Impact of Sedimentation on Biological Resources:
A Sediment Issue White Paper Report
prepared for the State of Kansas

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Executive Summary

In summary, the great body of scientific literature strongly indicates that sediment entering into aquatic ecosystems can cause the loss or impairment of fish, macroinvertebrates, and other aquatic organisms. However, our current ability to quantify relationships between aquatic sediment variables and the aquatic biota of Kansas water resources is limited by available data and the complexity of direct and indirect linkages between resource components. At present, turbidity appears to be the best indicator of suspended sediment for defining biological impairment in flowing water systems. Better coordination of existing and new research, along with the use and analysis of well-selected indicators of suspended and deposited sediment and ecosystem function, will allow for the identification and quantification of sediment impacts on aquatic ecosystems in Kansas.

Recommendations for Future Research

Based on reviews of available literature and recent research, the following recommendations are made to guide future research regarding the impact of sedimentation on biological resources:

1. **Adopt a multi-disciplinary approach.** The complex nature of sedimentation regularly spans areas of hydrology, geomorphology, aquatic ecology, water chemistry, soil and sediment chemistry, and landscape-level phenomena, such as urban development and best management practices for agriculture. Currently, sediment studies are usually approached from only one or two of these points of view.

2. **Observe both sediment loading and biological response.** Surprisingly, little overlap exists between datasets on sediment loading and biological indicators. Future studies should emphasize concurrent collection of physical, chemical, geomorphic, and biological data to gain a more comprehensive understanding of their complex and integrated relationships.

3. **Begin with gaged locations.** Often, sediment loading rates are the limiting factor in a multi-disciplinary suite of sediment data. To better estimate the impacts of sediment on biological resources, those resources should be evaluated at locations where sediment loading data is available. Typically, stream gaging stations provide available loading data or opportunities to calculate sediment loads.

4. **Determine reference conditions for sedimentation.** In order to evaluate the extent of impact on biological resources (*i.e.,* how "good" or "bad" a site is), a condition of high quality must be established for comparison. Currently, there is little agreement between hydrologic, geomorphic, and biological definitions of this reference condition, making assessment of sediment – biological quality interactions problematic.

5. **Consider the regional context.** In many cases, the full range of geomorphic, hydrologic, and biological characteristics of certain aquatic systems are not present within Kansas alone. However, such a range may be observable at a regional or multi-state scale. For example, reference conditions for certain Kansas systems may not occur within the state boundaries. Study of related systems in other states may then be appropriate.

6. **Record both intensity and duration of sedimentation events.** Research has shown that an ecotoxicological model (*i.e.,* one that considers both the amount of sediment and the length of time during sediment exposure) better predicts the impacts of sedimentation. However, for various reasons, only concentration (intensity) of sedimentation is reported in most studies.

7. **Distinguish between natural and induced sedimentation.** Some low gradient, high turbidity systems in the Central Plains have elevated natural sediment loads as an ambient condition. Significant study may be required to discern impairment in these systems.

8. **Use advanced statistical techniques.** The interactions between response and predictor variables are complex in ecological systems, and especially so with regard to the impacts of sediment on aquatic biota. Since the rate of variation in response may be unequal across data distributions, the statistical procedures used to analyze the response data must be robust to such changes in rate of variation. Techniques such as analysis of covariance, quantile regression, and structural equation modeling may be appropriate.

Introduction

A major disturbance threatening the ecological integrity of streams and rivers throughout the United States is increased sedimentation and sediment loading. Both "clean" and "dirty" sediment (*i.e.,* sediment either uncontaminated or contaminated by other compounds, respectively) can directly and indirectly affect the structure and ultimately the function of all aquatic ecosystems (Figure 1). Anthropogenic activities including urbanization, agriculture, and the alteration of riparian habitat and flow regimes have increased the concentrations and rates at which sediments enter lotic systems (Wood and Armitage 1997; Zweig and Rabeni 2001). As a result, sedimentation is currently listed as one of the most common stream impairments within the United States (USEPA 2000, 2004). While the effects of sedimentation are widespread, a comprehensive theory of these effects on benthic communities does not currently exist (Zweig and Rabeni 2001).

Streams and rivers provide many important services to humans including water for drinking, irrigation, waste dilution, power generation, transportation, recreation, and fish for harvest and sport (Allan 1995). Two major disturbances that threaten the ecological integrity of waterways are increased siltation and sediment loading. Instream sediments come from two sources: runoff from surrounding areas and erosion of sediments from the channel itself. The complex interaction of streams and the surrounding landscape can be characterized to a large extent by describing these sediment movements. Erosion and deposition of sediments helps determine many stream habitat features such as channel depth, shape, substrate, flow patterns, dissolved oxygen concentrations, adjacent vegetation, and aquatic communities.

Sedimentation is a natural process that occurs in most aquatic ecosystems, and sediments provide the primary food source for a number of filtering macroinvertebrates (Waters 1995; Wood and Armitage 1997). However, human activities such as urbanization, agriculture, and the alteration of riparian habitat and flow regimes have increased the concentrations and rates at which sediment enters streams and rivers (Wood and Armitage 1997; USEPA 2000; Zweig and Rabeni 2001). As a result, sedimentation has been reported to be the second most common stream impairment in the U.S., occurring in almost one-third of the river and stream miles that have been assessed by the United States Environmental Protection Agency to date (USEPA 2004). Sedimentation at higher than normal rates has been shown to reduce or impair habitat and production in wetlands as well (Gleason and Euliss Jr. 1998; USEPA 2002; Gleason et al. 2003). Similar habitat reduction in lakes has also been observed, with 10% to 40% decreases in conservation-pool water storage capacity in several Kansas reservoirs. If current sedimentation rates remain the same, the design sediment pools of these reservoirs will be filled by the 2020's (Juracek 2006).

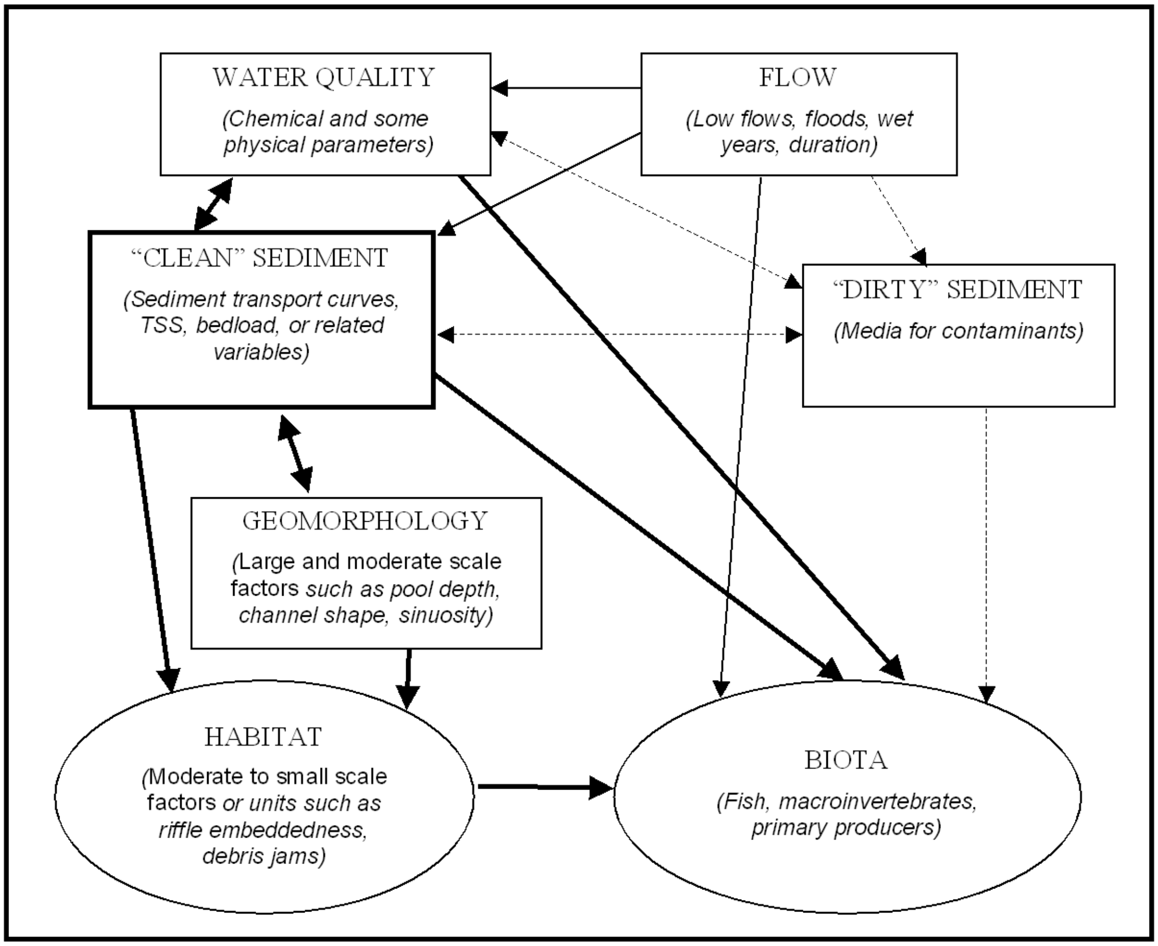


Figure 1. Conceptual framework showing interactions of sediment in aquatic ecosystems. Bold lines indicate direct and indirect effects of "clean" sediment on biological resources and represent the emphasis of this report. Dashed lines indicate relationships that are mediated by the nature of the contaminants in "dirty" sediment. "Clean" sediment is defined as sediment free from additional contaminants (*e.g.,* volatile organics, metals, or other toxic compounds), such that any impacts are caused by the nature and concentration of the sediment particles themselves. "Dirty" sediments are those sediments that harbor such contaminants. For the initial understanding of the impacts of sedimentation, primary concern is given to "clean" sediments, since "dirty" sediment issues are subject not only to the nature of the sediment, but the nature of the contaminant as well.

State of the Art: Review of Science to Date

Brief Literature Review

In general terms, sedimentation has been widely studied, though most research has concentrated on cold water systems. Representative works include basic research studies (Luedtke and Brusven 1976; Erman and Ligon 1988; Lisle and Lewis 1992; Goodin et al. 1993; Maund et al. 1997; Simon et al. 2003; Dodds and Whiles 2004), literature reviews (Cordone and Kelley 1961; Foess 1972; Newcombe and MacDonald 1991; Doisy and Rabeni 2004), and books (Ford et al. 1990; Waters 1995). Excluding the effects of sediment-mediated contaminants (“dirty sediments”), specific “clean sediment” effects such as physical light interruption, physical smothering of organisms, and physical coverage of sites used for germination, feeding, spawning, etc. have all been documented, as have biotic effects such as direct mortality, reduced fecundity, reduced resistance to disease, and inhibited feeding, growth, and reproduction. Previous reviews by Newcombe and MacDonald (1991) and later Doisy and Rabeni (2004) have categorized these direct effects of sedimentation on aquatic life into three categories: *lethal effects*, which cause direct mortality of individuals, cause reductions in populations, or damage ecosystem capacity for production; *sublethal effects*, which injure organism tissues or cause physiological stress, both without causing mortality; and *behavioral effects*, which alter the kind and/or pattern of activity of affected organisms. A brief sampling of studies in each of these categories appears as Table 1.

Both suspended and deposited sediments can impact aquatic ecosystems (Waters 1995; Zweig and Rabeni 2001; Richardson and Jowett 2002). For example, increases in suspended solid concentrations can lead to reductions in primary production (Van Nieuwenhuyse and LaPerriere 1986), the disruption of feeding and respiration rates of macroinvertebrates (Lemly 1982), and reductions in growth and feeding rates of many stream fish (Wood and Armitage 1997). Similarly, increased sediment deposition can reduce the complexity of stream habitat (Allan 1995), and smother aquatic organisms including macroinvertebrates, fish, and macrophytes (Waters 1995; Wood and Armitage 1997). As previously noted, no comprehensive theory of these effects on benthic communities currently exists (Zweig and Rabeni 2001).

A number of potential sediment and erosion indicators have been suggested for use in TMDL (Total Maximum Daily Load) development (USEPA 1998). These include water column indicators (i.e. suspended sediment, bedload sediment, and turbidity), streambed indicators (i.e. streambed particle size and embeddedness), and riparian indicators (i.e. buffer size and vegetation community composition). Several biological indicator groups have also been noted to respond to sediment-related impacts (Luedtke and Brusven 1976; Culp et al. 1986; Richards and Bacon 1994; Rier and King 1996; Birtwell 1999). Within the Central Plains region, relatively little has been documented about the linkages between sediment indicators and biological indicators both within and between streams (except see Whiles and Dodds 2002).

Table 1. Selected summary of data on effects of suspended sediment on aquatic invertebrates after Newcombe and MacDonald (1991). Citations of original sources remain with the original review.

Taxon	Exposure and Category			Effect	Original Source(s)
	category	mg/L	hours		
Zooplankton	sublethal	24	0.15	reduced capacity to assimilate food	McCabe and O'Brien (1983)
Cladocera	lethal	82-392	72	survival and reproduction harmed	Robertson (1957); from Alabaster and Lloyd (1982)
Cladocera and Copepoda	lethal	300-500	72	gills and gut clogged	Stephan (1953); from Alabaster and Lloyd (1982)
Macroinvertebrates	lethal	53-92	24	reduction in population size	Gammon (1970)
Zoobenthos	lethal	10-15 >100	720 672	reduction in standing crop	Rosenberg and Snow (1977)
Benthic invertebrates	lethal	16-32	1,440	reduction in standing crop	Slaney et al. (1977)
Benthic invertebrates	lethal	62-278	2,400	53-80% reduction in population	Wagener and LaPerriere (1985)
Benthic invertebrates	lethal	261-390	720	reduction in population size	Tebo (1955)
Stream invertebrates	lethal	130-25,000	8,760	85-100% reduction in population	Nuttall and Bielby (1973)

In addition to the predominating study of cold water systems, the majority of stream research in general (Dodds et al. 2004), and sedimentation research in particular, has been carried out in systems with either naturally high gradient or naturally low turbidity. However, aquatic systems in the Central Plains of the United States, especially those in agriculturally dominated areas like the Central Great Plains, Western Cornbelt Plains, and, to a lesser extent, the Central Irregular Plains, are generally characterized as warm water, low-gradient, high-turbidity systems, though some evidence from the earliest reports suggests that many Central Plains streams that have been turbid for the past 100 years may have been clear prior to widespread plowing in the region (Matthews 1988).

Compounding the issue, many systems in the Central Plains are also historically characterized with sand as the primary substrate. Such overlap of areas with induced sediment loading and areas with historically high natural sediment loading requires significant regional testing of widely held theories on the biological effects of sedimentation, in order to determine their relevance for ecosystems in the Central Plains.

Given this need for regional testing and the current lack of a comprehensive theory of sediment impacts, a conceptual framework for the interactions of sediment on lotic ecosystems was developed by a sediment workgroup sponsored by USEPA Region VII for scientific investigation of potential sediment indicators and impacts to flowing water ecosystems in the Central Plains (Figure 1). This framework provides the hypothesized linkages between both “clean” and “dirty” sediments, geomorphology, flow regimes, chemical and physical water quality parameters, habitat effects, and biotic components, including primary producers, macroinvertebrates, and fish. It is believed that the pathways between these ecosystem components identify the basic direct and indirect effects that exist between sediment related stressors and aquatic biota.

Recent Regional Findings

In order to analyze complex systems, it is often necessary to construct linked individual relationships to depict indirect effects. For example, the effects of “clean” sediment (*i.e.,* sediment only, without associated nutrient or chemical loading considerations) can be modeled by relating sediment loads (inorganic solids) to an indicator (total suspended solids), then relating that indicator to another (turbidity), and so forth. Where possible, relevant analyses are therefore presented in terms of the previously discussed theoretical framework for interactions of sediment in lotic ecosystems (Figure 1).

Clean Sediment – Water Quality Links

Using data from multiple dates and sites in 16 small watersheds (CPCB 1994), CPCB has found that inorganic suspended solids (ISS) were highly correlated with total suspended solids (TSS) throughout the Western Cornbelt Plains, as was turbidity (Figure 2). Furthermore, using a much larger regional database, the USEPA Region VII Regional Technical Assistance Group (RTAG) has also found turbidity to be highly correlated with TSS throughout the Central Plains and across ecoregions (USEPA Region VII Regional Technical Assistance Group (RTAG) 2006) (Table 2). Dodds and Whiles (2004) also found high correlation ($R^2 = 0.89$) between turbidity and TSS using nationwide data. Since turbidity is highly correlated with TSS and by extension, inorganic suspended solids, it appears that turbidity measurements can be used as a surrogate indicator for suspended “clean” sediment in streams in the Central Plains.

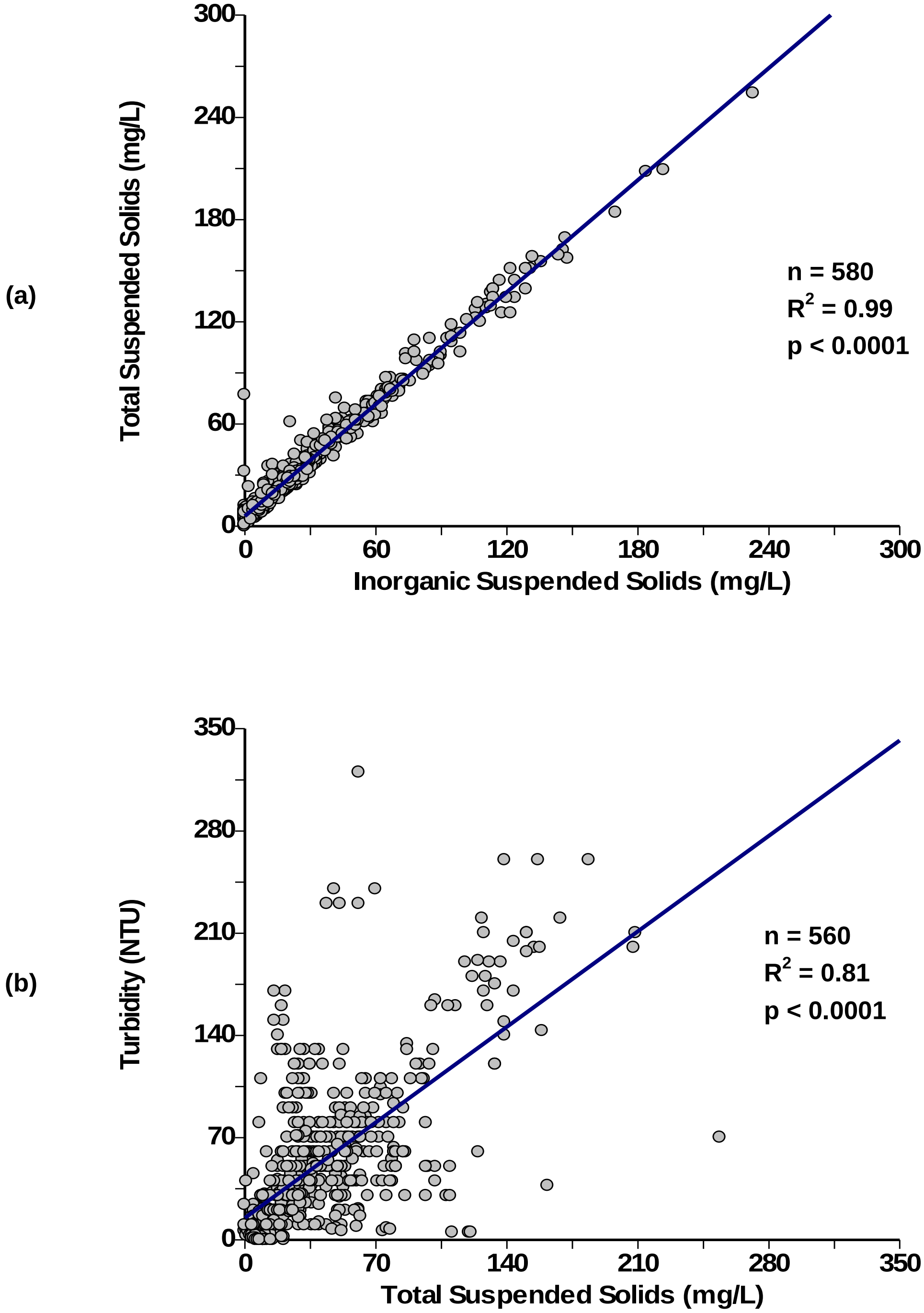


Figure 2. Scatter plots showing least-squares regression lines for (a) inorganic suspended solids (ISS) and total suspended solids (TSS) and (b) TSS and turbidity for five sites in each of six Western Corn Belt streams sampled five to nine times from 1992 to 1994. Streams included three northeastern Kansas streams (North Elm, Straight, and French Creeks) and three eastern Nebraska streams (Silver, Cedar, and Wolf Creeks) (CPCB 1994).

Table 2. Turbidity conversion factors for given total suspended solids (TSS), based on robust regression of data compiled by the USEPA Region VII Regional Technical Assistance Group (2006). Turbidities were measured in NTUs, and TSS in mg/L. All relationships shown were highly significant ($p < 0.001$).

Ecoregion	Turbidity Conversion	R^2	Number of Observations
Central Irregular Plains (CIP)	0.428975 * TSS + 1.303919	0.983551	4,047
Central Oklahoma/Texas Plains (COT)	0.6085414 * TSS – 0.8589433	0.930430	86
Western Cornbelt Plains (WCB)	0.3176404 * TSS + 2.261994	0.988871	1,707
Flint Hills (FH)	0.3361894 * TSS + 1.990286	0.991641	2,254
Central Great Plains (CGP)	0.3804407 * TSS – 0.1230823	0.977409	4,715
Southwestern Tablelands (ST)	0.4363467 * TSS – 1.290563	0.976931	525
Ozark Highlands (OH)	0.4493134 * TSS – 0.3472661	0.983601	209
Western High Plains (WHP)	0.4174888 * TSS – 0.7488838	0.989055	397
All Ecoregions Combined	0.3899303 * TSS + 0.6833382	0.977163	13,854

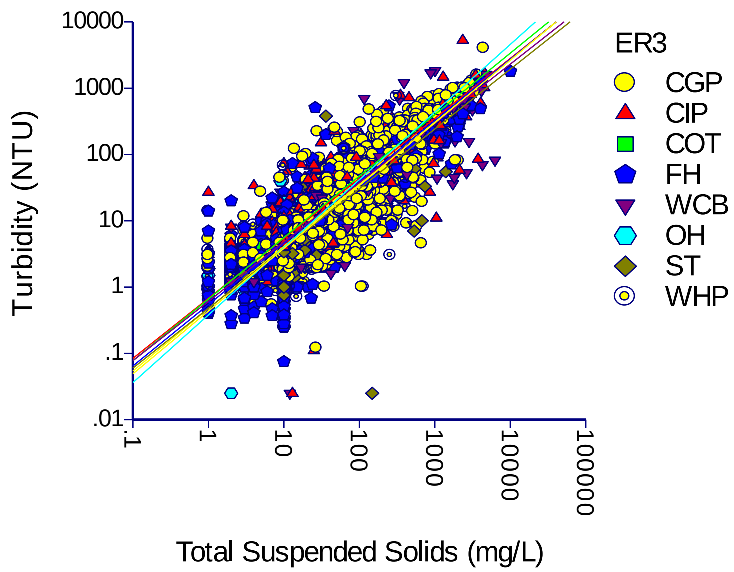


Figure 3. Scatter plot of TSS and turbidity (NTU) with linear trend lines for those portions of eight level 3 ecoregions (Omernik 1987) occurring within US EPA Region VII (Iowa, Kansas, Nebraska, and Missouri). The eight ecoregions and their models for linear regression appear in Table 2.

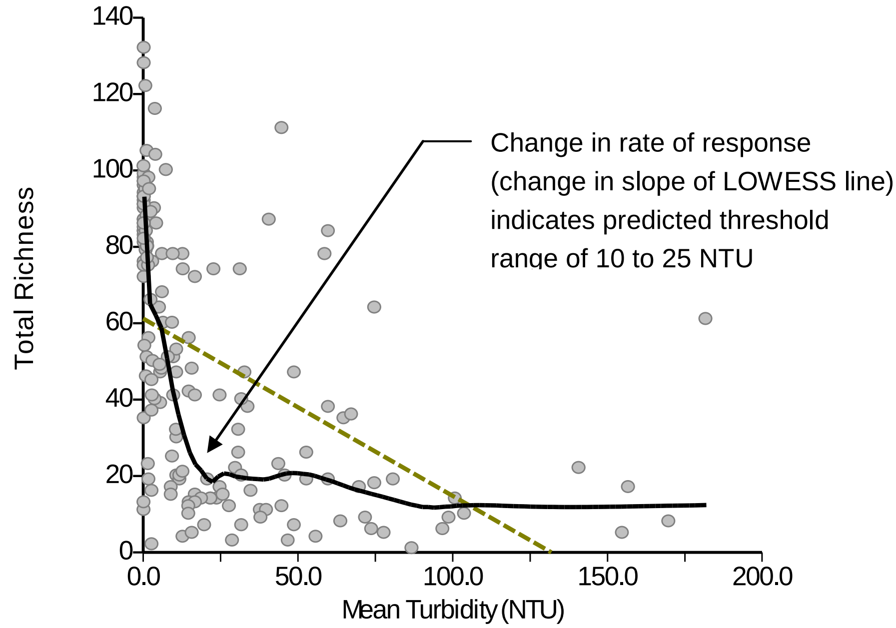
Clean Sediment/Water Quality – Biota Links
Using this regional database, several relationships have been observed between sediment and biotic indicators. Several models of total macroinvertebrate richness versus turbidity were developed using RTAG data (Figures 4, 5, 6). According to these models, macroinvertebrate richness significantly declines with increasing turbidity. Locally weighted least squares (LOWESS) regression and piecewise linear regression models of regional data suggest a threshold range of turbidity between 10 and 25 NTU (Figure 4). The sharp break in the LOWESS smoothed line (Figure 4a) and in the piecewise linear regression (Figure 4b) both indicate an area of rapid decrease in macroinvertebrate taxa richness (from 60 to 20) with increased turbidity. These thresholds represent levels of turbidity above which macroinvertebrate richness drops very little, presumably because some ecological limit of turbidity impairment has already been reached. Such threshold ranges are often used as the basis for benchmarks and criteria of other types of impairment (*e.g.,* nutrient loading).

RTAG data further show that taxa richness of Ephemeroptera, Plecoptera, and *Trichoptera* (EPT taxa richness), sediment sensitive macroinvertebrate richness, and percent sensitive fish taxa also decline with increasing turbidity (Figure 5). Data collected during the National Wadeable Streams Assessment (USEPA 2004) from 125 sites in Kansas, Nebraska, Iowa, Missouri, and Oklahoma showed similarly decreasing trends in richness of total macroinvertebrates and EPT taxa with increasing TSS (Figure 6). Similarly, EPT taxa richness and macroinvertebrate scraper richness also decreased with increasing percentage of fine substrates (*i.e.,* silt or mud, but not including sand), though macroinvertebrate shredder richness and macroinvertebrate predator richness were generally unaffected (Figure 7).

Habitat – Clean Sediment/Biota Links
Based on data from the National Wadeable Streams Assessment (USEPA 2004), mean percent embeddedness was positively correlated with turbidity ($R^2 = 0.12$, $p=0.0001$, $n=125$) and percent fines ($R^2 = 0.26$, $p < 0.0001$, $n = 125$) (Figure 8). Total macroinvertebrate richness and EPT taxa richness both appeared to decline with increased embeddedness of substrates (Figure 9). A significant ($p = 0.0008$) linear regression was found between macroinvertebrate richness and percent embeddedness, but the amount of variance explained by the model was limited ($R^2 = 0.13$).

Geomorphology – Clean Sediment/Habitat/Biota
Of the 62 geomorphic variables measured, 53 were numeric. Aside from the high number of cross-correlations (Parr 1999) within the geomorphic data (*e.g.,* cross-sectional area, channel width, and channel depth), few correlations were observed between geomorphology and other stream ecosystem variables.

(a)



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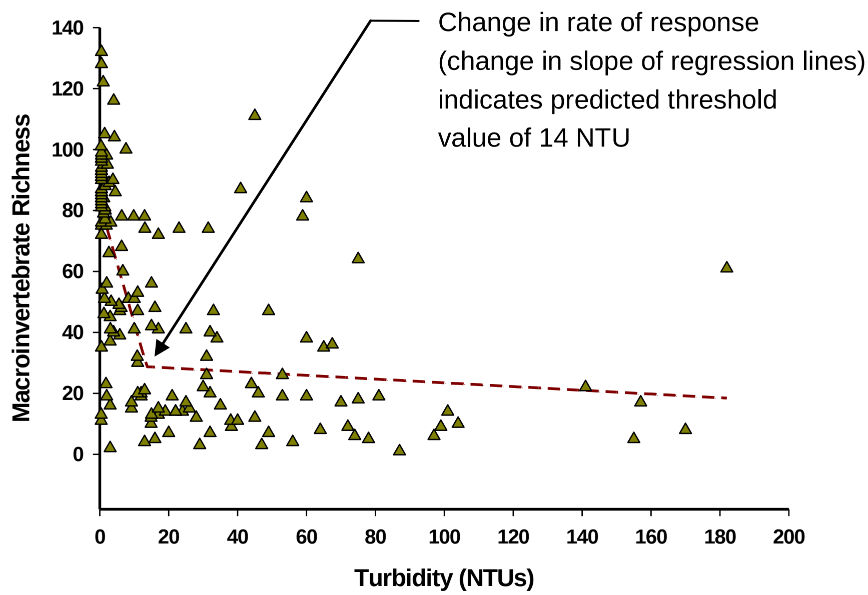


Figure 4. Scatter plots showing total macroinvertebrate taxa richness and turbidity for 163 stream sites in Kansas, Nebraska, and Missouri with (a) linear (dashed line = least-squares trend line) and smoothed (solid line = 30% locally weighted least squares or "LOWESS") fit relationships for the 30-day period prior to macroinvertebrate sampling and (b) a significant, piecewise linear regression model (r^2 *= 0.47, p* < *0.0001*) for the same data set. Note the predicted threshold range of 10 to 25 NTU in (a) and predicted threshold value of 14 NTU in (b).

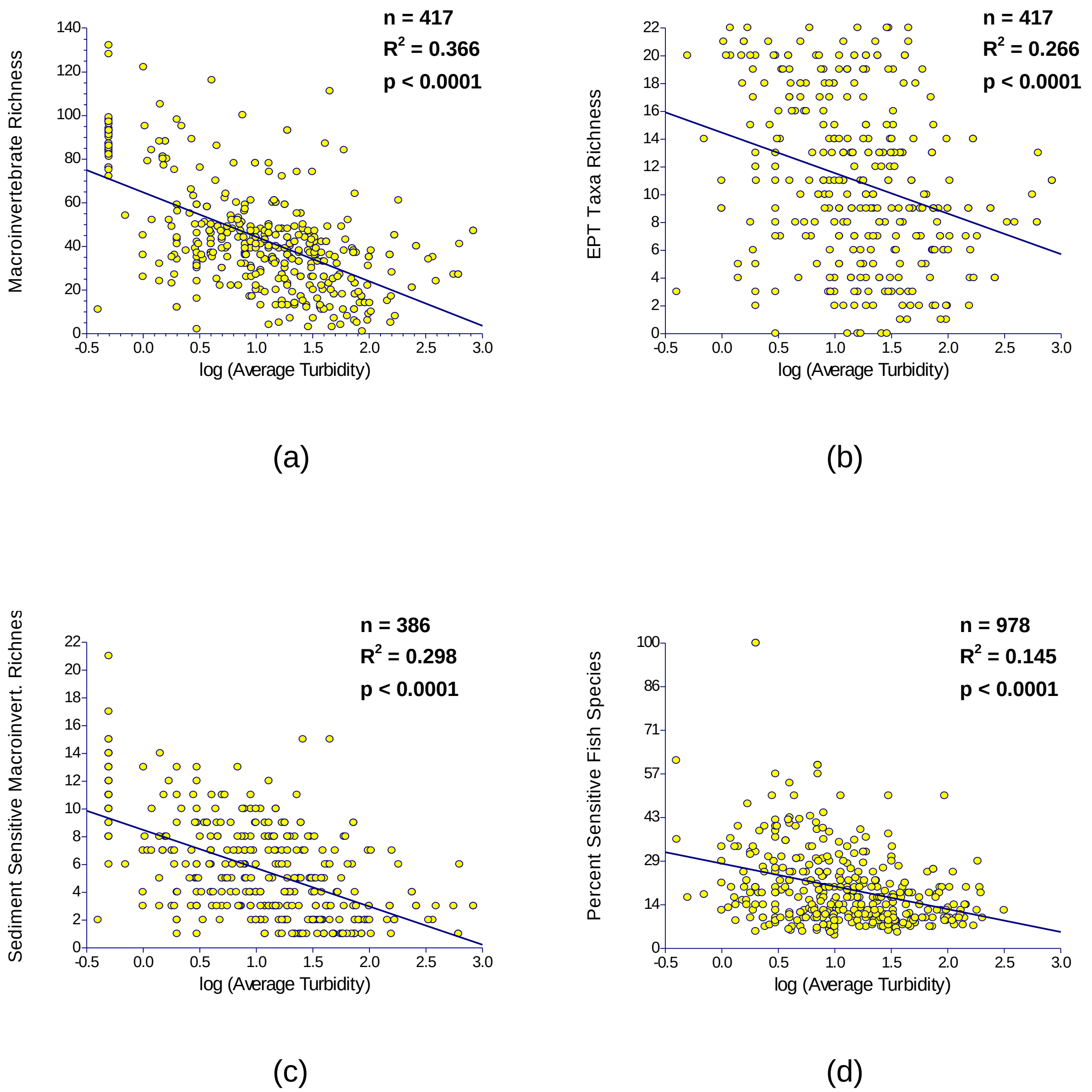
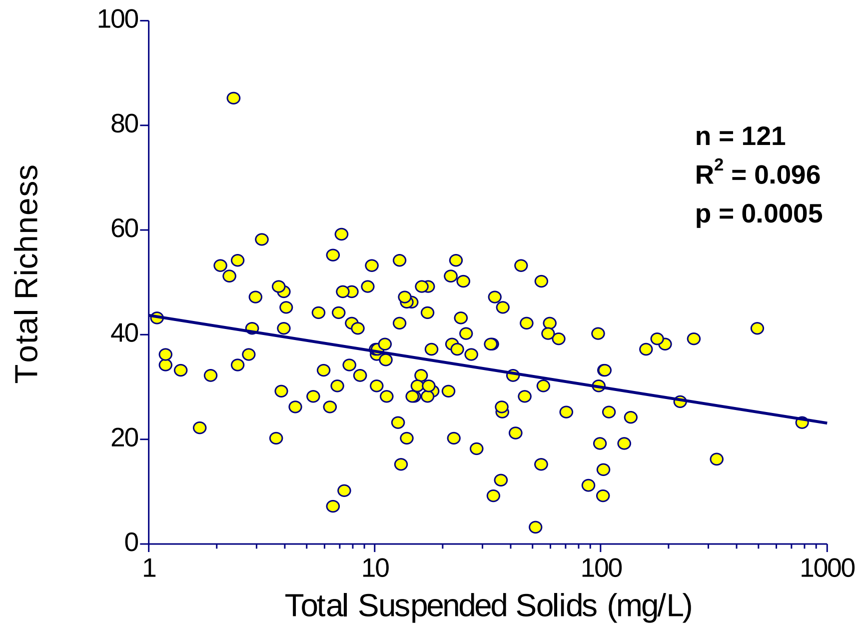


Figure 5. Scatter plots showing least-squares regression lines for turbidity versus (a) macroinvertebrate total richness, (b) Ephemeroptera, Plecoptera, and Trichoptera (EPT) taxa richness, (c) sediment sensitive macroinvertebrate richness, and (d) percent sensitive fish taxa. Data consist of over 13,000 records compiled for USEPA Region 7's Regional Technical Assistance Group for streams in Kansas, Nebraska, Iowa, and Missouri (USEPA Region VII Regional Technical Assistance Group (RTAG) 2006). Iowa sites were excluded, since these sites did not resolve chironomids beyond the family level.

(a)



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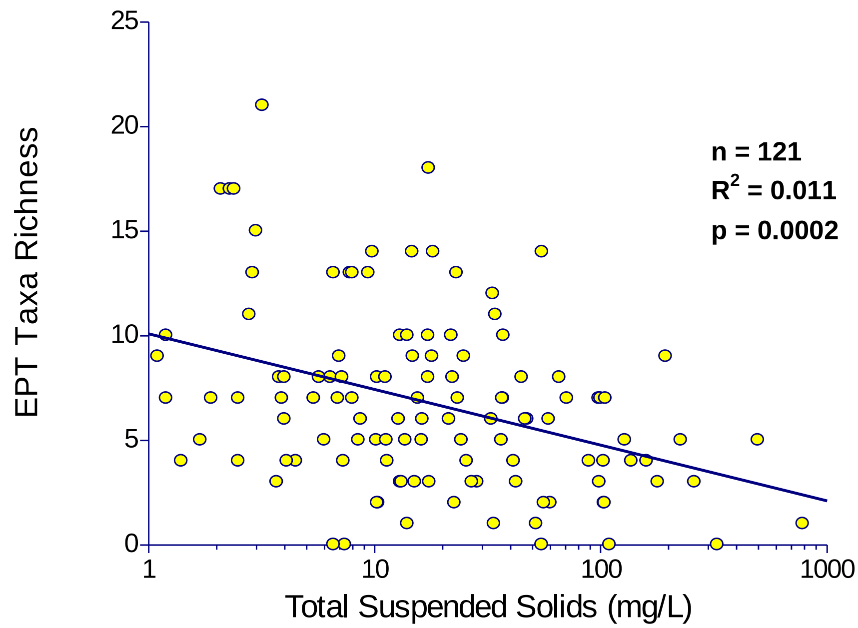
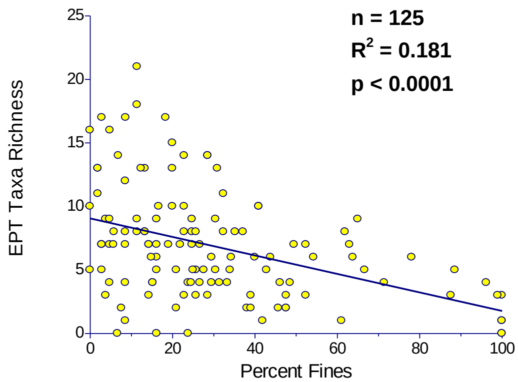


Figure 6. Scatter plots showing least-squares regression lines for total suspended solids (TSS) versus (a) macroinvertebrate total richness and (b) Ephemeroptera, Plecoptera, and Trichoptera (EPT) taxa richness. Data are from 121 sites in Kansas, Nebraska, Iowa, Missouri, and Oklahoma taken from the National Wadeable Streams Assessment database (USEPA 2004).



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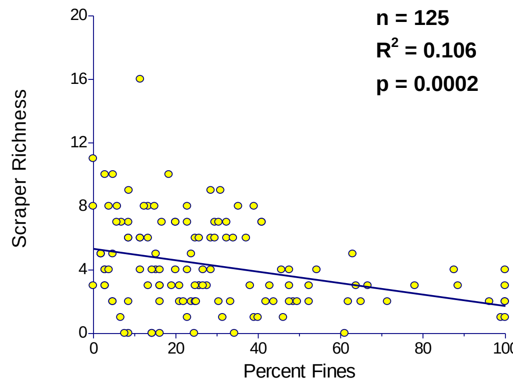

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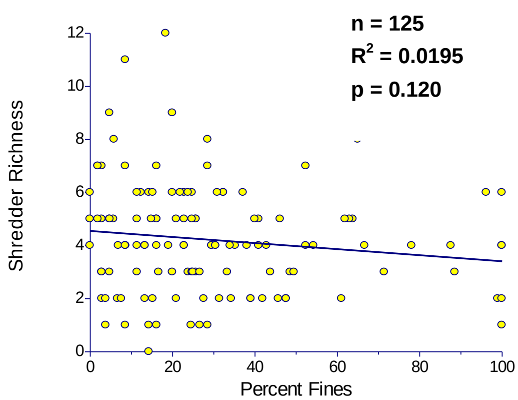

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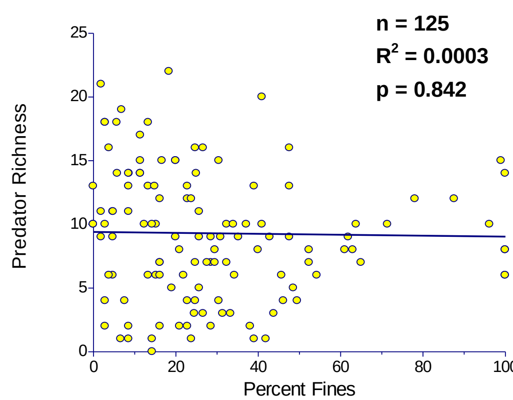
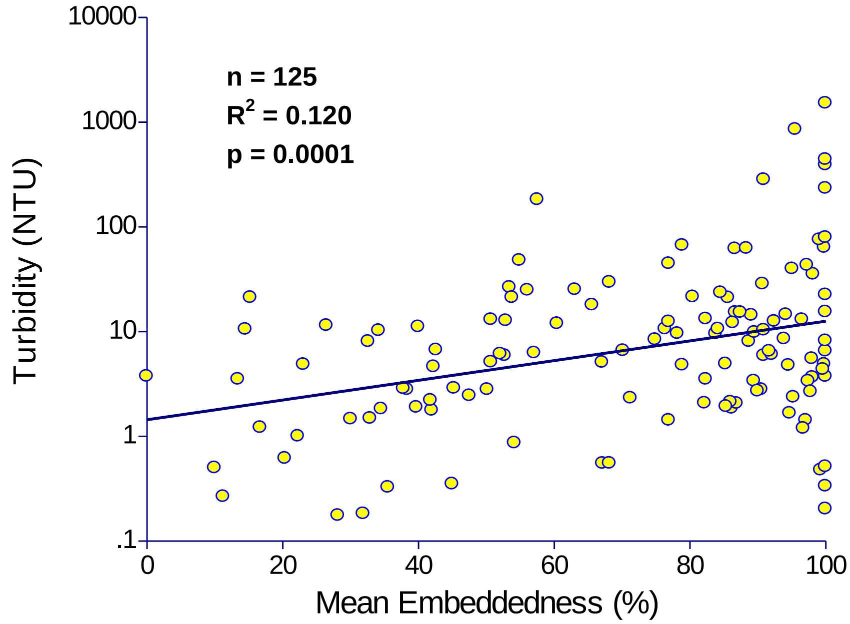


(d)

Figure 7. Scatter plots showing least-squares regression lines for percent of substrate categorized as fines (*i.e.,* mud, silt, muck, etc.) versus (a) Ephemeroptera, Plecoptera, and Trichoptera (EPT) taxa richness, (b) macroinvertebrate scraper taxa richness, (c) macroinvertebrate shredder taxa richness, and (d) macroinvertebrate predator taxa richness. Data are from 125 sites in Kansas, Nebraska, Iowa, Missouri, and Oklahoma taken from the National Wadeable Streams Assessment database (USEPA 2004).

(a)



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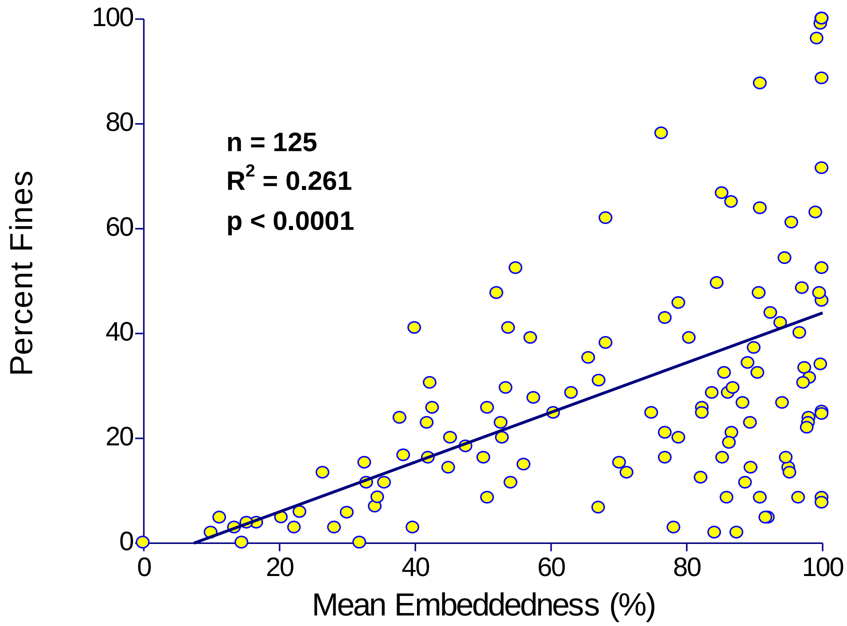
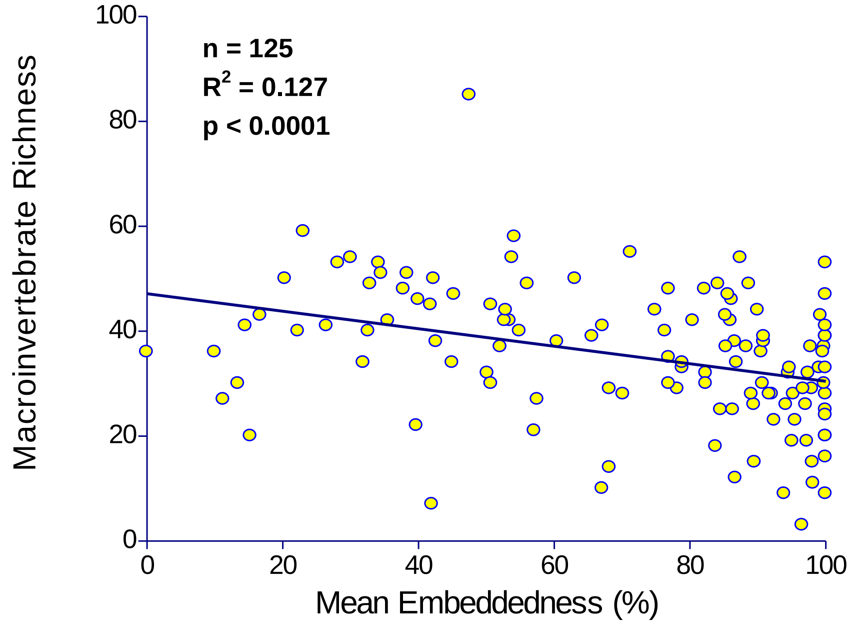


Figure 8. Scatter plots showing least-squares regression lines for mean embeddedness (%) versus (a) turbidity and (b) percent fines. Data are from 125 sites in Kansas, Nebraska, Iowa, Missouri, and Oklahoma taken from the National Wadeable Streams Assessment database (USEPA 2004).

(a)



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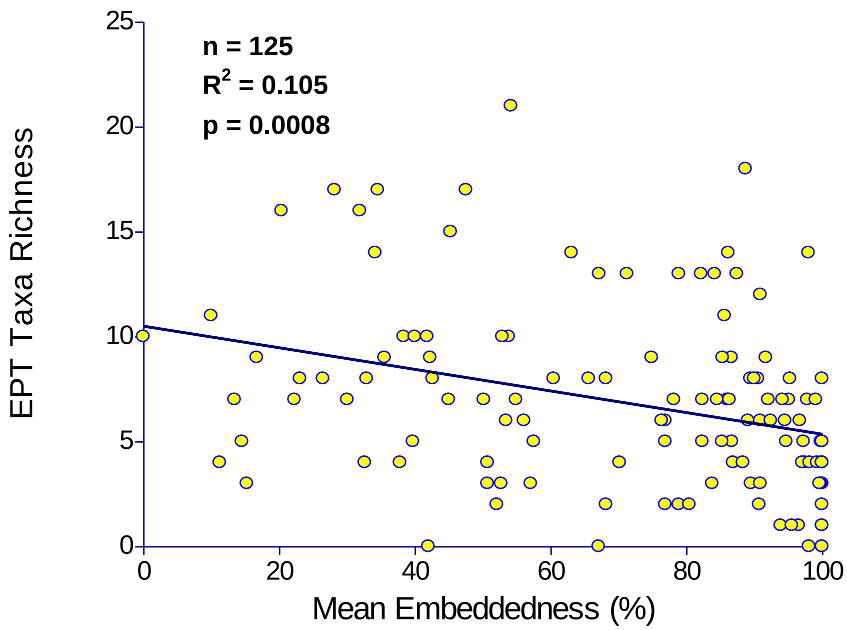

Figure 9. Scatter plots showing least-squares regression lines for mean embeddedness (%) versus (a) total macroinvertebrate richness and (b) Ephemeroptera, Plecoptera, and Trichoptera (EPT) taxa richness. Data are from 125 sites in Kansas, Nebraska, Missouri, and Oklahoma taken from the National Wadeable Streams Assessment database (USEPA 2004).

Specifically, since available geomorphic data were not concurrently measured with any other stream data, no connections between geomorphology and other indicators – including "clean" sediment, water quality, habitat, and biota – could be tested. Additionally, only 16 of 72 sites in the available dataset had both geomorphic and other relevant biotic stream data.

Conclusions

In general, effects of sedimentation in low gradient systems are complex and difficult to measure directly. Therefore, surrogate variables are often required to relate different structural groups (*i.e.,* habitat, biota, water quality, "clean" and "dirty" sediment, geomorphology, and flow). Based on Kansas and regional data, turbidity appears to be a reliable and easily measurable surrogate for "clean" sediment in lotic systems throughout the Central Plains. Further, increasing turbidity (and by extension, increasing "clean" sediment) tends to correlate with decreases in macroinvertebrate richness. Using two statistical techniques (locally weighted, least-squares regression and piecewise regression), a marked change in the rate of decrease of macroinvertebrate richness with increasing turbidity was observed (Figure 4). This change in the rate of response indicates a threshold region of turbidity around 14 NTU. In other words, macroinvertebrate richness is highly sensitive to low levels of turbidity (less than 14 NTU), but does not decline much at higher levels of turbidity (greater than 14 NTU), presumably because those taxa sensitive to higher turbidity levels have already been extirpated. These findings strongly suggest that sediment is impacting macroinvertebrate communities in Central Plains streams. Different mechanisms probably contribute to this effect, including direct effects (*e.g.,* shading, decreased sight predation success, etc.) and indirect effects (*e.g.,* decreased primary productivity, increased embeddedness, etc.).

Additional evidence indicating that sediment (as represented by several indicators) impacts stream biota appears in Figures 4 - 8. Cade and Noon (2003) have shown that wedge-shaped distributions are indicative of ecological conditions in which factors other than the measured variable may contribute to variation in the response variable (Figure 10). Due to complex interactions with unequal variation, more than one slope or rate of change may describe the relationship between response and predictor variables (Cade and Noon 2003). This pattern of variation complicates interpretation of parameter estimates and significance of ordinary least-squares (OLS) regression models (Terrell et al. 1996; Thomson et al. 1996). The observed wedge-shaped patterns of variation between various biological variables and indicator variables for clean sediment (Figures 4a,b; 5a,c,d; 6a,b; 7a,b,c; 8a,b) are consistent with the concept that sediment is an impairment (or limiting) factor for both macroinvertebrate and fish populations. The slopes of these regression models have little predictive value, but do indicate the presence of negative relationships between increased sediment and stream biota. Further investigation using quantile regression techniques developed by Koenker (1995; 2005), may lead to a better understanding of these complex sediment – biota relationships.

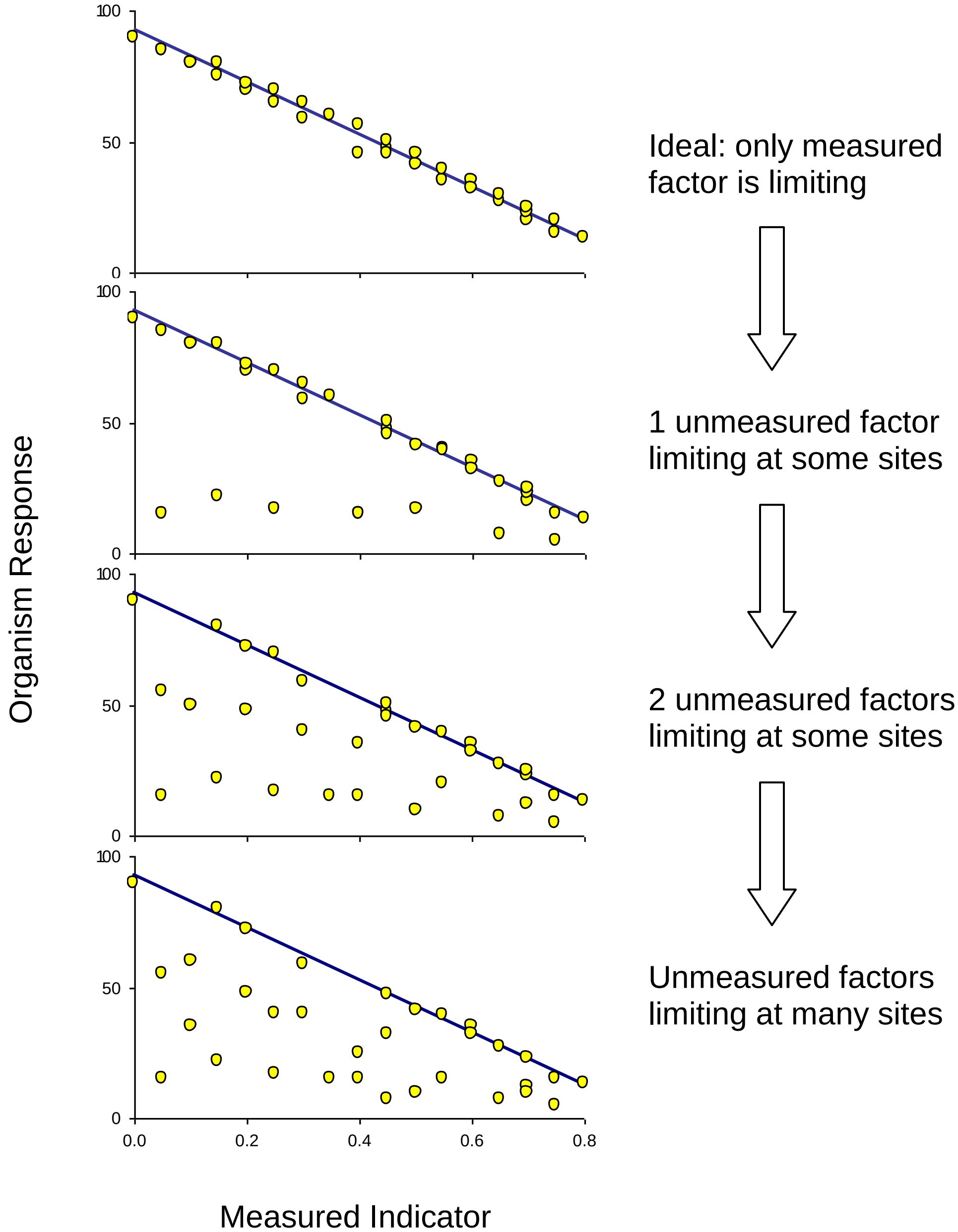

Figure 10. Explanatory figure after Cade and Noon (2003). This figure shows the progression from the ideal statistical situation (top) to the real-world statistical situation (bottom). In the ideal situation, organism response is primarily a function of the measured indicator, and all other potentially limiting, unmeasured factors are not currently limiting. Progressing down toward the real-world situation, increasing numbers of unmeasured factors become limiting for various sites and sampling times. This increase in the potentially limiting, unmeasured factors creates heterogeneity of organism response to the measured indicator with respect to the regression model.

In summary, by applying the conceptual framework and analyses presented in this report, a preliminary characterization of the impacts of sedimentation on biological resources can be made (Figure 11). Where data are available, preliminary estimations of the relative strengths of these impacts can also be made (Table 3). Interpretation of these and similar analyses must be made carefully, as statistical significance does not necessarily imply biological significance (and vice versa). Additional statistical analyses including analysis of covariance, quantile regression, and other advanced techniques may provide additional insight to this framework.

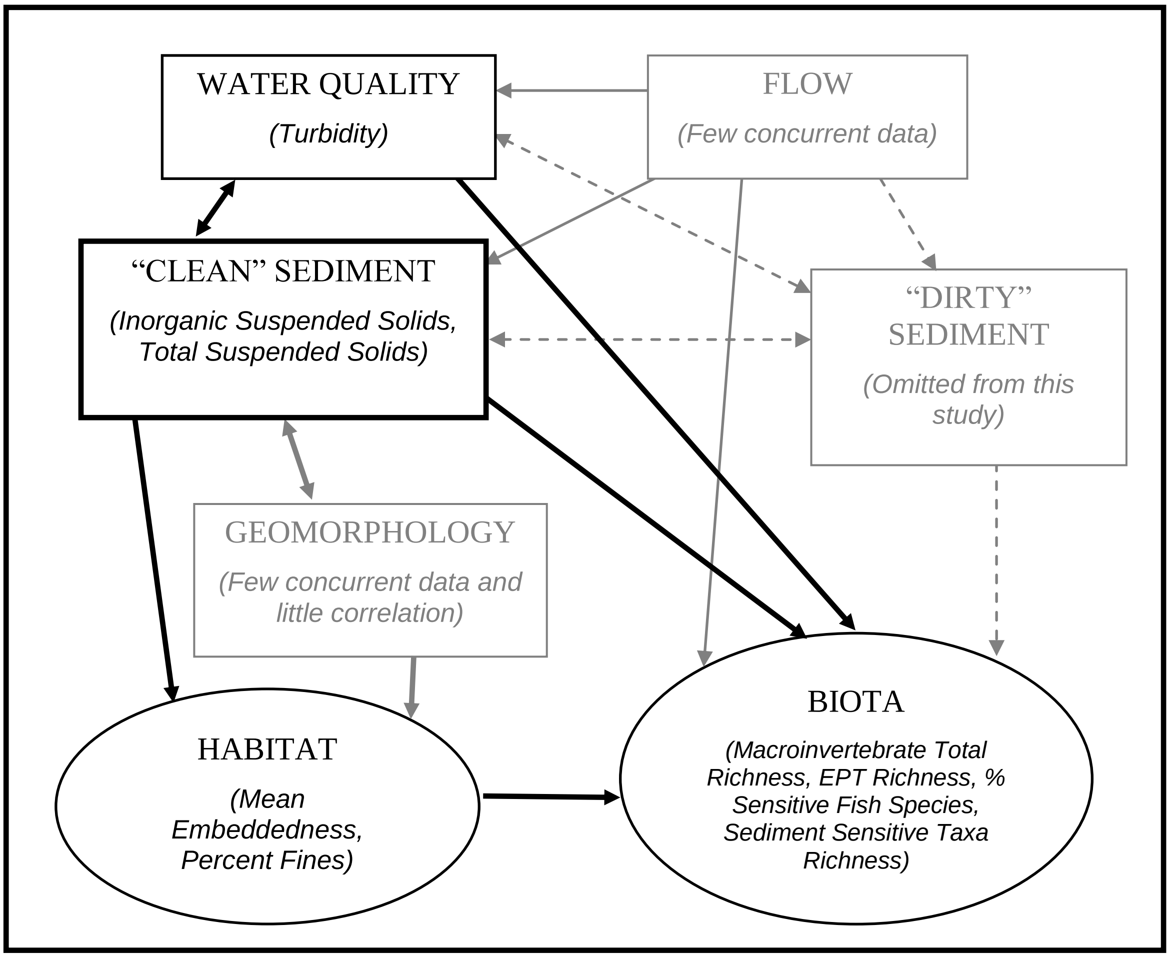

Figure 11. Conceptual framework with interactions of sediment in aquatic ecosystems as determined in this report. Portions in grayscale were either inconclusive and/or data poor (Geomorphology and Flow) or omitted from the scope of this report (“Dirty Sediment”). Other symbology is consistent with the original conceptual framework (Figure 1). The relative strengths of observed relationships, as characterized by R^2 values from significant least-squares regressions, appear in Table 3.

Table 3. Observed strengths of sediment indicator relationships, as characterized by least-squares regression. All relationships shown were statistically significant ($p < 0.001$). Indicator groups and relationships correspond to groups and arrows in Figures 1 and 11, respectively.

Indicator Group 1	**Factor 1 (Measured Indicator)**	**Indicator Group 2**	**Factor 2 (Response Indicator)**	**R^2**	**Associated Figure**
Clean Sediment	Inorganic Suspended Solids	Clean Sediment	Total Suspended Solids	0.99	2a
Clean Sediment	Total Suspended Solids	Water Quality	Turbidity	0.98	3
Water Quality	Log (30 day Average Turbidity)	Biota	Macroinvertebrate Richness	0.37	5a
Water Quality	Log (30 day Average Turbidity)	Biota	EPT Richness	0.27	5b
Water Quality	Log (30 day Average Turbidity)	Biota	Sediment Sensitive Taxa Richness	0.29	5c
Water Quality	Log (30 day Average Turbidity)	Biota	Percent Sensitive Fish Species	0.15	5d
Clean Sediment	Total Suspended Solids	Biota	Macroinvertebrate Richness	0.10	6a
Clean Sediment	Total Suspended Solids	Biota	EPT Richness	0.01	6b
Habitat	Mean Embeddedness	Water Quality	Turbidity	0.12	8a
Habitat	Mean Embeddedness	Habitat	Percent Fines	0.26	8b
Habitat	Mean Embeddedness	Biota	Macroinvertebrate Richness	0.13	9a
Habitat	Mean Embeddedness	Biota	EPT Richness	0.11	9b
Habitat	Percent Fines	Biota	EPT Richness	0.18	7a
Habitat	Percent Fines	Biota	Scraper Richness	0.11	7b

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