspension enables the wicks to exert a matric potential (suction) on the soil in contact with the tops of the wicks, extracting soil water at moisture contents below saturation. The bottoms of the wicks were grouped into four discrete two-liter HDPE collection bottles at the base of the sampler. These bottles were evacuated periodically by HDPE sampling lines that ran to the surface. A drain installed at the bottom of the box served to drain both any overflow from the collection bottles as well as any water that infiltrated into the box. A tile line was extended from the installation area to ensure the samplers were not flooded with water percolating from the uphill gradient. After adding a layer of gravel to ensure proper drainage to the tile line, the excavated pits were backfilled.

Percolation typically occurs predominantly from late fall until late spring or early summer, depending on annual precipitation patterns. Samplers were installed during 1998, but many experienced flooding during the 1998 and 1999 sampling season, so data from that time period are not shown. Sampler drainage was improved by deepening and extending pit area drains during 1999, and sampling resumed once percolate flow resumed. Thirteen samplings were collected during the period of percolate flow, which, due to dry conditions in the fall of 1999, did not commence until January 2000 and ended in June 2000. During each field visit the wick lysimeters were sampled by extracting water from the collection bottles using a portable vacuum pump and a four-liter glass vacuum flask. Percolate volumes were recorded. Grab samples from tile lines were also taken during each field visit.

Surface Water Sampling

Runoff, the primary contaminant vector from dairy farms to streams, is difficult to isolate for repetitive sampling due its nonpoint nature. Therefore streamwater samples were collected at various points on the farm to estimate effects of surface runoff (Figure 1). Stream Site 1 was located at the mouth of a subwatershed applied with commercial fertilizer and that contained fields with F1 and F2 plots. Stream Sites 2 and 3 drained a primarily biosolids subwatershed containing sites B1 and B2, Site 3 being upstream closest to the B plots. Stream Site 4 drained a watershed that is applied with both manure and biosolids. Stream Sites 5 and 7 are located on the same stream and drain areas of the watershed where manure was the primary application and contains the M1 site. The subwatershed also contains a silage bunker just upstream of Site 7 that was subsequently found to be draining into the stream. Stream Site 6 drains areas that have manure and biosolids applied to the fields. Grab samples were taken repetitively from these sampling points during the first six months of 2000.

Intensive stream sampling was conducted in the watersheds primarily applied with fertilizer (Stream Site 1) and biosolids (Stream Site 2). Stream stage was recorded daily at these sites using a pressure transducer calibrated to determine water depth (Model 10/D, Druck Inc., Fairfield, Connecticut) (0 to 1 psi range x 0.01 resolution). A 45-day capacity data logger was attached to the pressure transducer to allow constant monitoring of streams. The pressure transducer was protected from silt and rocks with a PVC pipe housing (10 cm diameter, 1.1 m tall) with 1.25 cm holes to allow proper water flow. The transducer was installed in the deepest part of streams, attached to permanent stakes to keep the transducer position constant. Stream cross sectional survey and transducer calibrations were performed upon installation. Five stream velocity surveys were conducted during April 2000 when the stream was at different stages to establish a regression between transducer readings and stream discharge. The discharge of these two streams was then calculated at 10-minute intervals throughout the month of April 2000. In addition to grab samples gathered during each site visit, intensive automated sampling was conducted daily during April 2000, during which 500 ml was sampled from Streams Sites 1 and 2 every four hours.

Soil and Crop Sampling

Soil samples were taken by depth (0 to 50 cm, 50 to 100 cm, etc.) across the exposed face of each pit excavated to facilitate wick sampler installation. Sampler installation depths of 0.6 to 0.7 m were below the surface of fragipan horizons observed at depths of 0.3 to 0.6m. Plow (Ap) layer depths were typically 0.18 to 0.21 m.

In 2000, all test fields were used for growing alfalfa. Ten grab samples of alfalfa hay were collected from each field site along a 500 m transect and were composited into one sample per field. The plant tissue samples were dried, ground, and analyzed for total elemental content using ICP (inductively coupled plasma) spectroscopy.

Analytical

Soils were tested for pH by mixing 1:1 v/v with distilled water and measuring with a calibrated meter after one hour. Soil elemental analysis was conducted using nitric acid microwave digestion, with 0.5 g soil placed in a capped fluorocarbon microwave vessel and digested in 10 ml 4M HNO_3 for 10 minutes using microwave heating. After cooling, the vessel contents were filtered, centrifuged, or allowed to settle and then diluted to volume and analyzed.

Percolate collected from wick lysimeters (each of which held four collection bottles) was composited at the laboratory in proportion to the volume collected in the field, resulting in one sample per wick lysimeter for analysis. This composite was then filtered (0.45 μm nominal porosity) to remove particulates. Streamwater samples were likewise filtered, and 100 ml of each sample was then archived and refrigerated at 4°C. Selected subsamples were subsequently analyzed for trace elements using ICP spectroscopy.

Samples from the April 2000 intensive sampling of Stream Sites 1 and 2 were frozen and subsequently analyzed colorimetrically for soluble reactive P using the ascorbic acid method (APHA/AWWA/WEF, 1999). A subset of the streamwater samples was selected for elemental analysis, including total P. Fifty ml of each sample was pipetted into a 250 ml beaker, brought to dryness, and then boiled for 30 minutes with 5 ml of acid solution (25 percent HNO_3, 25 percent HCl). After cooling, the solution was transferred and brought to 10 ml volume in a 15 ml plastic centrifuge tube for storage. The net five-fold concentration of the sample ensured low level detection by the graphite furnace atomic absorption (GFAA) spectrometer (Perkin-Elmer Model 5100 with Zeeman module).

RESULTS AND DISCUSSION

As noted earlier, the wide range of initial soil characteristics, cropping history, and waste application rates means that it is not meaningful to regard the two fields sampled for each management pattern as experimental replicates, nor to average the data from them. This precludes standard statistical treatments, so the results are framed in terms of observable trends and comparisons with relevant water quality standards.

As noted earlier, it was found that sewage biosolids cumulative metals loadings were disparate from the recent biosolids application history, resulting in a relative ranking of reported elemental loadings (Table 3) of B2>>F2>F1≈M1>B1>>M2. Of currently applied materials (Table 5), the dairy manure and mixed dairy/cheese waste had much lower concentrations of most trace elements of concern (As, Cd, Cr, Mo, Ni, Pb, and Zn) than did the solid or mixed liquid biosolids; only Cu had comparable concentrations.

Soil and Plant Analysis

Soil pH data in Table 6 include mean historical soil pH data from Table 2, which correspond well with this study's results. Soil pH levels in the plow layer (upper 20 cm) were greater in the sludge plots than in most other field sites, likely due to calcareous additions at the wastewater treatment plants as well as on-farm lime stabilization of the liquid sludge and other wastes. Soil pH generally became more acidic with depth. Field M2 was more acidic than other sites due to intrinsic soil characteristics, as reflected in subsoil levels.

TABLE 6. Soil pH Analysis.

Depth (cm)	Manured Plots		Biosolids Plots		Fertilizer Plots	
	M1	M2	B1	B2	F1	F2
Ap, From Table 2	*6.3*	*5.4*	*7.0*	*6.6*	*6.9*	*6.2*
0 to 20	6.18	5.31	6.67	6.70	6.62	6.35
0 to 50	6.58	5.19	7.01	6.05	6.52	6.30
50 to 100	5.52	5.20	6.99	5.63	5.87	6.19
100 to 150	5.62	4.75	5.01	6.03	7.01	6.13
150 to 200	5.20	5.51	–	–	–	–

Nitric acid extractable element profiles are presented in Table 7. Samples for the 0 to 20 cm layer were taken later and from slightly different locations in the test fields than the 0 to 50 cm and lower layer samples collected during wick sampler installation, so some variability in surface horizon results is anticipated. Results for the 0 to 50 cm sample at the M1 plot were extremely erratic and discarded.

Several patterns of trace element concentrations can be seen in the elemental profiles. Arsenic, chromium, copper, nickel, and magnesium tended to have greatest concentrations at depth, likely due to native subsoil concentrations, which thus masked any surface enrichment. One notable exception was elevated copper in the M1 field. It has been observed that copper sulfate solutions used as a hoof dip are dumped into manure and spread. This can occasionally greatly increase manure Cu concentrations (McBride and Spiers, 2001), with some field soil concentrations approaching 100 mg/kg observed in New York (McBride, personal communication).

A second pattern observed in Table 7 was surface enrichment. Significant enrichment of calcium (all B and F treatments) is not surprising in view of liming as well as the elevated Ca contents of many of the wastes applied. The clearest cases of surface enrichment were molybdenum, phosphorus, and sulfur. Most subsoil Mo levels were below detection limits. P levels were typically two to three times greater than subsoil levels (especially with biosolids treatments). As noted earlier, P levels exceeded crop requirements for all M and B treatment soils examined. Surface S concentrations were about 10-fold greater than subsoils. A small degree of cadmium enrichment may have occurred for the B and F treatments. Potassium enrichment was greatest in biosolids plots, although subsoil variability was high. Sodium suggested some surface enhancement but is confounded by the recent application history of high Na wastes. Concentration patterns of lead and zinc were inconclusive, with little relative difference in concentrations.

No trends were evident in alfalfa hay analysis results (Table 8), unsurprising in view of relatively low trace element loadings. The detection of Cd in alfalfa in the M2 field despite soil concentrations comparable to all other sites was likely due to increased bioavailability resulting from the substantially lower soil pH. Copper/molybdenum ratios can be a significant concern for forages grown for ruminants. Even single applications of advanced alkaline-stabilized sewage biosolids have resulted in Cu/Mo ratios approaching levels of concern (McBride *et al.,* 2000). All Cu/Mo ratios were greater than the recommended minimum 4:1 ratio, which is not surprising in view of the fact that the sewage biosolids applied were not alkaline-stabilized.

Wick Sampler Analysis

Percolate samples came from a six-month time period (December 29, 1999, to June 29, 2000) that constituted the primary period of percolation for 1999 to 2000. The total precipitation for this period was 39.3 cm, typical for this region. Table 9 indicates the mean volume (expressed as depth, based on the horizontal surface area of the sampler) collected from the two samplers in each test field. Most samplers collected between 90 and 120 cm during this period, which was greatly in excess of normal percolation rates. This was almost certainly due to accumulation of lateral flow across the soil fragipan. Percolate concentrations were likely not affected by this phenomenon because the samplers were positioned on the downhill side of each field so that any intercepted interflow would have been percolated through the treated soil at some point in the field. The only exception to this pattern was the M2 samplers which averaged 31.5 cm percolate depth, more consistent with expected percolation rates for the region.

Volume weighted mean percolate concentrations are summarized in Table 9. Also included in Table 9 are sample standard deviations to quantify the variation between the volume weighted concentrations of each site's pair of wick samplers.

Several general observations can be made. As a basis of comparison, relevant New York State drinking water standards (for ground water) are shown in Table 9 (percolate measured at this shallow depth is not a drinking water source, but could conceivably

TABLE 7. Soil Profile: 4M Hot Nitric Acid Extractable Elements in mg/kg.

		Manured Plots		Biosolids Plots		Fertilizer Plots	
Element	Depth (cm)	M1	M2	B1	B2	F1	F2
As	0 to 20	1.98	2.76	3.30	4.07	3.72	3.01
	0 to 50	-	3.71	3.88	2.66	5.27	3.07
	50 to 100	2.91	3.43	8.81	5.72	9.01	9.44
	100 to 150	4.56	7.58	9.74	8.00	9.67	8.64
	150 to 200	5.75	9.17				
Ca	0 to 20	1904	1208	3364	3320	2762	2625
	0 to 50	–	1657	4581	2168	3036	2061
	50 to 100	2584	1334	2228	941	1163	1655
	100 to 150	648	375	1089	1859	2091	1158
	150 to 200	752	1123				
Cd	0 to 20	0.176	0.468	0.490	0.593	0.501	0.503
	0 to 50	–	0.405	0.442	0.373	0.437	0.401
	50 to 100	0.289	0.397	0.389	0.301	0.372	0.439
	100 to 150	0.275	0.355	0.520	0.340	0.460	0.355
	150 to 200	0.288	0.430				
Cr	0 to 20	11.67	13.39	14.22	16.26	13.24	13.07
	0 to 50	–	13.02	13.72	14.62	15.29	9.13
	50 to 100	11.85	13.25	14.79	14.12	15.35	16.15
	100 to 150	12.19	16.26	17.42	16.35	16.96	15.74
	150 to 200	13.46	17.92				
Cu	0 to 20	34.48	17.12	18.77	16.7	12.55	15.72
	0 to 50	–	13.19	16.09	7.768	12.03	12.69
	50 to 100	10.17	12.92	26.2	16.71	22.23	24.98
	100 to 150	11.98	21.5	31.67	20.46	22.87	22.8
	150-200	14.92	26.52				
K	0 to 20	1181	1180	1473	1917	2011	1027
	0 to 50	–	1308	1103	919	1979	515
	50 to 100	740	1246	968	848	1360	1506
	100 to 150	441	762	1124	1555	1750	1660
	150 to 200	565	723				
Mg	0 to 20	2348	2484	2505	2658	2651	2332
	0 to 50	–	3004	3030	2943	3368	1898
	50 to 100	3182	2903	3612	3209	3713	3870
	100 to 150	3294	4091	3765	3727	3947	3710
	150 to 200	3622	4534				
Mn	0 to 20	245.5	419.4	487.1	610	348	481.5
	0 to 50	–	529.9	551.9	557.6	443	608.1
	50 to 100	230.4	828.8	322.8	270	402.2	434
	100 to 150	237.5	324.9	335.1	370.4	421.9	370.8
	150-200	305.7	538.3				
Mo	0 to 20	0.653	0.617	0.731	0.731	0.443	0.523
	0 to 50	–	0.177	0.122	<0.0004	<0.0004	0.199
	50 to 100	<0.0004	0.082	<0.0004	<0.0004	<0.0004	<0.0004
	100 to 150	<0.0004	<0.0004	<0.0004	<0.0004	<0.0004	<0.0004
	150 to 200	<0.0004	<0.0004				
Na*	0 to 20	148.8	29.82	112.2	184.1	2.213	10.77
	0 to 50	-	61.54	115.6	94.12	66.53	36.04
	50 to 100	99.83	56.81	60.94	55.17	23.98	51.69
	100 to 150	89.47	55.49	72.32	33.87	57.3	50.56
	150 to 200	49.57	52.93				

*Data influenced by food waste.

TABLE 7. Soil Profile: 4M Hot Nitric Acid Extractable Elements in mg/kg (cont'd.)

		Manured Plots		Biosolids Plots		Fertilizer Plots	
Element	Depth (cm)	M1	M2	B1	B2	F1	F2
Ni	0 to 20	17.83	17	16.48	19.49	17.89	16.16
	0 to 50	–	16.47	16.7	15.2	18.64	10.67
	50 to 100	16.87	16.69	25.81	19.61	25.18	28.57
	100 to 150	18.76	24.64	61.62	26.98	28.2	25.02
	150 to 200	20.94	31.54				
P	0 to 20	1013	954.5	1098	1298	686.9	712.1
	0 to 50	–	1229	1248	461.4	762.5	386.7
	50 to 100	785.5	960.1	350.3	314.6	353.6	346.9
	100 to 150	330.4	306.4	300.6	371.4	387.1	352.3
	150 to 200	379.8	419				
Pb	0 to 20	13.58	13.23	13.02	15.31	13.98	13.95
	0 to 50	–	13.40	15.65	15.64	18.57	13.30
	50 to 100	10.97	13.63	13.11	11.25	12.78	16.58
	100 to 150	11.07	12.72	14.62	13.4	16.05	12.74
	150 to 200	11.23	17.24				
S	0 to 20	349.3	293.4	266.6	398.9	341.8	345.8
	0 to 50	–	414.6	306.7	272.3	381.5	190.5
	50 to 100	283.4	360.8	27.75	29.72	55.23	20.74
	100 to 150	33.38	55.02	83.8	26.67	20.49	52.87
	150 to 200	34.91	18.22				
Zn	0 to 20	76.12	74.62	72.72	84.42	72.3	73.71
	0 to 50	–	80.64	76.46	72.14	84.24	52.55
	50 to 100	60.94	88.53	57.43	50.65	64.56	76.93
	100 to 150	45.57	63.1	57.93	66.28	73.8	63.71
	150 to 200	51.36	80.93				

TABLE 8. Plant Tissue Trace Element Concentrations (mg/kg) and Copper/Molybdenum Ratio.

Element	M1	M2	B1	B2	F1	F2
Al	56.6	52.3	106.0	140.0	73.3	94.9
Ca	12,600	12,980	12,970	14,270	14,230	16,110
Cd	nd	0.0174	nd	nd	nd	nd
Cr	1.58	1.09	0.381	0.466	0.338	1.05
Cu	10.2	11.3	11.0	11.7	11.9	12.6
Fe	105	103	203	179	117	170
K	24,220	36,910	33,600	28,450	37,250	21,170
Mg	2,823	2,619	2,274	2,605	2,290	2,701
Mn	40.7	52.9	34.7	55.7	30.6	38.7
Mo	1.1	0.82	1.93	0.96	0.74	1.04
Na	286	279	269	316	102	519
Ni	0.57	0.74	0.54	0.86	0.27	0.70
P	3,617	3,259	3,434	2,965	3,266	2,778
Pb	0.059	nd	nd	nd	0.012	nd
S	1,448	2,546	2,274	2,128	2,502	2,941
Zn	26.5	22.9	28.5	28.9	31.0	25.3
Cu/Mo Ratio	9.3	13.7	5.7	12.2	16.0	12.1

*nd indicates 'not detected.'

TABLE 9. Wick Lysimeter Results: Total Sample Depth Collected (cm), Volume-weighted Mean Percolate Concentrations (µ g/l), and Sample Standard Deviations Between Mean Concentrations from Each Pair of Wick Samplers.

	M1		M2		B1		B2		F1		F2		NY
Analyte	**Mean**	**SD**	**Mean**	**SD**	**Mean**	**SD**	**Mean**	**SD**	**Mean**	**SD**	**Mean**	**SD**	**Std.**
Depth (cm)	129.9		31.5		91.2		100.8		101.5		120.6		
							Concentration µg/l						
As	3.5	1.9	3.8	3.0	1.7	0.3	6.1	3.5	1.4	0.1	1.7	0.3	50
Ca	24,610	2,520	90,460	1,840	88,920	13,280	140,910	27,860	38,060	1,199	20,570	3,490	–
Cd	0.14	0.06	**1.11**	0.32	0.26	0.03	0.22	0.01	0.09	0.02	0.33	0.37	0.9
Cr	2.1	0.3	2.8	2.3	1.3	0.1	5.7	1.3	1.8	0.2	1.9	0.3	160
Cu	**11.5**	0.9	**12.6**	6.1	7.1	0.9	**11.5**	8.6	**10.1**	0.4	5.4	2.2	8
K	9,440	1,640	83,510	10,490	5,410	395	8,210	7,860	13,010	1,670	801	452	–
Mg	8,590	1,000	25,360	3,060	13,830	1,370	28,460	2,720	9,810	622	5,820	1,120	35,000
Mo	10.2	11.6	54.8	49.2	10.4	15.4	4.6	4.5	6.9	7.9	7.5	7.3	–
Na	**40,120**	1,940	**124,160**	7,860	**36,185**	6,090	**133,680**	8,700	7,840	544	8,220	451	20,000
Ni	5.1	0.2	22.1	1.5	16.4	0.8	23.5	0.6	5.5	0.1	3.7	0.5	70
Pb	**2.6**	1.1	**2.1**	0.1	1.8	0.2	1.6	0.4	1.7	0.1	**2.1**	0.1	2
S	5,767	540	16,004	527	26,253	3,303	24,766	10,447	9,385	171	4,242	35	250,000
Zn	15.7	3.6	42.2	10.7	8.4	0.6	12.3	3.8	6.4	1.3	6.7	5.6	59

Note: Bold indicates values exceeding New York State ground water quality standards.

contribute to recharge). The B2 site had the greatest measured concentrations for six of the 13 analytes reported in the table, followed by M2 with five maximum analyte concentrations. In only four cases (As, Cr, Na, and Ni) did the peak percolate concentration coincide with greatest topsoil concentrations, all of which occurred with B2 site. Values shown in bold type in Table 9 exceed New York drinking water standards. Percolate Cu exceeded the 8 µg/l standard at four sites: M1, M2, B2, and F1. Sodium showed the greatest exceedence of any analyte, exceeding the 20,000 µg/l standard on all M and B fields, most notably at M2 and B2 with levels over 124,000 and 133,000 µg/l, respectively. Three sites – M1, M2 and F2 – had Pb levels that slightly exceeded standards, but these levels were close to the lower detection limit (LDL) for Pb. Cadmium slightly exceeded standards in the M2 percolate, with the increased mobility again a function of substantially more acidic soil throughout the profile. While not exceeding drinking water standards, sulfur levels in M2, B1, and B2 were substantially greater than other fields. Molybdenum had the greatest relative variability (coefficient of variation), suggesting that transport was extremely flow path dependent and thus subject to spatial variation in flow paths, as reflected in the between sampler variability.

Stream Analysis

Surface streams were monitored for the first six months of 2000, with mean grab sample soluble elemental concentrations shown in Table 10. Many New York State surface water quality standards are a function of the individual water sample's hardness, thus the quality standard levels shown in Table 10 were calculated from measured hardness data. Standards are flagged to indicate whether the standard is based on human (H) or aquatic (A) health. Exceedence of those standards are indicated in the table. As noted earlier, Stream Site 1 was located at the mouth of the fertilizer subwatershed containing fields with F1 and F2 plots. Stream Sites 2 and 3 drained a primarily biosolids subwatershed containing sites B1 and B2, with Site 3 being upstream closest to the plots. Watersheds for Sites 4 and 6 were applied with both manure and biosolids. Stream Sites 5 and 7 drain a primarily manure applied watershed containing the M1 site.

The data for Sites 5 and 7 were most strongly influenced by the silage bunker just upstream of Site 7 that was subsequently found to be draining into the stream. As a result, almost all elements at Stream Sites 5 and 7 were substantially elevated. Most notably, Ca, Cd, Fe, K, Mg, Mn, Mo, and Ni concentrations were at least five-fold greater than other streamflow sites, with Mn and Na substantially

TABLE 10. Mean Streamwater Grab Sample Concentrations (µg/l).

Element	Fertilizer S1	Biosolids S2	Biosolids S3	Biosolids and Manure S4	Biosolids and Manure S6	Manure/Food Waste/Bunker S5	Manure/Food Waste/Bunker S7	NYS Quality Standard
As	1.1	1.1	1.0	1.0	1.0	1.3	1.3	50 H
Ca	22,390	28,790	39,960	20,070	38,070	106,320	103,650	–
Cd	0.16	0.16	0.19	0.34	0.21	1.35	1.13	0.90 A*
Cr	2.0	1.8	2.5	2.4	2.8	3.3	5.8	50 H
Cu	3.9	3.5	5.1	3.5	8.3	5.1	6.3	8 A*
Fe	9.7	4.3	3.2	1.7	4.6	100.3	116.1	300 HA
K	623	1,440	2,690	602	9,990	80,210	114,600	–
Mg	4,390	5,460	8,290	4,220	9,630	25,060	27,580	35,000 H
Mn	1.24	0.27	0.50	0.70	1.69	**3,320**	**3,020**	300 H
Mo	0.74	0.91	1.22	1.19	1.00	4.64	12.06	–
Na	6,780	**20,220**	**26,120**	**5,167**	**52,130**	**175,800**	**245,680**	20,000 H
Ni	4.3	5.2	7.2	3.3	6.8	25.2	25.1	70 A*
Pb	2.7	2.1	**2.4**	**2.8**	1.6	1.9	1.9	2 A*
S	5,570	5,530	6,490	6,790	7,680	6,640	15,840	250,000 H
Zn	5.1	5.7	4.7	5.9	10.2	11.6	**72.4**	59 A*

*Standard based on measured stream water hardness.
Notes: Bold type indicates values exceeding New York State surface water quality standards. Standards are coded H for human health or A for aquatic health.

exceeding the surface water standards. This situation has been corrected by installing a drain from the bunker silo to the recently-installed manure processing system. Because of this bunker silo impact, effects from the "primarily manure" treatment that Sites 5 and 7 were designated to represent could not be separated.

Of the remaining Stream Sites (1 through 4 and 6), measured levels of As, Cd, and Pb were all close to the LDL of the ICP instrument (Table 10). Site 6, representing biosolids plus manure application, tended to have the greatest measured concentrations of all other elements, exceeding Na and, to a lesser extent, Cu limits. Biosolids treatment Sites 2 and 3 also exceeded the Na standard. The exceedence of the Pb standard at Sites 1 through 4 is likely not significant due to the proximity of the results to the lower detection limit.

Time series results of streamflow grab sample results are shown for several representative elements from watershed Sites 1 through 4 and 6 in Figure 2. Also shown are sewage biosolids applications (cumulative mass per field) and meteorological data. Note that sewage biosolids applications during the monitoring period were significant only in the watershed for Sites 2 and 3. Na and Ni declined following the snowmelt and precipitation events in late February. All sites showed a steady increase in apparent response to the early April precipitation events. There was no clear correlation of stream concentrations with sewage biosolids loadings.

Intensive sampling allows detection of signals missed by random grab sampling. Total elemental concentrations for Stream Sites 1 (fertilizer watershed) and 2 (sewage biosolids) are shown in Figure 3, along with stream discharge results. Only a few elements had concentration changes that clearly correlated with the runoff events noted by the intensive monitoring in April. Soluble phosphorus strongly correlated with stream discharge (Figure 3), with a high correspondence of P peaks with increases in discharge for both watersheds, although the biosolids watershed peaks were notably greater and more persistent.

Analyses aside from P were less frequent due to cost considerations. Substantial increases in Mo concentrations from the Site 2 biosolids treatment lagged large runoff events by about five days (Figure 3). Similar delays in Mo transport have been observed in column leaching studies (Richards *et al.,* 2000). Patterns for Cu were less clear, with an increase from both sites lagging the second runoff event of April, although overall levels were greater from the sewage biosolids Site 2 treatment. Both Ca and Na showed no real correlation with runoff events, other than perhaps a slight dilution effect.

Total elemental exports (kg/ha/month) were calculated for April using the mean discharge multiplied by the mean daily concentration of each element (Table 11). While many elements were similar in total export, exports of Cu, Mo, Na, and soluble P (both ortho soluble and total soluble) were two to nearly eight times greater from the S2 watershed. Although

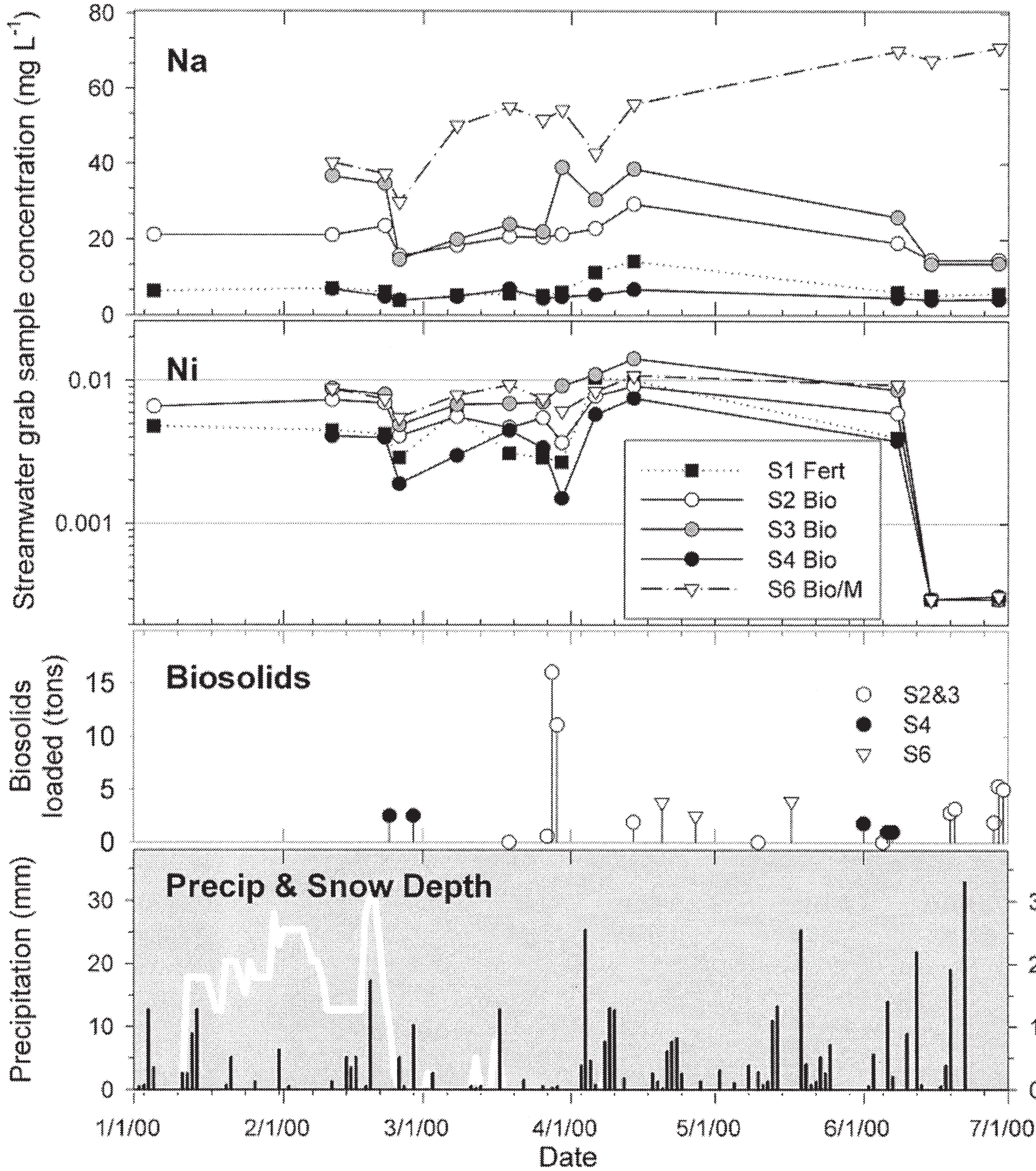

Figure 2. Streamwater Monitoring January to June 2000: Streamwater Grab Sample Na and Ni Concentrations; Biosolids Loadings on Subwatersheds; and Daily Precipitation and Snowpack Depth. Stream monitoring Site S1 (fertilizer treatment), Sites S2 through S4 (biosolids), and S6 (mixed biosolids/manure).

watershed areas were only 2 percent different, the mean discharge for the S2 biosolids watershed was 18 percent greater. Normalizing for equal discharge (dividing the W2/W1 ratio by 1.18), the S2 watershed exports for these four analytes were still 1.7 to 6.5 greater than the S1 fertilizer watershed, with the greatest difference being Mo.

Comparing Percolate and Surface Stream Concentrations

In order to assess the relative impacts on percolate and surface streamwater, concentration ratios were calculated by dividing wick sampler percolate concentrations (Table 9) by the concentrations of corresponding surface stream samples (Table 10), similar to the

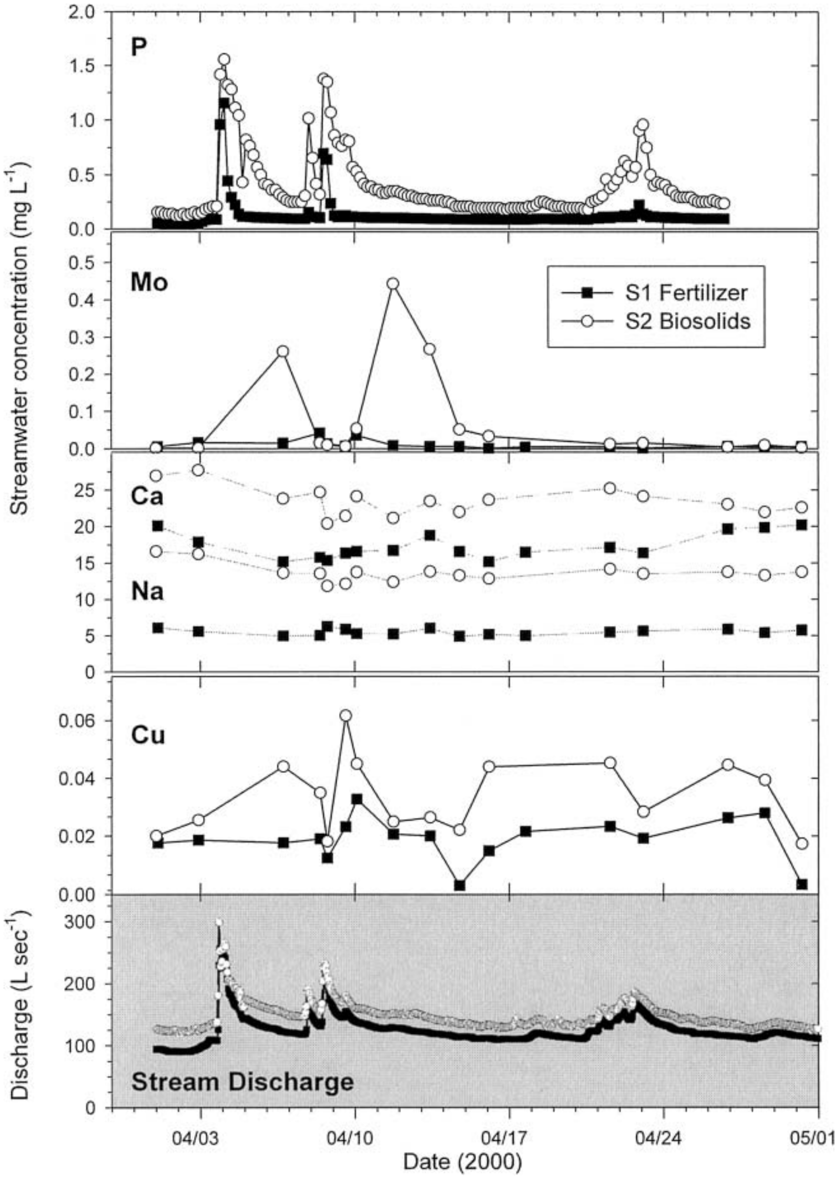

Figure 3. Intensive April 2000 Streamwater Monitoring at Sites S1 (fertilizer) and S2 (biosolids): Streamwater PO_4-P, Mo, Ca, Na, and Cu Concentrations, and Stream Discharge.

approach of Page (1981) who examined relative ground water and surface water contaminant concentrations. Ratios were calculated for the fertilizer treatment using the average of F1 and F2 wick sampler results divided by Stream Site 1 concentrations. For biosolids treatments, the B2 wick sampler results were divided by Stream Site 3 results. Percolate/streamwater concentration ratios are summarized in Table 12. The impact of the bunker silo drainage on Stream Sites 5 and 7 prevented any meaningful analysis of ratios from manured treatments. Note that both surface and percolate samples were filtered, thus the observed ratios presented in Table 12 are for soluble components only.

Percolate concentrations almost always exceeded corresponding surface stream concentrations for both treatments (Table 12 and Figure 4). The only exceptions were slight differences in Pb levels (and all were near lower detection limits), and Cr in the fertilizer S1 site. This pattern of percolate exceeding surface water concentrations is consistent with the findings of Page (1981). Percolate/streamwater ratios tended to be greater for the biosolids treatment (eight out of thirteen analytes had greater ratios: As, Ca, Cr, Mg, Na, Ni, S, and Zn). It should be noted that soil types and pH levels for the represented treatments (F1, F2, and B2) were similar, allowing greater weight for this observed pattern to be attributed to the respective land application treatments. Based on these ratios, it appears that the biosolids treatment impacted soluble percolate concentrations proportionally more than corresponding soluble streamwater concentrations. This observation is consistent with the fact that biosolids are spread with machinery that immediately cultivates the soil applied with biosolids, making the applied material more "available" to percolate than to surface runoff.

TABLE 11. Total and Relative Element Export From Watersheds 1 and 2 in April 2000.

	Stream S1 Fertilizer	Stream S2 Biosolids	Watershed Comparisons W2/W1 Ratio	Q-Normed W2/W1 Ratio
Area (ha)	253.0	248.4	0.98	–
Mean Q (l/sec)	125.3	147.5	1.18	1.00
		kg/ha/mo		
Ca	22.17	34.00	1.53	1.30
Cu	0.024	0.049	2.02	1.72
Fe	0.306	0.290	0.95	0.81
K	1.85	2.85	1.54	1.31
Mg	4.09	5.83	1.43	1.21
Mn	0.006	0.011	1.85	1.57
Mo	0.014	0.108	7.65	6.50
Na	7.07	19.78	2.80	2.38
PO_4-P	0.175	0.564	3.23	2.74
Total P	0.337	0.705	2.09	1.77
S	0.056	0.075	1.34	1.14
Zn	0.036	0.052	1.46	1.24

TABLE 12. Percolate/Streamflow Concentration Ratios for Fertilizer (mean F1 and F2/Stream Site 1) and Biosolids (B2/Stream Site 3).

Element	Fertilizer (F1 and F2)/S1	Biosolids B2/S3
As	1.6	5.8
Ca	1.5	3.5
Cd	1.6	1.1
Cr	1.1	2.3
Cu	2.2	2.2
K	11.9	3.1
Mg	2.0	3.4
Mo	11.8	3.8
Na	1.4	5.1
Ni	1.2	3.3
Pb	0.8	0.7
S	1.4	3.8
Zn	1.5	2.6

CONCLUSIONS

Waste Loadings

Recent application patterns differed from historical sewage biosolids loadings, with all but the M2 field having had biosolids applied at some point in time. The ranking of cumulative trace metal loadings was B2>>F2>F1=M1>B1>>M2. Because of this and the wide range of soil characteristics, it was not meaningful to regard the two fields representing each management pattern as experimental replicates. Overall loading rates were low, with maximum rates (Field B2) only approximately 1 percent of USEPA Part 503 cumulative limits. Of currently applied waste materials, dairy manure, and mixed dairy/ cheese waste had lower concentrations of most analytes of concern (As, Cd, Cr, Mo, Ni, Pb, and Zn) than did the solid or mixed liquid biosolids.

Soil and Crop Analysis

Surface enrichment of soil trace elements due to waste applications was most evident for Mo, P (two to three times greater than subsoil levels), and S (an order of magnitude greater than subsoils). Significant surface enrichment of calcium (all B and F treatments) was attributed to liming as well as elevated Ca contents of many wastes. No trends of concern were evident in alfalfa hay analysis results, unsurprising in view of relatively low trace element loadings. All Cu/Mo ratios were greater than the recommended minimum 4:1 ratio.

Percolate Analysis

Most wick samplers collected greater than normal percolation depths due to accumulation of water resulting from lateral flow across the soil fragipan. The only exception was the M2 site where samplers averaged 31.5 cm percolate depth, consistent with expected percolation rates.

The B2 biosolids site had the greatest observed concentrations for six of 13 analytes, followed by M2 with five maximum concentrations. The greater soil acidity at site M2 contributed to the observed mobility in percolate despite low to moderate soil concentrations. Percolate Cu exceeded the 8 µg/l ground water standard at four sites: M1, M2, B2, and F1. Sodium exceeded the 20,000 µg/l standard on all M and B fields, with concentrations up to 133,000 µg/l. Slight exceedences were observed for Pb (M1, M2, and F2) and Cd (M2). While not exceeding standards,

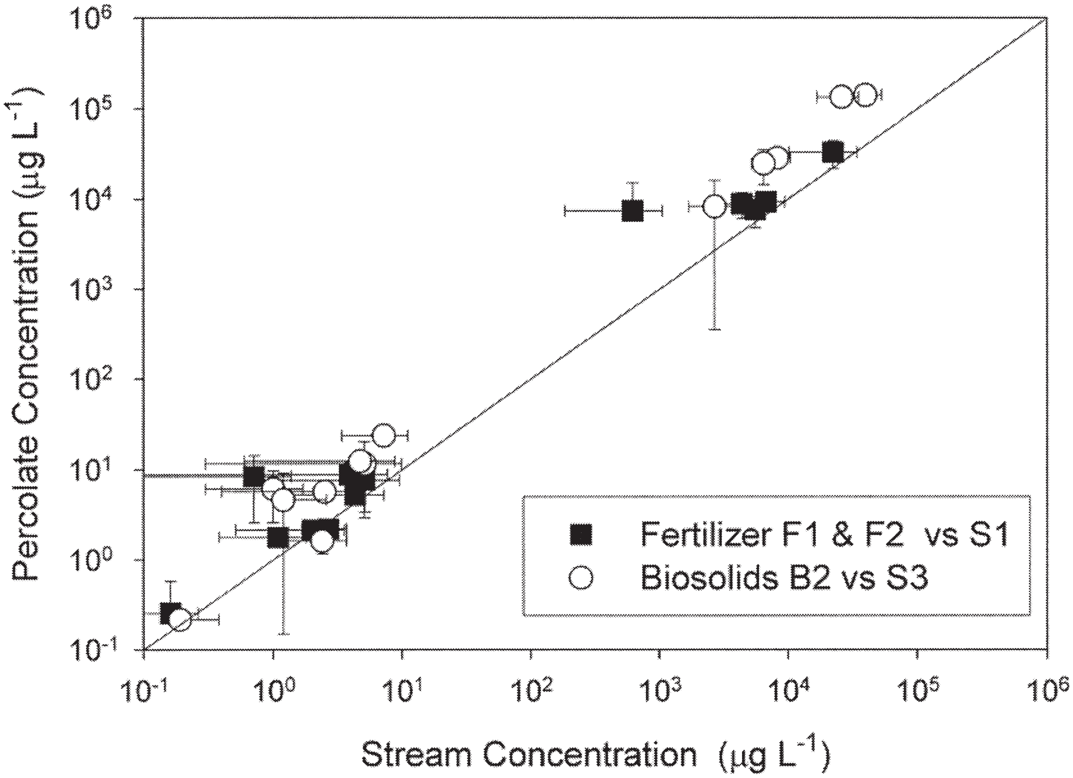

Figure 4. Mean Wick Sampler Percolate Concentrations Versus Corresponding Mean Grab Sample Streamflow Concentrations for Fertilizer Subwatershed (F1 and F2 versus Stream Site S1) and Biosolids Subwatershed (B2 versus Stream Site S3). The error bars represent standard deviations of concentrations among individual wick samplers (vertical bars) and among streamwater grab samples (horizontal bars).

S levels in M2, B1, and B2 were substantially greater than other fields. In only four cases (As, Cr, Na, and Ni) did the peak percolate concentration coincide with greatest topsoil concentrations, all of which occurred with the B2 site.

Surface Streamwater

Elements at manure only treatment Stream Sites 5 and 7 were substantially elevated due to upstream bunker silo drainage, which was the most significant impact noted in the study. However, this precluded evaluation of the streamwater from the manured watershed. For the other stream sites (1 through 4 and 6), soluble As, Cd, and Pb were all close to the LDL. Site 6, representing biosolids plus manure application, tended to have the greatest measured soluble concentrations of all other elements, with substantial exceedences for Na and P, and a slight exceedence for Cu.

All stream sites showed a steady increase in apparent response to the early April precipitation events. There was no clear correlation of stream concentrations with biosolids applications. Intensive monitoring of Stream Sites S1 (fertilizer) and S2 (biosolids) in April showed soluble ortho-P strongly correlated with stream discharge. Increases in Mo concentrations from Site S2 treatment lagged large runoff events by about five days.

Total exports of Cu, Na, soluble P (both ortho and total), and especially Mo were greater from the S2 (biosolids) watershed than from the S1 (fertilizer) watershed; even when normalizing for equal discharge the S2 watershed exports for these analytes were 1.7 to 6.5 times greater than S1.

Comparing Percolate and Surface Stream Concentrations

Percolate concentrations generally exceeded corresponding surface stream concentrations for the two treatments (fertilizer F1 and F2 wick samplers versus Stream Site 1, and sewage biosolids treatment B2 wick samplers versus Stream Site 3). Eight elements (As, Ca, Cr, Mg, Na, Ni, S, and Zn) had greater percolate/streamwater ratios for the biosolids treatment than the fertilizer treatment.

General

This study documented some detectable water quality impacts correlating with waste application practices, but few were severe, nor were they limited to one type of waste application. Some significant impacts – such as sodium concentrations in runoff and percolate – were traceable to less documented food waste applications. The strongest signal noted was the impact of bunker silo drainage on surface streamwater at several sites, which has been corrected by the farm.

It should be remembered that test fields had been manured for many years prior to the initiation of biosolids or food waste applications in the past two decades. As with the biosolids fields, all manured sites had available P in excess of crop requirements, with enrichment of other elements noted as well.

The most heavily loaded biosolids treatment site (B2) had the greatest number of maximum percolate and soil concentrations, as well as greater relative mobilization of elements to percolate when compared to the fertilizer site. Nutrient export via streamwater and percolate/streamwater concentration ratios appeared to be greater. It is important to remember that the farm has used low loadings of clean sludges, resulting in maximum cumulative loadings on the order of 1 percent of USEPA Part 503 cumulative limits. Observed results reported here cannot be extrapolated to the effects of greater cumulative loadings.

Percolate concentrations at manure Site M2 – with relatively low overall trace element loadings – highlighted the important role of soil pH on trace metal mobility. It is anticipated that mitigation of the soil acidity would reduce the observed percolate concentrations. Ongoing changes in manure management at the farm are intended to further reduce potential water quality impacts.

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