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SUSPENDED SEDIMENT, SOLAR RADIATION AND HEAT IN AGRICULTURAL RESERVOIRS

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ABSTRACT

Comparative studies conducted on two adjacent northern Mississippi reservoirs show that the suspended sediment has a measurable effect on the surface water temperature. Spectroradiometer measurements of reflected solar radiation from waters with different concentrations of suspended sediment indicate that up to 4 times as much energy is reflected from a reservoir with 200-ppm sediment concentrations than is radiated from an adjacent reservoir with 50-ppm concentrations. Epilimnion temperature measurements indicate a 1 to 4°C difference with clearer waters always warmer. Energy budget equations consistent with the experimental findings show that if turbid waters reflect more energy, then the temperature of these surface waters is cooler than that of clear water.

INTRODUCTION

Sediment, a major pollutant of streams, lakes, and reservoirs, occurs naturally but man's activities on watersheds can greatly increase it. Suspended sediment carries agricultural chemicals which may adversely affect aquatic ecosystems. In addition, when the suspended sediment settles, it fills the receiving reservoir basin and eventually renders it useless. Since suspended sediment must be measured and controlled for proper management of lakes and reservoirs, the various processes influenced by it must be fully understood. Temperature is a parameter which has an important influence on aquatic ecosystems and has been studied extensively. The objective of this study was to determine the influence on the temperature of reservoir waters of suspended sediments.

THEORY

Edinger et al. (1968) have introduced the concept of an equilibrium temperature to analyze the effect of meteorological parameters on the heat transfer across the air-water interface of an impoundment. If the imposed meteorological parameters are held constant for a long period of time, surface temperature will reach equilibrium and the terms in the heat budget equation describing the heat transfer across the air-water interface combine to give a zero net transfer. Using the Edinger model, there is a unique correspondence between the surface equilibrium temperature and any given set of applied meteorological parameters. In nature these meteorological parameters are highly transient and thus the equilibrium temperature is a continually fluctuating value as shown by Stefan et al. (1972) who compared computed equilibrium to measured lake temperatures. At any given time the actual water surface temperature is tending to the momentary value of the equilibrium temperature.

The three distinct processes, radiation, convection, and evaporation, involved in the heat transfer across the air-water interface are related by the following equation.

$$H_n = H_s - H_{sr} + H_a - H_{ar} - H_{br} \pm H_c - H_e \qquad (1)$$

where

H_n = net heat flux across air-water interface,
H^s = rate of incident short wave radiation,
H^{sr} = rate of reflected short wave radiation,
H^a = rate of incident long wave sky radiation,
H^{ar} = rate of reflected long wave sky radiation,
H^{br} = rate of long wave radiation from the water,
H^c = rate of heat convected from the water,
H^c_e = rate of latent heat loss by evaporation.

The first four terms on the righthand side of Eq. 1 represent the absorbed radiation and are independent of the water temperature. The last three represent processes dependent on the surface water temperature.

The equilibrium surface temperature as derived from a zero net heat flux conditions (Edinger et al. (1968)) is

$$T_e = H_r/K - \varepsilon\sigma\Delta/K + ((K - 4\ \varepsilon\sigma\Delta^3)/K(C_1 + \beta))\ (C_1\ T_a + \beta\ T_d) \qquad (2)$$

where

T_e = equilibrium temperature at the surface
H^r = $H_s - H_{sr} + H_a - H_{ar}$,
ε = emissivity of water,
σ = Stefan - Boltzmann constant,
Δ = absolute temperature constant = 273 Kelvin,
C_1 = Bowen ratio constant,
β = slope of linear approximation to the water vapor pressure temperature relationship,
K = thermal exchange coefficient = $4\ \varepsilon\sigma\Delta^3 + L\ \rho(C_1 + \beta)\ f(U, \Psi)$,
T_d = dew point temperature of the air,
T^a = air temperature,
L^a = heat of vaporization of water,
ρ = density of water,
$f(U, \Psi)$ = general function of wind speed and other variables.

A lake can be considered as an averaging system with a phase lag by operating on the applied fluctuating equilibrium temperature to produce the actual measured lake surface temperature, the output function of the system, which fluctuates much less.

If the equilibrium temperatures computed from actual meteorological conditions are averaged over several days, as indicated by the Stefan et

al. data, the computed and measured lake temperatures can be expected to be in close agreement.

It is assumed that the suspended sediment will influence only the reflected radiation term in Eq. 2. The incident solar radiation refracted across the air-water interface is scattered in all directions by the particles of suspended sediment and that which is scattered back and refracted through the water surface appears as reflected radiation. The albedo or reflectance from a water body loaded with suspended sediment has been shown by Ritchie et al. (3) as nearly proportional to the mass concentration of the suspended sediment. A typical example of this is shown in Figure 1. Examination of Eq. 2 reveals that only the first term on the righthand side can be influenced by suspended sediment.

When the equilibrium temperatures of two adjacent reservoirs with different concentrations of suspended sediments are compared, only the reflective radiation terms will differ since nearly identical meteorological conditions are presumed to exist. Also, presumably the incident long wave sky radiation is completely absorbed by both bodies of water. The difference between the equilibrium temperatures of the two reservoirs from Eq. 2 is then,

$$T_{eI} - T_{eII} = (H_{srII} - H_{srI})/K \tag{3}$$

or

$$\Delta T_e = H_s A_I \left((A_{II}/A_I) - 1\right)/K \tag{4}$$

where A_I and A_{II} are the short wave albedo or the reflectance values, H_{sr}/H_s, for reservoirs I and II, respectively. The thermal exchange coefficient, K, and the incident solar radiation are assumed to be identical for both reservoirs.

If the values of the incident solar radiation and the heat exchange coefficients are averaged over a period of time, Eq. 4 will predict the surface temperature difference between adjacent reservoirs with differing short wave albedos.

Eq. 3 may be further developed by incorporating the previously mentioned linear relationship between the reflectance or albedo and the concentration of suspended sediment.

$$A = a + bC \tag{5}$$

where a and b are constants and C is the concentration of suspended solids. Substituting Eq. 5 into Eq. 3 results in

$$T_{eI} - T_{eII} = H_s b(C_{II} - C_I)/K. \tag{6}$$

The measured temperatures and concentrations are in reality the results of a time averaged system and their values depend upon previous history.

If measured, paired values of surface temperatures and suspended sediment concentration from the two reservoirs are utilized in Eq. 6 then the combined function:

$$\theta = H_s b/K \qquad (7)$$

represents also a weighted time averaged value which depends upon the history of the applied meteorological parameters and a characteristic function representing the "memory" of the reservoir. Recent events in effect are given greater weight than events in the remote past.

METHODS AND MATERIALS

Two small reservoirs, (Fig. 2) located immediately adjacent to each other were selected for study. Descriptive data for these reservoirs are presented in Table I. The watershed of reservoir I is primarily forested, while that of reservoir II is primarily cultivated land. The watersheds are covered by highly erodible loess cap of varying thickness overlying sediments of Eocene Age. Lexington and Loring soils have developed in these watersheds (Ritchie et al. (1974)).

Table I
Reservoir Descriptive Data

Reservoir	I	II
SCS No.	Y-5-121	Y-5-122
Watershed Area	82.1 hectares	86.3 hectares
Normal Pool Area	2.68 hectares	2.80 hectares
Normal Pool Volume	36576 m^3	48593 m^3
Principal Land Use	Forest, Pasture	Cultivation, Pasture
Latitude	34 40'N	34 40'N
Longitude	89 40'W	89 40'W

Measurements were made over a 1-year period from August, 1973 to August, 1974 and these data are summarized in Table II. The water temperature profiles (Fig. 3) were measured at half meter intervals with (a Yellow Spring Instrument (YSI)) thermistor thermometer (Model 46 TV) and a submersible probe on a 15 meter cable[1/].

Radiation was measured with (Instrument Specialty Company (ISCO) Model SR) a spectroradiometer with a 2.95 meter light pipe equipped with a diffusing screen and calibrated with a spectroradiometer calibrator (ISCO Model SRC). The solar and reflected radiation were measured in micro watts per square centimeter per nanometer (μ watts cm^{-2} nm^{-1}).

1/ Names of products or companies are provided for information only and do not constitute an endorsement or preferential use by the U. S. Department of Agriculture.

Table II

Summary of Data for Two North Mississippi Reservoirs

Date	Surface Temp. [°C]		Secchi Depth [cm]		Suspended Sediment [ppm]		Solar Radiation [cal $cm^{-2}sec^{-1}$]x10^3			Albedo		[°C/ppm]x10^3
	I	II	I	II	I	II	H_s	H_{srI}	H_{srII}	I	II	
7-18-73	31.9	30.1	54.90	15.25	38	262	---	--	--	--	--	8.036
9-18-73	24.8	23.0	60.90	15.25	11	197	20.00	1.23	2.51	.061	.125	9.677
11-2-73	15.2	13.8	---	---	35	179	---	--	--	--	--	9.722
12-3-73	13.1	13.0	15.30	5.00	83	219	7.17	0.97	1.41	.135	.196	0.735
1-30-74	12.8	12.2	17.80	7.60	56	142	6.45	0.81	0.94	.126	.146	6.977
3-12-74	19.1	18.6	15.30	10.20	66	148	14.30	1.38	1.83	.096	.128	6.098
4-4-74	19.0	17.6	26.70	10.20	66	153	---	--	--	--	--	16.090
5-7-74	21.7	19.9	36.80	6.25	55	296	13.15	0.95	2.12	.072	.161	7.469
5-20-74	33.1	29.4	28.00	7.60	55	255	12.19	0.72	1.60	.059	.131	18.500
6-21-74	33.5	30.3	63.50	7.60	15	307	6.45	0.34	1.05	.052	.162	10.960

Reflected solar radiation was measured perpendicular to and approximately 20 cm from the water surface. Incident radiation was measured at 180° from the surface. Both the incident and reflected spectra were integrated between 400 and 1550 nanometers and the albedo determined by ratioing these two values. All radiation measurements (Fig. 4) were made on clear days within 2 hours of solar noon local time.

The Secchi depth was obtained in the standard manner using a 20 cm disk with alternate black and white quadrants.

The concentration of total solids in the surface waters was determined from two (100 ml aliquot) water samples collected at each sampling site, evaporated to dryness in a weighing beaker, and the total solids remaining were determined in parts per million (ppm). A second 100 ml aliquot was filtered with a 0.45 micron Millipore filter and processed as above to approximate the concentration of dissolved solids. The dissolved solids averaged 29 ppm for Reservoir I and 23 ppm for reservoir II. These values deviated very little. The suspended solids were determined as the difference between these determinations.

RESULTS AND DISCUSSION

Average Secchi depths are 35 and 10 cm in reservoirs I and II, respectively. The water in reservoir II looks red-brown because of suspended clay particles while the water in reservoir I has a light gray cast. The turbidity in both reservoirs is primarily due to suspended sediment.

The Secchi depth data in Table II indicate that reservoir I undergoes a definite cycle. During the winter rainy season the Secchi depth reaches a minimum. Reservoir I clears considerably during the summer months, whereas, reservoir II always appears red-brown and has a fairly constant low Secchi depth.

The surface temperatures and the albedo ratio values of the reservoirs are plotted in Fig. 5 for the 1-year period. The lake with the lowest albedo was always warmer. During the winter months when the albedo ratio was lowest the temperature difference was also lowest. Differences in surface temperature in the summer reached nearly 4°C with high albedo ratios.

The difference in surface temperatures generally indicates temperature differences at all depths (Fig. 3). The reservoir with the least turbidity contained the greatest amount of energy.

The combined parameter, θ, was calculated by ratioing the difference in measured surface temperature to the difference in measured surface concentration of suspended solids between the two reservoirs. No obvious seasonal tendency is noticed.

Since the data for 12-3-73 were obtained after very heavy rains, the heat advected by inflowing and outflowing water completely dominated the

reservoir temperatures on that date. If the value for 12-3-73 is deleted, the average of the remaining 9 values is 10.39×10^{-3} °C/ppm with a standard deviation of 4.238×10^{-3} °C/ppm. These data indicate that the turbidity levels in surface waters in a given locality do influence the temperatures of those waters.

It is evident from this and previous studies that suspended particles, such as sediment, in surface water scatter solar radiation and the back scattered radiation is rejected across the air-water interface as reflection. This loss of energy lowers surface temperature and the heat content of the entire reservoir.

Even slight shifts in temperature can cause significant alterations in the ecological system and thus affect the aquatic ecosystem. This effect may ultimately affect the species and numbers of fish which may be harvested from agricultural reservoirs.

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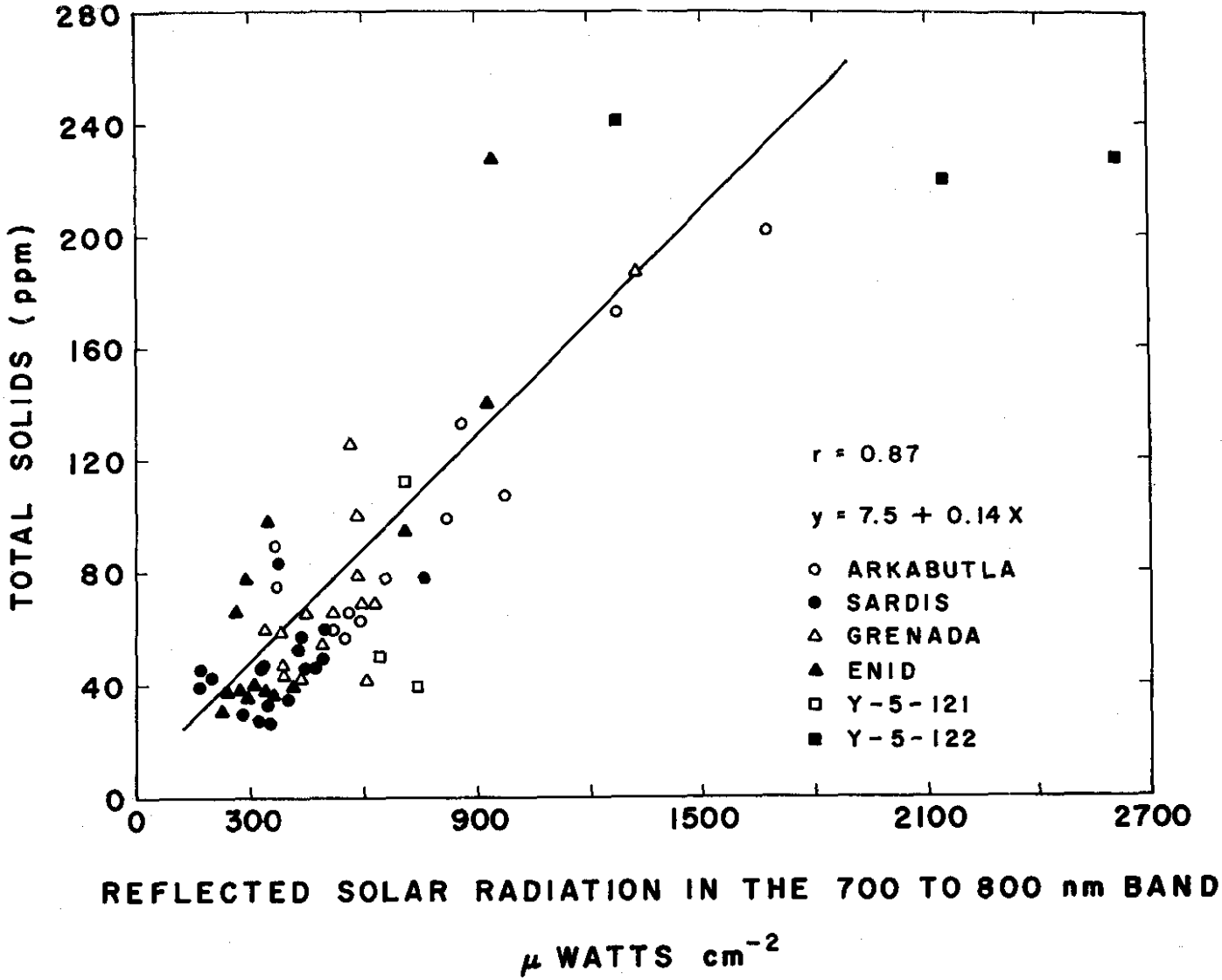

Fig. 1 Reflected Radiation - Total Solids Relationships for Six North Mississippi Reservoirs

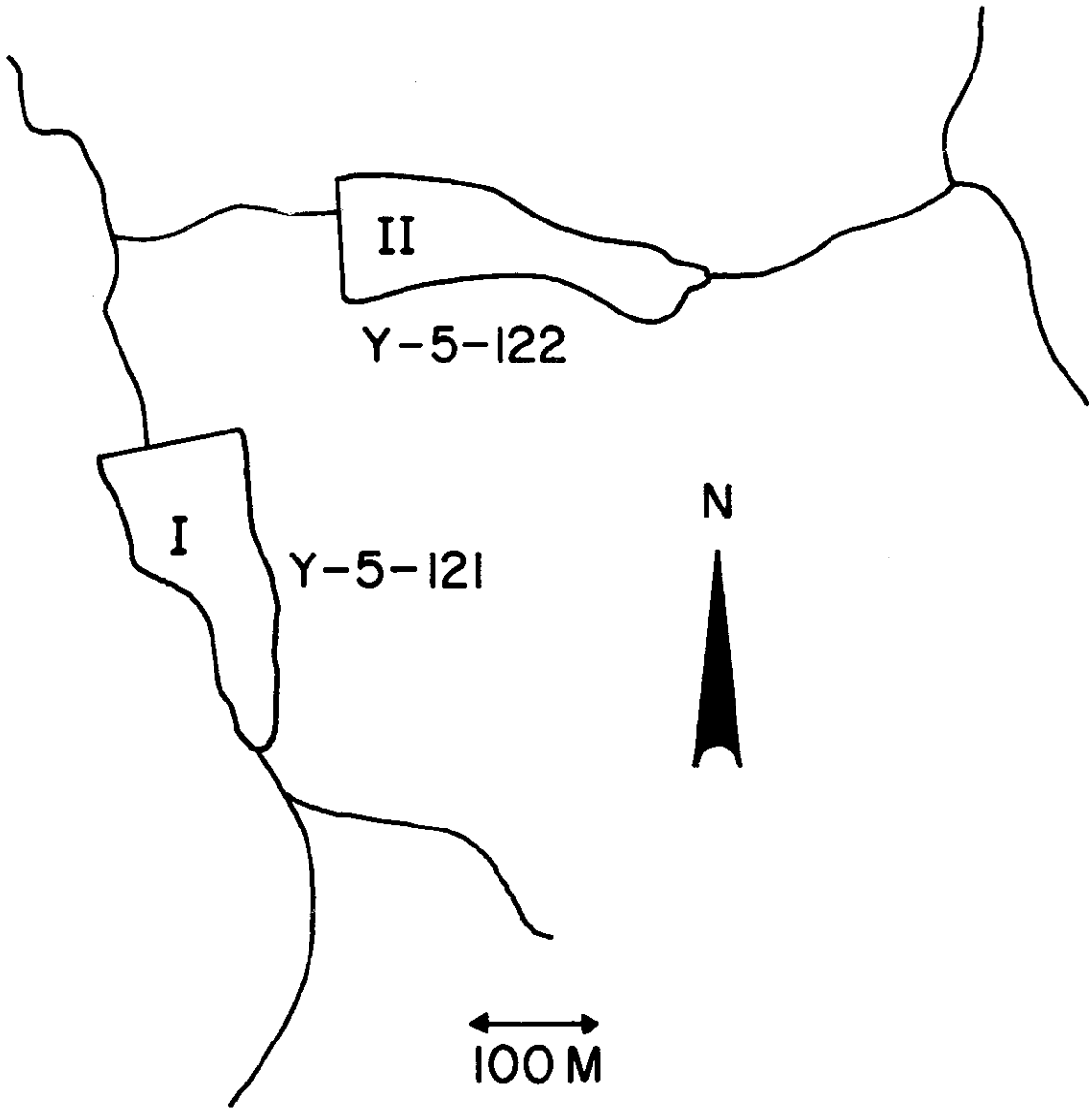

Fig. 2 Schematic of the Subject Reservoirs

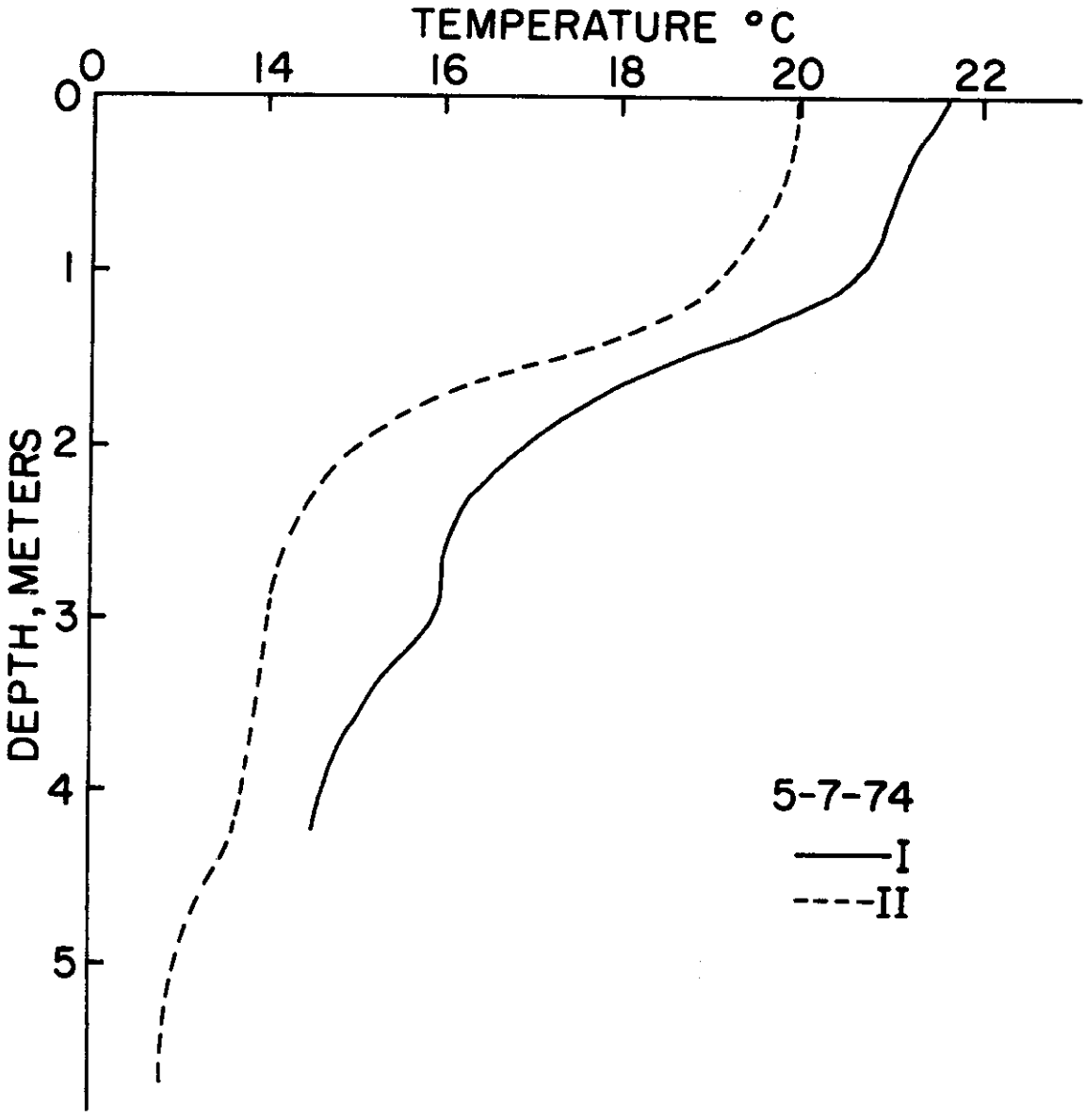

Fig. 3 Water Temperature Profiles for Two North Mississippi Reservoirs

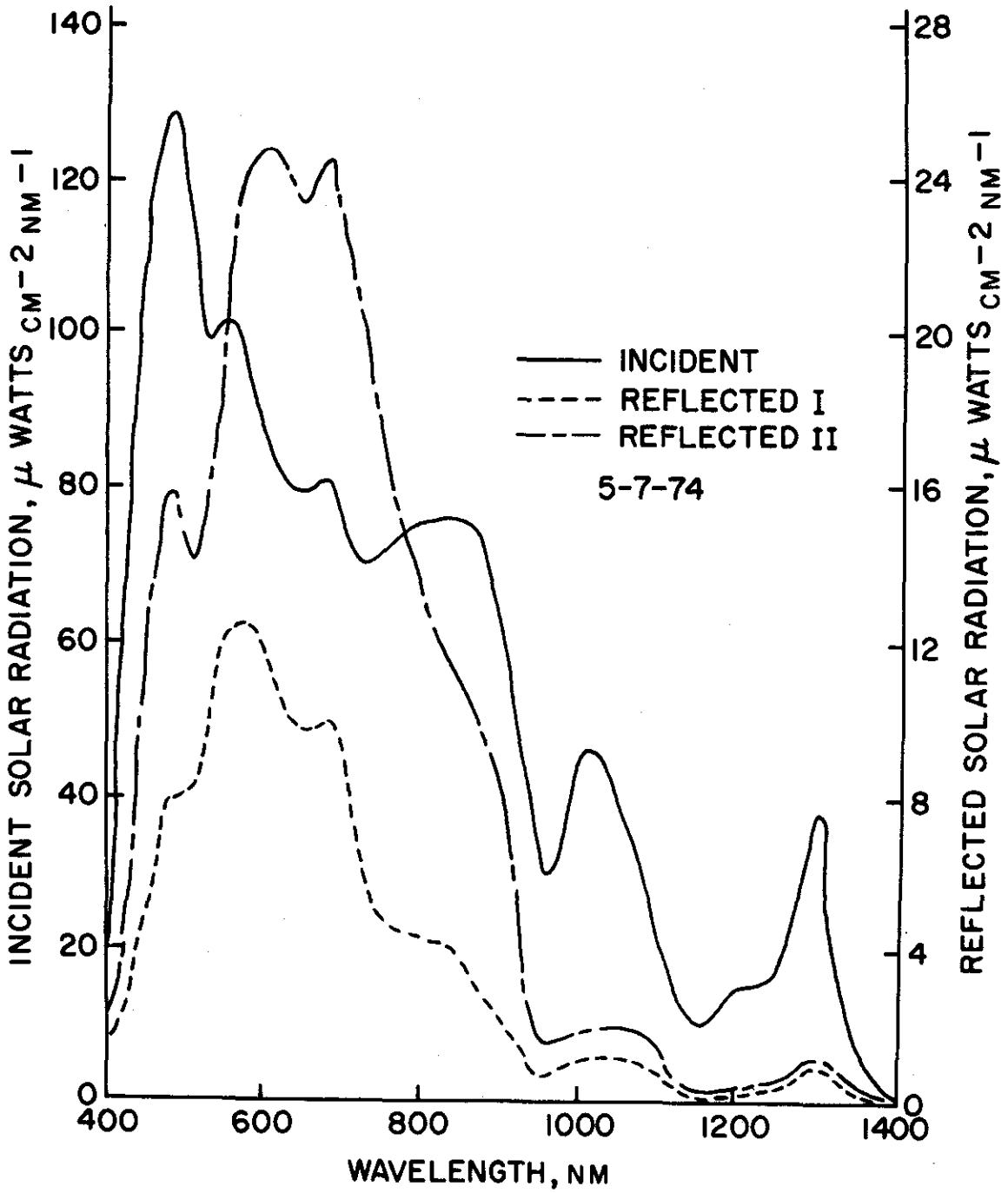

Fig. 4 Incident and Reflected Solar Radiation for two North Mississippi Reservoirs

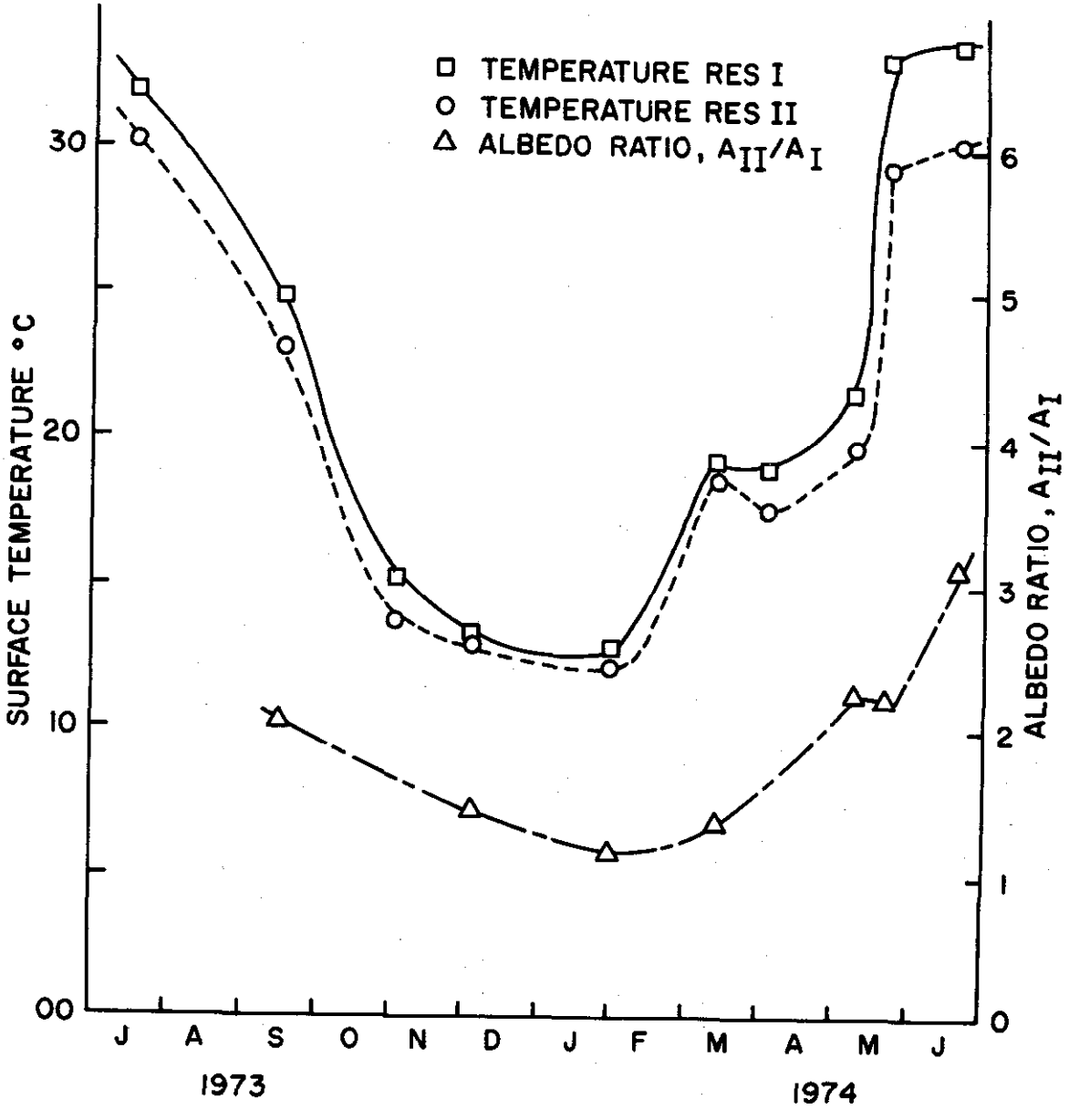

Fig. 5 Surface Temperatures and Albedo Ratios for Two North Mississippi Reservoirs

NUTRIENTS LOST IN DEBRIS AND RUNOFF WATER FROM A BURNED CHAPARRAL WATERSHED

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ABSTRACT

Runoff water and movement of debris were measured on a chaparral-covered watershed, in southern California, that was prescribe-burned in summer 1973. Plant nutrients in the runoff water and debris during the first rainy season were analyzed. The amounts of N, P, Ca, Mg, K, and Na in the soil, litter, and plant material after burning, but before erosion occurred, were measured. In the first year, an area with 50 percent slopes had about 7300 kg/ha (6300 lb/ac) of material eroded from the plots on the burned sites as compared to only 210 kg/ha (187 lb/ac) on similar unburned sites. On the burned area, less debris was lost from the plots having 20 percent slopes than from those on steeper sites. No erosion occurred on unburned slopes with a 20 percent slope. The total amount of nutrients lost in debris and runoff water from the steep slopes that burned, in kg/ha (lb/ac), were: nitrogen--15.1 (13.5); phosphorus--3.4 (3.0); potassium--27 (24.1); magnesium--31.6 (28.2); calcium--67.4 (60.2); and sodium--4.6 (4.1). Nutrients lost represented less than 6 percent of the total nutrients found in the upper 2-cm (0.79 in) of the soil and litter layer immediately after the fire.

INTRODUCTION

Fuel modification has been proposed as a possible solution to southern California's wildland fire problem. One technique that is gaining renewed interest is prescribed burning--fires set off under carefully prescribed environmental conditions. Questions have been raised, however, about the effects of prescribed burning on water quality.

Much of the proposed burning would take place in chaparral-covered mountains. The erosional processes in these mountains have been the subject of many studies (Krammes, 1963; Rowe, 1941; Sinclair, 1954). But little is known about the nutrients lost from burned chaparral areas. Studies of such losses in other areas are of limited use in southern California because of the differences in magnitude of erosion (DeByle, 1972; Fredricksen, 1970).

Erosion rates vary but some typical maximum rates reported for chaparral have been 49,000 kg/ha (43,750 lb/ac) on south-side slopes in the San Gabriel Mountains the first year after a fire (Krammes, 1963); 70,000

kg/ha (62,500 lb/ac) mainly channel scour material (Sinclair, 1954); 283,360 kg/ha (253,000 lb/ac) from annually burned plots in Sierra Nevada foothills (Rowe, 1941); and 346,000 kg/ha (308,900 lb/ac) for plots on side slopes during the first year after fire in the San Gabriel Mountains (Krammes, 1969). The amounts of nutrients in the debris lost after fire have not been quantified.

This paper describes an experiment to evaluate the nutrient losses from a chaparral brushland area that was prescribe-burned in the summer 1973, in southern California.

Study Area

The burned area lies on a 43-ha (105-ac) chaparral watershed in the Los Padres National Forest, northeast of Santa Maria, California. Elevation ranges from 550 m (1800 ft) to 610 m (2000 ft). Slopes are mostly between 10 and 50 percent, although some approach 100 percent. The burned watershed and main channel are generally oriented in a north-south direction. Therefore, most of the side slopes are at an east or west aspect.

The vegetation was climax chaparral containing dense stands of redshank (Adenostoma sparsifolium) and lesser amounts of chamise (Adenostoma fasciculatum), buck brush (Ceanothus cuneatus), manzanita (Arctostaphylos spp.), scrub oak (Quercus dumosa), and black sage (Salvia mellifera).

The underlying rock material is primarily Cretateous sandstones and conglomerates and Quaternary age alluvium and landslide breccia (Bergwall, 1973). Most of the material is highly calcareous, with many calcareous lithic tuft beds interspersed throughout the sandstones. The soils in the areas were mapped as gravelly loam Lithosols having an A horizon 0-10 cm (0-4 in) thick, dark yellow brown in color, a loam texture, weak to medium-fine granular structure and a pH of 6.6. The surface soil layers consisted of 50 percent sand, 36 percent silt, and 14 percent clay. The C horizon extended down to 46 cm (18 in) or deeper and had a rocky loam texture, with medium to blocky structure and a soft friable sticky consistency. As high as 80 percent of the C horizon was rock.

Study Methods

Before the prescribed burn, two study areas were designated within the boundary of the fire. One sampling site was on a steep slope (50 percent); and a second on a gentle slope (20 percent). Paired sites, having similar slope, aspect, soil, and vegetation were located in a nearby unburned area outside the boundary. Changes in dead and live biomass by species were determined by clipping 12-m^2 (131-ft^2) paired

plots before and after the prescribed burn. Litter and soil samples were also collected before and after fire. Litter was sampled by vacuuming all identifiable plant litter from a 0.49-m^2 (5.3-ft^2) plot. Soils were sampled at 1-cm (0.4-in) intervals down to 4 cm (1.6 in) and at 2-cm (0.8-in) intervals the remainder of the distance to 10 cm (4.0 in).[1/]

About 66 percent of the standing vegetation on the burned site was consumed during the prescribed burn on August 21, 1973. Maximum temperatures at the soil surface during the burn were between 340-370°C (650-700°F).

Debris Measurement

Plots for measuring debris and runoff were 3 m (10 ft) wide by 12 m (40 ft) long, on both burned and unburned areas. For replication, two plots were set up for each slope class on both burned and unburned areas (Fig. 1). Metal edging prevented water and debris running off upslope areas from entering the collection troughs placed at the lower end of the plots. Each trough was 3 m (10 ft) wide by .3 m (1 ft) deep and .3 m (1 ft) high. The steel troughs emptied into a pipe leading to a tipping bucket apparatus equipped with a Veeder counter mechanism for recording the volume of runoff water. From the tipping bucket water flowed into a can equipped with a siphon mechanism for filling a sample bottle. Once filled, no more water could enter the sample bottle. Water samples so collected represented only the first increments of runoff occurring during each storm because the sample bottle filled during the first two tips from the tipping bucket. Each tip represented runoff of about 630 l/ha (67 gal/ac). Because the sites were not easily accessible, we had to use a device that would remove a water sample automatically. Although these samples represented maximum concentrations it seemed better to report the concentrations as measured, accepting the possible overestimates, rather than trying to construct a curve describing the decreasing concentration over time which may or may not have been correct. The sample bottles were removed after each storm and analyzed in the laboratory for plant nutrients. The total amount of debris in the troughs was measured volumetrically after each storm. A representative subsample of the debris was analyzed in the laboratory for nutrients.

The amounts and intensities of rainfall were measured by a rain gage near the plots. Nutrients in the rainfall were not determined.

[1/] Details of methods presented in a paper in preparation; "The Impact of Prescribed Burning on the Nutrient Status of Vegetation, Litter, Soil, and Post-fire Erosion in Southern California Chaparral" by L. F. DeBano and C. E. Conrad.

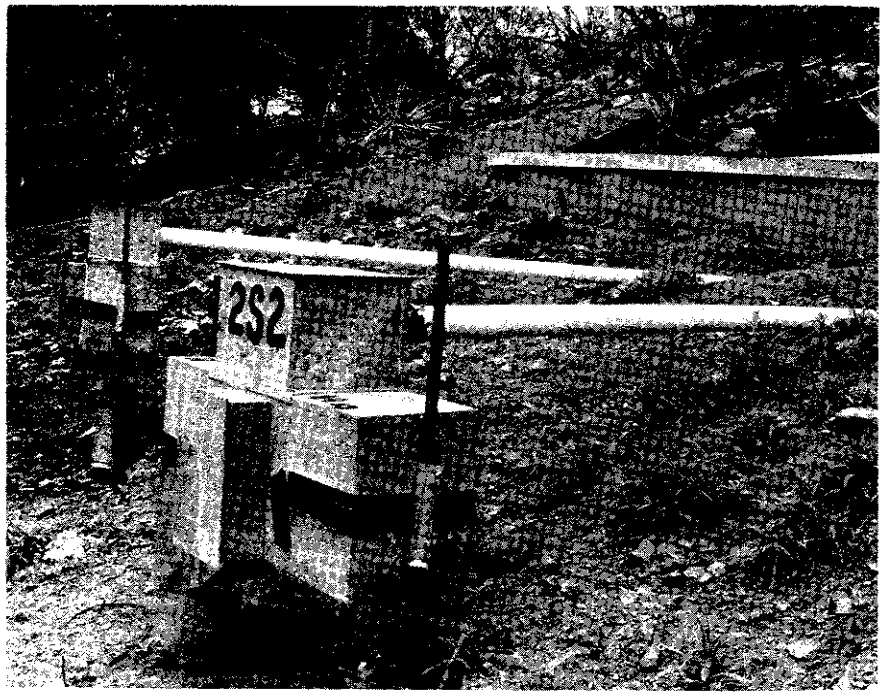

FIGURE 1. - Troughs and tipping buckets were used to measure erosion and nutrient loss from plots on a burned site.

Laboratory Analysis

Debris and runoff water samples were analyzed for nitrogen, phosphorus, potassium, magnesium, calcium, and sodium. Organic matter was also determined on the debris samples.

The debris samples were first air-dried and screened through a 2-mm screen. Any organic matter left was pulverized and added to the less-than-2-mm fraction for subsequent chemical analysis. The rock material greater than 2 mm was weighed and discarded. A subsample of the less-than-2-mm fraction was removed with a sample splitter for chemical analysis. Total nitrogen on the debris was determined by macro-Kjeldahl modified to include nitrates (Chapman, 1961; Bremner, 1965). Total phosphorus was determined by dry-ashing after saturating with magnesium nitrate (Piper, 1950). Phosphorus was measured colorimetrically either by the vanadate-molybdate-yellow method when the concentration was reasonably high or by the ammonium-molybdate-stannous chloride method when the concentrations were low (Chapman, 1961). Cations of potassium, magnesium, calcium, and sodium were determined by dry-ashing litter material for 4 hours at 510°C (950°F) after saturating with sulfuric acid and ethanol (Chapman, 1961). The cations were measured in the aqueous solution of ash by atomic absorption. A hydrogen-air flame was used for determining sodium and potassium and a nitrous oxide-acetylene flame, with either a cesium or potassium chloride suppressant, for determining calcium and magnesium (Varion Techtron, 1971).

Organic matter was measured by dry-ashing the litter samples at 510°C (950°F) for 4 hours. The water samples were filtered and analyzed for potassium, calcium, magnesium, and sodium. The concentration of dissolved phosphates and nitrogen checked in some of the earlier samples was so low that analysis was discontinued. The concentration of cations in the filtered solution was determined by atomic absorption, using a procedure similar to that for the ashed litter samples.

Results

Total Erosion and Nutrient Loss

The largest amount of debris produced was from burned plots on steep slopes, which yielded 7300 kg/ha (6500 lb/ac) (Table 1). The gentle slopes yielded about 2800 kg/ha (2500 lb/ac). The debris leaving the unburned control plots was only 210 kg/ha (186 lb/ac) from the steep slopes. No debris was produced on the unburned gentle slopes.

The difference in runoff was not as great as was that in debris yield from the two slope classes. The burned plots on the steep slopes yielded about 780,000 l/ha (84,000 gal/ac) of runoff during the first rainy season. The burned plots on gentle slopes yielded 500,000 l/ha

(54,000 gal/ac) of runoff. The amount of water running off the unburned control plots was smaller than that off the burned plots. The steep unburned plots yielded, on the average, 24,000 l/ha (2600 gal/ac) of runoff water; the gentle slopes, only 4500 l/ha (480 gal/ac). The surface depth of water yielded from the steep unburned areas was less than 0.25 cm (0.10 in).

The steep burned plots lost significant amounts of plant nutrients during the first rainy season. The greatest nutrient losses were in the debris with smaller amounts in the runoff water (Table 1). Most losses in the debris were probably in the organic matter. Organic matter content of the debris varied between steep and gentle slopes, indicating a difference in the composition of the eroded material (Table 1). The debris leaving the steep slopes contained 5.6 percent organic matter and that from the gentle burned slopes contained 10.9 percent. Debris from the gentle slopes had a higher organic matter content because the runoff water had less erosive power thereby removing material having a larger proportion of less dense organic matter. In contrast, the erosive power of runoff from steep slopes was greater and the debris contained higher proportion of dense mineral soil. In spite of the differences in organic matter content of the debris, the steep slopes lost about 100 kg/ha (89 lb/ac) more organic matter than did the gentle slopes.

The runoff water and debris from the steep burned plots contained: 15.08 kg/ha (13.46 lb/ac) of nitrogen, 3.37 kg/ha (3.01 lb/ac) of phosphorus, 27.01 kg/ha (24.12 lb/ac) of potassium, 31.65 kg/ha (28.26 lb/ac) of magnesium, 67.43 kg/ha (60.21 lb/ac) of calcium, and 4.57 kg/ha (4.08 lb/ac) of sodium. The unusually high amounts of calcium yielded by the plots probably occurred because the soil and parent rock material in the area was calcareous. Analyses of plant, litter, and soil showed high quantities of calcium. The amount of plant nutrients lost from the burned gentle slopes were: 7.50 kg/ha (6.70 lb/ac) of nitrogen, 1 kg/ha (0.89 lb/ac) of phosphorus, 10.90 kg/ha (9.73 lb/ac) of potassium, 8.06 kg/ha (7.20 lb/ac) of magnesium, 27.61 kg/ha (24.64 lb/ac) of calcium, and 2.13 kg/ha (1.90 lb/ac) of sodium.

Only small quantities of nutrients were lost from the unburned control sites (Table 1). On the steep slopes, only small amounts of nutrients were lost because the quantity of eroded material was small. On the plots having gentle slopes, no erosion occurred and so the only nutrient losses were trace amounts in the runoff water.

Nutrient Loss Over Time

Generally, the concentrations of plant nutrients in the debris were not related to the amount of debris lost during any sampling period. In many instances, the concentrations of nutrients in large and small amounts of debris were the same (Table 2). No debris was recorded on

January 10, 1974 because the troughs were not cleaned until January 16. However, water samples of runoff were collected on both dates. Aside from this irregularity the amounts of debris lost by each successive storm generally decreased, but the amounts and intensities of rainfall during the later storms did not necessarily decrease. The last sampling date for 1974 (April 5) yielded the least amount of nutrients--even during a storm totaling 6.52 cm (2.57 in) of rainfall and having a maximum 15-minute intensity of 0.91 cm/hr (.36 in/hr).

Because the concentrations of nutrients in the debris did not change during the season, the amounts of nutrients lost paralleled the losses in debris. Both debris and amount of plant nutrients generally decreased over time (Tables 2, 3). The concentrations of nutrients in the runoff water generally decreased over time (Table 3). This decrease would be expected because the soluble nutrients were generally removed from the site throughout the rainy season. At the last sampling period the concentrationsof the nutrients in the runoff water was at relatively low levels (Table 3).

Post-fire Nutrients Lost by Erosion

The nutrients remaining in the ash and upper soil layers after the fire were probably highly soluble and mobile. Solubility of nutrients in litter and plant material is greatly increased by burning (Sampson, 1944; White, 1973; Lewis, 1974). In this study, the nutrients lost in runoff water and debris represented only a small percent of the nutrients contained in the ash and upper two soil layers (0-1 and 1-2 cm) immediately after the fire (Table 4). The nutrients in the ash and upper soil layers were considered to be "readily erodible" because of exposure to raindrop impact and surface erosion. Soluble nutrients in the ash probably leached into the soil during the first stages of infiltration (Grier, 1971). Some of the leached nutrients were probably lost from the site by deep leaching and percolation but this loss was small because of limited rainfall.

Nitrogen and phosphorus losses were measured only in the debris because concentrations in the runoff water were low. Concentrationsof nitrogen and phosphorus in the few samples of runoff water analyzed were below 5 ppm. The 15 kg/ha (13.4 lb/ac) of nitrogen lost by erosion during the first year represented only 3.4 percent of the nitrogen contained in the ash and upper two soil layers immediately after the fire. Slightly less phosphorus was lost and represented only 2.4 percent of the readily erodible phosphorus on the site.

Potassium lost in runoff water and debris amounted to 9.7 percent of the total potassium in the ash and upper two soil layers. It is not surprising that the amount of potassium lost by erosion was high because more than 217 kg/ha (194 lb/ac) of this element was in the litter and ash layer, where it was readily erodible (Table 4). The amount of

potassium lost in debris was larger than that contained in runoff water although there were significant amounts lost in the water. About 10.9 percent of the magnesium, 3.5 of the calcium, and 11.4 of the sodium contained in the ash and upper two soil layers after the fire was lost in runoff water and debris.

Although the relative solubilities of the nutrients after fire were not evaluated, the percent of nutrients in runoff water found in the ash and soil layers immediately after the fire suggest the ease with which these nutrients were lost by leaching. For example, 2.8 percent of the total potassium in the ash and upper two soil layers was lost as runoff (Table 4). The percent sodium leaving the site in runoff water was greater than that of potassium (5 percent). Although much calcium was lost in runoff, the percentage was less than for any of the other cations (1.02 percent). Magnesium losses amounted to 1.26 percent. The loss of cations in runoff water seems consistent with the relative strength of adsorption of these ions on the exchange sites of soil and organic matter (Viro, 1974). The classical replacement series, in which the increasing strength of adsorption is Na<K<Mg<Ca, seems to agree with the percent loss of these nutrients, although the total amount of nutrient loss from the site seems more closely associated with the amount of nutrients in the ash and upper two soil layers. The mobility of these ions in decomposing litter follows the same series as the percent of the nutrients in the runoff water (Attiwill, 1968). Perhaps fire can be viewed merely as force accelerating the decomposition of litter and basic relationships concerning it still hold.

Discussion

The rates of maximum erosion in this study were larger than those reported for other areas. They were completely dwarfed however, by the amounts of debris reported by past experiments on burned chaparral watersheds (Rowe, 1941; Sinclair, 1954; Krammes, 1963; Krammes, 1969). The maximum annual erosion rate measured in our study was 7700 kg/ha (6900 lb/ac) from the plots on the steep slopes (Table 1). The amounts reported by those earlier studies ranged from 49,000 kg/ha (43,750 lb/ac) up to 346,000 kg/ha (308,900 lb/ac). We cannot account for the lower erosion rates, although obvious differences exist in geology, soils, and rainfall during the studies.

Although the loss of nutrients was large, the fertility of the site was not seriously depleted. Similarly, it is not likely that fertility would be seriously depleted by any one fire or series of fires at time intervals similar to the pattern that has apparently been part of the evolution of California chaparral (Burcham, 1960). It would be easy to conclude that the nutrients will become severely limiting within a couple of burning cycles. Of particular concern might be the loss of nitrogen where considerable amounts are removed from the site by erosion in addition to even greater losses during a fire. We estimated about

138 kg/ha (123 lb/ac) of nitrogen were volatilized by the fire. This loss added to the 15 kg/ha (13 lb/ac) lost by erosion amounts to 150-160 kg/ha (134-143 lb/ac) lost as a result of fire. If this amount of nitrogen is not replaced during each burning cycle, the site would quickly be depleted of nitrogen. The site probably had at least 500-600 kg/ha (446-536 lb/ac) of nitrogen in the soil, vegetation, and litter before the fire. The chaparral plant community apparently can adapt to a regular depletion of nitrogen by fire.

Mechanisms allowing chaparral plants to survive these vicissitudes of fertility are available. First, species in the chaparral community seem better adapted to low fertility levels than other species (Hellmers, 1955). Second, the nutritional well-being of the chaparral plants, from the standpoint of nitrogen, may be enhanced by the post-fire invasion of nitrogen-fixing plants. These plants are either long-lived shrubs, such as Cercocarpus betuloides (Vlamis, 1964) and Ceanothus leucodermis (Hellmers, 1959), short-term semi-woody plant (Lotus scoparius), or herbaceous annuals (Lotus sp.). The nitrogen-fixing capacity of a Ceanothus stand has been estimated to be about 60 kg/ha (53 lb/ac) annually (Delwiche, 1965).

Nitrogen may also be fixed by microorganisms. About 56 kg/ha (50 lb/ac) seem a reasonable estimate for the nitrogen fixed annually by nonsymbiotic bacteria (Jorgensen, 1971). From 10 to 100 kg/ha (9 to 89 lb/ac) of nitrogen have been reported fixed by blue-green algae-lichen crusts each year in the Great Basin desert (Rychert, 1974), but how important these organisms are in chaparral communities is not known. Rainfall provides still another mechanism for adding nitrogen to the site. Rainfall samples taken in the general area of this study, in 1975, showed the annual addition of nitrogen in rain water is probably less than 4 kg/ha (3.6 lb/ac).

Conclusions

Substantial amounts of plant nutrients are either lost by heating during the fire or by erosion after the burn. The quantities of nutrients lost by erosion in this study were probably insignificant when compared to the losses experienced in other areas of southern California after wildfires. And the losses are not necessarily irreparable. Plant nutrients may be temporarily depleted, but they can be replaced by various means, such as the production of new cations by the weathering of parent rock material or by the biological fixation of nitrogen.

Acknowledgment

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Table 1. Loss of nutrients from erosion on burned and unburned areas on steep and gentle slopes the first year after a prescribed burn, southern California 1973.

Slopes	Total debris	Total runoff	Nutrients in debris						Organic material		Nutrients in runoff water			
			N	P	K	Mg	Ca	Na			K	Mg	Ca	Na
	kg/ha x 10^2	l/ha x 10^4	kg/ha						kg/ha	pct	kg/ha			
Burned														
Steep														
Rep I	69.36	61.04	13.87	3.22	17.15	21.62	43.92	2.25	407.1	5.9	6.77	3.13	18.34	1.51
Rep II	77.44	96.14	16.29	3.52	21.53	34.42	50.87	2.89	413.6	5.3	8.57	4.12	21.74	2.48
Ave.	73.40	78.59	15.08	3.37	19.34	28.02	47.39	2.57	410.4	5.6	7.67	3.63	20.04	2.00
Gentle														
Rep I	29.80	79.45	7.25	1.04	7.02	6.16	18.53	0.87	313.6	10.5	4.54	2.55	12.56	1.64
Rep II	27.15	37.26	7.75	0.96	8.27	6.14	18.42	0.81	309.3	11.4	1.98	1.27	5.71	0.93
Ave.	28.48	58.36	7.50	1.00	7.64	6.15	18.47	0.84	311.5	10.9	3.26	1.91	9.14	1.29
Unburned														
Steep														
Rep I	0.47	2.5	0.07	0.02	0.15	0.14	0.15	0.02	2.1	4.5	0.09	0.10	0.55	0.11
Rep II	3.7	2.2	0.51	0.13	0.84	0.80	0.89	0.11	12.1	3.3	0.09	0.04	0.26	0.08
Ave.	2.1	2.4	0.29	0.08	0.50	0.47	0.52	0.07	7.1	3.4	0.09	0.07	0.41	0.10
Gentle														
Rep I	0	0.32	0	0	0	0	0	0	0	0	0.008	0.004	0.029	0.006
Rep II	0	0.57	0	0	0	0	0	0	0	0	0.014	0.006	0.050	0.013
Ave.	0	0.45	0	0	0	0	0	0	0	0	0.011	0.005	0.040	0.010

Table 2. Nutrients in debris during the first year after a prescribed burn from a burned plot on a steep slope, southern California 1973.

Date	Rain fall (cm)	Debris kg/ha x 10^2	Concentrations (ppm x 10^4)						Total amounts (kg/ha)					
			N	P	K	Mg	Ca	Na	N	P	K	Mg	Ca	Na
Nov 27/73	6.07	47.97	0.18	0.046	0.23	0.30	0.61	0.029	8.53	2.18	10.90	14.22	28.92	1.37
Dec 4/73	3.51	6.53	0.21	0.042	0.29	0.31	0.56	0.045	1.37	0.27	1.86	2.02	3.65	0.29
Jan 10/74	11.33	1/	1/	1/	1/	1/	1/	1/	1/	1/	1/	1/	1/	1/
Jan 16/74	2.59	5.36	0.31	0.058	0.27	0.32	0.85	0.046	1.65	0.31	1.46	1.70	4.52	0.24
Mar 6/74	9.07	7.60	0.24	0.049	0.30	0.39	0.73	0.039	1.81	0.37	2.23	2.92	5.51	0.29
Apr 5/74	6.52	1.90	0.27	0.048	0.37	0.41	0.70	0.034	0.51	0.09	0.70	0.76	1.32	0.06
Total	39.09	69.36							13.87	3.22	17.15	21.62	43.92	2.25

1/ Debris present but not measured until January 16, 1974.

Table 3. Nutrients in runoff water during the first year after a prescribed burn from a burned plot on a steep slope, southern California 1973.

Date	Maximum Intensity (cm/hr) 1/	Runoff cm	Concentrations				Total amounts			
			K	Mg	Ca	Na	K	Mg	Ca	Na
			ppm				kg/ha			
Nov 27/73	2.03	1.75	17.4	8.5	54.0	3.2	3.05	1.49	9.47	0.56
Dec 4/73	1.93	0.90	14.0	7.9	25.1	3.2	1.27	0.71	2.27	0.29
Jan 10/74	1.01	0.59	9.9	3.6	24.4	4.0	0.58	0.21	1.43	0.24
Jan 16/74	2/	0.09	7.0	3.3	23.5	2.5	0.06	0.03	0.21	0.02
Mar 6/74	0.91	1.78	8.2	3.1	22.7	1.3	1.46	0.55	4.05	0.23
Apr 5/74	0.91	0.99	3.5	1.4	9.2	1.7	0.35	0.14	0.91	0.17
Total		6.10					6.77	3.13	18.34	1.51

1/ Maximum rainfall intensities for a 15-min. period.

2/ Rainfall intensity gage malfunctioned.

Table 4. Amounts of nutrients on the steep slopes immediately after fire and the amounts lost by erosion, southern California 1973.

Item	N	P	K	Mg	Ca	Na
	Nutrients on slopes					
	kg/ha					
Plant remains	33.0	0.83	23.7	4.4	86.7	3.2
Litter and ash	138.3	31.3	217.4	233.3	601.4	20.7
Soil 0-1 cm	161.9	56.7	34.7	26.5	654.3	9.0
Soil 1-2 cm	146.7	51.7	25.1	29.4	700.6	9.3
Erodible nutrients[1/]	446.9	139.7	277.2	289.2	1956.3	40.0
	Erosion loss					
	kg/ha					
Debris	15.08	3.37	19.34	28.02	47.39	2.57
Runoff	2/	2/	7.67	3.63	20.04	2.00
Total	15.08	3.37	27.01	31.65	67.43	4.57
	Nutrient loss					
	Percent of erodible nutrients[1/]					
Runoff and debris	3.37	2.41	9.74	10.94	3.45	11.42
Runoff water	2/	2/	2.77	1.26	1.02	5.00

1/ Easily erodible from the ash, soil 0-1 cm deep, and 1-2 cm deep

2/ Trace amounts.

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PHYSICAL AND CHEMICAL CHARACTERISTICS OF SEDIMENTS ORIGINATING FROM MISSOURI VALLEY LOESS

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ABSTRACT

Nitrogen (N) and phosphorus (P) associated with sediment in surface runoff accounted for at least 85% of N and P discharged from agricultural watersheds. The N and P content of the sediment is related to its carbon (C) content. At low flow, the C concentration of the sediment increased, indicating the selectivity of the erosion process for organic material. The sediment showed an enrichment in the clay fraction over that contained in the surface soil, and this increase can result in a disproportionate increase in the discharge of those chemicals adsorbed to clay material.

INTRODUCTION

Recently much research has focused on agriculture's contribution to the deterioration in the quality of some of our water resources by eroded soil and agricultural chemicals. Assessing the problem has been difficult because of the diverse land and climatic situations involved.

Plant nutrient discharge occurs as water-soluble ions in runoff and adsorbed ions associated with the sediment portion of the runoff. Harms, Dornbush, and Andersen (1974) concluded that a large percentage of the nutrients discharged from seven sites in eastern South Dakota were soluble; however, sediment loss from these sites was quite low. Schuman et al. (1973a) found that N (nitrogen) losses in the soluble form were low and that sediment-associated N accounted for 94% of the N loss for a 3-year period on a contour-planted, corn watershed. However on a grass watershed where sediment discharge was low the soluble N loss accounted for 50% of the total N discharge. Burwell, Timmons and Holt (1975) also found that sediment-associated N accounted for a large percentage of the N discharged in surface runoff. Schuman and Burwell (1974) pointed out that at least one-half to two-thirds of the soluble N in surface runoff was equivalent to the amount of N in the precipitation; therefore, the soluble-N discharge attributable to fertilizer N may be

a very small fraction of the total N discharge. Schuman et al. (1973b) reported that a large percentage of the P (phosphorus) discharged in surface runoff is also associated with the sediment. Investigators have reported that the transport of sediment-borne chemicals from agricultural watersheds is disproportionately large in comparison to the chemical content of surface soils, due to selective erosion of clays (Massey et al. (1953); Scarseth and Chandler (1938).

Runoff and erosion are important to everyone because of economic and environmental concerns. Approximately 4 billion tons of sediment are carried into U. S. streams each year (Wadleigh, 1968). This figure could be reduced if good conservation and management practices were used to protect erosive cultivated cropland. Sediment is also a major cost item in reservoir sedimentation. Syers et al. (1973), Keeney (1973), and Kerr et al. (1973) thoroughly reviewed the literature on P, N, and C and their relationship to, and cycling in, the sediment water system. The purpose of the study reported here-in was to evaluate the N and P discharges and the N, P, and C content of sediment in surface runoff in western Iowa and to ascertain how these are related to the selectivity (enrichment) of the erosion process.

MATERIALS AND METHODS

The four experimental watersheds are located in southwestern Iowa near Treynor. The area is characterized by a deep loess cap, underlain by till, ranging in thickness from 25 m (82 ft) on the ridges to less than 5 m (16 ft) in the valleys. Gully and sheet-rill erosion are serious problems, and many valleys have deeply incised channels that extend upstream to an active gully head.

Principal soil types are Typic Hapludolls, Typic Haplorthents, and Cumulic Hapludolls. These soils are fine, silty, mixed mesics, with moderate to moderately rapid permeability. Slopes on the watersheds range from 2 to 4% on the ridges and bottoms to 12 to 18% on the sides.

The four watersheds were instrumented in 1964 to measure precipitation and streamflow. Precipitation measurements were obtained from recording raingages located on each watershed. Stream discharges were measured by waterstage recorders used with calibrated broad-crested, V-notch weirs. Watershed size, crop, conservation practice, and fertilizer application are shown in Table 1. The contour-planted corn watershed (WS-2) and the pasture watershed (WS-3) were fertilized at the recommended rate, 168 kgN/ha and 39 kgP/ha (150 lbN/A and 35 lbP/A). The level-terraced (WS-4) and a contour-planted (WS-1) corn watershed were fertilized at 2.5 times the recommended rates. These treatments were selected to enable us to evaluate the conservation watersheds (WS-3 and 4) in controlling or reducing nutrient loss due to surface runoff and erosion.

Table 1. Watershed description, crop, conservation practice, and fertilizer application rates for 1969, 1970, 1971, and 1972, Treynor, Iowa

Watershed	Size		Crop	Conservation practice	Fertilizer	
					N	P
	ha	(acres)			kg/ha (lb/A)	
1	30.4	(74.5)	Corn	Contour	448 (400)	97[1/] (87)
2	33.6	(83)	Corn	Contour	168 (150)	39 (35)
3	43.3	(107)	Brome-grass[2/]	Rotation grazing[2/]	168 (150)	39 (35)
4	60.8	(150)	Corn[3/]	Terraced	448 (400)	97 (87)

1/ Phosphorus fertilizer was applied at the rate of 39 kg/ha (35 lb/acre) on all watersheds in 1971 and 1972.

2/ Watershed 3 was changed to a mulch-tillage, contour planted corn Watershed in 1972.

3/ Watershed 4 was changed to mulch tillage, with terrace channels having surface pipe drains and fertilized at 168 kgN/ha (150 lbs N/acre) and 39 kgP/ha (35 lbs P/acre).

Samples of approximately 400 ml (16 oz) of the soil-water runoff were collected manually during each surface runoff event to determine sediment concentration and particle sizes and N, P, and C content of the sediment. The sediment content of each sample was determined gravimetrically. Total N was determined on the oven-dried sediment, using micro-Kjeldahl procedures as described by Bremner (1965). The sediment P content was determined on a $NaHCO_3$ extract (Olsen et al., 1954; Murphy and Riley, 1962). Total P was not determined on the sediment because it would not be meaningful from a pollution standpoint. Taylor (1967) states that less than 10% of the total P of soils will be considered a part of the "labile pool." Carbon content of the sediment was determined by dry combustion with a high-temperature induction furnace. The N, P, and C concentrations are expressed on a dry-sediment basis. Surface soil samples 0- to 10-cm (0- to 4-inch) were collected at Watershed 2 for comparison with sediment moved in storm runoff. Particle size distribution of streamflow and surface soil samples were determined by the sieve and pipette methods. Watershed 2 surface soils were sampled at 26 locations representing the ridges, side slopes, and colluvial deposits; the particle-size analyses showed that they were quite uniform within and among the slope classes. Seventy-three particle-size analyses of sediment in runoff at watershed 2 were the basis for calculating the discharge-weighted sediment outflow for the 11-year period of record, 1964-1974; this was accomplished by the flow duration-sediment transport curve procedure (Campbell and Bauder, 1940), where the 11-year runoff record was tabulated by 1- or 2-minute increments during periods of rapidly changing flow--and the runoff-sediment relation for each of four size classes was compiled (figure 3).

RESULTS AND DISCUSSION

Nitrogen, Phosphorus, and Carbon Content of Sediment

Nutrient discharges associated with the sediment was 37.59 kgN/ha (32.64 lbN/A) and 1.05 kgP/ha (0.936 lbP/A) for WS-1 and 23.16 kgN/ha (20.66 lbN/A) and 0.581 kgP/ha (0.518 lbP/A) for WS-2 (Table 2 and 3). Nitrogen and phosphorus associated with the sediment accounted for 94 and 85%, respectively, of the total discharge in surface runoff from the contour-cropped watersheds 1 and 2 at Treynor, Iowa. Sediment N losses were 86 and 51%, of the total N losses for the level terraced (WS-4) and pasture watershed 3, respectively. To evaluate the fertility treatment effect, the erosion differences for the watersheds must be taken into account. Weighted concentrations of nutrients in the transporting medium can be derived by dividing sediment N and P losses (Tables 2 and 3) by the corresponding sediment yields (Table 4). The average annual sediment N loss per unit of sediment for the recommended (WS-2) and high N fertilizer (WS-1) application rates was 1.40 and 1.45 kgN/MT of sediment (2.79 and 2.89 lbN/t), respectively. These data indicate little fertility treatment effect on the amount of N carried by the sediment. Sediment N loss per unit of sediment for Watersheds 3 and 4 was 2.01 and 1.97 kgN/MT of sediment (4.00 and 3.99 lb/t), respectively. The same rela-

Table 2. Nitrogen loss in surface runoff from agricultural watersheds, Treynor, Iowa

Watershed	Solution N		Sediment N[1/]	Total loss
	NO_3-N	NH_4-N		
	kg/ha (lb/acre)			
1969				
1	2.30 (2.05)	1.55 (1.38)	5.86 (5.23)	9.71 (8.66)
2	1.45 (1.29)	0.95 (0.85)	2.96 (2.64)	5.36 (4.78)
3	1.14 (1.02)	0.66 (0.59)	0.52 (0.46)	2.32 (2.07)
4	0.24 (0.21)	0.12 (0.11)	0.31 (0.28)	0.67 (0.60)
1970				
1	1.46 (1.30)	0.41 (0.37)	34.78 (31.02)	36.65 (32.69)
2	0.53 (0.47)	0.35 (0.31)	25.18 (22.46)	26.06 (23.24)
3	0.17 (0.15)	0.09 (0.08)	0.21 (0.19)	0.47 (0.42)
4	0.15 (0.13)	0.03 (0.03)	0.52 (0.46)	0.70 (0.62)
1971				
1	1.31 (1.17)	2.11 (1.88)	69.13 (61.66)	72.55 (64.71)
2	0.94 (0.84)	1.46 (1.30)	41.35 (36.88)	43.75 (39.03)
3	0.96 (0.84)	0.43 (0.38)	2.89 (2.58)	4.28 (3.82)
4	0.16 (0.14)	0.58 (0.52)	7.04 (6.28)	7.78 (6.94)
1969-1971 average				
1	1.69 (1.51)	1.36 (1.21)	37.59 (32.64)	39.64 (35.36)
2	0.97 (0.87)	0.92 (0.82)	23.16 (20.66)	25.05 (22.35)
3	0.76 (0.67)	0.39 (0.35)	1.21 (1.08)	2.36 (2.10)
4	0.18 (0.16)	0.24 (0.22)	2.62 (2.34)	3.04 (2.72)

[1/] Kjeldahl nitrogen

Table 3. Annual loss of P by surface runoff and sediment from experimental watersheds, Treynor, Iowa, 1969-1971

Watershed	Loss of phosphorus by surface runoff		Total loss of phosphorus[1]
	Solution	Sediment	
	kg/ha (lb/acre)		
1969			
1	0.189 (0.169)	0.306 (0.273)	0.495 (0.442)
2	0.093 (0.083)	0.164 (0.146)	0.257 (0.229)
3	0.193 (0.172)	0.058 (0.052)	0.251 (0.224)
4	0.081 (0.072)	0.009 (0.008)	0.090 (0.080)
1970			
1	0.085 (0.076)	0.949 (0.847)	1.034 (0.923)
2	0.046 (0.041)	0.477 (0.426)	0.523 (0.467)
3	0.064 (0.057)	0.017 (0.015)	0.081 (0.072)
4	0.005 (0.005)	0.015 (0.013)	0.020 (0.018)
1971			
1	0.237 (0.211)	1.893 (1.689)	2.130 (1.900)
2	0.189 (0.169)	1.101 (0.982)	1.290 (1.151)
3	0.386 (0.344)	0.126 (0.112)	0.512 (0.456)
4	0.059 (0.053)	0.229 (0.204)	0.288 (0.257)
1969-1971 average			
1	0.171 (0.152)	1.050 (0.936)	1.221 (1.088)
2	0.110 (0.098)	0.581 (0.518)	0.691 (0.616)
3	0.216 (0.191)	0.067 (0.060)	0.283 (0.251)
4	0.049 (0.043)	0.085 (0.075)	0.134 (0.128)

[1] Total loss values represent the inorganic P of the solution and the $NaHCO_3$-extractable P of the sediment.

Table 4. Annual precipitation, runoff, and sediment yield for experimental watersheds, Treynor, Iowa, 1969-1971

Watershed	Precipitation	Surface runoff	Sediment yield
	cm (inches)		metric ton/ha (ton/acre)
1969			
1	79.81 (31.42)	6.43 (2.53)	4.10 (1.83)
2	80.11 (31.54)	5.97 (2.35)	2.37 (1.06)
3	77.83 (30.64)	4.39 (1.73)	0.27 (0.12)
4	77.98 (30.70)	0.69 (0.27)	0.13 (0.06)
1970			
1	80.04 (31.51)	5.44 (2.14)	27.04 (12.06)
2	78.28 (30.82)	4.55 (1.79)	17.94 (8.00)
3	73.30 (28.86)	0.94 (0.37)	0.09 (0.04)
4	73.13 (28.79)	0.33 (0.13)	0.22 (0.10)
1971			
1	73.48 (28.93)	12.55 (4.94)	44.80 (19.98)
2	73.71 (29.02)	9.75 (3.84)	29.50 (13.16)
3	75.44 (29.70)	3.86 (1.52)	1.43 (0.64)
4	76.10 (29.96)	1.80 (0.71)	3.63 (1.62)
1969-1971 average			
1	77.78 (30.62)	8.14 (3.20)	25.31 (11.29)
2	77.37 (30.46)	6.76 (2.66)	16.60 (7.40)
3	75.52 (29.73)	3.06 (1.20)	0.60 (0.27)
4	75.74 (29.82)	0.94 (0.37)	1.33 (0.59)

tionship can be shown for sediment P per unit of sediment. Although these weighted concentrations are higher for watersheds 3 and 4, the small sediment discharge from these conservation watersheds held nutrient losses to a minimum.

Historically, lower erosion from WS-3 and 4 than from WS-1 and 2 may have resulted in organic matter accumulation differences, accounting for these sediment N differences. The organic C differences of the sediment from the various watersheds can be seen in Figure 1. The C:N ratio of the sediment ranges from 7:1 to 11:1. The sediment originating from the grass pasture and level-terraced corn watershed is generally one-half to two times higher in C than the sediment from the contour-planted corn watersheds. This difference in C content of the sediment is related to the watershed management which has resulted in the accumulation of organic material.

The sediment N and P concentrations are fairly constant during a runoff event except at low flows. Schuman et al. (1973a) and Burwell et al. (1975) postulated that the increase in sediment N concentration at low flow was related to a greater percentage of the total sediment sample being coloidal-size organic residue. Figure 1 shows that the sediment N and P vary with the C content of the sediment for the May 6, 1971, storm on all watersheds. Figure 2 shows that at low flows the C content of the sediment increased and is inversely related to the sediment concentration.

ENRICHMENT OF SEDIMENT

The particle size distribution of the sediment in transport from the contour Watershed 2 has shown a general increase in the clay size fraction from 26 to 29% over that of the 0- to 10-cm (0- to 4-inch) depth of surface soil (Table 5). For this watershed, the 11-year average sediment loss per year is 1452 MT/yr (1600 t/yr). Therefore, if we have a 3% clay enrichment, we would have a 43 MT (48 T) more clay discharge than without enrichment. Figure 3 shows that the concentration of all size fractions increased as runoff increased. The rate of increase is greatest with coarse fractions and least with the clay fraction; therefore, at low flows the percentage of clay in the total sample will be greater. The effect of clay enrichment on nutrient transport can be investigated best by considering the specific surface of the sediment and surface soil material. The specific surface (SS) can be described by equation 1 (Frere et al., 1975):

$$SS = 200\ (\%\ \text{clay}) + 40\ (\%\ \text{silt}) + 0.5\ (\%\ \text{sand}) \qquad (1)$$

For these calculations, the following specific surfaces of the separate fractions are assumed: clay, 200 m^2/g ($6.1X10^4$ ft^2/oz); silt, 40 m^2/g ($1.2X10^4$ ft^2/oz); and sand, 0.5 m^2/g ($1.5X10^2$ ft^2/oz). Therefore, the specific surface for the sediments and surface soil from Watershed 2 (Table 5) are 8482 m^2/g ($2.6X10^6$ ft^2/oz) and 7962 m^2/g ($2.4X10^6$ ft^2/oz),

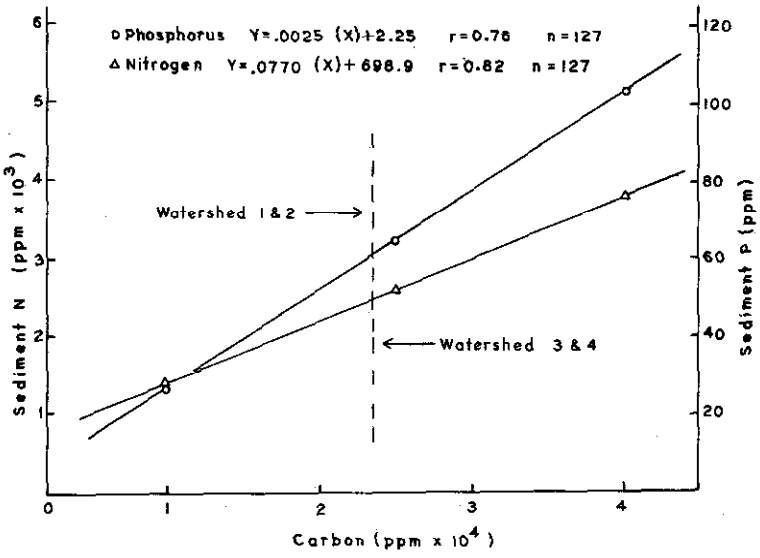

Figure 1. Relationship between N, P, and C content of sediment originating from agricultural watersheds on Missouri Valley loess.

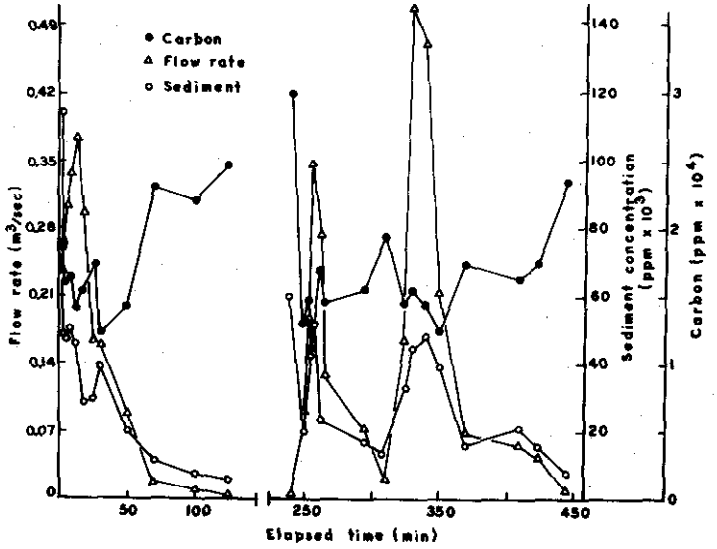

Figure 2. The effect of streamflow rate and sediment concentration on the C content of the sediment, Watershed 1, May 6, 1971, event, Treynor, Iowa.

Table 5. Size fractions of surface soil and streamflow samples from Watershed 2, Treynor, Iowa

Description of samples	Number of samples	% finer than indicated size mm (inches)		
		0.002 (0.0001)	0.020 (0.0008)	0.05 (0.002)
Surface soil	26	26	52	95
Weighted streamflow sediment 1/	73	29	57	96

1/ Based on flow duration-sediment transport relationship.

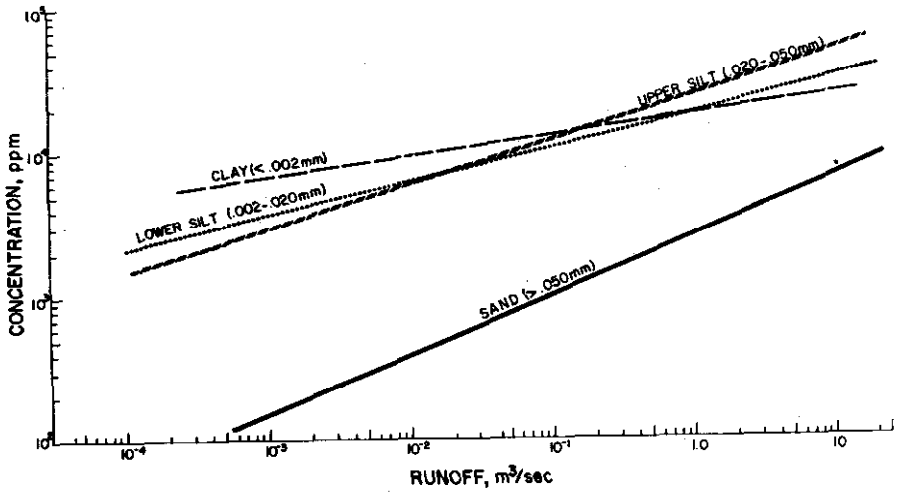

Figure 3. The effect of runoff rate on sediment concentration by particle size fraction.

respectively. These data indicate that the 3%-greater clay content of the sediment gives a 6.5%-greater specific surface and a similar capacity for adsorbed nutrient discharge. A small increase in clay content in eroded soil significantly increases the potential for nutrient transport during periods of runoff from agricultural lands. It is probable clay enrichment from the grass watershed or the terraced watershed would be greater than from the contour-corn watershed because (1) flows are generally lower so that the larger particles are not readily moved and (2) the greater roughness on the watershed due to vegetation results in deposition of the larger soil particles or filtering and thus favors clay movement.

The data presented point out the selectivity of the erosion process and how this can influence the chemical characteristics of the sediment discharged. The sediment fraction of surface runoff is responsible for the major nutrient discharges from cultivated cropland unless conservation and management practices are utilized to reduce erosion. The use of adequate management is becoming more important, both environmentally and to maintain maximum production.

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SEDIMENT-PHOSPHORUS RELATIONS IN SURFACE RUNOFF FROM IRRIGATED LANDS

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ABSTRACT

Phosphorus and sediment concentrations were measured in irrigation and drainage waters, and phosphorus and sediment inflows and outflows computed. Relationships between phosphorus and sediment were developed. Total phosphorus and orthophosphate concentrations measured in nonfiltered samples are closely related to the sediment concentration, but dissolved orthophosphate measured in samples filtered through 0.45 μm membrane filters is independent of the sediment concentration. A net sediment inflow was found on one large tract where sediment settles in drains and the amount of surface runoff is low, but a net sediment outflow was found for another tract with steeper drains and from which more surface runoff returned to the river. Net phosphorus inflows were measured on both tracts. Particle size segregation takes place in irrigation and drainage waters whenever the flow velocity is slow enough to allow suspended sediment to settle, and the quantity of phosphorus per unit of sediment remaining suspended increases. Actually, much more phosphorus settles with that portion of the sediment that settles than remains in suspension where sediments are eroded from silt loam or loam soils. Thus, conditions favoring settling are phosphorus conserving conditions.

INTRODUCTION

Phosphorus is tightly held by most soils and sediments, and the amount solubilizing, precipitating or combining with sediments depends upon equilibria conditions between the phorphorus in the ambient solution and that attached to or held by sediments. Sediments may scavenge phosphorus from solution (Taylor and Kunishi, 1971), and many sediments have a great capacity to do so (Latterell, Holt, and Timmons, 1971). Sediments also represent a vast phosphorus reservoir that can maintain a low and nearly constant water-soluble phosphorus concentration in a lake or stream for a very long time.

Sediments carry large quantities of phorphorus into waterways each year as a result of erosion (Viets, 1971; Wadleigh, 1968). Phosphorus is a limited, valuable, natural resource required by all living organisms, and reclaiming it from ocean and lake sediments would be extremely expensive. Therefore, the phosphorus entering waterways ultimately reaching the ocean or other large water bodies represents a phosphorus loss to man, animals, and plants, except those living in aquatic environments.

Irrigated agriculture involves very complex and dynamic solution-to-solid phase equilibria for phosphorus. Irrigation waters from rivers and streams

carry sediment loads and associated phosphorus that vary through the season. These waters have a dissolved phosphorus concentration that may also vary during the season. When this water is applied to close growing crops, like alfalfa and grass, the sediment and the associated phosphorus is usually deposited upon the land. Dissolved phosphorus is effectively removed from the water that percolates through the soil (Carter, Bondurant, and Robbins, 1971; Carter, Robbins, and Bondurant, 1973). In contrast when the water is applied to recently cultivated row crops, the soil may erode adding to the sediment and associated phosphorus loads in the runoff water. Surface and subsurface drainage waters from fields of various crops may enter surface drains and mix. In some cases flow velocities in drains are low or the water is passed through a sediment pond and sediments settle. In other cases, the flow velocity is high and there is no settling until the water reaches a river or a reservoir along a river, or in some cases, the ocean. All these dynamic processes can influence phosphorus equilibria.

During the past 6 years, we have been studying phosphorus inflows in the irrigation water and outflows in the drainage water from large irrigated tracts, and phosphorus-sediment relationships in irrigated agriculture in an effort to determine the extent of phosphorus losses from irrigated lands and to develop information useful for reducing such possible losses. This paper reports phosphorus and sediment concentrations in irrigation and drainage waters for two large tracts, functional relationships between sediment and phosphorus concentrations in irrigation and surface runoff waters, phosphorus inflows and outflows for two large tracts, phosphorus:sediment rations, and related information.

MATERIALS AND METHODS

The studies involved the 82,030 ha (202,700 a) Twin Falls Canal Company tract and the 65,350 ha (161,480 a) Northside Canal Company tract and the sampling sites previously described (Brown, Carter, and Bondurant, 1974). Three samples were collected at 2-week intervals at each site. They were: (1) a 200 ml sample filtered through a 0.45 µm membrane filter immediately upon sampling, (2) a 200 ml nonfiltered sample, and (3) a 10 liter nonfiltered sample. Biological activity was stopped by adding 40 mg $HgCl_2$/liter immediately as samples were collected. The 200 ml samples were refrigerated at 4 C until analyzed. The 10 liter samples were allowed to settle for 1 week in the laboratory, and then the supernatant was siphoned off and analyzed for phosphorus concentration. The sediment and a small amount of solution were transferred to weighed containers, dried at 105 C and weighed to determine sediment concentrations. The settling time exceeded that needed for 1.0 µm particles to settle the depth of the sample according to Stokes law. All samples were collected at a drop structure, culvert or turbulent zone to assure that the samples properly represented the stream and its sediment load. At some sites a fractional water-sediment sampler was used to assure representative samples (Heinemann and Brown, 1972).

All samples were analyzed for dissolved and total orthophosphate, dissolved and total hydrolyzable phosphorus and total dissolved and total phosphorus following chemical treatments previously described (Carter, et al., 1974). Dissolved organic and total organic phosphorus concentrations were obtained by subtracting orthophsophate plus hydrolyzable phosphorus from total phosphorus for the dissolved and total categories. The chemical treatments used were essentially those recommended by the Environmental Protection Agency (Environmental Progection Agency, 1974). Phosphorus concentrations were measured in the solutions following chemical treatment by the ascorbic acid method (Watanabe and Olsen, 1965). The pH was adjusted before adding the ascorbic acid-ammonium molybdate solution containing potassium antimony tartrate to assure reproducible results (Carter, et al., 1974).

Regression and correlation analysis were employed to evaluate sediment-phosphorus relations in irrigation and surface runoff waters. Such analyses were completed for dissolved orthophosphate and total phosphorus only because as studies progressed, hydrolyzable and organic phosphorus concentrations in irrigation and drainage waters seemed insignificant.

Sediments collected from the 10 liter samples were analyzed for total phosphorus by suspending 0.1 g subsamples in 40 ml of distilled water and following the persulfate digestion procedure for water samples (Carter, et al., 1974; Environmental Protection Agency, 1974). $NaCHO_3$ extractable phosphorus was determined by extracting 5 g samples with 100-ml 0.5 M $NaHCO_3$ solution and determining the phosphorus concentration by the ascorbic acid method (Watanabe and Olsen, 1965). Where only small amounts of sediments were obtained, 0.5 g of sediment was extracted with 10-ml of the solution.

Sediment, dissolved orthophosphate and total phosphorus inflows in the irrigation water and outflows in the surface drainage water were calculated for a typical irrigation season for the two tracts. This was done by multiplying the phosphorus and sediment concentration for each 2-week sampling period by the quantity of flow for that period and accumulating the results for the season.

The relationship of phosphorus concentration to particle size was investigated by collecting samples from the surface 15-cm of field soils from certain subbasins in the two tracts, mixing soil samples with water without the aid of dispersing reagents, and allowing samples to settle for the time periods that gave an aggregate or particle size segregation into sand, silt, and clay sizes. This segregation was assumed to simulate processes taking place as eroded soil enters drainage streams and subsequently settles as the flow velocity decreases. Complete dispersion was not expected under these conditions, and some aggregates containing clay settled in the sand and silt fractions. These size fractions were analyzed for total- and $NaHCO_3$-extractable phosphorus. Surface soil samples were collected from both tracts on a grid and analyzed for total and $NaHCO_3$ extractable phosphorus so that results could be compared with those from

Table 1. Numbers of samples with sediment and phosphorus concentrations in specific ranges.

Variables	Numbers of Samples									
	Concentration Range (ppm)									
	0-50	51-100	101-200	201-500	501-1001	1001-2000	Above 2000			
Sediment	33	21	32	13	7	2	3			
	Concentration Range (ppb)									
	0-25	26-50	51-75	76-100	101-150	151-200	201-250	251-300	301-400	Above 400
Phosphorus*										
Ortho-P (F)	21	23	23	23	20	1				
Ortho-P(NF)		10	14	11	24	20	7	10	5	11
Total-P(NF)			1	4	19	20	22	12	17	16

*Ortho-P(F) is the orthophosphate measured in a sample filtered through a 0.45 µm membrane filter. Ortho-P(NF) is the orthophosphate measured in a nonfiltered sample. Total-P is the total phosphorus measured in a nonfiltered sample.

sediments collected from surface drainage streams. Additional evidence that particle size segregation takes place in drainage streams was obtained by calculating phosphorus:sediment ratios (the quantity of phosphorus per unit of sediment) for some drainage streams with flow velocities slow enough to allow settling and some with high flow velocities so that there was little or no settling. Dissolved orthophosphate concentrations were subtracted from the total phosphorus concentrations so that the phosphorus involved was that attached to the sediment.

RESULTS AND DISCUSSION

The sediment concentration in surface runoff water returning to the river from the two large irrigation tracts varied widely during the irrigation season (Table 1). The lowest concentration measured was 10 ppm in the W drain on the Northside tract in which the flow velocity was very slow allowing the drain to serve as a sediment retention basin. The highest concentration measured was 2610 ppm in the J-8 drain on the Northside tract, but concentrations as high as 2250 ppm were measured in the Filer drain on the Twin Falls tract. Generally,the sediment concentrations were highest during the period when row crop cultivations were most frequent. After July, sediment concentrations were lower in all drainage water entering the river, except from those drains where the flow velocity was slow enough to allow sediments to settle. Little changes were observed in these drains.

The total phosphorus concentration paralleled the sediment concentration (Table 1). The relationship between the two parameters is given by the first equation in Table 2. A similar relationship was found between the

Table 2. Regression equation and correlation coefficients showing relationship between sediment and phosphorus concentrations in irrigation and drainage waters.

X	Y	Equation	r^2	r
Sediment in nonfiltered samples (ppm)	Total phosphorus in nonfiltered samples (ppb P)	Y = 140.52 + 0.72X	0.89	0.94
Sediment in nonfiltered samples (ppm)	Orthophosphate in nonfiltered samples (ppb P)	Y = 79.82 + 0.579X	0.90	0.95
Sediment in nonfiltered samples (ppm)	Orthophosphate in filtered samples (0.45 um) (ppb P)	Y = 57.71 + 0.033X	0.10	0.32

sediment and the orthophosphate concentrations determined in nonfiltered samples. These results showed that the concentration of both orthophosphate and total phosphorus in nonfiltered water samples depends upon the sediment concentration. These equations could be used to predict the total phosphorus and orthophosphate in nonfiltered samples. Better predictive equations could likely be developed for specific surface drains or subbasins where conditions are more uniform. The third equation shows that there is no such relationship between the orthophosphate in filtered samples and the sediment concentration. Thus, the dissolved orthophosphate concentration is independent of the sediment concentration. Dissolved hydrolyzable and organic phorphorus concentrations are very low in the 111 water samples included for comparison. Only two samples had dissolved hydrolyzable concentrations above 26 ppm and only four samples had organic phosphorus concentrations above that value. Total hydrolyzable and organic phosphorus concentrations were somewhat higher and related to the sediment concentrations, but still of little significance in irrigation and drainage waters.

There was a net sediment outflow or loss of 37,790 mt (41,368 t) from the Twin Falls tract or 0.46 mt/ha (0.20 t/a), in contrast to a net sediment inflow or accumulation of 45,170 mt (49,807 t) into the Northside tract or 0.69 mt/ha (0.30 t/a) (Table 3). These two tracts had contrasting results because of different amounts of surface drainage returning to the river and different slopes and resulting flow velocities in surface drains on the two tracts (Brown, Carter, and Bondurant, 1974). The surface drains on the Northside tract were constructed to a low slope, some of which serve effectively as sediment retention basins. One drain passes through a basin. The drains on the Twin Falls tract are mostly natural, steep channels in which the flow velocity is too high to allow much sediment settling.

Net total phosphorus and dissolved orthophosphate inflows were found for both tracts even though there was a net sediment outflow on the Twin Falls tract (Table 3). These results include only the phosphorus in the irrigation and surface drainage water, and do not include the phosphorus fertilizer applied to the land. The results indicated that the sediment leaving the Twin Falls tract carried less phosphorus per unit of sediment than the sediment entering the tract in the irrigation water. The sediment in the irrigation water was likely composed of smaller-sized particles than was the sediment leaving the tract in the surface drainage water.

The $NaHCO_3$ extractable and total phosphorus concentrations were greater in sediments in drainage waters from the Northside tract than that from the Twin Falls Tract (Table 4). These results show that more particle size segregation takes place in Northside tract drains than in Twin Falls tract drains. The sediment remaining suspended in Northside tract drains contained relatively more clay than that in Twin Falls tract drains. Both total and $NaHCO_3$ extractable phosphorus concentrations in sediments from both tracts were higher than concentrations found in surface soils, but the difference was greater for the Northside tract (compare tables 4 and 5).

Table 3. Water, Sediment, dissolved ortho-P and total-P inputs and outputs for two large tracts for an irrigation season.

	Metric				English			
	Water 10^4 m^3	Sediment	Ortho-P	Total-P	Water a ft	Sediment	Ortho-P	Total-P
		mt				t		
Inflow	Twin Falls Tract (82,030 ha)				(202,700 a)			
Diverted water	138,310	75,820	28.57	189.00	1,121,000	83,576	31.49	208.36
Outflow*								
Rock Creek	13,930	43,090	12.11	34.84	112,932	47,502	13.85	42.82
Cedar Draw	4,144	8,750	2.82	11.16	33,595	9,647	3.10	12.48
Filer Drain	1,084	11,280	1.13	10.16	8,971	12,434	6.22	18.16
Mud Creek	7,239	13,820	5.64	16.47	58,684	15,236	1.92	7.87
Deep Creek	4,732	5,630	1.74	7.14	38,363	6,208	1.25	11.25
13 small drains	6,295	31,040	2.31	15.06	51,036	34,217	2.55	16.60
Net Inflow	100,866	---	2.82	94.17	817,599	---	2.60	99.18
Net Outflow	---	37,790	--	---	---	41,368	--	---
Inflow	Northside Tract (65,350 ha)				(161,480 a)			
Diverted water	150,631	57,250	34.38	159.39	1,221,182	63,111	37.90	175.70
Outflow								
K	1,410	1,300	1.28	3.75	11,432	1,431	1.42	4.13
N-32	2,478	2,160	1.84	5.71	20,087	2,375	2.02	6.29
J-8	622	3,550	0.25	1.42	5,039	3,911	0.28	1.56
S	1,780	3,440	0.60	2.97	14,432	3,798	0.67	3.27
W-26	1,787	1,280	0.25	1.19	14,488	1,405	0.49	4.72
W	960	350	0.45	4.28	7,779	384	0.28	1.31
Net Inflow	141,594	45,170	29.97	140.08	1,147,925	49,807	32.74	154.42
Net Outflow	---	---	---	---	---	---	---	---

*Includes flow from subsurface drainage tunnels that enter the listed streams as well as water pumped from wells for industrial purposes. Surface runoff totalled 14% of the total water inflow to the tract (Carter, Bondurant and Robbins, 1971)

Table 4. $NaHCO_3$-extractable and total phosphorus concentrations in sediments collected from irrigation and surface drainage waters for two large irrigated tracts during the 1971 irrigation season (Carter, et al., 1974).

	$NaHCO_3$ Extractable P (ppm)					Total P (ppm)				
Source	May	Jun	Jul	Aug	Sep	May	Jun	Jul	Aug	Sep
					Northside tract					
Diverted water	138	80	46	--	--	1,100	1,243	1,040	1,100	1,278
K	195	148	98	95	61	1,206	1,040	1,036	1,185	1,018
N-32	345	277	104	--	72	1,395	1,392	1,076	1,338	941
J-8	47	75	59	84	68	649	808	1,018	1,110	948
S	166	157	62	53	60	1,075	1,086	966	999	908
W-26	171	191	55	44	64	1,345	1,153	985	1,018	940
W	278	404	--	--	318	1,370	1,760	1,000	1,228	1,006
					Twin Falls tract					
Diverted water	133	--	66	37	63	1,160	1,600	1,040	1,025	1,047
Rock Creek	36	122	38	28	33	955	936	831	672	939
Cedar Draw	34	74	44	36	30	895	1,038	902	915	819
Filer Drain	21	52	33	29	31	895	1,006	842	940	952
Mud Creek	30	48	21	26	22	1,040	1,110	1,040	1,138	956
Deep Creek	66	96	20	28	38	912	1,070	602	840	870
Hansen Drain	39	95	32	28	41	882	1,040	870	970	988
Kimberly Drain	54	59	40	38	80	963	1,013	897	912	965

Table 5. $NaHCO_3$-extractable and total phosphorus concentration in soil samples collected from the surface 15 cm (6 in.) of fields in two large irrigation tracts, ppm (Carter, et al., 1974).

	No. of fields sampled	$NaHCO_3$ Extractable P				Total P			
		Max.	Min.	Med.	Mean	Max.	Min.	Med.	Mean
Northside tract	44	64	4	22	24	885	488	763	722
Twin Falls tract	48	55	6	20	22	1007	580	842	839

The relative total and $NaHCO_3$ extractable phosphorus concentration in the sand, silt, and clay size fractions of soils separated without the aid of dispersion agents are presented in Table 6. A comparison of the values in Tables 4 and 6 shows that particle size segregation is more complete in drains on the Northside tract than in drains on the Twin Falls tract as suggested earlier, because both total and $NaHCO_3$ extractable phosphorus concentrations in sediments from Northside tract drains are closer to the values for the clay size fraction separated from soils. Also, the data in Table 4 indicate that sediments in the diverted water are comprised mostly of clay which is as expected because the river water is stored in upstream reservoirs where settling can occur, and is then released for irrigation downstream as the demand requires.

Table 6. Total and $NaHCO_3$-extractable phosphorus for various particle and aggregate size fractions of surface soils collected from four subbasins, ppm (Carter, et al., 1974).

Drainage subbasins	Total P			$NaHCO_3$-extractable P		
	Sand	Silt	Clay	Sand	Silt	Clay
Kimberly	875	1000	1400	33	55	137
Filer	650	975	1285	13	27	67
K	550	1150	1285	21	74	125
J-8	450	1020	1325	8	45	70

The data in Table 7 show that the phosphorus:sediment ratio (the quantity of phosphorus per unit of sediment) indicates the extent of the particle size segregation. The higher the value of the ratio the more extensive was the particle size segregation. For example, ratio values for the W drain which serves as a sediment basin were 1.60 and higher. Values for the K drain which passes through a sediment retention basin also generally exceeded 1.0. When the flow velocity was too high to allow settling, as in the Filer drain, the values were generally less than unity. Values for Rock Creek and Deep Creek varied because their flows vary widely and

Table 7. The quantity of phosphorus per unit of sediment in irrigation and drainage waters.

	Sampling date, 1971							
	6/15	6/29	7/13	7/26	8/10	8/28	9/8	9/28
	Phosphorus per unit of sediment X 1000							
Northside Canal	2.24	1.60	1.42	1.60	1.97	3.14	2.97	3.24
Twin Falls Canal	--	0.75	1.96	2.39	2.21	2.77	3.13	2.93
W	1.78	3.40	1.60	2.67	3.45	3.45	--	2.65
K	1.12	0.94	1.62	2.31	1.82	2.25	2.40	4.08
Filer	0.83	0.57	0.96	0.58	0.81	0.87	0.86	1.29
Rock Creek	0.81	1.04	0.87	1.09	0.60	--	1.45	1.17
Deep Creek	1.24	0.95	--	1.00	1.34	0.81	1.02	0.91

and the amount of settling is related to the flow velocity which is a function of the flow volume. The mean value for the total phosphorus for the Northside and Twin Falls tract surface soils, calculated in the same manner as for sediments were 0.72 and 0.84, respectively. Dividing values for each tract into those presented in Table 7 for drains of the respective tracts is another method of indicating the extent of particle size segregation. The numbers presented in Table 8 are particle size segregation indices. When these values exceed 1.0, particle size segregation has taken place, and the greater the number, the more extensive was the particle size segregation.

Table 8. Indices of particle size segregation in surface drainage streams.

	Sampling date, 1971							
Drain	6/15	6/29	7/13	7/26	8/10	8/24	9/8	9/28
W	2.5	4.7	2.2	3.7	4.8	4.8	--	3.7
K	1.6	1.3	2.2	3.2	2.5	3.1	3.3	5.7
Filer	1.0	0.7	1.1	0.7	1.0	1.0	1.0	1.5
Rock Creek	1.0	1.2	1.0	1.3	0.7	--	1.7	1.4
Deep Creek	1.5	1.1	--	1.2	1.6	1.0	1.2	1.1

The data presented in Tables 7 and 8 were obtained over a broad range of soil and cropping conditions, and varying stream sizes. Greater precision could likely be attained by restricting measurements to given stream sizes, subbasins, soils, cropping conditions, land slopes, time of season, etc. Also, in our studies, there was considerable variation in the total phosphorus measured in the surface soil so the numbers should be used only as indicators not as precise measures. Also sometimes canal water is spilled into surface drains, and the index values could

exceed unity because values of the phosphorus:sediment ratios in the irrigation water were generally greater than found in the surface soils.

Most of the sediment and associated phosphorus entering irrigated tracts is filtered from the water by soil as the water enters the soil or by close growing crops. Only a small fraction of that sediment entering an irrigated tract leaves that tract in drainage water, because only a small portion, 6% and 14% for the two tracts studied, applied water bebecomes surface runoff. Essentially all the sediment leaving an irrigated tract is from erosion of that tract, supplying different sediments in the drainage water than entered in the irrigation water.

CONCLUSIONS

Most of the phosphorus in irrigation water and surface drainage waters from irrigated lands is associated with or attached to the sediment. The total phosphorus concentration is directly related to the sediment concentration as is the orthophosphate measured on nonfiltered samples. Therefore, controlling the quantity of sediment returning to a stream from irrigated land controls to a large extent the quantity of phosphorus returning. The dissolved orthophosphate concentration is independent of the sediment concentration.

Sediment and phosphorus losses from irrigated tracts can be controlled by limiting the surface drainage water leaving the tract and by utilizing sediment retention basins or drains of low slope so that the flow velocity is slow enough for sediment to settle. Whenever the flow velocity is slow enough to allow settling, particle size segregation takes place. The clay size particles settle slowest and are most likely to be carried by the drainage stream to the river. These smaller particles contain higher phosphorus concentrations per unit weight so that as settling takes place the phosphorus per unit of sediment remaining in suspension increases. In a sense, this is phosphorus enrichment of the suspended sediment. However, in sediments eroded from most silt loam or loam soils the amount of phosphorus associated with the sediment that settles greatly exceeds the amount associated with the smaller size particles remaining in suspension so that conditions favoring settling are conservation conditions for both sediment and phosphorus. Ideally, preventing surface runoff from entering a river would prevent all of the sediment loss and the only phosphorus loss would be the very small quantity in subsurface drainage water.

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PESTICIDE CONCENTRATIONS AND YIELDS IN RUNOFF AND SEDIMENT FROM A MISSISSIPPI DELTA WATERSHED[1]

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ABSTRACT

Pesticide concentrations were measured in runoff from a 15.6-ha (38.5-acre) watershed planted to continuous cotton. The watershed, consisting of Sharkey silty clay soil, had been formed for drainage with mean slopes of 0.2 percent. Measurements during 1973 and 1974 indicated a linear relationship between pesticide yields in runoff and sediment yields. Mean sediment yield was 27.6 metric tons/ha/year (12.3 tons/acre) when mean annual rainfall was 43.8 cm (17.2 inches) above the 30-year average of 125 cm (49.4 inches). Toxaphene, DDT, DDE, and trifluralin yields from the watershed in 1973, when no toxaphene or DDT was applied, were 117 (0.104), 87 (0.078), 7 (0.006), and 2 (0.002) g/ha (lbs/acre), respectively. Corresponding yields in 1974, when 10.08 kg toxaphene/ha (9 lbs/acre) were applied, were 97 (0.086), 27 (0.024), 6 (0.005), and 2 (0.002) g/ha (lbs/acre), respectively. In 1974 the mean annual concentrations of toxaphene, DDT, DDE, and trifluralin in runoff were approximately 11, 3.1, 0.6, and 0.2 µg/liter (ppb). These sediment and pesticide yields are probably higher than would be expected in a year of normal rainfall amount and distribution. The data indicate the need, however, for improved erosion control practices on flat lands in the Delta to reduce the loss of valuable topsoil and to reduce pollution hazards from both sediment and farm chemicals.

INTRODUCTION

Agriculture in the lower Mississippi River Valley, one of the most intensively farmed areas in the United States, depends heavily on pesticide and fertilizer use. Various chemicals are applied frequently to cotton and soybeans during each cropping season. Evidence that persistent organochlorine pesticides may accumulate in food chains of nontarget species has greatly restricted their use. However, the potential pollution problems associated with these persistent compounds may continue for some time because of their previous extensive use.

In the Mississippi Delta, pesticides transported by runoff and sediment from flat to gently sloping agricultural lands have been implicated as

[1]Contribution from the USDA-ARS Soil and Water Pollution Research Unit, Baton Rouge, LA 70803, cooperating with the Louisiana Agricultural Experiment Station; and the USDA-ARS Sedimentation Laboratory, Oxford, MS 38655.

a pollution source. During 1972 and 1973, the Mississippi Game and Fish Commission closed several lakes to commercial fishing because of DDT and toxaphene residues in fish and bottom sediments. A combination of toxaphene, DDT, and methyl parathion was used extensively to control cotton boll weevils and bollworms until DDT was banned in 1972. Subsequently, toxaphene, alone and in combination with methyl parathion, has been used to control these and other cotton insects. Current cultural practices in the Mississippi Delta may be intensifying sediment and chemical transport from agricultural fields. After harvest, many farmers shred plant residues, till the soil, and form rows. The fields are left with little or no vegetative cover throughout the winter and early spring, and are subject to the erosive forces of rainfall and runoff until adequate cover develops.

It is recognized that pesticides are transported in agricultural runoff (Pionke, 1973; Willis, 1973, 1975). While it is often assumed that surface water contamination by the more water-insoluble pesticides results from eroded soil particles with adsorbed residues, the magnitude and significance of the problem are not well characterized. Moreover, our knowledge of the role of sediment in pesticide adsorption, transport, desorption, and degradation is limited. Knowledge concerning pesticide losses from flat lands is particularly lacking.

A cooperative study between the ARS Soil and Water Pollution Research Unit in Baton Rouge, Louisiana and the USDA Sedimentation Laboratory in Oxford, Mississippi was initiated in 1972 to investigate the relationship between surface runoff, sediment yield, and chemical yield from flat agricultural land in the Mississippi Delta.

MATERIALS AND METHODS

A 15.6-ha (38.5-acre) watershed on the G. L. McWilliams farm near Clarksdale, Mississippi was selected for the study. The watershed had been in continuous cotton for a number of years and had received a wide assortment of pesticides, including toxaphene and DDT. The soil, a Sharkey silty clay (1% sand, 52% silt, and 47% clay; 2.5% organic matter), had been formed for drainage with mean slopes of 0.2%. The drainage pattern was designed to direct runoff via turn-rows (shallow V-ditches) into a 1.6-ha (4-acre) pond located in the watershed. Instrumentation to measure surface runoff and sample sediment and chemical yields (Murphree, 1976; Parr, 1974) was installed where runoff entered the pond. Collection of water discharge and sediment yield data began in July 1972, and pesticide yield data, in March 1973.

The watershed was planted to cotton both years of the study. Cultural practices typical of those normally used in the Mississippi Delta were followed, i.e., shredding cotton stalks after harvest, disking and forming rows in the winter or early spring, applying preplant herbicides and fertilizers in the spring, followed by planting and normal cultivation

and pesticide application during the growing season. Pesticide application dates and rates for 1972, 1973, and 1974 are shown in Table 1.

Runoff samples (1000 ml) were collected at 10-minute intervals throughout each storm. A discharge-weighted-mean composite sample, representing each storm, was prepared from aliquots of the instantaneous samples. The volume of each aliquot was proportioned to the volume of water passing through the flume during each 10-minute period. Use of the composite sample for determining the sediment yield and chemical yield resulting from a storm eliminated the need for time-consuming and expensive analyses of the numerous instantaneous samples collected during each storm.

Pesticides were extracted by sonifying 100 ml of runoff suspension (water and sediment were not separated) and an equal volume of a 1:1 hexane-acetone mixture for 1.5 minutes. The hexane was separated from the acetone and water and dried with Na_2SO_4. The extract volume was then adjusted for gas chromatographic analysis.

The watershed soil was sampled each spring in 1973 and 1974 to determine residual concentrations of persistent pesticides. Pesticides were extracted by sonifying 40-g soil samples with three separate 80-ml volumes of a 1:1 hexane-acetone mixture. Gas chromatography was used to identify and quantify the pesticides.

RESULTS AND DISCUSSION

For brevity, the two sampling periods, March 1, 1973 through February 28, 1974, and March 1, 1974 through February 28, 1975, will be referred to as 1973 and 1974, respectively.

Total rainfall was 162.7 cm (64.07 in) in 1973 and 174.9 cm (68.86 in) in 1974. These values were 37.39 and 49.55 cm (14.72 and 19.51 in) above the 30-year average (1941-1970) recorded by the National Weather Service at Clarksdale, Mississippi, approximately 17.7 km (11 miles) to the southeast. Thus, the data reported in this paper were obtained during abnormally high rainfall years. Watershed runoff was 74.22 cm (29.22 in) in 1973 and 88.01 cm (34.65 in) in 1974. Corresponding sediment yields were 28.94 and 26.25 metric tons/ha (12.91 and 11.72 tons/acre), respectively.

The concentrations of toxaphene, DDT, DDE (a DDT degradation product), and trifluralin in the 0- to 15-cm (0- to 6-in) zone of the watershed soil in the spring of 1973 after seedbed preparation but before application of pesticides for the 1973 crop were 3.5, 1.8, 0.3, and 0.03 ppm, respectively. The coefficient of variability (c.v.) for these values was 7%. In the spring of 1974, soil concentrations of the same pesticides were 2.1, 0.6, 0.2, and 0.04 ppm, respectively (c.v. = 15%). Since neither toxaphene nor DDT was applied to the watershed in 1973, the decrease in residual concentrations should be representative of

Table 1. Pesticides Applied to McWilliams Watershed

Chemical	Class	Application	
		Date	Rate
	1972		kg/ha
Trifluralin	Herbicide	March	1.1
Fluometuron	Herbicide	April	1.7
Fluometuron + MSMA	Herbicide	May	1.1
Toxaphene*	Insecticide	May	2.24
Linuron	Herbicide	July	1.7
Methyl Parathion*	Insecticide	July-August	1.1
Toxaphene + DDT + Methyl Parathion†	Insecticide	July-Sept.	2.24 + 1.1 + 0.6
Tributyl Phosphorotrithioite	Defoliant	October	1.1
	1973		
Trifluralin	Herbicide	May	1.1
Fluometuron	Herbicide	May	1.7
Dicrotophos*	Insecticide	June	0.2
Prometryn*	Herbicide	June-July	0.3
Herbicidal Oil*	Herbicide	June-July	1.9 liter/ha
MSMA*	Herbicide	June-July	0.4
Dinitro	Herbicide	July	0.7
Chlordimeform*	Insecticide	July-August	0.1
Methyl Parathion	Insecticide	August	1.1
	1974		
Trifluralin	Herbicide	April	1.1
Fluometuron	Herbicide	April	1.7
Prometryn	Herbicide	July	0.3
Herbicidal Oil	Herbicide	July	1.9 liter/ha
MSMA	Herbicide	July	0.4
Methyl Parathion	Insecticide	August	1.12
Toxaphene + Methyl Parathion‡	Insecticide	August-Sept.	1.7 + 0.8
Tributyl Phosphorotrithioite	Defoliant	October	1.1

*2 applications at listed rate
†7 applications at listed rate
‡6 applications at listed rate

their natural dissipation rates under the climatic conditions observed. Although 1 year's data are inadequate for estimating field dissipation rates, the evidence suggests the rate may have been faster than normally expected (Edwards, 1973). Further evidence for rapid dissipation rates is indicated by the relatively low toxaphene and DDT concentrations in the soil in the spring of 1973, although 20.2 kg toxaphene and 7.8 kg DDT per hectare (18 and 7 lbs/acre) were applied to the watershed in 1972 (Table 1).

The relationship between pesticide and sediment concentrations in runoff was determined for each of the 1-year sampling periods, using discharge-weighted-mean composite samples to represent individual runoff events. Figure 1 depicts the relationship between DDT concentration and sediment concentration in runoff for 1973. The line slope of 0.977 and the correlation coefficient of 0.972 suggest a strong dependence of DDT concentration on sediment concentration over a wide range of runoff conditions. For toxaphene (data not shown), the correlation coefficient was 0.967, but the line slope was 0.712, suggesting a toxaphene enrichment at lower sediment concentrations. Such an enrichment could result when erosion energy is low (rainstorms of low intensity or dense vegetative cover) and relative proportions of clay-sized soil and organic particles in runoff are greater than when the erosion energy is higher. The finer soil fractions contain higher concentrations of pesticides per unit weight than do coarser fractions. The apparent toxaphene enrichment also may result from its higher water solubility than that of DDT (300 vs. 1.2 ppb). In the absence of high sediment concentrations, toxaphene may be transported in true solution to a greater extent than is DDT. Preliminary and incomplete data indicate a higher percentage of toxaphene associated with sediment at high sediment concentrations than at low sediment concentrations.

In general, the association between pesticide concentration and sediment concentration in runoff was greater in 1973 than 1974 for each of the four pesticides.

Pesticide yields from the watershed were calculated using the volume of storm runoff and storm discharge-weighted means of pesticide and sediment concentrations. The relationship between DDT yields and sediment yields for 1973 and 1974 is shown in Figures 2 and 3. Both figures indicate a very strong dependence of DDT yield on sediment yield. Similar relationships were obtained for DDE and trifluralin for both years. The discharge-weighted mean DDT concentrations in the sediments for the 1973 and 1974 water years were 3.0 and 1.03 ppm, respectively. Watershed soils sampled in the spring of 1973 and 1974 had DDT concentrations of 1.8 and 0.6 ppm, respectively. Thus, the sediments produced during both water years were enriched in DDT by a factor of about 1.7 relative to the soils from which they were derived. This enrichment is attributed to the selective erosion and/or transport of organic matter and clay (<2 μm) with sorbed DDT residues. Organic matter and clay in the sediments were enriched relative to the watershed soils by factors of

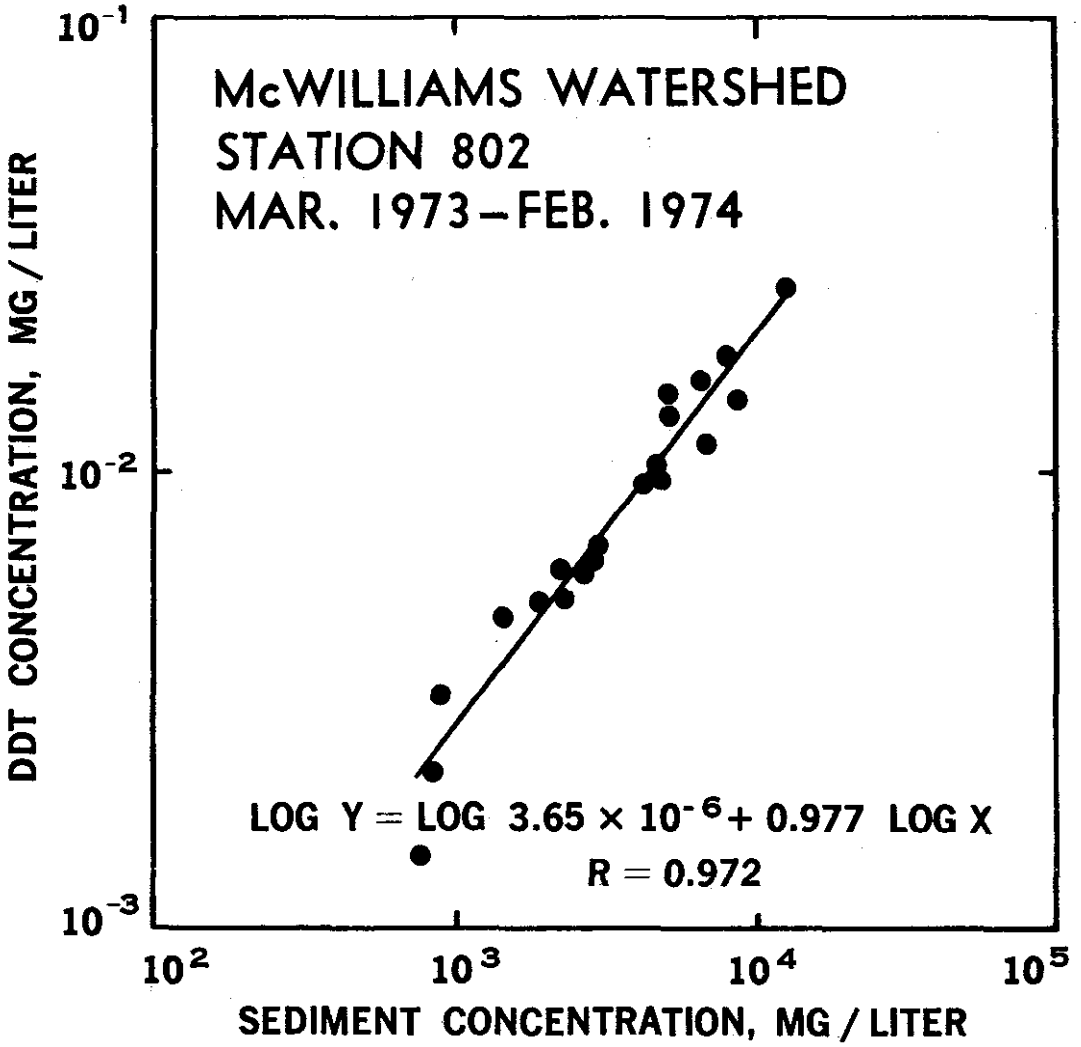

FIG. 1. RELATIONSHIP BETWEEN DDT CONCENTRATION AND SEDIMENT CONCENTRATION IN RUNOFF IN 1973.

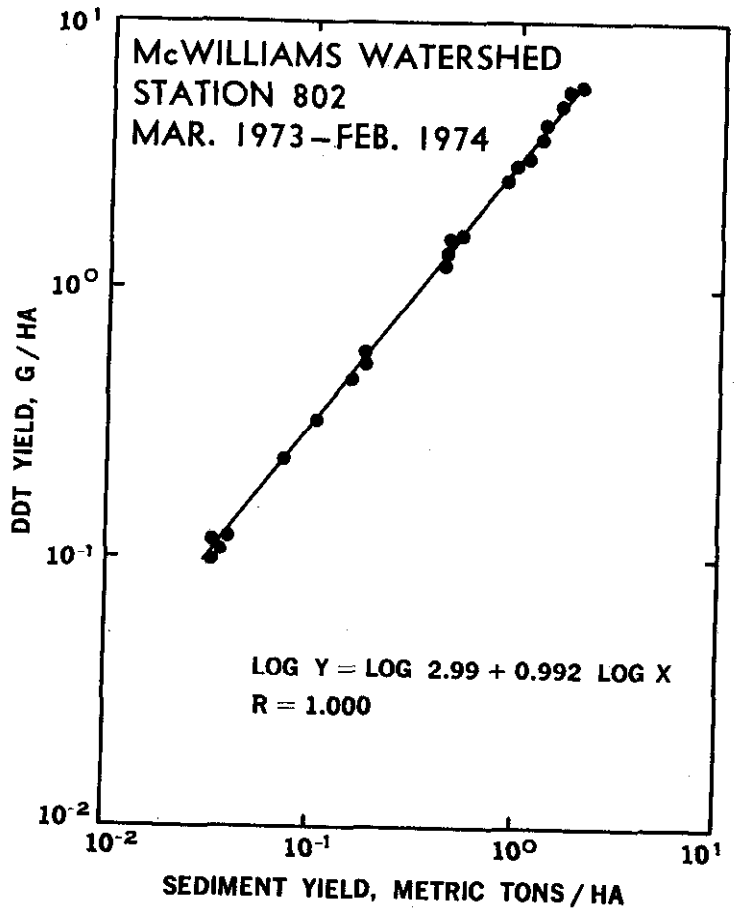

FIG. 2. RELATIONSHIP BETWEEN DDT YIELD AND SEDIMENT YIELD IN 1973.

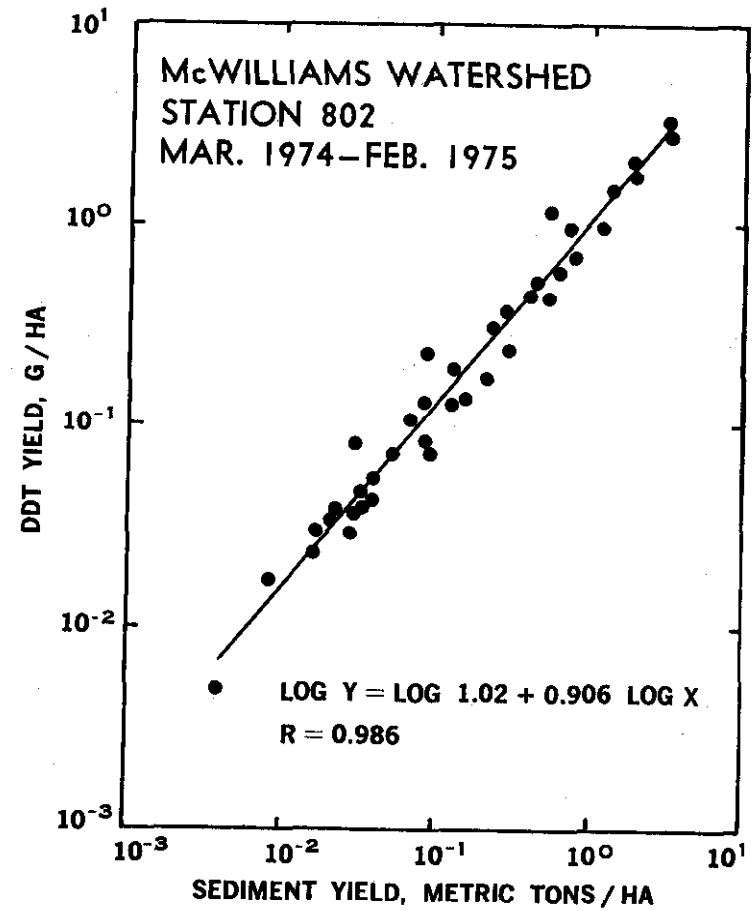

FIG. 3. RELATIONSHIP BETWEEN DDT YIELD AND SEDIMENT YIELD IN 1974.

2.5 and 1.5, respectively, during the water years of July 1972 - June 1974. The percent organic matter in the sediments was inversely related to the sediment concentration in runoff. The storm mean clay concentrations in runoff were related linearly to the storm mean sediment concentrations by the equation, $Y = 0.69X$, with a correlation coefficient of 0.975. The equation would be $Y = 0.47X$ with no clay enrichment. This clay enrichment illustrates the selective transport of the fine fraction of eroded soil particles generally accepted as responsible for the movement of the more water-insoluble sorbed chemicals (Letey, 1972).

Figure 4 depicts the relationship between toxaphene and sediment yields in 1973; although a close relationship is indicated, it is not as close as for DDT. Again this may be the result of toxaphene's higher water solubility.

The linear relationships between toxaphene yields and sediment yields in 1974 are given in Figure 5 for March-July, August-November and December 1974-February 1975. The slopes of the lines represent the mean toxaphene concentrations in the sediments in g/metric ton, or ppm, for the respective periods. The weighted mean concentration of toxaphene in sediments before application was 2.55 ppm (March-July). The concentration increased to about 16 ppm (August-November) after toxaphene application in August and September and subsequently decreased to 5.9 ppm for the remainder of the water year (December 1974-February 1975).

Normal cultural practices during the 5 months (March-July) preceding toxaphene application left the watershed soil very susceptible to erosion (Table 2), i.e., tilled (seedbed preparation, planting, and cultivation) and with less than complete vegetative cover (Murphree, 1976). Since no toxaphene was applied to the watershed during this period, eroded soil with sorbed residues was the source of toxaphene in runoff. The mean toxaphene concentration of 2.55 ppm in the sediments for this period is slightly higher than the value of 2.1 ppm found in the watershed soils sampled in the spring of 1974.

During this 5-month period (March-July), 93.58 cm (36.84 in) of rainfall [37.20 cm (14.65 in) above normal] produced sediment and toxaphene yields of 21.90 metric tons/ha (9.77 tons/acre) and 54.4 g/ha (0.049 lbs/ acre), respectively. Of the sediment and toxaphene yields observed during this tillage period, 81% occurred in May and June, months of abnormally high rainfall (Table 2). National Weather Service records indicate that at Clarksdale, 1974 was the only year since 1956 that rainfall during the month of May exceeded 20.32 cm (8.0 in).

The crop reached full canopy in late July and early August, coincident with the first toxaphene application to the cotton leaf surfaces [six applications between August 2 and September 16 (Table 1)]. Rainfall during the 4 months from August through November 1974 totaled 45.19 cm (17.79 in), 13.69 cm (5.39 in) above normal, and produced sediment and toxaphene yields of 1.58 metric tons/ha (0.70 tons/acre) and 27.82 g/ha (0.025 lbs/acre), respectively. The freshly sprayed plant leaves were

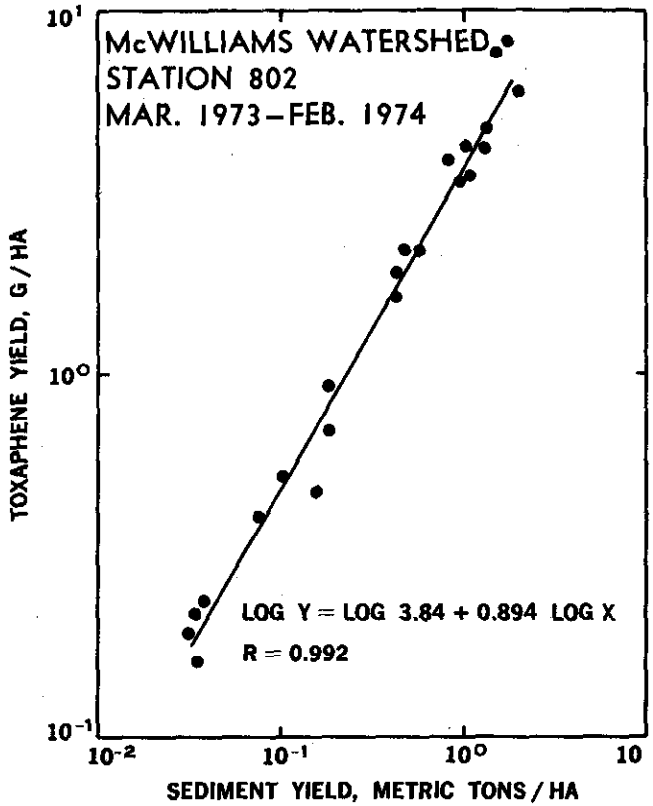

FIG. 4. RELATIONSHIP BETWEEN TOXAPHENE YIELD AND SEDIMENT YIELD IN 1973.

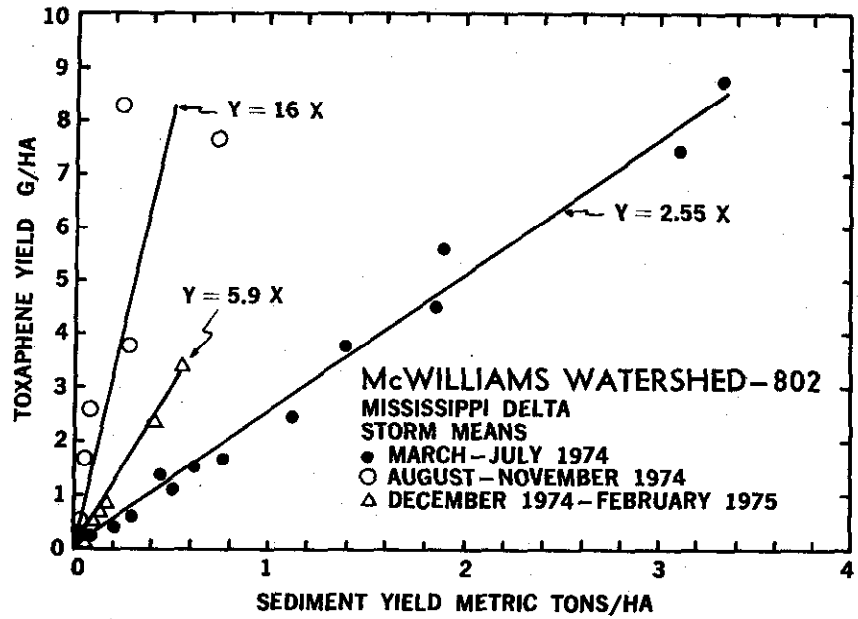

FIG. 5. RELATIONSHIP BETWEEN TOXAPHENE YIELD AND SEDIMENT YIELD IN 1974.

Table 2. Monthly Totals of Rainfall, Runoff, and Sediment and Toxaphene Yields for March 1974 - February 1975 Sampling Period

Month	Rainfall		Runoff	Toxaphene concentration	Yield	
	1974-75*	30-year average†			Sediment	Toxaphene
	cm			mg/l	t(m)/ha	g/ha
Tillage (Seedbed Preparation-Planting-Cultivation)						
March	4.88	13.79	0.23	0.006	0.04	0.14
April	12.95	13.31	6.07	0.010	2.29	6.07
May	37.85	11.96	26.64	0.010	10.09	26.64
June	22.99	7.72	15.88	0.011	7.71	17.47
July	14.91	9.60	3.43	0.012	1.77	4.12
Total(Mar-July)	93.58	56.38	52.25		21.90	54.44
No-Tillage (Full Crop Canopy-Defoliation in October)						
August	16.64	6.38	3.53	0.019	0.40	6.71
September	13.03	7.98	6.45	0.028	1.03	18.06
October	5.00	6.32	0.58	0.010	0.02	0.58
November	10.52	10.82	4.93	0.005	0.13	2.47
Total(Aug-Nov)	45.19	31.50	15.49		1.58	27.82
No-Tillage (Stalks Shredded-Crop Stubble)						
December	12.12	12.50	6.25	0.008	0.58	5.00
January	10.95	12.22	6.10	0.010	0.96	6.10
February	13.08	12.75	7.92	0.004	1.23	3.17
Total(Dec-Feb)	36.15	37.47	20.27		2.77	14.27
ANNUAL TOTAL	174.92	125.35	88.01		26.25	96.53

* Measured at Watershed 802

† National Weather Service records (1941-1970), Clarksdale, Mississippi

an important source of toxaphene in runoff in August and September, even though sediment yield was relatively low (Table 2 and Figure 5). The cotton was defoliated in October. After harvest stalks were shredded and left on the soil surface until late February when the crop residues were incorporated into the soil, reestablishing the soil as the primary source of toxaphene.

During the 3-month period, December 1974 through February 1975, 36.15 cm (14.23 in) of rainfall [1.32 cm (0.52 in) below normal] produced sediment and toxaphene yields of 2.77 metric tons/ha (1.24 tons/acre) and 14.27 g/ha (0.013 lbs/acre), respectively. Thus, rainfall during the 7 months from August 1974 through February 1975 totaled 81.34 cm (32.02 in), 12.37 cm (4.87 in) above normal, and produced sediment and toxaphene yields of 4.37 metric tons/ha (1.95 tons/acre) and 42.1 g/ha (0.038 lbs/acre), respectively. Although rainfall was above normal for the period August 1974-February 1975 and 10.08 kg of toxaphene/ha (9 lbs/acre) were applied during August and September, toxaphene yields in runoff were lower than the yields for March-July 1974. This lower yield is attributed to the lower runoff and sediment yield observed for the period August 1974-February 1975 (Table 2).

Monthly mean toxaphene concentrations in runoff ranged from 0.006 to 0.012 mg/liter in March through July 1974 (Table 2). Concentrations reached a maximum of 0.028 mg/liter in September and subsequently decreased to $\leq$0.010 mg/liter for the remainder of the water year. The rapid decrease in toxaphene concentrations both in the soil and in the runoff following application strongly suggests that other processes, such as volatilization and degradation, are responsible for its dissipation. Field studies are underway to determine the significance of these processes.

SUMMARY

Toxaphene, DDT, DDE, and trifluralin concentrations in runoff and sediments were measured from a 15.6-ha (38.5-acre) watershed planted to continuous cotton in the Mississippi Delta, a physiographic region noted for extensive use of pesticides. Measurements during 1973 and 1974 indicated a linear relationship between pesticide yields and sediment yields in runoff. Rainfall amounts 30 and 39 percent, respectively, above the 30-year annual average of 125.35 cm (49.35 in) produced abnormally high runoff (74.2 and 88.0 cm) and sediment yields (28.9 and 26.3 metric tons/ha/year). Since rainfall amount affects runoff and hence sediment yield, the sediment and pesticide yields reported are probably much higher than would be expected in a year of normal rainfall amount and distribution.

Toxaphene, DDT, DDE, and trifluralin yields from the watershed in 1973 were 117 (0.104), 87 (0.078), 7 (0.006), and 2 (0.002) g/ha (lbs/acre), respectively. Corresponding pesticide yields in 1974 were 97 (0.086), 27 (0.024), 6 (0.005), and 2 (0.002) g/ha (lbs/acre), respectively. The application of 10.08 kg toxaphene/ha (9 lbs/acre) to the watershed in 1974 did not increase the toxaphene yield over that of 1973 when no

toxaphene was applied. The decrease in DDT yield in 1974 reflects the decrease in soil residue levels from 1973 to 1974.

Toxaphene may be applied to cotton in the spring if cutworms are present, but the major portion is applied for bollworm and boll weevil control in late July and in August and September. This summer application period coincides with near-to-full crop canopy development and relatively low average rainfall, runoff, and sediment yield. Toxaphene concentrations in the runoff were maximum during the summer application period, but yields were lower than those during the spring tillage period. Data indicate that the major portion of the annual sediment and toxaphene yields is produced in the tillage period when the soil is most vulnerable to erosion. Reduced fall and winter tillage, which is now gaining acceptance in the Delta, will probably reduce sediment and sorbed pesticide yields, but improved erosion control practices are needed during the spring and summer tillage periods to minimize sediment and associated chemical yields from croplands.

ACKNOWLEDGMENT

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EFFECT OF AN ARTIFICIALLY INCREASED SAND BEDLOAD ON STREAM MORPHOLOGY AND ITS IMPLICATIONS ON FISH HABITAT

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ABSTRACT

Sand sediment has been added daily for nearly four years to a low gradient Michigan trout stream to determine effects on stream morphology, water temperature, and ultimately the trout population. Daily sand input is adjusted so that the average sediment concentration is increased by a factor of four over the pretreatment concentration of 20 mg/l. Sediment discharge is measured just upstream of the sand input point, and also at the lower end of the treated section, 1.6 km (1 mile) downstream. Permanent stream cross-section stations are spaced at 30-m (100 ft) intervals along the 1.6-km (1 mile) control and treated sections to facilitate the measurement of channel changes. The trout population is inventoried twice a year to determine its response to the increased sand bedload.

The added sand with its resultant streambed aggradation has produced increases in stream gradient and width, and decreases in stream depth and in the total static volume of water. The stream gradient and channel form are more uniform throughout the treated reach due to pool filling. Streambed composition has changed substantially; gravel areas have decreased, sand areas increased. Water temperatures are slightly warmer in summer and cooler in winter due presumably to the wider shallower stream. The impact of these changes on the trout population will be reported at a later date to allow time for population responses, if any, to occur.

INTRODUCTION

Anglers and biologists have for a long time believed that erosion of sand into streams and the movement of sand bedload is harmful to fish. The detrimental effects of excessive sand bedload in scouring off aquatic insects or otherwise decreasing food supplies, destroying cover by filling in pools, and reducing spawning by covering gravel areas are well known. However, most studies on sediment-trout relationships have dealt with effects of increased sediment due to dredging, hydraulic mining, logging, extreme floods, etc. (Cordone, et al. 1961). Even though most of these studies did not involve measurements of the sediment load, it is reasonable to deduce that sediment concentrations (including sand bedload) from these sources are considerably higher than those in most trout streams. The question logically follows: Do the smaller quantities of sand bedload common to most trout streams have important effects on fish habitat and on trout populations? Knowledge of effects of low

sediment loads on trout habitat and of the consequent response of the trout population is important because streambank stabilization programs and restrictions on sediment inputs from construction, are partially justified by the argument that they will benefit fish even when sediment amounts are relatively low.

The objective of this study was to determine the effects of a four-fold increase in sand bedload discharge on a brook trout *Salvelinus fontinalis* (Mitchell) population and its habitat in a previously undisturbed stream. This paper reports on the effects of such a sediment increase on stream morphology, bed type, temperature, and fish cover.

STUDY AREA

The study was at the Michigan Department of Natural Resources, Hunt Creek Fisheries Research Area near Lewiston, Michigan. Hunt Creek is a small, .57 m^3/sec (20 ft^3/s) trout stream flowing through sandy glacial drift country. The deep sand drift produces little surface runoff, a high groundwater yield, and consequently the stream discharge is extremely stable. For example, records for the Thunderbay River near Hillman, of which Hunt Creek is a major tributary, show that the stream discharge that is exceeded 2 percent of the time is only 4.4 times greater than that exceeded 98 percent of the time (Veltz, et al. 1960). This stable supply of cold groundwater, low stream gradient, and small sediment discharge, is typical of trout streams throughout much of the northern part of the Lower Peninsula of Michigan.

EXPERIMENTAL DESIGN

The stream was divided into two 1.6 km (1 mi) sections, with the lower section treated and the upper section serving as the control. An additional 0.8 km (0.5 mi) of main stream above the control and a major (2.6 km) tributary to the control served as backup controls.

Collection of sediment concentration data began 1 year prior to treatment. Treatment consisted of increasing the stream's sand bedload of approximately 20 mg/l, as determined by the 1 year of pretreatment data, up to a level of 80 mg/l. Sand was added daily at the upstream end of the treated section. The amount added was varied with stream discharge so that the sediment discharge was four times that which would have normally occurred for that particular mean daily stream discharge. This was done to more closely simulate natural sediment delivery patterns to the stream. Although the once-a-day input created a slug effect at the input point, it is believed that the slow moving sand eradicated any slug effect within a short distance downstream.

Data on fish populations were collected for 5 pretreatment years and for 4 posttreatment years as of October 1975. Thus, comparisons can be made in fish populations between treated and control sections,

both before and after treatment. The entire experimental area was closed to fishing. Consequently, aside from possible treatment effects and the controlled sampling of trout in both treated and control sections for diet analysis, only natural mortality altered the trout population.

METHODS

A stream gaging station with a water level recorder was established to provide a measure of mean daily flow. In addition, staff gages were installed at four sediment sampling stations. The stations were located so that the sediment discharge entering and leaving both the control and treated sections was sampled. Sediment samples were collected weekly by sampling with a DH-48 suspended sediment sampler over wooden sills in such a manner that the sampler intake traversed the entire vertical profile of flow (Hansen, 1974). Samples collected in this manner provide a measure of the total sediment discharge.

Samples of sand bed material were analyzed for particle size distribution and then compared with that from the intended sand borrow area to insure a similar size distribution.

Yearly supplies of sand were stock-piled at the upstream end of the treated section. Starting October 1, 1971, sand was added usually once a day to the stream with an endloader. The quantity added was three times the daily sediment discharge corresponding to the current staff gage reading. To simplify the procedure, a "sediment input table" was developed which gave daily sediment input in cubic yards based on stream discharge. Several times a year the content of the endloader bucket was weighed and converted to volume based on the sampled bulk density adjusted for moisture and gravel content. This calculated volume provided a check on the equipment operator's estimates of sand input. The sand contained a small amount of gravel which gradually formed a gravel riffle at the input point. These gravel deposits were removed with a backhoe whenever the damming effect became excessive and were replaced with an equal volume of sand added over several days with the normal daily sand input. The sand bedload was trapped at the lower end of the 1.6 km treated section by an 8 x 60 m (25 x 200 ft) sediment basin. The basin was cleaned with a dragline periodically throughout the study. The basin was surveyed before and after every cleanout by "leveling" on a 1.51 m (5 ft) grid of points. These data permitted calculation of the volume of deposits trapped by the basin. Sediment samples collected at sills above and below the basin provided a means for estimating trap efficiency.

Permanent stations were established for channel cross-section measurements at 30 m (100 ft) intervals along the treated and control sections of the stream. Initial measurements were made the summer prior to treatment and at 1-year intervals thereafter. Depth was measured at all major slope breaks in the channel cross-section. The streambed was also subdivided into widths as narrow as 0.3 m (1 ft) and classi-

fied as to streambed particle size (sand, gravel, cobble), biological materials (vegetation, wood, detritus), or various combinations thereof. These data permitted calculation of channel scour and fill, changes in cross-sectional water area (the static water volume in channel reach), and streambed composition changes. A "leveling" survey was made between selected cross-sections in the treated reach. From this survey the water surface profile was drawn and then updated with each cross-section remeasurement.

A water temperature recorder had been placed at the downstream end of the treated section many years prior to treatment. Maximum-minimum thermometers distributed throughout the study area were read weekly.

RESULTS

Sediment Size

The particle size distribution of the material added to the stream was similar to that of the pretreatment samples collected from the moving sand streambed (Figure 1). There was about 4 percent gravel (>2 mm) in the material added to the stream, which remained on the streambed at the input point and affected about 30 m of channel. "Fine" sediment (<0.062 mm) averaged less than 1 percent. These "fines" washed quickly through the 1.6 km treated section, reaching Sill 1 at the lower end about 1 hour after input. Intensive sampling of the turbidity wave on three different dates at Sill 1 showed that 0.3, 0.6, and 1.5 percent of the daily sediment input was present in the wave. This turbidity wave was a daily event and passed Sill 1 in 30 to 60 minutes. The average pretreatment concentration of "fine" sediment was about 5 mg/1 and was rarely above 20 mg/1 (maximum measured was 83 mg/1). In comparison, the added "fines" were the equivalent of a mean daily concentration increase of 0.3, 0.3, and 1.0 mg/1 for the three measured waves, and the peak "instantaneous" concentration was 60 mg/1.

Sand Wave Progression

Sediment input averaged about 1.7 m^3/day (2.2 yd^3/day)or 620 m^3/year (820 yd^3/year) (Table 1). The sand added to the stream moved along the streambed as a slowly advancing sand wave or dune (Figure 2). The front of the sand wave consisted of a transition zone over about 100 meters of stream. This transition zone graded from compact, dark-colored streambed material (due to silt and fine organic particles mixed with the sand) in the unaffected area, to loose, light colored sand in the advancing sand wave. Occasionally a sharp dune front was present also.

An increase in sediment concentration was noted at Sill 1 near the lower end of the treated section in June 1973, 21 months after the start of daily additions of sand nearly 1.6 km (1 mi) upstream (Figure 3). This indicated that sand added to the stream had finally traversed that length of the treatment section, and that essentially all of the sand (1000 m3) (1300 yd^3) added during the first 21 months went into channel deposits.

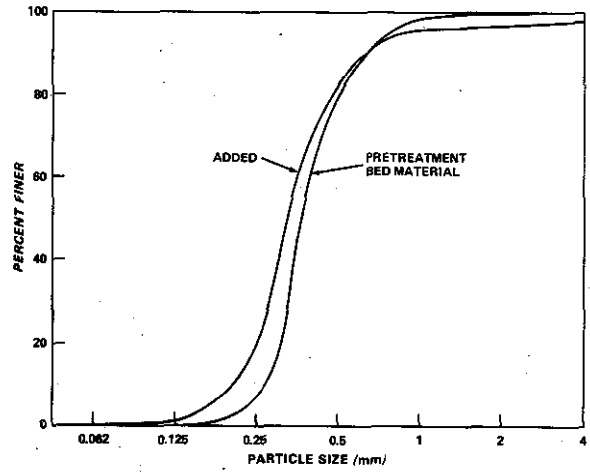

FIG. 1.--Comparison of particle size distribution of Hunt Creek bed material and sediments added to stream

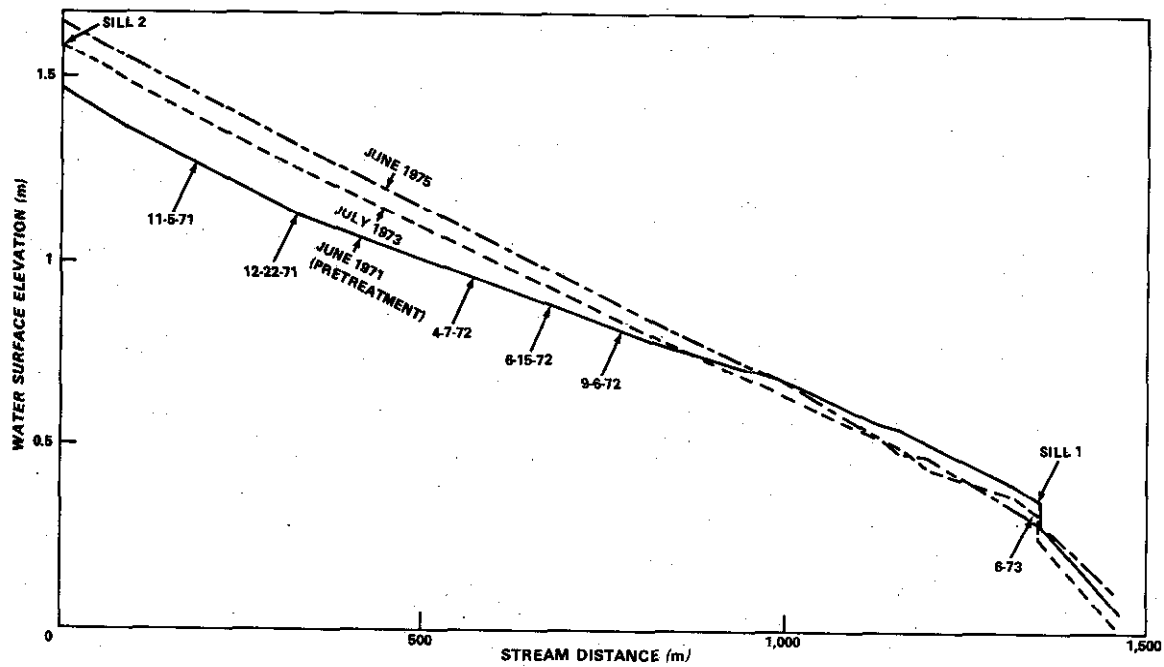

FIG. 2.--Water surface profiles for alternate years. Arrows show general location of sand wave front on given date.

Channel Response

Two months after the start of treatment, there was a general lowering of the water surface elevation throughout the entire treatment section (no comparable change in the control). This elevation drop of about 3 cm (.1 ft) was evident in the stage-discharge rating shift at Sill 1, and in elevation changes relative to the established cross-section datums throughout the entire reach. (This change was still evident in the lower half of the treatment section more than 3½ years later -- see lower 500 m (1600 ft) of water surface profile in Figure 2. Since the sand wave had progressed only 210 m (700 ft) at that time, and the water surface change occurred throughout the channel, it follows that the sand was not the causative factor. However, the turbidity wave was traversing the entire treated channel daily. In the absence of additional data, it is speculated that fine sediments settling out from the daily turbidity wave decreased the roughness of the channel boundary, thereby producing a slight increase in water velocity with a commensurate drop in head.

TABLE 1.--Annual sediment input

Water Year	Volume m^3	Volume (yd^3)
1972	665	(870)
1973	630	(824)
1974	550	(754)
1975 (Oct.-June)	346	(452)

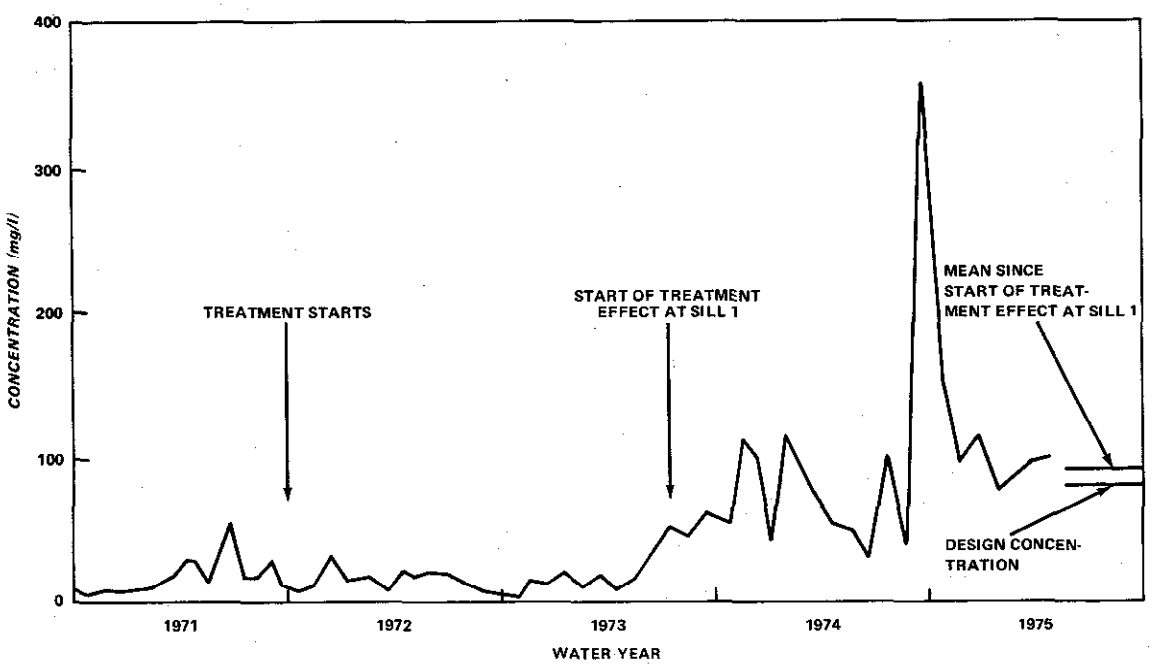

FIG. 3.--Sediment concentration at Sill 1. Points are means of five samples collected at approximately weekly intervals.

Other changes that occurred were more predictable. As the sand wave gradually progressed downstream, the streambed and water surface were both elevated. Eventually the water surface became higher than its initial datum which resulted in a wider stream. Channel gradient increased, resulting in increased water velocity. The greater water velocity together with pool filling resulted in a reduction in cross-sectional water area and, therefore, a reduced static water volume.

Streambed deposits averaged 14 cm (0.47 ft) in thickness (see "Bed Elevation" in Table 2). Maximum point depths of more than 1 m (3 ft) occurred in some pools which were at those cross-sections with maximum deposit thickness shown in Figure 4. Average stream depth decreased 14 cm (0.47 ft) from 42- to 28-cm (1.37 to 0.93 ft), almost all of which was due to a reduction in areas deeper than 0.3 m (1 ft) (Figure 5). Areas deeper than 0.6 m (2 ft) were reduced 86 percent (from 17 to 2 percent of the streambed area). There was essentially no change in stream depths in the control section.

TABLE 2.--Channel geometry changes relative to June 1971 base period.

Year	Water Elevation				Bed Elevation			
	Control		Treated		Control		Treated	
	mm	(ft)	mm	(ft)	mm	(ft)	mm	(ft)
6/72	- 3	(-.01)	- 6	(-.02)	-0.6	(-.002)	46	(.15)
6/73	-15	(-.05)	33	(.11)	0.6	(.002)	94	(.31)
6/74	--	--	--	--	0.6	(.002)	137	(.45)
6/75	-15	(-.05)	73	(.24)	-12	(-.04)	143	(.47)

Year	Stream Width				Water Volume					
	Control		Treated		Control			Treated		
1/	mm	(ft)	mm	(ft)	m^3	(yd^3)	%	m^3	(yd^3)	%
6/71	3960	(13.4)	5910	(19.4)	1274	(1665)	--	3566	(4662)	--
6/72	61	(0.2)	92	(0.3)	28	(36)	+2	-357	(-467)	-10
6/73	61	(0.2)	274	(0.9)	--	--	--	--	--	--
6/74	--	--	--	--	--	--	--	--	--	--
6/75	92	(0.3)	427	(1.4)	49	(64)	+4	-675	(-883)	-19

1/ Initial stream width and water volume given to provide a comparison for subsequent changes.

Stream width increased 43 cm (1.4 ft) (a conservative figure since the stream is now out of its low marshy banks over a considerable distance and most of the shallow "over-bank" width is not included). Static water volume decreased 19 percent (Table 2). Channel gradient between Sill 1 and Sill 2 increased from an initial .00081 to .00099 (from 4.3- to 5.2-ft/mi) and acquired a more uniform slope (Figure 2).

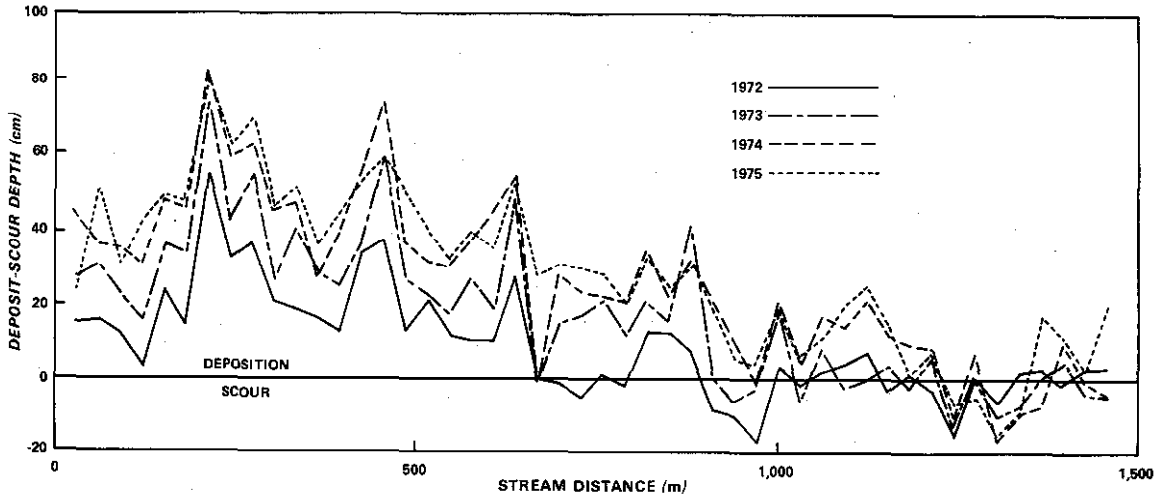

FIG. 4.--Average thickness of sand deposits at 30-meter (100 foot) intervals along the stream at annual intervals during treatment.

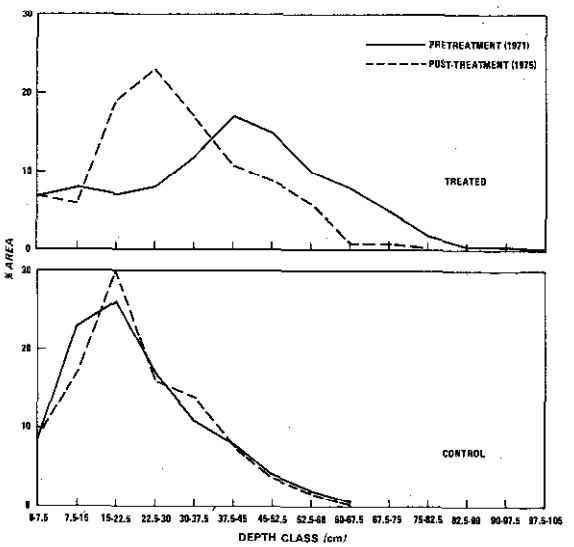

FIG. 5.--Change in stream depths 4 years after start of treatment.

Sand Budget

A comparison between the volume of sand metered into the stream, and that measured as deposits in the stream and sediment basin provides a check on the accuracy of the various measurements, together with some insight into the disposition of the added sand. Sediment samples collected above and below the sediment basin showed that almost all of the sand was trapped. Only small amounts of "fine" sand (<0.25 mm) were leaving the basin. Since this size fraction constitutes only a small portion of the bed material (Figure 1), it is believed that these losses had an insignificant effect on the budget calculations. Total adjusted sediment input as of June 3, 1975 was 2020 m^3 (2640 yd^3). Surveys indicated 1326 m (1733 yd^3) in streambed deposits and 350 m^3 (458 yd^3) trapped by the sediment basin leaving 343 m^3 (449 yd^3) unaccounted for. This unaccounted for volume is equivalent to a 4.0 cm (0.13 ft) deposit over the entire treated section.

Two areas of sand deposition not accounted for by survey were noted. Cross-section measurements on three different dates showed that immediately in front of the leading edge of the sand wave the streambed scoured to depths of 15 to 37 mm (0.05- to 0.12-ft). Since the scoured sediments were most likely predominantly organics and "fine" sediments that would pass completely through the sediment basin and since this area rapidly filled with sand from the advancing dune, it accounts for some of the "missing" sand. A second area is the compression of low density organic deposits on the streambed as they are covered by sand over-burden. Neither of these two probable areas of sand sediment storage were accounted for with the survey technique used. The "unaccounted for" sediments increased in magnitude each year until channel adjustment was essentially complete, and then remained constant during the last year of measurement.

Streambed Composition

As expected, the treatment produced a sizable (28 percent) increase in sand-covered streambed (Table 3). Gravel area decreased from the initial 17 percent down to 5 percent. These same bed types showed no trends in the control area during the same period. Areas with wood, vegetation, detritus, or various combinations of streambed types, showed large fluctuations from year to year but no definite trend. This could be due to both actual changes in their area and to changes in observer bias from year to year. It was observed that vegetation beds survived even with fairly thick sand deposits. Even when the vegetation was initially completely buried, it eventually penetrated the deposits and reestablished itself in much of its former area.

TABLE 3.--Streambed composition

Date	Sand		Gravel	
	Control	Treated	Control	Treated
	percent of area			
6/71	16	40	63	17
6/72	16	52	57	12
6/73	9	50	58	9
6/74	20	59	61	7
6/75	14	68	59	5

Water Temperature

Water temperature was slightly cooler in the spring and warmer during the winter in both the treated and control sections during the post-treatment years as compared to the pretreatment (Figure 6). An analysis of the net temperature change (Δtreated - Δcontrol) showed that temperatures averaged 0.8°C warmer in the treated section from March through September and .4°C cooler from October through February (Figure 6). Exceptions were May temperatures which were much colder (reason unknown) and December temperatures which were slightly warmer.

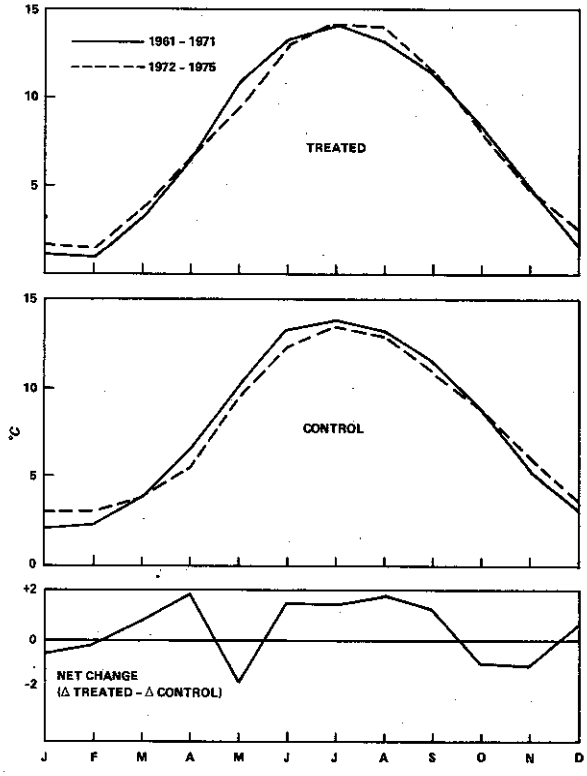

FIG. 6.--Change in mean monthly water temperatures during treatment.

DISCUSSION

Field surveys of channel deposits made in low gradient channels with an erodible bed may underestimate deposit volumes due to deposition below the initial datum. This is especially true if there are easily compressed deposits of organics on the streambed.

The sediment samples acquired a substantial high bias at Sill 2 as the sill became progressively submerged due to backwater effects above the sand input point, and a possible high bias at Sill 1 from streambed aggradation. (Submergence is defined as a reduction in the water surface elevation drop as water flows over the sill). Data from Sill 2 since treatment began was not used in any analyses, data at Sill 1 was. The mean concentration of 92 mg/l sampled at Sill 1 since the abrupt increase in sand bedload began, compares favorably with the treatment prescription of 80 mg/l. Discrepancies between these two values are due to bias in sampling and/or to additions of sand to the stream.

A low gradient stream may take a long time to adjust to an input of sand bed material. Movement rates may be a few hundred feet a month or less. Rates depend upon stream discharge, initial channel gradient, and quantity of added sediment -- factors that can vary widely from stream to stream. On Hunt Creek with an initial slope of .0008, the sand wave advanced at a rate of about 0.8 km/yr (0.5 mi/yr) with the given sediment input rate. It took between two and three years for the 1.6 km (1 mi) channel to undergo the major portion of the adjustment. During the fourth year, there was essentially no change in bed elevation (Table 2, also see Figure 4), there was very little change in the water surface profile, and the sand input and output from the section were about the same. All of these indicate that channel adjustment to the treatment effects is essentially completed.

Many changes occurred in stream morphology that might have a negative effect on fish habitat. The stream became wider and shallower, pools filled, and the center two-thirds of the stream became a uniform sand bed devoid of cover. These factors would make the trout more vulnerable to predation. The reduction in static water volume, filling of pools, reduction in channel diversity (by changing from a "pool-run" situation to essentially one long "run") -- all would tend to reduce the carrying capacity of the stream. Although deposition occurred in all areas of the stream, pools filled the most, with an 86 percent reduction in the deeper areas, thus producing a disproportionate impact on cover.

Loose, moving sand bed is the least desirable bed type from the standpoint of benthos production. Thus, the increase in this bed type might have undesirable effects on food production.

A wider, shallower stream would be expected to result in warmer water temperatures. On the other hand, the reduction in static water volume, increased velocities, and increased albedo of the streambed would tend to decrease temperature. The net effect of these contrasting changes on Hunt Creek was a slight warming of water in summer and cooling in winter.

This study illustrates that profound changes can occur from seemingly insignificant concentrations (but constant input) of sand sediment in a low gradient stream. Sand sediment is deceiving in that it does not produce the turbidity commonly associated with severe sedimentation. The treated section of Hunt Creek would still be classified as a clear, cold water trout stream. Also, 80 mg/l of moving sand bedload are not readily apparent in steeper gradient streams. Only when the gradients are low enough for significant deposition does the sediment become evident. Sampling for sand sediment over a natural streambed with standard sediment samplers in small low gradient streams will miss much of the sand sediment. This may lead the observer to erroneously conclude that there is no significant sediment discharge in that "clear" stream, when, in fact, there may be considerable sand moving in the unsampled zone adjacent to the streambed.

The long duration of channel adjustment and slow travel time of the sand wave caution against attempting to assess damages too quickly after an increase in sediment input to a low gradient stream. Allowances should be made for a long channel adjustment period, a long recovery period once the sediment source has been controlled, and since biological responses will presumably lag behind morphological changes, any monitoring or research (particularly on the fisheries aspects) will of necessity last even longer.

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ROLE OF THE SEDIMENTATION IN THE SELF-PURIFICATION OF THE SCHELDT ESTUARY

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and R. Wollast, Professor, University of Brussels, Belgium.

ABSTRACT

The hydrographic basin of the Scheldt covers a heavily populated and industrialised region and drains waters extremely polluted due to uncontrolled discharges.

In this partially stratified estuary, the mixing process of fresh and salt water is responsible for an important deposition of the suspended load of the river in a restricted area corresponding to the harbour of Antwerp. This important shoaling is explained by the physico-chemical properties of the suspended matter and the hydrodynamical characteristics of the estuarine region.

The mud accumulated in the estuary contains high levels of organic matter and heavy metals, and the sedimentation process contributes markedly to the removal of pollutants from the surface water. Mass-balances of input, output in the estuary and accumulation in the sediments were estimated for various pollutants. The role of the sediments on the oxygen budget was deduced from the previous mass-balances.

Large concentrations of nutrients persist in the brackish water zone where oxygen is available and turbidity is low. As a consequence, this zone is eutrophied and diatom blooms are frequent.

INTRODUCTION

The Scheldt estuary (fig. 1) constitutes the southern branch of the "Golden Delta" formed by the rivers Rhine, Meuse and Scheldt. The Scheldt river and his tributaries drain 21580 square kilometers in North-West France, West Belgium and South-West Netherland. The hydrographic basin covers one of the most heavily populated regions of Europ where a highly diversified industrial activity has developed. Most of the discharges are uncontrolled and as a consequence large amounts of domestic and industrial wastes are carried by the river. The tidal range varies along the estuary from 3,70 m at the mouth, increasing to 5 m near Antwerp, and decreasing to 2 m in Gent.

The mean riverdischarge amounts 120 m^3/sec at the mouth, or 5 million cubic meters during one tidal period while the volume of sea water flowing up the estuary during the flood tide is about one billion cubic meters.

The Scheldt may be considered as a well mixed estuary with only a small and local vertical salinity gradient. The mixing zone of fresh and salt water extends over a distance of 70 kms to 100 kms. A comparison of the river water at the head and of the brackish water at the mouth of the estuary reveales that important physical, chemical and biological processes occuring in the

mixing zone modifie strikingly the transport of pollutants to the sea. It was recognized that the intense sedimentation typical of slightly stratified estuaries is one of the most effective processes of removal of pollutants from the surface waters. We will present briefly the mechanisms and characteristics of the mud deposition and their influence on the mass-transfert, the accumulation and the transformation of some typical elements in the estuarine zone of the Scheldt.

MECHANISMS OF MUD DEPOSITION

One of the most important characteristics for the transport of pollutants in an estuarine system is the usually large residence time of the fresh watermasses. In the case of the Scheldt estuary, the average cross-sectional ebb or flood currents are about 0,7 m/sec, with instantaneous maximum values reaching 1,5 m/sec. However the residual currents averaged over one complete tidal cycle in a cross-section drop from 0,08 m/sec at km 100 to 0,02 m/sec at km 50. The total residence time in the brackish water zone, which extends over 100 km, is comprised between two and three months. From an environmental point of view, this implies a high accumulation of the persistent pollutants and intense modifications of the chemical or biological active substances in the estuarine region.

Mixing of fresh and salt water induces on the other hand complicated water movements and influences the physico-chemical behaviour of both suspended and dissolved species.

The measurements of vertical profiles of salinity, temperature and currents permit to distinguish two zones with different hydrodynamical characteristics. The lower one extending from the sea to km 50, is constituted by well defined flood and ebb channels which contribute to the intense mixing. The vertical stratification is generally small. The upper zone extending from km 50 to the fresh water zone (km 100) is characterized by a single and narrower channel. The vertical stratification is higher, chiefly in the area of the harbour of Antwerp, where average salinity gradients of about 0,2 $^{\circ}/_{oo}$ S/meter are observed. Despites their relatively low values, these salinity gradients influence sufficiently the vertical distribution of the currents in order to modify markedly the residual currents averaged over one complete tidal cycle. The density currents slacken th ebb movement and accelerate the flood movement near the bottom. A reverse effect in a surface layer compensates this bottom movement. Consequently, the residual currents near the bottom are orientated upstreams in the lower zone. They are however orientated downstreams in the fresh water zone, and the two opposed movements cancel out in the area of the harbour of Antwerp, which is thus a highly favorable zone for the accumulation of sediments.

The existence of a vertical gradient of turbidity, associated with these water-movements, will create a zone of maximum turbidity, which is well demonstrated in the case of the Scheldt (fig 2).
This region corresponds also roughly to the transition from fresh to brackish water. The suspended matter transported by the riverwater is mainly composed of colloïdal particles which floculate as the salinity increases. Laboratory experiments carried out with suspended matter of the river Scheldt show that

an intense floculation occurs as soon as the salinity reaches 2°/₀₀ S and is completed for a salinity of 5°/₀₀ S. A further increase of salinity produces larger flocs, but with lower densities and finally lower sedimentation velocities. The optimum values of salinity for floculation occur in the zone favorable to sedimentation and accumulation, leading to intense shoaling of mud in a restricted area (Fig. 3). The influence of salinity on the removal of suspended matter from the surface waters is well demonstrated by fig.4 .
The restricted accumulation of mud in the region of the harbour of Antwerp is also depicted in fig. 5 showing the organic content of the sediment along the estuary. The high values near Terneuzen are due to a local input of highly polluted waters transported by the Terneuzen canal (fig 1)

ESTIMATION OF THE TRANSPORT OF POLLUTANTS AND THEIR ACCUMULATION IN SEDIMENTS

Taking into account the physical characteristics of the Scheldt, the estuary was divided into two zones : an upper one from km 100 to km 55 and a lower one from km 55 to the mouth.
Four times a year fixed stations situated at the boundaries of these regions were managed during five days; hourly samples were taken at three depths and continuous measurements of the profile of the currents along a vertical were performed. A longitudinal survey was also executed monthly, following the low tide from the mouth to km 130. Approximately 50 surface samples are collected during each survey.
Observations over three years, enable us to estimate annual mass balances of input, transport and accumulation by sedimentation of various pollutants in the two estuarine regions.
The mass balances were established for each compartment by considering for both suspended and dissolved compounds the net flow due to river discharge, the longitudinal turbulent dispersion (estimated from the salinity profile), the sedimentation process, and the lateral input due to tributaries and sewers. In the case of organic matter the biodegradation was estimated by difference between the input and the output in the compartment.
Figure 6 shows the mass balance obtained for suspended solids. From a total load of 1520 10^3 Tons/y discharged in the first zone, 1200 10^3 Tons/y are deposited in this upper compartment and 320 10^3 Tons/y are transported to the second one, where the sedimentation is much smaller, and finally only 120 10^3 Tons/y reache the North-Sea. Coarse sand is also transported upwards by the strong bottom currents. The contribution of this process was evaluated by comparing the chemical composition of suspended matter carried by fresh water and by the bottom currents to the sediments deposited in that region. The estimation of 2 10^6 Tons/y of solid deposited in the upper region is in good agreement with an estimation of the shoaling in this region wich amounts to 10^7 m^3/y of mud, with a water content of 80% by weight.
As one may expect, pollutants discharged in the river strongly affects the composition of suspended matter especially in the case of organic matter and of heavy metals like Pb, Zn, Cu. The accumulation of sediments in the upper zone acts thus as a very efficient removal process which prevents further transport of sediments to the sea.
The evolution of organic matter in the Scheldt may be easily shown by a simple permanganate oxydability test (fig 2).
The mass balance obtained for the upper zone (fig 6) indicates that the removal of organic matter by sedimentation and biodegradation is spectacular :

only 20% of the total organic input is transfered downwards. These processes further continue between km 55 and the mouth and the organic matter reaching the North-Sea finally amounts to 15 10^3 Tons/y as solid and 1.8 10^3 Tons/y as dissolved organic matter.
The intense activity of heterotrophic bacteria is responsible for the existence of a large anaerobic region in the estuary especially during the summer (fig 6) The reaeration in the upper zone, which was estimated around 60 10^3 Tons O_2/y is insufficient to provide the oxydants necessary to account for the bacteriological activity. Other oxydants like NO_3, MnO_2, Fe_2O_3 and SO_4 can be successively used and a complete oxydation-reduction budget is necessary to describe correctly the evolution of the chemical composition of water under microbiological influence (Billen et all, 1975).
In the case of the upper Scheldt estuary, the degradation of the organic matter in the freshly deposited sediments is essentially related to sulfate reduction, producing measurable amounts of iron sulfides (greigite, pyrite).
A tentative mass-balance for oxygen is given in figure 6. It does not include the oxydo-reduction reactions for nitrogen, iron and manganese which are probably negligeable in the oxygen budget but are however important to describe the behaviour of these elements.
In the anaerobic zone, Mn^{4+}, NO_3^- and Fe^{3+} are successively reduced and in the downward zone they are regenerated in the opposite order as the oxydo-reduction potential increases.
Mass balances for copper, zinc and lead in the Scheldt estuary are given in figure 6. . A large fraction of these heavy metals are introduced in the river as solid compounds or are rapidly precipitated if they are discharged in a soluble form. The intense deposition of sediments in the upper zone constitutes thus again an efficient removal process. It should also be noticed that the ratio solute/suspended matter increases for heavy metals as the salinity increases. A carefull investigation of the longitudinal concentration profiles presently under progress indicates that copper and to a lesser extend Zn, are submitted to dissolution or desorption as the salinity increases. This may be possibly related to the evolution of the oxydo-reduction potential and the presence of even minute amounts of H_2S in the anaerobic zone.
As one may expect, the activity of the phytoplankton is also markedly influenced by the physical and chemical properties of the water masses of the estuary. This is well demonstrated by the behaviour of dissolved silica in the estuarine system. From May to early October, dissolved silica carried by the fresh water is rapidely consumed in a limited region situated between km 10 and km 50. During the winter, dissolved silica behave as a conservative compound. The seasonal variation suggests that this phenomenon is related to the activity of diatoms, abundantly present in the estuarine waters. The restricted area of their activity may be explained by the fact that they become active only when the turbidity is low enough to permit the photosynthesis to occur. This hypothesis is confirmed by primary productivity measurements.
It must be underlined that the rate of consumption of silica is unusually high for an estuary (1 to 2 mgr SiO_2/m^3h).
The large amounts of nitrogen and phosphorus discharged in the Scheldt persist in the lower part of the estuary which may considered as partially eutrophied. As a consequence, the supply of dissolved silica to the North-Sea is practically worthless during the summer, 95% of the dissolved silica carried by the river

water being consumed in the estuary itself. The same phenomenon occuring into the Rhine (Van Bennekom et all, 1974) it becomes that the North-Sea is actually deprived of an important source of a major nutrient, in contrast with nitrogen and phosphorus which are discharged in large amounts. This disequilibrium between the nutrients may affect the plankton population of the North-Sea.

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Billen, G., Nero, F., and Smitz, J., 1975, Mathematical Model of the Influence of Bacterial Activity on Water Composition and its Implementation on a Hybrid Computer, in Computer Simulation of Water Resources Systems (ed Vansteenkiste G.C.) North-Holland Publishing Cy, Amsterdam, 503-519.

Fig. 1. The Scheldt estuary.

Fig. 2. Longitudinal profile of salinity, turbidity, chemical oxygen demand and disolved oxygen content (january 1973).

Fig. 3. Mechanisms of mud deposition.

Fig. 4. Evolution of turbidity as a function of salinity in the Scheldt estuary. (The broken line represents the evaluation due to a simple mixing of fresh water with sea water).

Fig. 5. Organic matter and zinc content in sediments as runction of the distance to the sea.

Fig. 6. Mass balances of pollutants and oxygen in the Scheldt.

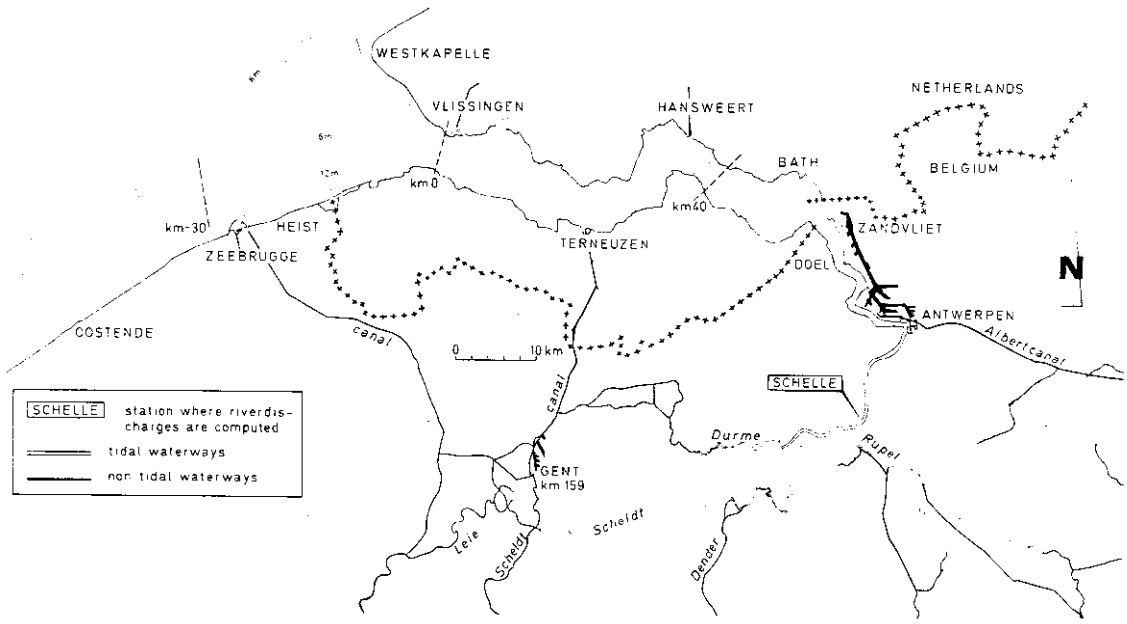
WESTKAPELLE
VLISSINGEN
HANSWEERT
NETHERLANDS
BELGIUM
BATH
km 0
km 40
km-30
HEIST
ZEEBRUGGE
TERNEUZEN
ZANDVLIET
DOEL
ANTWERPEN
N
COSTENDE
canal
0 10 km
Albertcanal
SCHELLE
station where riverdis-charges are computed
tidal waterways
non tidal waterways
Durme
Rupel
GENT
km 159
Leie
Schelde
Scheldt
Dender

Fig. 1

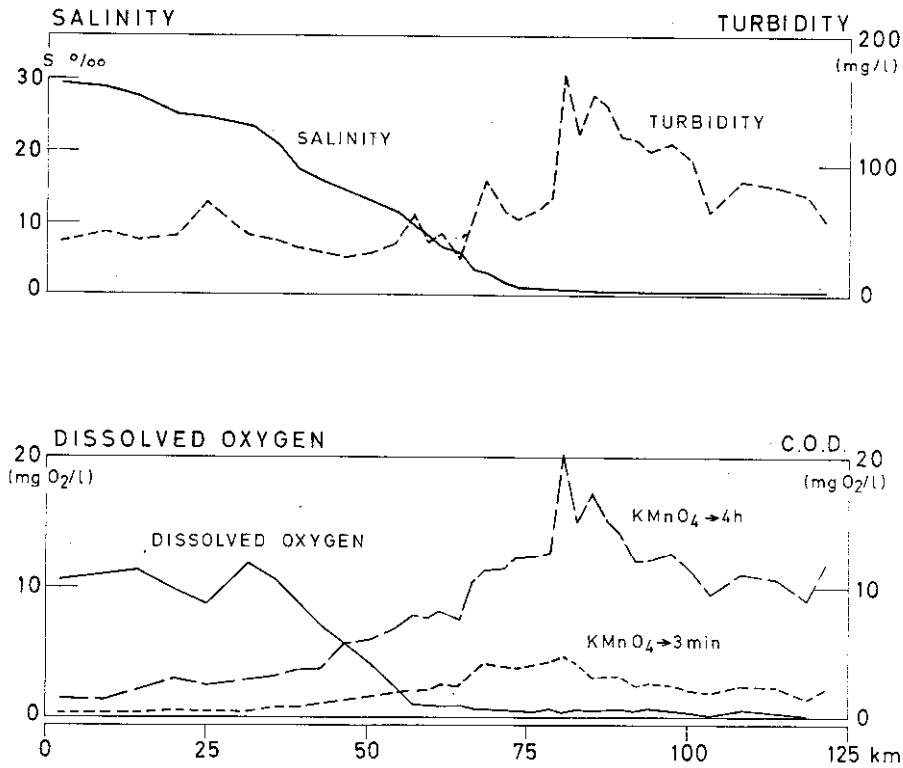
SALINITY
TURBIDITY
S °/oo
(mg/l)
30
20
10
0
200
100
0
SALINITY
TURBIDITY
DISSOLVED OXYGEN
C.O.D.
20
(mg O2/l)
10
0
KMnO4 → 4h
KMnO4 → 3 min
0
25
50
75
100
125 km

Fig. 2

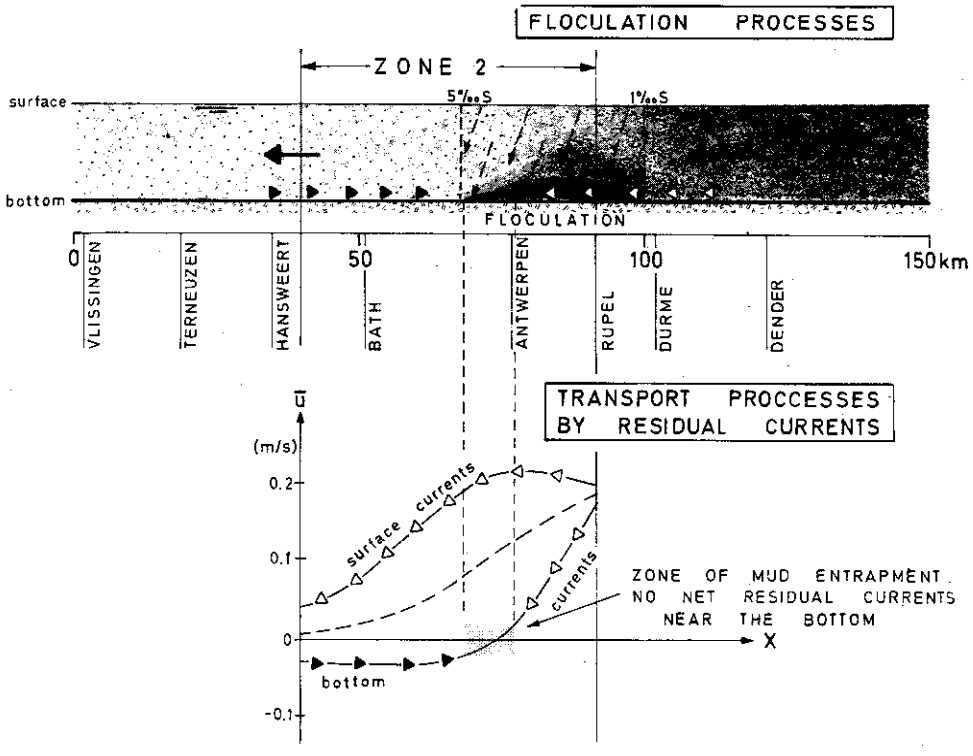
FLOCULATION PROCESSES
ZONE 2
surface
bottom
5‰S
1‰S
FLOCULATION
0
50
100
150km
VLISSINGEN
TERNEUZEN
HANSWEERT
BATH
ANTWERPEN
RUPEL
DURME
DENDER
TRANSPORT PROCCESSES
BY RESIDUAL CURRENTS
(m/s)
0.2
0.1
0
-0.1
surface currents
currents
bottom
ZONE OF MUD ENTRAPMENT
NO NET RESIDUAL CURRENTS
NEAR THE BOTTOM
X

Fig. 3

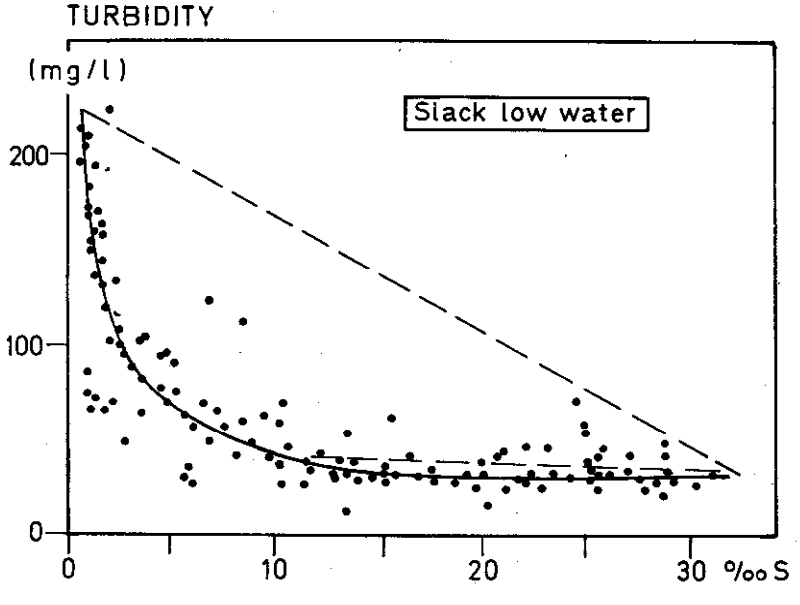
TURBIDITY
(mg/l)
Slack low water
200
100
0
0
10
20
30 ‰S

Fig. 4

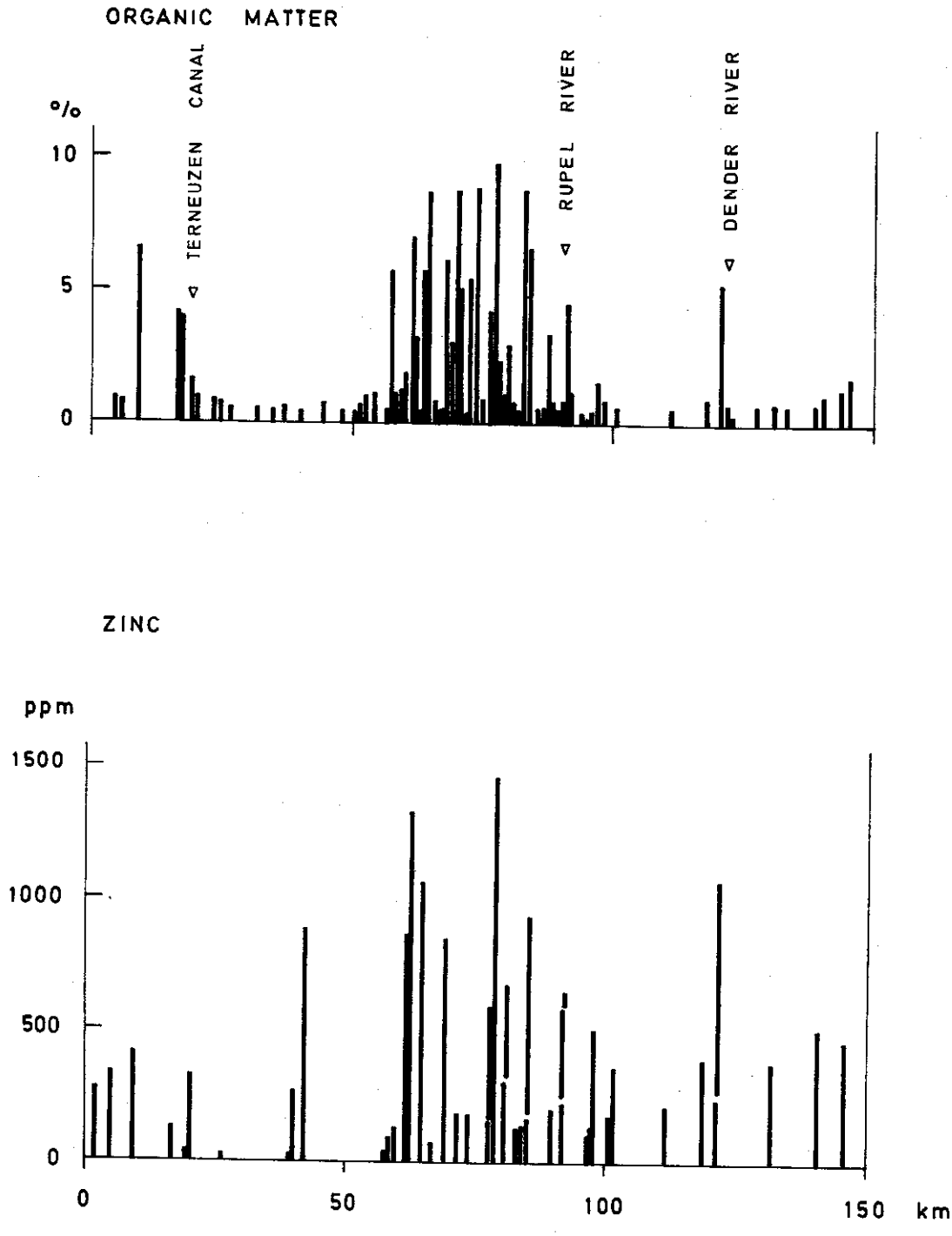
ORGANIC MATTER
%o
10
5
0
TERNEUZEN CANAL
RUPEL RIVER
DENDER RIVER
ZINC
ppm
1500
1000
500
0
0
50
100
150
km

Fig. 5

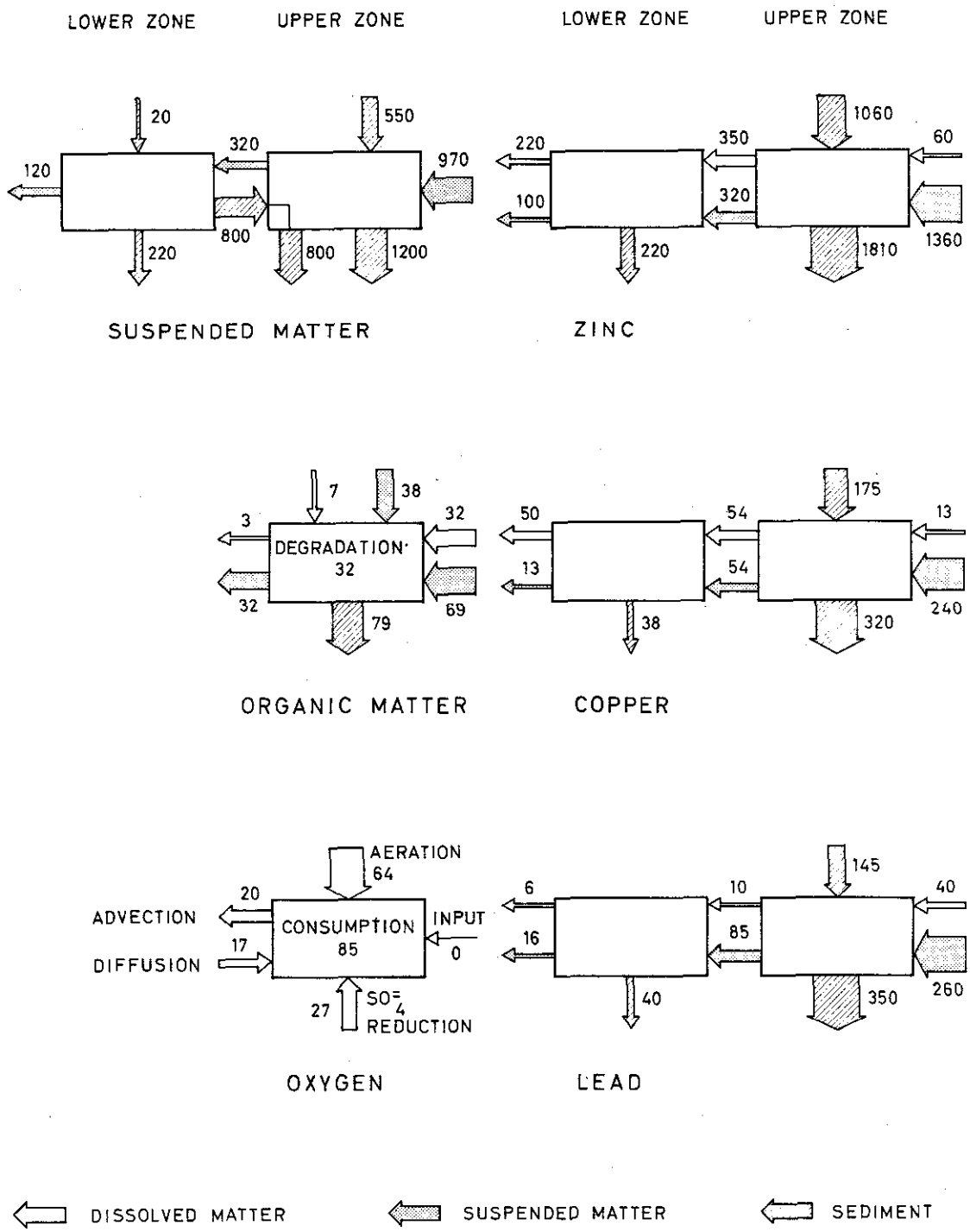
LOWER ZONE
UPPER ZONE
LOWER ZONE
UPPER ZONE
20
550
320
970
120
800
220
800
1200
SUSPENDED MATTER
1060
220
350
60
100
320
220
1810
1360
ZINC
7
38
3
32
DEGRADATION
32
32
79
69
ORGANIC MATTER
175
50
54
13
13
54
38
320
240
COPPER
AERATION
64
20
ADVECTION
CONSUMPTION
85
INPUT
0
17
DIFFUSION
27
SO=4
REDUCTION
OXYGEN
145
6
10
40
16
85
40
350
260
LEAD
DISSOLVED MATTER
SUSPENDED MATTER
SEDIMENT

Fig. 6