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ENVIRONMENTAL

MOVEMENT OF DREDGED SAND AT THALWEG DISPOSAL SITES

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ABSTRACT

Thalweg disposal experiments have been conducted with the U.S. Army Corps of Engineers, Rock Island District, at three sites on the Upper Mississippi River. During normal channel maintenance, hydraulically dredged sand was tagged with sand coated with fluorescent dye prior to disposal as a pile in the thalweg. In postdisposal surveys surficial bottom sediment samples were collected in the disposal area and in the thalweg and border areas downstream to determine the movement of the dredged sand relative to environmentally sensitive river habitats.

The experiments were initiated in successive years, and the tagged sand has been tracked for 1-3 years depending on the site. Although the downstream movement of the dredged sand was not the same at each site, the general pattern of behavior was similar. Downstream movement was confined primarily to the main channel and occurred in response to periods of high river discharge. There was no statistically significant evidence of dredged sand dispersing out of the main channel into nearby border areas or sloughs. The distributions of dyed sand in cores from one site suggest that the dredged sand has been incorporated into natural bed forms.

INTRODUCTION

Dredging for channel maintenance in the Upper Mississippi River creates a disposal problem. While the dredged material is generally clean sand and water quality is not an issue, loss of riverine habitat due to sand deposition in main-channel borders, backwaters, and sloughs is a concern. An alternative to dredged material disposal on beaches or upland sites is open water disposal in deep sections of the main channel or "thalweg disposal." This alternative is attractive from a logistical and cost perspective at dredging sites not far upstream from deep main-channel reaches and may result in less impact to the river environment than other means of disposal.

The Rock Island District of the U.S. Army Corps of Engineers has undertaken studies to assess the viability of thalweg disposal of clean dredged material at selected locations in the Upper Mississippi River. The principal issues addressed by these studies have been the impact of the disposal on the local sediment movement in the river and the potential loss of sensitive habitat areas. This paper deals primarily with the results of an experimental program conducted by Argonne National Laboratory with the Rock Island District to investigate these issues by means of field demonstrations.

An analysis of the physical consequences of thalweg disposal by Lagasse (1975) and an application of a one-dimensional model to the Upper Mississippi River by Lagasse, Simons, and Chen (1976) provided early estimates of potential impacts. The model showed a slight tendency for the downstream crossing below the disposal site to aggrade during the annual hydrograph after disposal. However, the one-dimensional model required that the material removed from the crossing and deposited in the pool be distributed evenly over the cross section in each case -- something that would not necessarily happen in practice where disposal in a discrete pile or piles in the thalweg was anticipated. Considering the large data requirements for high-resolution, two-dimensional modeling of thalweg disposal and the inherent limitations of such modeling, the District decided to study the viability of thalweg disposal through full-scale demonstration.

A study of methodologies for a field demonstration of thalweg disposal by Simons and Chen (1980) included the recommendation that the dredged sand be tagged with a tracer and suggested various tracer materials and detection techniques. After preliminary testing of one detection technique, the Rock Island District requested that Argonne National Laboratory work with the District to develop a suitable experimental program for study of thalweg disposal at dredging sites on the Upper Mississippi River.

EXPERIMENTAL PROGRAM

The objective of the experimental program developed by Argonne has been to tag the dredged sand with a tracer (dyed sand) prior to disposal so that inferences about the fate of the dredged sand can be drawn from the distribution of dyed sand on the river bed. A mass balance for the tracer was never intended; rather the presence or absence of the tracer was taken as an indicator of the dredged sand's presence or absence. The development of a plan for a large-scale tracer experiment in the dynamic environment of a large river required considerations of flexibility, reliability, and cost. From the need for rapid and repeatable field sampling and analysis at many stations in the disposal area and downstream, an experimental approach evolved for postdisposal surveys that coupled grab sampling of bottom sediments from a survey boat with on-board precision navigation and dyed-sand detection (Van Loon, et al., 1982). This approach has proved successful in three thalweg disposal experiments initiated in successive years during routine maintenance dredging. The sites at which tagged dredged sand was placed in deep sections of the main channel are: (1) Gordon's Ferry, 27.8 km south of Dubuque, Iowa, (2) Whitney Island, 6.3 km north of Hannibal, Missouri, and (3) Savanna Bay, about 1.7 km north of Savanna, Illinois. Summarized below are results from measurements taken during a 1057-day period following disposal at the Gordon's Ferry site, a 746-day postdisposal period at the Whitney Island site, and a 287-day postdisposal period at Savanna Bay. Detailed results are given in reports by McCown, Paddock, and Ditmars (1984) and Paddock and McCown (1984).

Tagging was accomplished by injecting sand coated with fluorescent dye into the suction side of the pump of a hydraulic dredge to create nearly uniform concentrations of dyed sand in the mixture of 100-1000 ppm. The size distribution of the sand that was dyed and used to tag the dredged sand was matched as closely as possible to the expected size of the dredged sand. The cost and the logistics problems of collecting large amounts of sand from the dredging site for dyeing precluded an exact match. Instead, sand with closely matching characteristics was obtained from other sources and dyed by Corps personnel. Dyeing was accomplished by combining in a concrete mixer fluorescent dye, vinyl plastic, sand, and enough acetone to dissolve the dye and plastic. The acetone evaporates readily during mixing so that the sand particles are individually coated and do not clump together.

In each experiment, surveys to trace the movement of the tagged sand were conducted immediately following the dredging and tagging operation, a week later, a month later, and then about three times a year thereafter. The survey boat was equipped with a depth sounder and a precise microwave positioning system. The positioning system was interfaced with a real-time position-plotting system which: (1) allowed easily repeatable bottom sampling and bathymetric measurements to be conducted with resolution of a few meters and (2) provided the flexibility to adjust sampling locations in response to previous sampling results. Surficial bottom samples were collected with a Ponar™ dredge and winch. A portion of the material collected by the dredge was spread evenly on a sample tray with a 23 × 23 cm surface area. The sample tray was placed into an ultraviolet light box where it was inspected visually to detect dyed sand grains and photographed for later detailed analysis. The sample was saved for reinspection if necessary. Fifty to two hundred surficial bottom samples were obtained during each survey in the vicinity of the disposal pile and downstream. These measurements provided relatively detailed information on the distribution of dyed sand in the surficial sediments and allowed inferences to be drawn regarding downstream movement and dispersal of dredged sand. Bathymetric surveys in the vicinity of the disposal pile identified gross changes in the topography of the pile and the river bed.

EXPERIMENTAL RESULTS

The first experiment was conducted in conjunction with dredging at Gordon's Ferry in October 1981. It provided an opportunity to evaluate the tagging operation and sampling procedures at full scale, as well as to study the fate of the disposal pile. The river reach in which the experiment was conducted is about 400-600 m wide and relatively straight. The dredging was necessitated by a longitudinal bar formed as a downstream, submerged extension of an island located in mid-river. About 6500 m^3 of dredged sand was tagged with 7.1 m^3 of dyed sand and discharged above the water surface about 0.8 km downstream of the dredge cut. The dredged sand formed a pile along the thalweg (8 m deep) of the river. The pile was about 300 m long, 25-45 m wide at the base, and 1.2-1.9 m high.

A plan view of the pile as determined from bathymetric measurements and the extent of the dyed sand from surficial bottom samples taken one day after disposal are given in Fig. 1. The pile created by the disposal operation is a distinct bathymetric feature. The extent of the dyed sand has been defined for experiment-specific statistical reasons as the region from which bottom samples produced three or more dyed sand grains on the surface of the sample tray. The fact that the downstream extent of the dyed sand is greater than that of the physical pile is attributed to downstream transport of sediment in the water column while it settled out after surface disposal. While one or two dyed sand grains were found in a few samples as far as 200 m downstream from the boundary of the "three-grain" region, no evidence was found in the first survey that dredged sand had moved from the thalweg region to border areas or sloughs.

Figure 2 is the hydrograph for this reach as recorded at Lock and Dam 12 located about 12 km below the experiment site. Superposed on the hydrograph are the times of site surveys subsequent to disposal and a brief indication of the changes in pile bathymetry and dyed sand distributions between successive surveys. Little change in dyed sand distributions and bottom topography was noted in surveys during the first five months after disposal. However, following high river flows during spring 1982, a survey in early June (Survey V) revealed that the pile was no longer bathymetrically distinguishable from other bed forms (sand dunes). An alongstream bathymetric transect through the area of

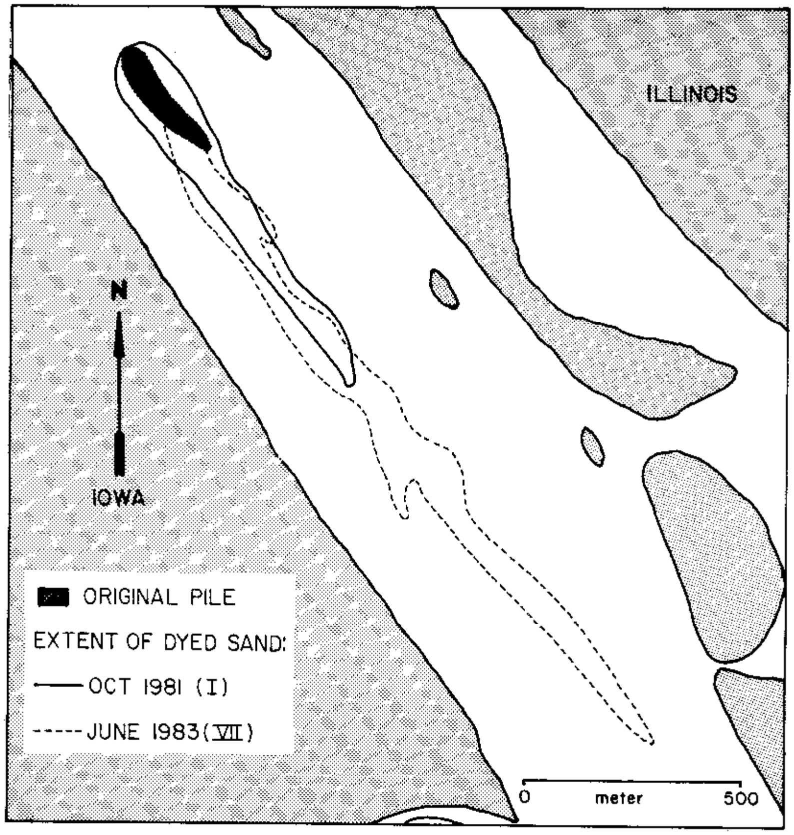
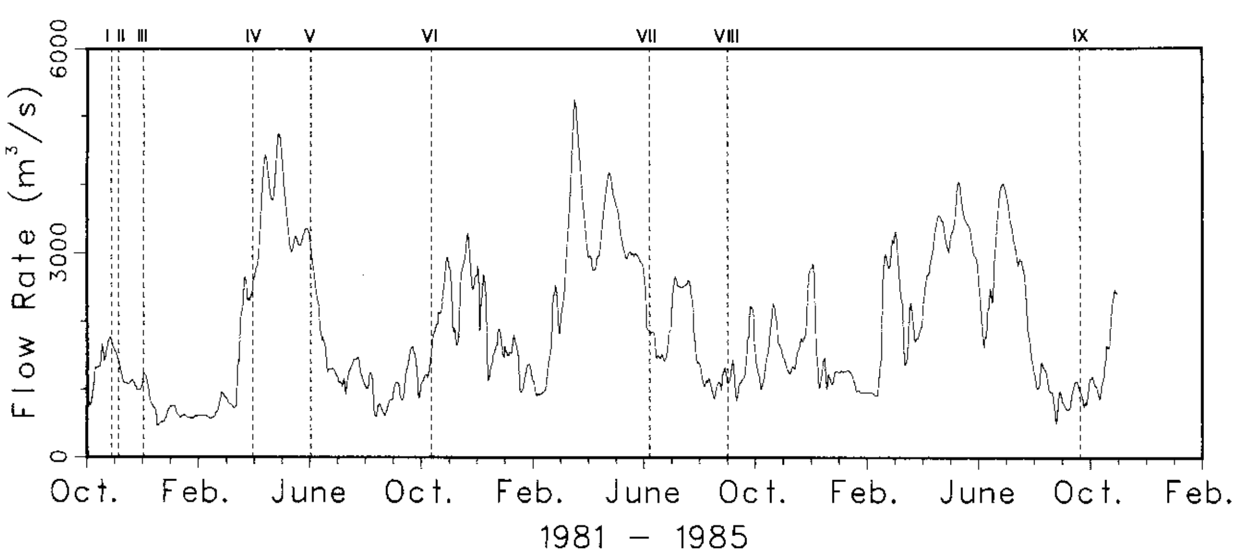


Fig. 1 Dyed Sand Distributions and Location of the Physical Pile at the Gordon's Ferry Site

the original pile showed sand dunes with wavelengths of 30-40 m and trough-to-crest amplitudes of 1-2 m. Subsequent bathymetric surveys through August 1983 indicated that the average elevation of the bottom in the disposal area had not returned to the elevation that existed prior to disposal and was about 1 m higher. In contrast to the bathymetry results, the bottom sampling results following the spring 1982 flood did not indicate a significant change in the distribution of dyed sand in the surficial sediments.

It was not until a survey conducted in early June 1983, following high river flows in spring 1983 (approximately a year after the physical pile disappeared), that a significant change in the extent of the dyed sand was observed. Those high flows were associated with a flood with a return period of about 5 years. As indicated in Fig. 1, the June 1983 survey (Survey VII) showed that the dyed sand then extended about 1000 m farther downstream and was spread over about twice the original area. The increased areal extent of the dyed sand was primarily due to downstream movement within the thalweg, and no evidence of significant lateral movement exists. Subsequent surveys indicated no change in the bottom conditions and no change in the length of the dyed sand region, although the area within the three-grain region has decreased slightly due to a narrowing of the upstream portion of the region. In the surveys at Gordon's Ferry there has been no evidence of migration of tagged dredged sand from the thalweg into nearby channel border areas, side channels, or backwaters.



Survey	Days after Disposal	Bathymetry in Disposal Area	Length of Dyed-Sand Contour
I	1	Distinct pile, 300 m long	900 m
II	8	No signnificant change	No change
III	35	No significant change	No change
IV	153	No significant change	No change
V	217	Pile gone, dunes present	No change
VI	349	No significant change	No change
VII	588	No significant change	1800 m
VIII	673	No significant change	No change
IX	1057	No significant change	No change (narrower)

Fig. 2 River Hydrograph for the Gordon's Ferry Site, Including Dates and Major Results of Surveys

The enlarged area of the bottom surficial sediments containing dyed sand and the bathymetric disappearance of the pile raised questions about the vertical distribution of tagged sand in the river bottom. Because the surficial sediment samples only provided clues as to how tagged sand had become incorporated into the river bottom, 25 cores of bottom sediments (1.2-2.0 m long) were taken in August 1983 in the original disposal area and downstream where surficial sediments contained dyed sand. A lightweight pneumatic coring device developed by Fuller and Meisburger (1982) was used for the coring operation. An analysis of the sediment cores for dyed sand as a function of depth below the sediment-water interface showed that the tagged sand existed not only in the surficial sediments but also throughout the dunes. Analysis of all 25 cores showed that cores taken in dune crests contained dyed sand as deep as 1.5 m, but cores taken in dune troughs contained little or no dyed sand. Figure 3 shows the positions of five consecutive downstream cores relative to the local dune profiles and the region of dyed sand within

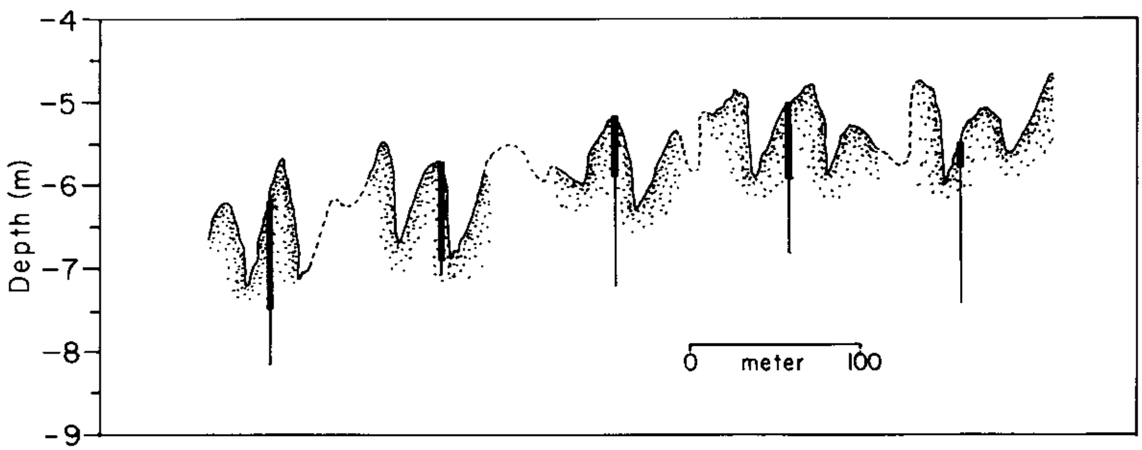


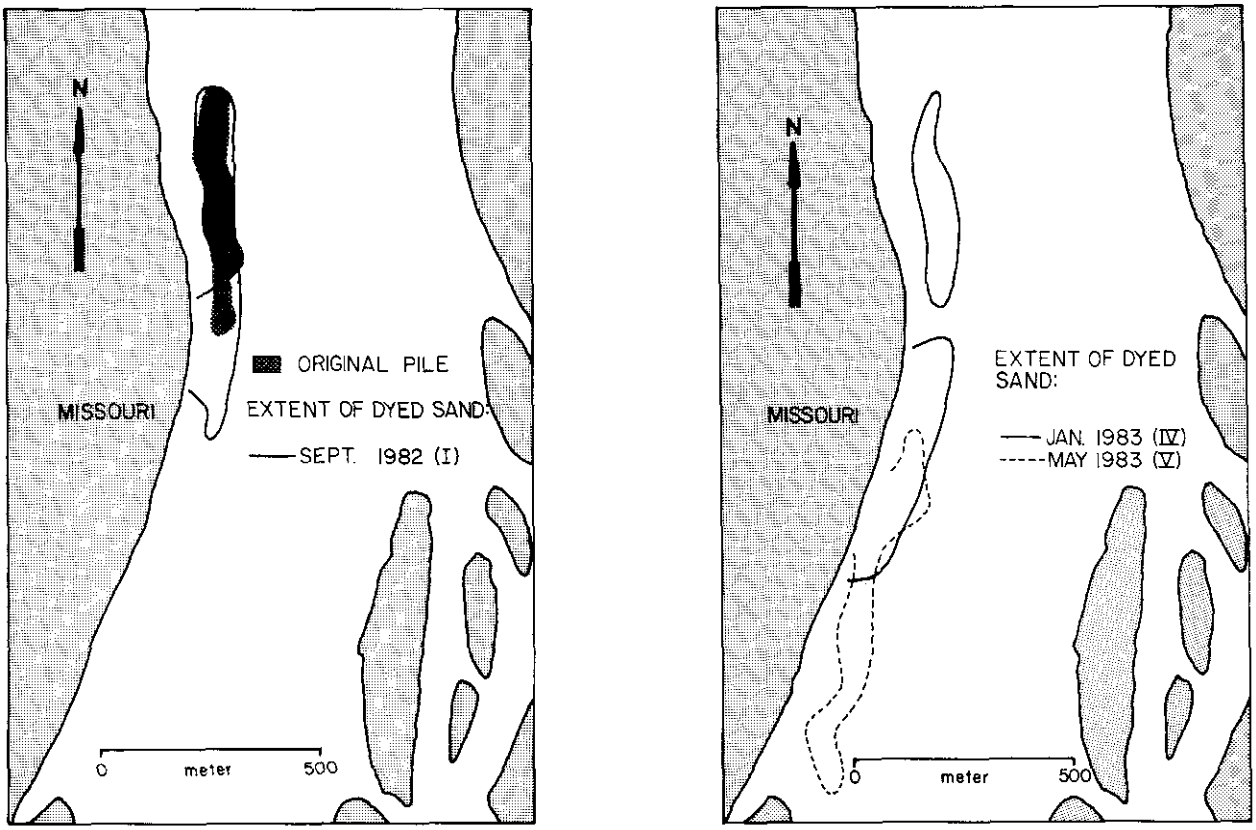
Fig. 3 Positions and Lengths of Five Consecutive Downstream Cores Relative to the Local Dune Profiles. Portions of the Cores Containing Dyed Sand are Indicated by Bold Lines.

each core. It appears that the dredged sand has become incorporated into the dune structure and is moving slowly in the dunes. Such transport is suggested by cores from the upstream portion of the disposal region containing layers of dyed sand and clean sand.

A second experiment, conducted in conjunction with routine dredging at Whitney Island, involved considerably more dredged sand and a different type of placement location and configuration. In contrast to the Gordon's Ferry site, the river reach for the Whitney Island experiment is not straight and has a more complex geometry. The river width varies from 400-730 m in the reach, and submerged wing dams protrude from the banks on both the Missouri and Illinois sides. The channel crosses from the Illinois to the Missouri side in this area, and shoaling between wing dams had reduced the opening between the bar from the Illinois shore and a wing dam on the Missouri shore. About 43,000 m^3 of dredged sand was tagged in September 1982 with 4.0 m^3 of dyed sand and deposited in an irregular pile of overlapping mounds about 600 m long, 40-80 m wide, and 0.7-5.3 m high near the west bank of the river, where the original depth was 5.7 to 7.3 m. The average height of the pile (about 3.0 m) was about one-half the predisposal water depth. At Gordon's Ferry, the height of the more regularly shaped pile is less than one-quarter of the water depth. Figure 4a shows the location of the disposal pile and extent of dyed sand as determined in the first survey.

During subsequent surveys at Whitney Island in the early fall of 1982, reductions in the heights of the higher peaks in the disposal pile were observed. By Survey III (38 days after disposal) the average height of the pile decreased from 3.0 m to about 1.7 m. Following a period of relatively high river flow, Survey IV in early January 1983 revealed that the average bottom elevation in the disposal area had returned to its original value and that irregular dunes had developed. Coincident with the disappearance of the bathymetrically distinguishable pile in January 1983, a major portion of the tagged sand (about the same areal size as the original pile) was found to have separated and moved about 400 m downstream leaving a smaller portion at the original disposal site (Fig. 4b).

Bathymetric measurements made in May (249 days after disposal), after a period of exceptionally high spring floods (about a 50-year flood), indicated that the average bottom elevation in the original disposal area was about 0.5 m lower than it had been

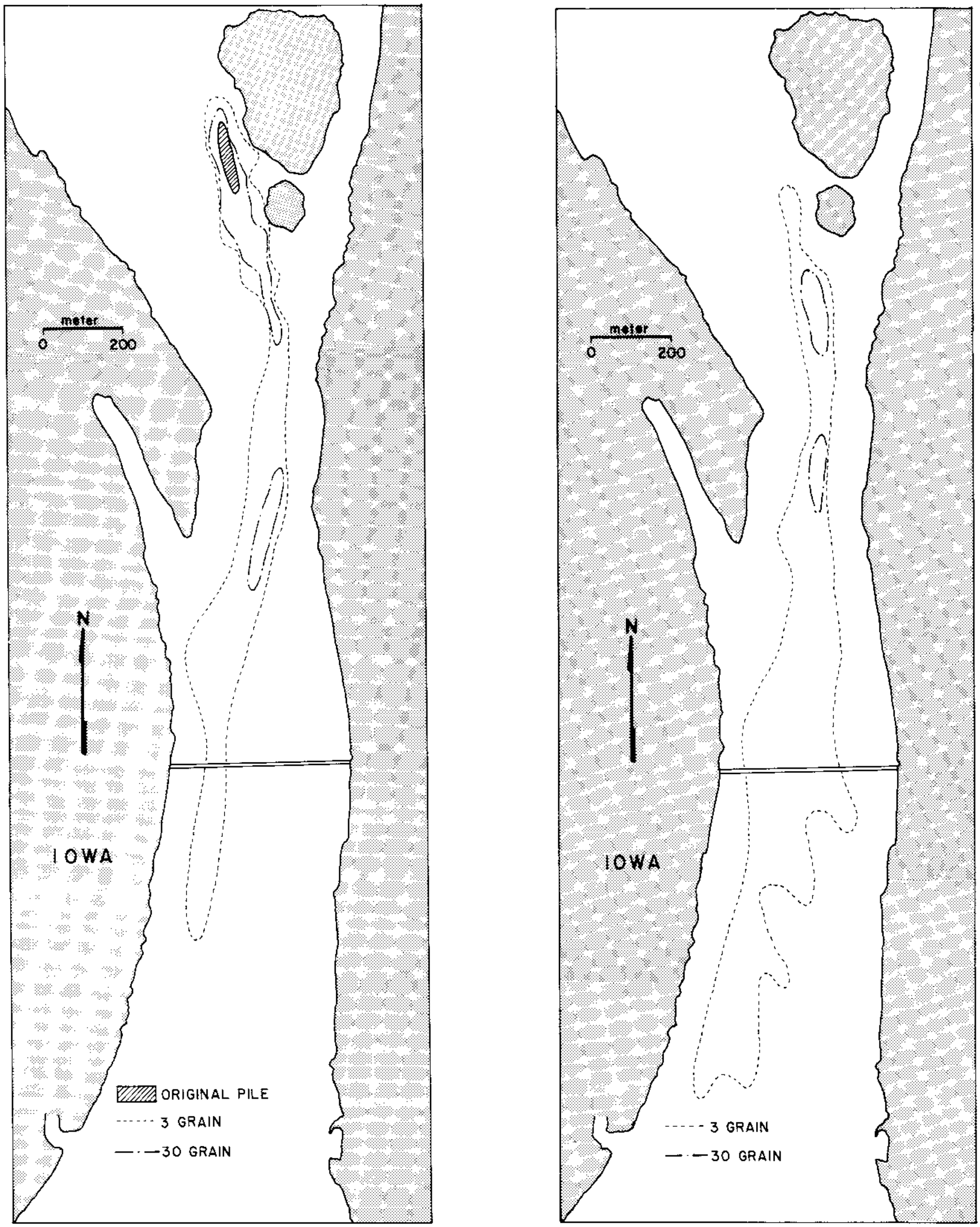


(a) One day after disposal **(b) After successive flooding events**

Fig. 4 Dyed Sand Distributions and Location of the Physical Pile at the Whitney Island Site

prior to disposal. It was also found that virtually no dyed sand remained in the area of the original pile. The areal extent of the portion of dyed sand that had moved downstream was about 45% larger than the original area and extended even further downstream (Fig. 4b) than it did in January. The maximum number of dyed sand grains per sampling station was found to be about 25% of the number found on the first postdisposal survey. Data from subsequent surveys indicate no change in the topography of the bottom and dyed sand concentrations at most sampling locations were below detection limits. Despite the more rapid movement of tagged sand downstream here than at Gordon's Ferry, no evidence of any migration of tagged sand into channel borders, backwaters, or sloughs was found. In fact, at Whitney Island the tagged material moved downstream in the thalweg directly toward the entrance of a slough but followed the thalweg away from the slough.

The third experiment, also conducted in conjunction with routine dredging, was at Savanna Bay (Fig. 5). The disposal site was 600 m upstream of a deep hole (>12 m deep) and in a deep, narrow channel (about 100 m wide) with rock revetment on one side and a shoal area (<2 m deep) on the other side. Dredging was necessary to widen a sharp bend just upstream from the disposal site to provide easier passage for barge traffic. Below the disposal site the river runs straight southward for several kilometers.



(a) One day after disposal

(b) 280 days after disposal

Fig. 5 Dyed Sand Distributions and Location of the Physical Pile at Savanna Bay

About 12,000 m^3 of dredged sand was tagged in October 1983 with about 5.6 m^3 of dyed sand and deposited in a series of overlapping mounds that extended about 170 m along the river in water originally about 8.5 m deep. The height of the disposal pile above the local natural bottom ranged from 0.7 m to 3.1 m, with an average height of 1.9 m. The width at the base of the pile ranged from 25 m to 55 m, with an average width of about 45 m. River conditions during the dredging operation were quite different than those that existed at the other two sites. River flow was unusually high for the time of the year and resulted in currents estimated to be as high as 3 m/s in the restricted channel where the disposal operation took place.

Bathymetric measurements indicated that the response of the disposal pile at Savanna Bay was similar to that at the other two sites. Prior to the first major flood, the average height of the pile had decreased and ripples had formed on the peak of the pile. However, as at the other sites, the pile was eradicated by the first major flood and the bottom returned to its predisposal form of dune structures.

The initital areal extent of the three-grain region at the Savanna Bay site was about three times larger than that at the other two sites (see Fig. 5). At the Gordon's Ferry and Whitney Island sites dyed sand was found to extend a few hundred meters downstream of the disposal pile, but it was found about 1.8 km downstream of the pile at the Savanna site. The large initial downstream dispersal of dyed sand at Savanna seems to be related to the high flows that existed during the disposal operations and not to the volume of the disposal (about twice that at Gordon's Ferry and one-quarter that at Whitney Island). The large initial extent of the three-grain region prompted the identification of a "thirty-grain" region from surficial sampling results as representative of dyed sand concentrations in the immediate vicinity of the physical pile.

The location and areal extent of the tagged sand (three-grain region) showed little change 153 days after disposal, but, as indicated in Fig. 5, at 280 days it had almost doubled in size and extended 500 m farther downstream following a spring flood of exceptional duration. The upstream end of the region moved about 200 m downstream as did the thirty-grain region in the vicinity of the original pile. The appearance during surveys of a thirty-grain region in the hole 600 m downstream from disposal varied from survey to survey and reflected changes in flow conditions. Despite the relatively large initial dispersal, no evidence was found at Savanna Bay that tagged sand had migrated from the thalweg into sensitive main-channel borders, backwaters, or sloughs.

CONCLUSIONS

The tagging of the dredged sand prior to disposal and the sampling procedures have provided an effective means to infer the gross behavior of the dredged sand during demonstration-scale experiments. The general pattern of behavior of the disposal pile and the tagged sand has been similar at all sites. No evidence was found at any site suggesting large-scale movement of tagged dredged material from the thalweg into sensitive habitat areas such as main-channel borders, backwaters, or sloughs. The pile was eradicated by the first flood at all sites, and the dredged sand appears to have been incorporated into the bed forms of the natural channel. The tagged sand moved downstream, within the thalweg, in response to floods. Differences in the details of movement (distance moved and rates) of the tagged sand at the individual sites did exist, but these differences are probably attributable to individual site topographic and hydraulic characteristics together with differences in initial conditions.

ACKNOWLEDGMENTS

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SEDIMENT-NUTRIENT TRANSPORT IN AGRICULTURAL RUNOFF

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ABSTRACT

Accurate predictions of sediment and associated nutrient transport are important from land use, management, and environmental standpoints. To predict sediment yield for individual runoff events the Modified Universal Soil Loss Equation (MUSLE) was employed for 23 grassed and cropped watersheds in the Southern Plains over study periods of 3 to 5 years. Use of MUSLE involved both measured and computed runoff energy factors. Corresponding losses of soluble P were predicted using a soil P desorption equation, and particulate P and N losses were predicted using a relationship between enrichment ratio (nutrient content of sediment/source soil) and soil loss. In general, the results indicate that MUSLE and the nutrient equations provided realistic estimates of sediment and nutrient transport in runoff.

INTRODUCTION

The need for accurate predictions of sediment and associated nutrient transport from agricultural watersheds is well recognized. The predictions may be utilized to 1) evaluate soil fertility losses, 2) determine the effectiveness of management practices for sediment and nutrient loss reduction and, 3) assess the environmental impacts of agricultural land use. For several years, we have participated in a series of cooperative watershed studies across the Southern Plains to assess the impact of various land uses on agricultural water quality (Jones et al., 1985; Sharpley et al., 1985; Smith et al., 1983; 1984). This paper presents an overview of our modeling approaches used to predict sediment, phosphorus, and nitrogen transport from agricultural watersheds in the Blackland Prairies (BP), High Plains (HP), Reddish Prairies (RP), and the Rolling Red Plains (RRP) major land resource areas of Oklahoma and Texas.

PREDICTIVE EQUATIONS

Sediment

Predictions of sediment yield are based on the modified universal soil loss equation (MUSLE). In this equation (Williams, 1975), the rainfall energy factor of USLE (Wischmeier and Smith, 1960) is replaced with a runoff energy factor. The replacement gives MUSLE certain advantages over USLE, i.e. 1) application to individual storms, 2) elimination of the need for sediment delivery ratios, and 3) greater accuracy because runoff generally accounts for more sediment yield variation than does rainfall. The MUSLE may be stated as:

$$Y = 11.8\ (Qq_p)^{0.56}\ KCPSL \quad [1]$$

where
Y = sediment yield in metric tons
Q = runoff volume in m^3
q_p = peak runoff rate in m^3/sec
K = soil erodibility factor
C = crop management factor
P = erosion-control practice factor, and
SL = slope length, gradient factor

Except for substituting the runoff energy factor, $11.8\ (Qq_p)^{0.56}$, for the rainfall energy factor in USLE, the remainder of the equation is identical to USLE.

For each watershed the MUSLE predictions were determined using both measured and computed runoff energy factors. The measured factors involved actual runoff volumes and peak flow rates from individual events. The computed runoff energy factors were determined using the EPIC (Erosion Productivity Impact Calculator) hydrologic model. Details of the model are given elsewhere (Williams et al., 1984), but the major processes include surface runoff, percolation, return flow, reservoir storage, and sedimentation. Surface runoff is computed from daily rainfall with the curve number technique (SCS, 1972).

The other factors K, C, P, and SL in Eq. [1] were obtained from Agricultural Handbook 537 (Wischmeier and Smith, 1978), or EPIC (Williams et al., 1984). For the croplands, P was unity and K, C, and SL ranged from 0.28 to 0.37, 0.003 to 0.680, and 0.22 to 1.06, respectively. For the grasslands, P was unity and K, C, and SL ranged from 0.28 to 0.37, 0.002 to 0.004, and 0.14 to 1.05, respectively.

Nutrients

Predictions of soluble P (SP) in runoff were described by a soil P desorption equation (Sharpley et al., 1981). The equation was selected because preliminary testing with simulated rainfall showed it to be more versatile than transport equations involving partition coefficients or equilibrium relationships (Ahuja et al., 1981; Sharpley et al., 1982). This is because the desorption equation incorporates parameters describing the depth of surface soil-runoff interaction, storm size, and the runoff water - suspended soil ratio. The equation may be written as:

$$P_r = \frac{K\ P_a\ EDI\ BD\ t^{\alpha}\ W^{\beta}}{V} \quad [2]$$

where
P_r = storm average SP concentration of runoff (mg L^{-1})
K, α, and β = constants for a given soil
P_a = soil available P content (mg kg^{-1})
EDI = effective depth of interaction between surface soil and runoff in SP transport (mm)
BD = bulk density (Mg m^{-3})
t = storm duration (min)
W = water: soil ratio ($cm^3\ g^{-1}$)
and V = total runoff (mm) during the event

In the case of soluble soil N, no desorption equation was employed since the primary constituent, nitrate, is generally not sorbed by surface soil material.

Particulate phosphorus (PP) and nitrogen (PN) contents in runoff were calculated using an enrichment ratio (ER) approach as described by Sharpley et al (1985), where

$$PP = \text{Soil TP} \cdot \text{Sediment concentration} \cdot \text{PER} \qquad [3]$$
$$PN = \text{Soil PN} \cdot \text{Sediment concentration} \cdot \text{NER} \qquad [4]$$

with soil units as mg kg^{-1} and sediment units as g L^{-1}. The enrichment ratios for PP (PER) and PN (NER) were predicted by the following equations (Sharpley et al., 1985):

$$\ln(\text{PER}) = 2.48 - 0.27 \ln(\text{soil loss}) \qquad [5]$$
$$\ln(\text{NER}) = 2.00 - 0.20 \ln(\text{soil loss}) \qquad [6]$$

where the units of soil loss for each runoff event are kg ha^{-1}.

MATERIALS AND METHODS

Watersheds and Sampling

The 23 watersheds used in the study were selected to provide good representation of the major land resource areas. General information about the watersheds is given in the left part of Table 1, with more specific details available in earlier publications (Sharpley et al., 1985; Smith et al., 1983). Suffice it to note here that the watersheds encompassed a range of sizes (1 to 122 ha), soils (Mollisols, Vertisols, and Inceptisols), slopes (1 to 9%), grasses (native and introduced), crops (wheat, oats, sorghum, and cotton), fertilizer P (0 to 40 kgP ha^{-1} y^{-1}) and N (0 to 134 kgN ha^{-1} y^{-1}) application rates, and study periods (3 to 5 years).

Watershed runoff was measured using pre-calibrated flumes or weirs equipped with FW-1 stage recorders. Sediment discharge was determined from suspended sediment samples taken automatically for the duration of each hydrograph. After comparison with the runoff hydrograph, samples for the subject watershed were composited in proportion to total flow to provide a single representative sample of liquid and sediment. Sediment concentration was determined gravimetrically after removal of liquid. Soluble P, particulate P, and particulate N were determined as described previously (Smith et al., 1983).

Statistical methods were conducted using standard procedures given in Snedecor (1956). In the case of the linear regression analysis, a slope greater than unity indicates sediment or nutrient discharge is overpredicted, whereas a slope less than unity indicate the same is underpredicted.

Table 1. Per event comparison of predicted and measured sediment yields of watersheds for various years 1976 through 1980

Major Land Use /Area	Watershed	Size	Average Slope	Study Period	Events	Mean Sediment Yield/Event			R^2		Regression Slope	
						Meas.	MUSLE[b]	EP/MUSLE[c]	MUSLE	EP/MUSLE	MUSLE	EP/MUSLE
Grasslands		(ha)	(%)			----------kg/ha------------						
BP	SW11	1.08	0.98	77-80	20	278	287	224	0.91	0.39	0.89	1.06
	W10	7.97	2.10	76-80	18	23	23	23	0.88	0.64	0.84	1.24
	Y14	2.27	1.38	77-80	28	124	128	149	0.77	0.83	0.93	0.91
RP	FR1	1.62	2.58	77-80	20	7	7	7	0.89	0.53	0.84	0.98
	FR2	1.62	2.88	77-80	17	8	8	10	0.96	0.54	0.95	0.76
	FR3	1.62	3.18	77-80	19	8	9	11	0.96	0.47	1.08	0.75
	FR4	1.62	3.64	77-80	21	15	15	10	0.99	0.35	1.05	0.59
RRP	WW1	4.80	7.00	77-80	34	10	11	7	0.94	0.04*	1.06	0.32
	WW2	5.56	8.20	77-80	54	111	111	120	0.88	0.17	1.10	0.43
Grasslands to Wheatlands												
RP	FR5	1.62	3.48	77-80	26	50	63	50	1.00	0.80	0.80	1.53
	FR6	1.62	2.88	77-80	23	46	47	44	0.94	0.89	0.87	1.44
	FR7	1.62	2.88	77-80	28	25	37	45	0.99	0.21	0.67	0.43
	FR8	1.62	2.73	77-80	29	26	18	12	0.69	0.11*	0.66	1.23
RRP	WW3	2.71	8.60	77-80	29	51	51	65	1.00	0.70	1.02	2.06
	WW4	2.91	7.40	77-80	40	38	37	46	0.88	0.51	1.09	1.39
Mixed Lands (Grasslands and Croplands)												
BP	Y	122	2.57	76-80	50	74	58	66	0.90	0.76	1.36	1.53
	Y2	53	2.86	76-80	34	148	174	138	0.83	0.75	0.75	1.40
Croplands												
BP	Y6	6.6	3.21	76-80	24	542	534	617	0.60	0.15*	1.02	0.53
	Y8	8.4	2.24	76-80	18	343	343	356	0.98	0.49	1.12	1.33
	Y10	7.5	1.88	76-80	23	812	632	790	0.84	0.59	1.32	1.01
HP	G10	3.3	1.80	78-80	16	396	386	475	0.99	0.87	1.09	0.97
	G11	2.6	2.00	78-80	19	364	286	222	0.98	0.44	1.23	1.61
	G12	2.1	2.20	78-80	15	428	531	807	0.94	0.78	0.76	0.44

[a] BP, Texas Blacklands Prairie; HP, Southern High Plains; RP, Central Rolling Red Prairie; RRP, Central Rolling Red Plains.
[b] Based on measured runoff energy term.
[c] Based on EPIC computed runoff energy term.
* Not significant at $P = 0.05$

RESULTS AND DISCUSSION

Per Event Amounts

Sediment

A comparison of the MUSLE predicted and actual measured amounts of sediment yield on an event basis for the individual watersheds is given in Table 1. The MUSLE predictions have been obtained using both measured and computed Qq_p terms. Computed Qq_p terms were included because, under most field situations, measured Qq_p terms are not available. Therefore, applicability of MUSLE on a general basis rests on how well the Qq_p terms can be computed. As shown in Table 1, both MUSLE estimates provided fairly close values to those actually measured. Moreover, this was the case over a wide range of results within and among watersheds, with mean sediment yields ranging from essentially none (i.e. <10kg ha^{-1}) on certain grasslands to as much as 812 kg ha^{-1} on the Y10 cropland watershed. Typically, R^2 values for MUSLE using the measured Qq_p term were 0.80 or higher and the regression slopes were close to unity. Corresponding statistics using a computed Qq_p term were not as good, but the regression coefficients were still significant at the 5% level for 20 of the 23 watersheds. Consequently, over a wide range of land uses and conditions the general utility of MUSLE seems promising whether a measured or computed Qq_p term is used.

The fact that realistic predictions were obtained using a computed Qq_p term with MUSLE is particularly encouraging. This indicates that MUSLE may be a useful tool for predicting sediment yield under field conditions beyond just the calibrated watershed situation.

Nutrients

Per event summaries for phosphorus and nitrogen are given in Table 2. No consistent differences (P = 0.01 or better) in predicted and measured values were obtained and the regression slopes were generally close to unity, indicating the overall utility of the prediction techniques for the subject nutrients. Considering the wide range of conditions and treatments, the predictions are quite good.

In the case of SP, however, low flow events (i.e. <75 mm) may sometimes be over predicted (Sharpley et al., 1985). Fortunately, such events typically constitute only a small portion (<10%) of the runoff. Consequently, in most cases the small events can be ignored when determining SP losses. For exceptional cases, the portion that can be ignored will be determined by the predictions errors considered tolerable.

Potential problems that may be encountered with SP and the low flow events result, in part, from the fact that a constant EDI was used in Equation [2]. The EDI will vary somewhat with raindrop kinetic energy, soil slope and length, surface soil shaping and presence of clods (Ahuja et al., 1982, 1983; Sharpley et al., 1981). Therefore, more exact pre-

TABLE 2-Per event comparison of predicted and measured nutrient yields of the watersheds for various years 1976 through 1980.

Major Land Res. Area[I]	Watershed	Study Period	Soluble P Meas.	Soluble P Pred.	R^2	Regression slope	Particulate P Meas.	Particulate P Pred.	R^2	Regression slope	Particulate N Meas.	Particulate N Pred.	R^2	Regression slope
			--g ha^{-1}--				---g ha^{-1}---				---g ha^{-1}---			
Grassland														
BP	SW11	77-80	17	18	0.90	0.94	196	207	0.95	1.19	779	906	0.95	1.33
	W10	76-80	28	22	0.89	1.07	30	34	0.91	1.27	510	550	0.91	1.01
	Y14	77-80	17	14	0.94	1.27	57	59	0.86	0.96	899	710	0.98	0.59
RP	FR1	77-80	16	19	0.93	1.12	16	16	0.90	0.98	300	280	0.71	1.25
	FR2	77-80	19	21	0.95	1.08	18	19	0.99	0.74	410	380	0.99	1.01
	FR3	77-80	13	14	0.95	1.10	21	24	0.89	1.14	260	260	0.74	1.00
	FR4	77-80	30	29	0.99	0.92	17	19	0.89	0.83	440	400	0.97	0.88
RR	WW1	77-80	3	3	0.93	1.01	12	12	0.90	1.18	56	60	0.83	1.12
	WW2	77-80	3	3	0.94	1.02	71	78	0.77	0.98	184	132	0.54	0.49
Grassland to Wheatlands														
RP	FR5	77-80	16	17	0.94	1.04	56	55	0.89	0.97	169	227	0.84	1.13
	FR6	77-80	14	15	0.96	1.00	82	89	0.97	0.92	349	378	0.99	0.89
	FR7	77-80	16	17	0.97	1.02	40	47	0.88	1.07	280	260	0.87	0.98
	FR8	77-80	5	6	0.85	1.09	30	35	0.96	0.98	200	190	0.98	0.98
RRP	WW3	77-80	4	4	0.94	0.98	78	83	0.89	1.05	420	390	0.99	0.89
	WW4	77-80	4	5	0.88	1.01	36	42	0.92	1.13	86	179	0.97	0.89
Mixed Lands (Grassland and Croplands)														
BP	Y	76-80	16	11	0.86	0.93	52	56	0.75	0.92	317	303	0.96	0.94
	Y2	76-80	67	57	0.97	0.93	136	131	0.98	0.95	630	580	0.94	0.88
Croplands														
BP	Y6	76-80	10	8	0.87	0.94	292	288	0.92	0.97	1120	1180	0.98	1.07
	Y8	76-80	10	16	0.96	1.02	232	266	0.84	1.30	813	747	0.97	0.95
	Y10	76-80	14	18	0.84	1.02	458	442	0.95	0.85	1470	1430	0.94	0.91
HP	G10	78-80	7	8	0.94	1.10	235	215	0.97	0.83	974	823	0.98	0.81
	G11	78-80	16	19	0.91	1.11	162	126	0.84	0.59	500	430	0.98	0.91
	G12	78-80	23	26	0.90	1.11	359	407	0.88	1.01	1090	950	0.99	0.78

+ All R^2 values are statistically significant at the 0.01% level and predicted values are not different at the 1% level from measured values using the t-test, paired data.

I BP, Texas Blackland Prairie; HP, Southern High Plains; RP, Central Rolling Red Prairie; RRP, Central Rolling Red Plains.

dictions of SP can be anticipated by additional calibration of EDI with storm intensity and prevailing management systems. Similarly, some adjustment in the SP predictions technique may be needed if a large runoff event occurs shortly after fertilizer P application (Sharpley et al., 1982).

No prediction data are presented regarding soluble N in surface runoff. This is because the main constituent, nitrate, is not closely related to that of the surface soil, due to rapid movement of nitrate into the soil profile with infiltrating water away from the zone of runoff removal (Sharpley et al., 1985; Smith et al., 1983).

Unlike SP, no delineation has been found between runoff volume and those small events for which PP and PN were less well predicted. Instead, discrepancies encountered with PP and PN predictions tend to revolve around the relationship between ER and soil loss. In the present study a constant value for slope and intercept was used to predict PER (0.27 and 2.48, respectively) and NER (0.20 and 2.00, respectively). For more exacting predictions, making regression slope and intercept values a function of factors affecting runoff energy, such as rainfall intensity and duration, vegetative cover, and management practices should improve the prediction of PER and NER, and thus PP and PN transport in runoff.

Annual Amounts

While knowledge of sediment and nutrient values on an event basis is necessary from an environmental/water quality standpoint, annual amounts are often desired from a soil fertility/land management standpoint. Figures 1 and 2 are plots expressing the amounts on an annual basis for sediment and nutrients, respectively.

Mean annual measured sediment amounts ranged from <100 kg ha^{-1} for the grasslands to nearly 4000 kg ha^{-1} for the Y10, BP cropland watershed. The MUSLE-predicted values were usually close to the measured values, using either a measured or computed Qq_p term. This was the case for both grasslands and croplands.

Tolerable annual soil losses for the RRP grasslands and other croplands are 5550 and 11,100 kg ha^{-1}, respectively (Smith et al., 1984). Therefore, none of the watersheds posed serious erosion problems during the study periods (3 to 5 years). It should be noted, however, that all watersheds had received recommended conservation treatments prior to and during the study periods.

Mean annual measured nutrient amounts ranged from 0.015 kg ha^{-1} SP for the WW1, RRP grassland watershed to nearly 7 kg ha^{-1} PN for the Y10, BP cropland watershed. Generally, annual losses were < 0.1 , < 1, and < 5 kg ha^{-1}, respectively, for SP, PP, and PN. For all nutrients, predicted values compared favorably to measured. While the values in Figure 2 are small from a soil fertility point, they clearly point out the magnitudes of the differences within and among nutrients that may be expected from a wide range of watersheds.

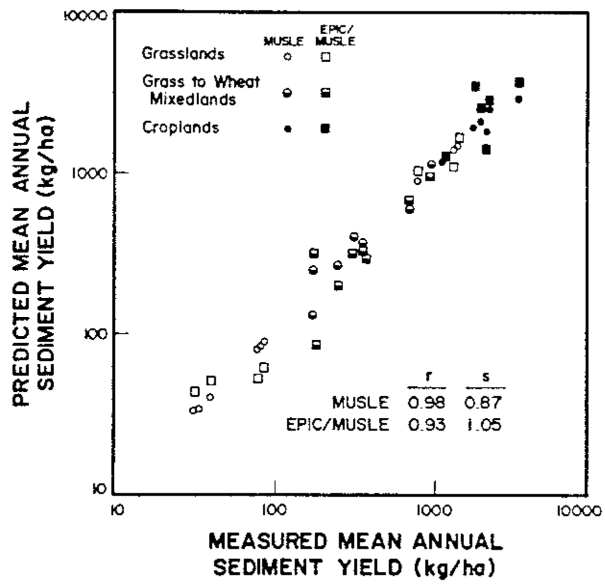

Fig. 1. Comparison of predicted and measured mean annual sediment yields for the watersheds.

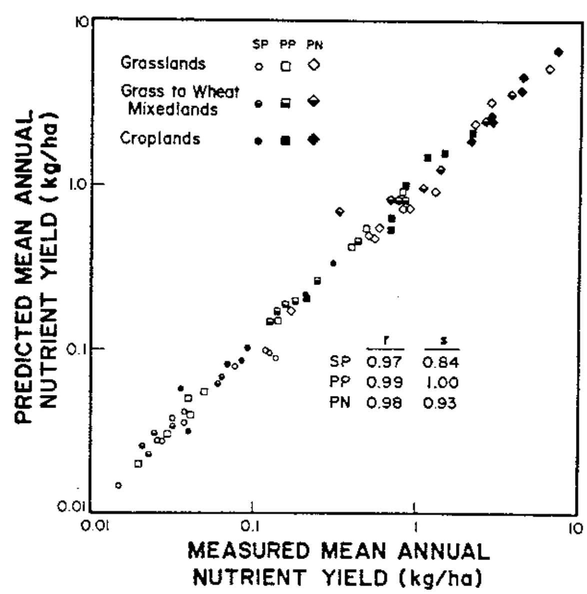

Fig. 2. Comparison of predicted and measured mean annual nutrient yields for the watersheds.

PERSPECTIVE

The results presented here indicate that the approaches used to predict sediment and nutrient yield gave realistic estimates on an event and annual basis. The fact that a computed runoff energy term performed fairly well for sediment yield predictions indicates that MUSLE may have application beyond just the calibrated watershed situation.

In the case of the nutrients, even more accurate predictions may be obtained, if desired, by making EDI and the slope/intercept values of the soil loss-ER relationship a function of soil and management characteristics. In addition, inclusion of soil nutrient cycling models simulating seasonal variation in surface soil nutrient content may improve transport prediction on an individual runoff event basis.

Because N is subject to atmospheric exchange and algal fixation, P is often the limiting nutrient regarding eutrophication. Current water quality models typically predict just the total amount of P associated with sediment. From an environmental standpoint, however, it is important to estimate what portion of the PP content is available for algal uptake in a water body. Therefore, future attention must also be directed to prediction of the bioavailable P content of sediment.

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EFFECT OF IMPOUNDMENTS ON NUTRIENT CONCENTRATIONS

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ABSTRACT

A three year study in the Little Washita River Basin (53,900 ha) in Oklahoma evaluated the effect of Soil Conservation Service (SCS) flood detention reservoirs on nutrient concentrations in surface water on an agricultural watershed. Concentrations of nitrate N, ammonium N, Kjeldahl N, soluble P and total P were determined in runoff from upland source areas, inflows to two reservoirs, impounded water, outflows from the reservoirs, and downstream from the reservoirs. Concentrations in runoff from selected upland source areas (0.55 to 4.15 ha) were similar to those in inflows to the reservoirs from watershed areas of 28 and 400 ha. The impounded water and outflows from the reservoirs had lower concentrations of all nutrient forms except ammonium N. Total N and total P concentrations in the outflows were about 20% and 5%, respectively, of those in the inflows. However, nutrient concentrations in water 7.5 km or 10 km downstream from the reservoirs were similar to those in upland source areas. Even though 51% of the Little Washita River Basin is controlled by SCS impoundments, there was no apparent reduction in nutrient concentrations in the stream.

INTRODUCTION

An important consideration for water quality management is how nutrient concentrations change in runoff moving through an agricultural watershed. Total nutrient concentrations depend largely on the amount of eroded soil carried in runoff (Menzel et al., 1978; Monke et al., 1981; Smith et al., 1983). Soluble nutrient concentrations depend more on watershed soil fertility and fertilizer management (Baker et al., 1979; McColl, 1978; Menzel, et al., 1978; Monke et al., 1981).

Changes in both total and soluble nutrient concentrations may occur as runoff interacts with soils, vegetation, and impounded water on the watershed. Runoff interaction with soil is usually limited to the surface one or two cm of soil (Ahuja et al., 1981; Ahuja and Lehman 1983). In some watersheds, tile drainage contributes appreciable quantities of nitrate nitrogen (Baker et al., 1979; Monke et al., 1981). Vegetation may act as a source of nutrients (Sharpley, 1981) or it may be used in buffer strips to retain nutrients in runoff from waste disposal areas (Bingham et al., 1980). In addition, marked effects on nutrient concentrations in agricultural watersheds may be caused by impoundments.

Nutrient concentrations have been measured in the inflows and outflows of many major lakes and reservoirs, but only in a few small agricultural impoundments. Data from European and North American lakes and reservoirs have been analyzed in the OECD Cooperative Program on Eutrophication (OECD, 1982). The geometric mean of total phosphorus (P) concentrations in 87 water bodies, excluding those with large internal P loadings, was about 1/3 of that in the corresponding inflows. Similarly, the geometric mean of total nitrogen (N) concentrations was about 1/2 of that in inflows. The reductions in nutrient concentrations generally were greater in lakes and reservoirs with longer water residence times. Higgins and Kim (1981) measured P concentrations in the inflows and outflows of major reservoirs in the Tennessee River Basin with water residence times generally shorter than those observed in the OECD program. In 9 tributary reservoirs with residence times ranging from 43 to 251 days, the outflow P concentrations averaged 49% of inflow concentrations. In 9 mainstream reservoirs with residence times ranging from 3 to 14 days, the corresponding average was 104%.

Schreiber et al. (1977) conducted a detailed one-year study of P inflows and outflows in an agricultural reservoir in central Missouri. The reservoir is representative of many upstream flood detention reservoirs designed by the USDA Soil Conservation Service (SCS). It has a total drainage area of 1460 ha and a normal surface area of 8.2 ha. Storm detention times during the year of study ranged from 1 to 32 days,with an average water residence time of 6 days (Rausch and Schreiber, 1977). The volume weighted average total P concentrations were 1.54 and 0.36 mg/L in the inflow and outflow, respectively. Thus, the outflow concentrations averaged only 23% of the inflow concentration. Soluble P constituted 10% of the total P in the outflow but only 4.5% of that in the inflow. Retention of P in the agricultural reservoir is related to the retention of suspended sediment. Schreiber et al. (1977) found that volume weighted average suspended sediment concentrations were 3663 mg/L in inflow, compared with 386 mg/L in outflow. Thus, in areas where runoff carries high suspended sediment loads, reservoirs may have a large effect on the amounts and forms of nutrients passing through them.

In this investigation, the effects of two SCS flood detention reservoirs on nutrient concentrations in agricultural watersheds in central Oklahoma were measured. The data show the concentrations and forms of N and P in water at several points in the watershed--runoff from upland source areas, inflows to the reservoirs, impounded water, outflows from the reservoirs, and downstream from the reservoirs.

METHOD OF STUDY

The study was done from 1980 through 1982 in the Little Washita River Basin in Oklahoma (Fig. 1). The area of the watershed above the lowest sampling point (Station 522 in Fig. 1) is 53,900 ha, which includes 9800 ha under cultivation, 35,500 ha in grassland, 3000 ha in timber, and 5600 ha in miscellaneous use. There are 6100 ha of alluvial soils in the watershed, mainly in the eastern half of the watershed, which will be referred to as the lower basin. The lower basin is predominantly grassland with cropland mainly on alluvial soils. The western half of the watershed, or upper basin, is about equally divided between grassland and cropland. Most of the timber occurs near the boundary between the lower and upper basins.

Since 1969, 37 reservoirs have been constructed above station 522 in an SCS upstream flood control program. Together with Lake Burtschi, these reservoirs control runoff from a total area of 27,500 ha (51% of the watershed). Two reservoirs, designated as site 11 and site 23, were selected as typifying the SCS reservoirs, with watersheds representing the lower and upper basins, respectively. The 535 ha watershed of site 11 is primarily grassland with less than 10% of the area cropped. Eufaula fine sand (thermic psammentic paleustalf) is the major soil type on the upper portion of the watershed with Minco silt loam (thermic udic haplustoll) dominating near the reservoir. The

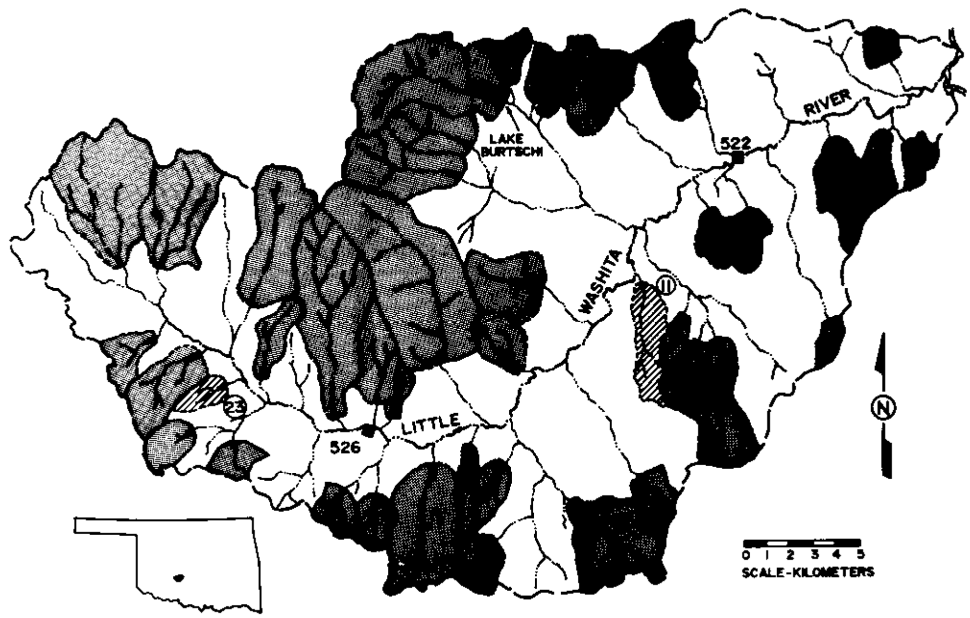

Figure 1. Map of Little Washita River Basin (see inset for location in Oklahoma) showing watershed boundaries of SCS upstream flood control reservoirs. The watersheds of the two reservoirs selected for this study are indicated by cross-hatching. Downstream sampling stations are located at 526 and 522.

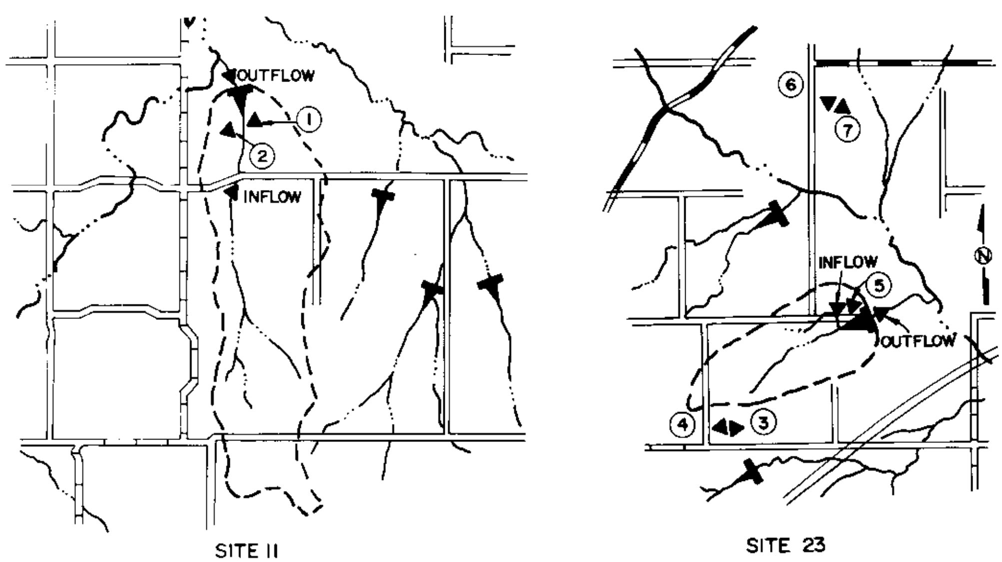

Figure 2. Watershed map of the two study reservoirs, showing locations of the unit source watersheds, inflow and outflow sampling points at each reservoir.

248 ha watershed of site 23 is approximately half grassland and half cropped to winter wheat (Triticum aestivum L.). Zaneis loam (thermic udic argiustoll) is the major soil type.

At the principal spillway level, the areas and depth of the reservoirs are:

	Site 11	Site 23
Surface area	4.5 ha	4.0 ha
Mean depth	2.2 m	2.1 m

The average water residence times for these reservoirs are approximately 100 days. However, during the period of observation, single storm flows had residence times as short as 23 days at site 11 and 5 days at site 23.

The upland source areas were relatively small watersheds that were representative of much of the contributing area for each reservoir (Table 1).

Table 1. Characteristics of upland source area watersheds at Sites 11 and 23 in the Little Washita River Basin.

Land Uses in Source Area	Soil Type and Classification	Slope %	Watershed Area ha
	Lower Basin Watersheds at Site 11		
1. Native grass, good condition (Primarily Andropogon scoparius Michx.)	Lucien-Nash complex (silt loam) thermic, shallow typic haplustoll thermic udic haplustoll	6.0	2.31
2. Native grass, overgrazed (Primarily Andropogon scoparius Michx.)	Lucien-Nash complex (silt loam)	6.0	1.16
	Upper Basin Watersheds at Site 23		
3. Idleland to improved pasture (Cynodon dactylon L.)	Zaneis loam thermic udic argiustoll	3.0	3.11
4. Wheat (Triticum aestivum L.) to improved pasture (Cynodon dactylon L.)	Zaneis loam	3.0	4.17
5. Bermuda grass (Cynodon dactylon L.)	Darnell-Noble complex (fine sandy loam) thermic, shallow udic ustochrept thermic udic ustochrept	7.0	1.42
6. Wheat, minimum tilled	Norge silt loam thermic udic paleustoll	3.5	0.55
7. Wheat, conventional tilled	Norge silt loam	3.5	0.60

Two grassland source areas at site 11 were located within the reservoir watershed (Fig. 2). Five source areas at site 23 represented different land uses and soil types of the reservoir watershed, but four of the five source areas were located outside the reservoir watershed to secure cooperation with landowners. The volumes of runoff water from the source area watersheds were measured with H-flumes equipped with FW-1 stage recorders. Automatic pumping samplers were used to collect 5 to 15 samples during each runoff event. These samples were composited on a flow-weighted basis to provide a single representative sample for the determination of nutrient concentrations from each runoff event and watershed.

Volumes of outflow from the reservoirs were measured from a stage-discharge relationship developed for each reservoir outlet. A V-notch weir was installed at the outlet from Site 11. The outlet from Site 23 was an SCS-designed horizontal spillway. Storm inflow volumes to both reservoirs were estimated from change in reservoir storage plus outflow minus precipitation. Evaporation and seepage were assumed negligible during storm inflow periods. Automatic pumping samplers were used to obtain samples of outflow and inflow for determining nutrient concentrations. However, as inflow rates were not known in detail, the inflow and outflow samples were composited on an equal volume rather than a flow-weighted basis for each event. The inflow stations sampled runoff from 400 ha and 28 ha at sites 11 and 23, respectively (75 and 11% of the respective watershed areas).

Surface water samples were taken in November, 1980 and at monthly intervals from July, 1981, to December, 1982, with few omissions, from both reservoirs for determining nutrient concentrations. The samples were taken from the top 30 cm in the 3 to 4 m deep central pool of each reservoir. Temperature profiles indicated that the reservoir water was well mixed except during June, July, and August. Downstream water samples were taken from the Little Washita River at site 522 approximately 10 km from reservoir site 11 and 34 km from reservoir site 23, and at site 526 approximately 7.5 km from reservoir site 23 (Figure 1). The samples were taken to include low flows in the winter and summer, and high flows after spring and fall rains. All water samples -- runoff, inflow, outflow, reservoir and stream water--were refrigerated at 0 to 4° C within a few hours of collection until they were analyzed, usually less than 6 weeks after collection.

Just prior to analysis, the samples were thoroughly shaken to resuspend all sediment. Aliquots of the unfiltered samples were removed for total Kjeldahl N(TKN) and total P analysis. Another aliquot of each sample was centrifuged and filtered through a 0.45- m Millipore filter. Nitrate- and NH_4^+-N, and soluble P were determined on the filtered samples.

Chemical analysis for TKN, NO_3^--N and NH_4^+-N were conducted using standard methods as described in the Federal Water Pollution Control Administration Manual (U.S. Dept. of Interior, 1971). Since a reducing agent was not added during digestion, TKN does not include nitrate. Both TKN and NH_4^+-N were determined using micro-Kjeldahl distillation apparatus and Nessler's reagent. Nitrate N was determined with brucine sulfanilic acid.

Soluble P and total P were determined on perchloric acid digests of filtered and unfiltered samples, respectively. All P determinations were made using the isobutanol extraction method described by Golterman and Clymo (1969). Color was developed by reducing the ammonium molybdate complex with ascorbic acid.

Suspended sediment concentrations were determined on all source area and downstream water samples but on only half of the inflow, outflow, or reservoir water samples. Sediment concentrations were determined by the evaporation method after flocculation with alum (USDA, 1979). Correction for dissolved solids was based on the conductivity of the samples.

Annual mean concentrations were calculated for the five sampling points (source areas, inflow, reservoir, outflow, and downstream) in the upper and lower basins for three years of sampling. Arithmetic means were used to give a common basis for comparison because the inflow, reservoir, and outflow samples were not flow-weighted. The logarithms of the annual arithmetic mean concentrations were then subjected to an analysis of variance to determine the major factors influencing concentrations of each nutrient form. The logarithmic transformation was appropriate because the variance increased in direct proportion to the nutrient concentrations (Box et al., 1978). The analysis of variance was run separately for each nutrient form. Differences between mean concentrations were tested for significance using Duncan's multiple range test (1955).

RESULTS AND DISCUSSION

With each nutrient form, most of the variance was related to basin and sampling point (Table 2). There was little effect of year of sampling on the nutrient concentrations. No analysis of variance was made on sediment concentrations because data for the inflow, reservoir, and outflow sampling points were incomplete.

Table 2. Results of the F-test for significance of factors affecting nutrient concentrations in Oklahoma watersheds. "F" values in parentheses relate to SxYxB interaction (8 degrees of freedom). Other "F" values relate to the sum of all interactions (22 degrees of freedom).

Source of Variance	Degrees of Freedom	"F" Values for NO_3-N	NH_4-N	Kjeldahl N	Soluble P	Total P
Sampling point (S)	4	9.40**	6.76**	16.32**	15.57**	37.86**
Year (Y)	2	0.04	0.95	0.65	0.15	2.53
Basin (B)	1	9.84**	18.05**	6.92*	15.36**	4.42*
SxY	8	(0.90)	(2.51)	(2.56)	(0.68)	(10.82)**
SxB	4	(2.30)	(2.20)	(9.63)**	(3.48)	(23.60)**
YxB	2	(0.13)	(0.57)	(0.37)	(0.41)	(1.22)
SxYxB	8	-	-	-	-	-

** Significant at the 0.01 level of probability

* Significant at the 0.05 level of probability

Arithmetic mean concentrations of sediment and nutrients at the sampling points associated with both basins are shown in Table 3. Standard deviations were approximately equal to the mean in most cases. Nevertheless, after logarithmic transformation, the Duncan's multiple range test showed that the reservoir and outflow concentrations of most nutrient forms were significantly lower than those in the source area runoff and inflow. The downstream concentrations were often as high as those from the source areas.

Table 3. Arithmetic means and standard deviations of sediment and nutrient concentrations observed at various points in the Little Washita River Basin.

Sampling Point	Number of Samples: Sediment	Number of Samples: Nutrient	Sediment Conc. mg/L	NO_3-N	NH_4-N	Kjeldahl N	Soluble P	Total P
				mg/L				
Lower Basin (Site 11 Reservoir)								
Source Areas	22	22	650 ± 850	0.20 ± 0.23 bc[1]	0.19 ± 0.11 bcd	1.94 ± 1.23 b	0.12 ± 0.10 abc	0.51 ± 0.44 c
Inflow	9	23	2330 ± 1560	.26 ± .19 abc	.11 ± .09 d	3.44 ± 4.37 ab	.08 ± .03 bcd	1.01 ± .71 abc
Reservoir	7	13	8 ± 5	.11 ± .23 cd	.17 ± .16 cd	0.77 ± 0.42 c	.03 ± .02 ef	0.06 ± .05 e
Outflow	4	13	21 ± 18	.05 ± .04 d	.13 ± .08 cd	.66 ± .14 c	.02 ± .01 f	.07 ± .09 de
Downstream (522)	34	34	8490 ± 8390	.73 ± .23 ab	.47 ± .63 a	7.96 ± 8.10 a	.07 ± .06 cd	2.20 ± 1.15 ab
Upper Basin (Site 23 Reservoir)								
Source Areas	98	98	2110 ± 3430	1.06 ± 1.73 a	.37 ± .27 a	5.89 ± 10.13 a	.15 ± .15 ab	1.12 ± 1.62 abc
Inflow	8	20	8120 ± 4320	0.67 ± 0.50 ab	.21 ± .18 bcd	7.05 ± 4.32 a	.19 ± .08 a	2.84 ± 3.32 a
Reservoirs	10	15	13 ± 13	.08 ± .09 cd	.33 ± .19 ab	1.48 ± 0.43 bc	.05 ± .03 de	0.10 ± 0.06 de
Outflow	8	20	62 ± 42	.24 ± .45 bc	.24 ± .21 abc	1.33 ± .48 bc	.07 ± .03 bcd	.15 ± .07 d
Downstream (526)	22	22	4760 ± 8670	.55 ± .67 ab	.42 ± .48 a	3.03 ± 1.95 ab	.06 ± .05 cd	.78 ± .76 bc

1 Values for the same nutrient form followed by the same letter are not significantly different from each other at the 0.05 level of probability.

Table 4. Regression equations relating particulate N and P to sediment concentrations in the Little Washita River Basin.

Source of Samples	No. of Samples	Regression Equation[1]	Coefficient of Correlation (r)
Site 522	34	ln N = -3.05 + 0.55 ln Sed	0.77
Site 526	22	ln N = -2.41 + 0.49 ln Sed	0.82
All samples	221	ln N = -1.02 + 0.31 ln Sed	0.74
Site 522	34	ln P = -7.27 + 0.89 ln Sed	0.86
Site 526	22	ln P = -7.28 + 0.83 ln Sed	0.88
All Samples	222	ln P = -4.94 + 0.62 ln Sed	0.86

1 N or P signify particulate N or P concentrations in mg/L. Sed signifies sediment concentrations in mg/L.

Table 5. Average annual runoff at various points in the Little Washita River Basin.

Sampling Point	Annual Runoff: Volume m^3	Depth mm	Volume m^3	Depth mm
	Lower Basin		Upper Basin	
Source Areas	3.0×10^2	10	1.2×10^4	103
Outflow from Reservoirs	4.0×10^5	74 1/	1.9×10^5	78 2/
Downstream	3.2×10^7	59	1.5×10^7	91

1/ Data for 1981 and 1982 only.

2/ Major storm outflows only, probably 65 to 75% of the annual outflows based on source area runoff.

Since most of the N and P at both downstream sampling locations was in particulate form, the correlations between sediment concentrations and particulate N and P were investigated. Particulate N was defined as Kjeldahl N minus ammonium N, particulate P as total P minus soluble P. At both downstream sampling points, particulate N and P were strongly correlated with sediment concentrations (Table 4). A similar strong relationship was found between sediment and particulate nutrient concentrations over all sampling points in the watershed. All regression coefficients (slopes) are less than one, indicating that low concentrations of sediment are enriched in N and P. The regression coefficients also show that sediment concentrations have a greater effect on particulate N and P concentrations at sampling points further downstream. This suggests that an important source of N and P in the downstream samples is resuspended sediment from streambed and bank materials.

In most cases there was no significant difference between the source area and inflow concentrations of any nutrient form. Although the watershed areas above the inflow sampling points were 400 ha at site 11 and 28 ha at site 23, the composition of inflow was similar to that of runoff from the much smaller source areas. There was, however, a significantly lower ammonium concentration in the inflow than in the runoff from source areas at site 23.

There was no significant difference between the reservoir and outflow concentrations of any nutrient form. The concentrations of total N and P were reduced greatly in the reservoirs, due largely to the deposition of sediment. Sediment concentrations in reservoir and outflow samples were less than 1% of the inflow concentrations (Table 3). Total N (Kjeldahl + nitrate) concentrations in the outflows were about 20% of those in the inflows. Total P concentrations were reduced to about 5% of the inflow concentrations. Soluble P and nitrate concentrations were also reduced significantly in the reservoirs. Only the ammonium concentrations were not reduced.

The reductions in total P concentrations in the Oklahoma reservoirs were greater than that in the Missouri reservoir (Schreiber et al., 1977). This is probably due mainly to the longer water residence time in the Oklahoma reservoirs. Our results agreed with those of Schreiber et al. (1977) in that there was a large increase in the proportion of soluble P in the reservoir outflows compared with the inflows. In our study less than 10% of the total P in the inflows was soluble, but up to 50% of that in the outflows was soluble.

There was a similar, but less pronounced increase in the proportion of inorganic N in the reservoir outflows. About 10% of the total N in the inflows was inorganic, but up to 30% of that in the outflows was inorganic.

Except at the downstream sampling points, the concentrations of nutrients were nearly always higher in the upper basin than they were at corresponding sampling points in the lower basin. However, the differences were statistically significant in fewer than half of the comparisons. The source areas in the upper basin include cropped watersheds, which generally provide higher concentrations of nutrients to runoff than do grassland watersheds (Menzel et al., 1978; Smith et al, 1983).

The downstream nutrient concentrations were generally higher than those in outflows from the reservoirs. However, the concentrations of soluble nutrients were not significantly higher ($p < .05$) at either downstream sampling point than they were at the outflow from reservoir site 23. Total P concentrations were significantly higher at both downstream sampling points, and Kjeldahl N was significantly higher at site 522, 34 km downstream, than at the site 23 outflow. The concentrations of all forms of nutrients at site 522 were significantly higher than those in the outflow at site 11. Increased nutrient concentrations at downstream sampling points are attributable to runoff from the uncontrolled watershed area, which includes all of the alluvial cropland, in addition to entrainment of streambed and bank materials.

The effect of the reservoirs on nutrient loads can be approximated from runoff volumes (Table 5) and mean concentrations (Table 3). These indicate that 280 tons of N and 70 tons of P flow past site 522 in an average year. Similarly, the two study reservoirs trap about 3 tons of N and 1 ton of P. The average depth of runoff (Table 5) indicates that the study reservoirs contributed to downstream flow nearly in proportion to their watershed areas. If we assume that other reservoirs in the Little Washita River Basin trap N and P in proportion to their watershed areas, the total retained would be 110 tons of N and 40 tons of P. According to this calculation, the amounts retained are smaller than the flows of N and P out of the basin. Thus, any changes in nutrient concentrations may be difficult to detect.

Even though the upstream flood detention reservoirs retain a large fraction of the sediment and nutrients that enter them, downstream concentrations of these materials in the Little Washita River were similar to those in upland source areas. The reduced concentrations in reservoir outflows were offset by sources of sediment and nutrients in the uncontrolled watershed area. These sources were not specifically identified. The main conclusion of this study is that an intensive upstream flood control program, with half of the watershed controlled by impoundments, had little apparent effect on downstream nutrient concentrations.

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POLYCHLORINATED BIPHENYLS IN THE HOUSATONIC RIVER

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ABSTRACT

Accumulations of PCB's (Polychlorinated biphenyls) in the Housatonic River in Massachusetts and Connecticut are predominantly located in impounded reaches of the river where sedimentation is greatest. The highest PCB concentrations are associated with fine-grained bottom sediments. An estimated total of 10,000 kilograms of PCB's are contained in the bottom sediments, 60 percent of which are in Massachusetts, predominantly in a stretch of the river called Woods Pond; and 40 percent are in Connecticut, primarily in Lakes Lillinonah and Zoar. PCB's are transported primarily in association with suspended sediment. At Great Barrington, Massachusetts, approximately 29 river kilometers below Woods Pond and 47 river kilometers below the source of the PCB contamination in Pittsfield, Massachusetts, a suspended-sediment discharge of 6,400 megagrams per year supported an estimated PCB transport of 220 kilograms per year.

In a sand and gravel aquifer adjacent to Woods Pond, the distribution of hydraulic heads and dissolved-oxygen, ammonia, and nitrate concentrations show that industrial water-supply wells have been inducing ground-water recharge through the PCB-contaminated bottom sediments of the pond over the last 25 years. At a distance of 1.5 meters from the pond's shoreline, the upper 12 meters of the 32-meter-thick aquifer contains water derived from induced infiltration. No PCB's were found in this vertical section at the detection limits of 0.0001 milligram per liter of water and 0.001 milligram per kilogram of sediment. It is concluded that induced recharge has not moved PCB's from the bottom sediments into the aquifer 1.5 meters downgradient from the edge of Woods Pond.

INTRODUCTION

The Housatonic River, located in western Massachusetts and Connecticut (fig. 1), is a valuable resource to the region. Known for its scenic beauty and excellent sport fishing, the river is used extensively as a source of hydroelectric power, water supply, and recreation. For many years, it has also been used for the disposal of industrial and domestic wastewater. From its origin near Pittsfield, Massachusetts, to its mouth on Long Island Sound near Stratford, Connecticut, the Housatonic River drains an area of 5,040 km^2. This drainage encompasses 3,190 km^2 in Connecticut, 562 km^2 in New York, and 1,300 km^2 in Massachusetts. The river is impounded by 18 dams, 5 of which are in Connecticut and 13 in Massachusetts. The average flow of the river is 73.8 m^3/s at Stevenson, Connecticut, about 31 river kilometers above the mouth. Streamflow is highly regulated in Connecticut by several hydroelectric plants, but less so in Massachusetts.

In the Housatonic River basin, bedrock is covered by a thin mantle of unconsolidated glacial deposits. These deposits consist predominantly of till in the upland areas and stratified drift in the valleys. Where the saturated thickness of the stratified drift exceeds about 8 meters, the drift forms an aquifer capable of sustaining municipal or industrial water supplies. Pumped wells adjacent to a stream or pond will intercept ground-water discharging to these water bodies and may also induce streamwater to infiltrate through the streambed and enter the aquifer.

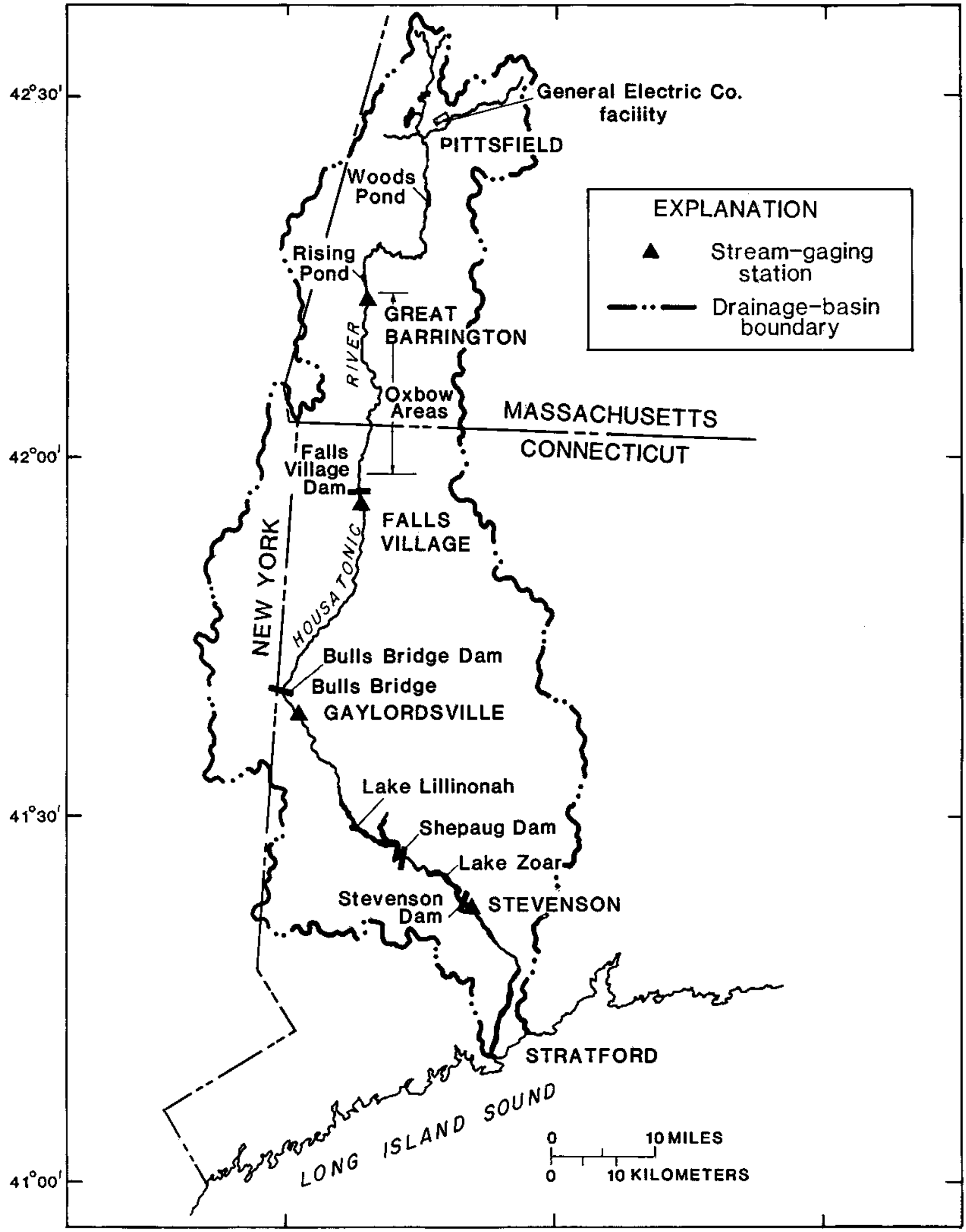

Figure 1.--The Housatonic River drainage basin.

PCB's (polychlorinated biphenyls) were detected in surficial bottom sediments in the Housatonic River in Connecticut by the Survey (U.S. Geological Survey) in 1974 and 1975 as part of a water-quality monitoring program being carried out in cooperation with the DEP (Connecticut Department of Environmental Protection). Further investigation of the Housatonic River by the Survey and the DEP in 1976-77 confirmed the presence of PCB's in the bottom sediments at concentrations up to approximately 2 mg/kg on a dry-weight basis. Samples of fish collected in 1977 from Connecticut reaches of the river were analyzed by the DOHS (Connecticut Department of Health Services) and found to contain concentrations of PCB's in excess of the U.S. Food and Drug Administration's maximum recommended level for consumption of 5 mg/kg in effect at that time. Data collected in Massachusetts during this same period indicated that fish and sediments contained even higher levels of PCB's than those that were found in Connecticut (Massachusetts Department of Environmental Quality Engineering data files).

PCB's are a group of synthetic organic compounds that are extremely stable, have high boiling points, are relatively nonflammable and nonconductive, and have low solubilities in water. Because of these properties, PCB's were used in a wide variety of industrial applications, but particularly in electrical equipment, where they served as dielectric and heat transfer fluids for over 40 years. In the 1970's, PCB's were found to have serious, adverse effects on health of humans and other animals. Also, PCB's were found to have a strong affinity for soils, sediments, and particulate matter present in the environment (Haque and others, 1974) and to bioconcentrate. As a result of their potential toxicity and environmental persistence, the manufacture and use of PCB's in the United States, other than for certain totally enclosed electrical applications, has been banned since 1977 (PL 94-469).

Growing concern over the severity and possible related health and environmental consequences of PCB contamination prompted several actions to be taken. In Connecticut, DEP and DOHS jointly issued a health advisory in 1977 that advised against eating fish taken from the river. In Massachusetts, concern over the potential contamination of municipal ground-water supplies adjacent to the river prompted the Massachusetts Department of Environmental Quality Engineering to prohibit the development of new municipal water-supply wells along the river until it could be demonstrated that PCB's in the river system were not a threat to ground-water quality.

A CASE (Connecticut Academy of Science and Engineering) report of 1978 stated that the primary source of PCB contamination to the Housatonic River was leakage from a General Electric Company facility located in Pittsfield, Massachusetts. This facility had used PCB's in the manufacture of electrical transformers from the early 1930's through March 1977, when the use of PCB's was discontinued. CASE also recommended that further studies be made to determine the extent and significance of the PCB contamination.

As a result of the CASE recommendation and the concerns of the States of Connecticut and Massachusetts, the U.S. Geological Survey, at the request of these States, entered into cooperatively funded agreements with the Massachusetts Division of Water Pollution Control and the Connecticut Department of Environmental Protection to determine the (1) distribution and mass of PCB's in the Housatonic River, (2) methods and rate of transport of PCB's downstream, and (3) effects of PCB's in stream-bottom materials on an adjacent aquifer under conditions of induced infiltration. The purpose of this paper is to describe the major findings of these investigations.

DISTRIBUTION AND MASS OF PCB'S IN THE HOUSATONIC RIVER

Analysis of the bottom-material samples collected during 1974-77 showed that the Housatonic River above the Stevenson Dam at Stevenson, Connecticut, was significantly contaminated by PCB's. Subsequent bottom-material sampling was concentrated in those areas of the river where large quantities of fine-grained sediment are deposited, such as at impoundments and low-velocity reaches. Limited sampling also was done in free-flowing reaches, where the streamflow velocity maintains fine-grained sediments in suspension, to test the assumption that areas which are relatively devoid of fine-grained bottom sediments contain few PCB's.

In areas where sediments were relatively coarse grained or contained thin layers of fine-grained material, a Ponar[1] dredge was used to collect the samples. In areas where fine-grained materials were thick, core samples were collected using piston or gravity corers. Core samples were divided into approximately 15-centimeter segments for analysis, in order to determine the concentration of PCB's with depth in the bottom material.

The mass of PCB's in the bottom sediments was calculated by integrating the PCB concentrations in the samples with the volume of bottom sediments represented by those samples. The volume of bottom sediments was calculated using sediment depths obtained from coring and seismic-reflection profiling, and the area of the river bottom. Because the river bottom and the PCB concentrations are not uniform throughout the study area, the mathematical methods and assumptions used to calculate PCB mass varied according to the characteristics of each portion of the river. These methods and calculations are described in detail by Frink and others (1982).

PCB's were found to have accumulated in the river bottom predominantly in areas where relatively fine-grained sediments are deposited. The concentration of PCB's in the bottom sediments generally decreases with distance downstream from Pittsfield, Massachusetts, as shown in figure 2. This decrease is due to PCB's settling to the streambed with the sediments to which they are sorbed and to the dilution of these PCB-contaminated sediments when they are in suspension with uncontaminated sediments entering the river downstream of Pittsfield. Samples of bottom sediments collected from sites upstream of Pittsfield had low concentrations of PCB's that average 0.04 mg/kg, which is considered to be the background concentration. Bottom sediments from sites located in free-flowing reaches of the river between Woods Pond, Massachusetts, and Bulls Bridge, Connecticut, were composed of relatively coarse-grained materials with an average PCB concentration of 0.32 mg/kg. In contrast, bottom materials from impounded and low-velocity reaches of the river in this same area were composed of relatively fine-grained materials with an average PCB concentration of 4.6 mg/kg. This indicates that most of the PCB's remain in suspension with fine-grained sediments until river velocities slow sufficiently to allow these sediments to settle out of suspension.

The total mass of PCB's in the study area is calculated to be approximately 10,000 kg, 60 percent of which is located in Massachusetts and 40 percent in Connecticut. A breakdown of the distribution of the mass by major sediment-deposition areas in the river is shown in figure 3, and their locations are shown in figure 1.

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[1]Use of the trade name in this report is for identification purposes only and does not constitute endorsement by the U.S. Geological Survey.

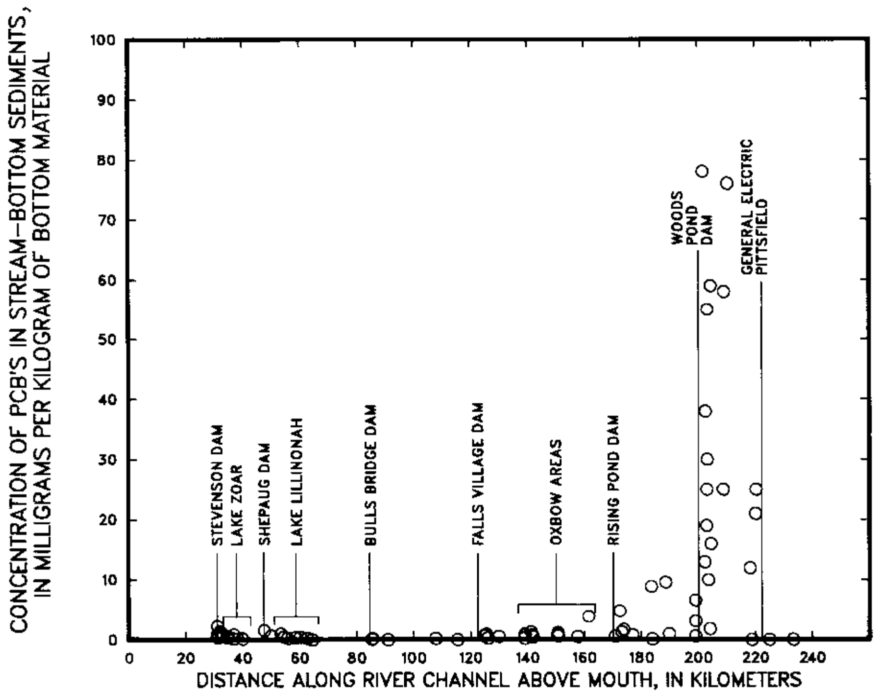

Figure 2.--Concentration of PCB's in the upper 15 centimeters of bottom sediments in the Housatonic River.

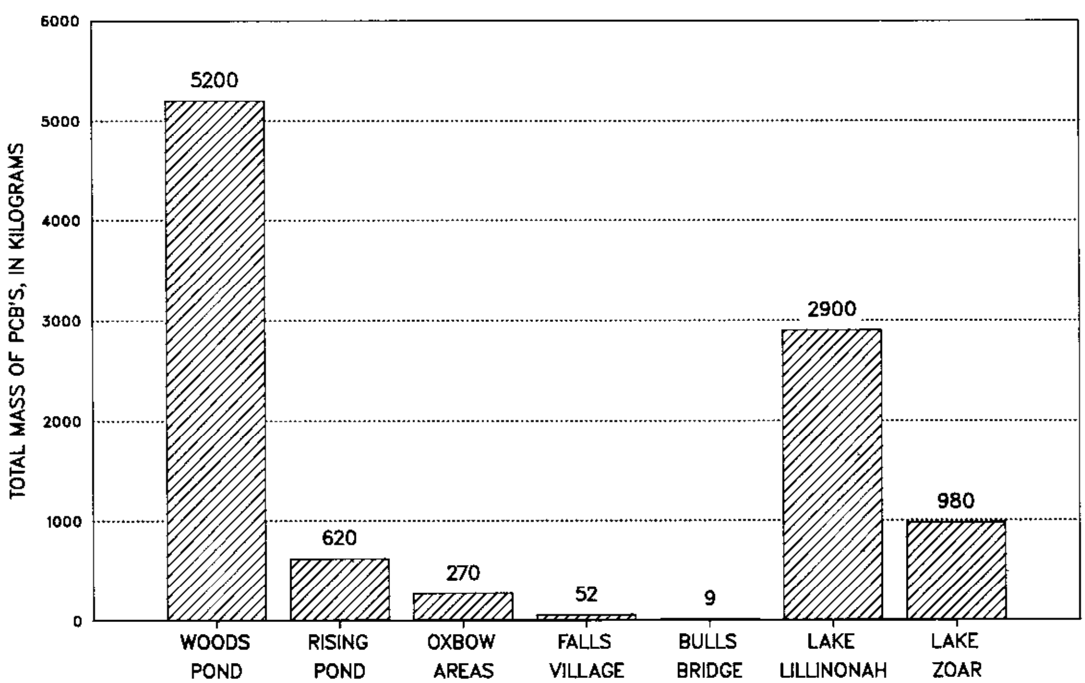

Figure 3.--Quantities of PCB's in the major sediment-deposition areas .

Although it was outside the scope of this study to analyze PCB concentration of individual sediment particle-size fractions or to separate the organic and inorganic components of the sediments for PCB analyses, linear-regression analyses of PCB concentration in relation to particle-size distribution and total organic carbon concentration of the bottom-sediment samples from several areas (Woods Pond and Lakes Lillinonah and Zoar) were performed. These regressions were calculated to determine if any relationships existed between PCB concentration and these sediment characteristics. In general, negative correlations were found between PCB concentration and the percentage of sand-sized particles (0.062 to 2 mm), and positive correlations were found between PCB's and the percentage of finer-sized particles (less than 0.062 mm) and with total organic carbon concentration; these correlations confirmed that PCB's are associated with the finer-grained sediments and organic particles in the Housatonic River. Correlations were strongest in the downstream impoundments (Lakes Lillinonah and Zoar), probably because the majority of the PCB's were transported to these areas in association with the finer-sized particles, which did not settle out of suspension upstream.

Woods Pond Area of the Housatonic River

The Woods Pond area is the first impoundment and major sediment-deposition site downstream of Pittsfield and contains more than 50 percent of the total PCB's (5,200 kg) in the river. This area is composed of a 49-hectare pond and a wetland that extends several kilometers upstream and contains many small coves and seasonal ponds. Most bottom-sediment deposits in this area are 1 to 2 meters thick, and 76 percent of the sediment is finer than 0.062 mm (silt and clay). The mean total organic carbon content of the bottom sediments is 130 g/kg. Because the Woods Pond area is the most significant sink for PCB's, it is described in detail here. Detailed descriptions of other areas are given in Frink and others (1982).

PCB concentrations in the surficial bottom sediments ranged from less than 0.001 to 110 mg/kg, with a mean of 21 mg/kg, and varied considerably from place to place and with depth (fig. 4). This spatial variation is the result of sedimentation characteristics in the area, and the depth variation represents the chronological order of PCB contamination of the river. Older, less-contaminated sediments were covered with younger, more-contaminated sediments as shown by the core samples where PCB concentration increases with decreasing depth in the bottom sediments. These concentration profiles are interpreted to represent the gradual increase in the load of PCB's to the river from their initial release in the early 1930's to the mid-1970's when the release was sharply reduced and finally stopped in 1977. At a few sites (core samples A, B, and C in fig. 4), the most recent sediments (0 to 15 centimeters in depth) contain less PCB's than sediments deposited earlier (15 to 30 centimeters in depth). At these sites, the lower PCB concentrations found in the top layer indicate that PCB-rich sediments are slowly being covered by more recent, less contaminated sediments. From these data, it seems that a thin layer of lower PCB-contaminated sediments is being deposited at the sediment-water interface. This layer was not detected at other sites because the sampling interval (15 centimeters) was too long at all but a few locations where the sedimentation rate must have been higher. This sedimentary burial in Woods Pond may limit the availability of the more PCB-contaminated sediments for erosion and transport downstream. Similar trends in PCB concentration with depth in bottom sediments were seen at downstream sites.

METHODS AND RATE OF PCB TRANSPORT IN THE HOUSATONIC RIVER

PCB transport was measured at the Survey's continuous record stream-gaging stations located at Great Barrington, Massachusetts, and Falls Village and Gaylordsville, Connecticut (fig. 1). Representative samples of the river water were collected at these sites using a US D-74TM sampler. The samples were analyzed for total and dissolved PCB concentrations. The arithmetic difference between the total and dissolved concentrations

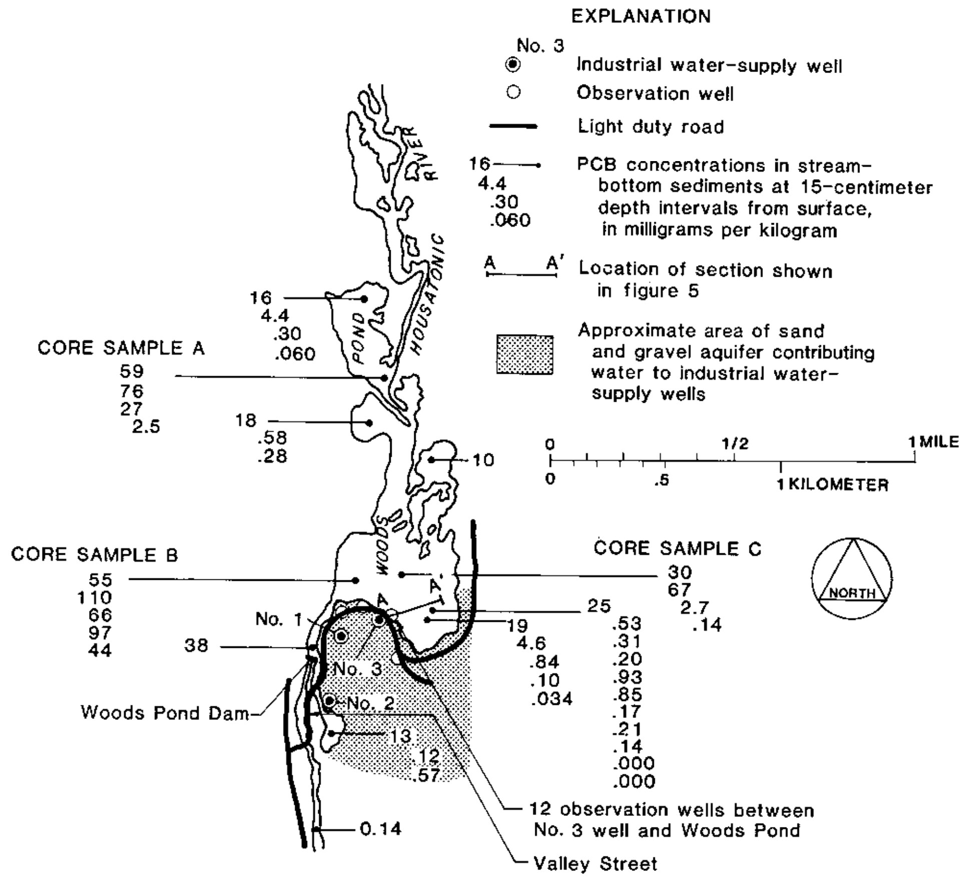

Figure 4.--Concentration of PCB's sorbed to bottom sediments in Woods Pond, and location of well field and sand and gravel aquifer.

is assumed to represent concentrations of PCB's that are attached to the suspended sediment. Samples for the determination of suspended-sediment concentration were collected at each site on a daily basis. The sediment concentration values from these samples and the streamflow data from the gaging stations were used to develop daily sediment concentrations and loads.

The data for PCB transport in association with suspended sediment were developed by dividing the PCB concentration in the sample by the suspended-sediment concentration, which yielded a PCB concentration per unit mass of suspended sediment. This concentration was then multiplied by the sediment load to obtain the quantity of PCB being transported.

PCB's were found to be transported predominantly in association with suspended sediments. In some instances, dissolved PCB's were detected at very low concentrations (0.0001 mg/L), but transport in the dissolved phase was judged to be insignificant in comparison to what is transported in the suspended phase.

Estimated transport rates of suspended sediment and PCB's for each of the three sites are: Great Barrington—6,400 Mg/yr of sediment and 220 kg/yr of PCB's; Falls Village, Connecticut—32,000 Mg/yr of sediment and 190 kg/yr of PCB's; and Gaylordsville, Connecticut—49,000 Mg/yr of sediment and 120 kg/yr of PCB's. Because the data used to calculate these rates were collected over a relatively short period of time (18 months), during which streamflows were generally below normal, their accuracy is uncertain.

However, a comparison of the PCB-transport rate estimated for Gaylordsville (120 kg/yr) with the total mass of PCB's calculated in the impoundments downstream of Gaylordsville (Lakes Lillinonah = 2,900 kg and Zoar = 980 kg) indicates that PCB's would have taken at least 32 years to accumulate in these impoundments; this suggests that the estimated transport rate is consistent with the historical release of PCB's to the river.

The concentration of PCB's sorbed to suspended sediment was found to decrease rapidly with increasing streamflow at the Great Barrington and Falls Village sites. This is probably caused by dilution of the PCB-contaminated suspended sediments with uncontaminated suspended sediments that are entering the river during periods of increased turbulence and velocity at higher streamflows. This relationship was not found at the Gaylordsville site, possibly because streamflow there is highly regulated by the Bulls Bridge hydroelectric plant located a short distance upstream.

EFFECTS OF PCB's IN WOODS POND ON INDUCED STREAM-WATER INFILTRATION

A stretch of the Housatonic River called Woods Pond was selected as the test site, because the pond's black, organic bottom sediments were found to contain some of the highest concentrations of PCB's along the river (up to 110 mg/kg). A sand and gravel aquifer adjacent to the pond contains a well field that pumps about 88 L/s from three large-capacity industrial water-supply wells (fig. 4) that have operated almost continuously since late 1965. Analysis of water-level data collected from a well-performance test conducted on No. 3 production well in 1965 (Geraghty and Miller, 1965) and from an aquifer test conducted by the Survey in December 1981 indicates that, even before the No. 3 well was added in 1965, wells No. 1 and 2 induced infiltration of river water into the aquifer along the southern shore of Woods Pond. Production wells (No. 1 and 2) have been pumping water since late 1956. Thus, river water has been induced to flow through PCB-laden bottom sediments in Woods Pond and into the aquifer for about 25 years at the time the aquifer materials and ground water were sampled for PCB content.

The saturated thickness of the aquifer is about 32 meters in the vicinity of the well field; the aquifer is 1.40 km^2 in area, of which about 0.50 km^2 contributes water to the industrial-supply wells. Quantities of water derived from different sources of water to the three wells were estimated from a water-balance analysis (Gay and Frimpter, 1984). This balance shows that about half the water pumped by the three wells, or 44 L/s, is derived from induced infiltration from Woods Pond.

To assess the potential PCB migration into the aquifer from Woods Pond, 14 observation wells were installed along the southern edge of the pond and opposite No. 3 production well (fig. 4) for water-quality sampling and hydraulic-head measurements; three wells were already present at this site. The group of 10 wells 1.5 meters from the edge of the pond and opposite No. 3 production well were screened (0.76-meter lengths) at incremental depths from 3 to 19.5 meters below land surface to test for PCB migration in the aquifer (fig. 5). Prior to placement of the well screens at selected depths in the aquifer, a split-spoon sample of aquifer material was collected for PCB analysis.

Ground-water discharge in this area would have been to the pond; however, continuous pumping of all three industrial water-supply wells has reversed the direction of ground-water flow by lowering the water table to below the level of Woods Pond. Water from the pond now infiltrates the bottom sediments and enters the aquifer. Water-level data from the wells located about 1.5 meters from the edge of Woods Pond indicate that a 1.2-meter decrease in hydraulic head occurs near the edge of the pond (fig. 5). Farther from the pond's edge, the water table slopes gradually toward the production well and steepens next to the well (fig. 5). Ground-water flow within the upper part of the aquifer is essentially horizontal and toward the production well. Hydraulic head also decreases with depth which indicates that there is downward movement of ground water toward the production well screen.

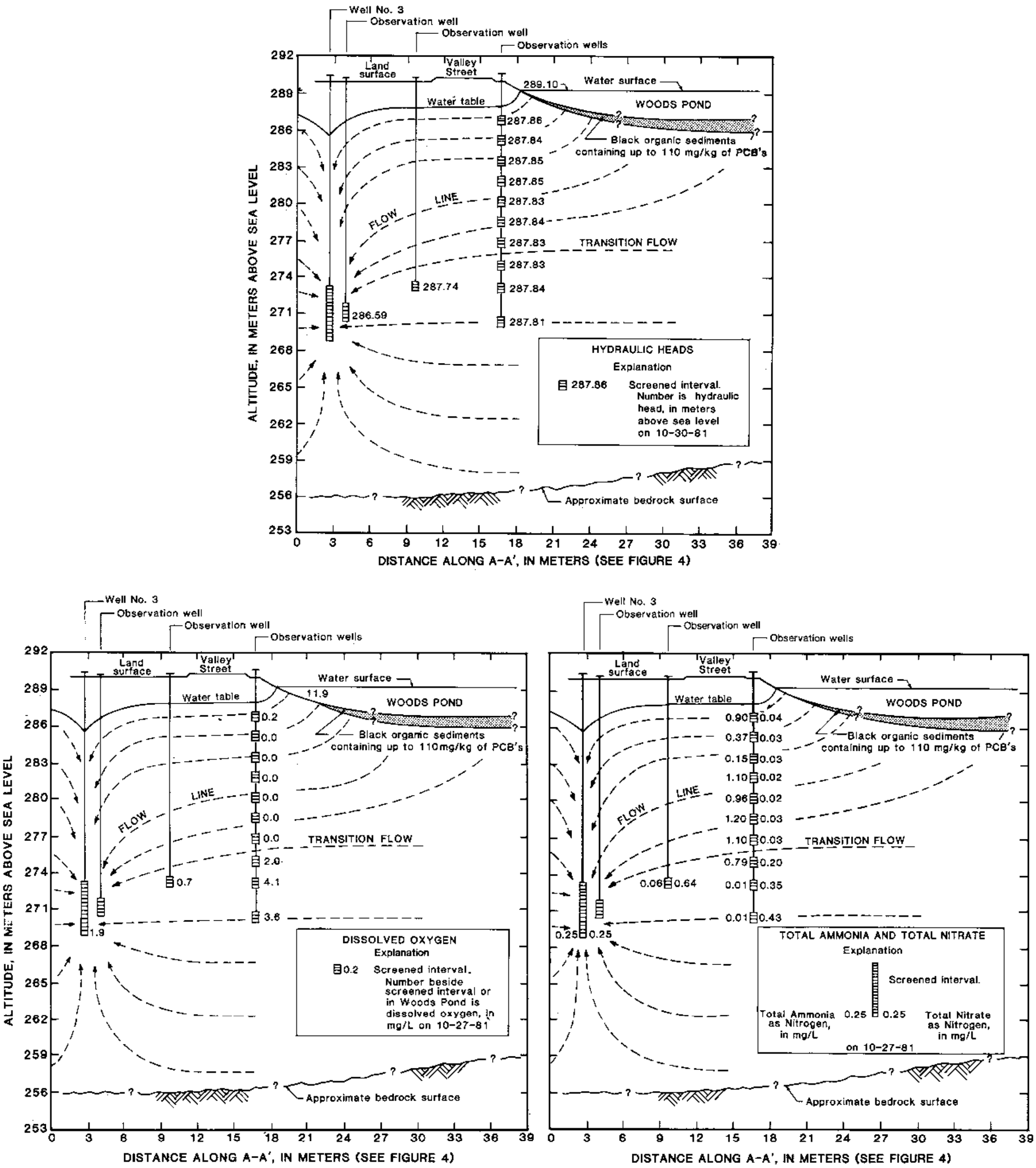

Figure 5.--Location of well screens and distribution of hydraulic heads and dissolved oxygen, total ammonia, and total nitrate concentrations in the aquifer adjacent to Woods Pond.

Physical and chemical data, including variations in water temperature, and concentrations of dissolved oxygen, ammonia, iron, and manganese show that water from Woods Pond is moving into the aquifer and discharging at the production well (Gay and Frimpter, 1984). For example, the absence of dissolved oxygen and the elevated ammonia concentrations (as much as 1.2 mg/L) in water pumped from the upper 12 meters of the aquifer near the edge of Woods Pond indicate that this upper zone of the aquifer contains water recharged through the bottom of Woods Pond. As oxygenated water in Woods Pond passes through the organic bearing bottom sediments, oxygen in the water is consumed as the organic matter decomposes. During this decomposition, ammonia is formed, which can be converted to nitrate by reacting with dissolved oxygen. Dissolved oxygen and nitrate found in the lower part of the aquifer (below the transition flow line in fig. 5) indicate that some of this ground water is a combination of direct recharge from precipitation, and recharge of overland runoff and leakage of ground water from the till and bedrock upland—probably from the east side of Woods Pond. This overturned hydrochemical profile is confirming evidence for recharge through the PCB-laden organic bottom sediments.

Chemical analyses of the aquifer material from selected intervals throughout a vertical section of the aquifer (3, 6.4, 9.8, 13.4, 16.8, and 19.5 meters below land surface) 1.5 meters from the edge of Woods Pond (fig. 5) show that PCB concentrations are less than the analytical method's detection limit of 0.001 mg/kg (Wershaw and others, 1983). If PCB's had moved through these aquifer materials in the past, they would be expected to have sorbed to the fine-grained fraction because of PCB's strong affinity for fine-grained sediment; however, none were detected.

PCB's were not detected in ground-water samples. Chemical analyses of water from 11 observation wells located between the pond and production well No. 3, and from the production well, did not detect PCB's in either the dissolved or suspended phase, at the detection limit of 0.0001 mg/L (Wershaw and others, 1983). The absence of PCB's from aquifer-material and ground-water samples indicates that PCB's have not reached a vertical section of the aquifer 1.5 meters from the edge of Woods Pond after 25 years of induced infiltration from the Housatonic River, nor do PCB's apparently pose a threat to public drinking-water supplies located in similar hydrogeochemical environments.

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SEDIMENT/FISH MODELING IN THE SOUTH FORK SALMON RIVER

by Philip N. Jahn, Soil Scientist, Payette National Forest and David C. Burns, Fisheries Biologist, Payette National Forest, McCall, Idaho.

ABSTRACT

Idaho's South Fork Salmon River has been controversial since the mid-1960's when a combination of factors including intensive logging, mining, roading activitites, wildfire, and an unusually large spring runoff severely reduced the spawning and rearing habitat of anadromous fish and other salmonids utilizing the drainage.

Following the events of the mid-1960's, major efforts were undertaken by several state and federal agencies, including the USDA-Forest Service, to mitigate the disturbances causing sedimentation and to monitor the recovery of the river system. The recent development of sediment production and fish response models by Regions 1 and 4 of the Forest Service affords the opportunity to quantitatively evaluate the history of the drainage comparing sediment production potentials to monitored changes in fish habitat.

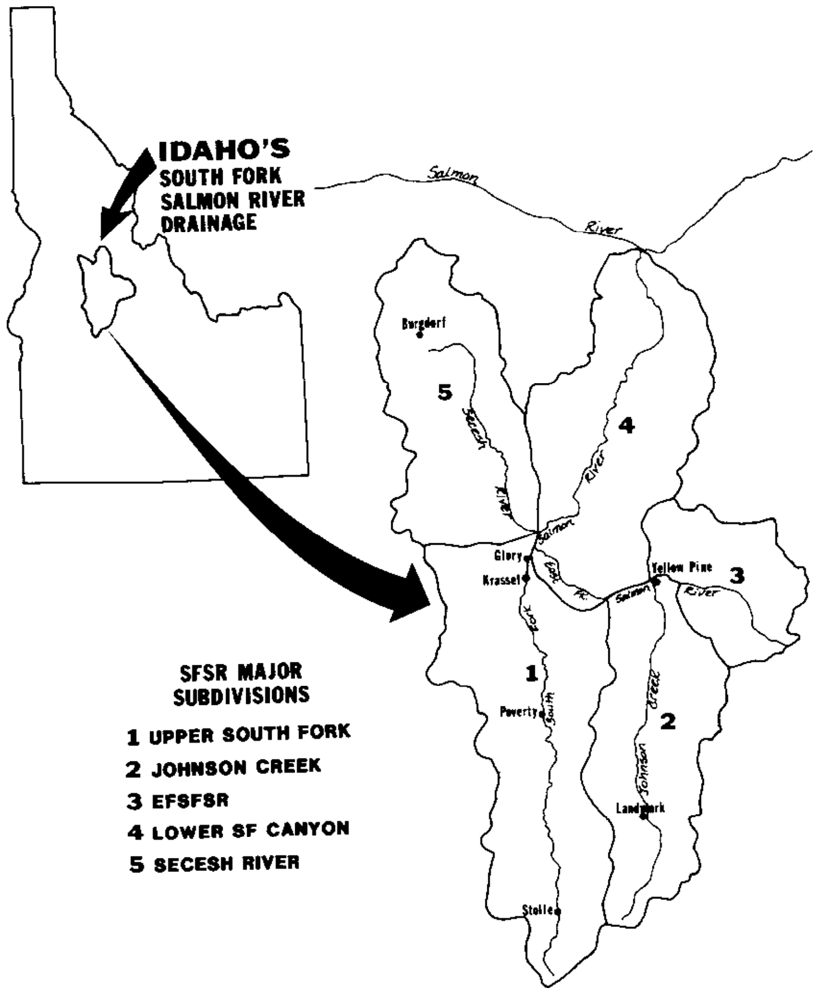
Today, accelerated sediment potentials are well below the levels of the 1950's and 60's and have stabilized. Fish habitat has recovered significantly from an estimated low of 20 percent of natural potential in 1966. Understanding the relationships between forty years of past management and concurrent changes in fish habitat within the drainage helps planners evaluate probable impacts and monitoring needs associated with future long-term management alternatives.

INTRODUCTION

The South Fork of the Salmon River is situated in central Idaho (Figure 1). The basin is approximately 800,000 acres in size and is administered almost entirely by the Boise and Payette National Forests. The drainage contains a wealth of diverse natural resources including minerals, timber, recreation, wildlife, and fish. The fisheries involve resident populations of trout and char as well as anadromous populations of salmon and steelhead.

Nearly all of the South Fork drainage is situated within the Idaho Batholith, an area comprised geologically of intrusive igneous parent rock materials. These geologic materials typically produce coarse textured soils which have little inherent resistance to erosion immediately following disturbance)USDA Forest Service, Soil-Hydrologic Reconnaissance for the Krassel Ranger District, 1970).

Figure 1



Chinook salmon and steelhead spawn and rear throughout the drainage with highest concentrations of spawning fish occurring in the Upper South Fork (Area 1, Figure 1). Major tributaries including Johnson Creek, East Fork South Fork Salmon River(EFSFSR), and the Secesh River (Areas 2, 3, and 5, respectively on Figure 1), also contain suitable habitat.

In the 1950's, the South Fork basin produced about 40,000 adult Chinook to downstream commercial and Indian treaty fisheries, with about 10,000 returning to Idaho. By the late 1960's, Chinook salmon and steelhead numbers were declining rapidly due to commercial over-harvest, passage problems associated with Columbia River dams, and loss of quality in upstream habitats from sedimentation. This combination of events resulted in only about 1,000 South Fork Chinook returning to Idaho by 1980. (Numbers of fish were developed from records of the Idaho Department of Fish and Game.)

Severe impacts to fish habitat occurred during the 1950's and 60's because land managers during this period failed to understand the relationships between fish habitat and sedimentation resulting from land disturbing activities, particularly roads constructed on steep slopes or adjacent to stream courses (Megahan, 1974, 1975). During the 1950's and early 60's, roads on slopes in excess of 50% steepness commonly reached densities of more than 12 miles per square mile of area logged in the South Fork basin (Lundeen, 1968). Over this period, about 800 miles of road were constructed to harvest more than 320 million board feet of timber from the area.

Forest Service management policy changed abruptly in the drainage following a series of major storm and runoff events in the mid-1960's. Extensive damage to the river system from natural and man-caused sediment became evident to most observers at this time. Concern for the protection of anadromous fish was also intensifying. To allow for the recovery of the river and to study the causes of the problem, the Forest Service imposed a moratorium on all logging in the Upper South Fork which lasted from 1965 until 1978. In an effort to rehabilitate overall watershed conditions, more than 500 miles of road were made impassable to car and truck traffic during this period. These road surfaces were ripped, seeded, and out-sloped to drain freely (Mickelson, et. al., 1973). At the same time, a research and monitoring program was intensified in order to identify the key management elements affecting sedimentation. The results of this research and monitoring form the basis for the sediment models currently being applied to the South Fork.

In 1981, a work group comprised of technical experts in watershed sciences from Idaho and Montana National Forests and the Intermountain Forest and Range Experiment Station developed data gathered from the South Fork studies and other research into a set of guidelines for predicting management related sediment from forested watersheds in the Idaho Batholith (Cline, et. al., 1981). These general guidelines were then refined into a modeling procedure involving the specific land type and soil characteristics of the Payette National Forest (Jahn and Kulesza, 1983). This procedure was used to calculate the sediment potentials displayed in the remainder of this report.

SEDIMENT MODEL RESULTS, 1945 - 1985

In the Forest Planning process, a sediment model may help planners compare probable impacts of future land management scenarios on water quality and fish habitat. The degree of confidence with which model results are used depends upon how well the model has been shown to fit actual situations which have been monitored. Because of its easily trackable development history and the record from more than twenty years of watershed and fish habitat monitoring, the South Fork Salmon River provides an excellent subject for retrospectively testing and refining sediment model assumptions.

The sediment model used in this analysis includes a large number of variables. The major variables related to land characteristics include soil type, slope, aspect, elevation, and distance from stream courses. The major variables related to land disturbance include disturbance type, extent, location, mitigation applied, elapsed time or age, and effects of subsequent disturbance. In order to obtain a picture of the historical development of sediment potential within the drainage, all disturbances

Figure 2

ACCELERATED SEDIMENT POTENTIAL
ENTIRE SFSR 1945-1985

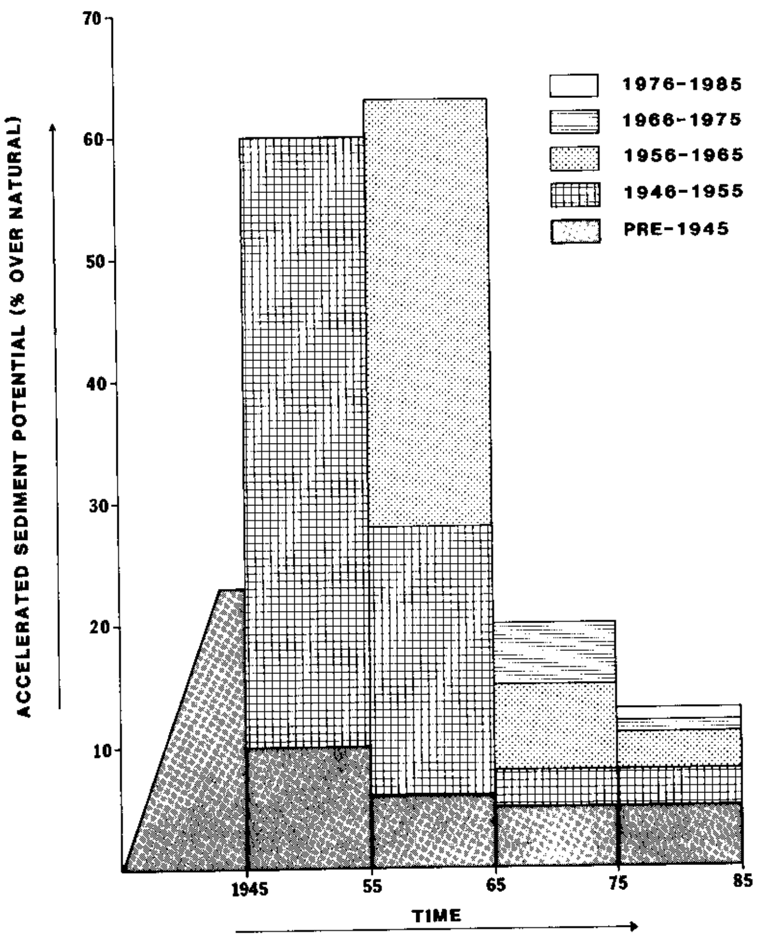
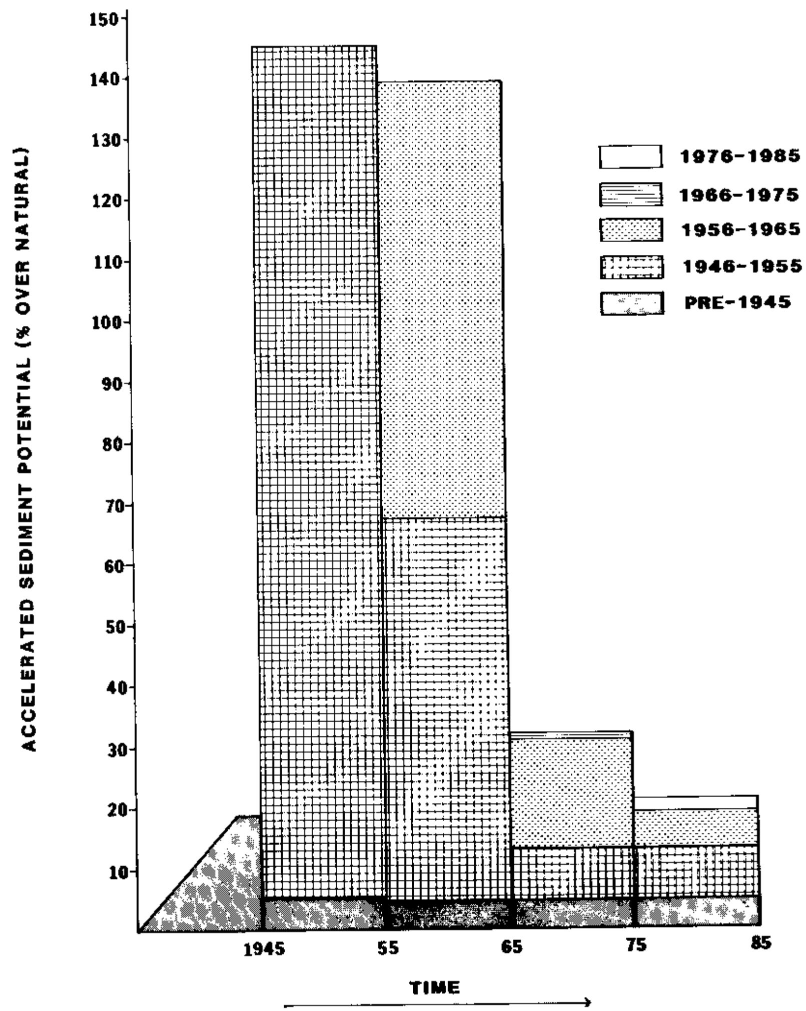


Figure 3

ACCELERATED SEDIMENT POTENTIAL
UPPER SFSR 1945-1985



since 1945 were measured and categorized according to appropriate sediment model variables. Disturbances occurring prior to 1945 were placed into more generalized categories because data were not as reliable. The results of this analysis show the history of accelerated sediment within the entire drainage (Figure 2), and for the Upper South Fork portion of the drainage (Figure 3) in particular. For tracking purposes, sediment potential was coded according to its decade of origin.

Drainage-wide, several important points stand out in Figure 2. First, sediment potential was already well above natural in 1945 due to an influx of mining roads which were typically located immediately adjacent to stream courses. By 1945, for example, there were more than 300 miles of recently constructed road in the drainage and a considerable amount of land disturbed by mining activities. Most mining occurred in the East Fork and Secesh portions of the drainage. Major logging efforts began in 1950 and lasted until 1965. During this time, more than 800 additional miles of road were constructed within the drainage, most of which were built on steep slopes with highly erodible soils (Megahan, Platts, and Kulesza, 1980). This rate of development corresponds to the relatively high accelerated sediment potential associated with the period from 1945 to 1965 when levels reached 63 percent over natural. The sharp drop in sediment potential between 1965 and 1975 is related to the moratorium on logging and road construction in the Upper South Fork combined with the major rehabilitation and closure of more than 500 miles of road. Although logging continued in other subdivisions of the drainage after 1965, disturbance levels were much lower and activities took place on less hazardous land types. The 1975-1985 column on the graph displays a cumulative accelerated sediment potential of 13 percent over natural for the entire drainage. According to this analysis, sediment production potential approaches an essentially stable condition in the 1975-1985 decade.

The history of development in the Upper South Fork portion of the drainage is reflected in Figure 3. This part of the drainage has been most intensively monitored over the past 20 years (Figures 4, 5, 6, 7, and 8). It was also the area most heavily developed in the 1950's and 60's. The accelerated sediment potential values, which are relatively larger than those in Figure 2, reflect this development intensity and the fact that the analysis area at 250,000 acres, is considerably smaller than the entire South Fork basin at 800,000 acres. The effects of the moratorium and rehabilitation appear more pronounced in the Upper South Fork than in the drainage-wide display.

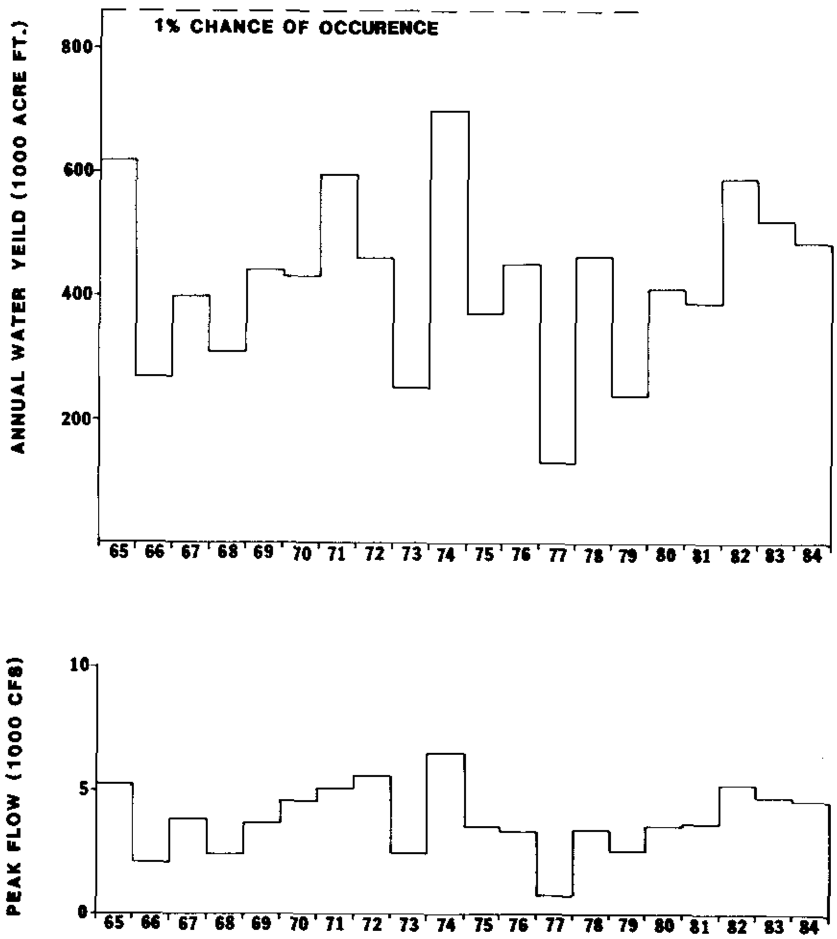
RESULTS OF CONTINUED MONITORING, 1965 - 1985

Peak Flow and Water Yield

The relationship between water yield, accelerated sediment potential, and the various parameters representing stream condition is not clearly understood at this time. The data, however, suggest that an important relationship does exist. For example, in the mid-1970's the river was still heavily loaded with sediment generated from the activities of the

Figure 4

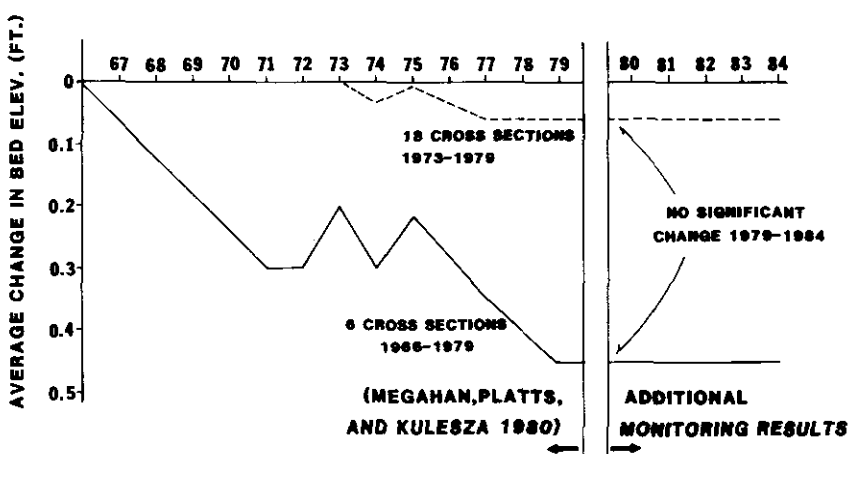
ANNUAL WATER YIELD [1/]
AND
INSTANTANEOUS PEAK FLOW AT KRASSEL GAGE



[1/] DATA PRIOR TO 1967 ARE ESTIMATED FROM A REGRESSION RELATION DEVELOPED WITH DATA FROM JOHNSON CREEK

Figure 5

SFSR SURVEYED CHANNEL
CROSS SECTIONS 1966-1984



previous decade. High peak flows during this period appear to be related to significant changes in the rate of channel degradation(Figure 5). During the mid-1980's, following the removal of most stored sediment of the 60's and 70's, large peak flows appear to have little if any effect upon channel morphology but may be related to more subtle changes in parameters such as depth fines at spawning areas.

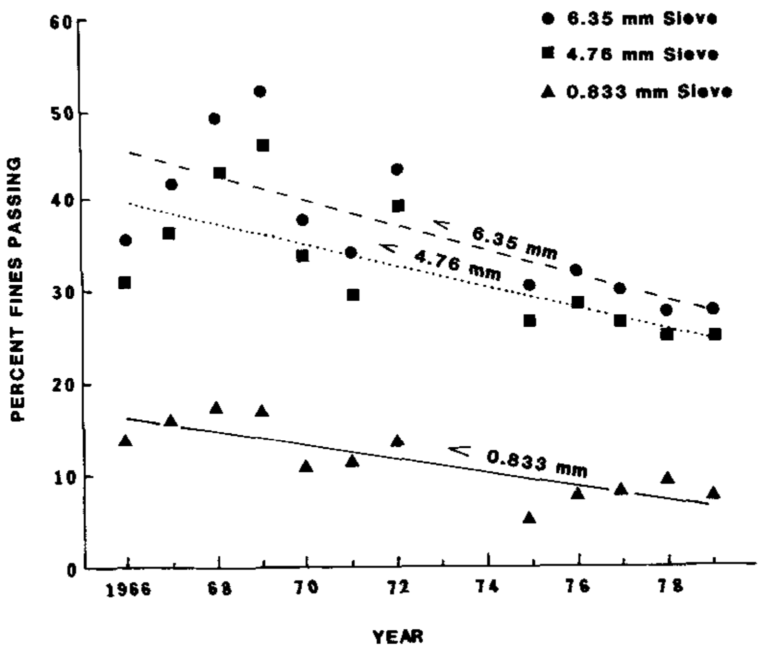
Water yield and peak flow as measured for the Upper South Fork area are shown in Figure 4.

Channel Morphology

Surveyed channel cross sections provide an indication of overall channel and streambed stability. These measured cross sections show pronounced recovery over the period of 1965 to the late 1970's, as large amounts of stored sediment were being removed from the stream channel (Megahan, Platts, and Kulesza 1980). The channel aggradation measured in 1973 and 1975 appears to be related to the high peak flows experienced in 1972 and 1974, respectively (Figure 5). Continued monitoring of these cross sections shows a stabilized condition to date since the late 1970's despite relatively large water years in 1982, 1983, and 1984 in terms of both total yield and peak flow.

Figure 6

CORE SAMPLES
SOUTH FORK SALMON RIVER



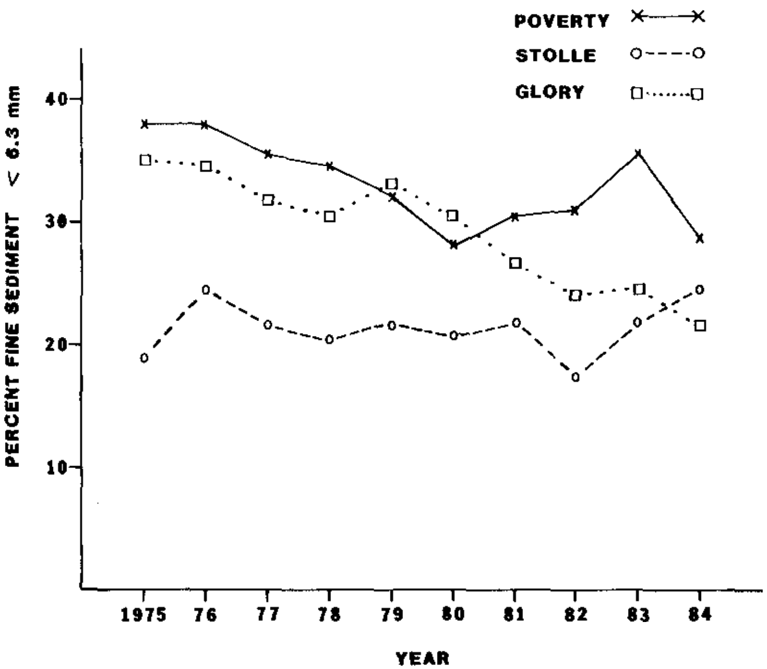
MEGAHAN, PLATTS, AND KULESZA, 1980)

Core Samples

The amount of fine sediment, less than 6.3 mm, contained within spawning gravels, is inversly related to successful salmonid fry emergence (Stowell, et. al., 1983). Megahan, Platts, and Kulesza (1980)

SPAWNING GRAVEL
CORE SAMPLES

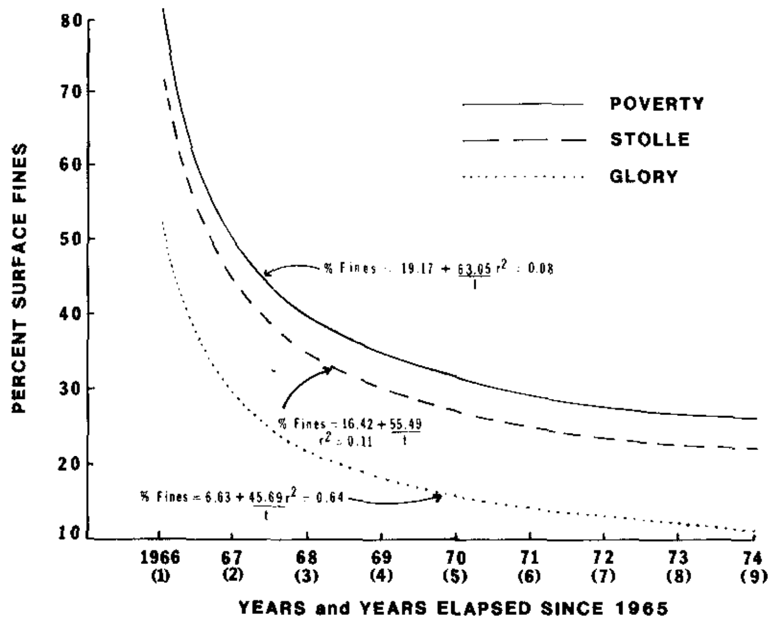
Figure 7



(CORLEY AND NEWBERRY, 1985)

SURFACE FINES
SOUTH FORK SALMON RIVER

Figure 8



(MEGAHAN, PLATTS, AND KULESZA, 1980)

demonstrated that fine sediment in spawning gravels declined over the 1970's (Figure 6). This decline is not as evident after 1978, throughout most of the Upper South Fork (Corley and Newberry, 1985), as shown in Figure 7. Such a relationship is consistent with modeled sediment potentials which imply that general stability has been reached. It is interesting to note that Stolle, the uppermost section of the river, appears to have reached stability in intergravel fines by the mid-1970's, while stability at Poverty, in the middle section of the Upper South Fork, was not achieved until about 1980. Significant improvement still appears to be occurring at Glory on the lower portion of the Upper South Fork.

Surface Fines

Similar to core sample results are observations of fine surface deposits (Megahan, Platts, and Kulesza, 1980), as shown in Figure 8. A sharp decline in the amount of deposited sediment is evident in the early 1970's, with the trend flattening out in later years.

SUMMARY AND CONCLUSIONS

Model results and monitored effects are consistent in showing that the potential for large amounts of sediment production from major climatic events on man-caused land disturbance has declined from the mid-1960's to the present in the South Fork Salmon River. The model nor the monitoring display cause and effect relationships developed from statistically sound experimental design. However, they do display relationships which can be used in decision making by stimulating knowledgable public debate about the effects of man-caused sedimentation on the South Fork. The model identifies that an essentially stable condition has been reached with further recovery being possible only in the minor amount of increased sediment potential generated from activities occurring since 1975 or from major drainage-wide mitigation of residual sediment sources which originated in previous decades and remain untreated to date. Most of these sources are associated with the nearly 600 miles of unsurfaced road within the drainage, which remain open to the public

Further refinement of sediment production/fish habitat response models, such as those used in the South Fork Salmon River, will provide forest planners with an increasingly effective tool for accurrately predicting the individual and cumulative effects of management activities on forest lands.

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BENTHOS IN A SEDIMENT-LADEN DELTA STREAM SYSTEM

By Charles M. Cooper, Ecologist, U. S. Department of Agriculture, Agricultural Research Service, Sedimentation Laboratory, Oxford, Mississippi

ABSTRACT

The macroinvertebrate fauna of Bear Creek, a 83 km stream which flows through 6 oxbow lakes, was studied for 2 years. Two off-stream lakes in the same watershed were also investigated. Bear Creek, a tributary of the Yazoo River in the Delta region of Mississippi, drains 330 km^2 of intensively cultivated agricultural land and carries a continuing sediment concentration during the rainy season of up to 1300 mg/L. Benthic organisms were collected monthly from August, 1976 through August, 1978 at 6 stream stations and at 14 lake stations, across a variety of substrates, and at depths ranging from 0.5 m to 6.5 m. In most lake and stream sections of the creek, Chaoborus (1 species), Chironomidae (19 genera), and Oligochaeta (3 species), dominated the benthos. Thirteen other genera of insects were represented as were Bryozoa (4 genera), Mollusca (9 genera), Hirudinea (5 genera), and Crustacea (3 genera). There was continuous stress from high rates of sedimentation (up to 7 cm/yr^1) as indicated by diversity (d) at a majority of the lake stations. Creek stations were subject not only to heavy sedimentation but also to fluctuations in water level and drying. A low standing crop of benthos in areas affected most by sediments and limited species diversity in potentially productive areas indicated a stressed aquatic environment. Invertebrate production from an abundance of snag habitat was significant at most creek sites. The mayfly Hexagenia bilineata, the clam Sphaerium rhomboideum, and the bryozoan Pectinatella magnifica, indicated significant negative responses to high suspended sediment concentrations and accompanying pollutants.

INTRODUCTION

The effects of massive amounts of suspended and deposited sediments on various components of aquatic ecosystems have been documented in assessing the processes governing natural systems and man-induced system changes. Individual species of benthos may exhibit specific levels of sensitivity or tolerance to environmental stresses, making them useful in ecosystem analysis. In addition, the mobility of benthos assures redistribution into marginal habitats. Several investigations have been made, with widely varying results, on the catastrophic effects of large quantities of sediment from logging, mining and highway construction sites on benthos (Tebo, 1955; Reed, 1977; Lenat et al. 1981; Duchrow, 1982; Cline et al. 1982). Generally, research has dealt with the effect of a particular disturbance and recovery following cessation of the perturbation. Stream recovery, as measured by re-establishment of macroinvertebrates, may begin within a few weeks following a disturbance depending on how efficiently a stream flushes out the sediments deposited during perturbation.

Results of long-term deposition of sediments on an annual basis are more subtle. Benthos studies in the flatland streams and lakes of the Mississippi River delta where this type of sedimentation is a problem are almost non-existent. The Mississippi Department of Wildlife Conservation conducted cursory investigations of benthos (Bingham, 1969) in several of their fisheries investigations in sediment stressed ecosystems, but no detailed taxonomic or ecological evaluations are available for these highly impacted ecosystems.

The objectives of this study were (1) to identify the benthic components of a typical alluvial stream system and (2) to determine what limits are imposed on such a system by sediment deposition.

STUDY AREA AND METHODOLOGY

Bear Creek, in Humphreys, Sunflower, and Leflore counties, Mississippi (Fig. 1) is a tributary of the Yazoo River in the intensively cultivated alluvial delta of the Mississippi River. The creek system (83.2 km long) can be divided into 2 reaches. The first reach is a sluggishly flowing stream that originates from Blue lake and meanders for 41 km. Flow in the upper one-third of the stream is intermittent. The second reach consists of 5 riverine oxbow lakes connected by short stream segments. The creek channel increases from 1 m deep and 5 m wide at its origin to 5 m deep and 10 m wide at maximum in-bank flow at its confluence with the Yazoo River. The 6 instream lakes vary in surface area from 3 to 142 ha. The 330 km^2 watershed also contains numerous off-stream lakes and old channel scars, many of which are in advanced stages of ecological succession and are partially filled with floating herbaceous vegetation or emergent woody vegetation.

Bottom substrate (0.28 m^2) was collected monthly for 2 years by Ekman grab, beginning in August, 1976 at 6 stream stations and 14 lake sites (Fig. 1). After sieving (>0.589 mm^2), samples were preserved, sorted and counted. Exclusive of Mollusca, organisms not permanently fixed for identification were sorted to species level and divided into 1-mm size classes when necessary before they were dried and weighed. Size-classes and life history information were used to separate generations or cohorts. It was necessary to separate generations where they overlapped because production estimates (Waters, 1971) were calculated by summing the amount of biomass produced by each generation of a species during a year. The mean annual density of any species in a square meter of bottom surface was calculated from monthly counts when species representation was too low to produce a reliable weight.

RESULTS AND DISCUSSION

Water Quality Results

Several chemical and physical water quality parameters were measured concurrently with benthos (Cooper and Burris, 1984). Of the parameters examined, turbidity and sediment accumulation were the most evident problems. The creek carried a sediment concentration during the rainy season of up to 1300 mg/L and flow through riverine lakes accumulated sediments at rates which averaged from 1.4 to over 5.6 cm/yr^{1} during the last 23 years (Ritchie et al. 1979).

Taxonomic Results

Representatives of 60 species of invertebrates were collected from Bear Creek (Table 1). Of these, Chironomidae (19 genera), *Chaoborus punctipennis* and Oligochaeta (3 species) composed the majority of the benthos. Aquatic insects

Odonata (6 species), Trichoptera (1 species), Coleoptera (3 species), Megaloptera (1 species), Hemiptera (1 species) and other Diptera (2 species). In addition, Bryozoa (4 species), Mollusca (9 species), Hirudinea (5 species) and Crustacea (3 species) were collected.

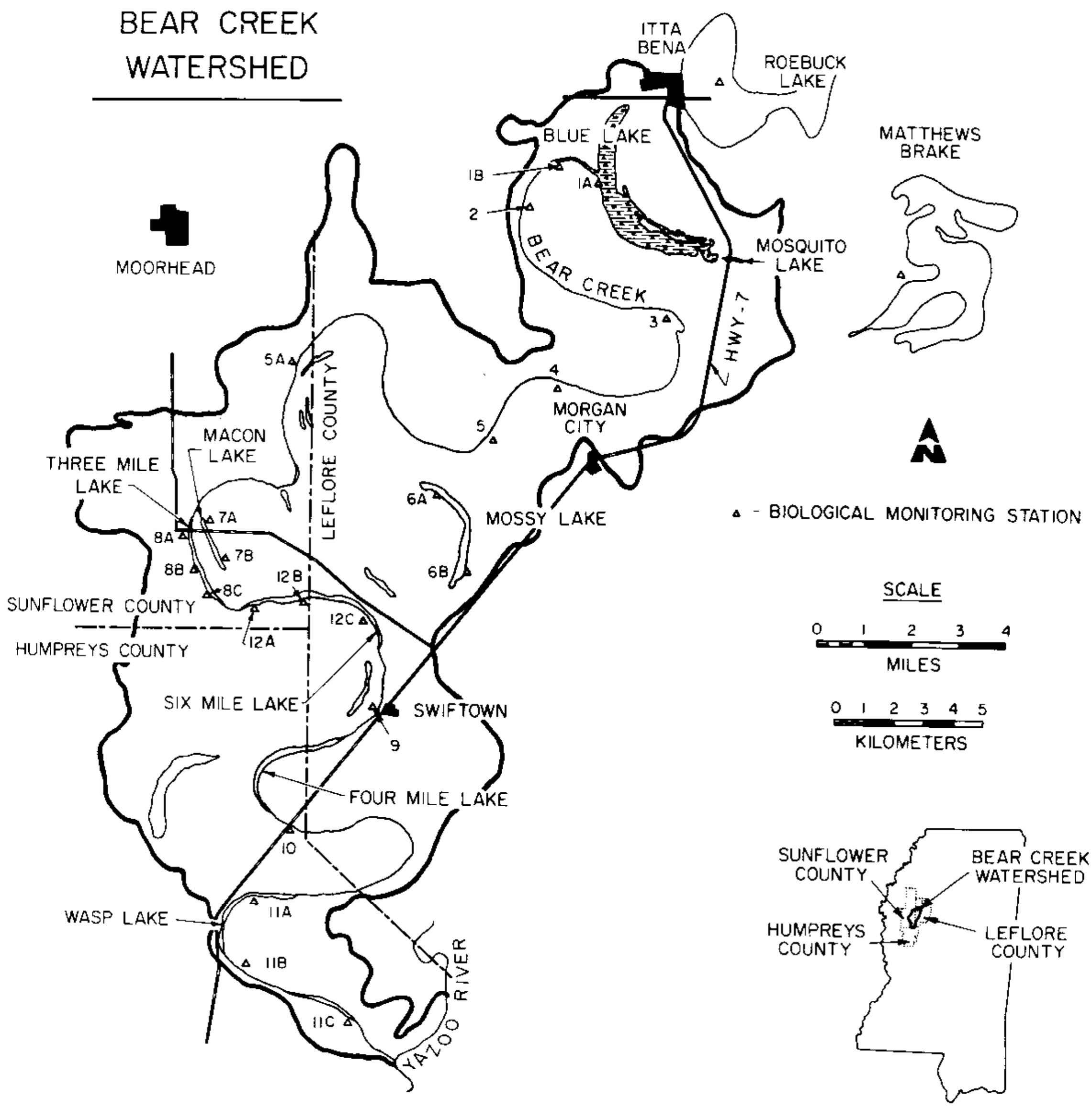

Figure 1. Map of Bear Creek, including sampling sites.

Table 1. List of species of macrobenthos collected from Bear Creek, Mississippi (1976-1978).

Bryozoa
- *Fredericella sultana*
- *Lophopus crystallinas*
- *Pectinatella magnifica*
- *Plumatella fruticosa*

Annelida
- *Tubifex tubifex*
- *Branchiura sowerbyi*
- *Limnodrilus hoffmeisteri*
- *Helobdella stagnalis*
- *H. elongata*
- *Placobdella montifera*
- *Illinobdella moorei*
- *Moorebdella microstoma*

Mollusca
- *Helisoma sp.*
- *Physa sp.*
- *Stagnicola sp.*
- *Corbicula manilensis*
- *Musculum sp.*
- *Sphaerium rhomboideum*
- *Anodonta sp.*
- *Carunculina parva*
- *Amblema costata*

Ephemeroptera
- *Baetis sp.*
- *Caenis sp.*
- *Hexagenia bilineata*
- *Ephemerella sp.*

Odonata
- *Libellula sp.*
- *Perithemus tenera*
- *Argia sp.*
- *Ischnura posita*
- *Dromogomphus spinosus*
- *Tauriphila sp.*

Trichoptera
- *Cyrnellus fraternus*

Coleoptera
- *Berosus sp.*
- *Dubiraphia sp.*
- *Hydrocanthus sp.*

Megaloptera
- *Neohermes sp.*

Hemiptera
- *Notonecta sp.*

Diptera
- *Chironomus riparius*
- *Leptochironomus sp.*
- *Cryptochironomus digitatus*
- *Dicrotendipes sp.*
- *Endochironomus sp.*
- *Coelotanypus concinnus*
- *C. scapularis*
- *C. tricolor*
- *Kiefferulus sp.*
- *Goeldichironomus sp.*
- *Pentaneura monilis*
- *Tanypus carinatus*
- *T. stellatus*
- *Tanytarsus sp.*
- *Ablabesmyia sp.*
- *A. annulata*
- *A mallochi*
- *A. ornata*
- *Polypedilum sp.*
- *P. halterale*
- *Glyptotendipes senilis*
- *Harnischia sp.*
- *Einfeldia natchitocheae*
- *Hydrobrobaeuus pilipes*
- *Procladius culiciformes*
- *Parachironomus sp.*
- *Chaoborus punctipennis*
- *Chrysops sp.*
- *Ceratopogonidae*

Crustacea
- *Hyalella azteca*
- *Procambricus sp.*
- *Palaemonetes kadiakensis*

Habitat Preference

Less than 50 percent of the species collected were found at more than two sites. Thus, the number of commonly-occurring species (20-25 species) compared favorably to other sluggish water or lacustrine habitats in the region (Cooper and Knight, 1985). Of the 36 species found at only one or two sites, 14 species occurred at Station 5A, 11 species occurred at Station 10 and 8 species occurred at Station 2. Nonparametric statistics were used to test for significant differences between sites for each species of benthos represented because benthos was not normally distributed [0.01 level, as determined by Lilliefors' test (Conover, 1971)]. The Kolmogorov-Smirnov two-sample test statistic (T_1) as described by Conover (1971) was used to test the null hypothesis (the distribution of any organism at sites x and y did not differ significantly during the 2 year study). T_1 was the greatest vertical distance between two empirical cumulative distribution functions. Distribution functions for 25 species found at 5A differed significantly from all other sites at the 0.1 level. The relative abundance of numerous species at stations 2 and 10 was also significantly different from other sites at the 0.1 level (Table 2). Substrate type and current were the major differences between these 3 sites (2, 5A, 10) and the other habitats sampled in Bear Creek. Because of the gradient, there was consistently enough flow at sites 2, 5A, and 10 to prevent sediment accumulation and to allow organisms that require current in food gathering activities to function properly. All other stream sites had less gradient and accumulated sediments seasonally. Lake sites normally had no visible flow and accumulated sediments at an average of 3.3 cm/yr^{-1} (Ritchie et al. 1979). Station 5A had an additional substrate variation. The collection site was 20 m downstream from a clay-gravel covered county road bridge crossing. Gravel, which does not naturally occur in Bear Creek, had gradually accumulated downstream of the bridge and created an artificial substrate unlike any other in the watershed.

Diversity

Diversity indices are used to measure the effect of induced stress on the structure of macroinvertebrate communities and, thus, measure the quality of the environment. Shannon-Weaver diversity (d) as presented by Lloyd et al. (1968) was calculated for each month and derived an annual mean diversity for each site (Table 2). Diversity calculations indicated stress at all sites. It is noteworthy that there was, without exception, more stress in the sites subject to measurable sediment accumulation, but d was not as sensitive to site quality as was taxa richness (number of species) (Table 2). Wilhm (1970) found unpolluted waters to generally have a diversity of between 3 and 4 and polluted waters to be less than 1, but many biologists have found naturally-stressed waters to also have low diversity and poor sensitivity to change (EPA, 1973).

Secondary Productivity

Annual secondary productivity values for benthos from several selected sites are included in Figure 2. The stream sites with the greatest taxa richness (2, 5A, 10) had much higher productivity than other habitats. Stream sites were quite productive in spite of water level fluctuations and seasonally high levels of suspended sediments. In addition, qualitative sampling of snag and emergent vegetation indicated that these habitats consistently provided additional secondary productivity. Estimates commonly revealed 500-1000 organisms/m^2 of snag surface in stream reaches with stable water level. Declines in standing crop of sediment-sensitive organisms reduced productivity in several instances [i.e., Station 5A in 1977-1978 (Table 2)].

Lake benthic productivity ranged from 1.5 $g/m^2/yr^1$ to 5.3 $g/m^2/yr^1$ (Figure 2). This range of productivity was more variable than stressed zones in other area lacustrine bodies. For comparison, productivity in stressed drawdown zones in flood control reservoirs in Mississippi may vary from <1 $g/m^2/yr^1$ to 2 $g/m^2/yr^1$ (Cooper and Knight, 1985). Area lakes with stable water levels may have littoral zone productivity of 4 to 5 $g/m^2/yr^1$. Lakes in Bear Creek watershed had fewer species and less density per species than stream sections and, thus, were generally less productive (Table 2).

Indication of stress

Several species of invertebrates demonstrated negative responses to high suspended sediment concentrations. Colonies of the bryozoan _Pectinatella magnifica_ were found in Blue Lake (Station 1) and Mossy Lake (Station 6), the two lakes in the system with the lowest concentrations of suspended solids (Cooper and Burris, 1984). In spite of suitable attachment sites, _P. magnifica_ did not colonize downstream reaches where levels of suspended and deposited sediments were higher, probably because of their susceptibility to ingestion of suspended material. Beds of the fingernail clam _Sphaerium rhomboideum_ disappeared or were greatly reduced (Table 2) during periods of high sediment movement at sites 5A and 10. Both Ephemeroptera and Odonata, whose gill structures are vulnerable to high concentrations of suspended sediments, responded similarly. _Hexagenia bilineata_ attempted to repopulate the lakes each spring during reproductive periods but were unsuccessful because of high suspended sediment concentrations. Low concentrations of dissolved oxygen created hypolimnetic stress in summer, creating an additional detrimental condition in deeper sites in riverine lakes.

SUMMARY

Research on Bear Creek demonstrated that the benthic component of aquatic productivity was affected in several ways by runoff-associated sediments. Direct effects included rendering the aquatic environment uninhabitable to sensitive larval forms and elimination of sediment-sensitive organisms during periods of deposition. The major indirect effect of sediment was a degradation of bottom habitat by deposited material so that both the number of taxa and the density of organisms were generally reduced when compared with areas not normally subjected to sediment deposition. Sixty species of invertebrates were identified from Bear Creek but less than 50 percent of those species were found at more than two sites. Thus, productivity was limited by substrate type or by extreme events. Several species responded negatively to seasonal sediment stress and their fluctuations caused wide variations in secondary productivity. The combination of number of organisms per unit area and taxa richness provided an accurate indication of habitat quality and relative environmental stress.

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Table 2. Mean density (organisms/m^2) of abundant benthic macroinvertebrates from the lakes and stream sections of Bear Creek, Mississippi (August 1976–July 1977).

	1A	1B	2	4	5	5A	8A	8C	12A	12B	12C	9	10	11A	11B	11C	6A	6B	7A	7B
Annelida																				
<u>Tubifex tubifex</u>	*	1	54	–	48	16	26	12	5	–	1	31	11	5	5	5	–	1	1	–
<u>Branchiura sowerbyi</u>	*	1	14	10	77	329	1	–	–	–	–	2	–	2	16	31	2	4	8	–
<u>Limnodrilus hoffmeisteri</u>	1	16	358	1114	234	551	175	73	125	638	58	137	57	116	138	274	22	26	76	5
Mollusca																				
<u>Sphaerium rhomboideum</u>	–	–	4	5	2	583	–	–	–	–	–	–	2	–	–	–	55	–	–	–
Ephemeroptera																				
<u>Hexagenia bilineata</u>	–	–	–	2	–	16	–	–	–	–	–	–	1	–	–	2	1	–	–	–
Diptera																				
<u>Chironomus riparius</u>	49	55	147	96	82	639	16	34	72	10	37	25	21	22	13	4	15	10	3	9
<u>Leptochironomus sp.</u>	–	1	1	–	–	–	–	–	–	–	1	1	1	–	–	4	–	–	–	–
<u>Cryptochironomus digitatus</u>	–	–	10	19	25	34	21	10	11	–	7	36	75	2	18	28	3	10	2	7
<u>Coelotanypus tricolor</u>	–	–	10	–	37	1	1	2	–	–	7	10	2	5	10	7	–	9	–	2
<u>Tanypus stellatus</u>	3	3	132	151	177	247	34	42	94	6	101	212	55	176	73	69	306	583	33	272
<u>Glyptotendipes senilis</u>	1	11	1	–	–	1	–	–	–	1	–	–	54	1	–	–	–	1	1	–
<u>Procladius culiciformes</u>	–	2	33	19	52	23	9	13	17	2	16	111	26	46	16	25	16	78	9	35
<u>Chaoborus punctipennis</u>	2801	2796	336	793	623	229	1476	2020	1458	3844	2628	888	207	854	654	411	2404	2636	2404	2937
<u>Ceratopogonidae</u>	–	–	2	8	2	44	–	–	–	–	–	1	1	–	–	–	–	1	–	–
Density (N)	2855	2887	1120	2221	1367	2725	1759	2206	1783	4505	2857	1459	538	1231	944	862	2825	3363	2537	3268
Taxa Richness (S)	11	14	27	17	18	33	11	10	11	11	12	17	30	14	12	16	14	15	11	10
Diversity ($\bar{d}$)	0.16	0.27	1.74	1.34	1.96	1.92	0.86	0.67	0.97	0.46	0.44	1.72	1.99	1.33	1.29	1.43	0.56	0.90	0.42	0.44

* Not found in reoccurring frequency.

Table 2. Mean density (organisms/m^2) of abundant benthic macroinvertebrates from the lakes and stream sections of Bear Creek, Mississippi (August 1977 - July 1978).

	1A	1B	2	4	5	5A	8A	8C	12A	12B	12C	9	10	11A	11B	11C	6A	6B	7A	7B
Annelida																				
Tubifex tubifex	–	–	99	–	5	2	4	1	–	–	–	–	–	–	–	1	–	–	–	–
Branchiura sowerbyi	–	–	–	–	43	28	46	29	21	38	6	12	11	19	14	23	2	1	–	2
Limnodrilus hoffmeisteri	1	5	205	412	311	312	246	88	111	200	24	110	67	165	135	401	19	29	52	39
Mollusca																				
Musculum sp.	–	–	–	1	–	10	–	–	–	–	–	–	–	–	–	1	–	–	–	–
Sphaerium rhomboideum	–	–	–	3	1	87	–	–	–	–	–	–	1	–	–	–	10	7	–	–
Ephemeroptera																				
Hexagenia bilineata	–	–	–	–	–	–	–	–	–	–	–	2	–	–	–	10	–	–	–	–
Diptera																				
Chironomus riparius	1	1	67	391	12	46	11	13	12	7	6	14	14	11	3	19	1	4	2	4
Cryptochironomus digitatus	1	1	–	391	12	46	11	13	12	7	6	14	14	11	3	19	17	17	13	10
Coelotanypus tricolor	–	–	–	1	27	2	1	2	3	–	–	29	–	10	48	4	11	39	1	11
Tanypus stellatus	1	2	30	16	226	67	273	368	770	229	536	597	136	618	317	372	351	642	266	629
Ablabesmyia ornata	–	–	1	–	1	10	–	–	–	–	–	2	–	–	–	–	–	–	–	–
Polypedium halterale	–	–	9	1	–	–	–	–	–	–	–	–	11	1	1	4	–	–	–	–
Glyptotendipes senilis	–	–	8	2	2	1	–	–	8	3	4	2	19	2	5	6	–	–	–	1
Harnischia sp.	–	–	6	4	1	4	1	1	–	–	1	3	–	1	–	–	1	–	–	–
Procladius culiciformes	–	1	9	7	50	10	16	35	60	24	24	49	16	65	31	19	35	32	19	34
Chaoborus punctipennis	2968	3204	333	1526	315	262	956	1263	1451	1624	2151	1568	190	1917	1239	408	1979	2267	4108	2595
Ceratopogonidae	–	1	2	–	4	19	1	–	–	–	1	3	2	1	1	7	–	1	–	–
Density (N)	2972	3215	781	2766	1011	918	1566	1813	2449	2132	2755	2406	491	2828	1798	1297	2427	3040	4461	3326
Taxa Richness (S)	9	13	21	21	19	31	15	12	13	10	13	16	24	15	14	20	14	13	9	13
Diversity ($\bar{d}$)	0.03	0.17	1.51	1.20	2.08	1.89	1.31	1.18	1.05	0.90	0.73	1.51	1.68	1.26	1.37	1.69	0.90	1.07	0.50	0.90

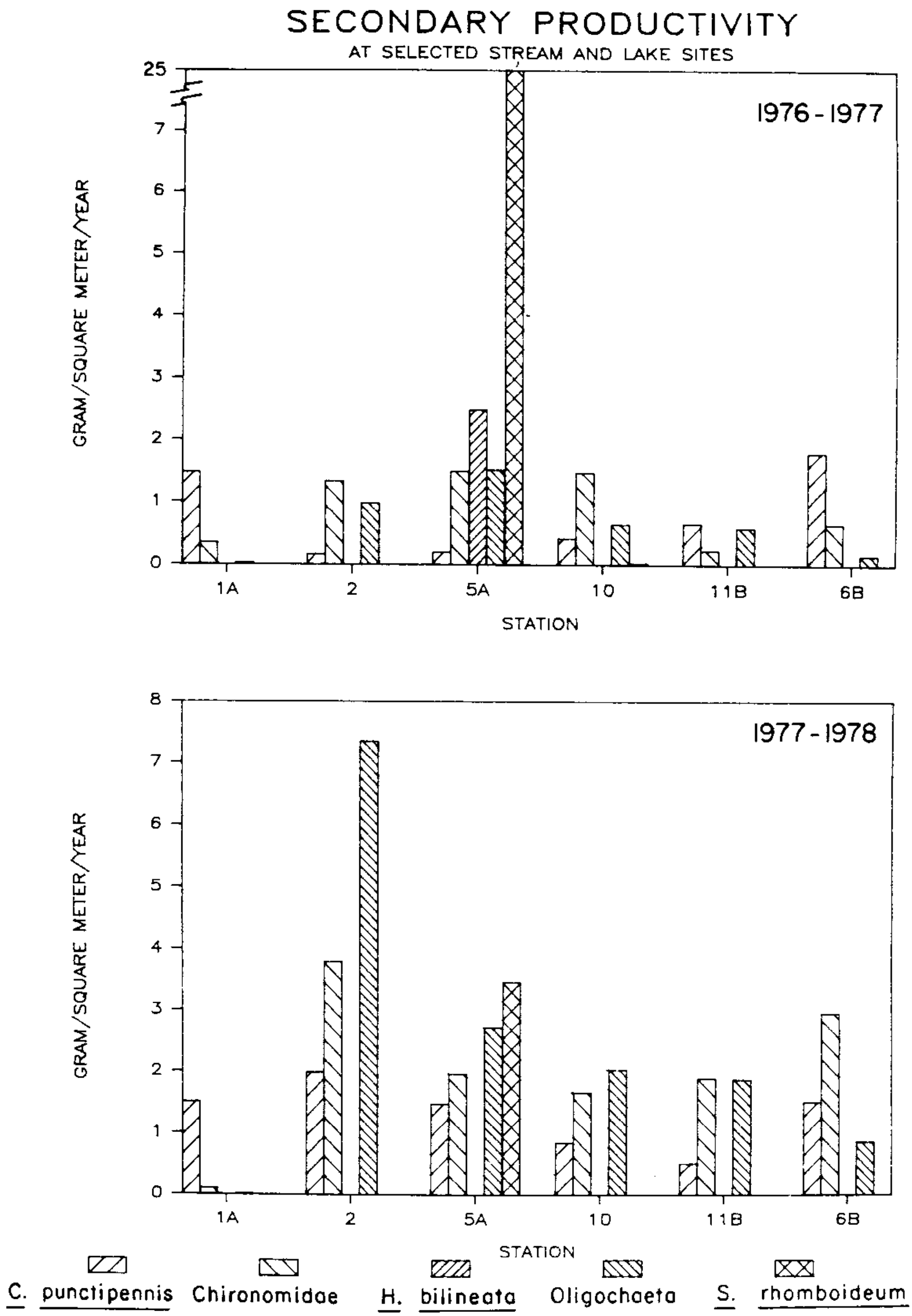

Figure 2. Secondary productivity at selected stream and lake sites in Bear Creek Watershed.

PLOT AND WATERSHED NUTRIENT LOSSES IN THE PALOUSE

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ABSTRACT

Water quality is of great concern in the Pacific Northwest. Only limited information is available on the contribution of cultivated areas to nutrients in runoff. In this study, comparison was made between soluble nutrients in runoff from a 7,019 ha watershed and adjacent research plots near Pullman, Washington. Runoff from the watershed was generally greater than from the mid-slope plots except during a concrete frost runoff event. Nitrate levels were higher from the watershed than from the plots. It was not possible to extrapolate data and relationships directly from mid-slope plots to watershed scale. Such extrapolation requires complex research and modeling techniques not within the scope of this study.

INTRODUCTION

Currently, much public and private concern exists over the issue of preserving water quality in our lakes, streams, and waterways. Maintaining water quality is a necessity in the Pacific Northwest. A deterioration of water quality would adversely affect sport fishing, tourism, the commercial fishing industry, and would adversely affect the economy of the region. In addition, many residents of the region consider water quality an important factor in selecting the Pacific Northwest as their home.

Identifying the source of potential nutrient contamination is clearly necessary for the preservation of water quality. Although not the only source of concern, agricultural practices are increasingly being scrutinized for their possible contribution to deterioration of water quality.

The purposes of this study were to determine (1) soluble nutrient losses in runoff from various crop management systems, (2) the quantities of nitrogen received in rainfall, and (3) the soluble nutrient levels of runoff from a typical cropped watershed in the Palouse.

PREVIOUS STUDIES

Several studies on nutrient losses in surface runoff have used small plots with natural or simulated rainfall. In Georgia, White et al. (1967) found 0.15 to 2.3% of broadcast nitrogen (N) (224 kg/ha as NH_4NO_3) in surface runoff from sandy loam soils with a 5% slope. Romkens et al. (1973) studied the effect of tillage methods on N and phosphorus (P) composition of surface runoff. Two successive simulated rainstorms of 51 mm/h were applied to a Bedford silt loam soil with slopes ranging from 8.2% to 12.4%. Losses of soluble nutrients were in the order coulter-plant > buffalo till plant (till) > chisel plant > disk twice, coulter plant > plow, disk twice, plant (conventional); whereas, sediment N and P losses were greatest from conventional and till systems. Timmons et al. (1973) found that incorporating broadcast fertilizer by plowing down and disking resulted in N and P losses in surface runoff about equal to those from unfertilized plots. Burwell et al. (1975) studied N, P, and potassium (K) losses in surface runoff water and sediment for five soil cover conditions on a Barnes loam soil in west-central Minnesota. Most sediment and insoluble nutrient losses occurred during the critical erosion period two months after corn planting, whereas snowmelt runoff accounted for much of the annual runoff and soluble nutrient losses.

On a highly erodible loessial soil in Mississippi, McDowell et al. (1980) reported total N and P losses from no-till soybeans were only 4.7 and 2.8 kg/ha, respectively, as compared with 46.4 and 17.6 kg/ha from conventional tillage. However, soluble P concentrations and losses from no-till were greater than from conventional tillage.

Baker et al. (1983) applied two simulated rains at 63.3 mm/hr for 105 and 68 min. in the fall after soybean harvest and fertilization (N, P, and K at 31, 35, and 80 kg/ha, respectively). NH_4-N, PO_4-P, and K concentrations in runoff water and on sediment from treatments where fertilizer was incorporated by either point injection or tillage were at the same levels as from the unfertilized treatment. Nutrient concentrations from the surface-application without an incorporation treatment were significantly higher than from all other treatments. Sediment concentrations in runoff from tilled plots were about three times higher than from the no-till plots.

Nutrient losses in surface runoff from different types of watersheds have also been studied. Taylor et al. (1971) found that near Coshocton, Ohio, N and P losses from a farmland watershed were significantly greater than those from a woodland watershed.

Johnson et al. (1973) studied water quality on two Palouse dryland watersheds in eastern Washington. They reported NH_4-N levels were generally less than 1 mg/L in rural waters, with no distinct seasonal variations. NO_3-N concentrations increased to 10 to 20 mg/L during the main erosion season while total inorganic N (NH_4-N + NO_3-N) concentrations dropped to 1 mg/L during midsummer low flow periods. These researchers found only small amounts of soluble phosphates in the surface runoff from agricultural lands.

Schuman et al. (1973) measured nitrogen losses in surface runoff from four agricultural watersheds in Iowa. The 3-year average annual soluble N loss from the contour-planted corn watershed, fertilized at 2.5 times the recommended rate, was 3.05 kg/ha. A comparable watershed, fertilized at the recommended rate, lost only 1.89 kg/ha or slightly greater than one-half as much. Ninety-two percent of the total N lost in the runoff from contour-planted corn watersheds was associated with the sediment.

Johnson et al. (1976) measured soluble and particulate P losses by stream transport from a predominantly rural watershed (330 km^2) in central New York. They reported 20% of the soluble P lost from the watershed was derived from diffuse sources associated with farming, 45% was derived from natural geochemical processes, and 35% from point source inputs. Less than 1% of the P applied to the landscape in chemical fertilizer and manure was lost from the watershed in soluble form. Menzel et al. (1978) measured N and P in runoff for four years from 5 to 18-ha watersheds on a deep Washita River alluvium 2 km east of Chickasha, Oklahoma. Soluble NO_3-N discharge was 4 kg/ha (10 to 30% of total N discharge) and soluble P loss was 2 kg/ha (20% of total P discharge). Rainfall averaged 5 kg N/ha and 0.15 kg P/ha.

Johnson et al. (1979) monitored runoff, soil loss, and nutrient losses for six small, paired watersheds planted to continuous corn near Castana, Iowa. Conservation tillage reduced runoff 40% and soil loss by 60 to 90%. Total N and P discharge was reduced, but soluble P losses and concentrations and available P concentrations in sediment increased with residue cover.

Langdale and Leonard (1981) found that for a 2.71-ha upland Piedmont watershed, total P and occasionally total N concentrations were greater from reduced tillage than from conventional tillage, but the total loss was decreased because of decreased runoff. Hubbard et al. (1982) measured losses of N and P over a two-year period in runoff and sediment from heavily fertilized agricultural watersheds. They reported 70 to 95% of the N and 90 to 98% of the P in runoff were carried with sediments, except during winter months when as much as 80% of the N and 33% of the P were in solution.

Smith et al. (1983) also measured N, P, and sediment in runoff from grassland watersheds in Oklahoma and Texas. Nutrient concentrations ranged from 2 to 10 mg/L for N and 0.3 to 2 mg/L for P. In most cases, less than half the nutrients occurred as soluble forms in the runoff water. Nutrient losses strongly correlated with sediment losses.

Several studies have addressed the contribution of rainfall to NO_3-N and NH_4-N in runoff (Burwell et al., 1975; Jawson et al., 1982; Schuman and Burwell, 1974; Smith et al., 1983; and Taylor et al., 1971). Taylor et al. (1971) reported that N in precipitation from a woodland and agricultural watershed in Ohio averaged 20.3 kg/ha annually for a two-year period and exceeded by six times the average annual N loss in runoff.

Schuman and Burwell (1974) reported that N in surface runoff from two small watersheds was only a fraction of that derived from the total precipitation. Their findings showed that on a watershed fertilized at 168 kg N/ha about two-thirds of the soluble N in surface runoff could be attributed directly to precipitation N.

MATERIALS AND METHODS

Missouri Flat Creek Watershed

Missouri Flat Creek (MFC) is a 7,019 ha (27.1 sq. mi.) watershed, located on the Washington-Idaho border, northeast of Pullman, Washington (Fig. 1) in the higher precipitation area of the Palouse. Most of the area is intensively farmed to winter wheat with barley, peas, lentils, and, occasionally, summer fallow in the rotation. Drainage patterns are well established. Hill slopes of 30 to 40% are common and can be farmed because precipitation rates are low and most precipitation infiltrates rather than runs off. However, erosion rates are severe. Annual soil and nutrient losses are highly variable and difficult to predict because of complex climatic and topographic factors.

Normal annual precipitation near the outlet of the watershed is 542 mm, approximately 20% of which is snow. From the outlet eastward toward the northeast boundary, where the nearby mountains retard the eastward air flow, precipitation increases to approximately 600 mm. The percentage of snowfall increases as well. Most runoff occurs in December through March and is

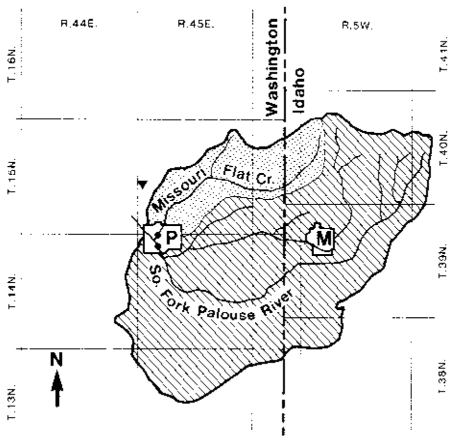

FIGURE 1. Missouri Flat Creek Watershed and PCFS Plots, Pullman, Washington

frequently associated with rain on snow, rain following snowmelt, or rain on frozen ground. Excessive soil erosion and high sediment concentrations frequently accompany these events.

Substantial amounts of fertilizer nitrogen, usually in excess of 100 kg N/ha, are used for winter wheat. Smaller amounts of N are used on other grain crops, but virtually none on peas and lentils. Application is normally by injecting either aqua or anhydrous ammonia at intended rates of up to 90 kg/ha in the fall. Then in early spring, 22 to 34 kg/ha of N as dry granules is commonly applied as a top dressing.

The U.S. Geological Survey (USGS) maintained a stream gaging station on the MFC near its confluence with the South Fork of the Palouse River (SFPR) at Pullman. The gage consisted of a water stage recorder and concrete control with a 90° V-notch weir inside a 0.61 m Parshall flume.

This gaging station was installed in 1934, deactivated in 1940, reactivated in 1960, and terminated following the 1979 water year (October 1 through September 30). Sampling and discharge analysis for our study began in October of the 1977 water year and continued through September of the 1980 water year. Discharge data are available in the annual U.S. Geological Survey publication, "Water Resources Data for Washington."

To avoid a potential nutrient point source, sampling for nutrients was conducted 6.1 km upstream of the gaging station. Soon after collection, nutrient samples were refrigerated at 4°C. Sediment was removed from the samples by filtration through a 0.45-μm membrane filter. NO_3-N and NO_2-N were determined according to Hendricksen and Selmer-Olsen (1970), NH_4-N according to Connetta et al. (1976), and PO_4-P by the single reagent method (USEPA, 1974). NO_2-N in the samples (which was checked frequently) was very low, so the results are reported as NO_3-N.

Nutrient samples were obtained every two to ten days during low flow periods and one or two times daily during high flow periods (major runoff events). Results of the nutrient analysis are reported in mg/L. Total nutrient discharge for the watershed was based upon weighted mean flow rates.

During the 1980 water year, samples for nutrient analysis were collected from MFC. However, the gaging station was deactivated, and no published report was available for calculating total discharge. An active gaging station was maintained on the SFPR through that year and a linear regression analysis for a 6-year period on a monthly basis demonstrated an extremely high correlation (average r^2 = .97 for monthly discharge volumes). The monthly discharge ratios used with the daily discharge for the SFPR provided the MFC estimated daily discharge data for the 1980 water year.

It was not within the scope of the project to quantify all variables contributing to the nutrient source inventory in the watershed. No attempt was made to determine the contribution of subsurface transport (including drain tile discharge) to soluble nutrient levels in Missouri Flat Creek.

RUNOFF PLOTS

The research plots were located 3.5 km northwest of the gaging station on a generally south-facing hillside at the Palouse Conservation Field Station (Fig. 1). All the plots were situated on a Palouse silt loam (fine silty, mixed, mesic Pachic Ultic Haploxeroll) soil.

The normal annual precipitation at the station is 542 mm, approximately 356 mm of which occurs during the main erosion season, October through March. Four basic crop rotations with associated tillage levels were used: (1) Winter wheat after summer fallow, tilled (WW/SF-T); (2) winter wheat after small grain, tilled (WW/SG-T); (3) winter wheat after peas, tilled (WW/P-T); and (4) winter wheat after small grain, no-till seeded (WW/SG-NT). These four rotation-crop management systems represented typical or potential crop managements for this area. In addition, a continuous bromegrass (BG) treatment was monitored.

Runoff plots were rectangular, 3.66 m wide by 22.2 m long. The plots were bordered by 200 mm galvanized sheet metal strips driven approximately 100 mm into the soil. In order to collect the runoff and sediment produced from the plots, the lower border of each plot was narrowed to a "V". The apex of the "V" was connected to a 51-mm diameter galvanized pipe which discharged into a large steel collection tank. Runoff volume was determined by measuring the depth of the runoff in the tank and using a volume vs. depth relationship for each tank.

Fertilizer was applied to the plots in dry form. In general, the bulk of the N was broadcast prior to final seedbed preparation, and a starter fertilizer with P was applied at winter wheat seeding. Application rates and formulations varied from year to year. To the WW/SF-T and WW/SG-T plots, from 72 to 121 kg N/ha and from 22 to 101 kg P/ha were applied. To the WW/P-T plots, from 88 to 121 kg N/ha was applied, except in the 1980 and 1984 water years when only 45 kg P/ha was applied. To the WW/SG-NT plots, from 67 to 112 kg N/ha was applied except in the 1980, 1981, 1983, and 1984 water years, when none was applied; from 46 to 101 kg P/ha was applied except the 1980, 1981, and 1983 water years, when none was applied. The BG plots were fertilized only in the 1977 through 1979 water years with 45 to 76 kg N/ha and in the 1977 and 1979 water years with 26 to 56 kg P/ha.

In general, runoff and erosion samples were collected from the plots daily. For extended runoff events, the samples represented the average composition of the runoff for that day. For most events, runoff remained in the tank for no more than 24 hours. Because winter air temperatures were low, temperature of the water in the tanks seldom exceeded 4°C.

Water samples were drawn from the tanks after resuspending the settled sediment with a recirculating jet on a pump. The water and sediment mixture was pumped through a sampling tee and the smaller portion into a small tank. The mixture in the small tank was resuspended, and a 1-L sample was collected for nutrient analysis. The sample was treated and analyzed the same as the watershed samples.

RESULTS AND DISCUSSION

Runoff data, soluble NO_3-N, NH_4-N, and PO_4-P data are presented in Tables 1 through 4, respectively. Precipitation and soluble N in precipitation data are presented in Table 5. The 1977 water year was a drought year with approximately 60% of normal precipitation (Table 5). The watershed discharged 10 mm (740,120 m^3) of water (Table 1), compared with a 25-year average of 110 mm (7,684,900 m^3). Soluble NO_3-N (Table 2) and NH_4-N (Table 3) exhibited low mean concentration levels (less than 1.0 and 0.19 mg/L, respectively) with very low total annual loss (0.09 and 0.02 kg/ha, respectively), presumably from a lack of overland and subsurface transport. During the 1977 water year there was no runoff from the plots. There was no significant precipitation during the main erosion season.

The 1978 water year was near normal in total precipitation and the watershed discharge volume at 110 mm (7,684,900 m^3) was equal to the 25-year average. Both the mean concentration level (7.0 mg/L) and the total NO_3-N loss (8.2 kg/ha) were high as compared with Midwest data. As expected, soluble NH_4-N losses were low (0.15 kg/ha). The plots, however, showed much lower losses with the WW/SG-NT cropping system showing the highest total losses of soluble NO_3-N, NH_4-N, and total runoff at 0.88 kg/ha, 1.26 kg/ha, and 75 mm, respectively.

The 1979 water year was somewhat atypical since there was only one extended major runoff event (during February) that lasted more than a week. This runoff event was caused by heavy rain on snow covering deeply and impermeably frozen ground. Consequently, soluble nutrients were primarily associated with the surface runoff from the rain and snow. This single event provided 100% of the year's runoff on all the plots except WW/SF-T, and it provided 80% of the discharge from that plot. The watershed discharge was 26% above average at 139 mm (965,600 m^3), but the total precipitation for the year was actually 12% below normal at 479 mm. In 1979, over 9 kg/ha NO_3-N was lost from the watershed. Although total discharge volume was high, resulting in above-average soluble nutrient losses, concentration levels generally were average to below average.

For the 1980 water year, total precipitation was near normal at 565 mm but discharge volume from both the watershed and the plots was considerably below average. However, mean soluble NO_3-N concentrations were higher for both plots (ranging from 6.7 to 15.6 mg/L) and watershed (9.1 mg/L). Mean plot concentration values were two to six times higher than all other years.

Nutrient sampling on MFC was terminated following the 1980 water year, but sampling continued on the research plots. In 1981, the plots exhibited

TABLE 1. RUNOFF (mm) FROM MISSOURI FLAT CREEK AND THE PCFS PLOTS

Water Year	MFC	PCFS PLOTS				
		WW/SF-T	WW/SG-T	WW/P-T	WW/SG-NT	Bromegrass
1977	10	NR[1]	NR	NP[2]	NR	NR
1978	110	NP	25	NP	75	5
1979	139	147	108	88	114	144
1980	67[3]	25	7	1	5	1
1981	ND[4]	63	1	22	1	4
1982	ND	94	5	29	18	18
1983	ND	91	7	33	NR	12
1984	ND	87	41	44	11	11

TABLE 2. ANNUAL NO_3-N LOSSES (kg/ha) FROM MISSOURI FLAT CREEK AND THE PCFS PLOTS

Water Year	MFC	PCFS PLOTS				
		WW/SF-T	WW/SG-T	WW/P-T	WW/SG-NT	Bromegrass
1977	0.09	NR	NR	NP	NR	NR
1978	8.20	NP	0.21	NP	0.88	0.02
1979	9.38	2.13	2.72	1.54	1.82	3.22
1980	6.08[3]	1.87	1.11	0.07	0.34	0.06
1981	ND	2.07	0.01	0.78	0.02	0.02
1982	ND	4.40	TR[5]	1.11	0.28	0.07
1983	ND	0.74	0.09	1.13	NR	0.02
1984	ND	1.58	0.78	0.99	0.04	0.03

TABLE 3. ANNUAL NH_4-N LOSSES (kg/ha) FROM MISSOURI FLAT CREEK AND THE PCFS PLOTS

Water Year	MFC	PCFS PLOTS				
		WW/SF-T	WW/SG-T	WW/P-T	WW/SG-NT	Bromegrass
1977	0.02	NR	NR	NP	NR	NR
1978	0.15	NP	0.28	NP	1.26	0.02
1979	0.25	0.47	0.22	1.32	0.07	2.23
1980	0.27[3]	0.39	0.08	0.01	0.04	TR
1981	ND	0.27	TR	0.33	TR	0.03
1982	ND	1.20	TR	0.10	0.03	0.06
1983	ND	0.10	0.04	0.13	NR	0.06
1984	ND	0.10	0.03	0.11	0.01	0.02

TABLE 4. ANNUAL PO_4-P LOSSES (kg/ha) FROM MISSOURI FLAT CREEK AND THE PCFS PLOTS

Water Year	MFC	PCFS PLOTS				
		WW/SF-T	WW/SG-T	WW/P-T	WW/SG-NT	Bromegrass
1977	ND	NR	NR	NP	NR	NR
1978	ND	NP	0.04	NP	0.33	0.03
1979	0.36	0.39	0.47	0.62	0.33	1.45
1980	ND	0.34	0.07	TR	TR	TR
1981	ND	0.56	TR	0.12	TR	0.03
1982	ND	0.68	TR	0.07	0.10	0.10
1983	ND	0.56	0.02	0.06	NR	0.07
1984	ND	0.43	0.35	0.09	0.08	0.10

TABLE 5. ANNUAL PRECIPITATION (mm) AND ASSOCIATED SOLUBLE N (kg/ha) AT THE PCFS, PULLMAN, WASHINGTON

Water Year	PRECIPITATION	NO_3-N	NH_4-N	TOTAL N
1977	335	ND	ND	ND
1978	545	1.76	3.12	4.88
1979	479	2.01	2.06	4.07
1980	565	1.69	2.31	4.00
1981	541	1.49	2.47	3.96
1982	680	ND	ND	ND
1983	633	ND	ND	ND
1984	595	ND	ND	ND

1 = No runoff; 2 = No plot installed; 3 = Estimated (see text); 4 = No data; 5 = Trace (<0.01 kg/ha).

typical discharge volumes but comparatively high mean nutrient concentrations. However, these concentrations were still not as dramatic as for the 1980 water year. Precipitation was near normal at 541 mm.

For the 1982-84 water years, only plot data were collected because gaging stations on both MFC and SFPR were deactivated. Precipitation in 1982 was about 22% above normal with discharge from the WW/SF-T plot again outperforming all others in volume and soluble nutrients. NO_3-N was over twice the five-year average (1980-84 water years) with a total loss of 4.40 kg/ha. Runoff for WW/SF-T was only 25% higher in 1982 than the five-year average. Runoff from the WW/SF-T plots in the 1983 and 1984 water years was only slightly less than in the 1982 water year, but nutrient levels were all lower. Especially noteworthy were the lower NO_3-N losses. All other plots performed more consistently during these three years.

The 1979 water year was the only year when soluble P data were collected on the watershed. Loss of soluble P for the year was 0.36 kg/ha which was similar to that from the four basic crop rotations (WW/SF-T, WW/SG-T, WW/P-T, and WW/SG-NT). The much higher-than-average soluble nutrient discharge from the BG treatment could be attributed to freezing of the soil shortly after the surface fertilizer application in the late fall, then a snowfall, and finally the runoff generated by rainfall on the snow and the frozen soil.

Total soluble N from precipitation in the forms NO_3-N and NH_4-N was analyzed from the 1978 through the 1981 water years. Average NO_3-N and NH_4-N delivery via rainfall for those years was 1.74 kg/ha and 2.47 kg/ha, respectively, giving average total soluble N in rainfall of 4.23 kg/ha. Total precipitation for those years averaged 533 mm/yr, which is slightly less than the 542 mm normal.

As compared with fertilizer applications, the amount of rainfall nitrogen is rather small. The direct contribution of rainfall to nitrogen in surface runoff was not investigated, but assuming that surface runoff from the plots carried the NO_3-N that fell with the precipitation that caused the runoff, up to 30% of the NO_3-N in plot runoff in water year 1979 came directly from the precipitation. An even larger proportion of the NH_4-N in plot runoff could have come from direct contribution of the precipitation. That the processes in the watershed are much more complex is evident from the NO_3-N data. The watershed losses were considerably larger than the total NO_3-N in the precipitation. The NH_4-N losses from the watershed are less than NH_4-N in the precipitation, consistent with fairly rapid adsorption by soil and sediment.

SUMMARY

During the period for which direct comparisons can be made, and based on the 25-year average, runoff from MFC was greater than from the runoff plots, with the exception of the WW/SF-T plot during the 1979 water year when nearly 100% of the runoff from the plots and watershed occurred during a single extended event. In that case, there was rain on snow above frozen ground and the runoff depth from all plots closely resembled the runoff depth from the entire watershed. Also, soluble NO_3-N losses from the watershed were always greater than from the plots. Possible explanations for the difference between plots and watershed are (1) the precipitation and snowpack on the watershed were somewhat greater than on the plots; (2) the soil types varied between plots and watershed; (3) the plots were located mid-slope where nutrient uptake is the highest (Halvorson)[1]; (4) the plots were primarily located on south-facing slopes; and (5) the plots include only surface flow but the watershed includes the subsurface transport component as well.

Concerning soluble nutrient transport in general, this study found that for the plots, climatological factors such as frost depth, snow-on-ground, and rainfall at the time of a runoff event seemed to contribute more to the soluble nutrient losses than did the tillage practices and crop rotation. However, sediment load and total nutrient loss data would presumably show a much different result. It was not possible to extrapolate data and relationships directly from mid-slope plots to watershed scale. Such extrapolation requires complex research and modeling techniques beyond the scope of this study.

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A BUDGET ANALYSIS OF TURBIDITY AND STREAMFLOW DATA

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ABSTRACT

Turbidity and streamflow data collected from an 8-year USDA-Forest Service monitoring project on Layng Creek, a 60 square mile Western Oregon watershed with high timber and water supply values, was analyzed to provide an indication of the turbidity production response of the watershed. A sediment budget model was used to quantify various production components which, in turn, were examined to determine relative quantities and annual trends. This study also showed deficiencies in the data which can be eliminated in future monitoring.

INTRODUCTION

The Layng Creek watershed is located on the Umpqua National Forest about thirty miles east of Cottage Grove, Oregon in the western Cascade Range. This watershed is part of the headwaters of the Willamette River system, and drains about 60 square miles with an elevation range between 1200 and 5000 feet.

The annual precipitation at the lower elevation averages 53 inches, most of it occurring during the winter months. Since the freezing level fluctuates over the entire elevation range, the winter runoff is supplied by both rainfall and snowmelt. Figure 1 shows a typical annual runoff pattern for the watershed. The topography is steep and extensively dissected; the streamflow response is rapid after the soils have exceeded their storage capacity.

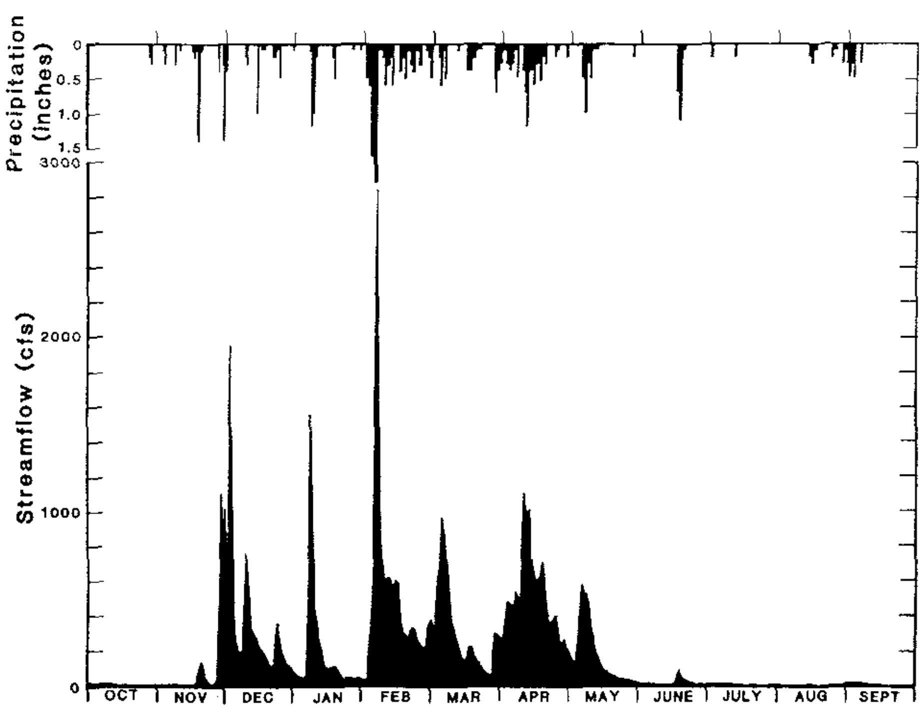


Figure 1. Daily precipitation and mean daily streamflow for Layng Creek, a watershed located in western Oregon for water year 1979.

The land management obligations of the USDA-Forest Service for the Layng Creek watershed are stated in the Multiple Use - Sustained Yield Act of 1960. The watershed currently experiences an annual timber harvest of about 10 million board feet which directly affects about 200 acres or 0.5% of the drainage. Also, about 2,600 acre-feet of water is withdrawn annually for consumption for the municipal water supply for the city of Cottage Grove (population 7,000). Water turbidity during the winter months is a primary concern since it is directly related to water treatment costs and can be adversely affected by timber harvest activities (Beschta, 1981).

A specific issue was whether or not turbidity production was increasing due to timber harvest activities within the watershed. Since turbidity of a stream at a particular location is affected by many factors such as streamflow and supply quantity, type, and location; it became apparent that a simple comparison of turbidity data was not sufficient to resolve this issue. In 1976 turbidity and streamflow monitoring was started to develop a data base which could be examined in more detail.

SAMPLING PROCEDURE

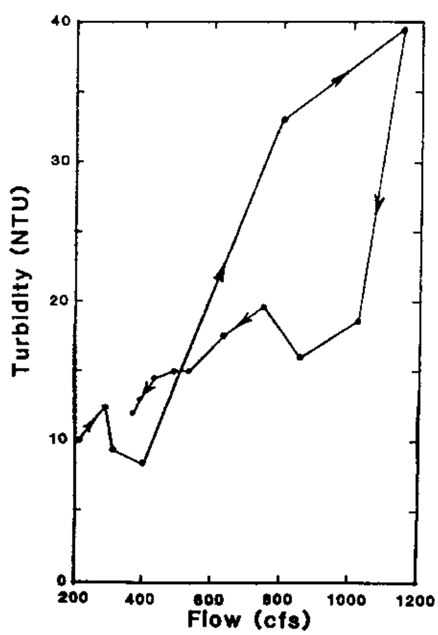
Water samples for turbidity measurement were collected at a monitoring site near the mouth of the watershed about 0.2 miles downstream from the municipal intake and were measured on a Hach 2100A turbidimeter. The samples were obtained by means of an automatic water sampler at 6 hour intervals and were collected weekly. Typically about 90 samples per month were collected during the winter storm season.

Continuous streamflow data was available from a Oregon Water Resources gauging station located about 0.7 miles upstream from the monitoring site. Data collection started in September 1976 and continued through the spring of 1984.

PRELIMINARY ANALYSIS

In Western Oregon the influence of streamflow on stream turbidity has been well documented (Beschta, 1981) and a preliminary analysis of the Layng Creek data confirmed the expected positive correlation. However, since the variability in the raw data was too great to detect annual trends, further analysis was required to refine the data. Earlier research (Paustian,1979) showed that the turbidity - flow relationship could shift during a storm event producing a "hysteresis loop" when the data are plotted as shown in Figure 2.

Figure 2. Sequential turbidity and flow values for the storm of 3/12/80.



Likewise, it was noted that the storms occurring later in the season tend to produce lower, more stable, turbidity/flow ratios; suggesting an annual "flushing" of stored material.

The preliminary analysis showed that the "flushing" effect could be quantified with cumulative plots of peak storm turbidity and peak storm flow as shown in Figure 3. Similar plots for each year indicate that, for peak flows of moderate size storms, the slope of these curves tended toward an annual stable value during the late season storms. Since the slope of the curve is a turbidity/flow ratio, these "stabilized" values provide an indication of annual turbidity production response as shown in Figure 4. Even though a recovery trend was indicated, more analysis was needed to examine the quantities attributed to the "flushed" material and to the large storms.

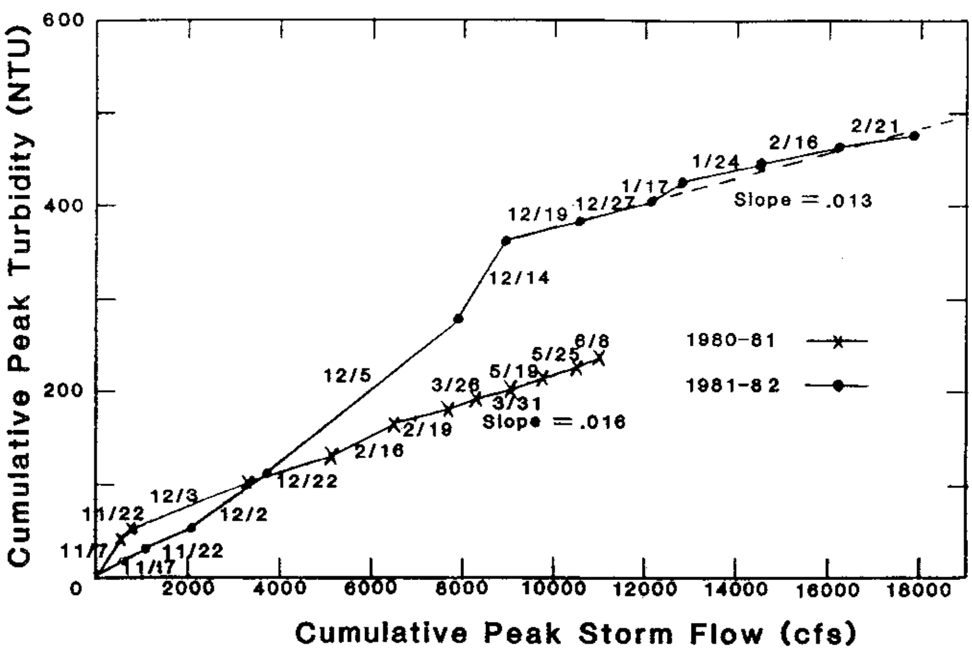


Figure 3. Cumulative peak storm turbidity and flow for two representative water years. The slope of the "stabilized" portion of these curves provided the turbidity/flow ratios used in Figure 4.

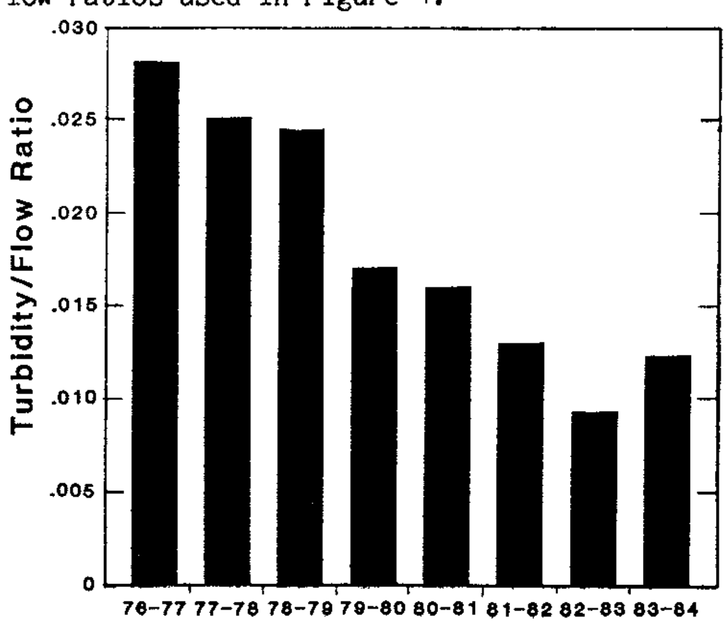


Figure 4. Turbidity/flow ratio after "stabilization" For each year of the study.

THE BUDGET MODEL

Since the data seemed to suggest that depletion of stored material was occurring, a conceptual model based on sediment budget principles was developed. Traditional sediment budgets typically consist of specific source components but, since the monitoring station was located near the mouth of the watershed, it was necessary to develop a model consisting of components defined by general process and transport concepts.

The first component considered was the sediment which was "flushed" from the system each year. Since this material appeared to be roughly time dependent and accumulated each year during the low flow period, it was named, for convenience, the "Weathering and Decay" component. The model assumed that the material generated by these processes would tend to accumulate during the low flow processes but would to be "flushed out" by the larger and more frequent winter storms. Even though these processes take place during the winter months, the amount accumulating between storms would be easily carried away by the rising limb of the storm hydrograph and not affect the peak turbidity/flow ratio.

The next component in the model was the material transported by the low and moderate storms after the flushing was completed. Since the production of this material seemed to be highly flow dependent, it was assumed that this material was being produced by the direct action of streamflow upon the long term supplies within the watershed. Consequently, this component was named the "Flow Generated" component.

The preliminary analysis also indicated that large storms (greater than 2000 cfs) tended to produce higher turbidity/flow ratios and that several subsequent storms were often required to regain "stability". The model assumed that new sources of sediment resulting from the higher flows such as activated bedload, mass wasting and rain related erosion could account for the apparent increased supply of sediment material. Since several storms were required to restore the watershed, it was thought that some of the sediment released by the large storm was temporarily stored upstream and later reactivated by the subsequent storms. For this model the excess sediment measured during the large storm was referred as "Large Storm Sediment" and the excess removed by the subsequent storms was referred to as "Post Storm Flush".

BUDGET DEVELOPMENT

Total Sediment Production

The first step in the sediment budget analysis was to obtain an indication of the suspended sediment associated with each turbidity-flow data pair. Several investigators (Brown,1973; Paustian,1979) have shown that turbidity can be used as an index of suspended sediment concentration in mountain streams. For this study the assumption was made that the turbidity was proportional to suspended sediment concentration and, as a result, the product of turbidity and instantaneous flow could then be used as an indication of instantaneous sediment production. Additional research (Beschta, 1980) showed that while the assumption of direct proportionality between sediment and turbidity may not be strictly correct, the sediment figures calculated on this basis would probably produce a conservative error. Nevertheless, the values obtained from the high flow conditions could differ significantly from direct sediment measurements. Consequently, the "sediment" values developed by this procedure were not converted to conventional sediment units to avoid comparison with studies using more direct sediment measurements.

"Flow Generated" Component

In order to isolate the flow generated component it was first necessary to identify the "flushed condition" and then attempt to establish the annual turbidity-flow relationship for this condition which could be used to calculate the amount of sediment attributed to the "Flow Generated" component. Several attempts were made to find a way to define the flushed condition which would consistently produce a "balanced" budget for each year of the study. The method used for this report involved finding monthly means for each flow class and using the annual minimum value of these monthy means as a indication of the "flushed" condition for that particular flow class. This method had the advantage of being rather easy to compute and it seemed to work reasonably well.

The first step was to sort the data by the flow classes shown in Table 1. Monthly means of the flow clases containing more than five samples were then plotted as shown in Figure 5. The minimum monthly means, representing the "flushed condition," were then plotted as shown in Figure 6 to obtain a family of curves representing the annual turbidity-flow relationship for the "flow generated" component. It should be noted that these curves show a decreasing pattern similar to the trend in Figure 4 which involved only the peak flows of the moderate sized storms.

Table 1. Flow Class Intervals Used to Sort Monthly Turbidity

Flow Class Number	Flow Interval CFS	Flow Class Number	Flow Interval CFS
1	1.0-1.	9	40-63
2	1.6-2.5	10	63-100
3	2.5-4.0	11	100-160
4	4.0-6.3	12	160-250
5	6.3-10	13	250-400
6	10-16	14	400-630
7	16-25	15	630-1000
8	25-40	161	1000-1600

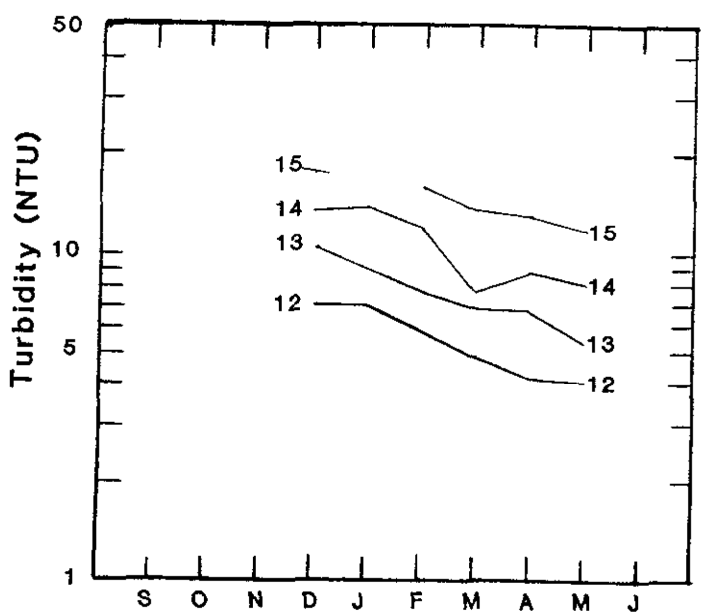


Figure 5. Mean monthly turbidity by flow class for year 1978-79.

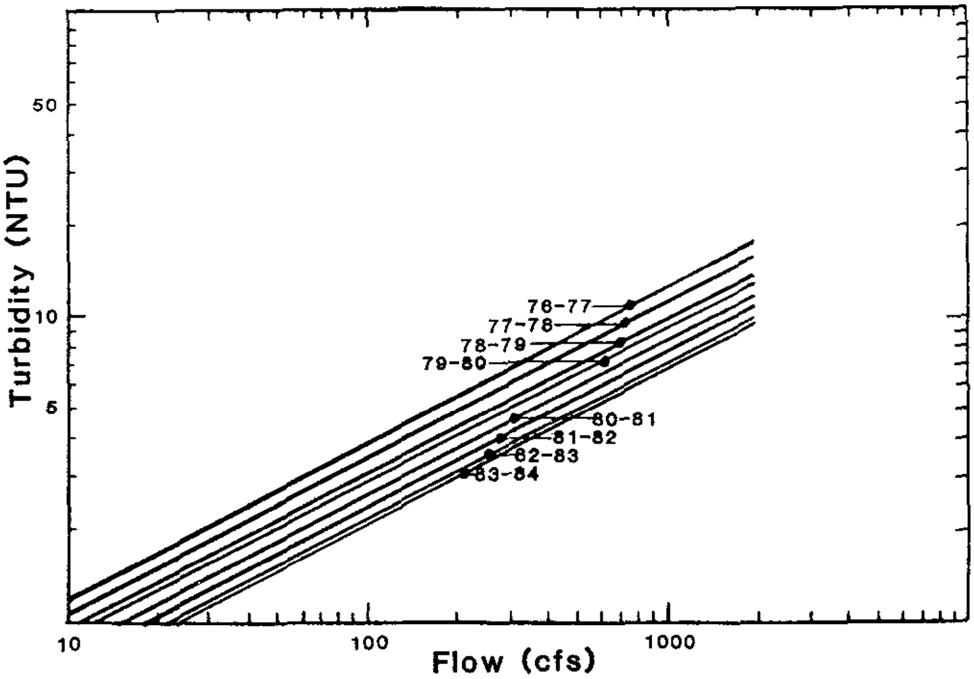


Figure 6. Annual turbidity - flow curves for moderate flows under "flushed" conditions. The general equation is: $T=KQ^{1/2}$ where K changes annually.

Since sediment concentration is assumed to be proportional turbidity the curves of Figure 6 indicate a sediment - flow relationship of the form:

$$\text{Sed} = KQ^{3/2}.$$

Where Sed is the "Flow Generated" budget component for one data point; K is an annual coefficient; and Q is streamflow.

These "Flow Generated"" values were then subtracted from the total sediment value for each data point to obtain the component attributed to other sources. Table 2 is a monthly summary of these calculations.

"Weathering and Decay" Component

The remaining components in the budget model are: (1) the "Weathering and Decay" and (2) the "Large Storm" components. The budget model dictates two conditions associated with these components. The first is that any year without a large storm will produce only the "Weathering and Decay" and the "Flow Generated" components. The other condition is that the quantity in the "Weather and Decay" component should remain relatively constant from year to year since the factors controlling these processes do not vary to the extent of the flow related processes. The results from the three study years that didn't experience large storms, 76-77,77-78, and 82-83 appear to support the model since, as Table 3 shows, the sediment attributed to this source remained fairly constant at a value of about 4 units.

Sediment Produced by Large Storms

To continue the analysis the additional assumption was made that the large storm years also experienced the same quantity of weathering and decay sediment. The sediment remaining after this component is removed was then attributed to large storm flow. The next step in the analysis was to calculate the sediment that was transported during the storm from the total storm sediment. An estimation of the sediment associated directly with the storm was calculated directly from the storm turbidity and flow data. Subtraction of this value from the remaining sediment in the budget gave an estimation of the "Post Storm" component.

Table 2. Monthly and Annual Sediment Production By Source Component. (Values are given as relative units.)

		Water Year							
		76-77	77-78	78-79	79-80	80-81	81-82	82-83	83-84
Oct.	Flow Gen.	0.2	0.1	0	0.8	0	0.2	0.5	0
	Other Source	0.3	0.1	0	1.8	0	0.3	0.4	0
	Total	0.5	0.2	0	2.6	0	0.5	0.9	0
Nov.	Flow Gen.	0.1	2.1	2.0	1.1	0.4	1.1	0.6	1.0
	Other Source	0.1	1.2	4.5	1.1	0.5	1.2	0.6	1.0
	Total	0.2	3.3	6.5	2.2	0.9	2.3	1.2	2.0
Dec.	Flow Gen.	0	6.1	4.0	1.3	3.0	5.1	3.4	6.7
	Other Source	0	1.8	12.7	1.2	3.4	13.1	0.6	8.5
	Total	0	7.9	16.7	2.5	6.4	18.2	4.0	15.2
Jan.	Flow Gen.	0.1	1.7	1.4	2.3	0.2	3.7	1.6	0.9
	Other Source	0	0	1.5	4.5	0.1	3.2	0.3	0.7
	Total	0.1	1.7	2.9	6.8	0.3	6.9	1.9	1.6
Feb.	Flow Gen.	0.3	2.5	5.0	0.2	1.5	4.1	2.9	3.1
	Other Source	0.9	0.5	19.3	0.1	1.2	1.4	0.7	6.4
	Total	1.2	3.0	24.3	0.3	2.7	5.5	3.6	9.5
Mar.	Flow Gen.	1.0	0.4	3.4	1.4	0.8	1.3	1.4	1.6
	Other Source	1.9	0	1.7	1.5	0.1	0	0.6	1.1
	Total	2.9	0.4	5.1	2.9	0.9	1.3	2.0	2.7
Apr.	Flow Gen.	1.3	1.0	3.9	1.5	1.3	3.1	1.1	1.6
	Other Source	0	0.8	2.2	0.4	0	0.7	0.1	0.3
	Total	1.3	1.8	6.1	1.9	1.3	3.8	1.2	1.9
May	Flow Gen.	2.5	0.7	0.1	0.6	0.8	0.1	0.6	0.9
	Other Source	0.5	0	0	0.2	0.2	0	0.1	0.1
	Total	3.0	0.7	0.1	0.8	1.0	0.1	0.7	1.0
June	Flow Gen.	0	0.1	0.1	0.5	0.7	0	0.1	1.8
	Other Source	0	0	0	.1	0	0	0.1	1.4
	Total	0	.1	.1	.6	0.7	0	0.2	3.2

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		76-77	77-78	78-79	79-80	80-81	81-82	82-83	83-84
Year Total	Flow Gen.	5.5	14.7	19.9	9.7	8.7	18.7	12.2	17.6
	Other Source	3.7	4.4	41.9	10.9	5.5	19.9	3.5	19.5
	Total	9.2	19.1	61.8	20.6	14.2	38.6	15.7	37.1

Table 3. Summary of Annual "Sediment" Budget 5/

Year	Annual Runoff Inches	Total "Sediment"	Sediment Budget Components			
			Flow Generated Component	Weathering and Decay Component	Large Storm Component	Post 4/ Storm Component
76-77	15.0	9.2	5.5	3.7	0	0
77-78	37.8	19.1	14.7	4.4	0	0
78-79	35.9	61.8	19.9	4 1/	18.5 3/	19.4
79-80	32.2	20.6	9.7	4 1/	3.5	3.4
80-81	27.8	14.2	8.7	4 1/	1.5	0
81-82	51.1	38.6	18.7	4 1/	5.6 2/	10.3
82-83	46.1	15.7	12.2	3.5	0	0
83-84	--	37.1	17.6	4 1/	10.3 3/	5.2

Notes:

1/ Assumed value based on production during years without large storms.
2/ Approximately 1/2 of storm data missing.
Actual value may be twice as large.
3/ Two large storms occured this year.
4/ This value is the difference between all other sources and total sediment.
5/ Sediment quantities are in relative units.

CONCLUSIONS

The data generated by the budget model provided a better understanding of the turbidity production of the Layng Creek watershed. Figure 7 shows the general form of the turbidity-flow relationship after the loosely held material has been flushed from the system. Determination of the turbidity-flow relationships for the "flushed" conditions as shown in Figure 6 confirmed the trend developed in Figure 4 from the preliminary analysis. Also, the values in Table 3, while not true sediment values, suggest that the "Weather and Decay" component is generally rather small compared to the other budget components. Also, plotting the rather limited data from the large storms as shown in Figure 8 indicates an improvement in this component as well.

It should be noted that the observed improvments in the various budget components reflect changes in the net response of the watershed and it is difficult to conclusively attribute these changes to specific management activities. However, the increased knowledge of the general trends as indicated by this study has been helpful in determining management direction. It is expected that an annual review of the peak flow/turbidity ratio and the response of the large storms will be sufficient to monitor future trends.

To improve the quality of future analysis, the monitoring site was moved to the intake location and continuous turbidity data will be taken. This change will provide an inproved indication of conditions at the intake, the main point of concern. Also, the use of continuous turbidity data will permit better analysis of the large storm data.

Since this method of analysis was developed specifically for the Layng Creek watershed, it may not work for other watersheds with different turbidity production characteristics. In particular, it would not work on watersheds which do not exhibit the "flushing" characteristics shown in Figure 3.

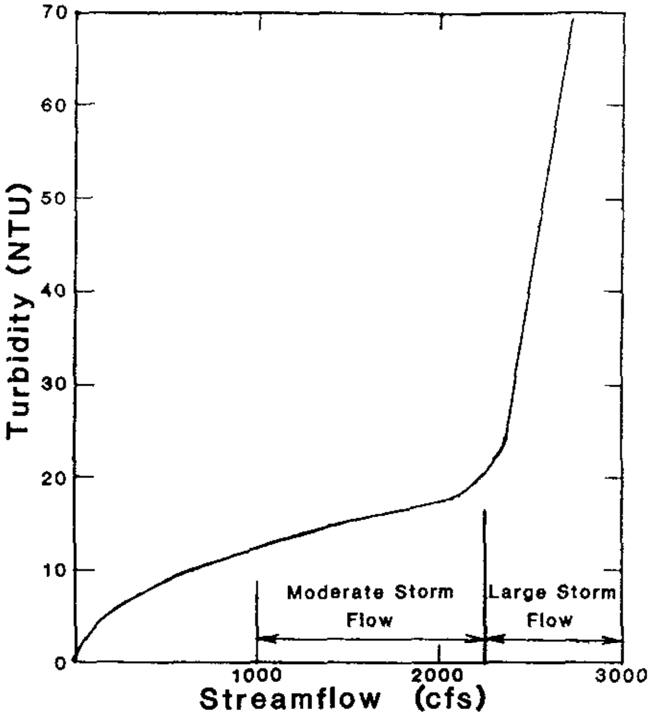

Figure 7. Model of turbidity and flow relationship after annual flush.

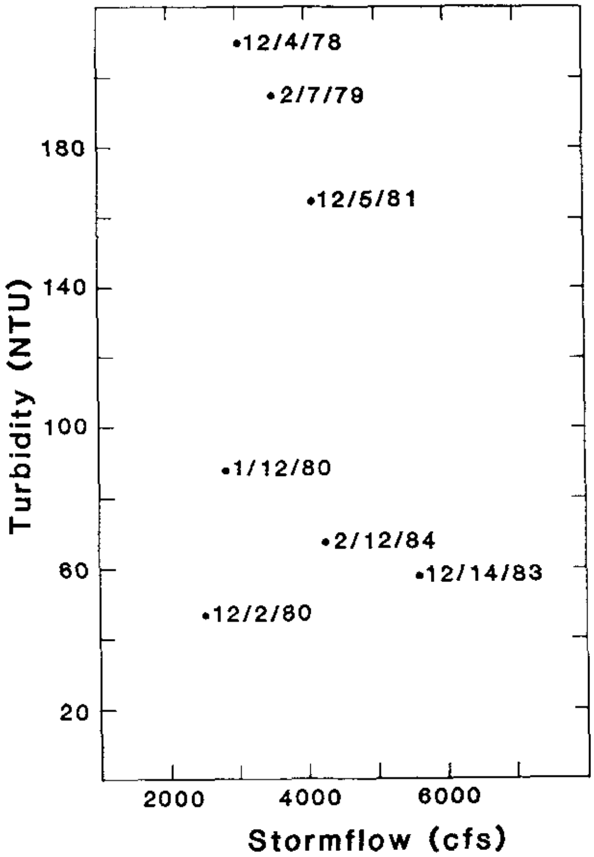

Figure 8. Peak turbidity - flow points for the large storms that occurred during the study period.

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ENVIRONMENTAL ASPECTS OF SEDIMENTATION

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ABSTRACT

Historically, the lower Colorado River annually transported over 180 million tons of sediment through the Parker Division, near Parker, Arizona. Since closure of mainstream dams, less than 2 million tons of sediment are transported out of the Parker Division each year. This present sediment load originates within the Parker Division due to lateral bank cutting and streambed degradation. Whereas the native aquatic community (under heavy sediment loads) was rather limited, the change in sediment transport has allowed a diverse, albeit exotic, aquatic community to develop.

INTRODUCTION

All aspects of the sedimentation process in running waters have pronounced effects on aquatic environments. When man-induced changes of river-born sediment occur, the change itself is easily determined, but its effect on the aquatic community is more difficult to measure. Suspended sediment levels which are harmful to one organism may often be favorable to another (EIFAC, 1964).

The Colorado River, prior to closure of mainstream dams, was often described as, "too thick to drink; too thin to plow." Closure of Hoover Dam (1935) and Parker Dam (1938) changed the sediment-laden Colorado into a relatively clear-water river. The physical and biological changes brought about by the desilting actions of these dams are looked at, herein, by examination of the 44-mile reach of the lower Colorado River known as the Parker Division.

STUDY AREA

For administrative purposes, the Bureau of Reclamation (Reclamation) divides the 265 miles of the lower Colorado River into ten operational divisions. Parker Division is a 44-mile reach between Headgate Rock Dam near Parker, Arizona and Palo Verde Diversion Dam near Blythe, California. Physical, chemical and biological surveys have been conducted by Reclamation within this division in preparation for operation and maintenance work conducted under the authority of the Colorado River Front Work and Levee System Act of 1927 (amended 1940, 1946, and 1958).

HISTORICAL CONDITIONS

Physical

Suspended sediment records indicate that before the closure of Hoover Dam in 1935, channel aggradation was active along the lower Colorado River. Between 1925 and 1935 an annual average of 212 million tons of suspended sediment were measured below Grand Canyon. Below the present site of Palo Verde Diversion Dam there was roughly 180 million tons measured and only 170 million tons reached Yuma, Arizona (Reclamation, 1969).

The river was actively building up areas such as Parker Valley, with the river channel frequently being higher than much of the valley bottom. Part of the sediment load transported by the spring/summer floods was carried out of the channel and deposited on the valley floor.

U.S. Geological Survey records for 1902-1903 show the river through Parker Valley as having a meandering channel with occasional braided reaches. Channel length at that time was 43.2 miles with an average slope of 1.7 feet per mile. The streambed was alluvial fill which had been graded and formed by sediment-laden flows. Dill (1944) characterized the suspended sediment historically carried through the lower river as mostly silt having the fineness of Portland cement. Only a little of the sediment could be classified as coarse sand and in most cases 50 percent passed a standard sieve of 200 meshes to the inch.

Streambed scour began throughout the division following closure of Parker Dam in 1938, and elevations declined an average of 1.27 feet per year. By 1944 a 4-foot decrease in water surface elevation at the intake for Palo Verde Irrigation District prompted the construction of Palo Verde Weir in 1945. The weir greatly reduced the rate of bed scour in the lower part of Parker Division.

Biological

Minckley (1979) reported only three native fishes from the Parker Division: bonytail chub (Gila elegans), Colorado squawfish (Ptychocheilus lucius), and razorback sucker (Xyrauchen texanus). While the latter two were believed to be more abundant than bonytail chub, none were thought to be abundant in swift, silt-laden main channel habitats. Minckley (1979) listed marshy backwaters and oxbow lakes as the major important features of the lower Colorado River prior to modifications. These areas provided food and shelter for young and adults of all three species. Trophic structure was simple and direct. Razorback sucker feed on the organic material (both plant and animal) within the bottom ooze of backwater and other low velocity areas; bonytail chubs fed primarily upon terrestrial invertebrates in backwater, sloughs, and channel margins; and Colorado squawfish fed on the young of all fishes, as well as opportunistic encounters with other vertebrates (Minckley, 1979).

Main channel habitats provided minimal food and therefore minimal fish habitat. Dill (1944) suggested four ways in which the heavy silt load of the lower river may have been deleterious to aquatic organisms: 1) lack of stability of streambed due to moving bed silt prevents establishment of fish food organisms, 2) the chemically inert character of the silt was not conducive to growth of aquatic organisms, 3) high turbidity precluded production of green aquatic plants, and 4) the mechanical effect of silt on aquatic organisms and their eggs must have been deleterious. Dill (1944) also credited off-channel areas as being major biological oases, stating, "Undoubtedly the bulk of the fish populations were found here before the dams were built."

PRESENT CONDITIONS

Physical

Following closure of Parker Dam sediment transport characteristics of the river changed little, and the energy formerly expended on transporting the sediment load was now expended in attacking the old channel bed. Construction of Palo Verde Weir in 1945 reduced scour and resulted in channel aggradation in the lower 10 miles of Parker Division. In the upper reaches the channel became entrenched. Sorting action resulted in the bed becoming more resistant to erosion and the river began attacking the banks. Bank cutting in many areas was estimated to be 45 feet per year (Reclamation, 1969).

Reclamation placed bankline riprapping, training structures, and dike fields in the upper 14 miles of Parker Division between 1967 and 1969. This action resulted in formation of 11 backwater areas and reduced or eliminated bank cutting in this upper section. This reach presently has a gravel-armored bed and banklines are mostly riprapped. Low velocity areas with soft sediments are limited to backwater habitats.

Immediately below this stabilized reach the river is again cutting laterally into the bankline. Channel bottoms are armoring along the thalweg and soft sediments are found in both backwater and drains and along the inside of meanders. Water transparency is high except in the immediate vicinity of bankline cave-ins and near the mouth of drains. This second reach covers roughly 10 river-miles.

The third reach within the division is classified as a transition zone. During periods of high river flow this 8-mile stretch shows characteristics of a degrading channel similar to the reach above it. During low flow periods the channel aggrades. Soft sediments accumulate in clumps or hummocks which become anchored by aquatic vegetation.

Below this transition zone the river bed is generally aggrading. The river channel has numerous wide areas with exposed sand bars, braided areas, and bankline marshes. Gravel bars and points are limited and localized in the vicinity of desert washes. While numerous slow-water areas are found within the channel, shifting sand substrates preclude development of weed beds. This aggrading reach is about 8 miles long.

The last four miles of the Parker Division show the effects of Palo Verde Diversion Dam, constructed in 1957 to replace the earlier rock weir. This reach is relatively deep with even-flowing water of moderate velocity. Considerable bankline sloughing occurs due to rapid water level fluctuations associated with sluicing at the diversion dam. Where banklines are vegetated with larger trees, considerable debris, snags and rootwads are found in the water at the river margins. Surface substrate is principally sand. Waters here are considerably more turbid than at most upstream reaches.

Biological

Aquatic habitats and their associated flora and fauna are both abundant and diverse in the present-day Parker Division (Minckley, 1979). Hard bottom areas in the main channel are colonized by filter feeding invertebrate organisms such as dipteran and trichopteran larvae and Asiatic clams. While the upper 14 miles of the division are primarily hard-bottomed, other nearby habitats, such as backwaters, provide quiet water areas with soft-bottomed substrates. Here again, Asiatic clams and dipteran larvae, as well as numerous other invertebrates, flourish to provide food for bait-, rough-, and game-fishes. Bait fish such as red shiner (Notropis lutrensis) are common in the main channel below Headgate Rock Dam where they feed on small drifting foods within the water column. Here and elsewhere in the division the shiners are food for largemouth bass (Micropterus salmoides), smallmouth bass (M. dolomieui), striped bass (Morone saxatilis), and other piscivorous fishes. Threadfin shad (Dorsoma petenense) are abundant in and near backwaters and in open areas of the channel where they are also fed upon by piscivorous fishes, bullfrogs, turtles, and fish eating birds. Recent aquatic surveys of the division have captured at least 18 different fish species (Minckley, 1979 and Reclamation, 1984).

DISCUSSION

The effects of mainstem dams on the aquatic ecology of the lower Colorado River are exemplified in the Parker Division. Physical changes are pronounced, and biological changes are tremendous. Historically the river flowing through the Parker Division was a rather monotonous ribbon of moving sediment, highly turbid and lacking substrate diversity. Fish ecology was both simple and unique. The channel of the largest river in the American Southwest had an assemblage of riverine fishes consisting of only three species (Minckley, 1979).

Closure of mainstem dams trapped sediments in newly formed reservoirs. Freed from its sediment load, the river scoured and sorted streambed substrates within the Parker Division. Construction of Palo Verde Weir (and later Palo Verde Diversion Dam) resulted in a hydraulic null area or transition zone to develop in the middle reaches of the division. Upstream of this zone the channel was armored or degraded, while downstream areas were aggraded. Now a wide range of streambed substrates of varying degrees of stability became available for plant and animal colonization.

Reduction of suspended sediment combined with both streambed sorting and organic material transport from upstream reservoirs offset the historical negative components for fish production given by Dill (1944). Fish food organisms were able to establish populations on and along the streambed; chemically inert silt was replaced in the water column by fine organic material from upstream reservoirs and this provided a food source for both benthic invertebrates and forage fish; reduced turbidities allowed light, necessary for primary production of green plantlife, to penetrate through the water column; and deleterious effects of suspended sediment upon aquatic organisms and their eggs were eliminated. Introduced (non-native) fishes flourished in these newly created habitats.

The rather simple ecological relationships between three riverine fish populations has been replaced by a complex food web containing at least 18 species of fish, as well as numerous other vertebrate animals.

Whether the replacement of the native fishes by 18 introduced fish species is good or bad is conjectural, if not dogmatic. Why this occurred appears to be a direct result of reduced sedimentation within the lower Colorado River.

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