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ANALYSIS OF POTENTIAL IMPACT TO A TROUT POPULATION
FROM FLOW DEPLETION IN MOYER SPRINGS CREEK
CAMPBELL COUNTY, WYOMING

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ABSTRACT

The worst case projection predicts a 27% flow decline in Moyer Springs during mining operations, reducing average flows from 1.27 cfs to .92 cfs. Without any mitigation or enhancement measures, the net effect would be a slightly more rapid increase in water temperature during the summer months as the stream flows from the spring sources. This effect would result in a slight upstream translocation of the marginal temperature zone for trout and slightly favor the native species of minnows and white sucker in the downstream area. No oxygen problem is anticipated. Physical habitat conditions would be essentially unchanged because of the low stream gradient. Gage height-flow data reveal a 27% flow reduction would reduce the surface level of the stream by less than one inch. A 27% reduction would provide an average daily flow 73% of the long term average flow--an optimum fishery flow.

With mitigation and enhancement measures, including the establishment of riparian vegetation to shade the stream and maintain lower water temperatures, structural devices to improve habitat conditions for various life history stages, and flow augmentation, if deemed beneficial, the abundance of the present trout population in Moyer Springs Creek could be greatly increased.

INTRODUCTION

Moyer Springs Creek, a tributary to the Dry Fork of the Little Powder River, Campbell County, Wyoming, is reputed to be the only perennial trout stream in the county and this unique attribute has stimulated considerable concern for possible environmental consequences that might result from a coal mining operation in the watershed. The major impact predicted from environmental assessment studies is a maximum depletion of spring flows by 27% during mining. The small stream, maintained by the springs presently contains a population of introduced brook trout, *Salvelinus fontinalis*, in the cooler upstream area and native longnose dace, *Rhinichthys cataractae*, lake chub, *Couesius plumbeus*, and white sucker, *Catostomus commersoni*.

CHARACTERIZATION OF PRESENT ENVIRONMENT

The source of perennial flow originates in Moyer Springs at about 4275 feet elevation. The stream courses about 2.5 miles to join the Dry Fork Little Powder River at about 4230 feet elevation. The gradient is gentle, dropping little more than 20 feet per mile; or about .4%. The stream averages about 6 feet in width with a total water surface area of about 1.9 acres. It provides good fish habitat with abundant aquatic macrophyte vegetation in the upstream reaches with deep pools and undercut banks as prominent features downstream. The streambanks lack woody riparian vegetation for shade, however, and the water warms relatively rapidly towards the junction with the Dry Fork due to incident solar radiation. Summer temperatures of 22°C (72°F) or more are reached near the mouth which are marginal for trout, but favorable to the native species. Water temperatures in July, to within a about a mile of the source springs, remain cool, about 13°C (55°F), an ideal temperature for trout feeding and growth. Gage recordings indicate continual input from springs along the route of flow. The upstream gage indicates an average flow of about .8 cfs and the flow near the mouth averages 1.27 cfs. The flow is extremely stable. The small size of the watershed minimizes the effects of precipitation. Generally, the runoff from a storm event can produce a brief spate that doubles the normal flow. The limited sampling data indicates a healthy brook trout population in the upper reaches of the creek along with longnose dace. Near the mouth, the white sucker is the dominant species.

The watershed is grazed by livestock. Livestock grazing probably restricts the establishment of shading riparian vegetation.

POTENTIAL IMPACTS

The following assessments are made assuming a worst case situation--a 27% flow depletion without mitigative or enhancing measures.

Temperature. The lack of shading vegetation and low stream gradient, resulting in low velocity flow, combine to optimize the warming of the water from solar radiation during the summer months. Marginal water temperatures with daily maxima from 21 to 25°C (70 to 77°F) are not lethal to brook trout but place them at a competitive disadvantage with warm water species such as suckers and chubs. The extension of the "warm-water" zone upstream due to flow depletion would favor the native fishes over the brook trout to some extent. A more precise quantification of this impact is not possible without before and after monitoring of stream temperatures and air temperatures at several sites along the stream correlated with fish sampling data. It can be roughly estimated, however, that present trout abundance and biomass may decline about 10 to 15% overall and native fishes would increase about 25% due to their lower trophic levels and higher production potential.

A solution to this problem would be the establishment of shading riparian vegetation that would extend the cool water environment downstream to the mouth.

Oxygen. There are no data available to me concerning the water quality parameters of the spring flows as they come out of the ground. Unless the Moyer Springs are highly unique in comparison to most springs, however, their waters contain little or no dissolved oxygen and high concentrations of carbon dioxide. Spring flows, at their source, are typically uninhabitable to fishes because of the lack of oxygen and/or high carbon dioxide levels. Gaseous exchange with the atmosphere and photosynthesis by plants create a suitable oxygen-carbon dioxide regime for fishes some distance downstream from a spring source, depending on the volume of flow. If the gas regime of Moyer Springs are comparable to most springs, a reduction in flow would not reduce dissolved oxygen levels in the stream; on the contrary, reduced volume of spring flows would cause the oxygen-carbon dioxide equilibrium to be attained more rapidly, nearer to the spring sources and extend the zone of the habitable trout environment upstream.

The only possible oxygen depletion problem from reduced stream flow concerns situations where dilution flows are needed in zones with excessive loads of organic matter such as a stream receiving raw sewage or coursing through a feedlot causing oxygen depletion from decomposition and respiration leading to fish kills unless adequate dilution with highly oxygenated flows are available. I can find no evidence of sources of excessive organic matter in Moyer Springs Creek nor any indication in any data that flow reduction will result in lower levels of dissolved oxygen in the stream.

Physical Habitat. The Wyoming Game and Fish Department long ago realized the threat to trout habitat from flow depletions and the department has established a leadership position in trout habitat research. This research has been led by Dr. Allen Binns (Binns 1979, Binns and Eiserman 1979). In conjunction with the Game and Fish Department, the Wyoming Water Resources Research Institute at the University of Wyoming, has conducted several research projects to quantify the correlation of trout habitat quality with various stream flows (Wesche 1973, 1974, Wesche and Rechard 1980). The critical limiting factor is the summer base flow. When the base flow falls "too low", water levels recede to a point that the prime trout habitat associated with undercut banks is lost. For most streams the "too low" point is reached when flows are 20 to 30% of the long term average daily flow. Dr. Binns found that the annual flow regime, of all habitat factors, was the major determinant of trout abundance. The best trout streams in Wyoming were those that exhibited the most stable flows and maintained a summer base flow of at least 55% of the average daily flow.

A flow reduction of 27% in Moyer Springs Creek would result in a flow of 73% of the average daily flow. Data from the two gaging stations on the creek indicate a 27% flow reduction lowers the stream surface by .8 of one inch at the lower gage and by .9 of an inch at the upper gage. This small decrease in stream surface level would not have a significant impact on the physical habitat characteristics of the stream. This conclusion is in agreement with predictions based on the Wyoming trout habitat research cited above.

MITIGATION, ENHANCEMENT SUGGESTIONS

If the major emphasis is placed on creating conditions that favor trout over the native fishes, then the most important objective would be to extend the zone of cool (less than 70°F) water downstream to the mouth of the creek during the summer months. This could only be accomplished by the establishment of abundant shading riparian vegetation, such as willow, aspen, or alder, to greatly reduce the incidence of solar radiation. It would probably be impossible to establish suitable plants with livestock grazing pressure. The riparian zone would have to be fenced to exclude livestock. The feasibility of planting and fencing could be tested by treatment of a short section of stream to observe the results.

Otter Creek, Nebraska, is a small spring fed creek draining to Lake McConaughy. The Nebraska Game and Parks Commission desired to improve the Otter Creek environment for trout. The stream bottom land was leased and a fence constructed to exclude livestock. Prior to fencing, livestock had virtually eliminated the riparian vegetation and water temperatures warmed rapidly downstream from the spring source. Before livestock were excluded from the riparian zone, the fish composition of Otter Creek was 1% rainbow trout (*Salmo gairdneri*), 17% brown trout (*S. trutta*), 22% white sucker, and 60% creek chub (*Semotilus atromaculatus*). After fencing, Otter Creek maintained lower summer temperatures and turbidity and siltation were reduced. Riparian vegetation flourished, the species composition changed to 97% rainbow trout, 2% brown trout, and suckers and chubs made up 1% of all fishes (Van Velson 1979).

Other measures for mitigation and enhancement of the trout population include flow augmentation from sediment ponds to replace the amount of flow lost from the springs. If the water from the sediment pond were to be taken from near the bottom and piped underground to the creek, water temperatures during the summer should remain below 70°F. If spring flows at 50°F made up 73% of the flow and pond water at 70°F were 27% of the flow, the maximum summer temperatures should be in the 55° to 60°F range near the source. If shading vegetation were to be established along the entire length of the stream, optimum to good water temperatures should extend to the mouth.

Modification and manipulation of the physical habitat can greatly increase trout abundance if the population is habitat limited rather than food limited (Behnke 1981). That is, a stream may produce a surplus of food that is unutilized by the trout population because of a lack of suitable habitat (areas of deep, low velocity water with adequate protective cover, such as undercut banks, pockets associated with boulders, logs, tree roots, etc.). In such situations, the creation or improvement of optimum habitat sites will result in an increase in trout abundance.

Before any extensive mitigation or enhancement measures are decided upon, experienced trout stream biologists should make a qualitative evaluation of the potential for improvement. For example, a knowledgeable biologist such a Dr. Binns of the Wyoming Game and Fish Department and a biologist experienced in trout habitat research representing the mining company could critically examine the creek for one or two days to qualitatively assess the following habitat components and limiting factors: Overall habitat quality in relation to a food limited vs. a habitat limited population, spawning habitat, rearing habitat for young-of-year and yearling fish, adult habitat, and overwinter habitat.

Soundly based recommendations for mitigation and enhancement could then be made. I would caution against a detailed quantitative research project designed to precisely determine the effects of a flow depletion such as the habitat models developed by the U.S. Fish and Wildlife Service (Instream Flow models and Habitat Evaluation Procedure [HEP] models). These models would not likely contribute much to what is already known or apparant in this situation and they hold the danger of erroneous (although quantitative) conclusions. This is due to the fact that not only habitat, but a species niche quantification is attempted. The niche of a species is a complex concept. It is an "n dimensional hypervolume" (Hutchinson, 1957). As such the niche is not static but expands and contracts under influence of biotic and abiotic factors and is especially influenced by interaction with the niches of other species (fundamental niche vs. realized niche). There is no way we can know all of the necessary information to accurately portray a niche mathematically. At best we can obtain a highly simplified abstract of nature that may be essentially correct for a particular situation (if lucky). A more qualitative, but critical and holistic interpretation by experienced and competent biologists would produce more efficaceous results and recommendations at far less cost to resolve the problem of the potential flow depletion in Moyer Springs Creek.

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A PROCEDURE AND RATIONALE

For

"SECURING FAVORABLE CONDITIONS OF WATER FLOWS"

On

NATIONAL FOREST SYSTEM LANDS In NORTHERN WYOMING

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SYNOPSIS

Rationale and procedures are presented for determining those streamflows needed as an integral part of management of National Forest System lands in the Big Horn River Basin, Wyoming, for the purpose of "securing favorable conditions of water flows." The procedures focus on streamflows required to maintain stability of the stream channel systems for the orderly conveyance of water from and through the National Forest.

The streamflows necessary for the maintenance of stream channel stability include "bankfull" discharge, a range of flows representing the rising and recession limbs of the hydrograph, and a "baseflow" discharge. The magnitude and durations of these discharges were determined for selected stream systems by hydraulic geometry measurements, drainage basin characteristics, and regionalized dimensionless flow-duration curves developed from long-term streamgage records within the Big Horn River Basin.

The required streamflows approximate the rising and recession limbs of the snowmelt hydrograph. This involves a series of flows which are distributed over a 69-day period starting at a flow comparable to the mean annual discharge, increasing to bankfull discharge for 3 days, and decreasing back to mean annual discharge. In addition, a "baseflow" discharge is required, which is approximately 11 percent of the mean annual discharge, or 1.7 percent of bankfull discharge. These

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flows represent approximately 78 percent of the total average annual water yield. Being non-consumptive in nature, these flows are available for both historic and new downstream appropriation.

These flows are necessary to provide for the self-maintenance of stream channel networks so as to retain their capability for passing flood flow discharges, and to reduce excessive channel erosion and/or sediment deposition associated with stream channel instability or disequilibrium conditions.

The potential consequences of channel instability that may develop as a result of streamflow regulation are presented including channel aggradation, channel erosion, floodplain encroachment, vegetation encroachment, changes in the hydraulic geometry of stream channels, and reduced channel capacity with resultant increased hazard for flooding and associated resource damage.

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A PROCEDURE AND RATIONALE For

"SECURING FAVORABLE CONDITIONS Of WATER FLOWS"

On NATIONAL FOREST SYSTEM LANDS In NORTHERN WYOMING

INTRODUCTION

The Organic Administration Act of June 4, 1897 (16 U.S.C. 475) authorized the establishment of National Forests to improve and protect them for the purpose of securing favorable conditions of water flows, and to furnish a continuous supply of timber. This authorization recognized that forests exert a most important resulting influence upon the flow of rivers. An important aspect of management in "securing favorable conditions of water flow" is the maintenance of stream channel stability and equilibrium.

Stability of stream channel systems

"Every river appears to consist of a main trunk, fed from a variety of branches, each running in a valley proportioned to its size, and all of them together forming a system of valleys connecting with one another, and having such a nice adjustment of their declivities that none of them join the principal valley either on too high or too low a level; a circumstance which would be infinitely improbable if each of these valleys were not the work of the stream which flows in it," from Playfair's classic law of streams, 1802 (Horton, 1945).

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The existing form and characteristics of rivers have developed in a predictable manner as a result of the water and sediment load imposed from upstream. Natural stream channels are self-formed and self-maintained. Their shape and size develop consistent with the basin's character and area. Hydrologic alterations within the basin, including changes in stream discharge, often lead to accelerated stream channel erosion and deposition of sediment in the channel creating the potential for reduced capacity for flood flows which, in turn, endangers downstream values.

A range of streamflows are necessary to maintain the form and characteristics of existing streams for their proper functioning. Fundamental concepts developed from research, actual stream gaging data, measured hydraulic geometry of stream channels, and drainage basin characteristics can be used to quantify these necessary "channel maintenance" streamflows.

Stable alluvial stream channels clearly exhibit certain consistent morphological characteristics which have developed over time from an integration of erosional processes and dominant stream discharges. Geomorphic evidence for the existence of such a condition of equilibrium between channel form and the independent variables of discharge and sediment load was presented by Leopold and Maddock (1953). They stated, ". . . the average river-channel system tends to develop in a way to produce an approximate equilibrium between the channel and the water and sediment it must transport. Thus, approximate equilibrium appears to exist even in headwater, ungraded tributaries, and in a given cross section for all discharges up to bankfull stage."

Equilibrium or a ". . . stable channel balance may be described as a balance that exists between the eroded material supplied to and stored in the stream channel and the energy available (streamflow) to transport the material" (Rosgen, 1980). Lane (1955) initially described a stable channel "balance" as one which shows a proportional relationship of sediment load and material size to stream discharge and stream slope (Figure 1). In addition, a channel is considered to be "in regime" (stable) if it can accommodate its flow over a period of years without a net change in hydraulic characteristics (Blench, 1969). Schumm (1973), in his presentation of the processes and feedback mechanisms which affect the response of river channels to flow regulation, emphasized the importance of a competent flow for maintaining the geomorphic thresholds of stream channels.

Human alteration of natural stream systems and the subsequent effects on stream-channel geometry and related flow conditions have been studied and evaluated for many years. To assure the proper function of the natural water conveyance system described, the maximum benefits of available water resources must be balanced with the stability of the system from which these resource benefits are derived.

Hydrology literature contains many studies which support the need for stream channel stability maintenance. For example, morphological

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changes (altered stability) of river channels due partly to streamflow changes have been documented by Daniels (1966), Orme and Bailey (1971), Mundorff (1967), Gregory and Park (1974), Wolman (1967), Leopold (1973), Dury (1973), Emmett (1974), and Einstein (1961). Petts (1979) summarized interpretations by Gregory (1976, 1977) and Hollis (1979) by describing the relationship between alterations in natural stream systems and resultant stream channel changes.

Hydraulic Geometry-Discharge Relationships

It has been well demonstrated that the size, shape, and pattern of river channels are closely related to the flow which they transmit (Leopold, et al., 1964). Hydraulic geometry has been used for many years for estimating both water yield and flood peaks. Methods of quantifying the interrelations between the flow characteristics of rivers and channel size have long been utilized by practitioners where actual streamgage data are not available. Leopold and Maddock (1953) described downstream hydraulic characteristics of stream channels as a function of a constant-frequency discharge such that, for width, depth, and velocity:

$$W = aQ^b$$

$$D = cQ^f$$

$$V = kQ^m$$

where: W = width; D = mean depth; V = mean velocity; Q = constant-frequency discharge, and a, c, k, b, f, and m are numerical constants.

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These basic concepts are shown in Figure 2, where the hydraulic geometry of stream width, depth, and mean velocity are related to mean annual discharge for various streams in the Big Horn River Basin. The most consistently employed hydraulic geometry parameter is bankfull width; for it, more than any other easily measured channel characteristic, is correlated with flow parameters having specific recurrence intervals (Dunne and Leopold, 1978). Thus, the shape and hydraulic character of individual stream reaches can be defined and related to specific discharges which are consistent over a wide range of conditions involving similar geographical areas. One particular characteristic of stream channels, uniform over a wide range of conditions, involves dimensionless rating curves where the ratio of mean cross-section depth to mean bankfull depth is related to the ratio of mean daily discharge to bankfull discharge for streams in the eastern United States and Idaho (Figure 3). This relationship is utilized in determining potential for flooding based on channel depths for various flood discharges for various recurrence intervals. The difference in depth/discharge relationships due to basin area can be eliminated if the rating curve is expressed in terms of dimensionless ratios (Dunne and Leopold, 1978).

Studies by the U.S. Geological Survey in most of the western states have tested the relationship of width and depth of stream channels at point bars in meandering channels to the mean annual discharge of these drainages (Hedman, et al., 1972). The relationships and statistical reliability of these methods are consistent to the degree that engineers have been able to utilize these procedures for discharge determinations on ungaged streams over widespread areas. Similarly, Leliavsky (1955) used the regime theory to describe empirical relations of hydraulic properties, where approximate equilibrium could be expressed as exponential functions of discharge.

Bankfull Discharge

The discharge at which a large portion of effective channel maintenance occurs is that which occurs at bankfull stage. Bankfull discharge is defined as "that water discharged when stream water just begins to overflow onto the active floodplain; the active floodplain is defined as a flat area next to the channel, constructed by the river, and overflowed by the river . . ." (Wolman and Leopold, 1957).

A commonly accepted principle of geomorphology is that channel geometry is influenced by a formative or dominant discharge that has a fairly frequent recurrence interval. Much of the early research work involving stream channels was oriented towards expressing channel dimensions as a function of the formative or dominant discharge. In recent years, investigative efforts have focused on using the dimensions of stream channels as indices of flow characteristics, particularly flood flows. An analysis by Wolman (1955) of Brandywine Creek in Pennsylvania related bankfull discharge to a wide range of channel-geometry properties, including bankfull width.

According to Leopold, et al. (1964), "The channel is formed and reformed during a range of flows lying between the lower limit of

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competence and an upper limit at which the flow is no longer confined within the channel. The range so defined consists of flows which reoccur more often than once a year or once in 2 years. Although great erosion does occur during exceedingly large flows in streams with developed meander patterns and floodplains, the channel forming discharge does not seem to be associated with the infrequent or catastrophic events."

Bankfull discharge is the discharge at which sediment movement, forming or removing bars, forming or changing bends and meanders, and generally "doing" work results in the average morphologic characteristics of channels (Dunne and Leopold, 1978). Very large hydrologic events are too infrequent to govern channel characteristics. It is the intermedite size flow that is sufficiently frequent so that the product of its frequency and magnitude of forces determines the effectivess for channel-forming characteristics (Wolman and Miller, 1960). This principle is shown graphically in a river mechanics example (Figure 4) where the relative effectiveness (line C) is the product of the frequency of discharge (line B) and the sediment transport rate (line A). Thus, the range of intermediate discharge, including bankfull, transports the largest part of the average annual sediment load.

Having described the significance of bankfull discharge, the question that next arises is how often does it occur? The average recurrence interval of bankfull discharge in the Yampa River Basin in Colorado

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and Wyoming is approximately 1.5 years (Williams, 1978a) with a duration of occurrence that ranged from 1.5 to 11 days per year or, on the average, 5.8 days per year (Andrews, 1980). The recurrence interval of flows that equaled or exceeded bankfull discharge on Brandywine Creek, Pennsylvania, ranged between 1 and 3 years (Wolman, 1955). In another study, Wolman and Leopold (1957) determined the discharge at bankfull stage for a number of streams in the United States. They found the annual peak ". . . will equal or exceed the elevation of the floodplain nearly every year."

Nixon (1959), in a study in England and Wales, concluded that discharge at bankfull stage was that which occurred at the 0.6 percent point on the flow duration curve based on the 1.5-year recurrence interval. This is the discharge that is equaled or exceeded, on the average 2.2 days per year.

Studies by Dury (1961) on the White and Wabash Rivers indicated that the recurrence interval of bankfull discharge was 1.1 years. Emmett (1972 and 1975) found from studies both in Alaska and Idaho that the recurrence interval of bankfull stage averaged about 1.5 years. Leopold, et al. (1964) included that the recurrence interval for bankfull state appears to be in the range of 1 to 2 years, or averaged to a 1.5-year value.

Bankfull discharge recurrence intervals for a number of streams, with drainage areas from 4.3 to over 13,000 square miles, are shown in

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Table 1. The mean recurrence interval value averaged 1.57 years. According to Leopold, et al. (1964), "The work of perennial streams in scour and fill and in transport of debris is accomplished principally by flows near or above bankfull state which occur less than 0.4 percent of the time or roughly once a year...thus, it appears that the channel and pattern forming discharge is one which reocurrs frequently."

Hydraulic geometry measurements were utilized in comparing the 1.5-year recurrence interval discharge (bankfull discharge) to field surveys of bankfull stage is shown in Figure 5 (Dunne and Leopold, 1978). Reasonable agreement as shown in this relationship for basins whose discharges range through three orders of magnitude.

According to Emmett (1975), "Bankfull discharge tends to have a constant frequency of occurrence among rivers, and discharges equal to a given percentage of bankfull discharge also appear to have a given frequency of occurrence." In relation to this, on the average, bankfull discharge is approximately 10 times "average annual discharge" and "baseflow discharge" is approximately 2 percent of bankfull discharge.

The evidence suggests, therefore, that the bankfull discharge in a river or stream will be equaled or exceeded 2 out of 3 years on the average.

Table 1 Observed bankfull flows, measured discharge, and computed recurrence interval (Leopold, et al., 1964)

River and Location	Drainage Area (sq. mi.)	Mean Annual Discharge (cfs)	Maximum Flow of Record: Discharge (cfs)	Maximum Flow of Record: Gage Height (ft.)	Bankfull Flow: Discharge (cfs)	Bankfull Flow: Gage Height (ft.)	Bankfull Flow: Recurrence Interval (yrs.)
Wabash River near New Corydon, Indiana	258	202	4,690	17.59	1,240	14.14	1.1
Eel River at North Manchester, Indiana	416	372	7,500	14.00	2,160	7.23	1.2
Wabash River at Delphi, Indiana	4,032	3,490	145,000	28.4	9,550	8.92	1.07
Wildcat Creek at Owasco, Indiana	390	366	10,000	13.3	3,390	7.58	1.7
Wabash River at Covington, Indiana	8,208	6,979	200,000	35.1	20,900	15.84	1.1
Wabash River at Riverton Indiana	13,100	10,880	250,000	26.4	31,900	16.02	1.24
Fall Creek at Millersville, Indiana	313	264	22,000	16.3	3,400	8.73	1.48
Bean Blossom Creek at Ben Blossom, Indiana	14.6	16.3	1,720	11.42	1,090	8.40	1.9
White River at Newberry, Indiana	4,696	4,475	130,000	27.5	14,700	12.77	1.1
Youngs Creek near Edinburg, Indiana	109	104	10,700	13.4	1,270	7.08	1.1
Driftwood Creek near Edinburg, Indiana	1,054	1,142	34,500	16.55	10,100	13.21	1.3
Trempealean River at Arcadia, Wisconsin	552				2,230	5.21	1.4
Wapsipinicon River near Dewitt, Iowa	2,330	1,320	26,000	12.07	5,080	9.36	1.15
Wahoo Creek at Ithaca, Nebraska	272	53.4	18,900	22.34	±2,500	±20.00	±1.0
Silver Creek at Ithaca, Nebraska	72	9.14	2,450	12.22	450	9.50	2.0
Nemaha River at Falls City, Nebraska	1,340	659	51,400	27.44	32,000	26.2	4.0
Powder River near Baker, Oregon	219	111	1,820	7.05	946	5.26	3.0
Watts Branch near Rockville, Maryland	4.3				430	4.05	2.2
Bogue Chitto Near Tylertown, Mississippi	502	875	45,700	33.50	7,000	16.0	1.2
Clear Creek at Bovina, Mississippi		36			2,770	21.73	1.25

Mean Recurrence Interval = 1.57 years

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Sediment Transport-Discharge Relationships

Although the mechanisms of sediment transport are extremely complicated and not entirely understood, there are some generally accepted concepts utilized by engineers and hydrologists which apply in this study.

As mentioned earlier, the stream channel stability balance is a function of the energy available for the transport of sediment and the sediment available to such transport (Figure 1). Lane (1955) described this "balance" as a proportionate relationship between sediment and water discharge using the equation:

$$Q_s D \propto Q_w S$$

where:

Q_s = sediment discharge

Q_w = water discharge

D = sediment particle size

S = stream slope

Any changes on one side of the proportionality creates an adjustment on the other to maintain equilibrium. Thus if frequenty-occurring stream flows were reduced, sediment supply and/or channel grade line would have to change to prevent channel aggradation. In most cases regulated flows in alluvial channels result in stream aggradation -

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because sediment supply is largely controlled by tributary input, channel storage, and the erosion of stream adjacent slopes. When the sediment supply is greater than the system's transport capability, a series of channel adjustments occur. These adjustments involve lateral migration of the channel, encroachment onto the active flood-plain during normal flood events, and damage to bridges and other structures.

It has been found that in natural channels, sediment load varies roughly as the square of the discharge (Heede, 1980). This is represented by the equation:

$$Q_s = K_a Q_w^{2}$$

where:

Q_s = sediment load in tons per day

Q_w = water discharge in cubic feet per second

K_a = a coefficient that changes from various streams

A temporal relationship for a mountainous snowmelt stream is shown in Figure 6 where sediment concentration is related to water discharge (Guy, 1970). This figure indicates the close relationship of variations in sediment concentrations due to natural variations in water discharge. Even at baseflow, sediment concentrations are influenced by water discharge. It may also be noted that the bulk of the

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sediment yield (product of water discharge x sediment concentration) occurs in a 2-month period representing the "rising, bankfull, and recession portions of the annual snowmelt hydrograph."

The size of sediment in transport is also important because it is related to stream discharge and channel stability. Andrews (1980) displayed the relation between discharge and grain size of bed material at the threshold of transport for the Little Snake River near Dixon, Wyoming, (Figure 7). The particle size ranged from 6 mm. at mean annual discharge to 17 mm. at bankfull. Extrapolating this curve, the potential grain-size for transport at baseflow discharge is approximately 2.0 mm., which is very coarse sand.

Another fundamental concept of sediment transport is that of stream power or rate of doing work. This expression of transport capability is a simple product of three terms; discharge, water surface slope, and a constant (Dunne and Leopold, 1978). This relationship takes the form:

$$\omega = \gamma \bar{\mu} d s$$

where:

ω = stream power in kilograms per meter of channel width per second

γ = unit weight of water in kilograms per cubic meter

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$\bar{\mu}$ = mean velocity in meters per second

$\bar{d}$ = mean depth in meters

s = river slope

The unit bedload transport rate (i_b) (in kilograms per meter-second) related to unit stream power (ω) shown in Figure 8 is taken from bedload transport data measured on the East Fork River, Wyoming, (Leopold and Emmett, 1976). Using this relationship, a reduction in flow would cause a similar reduction in the unit stream power and the corresponding transport rate. This relationship indicates the volume as well as the size of bed material that would be deposited in the channel.

Data from Andrews (1980) indicates that the initiation of bedload sediment transport on the Little Snake River near Dixon, Wyoming, occurred under moderate discharges. One might ask, at what discharges does the initiation of bedload sediment transport occur? In order to answer this question, the velocity at which motion begins for non-cohesive grains is needed. To deal with this question, a conversion was made of the basic Shields Diagram (ASCE, 1975, p. 96) by Sagan and Bagnold (1975) for sand and gravel particles. Their conversion involved the dimensionless functions Rg and θ of the Shields Diagrams:

$$Rg = \frac{\mu_* D}{\nu}$$

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and:

$$\theta = \frac{\mu_*^2}{(\frac{\sigma - p}{p})gD}$$

where:

Rg = Reynolds number of the grain

μ_* = shear velocity

D = grain diameter

ν = kinematic viscosity

θ = entrainment function

σ = density of the grain

p = density of the fluid

g = local acceleration due to gravity

By combining these two dimensionless functions and solving for critical shear velocity (μ_*), they derived the equation:

$$\mu_* = \left[Rg \ \nu \ \frac{\sigma - p}{p} \ g\theta\right]^{1/3}$$

Values of Rg and θ were then read from the Shields Diagram, computing corresponding values of D and μ_*, which are plotted in figure 9. This

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simplified version enables a determination of the sizes of material entrained under various critical velocities.

Streams also have the potential for sediment transport capability during low flows in order to maintain low flow features such as the winding thalweg, inner berms, and riffle-pool sequence. Leopold's (1969) study of the rapids and pools of the Colorado River revealed a condition where the migration of sediment from a riffle to a pool occurred during summer low flows. This is shown in Figure 10, where the elevation of the stream bed is decreased at very low flows and associated velocities. This same phenomenon was observed in the East Fork River, Wyoming, where sand-size particles were moved under baseflow stage from riffles to pools (Leopold, personal communication, 1980).

Streambank Erosional Processes

Many of the processes associated with bank erosion are controlled to a large degree by the full range of streamflow components. The natural rate and magnitude of stream or channel bank erosional processes are influenced by both the processes of fluvial entrainment and the actual weakening and weathering of bank materials (Thorne and Tovey, 1979). Fluvial entrainment is a process whereby the channel bed and bank materials can be detached and transported by flowing water. The "weakening and weathering" process includes those forces which operate on or within the bank to reduce its strength and loosen or detach particles of the bank surface materials. Such processes are creep,

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surface erosion, mass wasting, freeze-thaw, dry ravel, and soil piping due to positive pore water pressures.

Geomorphic processes affecting bank materials, and associated with fluvial entrainment include rejuvenation or over-steepening of stream adjacent slopes, where stream disequilibrium conditions occur; as well as liquefaction or the mass wasting of a fully saturated cohesionless soil due to extremely rapid changes in inundation (Thorne and Tovey, 1979). The relative contribution of these bank and adjacent slope erosional processes depends upon whether the bank and/or slope material is cohesive, non-cohesive, or composite. In addition, the processes of leaching and softening due to rapid movement of water through the bank can reduce the strength of cohesive banks causing them to behave similar to cohesionless banks. Rapid drawdown due to flow regulation or other conditions which reduce the wetted perimeter associated with frequently occurring long duration natural flows can increase the rates of weathering and weakening of these banks. For example, in a study by Thorne and Tovey (1979), accelerated downslope creep was decreased by providing a flow which inundated most of the bank area portions of the bank.

Consequences of Stream Channel Disequilibrium or Channel Stability Imbalance

Alterations in natural streamflow can change inherent stream channel stability, setting up both short and long-term adjustments in the conveyance system. When disequilibrium conditions occur as a result

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of alterations in the frequency and magnitude of natural discharges, a series of adjustments are likely to occur. These include:

a. Reduced flood flow capacity

b. Floodplain encroachment

c. Building of new floodplains

d. Lateral migration of channels and/or evulsion

e. Increased channel erosion with an associated increase in sedimentation

f. Damage to structures such as roads and bridges or property adjacent to the floodplain

g. Reduced surface water available at baseflow discharges creating intermittent and/or subterranean flow

h. Damage to riparian vegetation

i. Increased downstream hazards due to flooding from lower magnitude flows

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The physical processes responsible for these consequences can be grouped under the following major categories: (a) Channel Aggradation, (b) Hydraulic Geometry, (c) Bank Erosion, and (d) Vegetation Encroachment. Examples and discussion are provided for each major category.

(a) Channel Aggradation

Reductions in stream discharge result in reductions in sediment transport. The sediment material normally transported is deposited on the channel bed, bars, and other channel features. This process of deposition tends to reduce the channel cross-sectional area, reducing flood flow capacity. In addition, the process sets up a series of channel adjustments, some of which include lateral channel migration, floodplain encroachment, and rejuvenation of stream adjacent slopes. Under conditions of unregulated streamflow, side tributaries will normally contribute sediment yields to the main trunk during tributary runoff. If flow regulation were to reduce the main stem discharges, the delivered tributary sediment tends to be deposited near the mouth of the tributary of the main channel, causing an increase in the base level of the tributaries. This base level change can cause rejuvenation (over-steepening) of the unregulated tributary streams, supplying even higher sediment loads to the regulated trunk stream.

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Aggradation may also create a condition of reduced surface water availability as many streams become intermittent and/or subterranean.

The effect of lateral migration, typical of aggrading meandering streams, is to accelerate erosion on stream adjacent lands and rebuild floodplains. This is often undesirable as many stream adjacent lands support valuable timber resources, associated riparian values, developments, and structures. Lateral channel migration due to aggradation also results in increased sediment supplies, primarily due to accelerated erosion caused by fluvial entrainment and associated rejuvenation of stream adjacent slopes. Research results reporting such consequences are numerous. The most serious impact due to channel aggradation is an increase in the potential for downstream flooding. Several studies have reported the results of dam construction on a channel system's flood-discharge capability. Leopold and Maddock (1953) have contended that flow regulation due to dam constructions may adversely affect the flood discharge capability of the river channel. In further support of this contention, a review by Petts (1979) showed channel-capacity reductions below reservoirs which ranged from 25 percent to 73 percent on eight different streams. Although the mechanisms responsible for the loss of channel capacity are varied, the dominant processes

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summarized included aggradation, vegetation encroachment, and channel bank collapse.

(b) Hydraulic Geometry

The basic relationship of discharge and channel characteristics has been clearly demonstrated. Reductions in the frequently occurring discharges can be expected to result in corresponding adjustments in channel geometry. The Platte River system in Nebraska offers a good example of the result in flow reductions due to regulation and associated changes in channel geometry (Figures 11 and 12). From records dating back to 1865, as analyzed by Williams (1978b), these relationships show a reduction in channel width coinciding with a reduction in mean annual flow and annual peak flows. Vegetation now occupies much of the zone that used to be channel. Many flow regulation structures which result in reduced natural flows are not designed to store the larger flood events. The reduction in channel capacity in the years prior to such a flood event due to flow regulation causes a greater potential for flood damage than the same magnitude flood under pre-regulation conditions.

(c) Bank Erosion

Rapid drawdown of flows often results in increased pore water pressure in the channel bank materials, which accelerates the bank weakening and weathering rate (Thorne and Tovey, 1979). Cycles of wetting and drying are also extremely important as they

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cause swelling and shrinkage of the soil leading to downslope soil creep, ravel, and sloughing. As channel banks collapse, and the energy for fluvial entrainment is reduced in these regulated streams, deposition of sediment and reduced channel capacity results.

Lowered or regulated discharges occurring in channels with large flow capacities often result in the development of a winding thalweg (low flow channel) that may later cause severe channel erosion by concentrating flow against the banks (Mahmood and Shen, 1971). For this reason, it is recommended that the minimum discharge in a mature irrigation canal be controlled at flows of, at least, 55 percent of the design capacity. Similarly, the rate of water level fluctuations is frequently controlled to avoid severe variations during low flow, so as to prevent the failure of side berms due to positive seepage forces. Seepage flow caused by pore water pressure in banks and resulting from channel drawdown or low flows can be a major factor in the development of channel instability. Such bank or berm erosion may also change the channel bed form from dune to flatbed, which further reduces flow resistance during subsequent higher flows and thus decreases bed resistance beyond the limit of channel stability (Mahmood and Shen, 1971).

(d) Vegetation Encroachment

A reduction in the range and magnitude of frequently occurring discharges can result in the invasion of riparian vegetation into

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the channel area, with resultant reductions in the capacity of channels to transmit flood flows. Evidence of this process is documented by Turner and Karpiscak (1980) where, within 13 years following construction of Glen Canyon Dam, accelerated riparian vegetation encroachment occurred within the main channel, downstream from the dam. Williams (1978b), as stated earlier, described riparian vegetation establishment in the original channel of the North Platte River.

Vegetation encroachment on stream channels, as a result of flow regulation, has been documented by many other researchers. The growth of willows within the Republican River channel below Harlan County Dam initiated the formation of islands, the development of a new floodplain, and the reduction of channel capacity (Northrup, 1965). Other observations of stream channel aggradation and reduced flood flow capacity caused by vegetation encroachment are documented by Hathaway (1948) and Serr (1972). King (1961) examined a stream in the Wind River Basin of Wyoming where encroaching vegetation also reduced flood flow capacity.

Generally, the rising-recession and baseflow discharges inundate the channel bed and most of the banks during the growing season. These conditions normally prevent the invasion of riparian vegetation within the "active channel" and reduce the occurrence of vegetation encroachment.

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Application of Basic Concepts to Wyoming Streams

The preceding discussion provides the fundamental concepts involving stream stability and discharge relationships needed to secure "favorable conditions of water flow." These relationships were utilized in the development of a procedure which was used to calculate the stream channel stability maintenance flows for the Bighorn and Shoshone National Forests within the Big Horn River Basin in Wyoming.

The successful application of any procedure often depends on the verification of the fundamental relationships within the physiographic province under study. Most of the streams to which this quantification procedure was applied are not gaged. Collection and analysis of local data were therefore necessary for the development and application of regionalized channel geometry/stream discharge relationships.

The occurrence of direct channel impacts which are associated with flow reductions is evident from the earlier discussions of sediment transport-discharge relationships. There are very few streams in Wyoming with sediment data. However, two gravel-bed streams, the East Fork River near Pinedale and the Little Snake River near Dixon, have been studied and measured intensively, and, as such, provide sediment data which is typical for mountainous snowmelt runoff watersheds in Wyoming.

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Using the data from the Little Snake River, 86 percent of the total annual sediment yield is transported at discharges corresponding to bankfull, mean annual up to bankfull, and baseflow. Using Figure 7, the size of particles in transport ranged from approximately 2 millimeters at baseflow discharge (0.02 percent bankfull), 8.7 millimeters at mean annual discharge, and 16 millimeters at bankfull discharge (Andrews, 1980).

The same calculation when applied to the hydrograph-sedigraph for typical mountainous snowmelt watersheds (Figure 6) by Guy (1970) indicated that 79 percent of the total annual sediment yield was transported by discharges ranging from mean annual up to and including bankfull discharge; and 21 percent of the annual sediment yield was transported during low flow periods, of which 12 percent occurred from June 16 to September 30.

An analysis, conducted to determine a baseflow discharge requirement, was performed on data for the East Fork River in Wyoming, employing the "<u>inner berm</u>" concept (Leopold, 1980, personal communication). The U.S. Geological Survey utilizes this low-flow channel feature as a consistent index to mean annual discharge, in Colorado as well as other states (Hedman, et al., 1972). The inner berm concept relates the dimensions of the low flow or thalweg channel feature to the mean depth of flow for mean annual discharge. Since mean annual discharge is known to relate strongly to bankfull discharge, it follows that low flow channel dimensions are also related to bankfull discharge.

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The reasoning behind the use of the inner berm concept is that this particular low flow channel feature is an integral and necessary characteristic of stable channel systems which is consistently apparent and measurable. Thus, providing the low flow discharge necessary to maintain the capacity of the thalweg channel assists in the maintenance of overall channel stability.

Considering the magnitude of flows at baseflow, the question naturally arises, how effective is baseflow discharge in terms of the potential for sediment transport, and for the maintenance of these low flow channel features.

An analysis was made to determine the magnitude of flows associated with the inner berm feature. Channel cross-section profiles were available for the East Fork River in Wyoming, where the inner berm feature had been identified and evaluated (Leopold, 1980, personal communication). The plot of the cross section data (Figure 13) relates the mean depth of the inner berm (d) to the mean bankfull flow depths (d_B) in the same cross section.

A plot of the ratios of mean inner berm depth to mean bankfull depth versus the corresponding ratios of inner berm discharge to bankfull discharge for the East Fork River is shown in Figure 14. This relationship was compared to plots of similar data for other streams, from a variety of geographical areas and showed close agreement (Figure 15).

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The ratio of the inner berm depth to bankfull depth for the East Fork River was calculated as:

$$\frac{d\ (\text{inner berm depth})}{d_B\ (\text{bankfull depth})}$$

where:

$d = 0.20$ meters

$d_B = 1.60$ meters

and: $\frac{.20}{1.60} = 0.125$

In order to determine the ratio of inner berm or baseflow discharge to bankfull discharge, the depth of the inner berm was divided by the bankfull depth. This value (the ratio of d/d_b) was then used in figure 14 to estimate the percentage that baseflow discharge would be of bankfull discharge. The results of this analysis yielded a value of .019, or 1.9 percent of the bankfull discharge. Additional data from several California streams were provided by Leopold (1980, personal communication). The same analysis applied to these data indicated that baseflows would average 2.0 percent of the bankfull discharge.

A baseflow discharge requirement suggests an inherent sediment transport capability in order to maintain this low flow channel feature. A determination was made of the size of sediment materials that would be transported by the calculated baseflow, or inner berm discharge for the East Fork River (Leopold, 1980, personal communication). The analysis involved calculating a threshold shear velocity, utilizing

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the Shields equation as adapted by Sagan and Bagnold (1975) (Figure 9). The modified equation is shown as:

$$\mu_{\star} = \sqrt{gds}$$

where:

$\mu_{\star}$ = threshold shear velocity (feet per second)

g = gravitational force constant (32.2 feet per second per second)

d = depth of inner berm (feet)

s = slope of channel (decimal percent)

Once the threshold hear velocity was determined, the relationship of shear velocity and particle size (Figure 9) was used to determine the potential size of sediment materials that would be transported through the inner berm cross-sectional area. The results of this analysis are:

$$\mu_{\star} = \sqrt{(32.2)(0.69)(0.0007)}$$

where:

g = 32.2 feet per second per second

d = 0.69 feet

s = 0.0007

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thus, for the East Fork River, at this low flow condition,

$$\mu_{\star} = 0.125$$

From Figure 9, the corresponding size of transported sediment is 1.6 to 1.7 millimeters or approximately "very coarse" sand. These data agree very well with the size of material transported at baseflow discharge (2 percent of bankfull) on the Little Snake River at Dixon, where the potential transported sediment size was approximately 2 millimeters in diameter (Andrews, 1980). Based on this analysis, it is reasonable to assume that a baseflow discharge of approximately 1.9 percent of the bankfull discharge can be effective in transporting sand-sized sediment particles. Thus the cross-sectional area of the thalweg channel is maintained by reducing the potential for aggradation of the sand-sized and smaller sediment materials, in gravel-bed streams in Wyoming.

This agrees with observations made on the Colorado River where, under baseflow conditions, sand-sized particles were migrating into pools, with a corresponding decrease in the elevation of the stream bed at riffles. See Figure 10 (Leopold, 1969).

It was earlier demonstrated that stable channels generally have a channel capacity capable of transmitting the 1.5-year recurrence interval flow. This flow, known as bankfull discharge, is considered as part of the channel stability maintenance flow requirement. Since

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the channel geometry feature of bankfull width can be correlated with bankfull discharge, mean basin elevation, and drainage area, the relationships of these parameters were selected to calculate the bankfull discharge for ungaged streams. It was, therefore, necessary to analyze these basic relationships for streams in the Big Horn River Basin.

Fortunately, prior work by Lowham (1976) had established flow characteristics of Wyoming streams using channel geometry, drainage area, and weighted mean basin elevation for the mountainous regions of Wyoming. His work is widely used by engineers and hydrologists in the state for the calculation of peak flow events for various return periods on ungaged streams, in the design of drainage structures, bridges, etc.

For streams in Region 1 of Wyoming (mountainous, snowmelt watersheds) the regression relating the 2-year peak flow to bankfull width was positively correlated, with a correlation coefficient of 0.97 and a standard error of 35 percent. The 2-year peak flow was derived from long-term gaging station records using the Log-Pearson Type III method (Water Resources Council, 1976).

Where field measurement of bankfull width could not be practically done for large-scale flow determinations, Lowham (1976) utilized drainage area and weighted mean elevation to predict various recurrence interval flows. This effort has successfully demonstrated

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the practical application of the fundamental hydraulic geometry principles for mountainous snowmelt streams in Wyoming.

Similar work was also done by Emmett (1975), who developed similar relationships for snowmelt watersheds in Idaho. Examples of the approaches of both Emmett (1975) and Lowham (1976) will be used throughout the procedure section.

PROCEDURE

The objective of this section is to provide the basis for the quantification procedures as developed for the Big Horn River Basin. The procedures represent "state of art" applications of established principles of the sciences of fluvial geomorphology and engineering. The procedures incorporate site-specific verifications of stream channel geometry measurements and their relationships to measured discharges for selected streams in the Big Horn River Basin.

Rationale

These relationships, when combined with basin characteristics, can then be utilized for flow determinations on individual ungaged third, fourth, and fifth order alluvial streams in the Bighorn and Shoshone National Forests.

The range of flows required for self maintenance of the stream channel systems for ". . . favorable conditions. . ." involve those flows

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which transport the bulk of the total annual sediment load, inundate the "active" channel area to an extent sufficient to reduce vegetative encroachment, and maintain the banks and other integral features of the alluvial channel. Based on the relationships presented, the flows needed are "bankfull" discharge, a series of flows representing the rising and recession limbs of the snowmelt hydrograph, and a "baseflow" discharge.

Some 80 to 90 percent of the total annual sediment yield is transported during the snowmelt runoff season by those flows associated with the rising-recession limbs of the annual hydrograph.

This range of flows, considered critical for channel stability maintenance, is calculated with the use of the dimensionless flow duration curve, from bankfull discharge down to a point which represents mean annual discharge.

Since it has been previously demonstrated that the rapid increase and/or drawdown of stream flow results in accelerated bank erosional processes, a series of less abrupt rising and recession flows are needed for channel maintenance.

The importance of mean annual flow, in addition to peak flow, was demonstrated in a study where lowered annual peak flows <u>and</u> lowered mean annual flow resulted in reduced channel widths in the Platte River system (Williams 1978b.) These data further support the concept

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of a range in flows doing most of the "work" in the self maintenance of channel networks.

The basic interrelationship of mean annual, bankfull, and baseflow discharges make it easy to apply these principles to widely divergent hydrophysiographic provinces. For example, baseflow discharge is generally 10 percent of mean annual discharge and 2 percent of bankfull discharge. Furthermore, bankfull discharge is generally 10 percent of the mean annual discharge, thus the determination of one flow can be used to approximate the other two.

For the streams in the Big Horn River Basin, these flows, including the rising and recession flows, were determined individually, utilizing a regionalized dimensionless flow-duration curve specifically developed for this purpose.

In order to apply these procedures, the following assumptions were made.

Assumptions

1. The rate of sediment supply in the watershed will not be accelerated over time.

2. The dominant or effective discharge, defined as bankfull discharge, is the flow at bankfull stage and has a recurrence interval of 1.5 years.

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3. Bankfull discharge can be described as a function of bankfull width, which can be determined by using established regional relationships, verified with field measurements.

4. Bankfull width is related to drainage area and weighted mean basin elevation.

5. Mean annual and baseflow discharges can be reasonably estimated through the use of a regionalized dimensionless flow-duration curve.

6. The procedure as designed, is applicable to alluvial channels only.

Analysis Steps

A generalized flow chart showing the stepwise progression of the analysis steps used in the procedure is found in Figure 16.

Dimensionless Flow-Duration Curves:

Step 1 Develop regionalized dimensionless flow-duration curves from surface water data for local "representative watersheds" that have not been subject to streamflow regulation and for which there are at least 10 years of record.

Step 2 Perform a Log-Pearson Type III calculation to determine the 1.5-year return period discharge (bankfull discharge;Q_B)

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(Water Resource Council, 1976). An example is shown in Figure 17.

Step 3 Using standard techniques, plot flow-duration curves which relate mean daily discharge to a frequency of occurrence for each selected gaging site (Figure 18).

Step 4 Develop a dimensionless flow-duration curve for the individual selected gaged stations. This is accomplished by calculating the ratios of mean daily discharge to bankfull discharge (Q/Q_B) and plotting these ratios on probability paper for selected durations.

Step 5 Combine all dimensionless flow-duration curves into a composite regional curve representing the full range of drainage areas for the hydro-physiographic region to be analyzed. This process yields a single average curve valid for all locations in a local study area. An example is shown in Figure 19 (Emmett, 1975).

At this point, the ranges of required flow and their respective durations can be calculated for ungaged streams.

Baseflow discharge may be estimated by constructing two lines tangent to the lower segments of the flow-duration curve. A line, which then bisects the angle formed by the intersection of the two tangents, will intersect the flow duration curve at a point which represents the

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value of baseflow discharge as a percent of bankfull discharge. The value of mean annual discharge can be plotted directly on the duration curve from the long-term data summary; or it can be obtained using the procedure described in Step 13.

Relationship of Basin Characteristics to Bankfull Discharge:

Step 6 Obtain bankfull surface width of gaged stream, i.e., measure the stream channel width at the bankfull stage on an appropriate reach at or very near the gaging station, using measurement techniques described by Lowham (1976, Appendix I).

Step 7 Obtain bankfull discharge (Q_B) from the Log-Pearson flow frequency analysis (Step 2). Regress Q_B (dependent variable) against bankfull surface width W_B (independent variable) obtained in Step 6 for all selected stations within the hydrophysiographic region. An example is shown in Figure 20 (Emmett, 1975). With the data transformed to base 10 logarithms, a regression analysis is performed resulting in equations of the form:

Equation 1: $Q_B = a\ W_B{}^b$

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where:

Q_B = bankfull discharge represented by the 1.5-year recurrence interval discharge (in cubic feet/second)

W_B = bankfull width (in feet)

a and b = regression constants of slope and intercept

Obtain correlation coefficient and standard error statistics.

Step 8 For each selected gage site, calculate the area (in square miles) and area-weighted mean elevation of the contributing watershed.

Step 9 Analyze data from selected gaging stations: i.e., bankfull width (dependent variable), drainage area, and area-weighted mean elevation (independent variable) using a log transformed multiple regression analysis of the form:

Equation 2: $W_B = cA^dE^e$

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where:

W_B = bankfull width (feet)

c, d, e = regression constants

A = drainage area (square miles)

E = area-weighted mean elevation (feet in thousands)

Obtain correlation coefficient and standard error statistics. An example of this relationship (not including elevation) is shown by Emmett (1975) in Figure 21. (This allows for office calculations where bankfull widths cannot be measured.)

Step 10 Determine how well the relationships of drainage area, area-weighted mean elevation, and bankfull surface width for ungaged areas relate to gaged watershed data, using a wide range of watershed sizes and elevations. Plot these data over the relationships shown in Step 9. Determine if the regional relationships fit the gaged area data. If necessary, repeat the analysis in Step 9 and utilize local data for best relationships (i.e., best correlation coefficient and lowest standard error).

Step 11 To calculate bankfull discharge (Q_B) for ungaged streams:

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(a) Calculate drainage area in square miles and area-weighted mean elevation in thousands of feet.

(b) From equation 2, ($W_B = cA^dE^e$), determine W_B (bankfull surface width).

(c) From equation 1, ($Q_B = aW_B{}^b$), determine Q_B (bankfull discharge).

(d) To obtain the duration of Q_B, calculate the duration in days from flow duration curve (Step 5)(365 days x % time equaled or exceeded = duration in days).

Step 12 To calculate baseflow (Q_L):

Obtain ratio of Q/Q_B from dimensionless flow-duration curve. Convert to equivalent Q_L where $Q_L = Q/Q_B \times Q_B$.

Step 13 Calculate the rising and recession flows. Obtain mean annual discharge from the regionalized dimensionless flow-duration curve. Select flows from the flow-duration curve such that the slope of the rising/recession limbs of the snowmelt hydrograph is approximated at a rate of 2 percent of bankfull discharge per day up to bankfull and back down to mean annual discharge. The snowmelt rising/recession rate can also be calculated in uniform

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segments of the dimensionless flow duration curve as described in Figure 22. The inflection point indicating the change in slope of the flow-duration curve starting upwards from baseflow very closely approximates the mean annual discharge. Rising and recession flows are represented from this inflection point (mean annual discharge) up to bankfull discharge.

PROCEDURAL RESULTS

Step 1 Surface water records were obtained from the Cheyenne Office of the U.S. Geological Survey and from publications by Lowham (1976). APPENDIX II shows station locations.

Step 2 Discharge for The 1.5-year recurrence interval discharges were obtained from the U.S. Geological Survey for selected streams within the analysis area, using the Log-Pearson Type III analysis method as specified by the Water Resources Council (1976).

Step 3 Flow-duration curves were obtained from selected gaged stream sites in the region. A typical curve for Tensleep Creek is shown in Figure 23.

Step 4 Dimensionless flow-duration curves were developed for each stream with adequate long-term records. The basic data for

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the streams in the Big Horn River basin were derived from U.S. Geological Survey Water Supply Papers, flow-duration curves, and basic data from Lowham (1976). Figure 24 shows the dimensionless flow-duration curve, where the ratio of mean daily discharge to bankfull discharge is plotted as a function of duration (percent of time a flow is equaled or exceeded).

Step 5 All of the flow-duration curves for streams used in the analysis were combined into a composite regional dimensionless flow-duration curve (Figure 22).

Step 6 Water surface width measurements at bankfull stage for selected gaging sites were available through measurements completed by Lowham (1976).

Step 7 Bankfull discharge (Q_B) was regressed against bankfull width (W_B) for the representative gaged streams (Figure 26). The bankfull discharge used was that for the 1.5-year recurrence interval. The resultant relationship is:

Equation 1: $Q_B = 1.63\ W_B^{1.58}$

with a correlation coefficient of 0.975 and a standard error of 35 percent.

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Step 8 The mean elevations and drainage areas were obtained from maps and analyzed by the U.S. Geological Survey for those gaged streams in "Region 1" (high elevation snowmelt watersheds) (Lowham, 1976).

Step 9 Relationships, for the USGS stream gaging stations, of measured channel width at bankfull stage to drainage area are shown in Figure 18. An additional variable, area weighted mean elevation, was added (Figure 19) to this relationship. This improved the r^2 value from 0.88 to 0.93 and explained much of the variability shown in Figure 27.

The relationship took the form of:

Equation 2: $W_B = 0.028A^{0.55}E^{2.23}$

.28A

where:

W_B = bankfull width (feet)

A = drainage area (square miles)

E = Weighted mean elevation (M feet)

correlation coefficient = .93

standard error = 0.187

Step 10 Data gathered on numerous ungaged streams in the analysis area were plotted and the above regional curves developed

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from the gaged station data to determine the appropriateness of a regional relationship. The data points for the ungaged stations were in very close agreement with the regional curve (Figure 28).

Step 11 The calculation of bankfull discharge and its duration for streams on National Forest System lands in the Big Horn River Basin were calculated. Determination of bankfull width was accomplished using Equation 2, $W_B = 0.28A^{0.55}E^{2.23}$.

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Area-weighted mean elevation and drainage area were determined for each watershed. Once bankfull width was obtained, bankfull discharge was determined from Equation 1.

$$Q_B = 1.63\ W_B^{1.58}$$

The average duration of bankfull discharge, from the regionalized dimensionless flow duration curve, was approximately three days (Figure 29). This is an agreement with bankfull durations from other studies in Wyoming.

Step 12 "Baseflow" was determined from the regionalized dimensionless flow-duration curve. The procedure discussed in Step 5 was used, where two lines tangent to the lower straight segments were constructed. This resulted in a Q/Q_B ratio of 1.7 percent. To further verify this value,

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hydrographs from the gaged streams in the Big Horn Basin were analyzed for the percent of baseflow to bankfull for each respective stream. The average of these was 1.9 percent of bankfull, thus this procedure using flow duration curves was representative of baseflow conditions. The estimate of baseflow discharge can be determined by multiplying the bankfull discharge times 0.017.

Step 13 The series of rising and recession flows, starting with the point of mean annual discharge, as approximated on the dimensionless flow-duration curve and progressing to bankfull discharge is shown in Figure 29. These flows were distributed into nine segments where, beginning with bankfull discharge, the Q/Q_B ratio of 1.0 was reduced by a factor of 0.10 for each of nine segments to approximate the respective rising and recession flows.

The conversion of these required discharges into acre-feet yields is simply accomplished by the multiplication of the discharge in cubic feet per second, times the acre-feet per day conversion factor of 1.90, times the duration of flow in days. The results of this conversion indicated that the needed channel stability maintenance flows approximated 78 percent of the total annual water yield.

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SUMMARY

As more water becomes diverted from headwater areas, it becomes increasingly important to state the flow requirements necessary to prevent adverse changes in channel conditions due to these diversions. An range of in-channel non-consumptive discharges are required annually which will transport the bulk of the water and sediment in an orderly fashion for purposes of maintaining the stability of the natural conveyance system.

Since the required flows are non-consumptive in nature, they are available for historic downstream use and appropriation. The procedures presented are based upon a series of fundamental concepts supported by established research and measurement methodologies. The conceptual base is founded upon well-known principles of geomorpholgy and fluid mechanics. While the procedures involve in innovative combination of such principles and techniques, they do not depart in substance from the body of accepted scientific thought and principle. The use of the described procedures represents an attempt to apply current "state-of-the-art" techniques to accomplish the basic management objectives of ". . . securing favorable conditions of water flows . . .," in terms of maintaining the inherent stability of natural stream channel systems. Improvement in the procedure will obviously occur and is expected as a part of the normal evolutionary process associated with "state-of-the-art" techniques, but is not expected to result in consequential changes to the results.

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The procedures were applied on National Forest System lands in northern Wyoming, and may be employed elsewhere to make reasonable requests for a range of specific stream discharges and flow durations that will tend to assure ". . .favorable conditions of water flows. . .." Further, it is felt the recommended instreamflow values are reasonable in the context of existing appropriative downstream water uses. In addition, the long-term benefits derived from the maintenance of natural water conveyance systems will extend directly to the downstream water user as a result of securing these ". . .favorable conditions of water flows."

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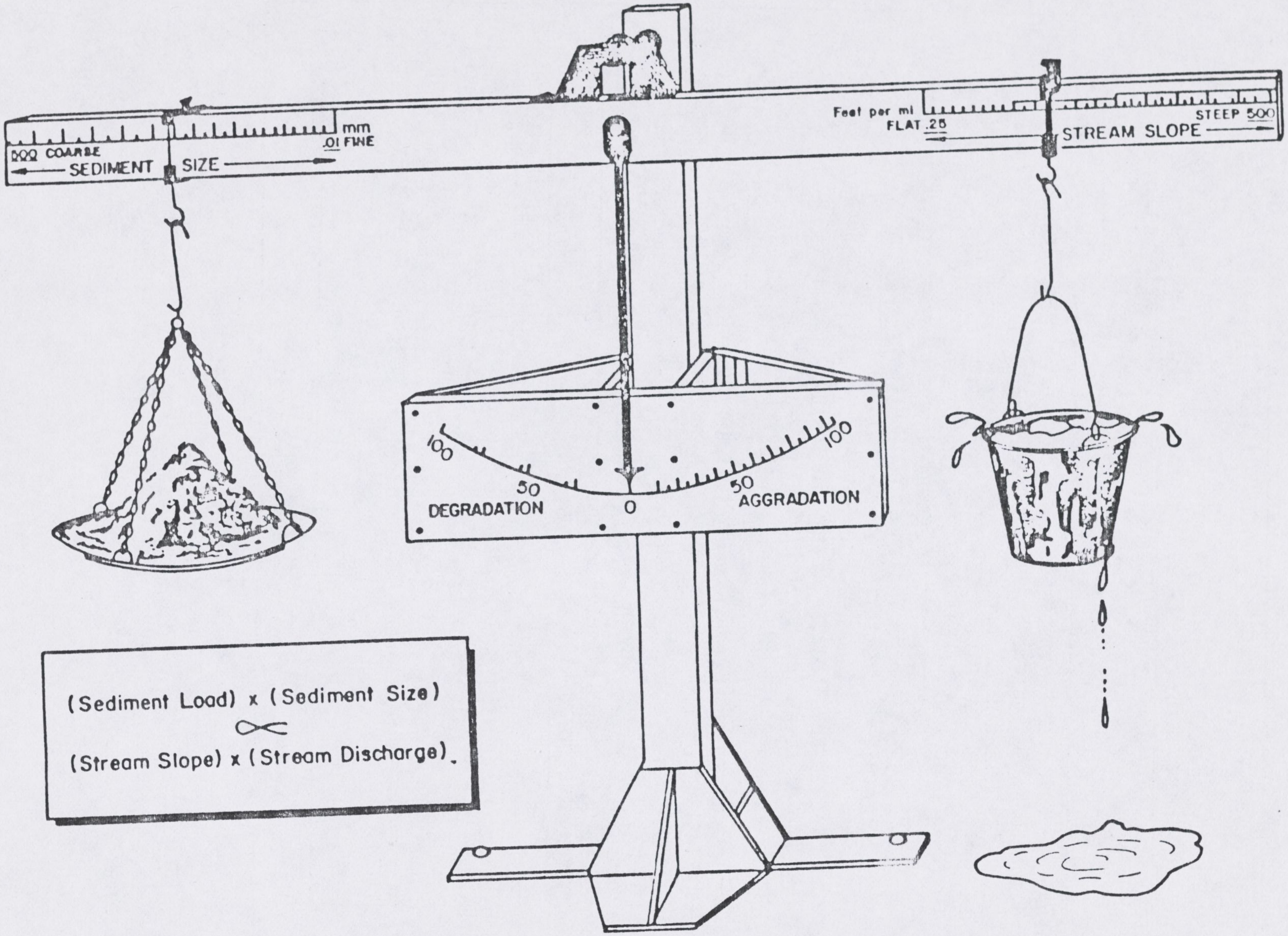

Figure 1.--Diagrammatic relationship of a stable channel balance. (Lane, 1955)

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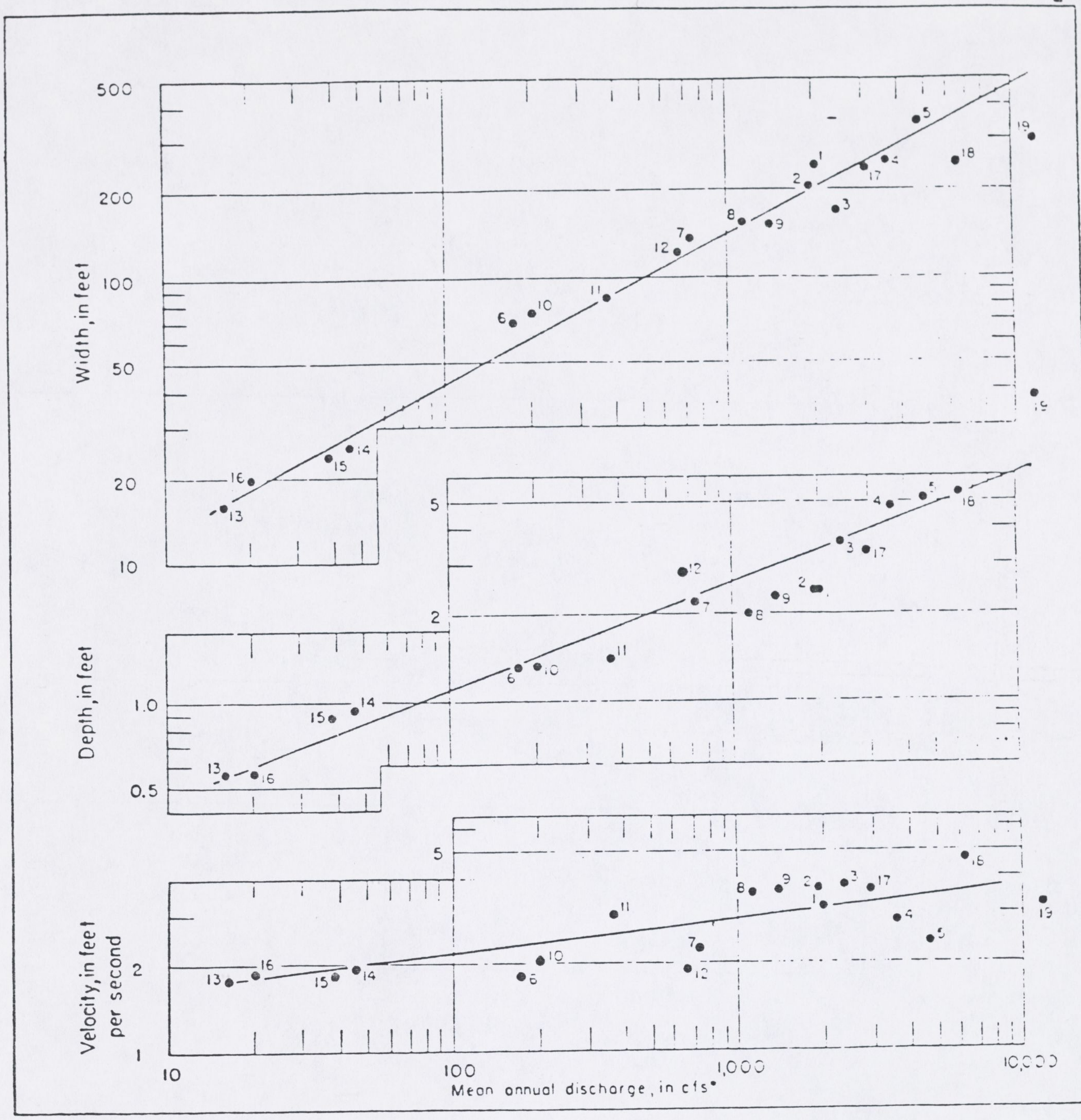

Figure 2.--Width, depth, and velocity in relation to mean annual discharge as discharge increases downstream. Big Horn River and tributaries. (Leopold and Maddock, 1953)

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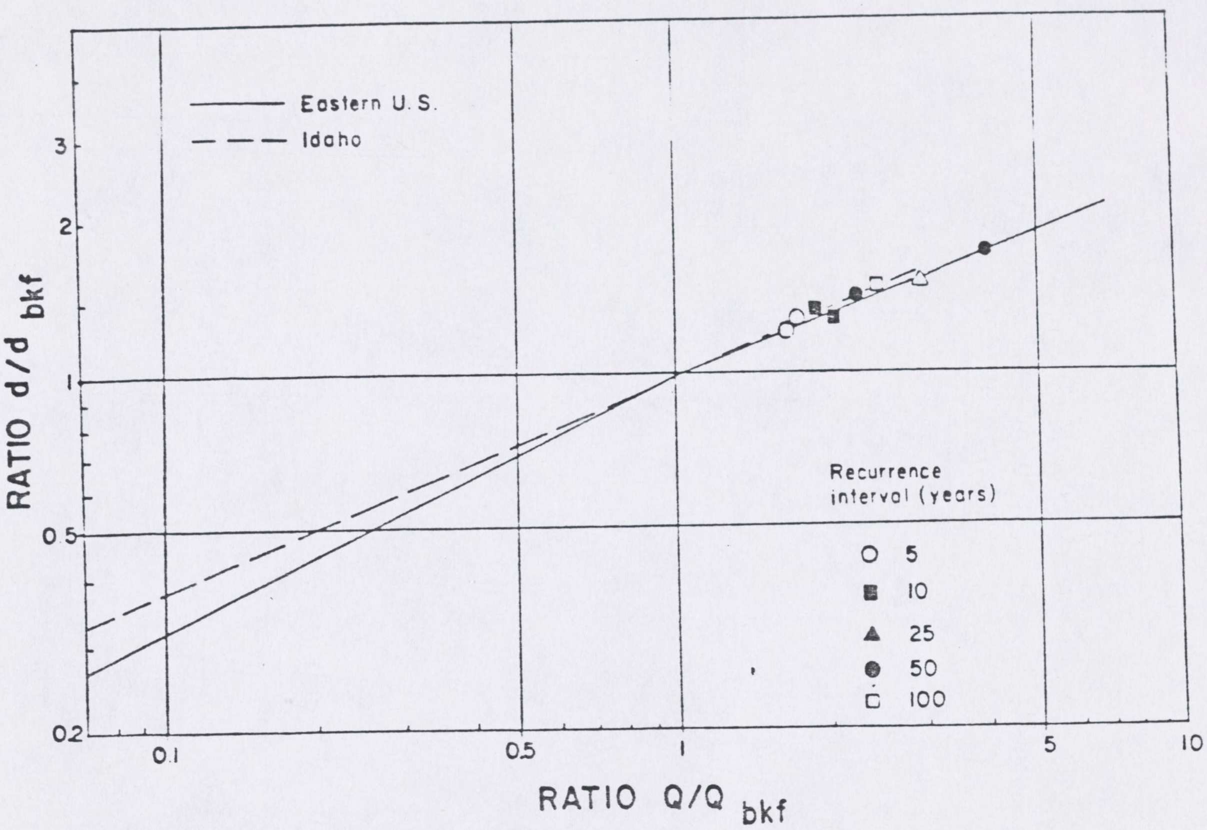

Figure 3.--Dimensionless rating curves for two geographical areas by recurrence interval. (Dunne and Leopold, 1978)

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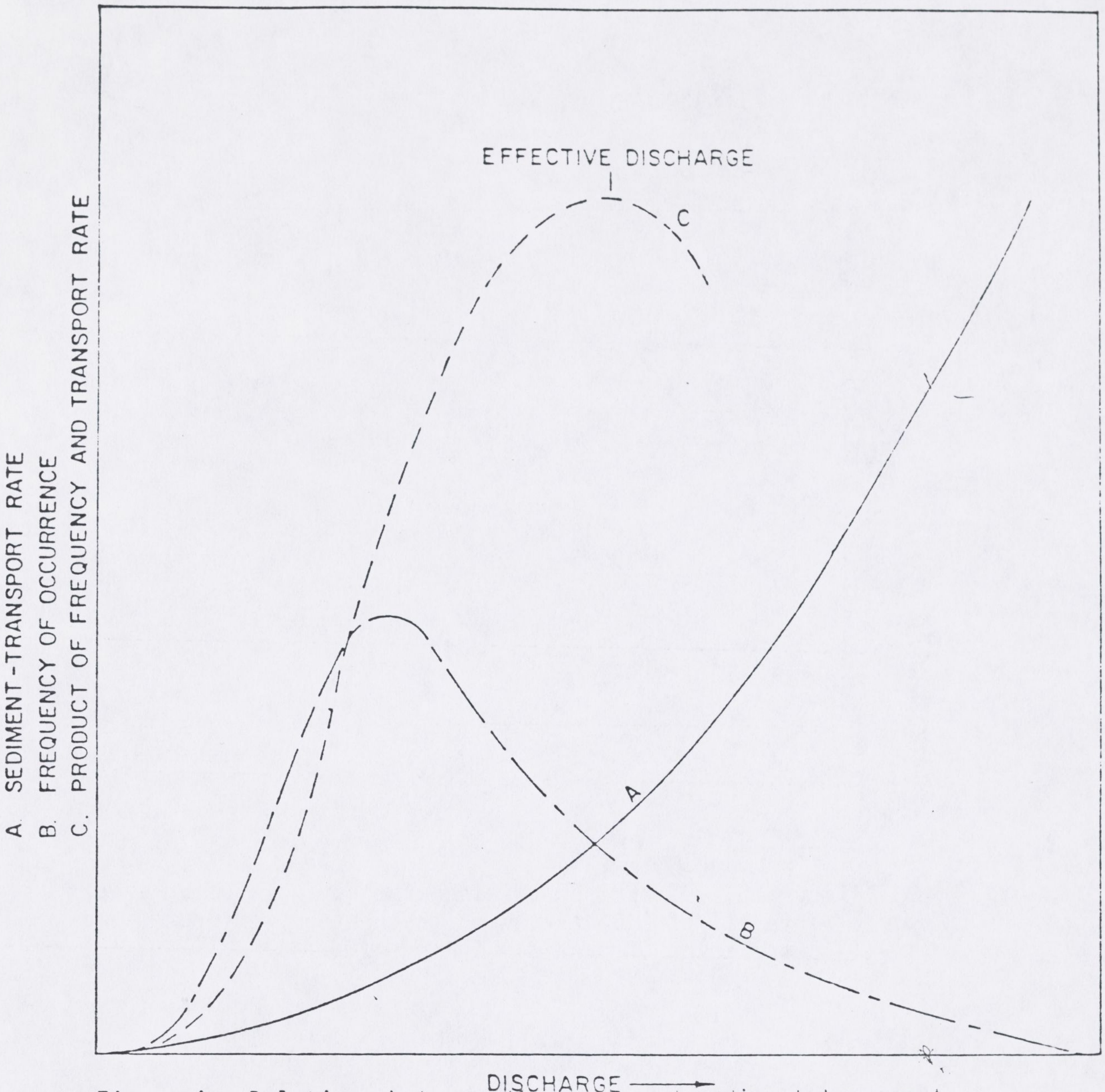

Figure 4.--Relations between discharge and sediment transport rate, frequency of occurrence and the product of frequency and transport rate (Wolman and Miller, 1960)

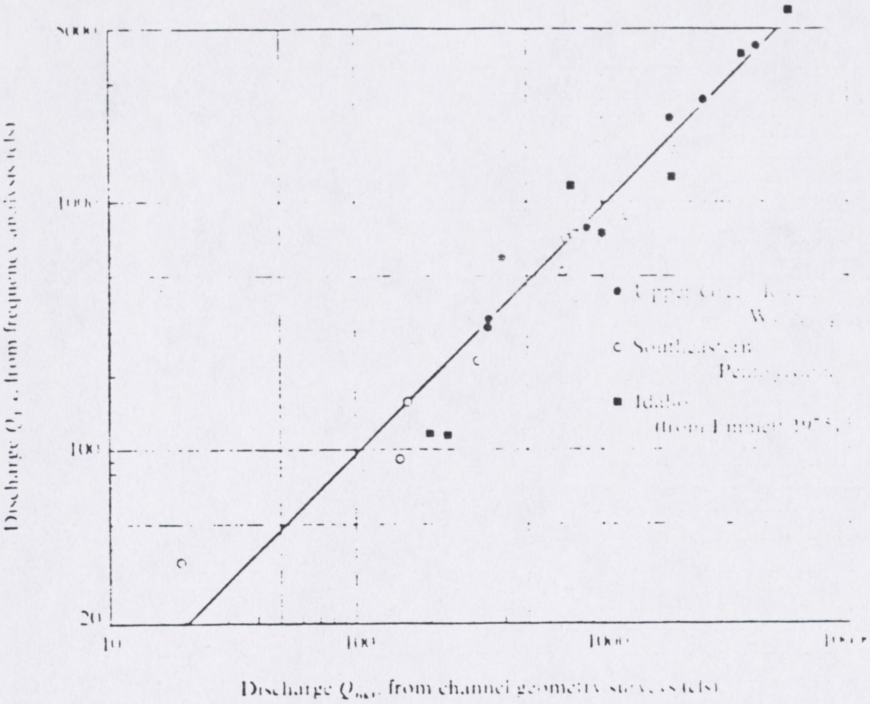

Figure 5.--Comparison of discharge having a recurrence interval of 1.5 years with that determined from field observation of bankfull state. (Dunne and Leopold, 1978)

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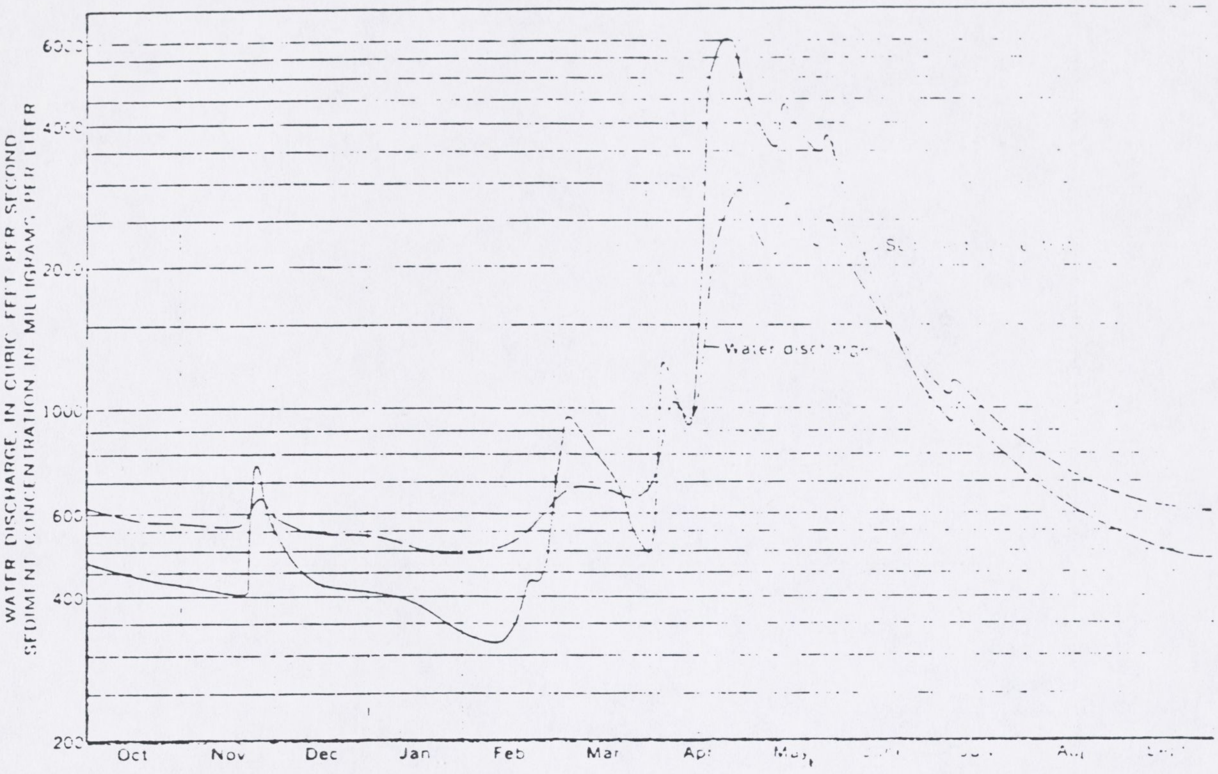

Figure 6.--Temporal relationship of sediment concentration to water discharge for an assumed "snowmelt" stream draining mountainous terrain. (Guy, 1970)

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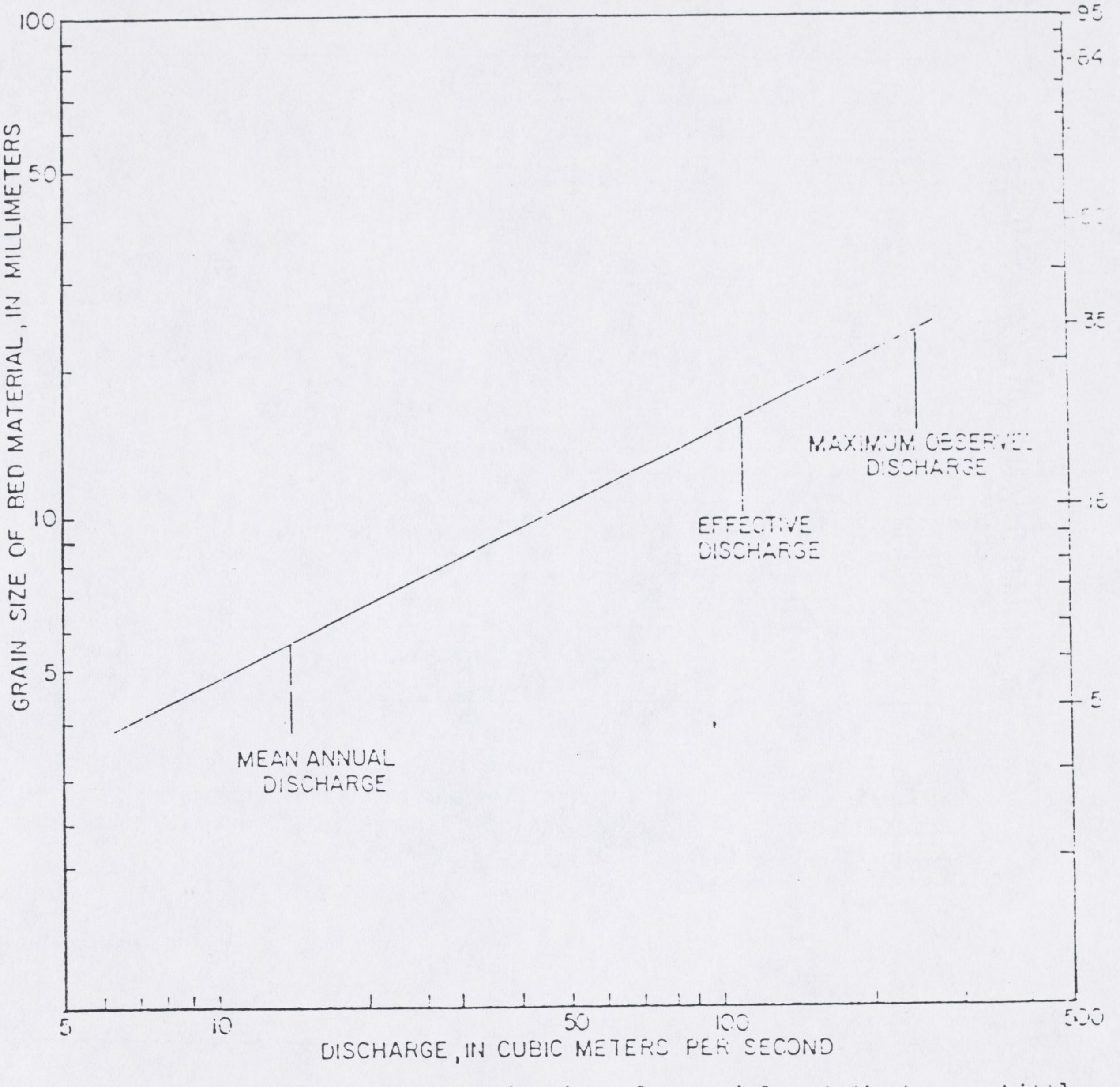

Figure 7.--Relation between grain size of material and discharge, Little Snake River near Dixon, Wyoming. (Andrews 1980)

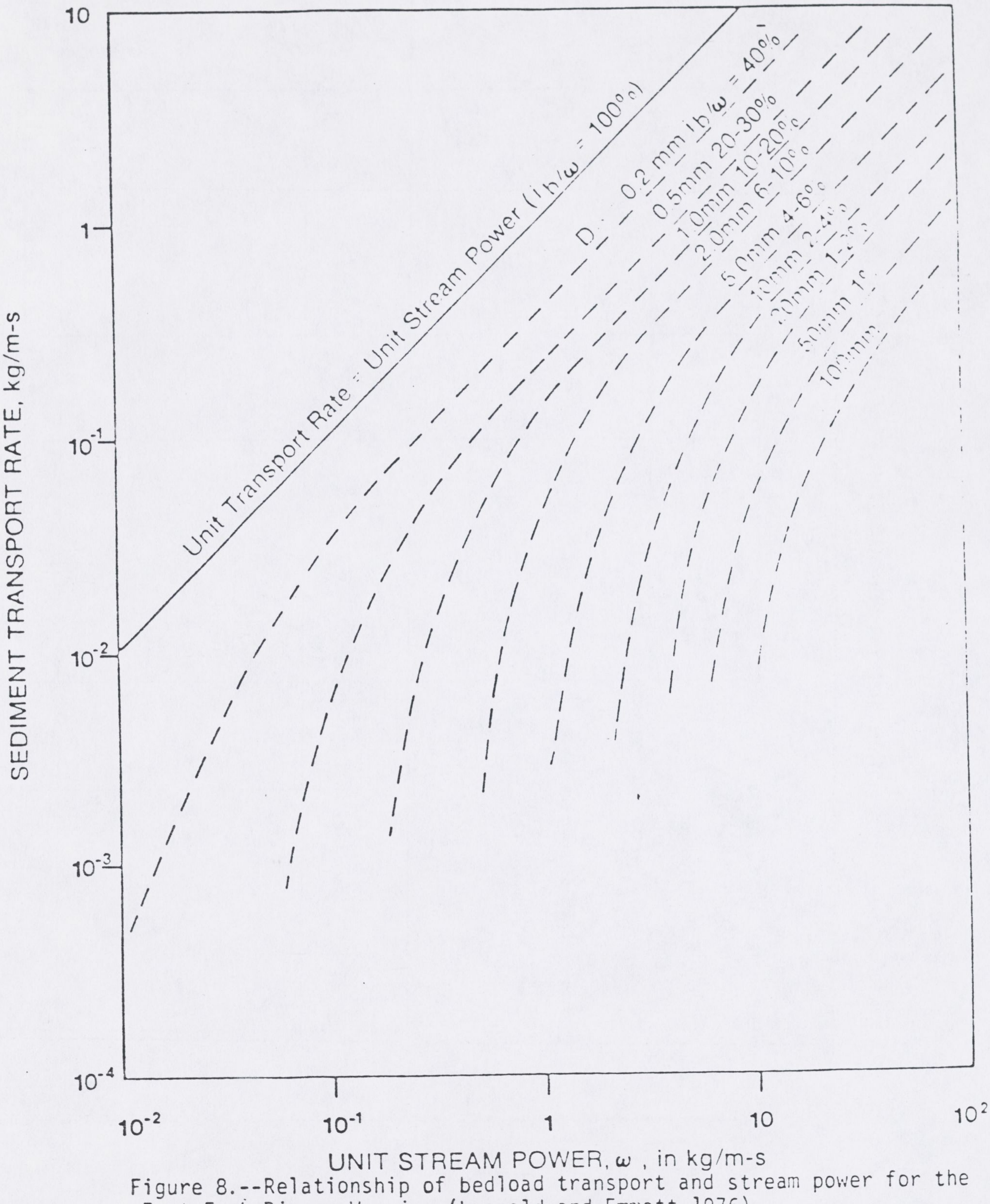

Figure 8.--Relationship of bedload transport and stream power for the East Fork River, Wyoming (Leopold and Emmett 1976).

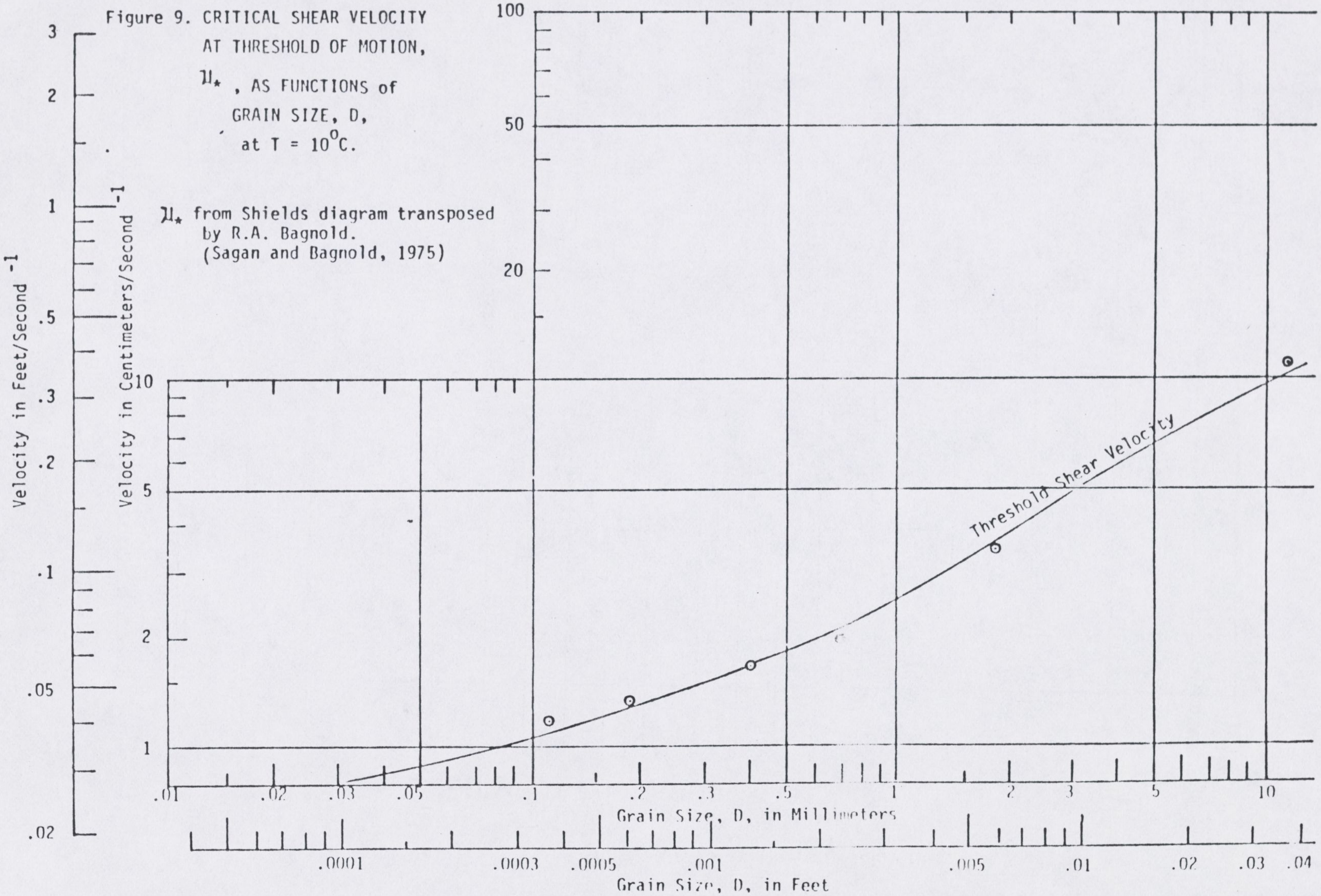

Figure 9. CRITICAL SHEAR VELOCITY AT THRESHOLD OF MOTION, u_*, AS FUNCTIONS of GRAIN SIZE, D, at T = 10°C.

u_* from Shields diagram transposed by R.A. Bagnold. (Sagan and Bagnold, 1975)

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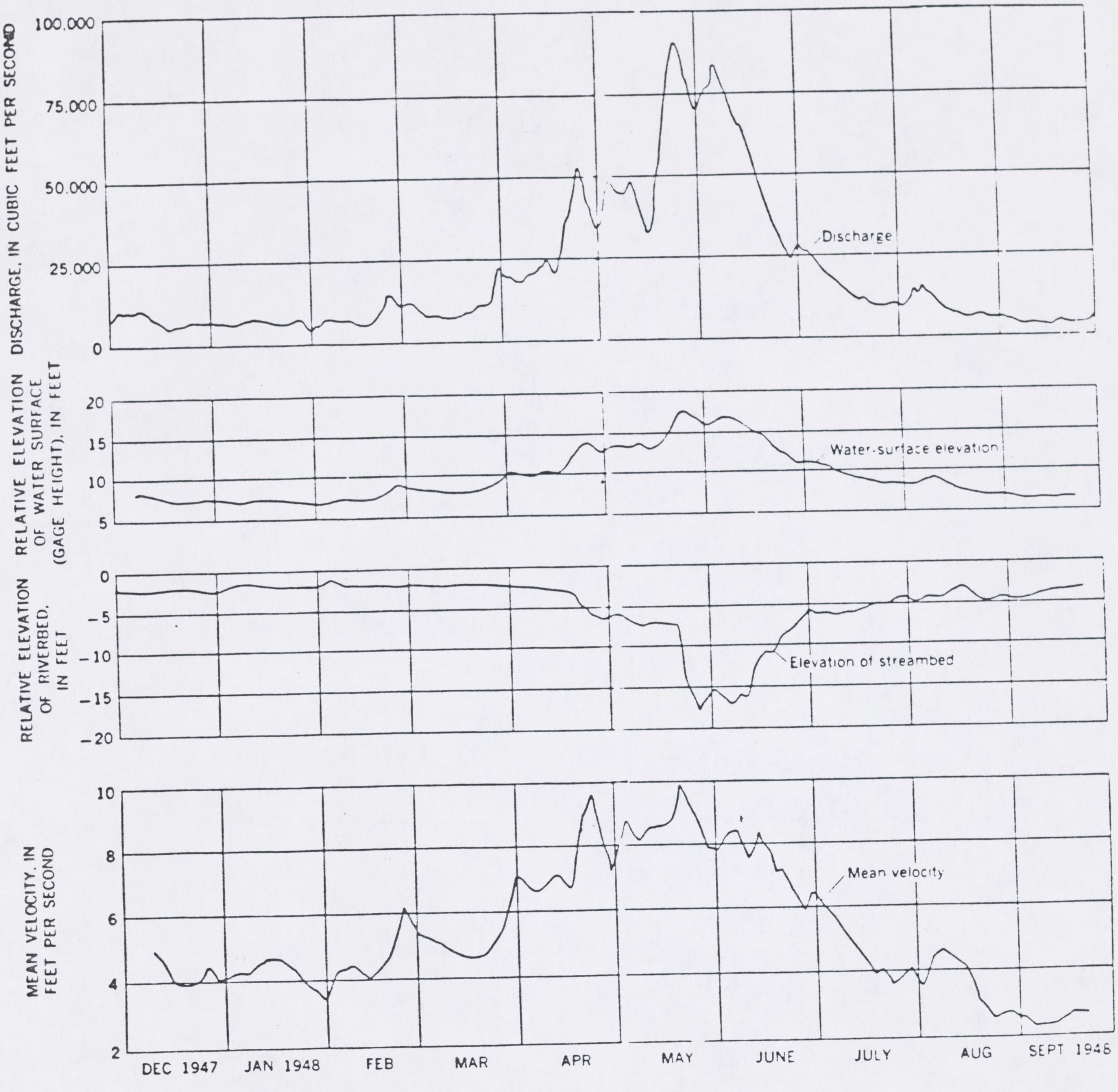

Figure 10.--Changes in discharge, water-surface elevation, bed elevation, and mean velocity during a 10-month period at the Lees Ferry measuring station, Colorado River. (Leopold, 1969)

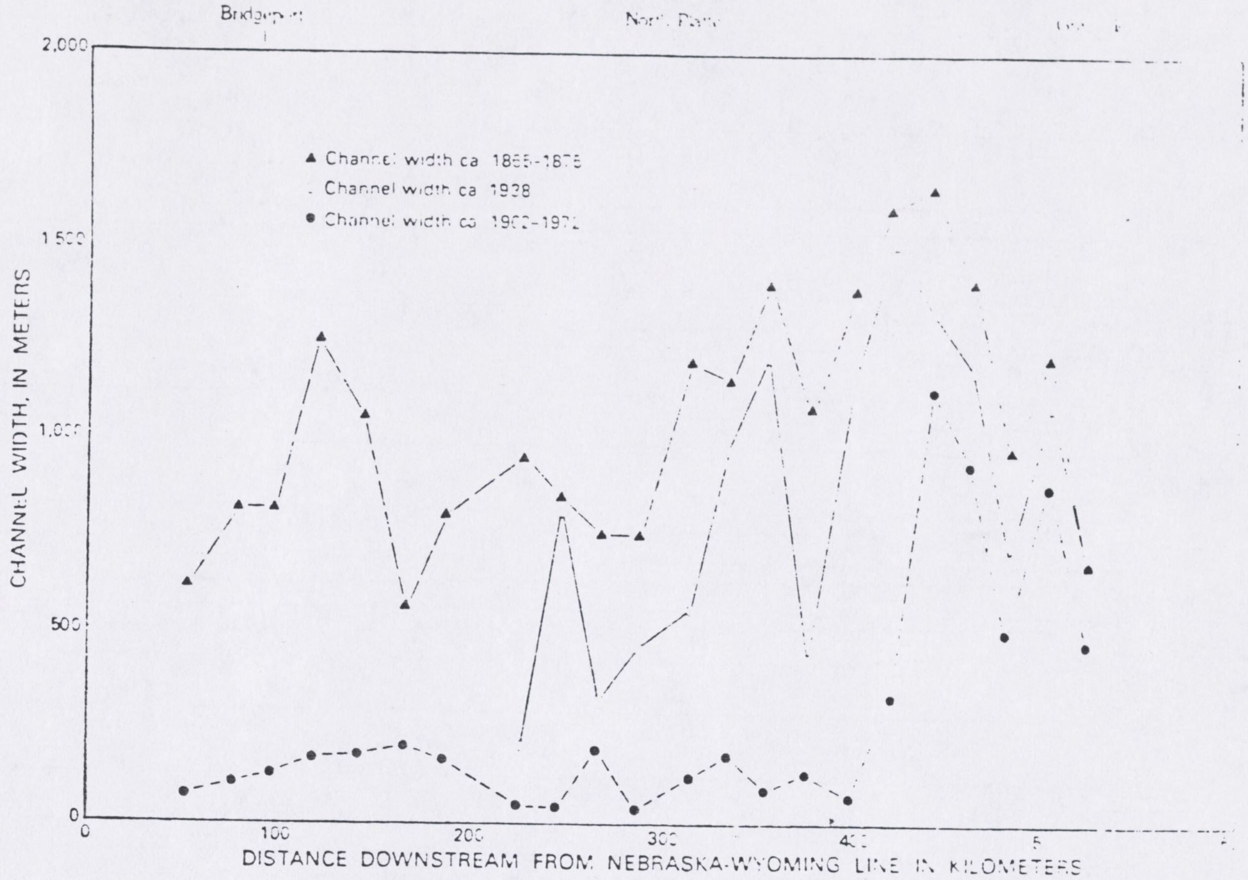

Figure 11.--Channel width at different time periods. (Williams, 1978b)

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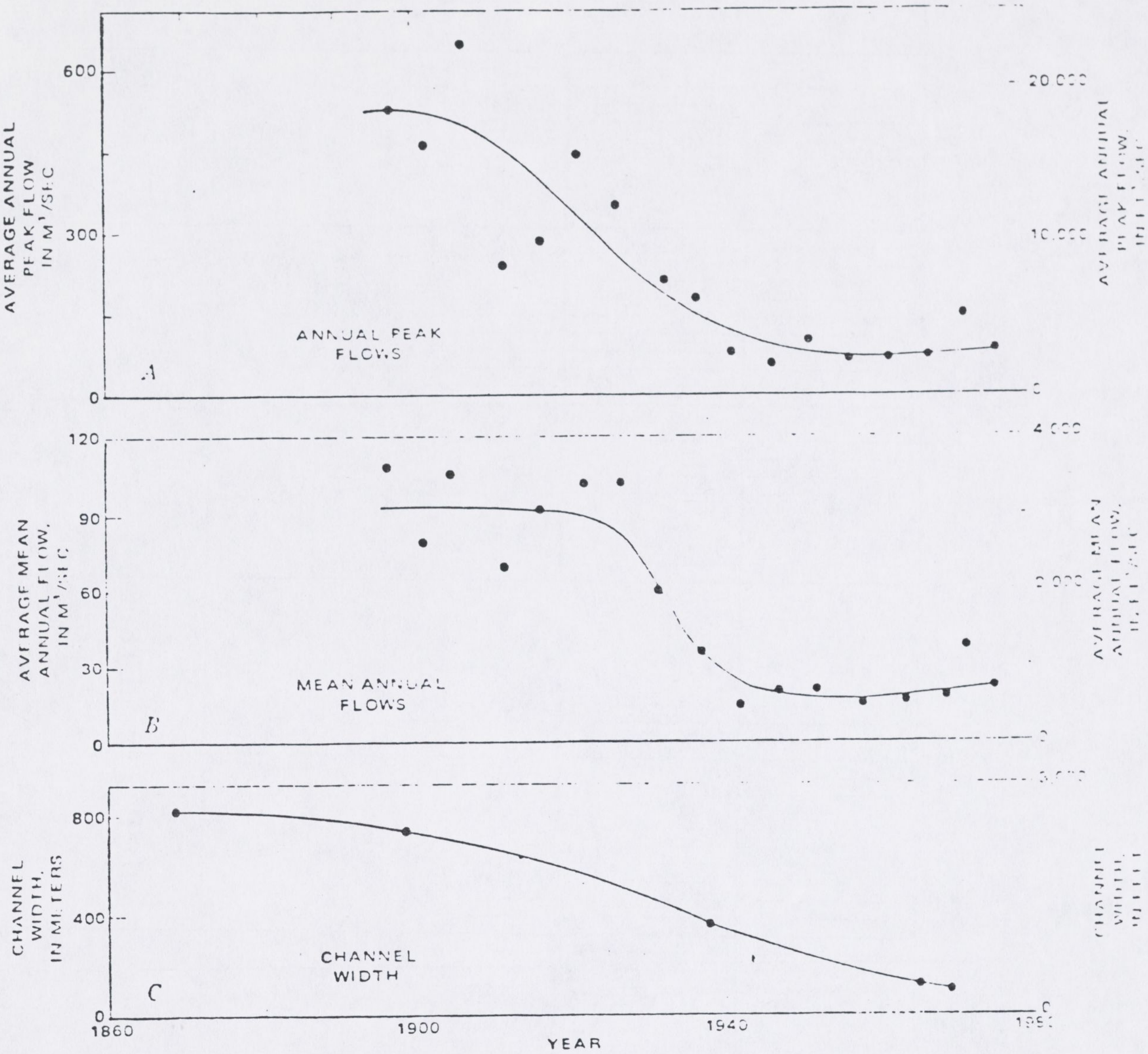

Figure12.--Historical trends of annual peak flows and channel width. North Platte River ai North Platte. (Williams, 1978b)

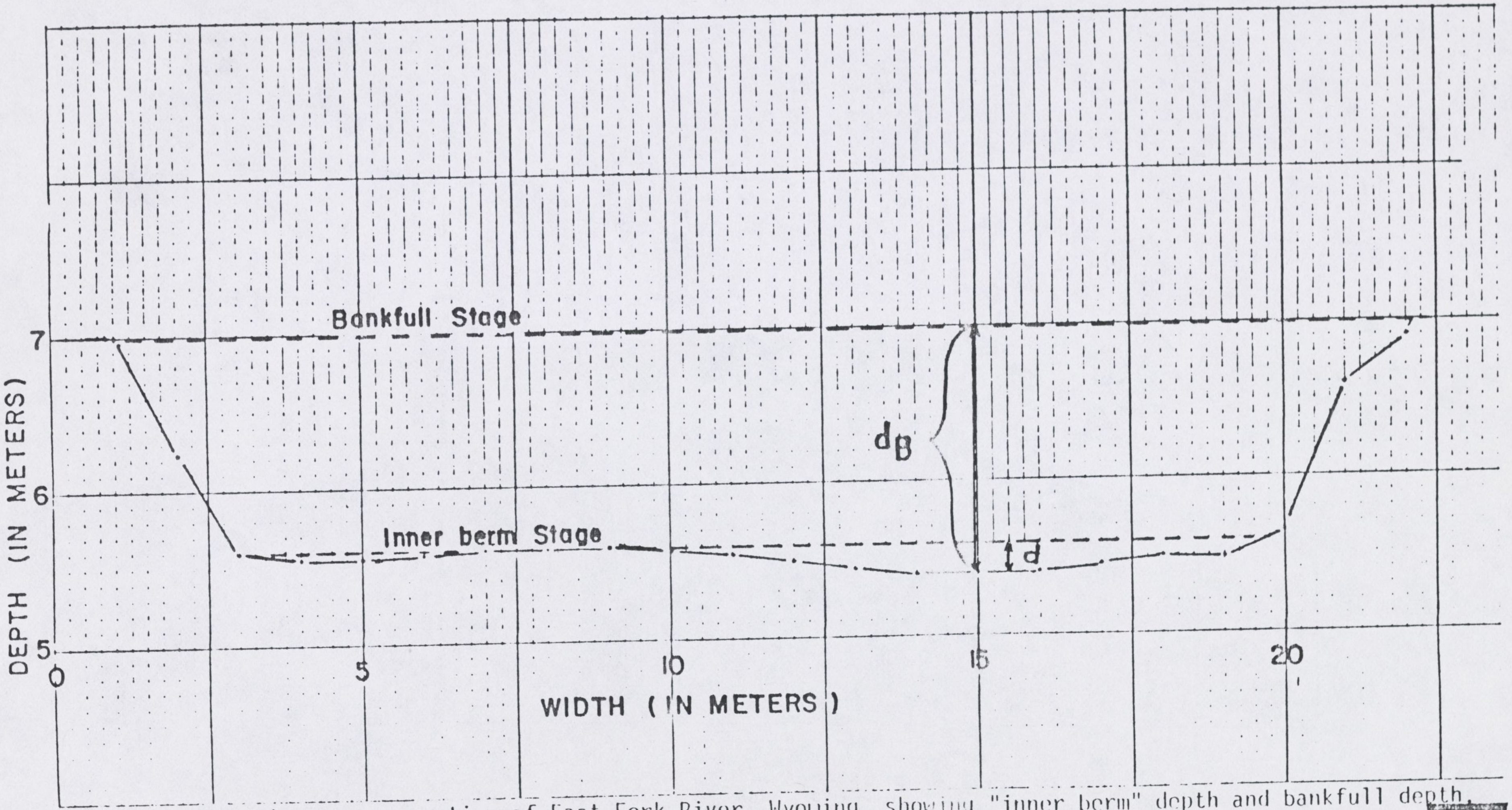

Figure 13. River cross-section of East Fork River, Wyoming, showing "inner berm" depth and bankfull depth. (Leopold, 1980 Personal Communication)

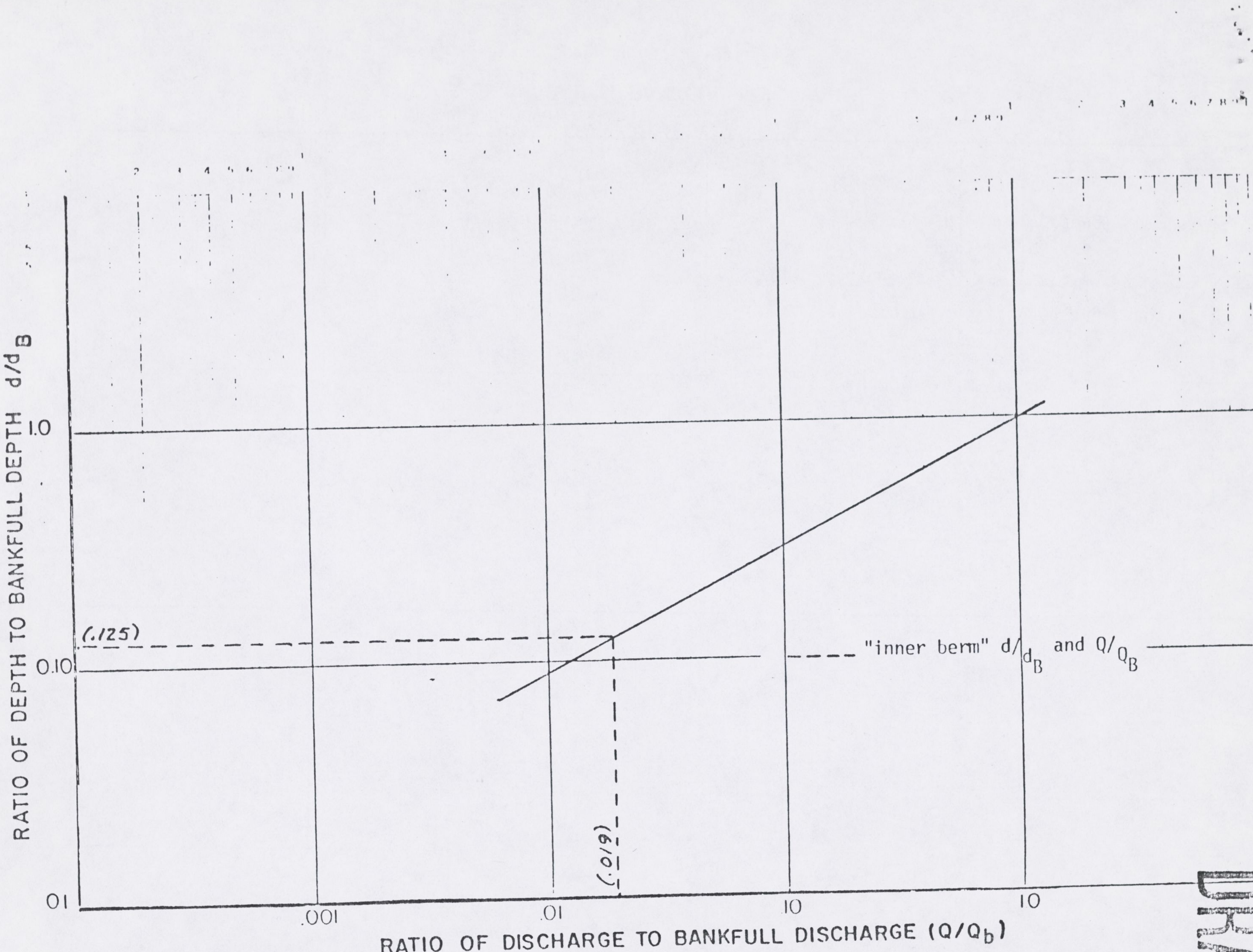

Figure 14. Dimensionless rating curve for East Fork River, Wyoming showing "inner berm" values.

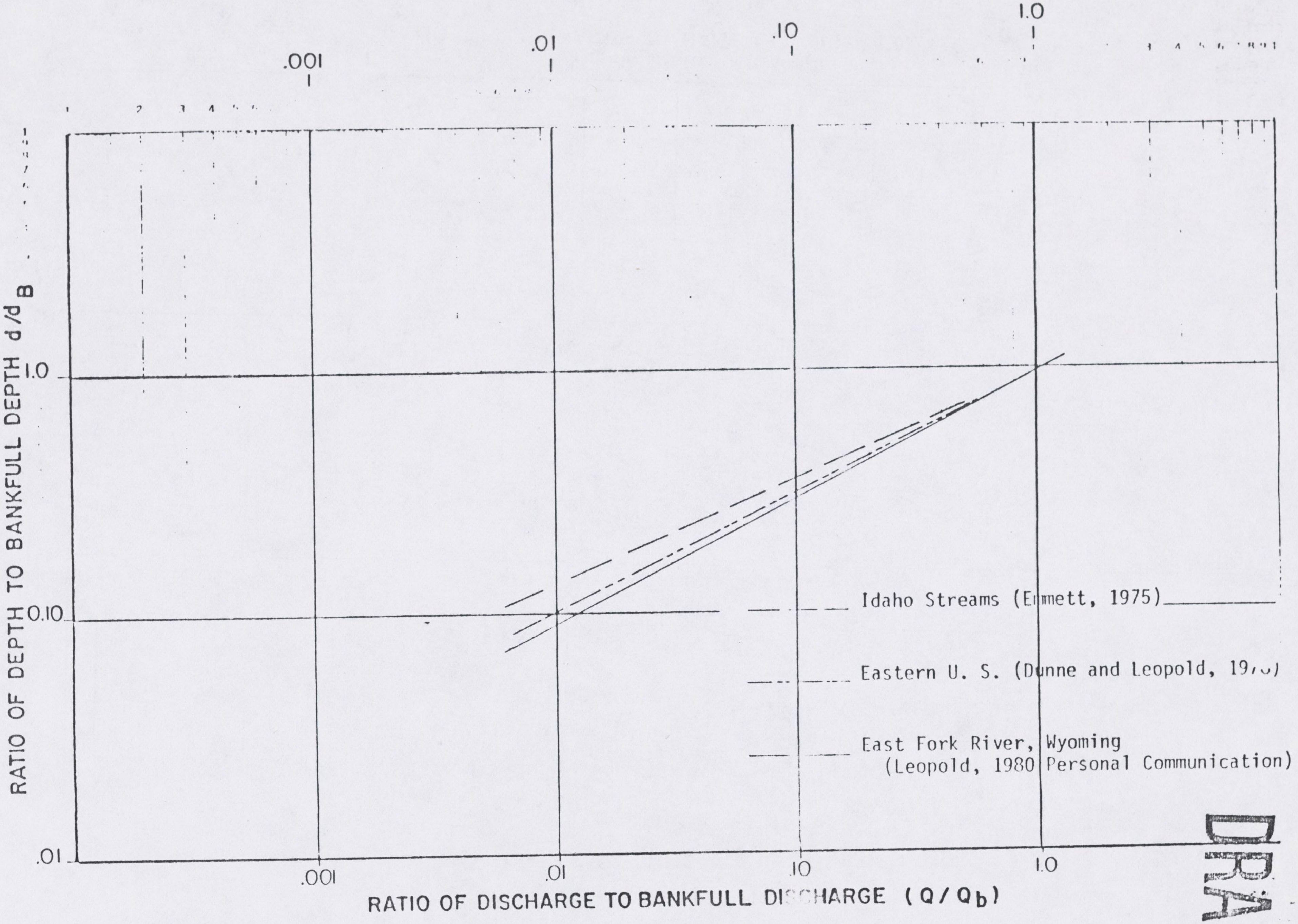

Figure 15. Dimensionless rating curve for three geographical re ons.

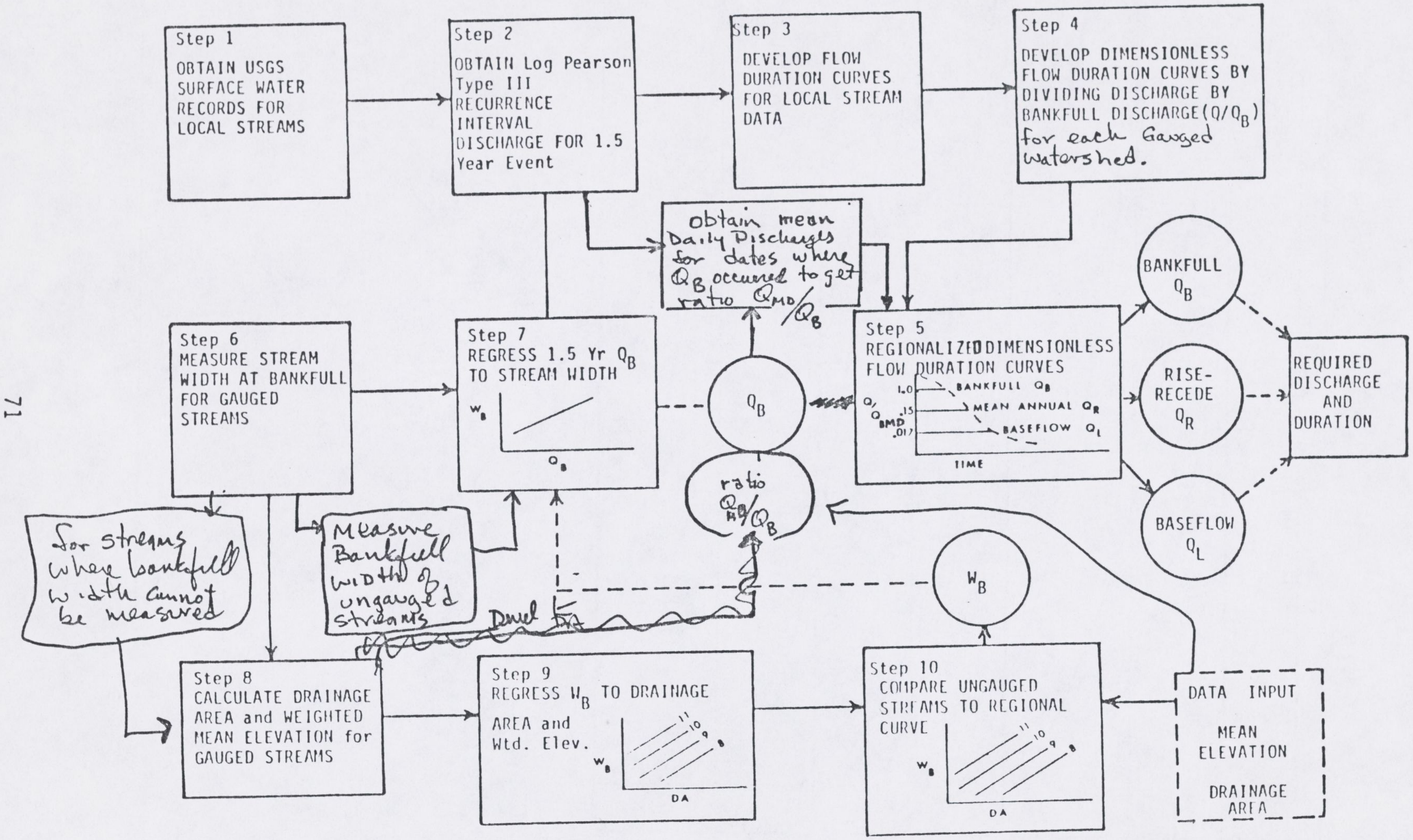

FIGURE 16. PROCEDURAL ANALYSIS STEPS REQUIRED FOR DISCHARGE DETERMINATION

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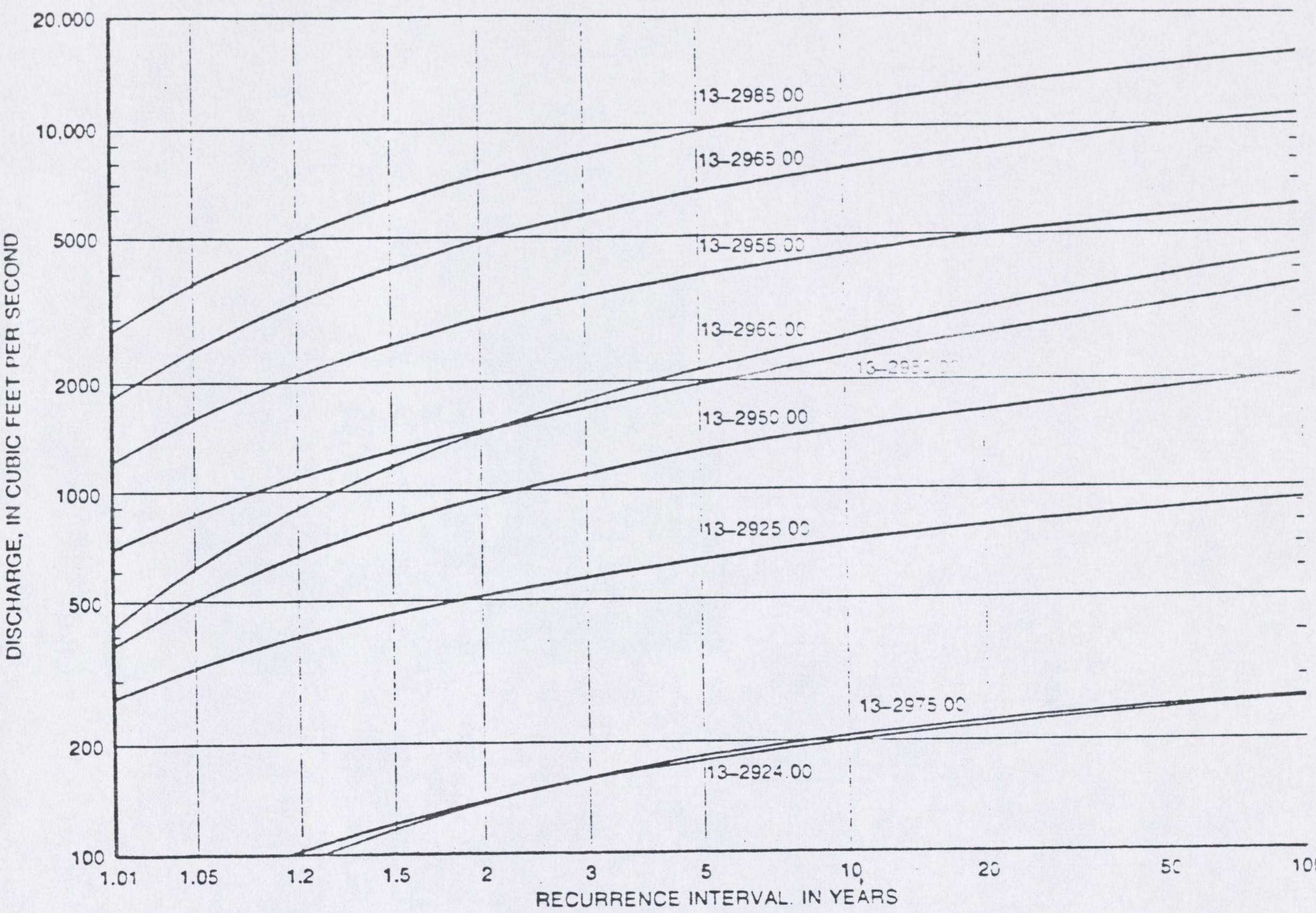

Figure 17.-Examples of flow-frequency curves for selected stream-gaging stations using Log-Pearson Type III analysis. (Emmett, 1975)

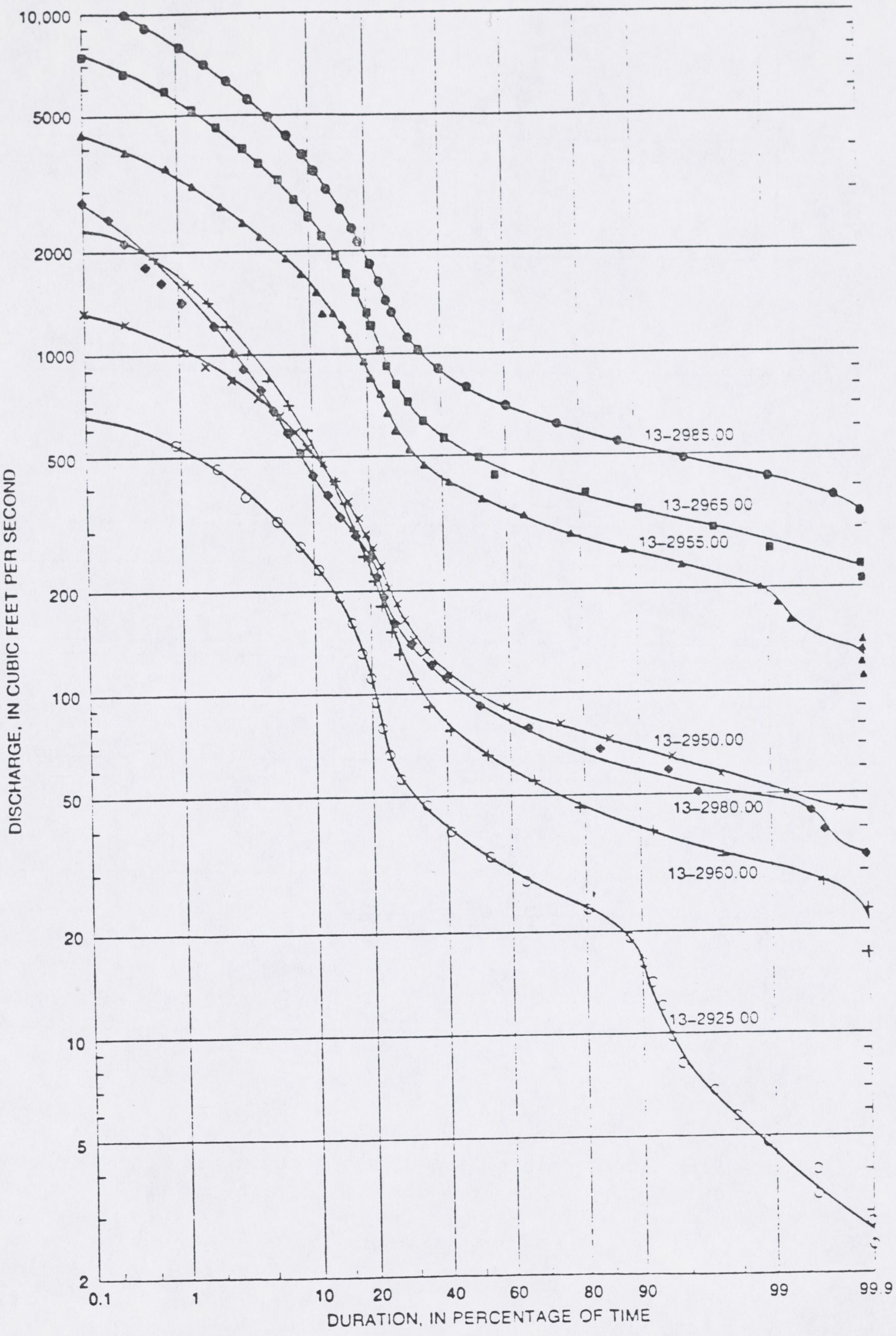

Figure 18.--Flow-duration curves for selected stream-gaging stations. (Emmett, 1975)

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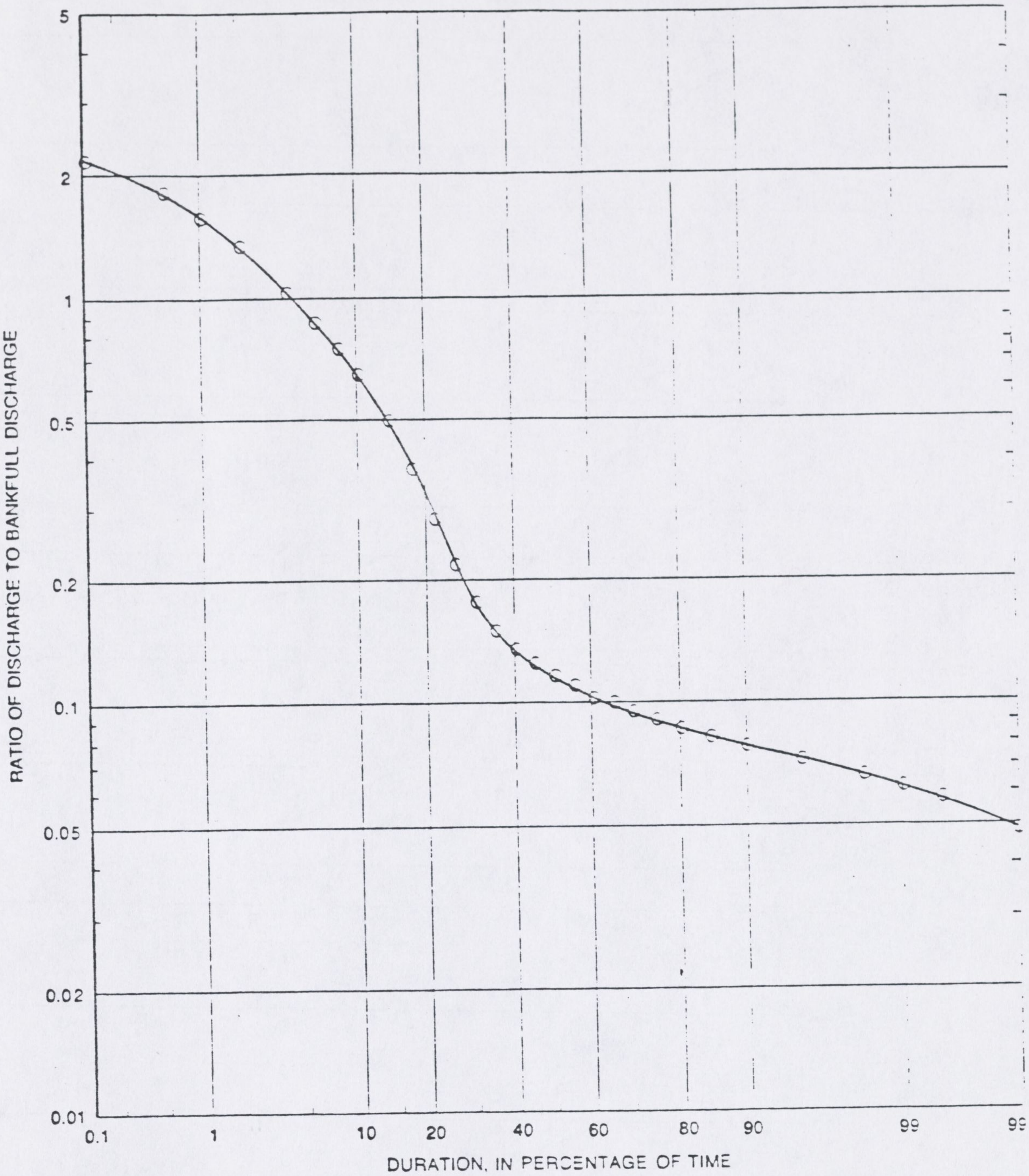

Figure 19.--Dimensionless flow-duration curve for streamflow in the upper Salmon River area. (Emmett, 1975)

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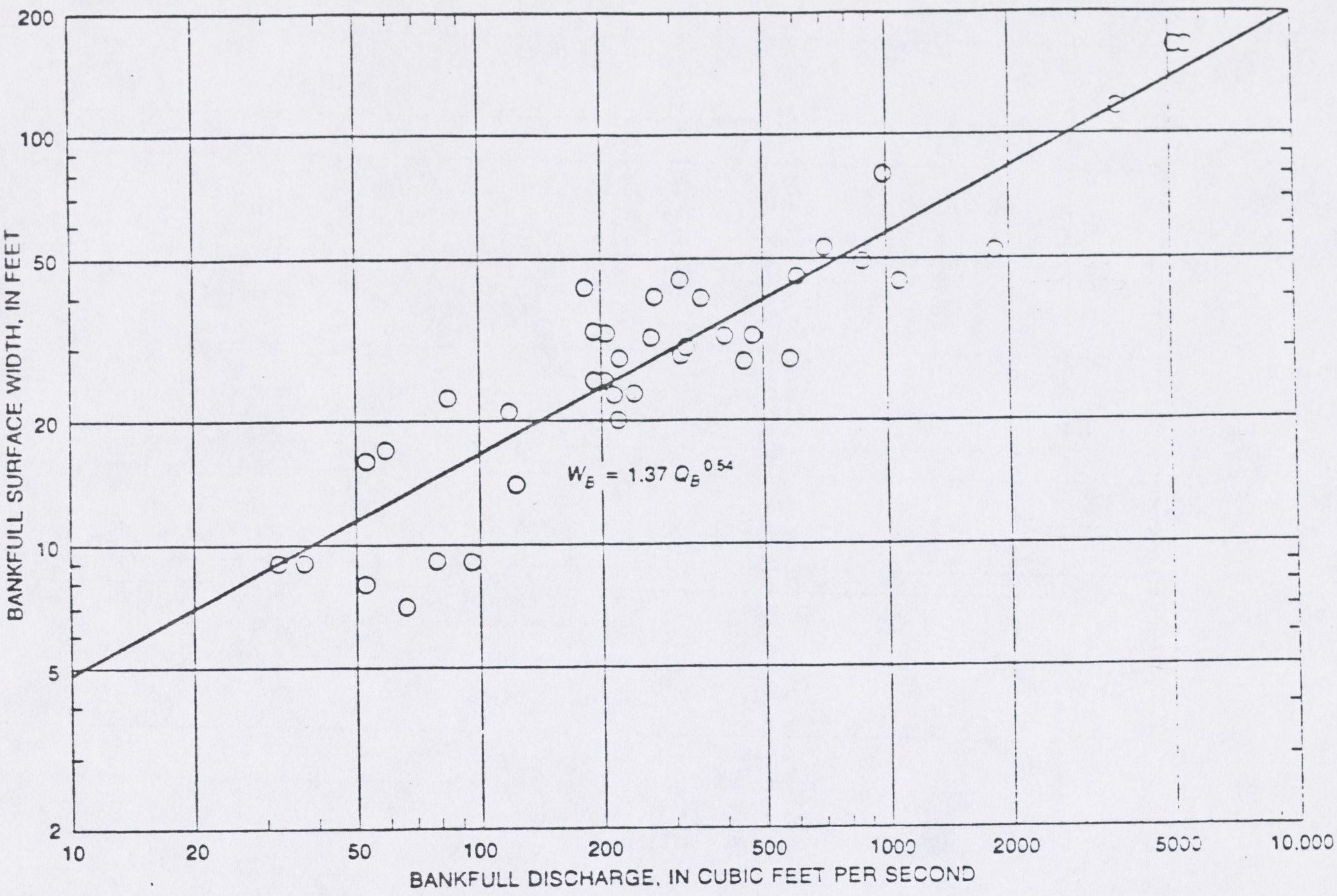

Figure 20.--Bankfull surface width as a function of bankfull discharge. (Emmett, 1975)

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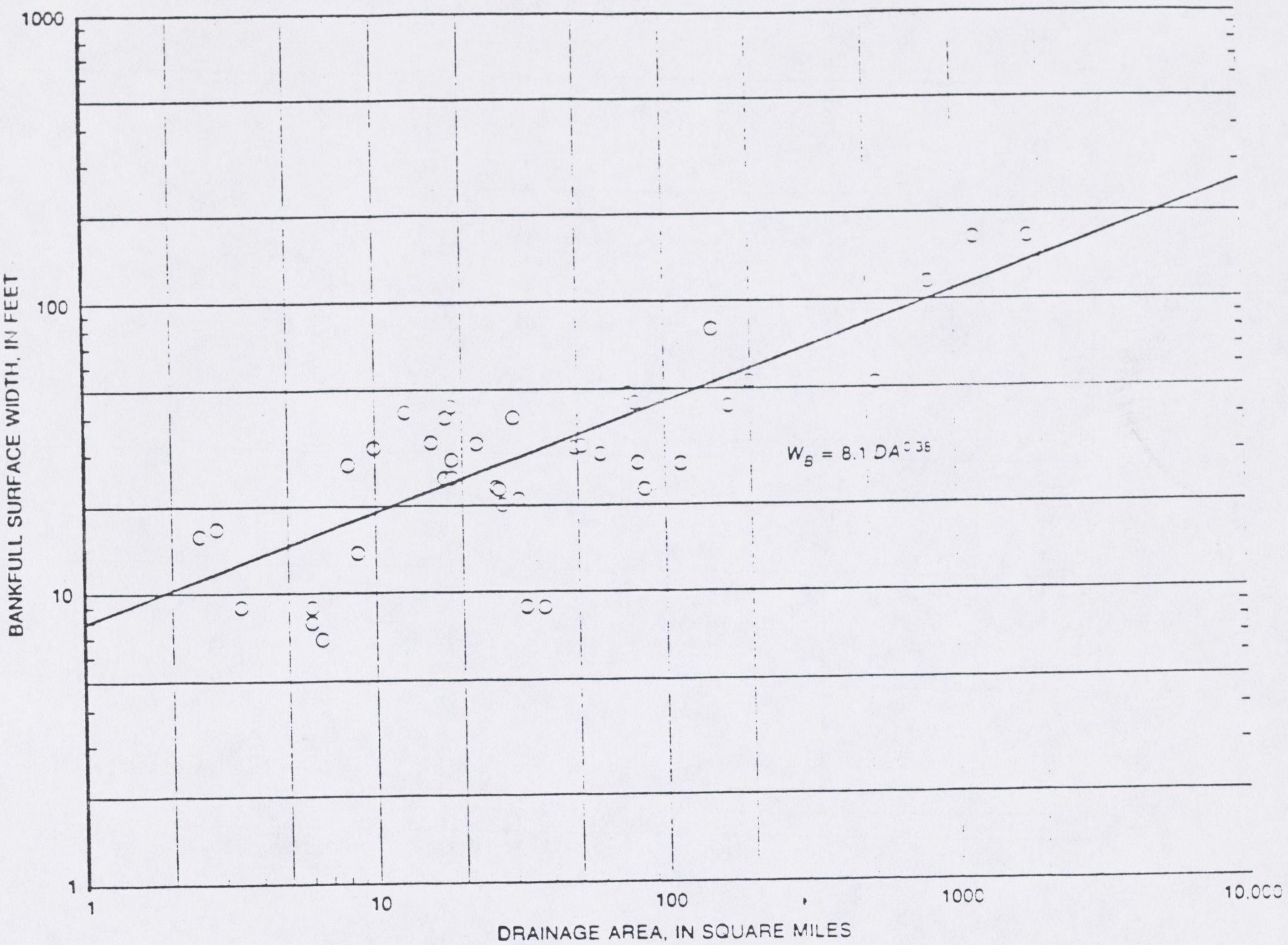

Figure 21.--Bankfull surface width as a function of drainage area. (Emmett, 1975)

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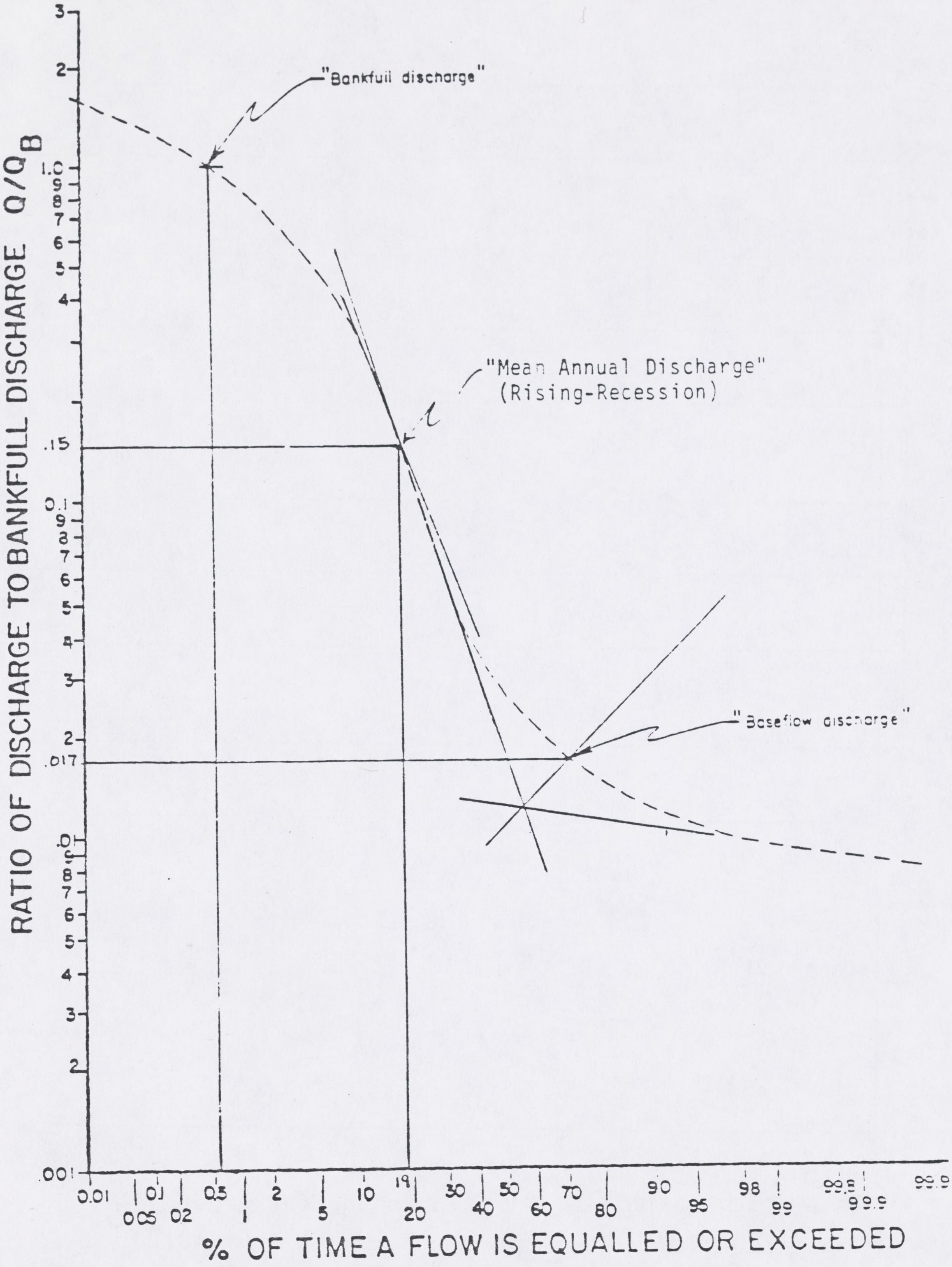

Figure 22.--Regionalized dimensionless flow-duration curve for Big Horn Basin streams.

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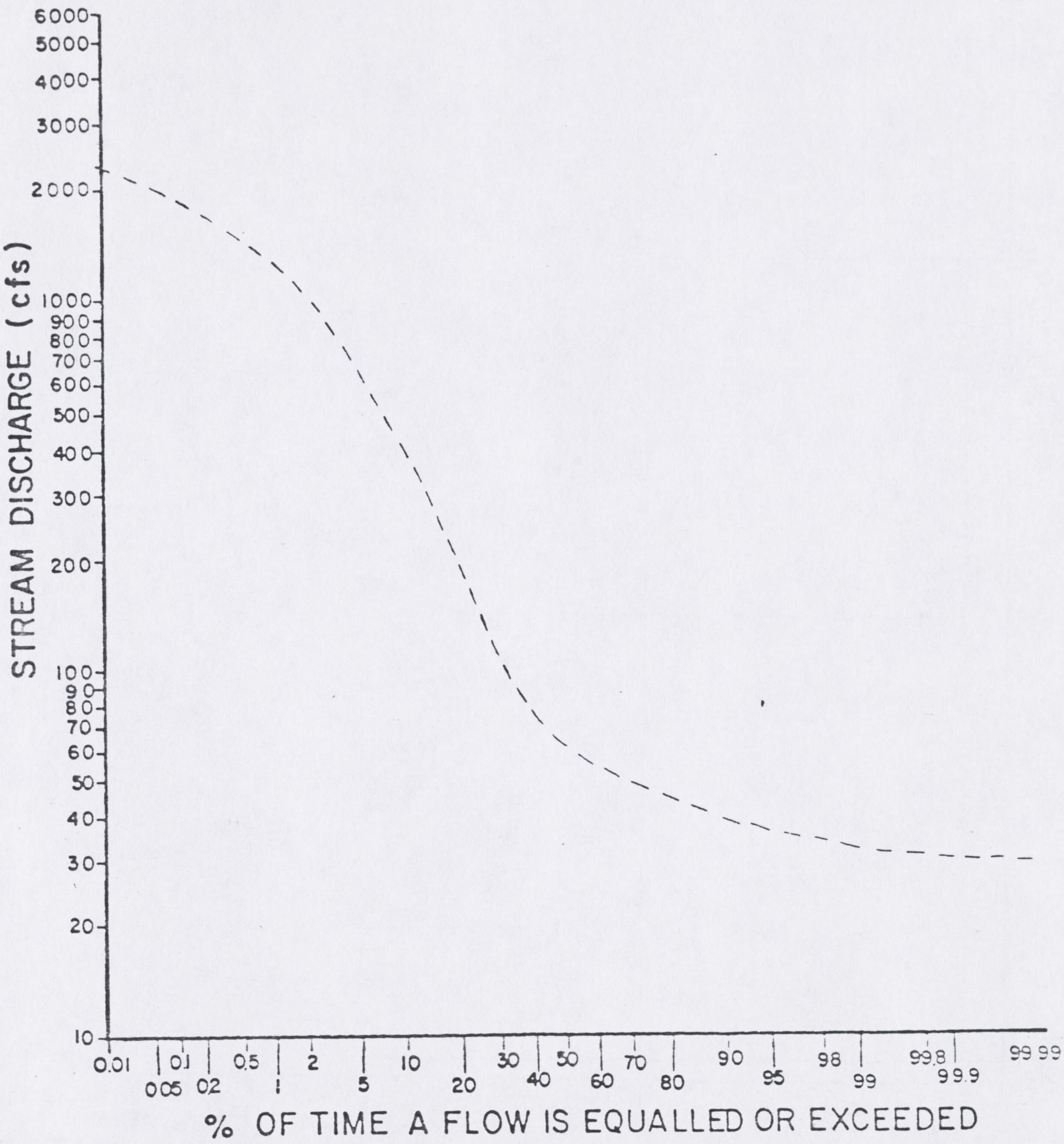

Figure 23.-- Flow duration curve for Tensleep Creek. 1911 to 1924 and 1944 to 1971. (Lowham, 1980 Personal Communication)

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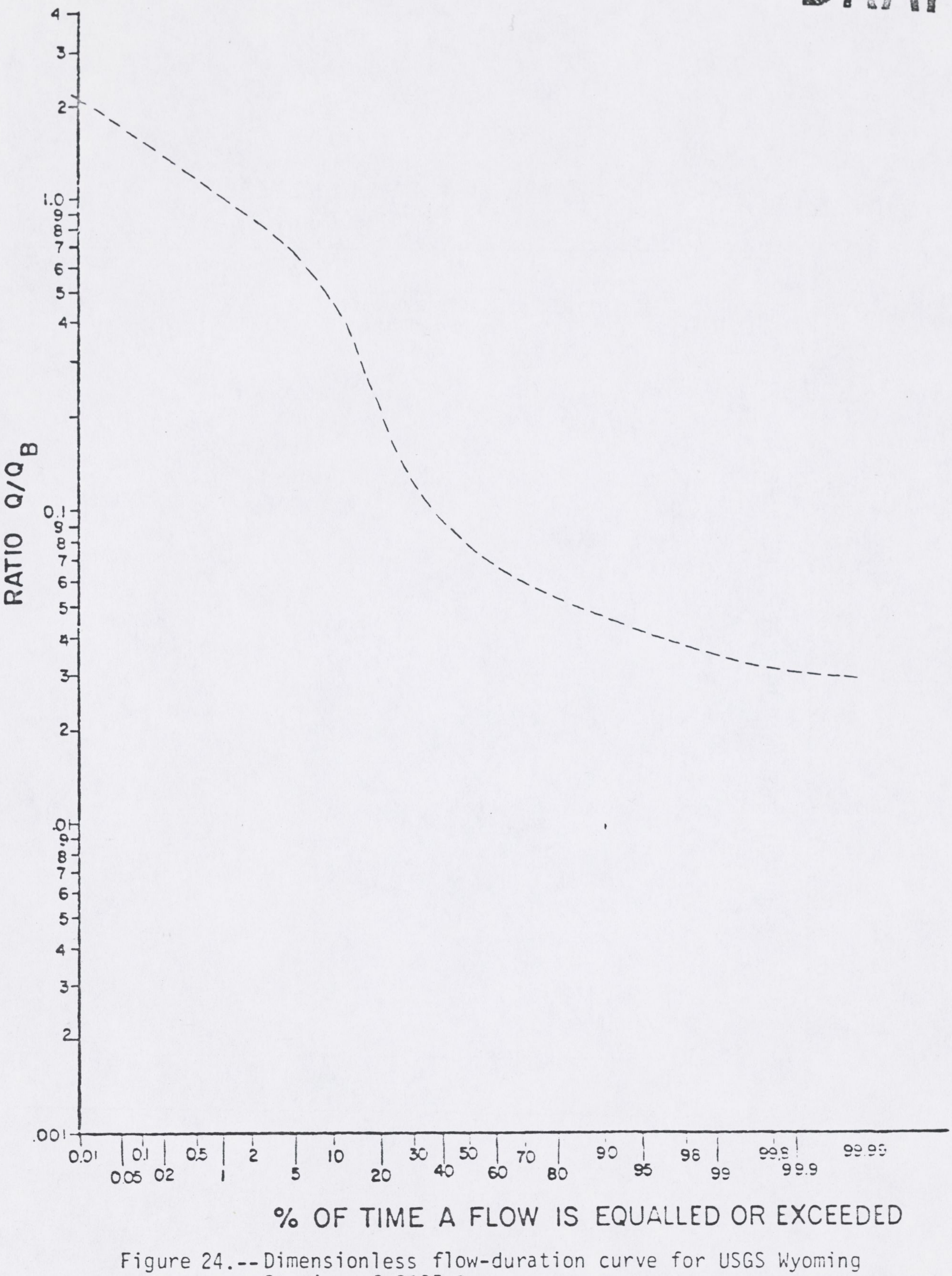

Figure 24.--Dimensionless flow-duration curve for USGS Wyoming Station, 6-2185-0.

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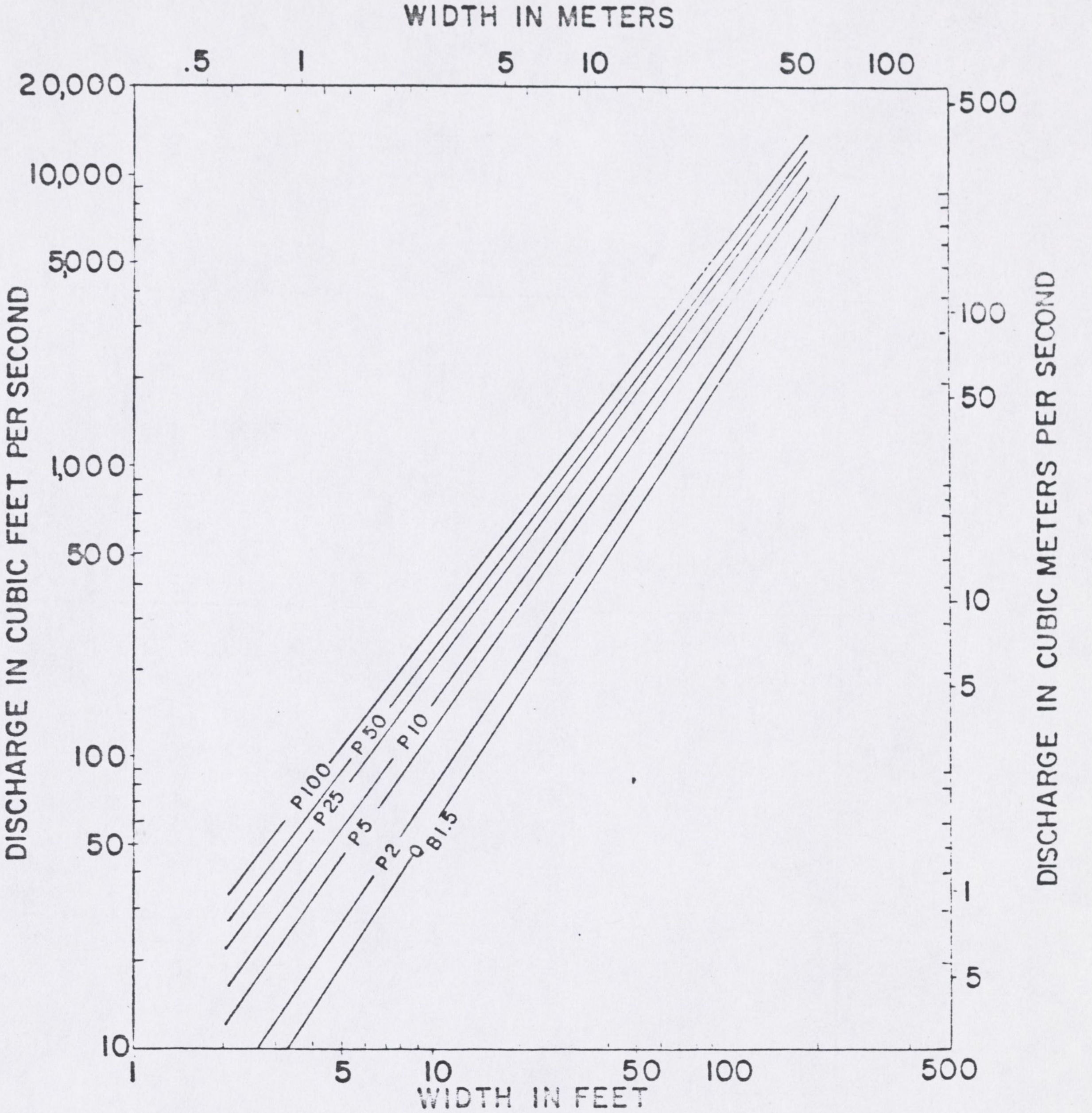

Figure 25.--Relationship of bankfull discharge ($Q_{B\ 1.5}$) to bankfull surface width compared to USGS data for streams in Region 1.

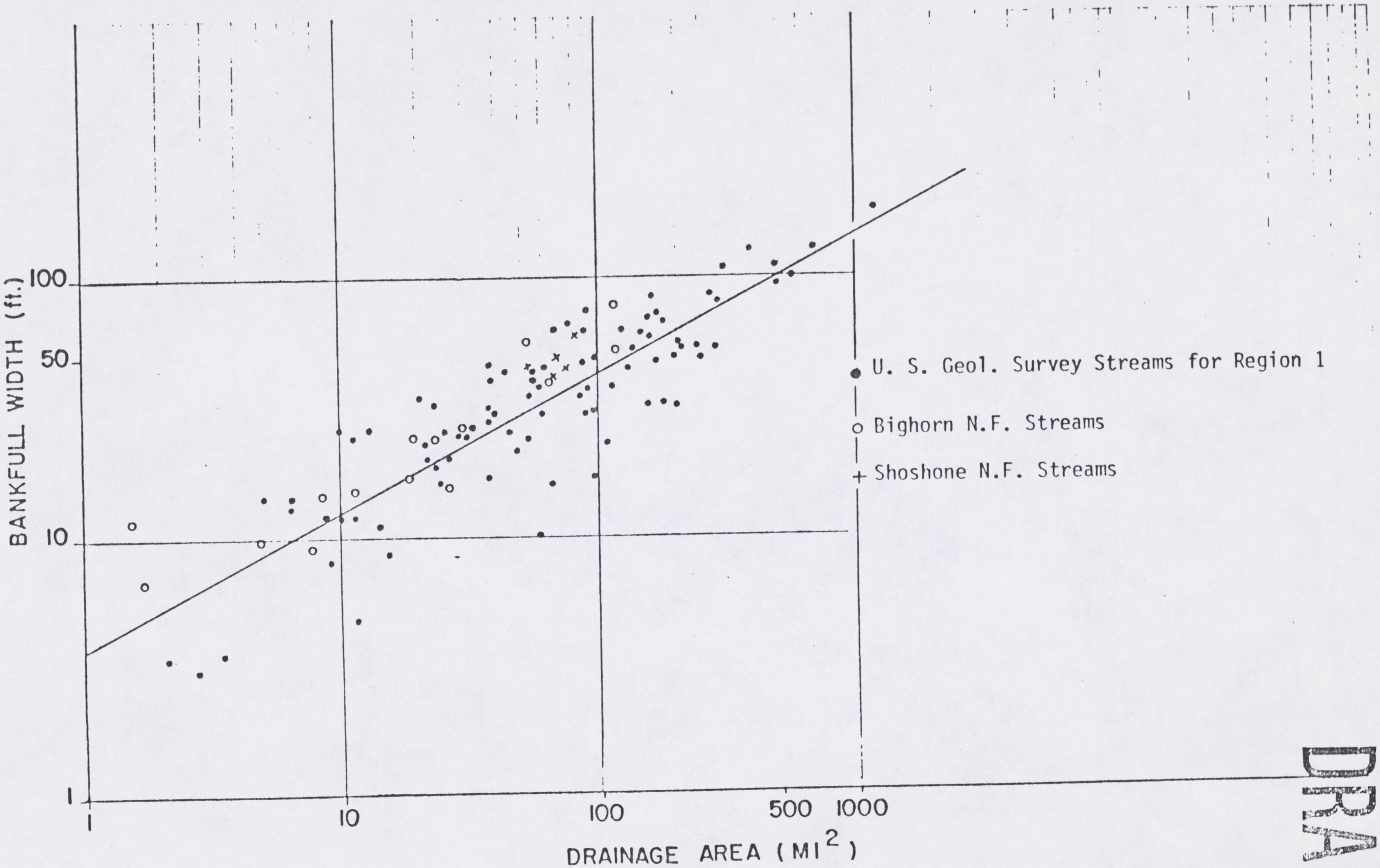

Figure 26.--Relationship of bankfull width to drainage area for streams in the Big Horn Basin and in USGS Region 1 for Wyoming.

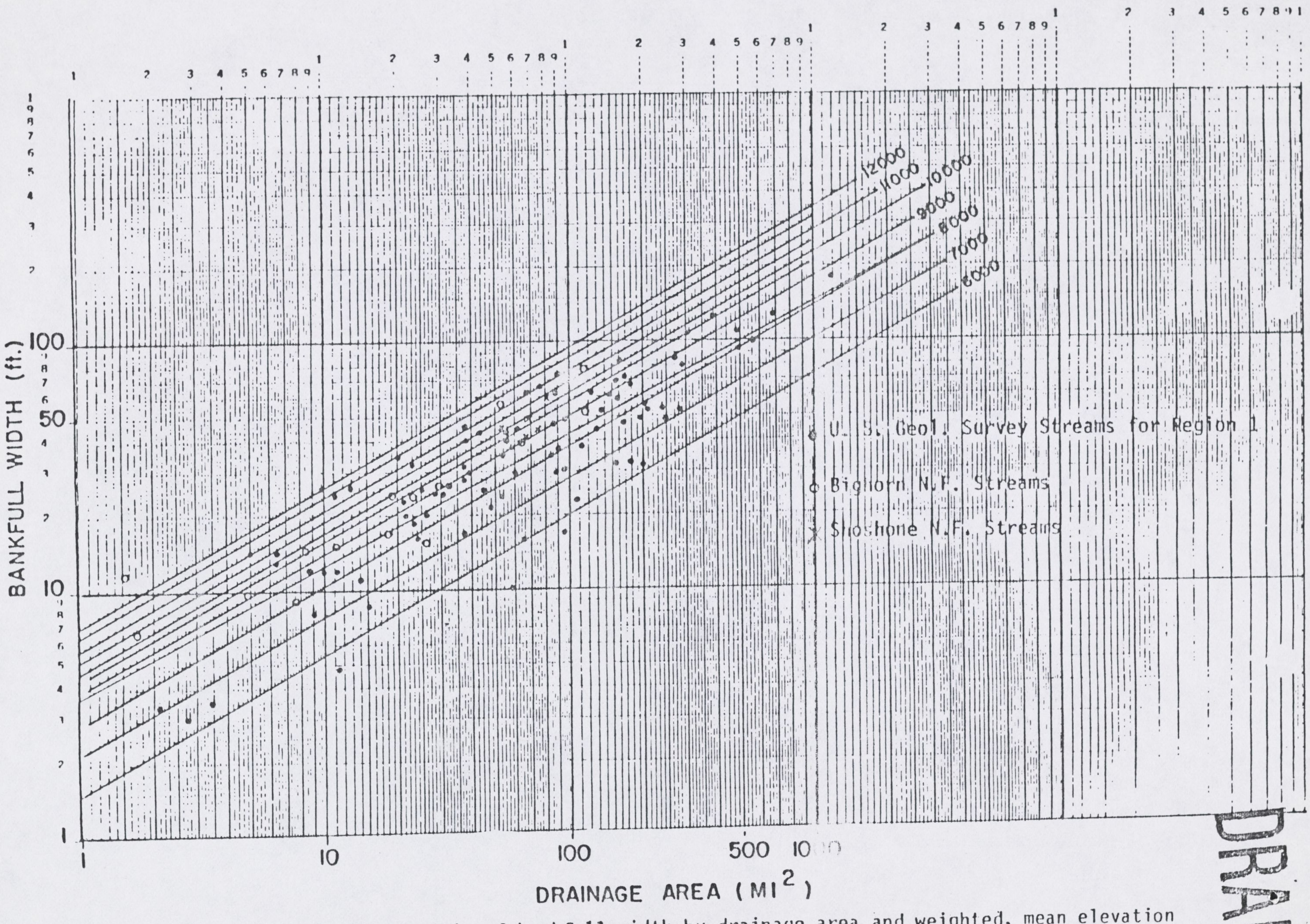

Figure 27.--Regional relationship of bankfull width by drainage area and weighted, mean elevation gaged streams for USGS Region 1 streams.

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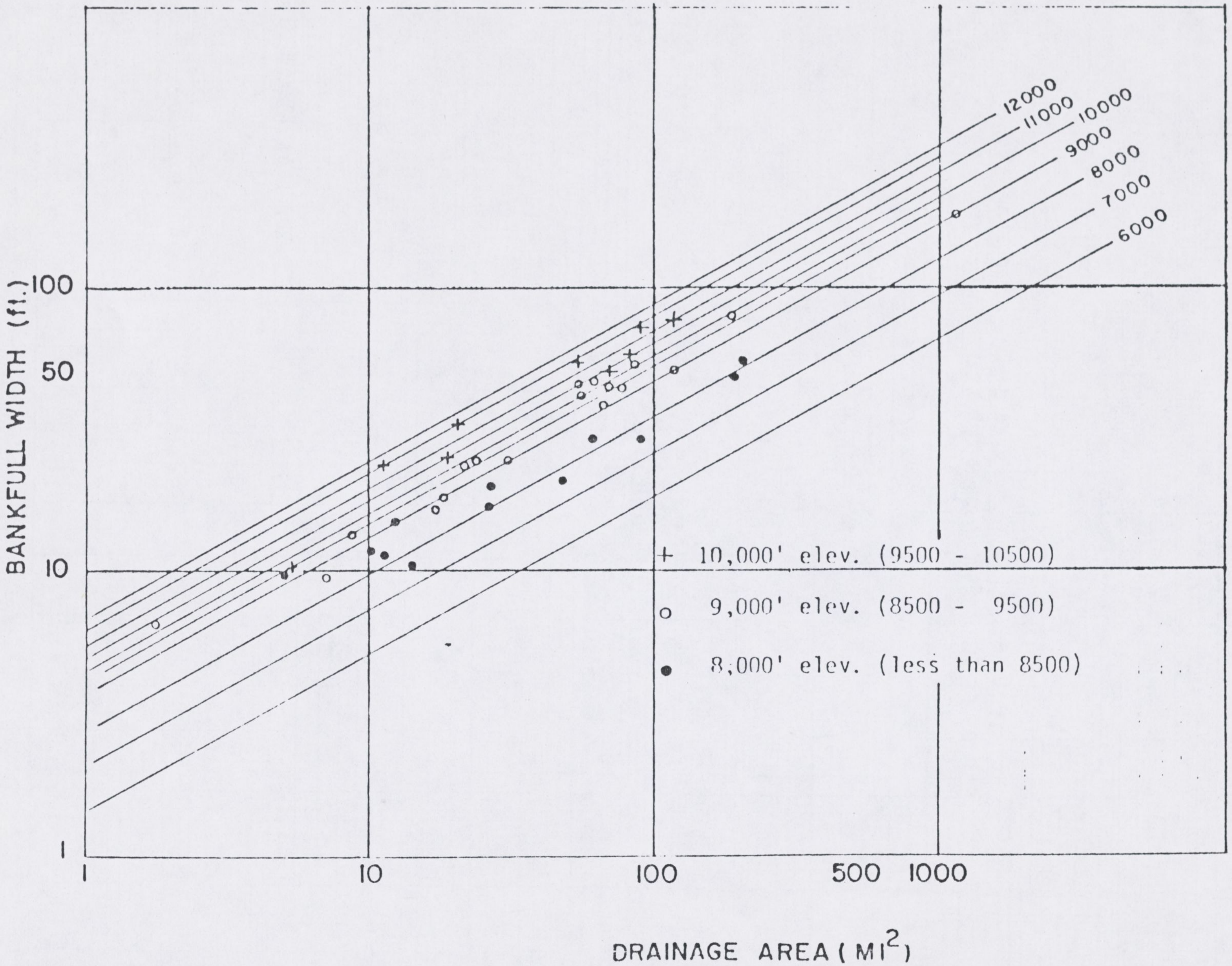

Figure 28.--Relationship of bankfull width by drainage area and weighted, mean elevation ungaged streams in the Big Horn River Basin.

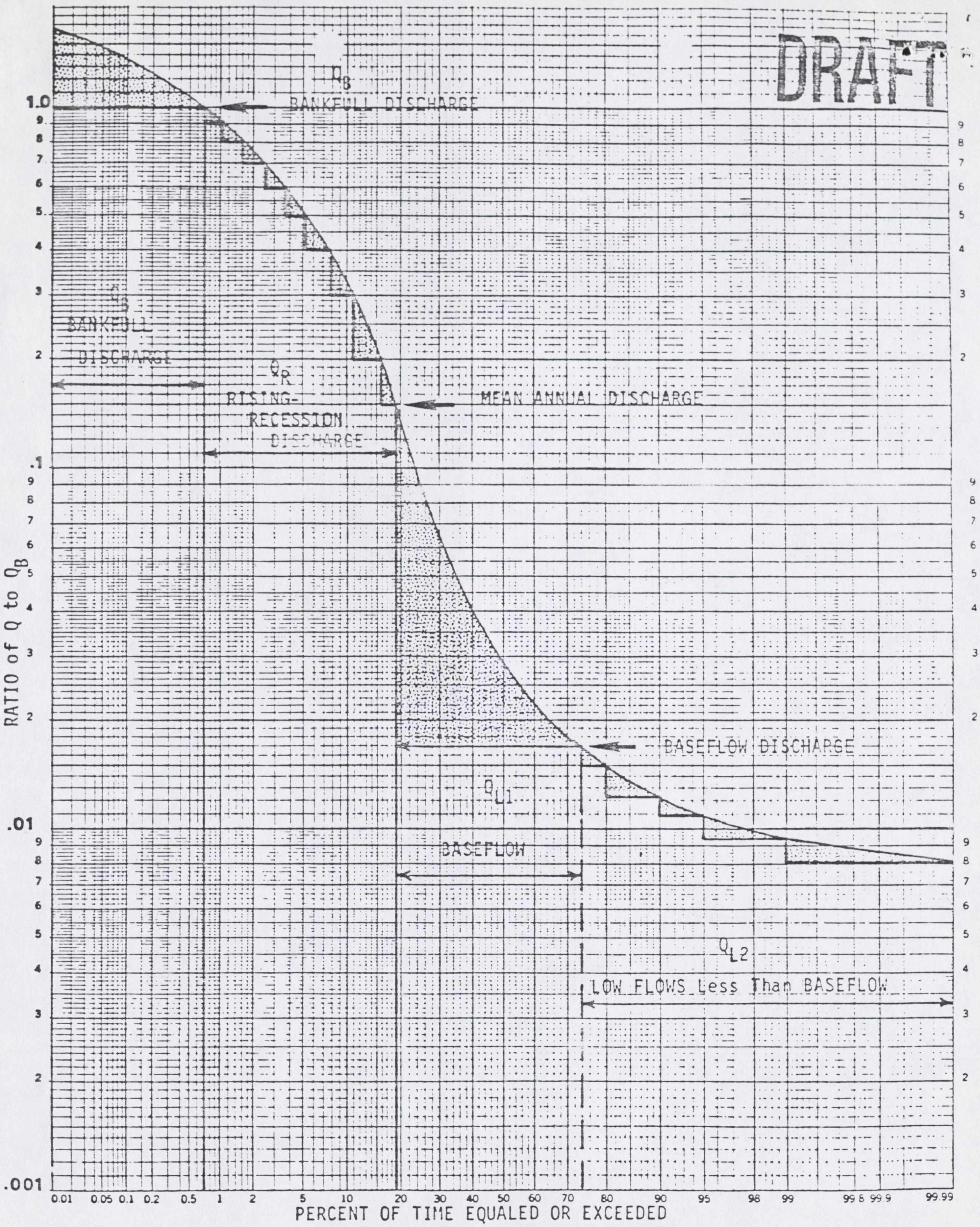

FIGURE 29. Regionalized dimensionless flow-duration curve; Big Horn River basin.

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APPENDIX I

Location of main-channel section for various types of streams. (Lowham, 1976).

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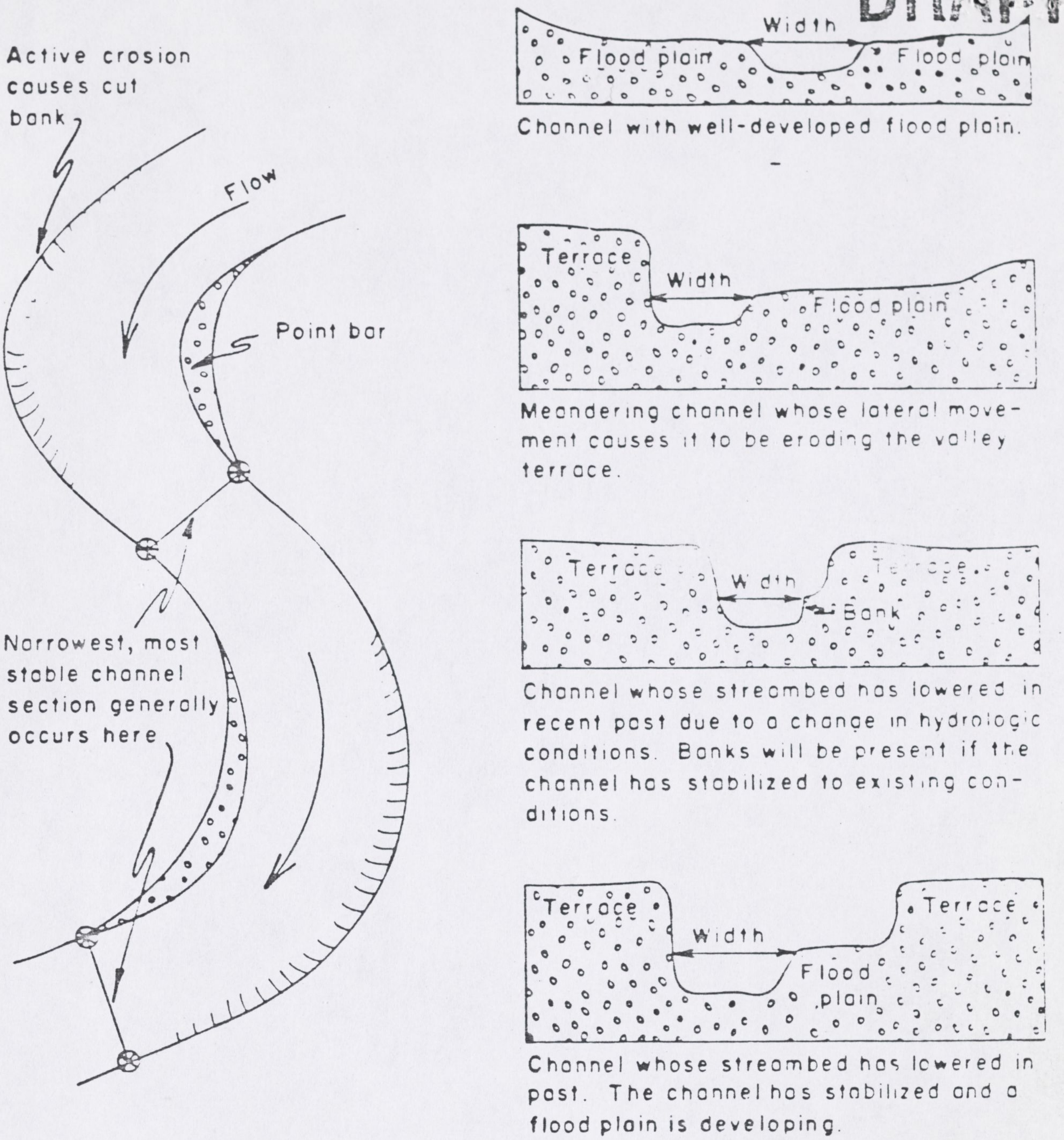

PLAN VIEW OF STREAM

CROSS SECTIONS OF VARIOUS TYPES OF STREAM CHANNELS

Figure 5.-Location of main-channel section for various types of streams.

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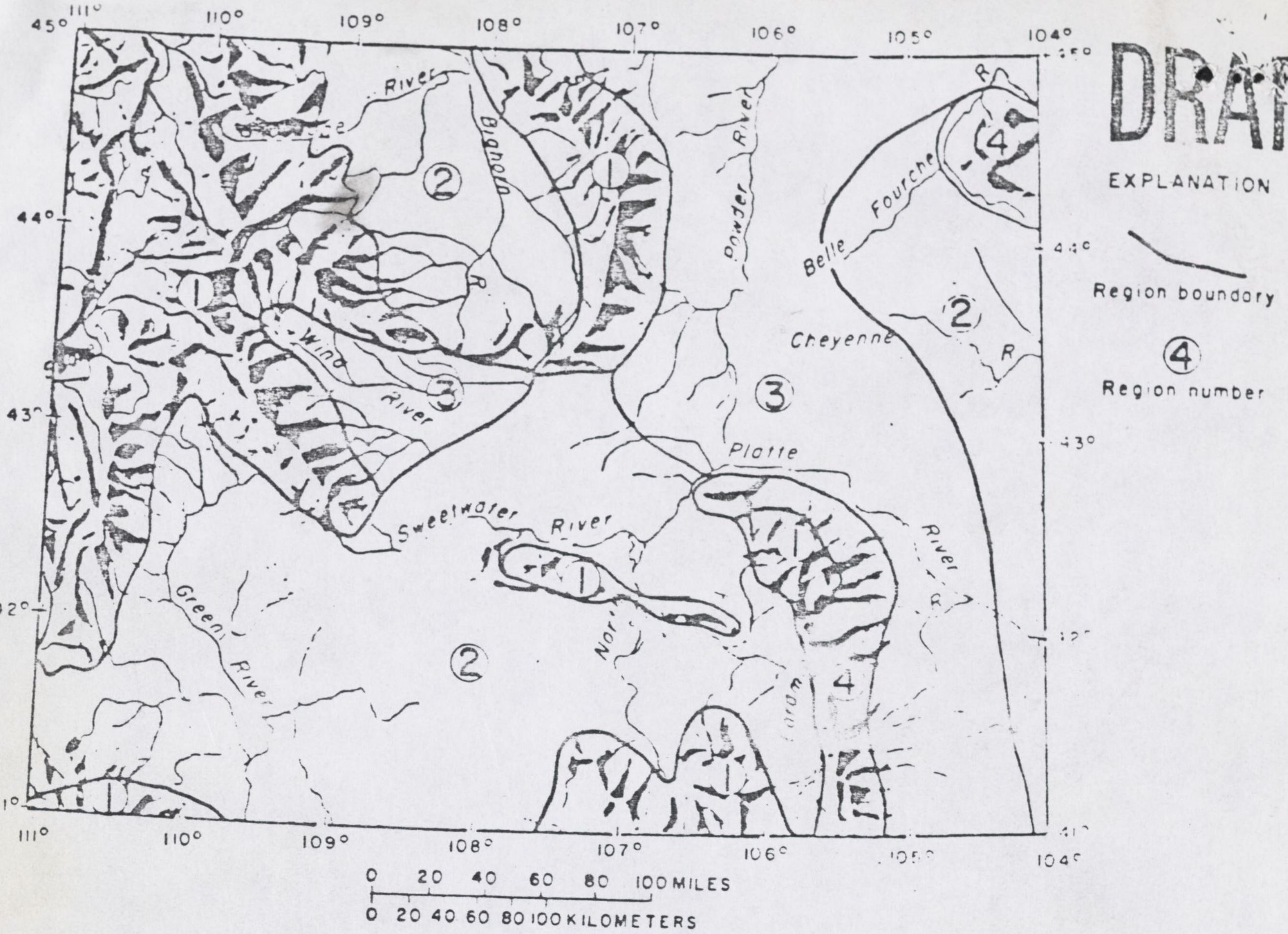
APPENDIX II

Vicinity Maps for Wyoming Study



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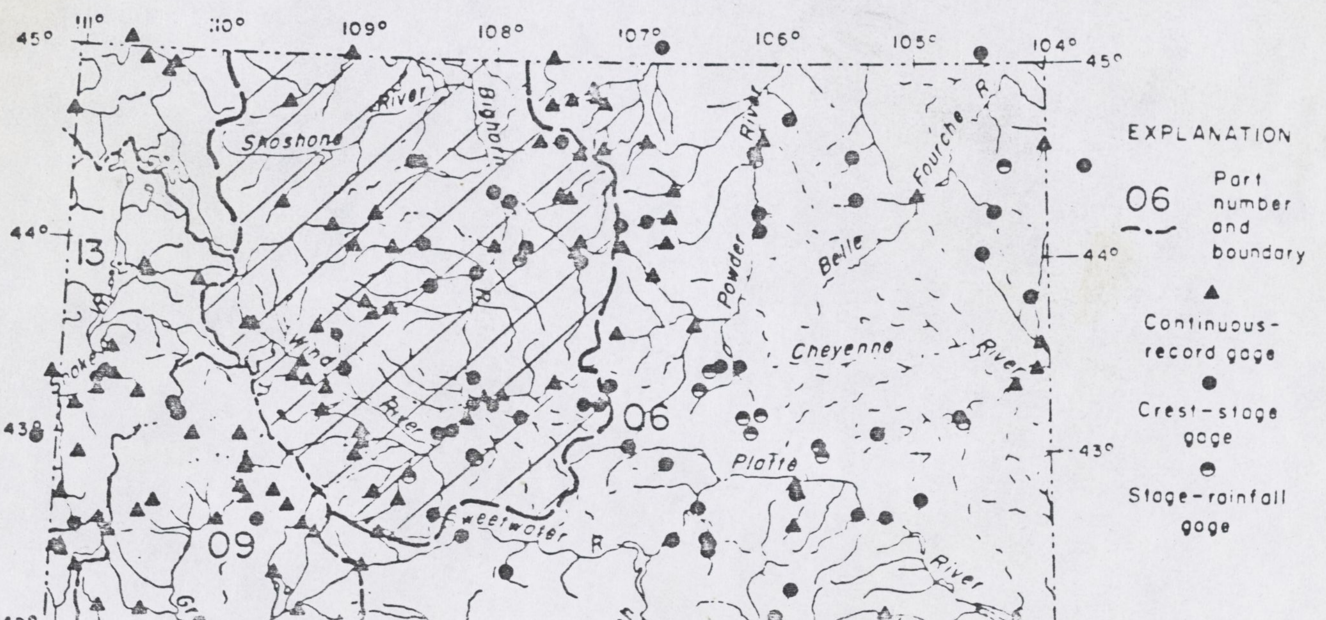
HYDROLOGIC REGIONS For USGS Stream Gage Data, For WYOMING. (Lowham,1976)



Locations of USGS Stream Gaging Stations Used in the Big Horn River Basin Analyses. (Lowham,1976)

[1986]
seminar guideline

STREAMFLOW PREDICTABILITY AND COMMUNITY STRUCTURE?

LeRoy Poff

9 December 1986

Abstract

Flow is one of the most characteristic attributes of streams, and, as such, it has played an important role in the evolution of the lotic fauna. Flow conditions in most streams are not static through time, although they frequently exhibit some type of periodicity. The natural variation in flow in a stream has important implications for the resident species, the fitness of which may be directly or indirectly influenced by a changing physical environment. Therefore, the patterns and magnitudes of flow fluctuations in streams are of interest to stream ecologists.

Streamflow variation can be considered a disturbance (*sensu* White & Pickett 1985) that can fall into one of two important categories: moderate, non-destructive fluctuations and extreme, destructive events (such as floods). The predictability of these moderate fluctuations can be evaluatecd mathematically using Colwell's index of predictability for periodic phenomena. Similarly, the frequency and periodicity of destructive flood conditions can be assessed for a stream, given a sufficient historical record and following accepted hydrological definitions and techniques of flood forecasting.

In this seminar, analyses of the daily flow records for 17 streams from across the continental United States will be presented. From these, quantitative regional comparisons of streamflow predictability (based on Colwell's index) and stream-specific histories of flood frequency and periodicity will be emphasized. These findings will be compared to regional generalizations commonly encountered in the stream ecology literature. Also, the theoretical implications of these findings for life history selection and community structure in streams will be addressed.

I. Introduction
  A. General consideration of patterns of flow
    1. Seasonal and year-to-year variations
    2. Watershed characteristics
  B. Ecologically important components of flow
    1. Pattern and extent of variation
      a. Measurement
    2. Occurrence and timing of extreme flows
  C. Implications of flow regimes for lotic biota
    1. Examples from the literature
  D. Current generalizations concerning regional distribution and significance of flow patterns
    1. Variable flow as "disturbance" mediating ecological processes
    2. Eastern vs. Western streams
  E. Purpose of presentation
    1. Attempt to examine *quantitatively* flow patterns from 17 different streams around the U.S.
    2. Compare results to literature generalizations
    3. Explore implications for life history selection and community structure in streams

II. Methods
  A. Stream selection criteria
  B. Necessary assumptions and confounding factors
  C. Analysis of data (daily flow records)
    1. Variable of interest -- "disturbance"
      a. Definition
      b. Two types
        i. Environmental "fluctuations"
          a. Measurement using Colwell's Index of Predictability
        ii. Destructive events
          a. Measurement using a graphical analysis of return times and periodicities
    2. Hydrologic and biological justification for flow categories used
      a. Environmental fluctuations -- 11 categories for Colwell's
      b. Destructive flow -- mean of the ranked annual peak flows occurring over the entire period of record (Morisawa 1968)

III. Results
  A. Predictability of Environmental Fluctuations
    1. Regional comparisons
    2. Patterns and generalizations
  B. Analysis of destructive flow patterns
    1. Return times and recurrence probabilities
    2. Periodicities and "predictabilities"

IV. Speculations concerning importance of flow patterns in structuring stream communities
  A. Emphasis on interactions between frequency and "predictability" of destructive flows

V. Conclusions
  A. Advantages of quantitative approach
    1. Testable community hypotheses?
  B. Comment on literature generalizations

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STREAM LOCATIONS

PREDICTABILITY

CONSTANCY / PREDICTABILITY	.65-.80	.38-.60	.30-.38
.78-.97	Augusta Cr. Blackearth Cr. Colorado R. East R. Upper 3 Runs Cr.	Brandywine Cr. McKenzie R. Piceance Cr. Satilla R.	
.45-.69		Martis Cr. Owego Cr. Prosser Cr. 1 Prosser Cr. 2 Sagehen Cr. Saline R. Smith R.	
.09-.32			Buckhorn Cr. Lusk Cr.

STATISTICAL SUMMARY

Stream	State	POR	Pred	C/P	M/P	Antilog Mean Q (cfs)	Mean Q/ Basin area	Annual CV ($\bar{x}$/s)	Antilog Mean T_r (days)	1 yr Prob
East R.	CO	63	.798	.68	.32	162.4	19.4	4.8	162	.33
Augusta Cr.	MI	21	.759	.92	.08	41.3	36.6	11.8	166	.34
Upper 3 Runs Cr,	SC	18	.733	.97	.03	104.6	41.5	25.4	446	.59
Blackearth Cr.	WI	31	.690	.95	.05	30.0	22.7	11.1	234	.39
Colorado R.	CO	21	.650	.66	.34	100.5	10.7	4.8	161	.31
Brandywine Cr.	PA	20	.602	.86	.14	64.1	36.5	6.0	214	.35
Prosser Cr. 1	CA	19	.583	.45	.55	35.5	23.2	3.1	151	.32
McKenzie R.	OR	41	.563	.78	.22	420.0	156.8	14.3	162	.32
Owego Cr.	NY	48	.533	.70	.30	116.7	21.8	3.8	245	.39
Smith R.	OR	25	.524	.55	.45	44.7	95.2	3.0	214	.35
Piceance Cr.	CO	21	.457	.66	.34	20.1	1.4	5.6	98	.27
Prosser Cr. 2	CA	22	.446	.70	.30	50.9	33.2	3.6	93	.20
Satilla R.	GA	48	.434	.83	.17	337.0	9.7	4.2	380	.52
Sagehen Cr.	CA	32	.430	.55	.45	7.9	26.0	2.6	162	.29
Martis Cr.	CA	27	.405	.62	.38	15.0	13.0	3.3	151	.25
Saline R.	IL	20	.377	.51	.49	40.0	9.4	2.4	145	.29
Lusk Cr.	IL	18	.301	.09	.91	11.3	9.0	1.4	20	.001
Buckhorn Cr.	NC	13	.298	.32	.68	18.9	8.6	2.2	46	.04

Disturbance : Any relatively discrete event in time that disrupts ecosystem, community, or population structure and changes resources, substrate availability, or the physical environment.

(White & Pickett 1985)

Two General Categories

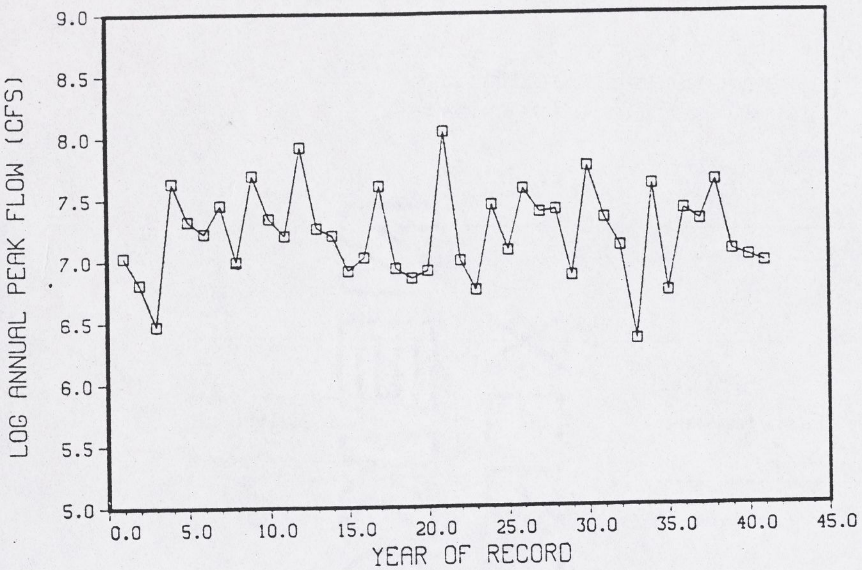
Environmental Fluctuations

Non-destructive
Diffuse, density-dependent effects
Predictable or non-predictable
Analysis : Colwell's Index

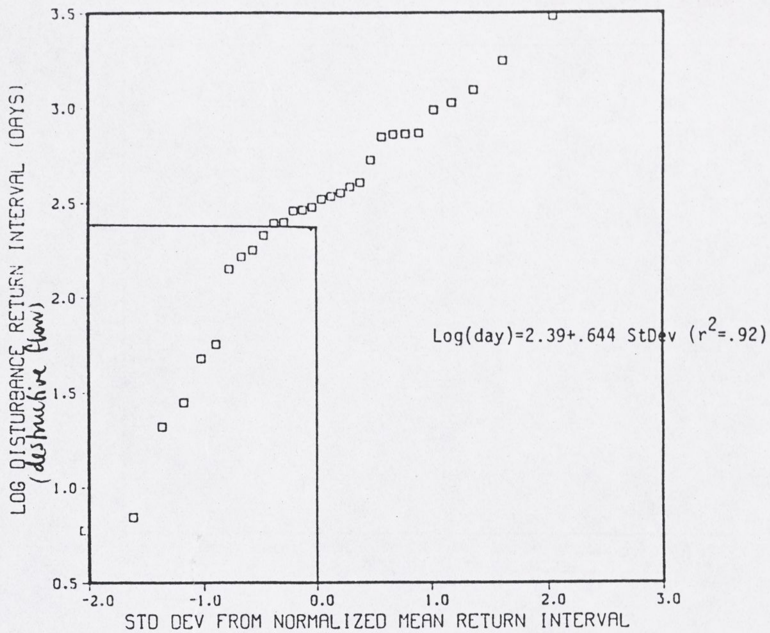
Destructive Events

Physical extreme
Direct, density-independent effects
Predictable or non-predictable
Analysis : Graphical "time-series"

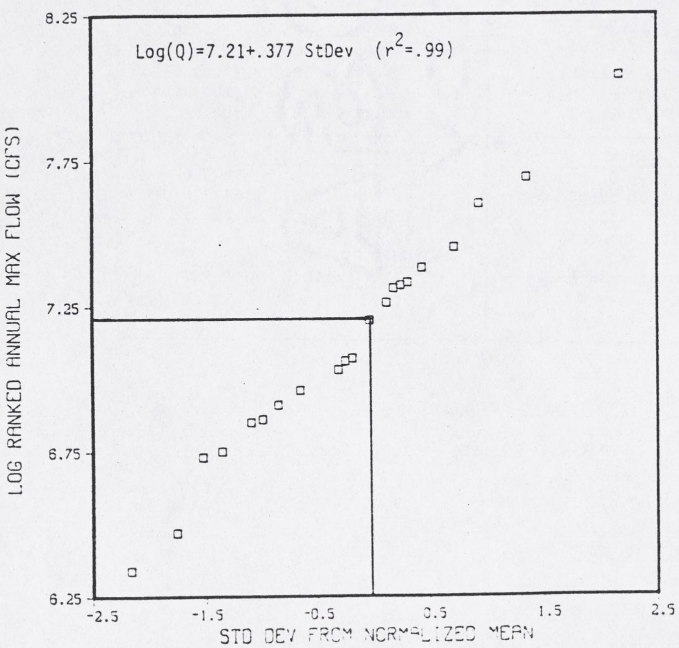
CHRONOLOGY OF ANNUAL PEAK FLOWS



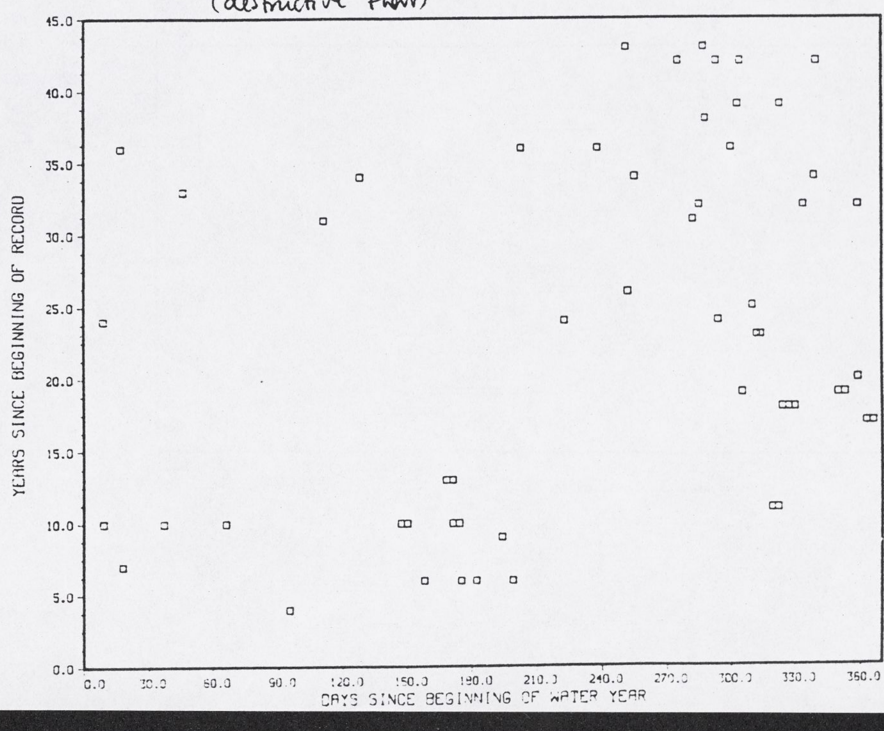
RETURN INTERVALS FOR BUCKHORN



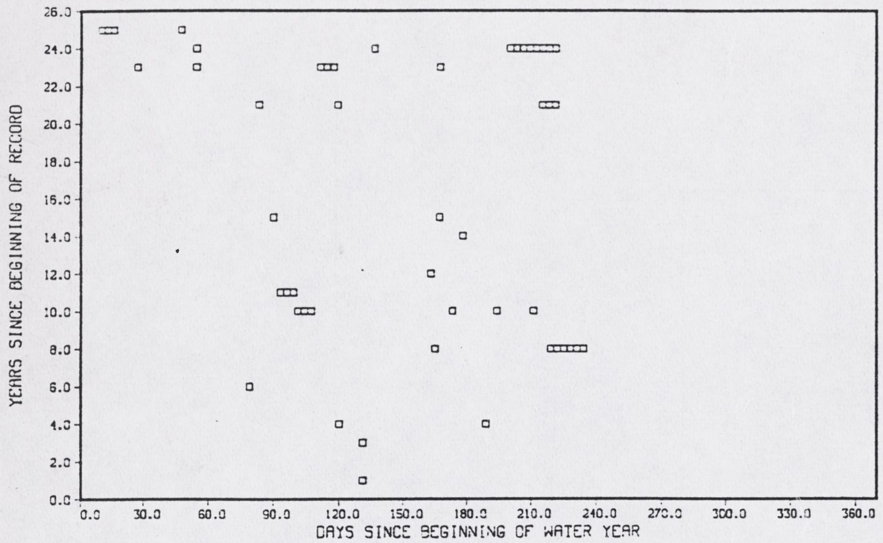
DETERMINATION OF DISTURBANCE FLOW
(destructive flows)



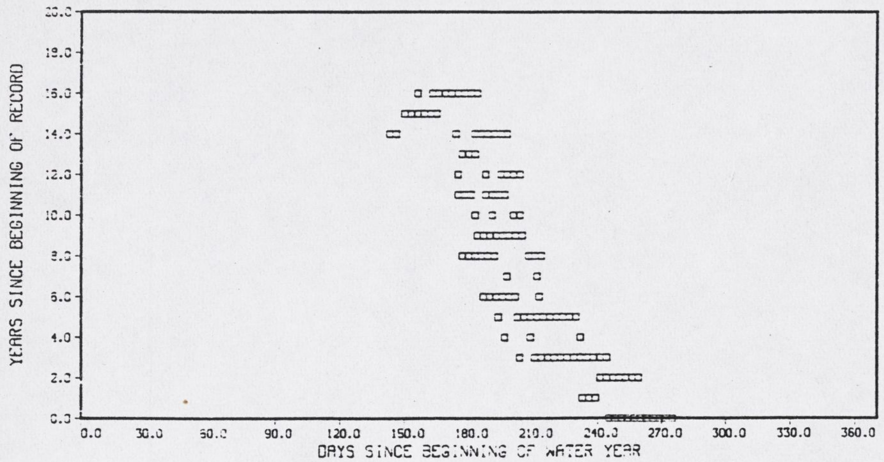
DISTURBANCE HISTORY FOR SATILLA RIVER
(destructive flow)



DISTURBANCE HISTORY FOR MARTIS CREEK
(destructive flow)



COLORADO RIVER DISTURBANCE HISTORY
(destructive flow)



HYPOTHETICAL SPECIES RESPONSE RANGES ALONG CONTINUUM OF LIFE HISTORY TRAITS

```
                                    /-------------- sp. F ----------------/
                /--------------------- sp. E ----------------/
/-------------------- sp. D -----------------/
/---- sp. A ----/              /---- sp. B ----/             /---- sp. C ----/

/-------------------------------------------------------------------------/
"r"                                                                    "K"
```

"r"	"K"
High development rate (multiple voltinism, resistance) Small adult size Inverts - "less" fecund Fish - "large" reproductive effort	Lower development rate (uni-,semivoltinism) Larger adult size Inverts - "more" fecund Fish - "small" repr. effort

DISTURBANCE PREDICTABILITY

DISTURBANCE PREDICTABILITY	LOW (frequency)	HIGH (frequency)
HIGH	<u>Predictable benign</u> <u>"K" > "r"</u> Life history adapt. to disturbance (not tightly synchronized) Semivoltinism <u>Diversity</u>:"variable" depending on intensity of ecol. interactions; Rel. constant in time <u>Stability</u>:High C, Low R	<u>Predictable harsh</u> <u>"K"< "r"</u> Life history adapt. to disturbance (tightly synchronized) Univoltinism <u>Diversity</u>:"variable"/"high" depending on intensity of ecol. interactions and timing of distubances <u>Stability</u>:High C, High R
LOW	<u>Unpredictable benign</u> <u>"K" and "r"</u> Few disturbance adapt. <u>Diversity</u>:"high" due to periodic disturbances; shifts in rel. abund. following disturbance <u>Stability</u>:Low C, Low R	<u>Unpredictable harsh</u> <u>"r"</u> Life history adaptation to disturbance is multi-voltinism <u>Diversity</u>:"low" <u>Stability</u>:Low C, High R

LOW HIGH

DISTURBANCE FREQUENCY

(destructive flow)

[1989]

The Columbia River — Toward a Holistic Understanding

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Abstract

EBEL, J. W., C. D. BECKER, J. W.MULLAN, AND H. L. RAYMOND. 1989. The Columbia River — toward a holistic understanding, p. 205–219. *In* D. P. Dodge [ed.] Proceedings of the International Large River Symposium. Can. Spec. Publ. Fish. Aquat. Sci. 106.

The Columbia River is one of the world's great rivers. It supports large runs of anadromous fish — several species of Pacific salmon and steelhead trout. Its watershed covers 671 000 km^2, including parts of British Columbia, Washington, Oregon, Idaho, Montana, and Wyoming, and the average annual flow rate at the river's outlet is about 6 655m^3• s^{-1}. Hydroelectric power, irrigation, and exploitation of regional resources other than water have greatly modified physical features throughout the Columbia River's vast system. Commercial and sport fishing, combined with alteration and degradation of riverine habitat, have reduced annual returns of anadromous fish from about 10 to 16 million originally to 2.5 million today. Efforts by management agencies to deal with the declines have focused on catch restriction, fish passage problems at dams, artificial propagation, habitat improvement, and identification of stocks at sea. Today, management is a joint effort by federal, regional, and state agencies and Indian tribes. Increased returns of anadromous fish to the river since 1980 are encouraging, but much remains to be done.

Résumé

EBEL, W. J., C. D. BECKER, J. W. MULLAN, AND H. L. RAYMOND. 1989. The Columbia River — toward a holistic understanding, p. 205–219. *In* D. P. Dodge [ed.] Proceedings of the International Large River Symposium. Can. Spec. Publ. Fish. Aquat. Sci. 106.

Le Columbia est l'un des grands fleuves du monde. Il s'y fait d'importantes remontées anadromes (plusieurs espèces de saumons du Pacifique et la truite arc-en-ciel). Il a un bassin hydrologique de 671 000 km^2, qui recouvre des parties de la Colombie-Britannique et des États de Washington, de l'Orégon, de l'Idaho, du Montana, et du Wyoming; il a un débit annuel moyen de 6655 m^3•s^{-1}. Les ouvrages hydro-électriques, l'irrigation et l'exploitation des ressources régionales autres que l'eau ont profondément modifié les traits physiques dans l'ensemble de ce vaste bassin. Les pêches commerciales et sportives, en plus de la transformation et de la dégradation des habitats riverains, ont eu pour effet de réduire les remontées annuelles, de 10 à 16 millions à l'origine jusqu'à 2,5 millions aujourd'hui. En réaction, les organismes de réglementation ont fait porter leurs efforts sur les limites de capture, les problèmes de remonte aux barrages, la multiplication par des moyens artificiels, la remise en état de l'habitat et l'identification des stocks en mer. La gestion des stocks est devenue une entreprise conjointe où sont réunis les efforts des gouvernements fédéral et régionaux, des services d'États et des bandes indiennes. Les remontées plus abondantes dans le fleuve depuis 1980 sont encourageantes, mais il reste encore beaucoup de travail à faire.

Introduction

The Columbia River is one of the world's great rivers. It drains 671 000 km^2, (259 000 mi^2) and discharges over twice the amount of water as the Nile River in Egypt. The Columbia River also produces large runs of Pacific salmon (*Oncorhynchus* spp.) and steelhead trout (*Salmo gairdneri*), and has served as a focal point for the evolution of northwest native cultures dependent on these fish. The river's discovery in 1792 by Captain Robert Gray and its exploration in 1805 by Lewis and Clark set in motion changes that profoundly altered the river and its watershed.

The river's capacity for sustained production of salmonids was greatest prior to 1930. Before encroachment by white settlers, the aboriginal fishery was estimated to take about 8.2×10^6 kg (18 million lb) (Craig and Hacker 1950) or 11.3×10^6 kg (25 million lb) (Hewes 1972) of fish each year. During the peak period of commercial fishing (1916 to 1920), catches exceeded 18.1×10^6 kg (40 million lb) each year. Even today, with the runs depressed, the annual combined catch of commercial, sport, and tribal fisheries exceeds 9.1×10^6 kg (20 million lb).

In terms of numbers, salmon and steelhead runs ranged from about 10 to 16 million fish, annually before major development of the Columbia River Basin (Northwest Power Planning Council 1986). Current runs average about 2.5 million fish, indicating that basin-wide losses have been about 7 to 14 million fish. Chief Joseph Dam on the Columbia River and Hells Canyon Dam on a major tributary, the Snake River, blocked return runs, and eliminated all habitat for anadromous fish production above them. Declines in runs of anadromous fish have been greatest in the upper Columbia and Snake rivers because of habitat loss and mortalities of upstream and downstream migrants at dams.

The history of the Pacific Northwest is marked by conflicts among fishermen, and between fishermen and other users over control of the Columbia River. Overfishing and resource allocation have been continuing problems. Economic development has, over the years, degraded or eliminated habitat and thereby decreased the system's capacity to produce anadromous fish. Hydropower leads the list, but agriculture and irrigation, logging, mining, stream channelization and clearing, and water pollution have all altered the river's ecosystem.

In this report, we first describe the ecological features of the Columbia River. We then focus on the salmonid resources: commercial and sport fisheries, effects of regional development and exploitation, smolt passage problems, artificial propagation, and institutional arrangements for management. We review needs and opportunities related to salmonid production.

Morphometry

The Columbia River begins at Columbia Lake in the Canadian Rockies. The river flows northwesterly in British Columbia for about 306 km (190 mi), then south 436 km (271 mi) across the Okanogan Highlands to Trail, British Columbia. It continues south across the international border to receive the Spokane River, then curves westward over the semi-arid Columbia Plateau to receive the Snake River near the Washington/Oregon border. At this point, the river turns west and flows about 483 km (300 mi) through the Cascade and Coast ranges to enter the Pacific Ocean near Astoria, Oregon (Fig. 1).

From source to outlet, the Columbia River extends over 1930 km (1200 mi) and drops 808 m (2650 ft). It passes through four mountain ranges: the Rockies, Selkirks, Cascades, and Coast; traverses several climatic zones from alpine to shrub-steppe to coastal; and receives flows from several large tributaries before discharging to the sea. The Snake River, the largest tributary, extends 1671 km (1038 mi) and drains 49 % of the system's watershed in the United States.

The Cascade Range forms a mountainous barrier to the passage of moisture inland from the Pacific Ocean. East of the Cascades is an open landscape, the Columbia Plateau, which was formed over millions of years from discontinuous flows of lava that solidified as basalt in nearly horizontal layers. As a result, parts of the mainstem Columbia and Snake rivers are entrenched in spectacular gorges.

Hydrology

The Columbia's average annual flow rate at its outlet is about 6655 $m^3 \cdot s^{-1}$ (235 000 $ft^3 \cdot s^{-1}$). Discharges from the Snake River average about 1300 $m^3 \cdot s^{-1}$ (46 000 $ft^3 \cdot s^{-1}$) annually (Pacific Northwest Regional Commission 1979). Nearly 25 % of the Columbia's total runoff originates west of the Cascade Range, an area less than 10 % of the total drainage, because of its higher precipitation.

Major tributaries of the mainstem Columbia River are: the Kootenai and Pend Oreille rivers in Canada; the Spokane, Okanogan, Wenatchee, Yakima, Snake, Cowlitz, and Lewis rivers in Washington; and the Umatilla, John Day, Deschutes, and Willamette rivers in Oregon. The interior drainage area extends to Idaho, Montana, and Wyoming.

In general, tributaries of the Columbia River originate in high, forested mountains where the climate is mesic, the gradient is steep, stream velocity is high, and scouring occurs. In the Columbia Basin proper, the climate is xeric, the gradient is less steep, stream flow is reduced, and sediment is deposited seasonally. The Cascade Range near the river's mouth is forested and receives heavy rainfall.

The main factors influencing hydrographs of Columbia River tributaries are changes in seasonal runoff and irrigation withdrawals. Hydrographs of the mainstem are influenced primarily by storage and release of water from impoundments for hydroelectric power production.

Mainstem Flow Regimes

Spring flows in the Columbia River are triggered by snowmelt and rain in headwater areas. Precipitation, primarily in the form of snow, is greatest in winter, and runoff increases with snowmelt during spring and early summer. Most major floods on tributaries east of the Cascades result from rapid snowmelt. The most severe spates are often accentuated by heavy, warm rain or warm wind. Convective storms accompanied by intense rainfall may also cause local floods.

Before impoundment of the mainstem, estimated discharges at the river's outlet averaged 18 690 $m^3 \cdot s^{-1}$ (660 000 $ft^3 \cdot s^{-1}$) from May through July and 1980 $m^3 \cdot s^{-1}$ (70 000 $ft^3 \cdot s^{-1}$) from September through March (Hickson and Rodolf 1957). Today, flows throughout the Columbia's drainage area are influenced by water storage projects. by 1973, the combined storage of Mica, Duncan, Arrow, Albeni Falls, Libby, Hungry Horse, and Grand Coulee dams (Fig. 1) provided capacity to store over 43 200 x 10 $^6 \cdot m^3$ of spring runoff for use later in the year when more electricity is needed. As a result, in most years,

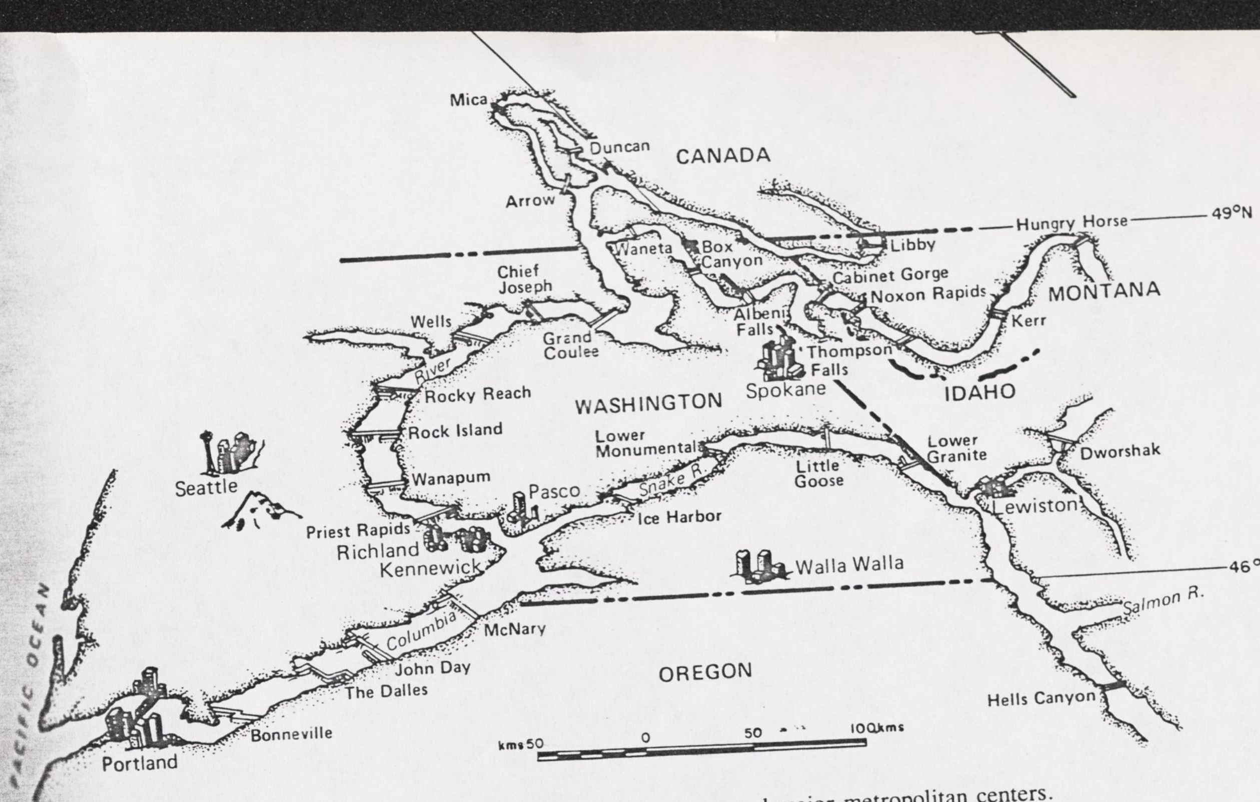

FIG. 1. The Columbia River system, showing the major tributaries, dams, and major metropolitan centers.

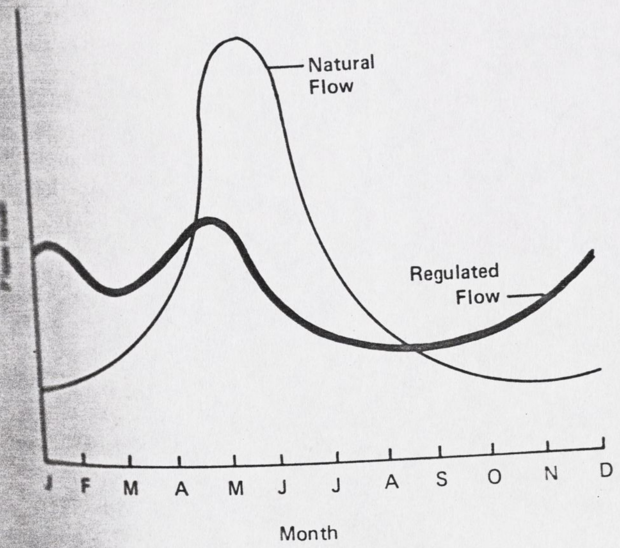

FIG. 2. Generalized effect of reservoir operations on mainstem Columbia River flows near The Dalles, Oregon. Tributary storage of water and mainstem production of hydropower eliminated the spring peak runoff that once transported juvenile salmonids downstream to the Pacific Ocean.

flows in the Columbia River when young salmonids migrate to sea in May and June have been reduced about 50% (Fig. 2). The system's total "active storage" capacity of 33 800 x $10^6 \cdot m^3$ represents about a quarter of the average annual runoff (Table 1).

Lake McNaughton of the Mica Project in Canada is the largest storage reservoir on the Columbia River system, and it may remain unfilled after seasonal drawdown. The Arrow Lakes in Canada are also used primarily for storage. Lake Roosevelt, behind Grand Coulee Dam, is the major storage reservoir in the United States; it contains 6400 x $10^6 \cdot m^3$) of active storage but has a total volume of 11 800 x $10^6 \cdot m^3$. Essentially, the whole mainstem below Lake Roosevelt is influenced by the storage and hydraulic capacities of Grand Coulee Dam.

The flushing rate of Lake Roosevelt is about 45 days. Below Grand Coulee Dam, flushing rates for river-run reservoirs vary from less than 1 day (Priest Rapids) to about 4 days (Lake Wallula). Current velocities in these impoundments average about 0.3 $m \cdot s^{-1}$. Lake Umatilla is primarily a storage reservoir, but it has a flushing rate of about 7 days.

Most storage reservoirs undergo major seasonal drawdown. For example, the elevation of Lake Roosevelt is lowered about 25 m (82 ft) each year prior to the spring spate (Stober et al. 1979). In contrast, water levels of most river-run reservoirs may fluctuate 0.3 to 1.5 m (1 to 5 ft) daily in response to power generation at their outlet dams.

The last unimpounded section of the mainstem of the Columbia River is the Hanford Reach, a 80.5 km (50 mi) section between the head of Lake Wallula and Priest Rapids Dam. It is not "free-flowing," but regulated by discharges at and above Priest Rapids Dam.

Discharge volumes from mainstem dams generally increase downstream. This is because of reduced reservoir storage ratios and increments of water from tributaries. Annual discharges at Bonneville Dam average near 164 000

TABLE 1. Storage characteristics of mainstem Columbia River reservoirs.

Dam	Reservoir (lake)	Location (RKM)	Length (km)	Total volume[a] ($\times 10^6 \cdot m^3$)	Mean annual discharge ($\times 10^6 \cdot m^3 \cdot yr^{-1}$)	Storage ratio[b]	Flushing rate (days)[c]
Mica	McNaughton	1638	209	25 040	18 260	1.37	499
Revelstoke	—	1498	129	1 480	70 440	0.02	7.3
Keenleyside	Arrow	1255	216	9 250	35 775	0.26	94
Grand Coulee	F. D. Roosevelt	960	243	11 800	96 220	0.12	45
Chief Joseph	Rufus Woods	877	71	616	96 470	0.007	2.6
Wells	Pateros	830	45	370	100 420	0.004	—
Rocky Reach	Entiat	761	68	493	102 390	0.005	1.8
Rock Island	Rock Island	729	34	123	105 600	0.001	—
Wanapum	Wanapum	668	61	740	105 600	0.007	2.6
Priest Rapids	Priest Rapids	639	29	247	105 720	0.002	0.7
McNary	Wallula	470	98	1 727	150 995	0.011	4.0
John Day	Umatilla	348	122	3 084	153 960	0.020	7.3
The Dalles	Celilo	309	39	370	158 890	0.003	1.1
Bonneville	Bonneville	325	72	616	163 700	0.004	1.5

[a] Total volume (table data) represents the maximum capacity of water storage in a reservoir, and is significant ecologically. Active storage (text data) represents only the storage capacity sufficient to provide daily or weekly streamflow regulations.

[b] Storage ratio (annual) = $\dfrac{\text{Total volume}}{\text{Mean annual discharge rate}}$

This is also called the exchange rate or flushing rate, and has a value in years, convertible to days.

[c] Flushing rate = annual storage ratio × 365 (days). This is the number of days required, theoretically, to completely empty a reservoir at the mean annual discharge rate.

$\times 10^6 \cdot m^3$. Also, reservoirs on the lower Columbia River are the widest and shallowest of the Columbia River system. The mean depth of Lake McNaughton near the river's origin is 58.5 m (192 ft), but Lake Bonneville near the river's outlet averages only 9 m (30 ft) deep. Flows below Bonneville Dam are under tidal influence.

Sedimentation

The drainage basin of the Columbia River contains a variety of igneous, metamorphic, and sedimentary rocks, as well as unconsolidated surficial deposits from ancient glaciers. Upstream, the sediments in Grand Coulee, Rocky Reach, Wanapum, and Priest Rapids reservoirs, are largely fine-grained, nonvolcanic, and carried in suspension (Whetten et al. 1969). Downstream, the sediments in Umatilla, Celilo, and Bonneville reservoirs are coarser, of andesitic volcanic origin, and make up most of the bedload. Erosion in the headwaters tends to be rapid because most andesitic formations are poorly consolidated and the local gradient is steep.

Amounts of suspended sediment in the mainstem of the Columbia river vary seasonally with input from tributaries, of which the Snake River is the greatest contributor. Most sediment is transported downstream during a few days or weeks of high spring discharge. During average or low flows, sediment is deposited in impoundments and slackwater areas. Much of this material is resuspended during high flows. Thus, maximum sediment loads enter the pacific Ocean during late spring and early summer, the period of maximum water discharge (Whetten et al. 1969).

Little sediment accumulates on the bed of the Columbia River except in slackwater areas and below Bonneville Dam. The river bed between reservoirs is either scoured to bedrock or covered with a thin deposit of coarse gravel. Bedload transport is evident only in the lower Columbia River and the amount transported is probably small, about 10 % of the total sediment load exclusive of dissolved materials (Whetten et al. 1969).

The Columbia River discharges about 10^7 t of sediment each year (Nittrouer et al. 1979). However, fine sediment is not deposited to any extent in the Columbia River estuary. Substrate in the estuary consists of about 1 % gravel, 84 % sand, 13 % silt, and 2 % clay; silt accumulates in only about 10 % of the estuary (Hubbell et al. 1972). Beyond the estuary, bottom currents along the shore remain northward throughout the year. Thus, most sediment leaving the Columbia River is carried northward. Sand tends to accumulate nearshore at <60 m depth, and most silt settles on the midshelf at the 60- to 120-m depth (McManus 1972).

Water Quality

The State of Washington has designated the mainstem of the Columbia River as Class A, or excellent, for water quality standards. This designation means that the water is suitable for use by the public, industry, and agriculture; for rearing livestock, fish, and shellfish; and for wildlife habitat, recreation, and navigation.

Water in the Columbia River is a dilute calcium-magnesium, carbonate-bicarbonate type, with a total dissolved solids content of about 90 $mg \cdot L^{-1}$ (range 71 to 158 $mg \cdot L^{-1}$), from the international border downstream to the confluence with the Snake River. Tributaries that drain the eastern parts of the Columbia Plateau are more mineralized from extensive irrigation and the higher amounts of solutes available from semi-arid land. Consequently, moderately higher mineralization of the mainstem of the Columbia occurs below the outlet of the Snake River.

Water quality in some tributaries used extensively for irrigation may be degraded by return flows from agricultural lands. For example, the lower portion of the Yakima River is seasonally laden with nutrients, pesticides, and coliform bacteria, and reaches temperatures about 4°C above levels expected otherwise. The outlets of the Okanogan and Umatilla rivers show similar impairment (Stober et al. 1979).

Mainstem Temperatures

Temperatures in the Columbia River are lowest in January and February and highest in August and September. The river is warmest near its outlet, where temperatures usually peak near 21°C. Thermal regimes in tributaries throughout the drainage basin differ widely with location, elevation, and input from rainfall, snowmelt, glaciers, and aquifers.

Studies in the 1960's showed that the construction of river-run reservoirs on the mainstem of the Columbia River caused no significant changes in the average annual water temperature. However, storage and release of water from Lake Roosevelt had delayed the timing of peak summer temperatures below Grand Coulee Dam since 1941. This delay was about 30 days at Rock Island Dam and was reflected, to a lesser extent, as far downstream as Bonneville Dam near the river's outlet. Temperature extremes were moderated by the reservoir complex so that the river below Grand Coulee Dam today is slightly cooler in summer and slightly warmer in winter (Jaske and Goebel 1967; Jaske and Synoground 1970).

Historically, average temperatures at the mouth of the Snake River during August and September have always been a few degrees higher than those in the mainstem Columbia (Roebeck et al. 1954; Jaske and Synoground 1970). During late summer of some years, high water temperatures (20° to 22°C) and low dissolved oxygen levels (<6 $mg \cdot L^{-1}$) make living conditions marginal for salmonids in lower Snake River reservoirs (Bennett et al. 1983).

Productive Potential

Reservoirs strongly affect energy dynamics in the mainstem of the Columbia River. Thermal stratification is restricted in river-run reservoirs, and their relatively high flushing rates limit primary productivity. Thermal stratification occurs seasonally in the lower end of Lake Roosevelt behind Grand Coulee Dam, which also has a definite density-flow regime (Jaske and Snyder 1967; Stober et al. 1977). The lower end of Brownlee Reservoir on the Snake River also stratifies thermally during the summer (Raleigh and Ebel 1968), but reservoirs on the lower Snake River rarely develop thermal layering (Bennett et al. 1983).

Development of plankton populations in mainstem impoundments depends, among other things, on water retention in relation to seasonal temperatures. Density and stability of plankton are maximum in reservoirs with long retention times such as Lake Roosevelt and Brownlee Reservoir. Development of indigenous plankton populations in Rufus Woods Reservoir is limited by flushing times of less than four days (Erickson et al. 1977).

Primary Production

Allochthonous detritus is the main contributor of organic material in forested tributaries of the Columbia River. However, a proportionally large population of autochthonous primary producers occurs in the mainstem today. Primary producers in the mainstem of the Columbia River originate largely in reservoirs and are essentially transient, passing from one impoundment to the other at rates related to water retention times. In large part, lentic forms of primary producers pass downstream while periphytic forms are retained. Some periphyton are dislodged by fluctuations of water levels in reservoirs and in the Hanford Reach, and these also pass downstream.

The Upper Arrow, Lower Arrow, and McNaughton reservoirs in Canada are oligotrophic with dissolved oxygen near saturation at all depths. Nutrient levels are low, as is typical of oligotrophic lakes, and diatoms are the dominant phytoplankton. Thermal stratification in these lakes is limited and may not occur in most years (B.C. Research 1977).

Diatoms are predominant in reservoirs in the mainstem of the Columbia River below the international border. Abundance usually peaks in April to June, followed by a second, lesser peak in September to October. Average primary production during the growing season (May to October) in the forebay of Lake Roosevelt is 620 mg $C \cdot m^{-2} \cdot d^{-1}$ (Stober et al. 1977). Carbon uptake values in the flowing Hanford Reach amount to 792 mg $C \cdot m^{-2} \cdot d^{-1}$ during June and September, but drop to near zero during the winter (Neitzel et al. 1982a). Most phytoplankton (and zooplankton) in the Hanford Reach originate above Priest Rapids Dam and are in transit downstream.

Phosphate, nitrate, and silica concentrations in the mainstem show a definite seasonal change, peaking in the winter and falling in the summer. The summer minima are greatly affected by primary productivity. Near Clatskani, Oregon, below the city of Portland, the nitrate–phosphate ratio is 3:1 during the summer and 19:1 at other seasons (Park et al. 1970).

Zooplankton

Zooplankton reach peak abundance in reservoirs in the mainstem of the Columbia River from June to September, but densities are relatively low the rest of the year. The main zooplankton species are the cladocerans *Bosmina longirostris* and *Daphnia* spp., and the copepods *Cyclops bicuspidatus* and *Diaptomus ashlandi*.

Zooplankton densities peak near 50 000 (Earnest et al. 1966) and 60 000 organisms$\cdot m^{-3}$ (Stober at al. 1977) in Lake Roosevelt; 25 195$\cdot m^{-3}$ in Rufus Woods Reservoir (Erickson et al. 1977); 4500$\cdot m^{-3}$ in the Hanford Reach (Neitzel et al. 1982b); and 12 500$\cdot m^{-3}$ in the lower Columbia River (Clark and Snyder 1970).

Secondary Production

Benthic communities in reservoirs in the mainstem of the Columbia River are dominated by populations of chironomids and oligochaetes (Stober et al. 1979; Beckman et al. 1985), but other benthic organisms may be abundant locally. The benthos is usually depleted in littoral zones where water levels fluctuate.

In the flowing Hanford Reach, caddisfly (Trichoptera) larvae (primarily *Hydropsyche cockerelli*), chironomid larvae, an encrusting sponge, annelids, and the crayfish

Pacifasticus leniusculus are common, but species diversity is low. Historically, the unimpounded Columbia River probably supported an average-to-rich bottom fauna in which caddisfly and chironomid larvae, mayfly nymphs, and molluscs predominated (Roebeck et al. 1954). Today, biomass estimates of benthic invertebrates in the Hanford Reach range from 6 to 237 $g \cdot m^{-2}$ during the winter period of maximum abundance (Beak Consultants, Inc. 1980).

General Productivity

The impounded Columbia River probably has greater primary productivity today than it did when still free-flowing. Mainstem reservoirs allow some development of plankton and periphyton populations, and additional nutrients are added from exogenous sources such as irrigation return water. Increased productive potential is paralleled by a general increase in the diversity and abundance of non-salmonid consumers in downstream impoundments (Mullan et al. 1986).

The period of highest primary productivity in mainstem Columbia River impoundments (June to September) may benefit juvenile salmonids that linger in them during outmigration. Some O-age chinook salmon (*0. tshawytscha*) now feed and grow in McNary, Umatilla, and John Day reservoirs (Miller and Sims 1984). The success of the fall chinook salmon population spawning in the Hanford Reach may depend, in part, on this enhancement to their nursery area.

Fish Species

At least 43 species of fish occur in the mid-Columbia River (Gray and Dauble 1977). While anadromous salmon and steelhead runs are the most important, the Columbia River has other valued fishery resources. Commercial species include the anadromous eulachon (*Thaleichthys pacificus*) and American shad (*Alosa sapidissima*), and a resident population of white sturgeon (*Acipenser transmontanus*). Sport catches include native salmonids such as cutthroat trout (*Salmo clarki*), rainbow trout, and mountain whitefish (*Prosopium williamsoni*), as well as introduced species such as largemouth (*Micropterus salmoides*) and smallmouth bass *M. dolomieui*), walleye (*Stizostedion vitreum*), yellow perch (*Perca flavescens*), crappie (*Pomoxis* spp.), and catfish (*Ictalurus* spp.).

Salmonid Resources of the Columbia River

The expanse of the Columbia River system and the anadromous life cycle of the Pacific salmons and steelhead trout tend to mask direct cause-and-effect relationships. In some cases, even heavy harvest in the lower river and ocean did not reduce return runs until spawning and rearing habitats were lost and mortality of smolts passing downstream had increased. Adverse effects occurred in varying degrees over several years, and an observable impact on any particular stock did not appear until successive generations, years later. Further, many stocks spawned in widely separated areas, making discovery of low returns more difficult.

Inordinately broad space and time scales contributed, in part, to policital attitudes and laws promoting development of natural resources in the Columbia River Basin — of which water for irrigation and power was the most vital. The salmon and steelhead runs, and the people who depended on them, received little consideration until recent years. Today, anadromous salmonids, hydroelectric power generation, and resource developments in the Columbia River Basin are interrelated to an extent unequaled anywhere.

The following sections provide insight into fishery management problems. The effect of regional resource development on anadromous salmonids, and concurrent present and potential solutions to these problems are described.

Commercial and Sport Fisheries

Runs of salmon and steelhead to the Columbia River have declined over the years. Landings of chinook salmon, for which the Columbia River is most famous, reflect this trend. They show: (1) an estimated peak catch of 2.3 million fish (19.5×10^6 kg) in 1883, followed by a decline until 1889; (2) catches of around 1.5 million fish (11.3×10^6 kg) annually until 1920; and (3) a decline until 1959, with only about 0.3 million fish landed each year from 1960 to 1980, mostly fall chinook salmon (Fig. 3).

From the 1860's to 1900, commercial fisheries in the lower 322 km of the Columbia River concentrated on and soon depleted runs of high quality chinook salmon from the peak summer return. Catches then shifted to the early "spring" and later "fall" runs (Thompson 1951). Today, returns of chinook salmon to the Columbia River are still separated into distinct spring, summer, and fall runs.

As early as 1878, the Columbia River was closed to commercial fishing during March, April, late August, and early September. Resourceful fishermen soon discovered that salmon could be harvested by trolling in the ocean off the river's mouth. An estimated 500 boats were involved in the new troll fishery in 1915. By 1975, about 3300 troll vessels were licensed in Washington; 2000 in Oregon; 2500 in California; and 1400 in British Columbia. Then, as today, many trollers were licensed in more than one state or province, and made extended trips to other fishing areas. Also, the ability of trollers to catch salmon vastly improved as

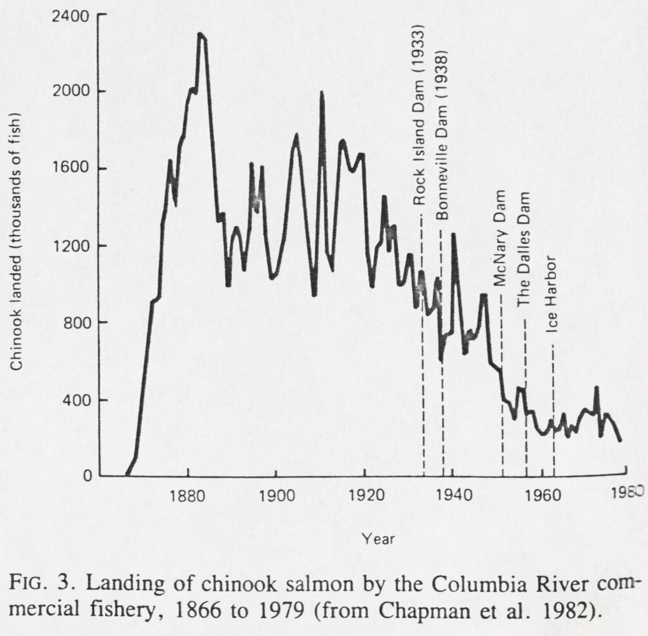

FIG. 3. Landing of chinook salmon by the Columbia River commercial fishery, 1866 to 1979 (from Chapman et al. 1982).

their range expanded and as gear efficiency improved.

Sport fisheries also expanded, albeit more slowly. By the late 1950's and 1960's, sport fishing became a major factor in reducing numbers of adult salmon and steelhead returning to the Columbia River.

Native Indian tribes have traditionally fished return runs in the mainstem Columbia River and various tributaries. Since 1979, certain treaty tribes have been legally entitled to 50% of the allowable harvest. This fishery takes place primarily above Bonneville Dam.

By the early 1930's, the number of adult salmon and steelhead in the annual returns had fallen precipitously (Fig. 3). Concern for the fate of both fish and fishermen led to a patchwork of management organizations and a tangle of state regulations. The early organizations were ill-equipped to manage migratory fish that crossed and recrossed regulatory boundaries. At the same time, new problems arose as hydroelectric dams were built on the mainstem of the Columbia River. However, completion of Bonneville Dam on the lower Columbia River in 1938 gave management a new tool — they could now enumerate adult returns and obtain good estimates of escapement size for each species and run.

In retrospect, three factors contributed to reduced catches of chinook and coho (*O. kisutch*) salmon before 1960: (1) overfishing; (2) decreased production of juveniles, resulting from loss and degradation of spawning/rearing areas; and (3) mortalities of upstream and downstream migrants at dams. Also, hatcheries built prior to 1960 had limited success in compensating for reduced natural production.

Few data exist before the 1960's on the contribution of sport fishing to the total harvest. From this point, the rise in ocean sport catches reflected increased popularity and expansion of the fishery, and more extensive and successful hatchery operations (Chaney and Perry 1976). Sport fishermen caught nearly 500 000 coho salmon of Columbia River origin in 1971 and 200 000 chinook salmon in 1976. Most of these catches were fish of hatchery origin.

Today, salmonids from the Columbia River are caught in the ocean from Monterey, California, to southeastern Alaska. They range widely in the ocean and freely cross political boundaries. Thus, different runs are harvested according to their dispersal patterns. Fall chinook salmon from the lower Columbia River dominate the ocean troll catches from central Oregon to mid-Vancouver Island. Fall chinook salmon from the upper Columbia River migrate farther north and are harvested heavily in waters off British Columbia and Alaska. The chinook salmon from the depleted summer run generally move north (Chapman et al. 1982; Fraidenburg and Lincoln 1985). Spring chinook salmon from the upper Columbia River move north of the river's outlet, whereas those from the Snake River go both north and south (Wahle et al. 1981). Coho salmon from the lower Columbia River are caught mainly off the coasts of California, Oregon, and Washington. In contrast, sockeye salmon (*O. nerka*) and steelhead are not greatly exploited at sea.

Regulating the harvest of mixed stocks of salmonids in the ocean remains a major management problem. Identification of individual stocks or specific fish by stream of origin is difficult. When summer chinook salmon from the Salmon River, Idaho, reach the ocean, they intermingle with runs of summer, spring, and fall chinook salmon from different streams along the Pacific coast. The mixed stocks, each composed of several age groups, are harvested by fishermen from Alaska, British Columbia, Washington, Oregon, and California. Thus, fish from weak runs may be caught along with fish from strong runs--runs still abundant enough to support sport and commercial catches.

Regional Exploitation and Development

As the fisheries expanded, physical changes began to affect adversely salmonid populations of the Columbia River. Exploitation of natural resources proceeded rapidly after 1830, but substantial impacts on salmon runs were not clearly documented until after 1902. In that year, President Theodore Roosevelt's administration passed the Reclamation Act, which eventually led to 28 major reclamation projects.

Economic growth in the Columbia Basin was inevitable. Regional development eliminated or altered fish habitat, caused fish passage and pollution problems, and imposed major constraints on anadromous fish runs.

Dam Construction and Operation

Dams were built primarily for irrigation and power, but they also enhanced navigation, flood control, recreation, and industrial production. The Grand Coulee project on the Columbia River and the Brownlee project on the Snake River had major impacts on salmonid runs. Grand Coulee Dam, operational since 1941, was built without fish ladders and thus prevented access for anadromous fish to over 1100 miles of habitat in the upper Columbia River. Brownlee Dam, operational in 1958, terminated all fish passage to the upper Snake River. Overall, 22 dams were completed on the mainstem of the Snake and Columbia rivers by 1975 (Table 2). They blocked about 50% of the inland headwaters from access by anadromous fish.

Wherever dams were installed, their impoundments inundated the spawning areas used by anadromous fish and significantly delayed the seaward migrations of smolts (Raymond 1979). Eventually, about 783 km (486 mi) of lotic river environment were converted into lentic or semilentic reservoirs.

TABLE 2. Mainstem dams adversely affecting anadromous fish runs on the Columbia and Snake river systems and their initial year of service.

Columbia River	Year of initial service	Snake River	Year of initial service
Rock Island	1933	Swan Falls	1910
Bonneville	1938	Lower Salmon Falls	1910
Grand Coulee	1941	Bliss	1949
McNary	1953	C.J. Strike	1952
Chief Joseph	1955	Brownlee	1958
The Dalles	1957	Oxbow	1961
Priest Rapids	1959	Ice Harbor	1961
Rocky Reach	1961	Hells Canyon	1967
Wanapum	1963	Lower Monumental	1969
Wells	1967	Little Goose	1970
John Day	1968	Lower Granite	1975

Reservoir habitats favor an increase in numbers of resident predator fish such as northern squawfish (*Ptychocheilus oregonensis*), largemouth and smallmouth bass, and walleye, all of which may prey on juvenile salmonids (Raymond 1979). Delays in downstream migration of smolts from the interconnected reservoir system can also extend their residence time and hinder osmoregulation on their entry to seawater (Adams et al. 1975; Zaugg and McLain 1972).

Mortalities of adult upstream and juvenile downstream migrants have caused great concern since the 1940's. Adult passage facilities at downstream dams were often ineffective, and delays were sometimes accompanied by adult mortality (Beiningen and Ebel 1970; Liscom et al. 1977; Johnson et al. 1982). Fishway designs were improved to attract and pass returning adults. Subsequent research showed that juvenile fish suffered high mortalities from passage through turbines at each dam, from predation on stressed fish below dams, and from delayed passage through consecutive reservoirs (Collins 1976; Ebel and Raymond 1976; Ebel et al. 1979; Raymond 1979).

Agriculture and Irrigation

Over 10×10^6 ha are used for agriculture in the Columbia Basin. Impacts on fish and fisheries arise primarily from water withdrawal, soil erosion and sedimentation, and from leaching of animal wastes, fertilizers, and pesticides to streams.

Early irrigation systems were unscreened and they entrained juvenile fish into canals and ditches to die. Today, most diversion intakes are screened on streams used by anadromous fish, but withdrawal of water still lowers flows on some tributaries to critically low levels, reducing or eliminating salmonid production.

The semi-arid ranges of the interior Columbia Basin were overgrazed from the time of early settlement. Uncontrolled grazing contributed to erosion and siltation of tributary streams. Grazing also destroyed riparian vegetation, and caused increases in water temperature and organic pollution. Overgrazing in riparian areas along tributaries remains a problem. Fish production in ungrazed streams is from 2.4 to 5 times higher than in grazed streams (Platts 1981).

Logging

Effects of logging include blockage to and alteration of stream habitat, sedimentation, and degradation of water quality through application of fertilizers, herbicides, and pesticides (NPPC 1986). The result is reduced productivity of tributary streams for salmonids.

Logging probably had its greatest impact from 1880 to 1910, especially in the Willamette River drainage, and in southwestern Washington where over 100 splash dams were built to transport logs downstream. The South Fork of the Salmon River, Idaho, was severely damaged between 1952 and 1965 when spawning gravels became heavily silted.

Mining

Mining activities in the Columbia Basin began early, particularly for gold and silver, and were extensive in the the Salmon, Boise, John Day, Powder, Coeur d'Alene, and Clark Fork rivers (NPPC 1986). Placer mining, often by dredges, displaced stream gravel, added sediment downstream, and eliminated salmonids from many productive areas in Oregon and Idaho. Lode mining degraded water quality by seepage from tailing ponds and mines, especially in Idaho.

Stream Channelization and Clearing

Stream channelization and clearing degraded and destroyed salmon habitat in many streams. Waterways were initially highways for transport of settlers and supplies. The advent and use of the automobile in mountains required that roads be placed along waterways. Boulders and woody debris were often cleared from streams, and this material was used to dike off sloughs and side channels to consolidate the main stream. The cleanup of debris in hundreds of streams during the late 1940's and early 1950's is now viewed as misguided effort.

Industrial Pollution

The discharge of untreated wastes from municipal and industrial sources into Columbia Basin streams accompanied population growth in the 20th century. Effluents from sewage treatment plants, pulp and paper mills, and aluminum plants produced much of the pollution. Runoff from urban nonpoint sources also increased pollution loads (NPPC 1986).

By the 1960's, pollution in the lower Willamette River became so severe that oxygen levels were critically low (0–3 ppm), juvenile migrants attempting to reach the Columbia River suffered losses, and adults attempting to enter the Willamette River were delayed (Fish and Wagner 1950). Plutonium production reactors in the Hanford Reach from 1944 to 1971 used once-through cooling and released radioactivity and heat. However, effluent monitoring and onsite studies showed no effects on fish or other aquatic biota from radioactivity, and minimal to no effects from heated water. More recently, sublethal concentrations of fluoride ($0.3-0.5$ mg•L^{-1}) in effluents from an aluminum plant caused excessive delays of adult upstream migrants at John Day Dam (Damkaer and Dey 1985); fluoride discharges were reduced in 1983, and delays are no longer apparent.

Point-source discharges are not viewed as a serious problem in most of the Columbia River drainage today because the Clean Water Act of 1972 has legislated improved treatment of both industrial and municipal wastes, and point-source discharges are regulated by National Pollution Discharge Elimination System permits.

Smolt Passage Problems

Upriver stocks of salmon and steelhead in the Columbia River have been severely stressed because juveniles must pass eight or nine mainstem dams and reservoirs to reach the sea. Losses at each dam are substantial. From 1968 to 1975, when the four latest mainstem dams were completed on the Snake and Columbia rivers, average survival of juvenile spring chinook salmon passing from the upper

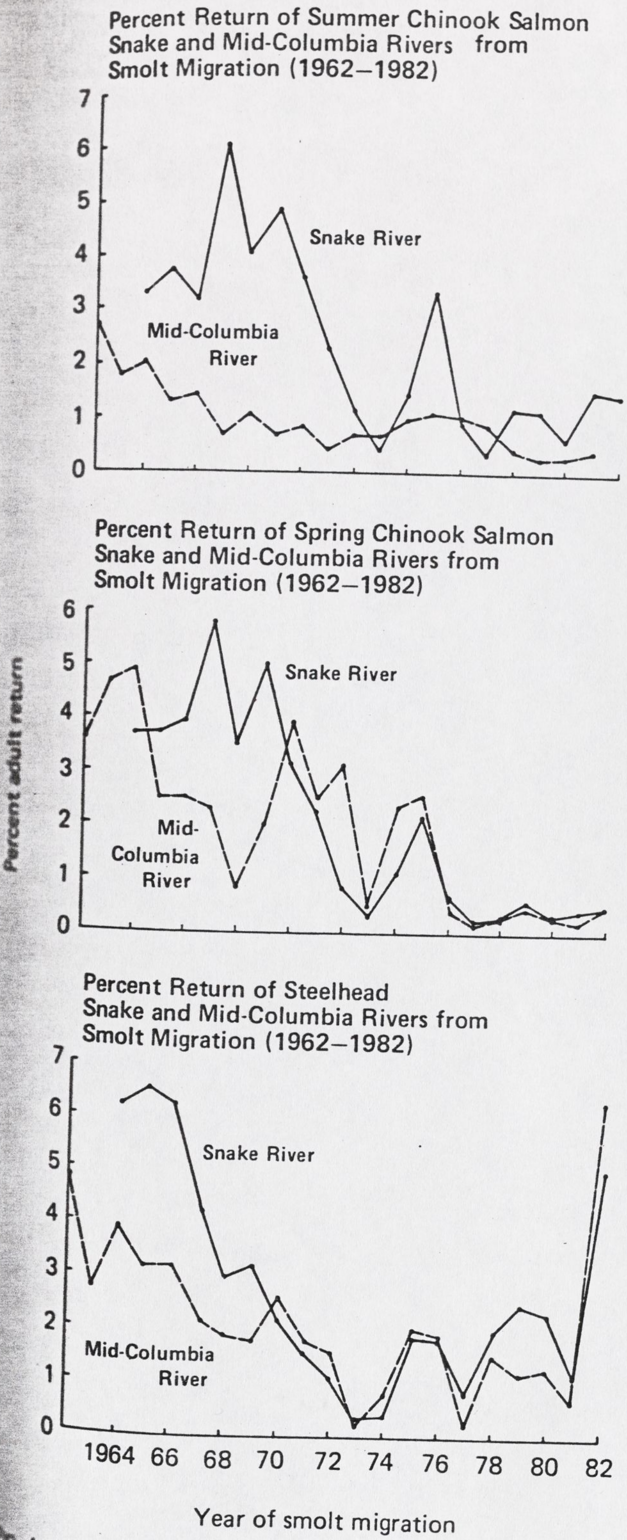

FIG. 4. Percentage return of chinook salmon and steelhead trout to the Snake River and mid-Columbia River dams based on smolt migrations during 1962 to 1982.

30 % (Raymond 1979).

In low flow years, most of the water is passed through turbines at dams, causing high mortalities of smolts from injury and, later, predation. Outmigrant losses range from 15 to 30 % at each dam, and cumulative losses during low water years are particularly severe (Ebel et al. 1979; NPPC 1986). During 1973 when flows were low, about 95 % of all juvenile salmon and steelhead emigrating from the Snake River were lost before they reached the Columbia River's estuary. By 1972, increased storage from the Mica and Arrow Lakes projects in Canada and increased turbine capacity of dams in the United States resulted in little or no spill at mainstem dams in average, as well as in low-flow years.

In high-flow years, when more water passes over spillways at dams, the river may entrain air in the plunge basin, leading to supersaturation and "gas bubble" disease among fish. During high-flow years from 1965 to 1975, mortalities ranged from 40 to 95 % of all outmigrants from the Snake River (Ebel 1971; Ebel and Raymond 1976). Losses of smolts from turbine passage and predation are minimal during high flows because most outmigrants pass with the spill and immediate mortalities are less than 3 % (Schoeneman et al. 1961). Thus, smolt losses at dams are usually lower in high-flow years than in low flow years.

Mortalities during smolt migration are reflected in low returns of spring and summer chinook salmon and steelhead to the Snake and mid-Columbia rivers (Fig. 4). Percentage returns declined between 1962 and 1974, when additional dams were constructed on the lower Snake River.

Actions taken to reduce smolt mortalities include: (1) installation of spillway deflectors (Fig. 5) at key dams, to reduce supersaturation of air in water; (2) installation of fingerling bypasses at dams, to direct smolts away from turbine intakes; (3) development of target flows, to provide more water during fish migrations; (4) implementation of annual spills at dams, to provide safe passage downriver; and (5) collection of smolts at upriver dams, for transport and release downstream below Bolleville Dam. In addition, hatchery production was increased to compensate for losses at dams.

In 1984, 90 % survival of outmigrants was set as a standard for each dam by the Northwest Power Planning Council. This standard can be met at most dams in most years without special spill. Yet cumulative mortality remains high today. Increasing smolt survival at each dam to 94 % in normal or high water years, and to 90 % in low-water years, would be a significant achievement.

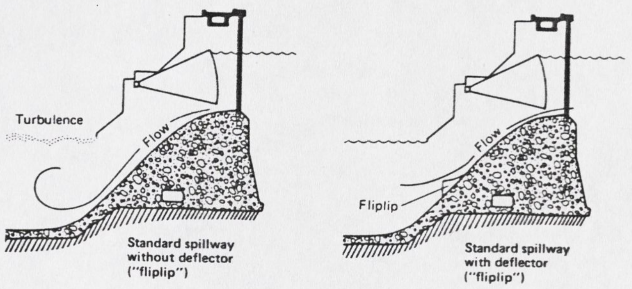

FIG. 5. Sketch of "Fliplip" deflector installed on spillways of Columbia River dams to reduce supersaturation of atmospheric gases.

Artificial Propagation

Compensation for losses of salmonids in the Columbia River system resulted in extensive rearing programs. By the late 1960's, hatchery production of fall chinook and coho salmon and steelhead far surpassed the remaining natural production. However, hatchery compensation is a two-edged sword. While releases supplement adult return runs, they also affect the survival of "wild runs." Large-scale production of hatchery fish on the lower Columbia River has been reported as detrimental to upriver runs of natural fish (NPPC 1986).

An estimated 395 million young salmonids weighing 3.0×10^6 kg (6.6×10^6 lb) were released in 1983 (GAIA Northwest Incorporated 1986). These fish were produced in 54 primary hatcheries or, including substations, 94 rearing facilities. Other hatcheries have begun operation since 1983. Thus, about 400 million anadromous fish are now stocked annually in the Columbia River system. The full potential of these hatcheries is near 1 billion smolts.

With such massive releases, hatchery salmonids now contribute substantially to catches of adult fish in the ocean and lower Columbia River. In 1977, about 75% of the coho salmon caught in ocean sport fisheries off the Oregon coast, and 85% of those caught off the Columbia River, came from hatcheries (Scarnecchia and Wagner 1979).

Young salmonids in hatcheries are not subject to high mortality in fresh water, as are naturally produced fish, because rearing conditions are controlled. Theoretically, hatchery stocks can withstand a greater harvest than natural stocks. While release of large numbers of fish from hatcheries as compensation for lost natural production did slow the decline of many runs, they also encouraged an increase in total harvest. Regulating mixed stocks for maximum allowable catches of hatchery fish caused the less productive natural stocks to erode (Lichatowich and McIntyre 1986). Further, it increased the dependence of many stocks on costly artificial propagation programs (PMFC 1982).

Institutional Arrangements

Until the early 1900's, anadromous fish in the Columbia River were harvested in estuaries and rivers as the adults returned to spawn. Initial effort to halt declines in the various runs was simply to restrict the commercial catch. Increasingly severe restrictions on inside fisheries failed to halt the downward trend. Analysis of the 1938 return runs led to the conclusions that the declines were caused primarily by overharvest (only 17% of the June–July chinook salmon run escaped to spawn), and that catch restrictions were limited by political infighting and, therefore, were largely ineffective (Rich 1941).

In subsequent years, the ocean fisheries expanded and more was learned about the behavior and migration of various species and stocks. Evidence now shows that protecting anadromous fish of the Columbia River requires regulation of catches in both fresh and salt water (Fig. 6). Early regulation of the ocean fishery was confined to seasonal catch restrictions, which were enforced by individual states with jurisdiction extending only 5.6 km (3 nmi) offshore. Federal jurisdiction was extended from 5.6 to 22.2 km (3 to 12 nmi) by passage of the Bartlett Act in 1966, and to 370.6 km (200 nmi) by the Fishery Conservation and Management Act in 1974. Further, early regulations were not always uniform among states and, generally, were not designed to achieve specific stock-by-stock escapement goals. Concepts of scientific fishery management did not begin to emerge until the 1940s.

In 1938, Congress passed the Mitchell Act authorizing

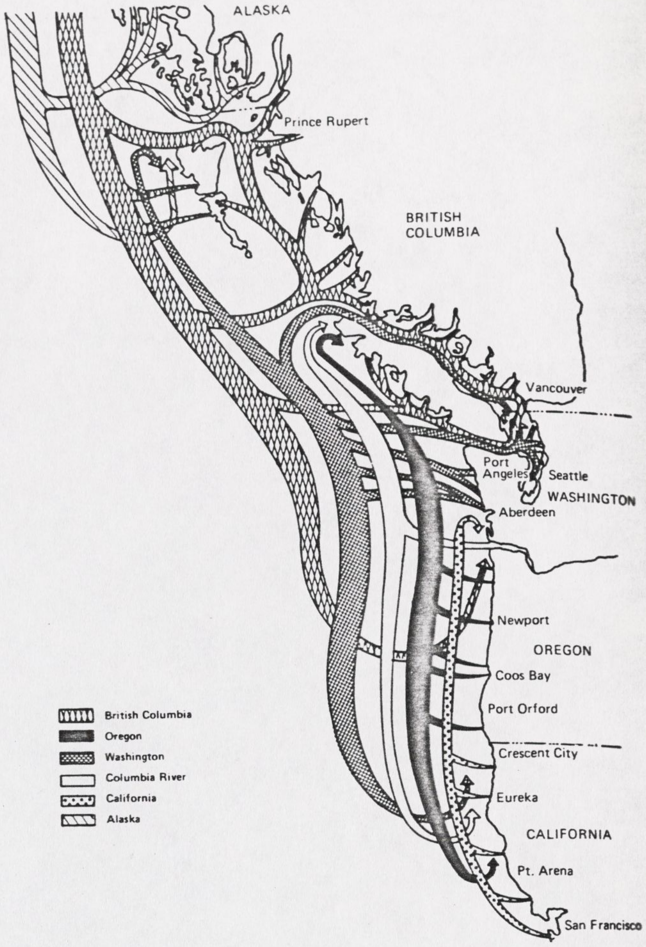

FIG. 6. General migration patterns of chinook salmon in northeastern Pacific Ocean. Migration patterns of coho salmon are similar.

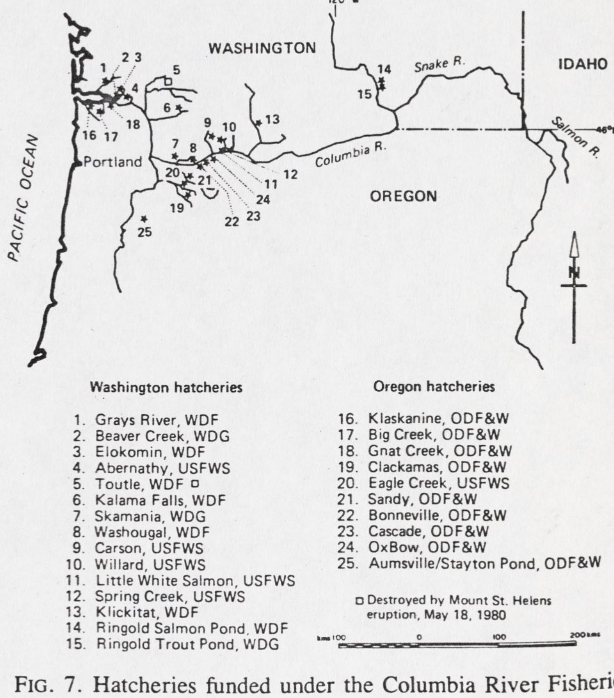

FIG. 7. Hatcheries funded under the Columbia River Fisheries Development program.

artificial propagation to compensate for destruction of habitat by hydroelectric development and other environmental changes. The Act was amended in 1946 to support state participation in stream improvement (fish passage over natural barriers, screening of irrigation diversions, and opening of streams blocked by debris), and in research to improve the quality of hatchery fish. Formalized as the Columbia River Fisheries Development Program, about 80 % of the available funds are now used to support 21 hatcheries and three rearing ponds (Fig. 7), with an annual production of about 100 million juvenile salmonids (Table 3).

Conflicts with various Indian tribes over regulation of tribal fisheries by states peaked in the early 1970's. These conflicts resulted in a 1979 ruling by the Supreme Court (the Boldt decision) that an 1855 treaty guaranteed 20 tribes the right to catch salmon and steelhead in their usual and accustomed places, and that the tribes were entitled to 50 % of the harvestable fish. Today, the treaty tribes participate in decisions affecting management of the resource through the Northwest Indian Fisheries Commission, the Columbia River Intertribal Fish Commission, and other liaison groups.

The federal government assumed management authority in 1974 with passage of the Fishery Conservation and Management Act. This Act established exclusive U.S. fishery authority over all salmon (and most other species) in the ocean within a 5.6–370.6 km (3–200 nmi) fishery conservation zone, except in territorial waters of other countries. Two regional councils for management were established under the Act: the Pacific Fisheries Management Council (PFMC) has jurisdiction off the coasts of California, Oregon, and Washington; the North Pacific Fishery Management Council (NPFMC) has jurisdiction off the coast of Alaska (Fig. 8).

The U.S./Canada salmon treaty, ratified in 1985, represented another milestone. It provided for regulating ocean fisheries to ensure meeting Indian treaty obligations, equitably distributing the catch between ocean and freshwater fisheries, and achieving spawning escapement goals (PMFC 1982).

With passage of the Water Resources Development Act of 1976, the Lower Snake River Fish and Wildlife Compensation Plan (SRCP) was authorized to compensate for losses of fish caused by hydroelectric projects on the Snake River. The SRCP called for hatchery releases sufficient to produce returns to the Snake River of 18 300 adult fall chinook salmon, 58 700 adult spring and summer chinook salmon, and 55 100 adult steelhead. Although such returns have not resulted to date, significant supplementation is occurring. The SRCP also called for release of 93 000 pounds of trout annually to compensate for loss of sport fish production in Washington and Idaho streams.

The Pacific Northwest Electric Power Planning and Conservation Act (NPCA) of 1980 and its accompanying Fish

TABLE 3. Numbers of salmonids released from hatcheries funded under the Columbia River Development Program from 1960 to 1984 (from Delarm and Wold 1985). All data × 10^6.

Year	Fall chinook salmon	Spring chinook salmon	Coho salmon	Steelhead trout	Totals
1960	89.1	1.8	6.4	1.0	98.3
1961	46.6	0.8	14.2	0.9	62.5
1962	55.8	1.7	12.9	1.6	72.0
1963	58.8	2.4	19.6	1.4	82.2
1964	65.5	7.6	16.5	1.7	91.3
1965	56.2	3.0	17.9	1.9	79.0
1966	54.9	3.8	19.7	2.5	80.9
1967	55.1	5.5	20.2	2.3	83.1
1968	55.5	3.8	15.7	3.0	78.0
1969	57.9	3.5	18.6	2.3	82.3
1970	62.2	2.6	17.4	2.9	85.1
1971	63.3	3.8	21.3	2.4	90.8
1972	67.1	3.6	23.9	2.5	97.1
1973	70.4	4.8	20.9	2.5	98.6
1974	65.5	4.4	20.2	2.3	92.4
1975	67.3	5.2	21.1	1.9	95.5
1976	84.0	5.9	22.2	2.1	114.2
1977	95.0	5.1	26.3	2.2	128.6
1978	89.3	5.5[a]	26.3	2.4	123.5
1979	89.1	7.5[a]	21.1	2.4	120.1
1980	80.1	7.2[a]	20.8	2.2	110.3
1981	73.3	7.6	19.2	2.3	102.4
1982	78.6	7.3	17.4	2.1	105.4
1983	74.5	6.9	21.7	2.1	105.2
1984	72.4	8.7	22.3	3.3	106.7

[a]Includes a small number of summer chinook salmon.

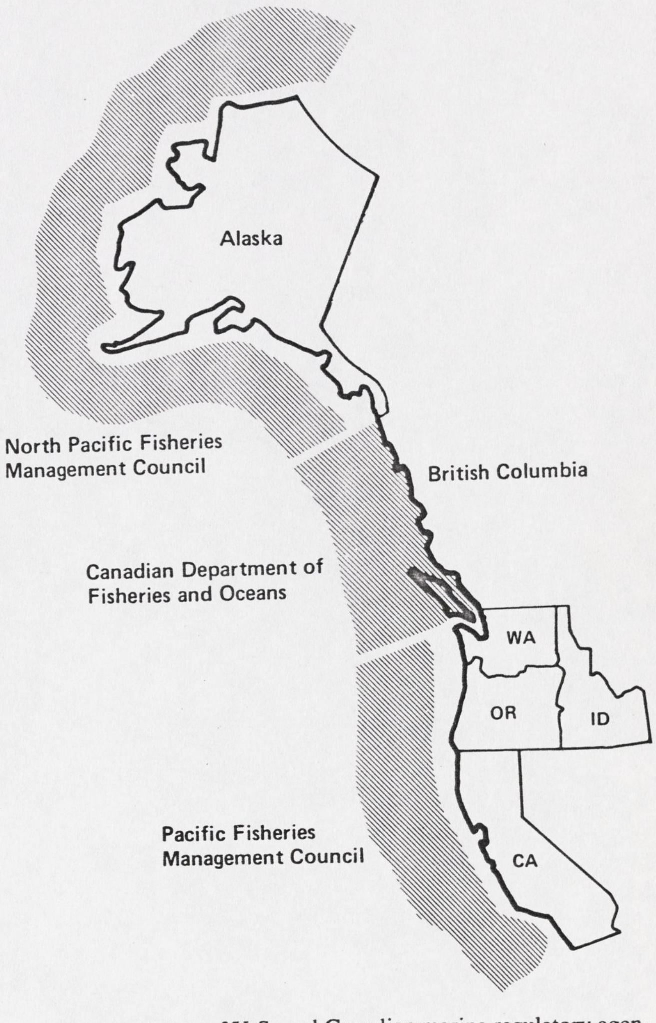

FIG. 8. Jurisdiction of U.S. and Canadian marine regulatory agencies.

and Wildlife Program (currently funded at about \$30 million annually) represented another milestone, particularly in improving degraded habitat in the Columbia River. The Program includes a water budget that sets aside $5724 \times 10^6 \cdot m^3$ of water each year to increase spills at dams from April 15 to June 15. Additionally, spill plans are drafted and implemented by fishery and tribal personnel working with public power utilities, the Corps of Engineers, and the Bonneville Power Administration. The intent is to reduce mortality of outmigrating smolts in turbines.

Public utilities and the Corps of Engineers have made substantial progress in compensating for salmon and steelhead losses at their dams on the Columbia River. Besides funding smolt production at hatcheries, they have spent millions of dollars annually to improve upstream and downstream passage of fish, and to provide adequate flows for fall chinook salmon spawning in the Hanford Reach below Priest Rapids Dam.

The results of these efforts are encouraging. Counts at Bonneville Dam since 1960 are listed in Table 4. About 440 000 salmonids were caught commercially in 1984. This was the highest catch since 1976 and may be attributed to a substantial increase in the size of runs.

Recent escapements are also encouraging. Several runs show an upward trend since 1980. Record returns of steelhead and fall chinook salmon in 1984 and 1985 reflect increased production of fall chinook salmon in the mid-Columbia River and benefits from improved passage of smolts. Upriver runs of summer chinook and sockeye salmon have also increased — this is especially evident from counts recorded in 1985.

Needs and Opportunities

The reasons that early efforts to compensate for declining salmon runs in the Columbia River were ineffective or failed include the unprecedented rate and magnitude of hydroelectric development on the Columbia River, and the limited knowledge and technology applicable to anadromous fish passage at the time. Many hatchery operations (e.g., hatcheries built to mitigate habitat loss from the Grand Coulee project) were conceived with inadequate background data as provisional experimental programs, subject to ongoing evaluation and updating, a proviso rarely done.

There was also the matter of priorities. Once dams were finished and operating, enthusiasm was often lost by construction agencies to see, and Congress to provide, compensation that was implied or promised when political support was sought for project appropriations. In addition, belated compensation programs were subjected to far more rigorous economic scrutiny than the projects that created the need (Chaney 1978). Nearly 15 years passed between completion by the Corps of Engineers of the first dam on the lower Snake River and the first compensation hatchery.

Numerous obstacles must still be faced to effectively manage anadromous salmonids of the Columbia River. New problems are likely to appear as greater insight is gained.

TABLE 4. Numbers of salmonids counted over Bonneville Dam on the lower Columbia River from 1960 to 1985.[a]

Year	Spring chinook salmon	Summer chinook salmon	Fall chinook salmon	Steelhead trout	Sockeye salmon	Coho salmon
1960	69 595	85 170	101 282	113 676	59 713	3 268
1961	98 695	66 461	119 916	139 719	17 111	3 456
1962	89 635	77 310	118 039	164 025	28 179	14 788
1963	75 473	64 013	139 079	129 418	60 319	12 658
1964	91 425	80 531	172 463	117 252	99 856	53 602
1965	84 261	75 974	157 685	166 453	55 125	76 032
1966	112 669	71 997	155 445	143 661	156 661	71 891
1967	84 935	95 659	185 643	121 872	144 158	96 488
1968	99 187	82 919	159 247	106 974	108 207	63 488
1969	173 566	102 153	231 838	140 782	59 636	49 378
1970	110 976	65 510	208 902	113 510	70 762	80 166
1971	125 517	77 911	202 274	193 966	87 447	75 989
1972	186 140	70 830	137 486	185 886	56 323	65 932
1973	142 148	45 360	211 127	157 823	58 979	54 609
1974	134 535	45 896	186 328	137 054	43 837	60 955
1975	104 104	44 351	277 111	85 540	58 212	58 307
1976	113 446	69 013	325 312	124 177	43 611	53 150
1977	119 508	41 023	206 126	193 437	99 829	19 408
1978	149 863	44 323	200 404	104 431	18 436	52 590
1979	51 462	34 217	190 613	113 979	52 628	45 328
1980	60 987	31 065	153 466	129 254	58 882	22 052
1981	65 009	26 929	193 712	159 270	56 037	30 510
1982	76 044	26 614	220 151	157 640	50 219	73 832
1983	56 838	23 458	164 180	213 779	100 527	15 176
1984	51 142	28 448	243 756	315 795	152 540	29 332
1985	90 964	29 353	334 436	326 194	165 928	55 529

[a] Dam counts indicate only escapement to the river, not total run size. Total run includes commercial and sport catches, and escapements to tributary streams below Bonneville Dam. Counts at dams today are further reduced by the Tribal fishery and a limited sport fishery above Bonneville Dam.

First, reducing smolt mortalities at dams remains an overwhelming need. Substantial progress has been made on developing and installing turbine screening and bypass systems at Columbia River dams. Yet technology still cannot reduce losses at each dam to less than 5% and few dams have yet attained such efficiency. Collection of seaward migrants in the Snake River at Lower Granite and Little Goose dams and at McNary Dam on the Columbia River, and barging or trucking them downstream, has produced favorable results. Steelhead and coho and fall chinook salmon have definitely benefited. The effectiveness of collection and transportation systems apparently varies with species, just as different abiotic factors influence losses at each dam.

Second, judicious allocation of spill over dams during seaward migration periods is required where effective turbine screen and bypass systems have not been installed. Not enough water is available in the mainstem of the Columbia and lower Snake rivers during years of low and average runoff for both hydroelectric power and downstream passage. The Water Budget of the NPPA Fish and Wildlife Program offers some improvement. Use of spill to reduce turbine mortalities may be an interim measure. Water for flushing smolts downstream and over dams will always be contested by other water users, particularly by power producers. The conflict emphasizes the need for adequate turbine screen and bypass systems.

Third, the problem of integrating hatchery releases and wild production without deleterious effects on wild stocks must be addressed. This includes density-dependent survival in fresh and salt water associated with large releases of smolts from hatcheries, and reduced survival of wild salmonids in tributaries laced with hatchery outplants to supplement natural spawning (Lichatowich and McIntyre 1986).

Fourth, disease continues to plague some hatchery stocks. Returns of upriver spring chinook salmon to federal hatcheries remain low. The presence of bacterial kidney disease (BKD) in hatchery stocks of spring chinook salmon that now contribute 70–80% of the seaward migration may cause poor survival (Banner et al. 1982; Fryer 1984). Juvenile spring chinook salmon are overwintered in hatcheries and released in early spring. Major BKD outbreaks occur near release time and may be induced by approaching smoltification. Could losses be reduced by new vaccines or by altering hatchery practices? Perhaps releases to impoundments in the fall for overwinter conditioning would increase survival.

Fifth, relationships between resident and migratory species need to be examined. Creation of impoundments and introduction of exotic fish has altered the composition and abundance of fish populations. Predation on salmonid outmigrants may be extensive, and other cause-and-effect relationships such as competition and disease transmission may exist.

Sixth, while pollution is not a major problem, information on current levels of contaminants in water, sediments, and fish tissue is incomplete. Salmonids in some areas now carry low burdens of polychlorinated biphenyls and chlorinated hydrocarbons, presumably derived from past industrial releases and return of contaminated irrigation water.

Seventh, mixed stock fisheries must be effectively regulated to prevent overharvest of wild stocks. This requires adequate data on the ocean distribution of stocks in the offshore fishery, the relative size of stocks available for harvest, and the response of stocks to oceanic conditions. Some data are available for hatchery fish from marking programs, but information on wild fish is meager. New techniques in stock identification (scale and parasite analysis, coded wire tags, genetic variant data) will help.

Eighth, efforts to increase natural production throughout the Columbia River system may be countered by construction of small hydropower dams on tributaries. The cumulative impact of hundreds of proposed projects on instream smolt production could be severe, depending on which projects are authorized.

According to one's philosophy, the resource base of salmon and steelhead in the Columbia River is either half empty (and declining) or half full (and increasing). The current program is not without critique (Anonymous 1986). However, the apparent success of recent management actions, which involve the cooperation of many agencies and governments, is encouraging. If current effort is continued, annual production of anadromous fish in the Columbia River system may increase to a level consistent with ecosystem changes and competing societal uses.

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[Ca. 1998]

MODIFICATION OF AN INDEX OF BIOTIC INTEGRITY

BASED ON FISH COMMUNITIES FOR STREAMS

OF THE NORTHEASTERN UNITED STATES

Doc,
Thanks for your note. The symposium went quite well. I thought you might be interested in this MS I hope to submit soon to N.A.J. of Fisheries Management.
Take care
Dave.

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ABSTRACT

The index of biotic integrity (IBI) developed for Midwest streams was applied to fish collection data from streams in the Mohawk, Merrimack, and Connecticut River drainages of the northeastern U.S. Because several fish taxa are poorly represented in this region, some of the original 12 IBI metrics needed to be either replaced, their criteria scores modified, or both. The number of darter species and number of sunfish species metrics were replaced with the integrative metrics, number of benthic insectivorous species and number of water column species, respectively. The proportion of individuals as white sucker (Catostomus commersoni) replaced the green sunfish (Lepomis cyanellus) metric in all drainages because green sunfish are not native to the Northeast. The proportion of individuals as insectivorous cyprinids metric was replaced with the proportion of individuals as insectivores in the Merrimack and Connecticut drainages. All replacement metrics followed the rationale for using each of the original metrics. A more objective way of evaluating number of fish per sample is presented that uses the empirical relationship between fish density and watershed area. Irrespective of the modifications, the basic theoretical underpinnings of the IBI remain unchanged. Our intent is to show the flexibility in the conceptual approach and content of the IBI, and provide a working model for use by water resource managers in the Northeast.

INTRODUCTION

Karr (1981) presented the index of biotic integrity (IBI) as a bioassessment procedure that integrates several attributes of stream fish assemblages. The IBI is a rapid and relatively inexpensive way to assess the general health of streams. It provides a useful measure for assessing or evaluating the environmental results of water quality improvements (Karr et al. 1985). The development of the IBI was a response, in part, to the prevailing use of water quality as a surrogate for biotic assessment, to the call for restoration and maintenance of biotic integrity in the Clean Water Act of 1977 (PL 95-217), and the need for a suitable broad-based methodology for evaluating biological conditions. Incorporation of biological monitoring and development of suitable biological indices should improve management of water resources (Karr et al. 1986) by providing a direct measure of water resource quality that can be easily quantified and understood.

Comprised of 12 measures or metrics of a stream fish community, the IBI encompasses the broad ecological categories of species richness and composition, trophic composition, and fish abundance and condition (Table 1). Initially developed for warmwater streams in Illinois, Fausch et al. (1984) extended the application of the IBI to watersheds throughout the Midwest. A recent peer-review workshop on the IBI (James Karr, Smithsonian Tropical Research Institute, pers. comm.) judged the approach not only sound with a firm foundation in ecological theory, but simple, yet providing a quantified basis for decision-making. Further, the IBI appears sufficiently robust to apply in areas outside the Midwest. The workshop participants recommended that further research on the IBI concentrate on regional applications and, if necessary, modifications of the IBI. Their recommendation stems from the structural and functional variability of stream fish communities across the U.S., and the concomitant need to assess metrics that are based solely on the structure and function of midwestern stream fish communities. Essentially, the IBI as proposed by Karr (1981) should serve as a template for development in other regions.

The IBI concept is being applied to coastal streams in California (Peter Moyle, U. of California-Davis, pers. comm.), the Willamette River, Oregon (Hughes and Gammon in press), West Virginia streams (Leonard and Orth in press, Angermeier and Karr in press), Colorado eastern plains streams (Kurt Fausch, Colorado State U., pers. comm.), Tennessee streams (Chris O'Bara, Tennessee Tech. U., pers. comm.), and estuaries (Thompson and Fitzhugh 1986). The estuarine application underscores the flexibility of this approach.

Our objectives are to (1) apply the IBI to stream fish communities of Atlantic slope drainages of the Northeast that comprise cold-, cool-, and warm-water fishes, (2) demonstrate the flexibility of the IBI by modifying metrics deemed inappropriate for use with the fish fauna of the Northeast, and (3) provide alternative versions of the IBI to help Northeast water resource managers to assess biological conditions and manage for environmental results. Our intent is not to supplant earlier methodologies, but rather to promote the conceptual content of the approach and encourage its use in water resource management.

METHODS

Establishment of scoring criteria for the metrics in the Midwest IBI (Table 1) requires stream fish assemblage data from the least degraded sites in a watershed. This provides a standard against which other sites in the basin can be scored (Fausch et al. 1984). For the Northeast, historical, quantitative stream fish community data are generally lacking; when available, the determination of least degraded sites is complicated by the long history of human settlement (Marston and Gordon 1938, Marston 1939).

The comprehensive surveys of the Mohawk River drainage of New York (864 collections, Greeley 1935; field catalogs of Mohawk River survey, housed at Olin Research Library, Cornell University), the Merrimack River (160, Bailey 1938) and Connecticut River (295, Bailey and Oliver 1939) drainages of New Hampshire represent the best available historical data on stream fish communities in this region (Fig. 1). However, they provide only species presence/absence data that can be used to evaluate the original species richness and composition metrics and their scoring criteria (Metrics 1-5, Table 1). Seining was the primary collection method in these studies, without reference to length of stream sampled or sample effort except that all available habitats were apparently sampled. At the time of these early surveys, environmental change had already affected the fish fauna through influences of channelization, pollution, dams, and species introductions. For example, Bailey (1938) found that of the 48 species comprising the known recent fauna of the Merrimack drainage, 14 were introduced, doubling the number of species of top carnivores.

The other Midwest IBI metrics (Nos. 6-12, Table 1) require quantitative (species abundance) data. These metrics and their scoring criteria were evaluated using more recently collected data. One of us (RAD) collected fish by seining or electroshocking from 85 sites in the Schoharie Creek drainage of the Mohawk River, 1981-1983 (Fig. 1). Madore (1974) collected fish from 42 sites in the Nashua River drainage of the Merrimack River, and Bergin (1972) collected fish from 77 sites in the Chicopee River drainage of the Connecticut River in Massachusetts (Fig. 1). The Nashua and Chicopee River sites were sampled with rotenone or by electroshocking.

For all collections, sites were rejected if too close ($\leq$ 1.6 km) to lakes or confluences of larger streams, canals, dams or other obstructions, or if data were obviously incomplete (Fausch et al. 1984). Sites which included bridges were rejected, but sites existing totally upstream or downstream of bridges were used. Only those sites having comparable sampling methods and effort were used to evaluate metrics and scoring criteria. Upland cold-water sites comprised of three or fewer species and including a trout species were rejected. Lowland cold-water sites usually have greater than three species and were included. We feel that upland cold-water sites do not fit into the IBI concept and are better evaluated by estimates of trout biomass/production or macroinvertebrate sampling. Because gradient changes may ultimately affect the scoring of species richness metrics (Leonard and Orth in press), we examined elevation as a surrogate measure for gradient, but observed no obvious species richness differences among similar-sized stream sites with different elevations.

Based on this data quality screening, the following sites/drainage were used: Mohawk (79), Merrimack (62), Connecticut (78), Schoharie (61), Nashua (10), and Chicopee (44).

The IBI requires that species be classified as to tolerance and trophic category. Our classification (Appendix) is a composite based on the following regional texts: Carlander (1969), Scott and Crossman (1973), Pflieger (1975), Smith (1979), Lee et al. (1980), Trautman (1981), Becker (1983), and Smith (1986). In particular, species were classified as intolerant if they appeared susceptible to general degradation of any major type (e.g. siltation, organic and inorganic pollution, channelization, and irrigation that degrade water quality, habitat, and food resources). The number of intolerant species should be limited to the 5-10% (10-15% in depauperate regions) that are most susceptible (Karr et al. 1986).

Scoring criteria for the species richness and composition metrics (Nos. 1-5, Table 1) were determined, where warranted, using the maximum species richness line (MSRL) concept of Fausch et al. (1984). The MSRL is based on empirical data that suggest an increase in species richness with increasing stream size. Unlike Fausch et al., we excluded nonnative species from the analysis of species richness and composition metrics because they may be considered as a biological disturbance. Accounts in Greeley (1935), Bailey (1938), Bailey and Oliver (1939), and Halliwell (1984) helped to determine nonnative species (Appendix). We used watershed area as a measure of stream size because of demonstrated problems with the use of stream order (Hughes and Omernik 1983). Topographic divides were drawn to encompass the watershed area drained by the stream at each collection site. A compensating polar planimeter was used to measure watershed area with an accuracy of 0.5 km^2. All maps were 1:250,000 in scale except the 1:25,000 scale maps for the Nashua and Chicopee drainage sites.

Because of the nature of stream fish communities in this region, several original metrics and scoring criteria were found inappropriate, and replacement metrics, new criteria, or both were developed. The nature of some replacement metrics required that species be classified for membership to these new metrics (Appendix). Two of the original metrics require data on presence of hybrids and anomalies (Table 1). Hybrid data were available from Schoharie Creek, but not for the Nashua and Chicopee data. Anomalies data were lacking in all studies. Where data were totally lacking to score a metric, a score of 3 (intermediate between a high of 5 and a low of 1) was given to each site for that metric. Karr et al. (1986) suggest using a score of 5 for such situations. However, we prefer to use a less conservative approach that does not inflate the scores of degraded sites by 4. Final IBI scores (summation of all 12 metrics with top score of 60) were calculated for sites in the Schoharie, Nashua and Chicopee drainages.

Results and Discussion

Table 1 compares our versions of the IBI for the Mohawk and Merrimack and Connecticut River drainages with the original IBI developed in the Midwest (Karr 1981; Fausch et al. 1984). A single IBI can be used for both the Merrimack and Connecticut drainages because these two adjacent watersheds support fish faunas with similar species composition and species number (44 vs 48, respectively; Bailey and Oliver 1939). The extent to which this "New England" IBI (four replacement metrics) differs from the IBI for the Midwest is a reflection of the Northeast's depauperate fish fauna. The Mohawk ~~drained by the stream at each collection site. A compensating polar~~ drainage has approximately double (81 species; Greeley 1935) the species complement of the Merrimack and Connecticut drainages. However, the IBI developed for the Mohawk drainage (three replacement metrics) is more similar to the New England IBI than the Midwest IBI.

These watersheds are typically depauperate of native sunfish and darter species (Appendix), so the use of MSRLs for these taxa was inappropriate. We felt that it was not possible to establish separate scoring criteria for these families; they contained too few species, and those species showed localized distributional patterns within watersheds. Therefore, replacement metrics of water column species and benthic insectivorous species were developed based on the rationale for using sunfish and darter species, respectively (Table 1; Karr et al. 1986). Sunfish are particularly responsive to declines in several aspects of habitat structure including loss of pool habitats. No other taxon in the Northeast is rich enough to suffice for this purpose, hence a metric that integrates across taxa was developed. All species classified as water column (defined as all non-benthic species; Appendix) exist in pool habitats and appear responsive to changes in habitat structure. The specificity of darters to reproducting and feeding in benthic habitats results in their sensitivity to degradation of those habitats (Karr et al. 1986). An integrative approach again was developed, incorporating benthic insectivorous species (Appendix) as a replacement metric for darters.

The green sunfish metric in the Midwest IBI evaluates the degree to which a single, very tolerant species dominates the community (Karr et al. 1986). However, the green sunfish is not native to the Northeast (Lee et al. 1980). In the Mohawk, Merrimack, and Connecticut drainages the white sucker is tolerant to a wide range of impacts, and becomes dominant in degraded areas. It was deemed an appropriate substitute for the green sunfish.

In the Merrimack and Connecticut drainages, cyprinids are a relatively depauperate group (Appendix), so we sought a replacement metric for insectivorous cyprinids. Karr et al. (1986) suggest that in such cases proportion of insectivores be used. They further suggest the use of criteria that almost doubles that used for insectivorous cyprinids. However, our criteria (Table 1) are a compromise between criteria used for insectivorous cyprinids and those suggested for insectivores. We found that the number of insectivorous species (19) in the Merrimack and Connecticut is only slightly larger than the number of insectivorous cyprinids (16) in the Mohawk (Appendix). Thus insectivores may not dominate the fish community in the Northeast to the extent that Karr et al. imply. Considering the general inverse relationship between omnivores and insectivores (Karr et al. 1986), comparisons of trophic structure across sites within watersheds also suggest less stringent criteria.

The only difference in the IBIs for the Mohawk and Merrimack and Connecticut drainages is the use of the insectivore metric (Table 1). Our rationale for maintaining this difference was based on the desire to maintain the integrity of the original IBI. In retrospect, we suggest the use of insectivores for both drainages. This allows the use of a single IBI to facilitate within-region comparisons of water resource quality. Drainage differences in species richness and trophic composition are then manifested in the different MSRLs and criteria levels for each drainage.

The proportion of individuals as omnivores metric for the Merrimack/Connecticut and Mohawk IBIs includes the white sucker even though this dominant omnivore is included in its own metric (Table 1). Initially, we did not include the white sucker because of a concern for confounding interactions similar to that described by Leonard and Orth (in press). They found inconsistent responses between the creek chub (*Semotilus atromaculatus*; their green sunfish metric) and insectivorous cyprinids. The creek chub, a very tolerant species, is classified as an insectivorous cyprinid. Inconsistencies resulted because those metrics have opposite expected responses to increasing degradation. However, the white sucker and omnivore metrics show similar responses to degradation. Inclusion of the white sucker does not appear to affect the efficacy of these metrics.

The MSRLs were developed for total number of native fish species (Fig. 2a, b), native water column species (Fig. 3a, b), and native benthic insectivorous (Fig. 4a, b) for both the Mohawk and Merrimack/ Connecticut drainages, and for total number of native intolerant species for the Mohawk (Fig. 4c). Initially, we plotted MSRLs separately for the Merrimack and Connecticut drainages. However, the MSRL plots showed approximately the same slope and intercept so they were combined. Attempts at establishing MSRLs for the other species richness and composition metrics provided no obvious species richness versus watershed area relationships. In these cases separate scoring criteria were established (Table 1) reflecting species presence/absence patterns across all sites within a watershed.

Occasionally, sites may either fall on or very near the lines that separate criteria score regions in MSRL-type graphs. Scoring a particular species richness and composition metric for these sites is problematic. Our solution is to assign positive or negative scores. For example, the boxes and triangles in Figures 2-5 represent sites given a score of 5(-) and 3(+), respectively. Pluses and minuses are summed during final IBI calculation, and IBI scores with two or more pluses are given a final IBI score upgraded by two. Minuses are handled in a similar manner. This method of scoring compensates for variability associated with drafting the graph, i.e., how the line was drawn, the thickness of the line, and the accuracy of point placement. We used this method for all metrics whose scoring criteria are determined by the MSRL and Maximum Density Line (MDL) (see below) procedures. Hughes and Gammon (in press) used the same scoring method for marginal proportional values of the other metrics, as well as the species richness metrics. If this scoring method is applied to all metric criteria scores, then final IBI scores should be adjusted where three or more pluses or minuses are present.

Karr (1981) and Fausch et al. (1984) did not give guidelines for establishing criteria for number of individuals per sample other than stating that relative criteria were used after conversion to catch/effort. For the Northeast, plots of number of individuals/100 m vs watershed area showed no obvious relationships. To provide a more objective method for establishing criteria, we replaced this metric with a density measure (number of individuals/100 m^2). Fish density has been shown to decrease as watershed area (stream size) increases (Thompson and Hunt 1930; Hallam 1958; Larimore and Smith 1963). This decrease may be wholly or in part a result of decreasing fish capture efficiencies in larger, deeper water streams. Because the relationship of fish density to watershed area differs between regions and watersheds (Hynes 1970), it should be evaluated for each watershed. Figure 5 depicts this inverse relationship for the Schoharie and Nashua/Chicopee drainages.

We call the line forming the upper bound for about 95% of the sites in Figure 5 the Maximum Density Line (MDL). The MDL is drawn with slope fit by eye (Fausch et al. 1984), and its intersection with the abscissa, provides the focal point to trisect the ordinate into criteria score regions similar to the MSRL. Originally, we calculated the upper 90% prediction band about each regression line (for Figure 5a $y = 1.95 - 0.42x$; $r = -0.57$, $p < 0.01$) as the MDL. Prediction bands about the sample are more appropriate than confidence bands about the mean (because of annual variability in fish population size), and 90%, in this case, approximated the upper limit of the data. However, we favor the simpler technique for determining the MDL because it corresponds to the way MSRLs are determined and differs little from the upper 90% prediction band.

Determination of the MDL requires that sample reach area be recorded in addition to species abundances. We believe that the extra time required in collecting this measurement is compensated for by increased objectivity in scoring this metric. However, this technique should be regarded as preliminary until similar relationships are established for watersheds in other regions.

Figure 6 compares the distribution of final IBI scores for Schoharie and Nashua/Chicopee drainage sites based on an interpretive scale (Karr 1981). These drainages have similar mean IBI values (representing the fair category), standard deviations, and ranges in IBI scores. However, the distribution of IBI scores for the Nashua/ Chicopee sites is more skewed to higher scores. For example, 78% of Nashua/Chicopee IBI scores are rated fair or better compared to 63% for Schoharie sites, although a greater percentage of Nashua/Chicopee sites are rated poor or worse (19% vs 16% for Schoharie). This skewness is, to a large extent, a result of the sites chosen for analysis. Mean watershed size for Schoharie sites is 273.3 km^2 (range: 2-2102 km^2) compared to a mean of 72.3 km^2 (range: 3-525 km^2) for Nashua/Chicopee sites. In the Nashua/Chicopee, these smaller (upstream) sites were less impacted than larger sites, hence the higher final IBI scores. Fausch et al. (1984) noted a similar trend for watersheds from the Chicago, IL area. Also, all sites were included in the analysis regardless of sampling method employed, because use of a particular method was apparently dependent upon stream size. Further data screening to insure a standard sampling method across sites would further skew the data in both drainages. Using sites without regard to sampling method assured a wider range of sites within drainages, and facilitated a general comparison between drainages. However, these results should not be considered as representative of the relative degradation of these watersheds. More recent data, collected in a standard fashion across sites of several sizes, should be utilized for those purposes.

Each of the 12 metrics in the Midwest IBI represents a different range in primary sensitivity to changes in biotic integrity (Angermeier and Karr in press). Angermeier and Karr (in press) show that the amount of information conveyed by a particular metric varied among data sets of different spatial scale from Illinois, Ohio, and West Virginia. Therefore, inclusion of all 12 metrics should increase the precision of an assessment of biotic integrity. This is a major strength of the IBI over other indices. Deletion of any metric should result in loss of information and reduction in overall IBI sensitivity and precision. For example, Leonard and Orth (in press) developed a six metric-IBI for use in cool-water, West Virginia streams affected by cultural pollution. Using the same data, Angermeier and Karr (in press) included all 12 original metrics. While both IBIs apparently work, it remains to be seen how well the six metric-IBI responds to a broad spectrum of degradation. We advocate the use of replacement metrics where original metrics are found to be inappropriate. By fragmenting the IBI, its appeal as a generally applicable measure of biotic integrity is reduced. Potential users should thus exercise caution when attempting to modify the IBI to their particular regions. Maintaining the theoretical integrity of the IBI should be a primary concern in attempts at modification. However, we do recognize that replacement metrics may be difficult to formulate in certain areas of the U.S. (e.g., west of the Rocky Mountains) where species richness is lower than the Northeast.

Conceivably, the IBI as modified for these study drainages can be applied elsewhere in the Northeast, e.g. other coastal watersheds to the south of the Merrimack and Connecticut drainages. Comparable conditions to the upland and lowland native fish fauna of the Mohawk exist in the Hudson, Raquette, Black, Oswegatchie, Champlain, Grass, and St. Regis drainages (Smith 1986), thus allowing possible extension of the IBI developed for the Mohawk drainage. However, drainages to the north of the Merrimack are more depauperate, so it may not be possible to apply the IBI concept. Gordon (1937) found 30 species each in the Androscoggin and small coastal watersheds of New Hampshire, and only 25 species in the Saco drainage. More work is needed before the IBI can be applied to these drainages.

Extension of the IBI to the Northeast presents additional problems not encountered in the Midwest application. In particular, Greeley (1935) noted that several species native to lowland waters of the Mohawk drainage had their ranges extended via introductions into upland waters. For example, the inundation of Devasego Falls on Schoharie Creek by Gilboa dam provided access to the upper drainage for smallmouth bass (Micropterus dolomieui). Such situations were taken into account when species richness expectations were developed.

However, users who lack a thorough knowledge of the stream under study could artificially inflate criteria scores for species richness metrics.

Also, eastern tributaries of the Connecticut drainage in New Hampshire represent three different stream conditions: those draining low, intermediate, and large numbers of lakes (Bailey and Oliver 1939). Lake-oriented species are naturally more abundant in systems with a generous supply of lakes. Fewer, cold-water species exist in such systems because lake outlets generally are warmer and there are fewer spring-fed streams. Bailey and Oliver (1939) showed that while most of these tributaries have between 20 and 27 species present, the Israel and Mohawk (of New Hampshire) Rivers have 14 and 11 species, respectively. Both of the latter drain few lakes. The difference in species richness is mainly due to absence of non-native species; however, native sunfishes and yellow perch (*Perca flavescens*) were notably absent. We did not use data from these rivers in setting criteria for the species richness metrics, but we wish to emphasize that indiscriminant use of the IBI in these drainages may produce uncharacteristic low scores. These systems should be evaluated individually for IBI usage.

We stress that the IBIs developed for the Mohawk and Merrimack/Connecticut drainages need to be critically evaluated, especially in response to environmental perturbations. The metrics may need to be further modified, new metrics developed, or scoring criteria may need to be changed in light of additional data. The individual IBIs for each of these watersheds should be regarded as preliminary models to be tested and further refined. However, the concept of the IBI apparently can be applied to regions outside the Midwest with considerably different lotic fish faunas. While the IBI has been modified to suit regional patterns in fish zoogeography and ecology, the overall concept and its theoretical underpinnings are retained.

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Table 1. Comparison of metrics and scoring criteria used in the original IBI for the Midwest (Karr 1981; Fausch et al. 1984) and the IBIs developed for the Mohawk and Merrimack/Connecticut River drainages. Only those metrics and criteria different from the Midwest IBI are listed for Mohawk and Merrimack/Connecticut IBIs. A criteria score of 5 is best.

Midwest	Scoring Criteria 5	3	1	Mohawk	Scoring Criteria 5	3	1	Merrimack and Connecticut	Scoring Criteria 5	3	1
Species Richness and Composition											
1) Total number of fish species				1)				1)			
2) Number and identity of sunfish species[a]		Varies with		2) Number and identity of native native column species		Use of MSRL		2) Number and identity of native water column species		Use of MSRL	
3) Number and identity of darter species		stream size and region.		3) Number and identity of native benthic insectivorous species[b]		MSRL		3) Number and identity of native benthic insectivorous species[b]		MSRL	
4) Number and identity of sucker species		Use maximum species richness		4) Number and identity of native sucker species	>1	1	<1	4) Number and identity of native sucker species	>1	1	<1
5) Number and identity of intolerant species		lines (MSRL)		5) Number and identity of native intolerant species		MSRL		5) Number and identity of native intolerant species	>1	1	<1
6) Proportion of individuals as green sunfish	<5%	5-20%	>20%	6) Proportion of individuals as white sucker				6) Proportion of individuals as white sucker			
Trophic Composition											
7) Proportion of individuals as omnivores[c]	<20%	20-45%	>45%	7)				7)			
8) Proportion of individuals as insectivorous cyprinids	>45%	20-45%	<20%	8)				8) Proportion of individuals as insectivores[d]	>65%	30-65%	<30%
9) Proportion of individuals as top carnivores[e]	>5%	1-5%	<1%	9)	>2%	1-2%	<1%	9)	>10%	5-10%	<5%

Table 1 (continued).

Midwest	Scoring Criteria 5	3	1	Mohawk	Scoring Criteria 5	3	1	Merrimack and Connecticut	Scoring Criteria 5	3	1
Fish Abundance and Condition											
10) Number of individuals in sample	Varies with stream size			10) Density of individuals in sample	Use of MDL[f]			10) Density of individuals in sample	Use of MDL[f]		
11) Proportion of individuals as hybrids	0	0-1%	>1%	11)				11)			
12) Proportion of individuals with disease, tumors, fin damage, and other anomalies	0	0-1%	>1%	12)				12)			

[a] Excludes blackbasses (Micropterus).

[b] Excludes insectivorous sucker species because suckers are included in own metric.

[c] Definition of omnivore is that used by Fausch et al. (1984): diet of 25% plant material and rest being animal material.

[d] Insectivore defined as: majority of diet includes insects (≧ 75%) with rest of diet including no more than 10% fish.

[e] Top carnivore classification includes species that are piscivores (diet more than 90% fish) and insectivores/piscivores (more than 10% fish, plus invertebrates only) (Lynn Schrader and Kurt Fausch, Colorado State University, pers. comm.).

[f] See text for explanation.

Appendix. List of species collected at only those sites used in the analysis (Greeley 1935; Bailey 1938; Bailey and Oliver 1939; Bergin 1972; Madore 1974). Species are classified as omnivores (OMN), insectivorous cyprinids (IC), insectivores (I), benthic insectivores (BI), water column (WC), top carnivores (TC), and intolerant (INT) for the Mohawk (M) and Merrimack/Connecticut (MC) drainages. Because of the difference in trophic metrics incorporated into the IBI for each drainage (see Table 1), not all species were classified identically for each drainage. Scientific and common names follow Robbins et al. (1980). Superscripts are: 1 = native to Merrimack and Connecticut drainages and 2 = native to Mohawk drainage.

Family Species	OMN	IC	I	TC	WC	BI	INT
Anguillidae							
American eel (*Anguilla rostrata*)[1,2]				M,MC	M,MC		
Salmonidae							
Brown trout (*Salmo trutta*)				M,MC			
Rainbow trout (*Salmo gairdneri*)				M,MC			
Atlantic salmon (*Salmo salar*)[1,2]					M,MC		M,MC
Brook trout (*Salvelinus fontinalis*)[1,2]				M,MC	M,MC		M,MC
Esocidae							
Redfin pickerel (*Esox americanus americanus*)[1,2]				M,MC	M,MC		M,MC
Northern pike (*Esox lucius*)				MC			
Chain pickerel (*Esox niger*)[1,2]				M,MC	M,MC		M,MC
Cyprinidae							
Central stoneroller (*Campostoma anomalum*)							

Appendix (continued).

Family Species	OMN	IC	I	TC	WC	BI	INT
Goldfish (Carassius auratus)	M,MC						
Redside dace (Clinostomus elongatus)[2]		M			M		M
Lake chub (Couesius plumbeus)[1,2]		M	MC		M,MC		
Common carp (Cyprinus carpio)	M,MC						
Cutlips minnow (Exoglossum maxillingua)[2]		M				M	M
Eastern silvery minnow (Hybognathus regius)[1,2]	M,MC				M,MC		M,MC
Hornyhead chub (Nocomis biguttatus)[2]		M			M		M
Golden shiner (Notemigonus crysoleucas)[1,2]	M,MC				M,MC		
Satinfin shiner (Notropis analostanus)[2]		M			M		
Blacknose shiner (Notropis heterolepis)[2]		M			M		M
Emerald shiner (Notropis atherinoides)[2]		M			M		
Bridle shiner (Notropis bifrenatus)[1,2]		M	MC		M,MC		
Common shiner (Notropis cornutus)[1,2]	M,MC				M,MC		
Spottail shiner (Notropis hudsonius)[1,2]		M	MC		M,MC		
Rosyface shiner (Notropis rubellus)[2]		M			M		M
Spotfin shiner (Notropis spilopterus)[2]		M			M		
Northern redbelly dace (Phoxinus eos)[1]					M,MC		M,MC
Bluntnose minnow (Pimephales notatus)[2]	M				M		
Fathead minnow (Pimephales promelas)[2]	M				M		
Blacknose dace (Rhinichthys atratulus)[1,2]		M	MC			M,MC	
Longnose dace (Rhinichthys cataractae)[1,2]		M	MC			M,MC	
Creek chub (Semotilus atromaculatus)[1,2]		M	MC		M,MC		
Fallfish (Semotilus corporalis)[1,2]		M	MC		M,MC		

Appendix (continued).

Family Species	OMN	IC	I	TC	WC	BI	INT
Pearl dace (*Semotilus margarita*)[2]		M			M		
Catostomidae							
Longnose sucker (*Catostomus catostomus*)[1,2]	M,MC						
White sucker (*Catostomus commersoni*)[1,2]	M,MC						
Creek chubsucker (*Erimyzon oblongus*)[1,2]			MC			M,MC	M,MC
Northern hog sucker (*Hypentelium nigricans*)[2]						M	M
Shorthead redhorse (*Moxostoma macrolepidotum*)[2]						M	
Ictaluridae							
Yellow bullhead (*Ictalurus natalis*)[2]			MC			M	
Brown bullhead (*Ictalurus nebulosus*)[1,2]			MC			M,MC	
Stonecat (*Noturus flavus*)[2]						M	M
Tadpole madtom (*Noturus gyrinus*)[2]			MC			M	M
Margined madtom (*Noturus insignis*)[2]						M	M
Percopsidae							
Trout-perch (*Percopsis omiscomaycus*)[2]					M		
Gadidae							
Burbot (*Lota lota*)[1]				MC	MC		
Cyprinodontidae							
Banded killifish (*Fundulus diaphanus*)[1,2]			MC		M,MC		
Gasterosteidae							
Brook stickleback (*Culaea inconstans*)[2]					M		
Percichthyidae							
White perch (*Morone americana*)[1,2]				MC	MC		

Appendix (continued).

Family Species	OMN	IC	I	TC	WC	BI	INT
Centrarchidae							
Rock bass (Ambloplites rupestris)				MC			
Banded sunfish (Enneacanthus obesus)[1]			MC		MC		MC
Redbreast sunfish (Lepomis auritus)[1,2]			MC		M,MC		
Pumpkinseed (Lepomis gibbosus)[1,2]			MC		M,MC		
Green sunfish (Lepomis cyanellus)				M			
Bluegill (Lepomis macrochirus)			MC				
Smallmouth bass (Micropterus dolomieui)[2]				M,MC	M		
Largemouth bass (Micropterus salmoides)				M,MC			
White crappie (Pomoxis annularis)				M,MC			
Black crappie (Pomoxis nigromaculatus)				M,MC			
Percidae							
Greenside darter (Etheostoma blennioides)[2]						M	M
Fantail darter (Etheostoma flabellare)							
Swamp darter (Etheostoma fusiforme)[1]			MC			MC	MC
Tessellated darter (Etheostoma olmstedi)[1,2]			MC			M,MC	
Logperch (Percina caprodes)[2]						M	
Yellow perch (Perca flavescens)[1,2]				M,MC	M,MC		
Walleye (Stizostedion vitreum)				M,MC			
Cottidae							
Slimy sculpin (Cottus cognatus)[1,2]			MC			M,MC	M,MC

Figure 1. Map showing general locations of watersheds used in developing the index of biotic integrity for Northeast streams: Letters are A = Schoharie Creek drainage, B = Chicopee River drainage, and C = Nashua River drainage.

Figure 2. Total number of native fish species versus $\log_{10}$ watershed area for sites in the Mohawk River drainage (A) and the Merrimack and Connecticut River drainages (B). Boxes and triangles represent sites with borderline scores (see text for further explanation). Numbers (5, 3, 1) represent criteria score regions.

Figure 3. Total number of native water column species versus $\log_{10}$ watershed area for the Mohawk (A) and Merrimack/Connecticut (B) drainages. Numbers, triangles, and boxes as in Fig. 2.

Figure 4. Total number of native benthic insectivorous species versus $\log_{10}$ watershed area for the Mohawk (A) and Merrimack/Connecticut (B) drainages, and total number of native intolerant species for the Mohawk (C) drainage. Numbers, triangles, and boxes as in Fig. 2.

Figure 5. Plot of $\log_{10}$ total individuals/100 m^2 versus $\log_{10}$ watershed area for the Schoharie (A) and Nashua/Chicopee (B) drainages. Solid line represents the maximum density line (MDL). The MDL forms the upper bound for about 95% of the sites. Dot-dash lines represent trisections of ordinate into criteria regions similar to the MSRL-type graphs. Numbers, triangles, and boxes as in Fig. 2.

Figure 6. Distribution of final IBI scores and summary statistics for sites in the Schoharie (S) and Nashua and Chicopee (N/C) drainages relative to an interpretive scale developed by Karr (1981). Final IBI score categories are as follows: NF = no fish, VP = very poor (final score $\leq$ 23), VP-P = very poor to poor (24-27), P = poor (28-35), P-F = poor to fair (36-38), F = fair (39-44), F-G = fair to good (45-47), G = good (48-52), G-E = good to excellent (53-56), E = excellent (57-60). Numbers above bars refer to number of sites. Further abbreviations are as follows: n = sample size, $\overline{x}$ = mean, sd = standard deviation, r = range.

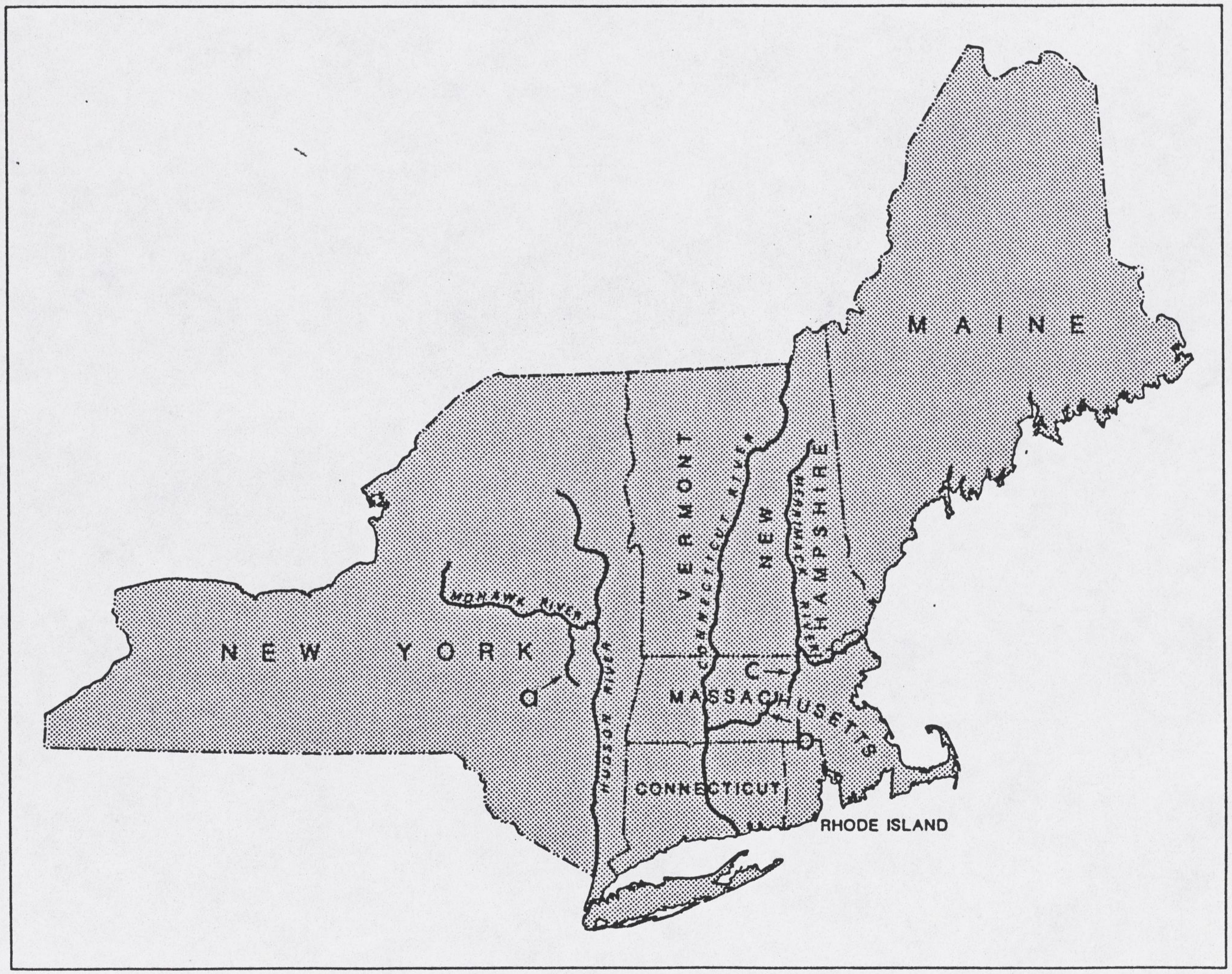
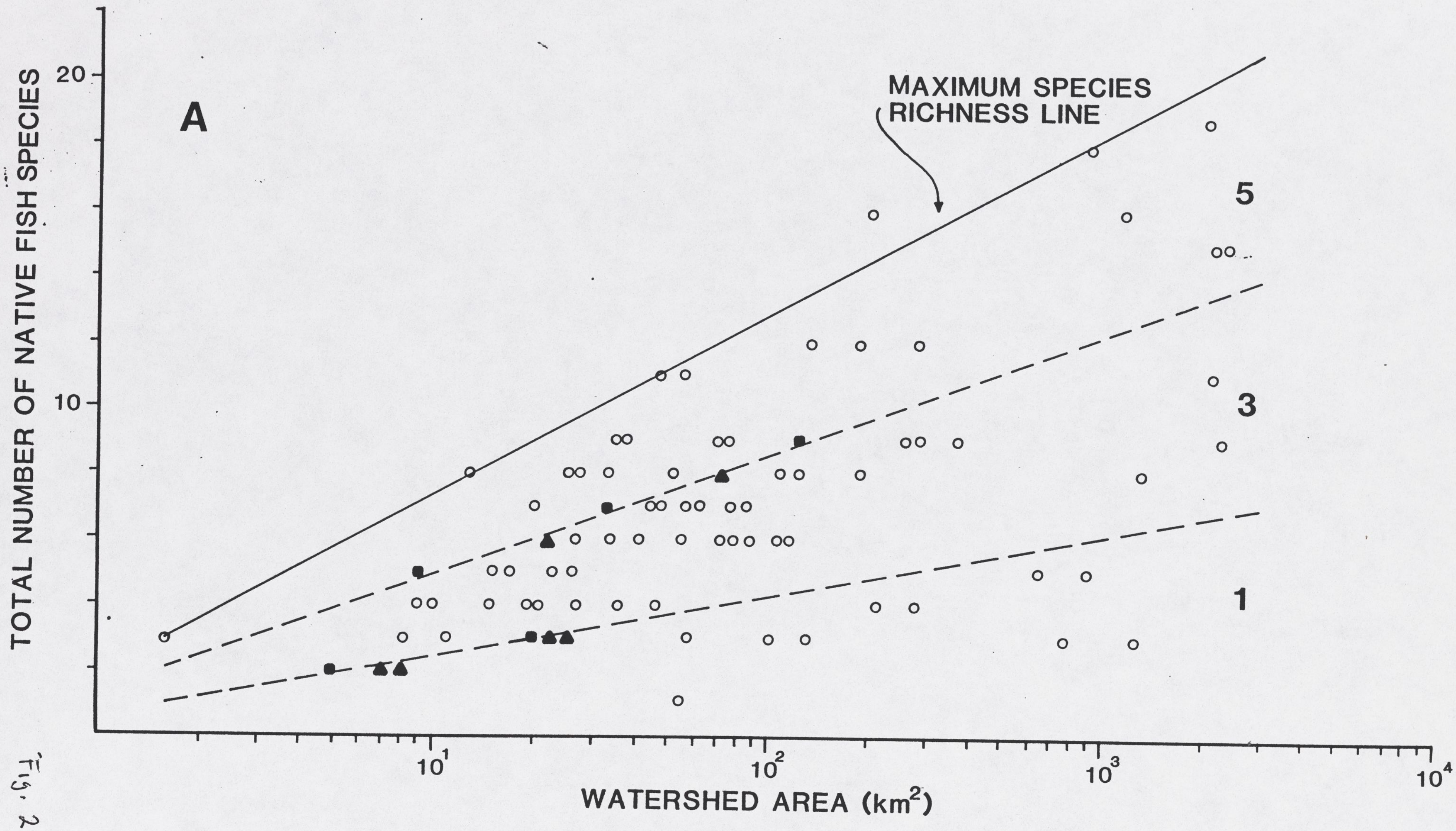

MAINE
VERMONT
NEW HAMPSHIRE
CONNECTICUT RIVER
MERRIMACK RIVER
MOHAWK RIVER
HUDSON RIVER
NEW YORK
MASSACHUSETTS
CONNECTICUT
RHODE ISLAND
a
c

Fig. 1


A
MAXIMUM SPECIES
RICHNESS LINE
5
3
1
TOTAL NUMBER OF NATIVE FISH SPECIES
20
10
10^1
10^2
10^3
10^4
WATERSHED AREA (km^2)
Fig. 2a

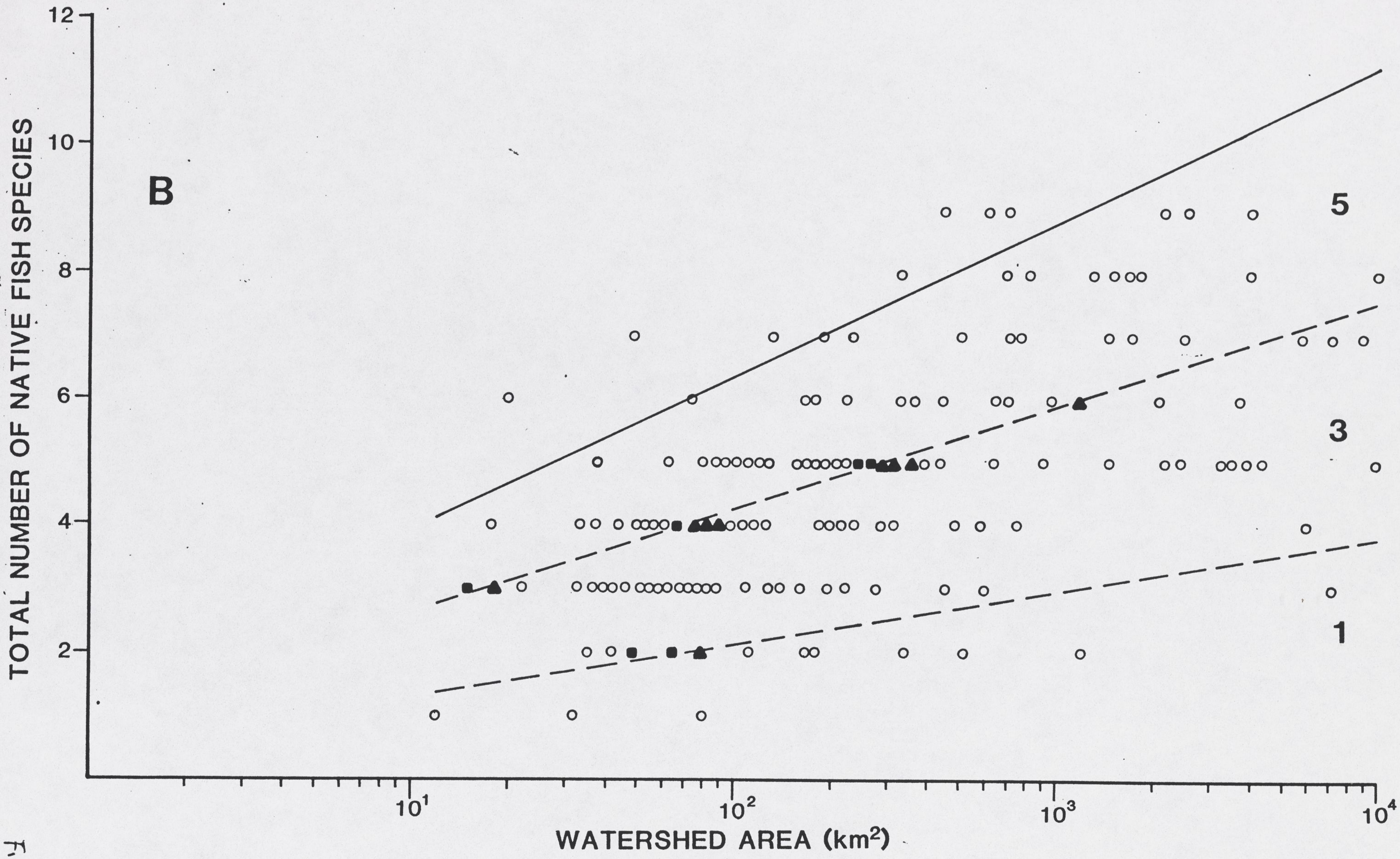

B
TOTAL NUMBER OF NATIVE FISH SPECIES
WATERSHED AREA (km2)
12
10
8
6
4
2
10¹
10²
10³
10⁴
5
3
1

Fig. 2b

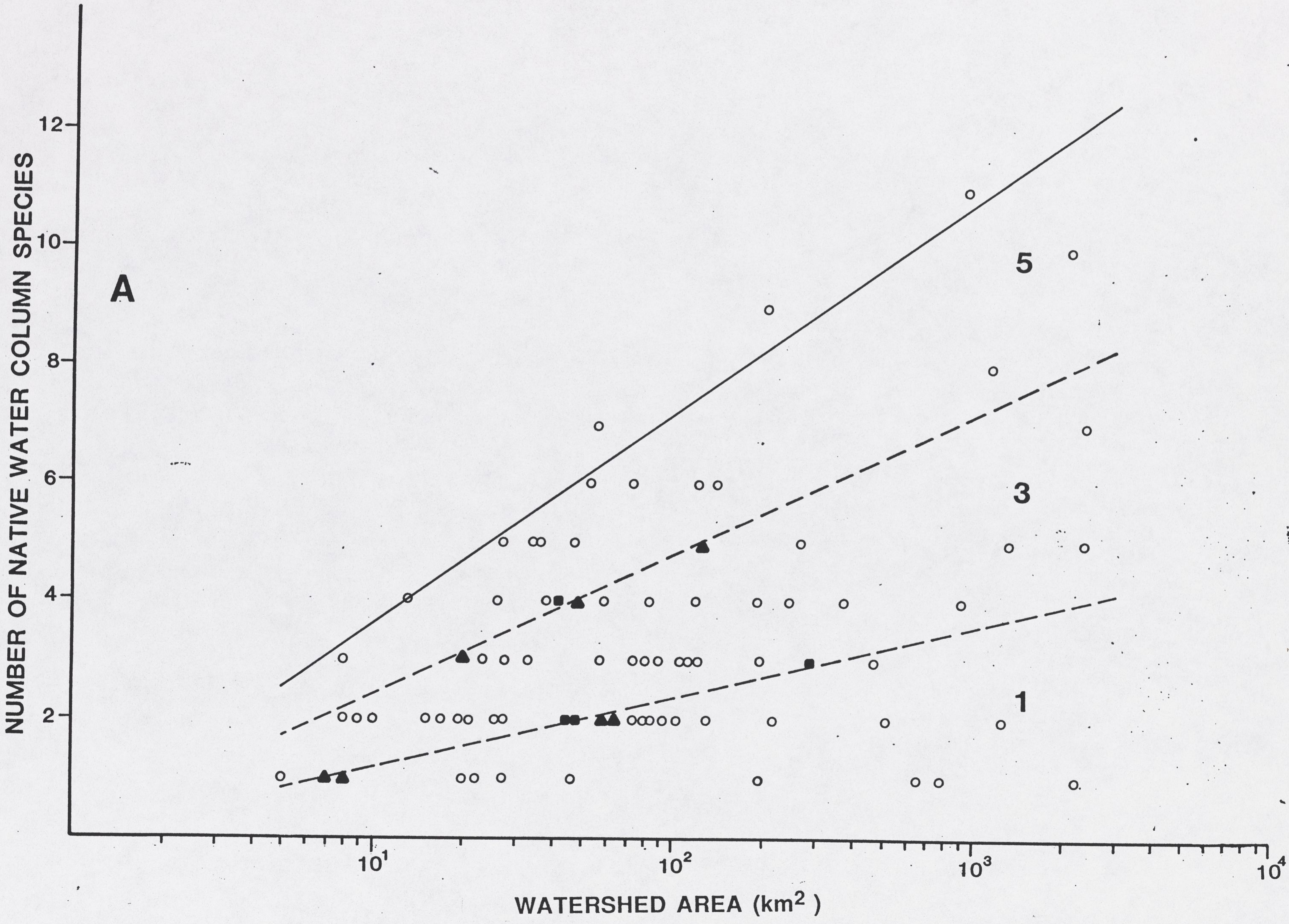
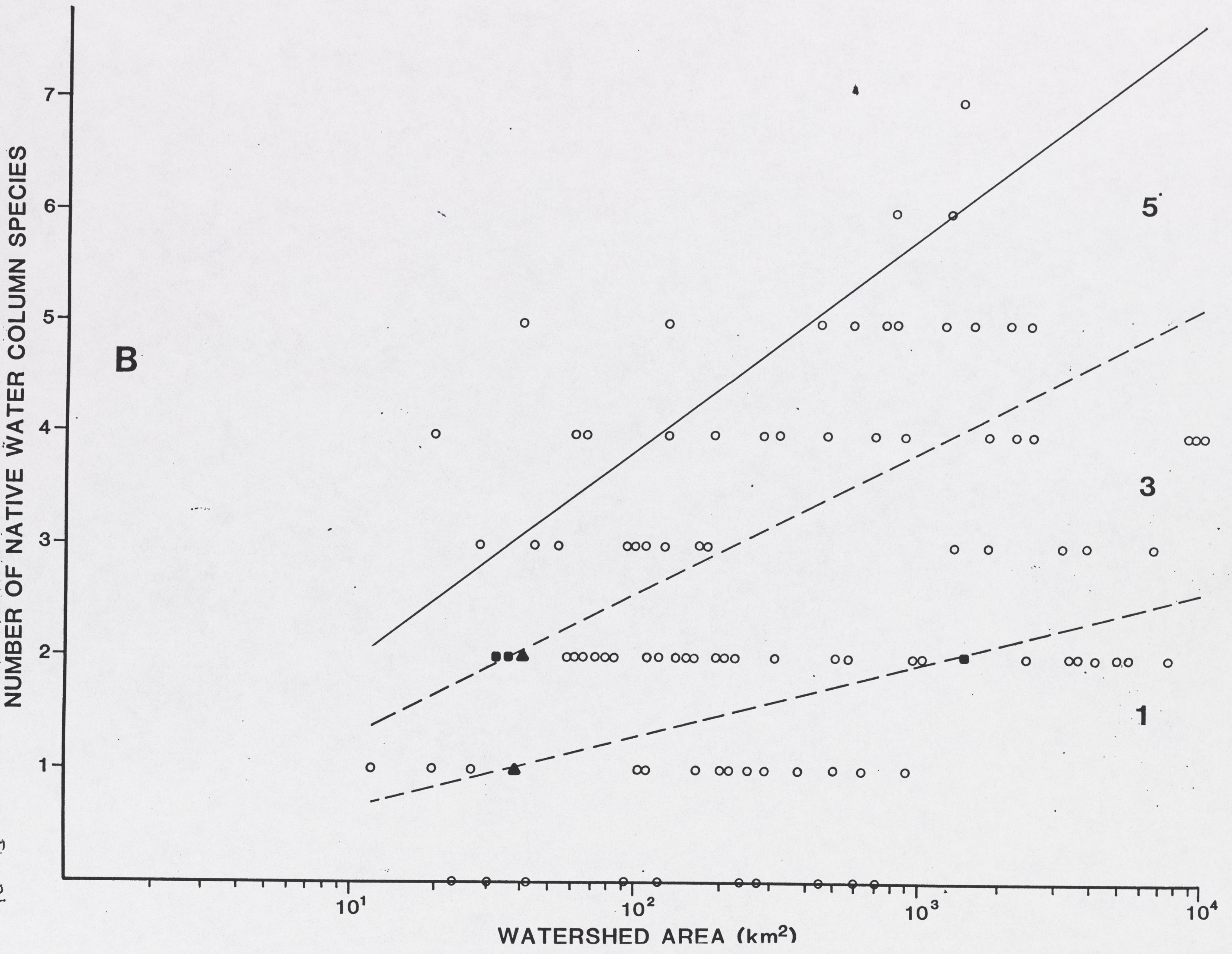
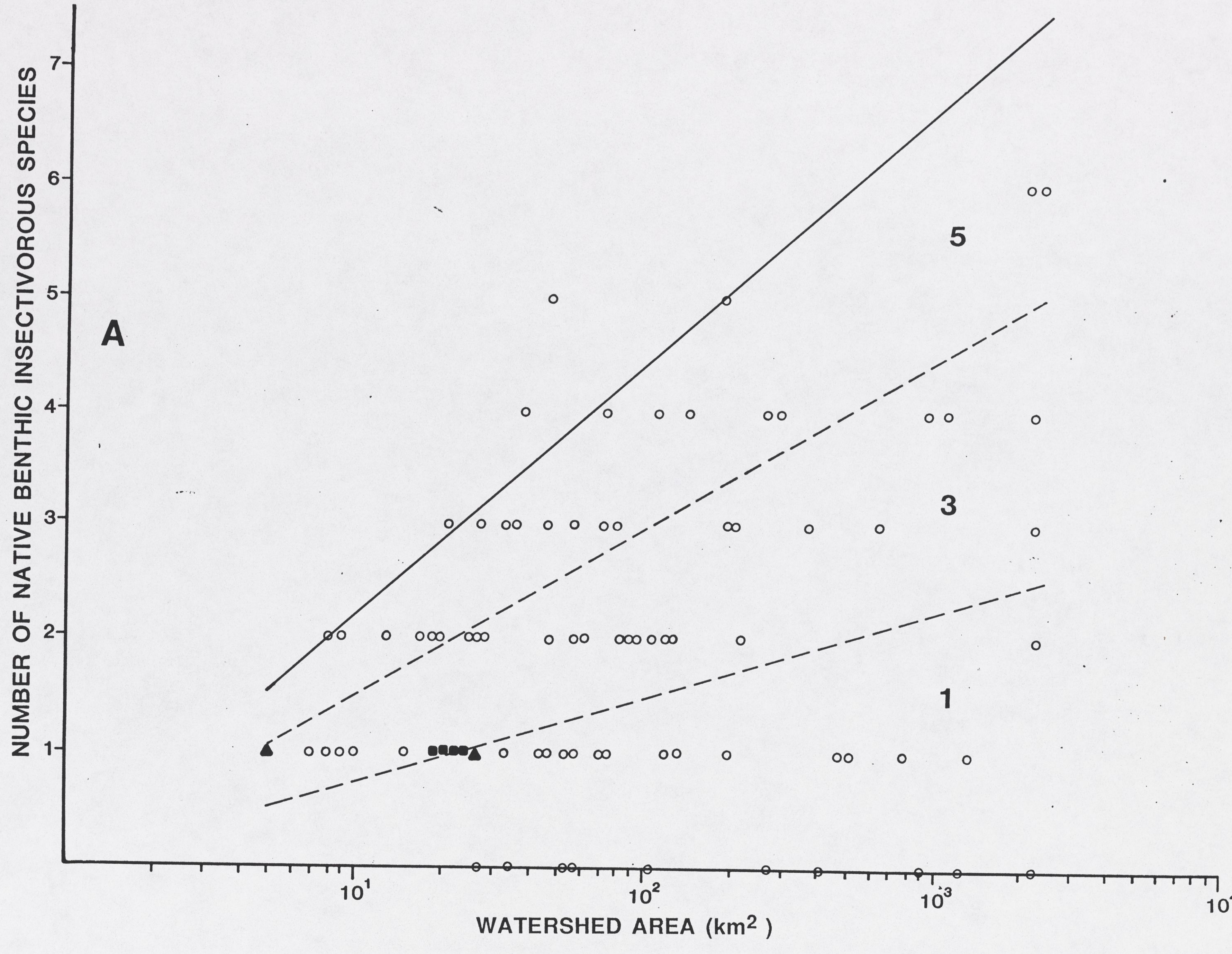

A
NUMBER OF NATIVE WATER COLUMN SPECIES
12
10
8
6
4
2
5
3
1
10^1
10^2
10^3
10^4
WATERSHED AREA (km^2)

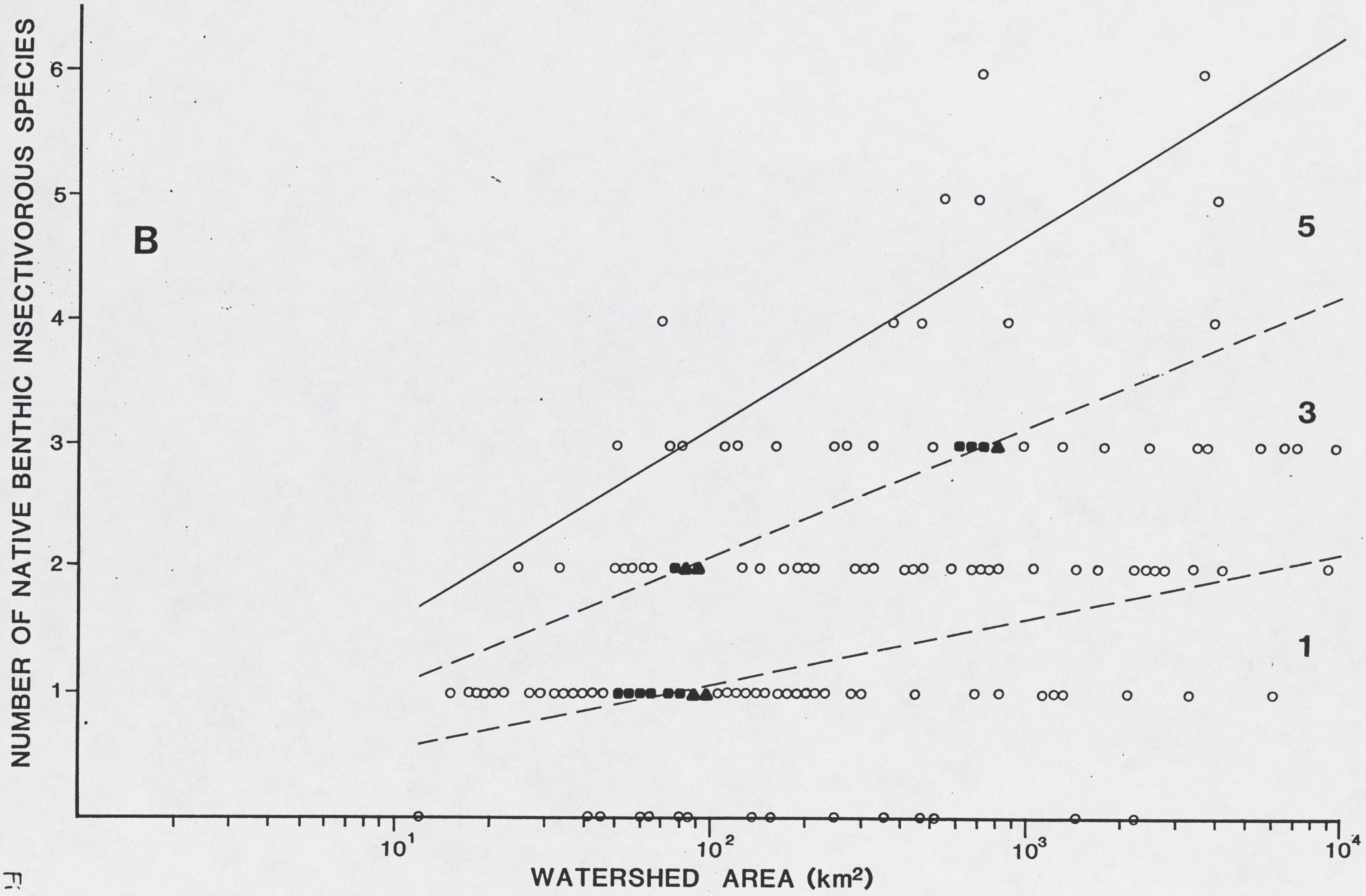
Fig. 3a



NUMBER OF NATIVE WATER COLUMN SPECIES
B
7
6
5
4
3
2
1
5
3
1
10^1
10^2
10^3
10^4
WATERSHED AREA (km^2)

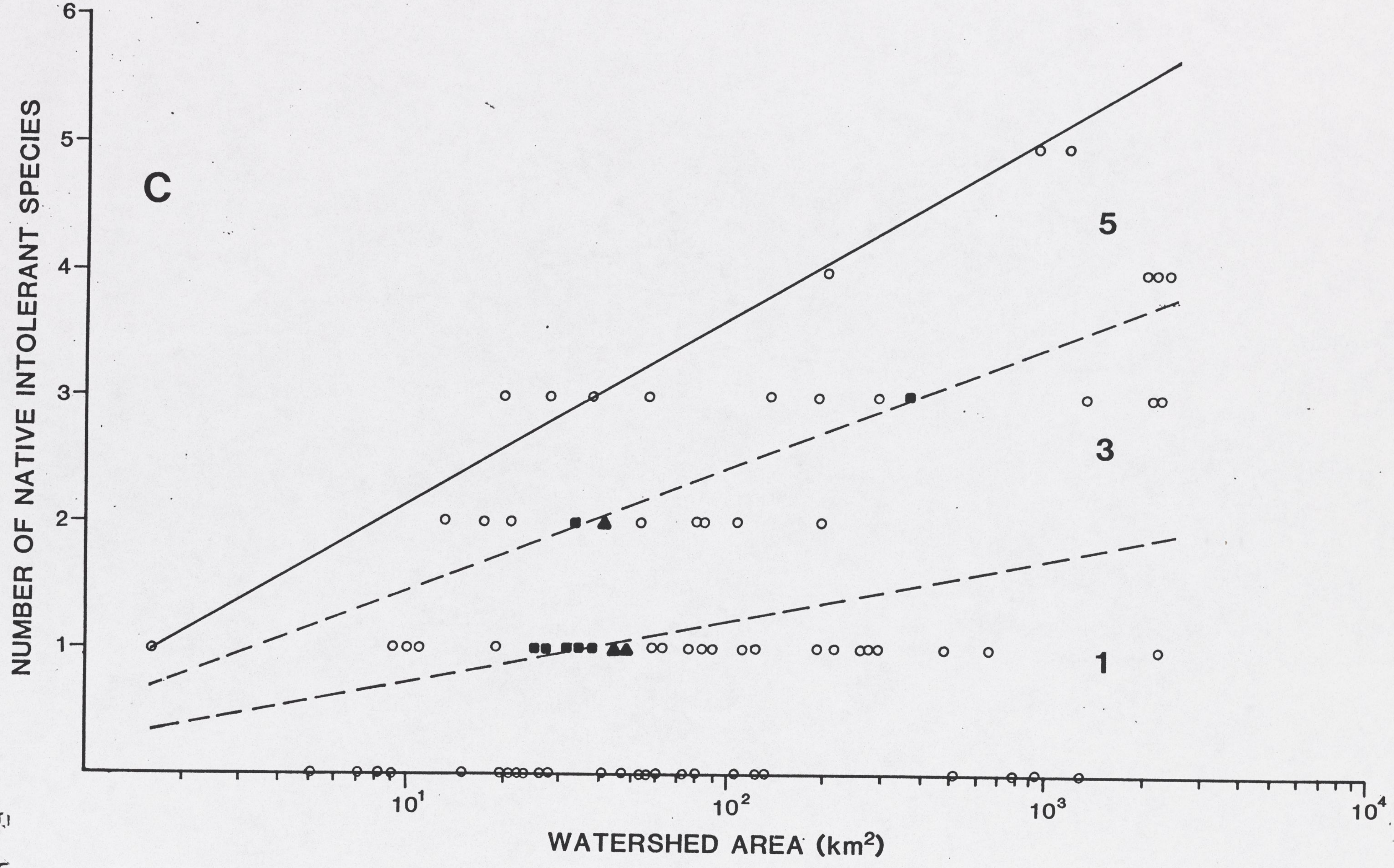
Fig. 3b


NUMBER OF NATIVE BENTHIC INSECTIVOROUS SPECIES
7
6
5
4
3
2
1
A
5
3
1
10^1
10^2
10^3
10^4
WATERSHED AREA (km^2)
Fig. 4a



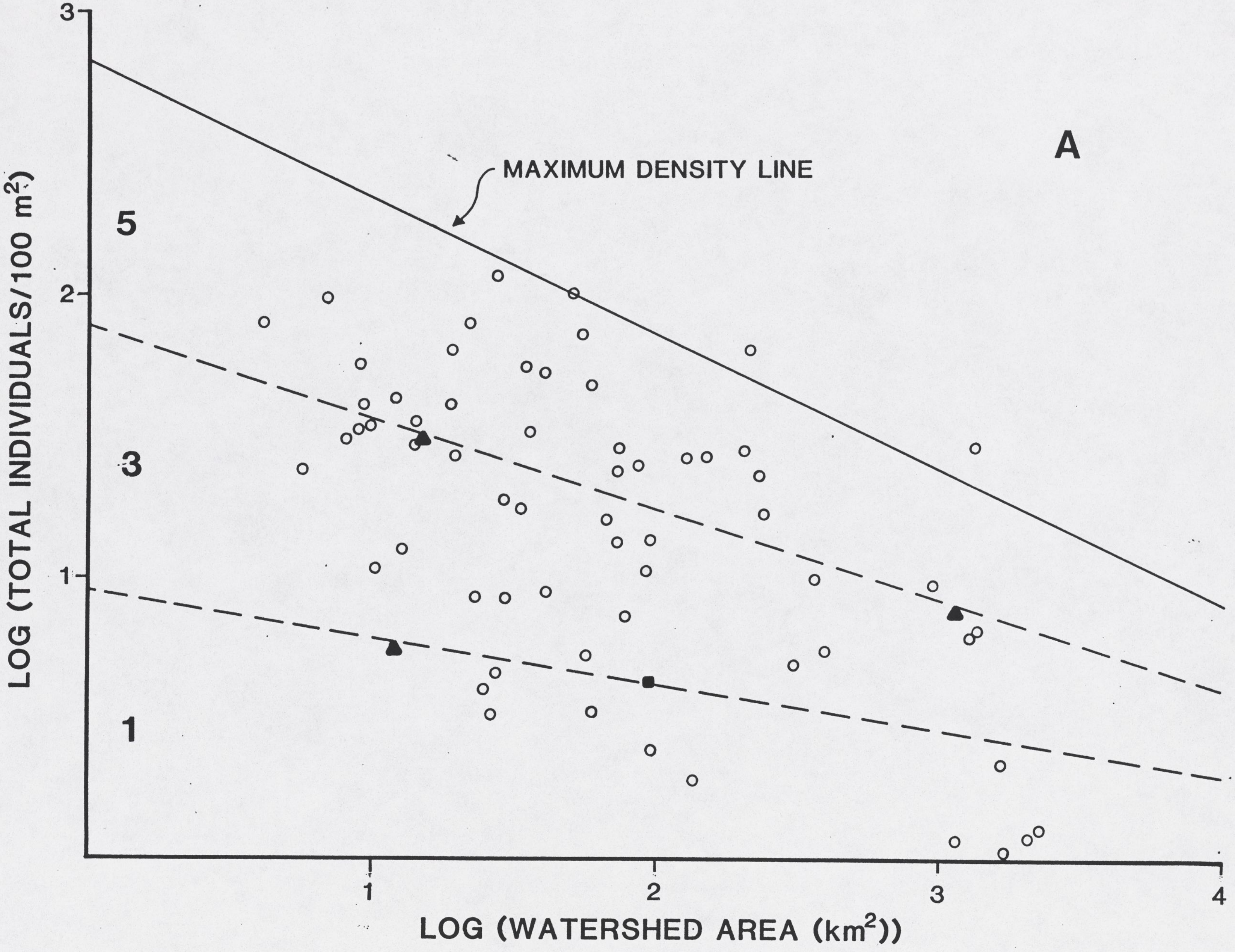
B
NUMBER OF NATIVE BENTHIC INSECTIVOROUS SPECIES
WATERSHED AREA (km2)
1
2
3
4
5
6
10^1
10^2
10^3
10^4
5
3
1

Fig. 4b



C
NUMBER OF NATIVE INTOLERANT SPECIES
WATERSHED AREA (km²)
5
3
1

Fig. 4c



A
MAXIMUM DENSITY LINE
5
3
1
LOG (TOTAL INDIVIDUALS/100 m²)
3
2
1
1
2
3
4
LOG (WATERSHED AREA (km²))

Fig. 5a

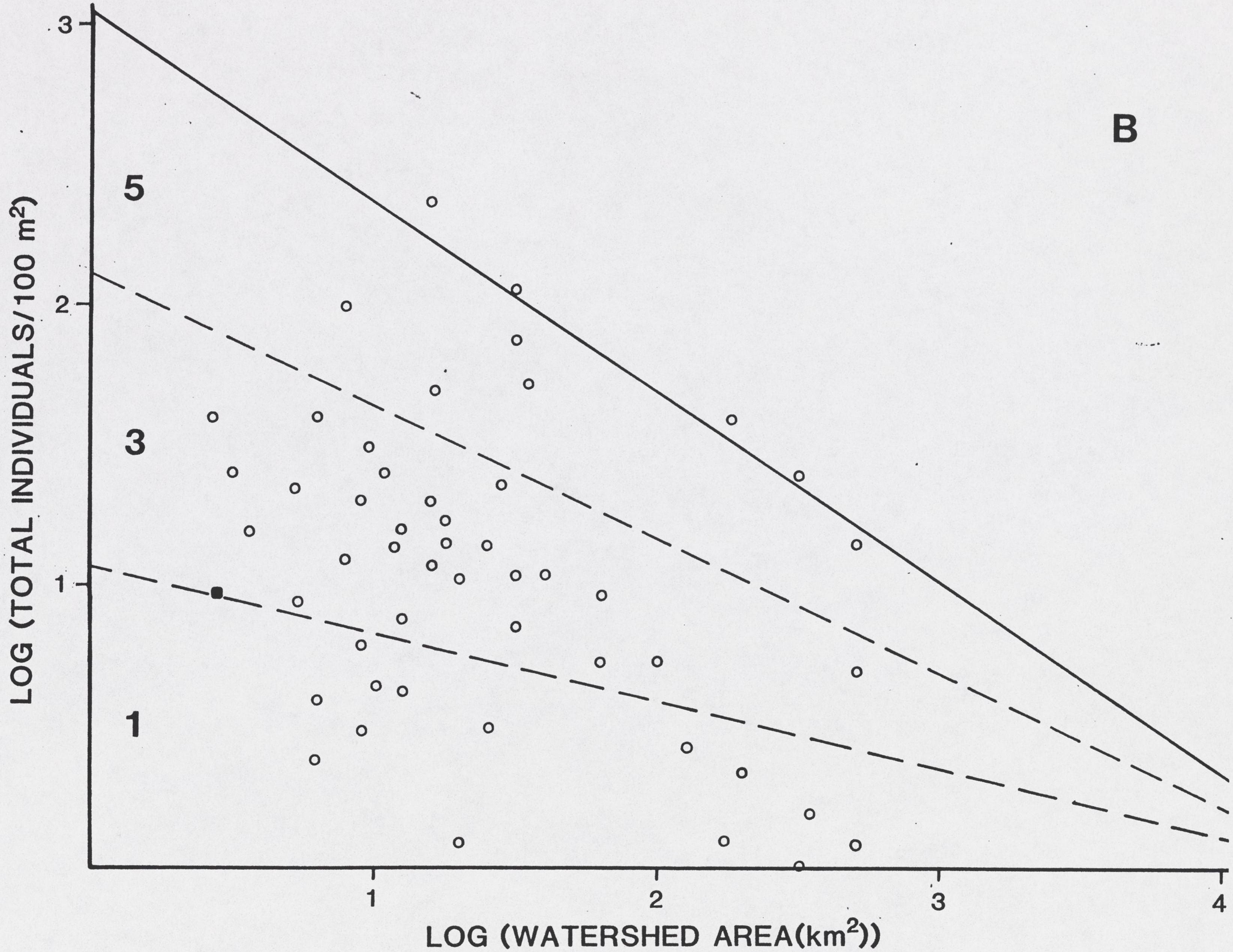
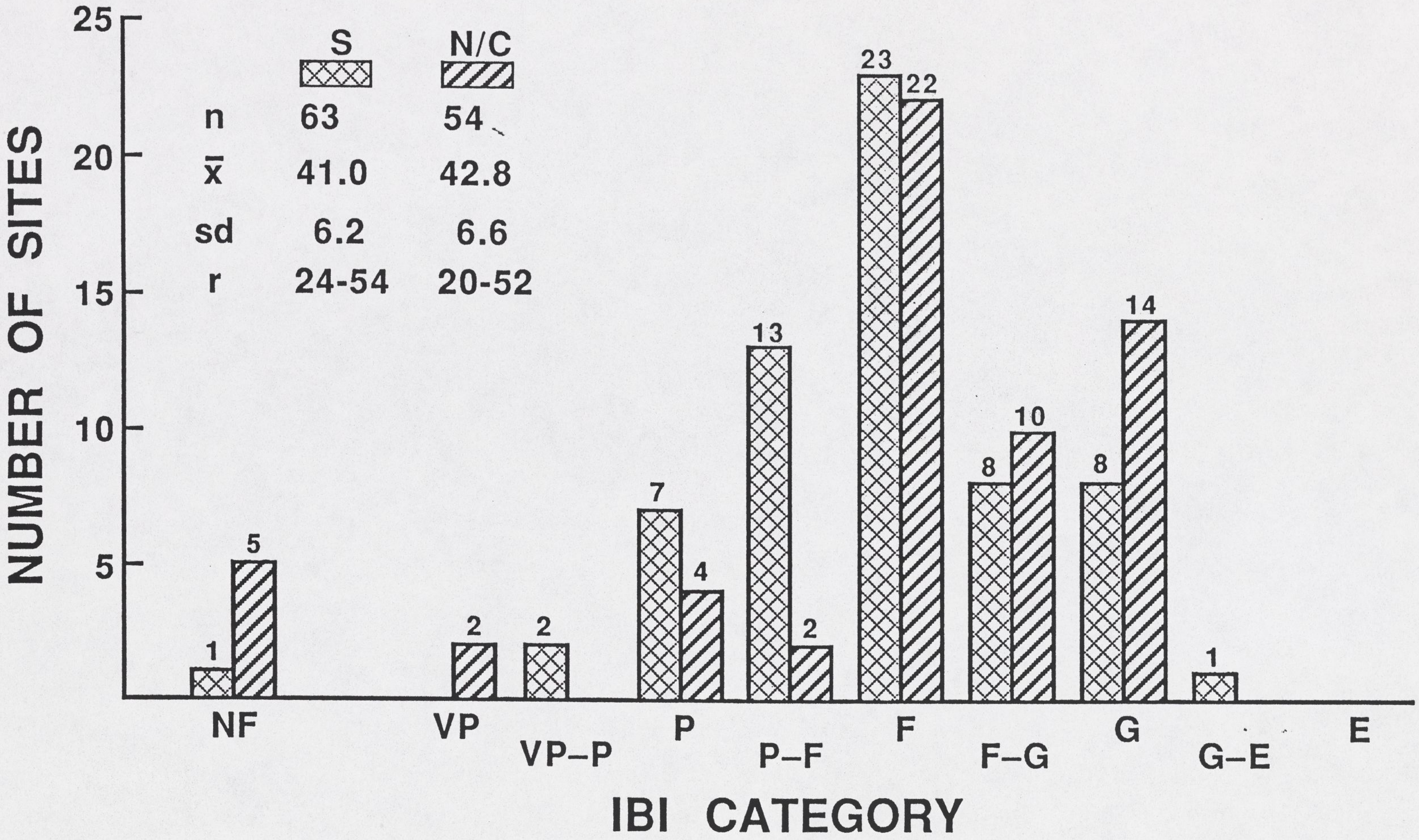

B
5
3
1
LOG (TOTAL INDIVIDUALS/100 m²)
LOG (WATERSHED AREA(km²))
1
2
3
4

Fig. 5b



NUMBER OF SITES
25
20
15
10
5
S
N/C
n 63 54
$\bar{x}$ 41.0 42.8
sd 6.2 6.6
r 24-54 20-52
1
5
2
2
7
4
13
2
23
22
8
10
8
14
1
NF
VP
VP–P
P
P–F
F
F–G
G
G–E
E
IBI CATEGORY

Fig. 6