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Q_c threshold departs from theoretical Q_c in urban watersheds: The role of streambed mobility data in managing the urban disturbance regime

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Abstract: The threshold discharge (Q_c) for streambed mobilization is both biologically and geomorphically relevant to stream ecosystems. Excess streambed mobilization can disturb benthic organisms and initiate cycles of channel instability. The mechanistic relevance of Q_c gives it great utility for aquatic ecosystem studies, stormwater management, and stream restoration design. However, field and laboratory data document considerable variability in Q_c across hydrogeomorphic settings, underscoring the importance of using field data to calibrate the Q_c estimate for a given stream or region. This paper shows how both high- and low-tech monitoring protocols can be used to constrain a Q_c estimate, depending on monitoring program goals and budgets. Data from 3 hydrogeomorphically distinct settings in the USA and Australia show that the departure of Q_c from theoretical estimates increases with watershed imperviousness. Although Q_c estimates derived from conventional critical Shields stress values tend to be a reasonable and conservative starting point for stormwater management in streams that lack site-specific or regional data, streambed mobility monitoring is recommended to calibrate and validate Q_c estimates for a stream or region prior to making large investments in stormwater interventions aimed at mitigating the urban streambed disturbance regime.

Key words: streambed mobility, incipient motion, benthic disturbance, stormwater management, channel stability, geomorphic equilibrium, critical discharge, critical Shields parameter, erosion, sedimentation, stream ecology, watershed planning

The threshold discharge (Q_c) for streambed mobilization varies across stream settings and has wide-ranging societal implications, such as river-delivered beach sand (Kondolf 1997, Shaghude et al. 2012), self-maintenance of flood conveyance capacity (Chow 1959, Nanson and Young 1981, Urbonas and Roesner 1993), and more geomorphically sustainable stormwater management in the context of the urban stream syndrome (Walsh et al. 2005). Urbanization is widely documented to increase peak flows and prolong durations of channel-eroding discharges (Sauer et al. 1983, MacRae 1997, Konrad and Booth 2002, Smith et al. 2002, Hawley and Bledsoe 2011, Hopkins et al. 2022). Excess erosion of streambed particles can initiate and exacerbate trajectories of geomorphic instability (Schumm et al. 1984, Simon 1989, Cluer and Thorne 2014) that can include incision (Ireland et al. 1939, Schumm and Parker 1973, Booth 1990), bank erosion and widening (Booth and Jackson 1997, Thorne 1998, Galster et al. 2008, Bevan et al. 2018, Hawley et al. 2020), and aggradation and braiding (Germanoski and Schumm 1993, Hawley et al. 2012, Booth and Fischenich 2015) (Fig. 1A–H). Channel enlargement can be especially pronounced in urban streams with conventional stormwater management, which amplifies the erosive power of the flow regime via prolonged durations of streambed mobilizing discharges (Hammer 1972, Hollis and Luckett 1976, Hawley and Bledsoe 2013, Taniguchi and Biggs 2015, Chin et al. 2017).

Intensified rates of erosion and instability in urban streams are associated with a range of ecosystem impacts, including degradation of habitat (Pizzuto et al. 2000, Booth and Henshaw 2001, Vietz et al. 2014, Russell et al. 2020, White and Walsh 2020) and water quality (Trimble 1997, Chin 2006, Wilson et al. 2007, Simon and Klimetz 2008, Russell et al. 2018, Kaushal et al. 2022), which can both adversely affect biological communities. Furthermore, excess streambed disturbance can directly affect aquatic taxa that require stable substrate for colonization, reproduction, or refugia (Poff 1992, Jordt and Taylor 2021). For example, Holomuzki and Biggs (2000) documented greater dislodgement of mayflies, caddisflies, and snails along with greater mortality to mayflies and snails in laboratory trials that exceeded Q_c. Furthermore, in a 7-y study of a reference stream, Hawley et al. (2016) showed that the macroinvertebrate biotic index decreased with fewer days between Q_c events. These linkages make the Q_c for streambed mobilization relevant to aquatic biology studies in both rural and urban streams in addition to stormwater management (e.g., Rieck et al. 2021, Wooten et al. 2022) and stream restoration (Jordt and Taylor 2021, Mayer et al. 2022) as well as to managers considering macroinvertebrate reintroductions (Clinton et al. 2022). In addition to biotic consequences, erosion and instability can have costly effects on property and infrastructure (Richey 1982, Hawley et al. 2013b, Vietz and Hawley 2019) and pose risks to human life. One example of such risk is when stream incision undermines the outfalls of large detention ponds (Fig. 1E–H), which can jeopardize the berm that detains water and risks sending large flood waves downstream during a failure.

Stormwater planning and watershed management can be substantially improved to reduce the risks of future effects to urban streams, acknowledging that existing risks to public safety will still need to be managed (e.g., via geotechnical repairs and bioengineered stream stabilizations). Building on decades of previous work (Wolman and Miller 1960, MacRae 1993, 1997, Biedenharn et al. 2001, Soar and Thorne 2001, Bledsoe 2002), Hawley et al. (2022) outlined how stormwater control measures can be designed to minimize increases in the cumulative erosive power of the flow regime relative to the undeveloped flow regime. The approach involves managing excess stormwater runoff such that it does not contribute to increases in the cumulative magnitude and duration of flows that exceed the Q_c for streambed mobilization (Hawley and Vietz 2016). The framework revolves around managing erosion potential (E), which is defined as the ratio of the post-developed to predeveloped sediment transport capacity. Across 3 case studies, Hawley et al. (2022) documented a geomorphic recovery in a retrofit watershed with substantial reductions in E (Hawley 2022), the lack of a geomorphic recovery in a retrofit watershed that did not substantially reduce E, and the continued recovery from legacy agricultural impacts downstream of a development site where stormwater controls were designed to match the predeveloped sediment transport capacity (E = 1). The framework needs substantially more implementation and monitoring across a gradient of hydrogeomorphic settings to understand the extent of E reductions that are necessary to induce geomorphic recoveries in previously developed watersheds. However, when implemented efficaciously, urban stream channels would experience no additional erosion or instability than would have occurred under undeveloped conditions and would be better aligned with society's goals for urban streams, such as water quality, instream ecology, vegetation, recreation, safety, and infrastructure (Somerville and Pruitt 2004, Polvi et al. 2020, Scoggins et al. 2022).

A central aspect to the success of such geomorphically tailored stormwater management is the calibration of the Q_c threshold to the given stream network. That is, managing stormwater to match the cumulative sediment transport capacity of the predeveloped watershed depends on knowing the discharge at which sediment transport begins. Much of the Q_c tailoring that has been used to date relies on standard measurements of median bed sediment diameter (D_{50}; Bunte and Abt 2001), channel cross section and profile surveys (Harrelson et al. 1994), and hydraulic and sediment transport calculations (Wolman and Miller 1960, Biedenharn et al. 2001, Soar and Thorne 2001). However, calibration efforts typically stop short of collecting streambed mobilization data in the field (e.g., Bledsoe 2002, Hawley and Bledsoe 2013, Hawley and Vietz 2016). Efforts to date have generally relied on literature values for the critical Shields parameter

(τ_{*c}; Shields 1936; see Boxes 1, 2, Eqs 1, 2). τ_{*c} is the dimensionless shear stress associated with the threshold of mobilization of a given sediment size in a fluid. Shields (1936) made shear stress (force/unit area) nondimensional by standardizing for the submerged weight of the sediment particle (Eq. 2).

Many decades of assembled laboratory data (e.g., Julien 1995) suggest standard τ_{*c} values by particle class/size, which vary over a narrow range (e.g., $\tau_{*c} = 0.054$ for boulders and large cobbles >128 mm, 0.047 for coarse gravel [16–32mm], 0.029 for very coarse sand [1–2 mm], etc.). The laboratory data, therefore, suggest very little variability in τ_{*c} across grain size and no variability across other metrics, such as slope, watershed imperviousness, etc. However, field-measured Q_c values can have wide departures from theoretical thresholds (Buffington and Montgomery 1997, Parker et al. 2011, Bunte et al. 2013, Petit et al. 2015). Because of the great variability between different estimates of streambed mobilization thresholds and uncertainty in the application of laboratory data to field conditions, literature-derived estimation methods (e.g., commonly used values of τ_{*c}) cannot be expected to be accurate without validation. Further uncertainty is produced by the application of hydraulic models in translating shear-stress type thresholds into discharge thresholds (i.e., Q_c). Field data on streambed mobilization in a study stream are, therefore, required to establish appropriate Q_c values and estimation methods for a given area. Indeed, Wilcock et al. (2009) noted that even using just a few samples to calibrate streambed mobilization thresholds and sediment transport equations is the single most effective way to increase modeling accuracy. In a systematic review of 8 decades of studies on the threshold of streambed mobilization, Buffington and Montgomery (1997) concluded that much of the variability in τ_{*c} is likely due to how researchers defined the threshold of streambed mobilization in their respective studies. Moving forward, they emphasized choosing defensible

τ_{*c} values for a particular streambed mobility application as opposed to relying on the conventional practice of assuming a universal value. Many field methods were used to monitor the threshold of streambed mobilization in field studies, including radio-frequency identification or painted tracer particles (e.g., Leopold 1994, Olinde and Johnson 2015, Cain et al. 2020) and bucket or slot traps (e.g., Cantrell et al. 2009, Russell et al. 2018).

Monitoring streambed mobility in a particular stream or region not only reduces reliance on theoretical values but also ensures that the threshold of streambed mobilization is defined based on overarching strategic goals of a watershed management program. For example, streambed mobilization in river systems with endangered mussels may define Q_c as the flow that mobilizes the relevant particle-size fractions that serve as mussel habitat at various life-cycle phases. Applications focused on geomorphic stability may need to define Q_c as the discharge that begins mobilizing the particles that make up the grade control elements in a particular system (e.g., the riffle particles in pool–riffle systems and the step-forming clasts in step–pool systems).

This paper aggregates streambed mobilization data from a gradient of urban streams in 3 regions with chronic channel instability attributable, in part, to excess streambed disturbance (e.g., Fig. 1A–H; Vietz et al. 2014, Hawley et al. 2020). Here, we define Q_c events as discharges that mobilize enough of the bed material to influence geomorphic stability, with events typically mobilizing particles up to and beyond the D_{50}. All study regions covered a gradient of watershed urbanization, spanning 0.3 to 65% total impervious cover (TI) across all regions. Among other factors, urban streams may be particularly sensitive to departures from theoretical mobilization thresholds because of increased disturbance frequency and less time to armor between disturbance events (Masteller et al. 2019) and amplified fine-sediment loads attributable to excess streambank erosion (Trimble 1997, Simon and Klimetz 2008, Russell et al. 2017, Hawley et al. 2020) that can make the native bed material more susceptible to mobilization, especially in cases where the fine sediment is sand as opposed to silt or clay (Wilcock and Kenworthy 2002, Wilcock and Crowe 2003).

We hypothesized that 1) as found by other more-comprehensive assessments (e.g., Buffington and Montgomery 1997, Parker et al. 2011, Bunte et al. 2013, Petit et al. 2015), field-derived Q_c values may depart from theoretically derived Q_c estimates that use standard values for τ_{*c} (e.g., 0.047 for coarse gravel); and 2) the magnitude of departure between field- and theoretically derived Q_c estimates would increase in more urban watersheds, potentially because of less recovery time to develop an armor layer between events (Masteller et al. 2019) or higher-than-typical sand content (Wilcock and Kenworthy 2002, Wilcock and Crowe 2003), among other factors.

METHODS

In this study, we synthesized data from 3 study areas in diverse physiographic contexts: Melbourne, Victoria, in Australia, and Northern Kentucky and Austin, Texas, USA.

Study areas and datasets

The 3 study regions have similar annual rainfall (~800–1200 mm/y) and climate classifications. The Victoria region has an oceanic climate, the Texas study region has a humid subtropical climate, and the Kentucky region lies on the boundary of the humid subtropical and humid continental climate zones (Peel et al. 2007). The Kentucky and Texas streams flow mostly through limestone surface geology, whereas the streams in Victoria flow through a mix of extrusive igneous (rhyolite/rhyodacite) and sedimentary (sandstone/siltstone) geology. The

Victoria study area has higher relief than the Kentucky and Texas study areas, with several catchments located in the Dandenong Ranges, which reach an elevation of 633 m a.s.l. The channels span a gradient of sizes and drainage areas; however, they are all single-thread channels in relatively gentle, unconfined valleys with channel slopes ranging from 0.08 to 2.4% occupying the dune–ripple and pool–riffle process domains (as developed by Montgomery and Buffington 1997).

The Victoria data are derived from 9 headwater streams in Eastern Melbourne (Table 1), previously described in Russell et al. (2018, 2020). The region has mean annual rainfall of ~800 to 1200 mm, which is fairly evenly distributed throughout the year. The streams range in catchment area from 4 to 15 km^2. The level of urban development in their catchments ranges from fully forested conservation reserve to fully suburban (dominated by detached single family dwellings but with some higher density developments), with TI ranging from 0 to 49% (mean of 19%). Of the Victoria study sites, the more urbanized catchments tend to produce greater sediment supply and transport rates and have coarser bedload grain sizes (Russell et al. 2018, 2020). Stream slopes ranged from 0.6 to 2.3%, and D_{50} ranged from 0.7 to 16 mm.

The Kentucky data are from a subset of long-term hydrogeomorphic monitoring sites (Hawley et al. 2020) that were selected for streambed mobility monitoring because of their proximity to a United States Geological Survey (USGS) discharge gauge or where a site-specific Q_c estimate was warranted for a particular stormwater intervention project (Table 1). Drainage areas range from 3 to 97 km^2, with TI ranging from 6 to 29% in the predominately suburban watersheds. Stream slope ranges from 0.5 to 2.4%, and D_{50} ranges from 57 to 73 mm. The humid climate delivers mean annual precipitation of 1170 mm (NCDC 2002) that is typically distributed with the highest precipitation in the spring and lowest in autumn; however, all seasons can experience large storms (e.g., hurricane-forced systems in autumn, convective thunderstorms in summer, and so forth).

The Texas data are from hydrogeomorphic monitoring sites established by the City of Austin Watershed Protection Department to explicitly collect streambed mobilization data (Table 1). The sites range in drainage area from 0.25 to 22 km^2 with 4 to 65% TI. Stream slopes range from 0.08 to 1.5%, and D_{50} ranges from 9 to 43 mm. Mean annual precipitation is ~890 mm, with seasonal peaks in the spring and autumn and the largest individual events being driven by autumn hurricanes.

Bed sediment mobility

For this study, we used a combination of volumetric and tracer particle methods to evaluate bed sediment mobility in the field. For the Victoria sites, custom-made bedload slot traps were used as described in Russell et al. (2018). Sediment deposited in the traps was collected after major storm events (>10 mm rainfall), 28× over a 1-y period (13 July 2016–2 August 2017). The amount of sediment collected (proportion of trap filled, by volume), the maximum particle size mobilized (D_{max}), and the 90th percentile particle size mobilized (D_{90}) relative to the bed surface D_{50} were used as indicators for bed sediment mobility.

The Kentucky and Texas sites both used a combination of bucket traps (modified after Cantrell et al. 2009) and spray-painted tracer particles on a bar in the channel (modified after Leopold 1994) (methods summarized in COADWP 2020). The monitoring setup included 2 rows of gravels and cobbles across standard template sizes of 32, 45, 64, 90, 128, and 180 mm, sized via an industry standard gravelometer/half phi template (Wildco®, Yulee, Florida; Potyondy and Bunte 2002). In both settings, events that mobilized tracer particles >D_{50} tended to coincide with events that captured appreciable volumes and sizes of sediment in the bucket traps. For each event, all mobilized tracer particles were recorded, along with the depth of sediment and largest particle in the bucket trap.

Bed surface sediment characterization

When studying streambed mobilization, it is important to systematically characterize the composition of streambed particles (e.g., cobbles vs gravels vs sands or combinations thereof). For the Victoria sites, 100 pebble counts were undertaken at each site via a diagonal zigzag walk to select a representative sample of bed surface particles (Bunte and Abt 2001). Each particle's *b*-axis (i.e., intermediate axis) was measured and the median was adopted as the bed surface D_{50}. Boulders, including rock protection boulders in urban streams, were not considered part of the mobile bed (because they were designed not to move) and were excluded from analysis. At 2 sites (Bungalook Creek [Bg] and Little Stringybark Creek [Lt]), the bed was sandy or clayey, and pebble count methods were not practical. For those 2 sites, the D_{50} of sediment trapped in the bedload traps was adopted as the relevant bed particle size.

For the Kentucky and Texas sites, 100-particle pebble counts were conducted (after Bunte and Abt 2001) and pebbles were measured via the handheld gravelometer/half phi template (Potyondy and Bunte 2002). All particles <2 mm were recorded as fines and were not further segregated. Particle counts were conducted at monitoring sites, which were established at representative riffles with relatively uniform streambeds (i.e., typically in straight reaches). One bed material sample at the beginning of the monitoring period for each of these gravel/cobble dominated sites was assumed to be representative for that site, with the exception of Tannehill (TH) in Texas, where visibly notable changes in the geometry and bed material composition suggested the need to resurvey and conduct a new pebble count. In the initial sample at TH on 19 November 2019, 17% of the particles were <2 mm and the D_{50} was 11.6 mm. In the follow up sample on 30 June 2020, only 3% of the particles were <2mm and the D_{50} was 23.9 mm. The characteristics from the June 2020 survey were adopted for the analysis because they better aligned with the bed mobility monitoring period, which ran from March 2020 to January 2021. However, we include the details from the original bed sample because it underscores some of the inherent complexities of conducting geomorphic monitoring in unstable urban streams with limited resources where repeat pebble counts might not always be possible. As a point of comparison, if the fine particles were withheld from the 1st sample, the D_{50} would have been 24.7 mm, nearly identical to the D_{50} of the 2nd sample that was collected after the fines were flushed from the reach.

Hydrology

Peak discharge is another important variable in streambed mobility studies, and multiple hydrologic monitoring methods were used across the study regions. For the Victoria sites, flow was gauged continuously at each site with a water-level gauge combined with a rating curve derived from repeated flow measurements in the field (Walsh et al. 2015, Russell et al. 2018). For 4 of the sites, field measurements over a limited period were supplemented by the United States Army Corps of Engineers Hydrologic Engineering Center River Analysis System (HEC-RAS) models to extend rating curves, as described by Russell et al. (2018). The peak discharge (m^3/s) for each sample period was extracted from the flow record for each site.

All the Kentucky and Texas sites had a simple crest stage gauge, after Sroka (2003), which served as a backup to pressure transducer gauges (Texas) and USGS gauges (Kentucky).

We converted the peak stage from the pressure transducers for each event to peak discharge via standard hydraulic models for each site in HEC-RAS. We then used HEC-RAS models to develop the theoretical Q_c estimates, after Hawley and Vietz (2016), discussed in more detail below.

Design flows with particular annual exceedance probabilities (i.e., probability of an event occurring in any given year) are commonly used in stormwater management (e.g., a 1-in-100-y event has a 1% chance of occurring in any given year, a 1-in-2-y event has a 50% chance of occurring in any given year, etc.). We estimated the 1-in-2-y peak discharge for undeveloped conditions (Q_2) for each site based on regional methods, as a basis for scaling flow for comparison between sites (Watson et al. 1997, Hawley and Vietz 2016). For the Victoria sites, the Regional Flood Frequency Estimation method developed for Australian Rainfall and Runoff (Rahman et al. 2015) was used to develop Q_2 estimates. In Kentucky and Texas, the USGS regional equations were used to develop Q_2 estimates, after Hodgkins and Martin (2003) and Asquith and Slade (1997), respectively. Estimating Q_2 with these regional equations produces non-urban flow estimates that are expected to underestimate actual flood frequency in urban streams. Rather than actual flow estimates, the Q_2 values in this study are conceived as scaling parameters to scale measured flows to the catchment area and climate of each site location (Watson et al. 1997, Hawley and Vietz 2016). Q_2 is a commonly computed and referenced flow across regions, making it even more tangible as a scaling parameter.

Q_c estimation from field data

The overarching goal of this analysis was to identify the Q_c threshold for discharges that can mobilize streambed material and reshape channels. Although changes to channel geometry typically require time to be measurable, repeat surveys at some of our longer-term sampling sites underscore that these channels are experiencing channel-shaping events (e.g., Hawley et al. 2016, 2020, 2022). Indeed, some of the Texas sites had such visibly obvious channel changes that they had to be re-surveyed during the bed mobility study.

With the aim of discerning geomorphically meaningful events, we estimated Q_c at each site by classifying each sampled event as either a bed-mobilizing or nominal event. No sediment sampling method can perfectly capture streambed sediment mobility. For example, sediment traps might interfere with hydraulics, may not span the full channel width, may not have a large enough opening to capture large particles, or may fill too quickly with smaller particles to have room for the larger particles moved later during the event. It is also possible that atypically coarse particles just happen to fall into the trap because of colluvial processes (e.g., bank collapse) or low-probability stochastic sediment movement (e.g., a particularly protruding particle rolls or is knocked into the trap). Similarly, tracer particles may not be imbedded in the armor layer prior to the event, like the native substrate, or may be placed at too high an elevation to be submerged during an event. Therefore, we used multiple lines of evidence to classify events as bed-mobilizing or nominal. The 1st line of evidence identified events that mobilized particles larger than the bed surface D_{50}, which we estimated from the maximum particle size captured in sediment traps or the maximum tracer particle size moved. The 2nd line of evidence was the proportion that sediment traps were filled during the event. A 3rd line of evidence was the proportion of tracer particles that were mobilized. All lines of evidence were tailored to the respective settings to keep the focus on geomorphically meaningful events that could drive changes to the channel size or shape.

In Kentucky, bed-mobilizing events were classified as those that mobilized large amounts of sediment (bucket traps 50–100% full) and those that mobilized tracer particles greater than the bed D_{50}. By contrast, events classified as nominal did not mobilize tracer particles larger than the D_{50} and typically had empty bucket traps, with the highest nominal event only having a 30%-full bucket. The coarse beds, larger watersheds, and longer durations of high flows at the Kentucky sites probably contributed to the clear difference between bed-mobilizing events and nominal events.

In Texas, the difference between bed-mobilizing and nominal events was fuzzier than for the Kentucky data. Bed-mobilizing events almost always mobilized tracer particles larger than D_{50}, and often larger than D_{84}. They also typically had very full buckets. However, sometimes the short-lived peak flows in the smaller watersheds did not have enough time to appreciably fill the buckets. In other cases, the bucket caught particles larger than D_{84}, but the tracer particles did not mobilize because of being set on too high of a bench to become submerged. Therefore, classification of events as bed-mobilizing or nominal also considered the proportion of tracer particles mobilized, with bed-mobilizing events not only mobilizing D_{50} and, often, D_{84} particles, but also the majority of both rows of bar tracer particles.

By contrast, nominal events in Texas typically had buckets that were empty, although they sometimes had buckets that were ≤8% full. One site (Metcalfe) had several nominal events with buckets between 13 and 68% full, but it was clear that the bucket sediment came from failure of the adjacent bank, which explained why none of the bar particles were mobilized. At Rinard downstream (R/DS), 3 nominal events were recorded as having 1 of the 128-mm tracer particles mobilized, but the buckets were only ≤5% full and did not capture particles larger than D_{50}. Furthermore, most tracer particles were not mobilized. In these cases, mobilization of a single large tracer particle (~10× larger than D_{50}) was suspected to be attributable to exceptional grain protrusion (e.g., Gob et al. 2010, Masteller and Finnegan 2017) as opposed to being indicative of a bed-mobilizing event. Additionally, events with missing data (e.g., events without installed bucket traps or missing data entries) were withheld from the analysis. The rare instances where the bucket trap was dislodged or the channel shifted so much that the geometry visibly changed required channel re-surveys to update the hydraulics as necessary to ensure comparability of the sediment event data.

In Victoria, we classified events as bed-mobilizing events if a substantial proportion (>10%) of the mobilized material was coarser than the bed D_{50} (i.e., D_{90} of the mobilized material > D_{50} of the streambed) or if the trap was >20% full and the bed D_{50} was mobilized (i.e., D_{max} of the mobilized material > D_{50} of the streambed). Particles >D_{50} tended to be mobilized when the slot traps were >20% full. The 20%-full threshold was lower than the 50%-full threshold found for the Kentucky data because of different trap designs. The wider, shallower design of the Victoria slot traps, and finer-grained bedload, meant that the Victoria traps were likely less efficient in capturing mobilized bedload sediment than the deeper bucket traps. Additionally, and similarly to the Texas sites, there may not have been enough time to fill the traps during flashy, short-lived events in the relatively small watersheds.

Once all events were classified as bed-mobilizing or nominal events, we adopted the mean of the peak flow for the highest nominal event and the lowest bed-mobilizing event as the field-derived Q_c. The nuances from all regional datasets underscore both the challenges of collecting this type of data in different settings and the importance of transparently minimizing bias to defensibly classify the threshold for streambed mobilization based on the goals of the program and management application (Buffington and Montgomery 1997). The threshold may indeed be fuzzier in some settings, but events that completely fill the buckets and mobilize tracer particles larger than D_{84} or D_{90} should consistently be above the field-derived Q_c threshold, whereas events that mobilize no tracers and capture no sediment in the traps should consistently be below the threshold. Beyond that, the precision of a field-derived Q_c threshold is not only a function of the quantity and quality of the data but also the magnitude of the events captured during the monitoring period.

Theoretical Q_c estimation

To compute theoretical Q_c for the bed surface D_{50}, we used conventional τ_{*c} values from laboratory studies assembled by Julien (1995), which document a narrow range of 0.029 to 0.052 for the particle sizes of interest from coarse sand to small cobbles (see Boxes 1, 2, Eqs 1, 2). Furthermore, as originally developed by Shields (1936), these commonly used estimates include no dependence between τ_{*c} and other variables, such as slope or watershed imperviousness, although slope-dependence has been observed in more recent lab and field studies (e.g., Lamb et al. 2008). To calculate critical shear stress (τ_c), we used the standard formula (Box 1, Eq. 2) by substituting standard τ_{*c} values for τ_* and then solving for τ.

Consistent with the previous analysis of 195 sites by Hawley and Vietz (2016), we presumed the bed D_{50} to be the representative particle for when most of the bed becomes mobilized. This commonly used assumption in the field of river mechanics is consistent with the concept of equal mobility (e.g., Parker and Klingeman 1982, Parker et al. 1982, Parker and Toro-Escobar 2002). The flow that coincides with this τ_c (i.e., the theoretical Q_c) depends on the hydraulic radius (R) and energy slope (S) (Eq. 1). Hydraulic models are, therefore, required to convert the τ_c to a Q_c (Box 2). For the Victoria sites, we used the HEC-RAS models developed by Russell et al. (2020). For the Texas sites, we used HEC-RAS models developed by staff at Sustainable Streams (Louisville, Kentucky). For the Kentucky sites, we used HEC-RAS at the Gunpowder and Horse Branch sites, where we had more expansive profile surveys, and simplified hydraulic models for the remaining sites that assumed a reach-averaged water surface slope that parallels the riffle-crest-to-riffle-crest slope (Hawley and Vietz 2016).

Working backwards using Eqs 1 and 2 (Box 3) and standard hydraulic models based on standard geometric cross-section and profile surveys after Harrelson et al. (1994), we converted the field-derived Q_c estimates to τ_{*c} values to allow comparison with the theoretical τ_{*c} estimates. The use of standard hydraulic models to translate measurements of depth to discharge and vice versa underscores the well-established parallelism between the 2 measures, whereas there has been much less ground-truthing of laboratory-derived τ_{*c} values in field settings. The selection of a single representative cross section is common for such reach-based analyses (e.g., Wolman and Miller 1960, Hawley and Vietz 2016). A single survey at the beginning of the monitoring period was presumed to be representative of the channel geometry of each site for the duration of the monitoring period, with the exception of East Bouldin (EB) in Texas, where visibly apparent changes suggested the need for additional surveys. The substantially reduced slope (0.33% post-May 2020 compared with 0.63% pre-May 2020) combined with the more-entrenched cross section (>100 cm deeper post-May 2020 compared with pre-May 2020) demonstrated the need for separation of the pre-/post-May 2020 data as independent sites. Indeed, both their theoretical and field-derived Q_c estimates were appreciably different (pre-May 2020 theoretical = 6% of Q_2 and field-derived = 5% of Q_2 compared with post-May 2020 theoretical = 14% of Q_2 and field-derived = 17% of Q_2).

Statistical analysis

We tested our hypotheses by fitting linear models to explore the relationship between Q_c and catchment/stream characteristics. Based on previous work by Hawley and Vietz (2016) that used theoretically derived thresholds at 195 sites, we expected Q_c to increase as a power function of bed D_{50}. In this study of field-derived thresholds, we also wished to test the dependence of Q_c on catchment urbanization levels, as expressed through TI. In addition, we initially tested slope dependency, but this effect was not detected for any region. We developed linear models for the response variables: theoretical Q_c, field-derived Q_c, and field-derived τ_{*c}. For the field-derived response variables, we performed censored Gaussian regression with the *survival* package (version 3.2-13; Therneau et al. 2021) in R (version 4.1.1; R Project for Statistical Computing, Vienna, Austria). Three sites in Victoria did not record any nominal events, limiting our ability to establish field-derived thresholds for Q_c and τ_{*c}, so we included the observed upper bound from these 3 sites as opposed to a known threshold. Censored regression is common in the fields of survival analysis (e.g., Klein and Moeschberger 2003) and flood frequency analysis (England et al. 2017) and allows information to be included for data points known only to be above or below a certain threshold. For theoretical Q_c, we performed normal multiple linear regression in R. Predictor variables included in all models were TI and bed D_{50} as well as a categorical dummy variable (0 or 1) indicating the country (i.e., USA or Australia). We confirmed the need for country differences by exploratory modeling. Country differences were only apparent in the intercept, not the coefficient or errors, nor was there evidence of an interaction between TI and bed D_{50}. We excluded variables whose model coefficients did not achieve p-values ≤ 0.05. We natural-log-transformed all continuous variables, producing power functions as their outputs.

RESULTS

Field-derived Q_c thresholds were estimated from field data and then tested for correlations against potential dependent variables such as TI and D_{50}. We estimated field-derived Q_c thresholds for all but 3 fine-bedded streams in Victoria (Bg, Lt, and Bushy Creek), where no nominal events were recorded because the beds were mobile during every monitored event (Fig. 2C). Across all regions, Q_c increased as a power function of both TI and D_{50} (Fig. 3C, D, Table 2). Theoretical Q_c had a meaningful but relatively weak relationship with TI, whereas the relationship for field-derived Q_c was much stronger and diverged from theoretical values at high levels of TI (e.g., compare the slopes of the respective theoretical and field-derived regression lines in Fig. 3D). Our expectation of a relationship between Q_c and D_{50} (following Hawley and Vietz 2016) was confirmed for both the theoretical and field-derived Q_c values for this study (both $p < 0.001$). There were also differences between the Australian and USA sites ($p < 0.001$), which were particularly apparent in their response to bed D_{50} (Fig. 3A, C). We detected differences between the Australian and USA sites in both the theoretical and field-derived Q_c values, which tended to be higher in Australia than the USA for a given bed sediment size. For the USA sites, the relationship between Q_c and D_{50} nearly perfectly aligns with the empirical relationship developed by Hawley and Vietz (2016) from 195 sites across California, USA, Kentucky, and Victoria that were based on theoretical τ_{*c} values (Fig. 3A, C).

As originally developed by Shields (1936), the τ_{*c} parameter should theoretically have little dependence on D_{50} and no dependence on imperviousness. Accordingly, we found no relationship between field-derived τ_{*c} and D_{50} (Fig. 4A). However, there was a strong positive correlation between τ_{*c} and TI, with no difference between regions (Fig. 4B, Table 2). Streams draining undeveloped watersheds (TI: <5–10%) tended to have field-derived τ_{*c} values lower than the theoretical τ_{*c} values, whereas the more urban the watershed, the more likely the field-derived τ_{*c} value exceeded the theoretical τ_{*c}. The relationship between field-derived mobilization threshold and TI persists despite different non-dimensionalization methods (i.e., % Q_2 vs τ_{*c}), providing strong evidence that streams with higher catchment urbanization levels have higher bed mobilization thresholds in this study.

Interestingly, 1 of the most impervious sites in Texas (TH, 62% TI) had 1 of the largest departures between field-derived Q_c (42% of Q_2) and theoretical Q_c (14% of Q_2 based on the 30 June 2020 bed sample with only 3% fines). The departure would have been even greater if we had relied on the earlier pebble count. For comparison, the theoretical Q_c drops to 3% of Q_2 when using the original 19 November 2019 bed sample that contained 17% fines. However, it is worth noting that when withholding the fine content from the original sample to compute a D_{50} without the influence of fines, the theoretical Q_c increases to 15% of Q_2, which is nearly identical to the Q_c estimate derived from the subsequent bed sample with only 3% fines.

DISCUSSION

The purpose of this study was to test whether field-derived bed mobility thresholds in urban streams departed from theoretical bed mobility thresholds and whether such departures were correlated with the magnitude of watershed urbanization. As documented, across 3 regional datasets, both Q_c and τ_{*c} were positively correlated with TI, suggesting that something about urban streams makes them exceptionally prone to depart from theoretical streambed mobility thresholds. Numerous potential causes for the departure of urban stream mobility thresholds from theoretical values are discussed below, including the amplified urban disturbance regime, higher rates of change in the channel geometry and bed composition, and potentially higher sand fractions easing gravel entrainment.

Above all else, this synthesis of data collected across continents underscores the importance of site-specific streambed mobility data in managing or designing interventions aimed at influencing streambed particle mobility. Other researchers in rural field studies have documented considerable departures from theoretical τ_{*c} values (Wilcock 1993, Buffington and Montgomery 1997, Parker et al. 2011, Petit et al. 2015), and this study from urban streams extends that trend. Furthermore, not only did our data show departures from theoretical mobilization thresholds, but the departure exhibited a positive correlation with watershed urbanization (note the relatively steep relationships in Figs 3D and 4B). It should be noted that the coefficient of determination is only 0.37 for the relationship of τ_{*c} as a function of watershed imperviousness, and the trend could benefit from additional data (especially in non-urban watersheds). However, urban watersheds in this dataset almost universally had higher field-derived Q_c and field-derived τ_{*c} values than would be expected from thresholds associated with theoretical τ_{*c} parameters (Figs. 3D, 4B).

Furthermore, there were important differences in some of the trends between countries. The coefficients of the Q_c power functions varied by country, whereas the differences between countries disappeared when the threshold was expressed in terms of τ_{*c}. Although the Australian streams had similar thresholds to comparable USA streams in terms of shear stress acting on bed particles, the flow that produced that shear stress was higher. This difference is most likely attributable to the fact that the Australian streams had higher form roughness (e.g., due to large wood, sinuosity, and vegetation—particularly for the non-urban streams), meaning that a greater proportion of flow energy is dissipated and not available to mobilize particles. The hydraulic modeling approach for Victoria assumed a drag partition that accounted for form roughness (Russell et al. 2020); therefore, the theoretical and field-derived Q_c both diverged from the Kentucky and Texas data in a consistent manner. τ_{*c} thresholds may be more consistent between regions than Q_c thresholds, but care needs to be taken to ensure that hydraulic models used to convert between the 2 reflect reality.

As noted above, although more data are needed to explore the relationship between Q_c and τ_{*c} relative to watershed urbanization, insights from previous studies may help illuminate why urban sites could depart from theoretical streambed mobilization estimates. First, amplified rates of streambed disturbance in urban streams (Papangelakis et al. 2019, Cain et al. 2020) attributable to more frequent discharges that exceed Q_c (Booth and Jackson 1997, Bledsoe 2002, Hawley et al. 2017) could make the streambed more prone to mobilization relative to a comparable rural stream that has more time to armor between disturbance events (Masteller et al. 2019). That is, a rural stream with less-frequent disturbance events would have more time to sort into a coarser (armored) surface layer and a finer subsurface layer, making the streambed less susceptible to mobilization even if it had an identical gradation to an urban site. Second, chronic streambank erosion (Chin 2006, Hawley et al. 2020, Sullivan et al. 2020) and channel enlargement (Hammer 1972, Hawley and Bledsoe 2013, Chin et al. 2017, Bevan et al. 2018) can load large volumes of fine sediment to the stream bed (Trimble 1997, Simon and Klimetz 2008, Russell et al. 2017), which can make otherwise resistant gravels and cobbles more prone to mobilization (Wilcock and Kenworthy 2002, Wilcock and Crowe 2003). However, these 2 mechanisms would only explain thresholds that are lower than expected from theory (e.g., EB, Threemile Creek in Kentucky), but not the more common situation of thresholds that are higher than expected. In the case of higher-than-expected thresholds, the sampling methods themselves may be less optimal for the flashy urban environment. For example, it is possible that, given supply limitations in urban streams (Russell et al. 2020) and flashy, short-lived flows, there is not enough time for bucket traps to fill adequately to record a response (although, given the flashy nature of urban streams, such flashy events still help to illuminate the variety of geomorphic responses in urban streams). Similarly, the probability of tracer particle movement increases with flow duration, and, thus, short-lived urban flow events are less likely to be recorded as bed-mobilizing (Olinde and Johnson 2015). Both of these potential methodological limitations underscore the value of using multiple lines of evidence to assess bed-mobilizing events in urban channels.

Furthermore, chronic instability and complex responses, such as headcuts and slope adjustments (Schumm et al. 1984, Hawley et al. 2012), bed coarsening (Pizzuto et al. 2000, Hawley et al. 2013a), and entrenchment (Anim et al. 2019, Hawley et al. 2020), among other common urban stream responses, have the potential to adjust hydraulics and bed composition more quickly than comparable rural streams, such that the geometry and bed sample collected prior to the Q_c monitoring may not be representative of the conditions during the Q_c sample events. As demonstrated at the EB site in Texas, geometric changes (more entrenched cross section with flatter slope) resulted in different theoretical and field-derived Q_c thresholds between the pre-May 2020 (6 and 5% of Q_2, respectively) and post-May 2020 (14 and 17% of Q_2, respectively) datasets. Additionally, had the field crew not collected a 2nd pebble count at the TH site in Texas, the theoretical Q_c estimate based on a finer D_{50} that was skewed by 17% fines would have departed even more from the Q_c that was measured in the field.

Another insight worth noting is the consistency in field-derived Q_c estimates from the Texas study at the Rinard upstream (R/US) and R/DS (both 2% of Q_2) compared with the wide discrepancy between their theoretically derived Q_c estimates (40 and 0.4% of Q_2, respectively). This unique upstream–downstream combination of sites on the same stream helps to underscore the sensitivity of theoretical Q_c estimates to D_{50} (R/US: 23 mm, R/DS: 14 mm). As the energy grade line flattens out with increasing depth, there is a large gap between the flow that can mobilize a 14-mm particle vs a 23-mm particle, whereas the actual Q_c is likely to be fairly consistent between 2 adjacent reaches in the absence of large sediment discontinuities, headcuts, or other geomorphic factors that would drive different Q_c thresholds on adjacent reaches.

In short, with τ_{*c} being so sensitive to slope, cross-section shape, and D_{50}, the dynamic nature of urban streams makes Q_c estimates that are exclusively based on theoretical τ_{*c} values naturally prone to space–time errors. For example, a single hydrogeomorphic survey and pebble count may not be representative of long-term average conditions. This sensitivity to slope, shape, and D_{50} also underscores the importance of site selection and the challenges of geomorphic monitoring in urban streams. As documented at several sites in this study, even the least unstable and most uniform reaches that were carefully selected for bed-mobility monitoring were prone to streambed composition and geometry dynamics that influenced streambed mobilization thresholds within the duration of the monitoring period.

The departure of field-derived from theoretical Q_c estimates demonstrates the importance of bed-mobility monitoring. Fortunately, monitoring streambed mobilization in urban streams does not need to be an expensive effort to provide value to stormwater management programs. The low-tech bucket traps and spray-painted tracer particles used in the Texas and Kentucky studies provided more than enough mobility data to bracket field-derived Q_c estimates. Following on the recommendations of Buffington and Montgomery (1997), the streambed-mobilization monitoring effort should be defensible for the desired application. For example, theoretically derived estimates may be a reasonable and conservative starting point to get an idea of Q_c in a stream or region. For urban streams (TI > 5–10%), theoretical τ_{*c} tends to consistently underestimate field-derived τ_{*c} and could, thus, provide an approximate lower bound on the mobilization threshold. Conversely, field-derived estimates may be more appropriate for individual projects where investments are aimed at reducing Q_c exceedances for a specific goal (e.g., reduced disturbance of benthic macroinvertebrates, improved geomorphic stability). In other instances, volumetric samplers may be useful to calibrate volumetric sediment transport equations to develop total sediment loads in a stream or watershed. Volumetric data become particularly important in fine-grained streams where thresholds are frequently exceeded and for long durations (e.g., the sand-bedded Victorian streams).

As mentioned by Wilcock (2009), even just a few samples can provide improved accuracy for estimating streambed mobilization thresholds. Although streambed mobility data collection can be challenging for inexperienced staff or for a single short-lived project, we hope that the low-cost methods described here will improve stormwater managers' ability to collect calibration data on their streams. Consistent methods by a centralized agency and data collection over a longer timeframe and across >1 site can help to build confidence in the data and Q_c thresholds and to reduce the influence of potential outliers on calculations of Q_c. Monitoring can also lead to regional insights, such as the apparent bed-stabilizing effects of large wood in fine-grained systems and the equal mobility behavior in the coarser gravel/cobble systems. Finally, additional bed mobility monitoring will likely lead to continued methodological advances for the urban stream monitoring community. As demonstrated by this synthesis, what works best in fine-grained systems (traps in Victoria) may be different from coarse-grained systems (tracers in Kentucky), whereas multiple strategies may be extremely helpful in mixed fine/coarse systems (tracers and traps in Texas). The best advice for assigning Q_c categories within a new system is to look for clear breaks in behavior between categories and assign Q_c based on the project or management goals. For example, interventions aimed at reducing juvenile mortality of endangered freshwater mussels may delineate Q_c for the specific particle class that is relevant to the juvenile lifecycle stage, whereas near full-bed mobilization may be more appropriate Q_c when trying to discern discharges that can substantially influence the macroinvertebrate community.

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Box 1: Glossary of select terms, symbology, and relevant equations.

Shear stress (τ): The force of moving water expressed along the boundary of the streambed (N/m^2). For gradually varied flow conditions in streams with mild to moderate slopes, τ can be calculated with the following equation:

$$\tau = RSg_{\mathrm{w}}, \quad \text{(Eq. 1)}$$

where R is the hydraulic radius (m) associated with the given flow depth. R can be simplified to flow depth (m) when partitioning the τ across the channel (e.g., to calculate the maximum τ in the channel thalweg). S is the slope of the energy grade line (m/m), which is similar to both the streambed slope and the water surface slope in gradually varied flow conditions in mild to moderately sloped streams. g_{w} is the specific weight of water (9810 N/m^3).

Critical shear stress (τ_{c}): The τ associated with the threshold for streambed mobilization.

Shields parameter (τ_*): The dimensionless form of τ as developed by Shields (1936) per the following equation, as presented by Julien (1995):

$$\tau_* = \frac{\tau}{(\gamma_{\mathrm{S}} - \gamma_{\mathrm{W}})D_{\mathrm{S}}}, \quad \text{(Eq. 2)}$$

where τ is shear stress (N/m^2), γ_{S} is the specific weight of sediment particles (26,000 N/m^3), and D_{s} is the diameter of the sediment particle on the streambed (m). In streambed mobility studies, the median diameter (D_{50}) is typically used as the representative particle size.

Critical Shields parameter ($\tau_{*\mathrm{c}}$): The Shields parameter associated with the threshold for particle mobilization. In streambed mobility studies, the D_{50} is typically used as the representative particle size for the threshold for streambed mobilization.

Critical discharge (Q_c): The Q associated with the threshold for streambed mobilization.

Box 2: Theoretical values associated with the threshold for streambed mobilization.

Critical Shields parameter (τ_{*c}): Default theoretical values for τ_{*c} come from many decades of assembled laboratory data (e.g., $\tau_{*c} = 0.054$ for boulders and large cobbles >128 mm, $\tau_{*c} = 0.047$ for coarse gravel [16–32 mm], $\tau_{*c} = 0.029$ for very coarse sand [1–2 mm], etc., after Julien [1995]).

Critical Shear stress (τ_c): The theoretical τ_c associated with streambed mobility calculated by setting τ_* in Eq. 2 equal to a theoretical value for τ_{*c} (e.g., 0.047 for coarse gravel) and solving for τ.

Critical discharge (Q_c): The theoretical Q_c for streambed mobilization can be calculated by setting τ in Eq. 1 equal to the theoretical value for τ_c and solving for R. Standard hydraulic equations or models can then be used to determine the discharge associated with that R value.

Box 3: Field-derived values associated with the threshold for streambed mobilization.

Critical discharge (Q_c): The field-derived Q_c between streambed mobilizing events and non-mobilizing events. Standard hydraulic equations or models can be used to determine associated hydraulic values, such as depth and hydraulic radius (R).

Critical shear stress (τ_c): The τ calculated with Eq. 1 by using the value of R that is associated with the field-derived Q_c.

Critical Shields parameter (τ_{*c}): The Shields parameter value τ_* calculated with Eq. 2 when using the value of τ_c that is associated with the field-derived Q_c.

FIGURE CAPTIONS

Fig. 1. Examples of excessive stream instability (A, B, C, D) from the Blackland Prairie ecoregion in Austin, Texas, USA. Excessive instability can lead to stream incision, channel deepening, widening, and bank erosion, including examples of how channel erosion can expose and damage urban infrastructure (D) and undermine the inlets and outlets of large detention facilities (E, F, G, H), posing even greater risks to downstream infrastructure, property, and potentially human life. Detention basins (E, F, G, H) were <10 y old at the time of the photos, underscoring the speed at which channel erosion can degrade streams and jeopardize infrastructure. City of Austin owns photos and grants permission for their use.

Fig. 2. Bed-mobilization field data for sites in Kentucky, USA (KY) (A), Texas, USA (TX) (B), and Victoria, Australia (VIC) (C). Discharge for each bed-mobilizing or nominal event is expressed as % of the 1-in-2-y peak discharge for undeveloped conditions (Q_2). The field-derived and theoretical critical discharge (Q_c) threshold for each site is shown on the same $\log_{10}$ scale. Each point represents an event, which are randomly jittered horizontally for clarity. Points with down arrows (panel C; sites Bg, Lt, Br) indicate the assumed upper limit of Q_c for 3 sites in Victoria that were mobile in every monitored event. For each region, sites are ordered by increasing total impervious cover (TI) from non-urban, peri-urban, suburban, to urban (not all categories were present in all study regions). Median bed sediment diameter (D_{50}) is also plotted for each site. Site names and characteristics are listed in Table 1.

Fig. 3. Relationships between logarithms of theoretical critical discharge (Q_c) and bed median particle size (D_{50}) (A), theoretical Q_c and total impervious cover (TI) (B), field-derived Q_c

and bed D_{50} (C), and field-derived Q_c and TI (D) for stream sites in Kentucky and Texas, USA, and Victoria, Australia (AUS). Discharge for each bed-mobilizing or nominal event is expressed as % of the 1-in-2-y peak discharge for undeveloped conditions (Q_2). All axes are log-scaled. Points with down arrows ($n = 3$ on panels C and D) indicate the assumed upper limit of Q_c for 3 sites in Victoria that were mobile in every monitored event. Fitted regression lines are shown for theoretical Q_c (dashed lines; $R^2 = 0.68$) and field-derived Q_c (solid lines; $R^2 = 0.79$) for Australia and the USA (see Table 2). Regression lines for theoretical Q_c are repeated in panels C and D for ease of comparison with fitted regression lines for field-derived Q_c. The regression lines shown for the relationship with D_{50} (A, C) are for the mean TI for each country (i.e., Australia: 19%, USA: 30%). Likewise, the regression lines shown for the relationship with TI (B, D) are for the mean D_{50} for each country (i.e., Australia: 7 mm, USA: 33 mm). The relationship between Q_c and D_{50} found by Hawley and Vietz (2016) across 195 sites in Australia and the USA is shown as a solid black line (A, C).

Fig. 4. Relationships between field-derived bed mobility threshold expressed as critical Shields parameter (τ_{*c}) and stream/catchment characteristics across stream sites in Kentucky and Texas, USA, and Victoria, Australia. We found no relationship between τ_{*c} and streambed median particle size (D_{50}) (A), as expected from theory. However, we did find a relationship ($r^2 = 0.37$, $p < 0.001$; solid line in panel B; see Table 2) between the log of field-derived τ_{*c} and the log of total impervious cover (TI) (B), in contrast with theoretical estimates of τ_{*c}, which are independent of catchment urbanization. Points with down arrows ($n = 3$ on both panels) indicate the assumed upper limit of τ_{*c} for 3 sites in

Victoria that were mobile in every monitored event. Theoretical values are from Julien (1995).

Table 1. Study site characteristics of 3 study regions in Kentucky and Texas, USA, and Victoria, Australia. CA = catchment area, TI = total impervious fraction, D_{50} = median bed sediment diameter.

Region	Site	Site code	CA (km^2)	TI (%)	D_{50} (mm)
Northern Kentucky, USA	Woolper Creek	WPC	62.7	6	61.8
	Horse Branch	HBR	3.2	20	64.0
	Threemile Creek	THC	8.1	21	57.3
	Gunpowder Creek	GPC	97.1	21	70.5
	Dry Creek	DRC	30.6	29	73.5
Austin, Texas, USA	South Fork Dry Creek	SFD	22.4	4	22.8
	Rinard downstream	R/DS	18.4	5	13.6
	Rinard upstream	R/US	18.4	5	23.0
	Marble	MB	10.1	9	24.9
	Williamson	WM	7.9	13	33.2
	Harris Tributary	HT	3.3	26	9.1
	Decker	DK	5.6	30	24.4
	Granada	GR	0.3	42	17.4
	Burleson	BR	1.3	45	21.6
	Carson Creek	CR	3.8	49	27.3
	East Bouldin[a]	EB	4.0	49	12.4
	Shoal Creek	SH	18.7	52	42.5
	Tannehill[b]	TH	5.3	62	23.9
	Metcalfe	MC	1.0	65	28.5

Melbourne, Victoria, Australia	Lyrebird Creek	Ly	7.2	0.3	5.0
	Sherbrooke Creek	Sh	6.6	2	16.0
	Olinda Creek	Ol	8.7	5	5.0
	Bungalook Creek	Bg	6.0	7	1.7
	Little Stringybark Creek	Lt	4.3	14	0.7
	Ferny Creek	Fe	6.4	18	7.0
	Brushy Creek	Br	14.8	29	6.0
	Scotchmans Creek	Sc	5.5	47	13.5
	Forest Hill Drain	Fo	5.4	49	5.5

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[a] Critical discharge threshold detection was separated into 2 time periods: January to March 2020 (EB/1) and June to September 2020 (EB/2) because of appreciable changes in the channel geometry between sample periods (i.e., a more entrenched cross section and flatter slope post-May 2020 compared with pre-May 2020).

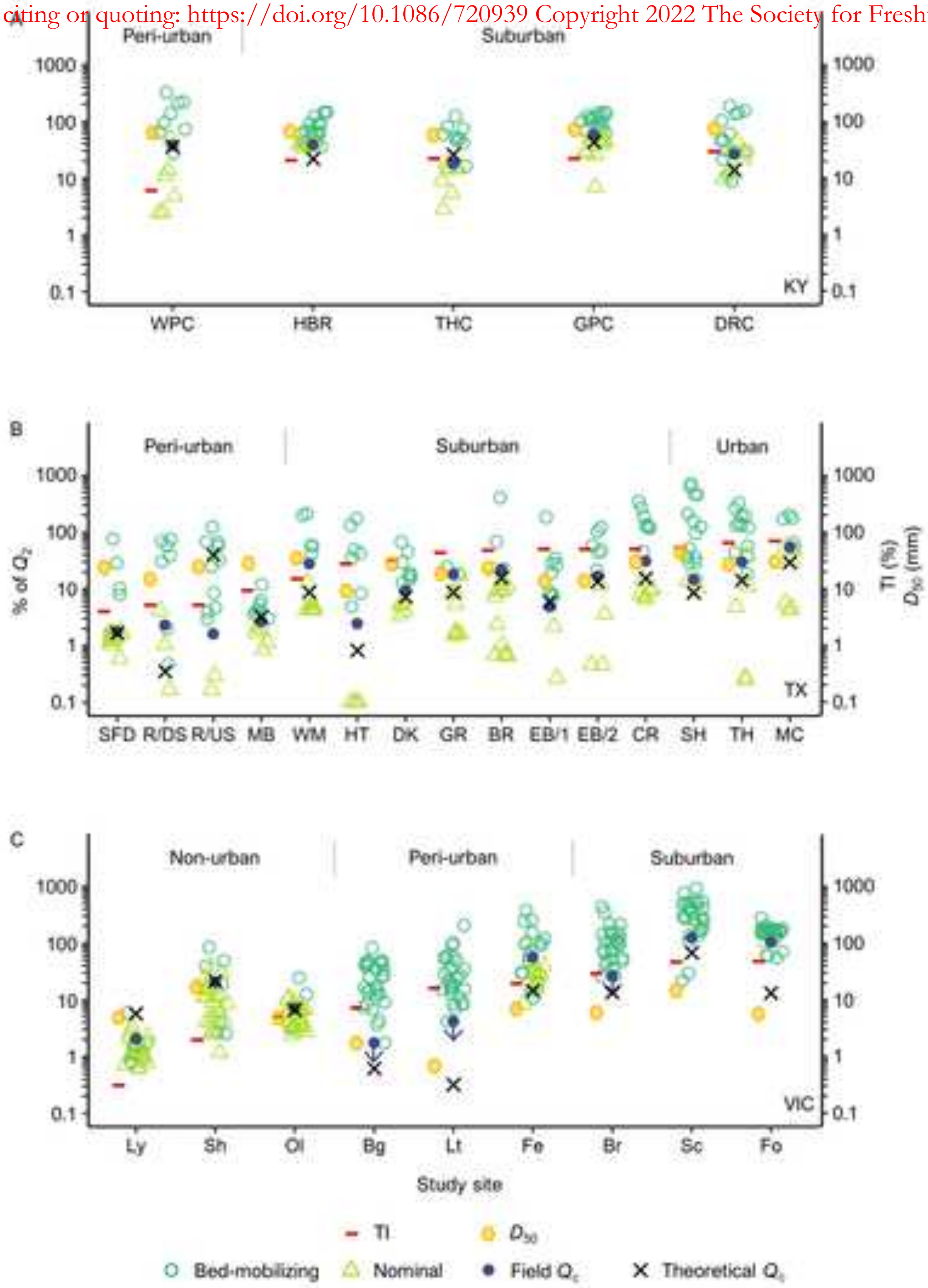
[b] Tannehill had visibly notable changes in geometry and bed material soon after the original survey, which led to a resurvey and new pebble count. The 2nd sample (June 2020) is reported in this table and was used for analysis because it better aligned with the bed-mobility monitoring period, which ran from March 2020 to January 2021.

Table 2. Fitted empirical relations between theoretical critical discharge (Q_c), field-derived Q_c, and field-derived critical Shields parameter (τ_{*c}) with total impervious cover (TI), median bed sediment diameter (D_{50}), and country (USA and Australia [AUS]). R^2 values for the fitted model and *p*-values for each coefficient are shown, as well as the fitted equation in power-function form.

Response variable	Model	R^2	*p*	Empirical equation
Theoretical Q_c	$\log(Q_c) \sim \log(TI) + \log(D_{50})$ + country	0.68	Model: <0.001	AUS: $Q_c = 0.299\ D_{50}^{1.51}\ TI^{0.33}$
			D_{50}: <0.001	USA: $Q_c = 0.022\ D_{50}^{1.51}\ TI^{0.33}$
			TI: 0.02	
			Country: <0.001	
Field-derived Q_c	$\log(Q_c) \sim \log(TI) + \log(D_{50})$ + country	0.79	Model: <0.001	AUS: $Q_c = 0.296\ D_{50}^{1.36}\ TI^{0.71}$
			D_{50}: <0.001	USA: $Q_c = 0.015\ D_{50}^{1.36}\ TI^{0.71}$
			TI: <0.001	
			Country: <0.001	
Field-derived τ_{*c}	$\log(\tau_{*c}) \sim \log(TI)$	0.37	Model: <0.001	$\tau_{*c} = 0.036\ TI^{0.15}$
			TI: <0.001	

Figure

Figure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Theoretical Q_c

Figure

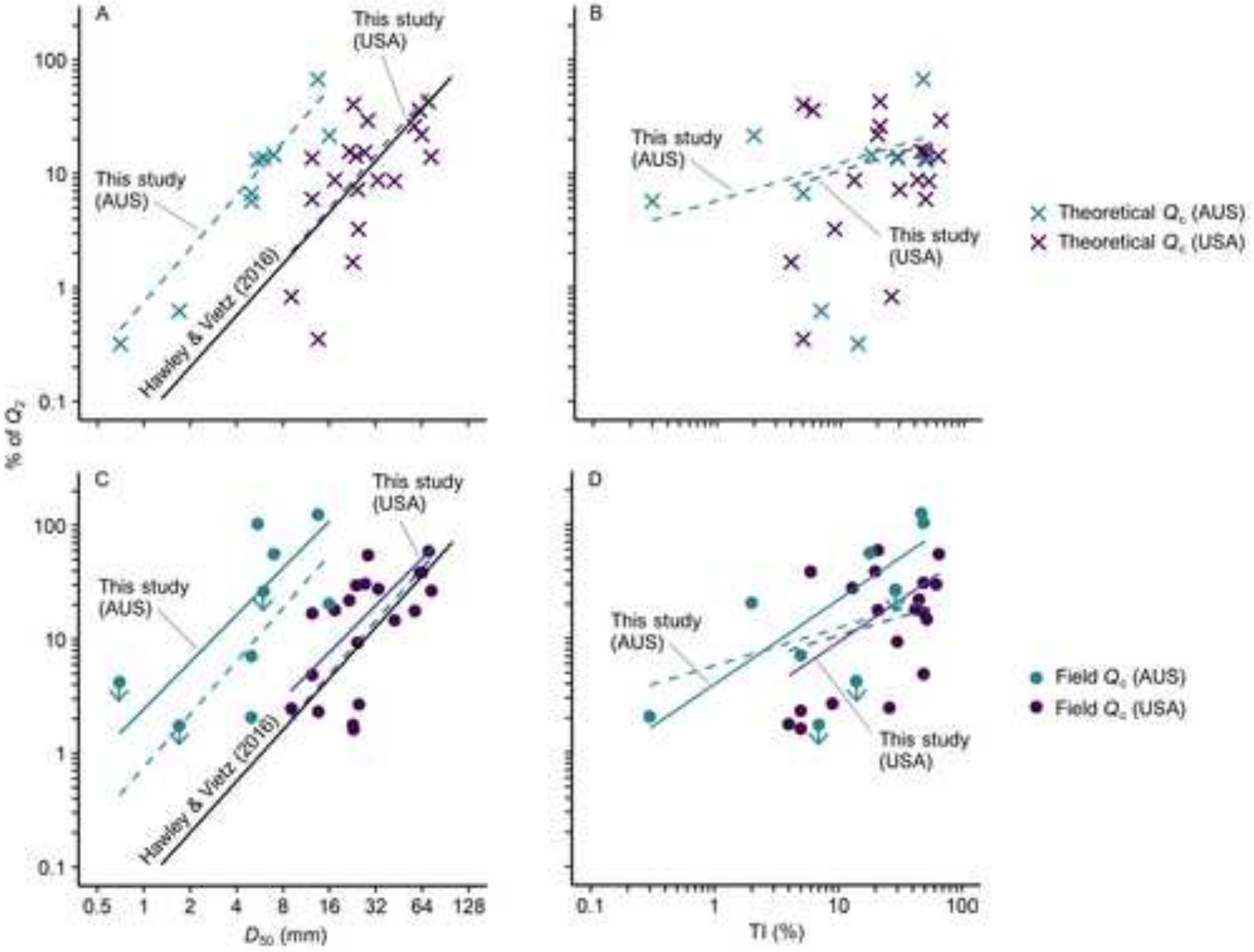
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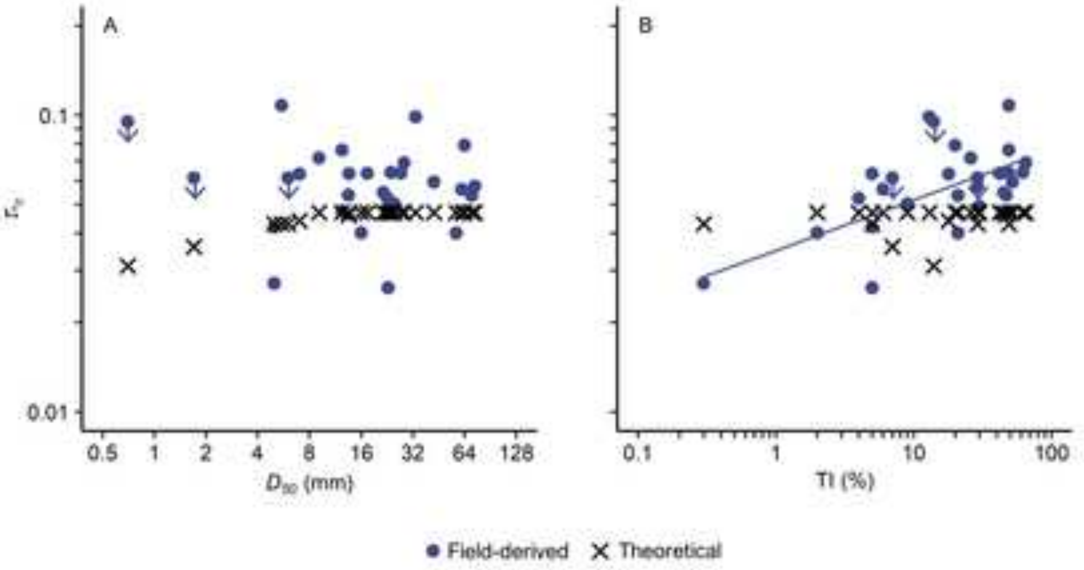
Figure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