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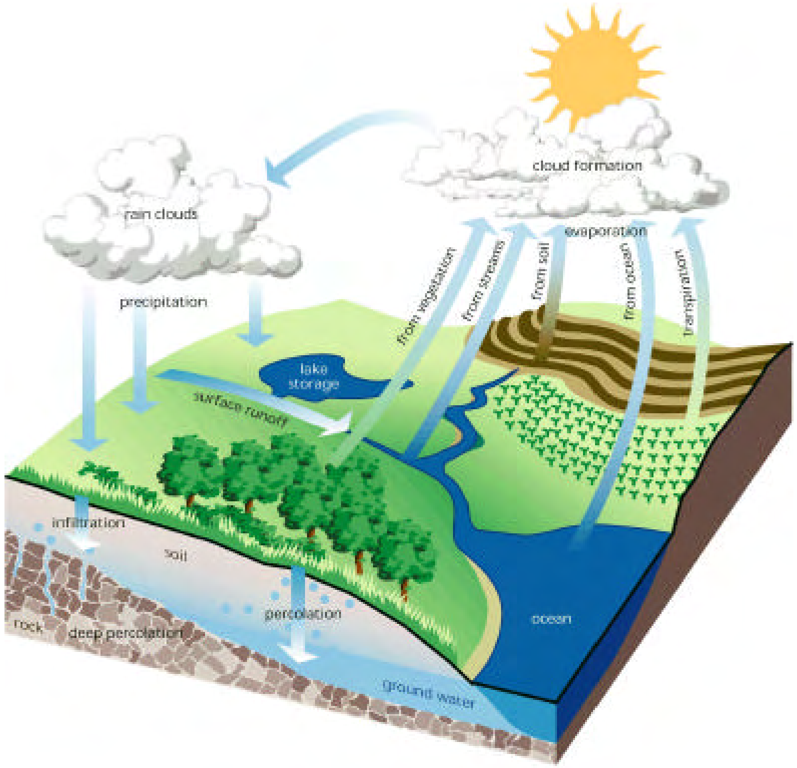

VII. Total Maximum Daily Loads (TMDL)

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TABLE OF CONTENTS

AGNPS 98: A SUITE OF WATER QUALITY MODELS FOR WATERSHED USE

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ABSTRACT: Watershed scale evaluation is an essential step in recommending best management practices and/or setting Total Maximum Daily Load (TMDL) pollutant allocations. Allocations established without comprehensive studies will likely require treatment of lands that will contribute little to load reductions and insufficient treatment of higher contributing lands. The Agricultural Nonpoint Source model (AGNPS 98) was developed to perform these necessary evaluations. AGNPS 98 is a suite of computer models developed to assist the user with quantifying the impacts of agricultural nonpoint source pollution on water quality and the environment. The models in AGNPS 98 include: (1) a watershed-scale, continuous-simulation, pollutant loading computer model designed to quantify & identify the source of pollutant loadings anywhere in the watershed for optimization & risk analysis; (2) a set of stream network, corridor, & water quality computer models designed to predict & quantify the effects of bank erosion & failures, bank mass wasting, bed aggradation & degradation, burial & re-entrainment of contaminants, and streamside riparian vegetation on channel morphology and pollutant loadings; (3) a watershed-scale, stream network, water temperature computer model to predict daily average, minimum, & maximum water temperatures; (4) a set of salmonid life-cycle models designed specifically to quantify the impact of pollutant loadings on their spawning & rearing habitats as well as include other important life-threatening obstacles; and (5) an economic model that determines the net economic value of Pacific Northwest salmonids restored to either the commercial or recreational catch. This paper will discuss these components and demonstrate the capabilities of the model using Goodwin Creek Watershed in Mississippi. By applying a watershed management approach, TMDL's can be better defined and practices can be better established in setting these water quality standards.

INTRODUCTION

The development of a continuous version of the single event AGricultural NonPoint Source model (AGNPS) watershed model (Young et al, 1989) has been in progress, in one form or another, since the 1980's. This continuous version, the ANNualized AGricultural NonPoint Source model (AnnAGNPS) (Cronshey and Theurer, 1998), is available through the Internet WEB address:

http://www.sedlab.olemiss.edu/AGNPS98.html

Since AnnAGNPS is designed to analyze the impact of non-point source pollutants from predominately agricultural watersheds on the environment, other models that simulate additional processes have been integrated with AnnAGNPS. These integrated models have been developed within the AGNPS 98 suite of modules. Each module provides information needed by other modules to enhance the predictive capabilities of each. The modules in AGNPS 98 include: (1) AnnAGNPS, a watershed-scale, continuous-simulation, pollutant loading computer model designed to quantify & identify the source of pollutant loadings anywhere in the watershed for optimization & risk analysis; (2) Conservational Channel Evolution and Pollutant Transport System (CONCEPTS) (Langendoen et al, 1998), a set of stream network, corridor, & water quality computer models designed to predict & quantify the effects of bank erosion & failures, bank mass wasting, bed aggradation & degradation, burial & re-entrainment of contaminants, and streamside riparian vegetation on channel morphology and pollutant loadings; (3) The Stream Network TEMPerature model (SNTEMP) (Theurer et al, 1984), a watershed-scale, stream network, water temperature computer model to predict daily average, minimum, & maximum water temperatures; (4) The Sediment Intrusion & Dissolved Oxygen (SIDO) model (Alonso et al, 1996), a set of salmonid life-cycle models designed specifically to quantify the impact of pollutant loadings on their spawning & rearing habitats as well as include other important life-threatening obstacles; and (5) an economic model that determines the net economic value of Pacific Northwest salmonids restored to either the commercial or recreational catch.

As part of the input data preparation process there are a number of modules that support the user in developing the needed AGNPS 98 databases. These include: (1) the TOpographic PArameteriZation program (TOPAZ) (Garbrecht and Martz, 1995), to generate cell and stream network information from a watershed digital elevation model (DEM) and provide all of the topographic related information for AnnAGNPS. A subset of TOPAZ, TOPAGNPS, is the set of TOPAZ modules used for AGNPS 98. The use of the TOPAGNPS generated stream network is also incorporated by CONCEPTS to provide the link of where upland sources are entering the channel and then routed downstream; (2) The AGricultural watershed FLOWnet generation program (AGFLOW) (Bingner et al, 1997; Bingner et al, 2001a) is used to determine the topographic-related input parameters for AnnAGNPS and to format the TOPAGNPS output for importation into the form needed by AnnAGNPS; (3) The Generation of weather Elements for Multiple applications (GEM) program (Johnson et al, 2000) is used to generate the climate information for AnnAGNPS; (4) The program Complete Climate takes the information from GEM and formats the data for use by AnnAGNPS, along with determining a few additional parameters; (5) A graphical input editor that assists the user in developing the AnnAGNPS database (Bingner et al, 1998); (6) A visual interface program to view the TOPAGNPS related geographical information system (GIS) data (Bingner et al, 1996); and, (7) A conversion program that transforms a single event AGNPS 5.0 dataset into what is needed to perform a single event simulation with AnnAGNPS. In addition to these input modules, there are procedures that utilize the Arcview program to facilitate the use of TOPAGNPS. There is an output processor that can be used to help analyze the results from AnnAGNPS by generating a summary of the results in tabular or GIS format.

This paper will provide some details and background on the AGNPS 98 modules. Also, included is a study of the runoff and sediment yield comparisons of simulated and measured values from the Goodwin Creek Watershed in Mississippi.

AGNPS 98 MODULE DESCRIPTION

AnnAGNPS

AnnAGNPS is the pollutant loading modeling module designed for risk and cost/benefit analyses. It is a batch-process, continuous-simulation, surface-runoff, pollutant loading (PL) computer model written in standard ANSI Fortran 90, which provides for studies of very large watersheds. The model was developed to simulate long-term sediment & chemical transport from ungaged agricultural watersheds. The basic modeling components are hydrology, sediment, nutrient, and pesticide transport. Land area (cell) representations of a watershed are used to provide landscape spatial variability. Each cell homogeneously represents the landscape within its respective land area boundary. The physical or chemical constituents are routed from their origin within the land area and are either deposited within the stream channel system or transported out of the watershed. Pollutant loadings (PLs) can then be identified at their source and tracked as they move through the watershed system.

The philosophy of the development of AnnAGNPS has been to maintain the simplicity of the single event version, AGNPS, while enhancing the features that are needed for a continuous simulation. The use of NRCS or ARS technology was adopted whenever feasable to ensure acceptance and readily available databases nationwide. This provides a watershed model that incorporates currently accepted science and databases from any location in the country, capable of simulating most of the management practices that are applied on farms.

The main components within AnnAGNPS are the incorporation of the SCS curve number technique (USDA, 1972) used to generate daily runoff and RUSLE 1.05 technology (Renard et al, 1997) to generate daily sheet and rill erosion from fields (Geter and Theurer, 1998). The parameters that are used for RUSLE are also used within AnnAGNPS. Each cell within AnnAGNPS can have different RUSLE parameters associated with describing the farm operations. This can provide a spatial and temporal variation of the management practices associated with a watershed system. Sheet and rill erosion is calculated for each runoff event during a user-defined simulation period and averaged for this same time period. A runoff event can occur from any combination of rainfall, snowmelt, and irrigation. All subsequent sediment is routed throughout the stream system down to the watershed outlet. An account of each individual field contribution to the sediment yield at any user-defined stream location can be determined.

Since RUSLE is used only to predict sheet and rill erosion and not field deposition, a delivery ratio of the sediment yield from this erosion to sediment delivery to the stream is needed. The Hydro-geomorphic Universal Soil Loss Equation (HUSLE) is used for this procedure (Theurer and Clarke, 1991). The procedure was initially developed to predict the total sediment yield at a user-defined point in the stream system using spatially- and time-averaged RUSLE parameters; and to ensure that sheet and rill-related sediment was properly calculated. This procedure utilizes the time of concentration (Tc) that is determined from parameters from AGFLOW and TOPAGNPS. Additionally, the instantaneous peak discharge of the runoff hydrograph is required for Tc and can easily be calculated using TR-55 (SCS, 1986) technology incorporated within AnnAGNPS.

Since RUSLE is used to calculate the amount of sheet and rill erosion and HUSLE is used to determine the delivery ratio for total sediment, the only factor remaining is to determine the particle-size distribution of the deposition in the field (Bingner et al, 2001b). This allows for the particle-size distribution of the sediment yield of the sheet and rill erosion to the receiving reach of the stream system.

The particle-size of the sediment deposited within the field is assumed to be proportional to the mass fall velocity of the individual particle-size classes. Since the density of both the large and small aggregates are noticeably less than the discrete particles of clay, silt, and sand, a product of the respective densities times its fall velocity is used to represent each particle-size class. This is called the deposition mass rate and has units of mass per length squared per time. The resulting deposition mass rate values for each particle-size class are summed and then normalized with respect to this sum. These normalized values are called deposition rate ratios. They are further normalized with respect to the smallest value, which will normally be clay, and are called the deposition ratio mass rate. From these calculations, the field deposition is determined, but careful consideration is given to exhausting any of the particular particle-size classes; i.e., when any of the particle-size classes are totally deposited, the calculations begin again at that point along the landscape with that particle-size class eliminated from further calculations. A modified Einstein equation is used to transport the sediment in the stream system and uses the Bagnold equation (Bagnold, 1966) to determine the sediment transport capacity of the flow (Theurer and Cronshey, 1998).

The soil moisture, nutrients, and pesticides are also tracked within each field and subsequent movement downstream. Soil databases developed by the NRCS are used to describe each cell or field. Crop information developed for RUSLE is also needed by AnnAGNPS, along with additional parameters that describe how the crop uses nutrients from the soil.

From any point in the watershed, any loadings that are produced from upstream can be determined along with the location that they originated. This can be used to provide source accounting information to planners to assess the impact of various management practices downstream. This can be used in the development of management plans to meet the needs of total maximum daily load programs (TMDLs) that states are having to formulate to meet EPA guidelines for the 1972 Clean Water Act.

CONCEPTS

CONCEPTS is a batch-process, continuous-simulation, instream-processes module that simulates unsteady, one-dimensional flow, graded-sediment transport, bank erosion processes, and pollutant transport in watershed channels. AnnAGNPS has a simple pollutant transport mechanism for stream channels, but lacks the ability to change the channel characteristics from storm event to storm event. CONCEPTS was designed to track the changes in the cross-sections of the channel, including: bed lowering or raising; bank widening from fluvial processes or from bank failures; and, channel particle size distribution changes resulting from sediment transport. Also a part of CONCEPTS is a stream corridor component that targets individual reaches for a detailed analysis of the channel segment (Langendoen, *In Press*). This provides options on the level of complexity to use in the simulations of pollutant transport in a stream system. For watersheds where channels are not a significant source of erosion or are fairly stable, then the stream channel component within AnnAGNPS would be adequate. When the channel is unstable or significant erosion is occuring or there are structures in the channel that would influence flow or sediment transport, then the stream network version of the CONCEPTS should be selected. If an analysis of a stream corridor is needed, then the stream corridor version of CONCEPTS should be selected. While AnnAGNPS is simple, there is not as much information required from the user as for CONCEPTS and does not require as much computer resources to analyze the watershed. Thus, the stream network version requires a more detailed description of the stream system than AnnAGNPS, while the stream corridor version requires an even higher level of description of the stream reach. The selection of a pollutant transport module offers flexibility in analyzing a watershed when lower or higher levels of detail used in simulating the processes are needed.

SNTEMP

A stream network water temperature component is included as part of AGNPS 98 to evaluate the effect of riparian vegetation (shading), water withdrawals, irrigation return flows, reservoir release operations, and changes in channel geometry on the instream water temperature (Theurer et al, 1984). The stream network temperature model (SNTEMP) predicts the minimum, maximum, & average daily water temperatures throughout a stream network. The input requirements to SNTEMP include the daily water discharge that is calculated and produced as output by AnnAGNPS. Most of the remaining SNTEMP input is duplicated in the input to AnnAGNPS. Obviously, the same climate data & stream network system, including the hydraulic geometry, used for AnnAGNPS should also be used for SNTEMP. The only significant additional input to SNTEMP would be any local topographic influences & riparian vegetation (trees); i.e., physical dimensions that produce important shade on the stream' s water surface. SNTEMP is fully supported by the USGS Biological Resources Division including software, documentation, training, and even some specialized subsidiary computer programs.

SIDO

A set of salmonid models is included to be able to evaluate the entire life cycle of salmonids (salmon, trout, and graylings) and their economic impact especially in the Pacific Northwest (Alonso et al, 1996; Miller et al, 1998; Theurer et al, 1998). The sediment intrusion & dissolved oxygen (SIDO) model predicts the daily sediment accumulation & DO status within a salmonid redd with special attention to the egg zone. In addition to local detailed hydraulic geometry for the redd and certain biological specifications particular to the specific salmonid species, SIDO requires daily water, sediment, and water temperature data that is output from AnnAGNPS & SNTEMP.

Climate Parameter Generation

GEM (climate generator) is a program that generates synthetic climatic data for locations in the United States. GEM generates daily precipitation, maximum and minimum temperature, and solar radiation. AnnAGNPS requires six climatic elements for each day which are precipitation, maximum and minimum temperature, sky cover, average daily dew point temperature, and average daily wind speed. An interim program named Complete_Climate was written to generate the missing climate elements and format the climatic data for input to AnnAGNPS. Input to Complete_Climate includes the GEM parameters generated previously and information on the monthly average sky cover, dew point temperature, and wind speed. These monthly averages are available from a climatic data atlas or climatic summary for the desired location. The output file of Complete_Climate then contains all six climatic elements (three generated by GEM and three generated by Complete_Climate).

AGNPS 98 SUMMARY

AGNPS 98 can be used to evaluate the long term impact of non-point source pollution from agricultural watersheds. Effects of implementing various conservation management alternatives within the watershed can be evaluated. The loadings predicted are: (1) water; (2) sediment by particle-size class & source of erosion; and (3) chemicals–nitrogen, phosphorus, organic carbon, & pesticides. Pollutant loadings are generated from land areas and routed through the stream system on a daily basis. Special land use components such as feedlots (nutrients), gullies (sediment and chemicals), and point sources (water and nutrients) are included.

The following is a summary of some of the more-important features included in AGNPS 98:

- Loading, transport, and tracking of pollutants from their source to the outlet of a watershed system.
- Nutrient concentrations from feedlots and other point sources can be simulated.
- Individual feedlot potential ratings can be derived.
- Inclusion of CONCEPTS.

- Inclusion of instream water temperature models for ungaged stream system networks and for additional special applications such as reservoir operations.
- A weather generator to provide long term climatic information from any location in the country.
- A graphical program allowing the automatic determination of land area boundaries & their hydrologic parameters such as the RUSLE LS-factors, flow routing sequence, and channel hydrologic parameters. This program utilizes topographic analysis tools using readily available DEM' s.
- A graphical input data preparation editor facilitates data input and revisions.
- An AGNPS 5.0 to AGNPS 98 input data converter, allowing backward compatibility with previously developed databases for AGNPS 5.0 or with the USDA–NRCS HU/WQ interface.

AGNPS 98 is expected to run on any PC (386 or higher) under Windows 95, 98, NT, & 2000. Actual memory requirements are dependent upon number of cells selected. A practical minimum memory limit would be 2MB for user data entry and 32 MB for AnnAGNPS.

The following components are planned as enhancements to AGNPS 98:

- Integration of stream corridor buffer technology (Riparian Ecosystem Management Model-REMM)
- Wetland and lake water quality components.
- Integration of NEXRAD technology.
- Land–atmosphere exchanges needed for global climate change evaluations.
- Integration with ArcView.

GOODWIN CREEK WATERSHED

Goodwin Creek Watershed (GCW) is in the Yazoo River Basin near Oxford, Mississippi. GCW is 21.3 km^2 in area with fourteen instream measuring flumes monitored by the USDA-Agricultural Research Service, National Sedimentation Laboratory since 1982 (Blackmarr, 1995). Data collected from each measuring flume include channel discharge, fine (<0.062mm) sediment concentration, plus climatological data from a central weather station and 32 spatially distributed raingages. Each measuring flume defines an outlet of a subwatershed of various drainage areas. Many of the parameters required by AnnAGNPS have been obtained for GCW, plus selected measuring flumes can be used for model validation purposes.

Figure 1 is a watershed map of Goodwin Creek showing the location of the measuring flume at Station No. 1 selected for this analysis. Measured rainfall and temperature for the period of 1982 to 1991 was used by AnnAGNPS to simulate the runoff and sediment yield at Station No. 1. Landuse changes were also incorporated into the management operations of the model. The landuse of Goodwin for this period was generally 12% cropland, 70% pasture and 18% wooded. The average annual rainfall for this period was 1460 mm, which compares to the long term normal rainfall of 1400 mm.

The measured rainfall, runoff, and sediment yield at Station No. 1 are compared to the simulated runoff and fine material sediment yield in Table 1. Simulated runoff is generally 10% lower than measured runoff. Simulated fine material sediment yield was 33% of the measured values.

The simulations were performed without calibration and used SCS curve numbers suggested from the literature. Some adjustments to those values could have been performed to improve the results, but AnnAGNPS will generally not be used on gaged watersheds where calibrations would be possible. These results show that AnnAGNPS provides a reasonable estimate of runoff without a large amount of time spent by the user in selecting the parameters. The sediment yield results represent only the fine material at the outlet of Goodwin Creek watershed. The measured sediment yield represents the fine material from all sources, including gullies, channels, as well as from fields. The simulated results only represents the fine material that is coming from the fields and eventually gets to the outlet. The results from AnnAGNPS are reasonably close to those from Grissinger et al (1991) who estimated that 25% of the fine sediment material transported in the channels of Goodwin Creek were from upland sources and the rest were from channel or gully sources. AnnAGNPS uses the technology of RUSLE, which predicts the long term average values of erosion. Thus, comparisons with individual events or years will be very difficult. In the evaluation of any management practice, long term results are necessary to properly determine the effects of the weather on the function of the practice.

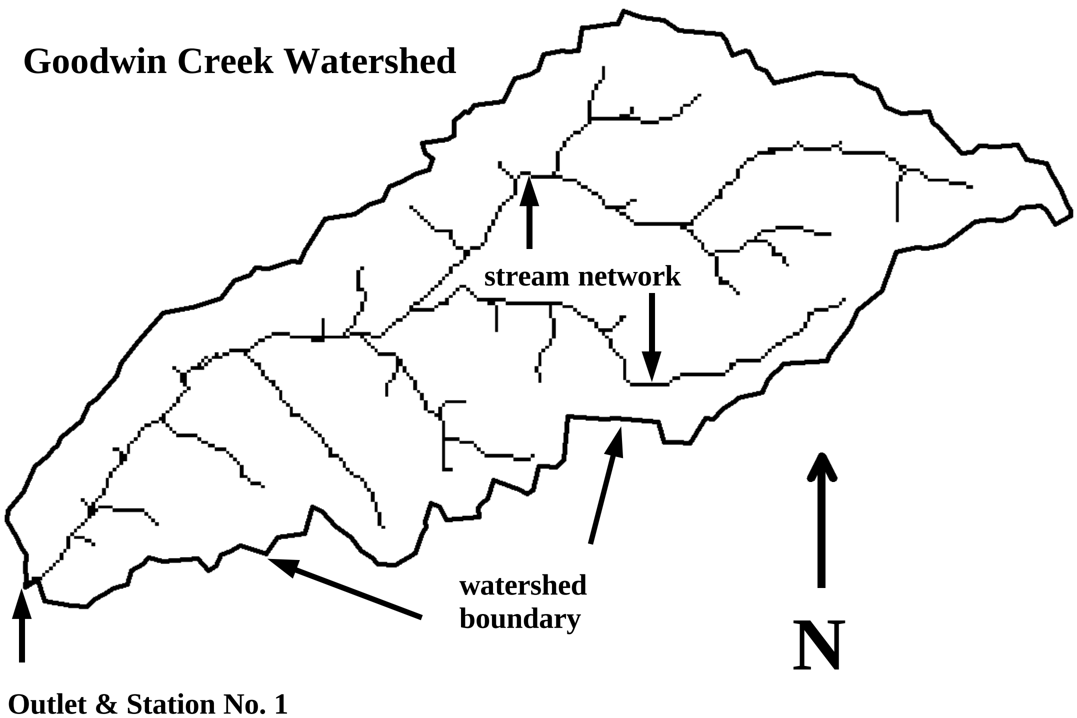

Figure 1: Goodwin Creek Watershed showing the location of Station No. 1 and the stream network and watershed boundary generated by TOPAGNPS from DEMs.

Table 1: Measured and simulated runoff and fine material sediment yield from Station No. 1 on Goodwin Creek Watershed from 1982-1991.

Year	Measured Rainfall (mm)	Measured Runoff (mm)	Simulated Runoff (mm)	Measured Sediment Yield (t/ha)	Simulated Sediment Yield (t/ha)
1982	1570	676	602	17.6	2.8
1983	1539	774	672	19.5	5.1
1984	1464	505	448	16.5	3.9
1985	1216	276	208	7.5	1.1
1986	1169	303	390	5.2	3.9
1987	1223	284	295	4.1	1.0
1988	1095	216	194	3.0	0.5
1989	1836	757	630	11.7	4.4
1990	1466	563	444	9.2	2.2
1991	2025	1021	903	12.6	9.8
Annual Average	1460	538	479	10.7	3.5

SUMMARY

AGNPS 98 is comprised of a suite of modules that can be used to predict the pollutant loadings within a watershed. This provides a powerful tool to perform watershed scale evaluations needed in recommending best management practices or setting TMDL pollutant allocations. In developing management plans to address TMDLs, states can utilize AnnAGNPS as science-based technology to meet their specific needs. Action agencies, such as NRCS, can utilize this technology on watersheds in conjunction with their application of RUSLE on individual fields by importing those databases to AnnAGNPS. Watersheds that have a significant source of pollutants originating from the channels can be evaluated using CONCEPTS. This provides a link between the simple watershed model and a more process-based channel routing model.

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SYSTEM EVALUATION OF RUNOFF USING ANNAGNPS FOR THE YALOBUSHA RIVER WATERSHED, MISSISSIPPI

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ABSTRACT

In the development of a TMDL for watershed systems, a linkage needed to produce a cause and effect pollutant loading relationship must be defined between a selected indicator and the identified sources. This relationship can vary seasonally with factors such as rainfall and farming practices. An integrated approach of utilizing simulation models in evaluating these climatic and man-made influences will enable researchers and action agencies to study and apply the most effective measures impacting TMDLs. The cost of studying a large watershed system requires that simulation models be used as a tool when trying to understand the complex relationships between many processes. All non-point pollutants are transported by means of surface or groundwater runoff from fields or channels. Thus, the first step in any TMDL evaluation is to determine the proper runoff from a watershed. Pollutant loadings transported or contained in the runoff can then be determined with greater confidence. AnnAGNPS was used for the development of a watershed analysis for runoff performed on the Yalobusha Watershed. This required the assembly of complete GIS layers for the watershed. From the GIS layers, individual subwatersheds were developed to provide spatial variability of land use, soils, topography, and identification of locations to analyze TMDLs used for the AnnAGNPS simulations. Simulations performed on the entire watershed system were used to evaluate the combined effects of upland management practices within fields on runoff, and eventually, sediment yield and water quality. Runoff volume and peak discharge can be determined at any point in the watershed by AnnAGNPS to provide stochastic analyses that are important in developing TMDL protocols.

USE OF INDIRECT FLUVIAL SEDIMENT MONITORING TO DEVELOP A TOTAL MAXIMUM DAILY LOAD FOR THE MIDDLE FORK PAYETTE RIVER SUBBASIN IN CENTRAL IDAHO

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Abstract: Direct fluvial sediment monitoring used in developing Total Maximum Daily Load (TMDL) for stream segments impaired by "clean" sediment is resource and time intensive. Our study examines the use of streambed sediment facies mapping to characterize and quantify sand storage (material < 8 mm) as an alternative to direct fluvial sediment monitoring. We quantified in-channel sand stored in two study reaches the Middle Fork Payette River, central Idaho, and compared the storage results with results from two years of direct monitoring. The river's streambed is characterized by sand dunes (d_{50} = 1.5 mm) moving over a coarser, armored layer (d_{50} = 57.0 mm). We delineated streambed facies by the percentage of surficial sand coverage and relative mobility of coarse fraction and determined sand volumes using a stratified, systematic sampling grid. The upstream and downstream reaches hold 0.20 and 0.62 ft^3/ft^2, respectively, or 259 and 898 yd^3 for equivalent 500 ft reaches. Presuming that the upper reach has the desired streambed conditions, then the sand stored in the lower reach needs to be reduced by 68%. Direct measurement of bedload transport yielded a required 68% reduction in supply at the lower reach. These remarkably identical results, given the error involved in sediment budget calculations, suggests that sediment facies mapping is a viable and cost effective alternative to direct monitoring for characterization and quantification of sand storage. Additional benefits of streambed facies mapping include increased sediment budget resolution, greater knowledge about available fish habitat, and increased predictive bedload transport equation accuracy.

INTRODUCTION

Idaho has 940 surface water bodies listed on Idaho's 303(d) list for water quality impairment due to excess fine-grained sediment, elevated temperatures, and/or elevated concentrations of chemical contaminants. Of the 940 surface water bodies, approximately 800 are streams impaired by excess sediment. Of the 800 streams listed for excess sediment, approximately 320 flow through basins underlain by granitic bedrock of the Idaho Batholith. The Idaho Batholith granitic bedrock decomposes into fine-grained sediment that ranges from silt to fine gravel in size. Excess sand impairs in-stream water quality for fish food, habitat, and spawning grounds by reducing gravel quality.

Each water body on Idaho's 303 (d) list requires that a Total Maximum Daily Load (TMDL) be established. Development of a sediment TMDL requires knowledge about the source, transport, and storage of suspended and bedload sediment. Such knowledge is typically obtained through expensive, time-consuming, and long-term sampling programs that are beyond the financial and resource limits of the TMDL authors. Bedload sampling is a particular problem because automated sampling techniques are not well developed. Thus, there is a need for a quick and accurate method to assess the bedload condition of a stream without long-term monitoring.

In this study we evaluate measuring sediment storage as an alternative long-term bedload transport monitoring. Specifically we compare a sediment budget derived from two years of sediment monitoring at upper and lower reaches along the Middle Fork Payette River (MFPR) to the quantity of sand (particles < 8 mm) stored at the same reaches. The upper reach (MFLT) is assumed to have the desirable water quality based on the quantity of surficial gravels and cobbles in the streambed and the presence of good fish habitat, a pool. The lower reach (MFDG) is considered the impaired reach because it is inundated with sand, covering the surficial gravels and cobbles and filling in pools. Based on the similar channel geometry at both reaches, we assume that MFDG reach's streambed should have similar quantities of sand and gravels as the MFLT reach. Fitzgerald (this volume) reports that bedload transport rates (97% of the bedload material is below 8 mm) needs to decrease by 68% per unit width for the MFDG to produce similar particle size distributions as the MFLT reach. The purpose of this study is to determine if mapping sand storage can produce the same result as sediment monitoring. To evaluate these methods, we conducted the following tasks:

1. Quantification of the sand stored at MFLT and MFDG reaches, and
2. Comparison of the two methods for determining TMDL targets.

Characterization of the lower MFPR's hydrology and sediment transport regimes is presented by Fitzgerald et al. this volume.

BACKGROUND

Sediment erosion within a basin involves sediment transport off the hillslopes, through the fluvial system, and out of the basin. While in the fluvial system, sediment is either being transported or in storage. Kelsey (1987) classified sediment storage into active, semiactive, inactive, or stable zones based on their frequency of mobilization. The actively stored sediment exists within, or juxtaposed to, the active river channel and is mobilized every 1 to 5 years. In gravel-bed streams, storage features characteristic of actively stored sediment include bars and pools. Sediment storage within these features varies temporally with sediment stored in a channel' s deeper sections being mobilized more often than shallowly stored sediment (Wathen and Hoey 1998). Bars form from hydraulic changes, non-fluvial effects, or excess supply of sediment (Church and Jones 1982). The latter represents the primary storage mechanism for bedload in gravel-bed rivers and sediment stored within them has a shorter residence time. Pools store sediment during low water discharges but scour during high water period (Gomez 1991).

Site Background: The MFPR basin is located in central Idaho, approximately 35 mi north of Boise, Idaho (Fitzgerald et al. this volume, Figure 1). The MFPR basin covers approximately 340 mi^2 and ranges in elevation from 8,696 ft at Rice Peak to 2,992 ft at the confluence with South Fork Payette River (SFPR). Valley profiles change from V-shaped with steep down-valley gradients and steep walls in the upper basin to a wide floor and gentler sloped walls near the confluence with the SFPR. The basin' s highly erodible soils are weathering products from the underlying Cretaceous granitic rocks of the Idaho Batholith. Average monthly temperatures are greatest in July and lowest in January and the wettest months are December through February with the average annual precipitation at Garden Valley Ranger Station and Deadwood Summit being 24.5 in. and 45.4 in., respectively. Snowfall ranges from 57.0 in. at Garden Valley Ranger Station and 186.5 in. at the Deadwood Dam station.

Distinct peaks in the MFPR' s annual hydrograph are caused by snowmelt, rain-on-snow events, and thunderstorms Snowmelt occurs from early May through late June. Rain-on-snow events typically occur from December through April and usually initiate at elevations below 6,500 ft (IDEQ 1998). Seventy seven percent of the MFPR basin at the confluence is below 6,500 ft. Thunderstorms occur as localized events from June through early September, but cause minor changes in the mainstem MFPR' s hydrograph due to their localized nature.

The study reach extends 12.4 mi. along the MFPR from the upper reach, located at Lightning Bridge (MFLT), to lower reach, located at the confluence with the SFPR (MFDG). At the MFLT and MFDG reaches, the MFPR drains 208 $mi.^2$ and 338 $mi.^2$, respectively. The average gradient and sinuosity is 0.0012 ft/ft and 1.56, respectively, and the valley is underlain by glacial outwash plain alluvium. Our study followed a 1997 rain-on-snow event that induced abnormally high quantities of debris flows within the MFPR basin. These debris flows delivered large quantities of sediment into the MFPR' s fluvial system. The MFPR at MFLT adjusted to the sediment pulse by transporting the sand fraction of the bed material. The MFLT and MFDG reaches are "in" and "out of" equilibrium, respectively, based each site' s dominant discharge,bedload rating curves, and quantity of sand stored (Fitzgerald et al. this volume).

METHODS

To quantify sand storage and assess sediment facies mapping as a potential tool for evaluating stream quality, we quantified the stored sand at the MFLT and MFDG reaches and compared the results to the two years of sediment monitoring results (Fitzgerald et al. this volume). We quantified the sand storage within the channel bankfull stage boundary for a 550 and 600 ft reach (MFLT and MFDG respectively). We delineated five streambed facies using the surficial sand abundance, coarse fraction' s relative mobility, and presence of vegetation. We defined "sand" as particles less than 8 mm along the b-axis based on the upper particle size limit for salmonid fry emergence (Kondolf 2000), and it was easily identifiable from the coarse bed material by its lighter color. In addition, material less than 8 mm represents 98 and 97% of the bedload samples' and dune material, respectively, suggesting the material less than 8 mm represents the majority of the lower MFPR' s bedload (Fitzgerald et al. this volume).

We determined the coarse fraction mobility based on the coarse particle' s packing, algal growth on the particle' s surface, and streambed firmness under foot. Visually, a distinct difference exists between the immobile armored layer and recently mobilized gravels and cobbles. The armored layer exhibits a firm bed, darker particles from aquatic plant growth, and matrix supported packing of the coarse particles. Characteristics of mobilized coarse fraction include sand supported matrix, loose under foot, and lighter colored particles. We found no facies comprised of exclusively mobilized coarse particles.

Facies Delineation Characteristics:

- *Dune* – streambed surface consisted of greater than 90% surficial sand coverage (D_{50} is approximately 1.4 mm). The Dune facies is usually loose under foot and appears lighter in color than coarse layer. If coarse material is present, it usually is associated with the largest particle in the underlying armored layer.
- *Dune Over Armored Layer (DOAL)* - streambed surface consisted of between 30-90% surficial sand coverage (D_{50} is approximately 1.4 mm) overlying an armored substrate of coarse-gravels to boulders (D_{50} approximately 40-85 mm). The sand is lighter in color and loose under foot while the armored layer is darker in color, has algal growth on it, and is firmly packed. We chose the lower threshold, 30% sand surficial coverage, based on the threshold where sand coverage appeared to change from interstitial to dune overriding the armored layer. We chose the upper threshold, 90% surficial sand coverage, because the armored layer usually consisted of a lone cobble or boulder sticking up through the sand with sufficiently deep sand surrounding it to be considered the Dune facies.
- *Armored* - streambed surface consisted of less than 30% surficial sand coverage laid overtop an armored layer of coarse-gravels to boulders (D_{50} approximately 40-85 mm). The Armored facies is characteristically darker in color, has algal growth on it, and is firmly packed.
- *Bar* - streambed surface consisted of less than 90% surficial sand coverage that is mixed with mobile coarse particles. The coarse particles are matrix supported and comprise 10-46%, averaging 25%, of this facies and have a D_{50} of approximately 32 mm.
- *Vegetation* – streambed surface consists of well vegetated areas with riparian grasses or willows. This facies represents regions lining the banks and established bars within the study area covered in vegetation.

Field Efforts: Field efforts included surveying facies boundaries, measuring sand depths through cross-sectional transects, measuring percentage of sand within the Bar facies by coring, and excavating pits to determine material underlying the Dune facies. We quantified sand stored in Dune, DOAL, Armored, and Bar facies, but not in the Vegetated facies because of its limited spatial extent. Additional field efforts included preliminary sampling to establish sampling density within the Dune facies, taking 30 cores at two Bar facies locations to establish particle size variability within this facies, and collecting data to derive the DOAL and Armored facies equation to calculate sand volume from the percentage surficial sand coverage.

To quantify the sand stored in the Dune, DOAL, and Armored facies, we surveyed facies boundaries and sampled sand quantities along transects established every 30 ft longitudinally. We surveyed facies boundaries to within $\pm$ 1 ft using a Topcon GTS-213 total station. Along each transect, we sampled depths every 1.5 ft for the Dune facies and percentage surficial sand coverage every 3.0 ft for the DOAL and Armored facies. We determined this sampling distribution using a semivariogram analysis of preliminary sampling at MFDG and MFLT. The sampling density chosen captures the majority of variation within 0.25 ft. However, we increased longitudinal sampling density with increasing heterogeneity in streambed facies.

At each sampling point along a transect in the Dune facies, we measured the sand depth by thrusting a graduated, acuminated rod until it hit the underlying coarse layer and recorded the depth. To confirm the underlying coarse layer' s presence at each site, we excavated at least one pit through the sand layer to the sand and armored layer interface. We also used these pits to verify the homogeneity of the particle size distribution throughout the sand column.

To determine the quantity of sand stored in the DOAL and Armored facies at each point along a transect, we noted the percentage surficial sand coverage and the D_{90} particle size representative of the 3 ft^2 sampling area surrounding the sampling point. We then entered the percentage surficial sand coverage into equations derived to convert the percentage surficial sand coverage to sand volume. We derived the equation by incrementally filling a 55-gallon barrel, cut in half endwise, with sand and noting the percentage of sand coverage. The test involved randomly selecting an area of coarse armored layer with less than 5% sand coverage, placing the barrel on the streambed, filling the barrel with 500 ml increments of sand, and noting the percentage of surficial sand coverage after each incremental addition. We placed the barrel such that no gaps existed between the barrel bottom and armored layer interface and that the armored layer was not disturbed. To consistently assess the percentage of covered sand, we used a mesh grid with 100 counts. We derived the equation using the data from nine tests conducted at three reaches along the river between MFDG and MFLT. Similar pebble counts of the coarse armored layer at seven reaches along the lower MFPR suggest these equations are applicable throughout the 12.4 mi. study reach.

To determine the sand present in each Bar facies, we used a 6-inch McNeil sampler to collect five spatially random cores, sieved each sample, and measured each fraction using volumetric displacement to determine its particle size distribution. The sieve sizes include 76.2, 38.1, 19.1, 9.7, 6.3, 2.0, and <2.0 mm (–6.3, -5.3, -4.3, -2.7, -1.0, and <1.0 ϕ, respectively). Use of the graduated, acuminated rod to measure sand depths was not practical in this facies due to the coarse particles suspended in the sand matrix. We determined the sample variance within the Bar facies by collecting 30 cores at two sites and checked analytical quality by reprocessing 12 % of the samples.

Office Analysis: We calculated facies spatial extent and sand volumes using Arcview 3.1. Calculating facies spatial extent involved warping a low-level aerial photograph, taken from 1,000 ft altitude, to survey data and then digitizing the facies boundaries from the survey data, site maps, and the aerial photograph. For the Dune, DOAL, and Armor facies, we calculated each facies' sand volumes by determining the area each transect represented of that facies, the average sand depth over that transect, and then summing the sand depths for all transects. For the Bar facies, we determined the sand volume by multiplying each sample's representative aerial extent by the percentage of sand and 6 inches depth, the McNeil sampler coring depth, and summing the volumes of each sample. We determined the representative area of each sample using the Theissen polygon method.

RESULTS

MFDG: The MFDG reach covers 46,910 ft^2 over a 600 ft reach and stores 28,921 ft^3 of sand (Table 1, Figure 1A). The Dune facies comprising nearly 60% of the streambed with the DOAL and Armored facies comprising 29.5 and 11.1%, respectively. Of the total sand stored within the reach, the Dune facies contains 97.2% of the sand with the DOAL and Armored facies comprising remaining 2.5 and 0.3%, respectively. The Dune facies covers the streambed throughout most of the low gradient section of the MFDG reach with an average and maximum sand depth of 1.01 and 3.56 ft, respectively. During low water discharges, this reach becomes shallow with a level sand surface that is poor fish habitat. Coarser material is exposed in the thalweg, along the right waters edge in transects 1 - 11, and in the higher gradient sections, transects 17 - 21. The thalweg and high gradient sections partially fill with sand during low water discharge periods. Without the excess sand, this reach would have pools 3-4 ft deep during low water discharges. There was no Bar or Vegetation facies mapped at this site.

Table 1 – Facies and storage statistics for the MFDG reach

	Facies Area		Sand Storage		
Facies	Area (ft^2)	% Reach	Stored Sand (ft^3)	% Reach	Ave. Depth (ft)
Dune	27,845	59.4	28,103	97.2	1.01
DOAL	13,815	29.5	728	2.5	0.05
Armor	5,250	11.1	91	0.3	0.02
Bar	0	0.0	0	0.0	0.00
Vegetation	0	0.0	n/a	n/a	n/a
Total	46,910		28,921		

MFLT: The MFLT reach covers 38,489 ft^2 over a 550 ft reach and stores 7,577 ft^3 of sand (Table 2, Figure 1B). The DOAL, Dune, and Armored facies comprise 49.0, 29.5, and 16.8% of the MFLT reach's aerial coverage, respectively, with Bar and Vegetation facies comprising the remaining 4.7%. Like the MFDG reach, the sand is primarily stored in the Dune facies, 85.6%, with lesser amounts stored in the DOAL, Bar, and Armored facies. While the DOAL facies covers 49.0% of the study reach, it only accounts for 10.0% of the stored sand. In the MFLT reach, sand is primarily stored in the Dune covers the middle of the low gradient sections of the reach with an average sand depth of 0.57 ft. The thalweg, running down the right water edge from transect 8 - 20, contains little sand and is underlain by the DOAL and Armored Facies. Transects 15 - 20 cross a 4.25 ft deep pool along the right waters edge that provides habitat for several resident native suckers and small trout. The Dune facies, in transects 14 -16, forms the upstream boundary of this pool.

Table 2 – Facies and storage statistics for the MFLT reach

	Facies Area		Sand Storage		
Facies	Area (ft^2)	% Reach	Stored Sand (ft^3)	% Reach	Ave. Depth (ft)
Dune	11,360	29.5	6,484	85.6	0.57
DOAL	18,853	49.0	759	10.0	0.04
Armor	6,468	16.8	53	0.7	0.01
Bar	338	0.9	281	3.7	0.83
Vegetation	1,470	3.8	n/a	n/a	n/a
Total	38,489		7,577		

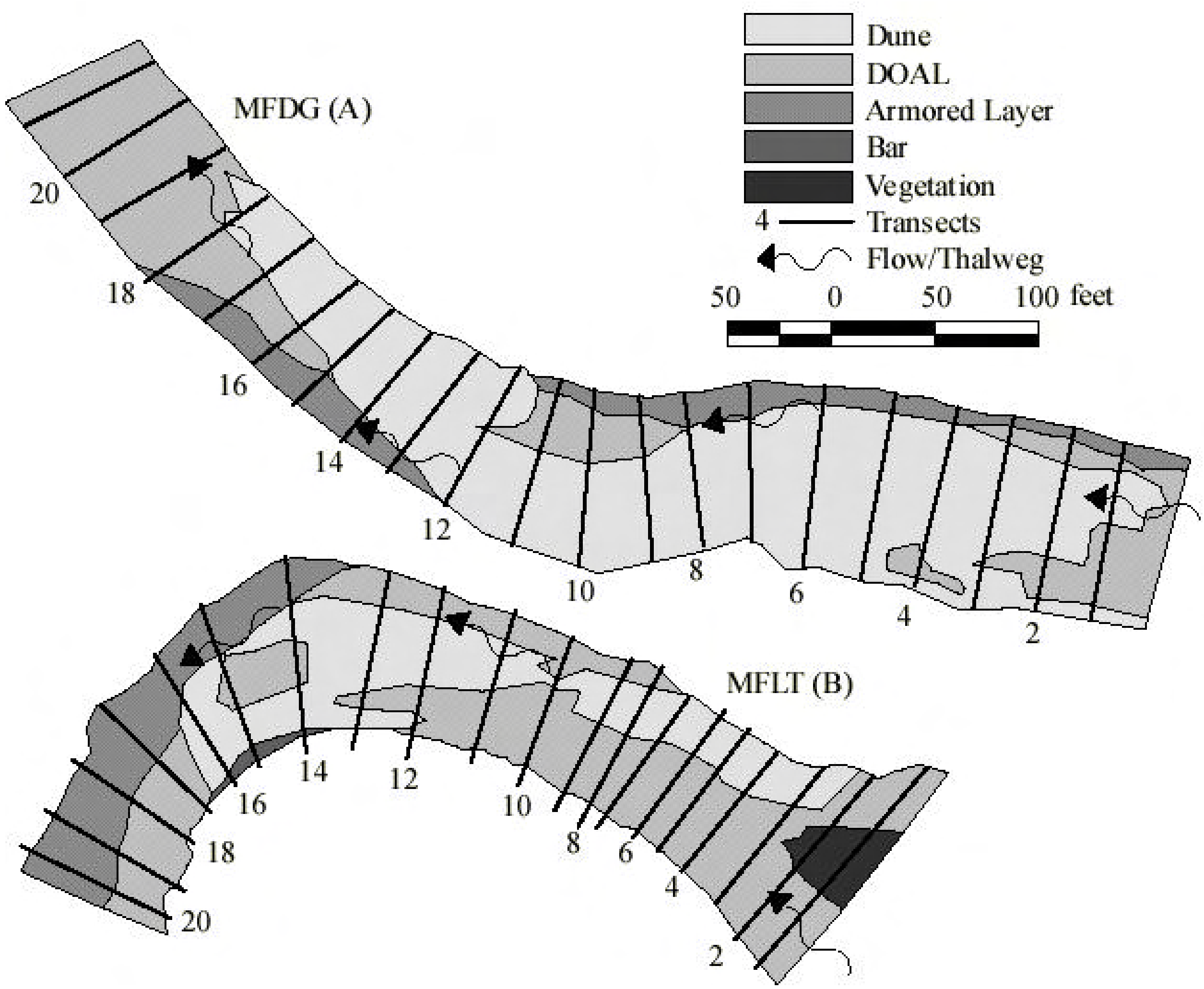

Figure 1 – Surficial facies maps and sampling transects of the MFDG (A) and MFLT (B) reaches.

__DOAL and Armored Facies Sand Volume Equations__: We derived two equations for determining the sand volume from the percentage surficial sand coverage in the DOAL and Armored facies (Figure 2). The equations are:

$''_s = 9.0*10^{-6}A_s^2 + 0.0003A_s$	$r^2 = 0.958$	0 – 75 % sand	Eq. 1
$''_s = 0.0019A_s - 0.0581$	$r^2 = 0.346$	75 – 90 % sand	Eq. 2

where "$_s$ is the sand volume per square foot (ft^3/ft^2) and A_s is the percentage of surficial sand coverage. For surficial sand coverage less than 75%, a r^2 value of 0.958 demonstrates a strong relationship between sand volume and percent sand coverage. We chose a second order polynomial due to slight parabolic nature of the data. Fitting a linear trend to the data slightly overestimated volumes from 30-55% and rendered a r^2 value of 0.895. Because we derived this equation from samples collected throughout the study reach, the high r^2 value is a function of the consistent coarse layer' s particle size distribution and packing the particle size. For surficial sand coverage greater than 75%, we used a linear equation to predict sand volume from percentage surficial sand coverage that had a r^2 value of 0.346. No strong correlation existed between largest particle present and volume of surficial sand stored.

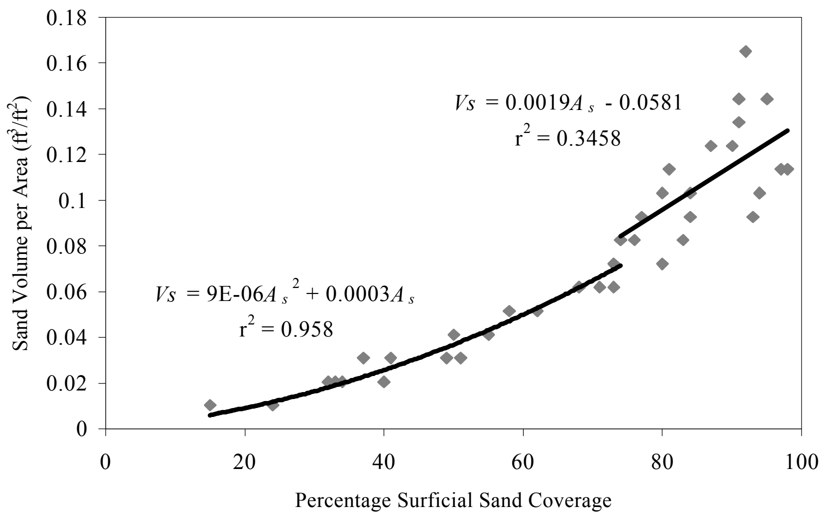

Figure 2 – Data from the nine test samples used to generate the equations relating sand volume to the percentage of surficial sand coverage for the DOAL and Armored facies.

We used two equations to calculate sand volume from percentage surficial sand coverage due to a fundamental change that occurs in the relationship around 75% surficial sand coverage (Figure 2). Below 75% surficial sand coverage, sand uniformly covers the coarse particles until only the largest particle is exposed. Above 75% surficial sand coverage, quantity of sand necessary for changing the surficial sand coverage depends strictly on the largest particle' s size, shape, orientation, and embeddedness. For example, if two identical particles are exposed at the surface, one with the a-axis and the other with the c-axis extending upward, then the former would require more sand to cover it due to its vertical protrusion above the streambed. We did not investigate using a particle' s size, shape, orientation, and embeddedness to better predict sand volume stored, however we hypothesize that measuring a particle' s protrusion above the streambed would provide a better relationship for the 75 – 90% surficial sand coverage range.

We made three assumptions in deriving these equations. First, the particle size distribution and packing remains similar throughout the study reach. Similar results of pebble counts of the coarse, armored layer at each reach and a r^2 value of 0.958 in the 0-70% surficial sand coverage suggest this assumption is valid. Second, the sand in an area is evenly distributed between coarse material and does not account for influences of water discharge in higher energy environments that would permit hiding. Third, the suspended fraction of the bed material load comprises negligible volume and therefore its loss during the incremental filling of the barrel is not important.

DISCUSSION

Fitzgerald et al. (this volume) showed that MFDG transports 68% more sand than MFLT per unit area and concluded that transport at MFDG must therefore be reduced by 68% to obtain the similar streambed conditions at MFLT. The sediment mapping presented here shows that MFDG has 68% more sand per unit area than MFLT. The answer to the objective stated in the introduction is therefore yes, sediment mapping produces the same result as sediment monitoring. The next question that must be asked is this: is it real or is it a coincidence? If bedload transport of sand is supply limited, then the stream by definition transports all of the available sand. Thus, if the amount of available sand is doubled, the amount of transport will be doubled. This is not true if bedload transport of sand is transport limited. Scour chains show that during high water discharges, the streambed surface scoured to the armored layer at MFDG in the channel. Since all of the available sand was transported, bedload transport of sand is therefore supply limited, and the matching results between the sediment mapping and sediment monitoring is real.

At both reaches, the Dune facies stores the majority of the sand. The Dune facies' average depth at MFDG is 0.45 ft greater. This excess sand influences the pool habitat at the reaches. At MFLT, sand quantities are insufficient to fill in the pool while at MFDG, sand depths indicate a 3-4 ft pool would exist in the sand's absence. Also, the excess sand shelters the sediment from experiencing shear stress, so mobilization of gravels for spawning is not likely.

The DOAL facies covered 29.5 and 49.0% of the MFDG and MFLT, respectively. Neither reach stored more than 10% of the sand in this facies but the MFLT reach had greater spatial extent and existed in lower gradient reaches. The average percentage surficial sand coverage between the two reaches is different to within a 99% confidence interval and the MFDG reach's average is 14.3% greater. The Armored facies at both sites covered less than 17% of the area, stored less than 3% of the sites' sand, had average sand depths of less than 0.02 ft, and primarily existed in the thalweg. At both reaches, the Vegetation and Bar facies represent insignificant spatial extent and sand storage.

The sediment facies mapping took 10 and 8 man days for the field work and office analysis, respectively, versus two years of sediment monitoring, sampling on a monthly and event basis, that used 32 and 16 man days for the field work and office analysis, respectively. Additional benefits of using sediment facies mapping includes that it is performed during low water discharges, requires minimal equipment, can be used to evaluate available habitat, and need only be performed once.

There are several limitations of using sediment facies mapping to assess reductions needed to set TMDL limits. First, this method only provides a snapshot in time so there it provides little information on time required to flush sand from the system or seasonal variation of sand deposition throughout the year. However, coupling the sediment mapping with bedload transport equation or conducting multiple sampling events may provide a means of answering these shortcomings. Second, because sampling needs to be conducted during wadable water discharges, it can not be used to evaluate sediment deposition during unwadable water discharges. For example, if one was interested in rainbow trout habitat for spawning, which occurs during spring runoff in the Rocky Mountains, then this would be difficult, if not impossible, to perform on most streams. Third, the practitioners must have some knowledge of the particle size mobility prior to sampling in order to establish facies based on relative transport of particular size fractions. Fourth, to assess target streambed conditions this method relies on comparing a desired reach and impaired reach to assess how much sediment storage needs to be decreased in the impaired reach. This requires having a reach along the same river with desired water quality conditions and similar geomorphic conditions or a reference reach in another drainage with similar physical, climatical, and hydrological conditions. Finally, sediment transport at both sites needs to be supply limited in order for sediment storage and transport rate correlation to be valid.

CONCLUSION

Sediment facies mapping of sand storage within two reaches of the lower MFPR show that the MFLT and MFDG reaches store 7,577 ft^3 over a 550 ft reach and 28,921 ft^3 over 600 ft reach, respectively. Sand is primarily stored in the Dune facies at both reaches. The identical results from the sediment facies mapping and sediment budget indicate that sediment facies mapping can be used as a surrogate to monitoring sediment transport saving time and resources.

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USE OF DIRECT FLUVIAL SEDIMENT MONITORING TO DEVELOP A TOTAL MAXIMUM DAILY LOAD FOR THE MIDDLE FORK PAYETTE RIVER SUBBASIN IN CENTRAL IDAHO

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Abstract: The tools commonly available to develop non-point source Total Maximum Daily Loads for instream "clean" sediments are direct and indirect fluvial sediment monitoring. We use two years of direct fluvial sediment monitoring to evaluate the type and quantity of sediment impairing the salmonid rearing uses of the Middle Fork Payette River Subbasin in central Idaho. Suspended sediment and bed-material data taken on a monthly and high flow event basis show that bedload was the dominant type of sediment transport in this watershed. The measured bedload accounted for about 80% of the total measured load during the study period. For all the sites, the bedload D_{50} = 1.5 mm. The study reaches tend to have year round sand transport via dunes over an armored very coarse gravel (D_{50} = 57 mm) substrate. For the lower mainstem-site, the average measured bedload flux was 179 tons/day with a minimum of 33 and a maximum of 837. For the upper mainstem-site, which is about 12 miles upstream of the lower site, the average measured bedload flux is 57 tons/day with a minimum of 0 and maximum of 616. We presume that the upper site is representative of the desired water quality condition and infer that the bed-material supply needs to be reduced about 68%. To help account for annual variability of the bed-material load and provide and effective monitoring tool, the sediment rating curves are used and indicate a 63 % reduction in slope. We show that direct fluvial sediment monitoring techniques were an effective way of describing the type and quantifying the relative amount of sediment contributing to poor water quality. Direct fluvial sediment monitoring, however, was resource intensive.

INTRODUCTION

Background: The Sediment Total Maximum Daily Load (TMDL) protocols allow the analyst flexibility to use the appropriate monitoring and analytical techniques given the watershed characteristics, time, and budget (EPA 1999). This is especially important for non-point source TMDLs where traditional water quality monitoring and analytical techniques are not adequate to set pollutant load reductions from dispersed land uses. In Idaho, "clean" sediment produced from public and private lands used for agriculture, forestry, mining, and urban development is the most common pollutant impacting salmonid species and placing streams and lakes on the State 303(d) list.

We monitored stream and sediment discharge of the Middle Fork Payette River (MFPR) from March 1998 to November 1999. The MFPR typified many streams listed on Idaho's 303 (d) list for sediment in that, prior to this study, its hydrology, channel morphology, and sediment transport had never been monitored. Our objectives were to:

1. Characterize the stream flow hydrology of the lower MFPR,
2. Quantify the type of suspended and bed-material sediments,
3. Quantify the amount of sediment transport; and
4. Set TMDL targets and sediment load reductions for stream reaches impaired as a result of "clean" sediments.

Site Description: The MFPR basin is located in central Idaho, approximately 35 miles north of Boise, Idaho (Figure 1). The study reach examines the lower 12 miles of the MPFR's channel, extending from the Lightning Road Bridge - downstream to the confluence with the South Fork Payette River (SFPR). In addition, a site was established on Lightning Creek, approximately 900 ft upstream from its confluence with the MFPR, for hydrologic and sediment characterization. We monitored an upper site (MFLT), a lower site (MFDG), and several tributaries.

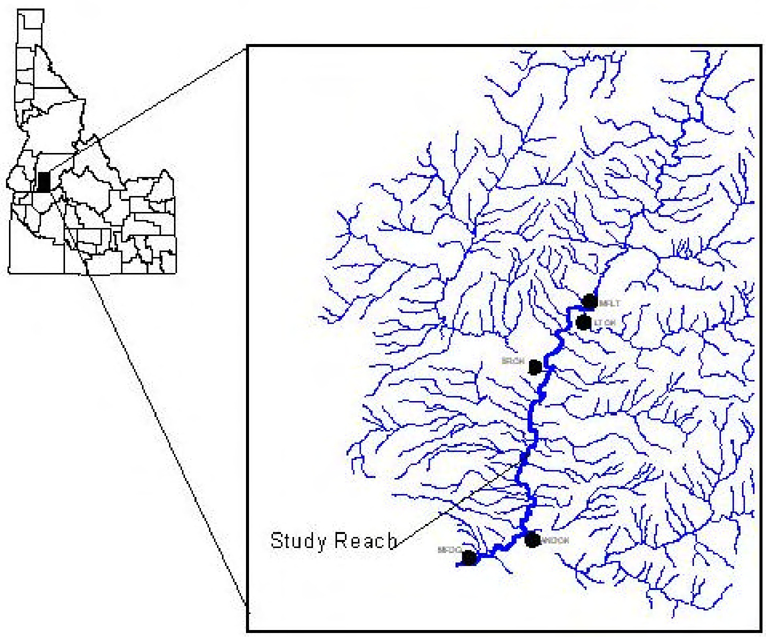

Figure 1. Site location map.

METHODS

Characterization of the lower MFPR's hydrology, channel morphology, and sediment transport rate involved fieldwork, laboratory measurements, and office analysis. Fieldwork included measuring water and sediment discharge, monitoring stage using outside staff gauges and pressure transducers, establishing and monitoring channel reference reaches, and deploying of scour chains. Laboratory analysis included measuring total suspended solids (TSS) and drying, sieving, and weighing bedload samples. Office analysis included developing stage-water discharge rating curves, bedload transport rating curves, and water and sediment budgets and determining dominant discharge and changes in channel geometry.

Field Techniques:

Sediment discharge: We measured sediment discharge directly at low and high flow using USGS protocols (Edwards and Glysson 1998). For suspended sediment, we collected depth integrated suspended sediment samples using the Equal Width Increment sampling technique and measured bedload using two passes of 20 cross-section points where each point was sampled for 30 seconds using a Helley Smith bedload sampler.

Laboratory Techniques:

Total suspended sediment: Suspended sediment samples were analyzed using the Total Suspended Solids (TSS) analytical technique at the USEPA Manchestor Laboratory. This analytical technique differs from the USGS suspended sediment technique and sometimes produces different results (Glysson et al. 2000). To quantify the differences, we ran duplicate samples using both methods on about 5 % of the samples and found substantial differences between the samples collected at high flow.

Bed-material: Grain size analyses were conducted at Boise State University's soil laboratory using standard techniques. Following these methods, the samples were dried, sieved, and weighed. The sieve sizes included -4.9, -4.3, -3.2, -2.2, -1.0, -0.5, 0.2, 1.2, and <1.2 phi. The results of the sieve analysis were plotted using cumulative grain size plots and the d_{16}, d_{50}, and d_{84} statistical breaks were used to describe the particle size distribution of bed-material samples.

Office Analysis: To generate sediment budget, we analyzed the discharge and sediment transport data collected during the study period to create hydrographs and sediment rating curves. Annual hydrographs developed for upper and lower sites were developed by either calculating water discharge from average daily stage, extending the discharge record using maintenance of variance extension type 1 (MOVE1) analysis (Hirsch 1982; Moog, Whiting et al. 1999), or regressing concurrent water discharge measurements between reaches.

Reach specific sediment rating curves were generated by regressing sediment and water discharge data from a reach. Sediment fluxes at upper and lower sites were determined by applying the respective sediment rating curve to reach's hydrograph to determine the quantity transported for the year. To calculate the study reach's sediment budget, the incoming sources and the lower site was the outgoing source to the control volume. Stated mathematically:

$$\Delta S = Qs_{MFLT} + Qs_{LTCK} + Qs_{SCRCK} + Qs_{ANDCK} - Qs_{MFDG} + \boldsymbol{e}$$

where ΔS is the change in storage, Qs_{XXX} is tons of sediment over the year at a particular reach, and $\boldsymbol{e}$ is the error associated with the calculations.

Random and systematic errors arose from measurements and computations, respectively. Measurement errors, the probability of a measurement's repeatability, are dependent on an instrument's accuracy and were assumed to have random normal distributions, with a mean and standard deviation equaling the measured value and instrument error, respectively. These errors can either be a fixed value, such as the accuracy of a tape measure, or fluctuating, such as water discharge velocity meter (± 2% of the measured velocity). Random errors (σ_R) associated with calculated variables (R) were a function of propagating measurement random errors through the calculations using:

$$R = f(x, y, ..., z)$$

$$s_R = \sqrt{\left(\frac{\partial R}{\partial x}\right)^2 s_x^2 + \left(\frac{\partial R}{\partial y}\right)^2 s_y^2 + ... + \left(\frac{\partial R}{\partial z}\right)^2 s_z^2}$$

where x, y, and z represent independent variables with corresponding errors of s_x, s_y, and s_z (Taylor 1997).

RESULTS AND DISCUSSION

Sediment: Suspended sediment comprised approximately 20% of the total sediment load in the MFPR. The low percentages of the total load, attributable to the suspended sediment load, is characteristic of forested, montane streams. The bedload transport regime in the mainstem MFPR was sand (d_{50} = 1.5 mm) overriding a coarse, armored layer (d_{50} = 57.0 mm). Bedload transport samples, scour chain data, sand dune excavations, and visual observations indicated that the mobile fraction of the streambed was primarily sand. Bedload samples collected at MFDG and MFLT shared similar particle size distributions (Figure 2) and mineralogic compositions. The consistent shape and size of particles in the bedload samples may indicate that the material entering the study reach is moving through without much reworking. The armored layer's relative immobility, for water discharges observed during the study period, was evident by the coarse particle's clast supported packing, algal growth on the particle's surface, streambed firmness under foot, scour chain results, lack of observed coarse material in the sand dunes and bedload samples, and incipient motion calculations. Flows observed during the period of study only locally mobilized small quantities of particles less than 64 mm (b-axis). In both reaches, incipient motion calculations, coupled with flood frequency analysis, indicated that an 8 year event would be necessary to fully mobilize the armored layer.

Bedload transport rates ranged from 29 to 854 tons/day and 0.0 to 562 tons/day for upper (MFDG) and lower sites (MFLT), respectively (Table 1). Due to the greater stream power and sand supply, the lower site transported considerably more bedload than upper. Gradation curve illustrating the particle size distribution of bed-material and armored layer.

Site: At bankfull conditions, ANDCK and SCRCK averaged 41 and 14 tons/day, respectively, about 10 % of the bedload that entered the study reach. The exponents of MFDG's, MFLT's, and LTCK's bedload rating curves were 0.76, 2.09, and 2.03, respectively, when regressing bedload in terms of tons/day. The MFLT and LTCK reaches' exponents were similar to those observed by Whiting (1999) for gravel bedded streams throughout the Idaho Batholith. Whiting's exponents ranged from 1.62 to 3.93 and determined the majority of the streams to be supply-limited. The MFDG reach's lower exponents was a function of the sand available for transport and is typical of a sand bed stream.

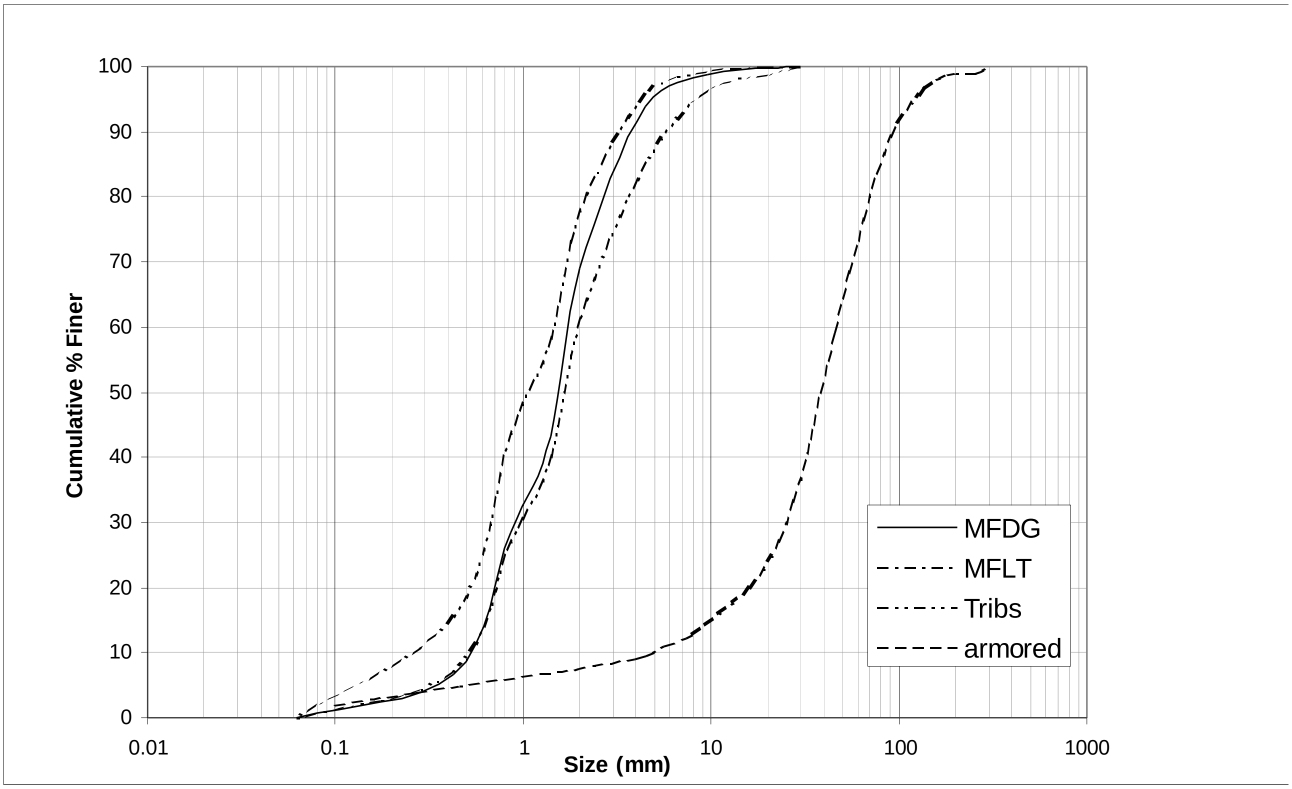

Figure 2. Gradation curve illustrating the texture of mobile and immobile bed-material.

Sediment Budget

The calculated the total bedload transported during the 1999 water year into and out of the study reach as 25,693 ± 7,708 and 64,855 ± 19,457 tons, respectively, with a net gain of 39,162 ± 11,760 tons (Table 1). The net gain reflected either uncertainty due to error propagation in the calculations, inputs not accounted for in the budget, a change in study reach's sand storage, or a combination of the three. Computationally, even after accounting for the error in the hydrologic and sediment transport rating curves, the study reach's net gain remained between 11,997 and 66,327 tons, or a 40 and 367% gain in the outgoing bedload. Since a net gain existed after accounting for the propagated error, then the study reach's net loss was not an artifact of the computations.

TMDL Development: IDEQ (1999) measured the biologic health of the MFPR and concluded that salmonids are impaired by excess "clean" sediment from the confluence with the SFPR to Bulldog Creek. Within this river reach, excess fine sediments inundating salmonid habitat are likely the major cause of water quality impairment. In addition, the level of impairment increases downstream with the lowermost reaches of the MFPR supporting very few salmonids (IDEQ, 1999). We use the upper site as a reference reach to infer what the desired substrate conditions are at the lower site.

Table 1. Study Reach Sediment Budget for the 1999 Water Year.

Reach	Bedload Transported (tons)	Bedload Transport Error (tons)
Input		
MFLT	16,746	5,024
LTCK	6,727	2,018
SCRCK	538	161
ANDCK	1,682	505
Output		
MDFG	64,855	19,457
Net	39,162	11,749

Data show that the dominant type of sediment is sand size bed-material; therefore, the TMDL likely does not need to consider the suspended sediment load. In the MFPR, the amount of sand stored and transported through a given reach is linked to the available salmonid habitat where pools are literally filled with sand size bed-material. The upper site, at Lightning road bridge, is located about 1 mile downstream of Bulldog Creek (Figure 1). We presume this site is an adequate reference reach to infer the desired conditions for the lower reach.

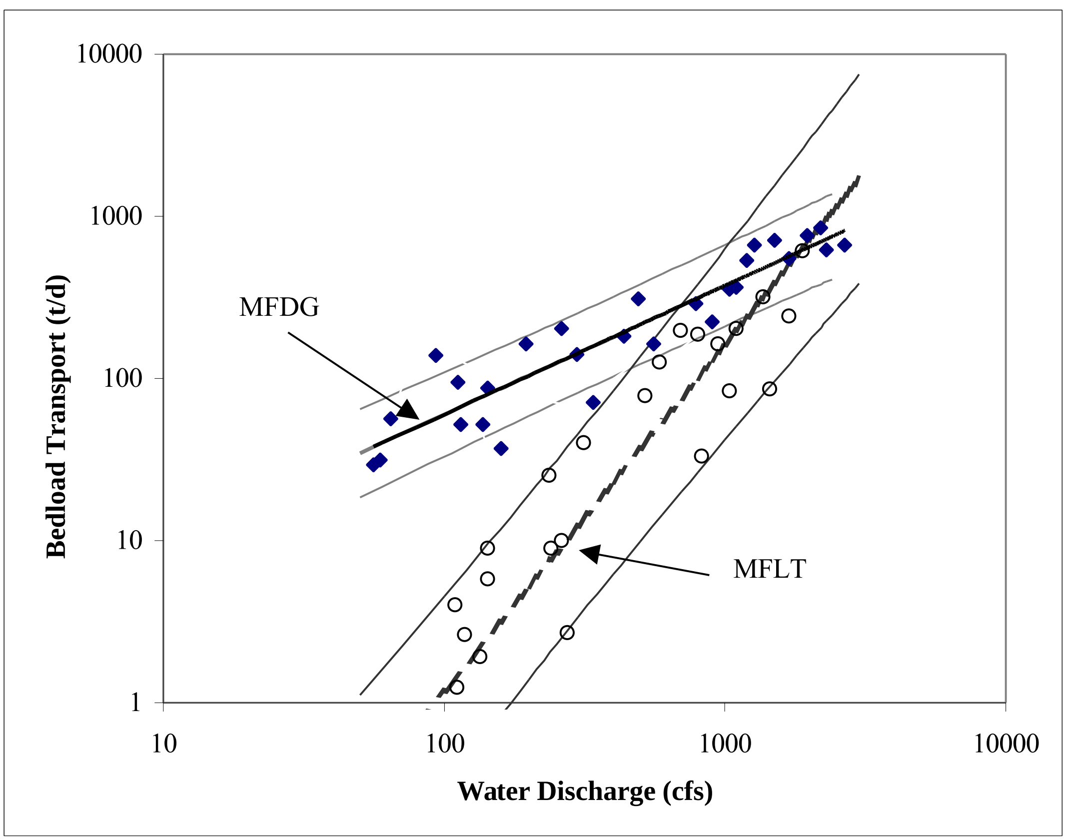

Figure 3. Scatter plot illustrating bed-material rating curves.

We use the sediment budget and sediment rating curves to estimate how much the sediment load needs to be reduced to meet the TMDL goals. The annual variability of the MFPR sediment load makes is difficult to establish a rigid quantitative load reduction much like a water quality standard. Using the average measured med-material load of the upper and lower sites indicates that the bed-material load needs to be reduced at least 68%.

To better define the desirable sediment load that will increase available salmonid habitat, we use sediment rating curves developed for the upper and lower sites. A common technique is to compare the slopes of the rating curves and estimate how much the slope needs to be reduced to meet the TMDL goals (EPA, 1999). Figure 3 illustrates the sediment rating curves for the upper and lower sites. The exponents of MFDG's, MFLT's bedload rating curves were 0.76 and 2.09, respectively. These curves indicate that to meet reference conditions measured at the MFLT, the slope of MFDG curve needs to be reduced 63 %. Indirectly, the sediment rating curves are a measure of biological health with an inferential link between the annual bed-material load and available salmonid habitat. As the goals of the TMDL are reached, we expect to see the slope of the lower site rating curve decrease and the volume of pools increase.

SUMMARY

The sediment transport regime measured during this study was dominated by bed-material transport comprising approximately 80% of the total sediment load. Bed-material samples consisted of well sorted sands (D_{50} = 1.5 mm) derived from granitic soils with angular to subangular shaped particles. In the lower reaches, sand overrides a coarse, armored layer, covering much of the bottom and filling in pools. The armor layer's particle size distribution was consistent throughout the study reach (D_{50} = 57.0 mm).

These data help focus TMDL restoration effort on bed-material sediments since suspended sediment does not appear to be contributing to poor water quality. We infer a threshold or goal for the "clean" sediment TMDL using sediment rating curves. These curves help account for annual variability and provide a consistent measure to track water quality improvements.

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ESTIMATION OF POTENTIAL RUNOFF-CONTRIBUTING AREAS

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Abstract: *Digital topographic, soil, and land-use information was used to estimate potential runoff-contributing areas in Kansas. The results were used to compare selected subbasins representing slope, soil, land-use, and runoff variability across the State. Potential runoff-contributing areas were estimated collectively for the processes of infiltration-excess and saturation-excess overland flow using a set of environmental conditions that represented very high, high, moderate, low, very low, and extremely low potential for runoff (in relative terms). Various rainfall- intensity and soil-permeability values were used to represent the threshold conditions at which infiltration-excess overland flow may occur. Antecedent soil-moisture conditions and a topographic wetness index (TWI) were used to represent the threshold conditions at which saturation-excess overland flow may occur. Land-use patterns were superimposed over the potential runoff-contributing areas for each set of environmental conditions.*

Results indicated that the very low potential-runoff conditions (soil permeability less than or equal to 1.14 inches per hour and TWI greater than or equal to 14.4) provided the best statewide ability to quantitatively distinguish subbasins as having relatively high, moderate, or low potential for runoff on the basis of the percentage of potential runoff-contributing areas within each subbasin. The very low and (or) extremely low potential-runoff conditions (soil permeability less than or equal to 0.57 inch per hour and TWI greater than or equal to 16.3) provided the best ability to qualitatively compare potential for runoff among areas within individual subbasins. The majority of the subbasins with relatively high potential for runoff are located in the eastern half of the State where soil permeability generally is less and precipitation typically is greater. The ability to distinguish the subbasins as having relatively high, moderate, or low potential for runoff was mostly due to the variability of soil permeability across the State. The spatial distribution of potential contributing areas, in combination with the superimposed land-use patterns, may be used to help identify and prioritize subbasin areas for the implementation of best-management practices to manage runoff and meet Federally mandated total maximum daily load requirements.

INTRODUCTION

The State of Kansas is required by the Federal Clean Water Act of 1972 to develop a total maximum daily load (TMDL) for water bodies throughout the State. A TMDL is an estimate of the maximum pollutant load (material transported during a specified time period) from point and nonpoint sources that a receiving water can accept without exceeding water-quality standards (U.S. Environmental Protection Agency, 1991). Requisite for the development of TMDL’ s is an understanding of potential source areas of storm runoff that are the most likely contributors of nonpoint-source pollution within a basin.

A study by the U.S. Geological Survey (USGS), in cooperation with the Kansas Department of Health and Environment, was begun in 1999 to estimate the spatial extent and pattern of potential runoff-contributing areas in Kansas. The specific study objectives were to:

(1) Estimate potential runoff-contributing areas for infiltration-excess and saturation-excess overland flows;

(2) Describe land-use patterns that may affect the potential for runoff; and

(3) Compare the potential for runoff between and within subbasins throughout the State.

This paper presents the methods and results of the study to estimate the spatial extent and pattern of potential runoff- contributing areas in Kansas using the Lower Arkansas River Basin as a representative example. The methods presented in this paper may be applicable nationwide as related to the development of TMDL’ s and the identification and prioritization of areas for the implementation of best-management practices (BMP’ s). This study was made possible in part by support from the Kansas State Water Plan Fund.

BACKGROUND

Runoff-contributing areas within river basins primarily are the result of two processes, both of which produce overland flow. The first process is infiltration-excess overland flow, which occurs when precipitation intensity exceeds the rate of water infiltration into the soil. This process may be dominant in basins where the land surface has been disturbed (for example, plowed cropland) or where natural vegetation is sparse. The second process is saturation- excess overland flow, which occurs when precipitation falls on temporarily or permanently saturated land-surface areas that have developed from "outcrops" of the water table at the land surface (Hornberger and others, 1998). A temporary water table can develop during a storm when antecedent soil-moisture conditions in a basin are high. The saturated areas where saturation-excess overland flow develops expand during a storm and shrink during extended dry periods (Dunne and others, 1975).

Both runoff processes would be expected to affect the load of water-quality constituents in streams, although possibly in different ways due to different flow paths. The identification of potential runoff-contributing areas in a basin can provide guidance for the targeting of BMP's to reduce runoff and meet TMDL requirements. Implementation of BMP's within potential runoff-contributing areas is likely to be more effective at reducing constituent loads compared to areas less likely to contribute runoff.

The spatial extent and pattern of runoff-contributing areas are affected by climate, soil, and terrain characteristics. Contributing areas of infiltration-excess overland flow are determined by the interaction of rainfall intensity and soil permeability. The least-permeable soils in a basin are the most likely to contribute infiltration-excess overland flow. As rainfall intensity increases, areas with more moderate permeability also may contribute overland flow.

Contributing areas of saturation-excess overland flow are determined by the interaction of basin topography and antecedent soil-moisture conditions. The effect of topography on saturation-excess overland flow can be quantified by an index called the topographic wetness index (TWI) (Wolock and McCabe, 1995). The TWI is computed as ln(a/S) for all points in a basin, where "ln" is the natural logarithm, "a" is the upslope area per unit contour length, and "S" is the slope at that point. The locations in a basin with the highest TWI values (large upslope areas and gentle slopes) are the most likely to contribute saturation-excess overland flow. When antecedent soil-moisture conditions are dry, only areas with the highest TWI values may be saturated and potentially contribute overland flow. When antecedent soil-moisture conditions are wet, areas with lower TWI values may be saturated and potentially contribute overland flow.

Land use is another important factor that affects runoff within a basin, both physically and chemically. Physically, such characteristics as vegetative cover, soil permeability, and the amount and connectivity of impervious surfaces combine to determine the relative magnitudes of runoff for various types of land use. For example, cropland and urban land uses are typified by higher runoff volumes than grassland and woodland. Increased runoff from cropland is attributable to several factors, including the removal of native vegetation and soil compaction, which decrease surface permeability. Increased runoff from urban areas is mostly due to the substantial increase in the percentage of impervious surfaces (for example, streets, parking lots, roofed structures). In contrast, decreased runoff from undisturbed grassland and woodland areas is due to such factors as the interception of falling precipitation by the vegetation and accumulated organic debris on the surface, as well as the dense network of roots that increases soil porosity. Chemically, land use is an important determinant of the sources, types, and amounts of contaminants that affect the water quality of runoff. The chemical effects of land use on runoff are not addressed in this paper.

Potential runoff-contributing areas with high percentages of cropland and (or) urban land uses would be expected to have higher potential for runoff compared to areas of similar soils and topography with high percentages of grassland and (or) woodland. Moreover, areas classified as noncontributing on the basis of soil and topographic characteristics may contribute runoff if the land use is mostly cropland and (or) urban. Thus, the importance of including land use in an assessment of the potential for runoff is evident. Implementation of BMP's in potential runoff-contributing areas with high percentages of cropland and (or) urban land uses is likely to be more effective at reducing runoff compared to similar areas with high percentages of grassland and (or) woodland.

ESTIMATION OF POTENTIAL RUNOFF-CONTRIBUTING AREAS

In Kansas, subbasins representing slope, soil, land-use, and runoff variability were selected for analysis. The selected subbasin boundaries were obtained from a statewide data base of 11- and 14-digit hydrologic unit (basin) boundaries that was developed at a scale of 1:24,000 (U.S. Department of Agriculture, Natural Resources Conservation Service, 1997). Geographic-information-system (GIS) techniques and available digital data were used to perform the spatial analyses required to estimate potential runoff-contributing areas. All analyses were done using the GRID module of the ArcInfo GIS software package. (Any use of trade, product, or firm names is for descriptive purposes only and does not constitute endorsement by the U.S. Geological Survey.)

In this study, digital topographic and soil data, as well as digital land-use data, were used to estimate and compare potential runoff-contributing areas in Kansas. The digital data included the U.S. Department of Agriculture' s 1:24,000-scale soil data base (U.S. Department of Agriculture, Natural Resources Conservation Service, 1996), the USGS 100-m-resolution digital elevation model (DEM) (U.S. Geological Survey, 1993), and 1:100,000-scale, land-cover data (Kansas Applied Remote Sensing Program, 1993). The soil and land-cover data were converted from polygon to grid format with a 10,000-m^2 (0.01-km^2) grid-cell size to match the resolution of the 100-m DEM. These three digital data sets are suitable for comparing potential runoff among areas tens of square kilometers in size. This statement is based on the fact that areas tens of square kilometers in size have sufficient numbers of unique 1:24,000- scale soil mapping units and elevation data points to compute representative mean values for the purpose of comparing areas. Thus, in this paper emphasis is placed on a comparison of potential contributing areas both between and within the subbasins that ranged in size from about 150 to 6,600 km^2.

The potential for infiltration-excess overland flow was estimated using the 1:24,000-scale soil-permeability digital data. A depth-weighted, mean soil permeability was used. In general, there is an inverse relation between soil permeability and the potential for infiltration-excess overland flow. Using GIS techniques, a statewide grid of depth- weighted, mean soil permeability was assembled from the soil data base.

An equal-interval approach was used to select six threshold soil-permeability values that represent the rainfall intensity at which infiltration-excess overland flow may occur. In Kansas, soil permeability ranges from 0 to 17.6 in/ hr. However, because about 93 percent of the State has a soil permeability of 4.0 in/hr or less, the effective range used in this study was 0 to 4.0 in/hr. Thus, the threshold soil-permeability values, representing very high, high, moderate, low, very low, and extremely low rainfall intensity (in relative terms), were set at 3.43, 2.86, 2.29, 1.71, 1.14, and 0.57 in/hr, respectively.

In general, lower rainfall intensities occur more frequently than higher rainfall intensities. For central Kansas, Hershfield (1961) estimated that 1-hour storms with rainfall intensities of 1.4 and 3.4 in/hr have recurrence intervals of 1 and 50 years, respectively. The higher soil-permeability thresholds imply a more intense storm during which areas with higher soil permeability potentially may contribute infiltration-excess overland flow. The threshold soil- permeability values were used to compare subbasins on the basis of the percentage of each subbasin with soil- permeability values that were less than or equal to the threshold value and thus potentially contribute infiltration- excess overland flow.

The potential for saturation-excess overland flow was estimated using DEM-derived TWI digital data. In general, there is a direct relation between TWI and the potential for saturation-excess overland flow. Derivation of the TWI digital data followed the approach described by Wolock and McCabe (1995). Elevation differences among the grid cells in the DEM were compared and used to create a flow-direction grid (Jenson and Domingue, 1988). The flow-direction grid was used to derive a flow-accumulation grid by computing the number of upslope cells that drain into each cell. The upslope area per unit contour length *(a)* for each cell in the flow-accumulation grid was computed as:

$$a = (\text{number of upslope cells} + 0.5) \times (\text{grid-cell length}). \quad (1)$$

Using the DEM and the flow-direction grid, the magnitude of the slope *(S)* was computed for each cell as:

$$S = (\text{change in elevation between neighboring grid cells}) / (\text{horizontal distance between centers of neighboring grid cells}). \quad (2)$$

The resultant slope (gradient) grid then was used in combination with the flow-accumulation grid to compute TWI for each cell as:

$$\text{TWI} = \ln\,(a\;/\;S). \tag{3}$$

Using GIS techniques, a statewide grid of TWI data was created.

An equal-interval approach was used to select six threshold TWI values that represented a range of wet-to-dry, antecedent soil-moisture conditions. For this analysis, the TWI grid cells that represent the streams were excluded because the TWI is considered a characteristic of the land surface that contributes runoff to the streams. In Kansas, the TWI (with grid cells representing the streams excluded) ranges from 4.5 to 18.3. Because the TWI had a normal distribution, the full range of values was used in this study. Thus, the threshold TWI values, representing very wet, wet, moderate, dry, very dry, and extremely dry antecedent soil-moisture conditions, were set at 6.5, 8.4, 10.4, 12.4, 14.4, and 16.3, respectively. The lower TWI thresholds imply wetter antecedent soil-moisture conditions during which areas with lower TWI values potentially may contribute saturation-excess overland flow. The threshold TWI values were used to compare subbasins on the basis of the percentage of each subbasin that had TWI values greater than or equal to the threshold value and thus potentially contribute saturation-excess overland flow.

The combined potential for runoff due to infiltration-excess and saturation-excess overland flows was estimated by merging the previously described hypothetical environmental conditions. A very high potential-runoff condition was created by combining very high rainfall intensity (soil permeability less than or equal to 3.43 in/hr) with very wet antecedent soil-moisture (TWI greater than or equal to 6.5) conditions. A high potential-runoff condition was created by combining high rainfall intensity (soil permeability less than or equal to 2.86 in/hr) with wet antecedent soil- moisture (TWI greater than or equal to 8.4) conditions. A moderate potential-runoff condition was created by combining moderate rainfall intensity (soil permeability less than or equal to 2.29 in/hr) with moderate antecedent soil-moisture (TWI greater than or equal to 10.4) conditions. A low potential-runoff condition was created by combining the low rainfall intensity (soil permeability less than or equal to 1.71 in/hr) with dry antecedent soil-moisture (TWI greater than or equal to 12.4) conditions. A very low potential-runoff condition was created by combining the very low rainfall intensity (soil permeability less than or equal to 1.14 in/hr) with very dry antecedent soil-moisture (TWI greater than or equal to 14.4) conditions. An extremely low potential-runoff condition was created by combining the extremely low rainfall intensity (soil permeability less than or equal to 0.57 in/hr) with extremely dry antecedent soil-moisture (TWI greater than or equal to 16.3) conditions. The combined conditions were used to compare subbasins on the basis of the percentage of each subbasin that potentially contributes runoff by one or both overland-flow processes. Also, the combined conditions were used to assess the spatial distribution of potential contributing areas within the subbasins.

Land use was addressed in two ways. First, the land-use composition of each subbasin was estimated as the percentage of each subbasin categorized as cropland, grassland, woodland, and urban land uses. This information may be used to quantitatively assess land-use differences between subbasins. Second, for each set of environmental conditions, the grid cells classified as potential contributing areas were color-coded by land-use type. The resulting maps provide information on the spatial distribution of potential contributing areas within a subbasin as well as the land-use patterns within the potential contributing areas. This information may be used to help identify and prioritize subbasin areas for implementation of BMP' s.

RESULTS

Results of this study indicated that the sets of environmental conditions that represented higher potential-runoff conditions generally were not useful for the purpose of distinguishing subbasins as having relatively high, moderate, or low potential for runoff. The inability to distinguish subbasins for the higher potential-runoff conditions was due to the fact that the percentage of contributing areas was in excess of 90 percent for virtually every subbasin. Thus, in this paper, only the results for the low, very low, and extremely low potential-runoff conditions are presented. The results are useful for the purpose of comparing potential runoff-contributing areas between and within subbasins. However, the results are not intended to be used for the purpose of inferring the magnitude of potential runoff within a given area. Complete results for the statewide analysis are presented in Juracek (in press). In this paper, only the results for the Lower Arkansas River Basin in south-central Kansas are provided as an example.

The ability to distinguish subbasins of the Lower Arkansas River Basin as having relatively high, moderate, or low potential for runoff was good for the low potential-runoff conditions and very good for the very low (figure 1) and extremely low potential-runoff conditions. Potential contributing areas for the very low potential-runoff conditions (soil permeability less than or equal to 1.14 in/hr, TWI greater than or equal to 14.4) ranged from 15.4 percent of the subbasin for Sandy and Little Sandy Creeks (subbasin 10) to 94.7 percent for Sun and Turkey Creeks (subbasin 12). Of the 12 subbasins in the Lower Arkansas River Basin, 1 had potential contributing areas in more than 90 percent of the subbasin, 3 had potential contributing areas in 70 to 90 percent of each subbasin, 2 had potential contributing areas in 50 to 70 percent of each subbasin, 2 had potential contributing areas in 30 to 50 percent of each subbasin, and 4 had potential contributing areas in 10 to 30 percent of each subbasin (table 1).

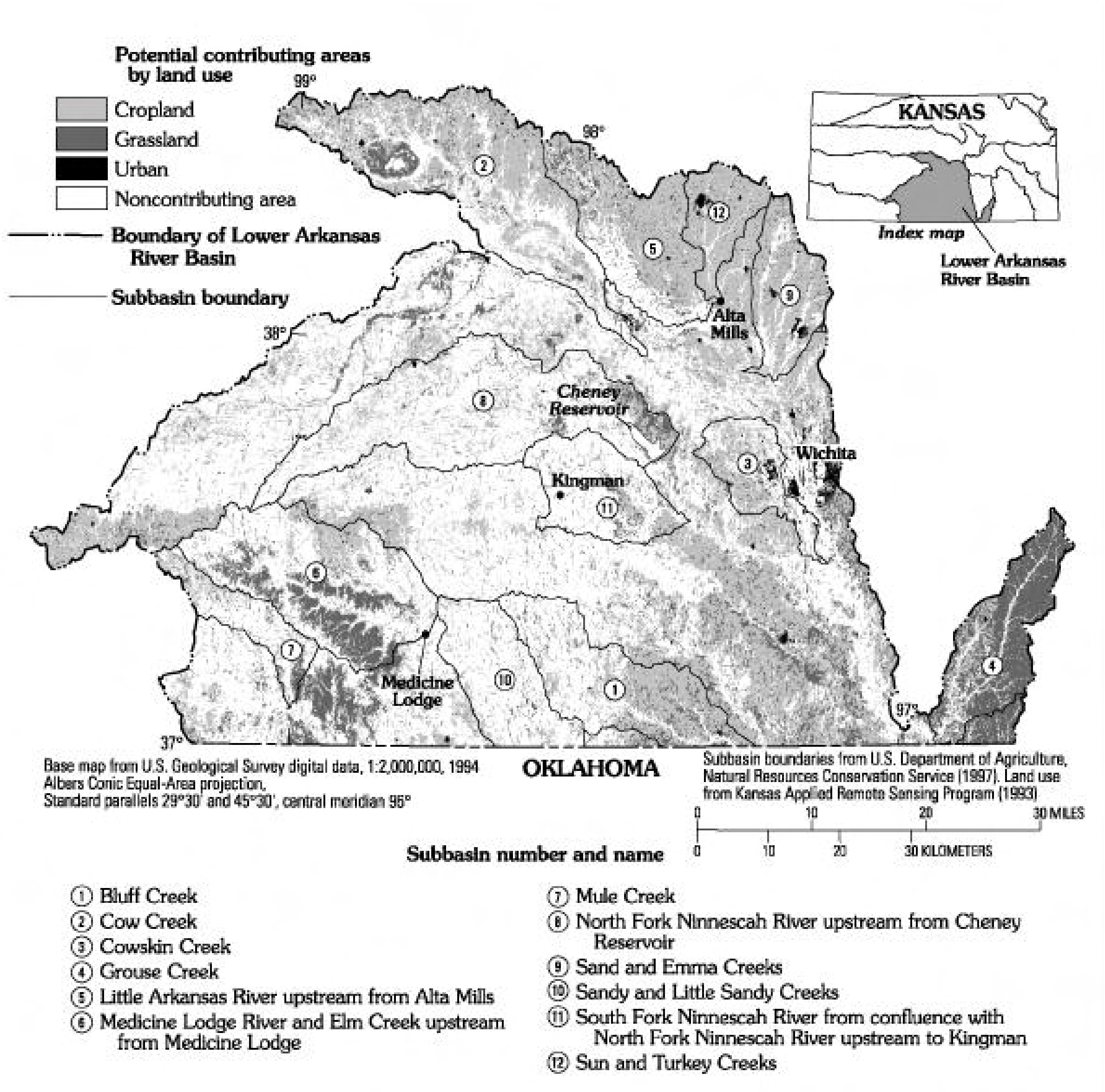

Figure 1. Potential contributing and noncontributing areas of combined infiltration- and saturation-excess overland flows for very low potential-runoff conditions in the Lower Arkansas River Basin, Kansas.

Table 1. Potential contributing areas for combined infiltration- and saturation-excess overland flows, and land use for selected subbasins in the Lower Arkansas River Basin, south-central Kansas

[P, soil permeability, in inches per hour; TWI, topographic wetness index. Land-use data from Kansas Applied Remote Sensing Program (1993)]

			Potential contributing area, in percentage of subbasin, for selected potential-runoff conditions			Land use, in percentage of subbasin			
Subbasin number (fig. 1)	**Mean P**	**Mean TWI**	**Low potential runoff**	**Very low potential runoff**	**Extremely low potential runoff**	**Cropland**	**Grassland**	**Woodland**	**Urban**
1	1.4	10.8	85.9	49.4	33.1	69.8	29.1	0.7	0.3
2	1.6	10.9	88.3	62.0	14.8	76.7	20.0	.9	1.4
3	1.9	11.4	89.5	54.1	17.7	76.1	14.9	1.0	6.7
4	.5	10.2	100	86.0	61.9	10.9	85.3	3.2	.1
5	2.4	11.0	86.4	71.5	24.3	66.5	31.8	1.2	.3
6	2.5	10.0	74.8	39.1	25.3	23.2	75.5	1.0	.2
7	2.9	9.9	71.8	28.9	11.7	23.6	75.8	.6	0
8	5.0	11.1	60.7	25.0	9.9	72.7	24.5	1.0	.2
9	.5	11.2	98.2	79.4	73.8	86.6	9.3	1.1	2.4
10	3.4	10.6	66.0	15.4	6.5	44.9	54.6	.4	.1
11	2.9	10.9	77.3	22.3	13.7	57.9	37.8	3.0	0.6
12	.5	11.1	100	94.7	31.6	90.0	6.3	.7	2.7

For extremely low potential-runoff conditions (soil permeability less than or equal to 0.57 in/hr, TWI greater than or equal to16.3), potential contributing areas ranged from 6.5 percent of the subbasin for Sandy and Little Sandy Creeks (subbasin 10) to 73.8 percent for Sand and Emma Creeks (subbasin 9). Of the 12 subbasins, 1 had potential contributing areas in 70 to 90 percent of the subbasin, 1 had potential contributing areas in 50 to 70 percent of the subbasin, 2 had potential contributing areas in 30 to 50 percent of each subbasin, 6 had potential contributing areas in 10 to 30 percent of each subbasin, and 2 had potential contributing areas in less than 10 percent of each subbasin (table 1).

The subbasins were categorized as having relatively high, moderate, or low potential for runoff using the average percentage of contributing areas for very low and extremely low potential-runoff conditions. The very low and extremely low potential-runoff conditions are meaningful because they provide the best ability to distinguish subbasins and because the 1.14 and 0.57 in/hr rainfall intensities occur more frequently than the higher rainfall intensities. A subbasin was categorized as having relatively high potential for runoff if the average percentage of contributing areas for the very low and extremely low potential-runoff conditions was greater than 70 percent. A subbasin was categorized as having relatively low potential for runoff if the average percentage of contributing areas for the very low and extremely low potential-runoff conditions was less than 30 percent. The subbasins having relatively high potential for runoff (average percentage of contributing areas greater than 70 percent) are Grouse Creek (subbasin 4) and Sand and Emma Creeks (subbasin 9). The subbasins having relatively low potential for runoff (average percentage of contributing areas less than 30 percent) are Mule Creek (subbasin 7), the North Fork Ninnescah River upstream from Cheney Reservoir (subbasin 8), Sandy and Little Sandy Creeks (subbasin 10), and the South Fork Ninnescah River from confluence with North Fork Ninnescah River upstream to Kingman (subbasin 11). The remaining subbasins have a relatively moderate potential for runoff (average percentage of contributing areas between 30 and 70 percent).

The spatial distribution of potential contributing areas for very low potential-runoff conditions varies considerably across the Lower Arkansas River Basin (figure 1). For Bluff Creek (subbasin 1) and the South Fork Ninnescah River from confluence with North Fork Ninnescah River upstream to Kingman (subbasin 11), most of the potential contributing areas are located in the downstream half of the subbasins. For Cow Creek (subbasin 2), Cowskin Creek (subbasin 3), and the Medicine Lodge River and Elm Creek upstream from Medicine Lodge (subbasin 6), the potential contributing areas are widespread with several areas of concentration. Potential contributing areas for the Little Arkansas River upstream from Alta Mills (subbasin 5) are widespread and uniformly distributed with the notable exception of a large noncontributing area located in the downstream half of the subbasin. For Mule Creek (subbasin 7), most of the potential contributing areas are located in the upstream and downstream one-thirds of the subbasin. Potential contributing areas for the North Fork Ninnescah River upstream from Cheney Reservoir (subbasin 8) are widely scattered with the exception of a large potential contributing area immediately north of Cheney Reservoir. Elsewhere, the potential contributing areas are widespread with a generally uniform distribution for Grouse Creek (subbasin 4), Sand and Emma Creeks (subbasin 9), and Sun and Turkey Creeks (subbasin 12). For Sandy and Little Sandy Creeks (subbasin 10), the potential contributing areas are generally sparse and widely scattered.

Land use in the subbasins of the Lower Arkansas River Basin is dominated by cropland or grassland. Cropland ranges from 10.9 percent of the subbasin for Grouse Creek (subbasin 4) to 90.0 percent for Sun and Turkey Creeks (subbasin 12). Grassland ranges from 6.3 percent of the subbasin for Sun and Turkey Creeks (subbasin 12) to 85.3 percent for Grouse Creek (subbasin 4) (table 1). The spatial pattern of land use in the potential contributing areas varies among the subbasins (figure 1). Throughout the Lower Arkansas River Basin and statewide, the use of BMP' s may be most effective at reducing runoff if implemented in the potential contributing areas where cropland and (or) urban land uses are widespread.

SUMMARY AND CONCLUSIONS

Digital topographic, soil, and land-use information was used to estimate and compare potential runoff-contributing areas for subbasins throughout Kansas. Potential contributing areas were estimated collectively for the processes of infiltration-excess and saturation-excess overland flow using a set of environmental conditions that represented very high, high, moderate, low, very low, and extremely low potential for runoff (in relative terms). Various rainfall- intensity and soil-permeability values were used to represent the threshold conditions at which infiltration-excess overland flow may occur. Antecedent soil-moisture conditions and a topographic wetness index (TWI) were used to represent the threshold conditions at which saturation-excess overland flow may occur.

Statewide results indicated that nearly all subbasins had a large percentage of potential runoff-contributing areas for the low to very high potential-runoff conditions. Thus, the ability to distinguish subbasins as having relatively high, moderate, or low potential for runoff for those conditions was very limited. The best statewide ability to quantitatively distinguish subbasins as having relatively high, moderate, or low potential for runoff, on the basis of the percentage of potential runoff-contributing areas within each subbasin, was provided by the very low potential-runoff conditions (soil permeability less than or equal to 1.14 inches per hour and TWI greater than or equal to 14.4). The best ability to qualitatively compare potential for runoff among areas within individual subbasins was provided by the very low and (or) extremely low potential-runoff conditions (soil permeability less than or equal to 0.57 inch per hour and TWI greater than or equal to 16.3). These results are evident in the example provided by the Lower Arkansas River Basin.

The ability to distinguish subbasins, as well as areas within subbasins, as having relatively high, moderate, or low potential for runoff was mostly due to the variability of soil permeability across the State. Because of this variability, the percentage of potential contributing areas for infiltration-excess overland flow varied considerably among the subbasins, especially for the very low potential-runoff conditions. In contrast, the topographic wetness index had a more spatially consistent distribution that typically followed the drainage networks within the subbasins. Because of this uniformity, the relative differences among subbasins in the percentage of potential contributing areas for saturation-excess overland flow remained typically small across the range of potential-runoff conditions despite substantial within-subbasin differences as the potential contributing areas expanded or contracted in response to changing conditions.

Together, the potential contributing areas for infiltration-excess and saturation-excess overland flows provide an understanding of how the spatial distribution of such areas may change in response to changes in environmental conditions. Under low potential-runoff conditions characterized by low antecedent soil moisture and low rainfall intensity, potential contributing areas for infiltration-excess and saturation-excess overland flows are limited to areas of lower soil permeability and saturated areas adjacent to rivers and streams, respectively. As antecedent soil moisture and rainfall intensity increase, the spatial distribution of the potential contributing areas for both infiltration-excess and saturation-excess overland flows increases. Under high potential-runoff conditions characterized by high antecedent soil moisture and high rainfall intensity, the distinction between infiltration-excess and saturation-excess overland flow becomes less meaningful as the ground becomes increasingly saturated and the potential contributing areas for both runoff processes coalesce.

In general, subbasins in eastern Kansas have higher potential for runoff than subbasins in western Kansas for the very low potential-runoff conditions. In eastern Kansas soil permeability generally is less, and precipitation typically is greater. The spatial distribution of potential contributing areas within the individual subbasins showed considerable variability, as is apparent in the Lower Arkansas River Basin. In many subbasins, the flood plains were determined to be mostly noncontributing areas for overland flow due to relatively high soil permeability. However, such areas may still represent a risk to in-stream water quality as contaminants may reach the streams through subsurface flow.

Land use in Kansas is predominantly cropland and grassland. The spatial pattern of land use varies regionally as well as between and within the subbasins. Potential runoff-contributing areas with high percentages of cropland and (or) urban land uses would be expected to have higher potential for runoff compared to similar areas with high percentages of grassland and (or) woodland. Implementation of BMP' s in potential runoff-contributing areas with high percentages of cropland and (or) urban land uses is likely to be more effective at reducing runoff compared to similar areas with high percentages of grassland and (or) woodland. The spatial distribution of potential contributing areas, in combination with the superimposed land-use patterns, may be used to help identify and prioritize subbasin areas for the implementation of BMP' s to reduce runoff and meet Federally mandated TMDL requirements.

This study had some limitations. The potential runoff-contributing areas that were determined may over or under estimate actual contributing areas for a particular location and precipitation event. A variety of factors may account for differences between potential and actual contributing areas including vegetation (type and density), soil compaction, impervious surfaces, BMP' s, and climatic variability. Such factors were not addressed in this study.

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AN INNOVATIVE DEVICE FOR STREAM BEDLOAD SEDIMENT REMOVAL AND MEASUREMENT

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Abstract

Streamside Systems, Inc. has recently patented an innovative stream sediment removal device. This new technology consists of a stainless steel "box" with a ramped front, and slot opening on top which allows the slowing particles ascending the ramp to fall into an internal hopper. Using a remote adjustable timer switch, the sand/sediment within the collector is periodically purged via suction, transporting the sand/water slurry to a contained upland site. The device is intended for removal of excessive stream sediments and as an instrument for accurately measuring bedload sediments.

INTRODUCTION

Excessive sedimentation in many rivers and waterways of the US and the world continues to be a significant problem. State of the art bedload sampling and related bedload quantity estimation techniques are less than optimum and limit the advancement of sediment science and sediment management. Streamside Systems Incorporated has invented, and patented, an apparatus suitable for long-term, efficient removal of bedload and near-bedload stream sediments. The patented collector will also lend itself to scientific measurement of bedload sediments.

DISCUSSION

Description Of Device The collectors are constructed as rectangular "boxes" with the top being a ramped edge (Fig. 1). They are fastened to the stream bottom with spikes or rods. The force of the stream passing over the units cause a significant downward pressure, even more so under floodflow conditions, ensuring the units are secure. Bedload sediments and water pass over the ramped top. Slowed particles fall into a slot on top of the collector, and remain in a collection hopper. The collectors can be constructed of various sizes and materials. The smallest unit, the Series I, is made of laser cut, TIG-welded 304 series, 11 gauge stainless steel. Larger units can be made of stainless steel, concrete, or poured in place.

An internal hopper, slotted pipe, and tubing system allows accumulated materials to be periodically pumped to an upland site. Pumping can be accomplished in several ways, using off-the-shelf pumps, hosing and water ejectors. There is a hinged trash rack, manually operated, which allows operators to clean the hopper or perform any maintenance. This rack is the only movable part on the collector.

A single Series I unit is available in a four foot or eight foot width. The ramp is available in four lengths (Fig. 2). They are designed to be used individually, or fastened end-to-end, providing a "seamless" unit of whatever desired width. The units can be equipped with a water pressure operated internal pinch valve. These pinch valves enable each individual unit to be purged of sediments one at a time. This enables a system to use a much smaller pump than would be needed to pump down multiple collectors in a simultaneous effort. The pump and pinch valves are operated by an electronic controller/sequencer which can be adjusted for each application. Some source of energy is needed to run the pump that periodically purges the collectors. Electric, gas or solar energy may be options to provide the necessary power.

The Series II collectors capture sediments in the same fashion as the Series I collectors. However, the Series II collector is designed to be purged with a direct drive pump, such as a diaphragm or "trash pump". These collectors are intended to be used for temporary installations associated with construction sites or other areas with anticipated short-term excessive sedimentation.

Testing The Series I collector is the result of several years of research by the inventor. Prototypes were tested in the Little Manistee River, Lake County, Michigan. This cold-water trout stream in northern Michigan suffers from excessive sand sedimentation. The current Series I collector reflects many refinements that have resulted in a functional, low maintenance device.

Recent testing in the Little Manistee River yielded a 91.7% capture rate of bedload sediments across the 4-foot width of the collector. Table 1 indicates technical details. Plans are underway for a long-term testing installation this fall (2000), on the Pine River in northern Michigan. This is a high quality trout stream, also suffering from excessive bedload sedimentation. We are also working with the University of Michigan Hydraulics Laboratory to model stream conditions and collector efficiency in a lab setting. Additional testing results will be available at the conference.

The system lends itself as a "portable sandtrap" or as a "portable weir", spanning an entire stream cross-section. To date, we have not tested the system as a multiple-collector installation spanning an entire stream. It is anticipated the heavily built collectors will provide years of trouble-free performance. Given the wide variability of stream hydrology, water quality and sediment qualities, it is difficult to anticipate testing challenges without site specific in-stream data. For example, in northern climates operation of the sediment collectors in the winter months may be a challenge, but not impossible. Having a functional sediment disposal site and avoiding freezing of disposal lines are the two largest concerns with a winter installation. In warmer months, the collector system operates very quietly with little maintenance.

Our collectors provide an elegant, non-intrusive means of sediment removal. The units are streamlined when installed, and pose little hazard to people or aquatic organisms in the stream. Organic debris in suspension or at the bedload level tends to pass over the collection slot, providing for collection of relatively clean sediments. Sediments typically migrate up the ramp in dune or ray formations, finally reaching the collection slot and pour inside.

More testing of the collectors over a variety of geographical locations will eventually enable us to design software which will allow potential users to spell out parameters such as stream flow rates, particle size and density, etc. and enable us to custom design a collector system. This could even include targeting certain size materials and allowing desirable sized materials to pass over the collector. For example, screening the collection slot would allow gravel-sized materials to pass by during a major storm event, yet allow finer grained material to be collected.

We have calculated that removing approximately 1,000 cubic yards of material annually, for a total of 5,000 cubic yards over 5 years costs less than .50 cents/cubic yard. This estimate includes costs for a three-collector installation, including all materials, design, electricity and maintenance. For any quantities over a few hundred cubic yards annually, this system is much less expensive than traditional dredging. Long-term placement for very large quantities of sand may be problematic for larger installations. We have several dewatering options available, including screening filters and storage hoppers that help address sediment management.

Applications Applications at the present time fall into two general categories. One category is the removal of commercial quantities of excessive sediments from rivers and streams. Specific applications include removal of excessive sediments to improve spawning habitat of salmon and trout rivers, maintaining navigation channels, temporary use during stream restoration projects, channel construction or dam removal. Improvement of water quality by capturing non-point source sedimentation, via stormwater discharges, county drains, or active construction sites are other applications. The collectors can be thought of as "portable sandtraps"; or "portable weirs". One potential client inquired about the possibility of spanning the Hudson River with a series of large collectors to capture contaminated sediments. There are many variations of this theme, including industrial applications.

The other main category includes the use of the collectors as scientific tools for measuring bedload sediments and calibrating other existing sampling equipment. The efficient nature of the collector lends itself to detailed point sampling techniques. We at Streamside Systems are not sediment engineers, however, and welcome input from all interested parties regarding applications for scientific purposes. As a private company originator of the collectors, we have the flexibility, willingness and technological skills to continue innovation in this field.

CONCLUSION

Streamside System's patented collectors represent a new, innovative technology, using simple physical principles to capture bedload sediments. There are many applications that will yield improved water quality and stream ecosystem improvements. For scientific purposes, the collectors can be used to make weirs across entire stream cross sections for the purpose of measuring bedload transport. They have the possibility to be used as calibration devices for the wide range of existing bedload sampling tools.

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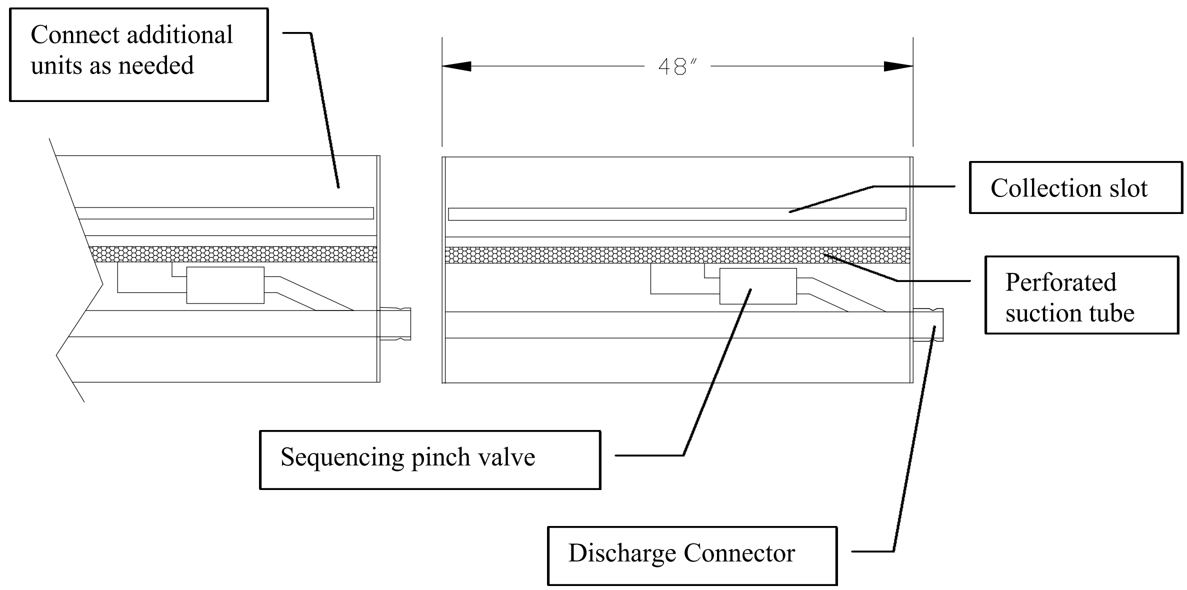
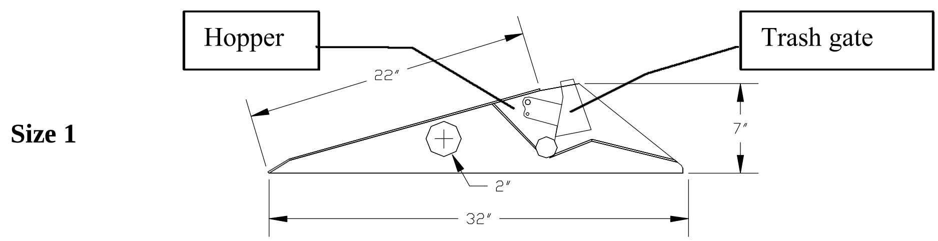
Figure 1

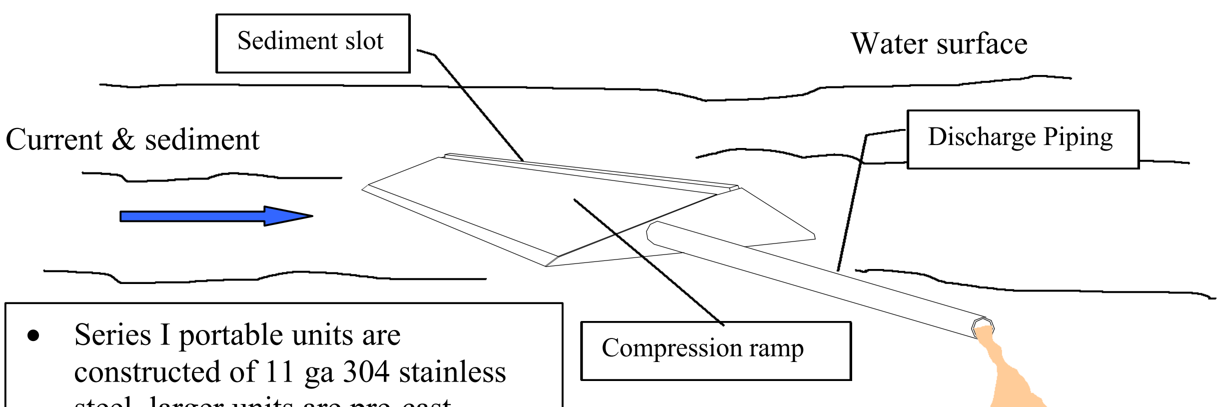
Top View



Cross-section



Oblique



- Series I portable units are constructed of 11 ga 304 stainless steel, larger units are pre-cast concrete or poured-in-place.
- Series I portable units are available in 4 sizes from 32" to 72".
- Units fit together end-to-end providing seamless "one-piece" inter-connection.

Streamside Systems, Inc.
Patented Collectors
Series I Model
www.Streamsidesystems.com
 Fig. 1

Figure 2

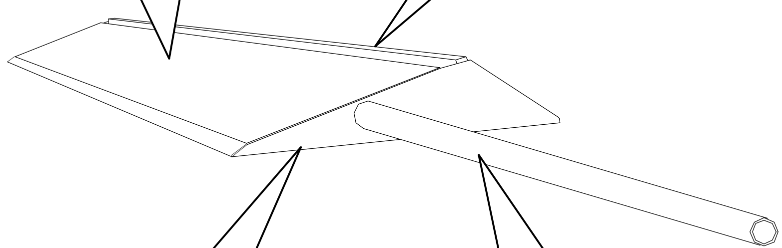
Sizing a Custom Collector

The Collectors hopper capacity determines the frequency that the collector needs to be suctioned down.

- Size 1 capacity is 3 cu/ft
- Size 2 capacity is 6 cu/ft
- Size 3 capacity is 9 cu/ft
- Size 4 capacity is 12 cu/ft

The Collectors height needs to be at least 20% of the stream' s depth.

- Size 1 height is 7"
- Size 2 height is 14"
- Size 3 height is 21"
- Size 4 height is 28"



The Collectors downstream length is in relationship to the stream' s velocity

- Size 1 length is 32"
- Size 2 length is 48"
- Size 3 length is 60"
- Size 4 length is 72"

The Collectors discharge piping Is sized to allow rapid suctioning

- Size 1 main suction is 2"
- Size 2 main suction is 2.5"
- Size 3 main suction is 3"
- Size 4 main suction is 4"

- Series I portable units are constructed of 11 ga 304 stainless steel, larger units are pre-cast concrete or poured-in-place.
- Units fit together end-to-end providing seamless "one-piece" inter-connection.

Streamside Systems, Inc.
Patented Collectors
Series I Model
www.Streamsidesystems.com
 Fig. 2

Table 1

IN-STREAM TEST RESULTS using STREAMSIDE SYSTEM'S patented sediment collector

Little Manistee River, Lake County, Michigan, June 2000

AN AVERAGE OF **91.7%** OF BEDLOAD SEDIMENTS PASSING OVER THE COLLECTOR WERE CAPTURED! *

(Samples were collected during normal summer flows using a Helly-Smith Bedload Sampler. "Before" samples were collected immediately in front of the collector slot, and "after" samples were collected immediately below, or downstream, of the collector slot. Collection time was one minute for all samples. Samples were oven dried and weighed to the nearst 100th gram using a Denver Model 400 XE series digital scale. Sand varied from fine to coarse) **

Sample #	Upstream Sample, grams			Downstream Sample, grams				
1	26.89			0.98				
2	5.06			0.63				
3	4.87			0.13				
4	3.19			0.54				
5	7.76			1.03				
6	15.57			0.57				
7	5.71			0.78				
8	7.2			1.21				
9	3.42			0.56				
10	9.21			0.94				
TOTAL	88.88 grms			7.37 grms				

* Stream velocity at collector apex was 0.6 meters/sec

** Bedload sediments primarily sand, moderate organics present were dried and weighed with all samples

CONTAMINANTS ASSOCIATED WITH SUSPENDED SEDIMENT IN URBAN STREAMS IN AUSTIN, TEXAS

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Abstract: Hydrophobic contaminants are frequently detected in bed sediments of urban water bodies, sometimes at concentrations posing a threat to aquatic biota, yet are rarely detected in whole-water samples of storm water from urban creeks. How, then, are these contaminants transported into receiving urban water bodies? This paper (1) describes a method for the isolation and direct chemical analysis of suspended sediments in storm water, and (2) presents the results of a study of hydrophobic contaminants associated with suspended sediment in storm water from four urban creeks in Austin, Texas. At each creek site, discrete whole-water samples were collected with an automated sampler over the hydrograph and combined into a flow-weighted composite sample. The flow-weighted composites were filtered, and the sediments obtained were analyzed for concentrations of trace elements, organochlorine compounds, and polycyclic aromatic hydrocarbons (PAHs). These concentrations, the discharge data, and the watershed area for each of the creeks were used to compute the load and yield for each storm. The contaminants of concern in the receiving water body—in particular chlordane, DDT, and PAHs—were also present in the storm-water suspended sediments. The presence of elevated concentrations of DDT (at concentrations equal to or greater than that of its metabolites DDE and DDD) and chlordane in storm water from several of the creeks suggests that these two organochlorine pesticides are still in use in the Austin area. Although the manufacture and sale of these organochlorine pesticides was banned in 1974 and 1988, respectively, use of existing stocks on private property is still allowed.

INTRODUCTION

A major fraction of hydrophobic contaminants (HCs) is transported in association with suspended sediment (Bradford and Horowitz, 1982; Garbarino and others, 1995; Rostad and others, 1995). HCs are often detected in high concentrations in the receiving water bodies of urban streams (Van Metre and Callender, 1997; Van Metre and others, 2000), yet they often are not detected in whole-water samples from these urban streams. As a result, city agencies may lack the information needed to determine occurrence, source, loads, and yields of HCs, making it difficult to evaluate the effectiveness of zoning and/or policies for pollution prevention.

Many HCs have serious environmental consequences—of the top 20 contaminants on the Agency for Toxic Substances and Disease Registry (ATSDR)/U.S. Environmental Protection Agency list of top-priority environmental contaminants (ATSDR, 1999), 15 are HCs. These 15 include trace elements such as lead and cadmium; organochlorine compounds, such as chlordane and DDT; and polycyclic aromatic hydrocarbons (PAHs), such as benzo[a]pyrene and dibenz[a,h]anthracene. Many HCs are carcinogenic or mutagenic. Sediment quality guidelines (SQGs) for evaluating whether concentrations of HCs in sediment may adversely affect aquatic biota have been adopted by Environment Canada (1995).

Although the importance of suspended sediments in the transport of HCs is widely recognized, suspended-sediment chemistry is not measured routinely. For example, the U.S. Geological Survey (USGS) National Water-Quality Assessment (NAWQA) program does not include routine suspended-sediment sampling (Gilliom and others, 1995). The USGS National Stream-Accounting Network (NASQAN II) program collected suspended-sediment samples for analysis of major and trace elements at selected sites in large rivers but does not analyze for organic compounds (Hooper and others, 1997).

The conventional way to determine the concentration of HCs on suspended sediment is to compare analyses of a whole-water sample and a filtered water sample, but this approach usually provides an incomplete picture of water quality. This is because HCs are not often detected in whole-water samples. Benzo(a)pyrene, for example, was not detected in any of 100 storm-event samples in residential sites in the Dallas-Fort Worth area and was detected in less than 5 percent of samples from commercial and industrial sites (Raines and others, 1997). Yet concentrations of benzo(a)pyrene have increased 20-fold in the Dallas-Fort Worth metropolitan area, coincident with urban development, based on an age-dated sediment core from White Rock Lake in Dallas (Van Metre and others, 2000). Similarly, lead was detected in only 30 of 120 samples collected from 1972 to 1996 in the Trinity River below Dallas (Jones and others, 1997), suggesting infrequent occurrence; however, over this same time period a dramatic trend in lead concentrations in sediment was revealed in the sediment core (Van Metre and Callender, 1997). The reason for this can be explained with a simple illustration. If, for example, a sample contains 50 milligrams per liter (mg/L) of suspended sediment, and the sediment contains 300 micrograms per kilogram (µg/kg) of PCB (a concentration likely to adversely affect biota health (Environment Canada, 1995)), the concentration of PCB in the whole-water sample will be 0.015 micrograms per liter (µg/L). This concentration is well below most laboratory method detection limits (for example, the USGS National Water Quality Laboratory (NWQL) method detection limit is 0.1 µg/L).

A better approach to determining the concentration of HCs on suspended sediments is to separate the sediment from the water column and analyze it directly. This paper presents a method for the routine collection of suspended sediments for chemical analysis of organic and inorganic constituents. The method allows for analysis of both inorganic and organic contaminants, uses equipment which is portable and relatively inexpensive, and can be used either in the field or in the laboratory. This method is currently being used by the U.S. Geological Survey in cooperation with the City of Austin for the routine monitoring of urban streams.

METHODOLOGY

An important consideration in suspended sediment sampling is the mass of sediment required to achieve reasonable analytical detection levels. Target analytes include major and trace elements, organochlorine pesticides, PCBs, and PAHs.

Elemental concentrations are determined on concentrated-acid digests using inductivity-coupled plasma mass spectrometry (ICP-MS); concentrations of mercury are determined by cold-vapor atomic adsorption (CVAA) (Fishman and Friedman, 1989). Analysis of all constituents by these methods requires about 300 milligrams (mg) of sediment.

Organochlorine compounds and PAHs are measured in organic-solvent extracts using a dual capillary-column gas chromatograph with dual electron capture detector. The method uses a soxhlet extraction with dichloromethane and methanol followed by permeation and adsorption chromatographic fraction. The extraction and clean-up procedures follow Foreman and others (1994) and Furlong and others (1995). Organochlorine compounds are quantified following the procedures of Wershaw and others (1987) and PAHs are quantified by selected ion monitoring (SIM) (E.T. Furlong, oral commun., 2000). Determining concentrations of individual PAHs and most organochlorine compounds at detection levels ranging from 1 to 10 µg/kg requires 2 to 5 grams (g) of sediment. Smaller masses of sediment result in proportionally larger detection levels.

The method described here involves separation of water by direct filtration. Depending on the concentration of suspended sediment in the water, water is collected in either a commercially-available 200-liter (L) plastic drum with a removable PTFE liner or in 9-L polyethylene carboys loaded into an automated sampler. For extremely low suspended sediment concentrations, if a point sample is adequate and streamflow is not changing rapidly, the filtration can be carried out in the field. This approach will have the effect of integrating short-term variations in sediment chemistry over the sampling period. In both cases, sand-size and larger particles are trapped with a polyester mesh before sample processing.

To obtain sufficient sample for analysis of organochlorine compounds and PAHs, water is pumped through a 298-millimeter (mm) diameter, 0.7-micrometer (µm) pore-size glass-fiber filter held in a stainless-steel plate filter holder. The filters are pre-cleaned by baking and wrapped in aluminum foil pending use. Water is pumped through the filter until it clogs; once the filter has clogged, air is pumped through the filter to remove as much water as possible—drying the filter in this way improves laboratory minimum reporting levels by decreasing interference during the analysis. The volume of water pumped is recorded, and samples for total suspended solids (TSS) concentrations are collected before and after sample processing. Experience suggests that from about 0.5 to 2 g of sediment is trapped on each filter, so that 2 to 3 clogged filters are sufficient for analysis. The filters are placed (together) in a baked glass jar and chilled for shipment to the laboratory. The reported concentrations (in mass of organic contaminant per liter of water filtered) are converted to mass of organic contaminant per mass of sediment based on the TSS of the original sample.

To obtain sufficient sample for trace element analysis, water is pumped through a PTFE filter held in an acrylic filter holder. A 145-mm diameter, 0.45-µm pore-size PTFE filter is sprayed lightly with methanol to allow water to pass through. Once clogged, the filter is placed in a sealable plastic bag with a few milliliters of distilled water and massaged until all the sediment has been removed from the filter and is suspended in the small volume of water. More than 98 percent of the initial suspended sediment can be recovered. The sediment is freeze-dried prior to analysis. From 0.1 to 0.5 mg of sediment can be recovered from a single filter, depending on the organic matter and algal content of the water. Results are reported as mass of element per mass of sediment. This approach is similar to that reported by Kimball and others (1995).

CASE STUDY: URBAN STREAMS IN AUSTIN, TEXAS

Large-volume suspended sediment sampling (LVSSS) is being used in Austin to determine the contribution of sediment-associated HCs from two urban creeks to Town Lake, the receiving water body. Chlordane and DDT are among the HCs that have been detected repeatedly in Town Lake surficial sediments (Gandara and others, 1995). High levels of chlordane in fish from the lake resulted in a fish consumption advisory imposed by the Texas Department of Health (1996) which was lifted in 2000. A sediment core from Town Lake revealed increasing trends in concentrations of chlordane and PAHs (Van Metre and Mahler, 1999). Much of the sediment entering the lake originates from Shoal Creek and Barton Creek. The entire Shoal Creek watershed and the lower part of the Barton Creek watershed are urbanized. After heavy rainfall, discharge from these creeks may exceed 1,000 cubic feet per second (ft^3/s) and suspended sediment concentrations can be as large as several thousand milligrams per liter. The USGS, in cooperation with the City of Austin, began using LVSSS in 1999 on one downstream site on Shoal Creek, two sites—one upstream and one downstream—on Barton Creek, and one site Boggy Creek, an urban creek discharging just downstream of Town Lake (Figure 1). Automatic samplers programmed to collect seven 9-L samples at specified intervals after the initial rise in stage have been installed at the four sites. The seven samples are combined to make a flow-weighted composite for each site, from which the suspended sediment is separated for analysis. The watershed area upstream of each of the samplers, the dates of the events sampled, the total sediment load for each event, and the peak flow for each event are given in Table 1.

Site	Watershed Area (acres)	Sample Date	Sediment Load (kg)	Peak Flow (ft^3/s)
Barton Creek, upstream	57,291	5/26/99	6,155	207
		5/01/00	502	41
		6/09/00	188,141	303
Barton Creek, downstream	76,927	5/26/99	2,231	43
		5/02/00	501	76
		6/09/00	70,051	1,050
Shoal Creek	8,269	4/26/99	20,266	175
		1/07/00	226,824	792
		3/17/00	14,711	221
Boggy Creek	8,488	5/10/99	137,441	796
		10/30/99	61,643	489
		1/07/00	212,397	1,020

Table 1. Characteristics of sampling sites and events.

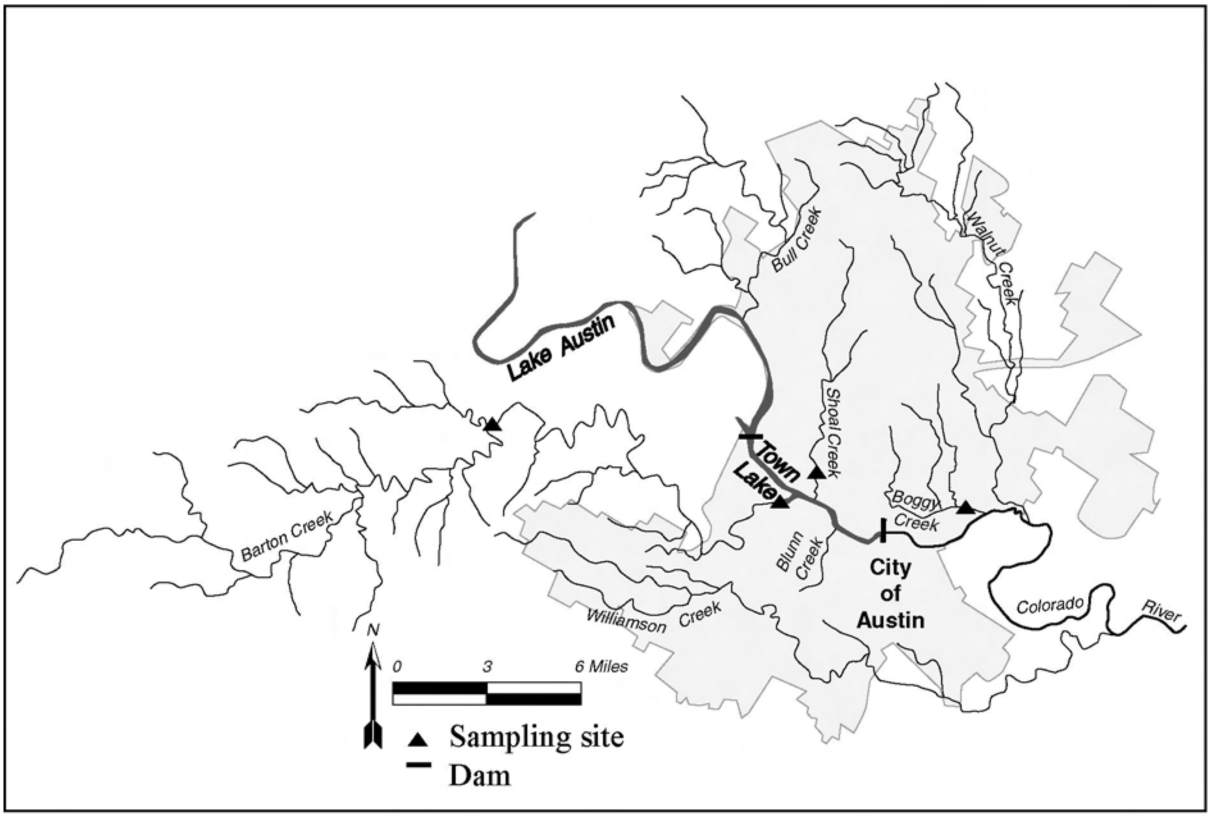

Figure 1. Location of sampling sites.

Results and Discussion: Contaminant concentrations, loads for the individual storm events sampled, and yields for the watershed are shown in Tables 2-4. Also shown are the Environment Canada Interim Sediment Quality Guidelines (ISQGs—the concentration below which no adverse effects to biota are likely to be seen) and the Probable Effects Levels (PELs—the concentration above which adverse effects to biota are likely to be seen) for comparison (Environment Canada, 1995).

DDT and its metabolites, p,p'-DDD and p,p'-DDE, were detected in one or more samples from all of the sites except the upstream Barton Creek site (Table 2). DDT has been banned from use since 1972. It readily metabolizes to DDE and DDD, and, when detected at all in surficial soils, is generally found at much lower concentrations than its metabolites (Van Metre and Callender, 1997), although in very rare cases soils have been found that have an unusually low capacity to degrade DDT (Hitch and Day, 1992). However, when DDT was detected in this study, it was found at concentrations of the same order of magnitude or exceeding that of its metabolites. This suggests that parent DDT may still be in use in the Austin area. DDT and DDE were detected at concentrations exceeding the Canadian PEL SQG in at least one sample at all sites except upstream Barton Creek, indicating a risk to aquatic biota from these sediments. In Boggy Creek, DDD was detected in one sample at concentrations almost 6 times the PEL.

Site	Sample Date	Chlordane			p,p'-DDE			p,p'-DDD			p,p'-DDT		
		Concen-tration (µg/kg)	Load (mg)	Yield (µg/ac)	Concen-tration (µg/kg)	Load (mg)	Yield (µg/ac)	Concen-tration (µg/kg)	Load (mg)	Yield (µg/ac)	Concen-tration (µg/kg)	Load (mg)	Yield (µg/ac)
Barton	5/26/99	<69.6	--	--	<10.7	--	--	<9.27	--	--	<44.8	--	--
Creek,	5/01/00	<63.4	--	--	<6.34	--	--	<6.34	--	--	<117	--	--
upstream	6/09/00	<5.46	--	--	<0.546	--	--	<0.546	--	--	<0.546	--	--
Barton	5/26/99	17.0	38.0	0.494	2.61	5.82	0.0758	2.27	5.06	0.0659	12.9	28.8	0.374
Creek,	5/02/00	39.3	19.7	.256	7.14	3.58	.0465	<6.79	--	--	<143	--	--
downstream	6/09/00	<11.4	--	--	<1.14	--	--	<1.14	--	--	<50.2	--	--
Shoal	4/26/99	10.7	217	263	7.56	153	14.0	<14.7	--	--	5.21	105	23.5
Creek	1/07/00	22.2	5,030	608	5.32	1,210	146	1.66	377	45.6	15.5	22.5	2.72
	3/17/00	56.0	824	100	9.35	137	16.6	1.32	19.5	2.35	19.2	282	34.1
Boggy	5/10/99	21.8	3,000	353	21.8	3,000	353	<1.09	--	--	6.54	899	106
Creek	10/30/99	<17.9	--	--	9.60	592	69.7	<1.79	--	--	9.60	592	69.7
	1/07/00	18.0	3,830	451	15.9	3,380	398	50.9	10,800	1,270	20.2	4,280	504
ISQG/PEL		4.50/8.87			1.42/6.75			3.54/8.51			1.42/6.75		

Table 2. Concentrations of organochlorine pesticides in suspended sediments.

Chlordane, banned in 1988, was also detected in one or more samples from all sites except the upstream Barton Creek site (Table 2). Concentrations in two or more samples from each of these three sites exceeded the Canadian PEL SQG. The multiple detections of chlordane, along with increasing chlordane concentrations in the Town Lake core, indicate that chlordane also may still be in use. A 1990 nationwide survey of pesticide use indicated that continued use of existing stocks in the possession of home owners was substantial, rivaling early 1970s agricultural use (Whitmore and others, 1992).

PCBs, banned in 1979, were detected at the Shoal and Boggy Creek sites (Table 3). Concentrations did not exceed the PEL SQG in any samples.

Concentrations of PAHs were similar at the three urbanized sites, but were 2 orders of magnitude lower at the upstream Barton Creek site. PAHs are largely a product of the incomplete combustion of petroleum, oil, coal, and wood. A recent survey of PAH trends in sediment cores in lakes and reservoirs across the country indicates that increases in PAH concentrations track closely with increases in vehicle use (Van Metre and others, 2000). PAHs represent the largest class of suspected carcinogens (Björseth and Ramdahl, 1985). Total PAHs exceeded the PEL SQG in one sample, the downstream Barton Creek site, and concentrations of benz[a]anthracene and benzo[a]pyrene exceeded the PEL SQG in samples from the Shoal Creek site and the downstream Barton Creek site.

Site	Sample Date	Total PCBs Concen-traction (µg/kg)	Total PCBs Load (mg)	Total PCBs Yield (µg/ac)	Total PAHs Concen-traction (µg/kg)	Total PAHs Load (mg)	Total PAHs Yield (µg/ac)	Benz[a]anthracene Concen-traction (µg/kg)	Benz[a]anthracene Load (mg)	Benz[a]anthracene Yield (µg/ac)	Benzo[a]pyrene Concen-traction (µg/kg)	Benzo[a]pyrene Load (mg)	Benzo[a]pyrene Yield (µg/ac)
Barton	5/26/99	<371	--	--	725	4.46	0.0779	26.3	0.299	5.23	41.7	0.257	4.48
Creek	5/01/00	<190	--	--	819	.41	.00718	26.8	.0134	.235	36.6	.0184	.32
upstream	6/09/00	<16.4	--	--	95.6	18.0	.314	2.52	.474	8.27	2.94	.553	9.65
Barton	5/26/99	<129	--	--	21,400	47.8	.621	808	1.80	23	1,290	2.88	37.4
Creek	5/02/00	<204	--	--	22,900	11.5	.149	679	.340	4	1,180	.590	7.68
downstream	6/09/00	<34.1	--	--	5330	374	4.86	237	16.6	216	351	24.6	319
Shoal	4/26/99	<58.7			15,200	173	20.9	438	7.62	921	647	7.62	921
Creek	1/07/00	22.5	5100	617	15,200	3,440	417	543	123.2	14,901	820	186	22,500
	3/17/00	23.0	338	40.8	19,200	283	34.2	618	9.09	1,100	975	14.3	1,730
Boggy	5/10/99	12.0	1,650	194	9,640	1,320	156	338	46.5	5,474	491	67.4	7,950
Creek	10/30/99	<36.8	--	--	14,400	889	105	155	9.53	1123	699	43.1	5,080
	1/07/00	<30.8	--	--	12,700	2,690	317	244	51.8	6,106	549	117	13,700
SQGs		59.8/6761[a]			1,610/22,800[a]			31.7/385			31.9/782		

Table 3. Concentrations of PCBs and PAHs in suspended sediments. Total PCBs were calculated as the sum of Aroclors 1242, 1254, and 1260. Total PAHs were calculated as the sum of 19 parent PAHs, 10 specfic alkyl-PAHs, and the homologous series of alkyl-PAHs, excluding perylene. ([a] Canadian SQGs not available; analogous SQGs taken from MacDonald and others (2000)).

Site	Sample Date	Arsenic Concen-tration (mg/kg)	Arsenic Load (g)	Arsenic Yield (mg/ac)	Cadmium Concen-tration (mg/kg)	Cadmium Load (g)	Cadmium Yield (mg/ac)	Lead Concen-tration (mg/kg)	Lead Load (g)	Lead Yield (mg/ac)	Zinc Concen-tration (mg/kg)	Zinc Load (g)	Zinc Yield (mg/ac)
Barton	5/26/99	11.2	68.9	--	0.540	3.32	--	39.2	241	--	141	868	--
Creek,	5/01/00	6.78	--	--	1.90	--	--	28.9	--	--	271	--	--
upstream	6/09/00	8.16	1540	--	0.349	65.7	--	15.7	2,950	--	57.6	10,800	--
Barton	5/26/99	12.0	26.8	0.348	0.546	1.22	0.0158	55.8	125	1.62	214	477	6.21
Creek,	5/02/00	15.9	7.97	.104	3.50	1.75	.0228	76.8	38.5	.500	326	163	2.12
downstream	6/09/00	11.7	820	10.7	0.708	49.6	.645	35.8	2,510	32.6	301	21,100	274
Shoal	4/26/99	11.3	228	27.6	0.535	10.8	1.31	47.2	956	116	182	3,690	446
Creek	1/07/00	9.79	2,220	268	0.447	101	12.3	36.0	8,170	988	140	31,800	3,840
	3/17/00	11.2	164	19.9	0.600	8.83	1.07	38.3	564	68.2	211	3,100	375
Boggy	5/10/99	10.2	1,400	165	0.540	74.2	8.74	40.9	5,620	662	151	20,800	2,450
Creek	10/30/99	9.71	599	70.5	0.563	34.7	4.09	33.0	2,030	240	136	8380	988
	1/07/00	8.56	1,820	214	0.723	154	18.1	36.7	7,800	918	134	28,500	3,350
SQGs		5.9/17.0			0.6/3.5			35.0/197			124/271		

Table 4. Concentrations of four trace elements in suspended sediments.

In contrast to the organic HCs, trace element concentrations were fairly similar from site to site (Table 4). Trace element concentrations tended to be slightly lower at the unurbanized site (upstream Barton Creek), but not enough data were available to test if this difference was statistically significant. On the basis of comparison to the SQGs, trace element concentrations from these urban creeks appear to present less a threat to aquatic health than organic HCs, with the exception of cadmium and zinc at the downstream Barton Creek site. These results are consistent with the trace element concentrations in the Town Lake core, in which trace elements had a much lower toxicity index than organochlorine compounds and PAHs (Van Metre and Mahler, 1999).

The results from the LVSSS are not only consistent with the results from the Town Lake sediment core, they help elucidate possible sources of contaminants. Clearly banned contaminants such as chlordane and DDT are still entering the lake on suspended sediments in storm water from at least two urban creeks: Barton and Shoal. Although concentrations of contaminants on sediments are similar at the Shoal Creek and downstream Barton Creek sites, and in some cases are actually higher at downstream Barton Creek, for the limited number of rain events presented here the overall load delivered to Town Lake by Shoal Creek exceeds that delivered by Barton Creek. Additional sampling events will be necessary before this result can be generalized.

The LVSSS method is successful at detecting HCs, particularly organic HCs, which would otherwise go undetected by the standard approach of analyzing a whole-water sample. Converting mass contaminant per mass suspended sediment to mass contaminant per liter of water shows that chlordane, PCBs, benz[a]anthracene, and benzo[a]pyrene would have gone undetected in whole-water samples from all sites, on the basis of minimum reporting levels of the NWQL (0.1, 0.1, 2.4, and 2.8 μg/L for chlordane, PCBs, benz[a]anthracene, and benzo[a]pyrene, respectively). Detection limits for DDT and its metabolites DDD and DDE in whole-water samples are two orders of magnitude lower than for the other compounds (NWQL minimum reporting levels of 0.002, 0.002, and 0.001 for DDT, DDD, and DDE respectively); consequently all seven of the Boggy Creek occurrences would have been detected, but only two of the eight occurrences in Shoal Creek and one of the four occurrences in the downstream Barton Creek site would have been detected.

CONCLUSION

The difficulties of detecting HCs in whole-water samples can be overcome by the separation and analysis of the suspended sediment in the water column. Use of filters of the appropriate size and material and filtration of large volumes of water allow efficient removal of sufficient sediment for laboratory analysis of trace elements and organic contaminants. By separating the solids and analyzing them directly, HCs that might have escaped detection in a whole-water sample may be detected, sometimes at concentrations sufficiently high to present a threat to the health of aquatic biota. A case study in Austin, Texas, revealed that occurrence of HCs on suspended sediments in urban creeks were consistent with those found in a sediment core from the receiving water body. By better elucidating occurrence and concentration of HCs in storm water from urban creeks, this approach promises to improve understanding of sources of HCs and transport processes affecting their occurrence in receiving water bodies.

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EVALUATION OF SEDIMENT TRANSPORT DATA FOR CLEAN SEDIMENT TMDLs

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Abstract: Excess sediment has been identified as a major cause of water quality impairments in U.S. streams. Scientifically-based techniques are needed for the development of Total Maximum Daily Loads (TMDLs) for clean sediment. A methodology is presented in this study to allow Problem Identification and Development of Numeric Targets for the development of TMDLs for clean sediment. Methods to study the link between the amount of excess sediment and the impact on the designated use are also presented.

INTRODUCTION

Excessive erosion, transport, and deposition of sediment in surface waters is a major problem in the United States. The 1996 National Water Quality Inventory (Section 305(b) Report to Congress) indicates sediments are ranked as a leading cause of water quality impairment of assessed rivers and lakes. A national strategy is needed to develop scientifically defensible procedures to facilitate the development of Total Maximum Daily Loads (TMDL) for clean sediment in streams and rivers of the United States. Clean sediment is defined here as sediment that is not contaminated by chemical substances. Pollution caused by clean sediment refers to the quantity of sediment, as opposed to the presence of pollutant-contaminated sediment (USEPA, 1999). The US Environmental Protection Agency has defined a seven-step procedure for the development of clean sediment TMDLs in impacted waterbodies (USEPA, 1999, Fig. 1-2). This study will present a methodology to define the first two steps of the TMDL process for clean sediments: Problem Identification, and Development of Numeric Targets. This methodology defines impaired and stable reference conditions in terms of the stage of channel evolution (Simon, 1989a) by relating these stages to sediment-transport ratings in different physiographic provinces of the country. Procedures for defining the link between impaired and designated uses of a stream and the level of excess sediment are also developed.

DEVELOPMENT OF A SEDIMENT TRANSPORT EVALUATION METHODOLOGY

Classification and Stage of Channel Evolution

A description of the form of the channel in question is important information to allow comparisons of different sites. For this purpose we used the classification scheme of Rosgen (1996). The channel sinuosity, entrenchment ratio, channel slope, and sediment particle size of the boundary are used by the Rosgen technique to classify the stream. Different primary variables than those used by Rosgen (1996) are needed to relate the channel to the likelihood that it may be impaired by excess sediment. A process-based classification scheme (Simon and Hupp, 1986; Simon, 1989a) relying on six stages of channel evolution was chosen for this purpose. The working hypothesis for using this scheme is that sediment transport rates will vary systematically by stage of channel evolution because stage of evolution is a surrogate for dominant channel processes and the relative stability of the channel boundary. Changes in sediment discharge which occur following disturbance to channel systems were related to the six

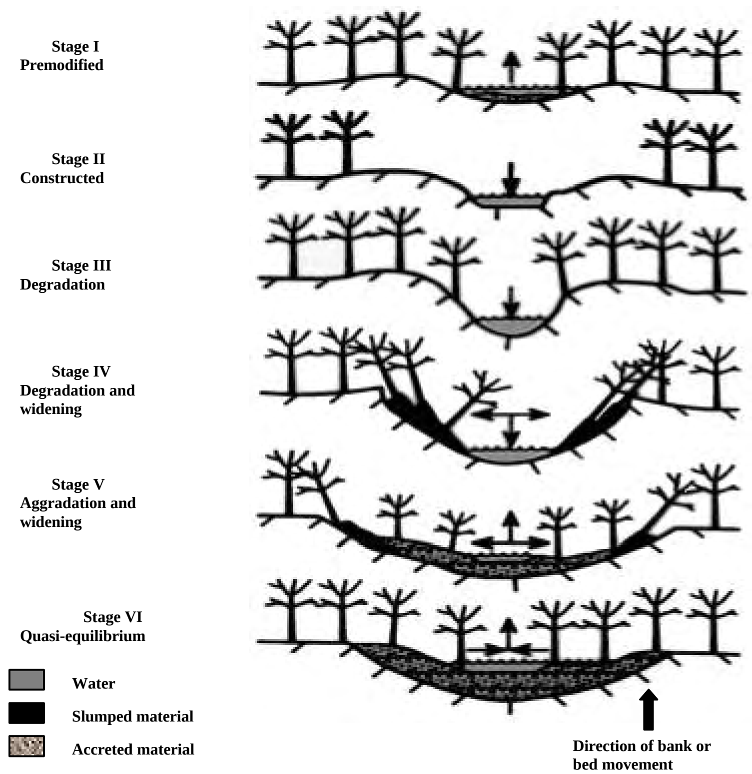


Fig 1.- Stages of channel evolution for disturbed alluvial channels (Simon, 1989a).

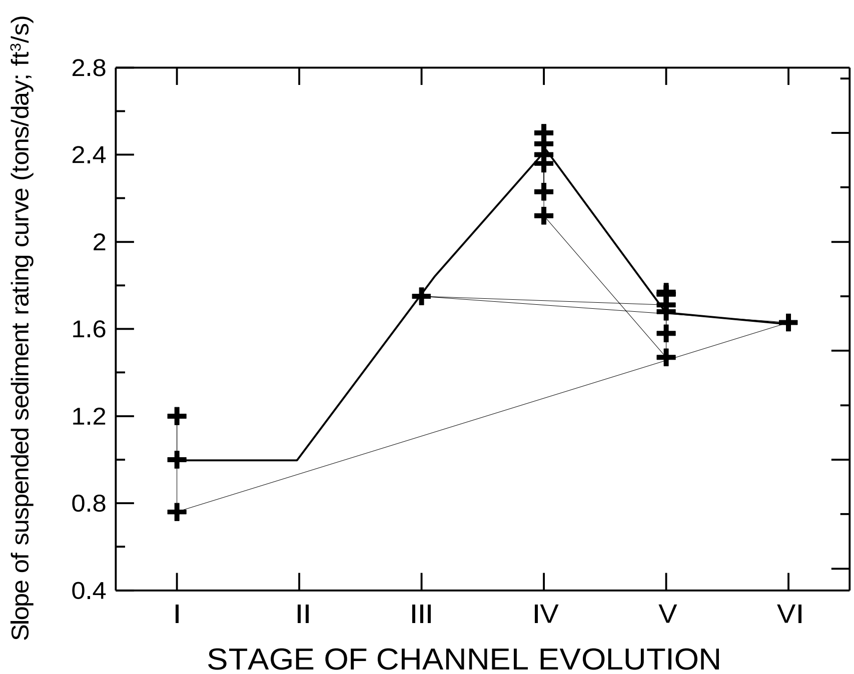


Fig. 2- Variation in suspended-sediment transport efficiency during the course of channel evolution (Simon, 1989b).

stages in a study of 14 sites on streams in West Tennessee. The six stages of channel evolution are shown schematically in Figure 1. Data from the West Tennessee streams have shown that slopes of suspended-sediment rating curves increase from stage I to stage IV, and then decrease in stages V and VI, but not to the levels of stage I (Fig. 2, Simon, 1989b).

Landscape Index

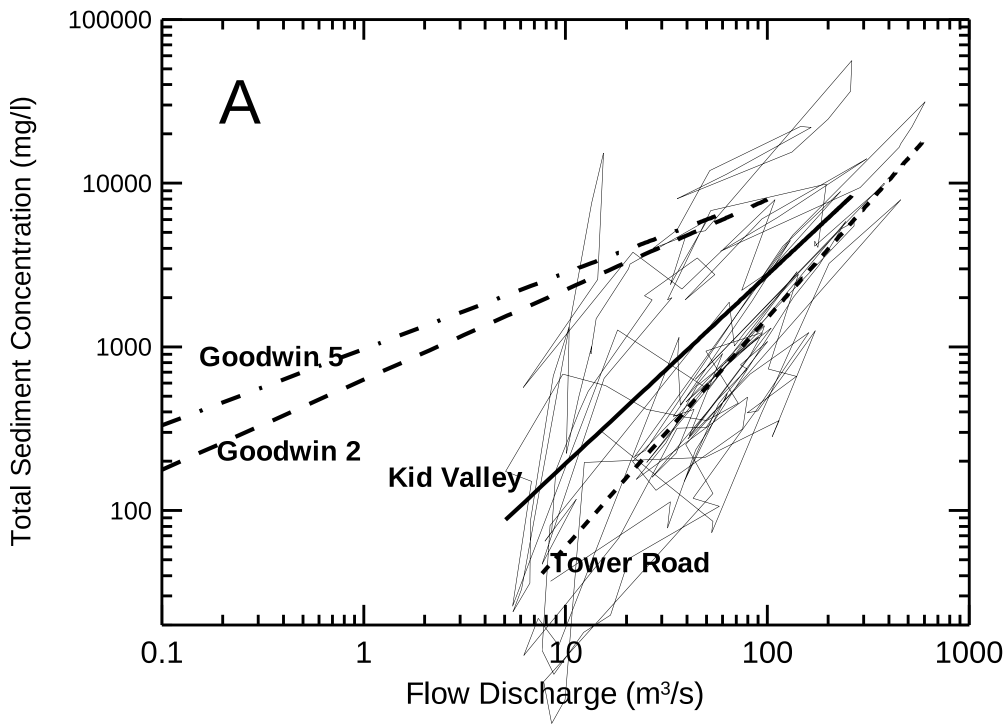
Determining the stage of channel evolution gives a useful indicator of the state of the channel and its banks. However, additional information is required on the state of the rest of the watershed land surface to fully determine the capability of the system to produce and transport sediment. The effect of land use change on sediment yield to the channels of a 21.3 km^2 mixed-land-use watershed in northern Mississippi has been documented by Kuhnle et al. (1996). Decreases in the percentage of land in cultivated crops were found to be related to decreases in the sediment reaching the streams. To incorporate the effect of the state of the watershed on sediment yield, a landscape index will be developed to reflect the influence of meteorological variables, physical characteristics of the land surface, and the type of land cover. This index is currently under development and will be refined with data from watersheds, which include information on the landscape and sediment transport.

Comparison of Different Size Streams.

To compare a given stream to a standard or reference stream, a method to scale the flow and sediment transport is needed. The flow discharge with a return period of 1 to 2 years (often associated with bankfull flow) has been shown to move the most sediment for many streams (e.g. Andrews and Nankervis, 1995; Kuhnle et al., 1999). Thus, bankfull flow (recurrence interval 1-2 years) will be used (similar to Troendle, 2000, written communication) to scale the flow and allow comparisons of streams of different flow discharges. Sediment transport will be expressed as a concentration (mg/l) or as mass per time per unit width. These units represent the sediment carried by a unit portion of the flow and allow comparisons of streams of different sizes. Examples of non-scaled and scaled rating curves for two sites in Mississippi and two sites in Washington are shown in Figure 3. Sediment concentration at bankfull flow and the slopes of the sediment rating curves are shown for the four sites, each with a different-sized drainage area in Figure 3b. Sediment rating curves from streams of different size may, therefore, be compared using the slope of the relation and the sediment concentration at bankfull flow (Fig. 3b).

Determination of Reference Conditions

The reference condition may be defined as the amount of sediment for a given flow that a stable non-impacted stream would produce. The level of sediment transport from a reference site is often used as a target or goal for an impacted stream of a similar type. In this study, reference conditions will be considered along with the stage of channel evolution. Channels that are in stages III, IV, or V are in various states of disequilibrium, are inherently unstable and, therefore, do not have a stable or reference analogue. Channels in stages I or VI have either suffered minimal disturbance and remained stable, or are close to becoming re-stabilized after a disturbance, respectively. Assuming that stages I and VI are stable, the expected range of



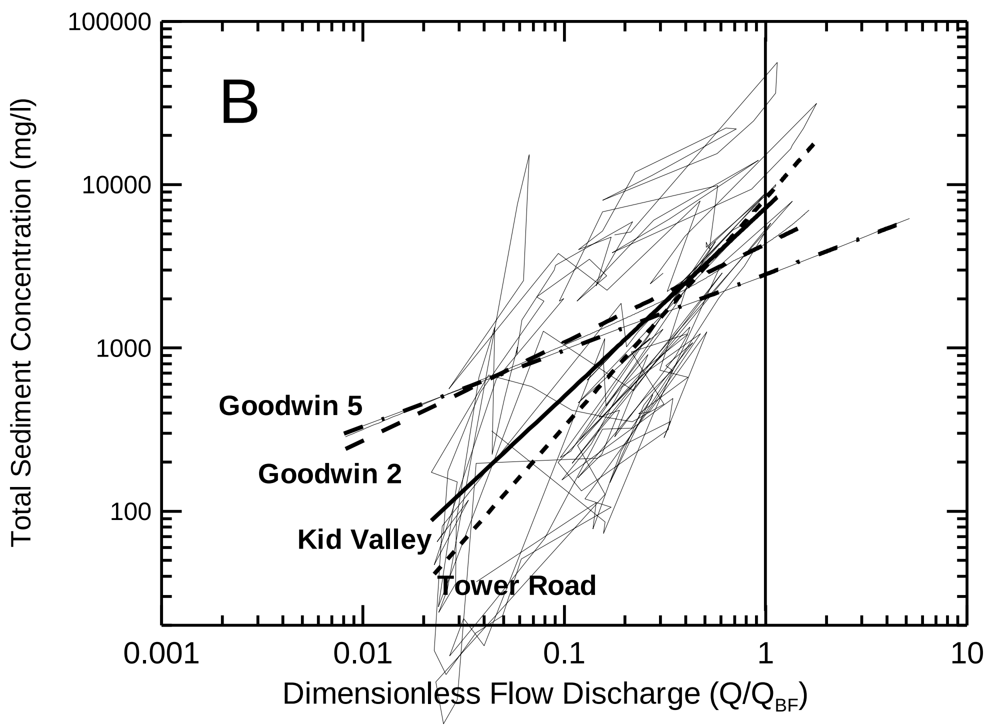


Fig. 3.- Sediment transport rating curves for sites from Mississippi (Goodwin 2 and 5) and from Washington (Kid Valley and Tower Road, Toutle River): (A) concentration versus flow discharge; and (B) concentration versus the ratio of flow discharge to bankfull flow.

sediment movement that is associated with streams in these two phases of evolution could serve as references for streams that have been impacted by excess sediment. Stage VI conditions, which represent re-stabilized streams, in many cases may serve as a more realistic reference for which to target rehabilitation of highly impacted streams.

RECOMMENDATIONS FOR DETECTING DEPARTURE

Our recommendation for detecting departure from stable conditions of a channel can be summarized as follows: 1) Classify the stream using the classification ofRosgen (1996); 2) Determine the stage of channel evolution of the stream (Simon, 1989a); 3) Collect information from the watershed and calculate a landscape index. 4)Use the stage of channel evolution and landscape index to determine the likelihood that the stream is departing from a stable condition. The third and fourth steps in this process are still experimental and need to be verified. The slopes of the sediment relations and the sediment concentration at bankfull flow for the four sites shown in Figure 3b support the hypothesis of the relation between stage of channel evolution and sediment transport. That is Tower Road and Kid Valley (both stage IV) have higher slopes and greater sediment concentrations at bankfull than the two sites from Goodwin Creek which were both stage V's (Kuhnle and Simon, 2000). The hypothesis that stages III, IV and V will show significantly higher sediment concentrations for a given flow than reference streams (stages I or VI) needs to be verified for streams in other physiographic provinces and climatic regions of the country, however. The landscape index and its relation to sediment yield also need to be established. Testing of the relation between the stage of evolution, the landscape index, and the transport of sediment for the different physiographic provinces of the country would provide information regarding the generality of this technique.

THE MISSING LINK: DEPARTURE VERSUS DESIGNATED USE

The determination that a given stream has a significantly higher rate of sediment transport than a corresponding stable reference stream is one facet of the TMDL problem. Another distinct problem is determining the link between excess sediment and a measurable impairment to the designated use of the stream. When the designated or existing use of a stream is defined as the aquatic life, the link between excess sediment and a measurable impairment to the biota needs to be established. Except for a few exceptions, such as for salmonid fish, very little information as to the levels of sediment that are harmful to the biota of a stream is available. In a related project at the USDA-ARS National Sedimentation Laboratory involving several researchers, an initial inquiry into the effect of sediment on the biota in streams is in progress. This study will initially concentrate on thirteen sites located on the Demonstration Erosion Control Watersheds (DEC) in northern Mississippi. These sites currently have data bases of flow, sediment transport, and biological indicators. Geomorphic assessments, landscape data, and additional biological data will be collected at these thirteen sites and added to the data base. The links among the geomorphic data, landscape index, sediment and biological data will be derived for these 13 sites. This will be the first study to our knowledge where these links have been explored in warm-water coastal plain streams. This type of study is also needed for other physiographic provinces of the nation.

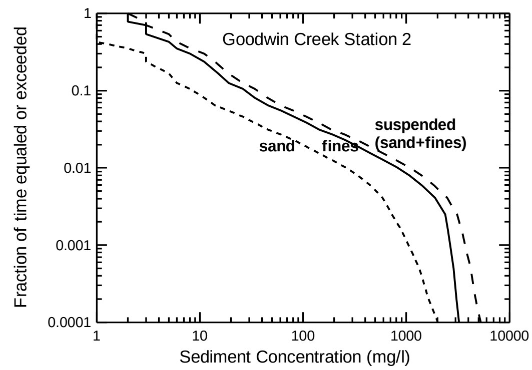


Fig. 4.- Fraction of time equaled or exceeded for suspended-sediment concentration, Goodwin Creek station 2.

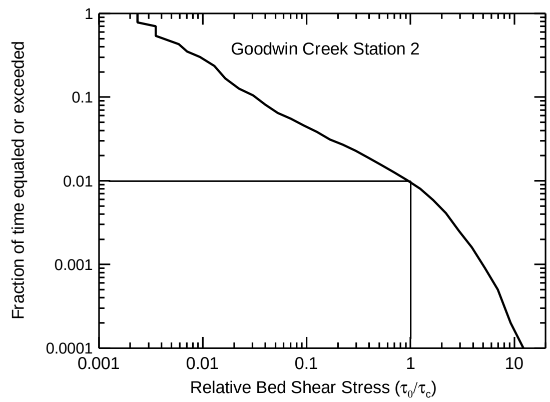


Fig. 5. –Fraction of time equaled or exceeded for relative bed shear stress (ratio of bed shear stress to critical shear stress for motion of the bed material).

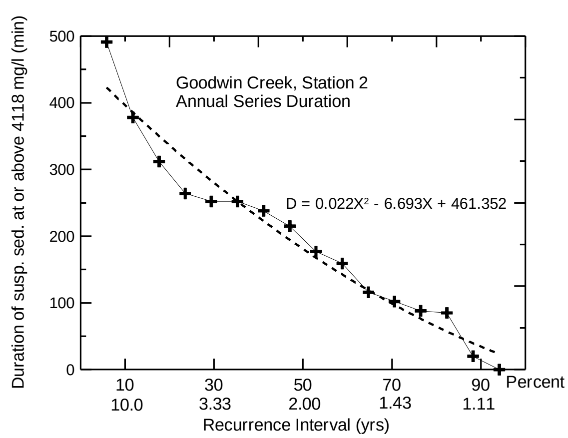


Fig. 6.- Duration of suspended sediment at or above 4118 mg/l as a function of recurrence interval. The concentration 4118 mg/l corresponds to the predicted sediment concentration at bankfull flow (1.1-year recurrence interval).

To make the comparison between sediment and biological impairment, information on the concentration of suspended sediment as well as on the duration and frequency of a given sediment concentration is required. Information of a similar type is also required to assess the stability or movement of the bed material sediment (substrate) of the study sites. Methods to calculate concentration, duration, and frequency of sediment movement using flow and sediment sampling data have been developed. The sediment and flow data from Goodwin Creek station 2 have been used as an example to illustrate these sediment parameters. The fraction of time that the sediment concentration in the water column is equal to or above a given value has been calculated (Fig. 4). Similarly, the fraction of time the bed material of the stream is in motion was calculated (Fig 5). The type of information in Figures 4 and 5 will be used with Goodwin Creek and other DEC sites to determine the ranges of shear stress and/or suspended sediment concentration that will adversely affect the biota. Information on the duration and frequency of the sediment movement events is also needed. Organisms may be able to survive a given concentration of sediment for a limited time without negative effects. Figure 6 contains information on the expected continuous duration of a given sediment concentration or higher (4118 mg/l) for a given recurrence interval. A similar relation for the expected continuous duration for the shear stress above 27 Pa is contained in Figure 7. The information on the fraction of time, duration, and recurrence interval, of sediment concentration and bed shear stress will be combined with biological data collected at the thirteen DEC sites. In addition to this field study, laboratory experiments on the toxicity of sediments to aquatic organisms are in preparation and will provide another important piece of information to the solution of this problem.

SUMMARY

A new methodology has been developed for the evaluation of Total Maximum Daily Loads (TMDLs) for clean sediment in streams and rivers. This methodology consists of the following steps: 1) Classify the stream according to Rosgen (1996); 2) Determine the stage of channel evolution of the stream (Simon, 1989a,b); 3) Collect data on the state of the watershed and calculate a landscape index; 4) Use the stage of channel evolution and landscape index to determine the likelihood that the stream is departing from a stable condition. This procedure has a high likelihood of being successful, although steps 3 and 4 need to be more fully developed and studied. In addition to the 4-step methodology proposed above, parallel studies of the magnitude and duration of increased sediment concentrations and instability of the bed that causes a measurable detriment to the designated use of the stream are also in progress.

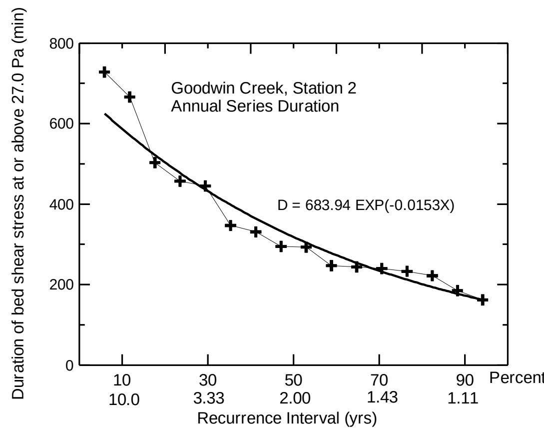


Fig. 7- Duration of bed shear stress at or above 27 Pa. The value of 27 Pa is the stress at which all sizes of the bed material are fully mobilized.

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PROCESS-BASED STREAM-RIPARIAN MODELING SYSTEM TO ASSESS STREAM TMDLS

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Abstract: This paper describes a comprehensive stream-riparian modeling system to evaluate the effects of channel-riparian zones on stream Total Maximum Daily Loads (TMDLs). Nonpoint source pollutants emanating from agricultural fields are major contributors to the ecological impairment of stream channels. Nutrients and sediments are the principle sources of surface water impairment. The edge-of-field system or riparian zone and stream corridor play an important role in the management of sediments and processing of contaminants. A particular challenge we face today is the lack of integrated, comprehensive modeling tools to evaluate Best Management Practices designed to meet proposed TMDL levels for agricultural watersheds. Scientists at the U.S. Department of Agriculture-Agricultural Research Service-National Sedimentation Laboratory (USDA-ARS-NSL) are integrating the ARS Riparian Ecosystem Management Model (REMM) and the CONservational Channel Evolution and Pollutant Transport System model (CONCEPTS). ARS operates field sites in Mississippi and Georgia to study the effects of riparian vegetation on streambank stability and pollutant transport. CONCEPTS has been validated using long-term morphological data from the Goodwin Creek in North-Central Mississippi. The hydrology, soil erosion, and nutrient transport components of REMM have been validated against data obtained at the Gibbs Farm near Tifton, Georgia.

INTRODUCTION

TMDL Development: Section 303(d) of the Clean Water Act requires that states, territories, and authorized tribes identify waters within their boundaries that do not meet water quality standards for their designated use. A TMDL (Total Maximum Daily Load) is a tool for implementing state water-quality standards based on the relationship between sources of pollutants and in-stream water quality conditions. The TMDL establishes the allowable loadings for specific pollutants that a waterbody can receive without violating water quality standards and is defined as:

$$\mathrm{TMDL} = \mathrm{LC} = \sum \mathrm{WLA} + \sum \mathrm{LA} + \mathrm{MOS}, \qquad (1)$$

where LC is loading capacity or the maximum amount of pollutant loading a waterbody can assimilate without violating water quality standards, WLA is wasteload allocation or the portion of the TMDL allocated to existing or future point sources, LA is load allocation or the portion of the TMDL allocated to existing or future nonpoint sources, and MOS is margin of safety. MOS accounts for the uncertainty about the relationship between pollutant loads and receiving water quality.

The development of TMDLs can be complicated because of the lack of adequate or proven tools or information on the fate, transport, and impact of each pollutant within the natural system. The U. S. Environmental Protection Agency (EPA) is therefore developing TMDL protocols to provide guidance on TMDL development. EPA (USEPA, 1991) divides the development of

TMDLs into seven components: (1) problem identification, (2) identification of water quality indicators and target values, (3) source assessment, (4) linkage between water quality targets and sources, (5) allocations, (6) follow-up monitoring and evaluation plan, and (7) assembling the TMDL.

During source assessment (step 3), the sources of loading for the pollutant of concern are identified and characterized by type, magnitude, and location. For each TMDL, a linkage between the selected indicator(s) and target(s) and the identified sources must be defined (step 4). This linkage establishes the cause-and-effect relationship between pollutant sources and the in-stream pollutant response and allows for an estimation of loading capacity. Consequently, pollutant loadings that will not exceed the loading capacity can be determined. These pollutant loadings are distributed among the significant sources of the pollutant (step 5), see also equation (1).

Sediment Pollution: Nonpoint source pollution is the major cause of surface water impairment in the United States (USEPA, 1998b). The nonpoint sources of pollution vary, but nutrients and sediments are the principle sources of surface water impairment. This paper, however, mainly focuses on tools to assess sediment TMDLs. Sediment is a vital natural component of waterbodies and the uses they support. Excessive amount of sediment can adversely impact aquatic life and fisheries. Excessive sediment deposition can choke spawning gravels, impair fish food sources, and reduce habitat complexity in stream channels. Excessive suspended sediments can make it more difficult for fish to find prey and at high levels can cause direct physical harm. Stream scour can lead to destruction of habitat structure. Sediments can cause taste and odor problems for drinking water, block water supply intakes, foul treatment systems, and fill reservoirs. High levels of sediment can impair swimming and boating by altering channel form, creating hazards due to reductions in water clarity, and adversely affecting aesthetics. Figure 1 shows the sedimentation processes across a landscape. Sediment production can occur both on hillslopes by surface erosion, gully erosion, or mass wasting, and in the channel through bank erosion and gross degradation of the channel bed. Sediments delivered to the stream channel move downstream. They often go through cycles of storage and removal.

USEPA (1999) categorizes sediment TMDL indicators as: (1) water column indicators, e.g., suspended sediment, bedload sediment, and turbidity; (2) streambed sediment indicators, e.g., particle size distribution and substrate properties, (3) other channel indicators, e.g., pool/riffle ratios, sinuosity, bank stability, and width/depth ratios; (4) biological and habitat indicators, e.g., presence, diversity, and productivity of invertebrate and fish species; and (5) riparian/hillslope indicators, e.g., riparian buffer width sizes and riparian vegetation character, large woody debris, and landslide area. Tools used in the TMDL development process must be capable of representing these indicators.

Watershed Approach: To address the combined, cumulative impacts of both point and nonpoint sources, EPA has adopted the same watershed approach that parallels those pioneered by the U. S. Department of Agriculture (USDA)-Agricultural Research Service (ARS) and Natural Resources Conservation Service (NRCS). EPA’s watershed approach is a coordinating framework for environmental management that focuses public and private sector efforts to

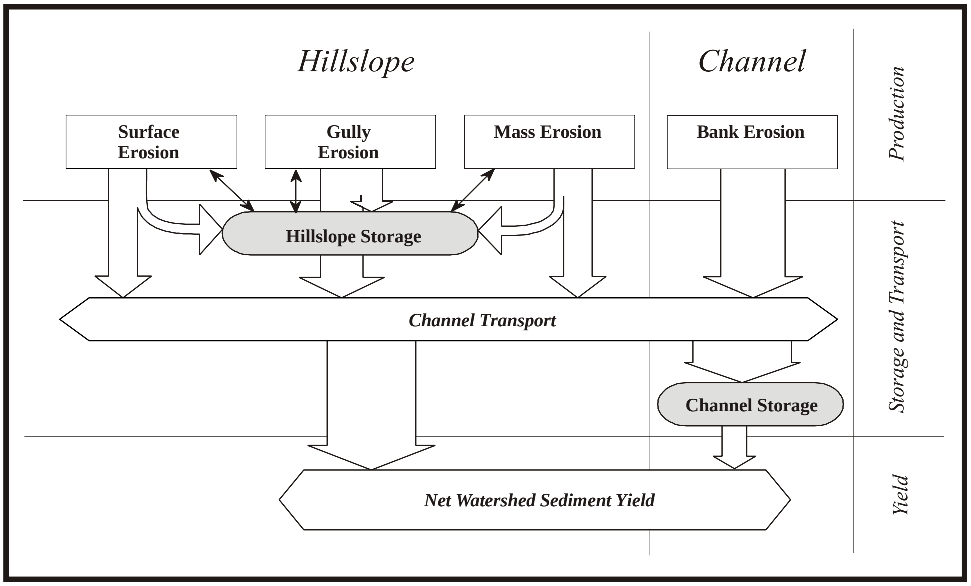

Figure 1 Sedimentation process (after USEPA, 1999).

address the highest priority problems within hydrologically-defined geographic areas, taking into consideration both ground and surface water flow (USEPA, 1996). This approach provides a means to integrate governmental programs and improve decision making by both government and private parties. This approach enables a broad view of water resources that reflects the interrelationship of surface water, groundwater, chemical pollutants and nonchemical stressors, water quantity, and land management. Accordingly, EPA has developed a system, BASINS (Better Assessment Science Integrating Point and Nonpoint Sources) (USEPA, 1998a), to meet the needs of agencies that develop TMDLs. BASINS addresses three objectives: (1) to facilitate examination of environmental information, (2) to provide an integrated watershed and modeling framework, and (3) to support analysis of point and nonpoint source management alternatives. BASINS is a sophisticated package comprising various EPA water quality models. However, BASINS has limitations when used to develop sediment TMDLs. Especially, if sediment delivery and in-stream processes are important.

ARS is developing technologies to characterize the movement of water and any associated constituents on agricultural watersheds. These technologies include models needed to rehabilitate degraded landscapes, stream corridors, and aquatic ecosystems. The NRCS and ARS have developed the AGNPS 98 suite of models and the Riparian Ecosystem Management Model (REMM) in partnership with other organizations. Integration of these technologies allows studies and assessments to be performed on the hydraulic, geomorphic, and biologic interactions between a stream, the riparian zone, and adjacent farmland.

MODELING APPROACH

EPA's TMDL protocols emphasize the use of rational, science-based methods and tools for TMDL development. "The availability of data influences the types of methods analysts can use. If long-term monitoring data are lacking, the analyst will have to use a combination of monitoring, analytical tools (including models), and qualitative assessments to collect

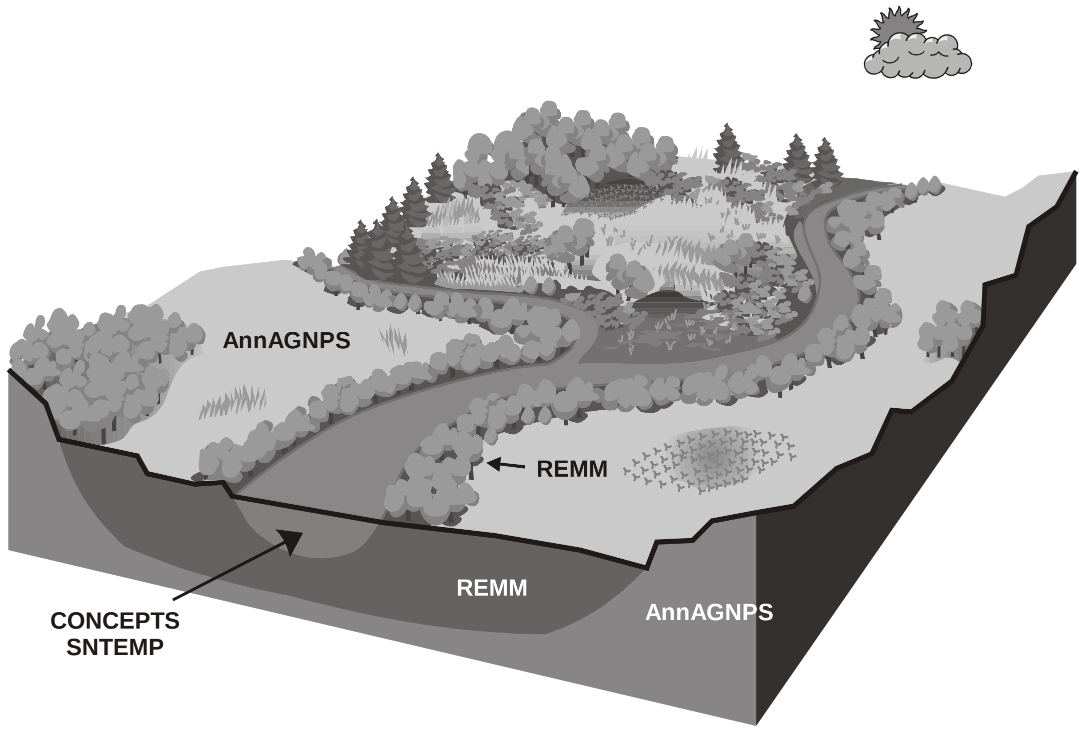

Figure 2 Landscape scales applicable to the ARS AGNPS 98 and REMM models.

information, assess system processes and responses, and make decisions. Although some aspects of TMDLs must be quantified (e.g., numeric targets, loading capacity, and allocations), qualitative assessments are acceptable as long as they are supported by sound scientific justification or result from rigorous modeling techniques" (USEPA, 1999).

The AGNPS 98 suite of models and REMM can be used in steps 3 through 5 of the TMDL development process. Figure 2 shows a view of the landscape scales applicable to the various models.

AGNPS 98: The AGricultural NonPoint Source pollution model 98 (AGNPS 98) is a joint NRCS-ARS system of computer models developed to predict nonpoint source pollutant loadings within agricultural watersheds (Bingner, 2001). The set of computer programs consist of: (1) a GUI for input generation and editing as well as associated data bases, (2) the annualized science and technology pollutant loading model AnnAGNPS, (3) output reformatting and analysis, (4) the integration of comprehensive routines for in-stream processes (CONCEPTS), (5) an in-stream water temperature model (SNTEMP); and (6) several related salmonid models (SIDO, Fry Emergence, Salmonid Total Life Stage, and Salmonid Economics).

AnnAGNPS: The Annualized AGricultural NonPoint Source pollution model (AnnAGNPS) (Cronshey and Theurer, 1998) is a continuous simulation, daily time step, watershed scale, pollutant loading model. AnnAGNPS analyzes a watershed divided into subareas of homogeneous landuse management, climate, and soils, which can adequately approximate site conditions. Runoff, sediment, and other pollutants are routed from each subarea through a channel network, including surface water impoundments, to the outlet of the watershed.

AnnAGNPS uses the Revised Universal Soil Loss Equation (RUSLE) (Renard et al, 1997) to predict soil erosion from agricultural landscapes.

REMM: REMM is a computer simulation model of riparian forest buffer systems (Lowrance et al, 1998). The riparian buffer consists of three zones between the field and stream. Each zone includes litter and three soil layers that terminate at the bottom of the root system, and a plant community that can include six plant types in two canopy levels. Surface hydrology, erosion, vertical and horizontal subsurface flows, carbon and nutrient dynamics, and plant growth that occur in each zone are modeled on a daily time step.

CONCEPTS: The CONservational Channel Evolution and Pollutant Transport System (CONCEPTS) is a distributed, long-term channel evolution and water quality model for use in ungaged watershed systems (Langendoen, 2000). The basic components are channel hydraulics, stream morphology, and transport of sediments. Integration of CONCEPTS and AnnAGNPS provides a means to model the evolution of large-scale channel systems in disturbed landscapes such as those in the Demonstration Erosion Control Project, Yazoo River Basin, Mississippi (Langendoen and Bingner, 1998).

The above suite of models covers the entire sedimentation process across a landscape (Figure 1). AnnAGNPS simulates hillslope erosion (sheet, rill, and gully erosion) and delivery processes. REMM simulates the storage of sediments alongside stream channels due to riparian buffers. CONCEPTS accounts for channel sources such as bank collapse, in-channel storage, bedload and suspended sediment transport, and net sediment yield from the watershed. During source assessment, step 3 of the TMDL development process, one uses the models to characterize the types, magnitudes, and locations of sources of sediment loading to the waterbody (USEPA, 1999). The results can be used to connect excess sediment load at a point of impact to sources of sediment generation and can thus be used to target load reductions (steps 4 and 5 of the TMDL development process). Further, one can use the models to evaluate sediment BMPs, such as landuse management alternatives, riparian buffers, and in-stream grade control structures.

STREAM-RIPARIAN CORRIDOR

The stream-riparian corridor is the conveyor of pollutants through the watershed. It determines the short- and long-term fate of pollutants both on-site and off-site. Riparian buffers or forests have well-known beneficial effects on bank stability, biological diversity, and water temperature of streams (Karr and Schlosser, 1978). Lowrance et al (1985) showed that riparian forests effectively reduce nonpoint source pollution from agricultural fields. A riparian buffer is a well-established BMP to reduce nonpoint source pollution. The stream-riparian corridor can also be a producer of sediments. Many streams in the US have been channelized for flood control often leading to incision with increased sediment transport rates, bank collapse, and habitat degradation. Plans to restore or rehabilitate the stream corridor are then required together with technology to assess or evaluate these plans.

Riparian Zone: Welsch (1991) specified guidelines for a riparian buffer system with three zones. Zone 1 is permanent vegetation immediately adjacent to the streambank. Zone 2 is managed forest occupying a strip upslope from zone 1. Zone 3 is an herbaceous filter strip upslope of zone 2. The primary function of zone 3 is to remove sediment from surface runoff and to convert channelized flow to sheet flow. The primary function of zone 2 is to block transport of sediment and chemicals from upland areas into the adjacent wetland or aquatic ecosystem. The primary function of zone 1 is to maintain the integrity of the streambank and a favorable habitat for aquatic organisms.

ARS has developed REMM to provide a tool to assess the nonpoint source pollution control functions of riparian buffer systems (Lowrance et al, 1998). To assess sediment TMDLS, REMM simulates soil and channel erosion, and sediment transport by particle size class (clay, silt, sand, and small and large aggregates) within riparian buffer systems (Bosch et al, 1998). Bosch et al (1998) tested REMM using a data set collected at a riparian site at the University of Georgia Gibbs farm near Tifton, GA. The soil in the riparian forest is an Alapaha loamy sand on a 2.5% slope. The riparian buffer consists of a 8 m long grass filter in zone 3, a 50 m long managed pine forest in zone 2, and a 10 m long hardwood forest in zone 1. Figure 3 compares computed and simulated yields leaving each zone. Annual predicted sediment yields entering zone 2 were approximately double the observed value, while yields entering zone 1 were somewhat less than observed. Overprediction of the simulated runoff caused the larger, predicted sediment yields entering zone 2.

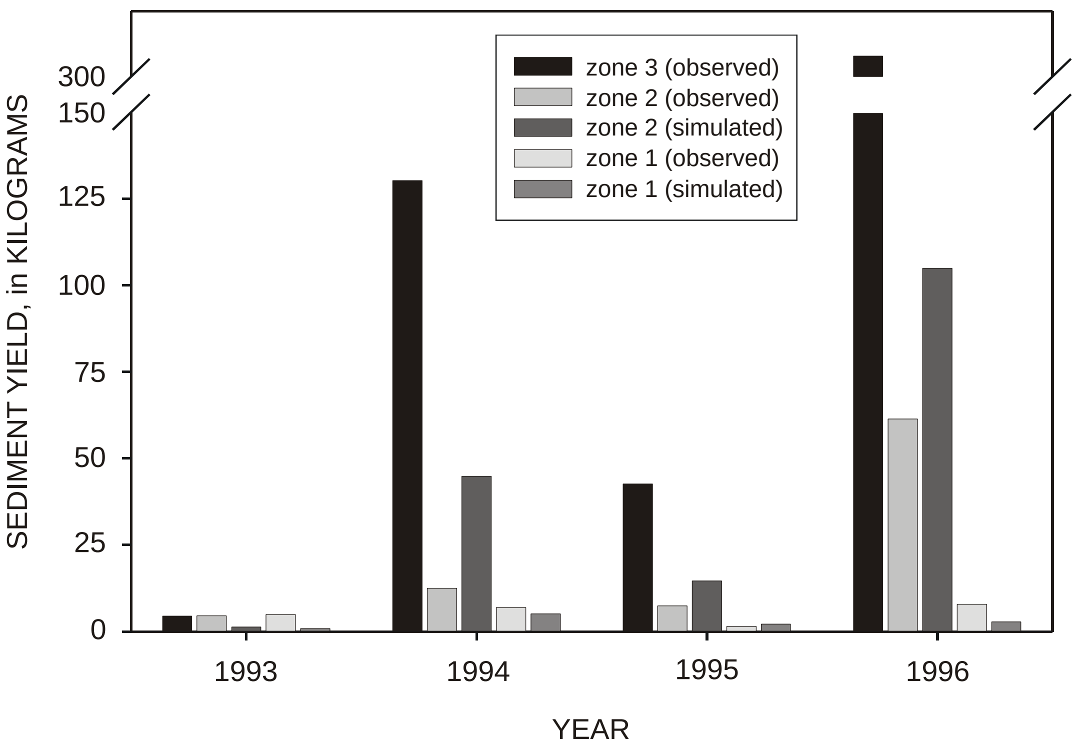

Figure 3 Comparison of observed and predicted sediment yields entering the three zones of a riparian forest system at the Gibbs Farm near Tifton, GA (after Bosch et al, 1998).

<u>Stream Channel:</u> Shields et al (1999) discussed various treatments to stabilize incised stream corridors. At the reach scale, restoration plans include re-alignment of the channel and bed and bank stabilization works, among others. Local, in-stream controls can be classified as 'hard' structures or vegetative treatments. Examples are drop structures, spur dikes, and large woody debris structures.

ARS has developed CONCEPTS to evaluate watershed scale, reach scale, and local control stream-corridor restoration projects. Using CONCEPTS, Langendoen and Bingner (1998) show that a system of 14 grade control structures in the stream network of the Goodwin Creek Watershed, MS, stabilizes the stream system and consequently reduces sediment yield. Langendoen et al (1999) showed the capabilities of CONCEPTS to simulate streambank failure processes. CONCEPTS accurately predicts timing and dimensions of bank failures along a bendway in the Goodwin Creek, Mississippi (Figure 4).

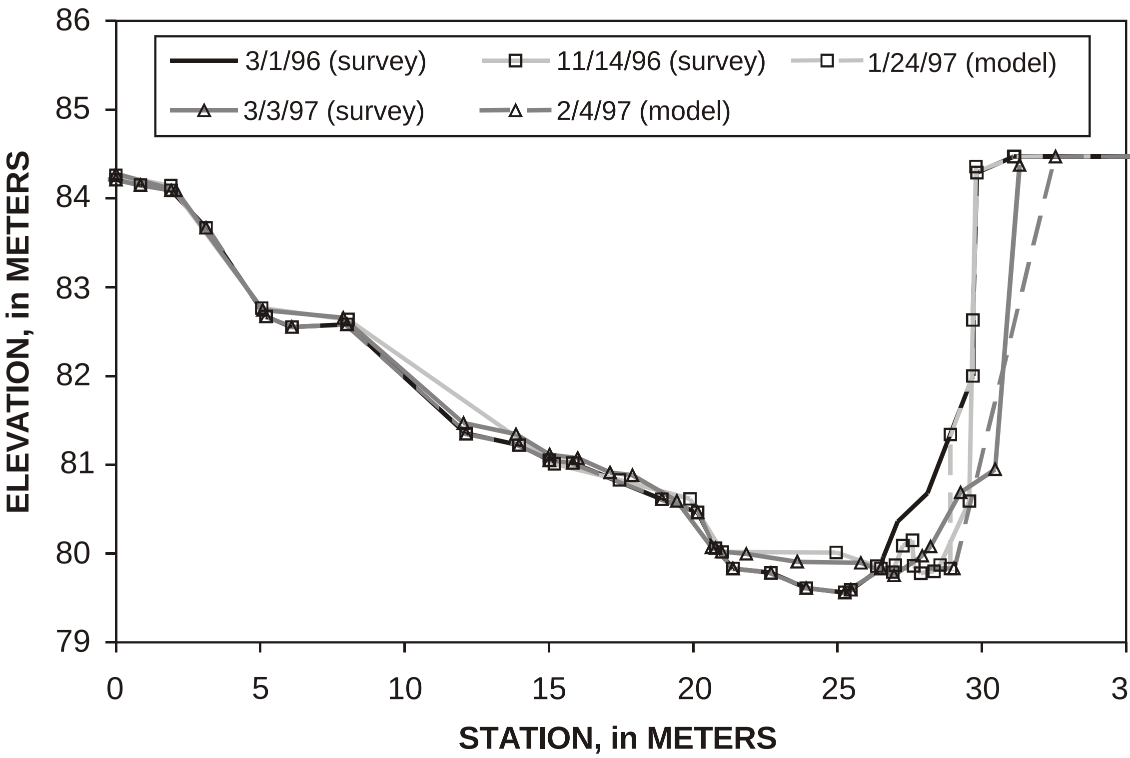

Figure 4 Comparison of surveyed and simulated profiles of a cross section located at the apex of a bendway of the Goodwin Creek, MS, between March 1996 and April 1997.

SUMMARY

The application of integrated watershed analytical tools provides a science-based foundation in the development of TMDLs. Without these tools, accurate assessments for ungaged watersheds will be very difficult.

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DRAINAGE DITCHES: NEW CONCEPTUAL BMPs FOR NON-POINT SOURCE POLLUTION AND TMDL DEVELOPMENT

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Abstract: During recent decades, researchers and managers have sought new techniques to reduce potential contamination from agricultural runoff. Most scientists have overlooked a valuable mitigation tool and best management practice (BMP) within the agricultural production landscape – drainage ditches. Historically, drainage ditches have primarily served as modes of water transport from fields following storm events or controlled releases. While performing this task, ditches have likewise been involved in agricultural contaminant transfer and transformation. That is to say, agricultural drainage ditches have the potential for significant mitigation of sediments, pesticides, and nutrients. Few, if any, recommendations for ditch management exist in the United States. Farmers manage their drainage ditches according to personal preferences and needs. Often times, they either dredge the ditches (thereby completely removing vegetation), or they cut most vegetation with mowers to allow better drainage. While water transport is the first and foremost objective of agricultural drainage ditches, our research has shown that herbaceous vegetation, allowed to remain relatively undisturbed within the ditch, has a tremendous capacity to sorb pesticides typically associated with storm water runoff. Most research conducted with drainage ditch systems in the U.S. has focused on monitoring concentrations of nutrients associated with movement of subsurface flow and tile drainage within ditches. We are advocating the use of drainage ditches as buffers for mitigation of agricultural contaminants. Through their mitigation capabilities, ditches will play an important role in the development of total maximum daily loads (TMDLs) for a variety of potential agricultural contaminants. By carefully designing our experiments, we are examining methods from which to offer ditch design parameters to farmers (e.g. length of vegetated ditch necessary to mitigate against some contaminant). In addition to examining chemical fate and ecotoxicological benefits of ditches, we are also exploring the macroinvertebrate and microbial assemblages located within these unique ecosystems, in order to better understand their functions and potential values. While this conceptual proposal of ditches as a mitigation tool has wide applicability, it is not without certain limitations. Scaling among ditches is an important issue. Contributing area (to ditches) is also a crucial variable. Current research focuses on individual farm scale ditches (i.e. those ditches immediately surrounding the agricultural field). Our data suggest that in some form, using vegetated agricultural drainage ditches as a BMP will offer farmers a low-cost, low-maintenance solution to help combat increasing concerns with the quality of non-point source pollution.

INTRODUCTION

Water quality issues have been at the forefront of public concern and legislative agendas for several decades. Such issues arise from both historical and current landscape-scale water use. Because agriculture affects a great deal of landscape alteration and water use, it has been inherent that agriculture take a leading role in both ground and surface water quality research (Cooper, 1990). The greatest pollutants, by volume, affecting water quality are instream suspended sediment and bedload materials (Fowler and Heady, 1981; Cooper and Knight, 1991). Other potential agricultural pollutants transported during runoff events include nutrients, pesticides, and bacteria. Once these pollutants enter receiving water bodies, such as rivers, lakes, and streams, they have the potential to adversely affect the established flora and fauna of the ecosystem.

In order to prevent potential agricultural pollutants from entering such aquatic receiving systems, focus has been placed on designing and implementing both in-field and edge-of-field best management practices (BMPs). Examples of suggested BMPs to decrease potential agricultural runoff impacts to receiving systems include the use of winter cover crops, conservation tillage (reduced or no-till), constructed wetlands, stiff grass hedges, riparian zones, grass filter strips, and other vegetative barriers. Performance of grass buffer strips for nutrient and sediment reduction has been well documented (Barling and Moore, 1994; Lee et al., 1989; Hayes et al., 1984; Hayes and Hairston, 1983; Barfield et al., 1979, 1977; Hayes et al., 1979a,b,c; Butler et al., 1974). The capacity of wetlands and other riparian areas to intercept and remediate nutrient and sediment-laden waters has likewise been well established (Barling and Moore, 1994; Brinson, 1993; Jordan et al., 1993; Pinay and Decamps, 1988; Peterjohn and Correll, 1986; Jacobs and Gilliam, 1985; Lowrance et al., 1984). Nitrate removal of 99% within the first five meters of a poplar vegetated riparian zone was reported by Haycock and Pinay (1993), while Chescheir et al. (1991) reported that 90% of sediment associated with agricultural stormwater runoff was removed by treatment using a forested wetland. Many of the above-mentioned BMPs require some minimum construction efforts (e.g. constructed wetlands, grass hedges and filter strips) in order to become established and functional. This research suggests that a valuable mitigation tool and potential new BMP already lies within the agricultural production landscape– agricultural drainage ditches. Historically, drainage ditches have served primarily as methods of water transport from fields following storm or controlled irrigation release events. During this process, ditches have likewise served as sites for transfer and transformation of potential agricultural contaminants. If indeed agricultural drainage ditches effectively mitigate contaminants such as nutrients, sediments, and pesticides, this will have significant implications in the development of practical, realistic total maximum daily loads (TMDLs).

MATERIALS AND METHODS

Two controlled-release experiments simulating pesticide runoff following a storm event have been conducted since 1998. Both experiments were conducted within the Mississippi Delta Management Systems Evaluation Area (MDMSEA) (Figure 1). In 1998, a 50 m portion of a vegetated agricultural drainage ditch within the Beasley Lake watershed was amended with a mixture of the herbicide atrazine (2-chloro-4-ethylamino-6-isopropylamino-s-triazine) and the insecticide lambda-cyhalothrin [8-cyano-3-phenoxybenzyl-3-(2-chloro-3,3,3-trifluoro-1-enyl)-

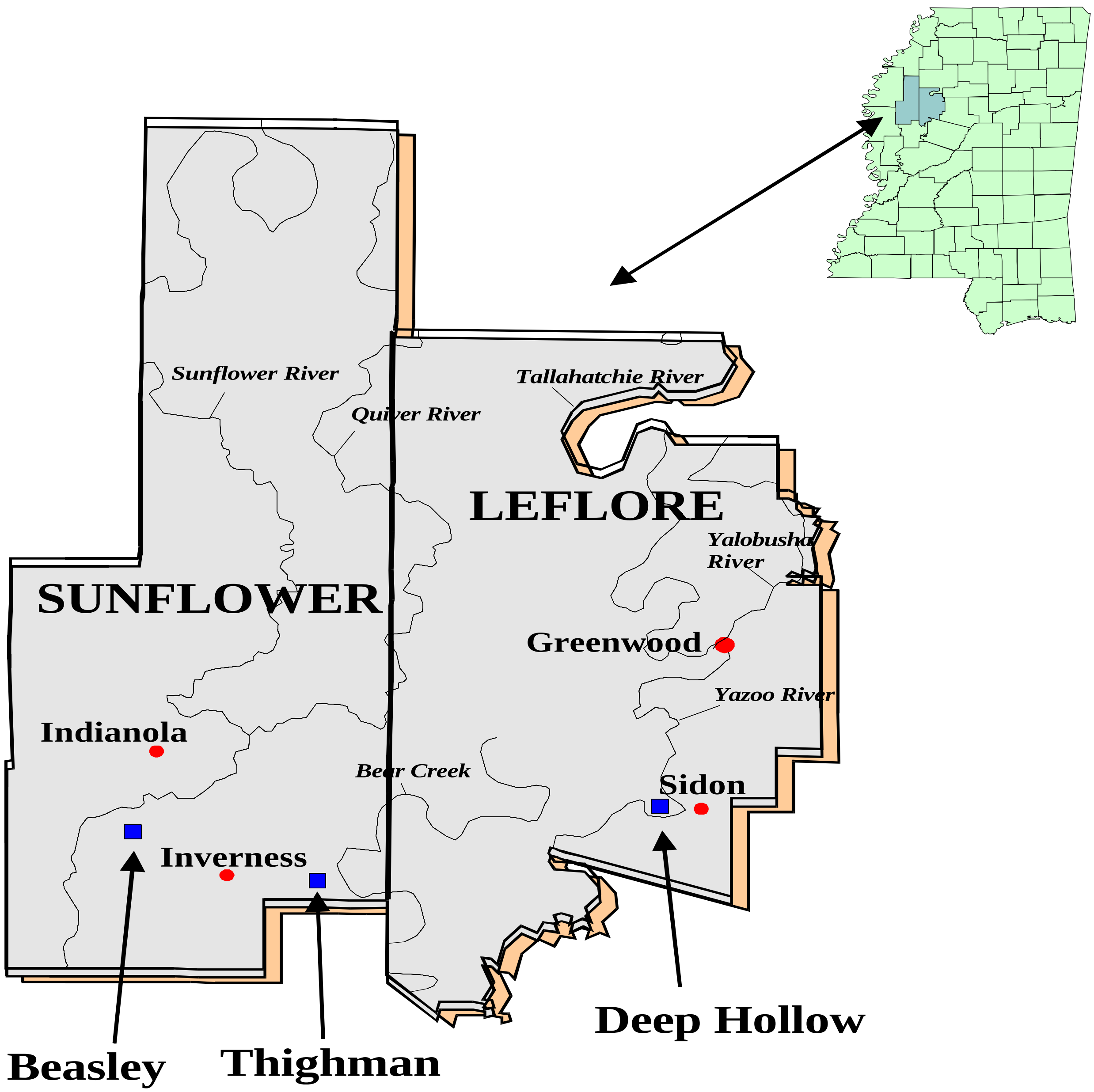

Figure 1. Mississippi Delta Management Systems Evaluation Area (MDMSEA) watersheds.

2,2-dimethyl cyclopropanecarboxylate] simulating pesticide runoff from a 10 ha field. Dominant aquatic plant species present in the ditch included *Polygonum* sp. and *Leersia* sp. The 1999 study was conducted on a 650 m vegetated agricultural drainage ditch leading into Thighman Lake, with a mixture of two pyrethroid insecticides, lambda-cyhalothrin and bifenthrin [[2 methyl(1,1'-biphenyl)-3-yl] methyl 3-(2-chloro-3,3,3-trifluoro-1-propenyl)-2,2-dimethyl-cyclopropanecarboxylate], simulating pesticide runoff from a 20 ha field. *Ludwigia* sp. and *Lemna* sp. were the dominant aquatic flora in the Thighman drainage ditch. For both studies, pesticide concentrations were based on label application rates and potential pesticide runoff percentages (Wauchope, 1978). Samples of water, sediment, and plants were collected spatially and temporally in both studies and analyzed according to Bennett et al. (2000) and Moore et al. (in review) for the presence of targeted pesticides.

RESULTS

Beasley Lake ditch: Vegetation within the 50 m portion of the agricultural drainage ditch was capable of mitigating 59-61% of the measured atrazine during the first 24 h following initiation of the simulated storm runoff. Fourteen days following the simulation, aqueous concentrations of atrazine at all five monitored sites were below the ecosystem toxicological threshold (20 ug/L) (Huber, 1993). Samples collected at the study' s conclusion (28 d) indicated than an average of 86% of measured atrazine was associated with plant material. Mean percentages of measured atrazine in water, sediments, and plants (from the end of the simulated storm event to the study conclusion) were 15 ± 24%, 28 ± 23%, and 57 ± 21%, respectively.

Approximately 97% of the measured lambda-cyhalothrin was associated with plant material only 3 h following the initiation of the storm event. Samples collected 28 d following the runoff simulation indicated that the remaining 3% of lambda-cyhalothrin was associated with the sediment. Likewise, 28 d following the simulation, samples collected at each of the five monitored sites indicated aqueous concentrations of lambda-cyhalothrin were at or below the suggested toxicological threshold of 0.02 ug/L (EXTOXNET, 1996). Mean percentages of measured lambda-cyhalothrin in water, sediment, and plants (from the end of the simulated storm event to the study conclusion) were 1 ± 1%, 2 ± 1%, and 97 ± 0.4%, respectively. Maximum aqueous concentrations measured both spatially and temporally indicated that, based on the assumptions associated with the experimental design (10 ha contributing area with estimated 5% atrazine and lambda-cyhalothrin runoff), both atrazine and lambda-cyhalothrin could be mitigated within 50 m of an agricultural ditch of similar physical dimensions.

Thighman Lake ditch: Three hours following the initiation of the simulated storm event, 96% of the measured lambda-cyhalothrin was associated with aquatic plant material, while the remaining 4% was associated with the ditch sediment. Samples collected at 12 h, 24 h, and 14 d indicate similar findings with 94 – 99% of measured lambda-cyhalothrin being associated with aquatic plant material in the ditch. For the entirety of the study (99 days), mean percentages of lambda-cyhalothrin associated with water, sediment, and plants were 1 ± 1%, 12 ± 16%, and 87 ± 16%, respectively.

Similar results were seen with concentrations of bifenthrin. Ninety-nine percent of the measured bifenthrin was associated with aquatic plant material, three hours following initiation of the simulated storm event. Indications from samples collected at 12 h, 24 h, and 14 d were that 98 – 99% of the measured bifenthrin was associated with aquatic plant material. Overall mean percentages of measured bifenthrin associated with water, sediment, and plants were 1 ± 0.5%, 18 ± 28%, and 81 ± 28%, respectively.

DISCUSSION

Current national water quality interests heavily emphasize the development and implementation of TMDLs (US EPA, 2000). Of great concern to agriculture and other parties not classified under the National Pollutant Discharge Elimination System (NPDES) permit process, is the scientific research used in the TMDL process. Poor, antiquated, unrealistic assumptions of non-point source pollution inputs due to agriculture threaten the continuance of proficient food and fiber production. Cooper (1991) conducted an intensive study in Moon Lake, Mississippi, which is located within an intensively cultivated watershed. Sporadic detection of permethrin was reported immediately after the spray season in soil, surface water, fish, and sediments, but was rapidly degraded and non-detectable by late fall. Knight and Cooper (1996) reported no permethrin in soil or water as part of a study within a mixed-cover hill land watershed, and only low concentrations in sediment and fish tissue (0.03 ng/g and 0.11 ng/g, respectively). Although these concentrations are extremely low, it continues to indicate the need for agriculture to strive for improvement in decreasing pesticide contaminants in runoff.

Many BMPs assume that the "spring flush" will be the timeframe for maximum pesticide concentrations in runoff. Coupe et al. (1998) reported that, due to a longer growing season in the Mississippi Delta, different pesticide applications for a variety of crops may result in pesticide concentrations being detected in surface water from April until August. Methyl parathion has also been reported to persist throughout the fall and into winter in sediment and fish (Cooper, 1991). Thus, a better approach to pesticide runoff mitigation would encompass the entire year, rather than focusing on only one season. The research described within this paper is a step toward providing baseline information on a proposed new BMP, agricultural drainage ditches. Within the two experimental designs, both ditches were capable of mitigating their respective pesticides (atrazine and lambda-cyhalothrin; lambda-cyhalothrin and bifenthrin) to or below toxicity threshold concentrations within the allotted distances. Results also provided evidence of the important role of vegetation in these agricultural drainage ditches (57 – 97% of measured pesticides associated with plants). By maintaining these ditches as potential runoff mitigation routes, agriculture can capitalize on their current use in the production landscape, while avoiding expensive construction costs associated with some other types of BMPs. Additional comprehensive studies are ongoing.

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A RANK-BASED APPROACH TO INTERPRETING SYSTEMATIC SAMPLING PROGRAM RESULTS USING REFERENCE LOADING CURVES

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INTRODUCTION

Systematic sampling involves collection of data at predetermined, usually evenly spaced intervals (Gilbert, 1987). As a water sampling strategy, it represents an affordable way of collecting information about a hydrologic system. It is useful for evaluating long-term trends, provided that sample collection intervals are sufficient to avoid bias introduced by over- or under- representing significant factors that may influence the parameters being measured in the sample. As an alternative to summary statistics for data analysis, we applied plotting positions to extract information from the systematic sampling record. Plotting positions are often used for preliminary flood frequency analysis to summarize and interpret flood flow data without imposing assumptions of specific distributions (Yevjevich, 1972). These take the general form of:

$$p^* = (i-\alpha)/(N+1-\alpha) \quad (1.1)$$

with i representing the rank of the observation in a data set, N the total of number of observations and α a constant, selected to conform with specific distributional assumptions (Cunnane, 1978). This approach is also referred to as the Weibul formula. Plotting positions are often used for empirical analysis of extreme events, to determine recurrence intervals, which are expressed as $1/F(x)$, with $F(x)$ representing an empirical distribution function of the variable under consideration (Cleveland, 1993). $F(x)$ for a specified value is expressed as a quantile p^*, which represents the probability that any value selected randomly from the data set will be less than or equal to the value under consideration ($P(X \leq x) \approx p^*$).

The plotting position approach can also be used to provide a reference loading criterion in the absence of a formally developed TMDL. Reference loading is based on ambient water quality standards and is developed using the plotting position approach. Observed flow data from the sampling record are transformed to daily loading rates, using ambient water quality standards as multipliers. When estimated loading rates, based on ambient water quality standards, are plotted against plotting position values, the resulting distribution defines a *reference loading curve*. Analysis takes place on a daily time step, for a full series analysis, which makes use of the entire record.

We then take the same approach with pollutant concentration data observed from systematic sampling results. When plotted with reference loading curves, the observed loading rates present a visual summary of systematic sampling data and can be used to estimate load reductions necessary to meet reference load criteria at specific quantiles of the systematic sampling data set. This allows planners to evaluate a potential TMDL with explicit performance standards and

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assess the levels of reduction needed to conform with loading rates implied by the ambient water quality standard. The procedure indicates the significance of failure, in terms of expected loading, when loading rates exceed specified performance standards.

Application to a river and stream in Northern Nevada

We applied the approach described above to two different drainage systems on two different geographic scales in Northern Nevada, which have decade-long records of systematic sampling results. Steamboat Creek (above U.S.G.S. gaging station 10346680) and the Carson River, (above U.S.G.S. gaging station 10311400) drain 632 and 8080 km^2 (244 and 958 mi^2), respectively (FIG. 1). The Nevada Division of Environmental Protection (NDEP) is developing TMDL for the Carson River, which is affected exclusively by nonpoint sources of phosphorus and sediment. Each has ambient water quality standards that correspond with specific beneficial uses and each has been systematically sampled for more than a decade, using a fixed schedule and location approach. Each is also fairly complex, because both natural and managed sources affect water quality and irrigation diversions and return flows complicate analyses of seasonal loading dynamics.

Steamboat Creek and Carson River, NV: Physical Descriptions

Steamboat Creek flows northeast approximately 18 miles from its source, Washoe Lake, and is tributary to the Truckee River (FIG. 1). It is the principal drainage for Washoe and Pleasant Valleys and the Truckee Meadows, an area that includes Reno, NV. The basin includes 632 km^2 (244 mi^2), which receives flow from a number of small watersheds from the eastern side of the Sierra Nevada Mountain, including Galena, Whites and Thomas Creeks. From 1987 to 1998 the Nevada Department of Environmental Protection carried out a systematic sampling program at three U.S.G.S. flow-gaging stations (gages 10349300, 10349980 and 10348800). The program collected samples and corresponding instantaneous flow measurements on a bimonthly schedule. Water samples were submitted to an EPA certified laboratory (the Nevada State Health Laboratory at the University of Nevada) and analyzed for a wide range of parameters, including total suspended solids (TSS) and total phosphate, as phosphorus (TP) using EPA methods 160.2 and 4500PE, respectively. Estimated flow rates reported with sample results correspond with those recorded at the time of sampling at the U.S.G.S. gaging stations.

The Carson River watershed, located in Northwestern Nevada, encompasses an area of approximately 8080 km^2 (3,966 mi^2) (FIG. 1). The river rises in the Eastern Sierra Nevada mountains in two main tributaries, the East and West Forks of the Carson River. The average yearly flow into the West Fork is 99.86 Mm^3 (80,990 acre-ft) per year, as recorded by U.S.G.S. gage 10310000 near Woodfords, CA (see TABLE 1 for description of gages on the upper watershed of the Carson River, including contributing watershed areas).

The Nevada Division of Environmental Protection's Bureau of Water Quality Planning conducted systematic sampling at nine U.S.G.S. gaging stations from 1987 to 1998 as part of efforts to establish water quality standards, to meet requirements of the Clean Water Act (sect 106). Data consist of instantaneous concentration and daily flow measurements collected at monthly (1987 - 1995) and bi-monthly (1995 - 1998) intervals. These include measurements of total phosphorus and total suspended sediment concentrations, carried out by the Nevada State Health Laboratory, using the methods referred to for Steamboat Creek.

Application: Reference and Observed Loading Curves for Steamboat Creek and the Carson River

To develop reference loading curves for Steamboat Creek, which has no standards for TSS and TP, we applied concentration-based standards for the Truckee River, set for a control point approximately 2.2 km (1.3 miles) upstream of the confluence of the Truckee River and Steamboat Creek. The standards for total phosphates (TP - annual average concentration) and total suspended solids (TSS - annual average concentration) are set to meet beneficial uses (TP) or maintain current water quality (TSS). Concentration standards for TP and TSS for the Truckee at the upstream control point are 0.1 and 15 mg/l, respectively(N.D.E.P., 1998).

We applied concentration-based standards for the Carson River, published by the Nevada Division of Environmental Protection in 1982 (Horton, 1997), to develop reference loading curves. The standards for total phosphorus (≤0.10 mg/l) and total suspended solids (≤80 mg/l) are set to protect aquatic life, (primarily cold water fisheries) and to support municipal or domestic supplies, irrigation and stock watering (N.D.E.P., 1998). Reference and observed loading curves for total suspended solids loading rates (kg/day) estimated for Steamboat Creek and the Carson River are included as FIG. 2 and 3, respectively. Reference and observed loading curves for total phosphorus loading rates (kg/day) are included as FIG. 4 and 5, respectively. The figures present distributions of the entire range of data, from $0.01 \leq p^* \leq 1.0$ (with $P(\text{Load} \leq l) \approx p^*$, in which l represents a specific loading rate (kg/day) and p^* represents the rank-based point estimate of probability associated with the loading rate).

Reference loading curves for TSS developed from the record for Steamboat Creek suggest that the observed loading rate at the downstream-most gaging station (10349980) rarely exceeds the loading rate implied by the reference loading curve (RLC). The estimated distribution of observed TSS loading (FIG. 2) crosses the RLC at approximately 17,000 kg/day ($p^* \approx 0.05$), which suggests that approximately 5% of observed loading rates equal or exceed this rate.

The differences in distribution of observed loading rates between the downstream-most gage and two upstream gaging stations are pronounced. At downstream stations quantiles for equivalent loading rates of total phosphorus decrease as distance from the upstream-most gage (10348800) increases, which indicates increased loading in the reaches between these stations. The most pronounced shift in distribution occurs between gages 10349300 and 10349980 (FIG. 4b), especially in loading rates with quantiles in the range of $0.1 \leq p^* \leq 1.0$. Loading rates estimated at the upstream gage are less than or equal to those of the reference loading curve with equivalent quantiles for approximately 85% of the data set. However, total phosphorus loading rates observed at gage 10349980 are always greater than those at equivalent quantiles for the reference loading curve.

Conversely, the observed loading rates for total suspended solids are approximately less than or equal to those of the reference loading curve developed at station 1034980 at equivalent quantiles for most of the data set. During extreme events (those with $p^* \leq 0.1$), the loading rate for TSS may exceed the reference loading rate for loading events in the upper extreme of the distribution, below the 0.05 quantile. This implies that for 95% of the daily loading rates observed at this gage, only the most extreme led to loading that would exceed a maximum daily loading rate developed using the ambient water quality standard.

Carson River: TSS and TP
Estimated loading rates of TSS at gage 10311400 exceed those at equivalent quantiles on the reference loading curves for approximately 25% of the daily loading rates observed in the sampling record. From the 0.6 to the 1.0 quantiles, values for loading rates of the reference loading curve are greater than those estimated for all gaging stations from the sampling record. Below the 0.6 quantile, estimated loading rates at the downstream-most station (10311400) exceed those at equivalent quantiles for the reference loading curve. This implies that TSS loading will exceed rates at equivalent quantiles on the reference loading curve approximately 40% of the time. For events with quantiles for equivalent loading rates that are greater than those estimated for the reference loading curve (at $p^* \leq 0.04$ in FIG. 3a), loading rates for all upstream stations except the West Fork of the Carson River (10310000) exceed those at equivalent quantiles on the reference loading curve for TSS. The degree of change in loading rate distribution between the gaging station on the East Fork of the Carson River (10309000) and downstream gaging stations below the confluence of the East and West Forks (10311000) and the most downstream gaging station (10311400) is most pronounced between the .03-.08 quantiles.

Total phosphorus loading distributions are very similar to those observed for Steamboat Creek, in that loading rates at quantiles on the reference loading curve are always exceeded by estimated loading rates at upstream stations, with the exception of 1031000, on the East Fork of the Carson River. Above the 0.6 quantile, estimated loading rates for station 10311400 and 10311000 are approximately equivalent. This implies that little additional loading takes place between these two stations under flow conditions represented in this part of the data set. At quantiles less than 0.6 loading rates at equivalent quantiles increase from station 10311400 to 1031100, which implies tht under these flow conditions the watershed area between the two gaging stations is a significant source of total phosphorus.

SUMMARY AND CONCLUSIONS

In watersheds with water quality problems due primarily to nonpoint source processes, a TMDL is certain to be exceeded under some conditions, such as during extreme events. Accordingly, land treatment programs to reduce nonpoint source contamination should be developed with a sense of how well management strategies must perform, with respect to current daily loading estimates. If management practices intended to control potential nonpoint sources are effective only under conditions in which loading does not normally exceed a TMDL, then the net reductions potentially will not yield the types of benefits anticipated by planners and managers. The data distributions and RLC's can be used to evaluate the degree of change needed in loading rates as a result of adopting or constructing management practices, if management practices have explicit performance criteria. Such criteria may be expressed in terms of loading events of a specified magnitude, which correspond with a quantile of the empirical distribution expressed as p*. Identifying an appropriate quantile as a design criterion helps to extract three types of information from the systematic sampling record. First, selection of a specific quantile (p*) focuses attention on the level of reduction needed to attain the loading rates that are implied to be permissible at an equivalent quantile of the RLC. Second, with several observed data distributions displayed on the same graph with the RLC, the portions of a watershed that have the greatest influences on loading rates at a point of interest can be identified. Third, the consequences of having design standards exceeded can be assessed, with respect to the likelihood that they will be exceeded and the type of loading dynamics that may be expected.

As an example, a design criterion of p*=0.5 for loading rates of TP on the Carson River, at gage 10311400, implies that approximately 46 kg/day of total phosphorus must be controlled to achieve the RLC loading rate of approximately 6 kg/day (FIG 5b). Observed loading rates from contributing upstream areas indicate that TP loading from the areas upstream of gages 10310000, 10309000 contribute minimally to the loads observed at gage 10311400. However, loading rates between the first gage downstream of the confluence of the East and West Forks (10311000) and downstream of the East Fork (10309000) exceed the RLC at this quantile. In fact loading rates observed at gages 10311000 and 10311400 at equivalent quantiles from $0.5 \leq p^* \leq 1.0$ are approximately the same, which suggests that for at least half of the data set, loading of conservative pollutants at the downstream most gage may be related to loading in the reach between the East Fork and the first gage that measures flow below the confluence of the East and West Forks (10311000). The estimated distribution function does not provide proof that land uses in this area are a source of TP, but does indicate that planning could begin by examining phosphorus use and associated land management practices that may be related to transport of total phosphorus to the Carson River.

The approach of using empirical distributions and reference loading curves as a method of summarizing information from systematic sampling programs yields information that is difficult to obtain from summary statistics. We applied plotting positions to transformed data from systematic samples to summarize information collected from differentareal scales and found that we could draw preliminary conclusions from the data, even in the absence of formally established maximum daily loading criteria. As a first step in analyzing results of systematic sampling programs, the approach avoids assumptions of underlying distribution and instead draws information about distribution of daily loading estimates from a transformed version of the observed data. This approach serves as an exploratory step to organizing and interpreting data, especially if other information, such as detailed land use data, modeling and sampling information are available. If TMDL are being considered, this type of exploratory data analysis can be used to evaluate the margins of safety incorporated in choices of daily loading thresholds.

One of the purposes of preliminary analysis may be to estimate the degree of loading reduction needed, in the absence of a TMDL. Establishing a TMDL for a river reach, based on assimilative capacity is likely to be very difficult because of the high rate of volume exchange. TMDL for large receiving water bodies, such as lakes or reservoirs, may be based on ambient water quality standards or indicators that are linked explicitly to a designated use or desirable condition in a receiving water body (U.S.E.P.A., 1999). We have applied ambient water quality standards to develop reference loading curves. However, the same approach could be used to explicitly depict permissible loading rates with respect to a desirable condition, which may be based on a maximum concentration related to a potential adverse effect. In either case, the difference between observed loading rates at a given quantile and those that lie on the reference loading curve suggests the degree of loading reduction that might be necessary to meet a maximum daily loading criterion at a specific point and which upstream areas may be most important to examine closely with respect to opportunities to manage potential sources.

Finally, guidance for selecting a margin of safety in setting a TMDL for nutrients suggests that both explicit and implicit methods be used (U.S.E.P.A., 1999). Explicit methods acknowledge that loading or concentration standards set to protect a desirable condition or meet an ambient water quality standard are based on assumptions or information that may not have a sound technical foundation. Implicit methods arbitrarily increase the level of protection without explicit concerns about reliability of information used to set the loading standards. In both cases, explicit or implicit margins of safety address a lack of confidence that preliminary estimates of loading limits are likely to provide effective protection. Empirical distributions developed from a record of daily loading estimates offer another perspective on the margin of safety incorporated in TMDL. By plotting the loading estimates against quantiles associated with each loading value, the reference loading curve and the empirical distribution function explicitly acknowledge the likelihood, given the data, that a TMDL will be equaled or exceeded. A TMDL corresponding with an estimated p* (determined empirically from the observed loading rates) can be evaluated in terms of the risk of exceeding a specified loading rate under a "no-action" alternative. This provides planners and regulators with information about the frequency and magnitude of daily loading events that would exceed pollutant loading thresholds considered as TMDL.

Overall this approach to analyzing systematic sampling data provides a useful first step for preliminary data analysis and supports both qualitative and quantitative assessments related to TMDL, contributing areas and seasonal trends in pollutant loading on a daily basis. The approach should be complementary to other analytic techniques used to develop strategies for nonpoint source management on a watershed scale.

ACKNOWLEDGEMENTS

The University of Nevada College of Agriculture, Biotechnology and Natural Resources and the Carson Water Subconservancy District contributed funds to compile and analyze the data represented in this paper. Two groups, the Steamboat Creek Restoration Committee and the Carson River Coalition, provided guidance and useful comments about the types of information needed from data sets of the type compiled under the supervision of the Nevada Division of Environmental Protection. NDEP's Bureau of Water Quality offered comments and suggestions that clarified the purpose and interpretation of these data analyses.

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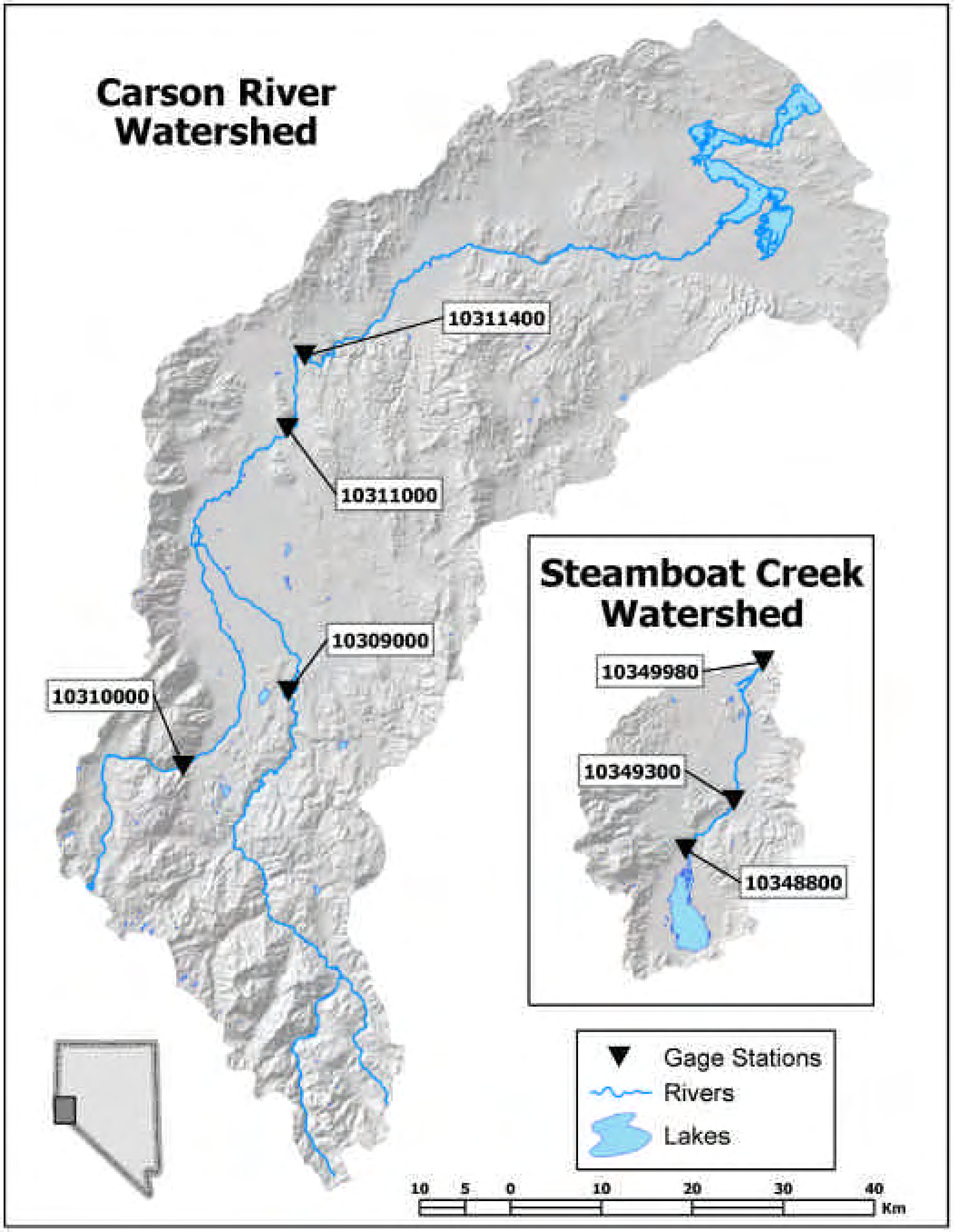

FIG. 1: Carson River and Steamboat Creek Drainages, Northwest Nevada, showing locations and designations of U.S.G.S. Gaging Stations

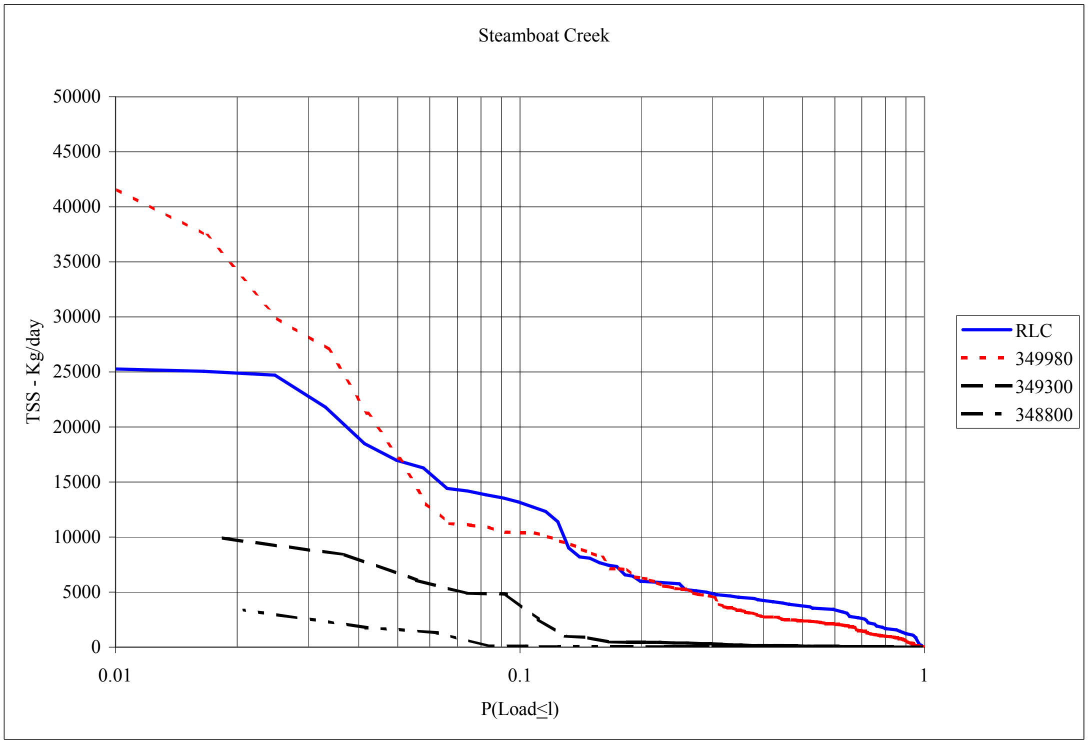

FIG. 2: TSS loading rates and reference loading curves, representing the full systematic-sampling record

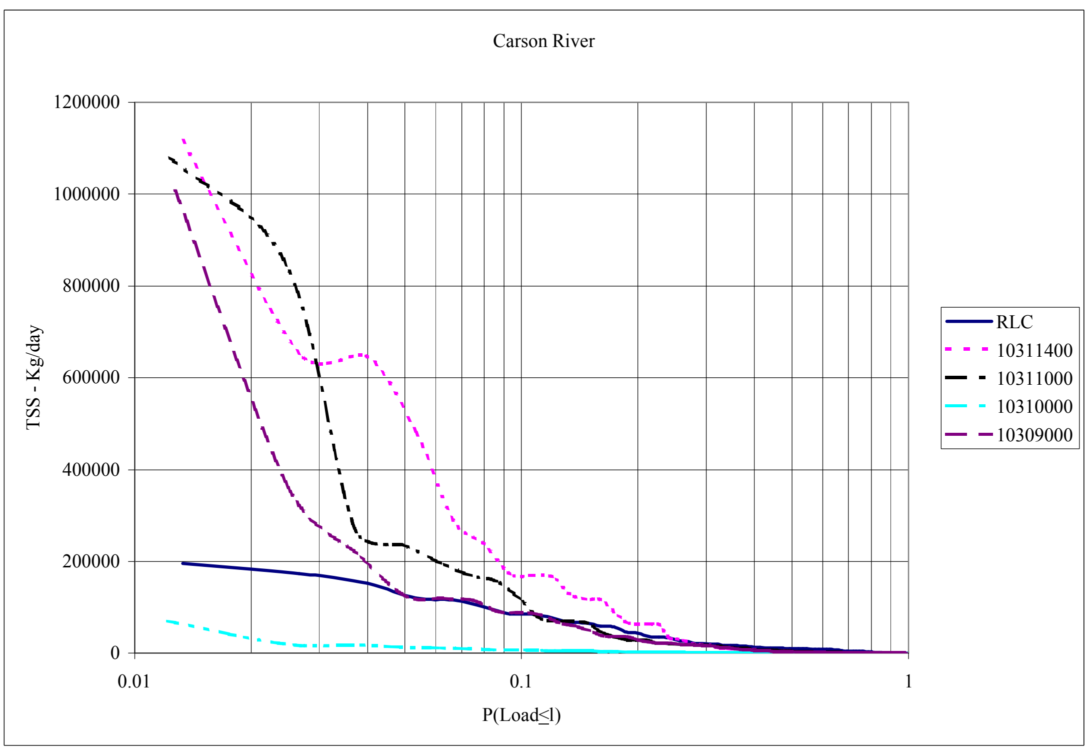

FIG. 3: TSS loading rates and reference loading curves, representing the full systematic-sampling record for the Carson River

A Spatially Referenced Regression Model (SPARROW) for Suspended Sediment in Streams of the Conterminous U.S.

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INTRODUCTION

Suspended sediment has long been recognized as an important contaminant affecting water resources. Besides its direct role in determining water clarity, bridge scour and reservoir storage, sediment serves as a vehicle for the transport of many binding contaminants, including nutrients, trace metals, semi-volatile organic compounds, and numerous pesticides (U.S. Environmental Protection Agency, 2000a). Recent efforts to address water-quality concerns through the Total Maximum Daily Load (TMDL) process have identified sediment as the single most prevalent cause of impairment in the Nation’s streams and rivers (U.S. Environmental Protection Agency, 2000b). Moreover, sediment has been identified as a medium for the transport and sequestration of organic carbon, playing a potentially important role in understanding sources and sinks in the global carbon budget (Stallard, 1998).

A comprehensive understanding of sediment fate and transport is considered essential to the design and implementation of effective plans for sediment management (Osterkamp and others, 1998, U.S. General Accounting Office, 1990). An extensive literature addressing the problem of quantifying sediment transport has produced a number of methods for estimating its flux (see Cohn, 1995, and Robertson and Roerish, 1999, for useful surveys). The accuracy of these methods is compromised by uncertainty in the concentration measurements and by the highly episodic nature of sediment movement, particularly when the methods are applied to smaller basins. However, for annual or decadal flux estimates, the methods are generally reliable if calibrated with extended periods of data (Robertson and Roerish, 1999). A substantial literature also supports the Universal Soil Loss Equation (USLE) (Soil Conservation Service, 1983), an engineering method for estimating sheet and rill erosion, although the empirical credentials of the USLE have recently been questioned (Trimble and Crosson, 2000). Conversely, relatively little direct evidence is available concerning the fate of sediment. The common practice of quantifying sediment fate with a sediment delivery ratio, estimated from a simple empirical relation with upstream basin area, does not articulate the relative importance of individual storage sites within a basin (Wolman, 1977). Rates of sediment deposition in reservoirs and flood plains can be determined from empirical measurements, but only a limited number of sites have been monitored, and net rates of deposition or loss from other potential sinks and sources is largely unknown (Stallard, 1998). In particular, little is known about how much sediment loss from fields ultimately makes its way to stream channels, and how much sediment is subsequently stored in or lost from the streambed (Meade and Parker, 1985, Trimble and Crosson, 2000).

This paper reports on recent progress made to address empirically the question of sediment fate and transport on a national scale. The model presented here is based on the SPAtially Referenced

Regression On Watershed attributes (SPARROW) methodology, first used to estimate the distribution of nutrients in streams and rivers of the United States, and subsequently shown to describe land and stream processes affecting the delivery of nutrients (Smith and others, 1997, Alexander and others, 2000, Preston and Brakebill, 1999). The model makes use of numerous spatial datasets, available at the national level, to explain long-term sediment water-quality conditions in major streams and rivers throughout the United States. Sediment sources are identified using sediment erosion rates from the National Resources Inventory (NRI) (Natural Resources Conservation Service, 2000) and apportioned over the landscape according to 30-meter resolution land-use information from the National Land Cover Data set (NLCD) (U.S. Geological Survey, 2000a). More than 76,000 reservoirs from the National Inventory of Dams (NID) (U.S. Army Corps of Engineers, 1996) are identified as potential sediment sinks. Other, non-anthropogenic sources and sinks are identified using soil information from the State Soil Survey Geographic (STATSGO) data base (Schwarz and Alexander, 1995) and spatial coverages representing surficial rock type and vegetative cover. The SPARROW model empirically relates these diverse spatial datasets to estimates of long-term, mean annual sediment flux computed from concentration and flow measurements collected over the period 1985-95 from more than 400 monitoring stations maintained by the National Stream Quality Accounting Network (Alexander and others, 1998), the National Water Quality Assessment Program, and U.S. Geological Survey District offices (Turcios and Gray, in press). The calibrated model is used to estimate sediment flux for over 60,000 stream segments included in the River Reach File 1 (RF1) stream network (Alexander and others, 1999).

SPARROW uses statistical methods to calibrate a simple, structural model of riverine water quality, one that imposes mass balance in accounting for changes in contaminant flux. As applied here, the mass-balance approach facilitates the interpretation of model results in terms of physical processes affecting sediment transport, and makes possible the estimation of various rates of sediment generation and loss associated with stream channels and features of the landscape. The statistical approach provides a basis for assessing the error of these inferred rates and of the error in extrapolated estimates of sediment flux made for streams in the RF1 network.

An important implication of the holistic modeling approach adopted in this analysis is that estimates of sediment production and loss are based on, and therefore consistent with, measurements of in-stream flux. Other ancillary information, such as direct measurements of long-term sediment storage and release from reservoirs (Steffen, 1996), is incorporated into the analysis by specifying additional equations explaining these ancillary variables. The imposition of cross-equation constraints affords this information a statistically consistent weight in explaining in-stream sediment flux. Thus, the methodology described here represents a general framework for synthesizing a wide spectrum of available information relevant to the understanding of sediment fate and transport.

METHODOLOGY

The SPARROW methodology (Smith and others, 1997) has been modified to incorporate greater spatial resolution. The primary spatial reference frame for the model continues to be the RF1 reach network: all point sources and landscape features are referenced to a particular RF1 reach. However, considerable internal structure has been added to each reach. Reach watersheds are delineated using the 1-kilometer HYDRO 1K digital elevation model (DEM) (U.S. Geological Survey, 2000a), and explicit pathways are defined between landscape features and their adjacent RF1 streams. The delineation method uses a “burn-in” process whereby the RF1 reach is first digitized in the 1-kilometer grid and then the elevations of RF1 grid cells are artificially lowered to insure that simulated flow from surrounding cells moves into them. Flow directions based on the steepest descent determine the extent of the reach watershed and the undefined tributary flow paths leading from the landscape to the RF1 channel cells. To insure the accurate determination of in-stream travel time, RF1 stream pathways continue to be defined by the line work of RF1 channels rather than by the grid-cell representation.

A schematic of a typical reach watershed, illustrating its spatial structure and associated features, is given in figure 1. Flow directions, represented by the arrows crossing each adjacent grid cell, define the movement of water in undefined tributaries leading to the RF1 stream. The “burn-in” method insures that all flow paths intersect a reach cell at some point within the watershed, although inconsistencies between the RF1 reach and the DEM-defined stream channel may artificially lengthen “off-RF1” flow paths and shorten “on-RF1” paths (see figure 1 for an example). The length of the flow path provides a rough estimate of the distance sediment must travel in smaller tributaries before reaching the larger streams included in the RF1 network. Travel time in small streams versus large rivers has been shown to be an important factor affecting the in-stream delivery of nutrients (Alexander and others, 2000) and could be of similar importance for sediment.

The enhanced spatial structure afforded by the DEM facilitates the incorporation of spatially integrating features into the model. “Off-channel” reservoirs, located on the grid net according to their geographic coordinates provided by the NID, act as potential sinks for sediment emanating from cells with flow paths that intersect the reservoir grid cell. Similarly, “off-RF1” monitoring stations can be located on the grid and given a basin representation. Although these stations are not useful for calibrating the delivery process within RF1 channels, they offer a high-resolution view of other processes affecting the movement of sediment across the landscape.

Other important spatial features identified in the model include point sources, located relative to RF1 streams based on geographic coordinates (S. Rubin, Environmental Protection Agency, written commun., 1999), and land associated with uses that serve as likely sources or sinks for sediment. Point-source loadings of total suspended solids are determined by methods developed by the National Oceanic and Atmospheric Administration for the National Coastal Pollutant Discharge Inventory (National Oceanic and Atmospheric Administration, 1993). Land use is taken from the 21-class, 30-meter resolution NLCD, and summarized according to the number of 30-meter cells of a given land-use class that are mapped to a corresponding 1-kilometer cell. NLCD land use is used to refine the areal extent of the various sediment erosion rates associated with different land covers identified in the NRI.

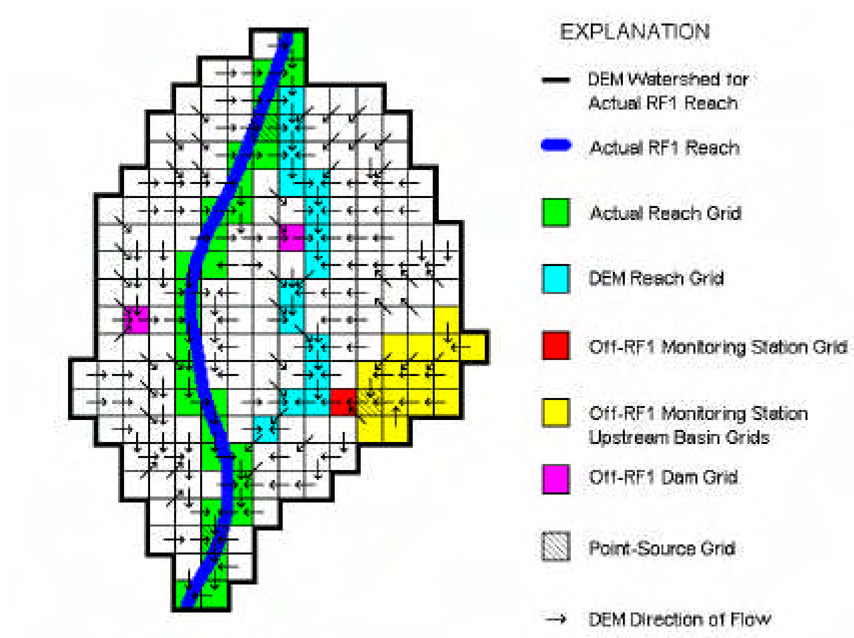

Figure 1. Schematic of a typical reach watershed illustrating the grid cell structure and identified attributes.

The mean annual suspended-sediment flux generated within and leaving reach watershed j, referred to as the incremental reach flux F_j, can be expressed as

(1)
$$F_j = \sum_{c=1}^{N_j} e^{-\mathbf{d}'\mathbf{d}_{c,j} + \mathbf{a}'\mathbf{Z}_{c,j}} \boldsymbol{\beta}'\mathbf{S}_{c,j} ,$$

where N_j is the number of 1-kilometer grid cells, indexed by c, in reach watershed j, $\mathbf{d}_{c,j}$ is a vector of factors describing the pathway from cell c to the outlet of reach j, dis a vector of coefficients associated with the pathway variables, $\mathbf{Z}_{c,j}$ is a vector of landscape and climatic characteristics affecting the delivery of sediment within cell c, a is a vector of coefficients associated with the **Z** variables, $\mathbf{S}_{c,j}$ is a vector of sediment sources, and b is a vector of associated source coefficients.

The vector **d** consists of (1) variables representing the landscape flow-path distance traversed to reach the RF1 stream, (2) the mean slope of the "off-RF1" flow path, the time of travel incurred along the RF1 stream, (3) variables affecting the retention of sediment in any reservoir located along the landscape or RF1 flow path, such as streamflow, reservoir age, and NID estimates of surface area or storage volume, and (5) other variables identifying possible sinks along the flow path, such as forested land or land classified by STATSGO as wetlands or alluvium. Variables included in the **Z** vector include runoff, overland flow, slope, and indicators of soils or other factors affecting the movement of sediment off the field to channels. The source vector, **S**, includes sediment erosion from the NRI and point-source loadings.

The 1-kilometer spatial detail used to determine F_j, corresponding to nearly 8 million grid cells for the more than 60,000 reaches in the conterminous U.S., places a heavy computational burden on the iterative, non-linear least-squares, calibration method. To reduce the number of computations, the reach model is simplified by assuming the **Z** variables take a single mean value $\overline{\mathbf{Z}}_j$ for all cells in the reach and, for the **d** variables, by substituting a second-order Taylor approximation about the reach-level mean $\overline{\mathbf{d}}_j$. The imposition of a common $\overline{\mathbf{Z}}_j$ value for all cells in a reach is not restrictive given the spatial coarseness of existing information. The resulting approximation is

$$(2)\quad F_j \approx e^{-\mathbf{d}'\overline{\mathbf{d}}_j + \mathbf{a}'\overline{\mathbf{Z}}_j} \sum_{c=1}^{N_j} \left\{ \left(1 - \mathbf{d}'\left(\mathbf{d}_{c,j} - \overline{\mathbf{d}}_j\right)\right)\mathbf{S}'_{c,j}\text{ß} + \left(\text{ß} \otimes \mathbf{d}\right)' \left(\mathbf{S}_{c,j} \otimes \left(\mathbf{d}_{c,j} - \overline{\mathbf{d}}_j\right)\left(\mathbf{d}_{c,j} - \overline{\mathbf{d}}_j\right)' \right)\mathbf{d} \right\}.$$

This approximation effectively converts the unit of observation in equation (1) from a 1-kilometer grid cell to a reach segment, replacing the non-linear terms dependent on individual cell values with non-linear and linear terms dependent on reach-level means, variances and covariances of the **d** and **S** variables.

To complete the model structure, individual reaches are combined to form a nested basin. Each nested basin i consists of the set $J(i)$ of reaches upstream from monitoring station i and below any monitoring station located further upstream (if such stations exist) (see figure 2). The sediment load for nested basin i, denoted L_i, is equal to the sum of the incremental fluxes from the nested reach segments $j \in J(i)$, plus the monitored sediment discharged from the set $U(i)$ of nested basins bounding the upper drainage of nested basin i (there may be more than one) and delivered to monitoring station i. The sediment load L_i is related to the upstream incremental fluxes, F_j, and monitored loads, L_u, according to a log-linear relation

$$(3)\qquad \ln\left(L_i\right) = \ln\left(\sum_{j \ni J(i)} e^{-\mathbf{d}'\mathbf{d}_{j,i}} F_j + \sum_{u \ni U(i)} e^{-\mathbf{d}'\mathbf{d}_{u,i}} L_u \right) + e_i\,,$$

where $\mathbf{d}_{j,i}$ represents a vector consisting of the same variables in $\mathbf{d}_{c,j}$, but corresponding to the RF1-reach path extending from the downstream-end of reach j to the i^{th} monitoring station (accordingly, $\mathbf{d}_{j,i}$ has values of 0 for all variables pertaining to "off-RF1" flow paths). In equation (3), an independent error term e_i has been added to represent the combined effect of measurement and model error introduced at nested basin i.

Data on reservoir storage can be incorporated directly into the model by introducing an additional storage equation. Let $\mathbf{d}^*$ and d^* pertain to the subset of path variables and associated coefficients determining the rate sediment is stored in reservoirs, and define R_k as the annual amount of stored sediment measured at a reservoir on reach k (a similar analysis can be done for "off-RF1" reservoirs). The reservoir storage equation takes the form

$$(4)\qquad \ln\left(R_k\right) = \ln\left(\sum_{j \ni J(k)} \left(e^{\mathbf{d}^{*\prime}\mathbf{d}^*_{j,k}} - 1 \right) F_j + \sum_{u \ni U(k)} \left(e^{\mathbf{d}^{*\prime}\mathbf{d}^*_{u,k}} - 1 \right) L_u \right) + w_k\,,$$

where w_k is a random error.

Joint estimation of equations (3) and (4), with the F_j and corresponding a, b, and dparameters defined by equation (2), is by non-linear three-stage least squares. To insure robust estimates and to facilitate the estimation of prediction error, the calibration of the model is repeated 200 times employing a bootstrap estimation algorithm (see Smith and others, 1997).

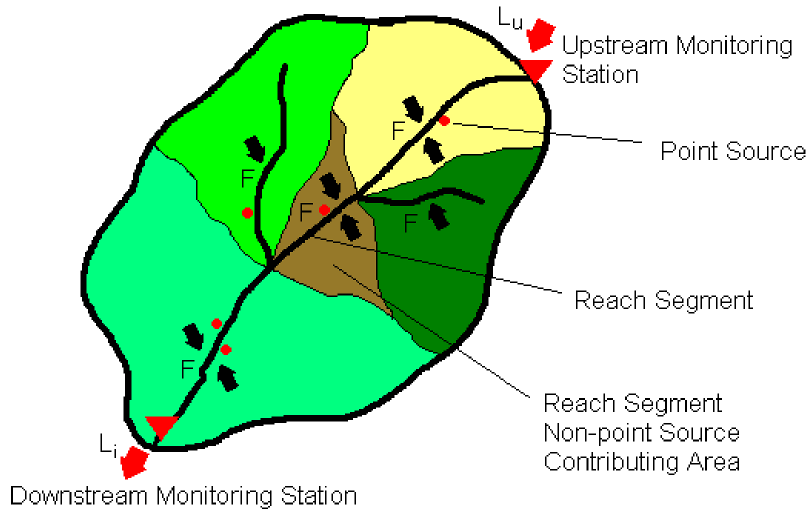

Figure 2. Schematic of a nested basin defined by upstream and downstream monitoring stations.

The flexible mathematical structure used in equations (1) - (3) is capable of accommodating a number of hypotheses concerning sediment fate and transport. Sites of sediment storage, identified in the model as a subset of the **d** variables, can act as sediment sources or sinks, depending on the sign of corresponding dcoefficients. A random coefficient form of the model allows storage sites to serve as sources in some regions and sinks in others. Such behavior can be inferred statistically by relating the prevalence of storage sites in nested basins to the magnitude of the squared residual ***e*** in these basins (Godfrey, 1988). Non-point sources of sediment, such as soil erosion included under **S**, are distinguished from sediment losses from storage (e.g., an alluvial plain) identified with **d**, on the assumption that the former is a primary process due to weathering whereas the latter is a consequence of the accumulation of previously weathered material which is later released to streams under changing hydraulic conditions. Accordingly, the potential for storage loss in the model depends on the extent of accumulated upstream soil erosion due to weathering. The empirical validity of the USLE estimate of soil erosion can be evaluated through statistical hypothesis tests conducted on the relevant b coefficients. Alternative measures of soil erosion can also be empirically evaluated in the model by substituting variables serving as determinants of the USLE for the USLE erosion estimate.

The estimation of long-term suspended-sediment load at a monitoring station is based on the regression of the natural logarithm of instantaneous suspended-sediment concentration on current and lagged values of the natural logarithm of daily flow and other variables representing seasonal and trend effects. If the station has concentration data collected more frequently than a weekly basis, the regression model is modified to account for serial correlation. To be included in the analysis, a station must have at least 3 years of data between 1985 and 1995. Only data within the period 1985-95 are included in the regression.

Mean-annual suspended-sediment load is estimated by first simulating load for each day over the 1985-95 period and then averaging daily values on an annual basis. Simulated loads are obtained by taking the exponential of the sum of the predicted daily load given by the calibrated regression model with the time trend variable set to a base year of 1992 and a randomly selected residual from the regression model. For days having actual monitoring data, the daily load is computed by multiplying the measured instantaneous concentration by the daily flow. If a station has a data record with sufficient frequency to estimate a serial correlation parameter, the simulated daily load is based on the conditional prediction associated with past and future observed loads, plus a normally distributed random error having a correlation structure consistent with the conditional prediction and with the variance estimated by the regression model. The Monte Carlo process used to estimate simulated daily loads for the 1985-95 period is repeated 200 times, providing 200 values for estimating the mean and standard deviation of the average annual sediment load for a site.

SUMMARY

The model described here is intended to empirically evaluate regional-scale processes affecting the long-term (i.e., decadal) transport of sediment in rivers. Additionally, the model will provide estimates of sediment mean annual flux for every reach included in the RF1 network. Error estimates for these process evaluations and stream predictions are determined using robust bootstrap methods. Future work will address the dynamic behavior of sediment flux associated with non-steady state streamflow conditions.

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