Volume 2

VIII. Monitoring and NonPoint Source Pollution

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OVERVIEW OF THE RECONFIGURED-CHANNEL MONITORING AND ASSESSMENT PROGRAM

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Abstract: The US Geological Survey Reconfigured-Channel Monitoring and Assessment Program was developed to monitor and assess increasingly popular and widespread channel reconfiguration efforts. A periodically updated data base available on the world wide web will enable land-management agencies and other interested parties to evaluate the long-term success of specific channel reconfiguration projects. Monitoring projects on the Lake Fork and the North Fork of the Gunnison River, Colorado, illustrate two different reclamation methods and an example of the monitoring program objectives and approach.

INTRODUCTION

Channel reconfiguration to mitigate a variety of riverine problems has become an important issue in the Western United States. Reasons cited for channel reconfiguration include restoration to more natural or historical conditions, improved water conveyance in flood-prone areas, mitigation of unstable streambed and streambanks, increased sediment transport, and enhancement of riparian habitat or recreation. Numerous private entities and resource-management agencies have attempted to reconfigure stream and river channels by using designs based on different geomorphic philosophies. However, little work has been done in assessing the channel response to and the effectiveness of these modifications over a long period of time (Kondolf and Micheli, 1995). The U.S. Geological Survey (USGS) is engaged in a program designed to monitor and assess selected river reaches that have undergone reconfiguration (Elliott and Parker, 1999).

The objectives of the USGS Reconfigured-Channel Monitoring and Assessment Program (RCMAP) are: (1) to develop uniform and versatile monitoring methods for reconfigured channel reaches and to apply these methods to selected reaches; (2) to create and maintain a data base consisting of numerous monumented stream reaches; and (3) to revisit these reaches periodically and assess regional and temporal trends in the geomorphic response of the stream to the channel modifications. Long-term monitoring of reconfigured channels will enable analysis of how and why a particular reconfiguration design may have remained stable or failed. If a channel modification fails, the analysis will focus on understanding the processes by which failure occurred. These processes could include bank erosion, streambed aggradation or incision, flood-plain deposition or scour, and loss of riparian vegetation through root scour, soil-moisture deficit, or prolonged submergence.

TWO-LEVEL APPROACH

The RCMAP is implemented at two levels to satisfy multiple objectives. Level 1 involves development of standardized sampling and monitoring methods, site-specific measurements, and analysis of channel characteristics. Level 2 involves long-term data-base development and periodic analyses.

Level 1 - Methods and Site-Specific Analysis: Level 1 activities consist primarily of descriptive measurements of channel characteristics prior to (if possible) and following channel modification and geomorphic and hydrologic evaluations of the river reach. These measurements are tailored to a specific reach and entail surveys of the channel cross section and longitudinal profile, measurement of sediment-size characteristics of the streambed and banks, and oblique photography from monumented locations through the reach. Other measurements may include aerial photographic interpretation and streamflow-regime analysis, if photographs and hydrologic records are available. River reaches are selected for study and inclusion in the RCMAP data base on the basis of: (1) cooperator interest and funding availability, (2) the potential for future channel-modification activity in the reach, (3) the proximity of a streamflow-gaging station, and (4) scientific research objectives.

Data are collected over a reach of at least several channel widths in length. A set of measurements are made prior to reconfiguration, if possible, and during the first year after reconfiguration. These measurements will be replicated in a subsequent year to evaluate channel change in the reconfigured reach.

The time interval between replicate measurements will be determined partly by the hydrologic history at the monitored reach. Some simple empirical relations also may be used to evaluate potential channel response.

Site-specific analysis provides descriptive information about a reconfigured channel reach in a timely manner and enables interested parties to assess whether the modification activities have resulted in persistent qualities deemed acceptable to land managers and the public. Another potential benefit of the USGS RCMAP is that it enables other agencies or researchers to expand upon and augment the geomorphic data collected by the USGS. Research topics might include hydraulic function of habitat-improvement structures, sediment transport, aquatic habitat, and riverine ecology.

Level 2 - Data-Base Development and Analysis Among Sites: The RCMAP is being expanded to include sites that represent a range of geomorphic, sedimentologic, and hydrologic stream types. RCMAP data is being archived in a manner similar to that of the USGS Vigil Network (Emmett and Hadley, 1968), and these data will be added to a USGS web site. The optimal size of the data base for subsequent analyses depends, in part, on the site-to-site variability in the data base.

The Level 2 analyses use the Level 1 data base, compiled over several years, to identify regional patterns or trends in channel processes and morphology and to assess the channel response to earlier modification efforts. This analysis among sites is ongoing as the data base periodically is updated and expanded. Level 2 analyses identify additional data collection or model applications needed to understand channel processes and responses at specific sites. Level 2 analyses could include: (1) an evaluation of the effects of observed streamflow on post-reconfiguration channel morphology; (2) a determination of flow velocity, shear stress, and sediment entrainment potential under a range of discharges; (3) an empirical determination of sediment-transport rates to identify sites of potential aggradation or scour; and (4) parametric and nonparametric statistical analyses to evaluate whether the success rate of channel reconfiguration efforts is a function of specific channel morphology, gradient, sediment type, flow regime, or other factors, such as specific design features.

EXAMPLES OF MONITORED REACHES

Two reconfigured river reaches are presented to illustrate slightly different engineering approaches to channel modifications and to illustrate the RCMAP monitoring methods.

Lake Fork of the Gunnison River: The Lake Fork of the Gunnison River is a perennial, snowmelt-dominated stream draining the northeastern side of the San Juan Mountains in southwestern Colorado (fig. 1). Streamflow records have been collected by the USGS since 1938 at a gaging station within the study reach (09124500 Lake Fork at Gateview). The mean annual streamflow in this reach, where the drainage area is 865 km^2, is 213 hm^3. Bankfull discharge is approximately 42.0 m^3/s (Andrews, 1984), and the 10-year flood is 69.6 m^3/s (U.S. Interagency Advisory Committee, 1982). The valley slope in the study reach is 0.0075 m/m.

Land use in the Lake Fork Valley is predominantly agricultural (hay meadows and livestock grazing). Aerial photography from 1977 and onsite reconnaissance in 1992 and 1998 indicated that segments of the river channel had been artificially straightened. Other segments were braided and prone to bank erosion and lateral shifting (fig. 2). Prior to reconfiguration, the reach near the Gateview gage was characterized by a wide, shallow channel with a streambed composed of gravel, cobbles, and boulders. Onsite reconnaissance and land-owner interviews in 1998 indicated that some segments of the channel were laterally restricted by an engineered levee and possibly dredged over a period of years.

A 3.2-km segment of the Lake Fork was reconfigured in late 1997 to mitigate past problems associated with flooding and gravel deposition on the flood plain, and to improve the trout fishery. The channel modifications included: (1) deepening of the channel by streambed excavation, (2) slight increases in sinuosity by constructing new cobble alternate bars within the former bank-to-bank channel area, (3) reduction of flow width and creation of streambank protection through addition of coarse sediment and tree-root wads to formerly vertical banks, (4) addition of large boulders as streambed roughness elements to improve fish habitat, and (5) construction of several grade-control and flow-directing structures composed of large boulders (fig. 3). The boulder structures placed in the stream were designed tore-direct high-velocity flow and to facilitate bedload transport through the reach (Rosgen, 1996).

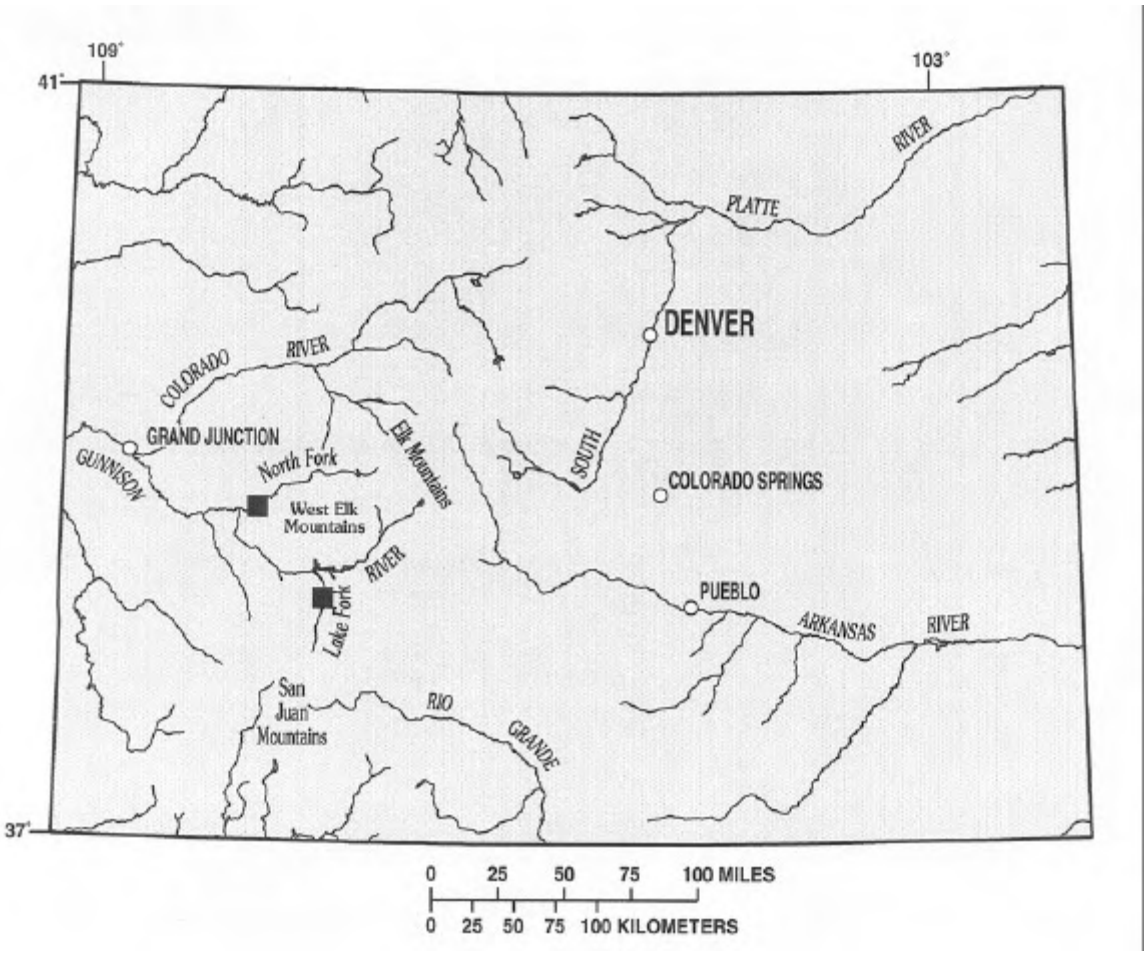

Figure 1. Location map showing two monitored reconfigured river reaches.

A 0.8-km reach of the Lake Fork was monumented and surveyed by the USGS in September 1998. Permanent reference marks were installed for vertical and horizontal control (fig. 4). The reference-mark locations (latitude and longitude) were determined with a global positioning system (GPS) receiver to facilitate replication of the survey at a future date. The survey consisted of longitudinal profiles of the streambanks, terraces, and the water surface at a streamflow of approximately 6.1 m^3/s. Nine channel cross sections were surveyed in the study reach. Cross sections were selected that represented the range of channel geometry in the reach or that were in locations likely to exhibit change should future adjustments occur in cross-section dimensions (fig. 5). The cross section endpoints were established on a relatively stable surface, monumented, and located with a GPS receiver.

The monumented reach and cross-section survey was augmented with additional measurements. Sediment characteristics of the streambed and banks were determined at eight locations along the Lake Fork study reach using the Wolman (1954) pebble-count method. Oblique photographs were taken from 23 monumented locations; these photographs provide a means to observe and quantify changes in channel dimensions, sediment characteristics, or vegetation.

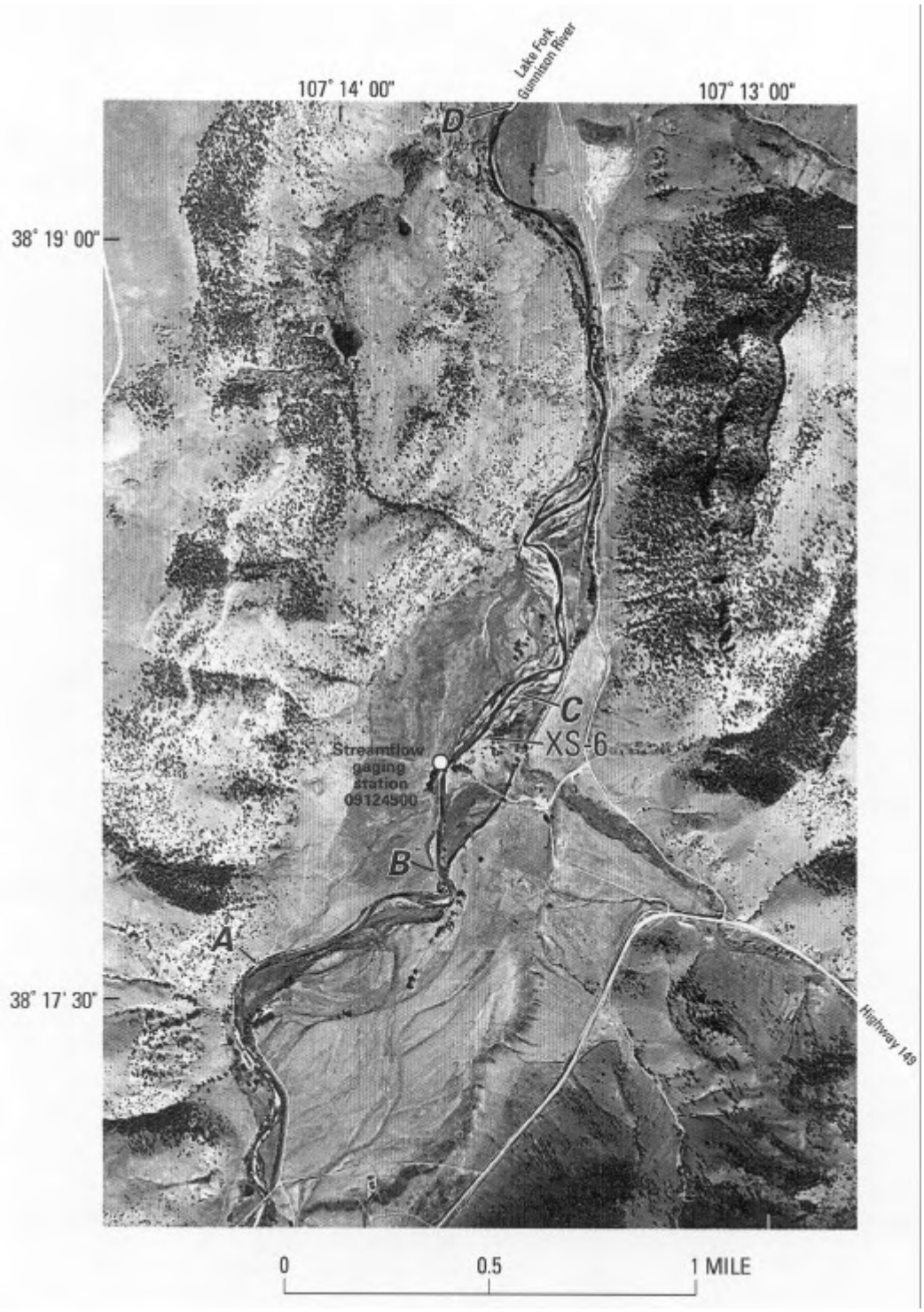

Figure 2. Aerial photograph of the Lake Fork of the Gunnison River at Gateview, Colorado, taken in 1977 before channel reconfiguration activities, showing location of streamflow gaging station 09124500, reach reconstructed in 1997 (A-D), and reach monitored by USGS in 1998 (B-C).

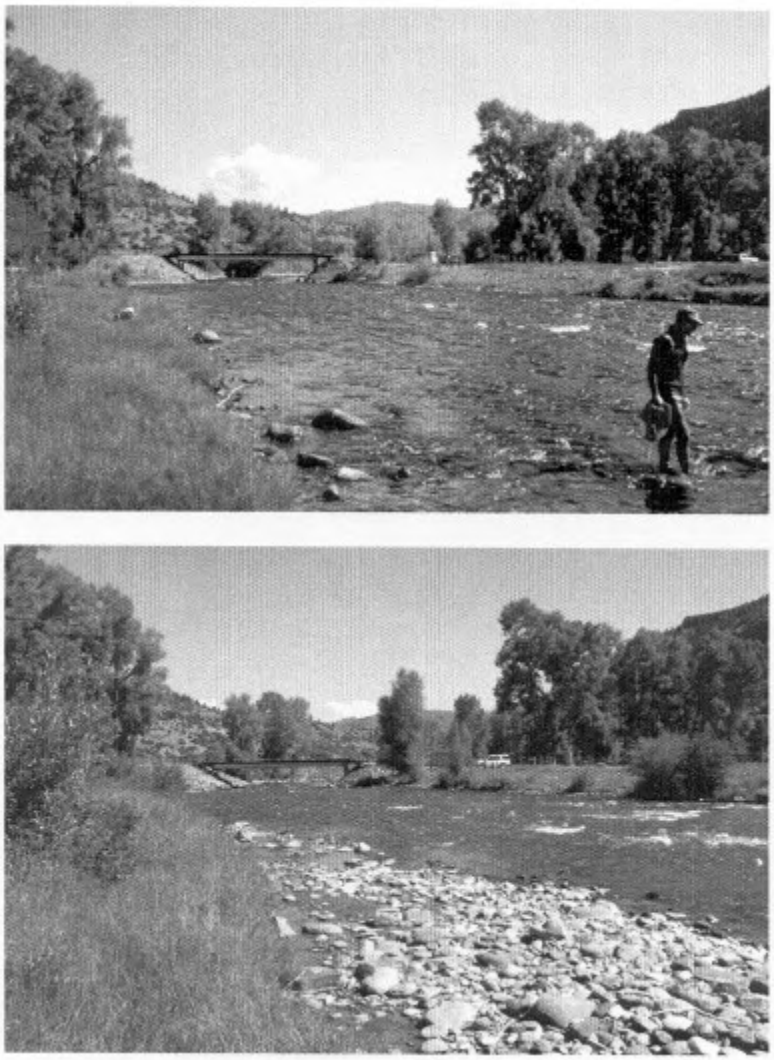

Figure 3. Lake Fork of the Gunnison River at Gateview, Colorado. Views looking downstream to the bridge near gaging station 09124500, (A) in 1992 prior to reconfiguration at a discharge of 7.6 m^3/s, and (B) in 1998 following reconfiguration at a discharge of 6.5 m^3/s. The cobble bar forming the left bank in (B) was constructed of material dredged from the streambed to the right. The large, partly submerged boulders in the right side of the channel in (B) were quarried off site and added to the streambed during reconfiguration.

North Fork of the Gunnison River: The North Fork of the Gunnison River is a perennial, snowmelt-dominated stream draining the West Elk and Elk Mountains in western Colorado (fig. 1). Streamflow records have been collected by the USGS since 1933 at a gaging station 29 km upstream from the study reach (09132500 North Fork Gunnison River near Somerset). The drainage area at the study reach is approximately 2,200 km^2 and the valley slope in the study reach is 0.0058 m/m. Bankfull discharge in the study reach is approximately 85.0 m^3/s (J.P. Crane, North Fork River Improvement Association, oral communication, 1999), and the 10-year flood is 215 m^3/s (U.S. Army Corps of Engineers, 1980).

Land use in the North Fork Valley is predominantly agricultural (livestock grazing and fruit orchards); however, extractive industries include underground coal mining and alluvial gravel quarrying. The North Fork of the Gunnison River channel has been extensively disturbed as a result of intentional riparian-vegetation removal and meander cut-off in the late 19th century, and by floods in 1912 and 1932 and subsequent channelization efforts (Crane, 1997).

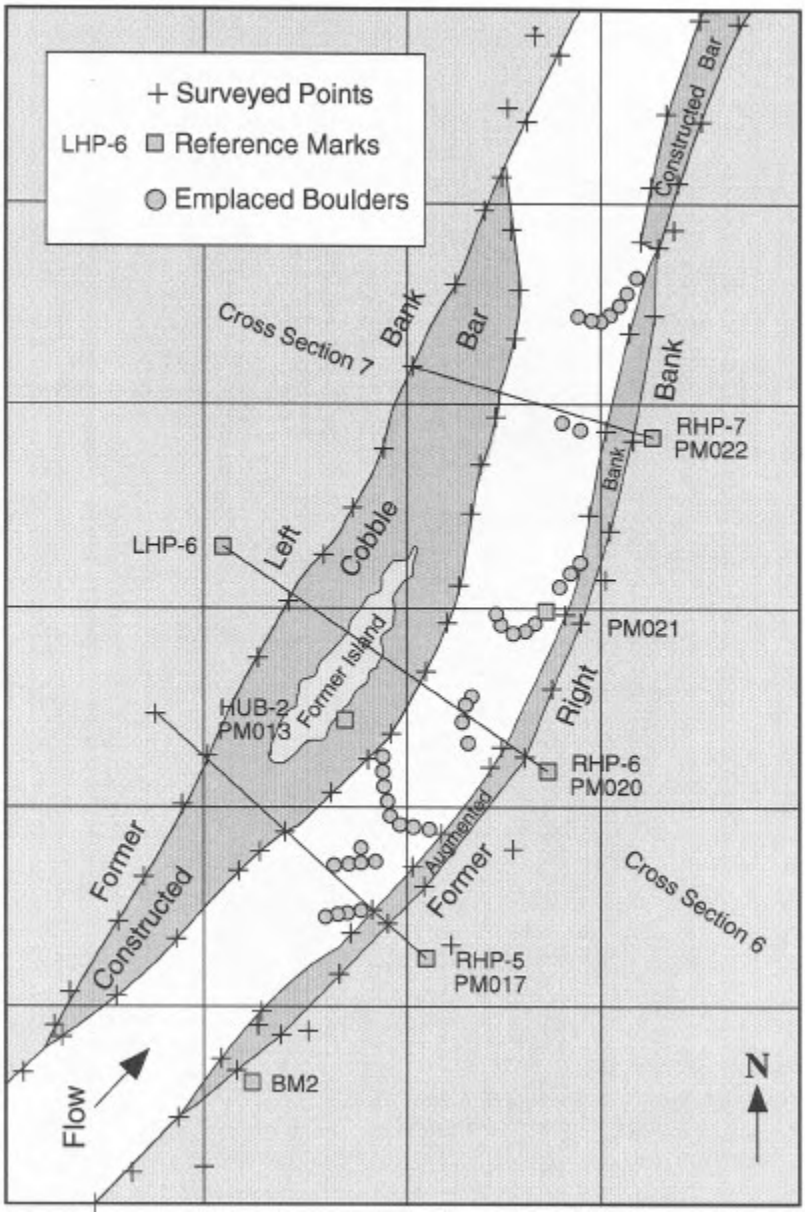

Figure 4. Detail of the Lake Fork of the Gunnison River study-reach survey showing the reconfigured channel at a discharge of 6.1 m^3/s, selected surveyed points, monumented cross sections 5, 6, and 7, and several reference marks. Sediment that composes the new right bank and the large cobble bar on the left was dredged from the streambed, which produced a narrower, deeper channel.

Prior to reconfiguration, the study reach was characterized by a wide, shallow channel with braided sections and a streambed composed of gravel and cobbles. Streambed aggradation and lateral bank erosion were common. A 2.2 km reach of the North Fork was reconfigured in the winter of 1999/2000 to improve the conveyance of runoff and sediment, to stabilize streambanks, and to improve aquatic habitat. The channel modifications included: (1) deepening of the channel by streambed excavation, (2) slight increases in sinuosity by excavation and realignment of the channel, (3) creation of streambank protection in limited areas through willow plantings and addition of coarse sediment to the new banks, (4) addition of a few large boulders as streambed roughness elements to improve fish habitat, (5) construction of a few grade-

control and flow-diversion structures composed of large boulders, and (6) stabilization of an irrigation-ditch intake structure (fig. 6).

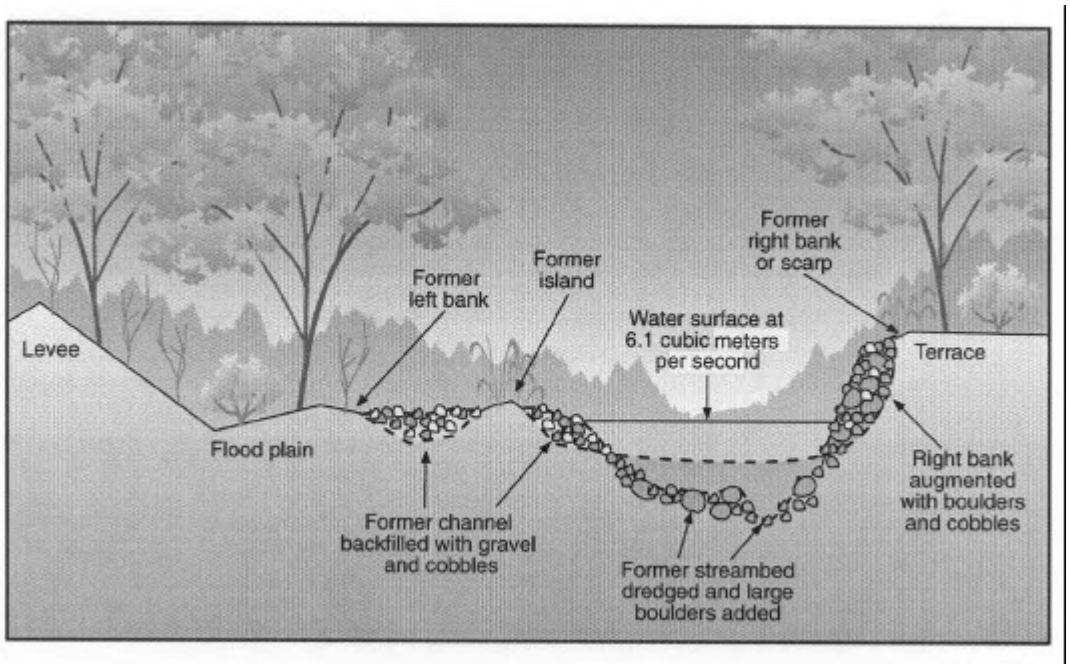

Figure 5. Lake Fork of the Gunnison River at Gateview cross section 6 showing reconfigured channel geometry in 1998. The formerly wide and shallow channel has been narrowed and deepened by redistribution of material from the streambed to the right bank and left cobble bar.

A 2.0-km reach of the North Fork was monumented and surveyed by the USGS in March 2000. As with the Lake Fork, permanent reference marks were installed for vertical and horizontal control. The reference mark locations (latitude and longitude) were determined with a GPS receiver to facilitate replicate surveys. The survey consisted of longitudinal profiles of the streambanks, terraces, and the water surface at a streamflow of approximately 4.5 m^3/s. Sixteen channel cross sections were surveyed in the study reach. Sediment characteristics were determined with the Wolman (1954) method and monumented oblique photographs were made.

The principal difference between the Lake Fork and North Fork reconfiguration methods was that the Lake Fork channel was designed to function with the assistance of several boulder structures whose purpose was to fix the location of the channel and high-velocity flow thread: whereas, the North Fork channel was designed to function as an alluvial channel, largely without reliance on fixed-location boulder structures.

FUTURE ACTIVITIES

Channel adjustments are the expected behavior of fluvial rivers; however, the rate of channel adjustment can range from imperceptible to dramatic and can affect river function and water-resource utilization. The recently monumented channels will be resurveyed and rephotographed, and the sediment will be recharacterized in the future. Replicate measurements will be made to quantify changes in channel geometry and sediment-size characteristics and to determine how and why a particular reconfiguration design may have remained stable or failed. The replication interval will be determined largely by year-to-year streamflow characteristics (recurrence of floods) and the presence or absence of geomorphic adjustment.

Figure 6. North Fork of the Gunnison River near Hotchkiss. View looking upstream . Prior to reconfiguration, the reach was braided. The main channel, previously on the left of the photograph, has been converted into a backwater and wetland area.

Channel modification and reconfiguration projects have been considered for many other river and stream reaches in the Western United States. The RCMAP will include surveys of other recently reconfigured stream reaches and will revisit previously monumented reaches as opportunities arise. Data from new river reaches and replicate surveys at previously monumented reaches also will be added to the data base and subsequently analyzed by the USGS.

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IS YOUR MESSAGE BEING HEARD?

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It is said, "A picture is worth a thousand words." That is probably true IF the receiver of that picture clearly understands the meaning of the picture. In other words, was the message received the way it was intended?

A visual image, just like the spoken or typed word, is only as good as the information contained in the message and how well it is received and understood by the audience. It is possible that the aforementioned picture may create more than a thousand words of discussion if it doesn't clearly transmit the intended message to the audience.

Communication, as defined by Webster, is "a process by which information is exchanged between individuals through a common system of symbols, signs, or behavior." Communication, or expressiveness, as Arthur Plotnik refers to it, is "an onslaught of stimulation that seizes and engages an audience." Oversimplified, communication is just about everything you do. Whether you realize it or not, whenever you do anything witnessed, heard or read, communication is happening.

A rather recent (Spring 2000) example of communication was the removal of Elian Gonzalez from his relatives' home in Florida. You may be visualizing that image of Elian facing a federal agent with an assault rifle. But how many of you are thinking of the picture taken several weeks later, as a smiling Elian and his father are getting on the airplane to return to Cuba? Messages sent and received often cause much rhetoric and enflame emotions. Were those the intended reactions of the photos?

Communication doesn't have to be quite as dramatic. Consider your normal daily activities, such as reading, writing, day dreaming, riding the elevator, watching a soccer game, looking at a backward baseball cap. Whatever you see or do, you are giving and receiving messages and images. And from them, you and others are drawing conclusions and making decisions. This is information transfer no matter what form that information transfer takes.

In all its forms, communication should ensure that your intended messages, which is what the sender wants a receiver to know or understand, are effectively delivered to a target audience. That is accomplished through the use of various signals such as words, visuals, colors, designs, or music. Then the receiver uses one or more senses to receive the information, decode and interpret the information. That is the "simple" part.

But what does it take to begin the process to communicate? Behavioral scientists say the sender must engage the audience - overcoming any resistance to the message (Plotnik). How that is done becomes a challenge. Therefore, a basic question is: "How do you make sure you communicate what you want communicated?"

First, understand your audience and plan to meet its needs. So, developing the message to meet an audience's needs becomes a critical factor in ensuring that your intended message is heard the way you want it to be and interpreted in such a way that an intended action does take place. Carefully consider the key to audience identification - what does the audience know about your topic?

Second, determine what is supposed to happen on the receiver's end. What does the sender want the receiver to do? Is it awareness, understanding, or motivation to action? The clever manager of the message may touch on a long-standing passion or dislike to incite a reaction. That comes from understanding the audience. The difficulty may come in judging how the audience received the message. Did the audience understand? Did the message motivate the audience to action? And was that action the kind of action the sender wanted? Another factor, which is often not possible to judge, is the temporal factor between receipt of the message and when action takes place.

Third, make sure your message is organized and transmitted in a way that your audience will clearly understand what you are trying to communicate.

Hence, is it fair to say that the most important thing any organization does is communicate? After all, what are technical documents? A communication method. And how are those technical references used? By someone wanting the information? In other words, reinforcing the classic definition of communication - a sender has a message, and a receiver wants information about that topic. The sender wants, indeed needs, feedback. That helps determine if the message was received, how it was received, and what the results were. And that is critical to achieving a communication objective.

So how does communication fit with sedimentation? Neatly. Let's look at a basic concern. How many people truly understand or care about sedimentation? Probably not many, except those few who are scientists, planners, or conservationists. <u>If</u> that is true, how do you get your work to the populace in general? Maybe you don't want to. That, in it, is a communication decision. If you do, though, who is the intended audience, and what do you want them to do? And why? What motivations do they have that match the intended outcomes of a communication strategy?

If you have as an objective, for example, reducing sedimentation though the use of conservation practices, how is that going to be accomplished? Who will be the most influential group to actually make that happen? Probably, private landowners who control 70% of the land and 80% of the water in the United States. Therefore, some of the information about sedimentation needs to be targeted to that group.

Now we are touching on developing a communication strategy that uses the many issues and science surrounding sedimentation, plus the technical work, human emotions, and ideas, and neatly packaging them into a plan that will create awareness, understanding, and action. Who you actually target will be determined by understanding your audience and what it needs to know. That may consist of just one speech or visual presentation, or a long-term series of coordinated activities. Whatever you decide, it needs to be focused on the needs of the targeted group.

Therefore, if communication is considered a critical element for your organization, it needs to approach it in a disciplined, planned, and systematic method, just as any other activity. After all, you do want results. But, without planning, how would you know what results you want and whether you've achieved them?

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CUSTOMIZED TECHNIQUES FOR INTERPRETATION OF SUSPENDED SEDIMENT DATA

By Donald E. Stump Jr., Hydrologist, Office of Surface Mining Reclamation and Enforcement, Pittsburgh, Pennsylvania

Abstract: With increased public awareness of the environmental impacts of land disturbance, regulatory agencies are requiring more information and analyses. Sediment transport is just one of the parameters required to be evaluated. Regulatory agencies need to have an analysis of the impacts of proposed land disturbance on sediment transport before a permitting decision can be made. The relationship of water discharge to suspended sediment concentrations varies from location to location depending on various factors. A simple and accurate method of evaluating this relationship is needed. Customized suspended sediment transport curves can be used as part of this evaluation. They characterize the suspended sediment transport for individual sites and are useful to predict the impacts of proposed land disturbance.

The USGS-Water Resources Division publishes daily values suspended sediment concentrations annually. The suspended sediment transport curves for daily value data tend to have significant scatter. Customized suspended sediment transport relationships created, using techniques to sort and filter the data minimize this scatter. These customized relationships are created by sorting the data for season, total storm period, time since the previous storm, precipitation intensity, etc.

These customized sediment transport curves provide the foundation for creating and calibrating predictive models for use by industry, government, and the public to support land disturbance decisions.

INTRODUCTION

Soil erosion and transport results when soil is exposed to the energy of rainfall and flowing water. It is not possible to conduct earth moving and disturbance without exposing soil to these erosive forces (Barfield and Warner, 1985). Land disturbances may cause temporary increases in soil transport while others may cause permanent increases. This transport can be accelerated if the surface drainage patterns are altered and the volume of excess runoff is increased along with the soil disturbance.

Government agencies must make decisions involving significant land disturbance for residential development, industrial use, agricultural use and resource extraction. The agencies should evaluate the land disturbance and its impacts to sediment transport as part of their permitting process. Water quality criteria and/or maximum allowable levels that have to be met may already be in the regulations for total suspended sediment.

Using a customized suspended sediment transport curve is a method that can be used to evaluate land disturbance impacts to sediment transport. The transport curves can also be used to provide

governmental agencies with a basis for making impact predictions for proposed land disturbances.

LAND DISTURBANCE

Earth moving equipment is used in many industries as part of their everyday operations and during construction and demolition. There is land development for housing, road construction, surface mining of minerals and ores, agricultural activities, etc. All of these activities require land disturbance and the potential to expose soil particles to the wind and water for transport.

Construction of roads can significantly increase erosion potential when surface vegetation is removed and the ground surface is unprotected.. At an interstate highway construction site the annual suspended-sediment discharge increased during construction from 100 to 300 percent, but returned to preconstruction rates after work was completed. (Reed, 1980)

Mining operations may introduce large volumes of sediment directly into streams. Mine dumps and spoil banks, which are left ungraded and not vegetated, often continue to erode many years after mining. (Vanoni, 1977) Sediment yield from surface mine spoil banks increased by 1,000 percent over pre-disturbance levels. Sediment loss from the mining haul roads also produced significant amounts of sediment. (Collier, 1970)

Agricultural activities have caused accelerated erosion through deforestation to clear the land, cultivation, and altering drainage patterns. The most significant source of sediment is from sheet and rill erosion. (Brakensiek, 1979) The results of plot studies indicate this conversion from forested cover to agricultural activity accelerates erosion from 100 to 1,000 fold. (Musgrave, 1957)

SEDIMENT TRANSPORT

Sediment transport is only one part a sediment particle's history. Transport is sandwiched between the conditions that mobilized (erosion) the particle and the conditions that will allow the particle to be deposited (sedimentation). When flowing water is evaluated for its quality and suitability for various uses, its sediment transport characteristics are important.

Suspended sediment samples provide the basic data from which sediment transport curves are developed. In the early days of fluvial sediment investigations, each investigator and at least each agency developed methods and equipment according to need. To insure comparable results standard sediment samplers and methods were developed through committees and laboratories. (Guy, 1973)

Surface runoff from rainfall create water hydrographs with characteristic rising limbs and falling recessions. Of importance is the peak discharges in the evaluation of flooding and sediment transport. The variation of sediment concentration with respect to the storm runoff hydrograph

may help to characterize the sediment transport. The relationship of the sediment peaks to the hydrograph peaks are identified as advanced, simultaneous, and lagging concentration graphs. The advanced type is the most common. The magnitude of sediment concentration for a typical sediment concentration graph at a given stream location will vary considerably depending on the season of the year, the changing patterns of land use, the antecedent moisture conditions and the nature of the precipitation intensity and pattern on the basin. (Guy, 1978)

Most of the sediment transport is done during periods of excess runoff from storms. Sediment yield generally increases geometrically with storm runoff rate. (Guy, 1978) Under certain conditions, it is possible for high-intensity runoff events to attain maximum concentrations at the beginning of runoff. This is true for watersheds where the weathering of soils or stream beds, or both, during long, dry periods has produced a large readily transportable load of fine material. If the storm is intense, the stream sediment concentration is comparatively high at the start of excess runoff and increases rapidly. These conditions result in a "loop" effect on the sediment transport graph for a given storm. (Vanoni, 1977)

SUSPENDED SEDIMENT TRANSPORT CURVES

The relation between water discharge and sediment discharge may be expressed by an average curve. This curve is called a sediment-transport curve. The types of sediment-transport curves are numerous. They may be classified as instantaneous, daily monthly, annual, or flood-period curves. (Porterfield, 1977) Traditionally sediment transport curves use daily values data to characterize the flow of suspended sediment past a site. These curves tend to have significant scatter because they include days of base flow, excess runoff, snowmelt, and reservoir releases.

To develop these transport curves there are usually two basic types of sediment records. They are published by the U. S. Geological Survey as daily values or periodic data used for research and interpretive reports. (Porterfield, 1977) in addition to this public sediment data individuals can collect information for their specific areas of interest.

The size and characteristics of the drainage area, the climatic conditions, and the magnitude and period of land disturbance will control the time scale and site locations that should be used for the analysis. Smaller drainage areas may need to be evaluated at a time scale that is less than one day while large drainages may be evaluated using daily values.

CUSTOMIZED SEDIMENT TRANSPORT CURVE

Sediment transport curves can be refined or customized to document the characteristics of sediment flow past a site, compare curves from multiple sites, and document changes at a site. These customized relationships are created by sorting the data for season, total storm period, time since the previous storm, precipitation intensity, etc.

To show how customized suspended sediment transport curves can give a better representation of

the impacts of land disturbance on sediment transport two sites will be used. One site was upstream of the land disturbance and the other site was downstream. The land disturbance evaluated was surface coal mining. The sites were located in Southwestern Pennsylvania. The upstream site was 0.9 square mile and the downstream site was 7.4 square miles. Automatic pumping sediment samplers were used to collect the suspended sediment samples. The samplers were configured to collect additional samples during storms. (Stump, 1985)

Two years of daily values suspended sediment data were used and are noted as “year 1" and “year 2". During year 1 little mining was occurring between the two sites. During year 2 data a portion of the area between the sites was disturbed by surface mining.

Monthly values of sediment load and discharge were graphed for both sites for the two years of data. Figure 1 is the graph of the data for the upstream site and figure 2 is the graph for the downstream site. The graphs show a significant amount of scatter. A best-fit line is drawn on each graph for each year of data. The best-fit lines intersect on each graph and do not indicate a change in the sediment transport at either site. The impact of the land disturbance from the surface mining would be expected to appear in figure 2 at the downstream site. Based on this analysis the impacts of the mining operation on the sediment transport appears to be negligible.

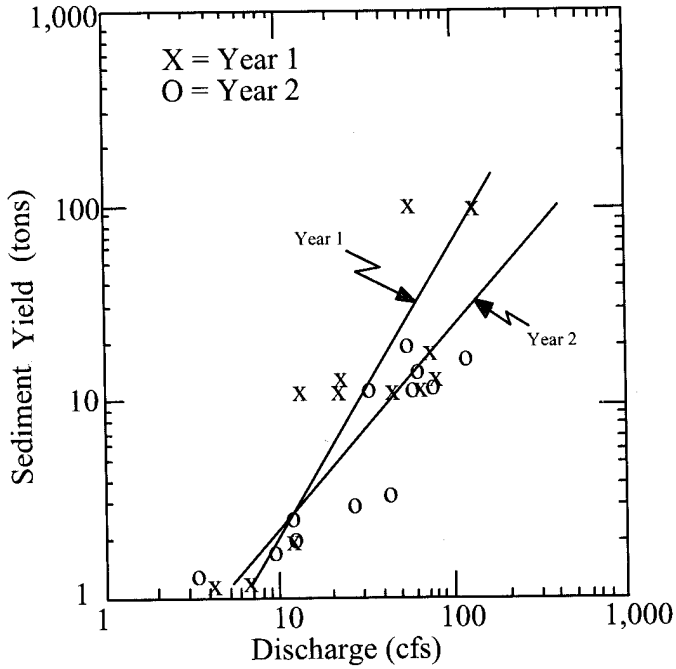

Figure 1 : Monthly Sediment Yield and Discharge
Upstream Site

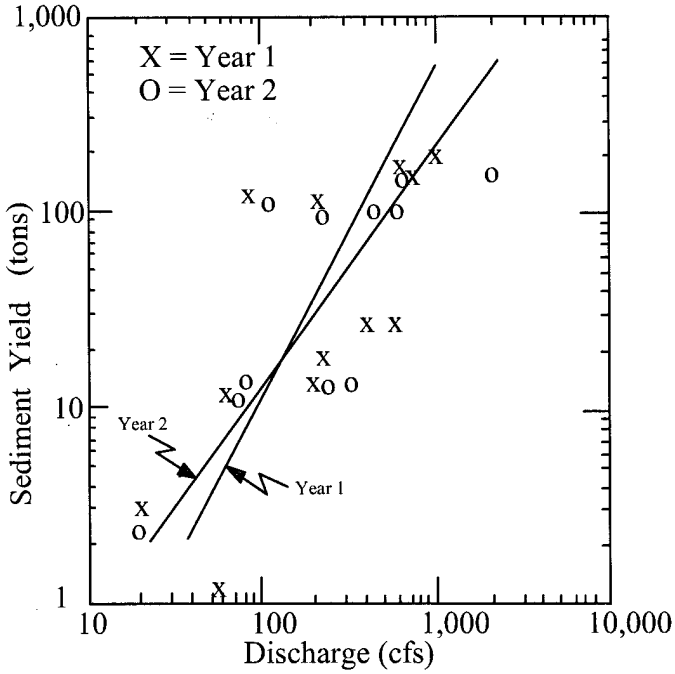

Figure 2 : Monthly Sediment Yield and Discharge
Downstream Site

A customized sediment transport curve was then graphed. The customized graph also used the daily values data but several filters were used to select the data for the sediment transport curve.

Only data from periods of excess runoff were used (storms). The sediment yield for each storm was calculated from the daily sediment data. This was done by summing the days associated with each storm. The increase in discharge from the beginning of the storm to the storm peak was used as the other parameter in the sediment transport curve. These customized sediment transport curves are shown in figures 3 and 4. Figure 3 is the storm sediment yield and discharge increase graph for the upstream site for years 1 and 2. Figure 4 is the sediment yield and discharge increase graph for the downstream site. All of the storms during the two year data collection period were plotted. The best-fit average lines on figure 3 shows the similarity of the sediment transport characteristics for the two years. The data scatter on figure 3 for the upstream site is less than the scatter in figure 1. The best-fit average lines on figure 4 shows how the sediment transport characteristics changed from year 1 to year 2 at the downstream site. Figure 4 shows a distinct migration of the sediment transport curve from year 1 to year 2. For any it given storm the sediment yield was greater in year 2 than in year 1.

Figures 3 and 4 would support the conclusion that the land disturbance from the mining activities between the upstream station and the downstream station increased the sediment transport. This conclusion contradicts the conclusion, of no impact, that was already drawn from figures 1 and 2. The customized sediment transport curves in figures 3 and 4 allow for a better interpretation of daily values sediment data to identify the impacts that land disturbance can have on sediment transport.

SUMMARY

The evaluation of sediment transport is only one part of the sediment system. Considerations need to be made to see where the sediment is coming from and where it is being deposited. This may be considered as a cradle to grave analysis similar to what is done with certain chemicals.

Customized sediment transport curves can be developed to evaluate the impacts of land disturbance. By selectively filtering the sediment data to minimize some of the scatter changes in sediment transport characteristics caused by land disturbance can be identified. These customized sediment transport curves can provide the foundation for creating and calibrating predictive models for use by industry, government, and the public to support future land disturbance decisions.

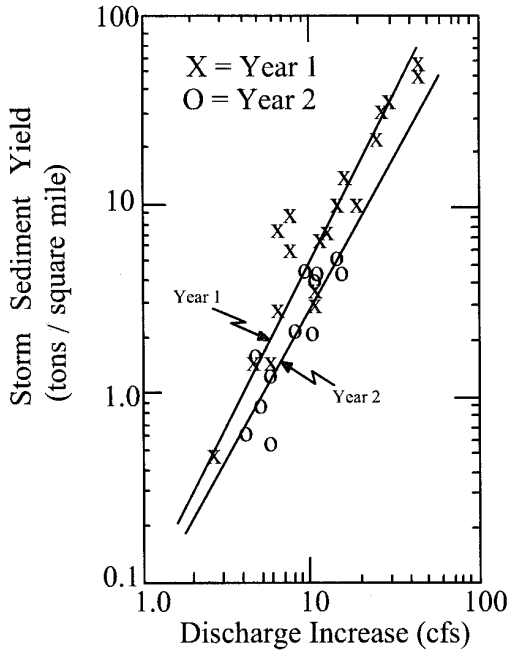

Figure 3 : Storm Sediment Yield and Discharge Increase, Upstream Site

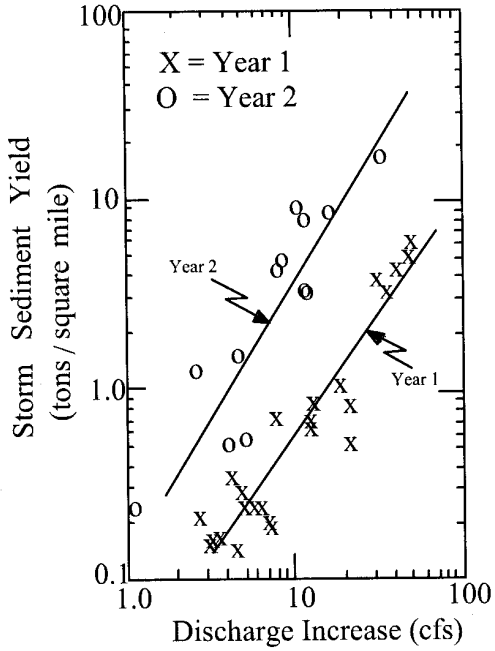

Figure 4 : Storm Sediment Yield and Discharge Increase, Downstream Site

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SEDIMENT SOURCES IN THE LOWER ATHABASCA RIVER, CANADA: IMPLICATIONS OF NATURAL HYDROCARBON INPUTS FROM OIL SAND DEPOSITS

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Abstract: The Athabasca River drains an area of 160,000 km^2 in northern Alberta, Canada. Oil sand deposits underlie a considerable portion of the lower Athabasca River basin. The oil sands primarily occur in the McMurray Formation of the Cretaceous Period, with outcrops evident along the banks of the Athabasca River, as well as the lower portions of several tributaries. Extraction of oil from these deposits constitutes a major and rapidly expanding industry, the environmental consequences of which, are a concern to government, industry and the public. The ultimate ecosystem consequences of these activities are unclear and are the subject of ongoing investigations. Many investigations have focused on the nature of effluent produced by the industry and its associated impact to biotic systems. Investigations have also been undertaken to assess the spatial distribution of these hydrocarbon contaminants, typically by determining the amount of hydrocarbon associated with deposited fluvial sediments. Natural hydrocarbon substrates, however, are exposed throughout the oil sands area and studies have suggested that these outcrops may be responsible for observed biological responses in areas not exposed to industrial effluent. Given that the oil sands represent a natural source of hydrocarbons to the environment and with no quantification of this source, it is currently difficult to determine the true impact of oil recovery activities within this aquatic ecosystem. In order to assess and predict potential impacts of hydrocarbon development activities occurring in the Alberta Oil sands area, it is necessary to distinguish these impacts from those produced by naturally occurring hydrocarbon deposits and releases. To accomplish this it is also necessary to have an understanding of the nature and extent of natural hydrocarbon input sources to this fluvial system. This paper describes some of the fluvial geomorphic characteristics of the lower Athabasca River and provides a context for assessing sediment-bound hydrocarbon contaminants within the context of the sediment regime of the lower Athabasca River basin.

INTRODUCTION

The Athabasca oil sands deposit in northeast Alberta (Figure 1) is the largest Cretaceous oil sands deposits in Alberta and is considered to be one of the largest single accumulations of oil in the world (Flach, 1984). As such the area has undergone varying levels of development over the past one hundred years. Recent increases in hydrocarbon development in the area coupled with the results of a five-year, $12 M research initiative, referred to as the Northern River Basins Study (NRBS) which included the Athabasca River Basin, has highlighted the issues of the potential impacts (NRBS, 1996). Specifically, NRBS studies found that there was evidence of ecological stress on fish found in the mainstem of the Athabasca River adjacent the oil sands development (Wrona at al, 1996). Of further interest is evidence of biomarker responses in areas where there are natural exposures of oil sands and no active development.

Both oil sands development and natural exposure to oil sands deposits may result in concentrations of hydrocarbons and other substances, in both the water and sediment, which can cause stress to fish and other biota. As a result of the NRBS studies and the recent expansion of the oils sands industry, a four-year program (1998-2002) to provide an *Assessment of Natural and Anthropogenic Impacts of Oil Sands Contaminants within the Northern River Basins* was developed (Wrona and Cash, 1997). This research initiative is supported by the Panel on Energy Research and Development (PERD) and Environment Canada (EC). The goal of this research program is to assess and predict potential impacts of hydrocarbon activities and separate these impacts from those produced by naturally occurring hydrocarbon deposits and releases. In order to meet this goal it is necessary to have an understanding of the spatial distribution, nature and extent of natural hydrocarbon releases to the aquatic environment.

As an initial step in understanding the availability of natural sources of sediment-bound hydrocarbon contaminants, an assessment of the fluvial characteristics of the lower Athabasca River basin was undertaken and

includes an evaluation of potential sediment source areas within the reaches of the lower Athabasca River. Emphasis was placed on quantifying the potential contribution of sediment derived from naturally occurring oil sand exposures and evaluating this in the context of the sediment regime of the lower Athabasca River basin.

STUDY SITE

The Athabasca River originates from the eastern slopes of the Rocky Mountains in Alberta, Canada. The river trends northeast across Alberta draining an area of approximately 160,000 km^2 and eventually discharging into Lake Athabasca (Figure 1). The main focus of this study is the lower portions of the Athabasca River, primarily downstream of the community of Fort McMurray (Figure 2). The lower portions of the Athabasca River drains an area of approximately 58,000 km^2 including the Clearwater River basin. The Athabasca-Clearwater rivers are located in a post-glacial spillway, with the lower portions of the Athabasca River cutting into a late Pleistocene braid delta (Rhine and Smith, 1988). Boreal forest and muskeg dominate the surrounding uplands. The mean annual flow of the Athabasca River at Fort McMurray is approximately 650 m^3s^{-1} with lower flows occurring in the late winter, when catchment runoff is at a minimum, and peak flows occurring in late June or July, corresponding with mountain runoff (HYDAT, 1994). Mean monthly winter flows typically do not exceed 180 m^3s^{-1} and monthly peak flows in July range from 1400-1500 m^3s^{-1} on average. The Athabasca River has had flows less than 90 m^3s^{-1} and in excess of 4700 m^3s^{-1}.

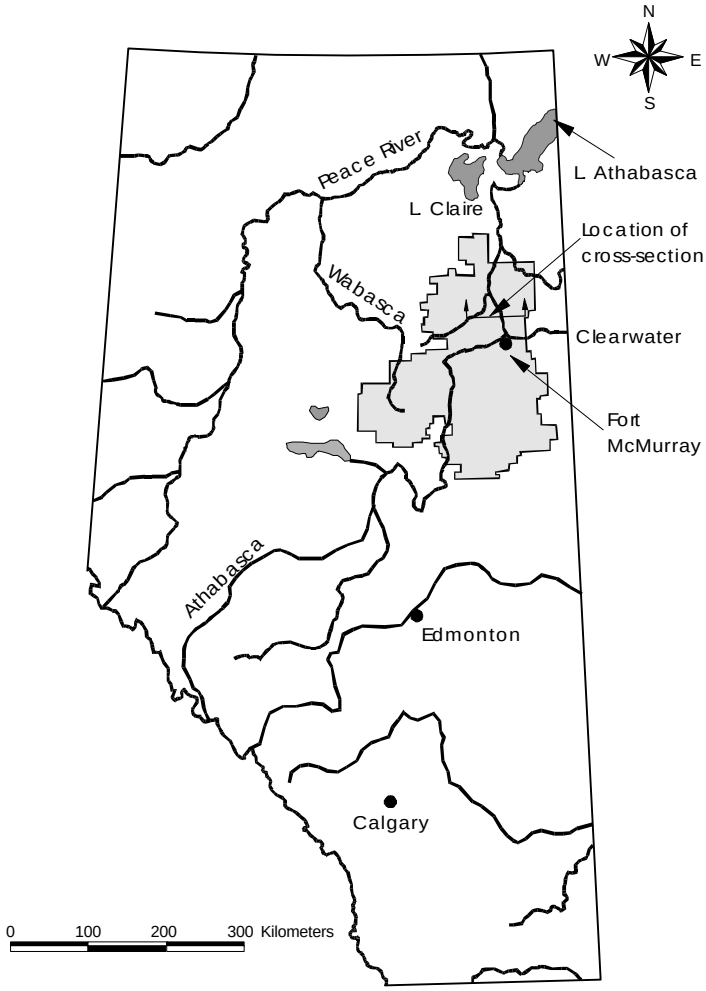

Figure 1: Location of Athabasca River in Alberta, Canada. Shaded area indicates area of active oil sands development in northeastern Alberta.

The climate of the lower Athabasca River basin can be generally characterized as relatively dry with cool summers and cold winters (Atmospheric Environment Service, 1993). Mean daily temperatures in January are approximately –20 ^oC and July mean daily temperatures are around 15 ^oC. Average annual precipitation is less than 500 mm with over 60% occurring as rainfall and the remainder as snowfall.

GEOLOGY OF THE OIL SANDS

Oil sands in the lower Athabasca region are found in the McMurray Formation of the Cretaceous Period, which is underlain by shales and limestones of the Waterways Formation (Devonian) and overlain by shale and sandstones of the Clearwater Formation (Figure 3). The thickness of the formation depends considerably

on the underlying unconformity and varies from over 150 m (500 ft) in the centre of the deposit to zero in the west where it eventually pinches out against a paleo-topographic ridge of Devonian limestone (Flach, 1984). The overlying Clearwater Formation, consisting primarily of marine shales, and on top of this is the dominantly sandstone Grand Rapids Formation (Figure 3).

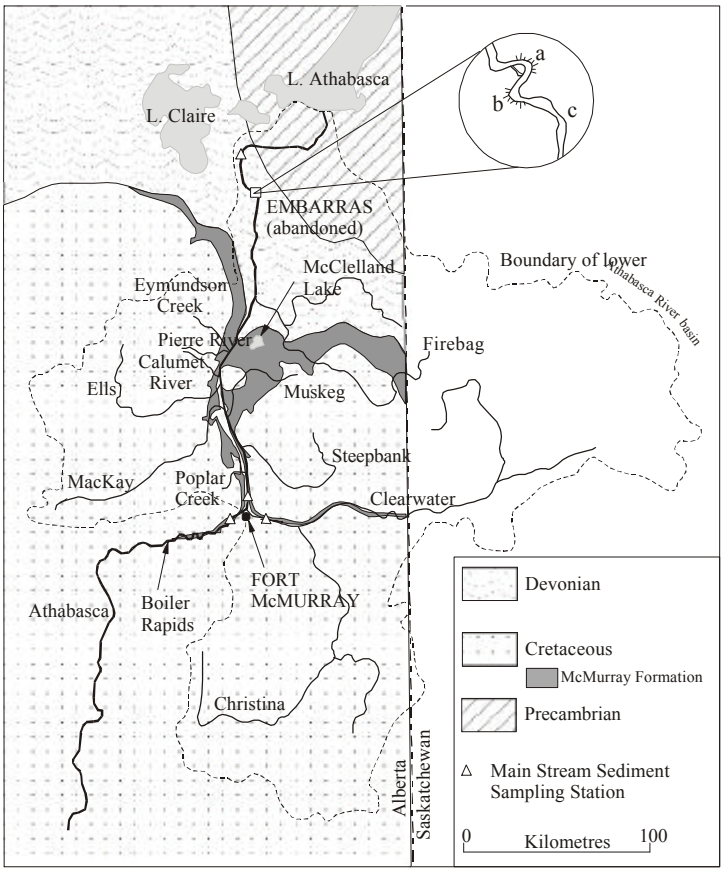

Figure 2: Simplified geology of the Lower Athabasca showing the extent of the McMurray Formation. Inset map shows the erosional meander bends near Embarrass – see text for discussion.

Although there is evidence of hydrocarbon reserves along a relatively thin layer of glauconitic sands at the base of the Clearwater Formation, they are small compared to those found in the McMurray Formation. Over all of the sedimentary rock is a spatially variable mantle of unconsolidated Quaternary deposits, primarily glacial, fluvio-glacial and glaciolacustrine sediments (Figure 3).

The texture of the McMurray Formation itself is variable, and includes shale, interbedded shale and sandstone as well as oil-impregnated sands. The McMurray Formation has been subdivided into a lower, middle and upper member, based on depositional history (Carrigy, 1959). Ease of bank erosion in the McMurray Formation may be quite variable depending upon the extent of shale interbedding with the oil sands. Flach (1984) notes, however, that while the oil sands are essentially uncemented, the oil found within the pores is virtually immobile at reservoir temperature, and thus the material (bitumen) behaves more as a solid than as a fluid.

Oil Sands Exposure on the Lower Athabasca

The McMurray Formation strata are first exposed in the Athabasca riverbed at Boiler Rapids approximately 50 km upstream of Fort McMurray (Figure 2). It is replaced in the riverbed about 15-km downstream by the underlying Waterways Formation, but are found in the valley walls, almost continuously, as far downstream as Eymundson Creek. The McMurray Formation becomes exposed to the riverbed again near the McKay River. Downstream of Eymundson Creek geologic maps indicate exposures of the Waterways Formation and older formations of the Middle Devonian (McPherson and Kathol, 1977), but bedrock exposures become increasingly masked by fluvial-glacial deposits and more recent alluvium.

Between Boiler Rapids and the Clearwater River Valley the Athabasca River flows through a narrow canyon that was carved in post-glacial times. In contrast, the present Clearwater-Athabasca (below Fort McMurray) drainage in the oil sands region follows a late-glacial trench excavated by melt water spillage from water impounded at the ice-sheet front to the east (Smith and Fisher, 1993), and the present drainage has incised further into this trench. The actual height of the walls flanking the present river varies and depends, to a large extent, on whether the new

valley sides have been cut laterally into the older higher-level walls, or whether the river has merely cut down into the floor of the older trench.

The largest tributary to the lower Athabasca River basin is the Clearwater River. Maps of bedrock geology indicate that the valley walls of the Clearwater River are flanked by the McMurray Formation from Fort McMurray to approximately the Alberta Saskatchewan border (Figure 3), a distance comparable with the length of exposure along the Athabasca River downstream of Fort McMurray. The basal parts of the slopes are Waterways Formation, and the tops of the slopes are Clearwater Formation, overlain by Grand Rapids Formation. The valley walls are masked by a layer of colluvium with no exposures of the McMurray Formation indicated on local bedrock geology maps, though outcrops of the overlying Clearwater Formation are indicated in the east (Alberta Research Council, 1970).

Several small tributaries are also incised into the McMurray Formation, typically along the lower reaches prior to discharging into the Athabasca River. In several circumstances the down cutting of the Athabasca River into the surrounding plain and particularly into the McMurray and Clearwater Formations, has resulted in some tributaries developing a steep gradient as they discharge into the Athabasca River.

SEDIMENT REGIME OF THE LOWER ATHABASCA RIVER

The primary sediment sampling station on the lower Athabasca River is located near Fort McMurray. Sediment sampling was first initiated at this site in 1967 and operated continuously between 1969 and 1979. The program shifted to a seasonal operation and continued until 1987 since then only miscellaneous samples have been collected. Another sediment sampling station was located further downstream near Embarras and operated seasonally between 1976-84 (Figure 2). A sediment budget analysis by Carson (1990) for the common period of record (1976-84) found the mean annual suspended sediment load at Embarras (at 6.3 Mt) to be 1.1 Mt greater than at Fort McMurray (5.2 Mt) (Table1). Using supplementary sediment data for tributaries and estimates of sediment contributions from non-monitored basins, Carson (1990) estimated that only around 55% of the suspended sediment originated from tributary inputs (the largest being the Clearwater- see Table 1). The balance, approximately 0.5 Mt, was assumed to originate from bank erosion between the two stations.

Table 1: Estimated mean annual suspended sediment loads at Environment Canada river gauging stations in the Athabasca oil sands area, Alberta, 1976-84 (based on data from Carson, 1990).

Station	Load (kt)
Mainstem stations:	
Athabasca River at Embarras	6340
Athabasca River at Ft. McMurray	5200
Difference	1140
Tributaries:	
Clearwater River at Draper (near Ft. McMurray)	400
Other tributaries	210
Total tributary inputs	610

A recent evaluation of channel stability along the mainstem of the lower Athabasca River, as part of the PERD/EC study, indicated only localized instability throughout the reaches that are exposed to oil sands strata. Moving downstream from Fort McMurray to approximately Poplar Creek, the Athabasca River hugs the right side of its valley bottom, with a floodplain separating the left valley wall from the channel. Valley wall erosion and instability is therefore restricted to the right bank through this reach, with some additional sediment inputs from the lower course of small right-bank tributaries. Continuing downstream, the old bed of a melt water channel flanks the Athabasca River downstream to the McKay River and thus the river has little opportunity to erode the valley walls. From just upstream of the McKay River, the Athabasca River becomes weakly sinuous, with abundant vegetated islands which cause an over-widening of the channel where they occur. Although there is no obvious pattern to the distribution of islands, a pattern of alternating lateral bars is apparent. Moving downstream towards Eymundson Creek, the clusters of vegetated islands take on a distribution along the channel similar to that of ‘ alternate bars’ . This rudimentary meandering thalweg morphology is characteristic of rivers where the bank material is too resistant to allow sufficient bank erosion to produce true meanders or for the channel to widen generally allowing true braiding.

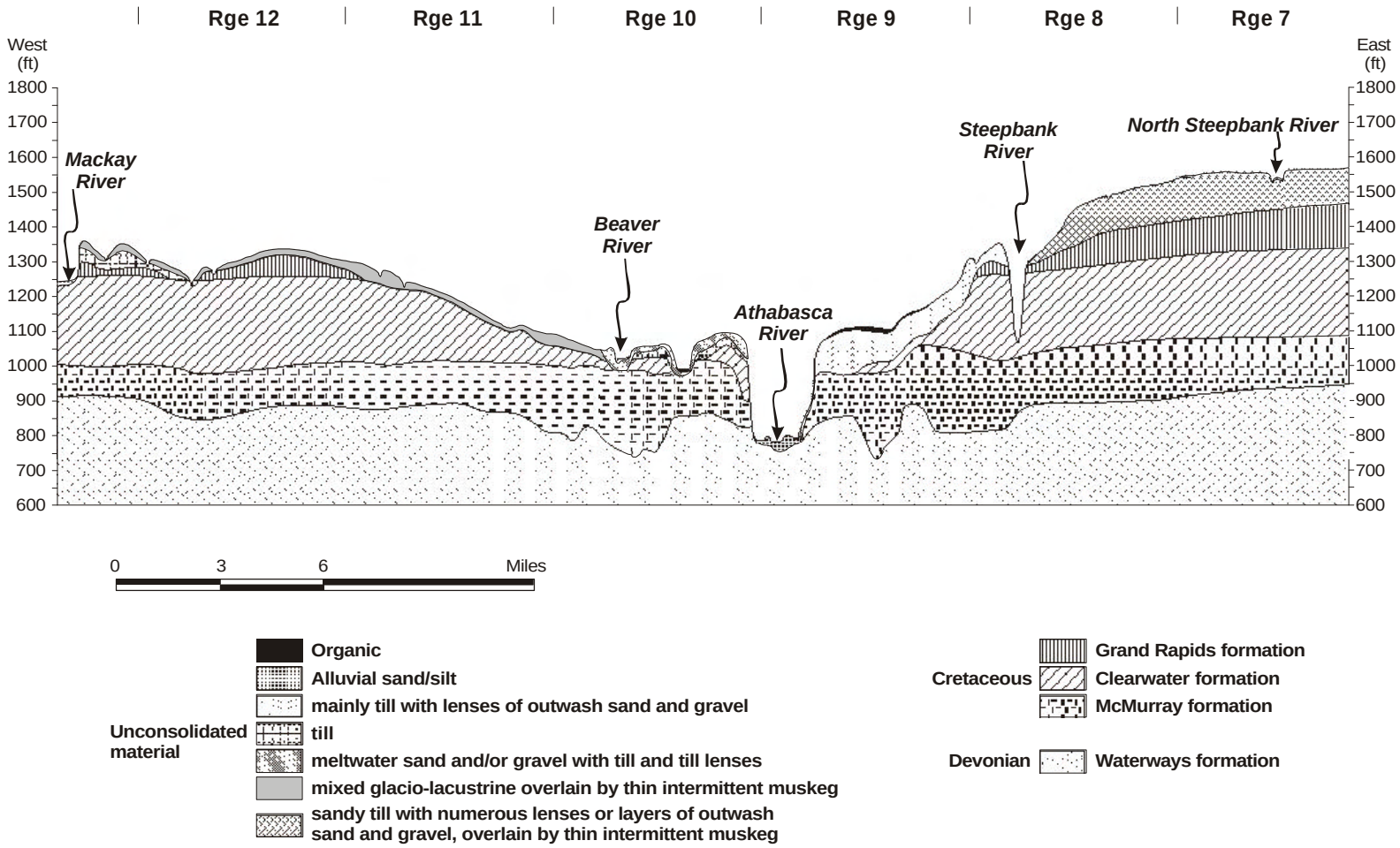

Figure 3: East-west geologic cross-section through the oil sands area in northeast Alberta Canada. Approximate location of cross-section is indicated on Figure1. (after McPherson and Kathol, 1977)

Downstream of Eymundson Creek (roughly the downstream limit of McMurray Formation exposures), the Athabasca is flanked by Quaternary sediments. Initially, this reach is relatively narrow, and variable degrees of re-vegetation along the valley walls, suggest old sites of sediment delivery to the river. In most cases, these old sites are now separated from the river by laterally accumulating floodplain sediment. As the paleo-delta area broadens down river, old cliffs have become insulated from the Athabasca River, over almost its complete length, by wide tracts of floodplain. While undercutting of the floodplain itself may be locally important, volumes of sediment would be expected to be much smaller than in the case of the tall cliffs, and the coarser sand sediment eroded from one bank site is likely to accumulate a short distance downstream.

There is, however, one major sediment-producing site on the lower Athabasca between the McMurray and Embarras sediment stations. This is the second (right bank – site ‘ a’ on inset in Figure 2) of two well developed, large meander bends immediately downstream of Embarras. The first (left bank – site ‘ b’ on inset in Figure 2) bend is cut into low alluvial deposits, with a bank height of less than 8 metres. The second bend, however, undercuts a terrace in the early post-glacial paleo-delta sediments (Rhine and Smith, 1988). This cliff is about 27 m high, based on the 21 m height above water level as described by Rhine and Smith (1988), and estimating another 5 to 6 m of submerged base, which would be expected given the location on the outside of a meander bend. Undercutting is indicated on air photographs along about 3200 metres of this bend. An examination of the position of this long cliff on aerial photographs taken in 1953, 1967 and 1984 indicates an average retreat rate through the bend in excess of three metres per year. Assuming a bulk density of 1.6 t/m^3, the mean annual sediment supply rate during 1976-84 is conservatively estimated at 0.4 to 0.45 Mt/yr.

Another site of active erosion of Quaternary sediments occurs at Embarras (site ‘ c’ on inset in Figure 2). Cliffs at this site are estimated at approximately 22 m tall and extending approximately 1300 m. Using a mean retreat rate of 0.7 m/yr yields and the same bulk density as above, the sediment contribution from this site is approximately 30 kt/yr or less than 10% of the downstream meander bend.

Although further photogrametric analysis and onsite morphometric surveys would be useful in refining these estimates, the sediment supply from these cliffs compares with a tentative estimate of bank erosion of about 0.5 Mt/yr as provided by Carson (1990) for the lower Athabasca River between Fort McMurray and Embarras. The implication seems to be that comparatively little bank sediment is produced along the rest of the Athabasca River between Fort McMurray and Embarras stations, which is consistent with the small degree of bank instability observed between these sites.

This comparison assumes that the bulk of the sediment undercut from cliffs becomes incorporated into the suspended load of the Athabasca River at Embarras. Rhine and Smith (1988) note that the bottom of the cliff sections are lacustrine mud. This lacustrine mud, which would certainly be moved in suspension, would likely constitute the lower third of the bank section. The rest of the section is dominantly sand, the texture of which is highly variable, depending on the facies. Granules and pebbles do occur, but are not abundant. Assuming, conservatively, that only half of the eroded cliff volume becomes incorporated into the suspended load, this sediment supply from the Embarras bend cliffs would still account for a significant portion of the suspended sediment load between Embarras and McMurray.

DISCUSSION

The data available indicate that tributaries and mainstem erosion in the Fort McMurray-Embarras reach could be supplying up to 1.1 Mt (18%) of the mean annual suspended sediment load of the Athabasca River. Approximately 55% of the sediment input along the lower Athabasca River (10% of the load at Embarras) originate from tributaries in the oil sands region, primarily the Clearwater River. It should be noted that tributaries that were sampled account for only 77% of the total drainage area of the lower Athabasca River, not including the Clearwater River. The load of the remaining 23% was determined by assuming that the specific sediment yield ($t/km^2/yr$) was the same as the average for those that were sampled (Carson, 1990). Rivers that were not sampled consisted primarily of those downstream of the oil sand exposures, though some, such as Calumet River, Pierre River and Eymundson Creek are underlain by McMurray Formation in their lower courses (Figure 2).

Analysis of these data suggests that roughly 45% of the 1.1 Mt increase in sediment load between McMurray and Embarras, that is about 8% of the Embarras suspended load, derives from valley wall erosion downstream of McMurray. However, most of the valley wall erosion in this reach is taking place in the general vicinity of Embarras, well downstream of the bed and bank exposures of the McMurray Formation. Thus valley wall erosion of *in-situ* oil sands along the main stem of the lower Athabasca River is not likely a significant source of sediment passing Embarras.

Of the tributary inputs, the Clearwater River is by far the most significant source of suspended sediment, with more than 60% of the total tributary contribution for the lower Athabasca River. How much of this suspended sediment of the Clearwater River is derived from McMurray Formation is difficult to assess. As indicated previously the valley walls are predominantly made up of colluvium. Although the genesis of this material is derived, in part, from the underlying basal material, it is unlikely that there would be a significant contribution of sediment from oil sand sources. Further research and site investigation will be required to confirm this speculation.

The Christina River, a major tributary to the Clearwater River, dissects oil sands strata in its lower course. Site investigation on the Christina River suggested that the majority of sediment supplied by this river occurred upstream of the exposed McMurray formation. As such, any sediment produced in the relatively short reach of the lower Christina River, that is exposed to the McMurray formation, would likely be highly diluted from sediment produced from upstream sources.

Assuming that the mainstem Clearwater River produces little suspended sediment itself, the corollary would be that the only major source of oil-sands sediment in the Fort McMurray-Embarras reach (apart from that derived from the Athabasca upstream of Fort McMurray), is from the tributaries to the lower Athabasca River. Using the figures provided in Table 1, the suspended sediment from these tributaries would account for at most 3.5% of the mean annual load passing Embarras. In addition, it is important to remember that some of the tributaries do not drain through areas that have oil sand exposures and that it is typically only the lower reaches of the tributaries that are incised into the McMurray Formation.

CONCLUSION

Based on the considerations of the fluvial characteristics and sediment regime of the lower Athabasca River basin, naturally occurring hydrocarbon-contaminated sediments are most likely to be detected in tributaries that are incised into the McMurray Formation. This assumes, of course, that there is no significant contribution of oil sand sediments from upstream of Fort McMurray. It is likely, however, that there are localized depositional accumulations oil sand derived sediments in the mainstem of the Athabasca River and almost certainly immediately downstream of these tributaries. By the time suspended sediment from the tributaries of the lower Athabasca River gauging site near Embarras, oil sand derived sediment from downstream of Fort McMurray will have been diluted to no more than 3.5% (an upper limit) of the river' s suspended sediment load.

A question remains, however, as to how much oil-sands sediment is produced upstream of Fort McMurray in the Athabasca River and its tributaries. Certainly there are similarities in the fluvial characteristics of the Athabasca River and its tributaries upstream of Fort McMurray, particularly in reaches exposed to the McMurray Formation, but the volume of oil sand derived sediment produced is currently unknown. Moreover, the degree to which this sediment would be diluted by hydrocarbon-free sediment from upriver is also a question that would need to be addressed.

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PREDICTION OF SEDIMENT TOXICITY USING CONSENSUS-BASED FRESHWATER SEDIMENT QUALITY GUIDELINES

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Abstract: Numerical sediment quality guidelines (SQGs) have been developed by a variety of federal, state, and provincial agencies across North America using matching sediment chemistry and biological effects data. These SQGs have been routinely used to interpret historical data, identify potential problem chemicals or areas at a site, design monitoring programs, classify hot spots and rank sites, and make decisions regarding the need for more detailed studies. Additional suggested uses for SQGs include identifying the need for source controls of problem chemicals before release, linking chemical sources to sediment contamination, triggering regulatory action, and establishing target remediation objectives. In the current study, the ability of probable effect concentrations (PECs) to predict sediment toxicity was evaluated using a database recently developed using matching sediment toxicity and chemistry data from throughout North America (1657 samples). The PECs are SQGs that were established as concentrations of individual chemicals above which adverse effects in sediments are expected to frequently occur. The database used in this evaluation was comprised primarily of 10- to 14-day or 28- to 42-day toxicity tests with the amphipod *Hyalella azteca* (designated as HA10 or HA28 tests) and 10- to 14-day toxicity tests with the midges *Chironomus tentans* or *C. riparius* (designated as CS10 test). Mean PEC quotients were calculated to provide an overall measure of chemical contamination and to support an evaluation of the combined effects of multiple contaminants in sediments. There was an increase in the incidence of toxicity with an increase in the mean quotients in all three tests. A consistent increase in the toxicity in all three tests occurred at a mean quotient >0.5, however, the overall incidence of toxicity was greater in the HA28 test compared to the short-term tests. The longer-term tests in which survival and growth are measured tend to be more sensitive than the shorter-term tests, with acute to chronic ratios on the order of 6 for *H. azteca.* Different patterns were observed among the various procedures used to calculate mean quotients. For example in the HA28 test, a relatively abrupt increase in toxicity was associated with elevated polychlorinated biphenyls (PCBs) alone or with elevated polycyclic aromatic hydrocarbons (PAHs) alone, compared to the gradual increase in toxicity observed with quotients calculated using a combination of metals, PAHs, and PCBs. These analyses indicate that the different patterns in toxicity may be the result of unique chemical signals associated with individual contaminants in samples. While mean quotients can be used to classify samples as toxic or non-toxic, individual quotients might be useful for helping to identify substances that may be causing or substantially contributing to the observed toxicity. An increase in the incidence of toxicity was observed with increasing mean quotients within most of the regions, basins, and areas in North America for all three toxicity tests. The results of these analyses indicate that the PECs can be used to reliably predict toxicity of sediments on both a regional and national basis.

INTRODUCTION

Numerical SQGs, when used with other tools such as sediment toxicity tests, bioaccumulation evaluations, and benthic community surveys, can provide a powerful weight of evidence for assessing the hazards associated with contaminated sediments (USEPA 1997, Long and MacDonald 1998). A critical component in the application of SQGs for assessing sediment quality is a demonstration of the ability of the guidelines to accurately predict the absence or presence of toxicity in field-collected sediments (Long *et al.* 1995, 1998a; Ingersoll *et al.* 1996, Smith *et al.* 1996, Field *et al.* 1999; Swartz 1999; Fairey *et al.* 2000; MacDonald *et al.* 2000a,b). A series of four papers has recently been published to evaluate the ability of various SQGs to predict toxicity in contaminated sediments. The first paper in the series focused on resolving the "mixture paradox" that is associated with the application of empirically-derived SQGs for individual PAHs. In this case, the paradox was addressed by developing consensus-based SQGs for total PAHs (Swartz 1999). A second paper developed and evaluated consensus-based SQGs for PCBs to address a similar mixture paradox for that group of contaminants (MacDonald *et al.* 2000b).

A third paper developed consensus-based SQGs for freshwater sediments (MacDonald *et al.* 2000a). The published SQGs for 28 chemical substances were assembled and classified into two categories in accordance with their original narrative intent. These published SQGs were then used to develop two consensus-based SQGs for each contaminant, including a threshold effect concentration (TEC; below which adverse effects are not expected to occur) and a probable effect concentration (PEC; above which adverse effects are expected to frequently occur; MacDonald *et al.* 2000a). A preliminary evaluation of the predictive ability of these consensus-based SQGs for freshwater sediment was conducted using a database of matching sediment chemistry and toxicity which included information on 347 samples obtained from 15 studies. The results of these three previous investigations demonstrated that the consensus-based SQGs provide a unifying synthesis of the existing guidelines, reflect causal rather than correlative effects, and account for the effects of contaminant mixtures in sediment (Swartz 1999, MacDonald *et al.* 2000a,b).

A fourth paper in the series evaluated the ability of consensus-based SQGs to predict toxicity in freshwater ecosystems using an expanded database of matching toxicity and chemistry (USEPA 2000a, Ingersoll *et al.* 2000). Results of the analyses presented in USEPA (2000a) are summarized in the current paper. The primary objectives of the analyses presented in USEPA (2000a) were to: (1) evaluate the ability of consensus-based SQGs to predict toxicity in a freshwater database for field-collected sediments in the Great Lakes basin; (2) evaluate the ability of SQGs to predict sediment toxicity on a regional geographic basis elsewhere in North America; and, (3) compare approaches for evaluating the combined effects of chemical mixtures on the toxicity of field-collected sediments.

METHODS

Individual SQGs for freshwater ecosystems have previously been developed using a variety of approaches (MacDonald *et al.* 2000a). Each of these approaches has certain advantages and limitations which influence their application in the sediment quality assessment process (Ingersoll *et al.* 1997). In an effort to focus on the agreement among these various published SQGs, consensus-based TECs and PECs were developed by MacDonald *et al.* (2000a) for 28 chemicals of concern in freshwater sediments (metals, PAHs, PCBs, and pesticides). The TECs were calculated by determining the geometric mean of the previously published SQGs that were included in this category. Likewise, consensus-based PECs were calculated by determining the geometric mean of the PEC-type values (MacDonald *et al.* 2000a). The geometric mean, rather than the arithmetic mean or

median, was calculated because it provides an estimate of central tendency that is not unduly affected by extreme values and because the distributions of the SQGs were not known (MacDonald *et al.* 2000a). Consensus-based TECs or PECs were calculated only if three of more published SQGs were available for a chemical substance or group of substances. The evaluations of sediment toxicity in the present study were based on the use of PECs because TECs were developed to provide an estimate of chemical concentrations that would not be expected to be toxic while the PECs were developed estimate chemical concentrations that are expected to be toxic. PECs were considered reliable if more than 75% of the samples with concentrations exceeding the PEC were toxic (MacDonald *et al.* 2000a). These dry-weight normalized PECs included: arsenic (33.0 µg/g), cadmium (4.98 µg/g), chromium (111 µg/g), copper (149 µg/g), lead (128 µg/g), nickel (48.6 µg/g), zinc (459 µg/g), total PAHs (22.8 µg/g), total PCBs (0.676 µg/g), and sum DDE (0.0313 µg/g). Evaluations of SQGs in the present study were based on dry-weight concentrations because previous studies have demonstrated that normalization of SQGs for PAHs or PCBs to total organic carbon or normalization of metals to acid-volatile sulfides did not improve the predictions of toxicity in field-collected sediments (Ingersoll *et al.* 1996; Long *et al.* 1998b; USEPA 2000a).

In USEPA (2000a), a database was developed from 92 published reports which included matching sediment chemistry and toxicity data for a total of 1657 samples. The database was comprised primarily of 10- to 14-day or 28- to 42-day toxicity tests with the amphipod *Hyalella azteca* (designated as the HA10 or HA28 tests) and 10- to 14-day toxicity tests with the midges *Chironomus tentans* or *C. riparius* (designated as the CS10 test). Endpoints reported in these tests were primarily survival or growth. Toxicity of samples was determined as a significant reduction in survival or growth relative to a control or reference sediment (as designated in the original study or determined using appropriate statistical procedures).

Because field-collected sediments typically contain complex mixtures of contaminants, the accuracy of a sediment assessment is likely to increase when SQGs are used in combination to classify toxicity of sediments. For this reason, the evaluation of the predictive ability of PECs was conducted to determine the incidence of effects above and below various mean PEC quotients (mean quotients of 0.1, 0.5, 1.0, and 5.0). A PEC quotient was calculated for each chemical in each sample in the database by dividing the concentration of a chemical by the PEC for that chemical. A mean quotient may be calculated for each sample by summing the individual quotient for each chemical and then dividing this sum by the number of PECs evaluated.

RESULTS AND DISCUSSION

Evaluation of approaches used to calculate PEC quotients: When mean quotients were calculated using an approach of equally weighting up to 10 reliable PECs (PECs for metals, total PAHs, total PCBs, and sum DDE), there was an increase in the incidence of toxicity with an increase in the mean quotient in all three tests (USEPA 2000a). For example in the HA10 test, the incidence of toxicity was 20% at mean quotients of <0.1 and increased to 67% at mean quotients of >5.0. Similarly, for the CS10 test there was a 20% incidence of toxicity at mean quotients of <0.1, increasing to a 64% incidence of toxicity at mean quotients of >5.0. In contrast, the incidence of toxicity in the HA28 test was only 8% at mean quotients of <0.1 and increased to 91% at mean quotients of >1.0. In all three tests, there was a consistent increase in the toxicity at mean quotients of >0.5. However, the overall incidence of toxicity was greater in the HA28 test compared to the short-term tests.

The incidence of toxicity at mean quotients of <0.1 was somewhat higher in the HA10 and CS10 tests (20%) compared to the HA28 test (8%). This toxicity at low mean quotients does not appear to be related to total organic carbon in sediment. There was insufficient information in the database to evaluate effects of grain size on toxicity. Unmeasured contaminants in these field-collected sediments or contaminants for which we do not have reliable PECs (i.e., pesticides, herbicides; MacDonald *et al.* 2000a) may have contributed to this toxicity at low mean quotients. Alternatively, the data for HA10 and CS10 tests were obtained from numerous laboratories, which may have contributed to variability in the data reported in these studies. In contrast, a limited number of laboratories conducted most of the HA28 tests.

We were also interested in determining the predictive ability of PEC quotients for major classes of compounds. Therefore, we evaluated the incidence of toxicity based on a mean quotient for metals, a quotient for total PAHs, or a quotient for total PCBs. Different patterns of toxicity associated with the various procedures for calculating quotients were observed. For example in the HA28 test, a relatively abrupt increase in toxicity was associated with elevated PCBs alone or with elevated PAHs alone, compared to the gradual increase in toxicity observed with quotients calculated using a combination of metals, PAHs, and PCBs. These analyses indicate that the different patterns in toxicity may be the result of unique chemical signals associated with individual contaminants. While mean quotients can be used to classify samples as toxic or non-toxic, individual quotients might be useful for helping to identify substances that may be causing or substantially contributing to the observed toxicity.

Evaluation of exposure duration and endpoints measured in toxicity tests: We evaluated the relationship between mean PEC quotients and the incidence of toxicity as a function of the duration of the exposure or of the endpoints measured in the toxicity tests (Figure 1). Samples within each test were ranked in ascending order by mean quotient. The incidence of toxicity and geometric mean of the mean quotients within groups of 20 samples for the HA10 and CS10 tests or within groups of 10 samples for the HA28 test was then plotted (Figure 1, USEPA 2000a).

In Figure 1, samples were classified as toxic based on an adverse effect on survival alone or based on an adverse effect on survival or growth in the three tests. The relationship between the incidence of toxicity and the geometric mean of the mean quotients was best described by a three parameter logistic model (Figure 1; see USEPA 2000a for the equations and coefficients). The best fit of the data was observed in the HA28 test (r^2 = 0.79 to 0.93) relative to the HA10 test (r^2 = 0.73 to 0.78) or CS10 test (r^2 = 0.56 to 0.76; Figure 1). The incidence of toxicity increased with increasing level of contamination in all three tests. This increase was particularly pronounced at mean quotients of >0.5

in all three tests. In the HA10 test, the relationship between toxicity and mean quotient was similar when either survival alone or survival and growth together were used to classify a sample as toxic. However, in the HA28 and CS10 tests, the relationship between the incidence of toxicity and mean quotient was different when survival or growth were used to classify a sample as toxic compared to survival alone (Figure 1).

The incidence of toxicity in the HA28 and CS10 tests based on survival or growth was often double the incidence of toxicity based on survival alone at mean quotients of >0.3. A 50% incidence of toxicity in the HA28 test corresponds to a mean quotient of 0.63 when survival or growth were used to classify a sample as toxic (Figure 1). By comparison, a 50% incidence of toxicity is expected at a mean quotient of 3.2 when survival alone was used to classify a sample as toxic in the HA28 test. In the CS10 test, a 50% incidence of toxicity is expected at a mean quotient of 9.0 when survival alone was used to classify a sample as toxic or at a mean quotient of 3.5 when survival or growth were used to classify a sample as toxic. In contrast, similar mean quotients resulted in a 50% incidence of toxicity in the HA10 test when survival alone (mean quotient of 4.5) or when survival or growth (mean quotient of 3.4) were used to classify a sample as toxic.

There was a slightly elevated incidence of toxicity at the very lowest mean quotient in all three tests. Long *et al.* (1998a) also observed an elevated incidence of toxicity with marine amphipods at low mean quotients. Long *et al.* (1998a) suggested that these samples were sometimes fine-grained sediments with low concentrations of organic carbon and detectable concentrations of butyltins, chlorinated pesticides, alkyl-substituted PAHs, ammonia, or other substances not accounted for with the SQGs. In the present study, the incidence of toxicity at low mean quotients did not appear to be related to total organic carbon in sediment. There was insufficient information in the database to evaluate effects of grain size on toxicity. However, ASTM (2000) and USEPA (2000b) reported that amphipods and midges were relatively intolerant to effects of sediment grain size.

Results of these analyses indicate that both the duration of the exposure and the endpoint measured can influence whether a sample is found to be toxic or not. Comparisons of the sensitivity between these tests needs to be made with some caution. There were very few samples in the freshwater database where tests were conducted using splits of the same samples. Therefore, the differences observed in the responses of organisms may also be due to differences in the types of sediments evaluated in the individual databases for each test. Nevertheless, it appears that longer-term tests in which survival and growth are measured tend to be more sensitive than short-term tests, with acute to chronic ratios on the order of 6 for *H. azteca.* Similar differences in sensitivity of *H. azteca* have been observed in 10- and 42-day water-only exposures to cadmium or DDD (unpublished data).

Evaluation of the predictive ability of mean PEC quotients across various geographic areas in the database: In USEPA (2000a), we chose to make comparisons across geographic areas using mean quotients calculated by equally weighting the contribution of the three major classes of compounds (metals, or PAHs, or PCBs). This approach assumes that these three diverse groups of chemicals exert some form of joint toxic action. Use of this approach also maximized the number of samples that were used to make comparisons across geographic areas. Generally, there was an increase in the incidence of toxicity with increasing mean PEC quotients within most of the regions, basins, and areas for all three toxicity tests. For the HA10 and HA28 tests, the incidence of toxicity for samples from each of the Great Lakes and within the areas of each Great Lake was relatively consistent with the overall pattern of toxicity in the entire database. However, the relationship between the incidence in toxicity and mean quotients in the CS10 test was more variable among geographic areas compared to either the HA10 or HA28 test. The results of these analyses indicate

that the consensus-based PECs can be used to reliably predict toxicity of sediments on both a regional and national basis.

This paper summarizes the results of the first analyses completed on the entire freshwater sediment database. Some of the additional analyses planned for the database include: (1) comparing approaches for designating samples as toxic; (2) evaluating logistic-regression models; (3) identifying a list of optimal analytes for broad scale application and testing the relative efficacy of the mean versus the sum of PEC quotients; (4) evaluating the influence of grain size and ammonia on the incidence of toxicity; and, (5) developing a guidance manual for conducting an integrated assessment of sediment contamination.

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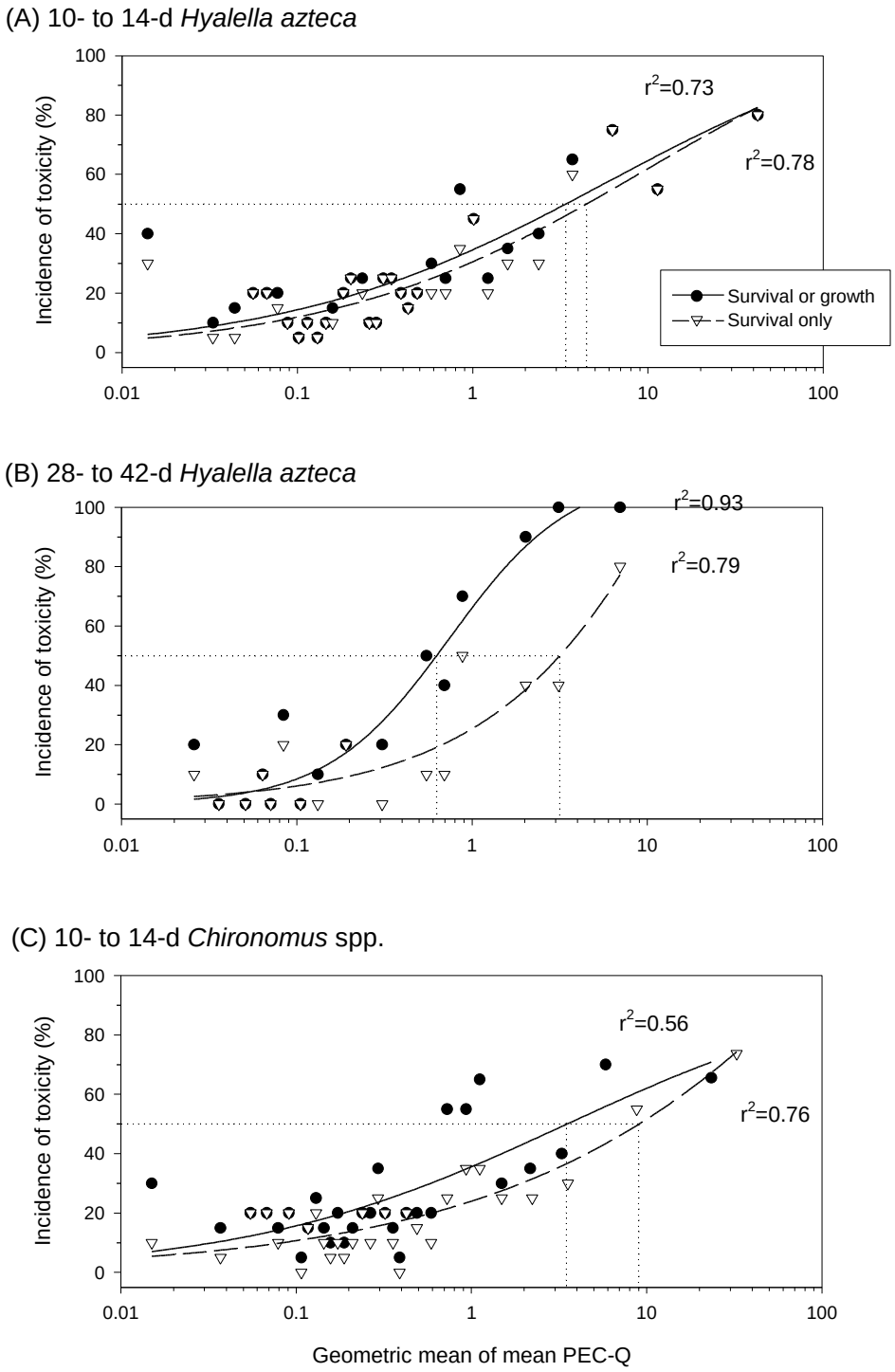

Figure 1. Relationship between the geometric mean of the mean PEC quotients and the incidence of toxicity in the three tests, based on survival or growth, or based on survival alone. The dotted line represents a 50% incidence of toxicity. See USEPA (2000a) for additional detail.

COMPARISON OF SEDIMENT AND NUTRIENT TRANSPORT AND BEST MANAGEMENT PRACTICES WITHIN THE YAZOO RIVER BASIN, MISSISSIPPI

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Abstract: The Yazoo River Basin in Mississippi is one of the most productive agricultural areas within the U.S. The basin is characterized with two distinct physiographic regions, the uplands, and the Mississippi River Alluvial Plain, locally referred to as the Delta. In recent years, much attention has been placed on understanding the effects of nonpoint source pollution within the entire Yazoo Basin. Such understanding requires information on the source of pollutants, quantities in transport, mode of transport, and the transient nature of pollution events. Data from three separate studies in the two physiographic regions of the Yazoo River Basin were compared to evaluate the effects of nonpoint source pollution. The studies were conducted at different periods of time from 1974 to the present. One of the studies was located in agricultural areas in the Mississippi Delta; another study was located in the agricultural areas in the Uplands; and the third study was located in forested areas in the Uplands (representing unimpaired conditions). In addition, best management practices (BMP's) implemented to reduce nonpoint source pollution were also observed as part of the studies. From all three general study areas, single large transient (temporary contaminating occurrences) storm events transported a significant portion of the annual sediment load. Similarly, for all the study areas, a few storms transported a significant portion of the annual nutrient loads generally following fertilization of the two agricultural areas. With regard to the reduction of nonpoint source pollution, the studies indicated that for the agricultural areas in the uplands and Mississippi Delta a conservation tillage BMP approach reduced sediment loads as much as 82 to 98%. For both the uplands and Mississippi Delta agricultural watersheds, a conservation tillage BMP approach also resulted in a substantial decrease in the nitrogen and phosphorus loads via sediment transport.

INTRODUCTION

Diffuse, or nonpoint source, pollution in the United States has only received national attention within the last decade. Prior to this time, most of the legislation concerning surface water protection has primarily dealt with point source pollution such as industrial and municipal discharges. Initially, states were the first to accept responsibility for surface water protection with little federal involvement. However, the U.S. Congress in the early 1960's was not satisfied with the states' progress in pollution control and passed the Water Quality Act of 1965 requiring each state to adopt water quality standards better than or equal to those of the federal government. Furthermore, the act fundamentally changed the role of the federal government from one acting as advisor to the states to one of taking the lead in water pollution control. In the early 1970's Congress felt that the states had failed to enact comprehensive water quality control legislation (Beck, 1991). This fact, along with the impact of the environmental movement of the time, led to the passage of the Federal Water Quality Control Act Amendments of 1972, commonly known as PL 92-500 and/or the 1972 Clean Water Act (CWA). With only relatively small revisions, this piece of legislation became the framework for water pollution control policy during the past 20 years [1].

When the CWA was passed in 1972, human health and the use of surface waters were the driving issues of the time. It is clear that in the 1990's ecosystem health and integrated management of water quality on a watershed basis are the issues of concern [2]. The CWA establishes a national goal of "fishable and swimmable" waters. Still, many waters in the U.S. do not meet this goal with diffuse pollution now being blamed for a large portion of the problem. Thus, CWA section 303(d) addresses these remaining waters by requiring the states to develop and implement Total Maximum Daily Loads (TMDL's) standards to achieve water bodies that are fully functional ecologically [3].

Diffuse pollution in the southern United States is of major concern because of the region's abundant water resources. Six major activities in the southern United States contribute to diffuse pollution; these include agriculture, silviculture, mining, construction, urban, and atmospheric deposition. In the southern United States agriculture is by far the most prevalent source of diffuse pollution, and sediment remains the single most important water quality problem.

The Yazoo River Basin has two distinct physiographic regions within its drainage area. The uplands area is characterized by streams that have relatively steep slopes and deep-incised channels. The region is largely forested with some agricultural areas. In contrast, the region locally referred to as the Delta is one of the most productive agricultural regions in the nation. Drainage in the Delta can be characterized by low slopes that cause frequent backwater flooding conditions. The waters are extremely sediment-laden, and potentially there is concern that excessive nutrients in the agricultural runoff may cause eutrophic conditions and ultimately extreme water quality problems such as fish kills. These are only a few of the general differences that exist within the regions of the Yazoo River Basin.

The purpose of this paper is to examine several decades of research within the Yazoo River Basin and to make comparisons among the physiographic regions. Specifically, this paper will address: (1) similarities and differences in diffuse pollution from these two regions; (2) the effectiveness of selected best management practices (BMP's) in the remediation of diffuse pollution.

STUDY AREAS, STUDY PERIODS, AND RESEARCH METHODS

Upland – Agricultural – Nelson watersheds: This water quality research was conducted on the Nelson Research Farm in Tate County in the northern part of Mississippi from the water years (WYs) 1990 to 1993. The study compared two watersheds: one with no-till soybeans (2.13 ha) and one with conventional-till soybeans (2.10 ha). The soils were highly erodible loess within the Loring Series. Each watershed was instrumented for discharge-weighted composite sample collection (Cullum et al., 1992). Following 0.45 Φm filtration, both sediment and aqueous phases of surface runoff were analyzed for plant nutrients (Schreiber and Cullum, 1998).

Upland – Forested – Coffeeville watersheds: The forested water quality research was conducted in Yalobusha County in the north - central part of Mississippi. The five study watersheds, 1.49 to 2.81 ha, were established to pine in 1939 on severely eroded agricultural land. Soils were developed from Coastal Plain sediments overlain by a shallow loess mantle and include the Providence, Memphis, Loring, Smithdale, and Collins Series. On each watershed a Coshocton wheel set below a 0.91 m H-flume diverted 0.5% stormflow into containers. Thus, one composite sample was collected during each storm event. Both sediment and aqueous phases of runoff were analyzed for plant nutrients. Sediment (Duffy et al., 1986), P (Duffy et al., 1978), and N (Schreiber et al., 1980) were measured for the WYs'74-'78, '74, and '75, respectively.

Delta – Agricultural – MSEA watersheds: The U.S. Geological Survey (USGS) began operation of a water quality monitoring network in 1995 as part of the Mississippi Delta Management Systems Evaluation Areas (MDMSEA) project. The USGS currently operates nine sites in three watersheds. The watersheds have various degrees of BMP's installed in each: the first watershed has no MDMSEA-sponsored BMP's, the second watershed has primarily structural-type BMP's, and the third watershed has a combination of structural and cultural BMP's (Rebich, 1997).

The scope of the USGS research presented in this paper focuses on two of the study sites. The first sampling site is located in the watershed that has no MDMSEA-sponsored BMP's. It is located at the edge of a 5.7 - ha field that has been in conventional-tillage (2-yr. cotton and 1 yr. soybeans) throughout the life of the project. Soils in this field are generally heavy clay. The second sampling site is located in the watershed that has a combination of structural and cultural BMP's. It drains both a conservation-till cotton field and conservation-till soybean field. In addition, winter wheat has been planted each year during the fall and winter after harvest to provide cover for these fields as a means of protecting the soil from erosion losses. The total drainage area is about 9.9 ha, and the soil types are a sandy and sandy-loam combination.

Each site is instrumented with flumes to measure total flow during rainfall/runoff events and automated samplers to collect discharge-weighted water-quality and sediment samples. Data presented in this paper were for the 1997-99 WY's

RESULTS AND DISCUSSION

Upland – Agricultural – Nelson watersheds

Sediment in surface runoff: For the '90 to '93 WY, the mean discharge-weighted sediment concentrations in the no-till soybean runoff were 319, 67, 105, and 80 mg/L, respectively (mean sediment concentration for the entire study period was 129 mg/L, Table 1); sediment loads were 1,050, 455, 375, and 136 kg/ha, respectively (mean sediment load for the entire study period was 504 kg/ha). In contrast, the '91 to '93 WY mean discharge-weighted sediment concentrations in runoff from the conventional-till soybean watershed were 3,569, 5,241 and 1,270 mg/L, respectively (mean concentration for the entire study period was 3,623 mg/L); mean sediment loads were 31,928, 22,994, and 2,898 kg/ha, respectively (mean sediment load for the study period was 19,273 kg/ha). Thus, for the study period, no-till soybeans resulted in over a 97% reduction in sediment load.

Transient pollution events can occur on a plot, watershed, or large drainage scale (Schreiber et al., 1996). Although temporary in nature, such events can be relatively short to long term and can occur diurnally as well as annually. The cause of these events may be natural, man made, or a combination of both. Transient events may be responsible for the transport of large quantities of sediment, nutrients, and pesticides; some storm events produce nearly the annual load. For example, over a four year study of soil loss from these soybean watersheds, the storm with the highest sediment concentration resulted in a percentage of the total yearly sediment load that ranged from 17-72% to 37-60% in runoff from the no-till and conventional-till watersheds, respectively. Rainfall distribution frequency as related to spring tillage appeared to play the dominant role in producing the high sediment concentrations in runoff. At other times antecedent soil moisture was important (Schreiber and Cullum, 1998).

Soluble nutrients in surface runoff: The mean discharge-weighted soluble nutrient concentrations in surface runoff for the Nelson watersheds are presented in Table 1. The higher soluble nutrient concentrations in runoff from the no-till compared with the conventional-till watershed are most likely due to the leaching of accumulated soybean and weed residues, and the lack of sorption surfaces in runoff. The distribution of PO_4-P concentrations in runoff differed significantly between the tillage systems throughout the study period. The distribution of NH_4-N concentrations differed significantly during the '91 WY, and those of NO_3-N for the '92 WY (Schreiber and Cullum, 1998). The much higher suspended sediment concentration runoff from the conventional-till watershed would sorb soluble phosphorus, thereby reducing soluble PO_4-P concentrations in runoff.

Soluble nutrient loads for each tillage system are presented in Table 1. For the study period, only the distribution functions of soluble PO4-P loads differed significantly between the two tillage systems. Despite lower runoff from the no-till watershed for all WY's, the soluble PO_4-P loads for the study period were about five times that of the conventional-till watershed due to the higher concentrations of soluble PO_4-P (Schreiber and Cullum, 1998).

In general, soluble nutrient concentrations in runoff showed similar seasonal trends for both tillage systems. Lowest nutrient concentrations in runoff were observed during the winter and early spring months, a time period of minimal microbiological activity. A number of biological, chemical, and cultural factors, along with rainfall distribution act concurrently to produce transient pollution events in surface runoff. Runoff-producing events shortly after broadcast applications of fertilizer to no-till and conventional-till soybeans can result in soluble phosphorus transient pollution events. These dramatic increases in PO_4-P were most noticeable in runoff from the no-till watershed due to the lack of sediments to sorb soluble PO_4-P. As an example, the highest PO_4-P concentration in runoff from the no-till and conventional-till watersheds was 23 and 6 mg/L, respectively, on May 16, 1991, just 2 days after a broadcast application of 0-20-20 fertilizer. Furthermore, these storm events transported 18 and 3 % of the yearly total soluble PO_4-P loads from the watersheds, respectively. Similar observations were made for each WY of the study period. Phosphorus is considered to be a key limiting nutrient in eutrophication processes.

Sediment nutrients in surface runoff: Soluble loads of nutrients from the no-till watershed represented almost all of the N and P nutrient loads because sediment concentrations and loads were low (Table 1). Compared to the no-till watershed, sediment-associated nutrient loads from the conventional-till watershed are a substantial

Table 1. Mean physical and chemical parameters of runoff from selected Mississippi watersheds for various study periods, (mm/yr, millimeters per year; mg/L, parts per million; kg/ha/yr, kilograms per hectare per year; conv.-till, conventional tillage; cons.-till, conservation tillage)

	PHYSICAL			
			Sediment	
	Rainfall mm/yr	Runoff mm/yr	Concentration[4] mg/L	Load kg/ha/yr
Nelson (agricultural)[1]				
conv.-till	1431	532	3,623	19,273
no-till	1393	392	129	504
Coffeeville (forested)[2]	1510	170	112	183
Delta (agricultural)[3]				
conv.-till	1100	742	1492	11,073
cons.-till	1189	564	593	3,342

	SOLUBLE CHEMICAL					
	Concentration			Load		
	PO4-P mg/L	NH4-N mg/L	NO3-N mg/L	PO4-P kg/ha/yr	NH4-N kg/ha/yr	NO3-N kg/ha/yr
Nelson (agricultural)[1]						
conv.-till	0.09	0.16	0.50	0.48	0.86	2.61
no-till	0.57	0.26	0.63	2.06	0.98	2.35
Coffeeville (forested)[2]	0.01	0.20	0.01	0.03	0.42	0.03
Delta (agricultural)[3]						
conv.-till	0.08	0.15	1.67	0.59	1.12	12.42
cons.-till	0.17	0.12	0.55	0.98	0.70	3.29

	SEDIMENT CHEMICAL			
	Concentration		Load	
	N mg/kg	P mg/kg	N kg/ha/yr	P kg/ha/yr
Nelson (agricultural)[1]				
conv.-till	617	636	11.60	12.21
no-till	4,552	1,905	1.54	0.92
Coffeeville (forested)[2]	3,553	515	0.40	0.21
Delta (agricultural)[3]				
conv.-till	2,122	416	23.50	4.61
cons.-till	2,977	811	9.95	2.71

[1] Water years 1990-1993. (Data from Schreiber and Cullum, 1998).

[2] Segmented research inclusive of water years 1974-1978. (Data from Duffy et al., 1978; Duffy et al., 1986; Schreiber et al., 1980).

[3] Water years 1997-1999.

[4] All sediment and nutrient concentrations are discharge weighted.

portion of the soluble plus sediment nutrient load (Table 1). A portion of the P associated with the sediment can be bio-available to aquatic organisms (Sharpley, 1993). For both N and P, total nutrient (soluble plus sediment) loads were reduced by nearly a factor of five with the no-till system for soybeans.

In addition, another important pollutant to aquatic ecosystems, sediment, was reduced by as much as 97 percent. However, under no-till soybeans, soluble P loads increased four to six fold compared to the conventional-till soybeans. As mentioned earlier, this increase in soluble P transport is related directly to the lack of sediment present to sorb the soluble P. The soluble loads of N (NO_3-N + NH_4-N) were about the same for both tillage systems (Schreiber and Cullum, 1998).

Upland – Forested – Coffeeville Watersheds

Sediment in surface runoff: Sediment transport research from five pine forested watersheds in northern Mississippi indicates that channel morphology plays an important role in sediment producing events (Duffy et al., 1978; Schreiber and Duffy, 1982; Schreiber et al., 1980). Sediment concentrations differed among watersheds (WS), but in general were not correlated with stormflow volume. Mean sediment concentrations ranged from 49 mg/L for WS 1 to 228 mg/L for WS 3 for the study period (mean sediment concentration for the study period was 112 mg/L, Table 1). Watersheds 1 and 2 with low sediment concentrations have well covered broad channels without steep banks. In contrast, WS 3, 4, and 5 with higher sediment concentrations have incised channels with exposed banks which are prone to frost heaving and wetting drying cycles that deposit loose materials in channels. Occasional (transient) large storms flush these deposits from the channels, and also cause some headward cutting. Over the period 1976-78 the largest storm event accounted for 26 to 97 % of the total yearly sediment load. Sediment loads for all watersheds averaged 183 kg/ha/yr for the study period (Table 1).

Soluble nutrients in surface runoff: Ammonium N in the stormflow was the dominant form of soluble nitrogen (Table 1). Solution N concentrations, NH_4-N or NO_3-N, in individual stormflows did not correlate with stormflow, show seasonal trends, or differ among watersheds (Schreiber et al.,1980). However, loads of both NO_3-N and NH_4-N differed among watersheds because of differences in stormflow. Consequently, yearly soluble N loads were positively and linearly related to yearly stormflow.

The yearly mean PO_4-P concentration was 0.01 mg/L, 33% of the soluble total P concentration, and did not differ among watersheds. In November, when litter fall was near maximum, concentrations of all soluble P forms were greatest. This phenomenon was likely associated with mineralization and/or leaching of fresh litter. Lowest P concentrations occurred in February. Concentrations of all soluble P forms in stormflow were not significantly correlated with either stormflow or sediment concentration. Annual soluble PO_4-P loads differed among watersheds because of differences in stormflow since concentrations were the same.

Sediment nutrients in surface runoff: Sediment N concentrations were greatest for sediments from watersheds with predominantly loess soils (WS 1 and 2). Sediment N correlated positively with sediment organic matter, but not with stormflow or sediment concentration for individual storm events. Yearly mean sediment N concentration for each watershed was 5.4 to 10.0 times the mean N concentration of the watershed soil (0-15 cm). Soil and sediment N concentrations among the watersheds were correlated with their respective soil and sediment organic matter concentrations as linear functions. Nitrogen enrichment of the suspended sediment was attributed to the selective-erosion of fine sediments (clay and organic matter) and/or deposition of coarse sediment in transport. Since sediment N concentrations for individual storms did not correlate with either sediment concentrations or stormflow, sediment N loads for each watershed were a linear function of sediment loads. Sediment N loads were 50, 50, 41, 49, and 45% of the total N load (solution total N plus sediment N) for WS 1 through 5, respectively (Schreiber et al., 1980).

Sediment total P concentrations differed among watersheds, and were also greatest for sediments from watersheds with predominantly loessial soils where soil P was also greatest. Sediment P concentrations were 2.0 to 8.9 times those in the watershed soils. This was attributed to the selective erosion of fine sediments and/or deposition of coarse sediments in transport.

Sediment P loads, like sediment N loads, were a function of sediment concentration, sediment P concentration, and stormflow, but in a more complex fashion (Duffy et al., 1978). The mean yearly load of sediment P for the watersheds was 0.21 kg/ha - 72% inorganic and 28% organic (Table 1). Yearly sediment total P loads did not differ

among watersheds, though sediment P concentrations and sediment loads did, since watersheds with the highest sediment P concentration had the lowest sediment load. Sediment P loads were 67, 64, 73, 74, and 76% of the total P (soluble total P + sediment total P) loads for WS 1 through 5, respectively.

As transient large single storms were responsible for the majority of the annual sediment load from five forested watersheds in northern Mississippi, we would also expect them also to account for a large portion of sediment associated nutrient transport in stormflow. For example, 10 % of the storms accounted for 67 % of the sediment N and 72 % of the soluble N load; and 63 % of the sediment P and 60 % of the solution total P loads. Other research indicates large single storm events transport a significant portion of the annual sediment load from forested watersheds (Martin and Hornbeck, 1994).

Delta – Agricultural – MSEA Watersheds

Sediment in surface runoff: The mean annual suspended-sediment concentration for the conventional-till site was 1,492 mg/L for the study period (Table 1). The mean annual suspended-sediment load was 11,073 kg/ha for the study period. In looking at the individual storm loads for each water year, the maximum sediment loads were 1,849, 3,402, and 2,883 kg/ha, respectively, representing 13, 40, and 28% of their annual loads.

The mean annual suspended-sediment concentration for the conservation-till site was 593 mg/L for the study period. The mean annual suspended-sediment load was 3,342 kg/ha for the study period. For the individual storm loads for each water year, the maximum sediment loads were 795, 731, and 828 kg/ha, respectively, representing 21, 21, and 31% of their annual loads.

In comparison of the two sites, the mean suspended-sediment load for the conservation-till site was about 70% lower than the mean load for the conventional-till site for the study period. For the individual storm loads, the maximum sediment loads for the conservation-till site were 57, 79, and 70% lower than the maximum sediment loads for the conventional-till site for the 1997-99 WY's, respectively.

Soluble nutrients in surface runoff: The mean annual soluble nitrate, ammonia, and ortho-phosphorus loads for the conventional-till site were 12.42, 1.12, and 0.59 kg/ha, respectively, for the study period (Table 1). The maximum nitrate loads for individual storms were 5.2, 4.1, and 2.5 kg/ha for the 1997-99 WY's, respectively, 37, 59, and 16% of their annual loads. A cumulative frequency of nitrate concentrations and loads show that very few storms generated significant concentrations or loads. For example, 80% of the concentrations were less than about 2.50 mg/L and 80% of the loads were less than 3 kg per storm event. Most of the high concentrations and loads occurred during the first few events following N fertilizer applications in the spring.

The mean annual soluble nitrate, ammonia, and ortho-phosphorus loads for the conservation-till site were 3.29, 0.70, and 0.98 kg/ha, respectively, for the study period. In looking at the individual storm loads, the maximum nitrate loads were 0.61, 1.41, and 0.79 kg/ha representing 23, 37, and 19% of their annual loads. Again, these maximum loads were observed in the first sampled runoff event after application of nitrogen fertilizers in the spring.

The mean annual nitrate load at the conservation-till site was about 73% lower than the mean nitrate load at the conventional-till site for the study period. Nitrogen uptake by the winter wheat cover crop, as well as N immobilization following a spring herbicide burn down, are the suspected reasons for the lower nitrate loads from the conservation-till site. Lesser reductions occurred in the mean ammonia loads. No significant reductions occurred for the ortho-phosphorus loads. In fact ortho-phosphorus concentrations and loads were higher from the conservation tillage sites, and in part, can be attributed to the leaching of cover crop residues.

Sediment nutrients in surface runoff*:* The USGS research in the MDMSEA project did not include analyses of sediment for attached water quality constituents such as nutrients. However, unfiltered samples were analyzed for total (Kjeldahl) nitrogen and total phosphorus. The amount of nutrients that are attached to the sediment can then be estimated by subtracting the soluble nutrients from the total nutrients. The mean annual sediment-nitrogen load from the conventional-till site was estimated to be 23.50 kg/ha for the study period (Table 1). By subtracting the annual ortho-phosphorus loads from the total phosphorus loads, the mean annual sediment-phosphorus load was estimated to be 4.61 kg/ha for the study period for the conventional-till site. The mean sediment-nitrogen load from the conservation-till site was estimated to be 9.95 kg/ha for the study period. By subtracting the ortho-phosphorus loads from the total

phosphorus loads, the mean annual sediment- phosphorus load was estimated to be 2.71 kg/ha for the study period from the conservation-till site

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In comparison of the two sites, the mean sediment-nitrogen load at the conservation till site was about 58% lower than the sediment-nitrogen load at the conventional-till site for the study period. The mean sediment-phosphorus load at the conservation till site was about 41% lower than the sediment-phosphorus load at the conventional-till site for the study period.

Total Maximum Daily Load (TMDL): Total maximum daily load (TMDL) is a new term to many, but the concept was presented in the original 1972 Clean Water legislation and in many more recent water-oriented documents in the United States. TMDL issues have greatly accelerated in the last four years because of multiple lawsuits regarding possible lack of enforcement at the state level. Although many of the issues involved are complex, the concept of TMDL is simple. The allowable load of a contaminant in a water body cannot exceed the amount (daily or seasonal) of the contaminant that impairs the ecological integrity of the water body. Thus, the TMDL is the sum of watershed point source loading plus the sum of non-point source loads plus a margin of safety. If the biological community is regarded as poor when compared to a reference community or monitored water quality criteria are exceeded, the stream reach, reservoir, or lake is declared impaired. If measurements show the biological community to be acceptable but borderline, the water body may be regarded as partially impaired or high risk. Nonpoint-source contamination, often associated with agriculture or silviculture, is generally analogous with runoff events. Major causes of impairment in most states include sediment, nutrients, pesticides and coliforms. Meeting TMDL goals in rural areas will necessarily include widespread use of existing and innovative, new best management practices. Better control of sediment and runoff from transient events by the practices discussed in this paper should bring many watersheds into compliance.

CONCLUSIONS

Data from three separate agricultural studies in the two physiographic regions of the Yazoo River Basin were presented in this paper. Some generalized trends were seen throughout each of the studies in spite of differences between time periods, time lengths, and different crops and soils. For example, for both the agricultural areas in the uplands and Mississippi Delta, a conservation tillage/BMP approach can reduce sediment loads as much as 70 to 97%. From all three general study areas, single large transient storm events transported a significant portion of the annual sediment load. Similarly, for the three study areas, a few storms transported a significant portion of the annual soluble chemical loads generally following fertilization of the two agricultural areas. Winter cover crops in the Mississippi Delta appear to play a complex role in the reduction of NO_3-N loads in runoff. For both the upland and Mississippi Delta agricultural watersheds, a conservation tillage/BMP approach results in a substantial decrease in the loads of N and P via sediment transport; and in the case of sediment P, loads approach those from forested areas that have received no fertilization. In addition to winter cover and conservation tillage, a multitude of other techniques are available to combat the diffuse pollution problems currently recognized as causing pervasive damage to the Yazoo River Basin environment as well as other watersheds throughout the southern United States. Such practices include stiff grass hedges, riparian zones, constructed wetlands, and more recently, alley cropping. Watershed scale reduction of sediment and nutrients plays an important role in broader issues. Hypoxia, a global phenomenon, accelerated by excessive nutrients, is of concern in coastal waters off every continent. Restoration of wetlands and riparian zones will help lessen hypoxia, but reduction of nutrients in agricultural runoff at the field and watershed scale will be necessary also. While it is certainly necessary to acknowledge the necessity of production agriculture to meet ever-increasing human consumption needs, it is also necessary to balance those production needs with ecological integrity to protect proper environmental functioning.

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DEVELOPMENT OF SAMPLING DESIGN METHODOLOGY TO REDUCE SUSPENDED SEDIMENT DATA COLLECTION ON THE MISSOURI RIVER

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Abstract: The U.S. Army Corps of Engineers, Omaha District, in cooperation with the U.S. Geological Survey, collects suspended sediment samples twice weekly at three sites along the Missouri River: Sioux City, Iowa; Omaha, Nebraska; and Nebraska City, Nebraska. An assessment of the suspended sediment data collection program was undertaken to determine if it was possible to reduce the sampling frequency, or even to halt sampling at one or more of these locations for a time, without compromising the precision and accuracy of annual sediment yield calculations. The study evaluated the adequacy of the existing data and the effectiveness of alternative sampling strategies. Each sampling site was analyzed independently. The recommended sampling strategy at each station was a seasonal, flow-stratified suspended sediment sampling design having the following properties: 1) suspended sediment samples that span the entire streamflow range for low, moderate and high water discharges; 2) more suspended sediment samples concentrated at streamflow intervals having the largest fraction of the total suspended sediment discharge; 3) the number of annual suspended sediment samples necessary to observe shifts in the suspended sediment rating curves that can occur in response to changes in the watershed or river; and 4) an adequate number of samples that will maintain or improve the precision and accuracy of the suspended sediment rating curves. The proposed sampling design required fewer annual sampling events than the current twice-weekly sampling scheme.

INTRODUCTION

The U.S. Geological Survey, under a cooperative stream gaging program with the U.S. Army Corps of Engineers, Omaha District, has collected suspended sediment samples at three sites along the Missouri River since October 1976 (Sioux City, Iowa; Omaha, Nebraska; and Nebraska City, Nebraska. Between October 1976 and September 1991, point-integrated suspended sediment samples were collected at six-week intervals during the open water season. Since July 1992, depth-integrated suspended sediment samples were collected twice weekly except when limited by weather conditions and equipment malfunctions or repairs.

The objective of this study is to develop a sampling strategy that would reduce the cost of suspended sediment data collection at each of the three sampling sites by either decreasing the number of samples collected at a site or discontinuing sampling at a site all together. Discontinuation of suspended sediment sampling is not recommended because a sufficient number of suspended sediment samples should be collected annually to measure the results of changes in the river or the watershed. Recent research has been directed at developing methods for reducing the number of samples required to obtain acceptable suspended sediment discharge estimates. In particular, rating curve estimators and statistically based stratified random sampling (Thomas, 1985; Thomas and Lewis, 1993; Thomas and Lewis, 1995) are the most popular methods (Cohn, 1995). Since stratified random sampling requires new sampling equipment (such as a programmable data logger and an automatic pumping sampler) at a significant additional cost, this methodology is not considered in this analysis. Therefore, rating curve estimator methods are used.

This paper presents theory used to develop the seasonal suspended sediment rating curves, with transformation bias correction. The sampling design methodology is then outlined and applied to the suspended sediment data at each sampling site. The frequency of sampling events from the current and proposed sampling strategies are compared and discussed.

SUSPENDED SEDIMENT RATING CURVES

Suspended sediment rating curve estimators are best applied to rivers flowing on an alluvial bed with a high percentage of the sediment discharge in the sand-size range of 0.062 to 2 mm (American Society of Civil Engineers, 1975). Since the suspended sediment in the Missouri River is generally within this sand-size range, the rating curve estimator approach is appropriate for this analysis.

A suspended sediment rating curve is an empirical relation between the suspended sediment discharge and the streamflow. This relationship is usually defined as a power function. The rating curve parameters can be estimated using a log-linear regression model and applying the method of least squares. The log-linear regression model is given by:

$$Y = \beta_0 + \beta_1 X + \varepsilon \tag{1}$$

where Q_w is the streamflow; Q_s is the suspended sediment discharge; X is $\ln(Q_w)$; Y is $\ln(Q_s)$; β_0 and β_1 are the regression parameters; and ε is the residual random error.

It is assumed that the regression residuals, ε, are independent and identically distributed normal random variables with a mean of zero and variance denoted by σ_ε^2, which can be estimated from the regression data set by:

$$\sigma_\varepsilon^2 \cong s^2 = \frac{1}{M-2}\sum_{i=1}^{M}\left(Y(i) - \hat{Y}(i)\right)^2 \tag{2}$$

where M is the number of observations in the data set. The estimate of Y, denoted by $\hat{Y}$, for a given streamflow is computed using the following equation:

$$\hat{Y} = \hat{\beta}_0 + \hat{\beta}_1 \ln(Q_w) \tag{3}$$

where $\hat{\beta}_0$ and $\hat{\beta}_1$ are the least squares estimates of β_0 and β_1 (for details, see Ross, 1987). Then, for any given streamflow Q_w, an estimate of the suspended sediment discharge, denoted by $\hat{Q}_{s,RC}$, can be obtained from the following equation:

$$\hat{Q}_{s,RC} = \exp\left(\hat{Y}\right) = \exp\left(\hat{\beta}_0\right)(Q_w)^{\hat{\beta}_1} \tag{4}$$

Under the assumption of normally distributed regression residuals, the rating curve estimator in Equation (4) has a bias and, in general, systematically underestimates the discharges (Ferguson, 1986; Cohn et al., 1989). The bias arises when $\hat{Y}$, the natural log of $\hat{Q}_{s,RC}$, is re-transformed into the original units using Equation (4). Therefore, a bias-correction factor must be applied to Equation (4) to obtain more accurate suspended sediment discharge measurements.

The Minimum Variance Unbiased Estimator (MVUE), developed by Cohn et al. (1989), is used to correct for the re-transformation bias. The MVUE bias-correction method is used for the following reasons: 1) the MVUE bias correction factor depends on the streamflow; 2) the MVUE method has zero bias in the long-term suspended sediment discharge estimates (Gilroy et al., 1990); and 3) the MVUE estimator has the smallest root mean square error associated with the long-term suspended sediment discharge estimates (Gilroy et al., 1990). The MVUE rating curve estimator, denoted by $\hat{Q}_{s,MVUE}$, for an individual streamflow is given by:

$$\hat{Q}_{s,MVUE} = \hat{Q}_{s,RC}\, g_m\left(\frac{m+1}{2m}(1-V)s^2\right) \tag{5}$$

where $\hat{Q}_{s,RC}$ is calculated from Equation (5), $m = M - 2$, g_m is the MVUE bias correction factor (Finney, 1941; Bradu and Mundlak, 1970) given in Equation (6), and s^2 is computed using Equation (2). The variance V, which is given by Equation (7), is a function of the streamflow value used to estimate the suspended sediment discharge. The variable $X(i) = \ln(Q_w(i))$ for the streamflows in the regression data set and are related as follows:

$$g_m(t) = \sum_{k=0}^{\infty} \frac{m^k (m+2k)}{m(m+2)\cdots(m+2k)} \left(\frac{m}{m+1}\right)^k \left(\frac{t^k}{k!}\right) \tag{6}$$

$$V = \frac{1}{M} + \frac{\left(\overline{X} - \ln(Q_w)\right)^2}{\sum_{i=1}^{M} (X(i))^2 - M\,\overline{X}^2} \tag{7}$$

$$\overline{X} = \frac{1}{M} \sum_{i=1}^{M} X(i) \tag{8}$$

Long-term estimates of suspended sediment discharges (such annual or seasonal values) are usually approximated by sums of daily suspended sediment discharge estimates computed from daily mean streamflows. Gilroy et al. (1990) developed exact expressions for the bias and variance of such sums for the MVUE rating curve estimator given in Equation (5). The equations developed by Gilroy et al. (1990) were used to compute the mean square errors of the long-term discharge estimates.

The relationship between streamflow and suspended sediment discharge may change with temperature, type of runoff (rainfall versus snowmelt), properties of the sediment, and characteristics of the drainage basin (topology, soils, land use, vegetation cover, etc.). To account for this variability, additional variables can be added to the regression equation (Cohn et al., 1992) or the data can be stratified (by season, discharge, temperature, etc.) and multiple rating curves can be developed (Colby, 1956). In this study, rating curves are developed for the fall/winter (October 1 through March 31) and spring/summer (April 1 through September 30) seasons.

SAMPLING DESIGN METHODOLOGY

The objective was to design a sampling strategy that minimized the number of suspended sediment samples collected at each sampling site within each season and discharge strata while 1) maintaining the precision and accuracy of the suspended sediment rating curve and 2) collecting enough samples to observe shifts in the rating curve due to changes in the river or the watershed. The existing suspended sediment data sets were systematically thinned to simulate different sampling strategies.

The suspended sediment data from each of the sampling sites were analyzed independently. Because of the logarithmic nature of the relationship between the suspended sediment discharge and streamflow, the measured flows were divided into equal streamflow intervals on the common log scale. Then the different sampling strategies were formulated at each sampling site as follows:

1. Each streamflow interval was analyzed independently. The number of samples in a particular streamflow interval was systematically thinned (i.e., 2, 4, 6 or more samples were taken out per year) and the regression parameters and associated mean square errors were computed. The results from the different thinned data sets were compared to those from the original data set. The reduced sample sizes within the given streamflow interval that had similar regression parameters, small changes in the estimated suspended sediment discharges, and acceptable mean square errors were identified.

2. Step 1 was completed for each streamflow interval.

3. Then the original data set was thinned simultaneously across all of the streamflow intervals based upon the results from Steps 1 and 2. The resulting regression parameters, total suspended sediment discharge estimates and their associated root mean square errors were compared to those from the original data set. The thinning scenario that had the smallest number of samples and that resulted in similar regression parameters, suspended sediment discharge estimates and acceptable mean square errors compared to the original data was recommended.

Only the depth-integrated samples, which were collected twice weekly since July 1992, were thinned from the data sets. There was not a sufficient number of suspended sediment samples collected annually during Water Years 1977 through 1991 to perform a meaningful thinning analysis. In each streamflow interval, the samples were thinned within each season to a specified number using the "last in, first out" scheme (LIFO) (i.e., for the fall/winter season, samples collected in February were removed before samples collected in January, in a time-backward sequence). The LIFO methodology was utilized because it simulated the process by which the sampling design would be implemented in the field. For example, during a particular water year, samples would be collected in Streamflow Interval 3 until the number of samples in that interval equaled the number specified in the sampling design (say in January). Then, for that year, further samples would not be collected in Interval 3 (say in February).

This approach was used to develop sampling designs for the fall/winter and spring/summer seasons at the Sioux City, Omaha and Nebraska City sampling sites on the Missouri River. Each sampling site and season were analyzed independently. In order to demonstrate the sample reduction associated with the proposed sampling designs, the current twice weekly and the proposed sampling schemes were implemented for future water years. It was assumed that the flow patterns for this period were the same as the historical patterns from Water Years 1977 through 1999. Then the numbers of samples collected under the two sampling schemes were compared. The results of these analyses are presented in the following sections.

RATING CURVES AND SAMPLING DESIGNS

At each of the sampling sites, the point-integrated samples were generally consistent with the depth-integrated samples. There were several suspended sediment measurements at each of the sites, called outliers, which deviated significantly from the other measurements for approximately the same streamflow intensity. Since it was assumed that all of the suspended sediment measurements were valid, these outliers were included in the regression analyses for rating curve development.

Sampling Site at Sioux City, Iowa: Rating curves and sampling designs were developed for the fall/winter and spring summer seasons at the sampling site in Sioux City, Iowa. The sampling site is located on the Missouri River at the USGS Gaging Station 06486000, "Missouri River at Sioux City, IA", at River Mile 732.2, which is approximately two miles downstream of the confluence with the Big Sioux River.

During the fall/winter season, 212 suspended sediment samples were collected from the Missouri River at Sioux City between Water Years 1980 and 1999. There were 10 point-integrated samples collected between Water Years 1980 and 1991 and 202 depth-integrated samples collected during Water Years 1993 through 1999. Suspended sediment samples were not collected during the fall/winter season of Water Year 1992. All of the point-integrated samples were collected during the month of October. The streamflow intervals and recommended samples sizes are listed in Table 1. The recommended sample size for Interval 1 is zero because the streamflows in this interval do not transport a significant amount of sediment. Since the streamflows in Interval 6b are high, infrequent and can transport a significant amount of sediment, suspended sediment samples should be collected for all flows in this interval. For future Water Years 2000 through 2019, the average annual number of samples collected during the fall/winter season under the current and proposed sampling schemes are 52 and 19, respectively. This corresponds to an average annual sample reduction of 60 percent during the fall/winter season.

During the spring and summer seasons, 370 suspended sediment samples were collected from the Missouri River at Sioux City. There were 43 point-integrated samples collected during Water Years 1980 through 1991 and 327 depth-integrated samples collected during Water Years 1992 through 1999. The streamflow intervals and recommended samples sizes for the spring/summer season are listed in Table 1. Since the streamflows in Interval 5 are high, infrequent and can transport a significant amount of sediment, suspended sediment samples should be collected for all flows in this interval. For future Water Years 2000 through 2019, the average annual number of samples collected during the spring/summer season under the current and proposed sampling schemes are 52 and 33, respectively. This corresponds to an average annual sample reduction of 36 percent during the spring/summer season.

Table 1. Streamflow intervals and recommended annual suspended sediment sample sizes for the sampling site at Sioux City, Iowa.

Streamflow Interval	Streamflow Range (ft^3/s)	% of 20-Year Suspended Sediment Discharge	Recommended Annual Sample Size
Fall/Winter Season: October 1 through March 31			
Interval 1	≤ 9,500	< 1%	None
Interval 2	9,501 – 15,000	6%	5
Interval 3	15,001 – 24,000	16%	5
Interval 4	24,001 – 38,000	40%	8
Interval 5	38,001 – 60,000	28%	6
Interval 6a	60,001 – 73,000	9%	7
Interval 6b	> 73,000	< 1%	All
Spring/Summer Season: April 1 through September 30			
Interval 1	≤ 24,000	< 1%	3
Interval 2	24,001 – 38,000	53%	19
Interval 3	38,001 – 60,000	33%	24
Interval 4	60,001 – 94,000	13%	40
Interval 5	94,000	< 1%	All

Sampling Site at Omaha, Nebraska: Rating curves and sampling designs were developed for the fall/winter and spring summer seasons at the sampling site at Omaha, Nebraska. The sampling site is located on the Missouri River at the USGS Gaging Station 06610000, "Missouri River at Omaha, NE", at River Mile 615.9, which is approximately 300 feet downstream of the Interstate 480 Highway Bridge.

During the fall/winter season, 194 suspended sediment samples were collected from the Missouri River at Omaha, Nebraska, between Water Years 1977 and 1999. There were 11 point-integrated samples collected between Water Years 1977 and 1991 and 183 depth-integrated samples collected during Water Years 1993 through 1999. Suspended sediment samples were not collected during the fall/winter season of Water Year 1992. All of the point-integrated samples were collected during the month of October. The streamflow intervals and recommended samples sizes are listed in Table 2. The recommended sample size for Interval 1 is zero because the streamflows in this interval do not transport a significant amount of sediment. Since the streamflows in Interval 6b are high, infrequent and can transport a significant amount of sediment, suspended sediment samples should be collected for all flows in this interval. For future Water Years 2000 through 2022, the average annual number of samples collected during the fall/winter season under the current and proposed sampling schemes are 52 and 34, respectively. This corresponds to an average annual sample reduction of 33 percent during the fall/winter season.

During the spring and summer seasons, 375 suspended sediment samples were collected from the Missouri River at Omaha, Nebraska between Water Years 1977 and 1999. There were 46 point-integrated samples collected during Water Years 1977 through 1991 and 329 depth-integrated samples collected during Water Years 1992 through 1999. The streamflow intervals and recommended samples sizes for the spring/summer season are listed in Table 2. The recommended sample size for Interval 1 is zero because there were no streamflows in this interval between Water Years 1977 and 1999. Since the streamflows in Interval 5 are high, infrequent and can transport a significant amount of sediment, suspended sediment samples should be collected for all flows in this interval. For future Water Years 2000 through 2022, the average annual number of samples collected during the spring/summer season under the current and proposed sampling schemes are 52 and 37, respectively. This corresponds to an average annual sample reduction of 28 percent during the spring/summer season.

Table 2. Streamflow intervals and recommended annual suspended sediment sample sizes for the sampling site at Omaha, Nebraska.

Streamflow Interval	Streamflow Range (ft^3/s)	% of 23-Year Suspended Sediment Discharge	Recommended Annual Sample Size
Fall/Winter Season: October 1 through March 31			
Interval 1	≤ 11,000	< 1%	None
Interval 2	11,001 – 18,000	7%	6
Interval 3	18,001 – 29,000	19%	9
Interval 4	29,001 – 46,000	41%	17
Interval 5	46,001 – 73,000	28%	10
Interval 6a	73,001 – 77,000	4%	5
Interval 6b	> 773,000	< 1%	All
Spring/Summer Season: April 1 through September 30			
Interval 1	≤ 24,000	0%	None
Interval 2	24,001 – 38,000	28%	16
Interval 3	38,001 – 60,000	48%	15
Interval 4	60,001 – 97,000	21%	47
Interval 5	97,000	3%	All

Sampling Site at Nebraska City, Nebraska: Rating curves and sampling designs were developed for the fall/winter and spring summer seasons at the sampling site at Nebraska City, Nebraska. The sampling site is located on the Missouri River at the USGS Gaging Station 06807000, "Missouri River at Nebraska City, NE", at River Mile 562.6, which is approximately one mile upstream of the Highway 2 Bridge.

During the fall/winter season, 200 suspended sediment samples were collected from the Missouri River at Omaha, Nebraska, between Water Years 1977 and 1999. There were 12 point-integrated samples collected between Water Years 1977 and 1991 and 188 depth-integrated samples collected during Water Years 1993 through 1999. Suspended sediment samples were not collected during the fall/winter season of Water Year 1992. All of the point-integrated samples were collected during the month of October. The streamflow intervals and recommended samples sizes are listed in Table 3. The recommended sample size for Interval 1 is zero because the streamflows in this interval do not transport a significant amount of sediment. Since the streamflows in Interval 6 are high, infrequent and can transport a significant amount of sediment, suspended sediment samples should be collected for all flows in this interval. For future Water Years 2000 through 2022, the average annual number of samples collected during the fall/winter season under the current and proposed sampling schemes are 52 and 33, respectively. This corresponds to an average annual sample reduction of 35 percent during the fall/winter season.

During the spring and summer seasons, 380 suspended sediment samples were collected from the Missouri River at Nebraska City, Nebraska between Water Years 1977 and 1999. There were 43 point-integrated samples collected during Water Years 1977 through 1991 and 337 depth-integrated samples collected during Water Years 1992 through 1999. The streamflow intervals and recommended samples sizes for the spring/summer season are listed in Table 3. The recommended sample size for Interval 1 is zero because there were no streamflows in this interval between Water Years 1977 and 1999. Since the streamflows in Interval 6 are high, infrequent and can transport a significant amount of sediment, suspended sediment samples should be collected for all flows in this interval. For future Water Years 2000 through 2022, the average annual number of samples collected during the spring/summer season under the current and proposed sampling schemes are 52 and 45, respectively. This corresponds to an average annual sample reduction of 13 percent during the spring/summer season.

Table 3. Streamflow intervals and recommended annual suspended sediment sample sizes for the sampling site at Nebraska City, Nebraska.

Streamflow Interval	Streamflow Range (ft^3/s)	% of 23-Year Suspended Sediment Discharge	Recommended Annual Sample Size
Fall/Winter Season: October 1 through March 31			
Interval 1	≤ 15,000	< 1%	None
Interval 2	15,001 – 23,000	6%	7
Interval 3	23,001 – 37,000	19%	7
Interval 4	37,001 – 59,000	42%	15
Interval 5	59,001 – 82,000	29%	19
Interval 6	> 82,000	4%	All
Spring/Summer Season: April 1 through September 30			
Interval 1	≤ 23,000	0%	None
Interval 2	23,001 – 37,000	8%	7
Interval 3	37,001 – 59,000	43%	30
Interval 4	59,001 – 93,000	35%	16
Interval 5	93,001 – 139,000	12%	7
Interval 6	> 139,000	2%	All

CONCLUSIONS

Flow-stratified sampling designs were developed for the fall/winter and spring/summer seasons for suspended sediment sampling on the Missouri River at three sampling sites: Sioux City, Iowa; Omaha, Nebraska; and Nebraska City, Nebraska. When hypothetically implemented for future water years, the proposed sampling designs required significantly fewer samples than the current twice-weekly sampling scheme. The average annual number of samples collected (and would have been collected) under the current and proposed sampling schemes for each of the sampling sites are listed in Table 4. More samples were required at the Omaha and Nebraska City sites because there was a more scatter in the suspended sediment measurements compared to the Sioux City measurements.

Table 4. Comparison of average annual sample sizes with the twice weekly and proposed sampling schemes at the three sampling sites.

Sampling Site	Average Annual Sample Size with Twice Weekly Sampling Schedule	Annual Average Sample Size with Proposed Sampling Scheme	Average Reduction in Annual Sample Size using the Proposed Sampling Scheme
Sioux City, Iowa	104	52	50%
Omaha, Nebraska	104	72	31%
Nebraska City, Nebraska	104	79	24%

The sample sizes for the spring/summer season were generally larger than the sizes for the fall/winter season. This is because during individual water years, the streamflows were historically concentrated in one or two intervals during the spring/summer season, whereas streamflows tended to span several intervals during the fall/winter

season. Also, there was more scatter in the spring/summer data compared to the fall/winter data. Because of this large amount of scatter in the spring/summer data, more data points were required to improve the accuracy of the rating curve. For the spring/summer season, the recommended number of samples generally increased with streamflow. More samples were required at higher streamflows for the following reasons: (1) during individual water years, higher streamflows transport a larger fraction of the suspended sediment discharge; and (2) historically, fewer samples have been collected at the higher streamflows. Therefore, more data points are required at higher streamflows to improve the precision and accuracy of the rating curve for these flows.

The approach used to formulate the sampling strategies resulted in seasonal, flow-stratified sampling designs having the following properties: (1) suspended sediment samples that span the entire streamflow range for low, moderate and high water discharges; (2) more suspended sediment samples concentrated at streamflows intervals having the largest fraction of the total suspended sediment discharge; (3) number of annual suspended sediment samples necessary to observe shifts in the suspended sediment rating curves that can occur in response to changes in the watershed; and (4) an adequate number of samples that will maintain or improve the precision and accuracy of the suspended sediment rating curves.

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