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Supplement of

Changing suspended sediment in United States rivers and streams: linking sediment trends to changes in land use/cover, hydrology and climate

Jennifer C. Murphy

Correspondence to: Jennifer C. Murphy (jmurphy@usgs.gov)

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S1. Additional description of methods

The weighted regressions on time, discharge and season model (WRTDS) was originally published in Hirsch et al. (2010). Hirsch et al. (2010) describe the need for improved methods for water-quality trend detection and layout an approach that uses weighted regression. Functionally, WRTDS fits the following equation for each day in the period of record,

$$\ln(c) = \beta_0 + \beta_1 t + \beta_2 \ln(Q) + \beta_3 \sin(2\pi t) + \beta_4 \cos(2\pi t) + \varepsilon \qquad \text{Equation (1)}$$

Where ln(c) is the natural log of concentration,
β_0 through β_4 are fitted coefficients,
t is time in decimal years
Q is daily mean streamflow
ε is unexplained variation.

For each day in the period of record, the data are weighted so that the concentration values most similar, in terms of time, streamflow and season, to the day being fit are given higher weight during calibration. Windows are used to control this weighting as well, such that data outside the windows were given a weight of zero while data inside the windows were weighted using a tricube weight function. The half-windows used in this study were 7 years for time, 1 log cycle for streamflow at sites where drainage area exceeded 250,000 square kilometers (km^2) or 2 log cycles for streamflow at sites with smaller drainage areas, and 0.5 years for season. For more details on model specifications see Oelsner et al. (2017). Computationally, Eq. (1) is fit at points across a grid of equally spaced intervals of time and streamflow, and results for a given day are determined using linear interpolations from this grid. See Hirsch et al. (2010) for details. The WRTDS approach thus gives coefficient estimates that vary over time, allowing for a flexible characterization of changing water quality.

Hirsch et al. (2010) also introduce the concept of flow normalization, an approach for identifying the “signal” of long-term systematic water-quality changes often due to human actions on the landscape from the “noise” of high year-to-year variability due largely to variable weather. In Hirsch et al. (2010), streamflow is considered the main source of this year-to-year variability and its influence on water quality is removed. This process includes compiling 365 empirical probability distributions of daily streamflow value, one probability distribution for each calendar day, using the observed streamflow record. Several “expected” concentrations for each day in the period of record are estimated using the weighted regression for that day and each of the observed daily streamflow values for that calendar day from the appropriate probability distribution. The mean of these “expected” concentrations is the flow normalized concentration for the given day. This process is repeated for each day in the period of record and daily estimates are aggregated to annual means. A shortcoming of this approach is that it assumes a stationary streamflow regime and if there are trends in streamflow at a site, this original implementation of flow normalization does not give the overall water quality trend. Instead the original implementation of flow normalization gives the change in water quality due to influences other than changes or variations in streamflow—whether that change in streamflow is from the noise of year-to-year variability because of fluctuating weather, or a systematic change in streamflow due to sustained changes in climate, land use or other human actions.

Choquette et al. (2019) address this need by extending the flow normalization process for use at sites with nonstationary streamflow. The updated flow normalization process presented in Choquette et al. (2019) follows the same steps as the original implementation described above, but instead of compiling 365 probability distributions of daily streamflow (which are the same set of probability distributions used every year), a probability distribution of daily streamflow is compiled for each day in the period of record. These probability distributions are compiled using the observed daily streamflow on the same calendar day within a 15-year moving window (7 years on each side of the target year plus the target year). Thus, each day in the period of record has a unique probability distribution of streamflow and from here the flow normalization process proceeds according to the original implementation. See Choquette et al. (2019) for a complete description of the updated flow normalization methods. WRTDS methods are implemented using the EGRET 3.0 R package (Hirsch et al., 2018a) which contains the updated flow normalization procedures.

Hirsch et al. (2015) present a block bootstrap approach for characterizing the uncertainty in WRTDS results. This approach was originally developed for water-quality trends determined using the original implementation of flow normalization. While the approach is still best described in Hirsch et al. (2015), it has since been updated to accommodate the updated flow normalization methods which allow for nonstationarity in the streamflow regime. These uncertainty methods are implemented using the EGRETci 2.0 R package (Hirsch et al., 2018b). For a detailed description of the block bootstrap approach, see Hirsch et al. (2015).

In addition to the recent update in the flow normalization procedure, Choquette et al. (2019) also present a novel approach for teasing apart the influence of various types of streamflow variability on a water-quality trend. This approach parses the water-quality trend into two components of change: (1) water-quality changes due to changes in the concentration-discharge (C-Q) relationship, and (2) water-quality changes due to systematic changes in the streamflow regime. This is accomplished by using the original implementation of flow normalization to determine the first component and subtracting this from the overall change in water quality to get the second component. See Choquette et al. (2019) for a complete description of these methods. See also Murphy and Sprague (2019) for an application of these methods at over 300 sites for 1-15 water quality parameters across the United States. Additionally, Fig. 1 in Murphy and Sprague (2019) provides a detailed example of this approach applied at four sites. Choquette et al. (2019) refer to these components of change as the concentration-streamflow trend component and the streamflow trend component (QTC). However, this paper follows the approach of Murphy and Sprague (2019) and refers to the concentration-streamflow trend component as the management trend component (MTC). Equation (2) shows how these components are related to the water-quality trend.

$$QTC = WQT - MTC \qquad \text{(Equation 2)}$$

Here, QTC is the streamflow trend component which gives the amount of change in the water-quality trend attributed to changes in the streamflow regime alone. WQT is the water-quality trend determined using the updated flow normalization procedures that incorporate the influence of streamflow trends on water quality. MTC is the management trend component, determined using the original implementation of flow normalization that assumes a stationary flow regime. The MTC gives the amount of change in water quality attributed to changes other than streamflow, conceptualized as changes from human actions in the watershed though could be due to other process that would systematically alter the C-Q relationship over time.

Note, this paper and its focus on suspended sediment, is a subset of a comprehensive trend assessment of US rivers and streams for multiple water quality parameters. The entire study is presented in Oelsner et al. (2017) which includes a detailed description of the data compilation methods. The modeling specifications described in Oelsner et al. (2017) were used to generate the WRTDS output published in the De Cicco et al. (2017) data release. These include the flow normalized estimates using the original implementation. Updated flow normalized estimates and the QTC and MTC estimates are described and published in Murphy et al. (2018) for the same set of sites using the same underlying data, fitted regression equations (Eq. 1) and non-flow-normalized estimates.

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Hirsch, R., De Cicco, L., Murphy, J.: EGRETci: Exploration and Graphics for RivEr Trends (EGRET) Confidence Intervals, version 2.0. https://CRAN.R-project.org/pack- age=EGRETci, 2018b.

Murphy, J.C. Farmer, W.H., Sprague, L.A., De Cicco, L.A. and Hirsch, R.M.: Water-quality trends and trend component estimates for the Nation's rivers and streams using Weighted Regressions on Time, Discharge, and Season (WRTDS) models and generalized flow normalization, 1972-2012: U.S. Geological Survey data release, https://doi.org/10.5066/F7TQ5ZS3, 2018.

Murphy, J., and Sprague, L.: Water-quality trends in US rivers: Exploring effects from streamflow trends and changes in watershed management. Science of the Total Environment 656, 645-658, 2019.

Oelsner, G.P., Sprague, L.A., Murphy, J.C., Zuellig, R.E., Johnson, H.M., Ryberg, K.R., Falcone, J.A., Stets, E.G., Vecchia, A.V., Riskin, M.L., De Cicco, L.A., Mills, T.J., and Farmer, W.H.: Water-quality trends in the Nation’s rivers and streams, 1972–2012—Data preparation, statistical methods, and trend results: U.S. Geological Survey Scientific Investigations Report 2017–5006, 136 p., https://doi.org/10.3133/sir20175006, 2017.

S2. Categorization scheme for determining land use of each site’s contributing watershed

Table S1. Rules for land use classification of watersheds

Land-use category	Rules for classification
Undeveloped	Undeveloped: (lu_11 + lu_12 + lu_46 + lu_50 + lu_60) > 75%
Urban	Urban: lu_43 < 25% AND (lu_43 + lu_44) < 50%, plus satisfying at least one of the “high”, “med” or “low” urbanization rules below
	High urbanization: (lu_21 + lu_22 + lu_23 + lu_24 + lu_25 + lu_26 + lu_27) > 50% OR lu_21 + lu_22 + lu_23 + lu_25) > 25%
	Medium urbanization: (lu_21 + lu_22 + lu_23 + lu_24 + lu_25 + lu_26 + lu_27) > 25%
	Low urbanization: (lu_21 + lu_22 + lu_23 + lu_24 + lu_25 + lu_26 + lu_27) > 12.5% AND (lu_21 + lu_22 + lu_23 + lu_24 + lu_25 + lu_26 + lu_27 + lu_31 + lu_32 + lu_33) > 25%
Ag	Agricultural: (lu_21 + lu_22 + lu_23 + lu_24 + lu_25 + lu_26 + lu_27) < 10%, plus satisfying at least one of the “high”, “med” or “low” ag rules below.
	High ag: lu_43 > 50% OR (lu_43 + lu_44) > 75%
	Medium ag: lu_43 > 25% OR (lu_43 + lu_44) > 50%
	Low ag: lu_43 > 12.5% AND (lu_43 + lu_44) > 25%
Mixed High	Does not meet criteria for Undeveloped, Urban, or Ag

Land-use categorization is based on the scheme used in Falcone (2015). The specific land uses associated with the “lu_XX” codes can be found in Falcone (2015) and the related data release.

S3. Proximal zone variables and results

In addition to the land use/cover change variables presented in the manuscript, 9 other proximal zone land use/cover change variables are presented here (Table S2). Recall the proximal zone is a near-site, near-stream zone and was computed as 25% of the watershed area nearest the site and stream (Fig. S1).

Table S2. Description of proximal zone land use/cover variables. Original data published in Falcone (2017).

Short name	Data description (original time-series or static variable name from referenced source)
Proximal zone land use/cover changes[1]	
Agricultural land	Percent change in agricultural land, excluding potential grazing lands, as a percentage of the proximal zone (RIP_NRSITE_NWALT##_AG4344_SUM)
Ag+Grazing land	Percent change in agricultural land, including potential grazing lands, as a percentage of the proximal zone (RIP_NRSITE_NWALT##_AG4346_SUM)
Cropped land	Percent change in row-cropped land as a percentage of the proximal zone (RIP_NRSITE_NWALT##_43)
All developed land	Percent change in developed and semi-developed land as a percentage of the proximal zone (RIP_NRSITE_NWALT##_DEV_SUM + RIP_NRSITE_NWALT##_SEMIDEV_SUM)
Developed land	Percent change in developed land as a percentage of the proximal zone (RIP_NRSITE_NWALT##_DEV_SUM)
Semi-developed land	Percent change in semi-developed land (land in close proximity to developed lands and partially used for same purposes) as a percentage of the proximal zone (RIP_NRSITE_NWALT##_SEMIDEV_SUM)
Impervious area	Percent change in impervious land cover as a percentage of the proximal zone[2] (RIP_NRSITE_NWALT_IMPV##)
Low-med density dwellings	Percent change in land with low-medium density residential development as a percentage of the proximal zone (RIP_NRSITE_NWALT##_26)
Low-use land	Percent change in land with little to no development or agriculture as a percentage of the proximal zone (RIP_NRSITE_NWALT##_50 + RIP_NRSITE_NWALT##_60)

[1] All land-use variables rounded to 1% of watershed or riparian zone area prior to calculating percent change.

[2] Variables not included in Falcone (2017) and estimated for this study using the same procedures described in Falcone (2017)

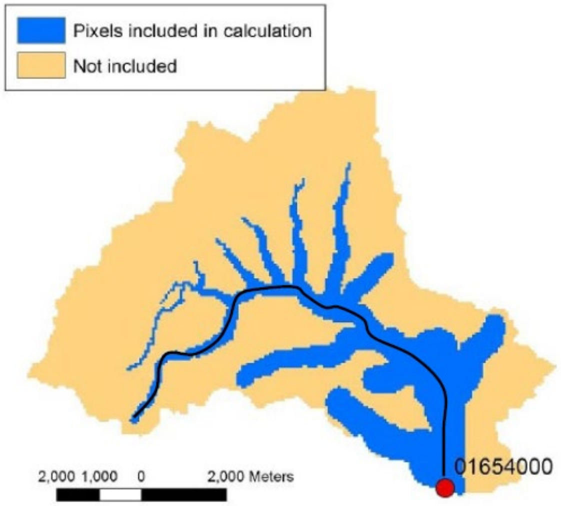

Figure S1. Example of pixels used in near-site, near-stream proximal zone calculation for site 01654000. Calculation of whole watershed values would use the tan and blue areas. Figure originally published in Falcone (2017).

Several studies have compared water quality to land-use/cover variables derived from the entire watershed and from a more confined riparian or buffer zone. For example, Johnson et al. (1997) found that buffer-zone characterizations of land use/cover were a better indicator of water quality compared to land use/cover across the watershed. Whereas, Sliva and William (2001) and Hunsaker and Levine (1995) found the opposite. In this current study, explicit characterization of land-use/cover changes in the proximal zone provided some additional insight, but typically did not yield more or stronger correlations with sediment trends compared to land-use/cover changes across the entire watershed (Fig. 4 and Fig. S2).

There are 11 variables that have both whole watershed and proximal zone determinations. These include the 9 proximal zone determinations in Fig. S2 and their corresponding watershed determinations in Fig. 4, plus the two proximal zone and watershed CRP variables in Fig. 4. Considering these 11 variables, slightly more SSC trends across the land-use categories were well correlated with land-use/cover changes using the whole watershed determinations compared to the proximal zone only. For TSS, 2 of the watershed determinations, compared to 4 of the proximal zone determinations were well correlated with TSS trends, and 7 watershed variables compared to 6 proximal zone variables were statistically significant (alpha ≤ 0.05) (Fig. 4 and Fig. S2). Various studies have speculated why riparian/buffer zone characterizations do not consistently provide more explanatory power given the known importance of riparian zone and near-stream conditions on water quality at a local scale (Hunsaker and Levine, 1995; Johnson et al., 1997; Sliva and William, 2001). In this study, most of the proximal zone land-use change variables that were correlated with the sediment trend were similar to the watershed-based estimates, however, there were some notable exceptions. For example, the percent change in the proximal zone agricultural land enrolled in CRP was well correlated with changes in SSC at agricultural sites, whereas the percent change in CRP across the whole watershed was not (Fig. 4).

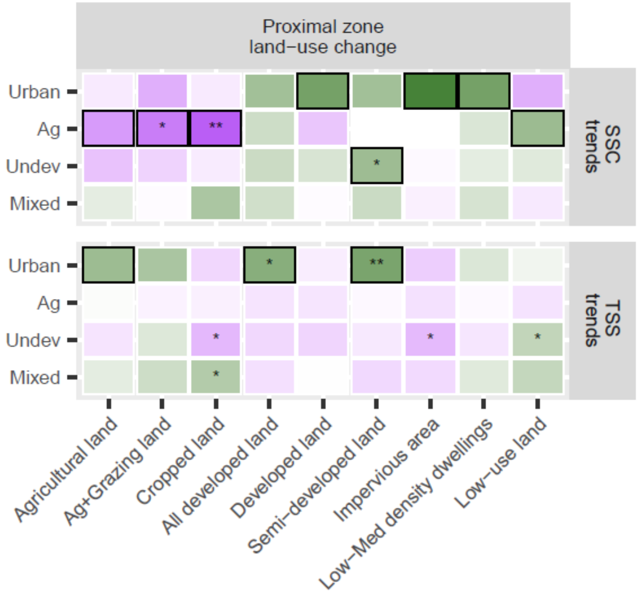

Figure S2. Correlations between 1992-2012 sediment trends and proximal zone land use/cover changes, grouped by the 2012 land use of the contributing watershed.

Falcone, J.A.: Watershed characteristics for study sites of the Surface Water Trends project, National Water Quality Program: U.S. Geological Survey data release, https://doi.org/10.5066/F7TX3CKP, 2017.

Johnson L. B., Richards C., Host G. E. and Arthur J. W.: Landscape influences on water chemistry on Mid-western stream ecosystems. Freshwater Biology 37, 193–208, 1997.

Sliva, L., and Williams, D.D.: Buffer zone versus whole catchment approaches to studying land use impact on river water quality. Water Research 35, 3462-3472, 2001.

Hunsaker, C.T. and Levine, D.A.: Hierarchical Approaches to the Study of Water Quality in Rivers. BioScience 45:3, 193-203, 1995.

S4. Additional figures

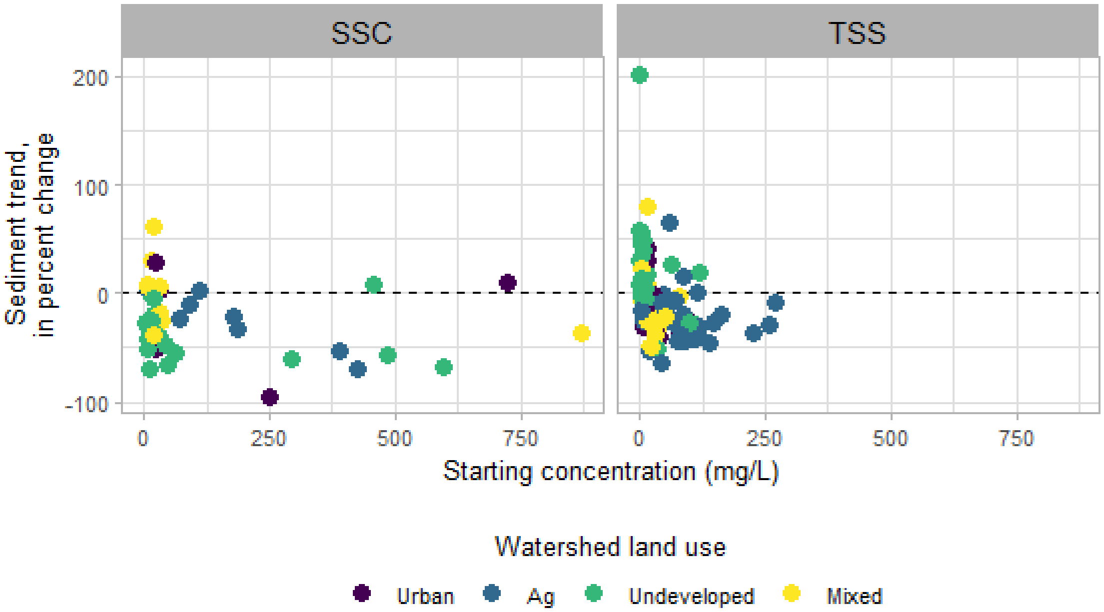

Figure S3. Sediment trend (1992-2012) in percent change versus starting concentration (1992), by sediment parameter (suspended sediment concentration (SSC) and total suspended solids (TSS)) and land-use category.

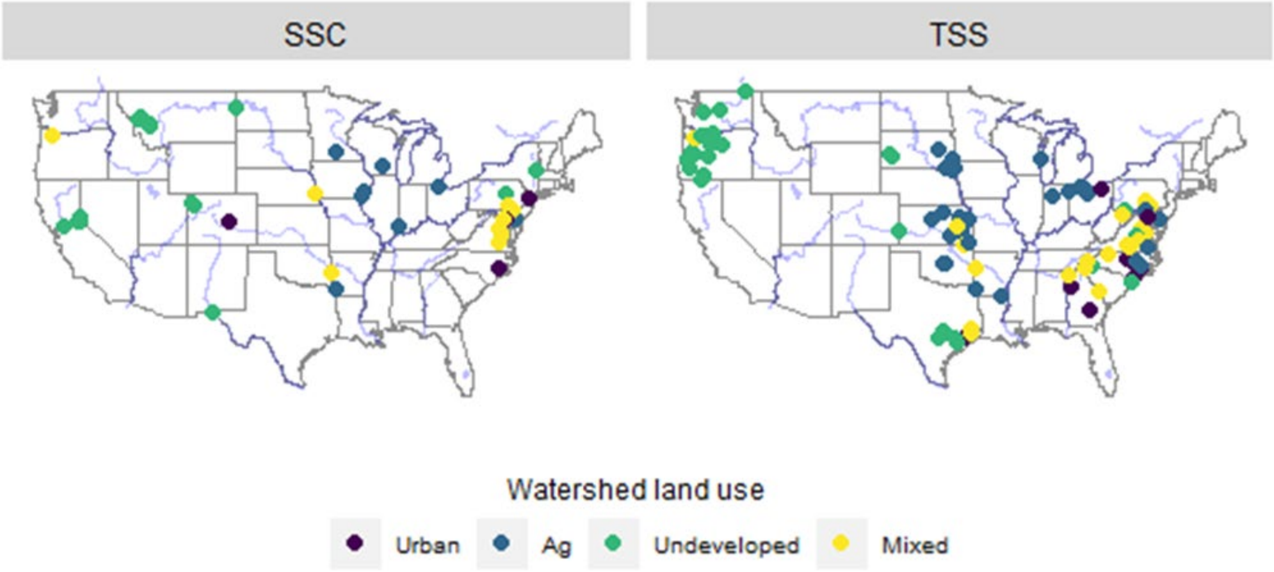

Figure S4. Land-use categorization of each site in 2012, shown by parameter

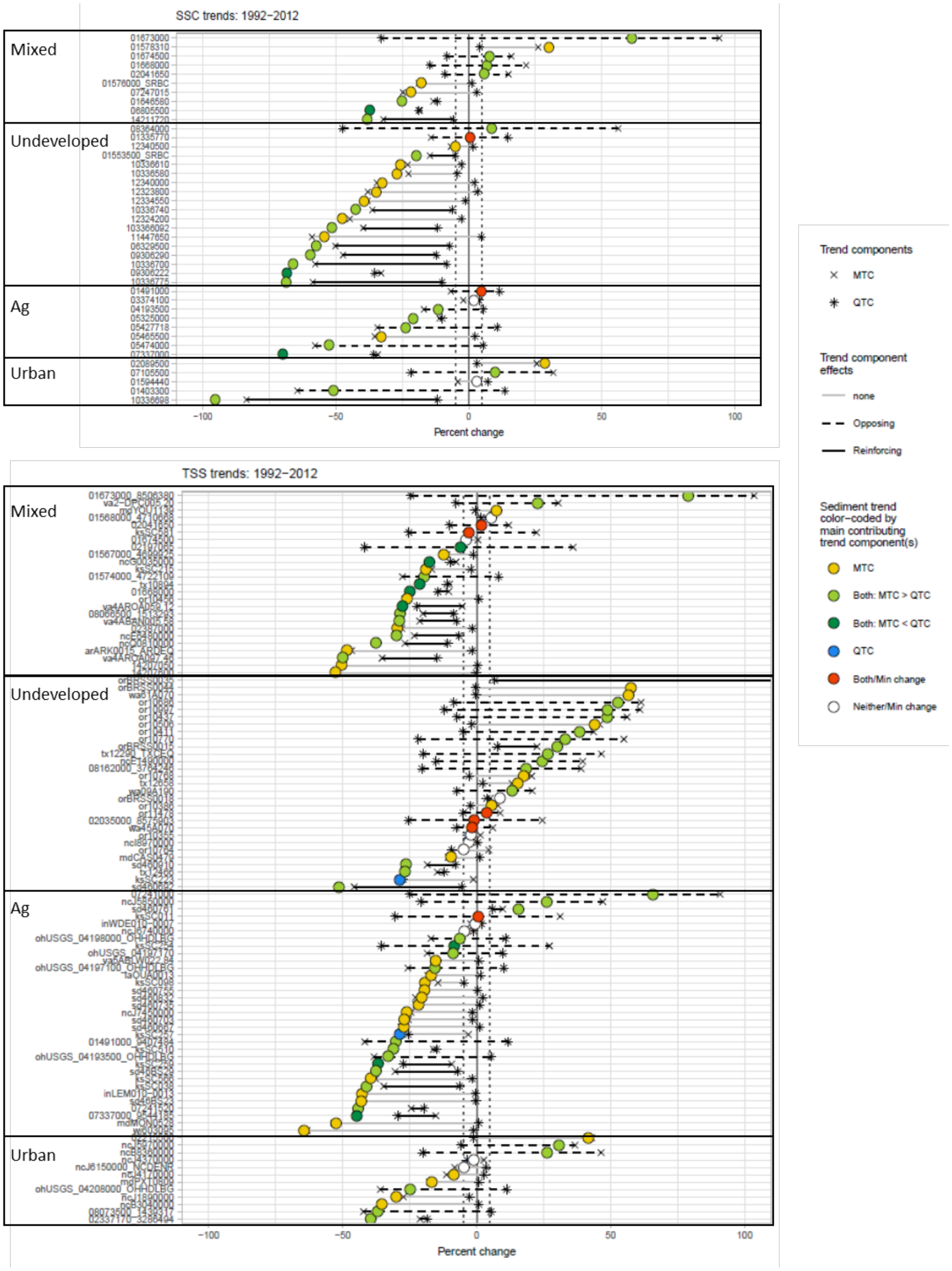

Figure S5. Percent change between 1992-2012 for sediment trend, management trend component (MTC) and streamflow trend component (QTC), grouped by 2012 land-use category and ordered by increasing magnitudes of change within each category. Dashed vertical lines are +/- 5%, an arbitrary threshold for differentiating between negligible and nonnegligible influences from the MTC and QTC, equates to rough 0.25% change per year. Recall sediment trend = MTC + QTC. Also, TSS trend and MTC estimate for site orBRSS0035 are 202% and 195%, respectively, and not shown on plot. Red and white symbols indicate sites where the sediment trend is < 5% and the MTC and QTC are also > 5% ("Both/Min change") or < 5% ("Neither/Min change")

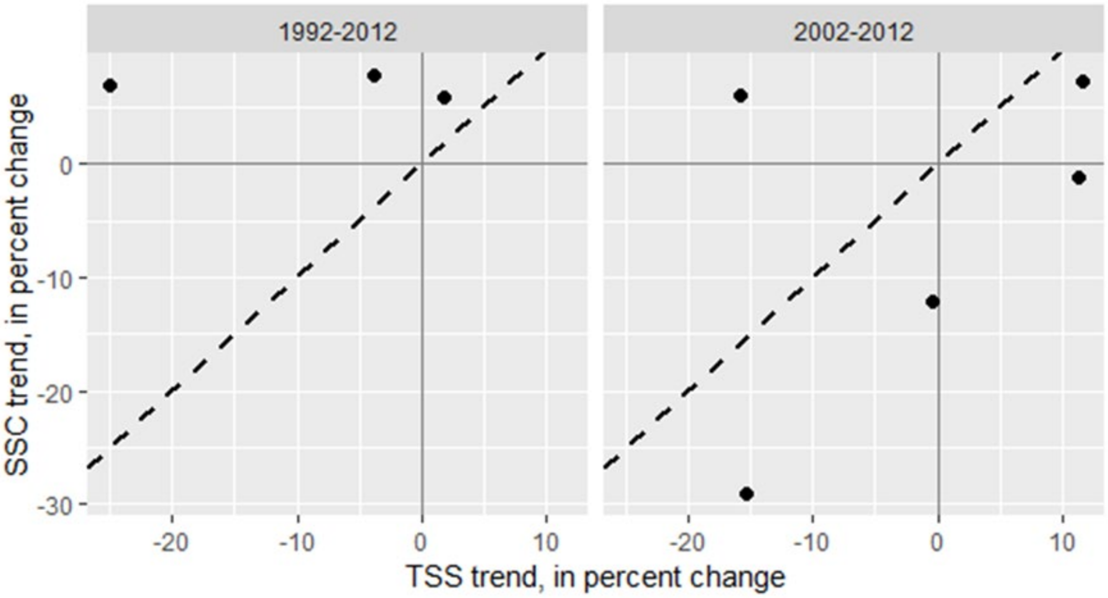

Figure S6. SSC trend versus TSS trend at sites where both sediment parameters were collected. 3 sites for 1992-2012 trends and 5 sites for 2002-2012 trends. Dashed line is a 1:1 line (SSC trend = TSS trend) and solid, gray vertical and horizontal lines are x=0 and y=0, respectively. Points in upper left quadrant and lower right quadrant indicate different trend directions for SSC compared to TSS.