

VALIDATING ENVIRONMENTAL FLOW RECOMMENDATIONS: DRIFTING
COARSE PARTICULATE MATTER, MACROINVERTEBRATES,
AND LARVAL FISHES

by

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ABSTRACT

Water quantity management in Texas and elsewhere is currently managed under the theory of the Natural Flow Paradigm, which states that sound ecological riverine environments are dependent upon the dynamic character of flow. Water quantity recommendations and standards prescribe a multi-tier flow regime, consisting of subsistence, base, and high-flow pulses, with magnitudes of each based on long-term averages of site-specific hydrographs. The next step in water quantity management is to validate that the recommended flow regimes are sufficient to maintain a sound ecological environment, although validation methodologies are rarely incorporated into water quantity management plans. Purposes of this study were to develop methodologies for validating flow recommendation and standards that are transferable and replicable and to quantify the value of flow tiers relating to organic drift. Objectives of this study were to assess drift biomass of coarse particulate matter (CPM) and drift rates of macroinvertebrates and larval fishes related to flow tiers (i.e., subsistence, base, two per season high flow pulse, one per season high flow pulse, and one per year high flow pulse) across four sites and two rivers (i.e., San Antonio and Guadalupe rivers). *A priori* predictions were that CPM, macroinvertebrates, and larval fishes were in greater biomass or densities at higher levels of flow tiers, though at some point flows would be sufficiently high and induce a washout effect. Findings were generally inconsistent with the predications. Biomass of CPM did not differ among flows ranging from subsisted to two per season high flow pulse events, and densities of macroinvertebrate did not differ

among flows ranging between base and one per season events. However, densities of fluvial specialist *Neoperla* (Order Plecoptera) were positively correlated with flow as a continuous variable rather than a categorical variable. Densities of larval fishes were greatest at subsistence, base, and two per season high flow pulses and decreased at 1 per season flow pulses. It is uncertain if a washout effect occurred or if survival of larval fishes was decreased at a higher flow pulses. With a validation methodology in place, replications from other sites and river basins can be added to the existing model to improve statistical power and inference, along with other flow-dependent variables, in order to fully assess the value of multi-tier flow regimes.

CHAPTER I

Validating Environmental Flow Recommendations: Drifting Coarse Particulate

Matter, Macroinvertebrates and Larval Fishes

Introduction

Since the 1940's, water quality (i.e., chemical, biological, and physical characteristics) has been a primary concern among state and federal resource managers and is monitored and regulated to protect agricultural, industrial, and municipal usages of the water, recreational activities, and instream biota. Since the 1970s, water quantity (i.e., amount of flowing water) is a concern but monitoring and regulation has lagged behind those in place for water quality. Tennant (1976) was among the first to recommend a generic water quantity recommendation with the Tennant or Montana Method, and several recommendation strategies have been developed since, such as the Savannah Process (Richter et al. 2006), and the Ecological Limits of Hydrologic Alteration (ELOHA) (Poff et al. 2009). Poff et al. (1997) provided the underlying theory of water quantity management, stating that aquatic communities are dependent upon the dynamic characters of a flow regime. Subsequently, water quantity strategies typically involve a combination of flow events. In Texas, water quantity monitoring and management is mandated under Senate Bill (SB) 2 and SB 3 for the maintenance of sound ecological environments in streams, rivers, bays, and estuaries (TCEQ 2008).

Texas SB 3 stakeholder process quantified natural characters of a river by identifying flow tiers that will ensure a sound ecological environment (GSA BBEST 2011). Flow tiers were classified into subsistence, base (dry, normal, or wet, depending on hydrological conditions), and high flow pulses (i.e., 2 per season events, 1 per season,

1 per year, 1 per 2 years, and 1 per 5 years). Magnitude and duration of the flow tiers are determined by estimating seasonal central tendencies of these events from historical hydrographs, provided by US Geological Service. When applied, the resulting environmental flow recommendation regulates when and how much flow can be harvested from surface waters within streams and rivers. Once an environmental flow recommendation is set, the SB 3 process requires validation of the recommendation. Specifically, does the recommendation (i.e., magnitude of subsistence, base, and high flow pulses) maintain a sound ecological environment? If not, what changes are necessary to maintain a sound ecological environment? However, few environmental flow recommendations have developed validation methodologies.

Legislatively, sound ecological environment is a loosely defined concept, which generally conforms to the following expectations: sustains the full complement of native species in perpetuity, sustains key habitat features required by these species, retains key features of the natural flow regime required by these species to complete their life cycles, and sustains key ecosystem processes and services (SAC 2011). A component of key ecosystem processes and services is organic drift. Organic drift is defined as organic matter of both allochthonous and autochthonous origins that is passively or actively transported downstream and contributes to the nutrient web of the riverine community (Bison and Bilby 1998). Organic drift includes coarse particulate organic matter (CPOM), macroinvertebrates of both aquatic and terrestrial origin, and drifting larval fish. Quality and quantity of organic drift is thought to be positively related to flow magnitude. As stream discharge increases, the amount of CPOM from both aquatic sources and terrestrial input increases (Poff et al. 1997; Bison and Bilby 1998; Junk and

Wantzen 2004). Likewise, densities of macroinvertebrates and larval fishes increase with stream discharge because of direct effects (i.e. increased success of reproduction) and indirect effects (i.e. nutrient pulses, increased spawning habitat) (Durham and Wilde 2008; Durham and Wilde 2009; Dettmers et al. 2001). However, high flow events, sometimes called catastrophic flood events (Brittain and Eikeland 1988), can generate sufficient flows to wash out any evidence of increase densities of CPM, macroinvertebrates, and larval fishes, thus potentially confounding the ability to detect value of flow pulse on densities of organic drift.

The direct relationship between flow and organic drift is well established and used in part to establish the Natural Flow Paradigm. However, converting the theory to application (SB 3 environmental flow recommendations) may or may not sufficiently maintain the necessary components of the natural flow character with respect to organic drift. Hence, the need for validation of the environmental flow recommendations. Goals of this study were to develop a validation process and then to validate environmental flow recommendations by observing their effects on organic drift within the lower Guadalupe River and lower San Antonio River. The San Antonio River and Guadalupe River were chosen for this study because both rivers have established SB 3 environmental flow recommendations, are representatives of lowland drainages of the western gulf slope drainages of Texas. Study objectives were 1) to characterize temporal occurrence and biomass of drifting CPM and occurrence and densities of macroinvertebrates and larval fishes for 1 year among four sites on the San Antonio and Guadalupe Rivers and 2) to assess the relationship between biomass or densities and flow tiers as a categorical variable and flow as a continuous variable among periods of CPM, macroinvertebrate,

and larval fishes occurrences. I predict that biomass of CPM and densities of macroinvertebrates and larval fishes will increase with each sequential flow tier and with flow up to a point where flows produce a washout effect.

Methods

Study Area

The Guadalupe River originates at the confluence of the North and South Fork Guadalupe Rivers in Kerr County, Texas (Perkin & Bonner 2011) and flows 700 km southeast through the Balcones Escarpment and onto the gulf coast plains (GSA BBEST 2011). The river is impounded by numerous low-head dams and one large dam (Canyon Dam) throughout its course. The San Antonio River originates at the confluence of Olmos Creek and San Antonio Springs. San Antonio Spring is now ephemeral, attributed to extensive groundwater pumping within the region. Source of base flow in the upper reach is wastewater return. The San Antonio River flows through the highly urbanized City of San Antonio and continues about 350 km downstream before emptying into the Guadalupe River and into the San Antonio Bay (Gulf of Mexico).

Sampling sites

Four sites were sampled, two on the lower Guadalupe River in Seguin, Texas at County Road 1117 (29°32'12"N,97°52'50"W) and Cuero, Texas at Highway 72 (29°08'60"N,97°18'57"W), and two on the lower San Antonio River in Falls City, Texas at Farm to Market 81 (28°57'25"N,97°58'48"W) and Goliad, Texas at Highway 183 (28°39'43"N,97°23'28"W) (Figure 1). All sites are located on the lower portions of their

respective systems in the Gulf Coastal Plains region of south central Texas (GSA BBEST 2011). Sites were selected because of their established environmental flow recommendations and because they each had a USGS gauging station; however, the USGS gauging station at Gonzales, Texas was used as a surrogate for the Seguin site because of its longer period of flow record (Gonzales: 1940 – present, Seguin: 2006 – present).

Field methods

Sampling protocol followed that of Williams (2011). Each site was sampled every other week from March 2012 to September 2012 during the peak reproductive season of resident stream fishes and once per month from October 2012 to March 2013 outside of peak reproductive season (Hendrickson and Cohen 2012). Three 500- μ m ichthyoplankton drift nets (Aquatic Sampling Company A211-500) with a mouth opening of 0.1394 m² were set just below the water's surface and were placed perpendicular to the shoreline, spaced among slow, moderate, and swift current velocities to adequately capture the spatial diversity in organic drift. If current velocities were relatively uniform across the channel, nets were set independent of velocity, and classified as near shore to mid channel. Nets were secured in place with metal fence posts. Each net was set two hours prior to sunset and were fished for three sets of two hours for a total sample time of six hours. The period before sunset and the hours immediately following sunset are peak hours for macroinvertebrate (Pendergrass 2006) and larval fish drift (Williams 2011). Water quality parameters including current velocity (Marsh-McBirney Flowmate Model 2000 flow meter & JDC Flowwatch), temperature, conductivity, dissolved oxygen (YSI

Model 85), depth, and distance from shore were measured at each net before and after each set. Nets were emptied and rinsed of their contents after each set or if the current velocity at the mouth of the net indicated clogging and placed in individual Whirl-Pak (2,041 ml) bags. Each sample was then fixed with 4% formalin. Nets were rinsed with river water for the next set.

Laboratory Methods

After at least seven days, the 4% formalin was replaced with 70% ethanol. Contents from each bag were emptied and rinsed on a 250 μm sieve (Fisher Scientific USA Standard Test Sieve). Whole samples were placed into trays and sorted to recover all larval fishes. This process was conducted by two separate sorters to ensure quality control. Once all larval fish were removed, samples were picked of all macroinvertebrates. CPM was then dried in an oven to a constant dry weight. Biomass of CPM was recorded to the nearest gram. Macroinvertebrates were identified to the lowest practical level of resolution and enumerated for only one set (3) and one net (1) because of the large number of macroinvertebrates captured in the nets. Life stage of larval fishes was classified as protolarvae, mesolarvae, metalarvae based on morphology (Cooper 1980; Hulbert et al. 2007), identified to the lowest practical level of resolution with assistance from larval fish keys (Auer 1982; Fuiman et al. 1983), and enumerated. Protolarvae were designated as those individuals with yolk (\approx 0-11 days post hatch), mesolarvae are individuals with a developed and active gut tract but lacking pectoral fins or fin buds (\approx 12-36 days post hatch), metalarvae are individuals with pectoral fins or fin

buds but less than the minimum length for a juvenile form of that particular species (\approx 37-60 days post hatch).

Statistical Methods

Metalarvae were not included in further analysis because it was determined that at their age and size class, metalarvae developed sufficient swimming ability to be less affected by the drift than the younger age groups. Drift rates of CPM, macroinvertebrates, and larval fish, by life stage, were calculated as catch per unit effort based on the volume of water filtered in each drift net (CPM: mg/l; macroinvertebrates and larval fishes: # individuals/100 m³). Prior to any analysis, drift densities were log transformed [$\log(x + 1)$] to help meet the assumption of normality and homoscedasticity. Maximum flow was assigned based off of the appropriate back dating and was log transformed [$\log(N)$]. Back dating was assigned to CPM and macroinvertebrates based on approximate residency time in the drift after a flow event. CPM was assigned a 2 day lag period (Webster 1987, 1999) and macroinvertebrates were assigned a 28 day lag period (McLay 1968). Larval fishes were assigned lag periods based on the approximate time spent in each age group: 11 day lag for protolarvae and 36 day lag for mesolarvae. Flow tiers were assigned based on this maximum flow observed within a lag period. Differences in drift rates among discharge (continuous variable) were assessed with simple linear regression. Differences in drift rates among flow tiers (i.e. subsistence, base, high flow pulses; categorical variable) were assessed with analysis of variance ($\alpha = 0.05$). Only flow tiers with ≥ 3 replicates (i.e., events) were included in statistical analysis.

Results

Temporal occurrences of organic drift

Coarse particular matter consisted primarily of detached macrophytes, riparian leaf material, and woody debris and was collected each month during the one year period. Mean \pm SE CPM densities ranged from $0.01 \pm < 0.01$ in September 2012 to 0.16 ± 0.12 mg/L in October 2012 (Figure 2). Densities of CPM were not different ($P = 0.55$) among collections dates.

A total of 45,096 macroinvertebrates taken monthly represented at least 52 families and 84 genera. Among fluvial specialists, order Ephemeroptera (combined genera *Homoeoneuria*, *Stenonema*, and *Traverella*) was most abundant at 1.9% of the total catch, followed by *Hydropsyche* at 1.8% and *Neoperla* at 0.2%. Mean \pm SE macroinvertebrate densities ranged between 28 ± 4.5 macroinvertebrates/100 m³ in November 2012 to 595 ± 220 macroinvertebrates/100 m³ in August 2012. Densities of macroinvertebrates were not different ($P = 0.51$) among collection dates. Among fluvial specialists, order Ephemeroptera differed ($P < 0.02$) among collection dates with greatest densities observed in May 2012. Densities of *Neoperla* (Order Plecoptera) differed ($P < 0.01$) among collections with greatest densities also observed in May 2012. Densities of genus *Hydropsyche* (Order Trichoptera) did not differ ($P = 0.55$) among collection dates.

A total of 1,958 protolarvae represented at least seven families (Table 1). Cyprinidae was most abundant, ranging in relative densities from 53% at Seguin to 95% at Goliad, followed by Centrarchidae (0.9% at Goliad to 29% at Seguin), and Percidae (1.8% at Seguin and Goliad to 9.5% at Cuero). Unidentifiable protolarvae ranged from 0.5% at Cuero to 14% at Seguin. Protolarvae were taken each month of the study period,

with the exception of October and November 2012. Among remaining collection dates, mean \pm SE protolarval densities ranged from 0.02 ± 0.016 larvae/100 m³ in May 2012 to 3.9 ± 1.06 larvae/100 m³ in August 2012. Densities differed ($P < 0.01$) among collection dates with greatest densities in June and August 2012. Among families, protolarval cyprinids were taken from March 2012 through March 2013, with greatest densities ($P = 0.03$) observed in June and August 2012. Protolarval centrarchids were taken between April 20, 2012 and September 2012 with densities not different ($P = 0.62$) among collection dates. Protolarval percids were taken in July and August 2012 and again from November 2012 through March 2013 with greatest densities ($P < 0.01$) observed in February 2013.

A total of 7,378 mesolarvae represented at least 14 species and 10 families (Table 2). Centrarchidae was the most abundant in density ranging from 1.4% at Seguin to 24% at Goliad, followed by Cyprinidae (6.2% at Seguin to 23% at Cuero), and Clupeidae (0% at Seguin to 14% at Cuero). Unidentifiable fish densities ranged from 4.3% at Seguin to 16% at Falls City. Mesolarvae were taken each month of the study period, with the exception of November 2012. Among remaining sampling dates, mean \pm SE monthly mesolarvae densities ranged between 0.05 ± 0.05 larvae/100 m³ in October 2012 to 13 ± 5.8 larvae/100 m³ in September 2012. Densities differed ($P = 0.03$) among months with the greatest number observed from June through September 2012, excluding the July II collection. Among families, mesolarval centrarchids were taken from April to October 2012 and February 2013 and densities were not different ($P = 0.59$) among collection periods. Mesolarval cyprinids were taken from April to October 2012 and February and March of 2013 with densities not different ($P = 0.74$) among collection periods.

Mesolarval clupeids were taken from May to August 2012 with densities not different ($P = 0.57$) among collection periods.

Densities among flow tiers

Densities of $CPM_{2\text{-day}}$ were estimated from 63 events among three flow tiers, ranging from subsistence to two per season high flow pulse and flows ranging from 27 to $512 \text{ m}^3/\text{s}$ (Table 3). Mean \pm SE densities ranged from $0.03 \pm 0.01 \text{ mg/l}$ at two per season high flow pulse to $0.04 \pm 0.01 \text{ mg/l}$ at both subsistence and base flow with no differences detected between CPM densities between flow tiers ($P = 0.92$, Figure 3) or among discharge ($P = 0.17$, Table 4).

Densities of macroinvertebrates_{28-d} were estimated from 52 events among five flow tiers, ranging from subsistence to one per year high flow pulse and flows ranging from 50 to $3,566 \text{ m}^3/\text{s}$. Mean \pm SE densities ranged from 72 ± 16 macroinvertebrates/100 m^3 at two per season high flow pulse to 271 ± 68 macroinvertebrates/100 m^3 at base flow. Macroinvertebrate densities did not differ among flow tiers ($P = 0.14$) or discharge ($P = 0.06$). Likewise, densities of Ephemeroptera did not differ among flow tier ($P = 0.47$) or discharge ($P = 0.86$), and densities of *Hydropsyche* did not differ among flow tiers ($P = 0.81$) or discharge ($P = 0.71$). However, densities of *Neoperla* differed by discharge ($P = 0.03$), but not by flow tier ($P = 0.74$).

Densities of protolarvae_{11-d} were estimated from 41 events among five flow tiers, ranging from subsistence to one per year high flow pulse and flows ranging from 30 to $3,566 \text{ m}^3/\text{s}$. Mean \pm SE densities of total protolarvae ranged between 0.2 ± 0.22 larvae/100 m^3 at one per season high flow pulse to 2.8 ± 0.6 larvae/100 m^3 at base flow.

Total protolarvae density differed among flow tiers with ($P = 0.01$) and by discharge ($P = 0.04$). Likewise, Cyprinidae densities differed among flow tier ($P = 0.03$, Figure 5) but did not differ by discharge ($P = 0.07$). Densities did not differ among flow tiers for protolarvae centrarchids ($P = 0.29$) or discharge for protolarvae centrarchids ($P = 0.36$) or percids ($P = 0.85$).

Densities of mesolarvae_{36-d} were estimated from 41 events among four flow tiers, ranging from baseflow to one per year high flow event and flows ranging from 67 to 3,566 m³/s. Mean \pm SE densities of total mesolarvae by flow tier, ranged from 3.5 ± 0.8 larvae/100 m³ at one per season high flow pulse to 16 ± 3.04 larvae/100 m³ at two per season high flow pulse. Differences were detected between flow tiers ($P < 0.01$) but not for discharge ($P = 0.36$) for total mesolarvae with the greatest densities at the two per season high flow event. Cyprinidae densities differed between flow tiers ($P = 0.01$, Figure 6) but did not differ by discharge ($P = 0.71$). Likewise, densities of Centrarchidae differed by flow tier ($P < 0.01$) but not by discharge ($P = 0.16$). Clupeidae did not differ among flow tiers ($P = 0.13$) or by discharge ($P = 0.72$).

Discussion

The predicted relationship between organic drift and flow was partially supported in this study. Densities did not differ for CPM ranging from subsistence flows to two per season pulses or for macroinvertebrates ranging from base flow to one per season pulses. Densities of larval fishes decreased at one per season pulses when compared to base flow densities (protolarvae, mesolarvae) and two per season densities (mesolarvae). As predicted, larval densities decreased at higher flow pulses, but at flow pulses lower than

expected. Also unexpectedly, densities did not differ among subsistence, base flows, and two per season pulses for CPM and protolarvae or among base flows and two per season pulses for macroinvertebrates and mesolarvae.

Estimates of mean monthly organic and inorganic CPM in this study (0.01 to 0.16 mg/l) were within the range of those reported in previous studies. Wallace et al. (1982) reported a range of POM means between 0.53-3.68 mg AFDM/l among forested Appalachian Mountain streams. Coarse (>5 mm) and large (0.864 – 5 mm) POM comprised 11% of the ash-free dry mass, and the reported ash-free dry mass was 36% of the total organic and inorganic CPM. Using these proportions to convert into similar units reported in this study, Wallace et al. (1982) organic and inorganic CPM for large and coarse material ranged from 0.17 to 1.17 mg/l. In graphical form, Gurtz et al. (1980) reported organic and inorganic, coarse (>5.0 mm) and large (0.864 – 5.0 mm) material ranged from 0 to <1.0 mg/l on one forested stream and from <1.0 to <21.0 mg/l on another forested stream in North Carolina. As such, estimates of CPM in this study (0.01 to 0.16 mg/l) were slightly less than those reported by Wallace et al. (1982; 0.17 to 1.17 mg/l) but within range of those reported by Gurtz et al. (1980; 0 to <21.0 mg/l). Heterogeneity in CPM through time and within and among basins is related to amount of litter fall within the watershed and the occurrence and number of macroinvertebrate shredders (Bison and Bilby 1998). As riverscapes transition from low order to high order streams, CPM from primarily allochthonous subsidies are converted to FPOM by mechanical and biological processes (Vannote et al. 1980; Naiman and Sedell 1980). High flow pulses exceeding effective discharge thresholds (Heitmuller and Raphael 2012) resuspend instream CPM along with new inputs from runoff, typically immediately

before peak discharge (Bilby and Likens 1979; Hill and Gardner 1987; Webster et al. 1987). However, a relationship between CPM and flow tiers was not detected in this study. I attribute the lack of detection to two plausible mechanisms: effective discharge was not reached during the period of observation and the lack of sampling sufficient number of high flow events within a two-day lag period. In a companion study, Ruppel (2014, unpublished) found greater abundance of detritus in the foreguts of larval fishes found in backwater habitats within 28-d following a high flow pulses, suggesting that effective discharge was reached during the period of observation. Only four replicates were taken in this study following a two per season high flow pulse; therefore, I conclude that insufficient number of replicates among the higher flow tiers likely lowered statistical power to detect differences.

Maximum mean monthly drift rates of macroinvertebrates ranged between 18 to 595 macroinvertebrates /100 m³ (range of mean annual discharge by site: 25 to 80 m³/s). Macroinvertebrate drift rates range between 200 and 500 macroinvertebrates/100 m³ among southeastern coastal streams (mean annual discharge: 28 to 62 m³/s; Benke et al. 1984) to 1,300 and 2,200 macroinvertebrates/100 m³ among mountain streams in Colorado (mean annual discharge: 21 m³/s; Allan 1982). In the lowland River Wye, Wales (annual mean discharge: 1.3 m³/s), mean macroinvertebrate density was 78 macroinvertebrates/100m³ during base flow and 106 macroinvertebrates/100m³ during simulated high flow (Brooker and Hemsworth 1978). Macroinvertebrate drift is related to active (e.g., seeking new resources) and passive (e.g., flow pulses) mechanisms (Brittain and Eikeland 1988; Gibbins et al. 2007a). With passive drift associated with flow pulses, shear stress for macroinvertebrates is the same as shear stress for sediment

entrainment and transport (Gibbins et al. 2007b; Gibbins et al. 2010) with shear stress and flow level dependent upon substrate size. Larger substrates require large flow pulses for entrainment and transportation (Webster et al. 1987). In addition, stream order influences macroinvertebrate communities (Vannote et al. 1980) and therefore patterns in macroinvertebrate drift. As such, substrate types and stream order differed among sampling sites in this study, which could explain the large amount of variation observed among sites. Nevertheless, drift rates of macroinvertebrates, though variable, were similar to those reported in lowland streams elsewhere (Southeastern USA, Wales) but much less than those reported for montane streams.

Maximum mean monthly drift rates were 6.3 fish/100 m³ for protolarvae and 13 fish/100 m³ for mesolarvae in this study with mean annual discharge by site ranging from 25 to 80 m³/s and generally represent similar numbers of larval drift compared to other published studies. In the lower Brazos River (mean annual discharge: 241 m³/s), maximum annual drift rates were 5.7 for protolarvae and 2.1 for mesolarvae in the lower Brazos River between April and September (Williams 2011). In other published studies, drift rates were reported for larval fishes independent of life stages (i.e., not split between protolarvae and mesolarvae, include metalarvae and at times juveniles). Maximum mean monthly drift rates range between 10 larvae/100 m³ in the Canadian River with a mean annual discharge of 40 m³/s (Durham and Wilde 2008) to 14 larvae/100 m³ in the Niobrara River with a mean annual discharge of 49 m³/s (Schainost 2008; Wanner et al. 2011) and Danube River with a mean annual discharge of 65 m³/s (Reichard 2001; Wanner et al. 2011). Only Williams (2011) reported larval fish drift estimates related to flow regime, though not directly comparable to this study since Williams (2011) reported

drift densities related to low vs. high flow year rather than individual flow events within a year. In the lower Brazos River, Williams (2011) reported mean annual drift densities of protolarvae did not differ whereas drift densities of mesolarvae increased from 1.2 to 2.1 larvae/100 m³ between a low water year (mean discharge during the spawning season: 192 ± 234 m³/s) and a high water year (378 ± 454 m³/s).

Linkages between flow pulses and spawning success (i.e., increase numbers of larvae, provision of suitable habitats; nutrients for larval fishes) are well established for riverine fishes (Geddes and Puckridge 1989; Lloyd et al. 1989; Sparks et al. 1990; Junk et al. 1989; Schiller and Harris 2001; Humphries et al. 2013), especially among short-lived species with prolonged spawning season strategy (Moore 1944; Bonner 2000; Ross 2001; Hendrickson and Cohen 2012); however, most studies do not directly test the flow-fish spawning relationship but instead use surrogates, such as gonadosomatic index (Bonner 2000; Williams 2011), otolith examination (Durham and Wilde 2009), gut content analysis (Ruppel 2014, unpublished), and statistical modeling (Durham and Wilde 2006). In contrast, Humphries (1999) observed and then developed the Low Flow Recruitment Hypothesis, which predicts greater spawning success in low flow condition among some fishes based on higher densities of small food items that low flows provide. Low Flow Recruitment Hypothesis is partially supported by Durham and Wilde (2006), who modeled numbers of larval fishes versus flow and found that the presence of flow is more important to reproductive success of fish than the magnitude of flow. Results of this study also partially support the Low Flow Recruitment Hypothesis. Drift rates of larval fishes at base flows were not different from those at a

two per season event and large flow pulse events were negatively correlated with larval drift densities.

Results of this study suggest that organic drift (i.e., CPM, macroinvertebrates, and larval fishes) is largely unaffected by or decreased under higher flow pulses (> 2 per season flow pulse events). This finding is constrained by low replication (four sites within two rivers) and unavailability of higher flow pulses during this study. Low replication can be remedied by adding additional studies from different sites and rivers as replicates given that numerous sites and river basins within the western gulf slope drainages have similar instream flow recommendations. Testing of higher flow pulses can be remedied by future selective sampling of higher flow pulses. In time, ecological value of prescribed flow pulses relating to organic drift might be validated.

Alternatively, additional replicates and sampling of high flow events might only further support findings of this study: greater organic drift is not detectable under higher flow events. If so, then the conceptual framework of organic drift and dynamic flows should be reassessed. Specifically, flow pulses are not necessary to transport nutrients and maintain riverine macroinvertebrate and fish communities. For larval fish success, perhaps the Low Flow Recruitment Hypothesis is the appropriate view and can be extended to CPM and to macroinvertebrates. However, oft cited “washout” effect might constrain our ability to detect values of high flow pulses on organic drift.

Washout effect is the export of riverine organics out of a study reach because of a high flow event, often referred to as catastrophic drift when referring to macroinvertebrates (Brittain and Eikeland 1988). Washout effect is described for CPM (Bilby and Likens 1979) and macroinvertebrates (Brittain and Eikeland 1988) and can

confound detectability of increased nutrients, macroinvertebrates, and larval fishes related to flow pulses. At a point along a river, temporary reduction in organic drift can follow a flow pulse. Indeed, organic drift in this study was initially predicted to decline at higher flow pulses, but organic drift declined at a moderate flow pulses rather than a higher flow pulses. However, reduction in organic drift at a point in a river does not necessarily equate to reduction in organic drift within the entire system. Instead, washout effect should be viewed more as a conveyer belt system. As organics enter the drift because of pulse events, regardless of magnitude, the conveyer belt increases speed of organic drift passing through a point in time, therefore decreasing chances of detectability. Nevertheless, whether or not greater amounts of organic drift are added to the system, either as CPM imports, macroinvertebrates entrained into the drift, or larval fishes because of spawning, the faster drift will be difficult to detect at a point in time. Instead, aggregation of multiple points along a river course after a flood pulse would increase detectable of organic drift, if linked to flow pulses.

In this study, the ecological value of individual flow pulses of various magnitudes to organic drift was not detected. However, the ecological values of flow pulses include more than organic drift. Each flow tier, even subsistence flows and overbank yearly flows provides a unique set of values to the riverine ecosystem. For example, subsistence flows can enable the recruitment of certain riparian floodplain plants while high flows can flush out a system or provide floodplain nursery habitat for young fish and other organisms (Risley et al. 2010). Using the validation process developed herein for organic drift, other theoretical flow-dependent variables can be tested, with flow as a categorical variable (to validate a flow recommendation) or as a continuous variable (to

adjust a flow recommendation). Success of flow recommendation validation depends on our ability to adequately replicate a suite of standardized flow recommendations. Currently, numerous western gulf slope streams in Texas have a standardized flow recommendation and can provide necessary replications. In time, perhaps lowland communities will respond differently from upland communities with respect to flow tiers, and these co-variates can be identified and adjusted as necessary. Likewise, in time, adequate lag times of assessments can be better understood and as well as our ability to separate wash out effects from community responses to high flow pulses. In the end, a systematic methodology with replication will be in place to validate the benefits of high flow pulses, or more generically the Natural Flow Paradigm. Then, river pulses, which are the targets for future water harvest, can be passed at appropriate levels to reach the goals of water quantity management in providing a flow regime capable of maintaining a sound ecological environment.

Table 1. Percent abundance of Protolarvae by family across sites, total number of protolarvae (N) per site.

| Family | Guadalupe | | San Antonio | |
|-----------------------|-----------|-------|-------------|--------|
| | Seguin | Cuero | Falls City | Goliad |
| <i>Cyprinidae</i> | 53 | 85 | 64 | 95 |
| <i>Catostomidae</i> | 1.4 | 0.3 | 2.0 | 0 |
| <i>Centrarchidae</i> | 29 | 3.5 | 24 | 0.9 |
| <i>Clupeidae</i> | 0.6 | 0.5 | 0 | 0 |
| <i>Ictaluridae</i> | 0.2 | 0.0 | 0 | 0 |
| <i>Percidae</i> | 1.8 | 9.5 | 6.7 | 1.8 |
| <i>Atherinopsidae</i> | 0 | 0.3 | 0.3 | 0 |
| Unidentifiable | 14 | 0.5 | 3.6 | 2.0 |
| Total | 325 | 297 | 537 | 799 |

Table 2. Percent abundance of mesolarvae by family across sites, total number of mesolarvae (N) per site.

| Group | Guadalupe | | San Antonio | |
|------------------------------|-----------|-------|-------------|--------|
| | Seguin | Cuero | Falls City | Goliad |
| <i>Lepisosteus</i> | 0 | 0.8 | 0 | 0 |
| <i>Clupeidae</i> | 0 | 14 | 1.8 | <0.1 |
| <i>Cyprinidae</i> | 6.2 | 23 | 17 | 22 |
| <i>C. lutrensis</i> | 0 | 3.6 | 5.5 | 2.0 |
| <i>P. vigilax</i> | 1.3 | 15 | 2.8 | 4.3 |
| <i>Notropis</i> | 1.1 | 2.6 | 2.5 | 1.7 |
| <i>O. emiliae</i> | 0 | 0.1 | 0 | 0 |
| <i>Cyprinus</i> | 0.5 | 0 | 0 | 0 |
| <i>Unknown species</i> | 3.3 | 0.9 | 6.1 | 14 |
| <i>Catostomidae</i> | 48 | 0.1 | 0 | <0.1 |
| <i>M. congestum</i> | 48 | 0.1 | 0 | 0 |
| <i>Unknown species</i> | 0 | 0 | 0 | <0.1 |
| <i>Centrarchidae</i> | 1.4 | 7.3 | 23 | 24 |
| <i>L. macrochirus</i> | 0 | 0.0 | <0.1 | 0 |
| <i>Lepomis</i> | 0 | 0.4 | 0 | 0 |
| <i>Micropterus</i> | 0 | 0.2 | 0 | 0 |
| <i>Unknown species</i> | 1.4 | 6.6 | 23 | 24 |
| <i>Percidae</i> | 33 | 0.6 | 0 | <0.1 |
| <i>P. carbonaria</i> | 12 | 0.1 | 0 | <0.1 |
| <i>Unknown species</i> | 20 | 0.5 | 0 | <0.1 |
| <i>Poecilidae</i> | 0 | 0.7 | 0.5 | 1.9 |
| <i>G. affinis</i> | 0 | 0.6 | 0.5 | 1.9 |
| <i>P. formosa</i> | 0 | <0.1 | 0 | <0.1 |
| <i>Unknown species</i> | 0 | 0.1 | 0 | 0 |
| <i>I. punctatus</i> | 0 | 0 | <0.1 | 0 |
| <i>H. plecostomus</i> | 0 | 0 | <0.1 | 0 |
| <i>Atherinopsidae</i> | 0 | 0 | <0.1 | 0 |
| Unidentifiable | 4.3 | 9.1 | 16 | 6.0 |
| Total Fish | 2789 | 2418 | 1170 | 1001 |

Table 3. Mean \pm SE [$\log(X + 1)$] of each drift group by flow tier, number of replications (N), F-value, degrees of freedom (df), and P-value all based on analysis of variance, (+) CPUE expressed as mg/l, (*) CPUE expressed as # individuals/100m³, flow tiers with N < 3 were not included in analysis.

| Group | Subsistence | | | Base | | | Two per season | | | One per season | | | One per year | | | F | df | P |
|---------------------|-------------------|------|---|-------------------|------|----|-------------------|------|---|------------------|------|----|--------------|------|---|------|------|-------|
| | Mean | SE | N | Mean | SE | N | Mean | SE | N | Mean | SE | N | Mean | SE | N | | | |
| CPM [†] | 0.04 | 0.01 | 8 | 0.04 | 0.01 | 51 | 0.03 | 0.01 | 4 | - | - | - | - | - | - | 0.1 | 2,60 | 0.92 |
| Macroinvertebrates* | 73 | - | 1 | 271 | 68 | 27 | 72 | 16 | 8 | 125 | 51 | 14 | 609 | 506 | 2 | 2.1 | 2,46 | 0.14 |
| Ephemeroptera | 0.6 | 0 | 1 | 0.7 | 0.2 | 27 | 0.6 | 0.2 | 8 | 0.4 | 0.2 | 14 | 0.9 | <0.1 | 2 | 0.8 | 2,46 | 0.47 |
| Hydropsyche | 0.0 | - | 1 | 5.4 | 2.4 | 27 | 0.7 | 0.3 | 8 | 2.5 | 1.1 | 14 | 2.9 | 1.2 | 2 | 0.8 | 2,46 | 0.45 |
| Neoperla | 0.3 | - | 1 | 0.4 | 0.2 | 27 | 0.4 | 0.3 | 8 | 1.0 | 0.6 | 14 | 0.1 | 0.1 | 2 | 0.3 | 2,46 | 0.74 |
| Protolarvae* | 0.5 ^{ab} | 0.3 | 3 | 2.8 ^a | 0.6 | 27 | 1.0 ^{ab} | 0.8 | 3 | 0.2 ^b | 0.2 | 7 | 0.2 | - | 1 | 4.3 | 3,36 | 0.01 |
| Cyprinidae | 0.4 ^{ab} | 0.3 | 3 | 2.2 ^a | 0.6 | 27 | 0.8 ^{ab} | 0.7 | 3 | 0.2 ^b | 0.2 | 7 | 0.2 | - | 1 | 3.3 | 3,36 | 0.03 |
| Centrarchidae | <0.1 | <0.1 | 3 | 0.5 | 0.2 | 23 | 0.1 | 0.1 | 3 | <0.1 | <0.1 | 7 | 0 | - | 1 | 1.3 | 3,32 | 0.29 |
| Percidae | 0 | 0 | 2 | 0.2 | <0.1 | 12 | - | - | - | - | - | - | - | - | - | - | - | - |
| Mesolarvae* | - | - | - | 7.3 ^a | 1.4 | 8 | 16 ^a | 3.0 | 9 | 3.5 ^b | 0.8 | 22 | 8.6 | 8.4 | 2 | 11.8 | 2,36 | <0.01 |
| Cyprinidae | - | - | - | 1.8 ^{ab} | 0.4 | 8 | 5.7 ^a | 2.0 | 9 | 1.6 ^b | 0.4 | 22 | 1.6 | 0.9 | 2 | 5.4 | 2,36 | 0.01 |
| Centrarchidae | - | - | - | 2.3 ^{ab} | 0.8 | 8 | 7.3 ^a | 2.4 | 9 | 0.9 ^b | 0.3 | 20 | 7.1 | 7.1 | 2 | 9.3 | 2,34 | <0.01 |
| Clupeidae | - | - | - | 2.1 | 1.8 | 5 | 0.6 | 0.2 | 5 | <0.1 | <0.1 | 9 | - | - | - | 2.3 | 2,16 | 0.13 |

Table 4. CPUE vs. discharge (continuous variable), F value, degrees of freedom (df), *P*-value, and R^2 value all based on simple linear regression analysis.

| Group | F | df | <i>P</i> | R^2 |
|--------------------|------|------|----------|-------|
| CPM | 1.92 | 1,61 | 0.17 | 0.03 |
| Macroinvertebrates | 3.7 | 1,50 | 0.06 | 0.07 |
| Ephemeroptera | 0.03 | 1,50 | 0.86 | <0.01 |
| Hydropsyche | 0.14 | 1,50 | 0.71 | <0.01 |
| Neoperla | 5.24 | 1,50 | 0.03 | 0.1 |
| Protolarvae | 4.7 | 1,39 | 0.04 | 0.11 |
| Cyprinidae | 3.54 | 1,39 | 0.07 | 0.08 |
| Centrarchidae | 0.85 | 1,35 | 0.36 | 0.02 |
| Percidae | 0.04 | 1,12 | 0.85 | <0.01 |
| Mesolarvae | 0.85 | 1,39 | 0.36 | 0.02 |
| Cyprinidae | 0.14 | 1,39 | 0.71 | <0.01 |
| Centrarchidae | 2.06 | 1,37 | 0.16 | 0.05 |
| Clupeidae | 0.13 | 1,17 | 0.72 | <0.01 |



Figure 1. Sampling locations on lower reaches of San Antonio River and Guadalupe River, Texas.

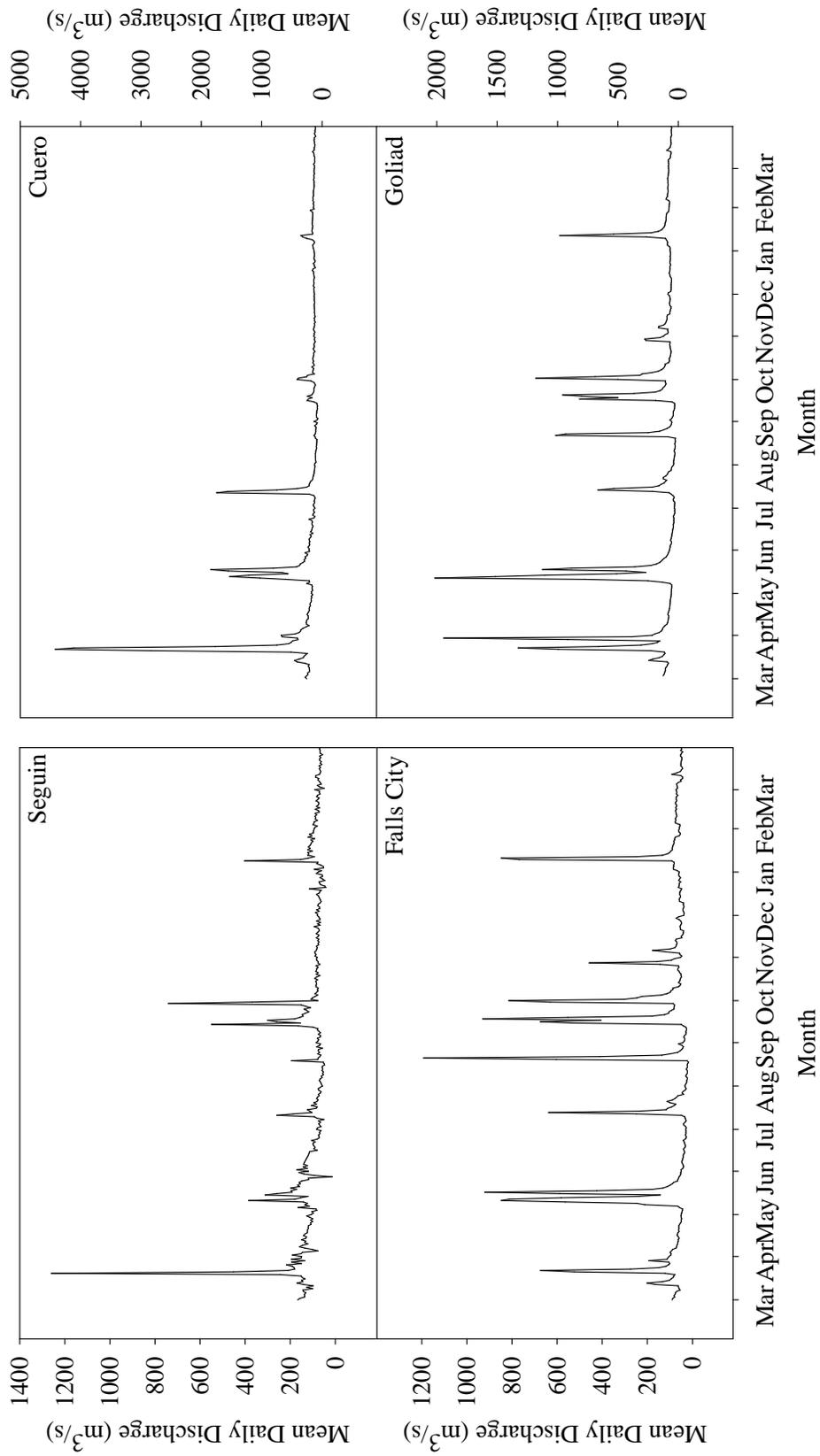


Figure 2. Mean daily discharge rates for each site from March 2012 to March 2013, data from USGS gauging stations at each site.

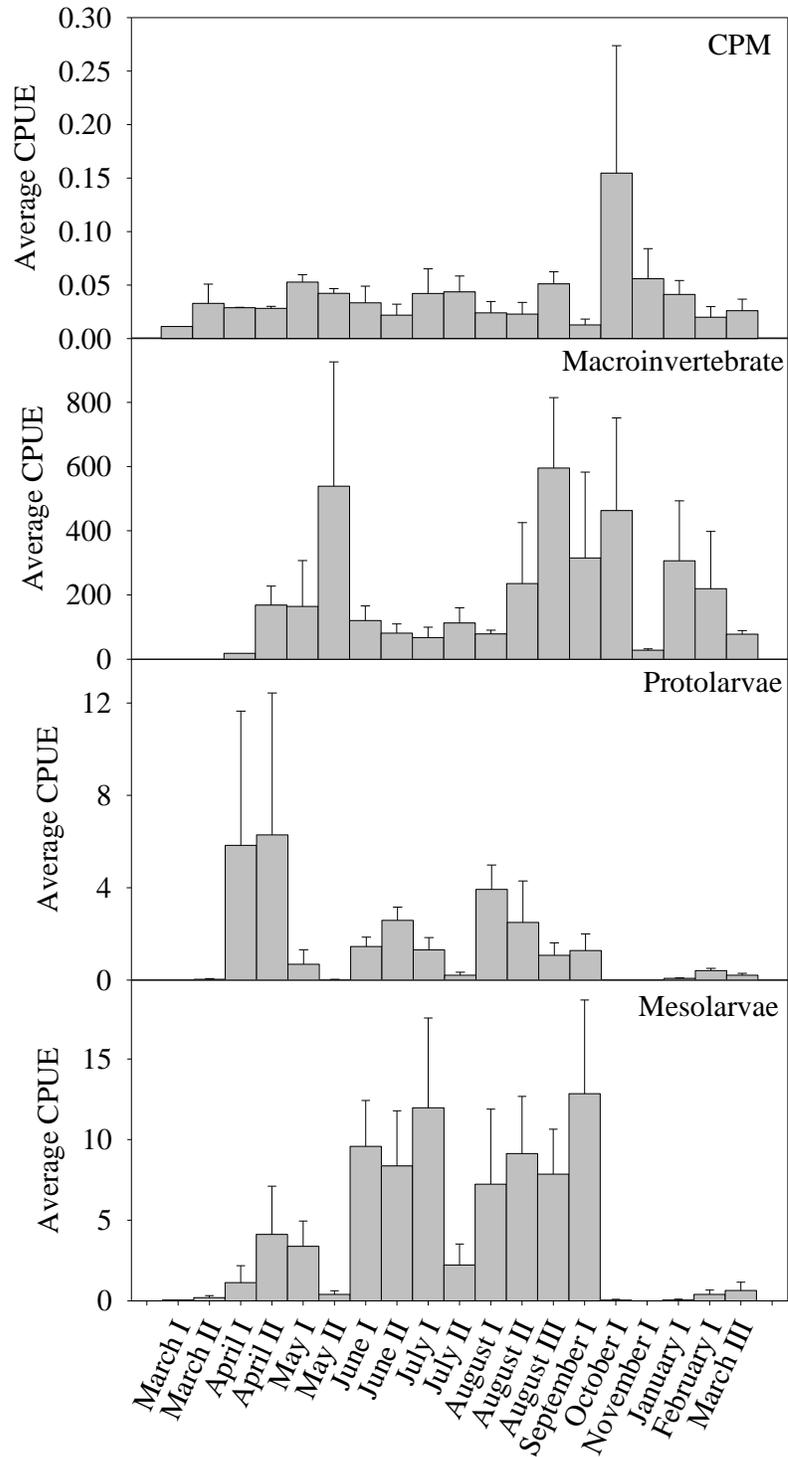


Figure 3. Mean \pm SE CPUE among four sites on the San Antonio and Guadalupe rivers from March 2012 through March 2013, during fish reproductive seasons, collections were made every two weeks, multiple collections within a month are denoted with roman numerals.

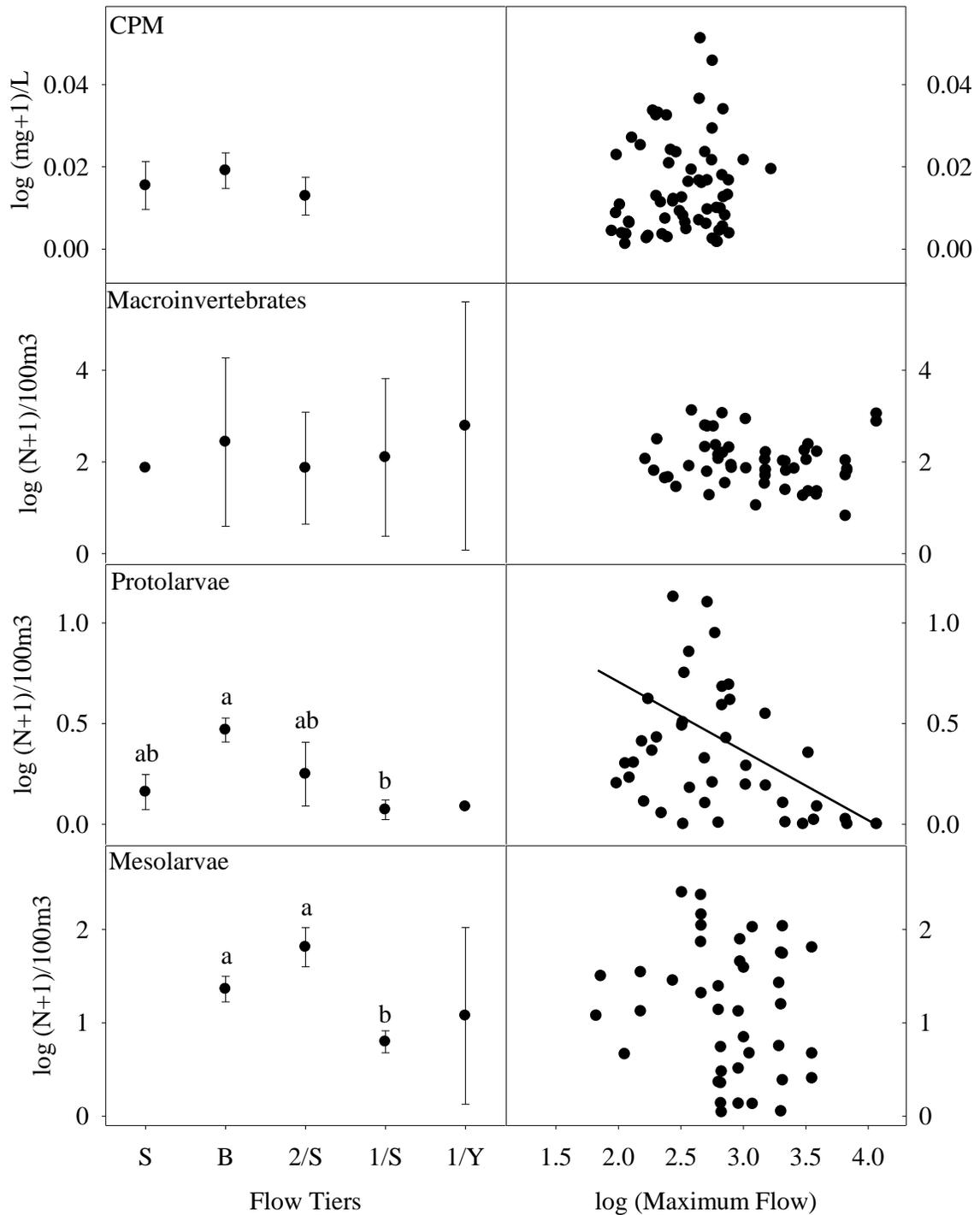


Figure 4. Mean \pm SE [$\log(N+1)$] CPUE response to flow tiers (left panel) and discharge [$\log(\text{maximum flow})$] (right panel) among four sites on the San Antonio and Guadalupe rivers from March 2012 through March 2013.

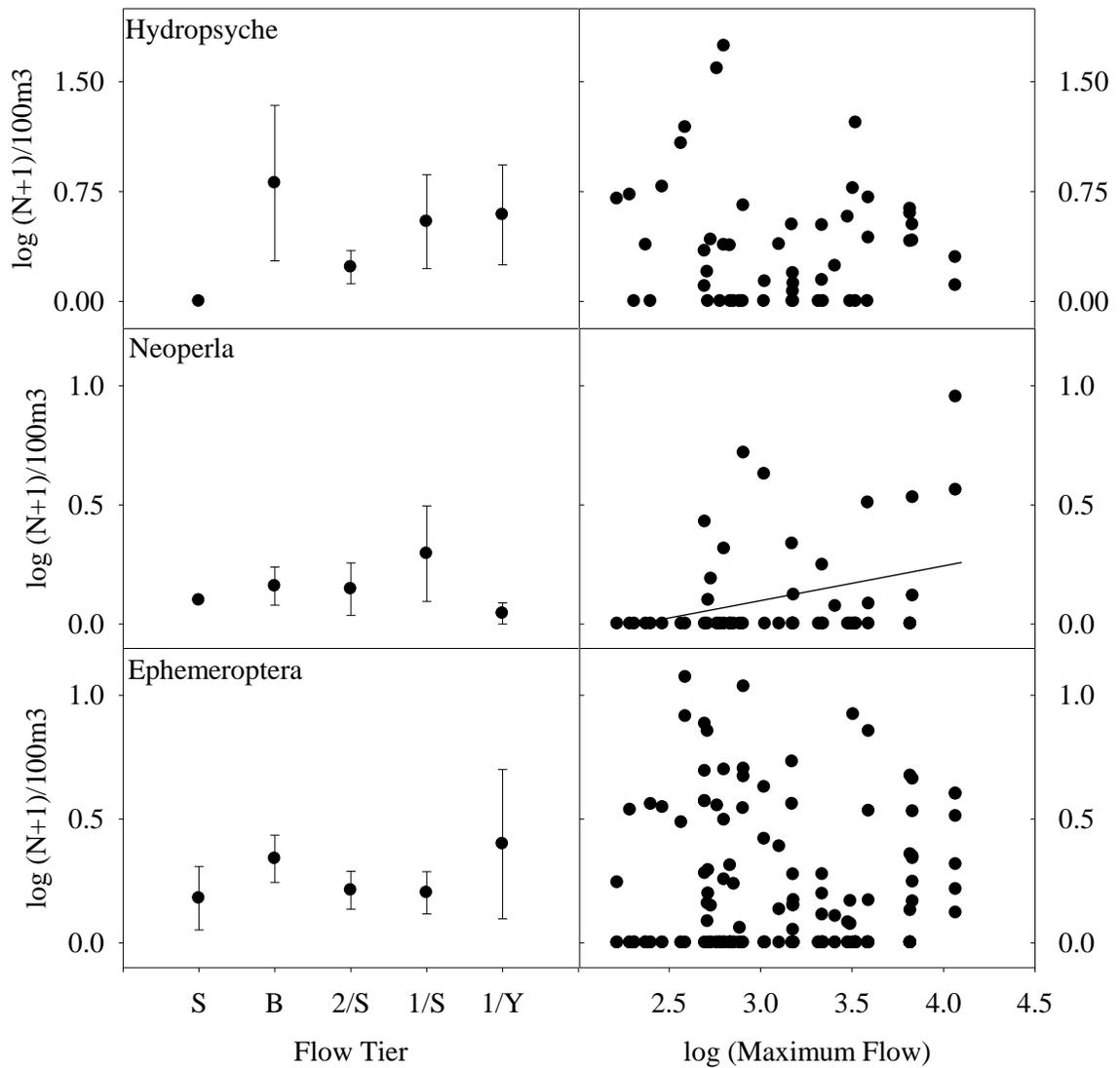


Figure 5. Mean \pm SE [log (N+1)] CPUE fluvial macroinvertebrates response to flow tiers (left panel) and discharge [log (maximum flow)] (right panel) among four sites on the San Antonio and Guadalupe rivers from March 2012 through March 2013.

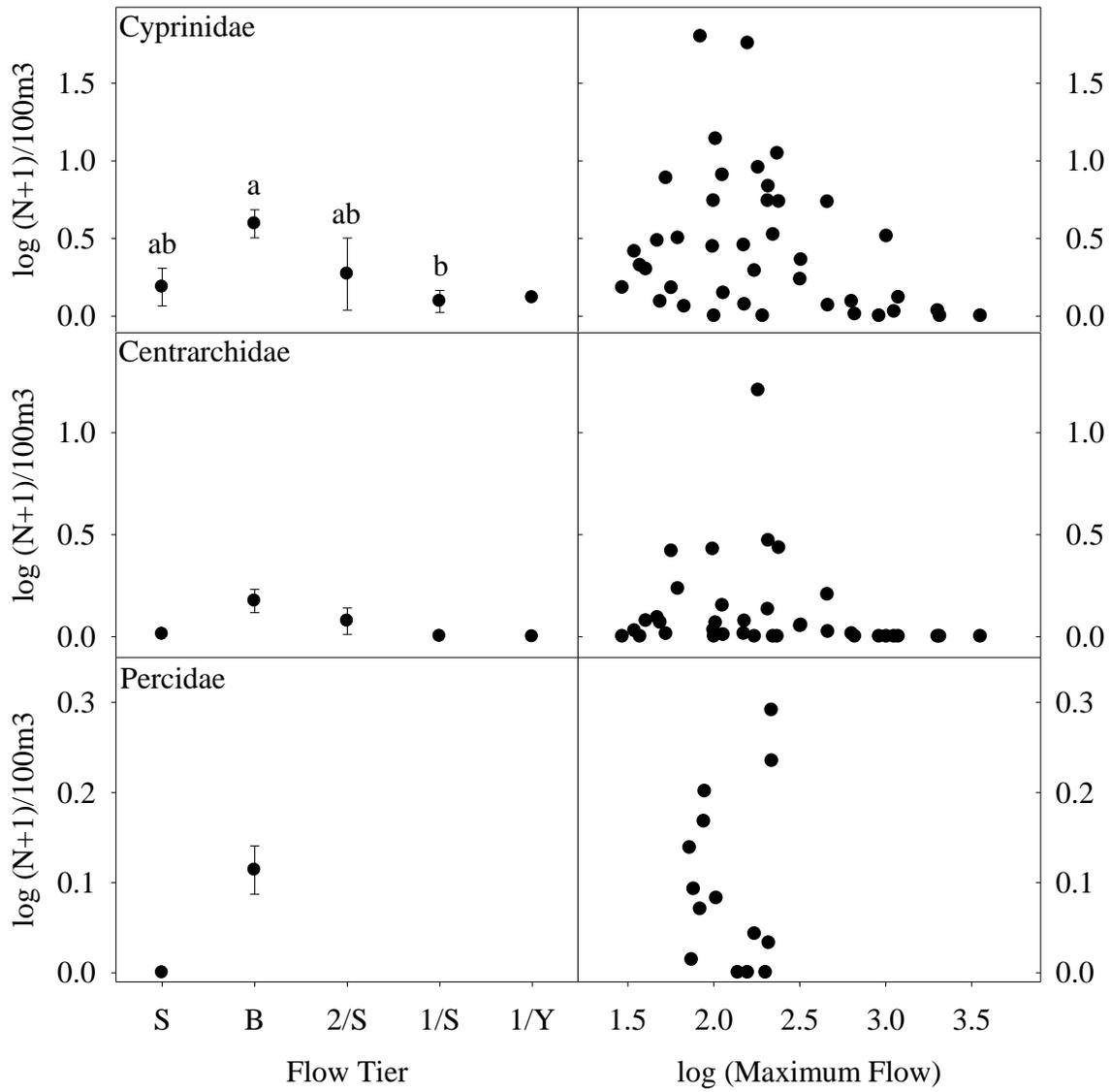


Figure 6. Mean \pm SE [log (N+1)] CPUE of protolarvae response to flow tiers (left panel) and discharge [log (maximum flow)] (right panel) among four sites on the San Antonio and Guadalupe rivers from March 2012 through March 2013.

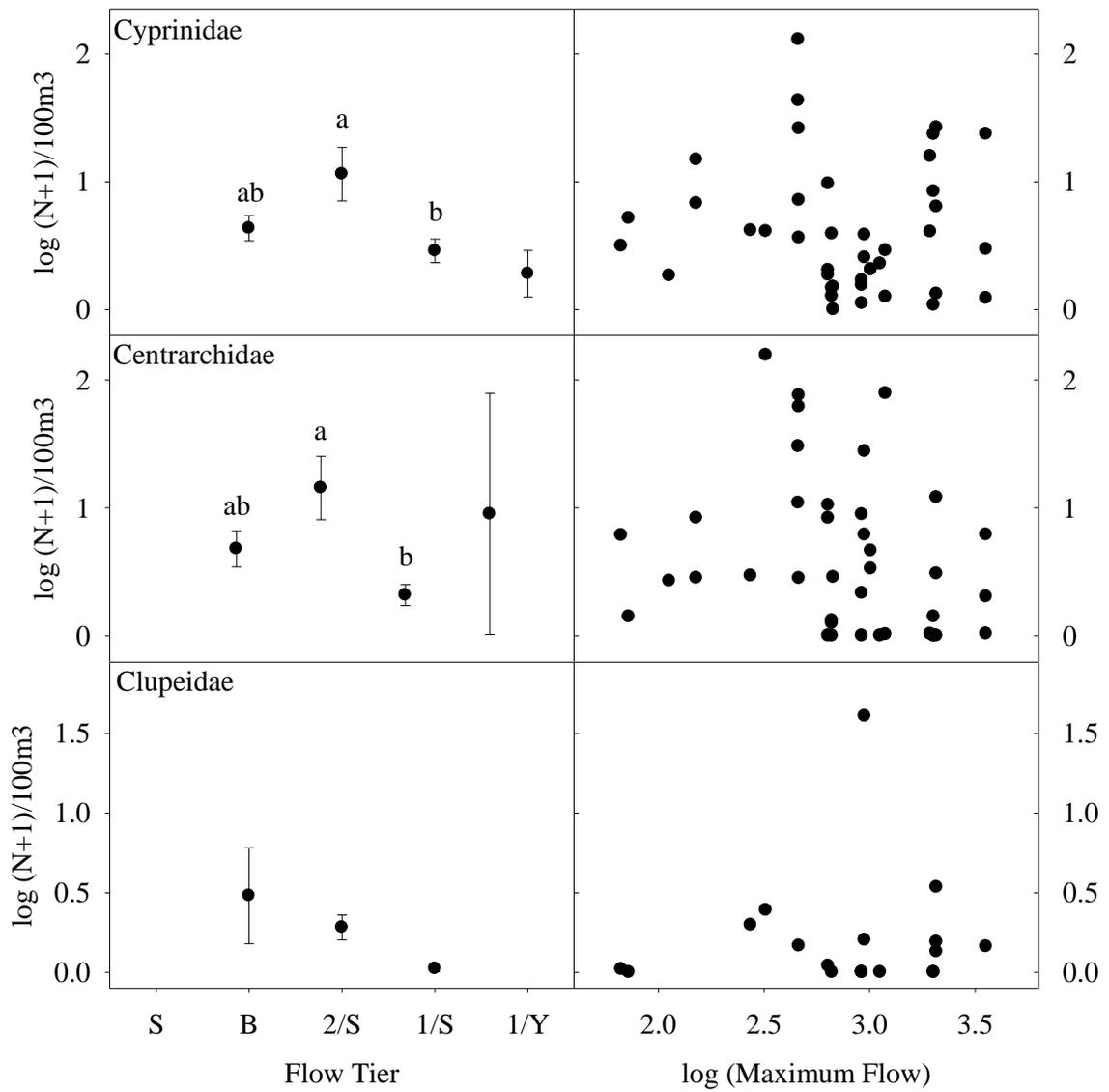


Figure 7. Mean \pm SE [$\log(N+1)$] CPUE of mesolarvae response to flow tiers (left panel) and discharge [$\log(\text{maximum flow})$] (right panel) among four sites on the San Antonio and Guadalupe rivers from March 2012 through March 2013.

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