New Isokinetic Version of LISST Technology Targets Needs of the Federal Subcommittee on Sedimentation

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Overview: Optical turbidity sensors, optical transmissometers, and acoustic backscatter sensors have been well entrenched in the monitoring of suspended sediments. However, definitive results published recently by (Sutherland, 2000) note the two difficulties with turbidity sensors: the calibration is changed whenever grain size changes, and further, calibration also changes significantly with particle color. Similarly, a survey by (Davies-Colley, 2001) notes that transmissometers also change calibration with grain size, and have upper size cut-offs, (Voss, 1993). These results confirm what is expected from Mie's classic theory of light scattering by spheres. Acoustics usually operate at frequencies where $a/\lambda << 1$, (a is grain radius and λ is acoustic wavelength) where scattering varies as *volume-squared*, again not suitable for a mixture of grain sizes. In contrast to these 3, **laser diffraction** methods measures multi-angle scattering at small angles from which size-distribution and concentration is computed. Measurements of concentration, TSS, are unaffected by changes in grain size or color (refractive index). The technique is widely used in industry. The present authors pioneered its use in the aquatic environment. In this paper, we describe the fundamentals of the technology, we note research currently in progress in the Grand Canyon by USGS scientists, we describe a new instrument that permits measuring TSS in a *size-subrange*, we conclude with a preview of an *isokinetic* version of the instruments, the LISST-SL and with effects of particle shape.

What is Laser Diffraction: Laser diffraction is a technique pioneered in the 70's (Swithenbank et al., 1976). At the time, it was widely known from light scattering physics (Mie theory) that when angular scattering from a particle is examined in small forward angles, it appears identical to the diffraction pattern from an aperture of the same diameter . There is a simple conceptual reason for it. A particle blocks light waves. Some enter the particle, others are diffracted *around* the particle. The diffracted rays appear in the small-angle region. The rays that enter the particle are scattered over the full π angle range, so that their contribution to the small-angle region is minimal. This property permitted the replacement of particles with apertures. Particle composition and color, which are represented by the refractive index as a function of light wavelength[1], became irrelevant. From the diffraction signature, which has a characteristic shape termed the Airy function (Born and Wolf, 1975), particle size and concentration of particles could at once be determined by inversion of the small-angle light scattering data. In other words, if the small-angle scattering signature is observed, it leads via inversion to the size-distribution. When the size-distribution is summed, one has the total concentration, TSS. The mathematics of interpreting the multiple-small-angle scattering are briefly by us in our Marine Geology paper (Agrawal and Pottsmith, 2000).

Thinking of particles as same-size apertures, clearly, is a great convenience. For this reason, the method was called laser diffraction. Due to its ability to size particles regardless of their composition, it is now widely used in diverse industries – from chocolates, paints, cements, to pharmaceuticals. In 1994, we published the first use of this technology in the sea from an autonomous instrument, equipped fully with a computer and datalogger, running on battery (Agrawal and Pottsmith, 1994). Refinements to the idea of pure diffraction occurred for 2 reasons. First, there is indeed a small sensitivity in small-angle scattering to refractive index. Thus the desire for better accuracy was behind replacing the pure diffraction approximation with the full Mie theory model for scattering. The second such factor was the use of larger angles, reaching all around to 170 degrees as extensions of laser diffraction. At such large angles, it became essential to abandon the diffraction approximation, and use Mie theory.

Basic Implementation, LISST-100: Refer now to the optics shown in figure 2. A collimated beam illuminates particles. A receiving lens of focal length f collects scattered light. A detector is placed at thefocal plane of the receiving lens. All rays originating at a particle at an angle θ arrive at the detector at a distance from center r such that $\theta = \mathrm{atan}(r/f)$. For mathematical reasons of inverting the measured scattering to get size distribution, instead of measuring the scattered light at single points (representing single angles), ring detectors are used. These *rings* integrate all light scattered into a cone of angle centered

[1] e.g. a red particle has an imaginary component in its refractive index that has a minimum at red wavelength

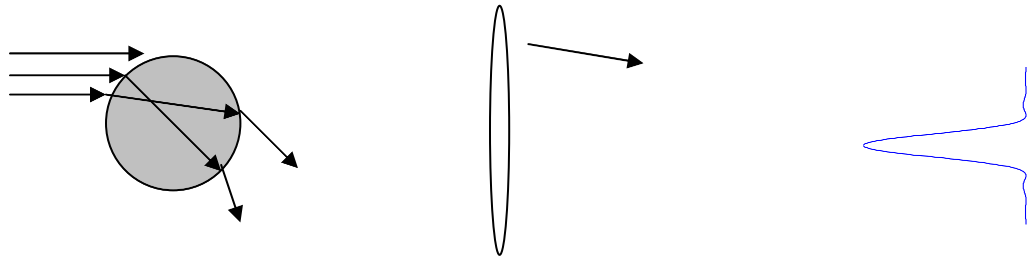
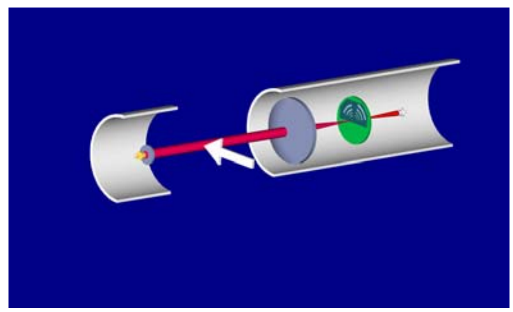


Figure 1: This sketch shows a parallel beam of light striking a spherical particle. The light that enters the particle – and that therefore feels its composition – exits at large angles to the original beam. It makes a very small contribution to the very small angle scattering. Only rays diffracted around the particle appear at the small angles, producing the Airy pattern shown on right. This is why the name: *laser diffraction*.

on θ. The radii of the rings increase in fixed proportion, i.e. the radius and width of each ring is a constant multiplier times the corresponding value for the previous ring. This *logarithmic* spacing of the rings also corresponds to a logarithmic spacing of particle sizes in the inversion. In other words, the size-distribution represents the concentration of suspended sediment in logarithmically spaced size bins. Logarithmic size-bins are familiar to geologists as sizes that are linearly spaced in φ units. This is our **LISST-100** instrument.

Figure 2: This is the **LISST-100.** A collimated laser beam emanates from left. Particles in the flow scatter light. A receiving lens collects the scattered light, which is detected by the *ring detector.* A hole in the center of the ring detector permits the focused laser beam to pass through, where its power is sensed. This constitutes a transmission measurement. This measurement corrects for attenuation of the scattered light that is sensed by the rings.



<u>New Developments</u>: As a precursor to the newest developments, we note first the development of the LISST-25 TSS sensor. The principle of the LISST-25 is based on ideas from laser diffraction, as follows. According to diffraction, the scattered light energy falls at larger angles on the ring-detector plane for finer particles, and vice versa. To measure true TSS, the sensed scattered light energy per unit sediment concentration should be identical for any size. Thus, crudely speaking, if the width of a ring at a large angle is proportional to the scattering per unit volume for the corresponding fine particle, and so on down to all rings, then the sum of these *modulated* rings would represent the true TSS. These rings can be joined together to form a single detector. Such a detector takes the shape of a *comet (lower form, right)*. The comet detector accomplishes an angle-weighted sum of scattering, which is directly proportional to TSS. Thus, unlike the old turbidity sensors or transmissometers, the LISST-25 responds directly to TSS, and since it too is grounded in laser diffraction principles, its calibration is held for all sizes and colors of particles. The upper wedge shaped detector senses total particle area. From these two detectors, the Sauter Mean diameter is computed as the ratio of volume/area concentrations.

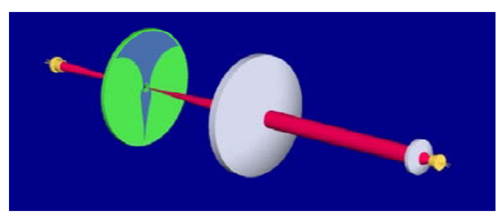


Fig.3: The use of shaped focal plane detector in LISST-25 for direct TSS measurement.

<u>LISST-25X</u>: The first new development since Reno-2001 is the LISST-25X. This instrument was designed in response to a need of the USGS Flagstaff scientists, wherein the required suspended sediment concentration was not to include fines below 63 micron in size. In response to this need, a family of new focal plane sensor geometries was invented. This family of geometries permits measuring the concentration in any sub-range of sizes. For example, one may measure concentration of particles greater than a threshold (*high-pass*), smaller than a threshold(*low-pass*), or within a band of sizes(*band-pass*). These detectors, replacing the ring detector, take the shape of truncated comets for high-pass, or *blobs* for low-pass. The first of these instruments are being tested in the Grand Canyon at about the time of this Conference (see Melis, this conference).

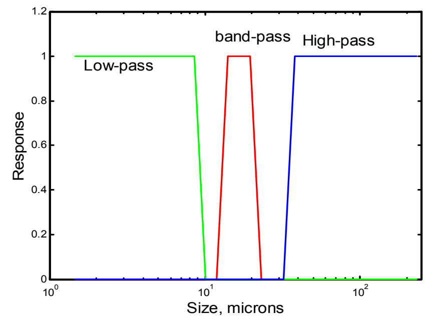


Figure 4: The LISST-25X embodies specially shaped focal-plane detectors that can permit the user to select the size-range over which TSS is to be measured. As example, a user may choose to ignore the wash-load in a stream, or use the LISST-25X to measure the wash-load only.

LISST-SL: The newest development underway at Sequoia is a streamlined, low-drag vehicle that encloses an isokinetic withdrawal LISST-100 instrument. This device includes pressure transducers to record depth of sampling. It actively equalizes the free-stream velocity and the withdrawal speed into the nose of the vehicle using a tiny pump. The device will run on external power and will use 2-wire communication protocol. Isokineticity is assured by measuring the free-stream velocity and adjusting an in-built flow-assist pump to control withdrawal rate. The LISST-SL will have the full size-distribution measuring capabilities of the LISST-100, although the housing can enclose the LISST-25 or 25X. Field trials are scheduled for summer of 2002.

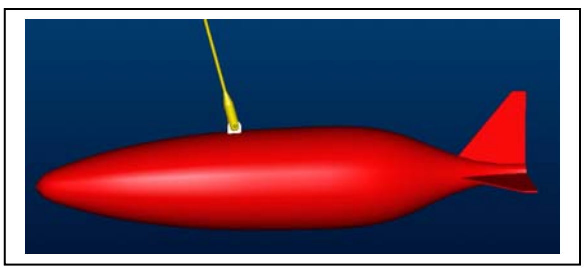
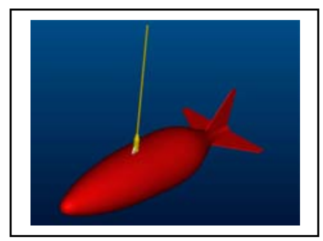


Figure 5: Two *artist's views* of the **LISST-SL**. A 2.5cm diameter opening at the nose draws water in.



Studies on Effect of Particle Shape: New research on the small-angle scattering properties of natural AC Coarse particles have been underway at Sequoia. Sorting the particles by settling velocity, scattering properties are measured using a LISST-100. Early data reveal differences from scattering by spheres. This work will be published elsewhere. The consequence of shape effects appears to that when small-angle scattering from random-shaped natural grains is inverted with a model based on apertures/spheres, fines are invented by the inversion, slightly biasing the TSS. In future, we envisage replacing the spheres model with a model for these natural grains, so that the data on small-angle scattering are inverted with a suitable model.

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LESSONS LEARNED FROM TURBIDITY FIELD MONITORING OF 12 METROPOLITAN ATLANTA STREAMS

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ABSTRACT

Introduction Gwinnett County, in metropolitan Atlanta, Ga., is one of the most rapidly growing areas in the United States. Nonpoint-source pollution is highly complex because it arises from varied, but unknown sources especially in areas of urban growth. The U.S. Geological Survey (USGS), in cooperation with Gwinnett County, Department of Public Utilities, established a water-quality monitoring program in 1996 to assess and analyze the impacts of nonpoint-source contamination. The program provides information that can aid land and water-resource managers in making resource management decisions that can affect water quality. The Gwinnett County Watershed Monitoring Program (GCWMP) includes the development of a network of real-time, continuous water-quality stations, which provide continuous monitoring of turbidity, specific conductance, flow and precipitation, augmented with intensive water-quality sampling and analysis of likely contaminants. Long-term monitoring may help to quantify and describe the fluctuation of contaminants within a stream. Analysis of water-quality within a stream, over time, may aid in delineating possible water-quality trends in the watershed; thereby identifying how land use and development may impact a watershed. A real-time monitoring network was installed and has been fully operational since September 2001. During the installation and monitoring phase of the project, many deployment concerns were addressed, and several adaptations were made to collect the best data possible. This paper describes the sonde deployment strategy, which includes the project design, implementation and modifications made to the water-quality monitoring network in Gwinnett County.

Scope and Study Area Gwinnett County, located in the Piedmont physiographic province, is one of the most rapidly growing areas in the Unites States. Gwinnett County is a mostly headwater area where streams drain into one of three major river basins the Chattahoochee, Ocmulgee, and Oconee. Land use varies greatly throughout the County; however, residential land use is more than 50 percent of the county's total land area when grouping all classes of residential land use. Twelve watersheds were selected for the network based on land use and watershed features. Drainage area, point-source discharges, suitability for instrumentation installation, stage-discharge control, flow characteristics, and availability of existing stage-discharge relations were considered in selecting monitoring locations within each watershed. The stations provide real-time continuous, water-quality data in watersheds that represent a wide range of land-use conditions and encompass more than 70 percent of Gwinnett County. The monitored basins range in size from 1.42 to 162 square miles. Six stations have operated since 1996 as stage and periodic water-quality sampling sites, and six additional stations were added in 2001 when the project became real-time. Of the twelve sites, five are located at culvert sites, the remainder are located at bridge sites.

Sonde Deployment Strategy The first step in developing the sonde deployment strategy for the 12 streams in the GCWMP was to perform a reconnaissance to identify a stream-reach where gage construction would be practical. Once a stream-reach was selected the next step was to

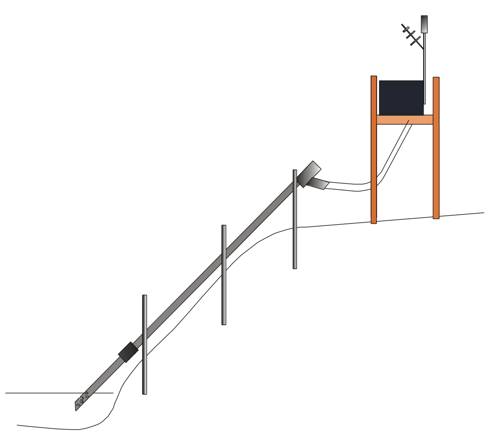
verify that the sonde location was representative of the stream cross section; urban streams often have large flow variability, low flow or base flow during dry periods, and relatively large flows during runoff events.

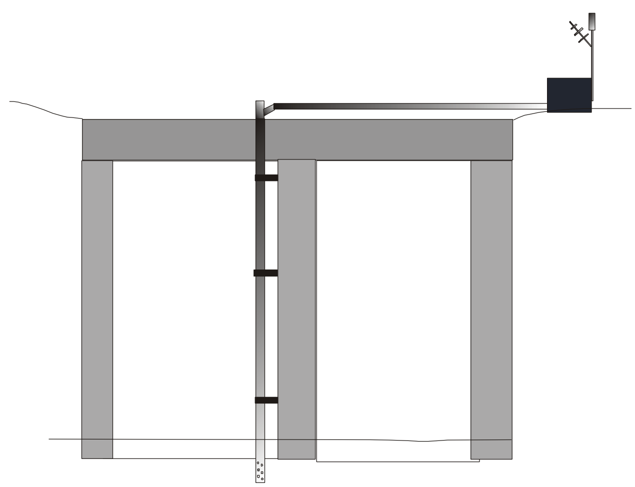
A final location was chosen when the following criteria were met:

- Adequate mixing of the stream where-by the position of the insitu sonde was representative of the whole cross-section at low, medium, and high flow,
- Sufficient velocity to create a natural flushing of the sonde to reduce fouling caused by debris,
- The sonde must be safely serviced/retrieved at all ranges of stage,
- Adequate protection of the sonde during high flow,
- Adequate depth during low flow.

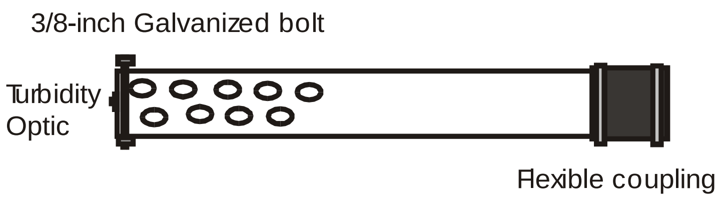
The position of the sonde relative to stream-depth is an important consideration. For example, it was estimated that if the sonde were deployed at least twelve inches off the streambed, the effects of bedload during high flow would be minimized. However deploying the sonde a minimum of twelve inches off the streambed led to concerns that the sonde sensors would not remain submerged, or only half of the sonde bulkhead would be submerged. At two installations, the control was modified to "build up" the sonde pool to ensure that the sensors would remain submerged. The manufacturer of the sonde alleviated concerns regarding the necessity of submerging the sonde bulkhead. Therefore it was decided that if the proposed deployment would guarantee total submersion of the sensors during all conditions of flow, the installation would proceed.

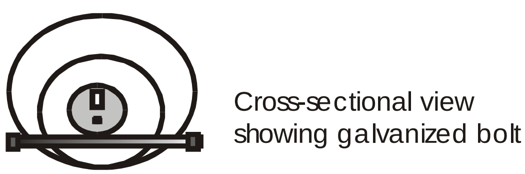
Two types of sonde deployment configurations were used depending on the conditions at the site. Where possible, a bank installation was chosen over a headwall mount as shown in the diagram below.





Both configurations use four-inch schedule 40-PVC pipe supported by either signpost rails or four-inch "U" brackets. At the landward end of the pipe there is a PVC "Y" connector with a locking four inch well cap, which allows for easy retrieval of the sonde. The communication cable is run through the 45-degree sweep of the "Y" connector, through a four-inch by two-inch slip reducing bushing, which connects to a length of two-inch flexible conduit, which is then run to the gagehouse. The streamward end of the pipe was modified several times until a satisfactory design was reached. The design development is as follows. At the bank installations, the first approach utilized a four-inch "T" placed inline with the flow of the stream. It was hoped that the "T" would funnel stream water across the sonde; however, the "T" proved to be a debris trap. A four-inch landscaping screen was added to the upstream opening. The screen slowed the stream velocity and created an eddy, which in turn directed debris into the "T" from the downstream end. The "T" was removed and ¾ -inch holes were drilled into the four-inch pipe. The pipe was left open-ended with a set bolt to ensure that the sonde was installed at the same position after each service. The open-ended vertical pipe was the configuration used at the culvert sites. At this time a suggestion was made by the visiting sonde representative that a screen should be wrapped around the outside of the four-inch PVC pipe to reduce the collection of debris inside the pipe. A ¼-inch landscape netting was used; however this proved to be an attachment point for filamentous algae growth which often produces false readings by the turbidity optic. The final modification proved to be the most successful. The devices used to protect the sensors were acting as traps for debris. The turbidity optic needs an unobstructed view of the creek. Therefore the bottom of the sonde guard was cut off and a new section of 4-inch PVC pipe was installed with a set bolt that positioned the turbidity optic flush with the end of the pipe as shown in the diagram below. The new PVC pipe was drilled with 1&1/8 –inch holes and the sonde guard, sonde, including the sensors, and PVC pipe were treated with an anti-fouling spray. The new length of PVC pipe was attached to the existing PVC via a flexible coupling.





The flexible coupling allows for easy removal and cleaning of the sonde housing, and absorbs the impact of debris during high flow. The sonde within the pipe is secured and retrieved with a steel cable.

Conclusions The Gwinnett County Watershed Monitoring Program, which includes the real-time monitoring of turbidity, has been fully operational since September 2001. A sonde deployment strategy was used to identify suitable locations for the deployment of the water-quality sonde. During the construction and operating phases of the project, several modifications were made to the original design. The current design, which will be used in upcoming and developing projects within the USGS, Georgia District, allows for the collection of the best data possible, and is used by water resource managers to make timely decisions regarding water-quality within twelve watersheds in the County.

METHODS FOR CONTINUOUS AUTOMATED TURBIDITY MONITORING IN BRITISH COLUMBIA, CANADA

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ABSTRACT

Introduction: Continuous automated water quality monitoring (AWQM) of turbidity is relatively new in Canada compared to traditional discrete monitoring methods.

The *Constitution Act, 1867* of Canada delineates federal and provincial legislative powers. Section 91 establishes federal jurisdiction over seacoasts and inland fisheries. Section 92 and Section 109 establish provincial jurisdiction over natural resources, which includes water. Both levels of government monitor water resources.

The Department of Environment Canada (EC) and the Department of Fisheries and Oceans Canada (DFO) regulate and monitor water resources. In BC, several ministries monitor freshwater resources including the Ministry of Forests, Ministry of Water, Land and Air Protection and the Ministry of Sustainable Resource Management (MSRM). MSRM develops and administers standard methods and protocols for monitoring water resources including water quantity and water quality. MSRM established the research and development AWQM station on the Sooke River on Vancouver Island, BC to research, test, and document new methods and protocols in the *Automated Water Quality Monitoring Field Manual* (Burke 2002).

Characteristics of the Study Area: Vancouver Island is comprised of accreted terranes. The bedrock consists of metamorphic sedimentary and volcanic rock and igneous complex, sandstone, shale and conglomerates. The overlay consists of glacial and fluvial deposition. The dominant soils include brunisols and podzols of porous gravel and quartz sand with a slightly acidic signature. The dominant biogeomatic classification is Coastal Western Hemlock (*Tsuga heterphylla*), Western Red Cedar (*Thuja plicata*) and Douglas fir (*Pseudotsuga menziesii*). The climate is wet maritime with mild wet winters and warm dry summers with a mean annual precipitation of 50 inches and mean temperature of 48 degrees Fahrenheit.

The Sooke River watershed area is 150 square miles. The headwaters consist of the Leech River complex and the Sooke Lake, which provides the drinking water for the city of Victoria. Historically, the watershed has been logged and mined. The lower Sooke River lies in a floodplain that is rural residential with homes and small hobby farms. Other stakeholder interests include active timber harvesting, development, and the T'Sou-Ke First Nations.

The mean annual discharge of the Sooke River is 335 cubic feet per second. The substrate is cobble, boulder and fines. The river supports freshwater fish species and anadromous salmon including Chum (*Oncorhynchus keta*), Chinook (*O. tshawytscha*), Coho (*O. kisutch*), and

Steelhead trout (*Salmo gairdneri*). Wildlife includes deer, bear, cougar, small mammals, raptors such as bald eagles and waterfowl.

The Sooke River AWQM Station Design: The Sooke River AWQM station is located at 48°25’28”N and 123°42’45”W. The station is a passive angle bank deployment design. Two equipment system configurations have been deployed. System A, deployed from November 2000 to October 2001, was comprised of a Forest Technology Systems (FTS) data logger, Stevens vented pressure transducer, YSI 600XL multi-sonde that measured conductivity, dissolved oxygen, pH, and temperature, and an analite turbidity sensor with a mechanical wiper arm. System B, deployed in October 2001 and currently in operation, is comprised of a Handar 555 data logger, Stevens vented pressure transducer, YSI 6820 multi-sonde that measures conductivity, dissolved oxygen, pH, temperature and turbidity with a mechanical wiper arm. The data are logged in fifteen-minute intervals and retrieved manually.

The calibrated range of the turbidity sensors is 0 to 400 NTU. The normal reported range of turbidity at this location is 0 to 5 NTU with an annual mean of 2.7 NTU where the sample number is 48 discrete measurements based on a twelve month baseline study from September 1999 to August 2000 (Burke 2000). Precipitation events elevate water flow and turbidity readings.

The Sooke River AWQM Station Operation: Station operation includes certification, bench testing, verification, and quality control and quality assurance (QA/QC). The equipment system components must be calibrated and certified by the manufacturer or authorized representative. The system must be bench tested by the AWQM technician prior to deployment The sensors must perform within specified criteria, such as within 10% of a certified standard turbidity solution, prior to deployment. The AWQM technician completes maintenance visits on a bi-weekly or monthly basis.

Verification of AWQM Turbidity Data: The AWQM technician cleans the optics on the turbidity sensor and rotates the mechanical wiper arm. The turbidity data are verified by three methods.

First, the performance, or drift, of the turbidity sensor is verified by measurements in certified standard turbidity solutions and distilled water. The solutions must be in containers that have a flat black surface to minimize local interference. The sensors have been verified using 100 NTU polymer bead solution manufactured by FTS and YSI INC. and 100 NTU formazin solution manufactured by HACH INC. The results indicate that the stability of the standard solutions varies between manufacturers.

Second, the turbidity data are verified by obtaining a discrete sample of surface water and comparing turbidity data between the AWQM turbidity sensor and a certified and calibrated HACH 2100P turbidity field meter. The results indicate that the field meters usually provide a sound comparison for low turbidity conditions; however, variance increases for higher turbidity conditions. Even so, the question remains “Which meter?” Consequently, field meter comparisons are used only as a general comparison.

Third, the turbidity data are verified by obtaining two discrete surface water samples for laboratory analysis. The first sample is obtained adjacent to the AWQM turbidity sensor *in situ* and verifies the data obtained by the AWQM turbidity sensor. A second sample is obtained from *in situ* mid-stream and is used as a measurement to determine if the AWQM sensor is obtaining data that is representative of the environmental conditions of the water body. The QA/QC for discrete samples is ten to twenty percent for replicates and blanks. The results show strong agreement between the AWQM turbidity sensor and the adjacent water column and the midstream water column. The laboratory results are the primary basis to determine if the AWQM turbidity sensor is measuring data that are representative of the environment.

Potential Interference's to AWQM Turbidity Data: The AWQM turbidity sensors are subject to specific interferences that include bio-fouling, physical fouling, signal noise, optic damage, entrained gas bubbles, sunlight spikes, hydrodynamic noise, calibration drift, temperature effects, and power-up interference (White 1999). Each potential interference must be taken into account in the system design, operation, maintenance, and data management.

Data Management: The BC MSRM developed three primary data bases: Environmental Monitoring Systems (EMS) for location information and laboratory results; Water Inventory Data Management (WIDM) for hydrometric data; and Water Quality Data Management System (WQDM) for AWQM time series and meta data. AWQM data are defined by data grades A, B, C and D, which reflect the quality of the data. The criterion are based on the performance of the equipment and the level of required QA/QC. All raw data are entered into the data warehouse and can be corrected based on data shift or drift. Data are approved and audited.

Future Study: The BC MSRM anticipates to continue to develop standard methods for other water quality variables, develop an internet based interface for data users, and integrate environmental monitoring into one data warehouse.

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COMPARISON OF ESTIMATED SEDIMENT LOADS USING CONTINUOUS TURBIDITY MEASUREMENTS AND REGRESSION ANALYSIS

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ABSTRACT

Suspended-sediment loads commonly are estimated with a streamflow regression model. However, turbidity, which is the reduction in the transparency of water due to suspended and dissolved particles, may be a better surrogate than streamflow in estimating suspended-sediment loads. To test this hypothesis, regression equations that relate suspended-sediment concentrations to discrete turbidity measurements were developed for eight U.S. Geological Survey stream-gaging stations in Kansas. For comparison, estimates also were calculated using simple regression equations with streamflow and multiple regression equations with streamflow and turbidity.

Turbidity Measurements and Regression Analysis: Between 1998 and 2001, about 20 discrete water samples were collected at each of the eight stream-gaging stations and analyzed for suspended sediment. Samples were collected throughout a range of streamflow conditions and sediment concentrations. In addition, samples collected for suspended-sediment analysis represented the range of recorded turbidity values, with a nearly equal representation of high and low values at most stations. The eight stations are equipped with water-quality monitors that provide relatively inexpensive, continuous (hourly) measurements of turbidity. The water-quality monitors are serviced at least monthly to check calibration and to verify that the continuous monitor is representing the stream cross section. Site-specific regression equations were developed relating laboratory analyzed suspended-sediment concentrations in the discrete samples to turbidity measurements recorded by the water-quality monitors. Suspended-sediment loads were estimated using the continuous turbidity measurements and were compared to suspended-sediment loads estimated with streamflow measurements. Examples from two of the eight gaging stations are shown in figure 1.

Results: Suspended-sediment concentrations in the Kansas River at DeSoto were strongly related to turbidity with a coefficient of determination (R^2) of 0.987, compared to an R^2 of 0.792 for streamflow. The suspended-sediment concentrations in the Little Arkansas River at Sedgwick were strongly related to both streamflow and turbidity, and the estimated average daily load did not differ substantially between the streamflow and the turbidity equations.

The results showed that, in general, suspended-sediment loads at stream-gaging stations where flows are affected by human-related factors (for example, reservoir releases) were more strongly related to turbidity than to streamflow. The Kansas River is affected by a series of reservoirs that act as sediment traps. During large reservoir releases, the streamflow at DeSoto increased, whereas turbidity increases were relatively small. On the other hand, during periods of substantial storm runoff, large increases were seen in both streamflow volume and turbidity values. The difference in sediment loads estimated using streamflow and turbidity regression equations (fig. 1) is about 8 million tons per year for the Kansas River at DeSoto, which demonstrates the need to determine whether streamflow or turbidity is a better surrogate. There are no large reservoirs on the Little Arkansas River to affect the flow, which may be why suspended-sediment concentration is more strongly related to streamflow at this gaging station (compared to the DeSoto station).

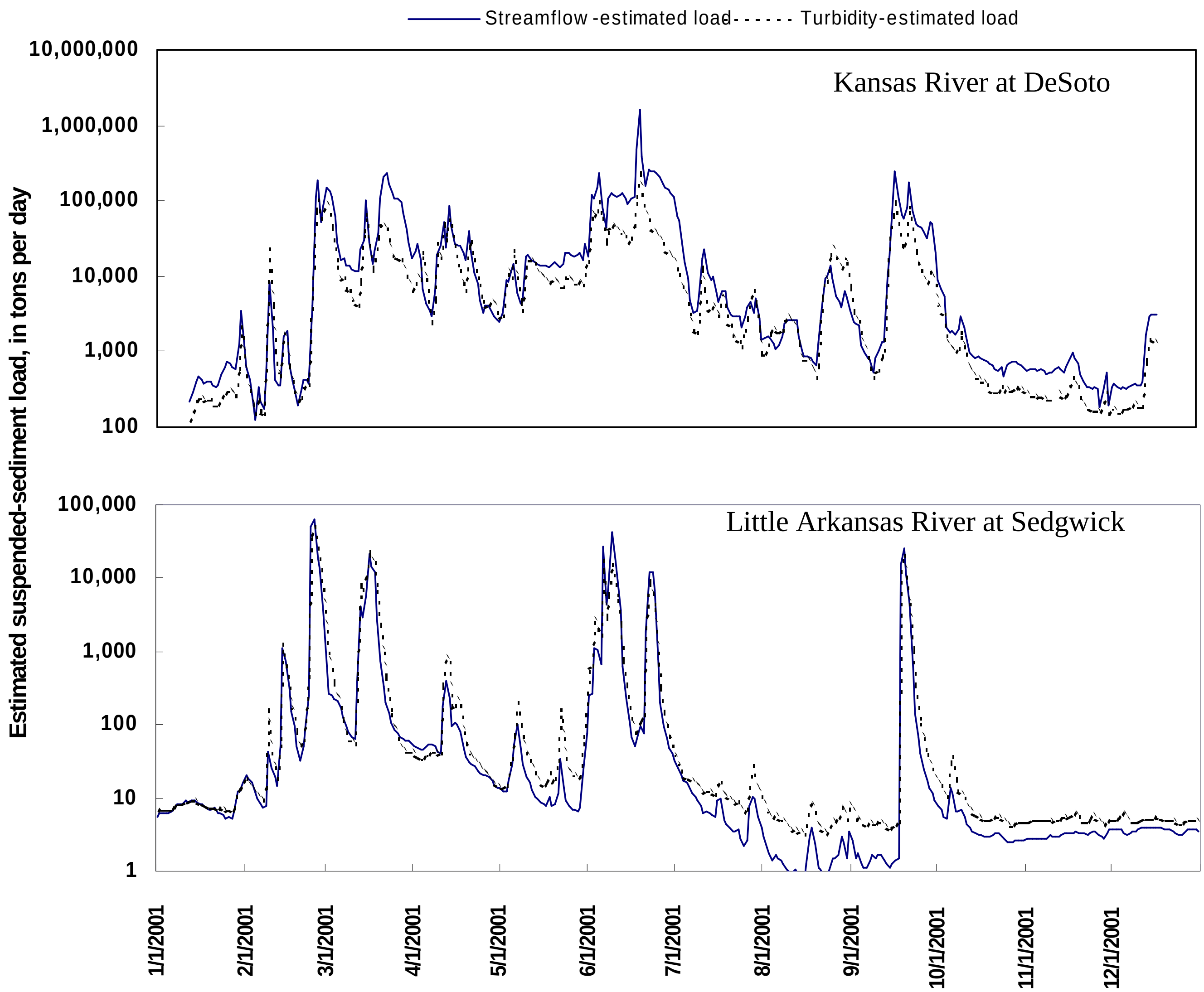

Figure 1. Estimated suspended-sediment loads for the Kansas River at DeSoto and the Little Arkansas River at Sedgiwck, Kansas, 2001.

The suspended-sediment concentrations for all eight gaging stations were significantly correlated to turbidity. The suspended-sediment concentrations at four of the eight gaging stations were also correlated to streamflow. The relation between suspended-sediment concentration and turbidity is affected by particle-size distribution (samples with the same sediment concentration but different particle sizes may have different turbidity measurements). However, the median particle size for all samples used in the regression analyses was 95-percent fines (particles smaller than 0.065 millimeters). This may indicate that suspended particle sizes in Kansas streams are generally small and have a relatively consistent relation to turbidity.

To determine whether streamflow or turbidity is a better surrogate for suspended sediment, a comparison was made between instantaneously measured suspended sediment loads and streamflow- and turbidity-estimated loads (table 1), using all the manually collected suspended-sediment samples used in the regression analyses (1998-2001). For the Kansas River at DeSoto, the difference between the measured and the streamflow-estimated suspended load is more than 100 percent, whereas the difference between the measured and turbidity-estimated load is about 4 percent. For the Little Arkansas River at Sedgwick, the difference between the measured and the streamflow-estimated suspended load is about 50 percent, whereas the difference between the measured and turbidity-estimated load is 6 percent.

Based on the results for these two Kansas stations, turbidity is a more reliable surrogate for determining suspended-sediment loads.

Table 1. Comparison of measured instantaneous suspended-sediment loads to streamflow- and turbidity-estimated suspended-sediment loads in the Kansas River at DeSoto and the Little Arkansas River at Sedgwick, Kansas, 1998-2001.

	Kansas River at DeSoto	**Little Arkansas River at Sedgwick**
Number of samples	24	33
Mean measured suspended-sediment concentration (milligrams per liter)	679	434
Mean measured streamflow (cubic feet per second)	8,520	1,530
Mean measured instantaneous suspended-sediment load (tons/day)	49,500	3,010
Mean streamflow-estimated instantaneous suspended-sediment load (tons/day)	106,000	4,610
Percentage difference from measured load	**-110**	**-53**
Mean turbidity-estimated instantaneous suspended-sediment load (tons/day)	47,200	2,830
Percentage difference from measured load	**4.6**	**6.0**

<u>Limitations:</u> Turbidity meters may have an upper limit that should be considered before using continuous turbidity measurements to calculate suspended-sediment load. Typically, limits vary for each meter and range from about 1,200 to 1,800 nephelometric turbidity units (NTU). The limit for the meter in the Little Arkansas River at Sedgwick is approximately 1,750 NTU. This limit was not reached in 2001. However, the limit for the meter in the Kansas River at DeSoto is about 1,200 NTU. The turbidity measurements reached this limit for parts of 7 days during 2001. The turbidity record was truncated during these periods. If the turbidity measurements are not adjusted, the estimated suspended-sediment load could be underestimated. In addition, there are 12 days of missing measurements in January 2001 due to ice at the Kansas River gaging station.

For more information on the real-time, continuous monitoring and regression analysis to estimate constituent concentrations and loads refer to http://ks.water.usgs.gov/Kansas/rtqw/

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CONTINUOUS TURBIDITY MONITORING IN STREAMS OF NORTHWESTERN CALIFORNIA

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ABSTRACT

Overview. Redwood Sciences Laboratory, a field office of the USDA Forest Service, Pacific Southwest Research Station has developed and refined methods and instrumentation to monitor turbidity and suspended sediment in streams of northern California since 1996. Currently we operate 21 stations and have provided assistance in the installation of 6 gaging stations for agencies, municipalities, universities, and citizens groups.

These installations employ a method called Turbidity Threshold Sampling (TTS), an automated data collection and sampling system in which a data logger employs real-time turbidity to control a pumping sampler (Lewis and Eads, 2001; Lewis, 1996). It is common in streams and rivers for most of the annual suspended sediment to be transported during a few, large rainstorm events. Automated data collection is essential to effectively capture such events.

TTS was designed to permit accurate determination of suspended sediment loads by establishing a relation between suspended sediment concentration (SSC) and turbidity for each sampling period with significant sediment transport. It does so by collecting pumped suspended sediment samples when pre-selected turbidity conditions, or thresholds, are satisfied. During analysis the relations are applied to the nearly continuous turbidity data for the respective sampling periods to produce a continuous record of estimated SSC (Lewis, 2002). The product of discharge and estimated SSC is then integrated to obtain accurate suspended sediment yields. Additional benefits of TTS are (1) it provides samples that can be used to determine whether turbidity spikes resulted from fouling or actual sediment transport, and (2) the continuous record of turbidity is useful for revealing the timing of erosion events, assessing impacts on beneficial uses, and enforcing water quality regulations.

Installation, fouling, and maintenance. Key requirements for collecting good turbidity data are real-time data filtering, proper mounting and housing of the sensor, selecting a sensor with a reliable wiper, regular inspection of the data, and maintenance of the equipment.

Real-time data filtering replaces a series of values taken rapidly over a short time period with a measure of central tendency of the series. We record the median of 60 values taken at half-second intervals. Examination of individual values from such short-duration series' reveals that elevated values commonly occur with no change in SSC. These contribute to a noisy record if recorded without filtering. The arithmetic mean is sensitive to outliers and, as such, is not nearly as effective as the median in removing the influence of stray values.

We have experimented with several different types of sensor mounting configurations

1. fixed-bracket mounted to the streambed
2. depth-proportional boom anchored to the streambed
3. articulating boom mounted on the stream bank
4. articulating boom mounted on a bridge
5. articulated cable-mounted boom spanning the channel

The first two configurations are not recommended because (1) sensors mounted too close to a mobile bed produce erratic turbidity readings, and (2) the sensor is not accessible at high flows. Articulating booms are designed to keep the sensor out of the bedload zone but adequately submerged at all flows. The booms are retractable, permitting access to the sensor at all flows, and they pivot both longitudinally and laterally upon impact to release large woody debris. The boom swings downstream and the sensor rises in the water column as velocity and depth of flow increase, so the boom must be appropriately weighted to keep the sensor from hydroplaning at the highest flows.

We have deployed the OBS-3 probe, manufactured by D&A Instrument Co., at all of our sites. In recent comparisons with the DTS-12, manufactured by FTS, Inc., the self-cleaning wiper on the DTS-12 prevented most episodes of fouling experienced by an OBS-3 mounted beside it on the same boom. However, a wiper can only prevent fouling from small contaminants such as fine organics and sediment, algae, and macroinvertebrates. Larger debris must be manually removed.

We have experimented with flow-through housings but now deploy a design made from square aluminum tubing that is open on the downstream end, and cut on an angle, allowing the sensor's optics to look across the flow or downstream, depending on the optical configuration. The housings are fastened to the downstream side of the boom in approximate alignment with the flow. The flow-through housing was screened at the upstream end, and the sensor required an intercept offset to remove the bias of viewing the pipe wall during low turbidities. The velocity inside the housing was restricted by this design, especially when the screen was clogged with debris, and readings were often unresponsive at lower flows or elevated by sediment that had settled inside the pipe. The housings are designed to shed debris that could potentially interfere with the sensor's viewing area, but have been only partially successful in that respect. Further design modifications, such as increasing the distance from the boom to the optical viewing area, might reduce the amount of fouling. Additionally, a sensor with a small viewing volume is less likely to view trapped debris. The OBS-3 has a relatively large viewing volume and the manufacture recommends placing it at least 20 cm from the nearest object.

In shallow streams, it is difficult to keep the sensor submerged at all flows without placing it close to the stream bottom. Therefore, we have had the most success positioning sensors in natural or artificially created scour pools. However, pools that scour and fill with each event, or that have excessive turbulence, are poor choices for sensor deployment. Close proximity to the water surface is also to be avoided to prevent entrainment of air at high flows or saturation of the sensor's detector with solar radiation. In shallow streams we shield the sensor's optics with a visor that prevents direct exposure to sunlight.

Routine site maintenance related to the turbidity sensor includes

1. inspecting the sensor and removing debris or cleaning as necessary
2. downloading and plotting the data to ensure the sensor is functioning properly
3. recording detailed field notes, including the times of any disturbances or manipulations
4. comparing the in-stream turbidity readings to Hach 2100P manual samples and adjusting the calibration offset if necessary (see Calibration section below).

Calibration. We consider two types of calibration here: (1) the calibration of the turbidity sensor to formazin, and (2) the calibration of SSC to turbidity. The first should be relatively stable while the second varies substantially throughout the year. The first needs to be checked upon shipment and once or twice a year. The second should not be considered fixed except during individual episodes of sediment transport. It is also possible to directly calibrate SSC to electronic output. Such calibrations fall in the same category as (2) above, i.e. they are very dynamic and need to be adjusted frequently. The TTS method is designed specifically to provide SSC data for type (2) calibrations of each event.

In estimating sediment yields, the absolute accuracy of the turbidity record is secondary in importance to obtaining reliable relationships between turbidity and SSC. Nevertheless, because turbidity is used by regulatory agencies to determine impacts on the beneficial uses of water, we now regularly check the continuous turbidity data with readings from portable Hach 2100P manual samples taken under low turbidity conditions. If necessary, the turbidity offset (calibration intercept) in the data logger program can be adjusted to bring the readings into agreement. We do not consider manual samples taken under high turbidity conditions to be reliable enough for such purposes.

Data processing. Data recorded at 10- or 15-minute intervals from a network of gaging stations is very difficult to manage without custom programs for plotting and processing the data. Processing programs are needed for interpolation, reconstruction, adjustment, and for adding quality codes to the data.

Routine processing starts with plotting the raw data and annotating the plots using field notes that might aid in the interpretation of the data. Such notes are invaluable for identifying problems and explaining anomalies. Some types of fouling can be readily identified on the plots with experience. However, fouling that occurs during storm events can often be identified only by plotting the turbidity against SSC from corresponding field samples or by comparing the turbidity with independent readings from a second sensor.

Once problems have been identified, the data must be corrected, omitted, or coded as suspect. In cases of ephemeral fouling, simple linear interpolation may often be satisfactory. Extended fouling is usually not correctable unless conditions are changing very slowly, such as late on the recession limb of a hydrograph or during an extended dry period. In such cases, the reconstructed data must be clearly coded as questionable. We do not recommend that raw turbidity data be released for any purpose before being carefully examined and corrected or quality-coded. Even with the proper caveats, provisional raw turbidity data is likely to be misinterpreted and misused.

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ESTIMATION OF SUSPENDED SOLIDS CONCENTRATIONS BASED ON ACOUSTIC BACKSCATTER INTENSITY: THEORETICAL BACKGROUND

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ABSTRACT

Widespread use of acoustic instruments to measure current velocity has led to interest in the technique of using acoustic sensors to estimate suspended solids concentration (SSC) from acoustic backscatter intensity (ABS). These measurements are non-intrusive, much less susceptible to biological fouling than are measurements from optical instruments, and provide time series of ABS (profile) for improved temporal resolution of SSC estimates. Successful estimates of SSC from ABS provides promise that this technique might be appropriate and useful for determining SSC from commercially available instruments such as acoustic Doppler current profilers (ADCPs). In spite of significant advantages to the method, users must be aware of important limitations to the technique

Introduction: The transport, deposition, and suspension of sediments in rivers, estuaries, and bays are of critical importance to understanding overall condition and health of these complex systems. Sediments carry nutrients and potentially toxic materials; transport of sediments is the mechanism to re-distribute these materials within the system. High concentration of suspended materials may limit light transmission and thus inhibit photosynthesis. In addition, deposition of suspended sediments in shipping channels requires periodic dredging to maintain navigable waterways. While knowledge of SSC is needed to begin to understand these processes, quantitative measurement of this highly variable property is difficult at best. Use of in-situ optical instruments such as optical backscatterance sensors and transmissometers has provided estimates of SSC, but they do not measure SSC directly and are subject to biological fouling in highly productive waters. Collection and analysis of water samples provides direct measurement SSC and is not subject to biological fouling; however, this procedure is extremely labor intensive and tends to under sample in most cases because of the variable nature of suspended material.

Acoustic sensors that are routinely used to measure time series of water velocity overcome some of these difficulties and hold promise as a means of quantitatively estimating SSC from ABS intensity, a by-product of velocity measurements. An additional advantage of acoustic techniques is that backscattered signal is range-gated to provide time series of data profiles rather than single point measurements. Initial studies utilizing the acoustic technique provided qualitative results, for example, Schott and Johns (1987), Flagg and Smith (1989), and Heywood et al (1991). Laboratory experiments designed to calibrate ABS to sand concentration were conducted by Thorne et al (1991) and Lohrmann and Huhta (1994). Hanes et al (1988) used a 3 mHz acoustic source to estimate suspended sand concentration near Prince Edward Island and Thevenot et al (1992) developed calibration parameters as part of a study to monitor dredged material near Tylers Beach, Virginia using Broadband-ADCPs (BB-ADCPs). Hamilton et al (1998) provided comparison of optical and acoustic methods in a study describing measurements of cohesive sediments using an acoustic suspended sediment monitor and Thevenot and Kraus (1993) compared optical and acoustic methods using a 2400 kHz BB-ADCP in the Chesapeake Estuary. This is only a partial list of research in the field; however, in general, previous studies have been qualitative in nature or limited to large (sand-size) particles. Some studies used non-commercial, specially designed acoustic sensors. Many required extensive laboratory calibrations or were used for short duration (hours). Others did not account for acoustic losses in the near field of the acoustic transducer. Recently, Byrne and Patino (2001), Land and Jones (2001) and Gartner and Cheng (2001) described techniques to estimate time series of SSC utilizing standard commercial ADCPs. Theoretical background and some limitations of the technique are described, however the present discussion deals only with use of acoustic sensors to estimate SSC. Potential for using multi-frequency acoustic sensors to estimate size distribution of suspended solids is beyond the scope of this discussion.

Acoustic Method: The method of estimating SSC from ABS is based on application of the sonar equation for sound scattering from small particles. In its simplified form for reverberation level, the sonar equation (Urick, 1975) contains terms for the ensonified volume, volume scattering strength (a function of particle shape, diameter, density, rigidity, compressibility, and acoustic wavelength), source level (intensity of emitted signal, known or measurable), and two-way transmission loss. The transmission loss is a function of range to ensonified volume, and absorption coefficient for the water; it contains terms for losses due to spreading and absorption. Attenuation due to sediment must also be accounted for if it is shown to be significant at ranges and levels encountered during a study. The absorption coefficient for water is a function of acoustic frequency, salinity, temperature, and pressure and can be found using equations from Schulkin and Marsh (1962). Spreading loss is different in near and far transducer fields. The transition between near and far transducer fields is a function of transducer radius and acoustic wavelength. The correction for spreading loss in the transducer near field can be calculated from the formula in Downing et al (1995). From a practical standpoint, it is not possible to measure all the characteristics of the suspended material and the acoustic source that are required to directly estimate SSC from ABS. The approach described here involves casting the sonar equation in an exponential or log form by relating the SSC to a relative backscatter utilizing calibration parameters and single particle theory following the technique of Thevenot et al (1992). In exponential form, the estimation equation is

$$SSC_{(estimated)}=10^{(A+B*RB)}. \qquad (1)$$

The exponent of Eq. 1 contains a term for the measured relative acoustic backscatter, *RB*, as well as terms for an intercept, *A* and slope, *B* that are determined by regression of concurrent ABS with known SSC on a semi-log plane in the form of $log(SSC_{measured})=A+B*RB$. The procedure to estimate a profile of SSC from a measured profile of ABS (say from ADCP) is a multi-step process that includes: 1) calculating transmission loss from spreading and absorption as a function of range and absorption coefficient including the near field transducer correction for spreading loss; 2) determining relative backscatter as a function of range by removing reference level (baseline), correcting for transmission loss and converting backscatter units to dB utilizing an (instrument dependent) scale factor; and 3) determining slope and intercept for a regression between logarithm of measured SSC and relative backscatter. Eq. 1 can then be used to estimate a profile of SSC.

Theoretical Limitations: There are two practical limitations to the method of predicting SSC from ABS. The first is a limitation common to any single frequency (optical or acoustical) instrument. Since single frequency instruments cannot differentiate between changes in concentration and changes in particle size distribution, a change in size distribution will be interpreted as a change in concentration unless independent particle size distribution measurements indicate need for additional calibrations. In addition, acoustic and optical methods respond differently to particle size with acoustic sensors more sensitive to large particles (proportional to volume) and optical sensors more sensitive to small particles (proportional to cross sectional area).

The second limitation is associated with the relation between instrument frequency and particle size distribution. The theoretical basis for acoustic analysis is Rayleigh (long wavelength) scattering model that is restricted to particles whose ratio of circumference to wavelength is less than unity. For a fixed frequency acoustic instrument, this condition restricts the maximum particle size for which the method is appropriate, beyond which estimates of SSC can be expected to have increasing errors. In addition, attenuation falls off significantly below circumference to wavelength ratios near 0.01-0.1 a situation that may create errors at small particle sizes. This limits the approach to a range of particle sizes beyond which estimates of SSC would be expected to display increasing errors in addition to errors from changes in particle size distribution. For a 1200 kHz acoustic source, particle diameters of 400, 40, and 4 μm correspond to circumference/wavelength ratios of 1.00, 0.10, and 0.01 respectively. Thus, the acoustic method is most appropriate for particle size distributions on the order of tens to hundreds of microns. Because of the inherent mismatch of frequency versus particle size, acoustic sensors are more appropriate for suspended material that is larger than that for which optical instruments are optimized. At very high frequencies (10-20 mHz) necessary for wavelengths to match un-aggregated clay particle sizes, sound attenuation is very high and acoustic range is unacceptably low for instruments designed primarily to measuring velocity profiles.

Summary: The technique of using ABS may provide reasonably accurate estimates of SSC under favorable circumstances. The method has some advantages over other methods but suffers from the same limitation as any single frequency sensor as far as being unable to differentiate between changes in size distribution and concentration. Although optical and acoustical instruments react differently to grain size, ABS measured by velocity sensors such as ADCPs provides SSC estimates concurrent with velocity measurements without the use of an additional sensor. It overcomes the problem of biological fouling, a major limitation of optical instruments. Another significant feature is that when utilizing acoustic measurements from ADCPs for estimates of SSC they are in the form of profiles rather than single point measurements. This method may be an extremely useful research tool if additional tests show that it can provide consistent and reasonably accurate results (within theoretical limitations), in spite of some minor changes in particle size distribution.

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TOTAL SUSPENDED SOLIDS DATA FOR USE IN SEDIMENT STUDIES

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ABSTRACT:

The U.S. Environmental Protection Agency identifies fluvial sediment as the single most widespread pollutant in the Nation's rivers and streams, affecting aquatic habitat, drinking water treatment processes, and recreational uses of rivers, lakes, and estuaries. A significant amount of suspended-sediment data has been produced using the total suspended solids (TSS) laboratory analysis method. An evaluation of data collected and analyzed by the U.S. Geological Survey and others has shown that the variation in TSS analytical results is considerably larger than that for traditional suspended-sediment concentration analyses (SSC) and that the TSS data show a negative bias when compared to SSC data. This presentation presents the results of a continuing investigation into the differences between TSS and SSC results. It explores possible relations between these differences and other hydrologic data collected at the same stations. A general equation was developed to relate TSS data to SSC data. However, this general equation is not applicable for data from individual stations. It also compares estimates of annual suspended-sediment loads that were made using regression equations developed from paired TSS and SSC samples with annual loads computed by the USGS using traditional techniques and SSC data. Load estimates were compared for 10 sites where sufficient TSS and SSC paired data were available to develop sediment-transport curves for the same time period for which daily suspended-sediment records were available. Results of these analyses indicated that as the time frame over which the estimates were made increases, the overall error associated with the estimates decreases. Using SSC data to compute loads tends to produce estimates with smaller errors than those computed from TSS data. Loads computed from TSS data tend to be negatively biased as compared to those computed from traditional techniques. There does not appear to be a simple way to examine SSC and TSS paired data sets to determine if the TSS data will give as good as or better estimate of the suspended-sediment load than the estimates obtained using the SSC data.

Differences Between the SSC and TSS Analytical Methods. The fundamental difference between SSC (ASTM, 1999) and TSS (APHA and others, 1995) analytical methods arises during the preparation of the sample for subsequent filtering, drying, and weighing. A TSS analysis generally entails withdrawal of an aliquot of the original sample for subsequent analysis, although as determined in a previous study, there may be a lack of consistency in methods used in the sample preparation phase of the TSS analyses (Gray, Glysson, and Conge, 2000). The SSC analytical method uses the entire water-sediment mixture to calculate SSC values.

Data: A total of 14,466 sample pairs analyzed using the SSC (USGS parameter code 80154) and TSS (USGS parameter code 00530) methods were retrieved from the electronic files of the USGS (U.S. Geological Survey, 2000a). Data were available from 48 States and Puerto Rico. Samples were collected sequentially in-stream using methods described in Edwards and Glysson (1999). Daily suspended-sediment records, obtained from the USGS Daily Suspended-Sediment Load database (U.S. Geological Survey, 2000b), were computed using the standard USGS methods described by Porterfield (1972) and normally have 200 to 300 samples per year available for the computation and are referred to hereafter as loads produced by "traditional techniques."

Findings:
1. An analysis of 14,466 paired SSC and TSS environmental samples from 48 states showed that the TSS tended to be smaller than SSC throughout the observed range of suspended-sediment concentrations encountered in this study. This is consistent with the assumption that most of the subsamples used to produce the TSS data were obtained by pipette, or by pouring from an open container. Subsampling by pipette or by pouring will tend to produce a sand-deficient subsample. (Glysson, Gray, and Conge, 2000)

2. No consistent relation between either the percent sand or percent difference between TSS and SSC, and water discharge or sediment concentration was identified for the stations used in this investigation. (Glysson, Gray, and Conge, 2000)

3. Although TSS and concentration of fines from SSC samples are generally in better agreement than TSS and SSC whole-sample concentrations, the degree of agreement can vary appreciably between stations (even stations with low sediment concentrations and low sand content.) (Glysson, Gray, and Conge, 2000)

4. The relation between SSC and TSS at a station will give a better estimate of the conversion factor needed to correct TSS data at that station than simply using the general equation of SSC = 126 + 1.0857(TSS)that was developed using the entire data set. Caution should be exercised before relating SSC and TSS using this general equation because of the potentially large errors involved. (Glysson, Gray, and Conge, 2000)

5. Using regression analysis in the estimation of suspended-sediment loads will have errors that can be substantial. The absolute value of errors in this study ranges from as large as 4000% for the estimation of a daily load to 2% for the estimation of the sum of the loads for the period of record. In all cases, the differences found between the actual suspended-sediment loads computed by the traditional methods used by the USGS and the estimated loads decreased as the time period over which the loads were estimated increased. (Glysson, Gray, Schwarz, 2001)

6. Using SSC data tends to produce load estimates with smaller errors than those for which TSS data were used. Six of the 10 sites included in the analysis had errors in the sum of the loads larger than 40% when the TSS data were used, compared to only one when the SSC data were used. No stations had the errors in the sum of loads using TSS data significantly smaller than those using SSC data. (Glysson, Gray, Schwarz, 2001)

7. There does not appear to be a simple, straightforward way to compare the SSC and TSS paired data sets to determine if the TSS data will give as good or better estimate of the suspended-sediment load. (Glysson, Gray, Schwarz, 2001)

Conclusions: The differences between TSS and SSC analyses of paired samples can be significant. If TSS and SSC paired samples exist or can be collected, it might be possible to develop a relation between SSC and TSS. It appears from the results of this study so far, that in order to attempt to adjust TSS data, one would have to have a significant number of paired data sets from the station of interest. Even then, this method may not be a guaranteed way to adjust the TSS data accurately. There appears to be no simple, straightforward way to adjust TSS data to estimate suspended-sediment concentrations if paired samples are not available. Additional work needs to be done before any definite procedure can be recommended to adjust TSS data to better estimate SSC values. Using SSC data tends to produce load estimates with smaller errors than those for which TSS data were used.

The TSS Method, which was originally designed for analyses of wastewater samples, has been showed to be fundamentally unreliable for the analysis of natural-water samples. In contrast, the SSC method produces relatively reliable results for samples of natural water, regardless of the amount or percentage of sand-size material in the samples. SSC and TSS data collected form natural water are not comparable and should not be used interchangeably. The accuracy and comparability of suspended solid-phase concentrations of the Nation's natural water would be greatly enhanced if all these data were produce by the SSC analytical method.

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THE NEED FOR SURROGATE TECHNOLOGIES TO MONITOR FLUVIAL-SEDIMENT TRANSPORT

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The need for reliable, nationally consistent fluvial sediment data in the U.S. arguably has never been greater since the U.S. Army's Captain Talcott first sampled the Mississippi River in 1838. In addition to the traditional uses for these data, which focused on the engineering aspects related to design and management of reservoirs and instream hydraulic structures, and on dredging, information needs over the last two decades have also included those related to the expanding fields of contaminated sediment management, dam decommissioning and removal, environmental quality, stream restoration, geomorphic classification and assessments, physical-biotic interactions, and legal requirements such as the U.S. Environmental Protection Agency's Total Maximum Daily Load (TMDL) Program.

Ironically, the dramatic rise in the Nation's sediment-data needs has occurred more or less concomitant with a general decline in the amount of sediment data collected by U.S. Geological Survey (USGS). After the end of World War II, the number of sites at which the USGS collected daily suspended-sediment data increased rapidly, peaking at 360 in 1982 (Glysson, 1989; Osterkamp and Parker, 1991). By 1998, the number of USGS-operated daily sediment stations had fallen by 65 percent to 125, with an average of 140 over the 5-year period ending in September 2001 (USGS, 2002). This substantial decrease in sediment monitoring is of particular concern in that the USGS bears primary responsibility for acquisition and management of the Nation's water data including suspended-sediment, bedload, and bottom-material data (Glysson and Gray, 1997). This paper examines some factors behind the decline in collection of new suspended-sediment data, and presents a vision and proposed first step toward reversing the general trend toward reduced Federal sediment-data acquisition.

Traditional Methods for Collecting Suspended-Sediment Data: The samplers, deployment techniques, and methods of sample processing and analysis used to produce the bulk of Federal sediment data have their roots in the Subcommittee on Sedimentation, a Federal cooperative effort that started in 1938, and its subordinate Federal Interagency Sedimentation Project (FISP) (Skinner, 1989; FISP, 2002). The FISP's calibrated depth- and point-integrating isokinetic samplers collect a water sample at a rate within ten percent of the flow velocity incident on the sampler nozzle. When deployed using the Equal-Discharge Increment or Equal-Width Increment Methods, these samplers provide representative samples for subsequent processing and (or) analysis (Edwards and Glysson, 1999). When processed and analyzed using standard methods (USGS, 1998, 1999; American Society for Testing and Materials, 1999), and served online from a nationally consistent database, the most reliable and consistent set of fluvial sediment data are made available to the widest audience.

The previously described equipment, deployment techniques, and analytical methods have been used to provide the bulk of USGS fluvial-sediment data collected since the 1940's (Turcios and Gray, 2001; Turcios and others, 2002). Although these data are widely considered the "best" available – the most accurate, reliable, and comparable – their cumulative accuracy is unquantified, and the manually intensive data-collection techniques are in some cases considered too expensive and, under some circumstances, potentially unsafe to collect. Continuous monitoring using sediment-surrogate technologies may provide a viable alternative to traditional equipment and techniques.

Accuracy: The accuracy (bias and variance) of suspended-sediment concentration and particle-size distribution data is dependent on a number of factors, including instream spatial and temporal variability; the computational time frame; the ability to representatively sample and quantify flows of interest; proper deployment of an appropriate sampler; use of reliable sample-processing and shipping procedures; and use of quality-assured analytical techniques by a certified, reliable laboratory to analyze samples collected in open-channel flows (USGS, 1998). Two key problems associated with traditionally computed daily sediment records are the need for interpolating between dozens or hundreds of sediment-concentration values to estimate concentration values for unit values (35,040 values per 365-day year for data computations at 15-minute intervals); and the need to estimate concentration values for periods lacking samples. Continuously measured surrogate technologies would provide the unit-value data that could be adjusted based on periodic calibrations to yield more reliable and consistent sediment-load data. Statistical methods could be applied to provide an estimate of the accuracy of those time-series data.

Cost: The cost to collect and manage USGS sediment data is also dependent on a number of factors. These include the gage location, site accessibility, safety requirements, the range in size distribution of suspended sediments, the variability in runoff at the site, and the human and mechanical resources required to collect and process the data. An informal poll of selected USGS offices in 2001 yielded a estimated range of about $20,000 to $65,000 gross funds to provide a year's worth of daily suspended-sediment discharge values. Although Osterkamp and others (1998) showed that a sediment monitoring network in the U.S. consisting of 120 daily sites and 2,000 periodic sites would exceed a cost-benefit ratio of unity forty-fold if the data produced by the program resulted in a 1-percent decrease in sediment-related damages, some consider perceived high sediment-data costs to be partly responsible for the decline in Federal data production. Use of appropriate sediment surrogate technologies at a gage would probably reduce the cost of producing sediment data by reducing the number of water-sediment sample analyses and site visits, in both cases from as many as hundreds to about one or two dozen annually. Other benefits would be reduction in time and effort because time-consuming interpolations and concentration estimates would no longer be a common part of the computational process.

Safety: Although equipment and techniques for collection of sediment and flow data are generally quite safe, site conditions may render safe collection of these data difficult or impossible. For example, sampling in poor lighting conditions, from a narrow bridge, and (or) in a debris-laden stream can be unsafe. There are conditions where sediment data cannot and should not be collected manually. Unfortunately, these conditions tend to occur at times where the sediment data would be most influential in a transport computation or managerial decision. Monitoring by sediment-surrogate technologies would automatically provide a continuous concentration time series under many of the circumstances considered unsafe for manual sampling.

In summary, although the traditional equipment and techniques used by the USGS nationwide to collect fluvial sediment data may seem ill-suited for many of the limitations and needs of the 21st century, no alternatives have been documented to work under the range of stream and transport conditions characteristic of the Nation's rivers.

A Vision for Future Federal Sediment-Data Production According to Osterkamp and others (1992; 1998) and Trimble and Crosson (2000), the Nation needs a permanent, based-funded, national sediment monitoring and research network for the traditional and emerging needs described previously, and to provide reliable values of sediment fluxes at an adequate number of properly distributed streamgages. The short-term benefits would include relevant and readily available data describing ambient sedimentary conditions and loads, and the requisite data to calibrate models for simulating fluvial sedimentary processes. The long-term benefits would include identification of trends in sedimentary conditions, and a more complete data set with which to calibrate and verify simulation models. Fundamental requirements for an effective national sediment monitoring and research program would include:

- **A CORE NETWORK OF SEDIMENT STATIONS** that is equipped to continuously monitor a basic set of flow, sediment, and ancillary characteristics based on a consistent set of protocols and equipment at perhaps hundreds of sites representing a broad range of drainage basins in terms of geography, areal extent, hydrology, and geomorphology. The focus of these sites would be measurement of fluvial-sediment yields. It would be most beneficial to collect these data at sites where other water-quality parameters are monitored.

- **A SUBSET OF THE SEDIMENT STATION NETWORK FOR SEDIMENT RESEARCH** at which testing on emerging sediment-surrogate technologies and new methodologies can take place at a minimum of additional expense. A major focus of this effort would be to identify technologies that provide a reliable sediment-concentration time series that can be used as the basis for computing daily suspended-sediment discharges. A secondary focus would be to identify surrogate technologies for measuring characteristics of bedload, bed material, and bed topography.

- **AN EQUIPMENT AND METHODS ANALYTICAL COMPONENT** that addresses development of equipment and techniques for collecting, processing, and laboratory analysis of sediment samples.

- **A DATA-SYNTHESIS RESEARCH COMPONENT** that focuses on identifying or developing more efficient methods of measuring and estimating selected fluvial sediment characteristics; developing a means to estimate the uncertainty associated in these measurements and estimates; and on performing syntheses on historical and new sediment and ancillary data to learn more about the sedimentary characteristics of our Nation's rivers.

- **A COMMON DATABASE** that can accept all types of instantaneous and time series sediment and ancillary data collected by approved protocols, including specific information on the instruments and methods used to acquire the data.

A First Step: Development and Verification of Sediment Surrogate Technologies for the 21'st Century

Traditional techniques for collecting and analyzing sediment data do not meet all of the above-stated requirements of a national sediment monitoring and research network. Before such a program can become operational, new cost-effective and safe approaches for continuous monitoring that include uncertainty analyses are needed.

An ideal suspended-sediment surrogate technology would automatically monitor and record a signal that varies as a direct function of suspended-sediment concentration and (or) particle-size distribution representative of the entire stream cross-section for any river in any flow regime with an acceptable and quantifiable accuracy. Although there is no evidence that such a technology is even on the drawing board, let alone verified and ready for deployment, the literature is rife with descriptions of emerging technologies for measuring selected characteristics of fluvial sediment (Wren, 2000; Gray and Schmidt, 1998). Considerable progress is being made to devise or improve upon available new technologies to measure selected characteristics of fluvial sediment. Instruments have been developed that operate on acoustic, differential density, pump, focused beam reflectance, laser diffraction, nuclear, optical backscatter, optical transmission, and spectral reflectance principles (Wren et al., 2000). Although some surrogate technologies show promise, none is commonly accepted or extensively used.

Formal adoption of any sediment-surrogate technology for use in large-scale sediment-monitoring programs by the Subcommittee on Sedimentation must be predicated on performance testing. Isokinetic samplers – primarily those developed by the Federal Interagency Sedimentation Project (FISP) and described by Edwards and Glysson (1999) – generally are considered the standard against which the performance of other types of samplers are compared. Ideally, a controlled setting such as a laboratory flume would provide flow and sedimentary conditions enabling direct assessments of the efficacy of the new technology. Even in that case, direct comparisons between an adequate amount of comparative data from the surrogate technology and isokinetic samplers collected for a sufficient time period over a broad range of flow and sedimentary conditions, would be needed to determine if any bias, or change in bias, would result from implementation of the new technology (Gray and Schmidt, 2001).

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BIOLOGICAL ASPECTS OF TURBIDITY AND OTHER OPTICAL PROPERTIES OF WATER

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ABSTRACT

Introduction: Water clarity and light penetration have significant effects on both ecology and recreational water use. Visual water clarity and light penetration are closely related, with both depending on the absorption and scattering of light. Suspended particles are the dominant influence on light penetration in most natural waters (Davies-Colley and Smith (2001), with the exception of highly colored waters where absorption can be more significant.

Light penetration is of great ecological significance because of its impact on photosynthesis. Visual clarity impacts the behavior of aquatic organisms that rely on sight to catch their prey, and also influences human perception of water quality.

Limnologists have long used the Secchi disk to measure water clarity. It is often argued that Secchi depth measurements are highly subjective, with the implication that they are not as reliable as other instrumental measurements. In a recent review, Davies-Colley and Smith (2001) assessed methods for measuring turbidity, suspended sediment, and water clarity, as measured with a Secchi disk, is a true scientific measurement that can be measured with better precision than either turbidity or suspended sediment concentrations.

History: Carlson (1995; 1997) performed extensive research on the origins and use of the Secchi disk. Sailors have long reported sighting of various objects as a means of determining water clarity. Based on reports of some of these earlier observations, Commander Cialdi, head of the papal navy, used disks of white clay and canvas stretched over circular iron frames to measure transparency in the Mediterranean Sea. He enlisted the help of Fr. Peitro Angelo Secchi, an astronomer and the scientific advisor to the Pope, to test the utility of the disks. Beginning on April 20, 1865, Secchi initiated a series of seven experiments over a six-week period using disks of various sizes and colors, on the sunny and shady sides of the ship, on bright and cloudy days, and at different times of the day. The result was the selection of an all-white disk that was very similar to the modern Secchi disk.

G.C. Whipple modified the white Secchi disk by adding alternating balck and white quadrants to improve contrast in 1899. Whipple also viewed the disk through a tube, the forerunner to today's viewscopes.

The first recorded Secchi disk reading was probably made in 1804, 8 years before Secchi was born, when someone on the U.S. Navy frigate President lowered a white china plate on a log line. That plate was observed at a depth of 45 m (148') off the southern Mediterranean coast of Spain. The first recorded Secchi depth measurements in

freshwater were recorded on August 28 and September 6, 1873. He lowered a white dinner plate, 9.5 " in diameter, into lake Tahoe and was able to see the plate at a depth of 33 m (108.27'). The deepest Secchi depth is 80 m, recorded on October 13, 1986, in the Weddell Sea near Antarctica.

Theory of Operation: The Secchi disk measures the depth of visibility in water. This depth depends on both absorption due to dissolved substances and scattering by suspended particles. While an all-white disk is still commonly used in oceanography, most limnoligists use a disk with alternating black and white quadrants. There is a scientific basis for this difference. The Secchi disk acts as a contrast instrument, disappearing when there is no longer any contrast between the disk and its background. A white disk would remain visible at the greatest depth when viewed against a completely black background. The background color in the deep ocean, as well as in deep lakes, would be black. In contrast, light can be reflected off the bottom in shallow lakes, or off suspended particles in turbid lakes. In these cases, a white disk disappears sooner than would be the case if the background was black. The black quadrants may help provide the standard black background.

The apparent difference in brightness between the disk and surrounding water is represented by the following equation, presented by Hutchinson (1957):

$(I_o d_1 d_2 r_d - I_u) / (I_u + I_u' + I_R)$,

where I_o = the light penetrating the surface, I_u = the light scattered upward from below the level of the disk, I_u' = the light scattered upward between the disk and the surface, I_R = the light reflected from the lake surface, d_1 = the loss in intensity of the light passing from the surface to the disk, d_2 = the loss in the intensity of light passing from the disk to the eye, and rd = the reflectivity of the disk. In general, the human eye can perceive a difference in intensity of the quantity defined by the above equation of not less than 1/133.

In those cases where light transmission depends only on absorption, only $d_1 d_2$ is decreased. If the loss in transmission if affected by scattering both I_u and I_u' increase. Because of these variables, correlations between Secchi depth and light penetration are limited. However, when a relatively homogeneous group of lakes is compared the can be a high correlation between Secchi depth and light transmission. For example, the Indiana Lake Enhancement Program requires measurement of both Secchi depth and the depth of 1% light penetration, as measured with a photometer. Jones (2002) reported the relationship:

1% light depth (m) = 1.73 x Secchi depth (m), with r2 = 0.52 (n = 681).

Scheffer (1998) also reported that the euphotic depth can be estimated as 1.7 times the Secchi depth. In contrast, measurements the Salton Sea, California, a highly saline body of water, found the 1% light depth = 4 x Secchi depth (Holdren, unpublished).

Application: The use of the Secchi disk to measure water clarity is an extremely valuable tool for limnologists. The Secchi disk is inexpensive, durable, easy to use, and produces readings that are directly related to key ecological variables and human perceptions of water quality. Applications and examples of readings with different styles of disks will be discussed.

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THE USE OF RATING (TRANSPORT) CURVES TO PREDICT SUSPENDED SEDIMENT CONCENTRATION: A MATTER OF TEMPORAL RESOLUTION

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ABSTRACT

In the absence of actual suspended sediment concentration (SSC) data, hydrologists have used rating (sediment transport) curves to estimate (predict) SSCs for flux calculations. Evaluations from a long-term, ongoing, daily sediment-measuring site on a large river, indicate that relatively accurate (<±20%) annual suspended sediment fluxes can be obtained from hydrologically based monthly sampling. For a 5-year period, similar results can be obtained from sampling once every 2 months. Over a 20-year period, errors of <1% can be achieved using a single rating curve based on data spanning the entire period. However, better annual estimates, within the 20-year period, can be obtained if individual annual rating curves are used.

INTRODUCTION

Since the 1970's, there has been growing interest in estimating the fluvial transport of suspended sediment. The reasons are numerous and diverse, and include such issues as contaminant transport, water-quality trends, reservoir sedimentation, channel and harbor silting, soil erosion and loss, as well as ecological and recreational impacts (Walling, 1977; Ferguson, 1986; de Vries and Klavers, 1994; Horowitz et al., 2001). The calculation of fluxes or loads requires both discharge and concentration data (e.g., de Vries and Klavers, 1994; Phillips, et al., 1999). Typically, continuous, or near-continuous discharge data are available from *in situ* devices such as a stage/discharge recorder. On the other hand, suspended sediment concentration (SSC) data typically result from manually collected individual samples taken at fixed temporal intervals; occasionally, the fixed interval samples are supplemented by event samples. More recently, continuous or near-continuous SSC data have been generated by employing automatic samplers, or by measuring applicable surrogates such as turbidity (e.g., Horowitz, 1995). These newer approaches for determining SSC require site-specific calibrations to produce cross-sectionally representative data. Further, whereas the requisite equipment (e.g., *in situ* turbidometers, automatic samplers) is fairly inexpensive to obtain, operational and maintenance costs are relatively high. Hence, currently, continuous or near-continuous SSC data are rare.

For more than sixty years, in the absence of actual continuous or near-continuous SSC data, hydrologists have used rating (sediment transport) curves to estimate (predict) SSCs for flux calculations. Although there are more than 20 methods for developing rating curves, the most common is a power function (regression) that relates SSC to water discharge, with the discharge measurement constituting the independent variable (e.g., Phillips, et al., 1999; Asselman, 2000). This requires the log-transformation of SSC and discharge data prior to the analysis. Comparisons of actual and predicted SSC, partially as a result of scatter about the regression line, as well as the conversion of results from log-space to arithmetic-space, indicate that rating curves can substantially underpredict actual concentrations (Walling and Webb, 1988; Asselman, 2000). To compensate, various method modifications have been applied; these include dividing the SSC/ discharge data into seasonal or hydrologic groupings, developing various correction factors, or using non-linear regression equations (Duan, 1983, Ferguson, 1986; Walling and Webb, 1988; de Vries and Klavers, 1994; Phillips, et al., 1999; Asselman, 2000).

In 1995-1996 the U.S. Geological Survey's (USGS) National Stream Quality Accounting Network (NASQAN) was revised from an occurrence and distribution-based network to a large-river flux-based water-quality monitoring network (Horowitz, et al., 2001). SSCs were required to calculate fluxes for sediment, as well as for various sediment-associated constituents (e.g., trace elements, nutrients). Due to resource constraints, the requisite SSCs/fluxes had to be determined from site-specific rating curves. Over the past 7 years, the effect of using the rating-curve approach, relative to such issues as sampling frequency, temporal resolution, and errors associated with flux estimates continue to be evaluated. Some of these evaluations are discussed herein.

METHODS

Within NASQAN, the Mississippi River at Thebes site is unique because it constitutes the only long-term, ongoing, daily sediment-measuring site in the network. As such, the data from this site are uniquely suited to evaluating such issues as sampling frequency, temporal resolution, and flux calculation/estimation errors. All calculations used in these evaluations are based on a 20-year data set covering water years (October-September) spanning 1981 to 2000.

All regression analyses were performed using Statview® 5.0 on a desktop computer. Linear and non-linear regression equations were calculated; comparison with actual data indicated that the predicted concentrations represented underestimates. These underestimates were substantially reduced by applying a 'smearing' correction (Duan, 1983).

RESULTS AND DISCUSSION

The first evaluation entailed an examination of the temporal resolution, and associated errors, of estimated suspended sediment fluxes at the Thebes site covering the first 5 years (1996-2000) of the revised NASQAN program. The actual flux for that 5-year period was 414 Mt (megatonnes), whereas the predicted flux, using daily values, was 404 Mt, a 3% underestimate. Despite this close agreement for the entire 5-year period, maximum errors in daily estimates of SSC ranged from -290% to +260%. The 5-year suspended sediment flux estimate using the approximately monthly NASQAN samples was 439 Mt, a 6% overestimate. The various errors associated with different levels of temporal resolution also were calculated for the same 5-year period; the errors tend to decline with increasing temporal resolution (table 1). This accrues because the rating-curve approach underestimates highs and overestimates lows. Hence, the longer the period of interest, the greater the chance for the over- and underestimates to balance each other out.

Table 1. Various levels of temporal resolution and their asociated errors for the Mississippi River at Thebes site for the five-year period 1996 - 2000.

Temporal Resolution	Maximum Underestimate Relative Percent	Maximum Overestimate Relative Percent	Average Absolute Error Relative Percent
Daily	-61	+65	27
Weekly	-60	+135	23
Monthly	-42	+35	18
Quarterly	-32	+28	13
Yearly	-13	+6	6

The effect of sampling frequency on the accuracy and associated errors of 5-year suspended sediment flux estimates also was investigated as part of the same evaluation. This entailed using the daily SSC values for the Thebes site and calculating a large number of rating curves to predict daily SSC values assuming different levels of sampling intensity. The sampling frequencies evaluated in this way corresponded to: (1) once a day; (2) once every other day; (3) once every 3 days; (4) once every 4 days; (5) once every 5 days (weekly); (6) once every 10 days, (7) once every 25 days (monthly); and once every 50 days (every other month). Not surprisingly the accuracy of the 5-year estimates decreased, and the size of the associated errors increased with decreasing sampling frequency (fig. 1a). Interestingly, there was little difference between sampling frequencies ranging from 1- to 5-days. On the other hand, estimation errors from sampling frequencies on the order of once every 2 months (once every 50 days) were little compromised, and tended to fall within a range of ±20%. As the calculations were based solely on calendar distributions, they probably represent the maximum error likely to occur with this level of sampling frequency (fig. 1a). If the same level of sampling (once every 50 days) were hydrologically distributed such as to encompass some 80 to 85% of the typical range of discharge, the associated estimation errors likely would be substantially less (e.g., Horowitz, 1995). The effect of sampling frequency on the accuracy and associated errors of annual suspended sediment

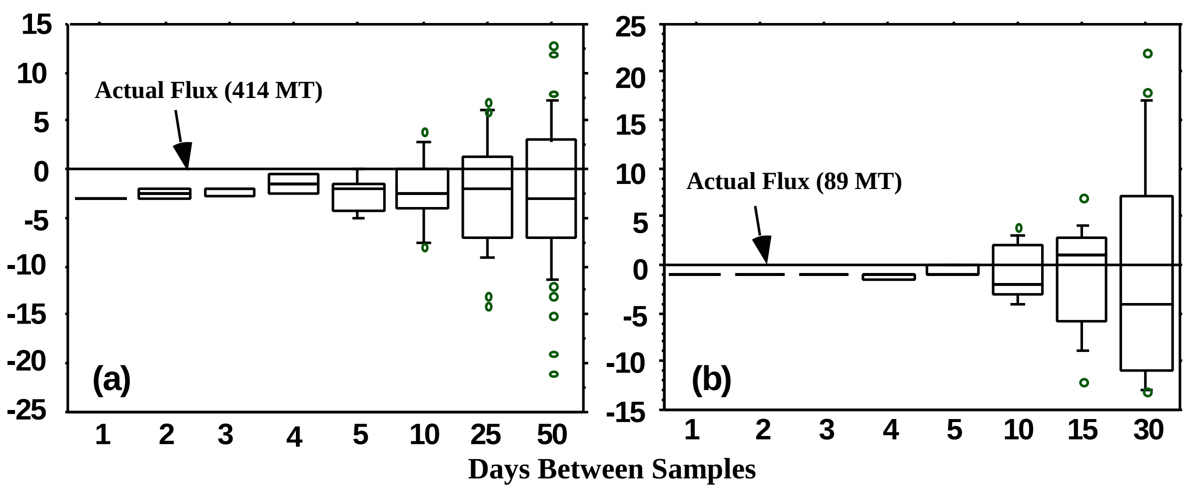

Fig. 1. Effect of sampling frequency on the errors associated with the estimation of suspended sediment fluxes over a 5-year (1996 - 2000 WY) period (a) and for a 1-year (1995 WY) period (b) for the Mississippi River at Thebes.

flux estimates also was investigated concurrently (fig. 1b). These evaluations covered high (1982), median (1995), and low (1989) flux years. The sampling frequencies evaluated in this way corresponded to: (1) once a day; (2) once every other day; (3) once every 3 days; (4) once every 4 days; (5) once every 5 days (weekly); (6) once every 10 days, (7) once every 15 days (fortnightly); and once every 30 days (every month). Note that as with the 5-year study, there is little difference between 1- and 5-day sampling. Further, even collecting a sample as infrequently as once a month only produced errors on the order of ±20%, regardless of the flow conditions (high, low, or median). The same caveats apply to the annual study as to the 5-year study, hence, hydrologically based sampling, as opposed to calendar-based sampling, is likely to produce substantially more accurate estimates.

The actual 20-year suspended sediment flux for the Thebes site for the period 1981 to 2000 was 1,200 Mt. A single rating curve, using the entire 20-year data set, yielded a similar estimate, representing an error of <1%. This is a fairly standard approach for generating site-specific rating curves where long-term data are available, and is based on the assumption that all the data from the site are part of the same statistical population. Note that the annual errors associated with this single rating-curve approach can be significant (fig.3a). However, when individual annual rating curves are calculated for the same 20-year period, it is apparent that the data are not part of the same statistical population. Some curves are linear whereas others are non-linear (both concave and convex). Interestingly, the sum of the annual fluxes for the 20-year period is still 1,200 Mt; however, the individual annual estimates are significantly closer to the actual fluxes (fig. 3b). Hence, although the estimate of total flux does not improve through the use of annual rating curves as opposed to a single rating curve, better annual estimates within the 20-year period can be obtained if individual calculations are used.

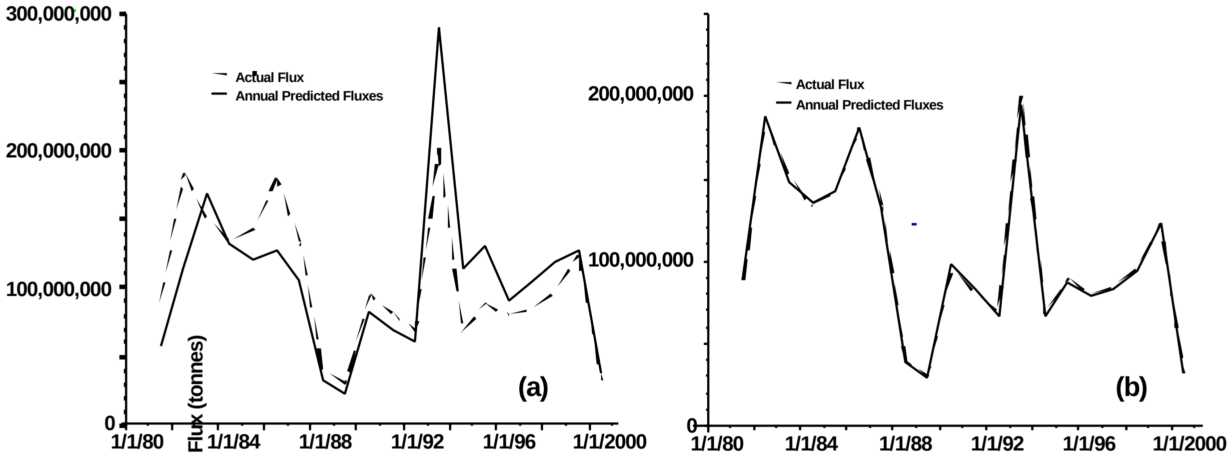

Fig.3. Comparisons of annual fluxes calculated by using a 20-year rating curve (a) and by calculating individual annual rating curves (b) for the Mississippi River at Thebes site for the period 1981 - 2000.

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ESTIMATION OF SUSPENDED SEDIMENT FLUX IN STREAMS USING CONTINUOUS TURBIDITY AND FLOW DATA COUPLED WITH LABORATORY CONCENTRATIONS

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The widening use of sediment surrogate measurements such as turbidity necessitates consideration of new methods for estimating sediment flux. Generally, existing methods can be simply be used in new ways. The effectiveness of a method varies according to the quality of the surrogate data and its relation to suspended sediment concentration (SSC). For this discussion, it is assumed that for each estimated period the surrogate data are accompanied by corresponding SSC data. If they are not, then the suspended sediment flux (i.e. yield or load) estimates are likely to be very poor. The accuracy of estimates is probably more dependent on sampling design and data quality than on the estimation method (Eads, 2002)

Sampling Design. Effective sampling designs focus on the important sources of variability. For example, if most of the variation in sediment flux occurs during summer thunderstorms, then sampling should target summer thunderstorms. In most streams the relation between turbidity and SSC varies significantly between events. Differences in turbidity for a given SSC can easily vary by a factor of 2 or 3. Therefore, numerous events must be sampled to properly represent the average relationship. And a relationship from one event will not serve well to estimate SSC in another.

A hypothetical sample was simulated from an intensively monitored storm event (Figures 1a-b) using the Turbidity Threshold Sampling (TTS) method (Lewis, 1996; Lewis and Eads, 2001), which obtains regression data (SSC vs. turbidity) covering the range of SSC in each episode of sediment transport. In practice, a data logger uses real-time turbidity data to govern the collection of pumped SSC samples. Regressions are later applied to the continuous turbidity data to obtain continuous SSC estimates. Depth-integrated samples are also collected for a subset of pumped samples so that SSC can be adjusted if necessary to reflect cross-sectional averages, but spatial variability of SSC in streams is generally small compared to temporal variability.

Data Quality. The importance of turbidity data quality cannot be emphasized enough (Eads and Lewis, 2002). Turbidity sensors with mechanical wipers can prevent fouling by detritus, and proper mounting of the sensor can reduce fouling from larger debris, but it is virtually impossible to collect perfect turbidity data. All data must be plotted and scrutinized carefully with reference to detailed field notes in order to properly identify, flag and correct problem areas. Some patterns of fouling are readily identifiable with experience, but others require comparison with SSC such as those from pumped TTS samples. Ephemeral fouling can usually be corrected by interpolation, but extended fouling is usually not correctable and can only be omitted or flagged with quality codes.

Flux Estimation. Custom computer algorithms are essential for sediment flux estimation, but the process cannot be entirely automated because many subjective decisions are required. Suitable models can vary between and within transport events. The choice of appropriate models for an event depends on the completeness and quality of the surrogate data and its relationship with SSC. If the turbidity sensor was fouled during a portion of an event, then the SSC may have to be estimated from its relationship with flow. When the sensor is fouled and the turbidity readings are fluctuating, TTS can trigger extra pumped samples. If both turbidity and flow are poorly related to SSC, there are often enough pumped samples to permit reliable estimation of SSC by linear time-interpolation (Figure 1b).

Regressions of SSC vs. turbidity (turbidity-SSC rating curves) are often quite linear with low variance. Therefore, when reliable turbidity data are accompanied by SSC samples, sediment flux can be estimated quite accurately (Figures 1c-d). Sometimes a quadratic or power model, or two linear models, are superior (Lewis, 1996), but in most cases, the variability and small sample sizes are inadequate to support a nonlinear model. A nonlinear model relating SSC to turbidity may improve variance estimation somewhat, but will not usually improve flux estimates (Lewis, 1996). Additionally, the preferred method of flux variance estimation with models for log-transformed SSC (Gilroy et al., 1990) is quite complex.

During periods when turbidity is of poor quality, relationships between flow and SSC may be needed (Figures 1e-f). As with turbidity-SSC rating curves, it is generally best to use only data collected during or immediately

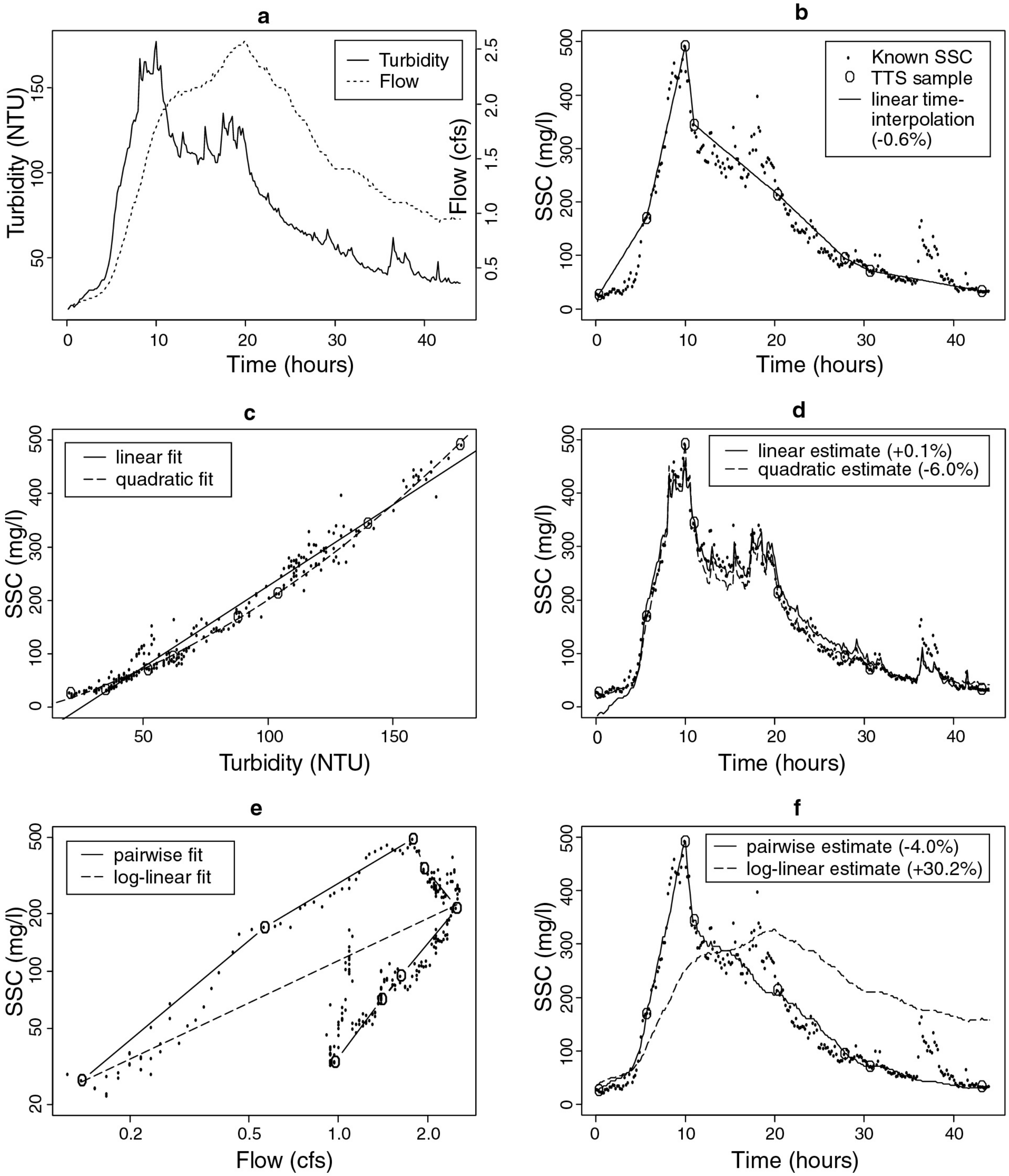

Figure 1. Estimation of suspended sediment concentration (SSC) and flux using 5 different methods for a storm event at Arfstein station, Caspar Creek, California. (a) Continuous turbidity and flow. (b) Measured SSC and 8 hypothetical samples obtained using Turbidity Threshold Sampling, showing linear interpolation of SSC as a function of time. (c) A linear and quadratic model of SSC vs. turbidity. (d) Estimated SSC from models shown in frame **c**. (e) Pairwise hysteresis fit and log-linear discharge-SSC rating curve. (f) Estimated SSC from models shown in frame **e**. Errors in estimated flux associated with each method are shown in parentheses in legends of frames **b, d,** and **f**.

surrounding the estimated event. When modeling SSC as a function of flow, it may be useful to employ a piecewise or pairwise model, in which each segment of the curve is applied only to the period of time between the sampling times of its endpoints (Figures 1e-f). Such a model can handle hysteresis, but produces inverted sedigraphs for negatively-sloping segments, in which flow peaks are modelled as SSC troughs. And if no smoothing is applied, pairwise models often include over-steepened segments that produce wild predictions. Discharge-SSC rating curves and piecewise hysteresis models are often more useful for representing segments of events than entire events.

Custom software could greatly simplify implementation of the above processes. A useful procedure would present the analyst with a series of choices as follows:

1. Select a time period to be estimated.
2. Select a set of SSC samples (default selection would be those from the selected time period) .
3. If needed, adjust SSC to the cross-section average using a user-supplied equation.
4. Select surrogate and constituent (SSC or adjusted SSC) variables.
   - For time-interpolation between concentrations, select only constituent variable.
5. View a scatter plot of the variables selected in step 4.
   - Omit erroneous points or add new points.
6. Choose an appropriate model (linear, power, polynomial, piecewise).
7. View a time series plot of estimated and measured concentrations for the period of estimation.
8. View statistics such as estimated total and variance, and sample size.
9. Save the results and repeat steps 1-8 for next time period.
10. Finally view complete time series plots and summary statistics.

As with TTS, the above procedure would be equally applicable to any water quality constituent for which a continuous surrogate measurement is available.

The TTS method was designed for, and has been applied to, storm event flux estimates. It can also be used for annual flux estimation, although it collects more samples than may be needed. The total flux can be estimated by summing individual event fluxes, or by applying a single model to all the data. The latter approach is easier to apply but less accurate, and requires surrogate data that are complete and correct. Intermediate approaches are also possible, for example using submodels for snowmelt and storm runoff, or early season and late season flux.

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THE CONTRIBUTION OF SUSPENDED ORGANIC SEDIMENTS TO TURBIDITY AND SEDIMENT FLUX

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ABSTRACT

Background

For over three decades, geologists, hydrologists and stream ecologists have shown significant interest in suspended load in running waters. Physical scientists have focused on turbidity, the development of sediment-rating curves and estimation of sediment yields, often as an indicator of changing land uses (Beschta, 1981). Stream ecologists, on the other hand, have focused on: 1) the role of suspended sediments in water quality degradation and its deleterious impacts on biological communities (e.g. Waters 1995); or 2) its beneficial roles in providing food resources for filter-feeding invertebrates and as the major pathway of organic matter transport and export, linking upstream and downstream reaches and affecting such ecosystem processes as nutrient spiraling (Minshall et al., 1983; Minshall et al., 1985; Wallace and Grubaugh, 1996). The focus of these interests has dictated the way in which sediment samples are examined. In many cases, the organics in suspended load samples are removed by ashing or chemical digestion. But physical scientists and stream ecologists concerned with the deleterious role of suspended sediments tend to discard data on the organic fraction (ash-free or carbon digested), while ecologists interested in its beneficial role discard information on the mineral fraction (ash or digestion residue). When data are reported on suspended load as derived from turbidity readings, it is seldom made clear whether reported values have been "corrected" for the organic fraction or whether, as is the usual case, both inorganic and organic components are combined. Nevertheless, a few studies have demonstrated the importance of suspended organic matter in sediment transport regimes. LaHusen (1994) reported the mean percentage of organic material causing stream turbidity was 64 percent of the total dry weight of suspended sediment. In coastal Oregon, coarse particulate organic matter (>0.2 mm) comprised 10 to 50 percent of the material transport along the stream bottom (Beschta, 1981). These studies suggest a closer look at the role of suspended organic sediment is warranted.

Problem: Failure to distinguish between organic and inorganic components of the suspended load or to consider the full suite of information present in suspended sediment samples has hindered full understanding of sediment dynamics as it affects stream health and reflects watershed condition. For example, because organic sediments remain in suspension longer than do similarly-sized inorganic components, and therefore contribute more to turbidity, they may have a greater effect on light reduction. An increased proportion of suspended organic sediments would thus be expected to decrease primary production and lead to a loss of invertebrate scrapers

that feed on periphyton. At the same time, an increased proportion of organic suspended sediments in the appropriate size range would benefit filter-feeding invertebrates (filtering collectors). Both scrapers and filtering collectors are important components in the diets of salmonids and other drift-feeding fishes, and the net effect of organic:inorganic ratios on prey availability for fish is not known. Apart from indirect effects on fish through their food base, the effect of relative percentages of organic and inorganic components on light attenuation would also directly impact fish through loss of visual capability, leading to reduced feeding efficiency, feeding rate, and depressed growth (e.g. Berg, 1982; Wilzbach et al., 1986).

The particle size distribution of the organic suspended load is another important attribute that has not often been considered in previous studies. The particle size distribution and qualitative nature (e.g. microbial activity, relative amounts of plant, animal, and detrital material) of the constituents of the organic fraction of the suspended load predict the response of invertebrate filtering collectors.

Thus, the separation of suspended load material into inorganic and organic fractions, and particle size distribution of both fractions together with qualitative aspects of the organic fraction will provide far greater resolution of physical and biological conditions of watersheds than is currently being provided.

Objectives and Methods

One objective of this research is to establish the contribution of size-specific, inorganic and organic components to turbidity and sediment flux. The role of these components in influencing stream health, as reflected in the efficiency of growth of juvenile salmonids and their invertebrate food base, will also be assessed. The study involves sampling on both within-storm and seasonal time scales at a range of stream sites in northern California which differ in land use, watershed area, riparian cover, and salmonid use, and for which records of continuous turbidity values and suspended load are available. Suspended sediment, turbidity, and water discharge are sampled on rising and falling limbs of flood hydrographs throughout the year, and sediment concentration, particle size and organic content are analyzed by standard laboratory techniques (Guy, 1969). In addition, sediment and biological sampling at each site are made at each site throughout the year so as to capture a full range of discharge and turbidity conditions. Parameters that are assessed include the following:

Field:

1) physical parameters: turbidity, fluorescence (as an index of chlorophyll-a)
2) microbial respiration, indexed by measurement of dissolved oxygen in the field
3) abundance of macroinvertebrate functional groups: collected with D-frame net samples of cobble and large wood
4) foraging efficiency and condition of juvenile salmonids: foraging efficiency is estimated in field feeding trials, using the experimental feeding apparatus and procedures described in Wilzbach et al (1986). Effects of suspended sediments on fish foraging has previously been evaluated only in laboratory experiments (Waters, 1995). Condition is estimated from length, mass, and age determinations of individuals collected from minnow traps.

Lab:

5) total particulate (suspended load) mass

6) total inorganic fraction
7) total organic fraction
8) inorganic particle size fractions
9) organic particle size fractions

Preliminary Results

Although this project is still in its initial phase, preliminary results suggest that the role of organic sediment will be important. For example, in an early season flood, the fraction of suspended load composed of organics ranged from only 3% at the peak to 60 to 80% on the rising and falling limbs of the hydrograph. Although the total mass of suspended organic transport was greater on the rising limb, the contribution of organics, as a percentage of the total suspended sediment, was greater on the falling limb. These results imply that the turbid 'tail' of the hydrograph, important biologically, is greatly influenced by the suspended organic sediment (which constitute 20 to 60% of the sediment load on the falling limb.)

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TESTING LASER-BASED SENSORS FOR CONTINUOUS, IN-SITU MONITORING OF SUSPENDED SEDIMENT IN THE COLORADO RIVER, GRAND CANYON, ARIZONA

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ABSTRACT

Overview: The Grand Canyon Monitoring and Research Center (GCMRC) was established in 1995, following completion of a major environmental impact statement on operations of Glen Canyon Dam (DOI, 1995). The GCMRC supports the Glen Canyon Dam adaptive management program (AMP) by providing research and monitoring information on a variety of resources associated with the Colorado River ecosystem within Glen Canyon National Recreation Area and Grand Canyon National Park. Resources of special concern include native fishes, cultural and recreational resources, as well as fine-grained sediment deposits located along channel margins of the river. Owing to the ecosystem's supply-limited sediment- transport behavior (Rubin et al., 1998; Topping et al., 2000a), intensive monitoring of fine sediment below Glen Canyon Dam is an AMP requirement for environmental management of the Colorado River ecosystem. One objective of the GCMRC's monitoring program is measurement of the ecosystem's monthly sand mass balance between influx from tributaries and efflux downstream in the main channel.

Daily or near-daily measurement of suspended-sand concentration and grain-size using standard suspended-sediment sampling methods is currently required to estimate monthly sand flux between the dam and upper Lake Mead. The current program is logistically complicated, costly and provides limited spatial and temporal resolution. In-situ, laser-based sensors are being investigated as one alternative method for measuring sand export to Lake Mead.

Results of 2001 LISST Testing: Initial point data collected at a fixed-depth, near-shore site were obtained by averaging 16 measurements at 2-minute intervals during a 24-hour deployment starting at 16:00 on July 19, 2001. The data were collected using a LISST-100 "Type-B" sensor (Laser In-Situ Scattering and Transmissometry). The Type-B is a laser-diffraction based sensor designed to detect suspended particles over a size range of 1.3-250 microns. The LISST can also determine suspended concentrations over a variable range, depending on grain size and the instrument's adjustable sample-path length. The standard sample path of the LISST-100 is a cylindrical volume with a diameter of 6 mm and a length of 50 mm. Additional description of this technology is reported by Agrawal and Pottsmith (2001). The LISST-100 used during the July test was previously evaluated under laboratory and field conditions and its performance was reported by Gartner et al. (2001). The 720 LISST point measurements collected at the Grand Canyon gage in July 2001, compare very well with cross-sectionally integrated suspended-sand and silt and clay data obtained from 13 samples collected at a cableway near the test site using a D-77 bag sampler. During the test, fluctuating releases from Glen Canyon Dam ranged from about 9,000 to 17,000 cubic feet per second; a typical diurnal summer pattern of discharge related to hydropower generation at the dam. In addition to accurately tracking the sand concentration, the LISST-100 also recorded the physically expected increase in sand-concentration variance with increasing flow, with peak values ranging up to 150 mg/l (Figure 1). As predicted, concentrations of silt and clay obtained by the LISST were much less variable and ranged from about 50 to 100 mg/l (Figure 2). It is worth noting that the highest concentrations of fines occurred during the daily minimum discharge, which at this location occurs at night when conventional sampling does not occur. The LISST also provided median grain size data for sand that closely matched sand sizes obtained using the D-77 sampler (Figure 3).

A second field test was implemented from September 2001 to February 2002, to explore performance characteristics of three different LISST instruments during longer, continuous deployments. Our preliminary results from the fall through winter 2002 testing, indicate that in-situ, laser-based sensors can provide continuous data with appropriate maintenance, albeit under a limited range of grain-sizes and concentrations.

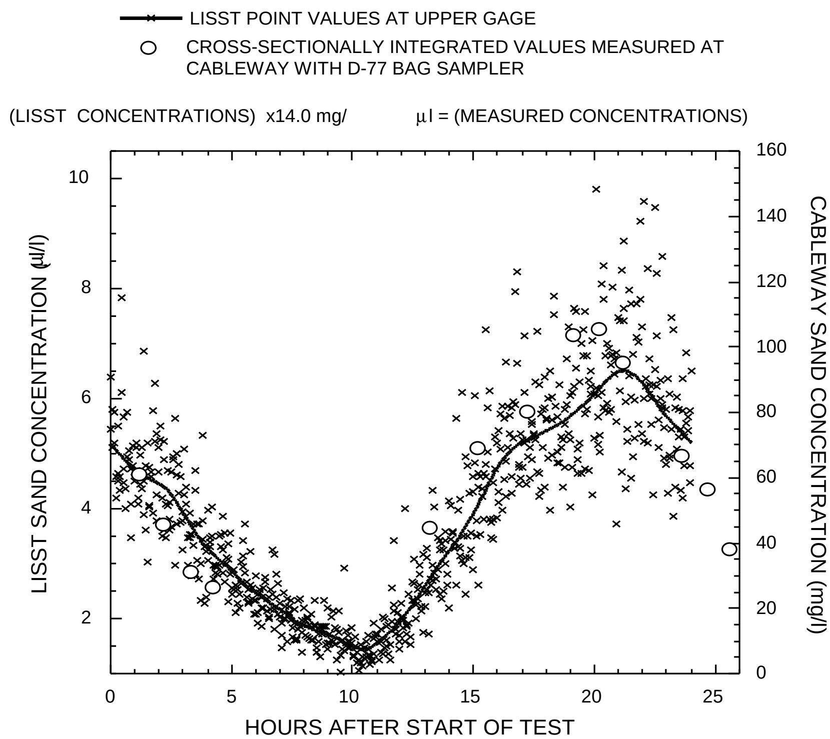

Figure 1. Comparison of sand concentrations measured at Grand Canyon using LISST-100 and the D-77 bag sampler.

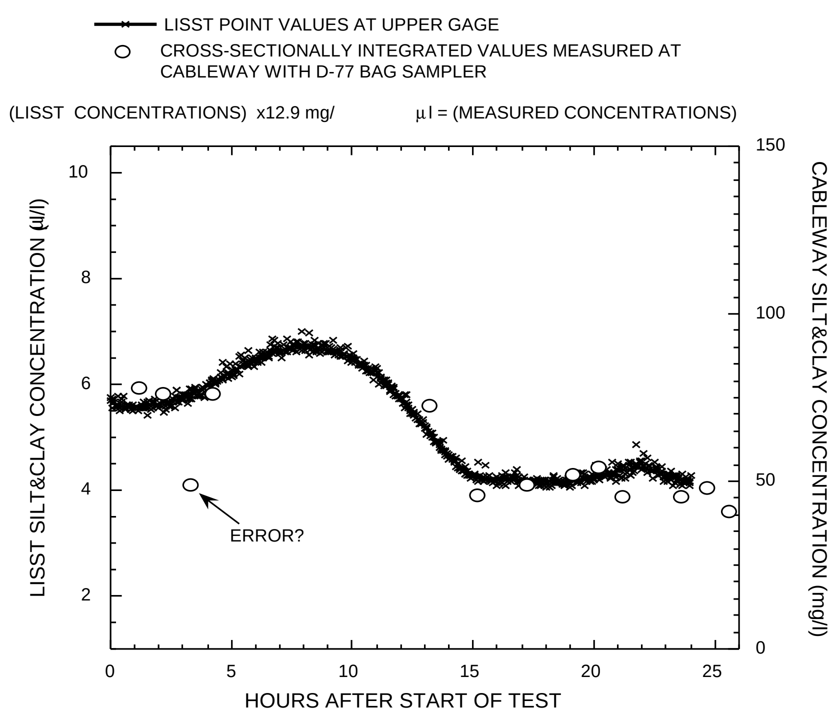

Figure 2. Comparison of fines measured at Grand Canyon using LISST-100 and the D-77 bag sampler.

<u>Monitoring Sediment Supply Conditions Using LISST and *Beta*</u>: Our previous work has shown that suspended-sediment concentration and grain-size data can be used to back-calculate grain size of sediment on the bed upstream (Rubin and Topping, 2001). The *beta* value, derived by the above method, is a surrogate for how enriched a river segment is in fine sediment, and thus provides an indirect, reach-integrated measure of a river's sediment mass balance (in non-armored conditions). The approach can also be applied to other sediment transport environments. Preliminary results suggest that LISST data will be suitable for calculating *beta* at higher spatial and temporal resolutions than those that are presently obtained using conventional suspended-sediment sampling methods.

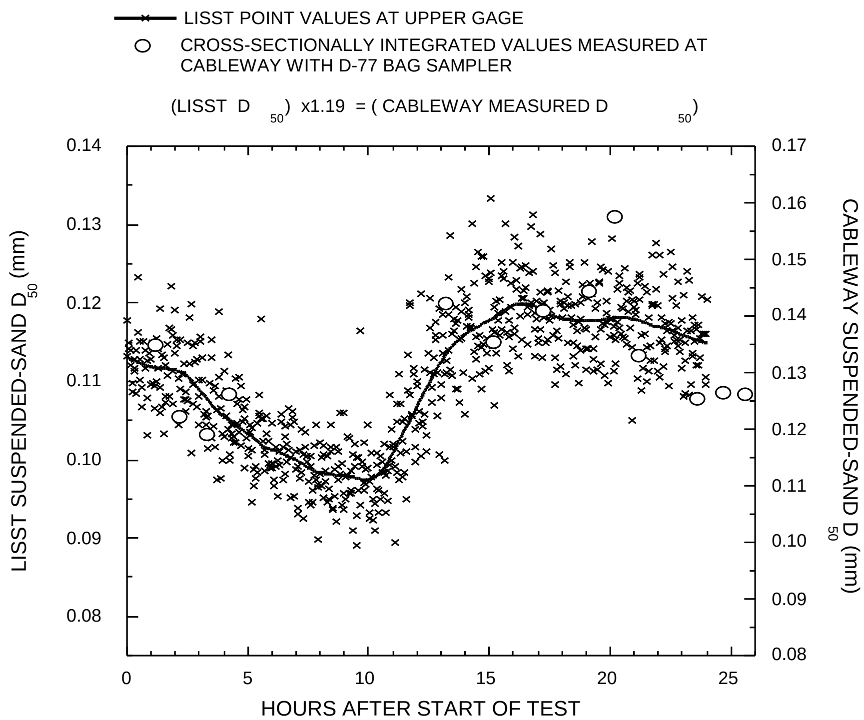

Figure 3. Comparison of median grain size of sand measured at Grand Canyon using LISST-100 and the D-77 bag sampler.

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TURBIDITY CALIBRATION STANDARDS EVALUATED FROM A DIFFERENT PERSPECTIVE

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ABSTRACT

History: Formazin was established as the first calibration standard for turbidimeters in the 1950's. Machine performance and Environmental Protection Agency (EPA) approval for turbidimeters was structured around formazin as the calibration standard. The EPA method 180.1 also outlined design parameters for turbidimeters used for testing surface source drinking water. The design parameters include a white light source and photodetector(s) positioned at 90° to the light source. The nephlometric design was to optimize the detection of sub-micron particulate. Refer to Brumberger et al, Light Scattering is a Function of Light Wave Length and Particle Size. That is, the characteristics of a given particle depend on its refractive index, shape, and size. Sub-micron particles scatter short wavelengths light (white light) at optimally 90°.

Current EPA Approved Standards: Today the scenario is unchanged except for additional EPA approved calibration standards. Besides "scratch" formazin, there is formazin concentrate (4000 NTU – Nephelometric Turbidity Units), stabilized formazin and submicron polymer suspensions.

The polymer suspensions are unique among the approved standards in several ways:

- non-toxic
- ready to use
- accurate +/- 1% of stated value lot to lot
- submicron in particle size distribution
- size, shape, and particle size distribution is always the same, regardless of lot.

It has been argued that since real world water samples have a wide distribution of particle shapes and sizes; the perfect turbidity standard should be of the same matrix. Perhaps true if the filtered final water still consisted of that composition, however, this is not the case. The large particles have been removed. Remember that turbidity reporting is done on finished water.

Particle Size / Light Scatter of Approved Standards: See Figure 1 of the three particle sizes. Figure 1(A) most closely resembles the remaining particulate in finished treated water. The size 1/10th the wavelength of white light; less than 60nm = 0.06µ. White light wavelength is 400 to 600 nm = 0.4 to 0.6microns. Again to emphasize the fact, the EPA protocol of nephelometric turbidimeter design optimizes detection of submicron particulate that scatters light in a 90° direction. Formazin is represented by Figure 1(C), 6000nm = 6.0µ. Formazin is outside the box; too large in size by several factors to equate to the particles that are analyzed. Does Formazin represent real world samples?

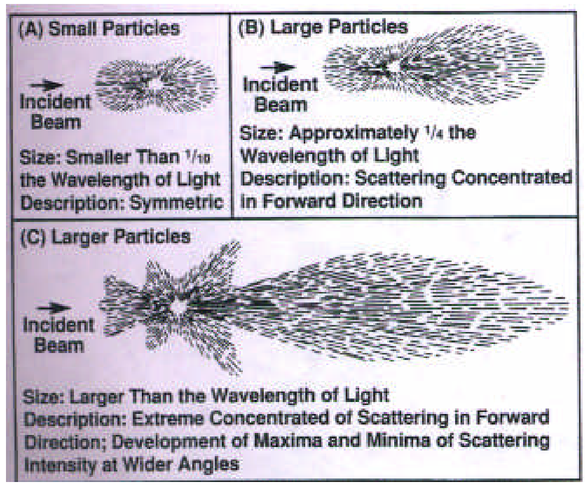

Figure 1

Formazin can be reproduced +/- 1% batch to batch. This is true under ideal conditions; which involves quality chemicals, precise volumetric glassware, ultra pure water and excellent laboratory technique. The formulation process is tedious and timely. The final diluted standards are time sensitive and it is commonly recommended not to prepare standards below 2.0 NTU. The EPA requires turbidity values not to exceed 0.3 NTU for surface source drinking water.

Turbidimeter Design Versus Particle Size of Standards: To demonstrate the relationship of machine design on formazin and the polymer calibration standards, three different lots of stock formazin data (1,2) and instrument specific polymer standards were tested in four different turbidimeters. The difference in this analysis is that the machines are calibrated with both types of calibration standards instead of just formazin and compared against each other.

Each machine employs a different optical and photo detector design. Analyzing the test results demonstrates several key points.

The different formazin lots do not stay within the 1% variance that is claimed by the manufacturer. The importance of the variance relates to the premise that it is reproducible by any end user.

Evaluating the data sheets for the HF Micro 100 and the McVan 160 probe, the worst case variance is 6.8% per NTU value. Discarding the outliers (2) the average variance is 1.56%. The machines do not change into ratio mode above 40.0 NTU. The polymer calibration standards are instrument specific due to the wavelength of the light sources, which are extremely different; HF 400-600nm and McVan 870 nm. The light source wavelength for the McVan is almost twice that of the HF. The impact of the difference is realized in what the two machines see. Imagine two wire mesh screens; one sized 0.4µ and the other size at 0.82µ, which one is going to trap smaller particles? Remember the EPA turbidimeter design criteria for filtered drinking water wavelength? The white light HF machine with its shorter wavelength, 400-600nm, will strike more small particles than the McVan machine. Visualize ping-pong balls verses basketballs.

The two Hach machines data sheets are the most complex to decipher. First, only the Hach 2100 AN instrument specific polymer calibration standards were used in the testing of both machines. At the 20 NTU reading, overall variance is 1.45 %. At the 200 NTU value, variance is 3.7%. At the 1000 NTU value, the variance is 10.43%. Lastly, the variance is 4.38% at the 4000 NTU calibration point. The percent error is large for both machines at the 1000 NTU and 4000 NTU polymer standard, why?

One, the polymer standards are specific for the 2100AN machine. Two, the machines are in the ratio mode at the 200 NTU, 1000 NTU, and 4000 NTU calibration points. Thus, multiple detectors at different angles other than 90° are being used, and transmitted light is also measured. These additional detectors are not seeing as much of the polymer suspension as with the 90° photo detector. Three, the ISO machine uses an infrared light source, 860 nm, as opposed to a white light source, 400-600 nm. Four, when calibrating the AN machine with the polymer suspension, the formazin standards read high in the ratio mode. The additional detectors are seeing the formazin therefore, inflating their turbidity readings. Also, more polymer suspension is needed to read matching formazin values at 200 NTU, 1000 NTU, and 4000 NTU. This is demonstrated in the ISO machine where the polymer suspension standards are not instrument specific. Once the ISO machine is calibrated with the non-instrument specific standards the calibration points are undervalued. This is shown by low formazin readings.

Is that a flaw in the polymer standard? No, because in the ratio mode the machines are "tuned" to measure large particles and to compensate for color. Neither of which is a parameter in the analysis of finished drinking water.

Polymer "Generic" Standards: The last test results demonstrate the variance of the generic EPA formulated polymer calibration standard in six different design parameter machines. The term generic is defined as the standard to be used to calibrate any turbidimeter that meets the EPA design parameters.

A criticism of the polymer calibration standards is that the turbidity values are established by comparing point to point against formazin, down to 0.1 NTU. Discard the outliers and factor this into the variance from 0.1 to 1000 NTU, then deduct 5% for the expected accuracy of formazin. The compared deviation is 3.37%!

Obviously, machine design can make radical differences in readings but they are outside of the EPA design parameters. Reverse the standard comparison. Let the polymer calibration standards be the gauge. Consider the benefits:

A. The polymer concentrate is formulated in batches that could be a 10 - 20 year supply. Batch to batch particle size variance +/- .001%.
B. Retention samples that could last indefinitely.

Defining New Turbidity Units: Realizing that turbidimeter design relates to its performance it is appropriate to define new application specific turbidity units.

Independent Study: Syracuse University under the sponsorship of AwwaRF conducted a one-year study of the performance of the calibration standards and turbidimeters. All of the EPA approved calibration standards were evaluated. On Page 76 of the study it states "the calibration method does not seem to have a significant effect on the agreement or lack of agreement between instrument-modes."

In summation

- The polymer calibration standards being instrument specific reveals a deviance of machine design as opposed to a shortcoming of the standard.
- Factoring in the machine design variance regardless of application, the generic polymer calibration standards on average are well within the tolerance of formazin (+/-5%).
- The final consideration is safety.

Use of Acoustic Instruments for Estimating Total Suspended Solids Concentrations in Streams -- The South Florida Experience

Eduardo Patino and Michael J. Byrne, U. S. Geological Survey, Ft. Myers, FL.

An acoustic velocity meter (AVM) and an acoustic Doppler velocity meter (ADVM) were used in a study to estimate total suspended solids (TSS) concentrations in southern Florida streams. The AVM system provides information on automatic gain control (AGC), an index of the acoustic signal strength recorded by the instrument as the acoustic pulse travels across a stream. The ADVM system provides information on acoustic backscatter strength (ABS), an index of the strength of return acoustic signals recorded by the instrument. Both AGC and ABS values increase with corresponding increases in the concentration of suspended material.

The study was conducted at two sites in southern Florida (fig. 1). An AVM was installed in 1993 in L-4 Canal (below structure G-88), a narrow manmade channel in northwestern Broward used to drain excess runoff from agricultural fields. Water velocities in this freshwater canal, which is about 40 feet wide and averages between 7 and 8 feet in depth, range from –0.5 to 2.5 feet per second. An ADVM system was installed in 1997 in North Fork stream (in Veterans Park), a tidal channel that discharges into the St. Lucie River Estuary along the southeastern coast of Florida. Water velocities in this tidal stream, which is about 280 feet wide and averages about 8 feet in depth, range from about –1.5 to 1.5 feet per second; salinity varies from fresh to brackish (0.2 to about 15 milligrams per liter). In addition to the acoustic instruments, water-quality sensors were installed at both sites to record specific conductance (or salinity) and temperature data. These data were used to monitor the potential effects that density changes could have on the AGC/ABS to TSS relations.

Depth-integrated samples for TSS analysis were collected at the L-4 Canal site using a DH-59 sampler and the equal discharge increment (EDI) methodology. Samples at the North Fork site were collected using a point sampler at the depth of the ADVM system and about 9 feet away from the transducer faces (near the start of the sampling volume). Samples for determining TSS and volatile suspended solids (VSS) concentrations were analyzed at the U.S. Geological Survey Laboratory in Ocala, Florida.

TSS concentrations ranged from 22 to 1,058 milligrams per liter at the L-4 Canal site and from 3 to 25 milligrams per liter at the North Fork site. The organic content of samples used in the analysis varied from 30 to 93 percent at the L-4 Canal site and from about 50 to 75 percent at the North Fork site. No sand splits or particle-size distribution analyses were performed for samples at either site.

Regression analysis techniques were used to develop empirical and site-specific relations between AGC and ABS to TSS concentration at the L-4 Canal and North Fork sites. The general form of the equation used to determine the AGC/ABS to TSS concentration relation at the study sites is:

$$TSS = 10^{\{A * [a + b * \log(salinity) + c * \log(temperature)] + d * \log(velocity) + e\}} \tag{1}$$

where A represents AGC or ABS; a, b, c, and d are regression coefficients; and e is the intercept. The relations obtained using site-specific forms of equation 1 produced good correlation as shown in figures 2 and 3. Correlation coefficients of 0.91 and 0.87 were obtained at the L-4 Canal and North Fork sites, respectively. The results suggest that this technique is feasible for estimating TSS concentrations in streams using information from acoustic instruments.

Figure 1. Location of the L-4 Canal and North Fork monitoring sites.

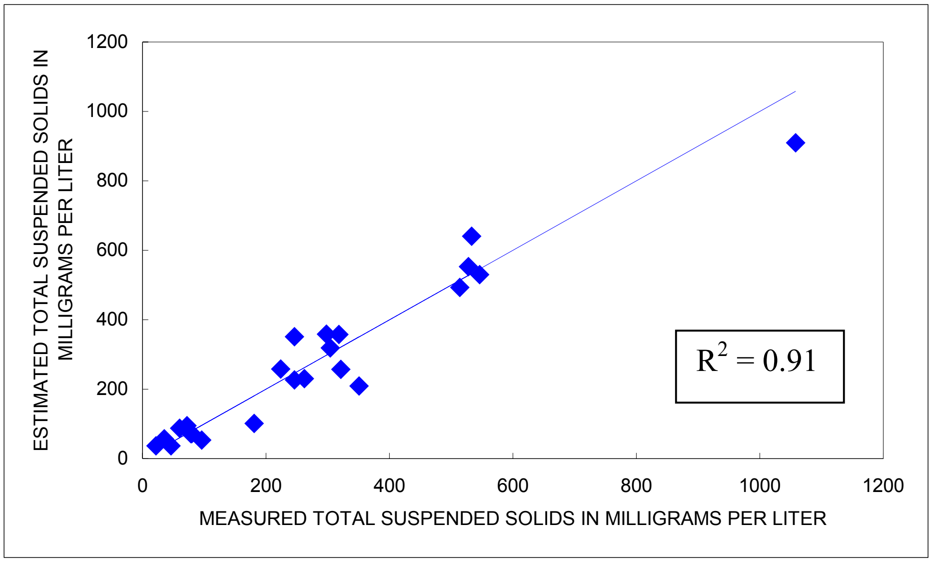

Figure 2. Estimated total suspended solids (TSS) concentrations for the L-4 Canal site. Relation developed using $TSS = 10^{\{AGC * [0.1968 - 0.017 * \log(temperature)] + 0.7096 * \log(velocity) - 4.4561\}}$

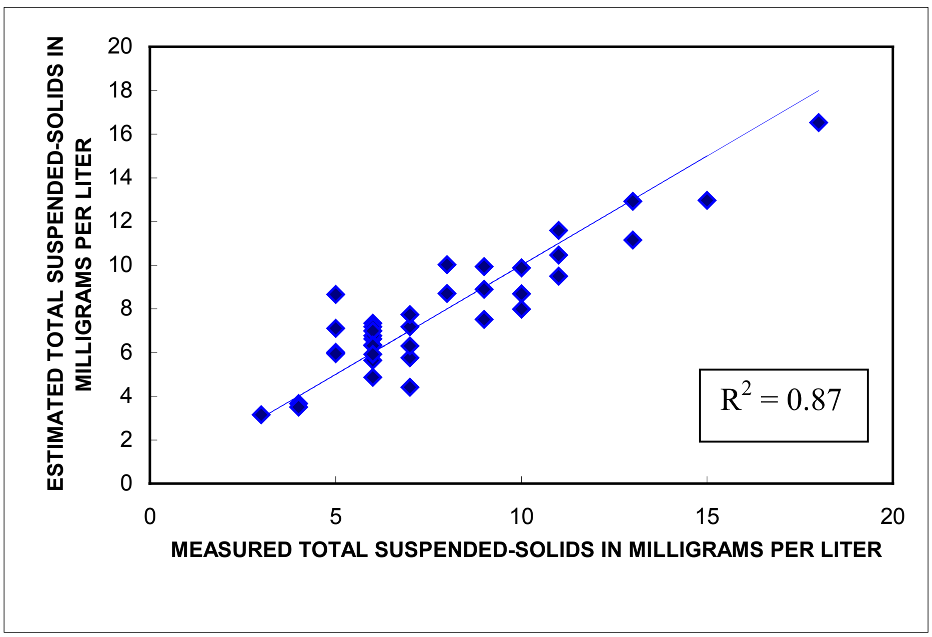

Figure 3. Estimated total suspended solids (TSS) concentrations for the North Fork site. Relation developed using $TSS = 10^{\{ABS * [0.07462 + 0.00084 * \log(salinity) - 0.02957 * \log(temperature)] - 1.4615\}}$

TURBIDITY STUDIES AT THE NATIONAL WATER QUALITY LABORATORY

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EXTENDED ABSTRACT

Turbidity interference caused by color: The U.S. Geological Survey National Water Quality Laboratory (NWQL) observed different results for sample turbidity measured on 1st and 2nd generation instruments manufactured by the same vendor. The NWQL purchased a Hach * 2100AN nephelometer to replace the Hach 2100A instrument that had been used, and the two instruments were compared in the laboratory. The turbidity of formazin standards and purchased references was comparable for both instruments (U.S. Geological Survey, 2000), as shown in Figure 1. Different turbidity results, however, were generated for environmental samples with turbidity greater than about 25 nephelometric turbidity units (NTU) (Figure 2). The cause of these differences is discussed in the following paragraphs.

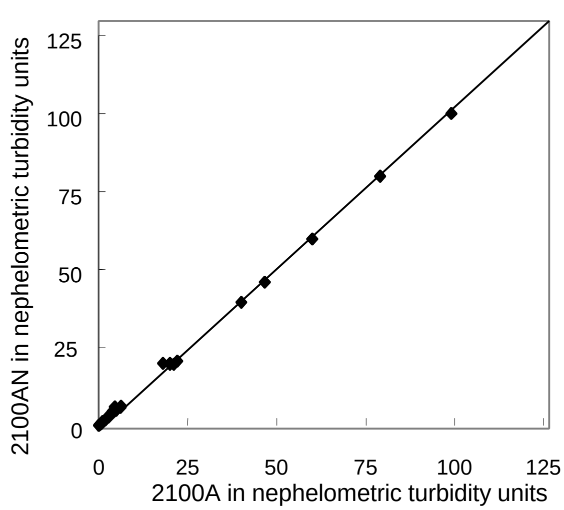

Figure 1. - Reference check

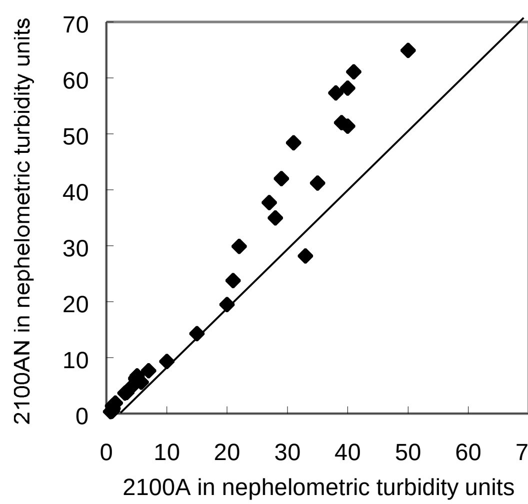

Figure 2. - Hach 2100A < 50

The major difference between the two instruments is the number of light detectors. The Hach 2100AN has several detectors, with the main detector at 90 degrees to the incident light. The secondary detectors measure light that is transmitted, forward scattered, and backward scattered. The signals from these detectors are combined (ratioed) mathematically to calculate the turbidity. The older instrument (2100A) had only one detector at 90 degrees to the incident light. Both instruments satisfy Standard Methods (American Public Health Association and others, 1998)

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* The use of trade, product, or firm names in this report is for descriptive purposes only and does not imply endorsement by the U.S. Government.

and U.S Environmental Protection Agency (1999) design criteria by using a tungsten filament lamp source and measuring scattered light at 90 degrees to the incident light.

The tungsten-filament lamp emits light in a wide band of spectral wavelengths. The advantage gained is the ability to see a large range of particle sizes (Sadar, 1998). The disadvantage is that color typically produces negative interference with turbidity measurement. Turbidity is defined by the amount of light that is scattered at 90 degrees, and so any light that is absorbed in the sample cannot be scattered to the detector. The color may derive from dissolved material that produces a colored matrix or from particles that are colored, or both. Two benefits are gained from using the ratioing detectors: (1) they effectively compensate for color as an interference, and (2) they extend the operational range of the instrument so that fewer dilutions of turbid samples are required.

We added varying amounts of IHSS Nordic Aquatic fulvic acid to samples of formazin at an initial concentration of 58 NTU and measured them on both instruments to evaluate the effectiveness of the ratioing nephelometer (Figure 3). Samples were measured on both instruments at NWQL for about 8 months and the results were compared (Figure 4). No universal correlation could be made between the two instruments and the different results. All the variability in results seemed to be site specific.

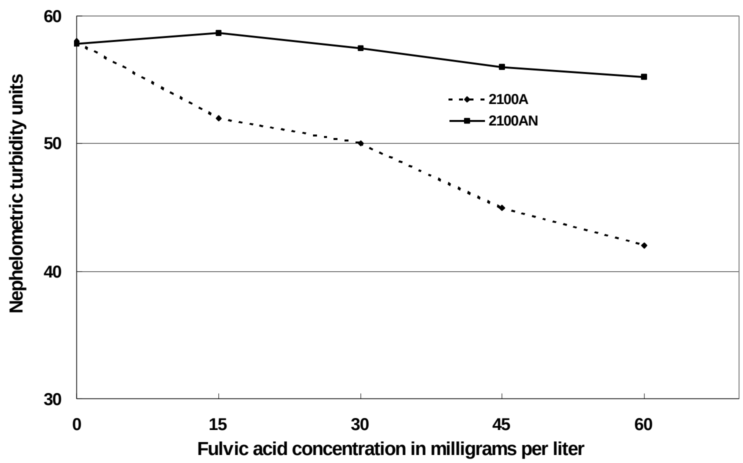

Figure 3. Formazin with fulvic acid added

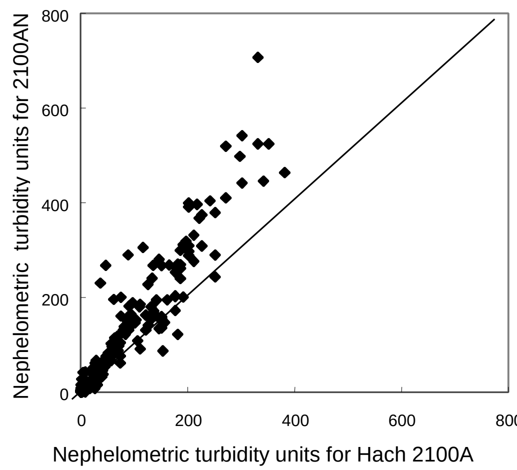

Figure 4.- long term comparison of instruments

Wavelength studies: As a follow-up to the initial study concerning differences in turbidity caused by changes in the number of light detectors, the NWQL also studied the effect that changing the wavelength of the light source has on turbidity measurements. The International Organization for Standardization (1990), in the International Standard (ISO 7027) for measuring diffused radiation, requires that the light source be at the specific wavelength of 850 nanometers (nm), with a spectral bandwidth of less than or equal to 60 nm. At this wavelength, light absorption caused by naturally occurring color usually is not a concern. As with the Standard Methods 2130 B (American Public Health Association and others, 1998), light is measured at 90 degrees to the incident light. Diffused radiation, turbidity, measured under ISO 7027 instrument criteria, is expressed as formazin nephelometric units (FNU).

An 860-nm filter was placed in the Hach 2100AN and then the instrument was calibrated with formazin standards. The 860-nm filter effectively allowed only light at about 860 nm to be read by the detectors. The results for samples were compared to results obtained from a YSI 6920 sonde field monitor with a light-emitting diode light source that also was calibrated with formazin standards. The YSI and the Hach instrument that was calibrated with the 860-nm filter (both FNU) had good agreement, but usually were higher than results obtained by using the Hach instrument and USEPA filter (NTU).

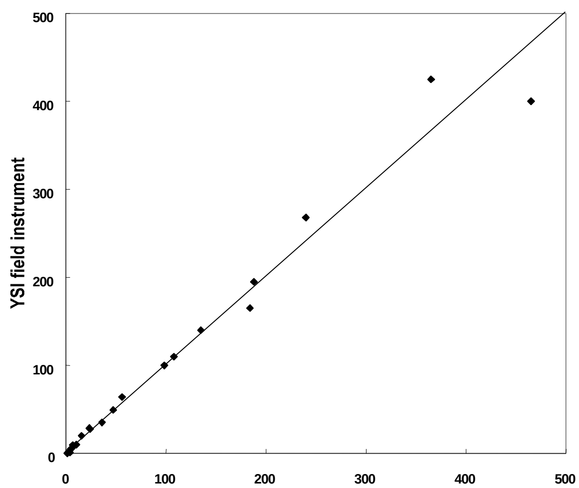

Figure 5. – Comparison of Hach to YSI field instrument

These studies show the importance of clearly limiting the variables and defining the instrument characteristics used to measure turbidity.

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OBS CALIBRATION AND FIELD MEASUREMENTS

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ABSTRACT

INTRODUCTION

Several major engineering projects and their components are related to water. These include cooling water intakes and hot water outfalls for thermal and nuclear power station, dams and irrigation canals, flood control and river bank stabilization works, navigation channels for harbors, and so on. The engineering design challenges may include design of sediment-free water intakes, avoiding local scour, estimation of siltation in navigation channels etc. The design problems and determination of counter measures are tackled through numerical or physical modeling, analytical methods or desktop studies. Field measurement of suspended sediment concentration is an essential requirement in all such problems. The data are used for validation of models or for drawing conclusions based on data analysis. Suspended sediment also has adverse environmental impact. A high concentration of fine sediment in suspension may clog fins of fishes resulting in their death. Deposition of fine sediment on leaves of aquatic plants reduces photosynthesis and hinders generation of new biomass. Suspended fine sediments cause substantial reduction in the amount of natural sunlight reaching sediment beds, which adversely affects growth of submerged aquatic vegetation and if it occurs over a large area, it may adversely affect the local ecosystem. Here again, data on suspended sediment concentration in the field is essential for evaluating the level of environmental impact and taking mitigation measures.

The traditional method consists of collection of field water samples in bottles, filtering them to separate out the suspended matter and determine its percentage with respect to the quantity of water sample used for analysis. Although this method is reliable and accurate, it has several disadvantages. The method is cumbersome, time-consuming, expensive, labor-intensive, and the results are not available quickly. Additionally, the water samples need to be preserved at low temperatures until they are analyzed in the laboratory. These disadvantages are overcome by using an Optical Backscatter Sensor, which is used for determining total suspended matter or turbidity. In addition, there is minimum disturbance to flow due to its small size. The sensors are commercially available. Pratt (1990) has described these sensors, their operating principles and the measuring system. He has also provided sensor thresholds and sensing limits to ensure accurate data collection.

OBS CALIBRATION

One of the major limitation for the use of OBS for obtaining reliable data is the requirement of their frequent calibration using the sediment that is present in the area of measurement. It is essential for the field group to have a facility for calibrating OBS sensors. Pratt (1990) conducted a study to offer operational guidelines and calibration techniques for using OBS. He has described the laboratory set-up and procedure to be followed for a satisfactory calibration of OBS sensors. Since different materials absorb and scatter light differently, calibration curves need to be developed for each sediment type because the calibration is material-specific. It may be noted that in addition to suspended sediment, other suspended substances such as diatoms, algae, and organic detritus cause turbidity in water column. The OBS cannot distinguish these substances from sediment. If the concentration of organic matter is high, measured turbidity does not give concentration of suspended sediment. It is advisable to always collect some water samples and determine the amount of organic content by standard ignition method.

FIELD MEASUREMENTS

Measurement of suspended sediment concentration alone is seldom done in the field. These measurements are invariably coupled with measurement of other field parameters. These include the sample positions, tidal water level, local water depth, magnitude and direction of current, depth of submergence of sensor, date and time of measurement and so on. It is quite common to obtain a time series of data on suspension concentration over a long duration extending to several weeks along with simultaneous time series data on other related parameters. Several instruments are attached to a mooring string, which is anchored to the bed with a floating buoy at the surface to mark its position. Sometimes a series of instruments may be attached alongside of a fixed platform. Fagerburg and Pratt (1998) collected extensive field data for the Upper Mississippi River Project. The objective was to measure increase in concentration of sediment suspension over the background resulting from passage of a vessel. An example of data is given in Figure 1. Such data are extremely useful in offering solutions to engineering problems. Field observations using OBS were also conducted for monitoring sediment plume of deposited dredged material in Delaware River. The data were analyzed and plotted on the relative acoustic intensity measured over a river cross-section as shown in Figure 2.

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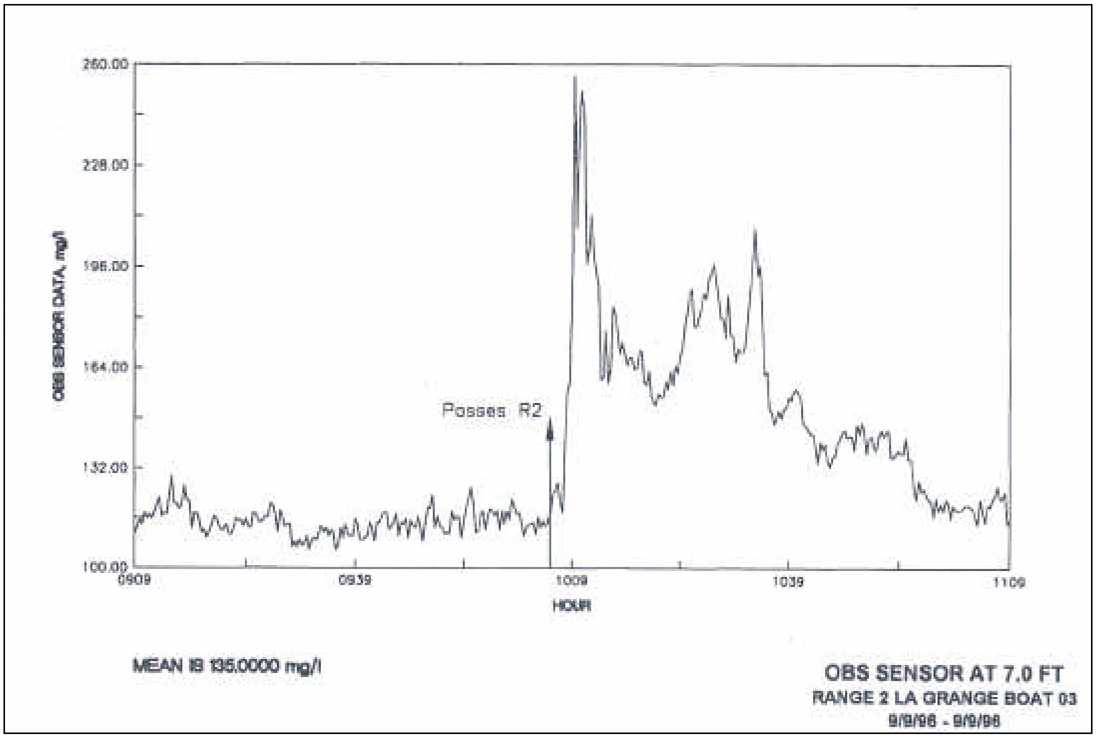

Figure 1: OBS measurement of sediment resuspension caused by passage of vessel in Upper Mississippi River

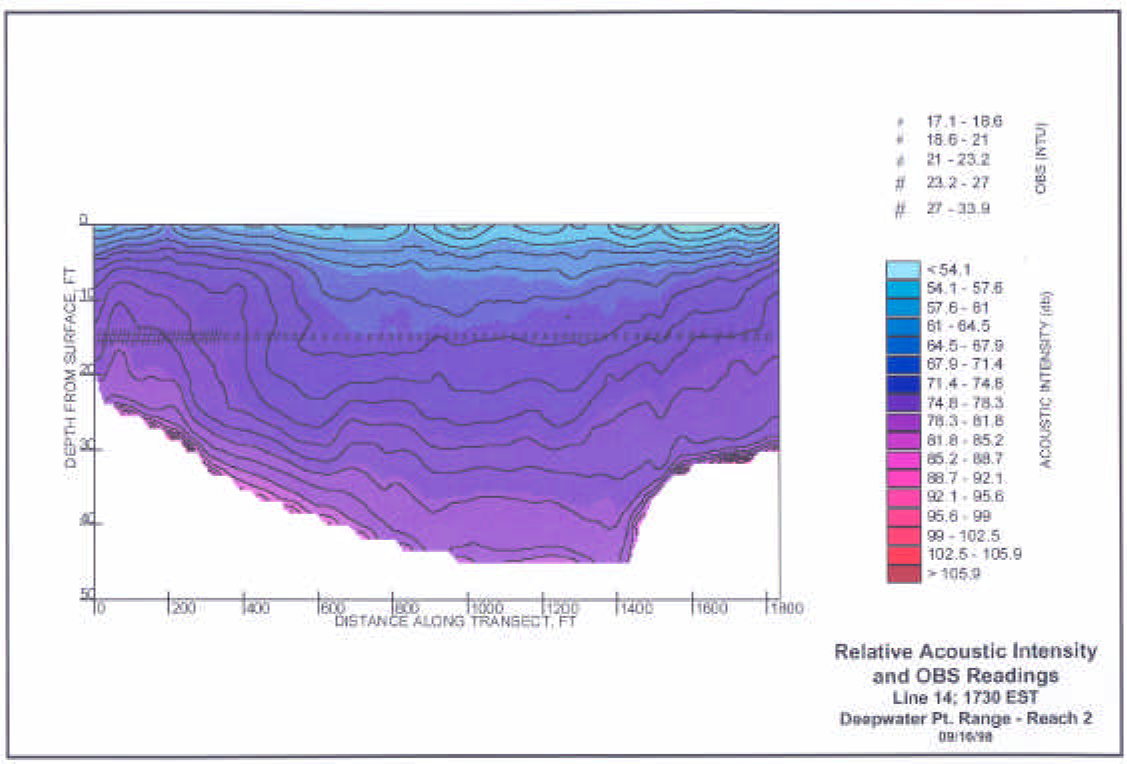

Figure 2: Relative acoustic intensity and OBS readings in Delaware River

USES OF TURBIDITY BY STATE AGENCIES

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ABSTRACT

Objective A questionnaire on uses of turbidity was submitted to state water quality coordinators. The objective of the questionnaire was to determine how turbidity was being used in addressing water quality issues including water quality criteria, what water bodies were being measured using turbidity including ranges observed, what technology was being used to measure turbidity, how turbidity was being calibrated, and how turbidity measurements could be improved in the future. The query also included questions pertaining to TSS, SSC, bedload, and particle size analysis.

Results Thirty-two of the fifty states responded to the questionnaire. The majority of the states that responded either used Oracle, STORET, or a "local" database or spreadsheet for data storage and analysis. The primary objective of the majority of the states was the establishment of a water quality criterion for turbidity that was protective of aquatic life. The majority of the states are using EPA method 180.1 for turbidity and method 160.2 for TSS. Turbidity measurements between states range from 0.4 to 2552 NTU. Numeric standards ranged from 5 NTU above ambient conditions to 50 NTU instantaneous measurements. Some states have established numeric standards that are basin-specific, while others vary with water bodies or presence of Salmonids. In general, most states were concerned with the effects of water clarity and light scattering on aquatic biota. Most states are presently using optical backscatter or optical transmission technology either by measuring *in situ* or on an environmental sample collected by grab or single-point, automatic sampler. The majority of the states are using formazin as a standard. Only three of the states that responded are using integrated sampling methods. Only three states are attempting to correlate turbidity with TSS or biological impairment. Only three of the states are presently using or planning to use SSC. The rest are using TSS. Four states are measuring particle size distribution using a wet sieve method. No states are presently measuring bedload. Most states recognize interferences (e.g., algal blooms), however, no states are attempting to adjust turbidity measurements accordingly.

Future Needs Most states agreed that more effort should be devoted toward improving the relationship between turbidity, TSS, SSC, channel stability, and biological impairment. In addition, many states expressed a need for establishing reference fluvial sediment conditions and means of measuring significant departure from reference conditions. Improvements need to be made in depth integrated isokinetic samplers. Many states were in favor of a consistent procedure and less expensive probes that can be rapidly deployed and are stable in the field. Several states expressed the need for additional long-term, stream discharge, suspended and bedload data. Instrumentation used for *in situ* measurements needs to be specially equipped for high bridge deployment with stabilization fins.

CONTINUOUS IN-SITU MEASUREMENT OF TURBIDITY IN KANSAS STREAMS

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ABSTRACT

Continuous, in-situ measurements of turbidity to estimate suspended-sediment concentrations are being made at stream monitoring sites throughout the United States. Considerations for selecting instrumentation, proper installation, methods for verifying sensor performance, and collection of point in-situ data that are representative of the channel cross section need to be well thought out for the data to be of acceptable quality. Experiences and specific examples for selected monitoring sites in Kansas are discussed.

Choosing an Instrument: There are many turbidity/optical backscatter probes suitable for continuous in-situ measurements. Sensors can measure turbidity values ranging from 0 to 1,000 nephelometric turbidity units (NTU), with some capable of measuring up to 4,000 NTU. Most turbidity probes conform to ISO method 7027 or GLI Method II. Currently, the only method approved for measuring turbidities > 40 NTU in stream source water is ISO method 7027. Some manufacturers offer features that help improve the quality of the data and extend the time between maintenance trips. Such probes are equipped with mechanical wipers or shutter technology that activate prior to a measurement and keep the sensor clear of interference. Probes that are SDI-12 (serial data interface at 1200 baud) compatible are easily installed at U.S. Geological Survey (USGS) stream-gaging stations that have data-collection platforms (DCPs), and the data can be displayed on the World Wide Web in real time. Daily review of real-time turbidity data is essential for timely troubleshooting of equipment malfunctions.

Installation: Several factors need to be considered prior to installation of a turbidity sensor as a surrogate for determining suspended-sediment concentrations in streams. First, a monitoring site that represents the area of interest and is located at a cross section of the stream that is well mixed needs to be selected. In Kansas, the USGS has selected mostly sites with existing stream-gaging stations. Adding a turbidity sensor to an existing stream-gaging station has several advantages: (1) continuous flow data are available for load calculations, (2) the equipment infrastructure for logging and transmitting the data is in place, and (3) sample collection is possible at all flow regimes. For ungaged installations, site selection for the turbidity sensor should be based on the same criteria for choosing the location of a stream-gaging station (that is, accessibility during all flow regimes, total flow is confined to one channel, the general course of the stream is straight within a few hundred feet of the stream, etc.. Rantz and others, 1982).

After the site is selected, the type of installation needs to be determined. In Kansas, the USGS has successfully used two types of installations, horizontal bank (or fixed) and vertical suspension. Bank installations have been limited to sites with small drainage areas. This type of installation has failed at sites with large drainage areas because, during extended periods of high flow, floating debris damages the equipment and high sediment concentrations fill the protective plastic pipe with mud and silt to the point that the turbidity probe becomes extremely difficult to retrieve. Most of the USGS turbidity monitoring sites in Kansas use a vertical suspension installation from the bridge deck to the stream. Vertical suspension is the most adaptable and convenient for installation and maintenance. The installation is made up of a turbidity probe, 10 feet of plastic pipe, a chain, a 12-volt winch, and sometimes a radio transmitter. The pipe and turbidity sensor typically are suspended behind a bridge pier so that the sensor is protected from debris. The pipe and turbidity sensor are tethered from the bridge deck using the chain. The DCP inside the gage house logs data every 15 or 30 minutes, either directly from the sensor or via radio communication from the sensor. The DCP then transmits the logged data every 4 hours via satellite for display of the data on the World Wide Web. A watertight aluminum box encloses the transmission equipment and is mounted to the bridge rail using clamps so that no holes are necessary in the bridge rail. The winch is used to raise the pipe to the bridge deck for servicing or repairing the sensor. The versatility of this type of installation is that it can be installed on any bridge and at any point along that bridge. This type of installation can be easily adjusted during high-flow conditions and relocated on meandering streams.

Continuous Measurements: The turbidity sensor is serviced several times a year. The USGS in Kansas uses a turbidity sensor equipped with a mechanical wiper that impedes the accumulation of silt and microbial growth on the optic sensor, reducing the number of cleaning visits. The transmitted turbidity data are reviewed daily to verify sensor performance. The sensors are inspected monthly to verify the most-recent calibration. Most sensors are designed to be calibrated with a formazin standard. Using standards that are not approved by the sensor's manufacturer most likely will not provide accurate readings. During these inspections, a calibrated field sensor is used to measure the turbidity at a minimum of 10 locations throughout the cross section of the stream. These data are used to verify that data from the continuous turbidity sensor are adequately representing the entire stream cross section. If the comparison differs by more than 10 percent the sensor can be relocated to a more representative location. The sensor is not relocated on the basis of temporary situations, but only as a result of long-term variations. A good check of the continuous in-situ turbidity sensor is determined by regressing the average cross-section measurements with the in-situ sensor values (fig. 1).

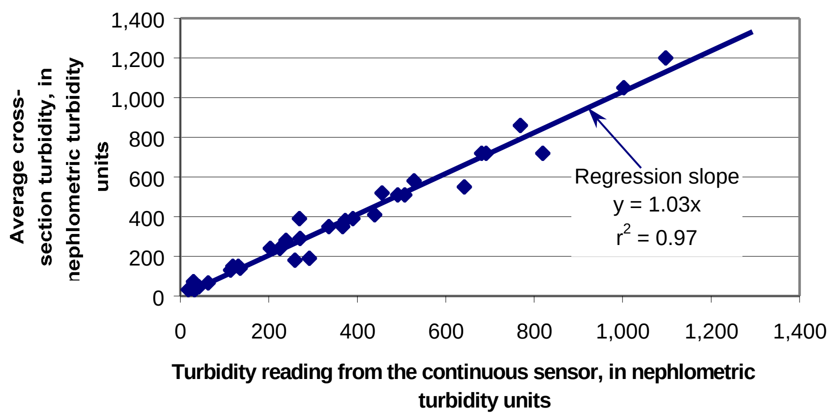

Figure 1. Comparison of continuous-sensor and cross-section turbidity values for Kansas River at Topeka, Kansas, October 2000 through January 2002.

The closer the slope is to 1.0, the more representative the data from the continuous sensor are of turbidity in the stream cross section without correction. At least 20 to 30 measurements throughout the entire range of turbidity values (0-1,500 NTU) are necessary to develop a robust relation. Regression results made on the basis of fewer measurements can lead to false conclusions. An effective method for determining at what turbidity level a cross-section measurement is necessary is to construct a turbidity duration curve (fig. 2).

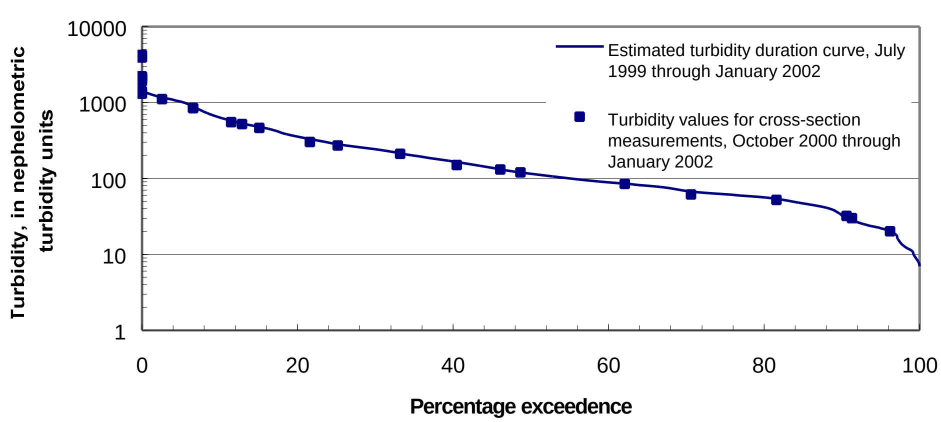

Figure 2. Turbidity duration curve for Kansas River at DeSoto, Kansas, October 2000 through January 2002.

Cross-section turbidity values plotted on the duration curve represent ranges of turbidity values for which cross-sectional measurements would be required and when samples need to be collected. The duration curve also provides an excellent summary of the turbidity conditions at a particular site.

Turbidity as a Surrogate: Continuous turbidity measurements have been shown to reliably estimate concentrations and loads of several constituents with defined uncertainty. Using methods explained in Christensen and others (2000), estimates for suspended-sediment load (fig. 3), total suspended solids, fecal coliform, *E. coli*, and total nitrogen and phosphorus can be estimated continuously and in real time

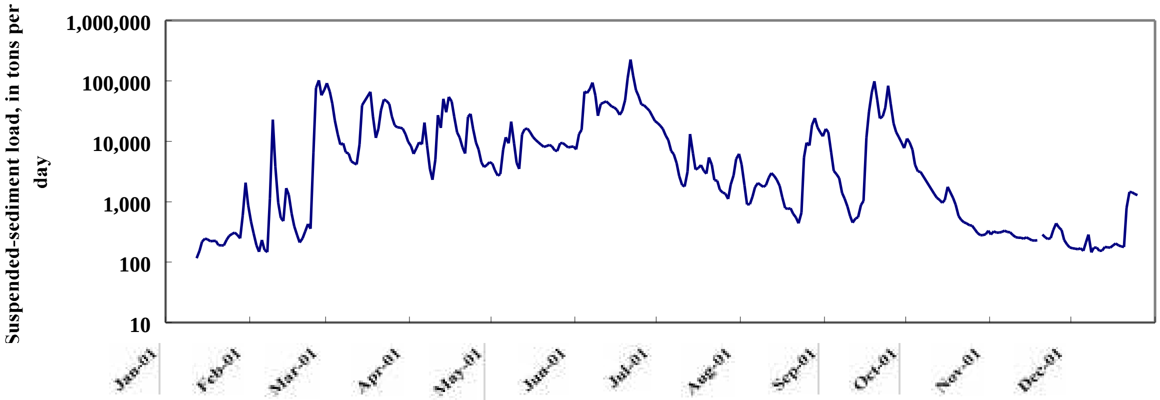

Figure 3. Turbidity-estimated suspended-sediment load for Kansas River at DeSoto, Kansas, 2001.

(http://ks.water.usgs.gov/Kansas/rtqw/). The advantage of continuous regression estimates using continuous turbidity measurements over discrete sample collection is that continuous estimates represent all flow conditions regardless of size or duration. This can be an advantage when determining total maximum daily loads or assessing resource-management practices.

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REAL-TIME WATER-QUALITY MONITORING IN KANSAS

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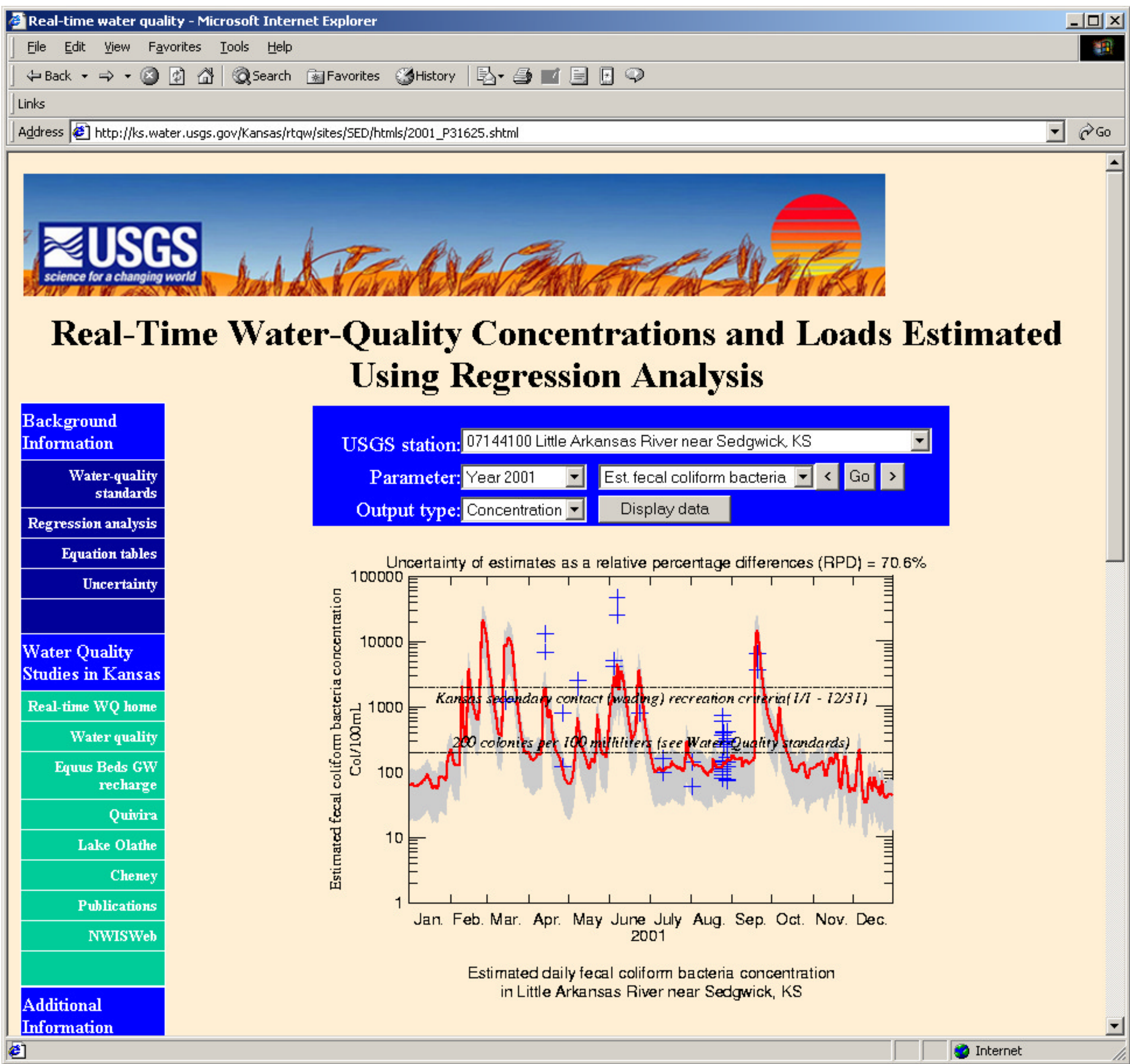
ABSTRACT

The U. S. Geological Survey (USGS) has established a real-time water-quality notification system for 12 surface-water sites in Kansas. Real-time water-quality data, including suspended sediment, fecal coliform bacteria, nutrients, and atrazine, are estimated and displayed in real time. Information is updated every 4 hours and is available on the Internet at http://ks.water.usgs.gov/Kansas/rtqw/.



This system was developed by the USGS in cooperation with the city of Wichita, city of Olathe, Groundwater Management District No. 5, Kansas Department of Health and Environment (KDHE), U.S. Fish and Wildlife Service, and the U.S. Environmental Protection Agency. The system allows water-resource managers to make decisions on the basis of real-time water-quality estimates, which can improve response times for drinking-water treatment and environmental monitoring. Long-term continuous monitoring will allow users to better determine and monitor the effectiveness of total maximum daily loads and the effects of resource management practices on stream water quality.

SELECTED REFERENCE

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TURBIDITY INSTRUMENTATION - AN OVERVIEW OF TODAY'S AVAILABLE TECHNOLOGY

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ABSTRACT

Introduction: Turbidity been used for many years as a surrogate for monitoring the combined quantity of particulate material in a water sample and as such, has been one of the parameters used to provide a basic assessment of water quality. The development of the first analytical turbidimeters was in the 1960s and the fundamental optical technology remained unchanged until the mid-1980s. Since then, instrument design technology has advanced dramatically and many new designs have resulted. These new designs have evolved to address many of the traditional interferences associated with turbidity. Because different technologies (such as light sources and detector design) have been used to compensate or eliminate interferences such as color, bubbles, stray light, absorption, and path length, it is often difficult or impossible to compare measurements.

This paper will provide a brief description of turbidity and review the current instrument designs. How each design attempts to address specific interferences will be addressed and a proposal will be made to assign specific units to each instrument design. The author proposes that a more meaningful turbidity value will be produced.

General Overview of Turbidity Measurement: In its simplest terms, turbidity is the optical measurement of scattered light resulting from the interaction of incident light with particulate material in a liquid sample. Typically, the liquid is a water sample and the suspended material causing the light to be scattered can be composed of a broad variety of components. Examples of particles include: suspended solids such as silt, clay, algae, organic matter, various microorganisms, colloidal material, and even large molecules that are dissolved in the sample such as tannins and lignins.

The Theory of Light Scattering and Common Interferences: Particulate matter in a water sample will cause the incident light beam to be scattered in directions other than a straight line through the sample. The scattered light that returns to the detector causes a response correlating to the level of turbidity in the sample. A higher level of scattered light reaching the detector results in a higher turbidity value.

The measurement of turbidity is not directly related to a specific number of particles or to particle shape. As a result, turbidity has historically been seen as a qualitative measurement. In an attempt to make turbidity methods more quantitative, we can use standards and standardization methods.

Although interferences have a dramatic and ever-present impact on turbidity measurements, the type and magnitude of the interference often depends on the turbidity level being measured. When performing low-level turbidity measurements (<5 NTU),

primary interferences are stray light, bubbles, ambient light, and contamination. For high turbidity testing (5 NTU or greater), a greater impact from color, particle absorption, and particle density is seen. Table 1 summarizes these interferences:

Table 1 – Typical Interferences Associated with Turbidity Measurement

Interference	Effect on the Measurement
Absorbing particles (colored)	Negative bias (reported measurement is lower than actual turbidity)
Color in the matrix	Negative if the incident light wavelengths overlap the absorptive sprectra within the sample matrix
Particle Size	Either positive or negative (wavelength dependent) a) Large particles scatter long wavelengths of light more readily than small particles. b) Small particles scatter short wavelengths of light more efficiently than long wavelengths
Stray light	Positive bias (reported measurement is higher than actual turbidity)
Particle Density	Negative bias (reported measurement is lower than actual turbidity)
Contamination	Positive bias (reported measurement is higher than actual turbidity)

In an attempt to minimize interferences, several new turbidity measurement methods have been developed. Many of these methods have been designed to maximize sensitivity and minimize the effects of interferences. It is important to understand and identify the prominent interferences in your sample stream. Doing so can help identify the instrument design that will provide the most accurate and "interference-free" measurement. Instrument designs can be categorized as shown in Table 2 below:

Table 2 – Summary of Instrument Designs:

Design	Prominent Feature and Application
Nephelometric non-ratio	White light turbidimeters – Comply with EPA 180.1 for low level monitoring.
Ratio White Light turbidimeters	Complies with LT1 and SM. Uses a nephelometric detector as the primary detector, but contains other detectors to minimize interference. Can be used for both low and high level measurement.
Nephelometric near IR turbidimeters	Complies with ISO 7027 – The wavelength (860-890-nm) is less susceptible to color interferences. Good for samples with color and good for low level monitoring.
Nephelometric Near IR turbidimeters	GLI method 2, ISO 7027 and USEPA approved. Compliant and contains a ratio algorithm to monitor and compensate for interferences.
Surface Scatter Turbidimeters	Turbidity is determined through light scatter from or near the surface of a sample. The detection angle is still nephelometric, but interferences are not as substantial as nephelometric non-ratio measurements. This is primarily used in high-level turbidity applications.
Back Scatter/Ratio Technology	Backscatter detection for high levels and nephelometric detection for low levels. Backscatter is common with probe technology and is best applied in high turbidity samples.
Light attenuation FAU	The use of a transmitted detector (180 degrees to the incident light beam). Most susceptible to interferences, best applied at medium turbidity levels (5-1000).

The dilemma: The units for reporting turbidity are commonly the same, no matter which turbidimeter design is being used. Depending on the interferences present (especially in high level reporting), the instrument design can have a dramatic effect on the reported result. For example, if a high level sample is measured with a white light non-ratio instrument, the results will be dramatically different from a reading obtained using a 4-beam, IR ratio method. One solution is to apply the correct measurement units to the measured value to help rationalize the results. Table 3 contains a proposal for using standardized units to report turbidity.

Table 3 – Proposed Units for Technology Traceability

Unit	Name	Description of Compliant Technology
NTU	Nephelometric Turbidity Unit	White light, 90 degree detection only
NTU_R	Ratio Nephelometric Turbidity Unit	White Light, 90 degree detection with additional correction detectors
FNU	Formazin Nephelometric Unit	860-nm Light (near IR) with 90-degree detection (ISO7027 compliance).
FNU_R	Formazin Nephelometric Unit	860-nm Light with 90-degree detection and additional interference correction detectors.
FNU_{2B}	Formazin Nephelometric Unit – Dual Beam Detection Technology	4 beam IR Detection utilizing 2 light sources and two detectors.
FNU_{BS}	Formazin Nephelometric Unit using Backscatter Detection	860-nm detection angle with a backscatter detector (270 – 285) degrees angle relative to the incident beam
FAU_{XXX}-nm	Formazin Attenuation Unit using a defined wavelength	The detection angle is 180 degrees of the incident light beam

Conclusion: The correct assignment of turbidity units to the recorded turbidity result is critical in understanding if interferences were addressed to some level. Currently, the NTU unit is used for all turbidity measurements and the reported value does not have any traceability to the instrument technology used. At the very least, the units should be listed to the level of NTU, FNU, or FAU to the measured unit.

The ability to accurately trace the measurement to an instrument design technology is necessary to effectively quantify the turbidity measurement. Attaching more specific units to the results will help to clarify the turbidity value and will allow the user to determine when it is appropriate to directly compare results obtained with different instruments.

TEN YEARS OF CONTINUOUS SUSPENDED-SEDIMENT CONCENTRATION MONITORING IN SAN FRANCISCO BAY AND DELTA

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ABSTRACT

Oceanographers began to commonly use optical sensors for measuring turbidity or suspended-sediment concentration (SSC) in the 1980s on the continental shelf, in nearshore waters, and in estuaries (Sternberg 1989). In December 1991, the U.S. Geological Survey (USGS) installed the first optical sensor for continuous monitoring of SSC in San Francisco Bay. Suspended sediment is an important component of San Francisco Bay and the tributary Sacramento-San Joaquin River Delta because it transports adsorbed toxic substances, provides habitat for benthic organisms, limits light availability and photosynthesis, contributes to wetland restoration, and deposits in ports and waterways that require dredging. In December 2001, SSC was monitored at 13 stations in the Bay and Delta. As of 2002, 159 sensor years of data have been collected, and the network is believed to provide the longest, continuous SSC time series collected in an estuary. Despite data losses due to biological fouling, the network provides a wealth of data that are used to monitor SSC and to determine the processes that affect SSC at tidal to annual time scales. A complete listing of publications describing the data-collection methods and data analyses is available at http://ca.water.usgs.gov/abstract/sfbay/sfbaycontbib.html.

Sampling Design: The SSC monitoring network is designed to capture the spatial and temporal variability of SSC (Buchanan and Ruhl 2001). Stations were established in each major subembayment of San Francisco Bay and in the primary Delta channels. Bay stations originally were established in a deep channel (depth about 25 - 50 feet), often at salinity monitoring stations. Near-bottom and mid-depth optical sensors were deployed in the deep channel. In 1998 a shallow water station (mean lower low water depth about 6 feet) in San Pablo Bay was added to the network. Semidiurnal tides and lower-frequency tidal constituents drive temporal variability of SSC, so measurements are recorded every 15 minutes. In addition to the continuous monitoring network, we have deployed optical sensors at as many as 14 sites for periods of several months as part of focused studies of sediment transport in shallow subembayments and Bay locales of special interest.

Installation, fouling, and maintenance: Optical sensors are positioned in the water column using polyvinyl chloride (PVC) pipe carriages coated with an antifoulant paint to impede biological growth (Buchanan and Ruhl 2001). Carriages were designed to align with the direction of flow and to ride along a stainless steel or Kevlar-reinforced nylon suspension line attached to an anchor weight, which allows sensors to be raised and lowered easily for servicing (fig. 2, Buchanan and Ruhl 2001). The plane of the optical window maintains a position parallel to the direction of flow as the carriage and sensor align itself with the changing direction of flow. An electronic data logger controls data acquisition.

The greatest problem in using optical sensors in San Francisco Bay and Delta is biological fouling that invalidates about one-half of the data. Fouling begins to affect sensor output from 2 days to several weeks after cleaning, depending on the level of biological activity in the Bay. Generally, biological fouling is greatest during spring and summer and at stations in saltier water. Optical sensors require frequent cleaning but, due to the difficulty in servicing some of the monitoring stations, they are cleaned every 1-5 (usually 3) weeks. Self-cleaning sensors have proven to reduce data loss only in relatively fresh water because they are ineffective when fouling is excessive and they are prone to leak and malfunction in saltier water.

On-site checks of sensor accuracy are done using 50 to 200-nephelometric turbidity unit (NTU) solutions prepared from a 4,000-NTU formazin standard. Solutions are prepared by diluting the 4,000-NTU stock standard with high-purity water in a clean, sealable bucket. At the field site, the cleaned sensors are immersed in the solution and the sensor output is recorded on the station log to help identify output drift and sensor malfunction.

Calibration: Calibration is needed to determine the relation between sensor output and SSC. This relation varies according to the size and optical properties of the suspended sediment; therefore, the sensors must be calibrated for each site using suspended material from the field (Levesque and Schoellhamer 1995). Water samples are collected before and after sensor cleaning during site visits (Buchanan and Ruhl 2001). The water samples are analyzed to determine SSC, which ranges from nearly zero to more than 1,000 mg/L.

At Bay and Delta sites, suspended particles primarily are fine sediments and particle size variability does not affect calibration of the sensors (Schoellhamer 2001) and sensor output is proportional to SSC (Buchanan and Ruhl 2001). (Schoellhamer 2001, presents a contrasting example of particle size variability affecting sensor calibration in the Colorado River). The output from the optical sensors is converted to SSC using the robust, nonparametric, repeated median method (Siegel 1982, Buchanan and Ruhl 2001). We no longer use ordinary least-squared regression because the calibration data usually are not homoscedastic (Helsel and Hirsch 1992). Bay sensors are calibrated to point SSC measurements and Delta sensors are calibrated to discharge-weighted cross-sectionally averaged SSC, which can be multiplied by water discharge to determine suspended-sediment discharge (Schoellhamer 2001). Data from several years are used to develop the calibrations if the same sensor has been operating at a site and there is no evidence of sensor output drift. At some of the landward sites, the calibration line shifts slightly during periods of relatively large freshwater inflow.

We prefer to use a relatively unprocessed signal to determine SSC rather than a calculated value, such as turbidity in NTU. The benefit of this approach is illustrated by the following. A commercially available multiprobe, which was used at some stations, included a software error in the interpolation table that converted the raw signal to NTUs (the unit's standard output). Scatter plots of the turbidity data from all identical probes indicated that there were minimal data between 50 to 70-NTUs due to an incorrect value in the table for the 60-NTU conversion, resulting in too few values in the 50 to 70-NTU range and too many values in the 0 to 50-NTU range. The manufacturer corrected the error and subsequent data do not display this characteristic.

Data processing: The raw time series data are archived and edited to remove invalid data. Recorded data are downloaded from the data logger onto a data storage module or laptop computer during site visits. Raw data are loaded into the USGS automated data-processing system (ADAPS).

The time series are retrieved from ADAPS and edited. As biological growth accumulates on the optical sensors, the output of the sensors increases or decreases, depending on the type of sensor. Invalid data collected prior to cleaning cannot be corrected because fouling masks the desired signal. Such data are removed from the record (fig. 3, Buchanan and Ruhl 2001). A correction is applied to the data, however, on the rare occasions when incomplete cleaning of a sensor causes a small, constant shift in sensor output that can be corrected using water-sample data. Spikes in the data, which are anomalous outputs probably caused by debris temporarily wrapped around the sensor or by large marine organisms (fish, crabs) on or near the sensor, also are removed from the raw data record. Processed SSC data are stored in ADAPS, published (Buchanan and Ruhl 2001), and are available on the Internet at http://sfports.wr.usgs.gov/Fixed_sta/ .

Acknowledgements: We thank Rick Adorador, Greg Brewster, Tom Hankins, Rob Sheipline, and Brad Sullivan for helping install and maintain the SSC monitoring network. Support for the network has come from the San Francisco Bay Regional Water Quality Control Board, U.S. Army Corps of Engineers as part of the Regional Monitoring Program for Trace Substances, CALFED Bay/Delta Program, USGS Federal/State Cooperative Program, and USGS Place-based Program.

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MANAGING TURBIDITY, SUSPENDED SOLIDS AND BEDDED SEDIMENTS UNDER THE CLEAN WATER ACT– THE EPA PERSPECTIVE

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ABSTRACT

Excessive erosion, transport and deposition of suspended solids and bedded sediments in surface waters is a major form of pollution resulting in extensive water quality problems throughout the Nation's waters. The 1998 National Water Quality Inventory ranks suspended solids and sediments as the leading cause of water quality impairment of rivers and lakes.

The States and Tribes are required by the Clean Water Act (CWA) to adopt water quality standards to protect public health and welfare, protect designated uses, enhance the quality of water and serve the purposes of the CWA. Water quality standards consist of designated uses, water quality criteria to protect those uses, and an antidegradation policy.

States and Tribes may adopt numeric water quality criteria into their water quality standards using CWA Section 304(a) criteria guidance; Section 304(a) criteria guidance modified to reflect site-specific conditions; or other scientifically defensible methods. EPA has published aquatic life criteria guidance for 31 chemicals and human health criteria for 110 chemicals. However, EPA has not yet published new criteria guidance for turbidity, suspended solids, bedded sediments or other indicators. Only an old solids and turbidity criterion remains from the 1970s.

In lieu of useful criteria for turbidity, suspended solids and bedded sediments, and given the large number of impaired water bodies and potential litigation, the States and Tribes are using a variety of approaches to managing these pollutants through the Total Maximum Daily Load (TMDL) program and are imposing National Pollutant Discharge Elimination System (NPDES) permit requirements on point sources and recommending best management practices (BMPs) on non-point sources to control turbidity and sediment throughout watersheds across the Country.

This presentation provides an overview of EPA's approach for dealing with turbidity, suspended solids and bedded sediments, top priority research needs, EPA research strategy to help resolve this problem, what States are currently doing, EPA's plans for developing water quality criteria and how EPA envisions suspended solids and embedded sediments be dealt with under the CWA legal and regulatory framework.

THE ADVANTAGE OF CONTINUOUS TURBIDITY MONITORING: A LESSON FROM THE NORTH SANTIAM RIVER BASIN, OREGON, 1998-2002

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Background

The North Santiam River Basin in Oregon drains the western Cascade Range and has a drainage area of 690 mi^2 upstream from the City of Salem Water Treatment Plant. Most river channelsin the basin are steep sloped with high stream velocities, especially during the high flow period from November to March. Normal summer to fall turbidity at base flow, as well as steady-state flow during other times of the year, ranges from 0 to around 3 Nephelometric Turbidity Units (NTU). Turbidity during winter highflow and storms approachs 300 NTU, and can peak at 1,400 NTU or higher for brief periods during landslide and glacial events. Most high turbidity (median daily turbidity greater than 10 NTU) occurs less than 10 percent of the time. There is little algal or organic growth in the North Santiam system.

Equipment

Selection of turbidity equipment depends on the objectives of the study. The North Santiam study required monitoring additional water quality parameters, so a multi-parameter datasonde device was used. The North Santiam study uses eight continuously monitoring datasondes, connected by either a hard-line or cellular phone system and are interrogated every 3 to 4 hours, providing data in 30-minute increments. Other stand-alone turbidity probes are available, although most datasondes provide internal logging, which is an advantage over single turbidity probes, since the datasonde logger can serve as a backup to the gaging station data logger. Also, additional parameters can help verify turbidity spikes that occur from land disturbances or glacial activity, as their adjoining readings usually change in conjunction with the turbidity values during these events.

The North Santiam project uses wiper-type turbidity probes, which rotate before each reading in two directions to clean the lens. Proper wiper rotation depends on wiper quality. If the wiper becomes dirty, corroded or torn it will affect the parking of the wiper and subsequently the turbidity readings, as the wiper will interfere with the lens and the infrared reflectance, causing erroneous readings. Wiper maintenance is critical to proper turbidity monitoring, although there are stand-alone probes that alleviate this problem by parking the wiper magnetically.

Installation

Because the high stream velocities in the North Santiam River Basin can damage or impede the stability of the datasonde, the units were housed inside 4-inch, schedule 80 PVC pipe, which in-turn was housed inside 6-inch cast-iron well casing. Both were securely mounted to permanent structures along the stream bank, such as gaging station houses, large rocks or trees. The PVC pipe was perforated on the end to allow for water flow and extended out from the well casing by about 2 feet, with a stainless steel bolt through the end, to provide both a resting place for the datasonde and to prevent it from passing through the pipe bottom.

Probes should be placed away from any channel obstruction, such as large rocks, bridge piers or abutments, and at least 1-2 feet from the river bottom to prevent bed material and other obstacles from affecting the readings. For best performance, the probe should be located in moving water, but the velocities should not be so turbulent as to cause air bubbles surrounding the probe.

Calibration

The North Santiam project developed protocols to calibrate turbidity and other water-quality probes that use a backup datasonde to compare readings to the station datasonde. Calibrations are conducted routinely on a 2-3 week basis or more if readings indicate a problem. All calibrations are conducted at each site with standards at stream temperatures. Each turbidity probe is initially calibrated to 0 (dionized water, DI), 10, and 100 NTU stabilized formazin (Sadar, 1999), after which the probe calibration is checked using a polymer-bead standard.

Initial readings are collected from the station probe and backup probe, after the backup probe has equilibrated in the stream at close proximity to the station probe. The station probe is then cleaned and another set of duplicate readings are recorded from both probes. These cleaning corrections are applied in ADAPS similar to datum corrections in working discharge record, although very few cleaning corrections are necessary, due to the wiper cleansing process. Next the station probe calibration is checked in 0 (DI), 10, 100, and 1000 NTU polymer-bead standards. If the readings vary by more than 5 percent from the previous calibration, the instrument is recalibrated using formazin, otherwise the discrepancies less than 5 percent are handled as regular ADAPS variable-shift corrections adjusted to the turbidity calibration points.

Cross-sectional measurements, either from a bridge or cableway, also are collected and correlated to the instream turbidity readings. This is especially important for large stream widths where the streambank turbidity may not represent the entire cross-section turbidity. Also correlated to the instream station readings are samples collected for turbidity. These cross-sectional equal-width-increment, and/or dip samples collected near the datasonde pipe, are measured on site directly after the sampling.

Standards

Formazin is a suspected carcinogen and experimental mutagen with a short shelf life; the polymer-bead standard is less toxic and has a longer shelf life. For this reason the polymer-bead standard is used more frequently, but is considered the secondary standard and is used only for checking calibration. The U.S. Environmental Protection Agency recognizes formazin as a primary standard for calibration, at least until instream turbidity probe standard methods are developed.

Polymer-bead standards are instrument specific, particularly for instream turbidity probes, and will not calibrate correctly if they are not referenced to formazin using the same turbidity instrument. The North Santiam project worked with the standard and probe manufacturers, to prepare a polymer-bead standard referenced to formazin, using an identical instream turbidity probe.

Turbidity Records

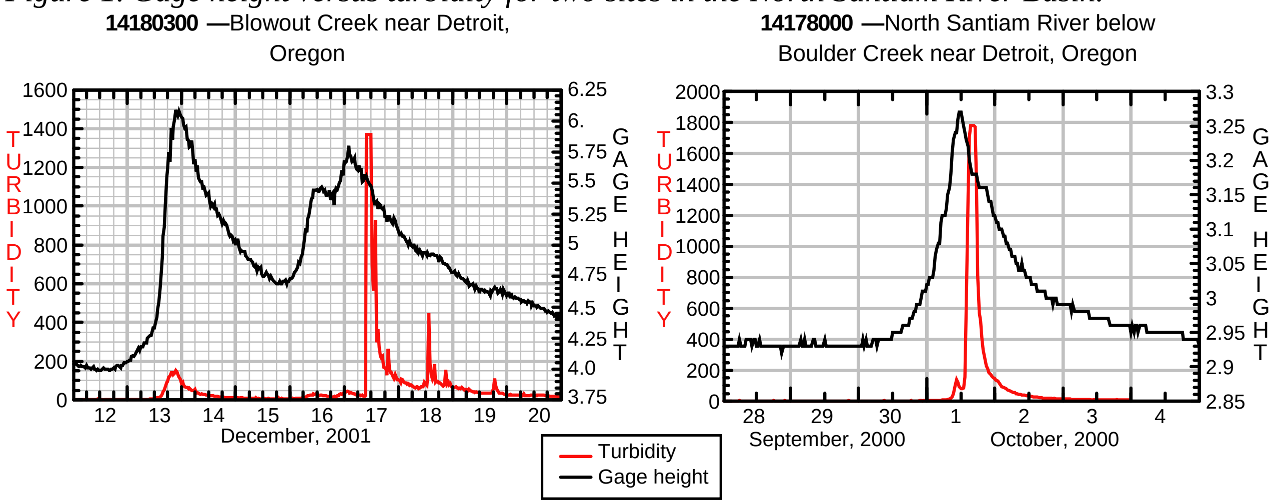
Instream turbidity is highly variable, especially in moving, dynamic river systems; even during normal base flow conditions. Most probes provide some data filtering, but occasional spikes will always occur in the turbidity record. If the spikes occur during high-flow storm conditions they are usually left as is; if they occur during quiescent conditions they are scrutinized carefully and removed if they vary by 10 percent or more from the previous value. For other periods of unexplained turbidity, the data are compared to the station streamflow and any local precipitation record, along with other neighboring continuous turbidity stations. Appropriate corrections are applied and probe operation is checked and/or recalibrated. Turbidity is published in whole numbers as the maximum, minimum and median daily value; turbidity from 0 to 1 is published as less than 1.

Turbidity versus Discharge as a Surrogate for Suspended-Sediment Concentration: Figure 1 illustrates two examples of why continuous turbidity monitoring is a more accurate and reliable surrogate for suspended-sediment concentration than discharge. The Blowout Creek Basin experienced a large landslide on December 17, 2001, after a 2-day, 2.5 inch precipitation event, causing abnormally high turbidity spikes. Turbidity values reached the high threshold of the probe at near 1,400 NTU, following an approximate 1-foot rise in gage height. Prior to that, on December 13 and 14th, a 3-day, 3.5 inch precipitation event occurred that caused a 2-foot rise in gage height, causing the turbidity to peak near 180 NTU. This later turbidity peak was the normal response to storm events for this basin. Conversely, the December 17th peak was caused by massive slope failure, undetectable using discharge correlated to suspended-sediment concentrations. Suspended-sediment loads calculated using discharge as a surrogate would not have provided accurate data in this instance.

On October 1, 2000, a glacial outburst episode occurred on Mt. Jefferson, a volcano in the upper-most basin of the North Santiam River. In this case, turbidity spiked again at the probe threshold (near 1,800 NTU). The stage rise was only 0.3 feet, almost imperceptible, yet the river turned into a muddy-brown slurry. Again, discharge correlated to suspended-sediment concentration would not have computed an accurate rise in suspended-sediment load through this period.

Continuous turbidity monitoring is basin specific. Relationships developed between one basin are usually not directly compared to other basins, especially between areas of dissimilar topography and geology. A diligent calibration routine coupled with proper probe placement will yield good turbidity record with little missing data.

Figure 1. Gage height versus turbidity for two sites in the North Santiam River Basin.



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DETERMINATION OF TOTAL AND CLAY SUSPENDED-SEDIMENT LOADS FROM INSTREAM TURBIDITY DATA IN THE NORTH SANTIAM RIVER BASIN, OREGON; 1998-2000

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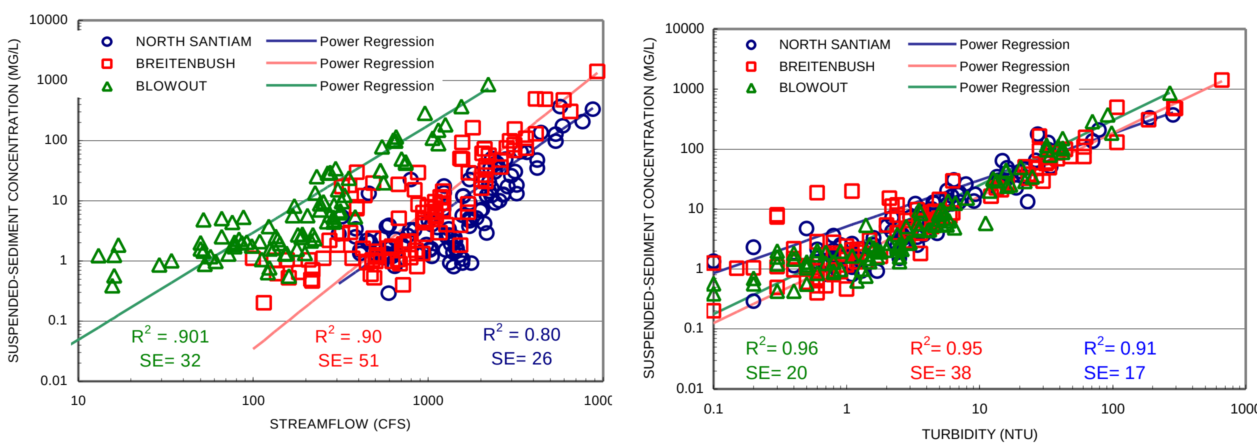
A method to estimate suspended-sediment load was developed using linear regression to correlate continuous turbidity-monitor data and suspended-sediment concentrations (SSC). In 1998, the U.S. Geological Survey began a cooperative study with the City of Salem, in Oregon to investigate the sources and dynamics of turbidity and suspended sediment within the North Santiam River-reservoir system. Three real-time sampling sites were established in October 1998 in the upper North Santiam River Basin upstream of Detroit Lake, a large, controlled reservoir, to collect water samples and continuously monitor turbidity, streamflow, water temperature, specific conductance, and pH from the three main tributary inputs to the lake. The sites were interrogated via telemetry every 3 to 4 hours, providing data in 30-minute increments. Approximately 75 equal-width-increment (EWI) and 15 dip samples (dipped and composited at vertical points in the cross-section similar to the EWI samples) were collected from October 1998 through September 2001 at the three sites.

Estimating Suspended-Sediment Concentrations from Turbidity

Regression correlations were developed for each site using the average instream turbidity values recorded during the sample collection and the sample SSCs. Estimates of SSCs were determined from the continuous turbidity data for each 30-minute reading. As a comparison, power transformed streamflow also was regressed with SSCs. The power regression equations for both turbidity and streamflow were each assessed as potential surrogates for SSC in the North Santiam River Basin (fig. 1). The turbidity and SSC plot clearly shows less scatter than the streamflow and SSC plot, as indicated by the higher coefficient of determination values (R^2) and lower standard error of estimate (SE). One reason for the higher scatter when using streamflow as the surrogate is erosion in the North Santiam River Basin, caused by glacial and landslide activity, can affect suspended sediment production disproportionately to streamflow, making streamflow unreliable for estimating SSC.

Figure 1. Comparison of streamflow and turbidity measurements versus suspended-sediment concentrations for three sites upstream of Detroit Lake (1998-2001).



Suspended-Sediment Load Calculations

Suspended-sediment loads (SSL) were computed from the estimated SSCs and corresponding streamflow data. The resulting 48 estimates per day were averaged and provided as the estimated mean daily SSL reported in tons per day (Porterfield, 1972). A graph of 1999 and 2000 annual SSLs using power equations between both instream turbidity and SSC and streamflow and SSC are presented in figure 2A. Most SSLs using streamflow as a surrogate for SSC were greater than the estimates using turbidity as the surrogate, except for Breitenbush in 2000 which was less, and Blowout which was about the same for both years, varying less than 10 percent between the surrogates. SSLs using better-fit regressions (usually not power equations) with turbidity as the surrogate were less than the SSLs using power regressions with turbidity for all sites and years (fig. 2B).

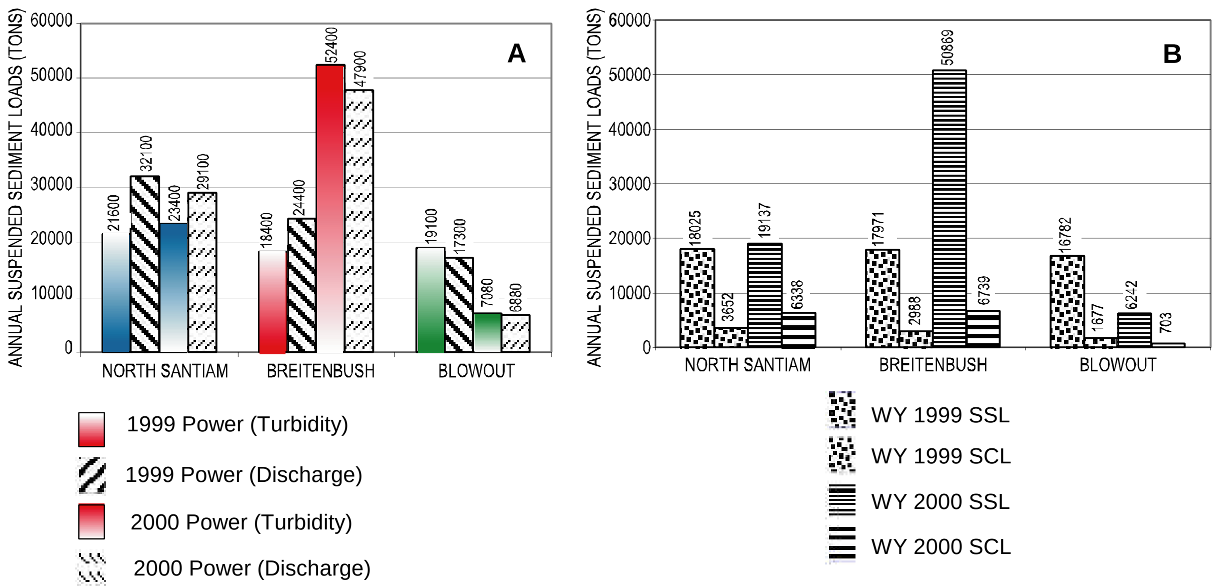


Figure 2. Estimated suspended-sediment loads using power equations with streamflow and turbidity as surrogates (A) and estimated suspended-sediment load (SSL) and suspended-clay load (SCL) using better-fit equations with turbidity as the surrogate (B).

Estimating Persistent Turbidity (Suspended-Clay Concentrations) from Turbidity

Colloidal particles held in suspension have been difficult and expensive to remove by the City of Salem slow-sand filtration system, which supplies drinking water from the North Santiam Basin. A method for predicting suspended-clay load from the persistent or residual turbidity was developed. Separate samples evaluating the change in turbidity over time were collected during the suspended-sediment sampling. Clay fraction ($\leq$ 2 µm diameter) estimates were derived from regression analysis of the turbidity decay curves and particle fall times computed using Stoke's Law (see equation 1, below).

$$\textbf{1. Fall time (in sec)} = \frac{\text{(0.1113) (viscosity at sample temp, in °C) (fall distance, in mm)}}{\text{(diameter of spherical particle, in mm)}^2}$$

The method used to determine persistent turbidity of fine sediments is similar to the pipet method for particle-size analysis (Guy, 1969), except that dispersion agents and mechanical agitation are not used, and the settling medium is native water. Aliquots are withdrawn from the same depth below the sample water surface at specific time intervals that correspond to the fall times of defined particle sizes (samples are refrigerated and settle at 4° C, the average winter temperature of Detroit Lake, Table 1).

Table 1. Fall times for persistent-turbidity samples (at 4° C)

Class Name	Particle Size Diameter	Fall Time for 2.75 cm (in lab)	Lab Aliquot Schedule	Fall Time for 70 feet (in lake)
Coarse to medium silt	.062 mm	34 seconds	Initial after shaking	2.7 minutes
Fine to very fine silt	.008 mm	32 minutes	30 minutes	6.7 days
Very fine silt to coarse clay	.004 mm	2.1 hours	2 hours	26.9 days
Coarse clay	.003 mm	3.8 hours	4 hours	47.8 days
Medium to fine clay	.002 mm	8.5 hours	8 hours	107.7 days (3.5 months)
Fine clay	.001 mm	34 hours	28-34 hours	1.2 years
Very fine clay	.0005 mm	5.7 days	5-6 days	4.7 years

Using table 1, persistent turbidity in Detroit Lake is defined as the time it takes 0.002 mm size particles (silt-clay breakpoint) and smaller to settle 70 feet in Detroit Lake to the penstock outlet port, approximately 3.5 months or longer at 4° C. In the laboratory, if we select the 0.002 mm diameter particle as the defining clay size, then the turbidity value after 8.5 hours of settling is considered the persistent turbidity value.

Persistent-Turbidity (Clay Load) Calculations

Suspended-clay loads (SCL) can be estimated using these correlations and corresponding streamflow. Regression equations were developed using the initial (or whole water) turbidity (independent variable) and the turbidity after 8.5 hours of particle settling (dependant variable). That is, the instream turbidity values are converted to persistent-turbidity values and used to compute suspended-clay concentration in the same manner as with computing suspended-sediment concentration. SCLs are computed using the suspended-clay concentrations. A comparison of annual SSLs and SCLs is shown in Figure 2B. The SCLs were 10 to 20 percent of the SSLs for all sites and years.

Data presented from this study will assist the City of Salem water treatment planners in understanding the water quality of their watershed and municipal managers in allocating drinking-water supplies from surface-water sources with persistent turbidity problems.

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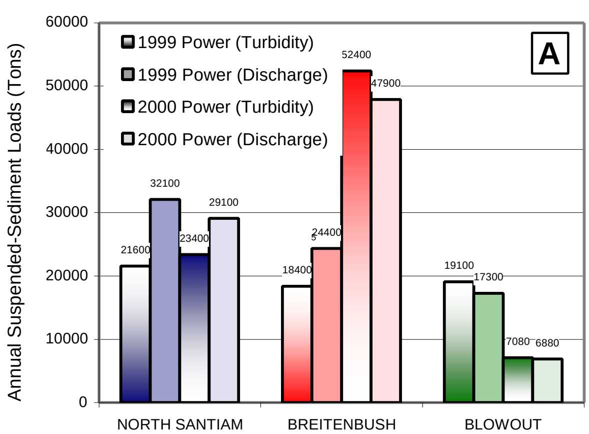

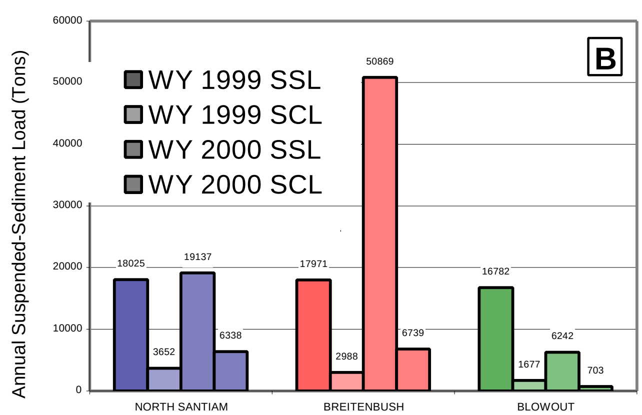

GUIDELINES AND STANDARD PROCEDURES FOR MONITORING TURBIDITY

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ABSTRACT

Water quality is continuously monitored nationwide by the U.S. Geological Survey to assess variations in the quality of surface water. Turbidity is one of the properties commonly monitored. The sensor that is used to measure turbidity requires frequent cleaning and calibration checks and computation and publication of final records can be complex.

Quality assurance of continuous turbidity data collection and publication is important to obtain consistently high-quality information. To help in this effort, the U.S. Geological Survey recently published guidelines and standard procedures for sensor site selection, test methods, calibration, and error correction, and for data computation, review, and publication processes (Wagner and others, 2000). These guidelines have evolved over the past three decades and continue to evolve as technology changes. High-quality data from turbidity sensors can be used in conjunction with chemical analyses and discharge data to estimate chemical loads and as a surrogate for suspended sediment and other water-quality constituents.

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TURBIDITY AS A SURROGATE TO ESTIMATE THE EFFLUENT SUSPENDED SEDIMENT CONCENTRATION OF SEDIMENT CONTROLS AT A CONSTRUCTION SITE IN THE SOUTHEASTERN UNITED STATES

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ABSTRACT

Study Objective: The objective of this applied research was to explore the interaction between effluent sediment concentration and turbidity for sediment controls that are currently being implemented at construction sites. Turbidity can be continuously monitored through an effluent pipe, a flume or in the receiving stream. Monitoring throughout a storm event enables making a more informed decision about the potential impact of effluent on the receiving waters. The impact of Total Suspended Solids (TSS) on various aquatic invertebrates and fish has been extensively documented for certain species. Additionally, the relationship between stream TSS and other environmental factors such as light penetration, growth of aquatic plants, temperature, etc. has been developed for some streams and lakes. If a reliable relationship can be developed between turbidity (TUR) and either suspended sediment concentration (SSC), measured in terms of the mass of sediment in the entire sample, or TSS, then turbidity can be potentially used as a surrogate enabling monitoring that can be readily accomplished at a construction site discharge point.

Regulatory Setting: Many government entities are now considering a maximum sediment concentration or turbidity value. These are often applied at the effluent point or sometimes as an in-stream increase, depending upon the type of stream receiving the sediment-laden discharge. Similarly, methodologies are currently being explored to determine the Total Maximum Daily Load (ASAE, 2002) for sediments. When setting regulations it is advisable to not only consider a maximum value based on a large design storm, e.g. 10-year, 24-hour, but to also consider a broader perspective encompassing (1) the occurrence of smaller, more frequent storms during the construction period, (2) the ability to efficiently control the sediment effluent concentration from these many smaller events, (3) the overall impact to the fluvial system and (4) the effect of land

disturbance on the complete sedimentgraph versus just the peak value. The impact to fish and aquatic invertebrates as well as aesthetic impacts are highly correlated to both sediment concentration and duration. Continuous monitoring of the entire storm event via a turbidity meter can afford greater flexibility in developing meaningful regulations.

Sediment Controls Analyzed: The database for this applied effort was obtained from an active construction site north of Atlanta, Georgia (Warner and Collins-Camargo, 2001). The objectives of the overall study were to design, implement and monitor a system of erosion and sediment controls that would be cost-effective and environmentally-efficient, integrate the riparian zone as a secondary synergistic passive treatment system, and to influence management decisions with respect to timing of installation of controls and construction. The types of controls monitored, for the effluent portion of this research, include: (1) two external sand filters receiving discharge from sediment ponds, (2) a floating siphon that discharged from a multi-chamber (in series) sediment basin and (3) a perforated riser installed in one sediment basin and one seep berm system.

Sand filters were employed to further reduce the effluent sediment concentration below that which is normally discharged through a sediment pond. The sand filter was an intermediate treatment process inserted between the sediment basin and a forested riparian zone. It was nominally 37-m^2 in surface area and constructed with a 15-cm depth of river-washed sand overlying an 8-cm gravel bed. The floating siphon was installed in one sediment basin and passively decanted the upper 5 to 15-cm of surface water once the first flush of sediment was retained below the outlet crest of the siphon. Perforated risers were installed in one sediment basin and one chamber of a seep berm. A seep berm is essentially an elongated basin with a large number of passive dewatering outlets along its length. Discharge from the seep berm spreads through a forested riparian zone where it partially or totally infiltrates prior to entering a stream. The Sediment, Erosion and Discharge by Computer Aided Design (SEDCAD) model was used for the design of the system (Warner and Schwab, 1998).

Turbidity Function of Particle Size and Sediment Concentration: Ideally, the prediction of turbidity would be linked with the effluent sediment concentration and the effluent particle size distribution being discharged from a sediment control. Knowing these values over the entire discharge time for the sediment control should enable the best prediction of turbidity. SEDCAD 4 has the capability to predict the complete sedimentgraph and a temporal-composite effluent particle size distribution. Controls such as the sand filter and the floating siphon have very high efficiencies resulting in low sediment concentrations. Sampling from these devices yields very small quantities of sediment. Suspended sediment concentration and turbidity were determined for 92 samples. To obtain a sufficient quantity of sediment for particle size distribution analysis, composite samples were used. Since the number of composite samples was too small to reliably be used in developing a methodology based on an effluent particle size distribution, a linkage between a predictive equation and functionality (efficiency) of sediment controls was developed. Some sediment controls inherently perform better than others. The monitoring period was during active construction, June 29 through Sept. 22, 2000. The resulting ratios of turbidity to suspended sediment concentration in the effluent from different erosion controls were based on 77 automatic pumped samples and 15 grab samples that were obtained from three storm events.

Turbidity – Suspended Sediment Concentration Predictive Relationships: To explore potential relationships between turbidity (TUR) and suspended sediment concentration (SSC), the ratio of TUR/SSC was calculated for all automatic and grab samples. Within a sample set from a given event and sediment control, the resulting ratios were summed and averaged to determine a single value representing that sediment control type for a given storm event. Specifically, a weighted ratio was calculated in which the average ratio of the automatic samples for each event is multiplied by the total number of samples in each event data set and then individual grab samples are added into the data set. The summation value for both automatic and grab samples is simply divided by the total number of samples taken from the sand filter resulting in a ratio of 1.7:1 (TUR/SSC) for the sand filter. Similarly, the resultant TUR/SSC ratios for the floating siphon and perforated riser are 1.7 and 1.4, respectively. To predict the effluent turbidity from the suspended sediment concentration, it is only necessary to multiply the concentration, which is the output of SEDCAD, by the ratio of 1.7, 1.7 and 1.4 for the sand filter, floating siphon and perforated riser, respectively. It should be noted that these are very preliminary values for the specific soils tested in the Atlanta area and these ratios may not be applicable to other soils or certainly not to other sediment controls.

Discharge from the sand filter and the floating siphon contains a higher fraction of finer grain particles than the perforated riser due to the filtering and skimming actions of these devices. The sand filter and floating siphon consequently have a lower effluent sediment concentration than the perforated riser. The derived TUR/SSC ratios of these two devices are higher than the perforated riser due to the higher contribution per unit mass of the finer grain particles. The perforated riser discharges sediment throughout its vertical height wherever there is an outlet hole. Hence, there is a higher potential for sand and/or larger silt particle release than for the floating siphon.

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SURROGATE TECHNIQUES FOR SUSPENDED-SEDIMENT MEASUREMENT

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ABSTRACT

Introduction: Physical, chemical and biological sediment damage in North America has been estimated to be up to \$16 x 10^9 annually (Osterkamp et al., 1998). Accepted methods of collecting sediment data are labor intensive, expensive and may be of unknown accuracy due to the large spatial and temporal variability associated with the transport of suspended sediment. To fill this data void, automatic, cost-effective techniques are needed to collect high quality data on suspended sediment load.

The following paragraphs describe, in no particular order, methods for measuring suspended-sediment concentration. The operating principle of each method is briefly described and, where the information was available, the particle size and concentration ranges are included. For more information and additional references, see Wren et al., 2000.

Optical backscatter (OBS): Infrared or visible light is directed into the sample volume where a portion of the light will be backscattered if particles are in suspension. A series of photodiodes positioned around the emitter detect the backscattered light. An empirical calibration is used to convert backscatter to concentration. The measurement volume varies according to turbidity but is on the order of several cubic centimeters. OBS devices are readily available and relatively inexpensive. The particle size range for best operation is 200-400 µm, and concentrations may range up to 100 g/L. (Black & Rosenberg, 1994)

Optical transmission: Light is directed into the sample volume where sediment will absorb and/or scatter a portion of the light. A sensor located opposite the light source measures the attenuation of the light beam. The sediment concentration is determined using empirical calibration information. The size of the measurement volume will vary according to the geometry of the device. Optical transmission devices are relatively inexpensive. (Clifford et al. 1995)

Focused beam reflectance: A laser beam focused to a very small spot (<2 μm^2) in the sample volume is rotated very quickly (many times per second). As it rotates, the beam encounters particles that reflect a portion of the beam. The time of this reflection event is used to determine the sizes of particles in the path of the laser. The particle size range is 1-1000 µm and the concentration range is 0.010-50 g/L. Few references to this type of device are found in the literature. (Phillips and Walling, 1995)

Laser diffraction: A laser beam is directed into the sample volume where particles in suspension will scatter, absorb, and reflect the beam. Scattered laser light is received by a detector or array of detectors that allow measurement of the scattering angle of the beam. Particle size can be calculated from knowledge of this angle. By basing concentration measurements on measured particle sizes, particle size dependency is eliminated. The optical path length is either 2.5 or 5 cm, the particle size range is 1.25-250 µm or 2.5-500 µm, and the concentration may range up to about 5 g/L. These devices are relatively expensive and are readily available. (Agrawal and Pottsmith, 1994)

Acoustic: Short bursts (≈10 µs) of high frequency sound (1-5 MHz) emitted from a transducer are directed towards the measurement volume. Sediment in suspension will direct a portion of this sound back to the transducer. The strength of the backscattered signal allows the calculation of sediment concentration. Backscatter amplitude depends on the concentration, particle size, and acoustic frequency. This can be exploited by using multiple frequencies to determine both particle size and concentration. Acoustic devices measure the concentration in a range-gated vertical profile of 1-2 m in depth. Using typical ultrasonic frequencies, the particle size range is approximately 62-2000 µm and concentrations may range up to 30 g/L, although the available sampling depth will be limited at high concentrations. Acoustic technology is still under development. Appropriate hardware is available, but there is no commercially available hardware/software system to acoustically measure suspended-sediment concentration profiles. (Thorne et al., 1991; Hay and Sheng, 1992)

Nuclear: This technique relies on the attenuation or backscatter of radiation, usually X or gamma rays, by sediment particles. An empirical calibration is used to convert backscatter to concentration. The concentration range is approximately 0.5-12 g/L. The measurement volume will depend on instrument geometry. Nuclear devices are not readily available, and there is little evidence that these devices are currently being used for fluvial sediment measurement. (McHenry et al., 1967)

Spectral reflectance: This technique is based on the relationship between the amount of radiation, generally in the visible or infrared range, reflected from a body of water and the properties of that water. The radiation is measured by a hand held, airborne, or satellite based spectrometer. The size of the measured area is much larger than the other devices discussed here and may range from m^2 to km^2 of the surface of the water body. This technique is better suited to marine environments where large areas are under observation or in other situations where concentration variations over large areas are of interest. (Novo et al., 1989)

Digital optical: A charge-coupled device (CCD) records the sediment/water mixture in-situ. This recording can be analyzed so that, among other things, the size and concentration of suspended-sediment particles can be determined. It can also be used to visually confirm the nature of the sediment. Recent improvements in computer and imaging technology should expand the usefulness of this technology. The device is under development in the laboratory with plans to expand into field application. The size of the measurement volume will be dependant on light penetration in the water. (Gooding, 2001)

Vibrating tube: Water is routed through a vibrating tube in a stationary housing located either on the stream bank or in the stream. The frequency of the vibration will be affected by the density of the water in the tube and can be used to determine the sediment concentration. However, several other factors such as temperature, debris on the tube walls, and dissolved solids concentration also affect the vibration frequency. All of these must be accounted for to obtain an accurate measurement. The device works best in concentrations over 1 g/L. (Skinner, 1989)

Differential pressure: A differential pressure transducer may be used to determine differences in the specific weight of sediment bearing water versus water nearer the surface with lower concentrations. This difference in pressure can be used to determine the average suspended sediment concentration between the two inlets of the differential pressure transducer. The size of the measurement volume will depend on the separation of the pressure inlets of the differential transducer. The concentration range is dependant on the sensitivity of the transducer. The hardware for this device is readily available and relatively inexpensive. Changes in temperature gradient, turbulence, and dissolved solids concentration will affect measurements. (Lewis and Rasmussen, 1996)

Impact sampler: The sampler works on the principle of momentum transfer. The impact rate of sediment particles hitting a sensor is measured. The detected impact rate is dependent on the mass, velocity, and angle of particle impact. Few references to this type of device are found in the literature. There are many technical problems with the use of this device in a fluvial environment. (Salkield et al., 1981, as referenced by Van Rijn and Schaafsma, 1986)

Conclusion: At the present time many options exist for the measurement of sediments suspended in water. All of the techniques reviewed above, however, suffer from limitations that render the techniques inadequate in some environments. Perhaps the best option for suspended sediment measurement remains a hybrid approach that relies on more than one technique and maintains a manual component. Continued improvements in technology will undoubtedly translate into improved methods to collect suspended sediment data in the future.

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ISSUES RELATED TO USE OF TURBIDITY MEASUREMENTS AS A SURROGATE FOR SUSPENDED SEDIMENT

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ABSTRACT

This abstract summarizes issues related to the use of turbidity measurements as a surrogate for suspended sediment. Issues discussed are: (1) methods used for measurement, (2) wavelength of light, (3) detector orientation, (4) standards for calibration, (5) grain-size and color effects, and (6) data reporting.

Turbidity Definition and Methods: Turbidity can be defined as a decrease in the transparency of a solution due to the presence of suspended and some dissolved substances, which causes incident light to be scattered, reflected, and attenuated rather than transmitted in straight lines; the higher the intensity of the scattered or attenuated light, the higher the value of turbidity. Turbidity can be expressed in nephelometric turbidity units (NTU). Depending on the method used, the turbidity units as NTU can be defined as the intensity of light at a specified wavelength scattered or attenuated by suspended particles or absorbed at a method-specified angle, usually 90 degrees, from the path of the incident light compared to a synthetic chemically prepared standard.

Currently approved methods for use by USGS include USEPA Method 180.1, (U.S. Environmental Protection Agency, 1979), ISO 7027 (International Organization for Standardization, 1999), GLI Method 2 (Great Lakes Instruments, Inc., 1992), ASTM Method (American Society for Testing of Materials, 2000), and Standard Methods (SM) (Clesceri and others, 1998). ASTM and SM methods are similar to USEPA Method 180.1 and are not discussed here. Because results from these methods typically are all reported in NTU, it is important that the method of measurement and type of instrument be identified when storing or reporting the data.

Discussion: As shown in table 1, turbidity methods, standards, reporting of units, and instruments are not identical. For each applicable method, the range in turbidity measurements does not cover all values for natural water. Because turbidity is an apparent optical property of water, it is likely that dilution of samples would not result in a physically reproducible measurement (Davies-Colleys and Smith, 2001). Light wavelengths are different, and color can affect the measurements. Different instruments may use forward or backscatter detection devices and multiple incident light sources and detection devices at different orientations that can compensate for the effects of color and grain size (Sadar, 1998). The detector-orientation measurement angles can be wide (USEPA Method 180.1) or narrow (ISO 7027). Therefore, measurements of the same water by different methods and different instruments are not likely to yield similar values.

Table 1.—Comparison of selected turbidity methods.
[NTU, nephelometric turbidity units; FTU, formazin $(C_2H_4N_2)_n$ turbidity units; FAU, formazin attenuation units; nm, nanometers; cm, centimeters]

Characteristic	**USEPA Method 180.1 (nonratio mode)**	**ISO Method 7027 (diffuse radiation)**	**ISO Method 7027 (attenuated radiation)**	**GLI Method 2**
Use of data	Drinking water	Drinking water	Wastewater	Drinking water
Range of method	0-40 NTU (dilution permitted)	0-40 FTU (diluted permitted)	40-4,000 FAU	0-40 NTU (dilution permitted)
Light source	Tungsten Lamp	Photodiode	Photodiode	Photodiode
Wavelength	400-600 nm,	860 nm	860 nm	860 nm
Spectral bandwidth	Not specified	60 nm	60 nm	60nm
Detector orientation measurement angle	90+/-30 degrees	90 +/-2.5 degrees	90 +/- 2.5 degrees	Two sources, two detectors at 90 +/- 2.5 degrees
Aperture angle	Not specified	20-30 degrees	20-30 degrees	unknown
Path length	Less than 10 cm	Less than 10 cm	Less than 10 cm	Less than 10 cm
Primary standard	Formazin polymer	Formazin polymer	Formazin polymer	Formazin polymer
Secondary standards	Polymer microspheres	Polymer microspheres	Polymer microspheres, cubes, or filaments	Polymer microspheres

Primary formazin standards can be unstable and have a wide variability in particle size and accompanying light-scattering characteristics (Papacosta, 2002). Secondary standards using other polymers may have a more defined (0.02 to 0.2 micron) size range, but can have different instrument and manufacturer response readings relative to formazin (Papacosta, 2002). The nephelometric design, with a detector at 90 degrees, is optimized for particle sizes of 1.0 micron or less (Papacaosta, 2002), which is much smaller than possible particles sizes of sediment.

The color of water can cause a negative bias in measurements by attenuating the light in colored samples using USEPA Method 180.l. The color of the darkened (more "black" colored using Munsell soil charts) sediment particles has been shown to substantially affect measurements with optical backscatter meters, and it is expected that nephelometers would give a similar negative bias in measurements depending on the mineralogy of the sediment (Sutherland and others, 2000). All nephelometers can be affected by the grain size and orientation of the sediment in a sample (Sadar, 1998).

Storage of turbidity data and comparability of measurements are concerns, especially when developing a relation with suspended sediment. Because instruments of widely different configurations, methods, and potential color effects are used and commonly report in NTU, it is not likely they will yield similar turbidity values. However, the ability to measure turbidity continuously and to relate these measurements to suspended sediment and sediment-associated constituents, such as fecal coliform bacteria (Christensen and others, 2000), is a valuable tool in describing transport of these constituents.

Research and standard protocols are needed in the following areas in order to improve the use of turbidity as a reliable surrogate for suspended sediment:

(1) Data storage needs to identify the method and instrument used. A suggested reporting convention would include the method, light wavelength, detector orientation, and number of sources and detectors. Data possibly could be reported as a beam attenuation coefficient value rather than relative to an arbitrary standard of formazin.

(2) The effects of grain size, color, and mineral composition need to be defined and documented. These effects probably can be calibrated with suspended-sediment samples collected over the range in turbidity conditions at the same time that continuous turbidity measurements are made.

(3) A priority should be given by standard organizations to approve a reproducible method and instrument design that will provide reliable readings for different water types—drinking water, natural water, and wastewater. A draft certification program for continuous turbidity monitors written by the United Kingdom Environment Agency (2001) is under review to improve instrument/method comparability.

(4) Comparisons need to be done between different turbidity meters and methods and samples collected and analyzed for suspended-sediment concentration and grain size.

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