

EFFECTS OF ARTIFICIAL LIGHT ON THE DRIFT OF MACROINVERTEBRATES
IN URBAN CENTRAL TEXAS STREAMS

THESIS

Presented to the Graduate Council of
Texas State University-San Marcos
in Partial Fulfillment
of the Requirements

for the Degree

Master of SCIENCE

by

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San Marcos, Texas
August 2013

EFFECTS OF ARTIFICIAL LIGHT ON THE DRIFT OF MACROINVERTEBRATES
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ACKNOWLEDGMENTS

I would first like to thank my major advisor Dr. Yixin Zhang for his assistance throughout this process. I would also like to thank my committee members Dr. Timothy Bonner and Dr. Clay Green for their comments and guidance.

Many thanks to my fellow members of the Stream Ecology lab: Leilani Williams and Mario Sullivan for their support and advice. Leilani especially has been a great friend and sounding board throughout this process. In addition, I would like to thank Harlan Nichols for all of his hard work in the field and lab. I would not have been able to pick and identify so many bugs without his assistance.

Finally, I would like to thank my family and friends for their continued support. I thank my mother and father, Ronald and Jayne Henn, for all of their love and encouragement during this time. I also thank them for fostering my love of science and animals by taking me to zoos and museums as a child. I thank my brother Steven and his family for always providing me with support and motivation. Last but not least, I thank my boyfriend Tommy, not only for his support, but also for providing me with a good reason to move to Texas and begin this entire process.

This manuscript was submitted on May 17, 2013.

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ABSTRACT

EFFECT OF ARTIFICIAL LIGHT ON THE DRIFT OF MACROINVERTEBRATES IN URBAN CENTRAL TEXAS STREAMS

by

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August 2013

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Since the majority of organisms operate on a circadian rhythm, light pollution in urban areas can possibly influence the aquatic community by affecting the drift of macroinvertebrates. The objective of this study is to examine if artificial night lights reduce drifting macroinvertebrates in the Edwards Plateau, by quantifying macroinvertebrate drift under ambient night light conditions and under extreme artificial lighting. This study was conducted in five streams (two large streams and three smaller streams) within urbanized areas of Central Texas. Drifting macroinvertebrates were sampled using drift nets under two treatments: ambient lighting (control) and extreme artificial lighting. Among all streams, both taxon diversity and richness of drifting insects was similar between treatments, but average abundance of drifting insects was 37% less in artificial lighting treatment than under the control treatment. Treatment effects were more evident in larger streams than smaller streams. Average abundance of drifting insects was 40% less in artificial lighting treatment with larger streams with

notable decreases in Simuliidae (58% less than the control), Baetidae (51% less), and Coenagrionidae (50% less). Reduced drift by artificial light conditions found in this study suggests the potential of artificial lighting disrupting dynamics of macroinvertebrate drift. Results of this experiment support a growing body of knowledge on how urbanized systems will influence stream communities and provide evidence to support various management strategies to minimize the effects of artificial lights on aquatic communities.

CHAPTER I

EFFECT OF ARTIFICIAL LIGHT ON THE DRIFT OF MACROINVERTEBRATES IN URBAN CENTRAL TEXAS STREAMS

INTRODUCTION

Physical characteristics of stream ecosystems including water quantity, stream morphology, and photoperiod, are susceptible to the influence of urbanization (Feminella and Walsh, 2005). Natural photoperiod can be disrupted by artificial night lights (i.e. street lights, safety lights, commercial lights) near or distant from aquatic systems (Longcore and Rich, 2004). Since many organisms operate on a circadian rhythm, ecological light pollution affects the biological community by disrupting drift, migratory, and reproductive habits of aquatic macroinvertebrates, thereby possibly affecting macroinvertebrate populations and aquatic communities through changes in productivity, and interspecific and intraspecific interactions (Longcore and Rich, 2004; Hölker et al., 2010b; Perkin et al., 2011). However, effects of light pollution on aquatic systems are reported in a limited number of studies. To study the effects of light pollution on the environment, scientists have created a new term called “ecological light pollution” that refers specifically to artificial lights which can disrupt ecosystem functioning (Longcore and Rich, 2004). This differs from “astronomical light pollution,” which refers to the obscuring of celestial bodies from humans (Longcore and Rich,

2004). Since 99% of the United States population experiences some level of light pollution (Cizano et al., 2001), it is expected that urban aquatic environments will also be affected by artificial lighting. In this study, I assessed the effects of artificial light directly on a stream ecosystem in order to document disruption on macroinvertebrate communities. Greater understanding of the extent and effect of artificial lighting is necessary to inform policy and management (Longcore and Rich, 2004; Perkin et al., 2011).

Artificial lights disrupt dispersion of aquatic macroinvertebrates (Perkin et al., 2011). Downstream drift is an essential part of dispersion among many aquatic invertebrates, and disruption of drift can alter community composition (Smith et al., 2009). Macroinvertebrate populations are connected through drift, so a reduction in drift can ultimately decrease total population size and reduce gene flow (Palmer et al., 1996; Bilton et al., 2001). Macroinvertebrate drift rates are highest immediately following dusk and continue through the night (Ciborowski, 1982). A number of taxa within Ephemeroptera undergo diel drift, including Baetidae, Caenidae, Heptageniidae, and Leptophlebiidae (Elliott and Minshall, 1968; Pearson and Franklin, 1968; Steine, 1972; Cloud and Stewart, 1974; Casey, 1987). Other families, such as Chironomidae, Elmidae, and Simuliidae drift primarily at night as well as Plecoptera and Trichoptera taxa (Elliott and Minshall, 1968; Pearson and Franklin, 1968; Brusven, 1970; Steine, 1972). As such, I predict that nighttime drifting taxa will be most affected by artificial lighting.

Bright light reduces macroinvertebrate drift under laboratory conditions (Bishop, 1969, Chaston, 1969) and natural settings (Perkin et al., 2011). Behavioral drift is actively undertaken to locate optimal living conditions (Müller, 1974) and the propensity

for nighttime drift in macroinvertebrates is attributed to predator avoidance (Flecker, 1992). Whereas drift of aquatic insects is often overlooked in ecosystem level studies, research suggests that a decrease in drift often leads to changes in local ecological processes (Palmer et al., 1996). Disruption of nighttime drift due to artificial lighting could cause insects to stop drifting over areas with suboptimum substrate or increased predation, leading to changes in recruitment for individual species and overall community composition (Blakely et al., 2006; Smith et al., 2009). Additionally, artificial light could reduce the number of benthic macroinvertebrates leaving the substrate, possibly causing a buildup in the larval population and increasing interspecific interactions (Palmer et al., 1996).

Many studies concerning light effects on invertebrate drift take place in natural environments in northern latitudes and do not investigate the possible effects of artificial light in arid climates. The purpose of this study is to conduct a preliminary assessment on the effects of artificial lights on nighttime drift of macroinvertebrates in headwater streams within urbanized areas of the Edwards Plateau. Edwards Plateau streams are supported predominantly by groundwater discharge (Hubbs, 1995; Groeger et al., 1997) and offer a unique opportunity to assess the effects of artificial lights in streams with high water clarity. As spring-fed systems with high clarity, artificial lighting has the potential to reach further down into the water column, possibly influencing the surrounding aquatic system. Headwater streams along the eastern edge of the Edwards Plateau between Austin and San Antonio are also within one of the fastest developing urban population centers of Texas (Texas Almanac 2012-2013). The objectives of this study are to quantify aspects of macroinvertebrate drift (diversity, abundance, and biomass) under

ambient night light conditions and under intense night light conditions in urbanized streams and to determine if artificial night lights affect all drifting macroinvertebrates equally. I hypothesize that macroinvertebrate drift diversity, richness, abundance, and biomass will be reduced by artificial night lighting within the headwater streams.

METHODS

Study sites

Study sites were located on two large spring runs with moderate levels of urbanization and three small spring runs with high levels of urbanization. The two large spring runs were San Marcos River (29.869395 °N, -97.930194 °W; Hays County, Texas) near Ed Capes Road and Comal River (29.710164 °N, -98.129171 °W; Comal County, Texas) near Elizabeth Avenue. The three small spring runs were Onion Creek (30.188457 °N, -97.71964 °W; Travis County, Texas) near McKinney Falls Parkway, Bull Creek (30.40468 °N, -97.789655 °W; Travis County, Texas) near Spicewood Springs Road, and Barton Creek (30.295928 °N, -97.92642 °W; Travis County, Texas) near Route 71. Discharge at each site was near base flow conditions. Each site included moderately flowing riffle or run mesohabitats with low levels of ambient light pollution. All streams had healthy riparian zones with moderate to high levels of riparian cover.

Physical habitat characteristics and water quality parameters were similar within large streams and within small streams (Table 1). Large streams (>10 m width; San Marcos River and Comal River) consisted of moderate depths (>0.5 m), swift current velocities (>0.30 m/sec), and gravel to cobble substrates. Smaller streams (<5 m; Barton

Creek, Bull Creek, and Onion Creek) consisted of shallow depths (<0.3 m), slow current velocities (<0.1 m/sec), and sand to bedrock substrates. Water temperatures were 22 - 23°C in large streams and cooler (<16°C) in small streams. Dissolved oxygen levels were relatively low across all streams except at Bull Creek (10.6 mg/l). Specific conductance ($\mu\text{S}/\text{cm}$) ranged between 491 and 650 $\mu\text{S}/\text{cm}$ across all streams. Ambient light was 1-2 lux across all streams.

Field collection

Five streams in Central Texas were sampled under two treatments: ambient lighting (control) and extreme artificial lighting. Extreme lighting was necessary to ensure that sufficient artificial light can produce an effect. Each stream represented a replication and each drift net represented an experimental unit. Each night of sampling utilized one drift net under one of the two treatments. Streams were sampled from November to December 2012. For each sampling night, a net was assigned the ambient light or artificial light treatment. Drift nets for each treatment were placed on separate evenings to ensure independence between nets within the same stream. Nets were set for two hour time periods starting within an hour after local sunset since drift is highest during the first few hours after sunset (Cloud and Stewart, 1974). Both light and no light treatments for each site were conducted within a week of one another to limit environmental differences between treatments. Additionally, a distinction was made between small and large stream segments. The San Marcos river and Comal river were categorized as large streams (>10 m width; >0.3 m/s velocity). Barton creek, Bull creek, and Onion creek were categorized as smaller streams (<5 m width; <0.1m/s velocity).

Drift nets were placed in an area of stream with the highest current velocities, typically downstream from a riffle, and supported by two t-posts. Nets were placed in the water column at least 5 centimeters above the substrate to minimize the likelihood of benthic insects crawling into the net. In large streams with a depth greater than the height of the net, the net was placed immediately below the water's surface, since invertebrate drift has been shown to be greatest in the surface current layer (Furukawa-Tanaka, 1992). For the artificial light treatment, two t-posts were set 1 meter apart and held up a 1.2 meter wooden plank with four portable work lights attached. Each portable work light held a 300 watt light bulb and placed about 0.6 meters above the stream surface. The lights were powered by a portable gas-powered generator. Artificial lights intensely illuminated an area one meter wide and four meters long and were placed 1 meter upstream of the drift net. The ambient light treatment in smaller streams consisted of one drift net. In large stream segments, the ambient light treatment included an additional two t-posts to simulate the setup of the light array. This addition was intended to control for the possibility of vegetation from the streams with high current velocity attaching to the t-posts and disrupting insects as they drift past. After each sampling period, all contents collected from the drift nets were stored in 95% ethanol.

In the field, a lux meter was used to measure luminous flux per unit area under each treatment. The measurement of lux is not necessarily biologically relevant as it only takes wavelengths visible to humans into account (Longcore and Rich, 2004); however, the light treatment was at least an order of magnitude greater in lux value. Current velocity (m/s) was measured at the beginning and end of each sample period using the Marsh-McBirney Flo-mate 2000 flow meter. Water depth and width were measured in

meters. Water quality parameters [temperature ($^{\circ}\text{C}$), conductivity ($\mu\text{S}/\text{cm}$), and dissolved oxygen (mg/L)] were measured with a YSI consisting of. Percent substrate was visually estimated and categorized as percent sand, gravel, cobble, boulder, or bedrock.

Laboratory analysis

Invertebrates were separated from the rest of sample debris and placed in 95% ethanol for each sample site and treatment designation. Invertebrates were then identified to the family level and counted for abundance, richness, and diversity at each site and for each treatment. The three most abundant taxa across all five streams were selected to calculate total biomass using taxon-specific, length-mass relationships (Benke et al., 1999). Using the method from Benke et al., 1999, body length was measured for at least 30% of each treatment sample and biomass was calculated. There were no constants available for Leptohyphidae individuals and so length mass values from Ephemerellidae, a member of the same subfamily, were used instead.

Data analysis

Renkonen Similarity indices were used to assess similarities between macroinvertebrate communities among streams. Experimental design was initially developed to accommodate a paired t-test in order to block differences in macroinvertebrate communities among streams. Drift densities of all taxa among five streams did not differ ($P > 0.05$) between treatments; however trends in the data suggested an unanticipated stream size effect. Number of drifting taxa were more abundant and differed substantially between treatments in larger streams (San Marcos

River and Comal River) than in smaller streams (Bull, Barton, and Onion Creeks). With low power to detect differences with parameter statistics for large or small streams, I calculated differences in dependent variables (density, richness, and total number) between ambient light treatment and artificial light treatment for all streams, large streams, and small streams, calculated mean and 1 standard error (SE), and visually estimated distributional differences. Treatment effects were considered detectable if mean and 1SE did not include zero. Differences in biomass were calculated through the use of a paired t-test comparing results across all streams and within large streams.

RESULTS

Among ten sampling dates, we captured 3,190 larval individuals representing six insect orders: Ephemeroptera (53%), Diptera (24%), Coleoptera (11%), Odonata (6.6%), Trichoptera (4.7%), and Lepidoptera (1.3%). Thirty-six separate insect families were represented within the sample (Table 2). Most abundant families captured were Leptohypidae (33%), Baetidae (19%), Chironomidae (14%), Simuliidae (8.3%), Elmidae (6.8%), and Coenagrionidae (5.7%). The total number of individuals captured among sites varied by stream size, with 90% of the insects captured from large rivers (San Marcos River: 62%; Comal River: 28%). Comal, San Marcos, and Bull were on average 52% similar to one another in family abundances. These streams were similar in percent abundance of Baetidae, Chironomidae, and Coenagrionidae, while dissimilar in Elmidae and Simuliidae. Barton was 34% similar to the other four streams as it contained the

most abundant taxa across all streams, but few other taxa. Onion was 27% similar, and dissimilarity occurred due to high numbers of Scirtidae.

A Comparison of Light to No Light Treatments

Greater numbers of macroinvertebrates were captured in the no light treatment (62%) in comparison to the light treatment (38%). Mean differences (artificial – ambient light) in total number of larvae and richness were negative between treatments of artificial light and ambient light among all streams and within larger streams (Figure 1) but near zero within smaller streams. Average abundance of drifting insects was 37% less in artificial lighting treatment than under ambient treatment among all streams and 40% less in artificial lighting treatments within large streams. Mean differences in diversity were near zero between treatments among all streams and within large and small streams (Figure 1); therefore differences in diversity were not detected.

Mean differences (artificial – ambient light) among orders were visibly negative for Diptera and Ephemeroptera but near zero for Coleoptera, Odonata, Lepidoptera, and Trichoptera among all streams and within large streams (Figure 2). Among small streams, mean differences among all orders were near zero.

Mean differences (artificial – ambient light) among families were negative for Baetidae, Chironomidae, and Simuliidae among all streams (Figure 3). Mean differences were negative among Baetidae, Chironomidae, Coenagrionidae, Leptohyphidae, and Simuliidae among large streams and close to zero among small streams (Figure 3). Since taxa (both Order and Family) were not always found across all three small streams, differences between light and no light treatments within small streams were analyzed

between only 2 streams. In addition, some families were not present in large enough numbers to conduct this analysis within the small streams.

As the most abundant taxa across all streams, biomass results were calculated for Baetidae, Chironomidae, and Leptohiphidae. Average individual biomass (mg) of drifting invertebrates for each treatment did not vary in any of the three families across all streams or within large and small streams (Figure 4). Total biomass, calculated by multiplying average biomass by the total number of captured individuals per treatment differed significantly ($p < 0.5$) across all streams for Chironomidae, but not significant for Baetidae and Leptohiphidae (Figure 5). Within large streams, results were significant for Baetidae ($p < 0.001$) and Chironomidae ($p = 0.001$) with greater biomass found in the no light treatment than the light treatment.

DISCUSSION

From the ambient to artificial light treatment there was a decrease in total abundance of larval insects across all streams, partially supporting the predicted decrease in larval insect drift due to the addition of artificial light. When divided into smaller ($N=3$) and larger streams ($N=2$), differences were only apparent in larger streams. Among all streams, there was also a difference in a number of larval aquatic insect taxa between the light and no light treatment with an average difference of 38% across the eight families focused on during the analysis. These families included Baetidae, Chironomidae, Coenagrionidae, Leptohiphyidae, and Simuliidae and are all known to undergo diurnal drift (Elliott and Minshall, 1968; Pearson and Franklin, 1968; Brusven,

1970; Steine, 1972; Cloud and Stewart, 1974; Casey, 1987). This pattern was more evident and with greater differences (45% average difference across the eight families of focus) in larger streams. Results from the total biomass t-tests show that there were significant differences due to treatment for Baetidae and Chironomidae within large streams, providing further evidence of a treatment effect. Detectable effects among these families are similar to the literature for the Baetidae and Simuliidae families which decreased in drift abundance under artificial lighting (Bishop, 1969; Chaston, 1969). Baetids decreased from 92% (number expressed as percentage of individuals drifting in that light level versus drifting in control with complete darkness) in a low light treatment to 5.5% in the highest light intensity treatment and Simuliids decreased from 102% to 0% (Chaston, 1969). While a number of studies show that these families are diurnally regulated or possibly regulate under an artificial light regimen, I report here for the first time light effects on three of the families: Chironomidae, Coenagrionidae, and Leptohephidae which all experience a decrease in drift due to the addition of artificial light.

The prediction of decreased larval drift due to the addition of artificial light was not supported within the smaller stream segments of this experiment. Small streams showed little difference in total abundance and richness. Finally, each family captured in small streams on average showed no difference between the two treatments. Several plausible scenarios could explain these unexpected results. One possible explanation for this result is that the smaller streams were part of a more urbanized ecosystem, which can cause a simplification in the invertebrate stream community (Morse et al., 2002; Gray, 2004), potentially preselecting individuals with greater tolerance for urbanization and

light tolerance. Additionally, the site at Barton was the only one where the number of drifting individuals was greater in the light treatment than the no light treatment. A plausible explanation for this result is the primarily bedrock substrate (90%) which would provide limited cover from light for insects hoping to drop out of the drift and causing them to continue drifting. Finally, taxa with the highest rate of nighttime drift have been shown to continue drifting when undergoing high rates of predation (Forrester, 1994) and with some fish spotted under the artificial light setup the insects may have considered predation a greater threat than artificial light pollution.

The decrease in drift shown across all streams and within larger streams has implications for the community dynamics within streams and their adjacent riparian zones. Decreases in colonization could result in additional loss of ecosystem structure and function. Light barriers could limit dispersal of larval invertebrates to downstream reaches, changing the community's structure through simplification (Longcore and Rich, 2004; Blakely et al., 2006). Benthic macroinvertebrates are often overlooked contributors to the ecosystem and play a key role in the aquatic trophic system (Covich et al., 1999). Stream insects are a large source of secondary productivity and an uneven distribution of drifting macroinvertebrates could result in a net loss of stream-wide productivity (Huryn and Wallace, 2000). Since 82% of colonization movements are hypothesized to be caused by drift, this could lead to an uneven distribution of insects within the benthic substrate (Townsend and Hildrew, 1976). Disruption in drift due to artificial lighting could cause insects to drop out early, leading to changes in recruitment for individual species and overall community composition (Blakely et al., 2006; Smith et al., 2009). Urbanization, a major cause of the increase in artificial lighting is known to

create subpar aquatic habitat, potentially causing drifting insects to land in areas of poor substrate (Blakely et al., 2006). Additionally, as an ideal biological indicator, benthic macroinvertebrates can be used to provide insight to environmental managers on the general health of an urban stream (Purcell et al., 2009).

Across aquatic and terrestrial ecosystems, artificial light influences ecological processes such as dispersal, reproduction, communication and predation (Longcore and Rich, 2004). Artificial light influences other aquatic taxa, including fish, which require a base level of light for larval growth (Boeuf and Le Bail, 1999). In addition, light can also be used as a management technique for maneuvering adult fish around anthropogenic stream disturbances (Longcore and Rich, 2004). While these behaviors can be exploited by environmental managers, a constant light regime could become stressful and even lethal to larval fish (Boeuf and Le Bail, 1999). Terrestrial insects, particularly predators and scavengers are drawn to artificial lighting which changes the structure of both the insect community but also an entire community (Davies et al., 2012). Larger predators like bats exploit the numerous insects around urban artificial lighting but the bats may have trouble surviving in a highly developed and fragmented environment (Rydell, 1992). While potentially beneficial, bats may be attracted to artificial lights in areas that have depleted insect populations so further predation may alter community dynamics (Rydell, 1992). Artificial light may also alter the behavior of migratory birds and songbirds due to disorientation (Longcore and Rich, 2004). Since many organisms operate on a circadian rhythm, light pollution has the potential to impact the biological community by disrupting an organism's migratory or reproductive habits, thereby affecting biomass and population size through changes in interspecific and intraspecific

interactions (Longcore and Rich, 2004; Hölker et al., 2010b). Without substantial evolutionary change, some species that are highly sensitive to light regulation could go extinct (Hölker et al., 2010b).

Although there is evidence that artificial light changes ecosystems, the implications of these changes is still to be determined. Research in this area is relatively new (Perkin et al., 2011), and there are still predictions to be tested. Given sufficient evidence that larval drift in urban semi-arid Texas streams decreases due to artificial light, further studies should assess artificial light effects on larval insects with the addition of other urban stressors. As urbanization continues, artificial lighting is expected to increase and the environmental effects could intensify (Navara and Nelson, 2007). Streams in urban environments can be difficult to restore due to the numerous stressors at play (Purcell et al., 2009), which includes artificial light. It is necessary to determine how other urban stressors can change an urban stream's response to artificial light in order to get a full picture of potential effects.

The results of this experiment will inform researchers and environmental managers working with spring fed ecosystems in a semi-arid climate. This experiment provides support for the hypothesis that extreme artificial light has negative effects on the abundance of drifting larval insects creating a need for conservation managers to look for ways to mitigate the effects. One possibility is the development of riparian buffer zones to decrease the amount of artificial light reaching the stream environment. The decrease of riparian buffer zones has been shown to cause an increase in the light reaching streams and a subsequent increase in periphyton as well as varying effects on the invertebrate community (Gjerløv and Richardson, 2010). Since there is support for the idea that the

width of riparian buffer zones plays a role in shaping the biotic community (Kiffney et al., 2004), it is reasonable to assume that an increase in an urban buffer zone, may decrease the total amount of light reaching a stream environment. Diptera larvae have been shown to negatively respond to UV radiation within a short time frame (Donahue and Schindler, 1998), so buffers could reduce the total amount or duration of light influencing a sensitive biological process. However it may be important to utilize larger and healthier stream environments in these studies, as significant results could be difficult to obtain within overly simplified communities of small urban streams. Another possibility is the implementation of new artificial lighting programs (Hölker et al., 2010a) especially near aquatic systems. Since high intensity lights are thought to have the greatest effect on insect drift (more than wavelength) (Bishop, 1969; Chaston, 1969) and light intensity varies by light source (Perkin et al., 2011), changes in the type of artificial lighting could decrease the overall effects of artificial lights on the aquatic system. Installing low intensity light bulbs, directing light away from the water's surface or smart grids that turn off lights when not in use are among the possible solutions to light pollution. Some of these solutions are already being implemented in major urban centers. For instance, the city of Austin in central Texas has decided to spend \$15 million on higher efficiency LED lights and smart grids to reduce the amount of artificial light produced by the city. Tests of how stream communities respond to these higher efficiency lights could provide biological support to these initiatives as well as guide selection of future light efficiency measures. So while this experiment lends evidence to the negative impacts of artificial lights from urbanization, future research will be

necessary to determine how best to mitigate these effects and whether investments in mitigation have actual environmental benefits.

Table 1. Mean physical habitat parameters for each of the five sampling locations from November 2012 – December 2012.

	San Marcos River	Comal River	Barton Creek	Bull Creek	Onion Creek
Current velocity (m/s)	0.6	0.3	<0.1	<0.1	<0.1
Depth (m)	0.73	0.58	0.30	0.24	0.15
Width (m)	15.2	10.1	4.6	3.4	1.5
Substrate:					
% sand					33
% gravel	20	80		40	33
% cobble	80	20	10	60	33
% bedrock			90		
Temperature (°C)	21.8	22.9	12.8	15.1	14.1
Dissolved oxygen (mg/l)	5	5	4	10.6	4.1
Specific conductance (u/cm)	577	555	491	650	609
Ambient light (lux)	1	2	1	2	1

Table 2. Total number of individuals (N), richness (S), and Shannon-Weiner diversity (H') during the artificial light and ambient light treatments at each of the five sampling sites

	Barton Creek		Bull Creek		Onion Creek		Comal River		San Marcos River	
	Light	No light	Light	No light	Light	No light	Light	No light	Light	No light
Coleoptera										
Elmidae	6	3	0	0	0	0	12	12	71	112
Dropidae	0	0	0	0	1	0	0	0	0	0
Dystiscidae	0	0	0	0	0	2	1	0	0	0
Gyrinidae	0	0	0	0	0	0	0	0	0	1
Hydrophilidae	0	0	2	1	1	3	0	0	0	0
Lampyridae	0	0	0	0	0	0	0	4	0	0
Scirtidae	0	0	1	0	26	48	1	2	17	10
Tipulidae	0	0	0	0	0	1	0	0	0	0
Diptera										
Ceratopogonidae	0	0	0	0	0	0	0	1	0	2
Chironomidae	11	1	5	12	7	22	73	134	78	115
Culicidae	0	0	0	0	1	0	0	0	0	0
Empididae	0	0	0	0	0	0	1	5	2	0
Ephydriidae	0	0	0	0	0	1	0	0	0	0
Simuliidae	0	0	1	2	1	7	75	171	0	8
Stratiomyidae	0	0	2	1	0	1	0	1	6	3
Tipulidae	0	0	0	0	0	0	0	1	0	1
Ephemeroptera										
Baetidae	4	3	12	15	7	6	54	139	132	237
Caenidae	2	0	0	0	2	2	0	2	3	1
Ephemeridae	0	0	0	0	0	0	0	0	5	3
Heptageniidae	2	0	3	1	0	0	0	0	0	0
Leptohyphidae	22	0	9	7	0	0	41	68	377	512
Leptophlebiidae	1	0	1	0	0	0	4	0	5	18

Table 2 – Continued: Total number of individuals (N), richness (S), and Shannon-Weiner diversity (H') during the artificial light and ambient light treatments at each of the five sampling sites

Lepidoptera										
Crambidae	0	0	0	0	0	2	7	12	9	12
Odonata										
Aeshnidae	0	0	0	0	0	0	0	2	2	3
Calopterygidae	0	0	0	0	0	0	0	5	3	9
Coenagrionidae	9	0	11	12	0	0	6	20	44	80
Cordulidae	0	0	0	0	0	0	0	0	0	2
Libellulidae	0	0	0	0	0	0	0	0	1	0
Protoneuridae	0	0	0	0	0	0	0	0	0	1
Trichoptera										
Brachycentridae	0	0	0	0	0	0	1	1	2	7
Hydrobiosidae	0	0	0	0	0	0	0	0	1	2
Helicopsychidae	0	0	0	0	0	0	0	1	0	0
Hydropsychidae	0	0	1	3	0	2	2	10	2	1
Hydroptilidae	0	0	1	3	0	0	5	4	1	5
Philopotamidae	0	0	0	2	0	1	3	6	26	44
Polycentropodidae	0	0	0	0	0	0	0	0	6	7
Total	57	7	49	59	46	98	286	601	791	1196
Shannon Diversity	1.706	1.004	2.053	1.988	1.365	1.656	1.89	1.918	1.769	1.842
Richness	8	3	11	11	5	9	15	21	21	25

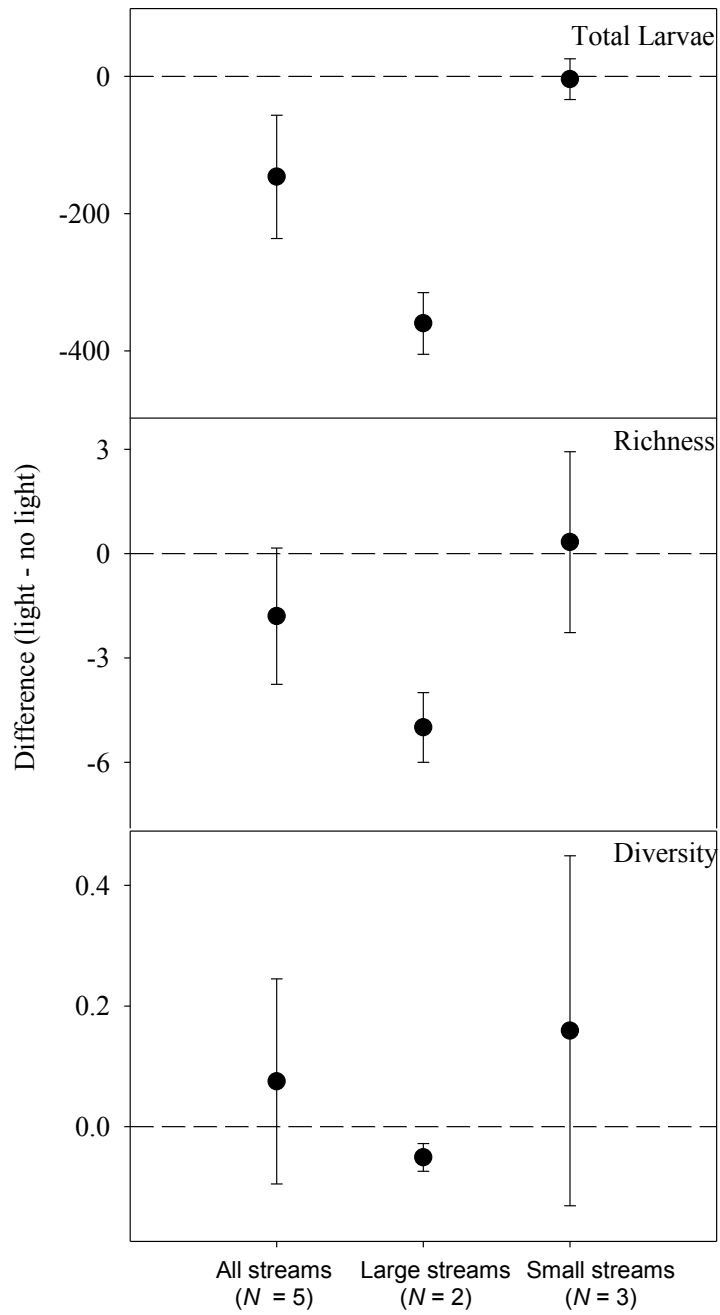


Figure 1. Mean abundance, richness (S), and diversity \pm SE across all streams and within large and small streams from November 2012 – December 2012.

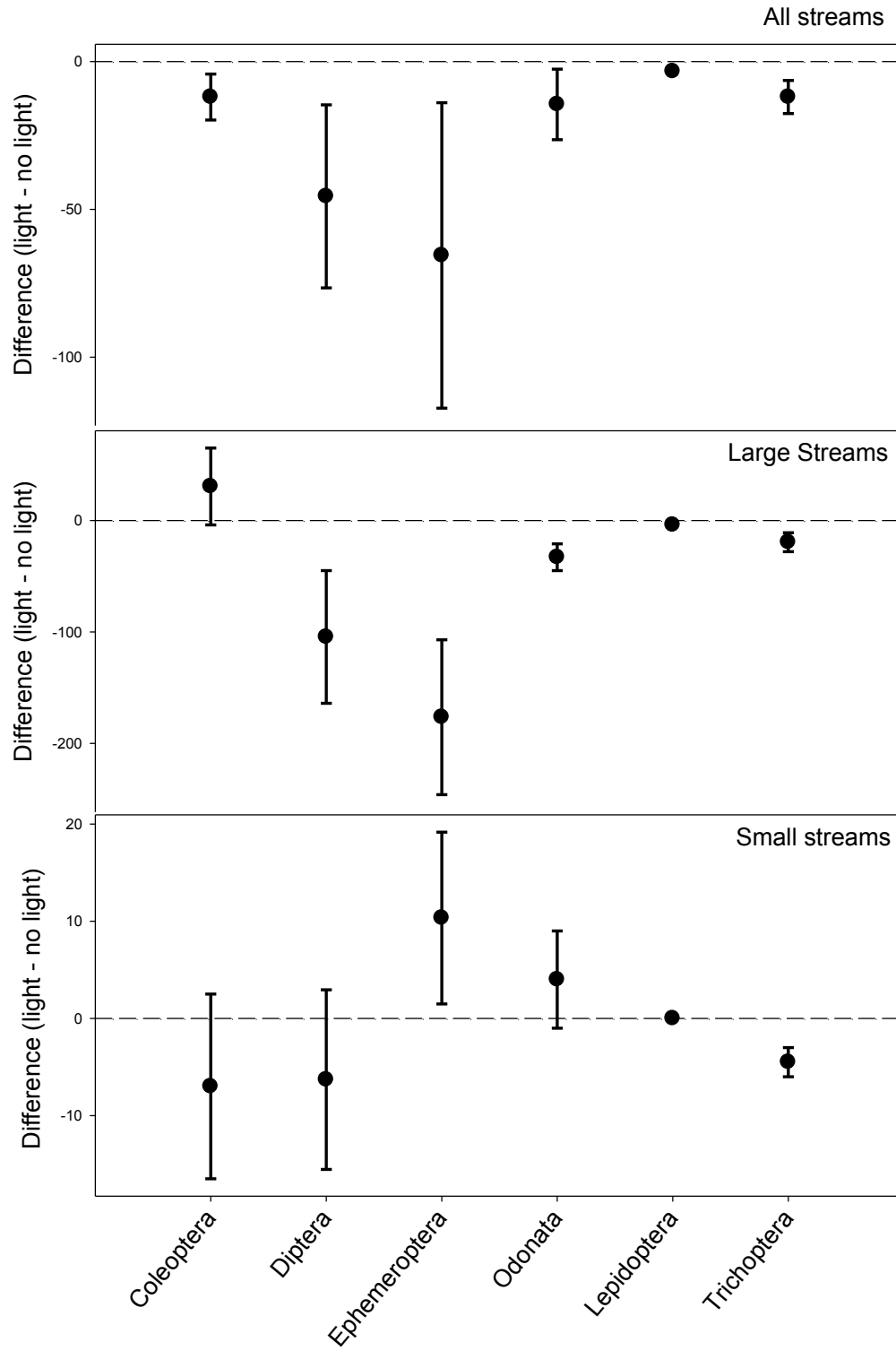


Figure 2. Mean abundance \pm SE for each order of aquatic macroinvertebrates across all streams and within large and small streams from November 2012 – December 2012.

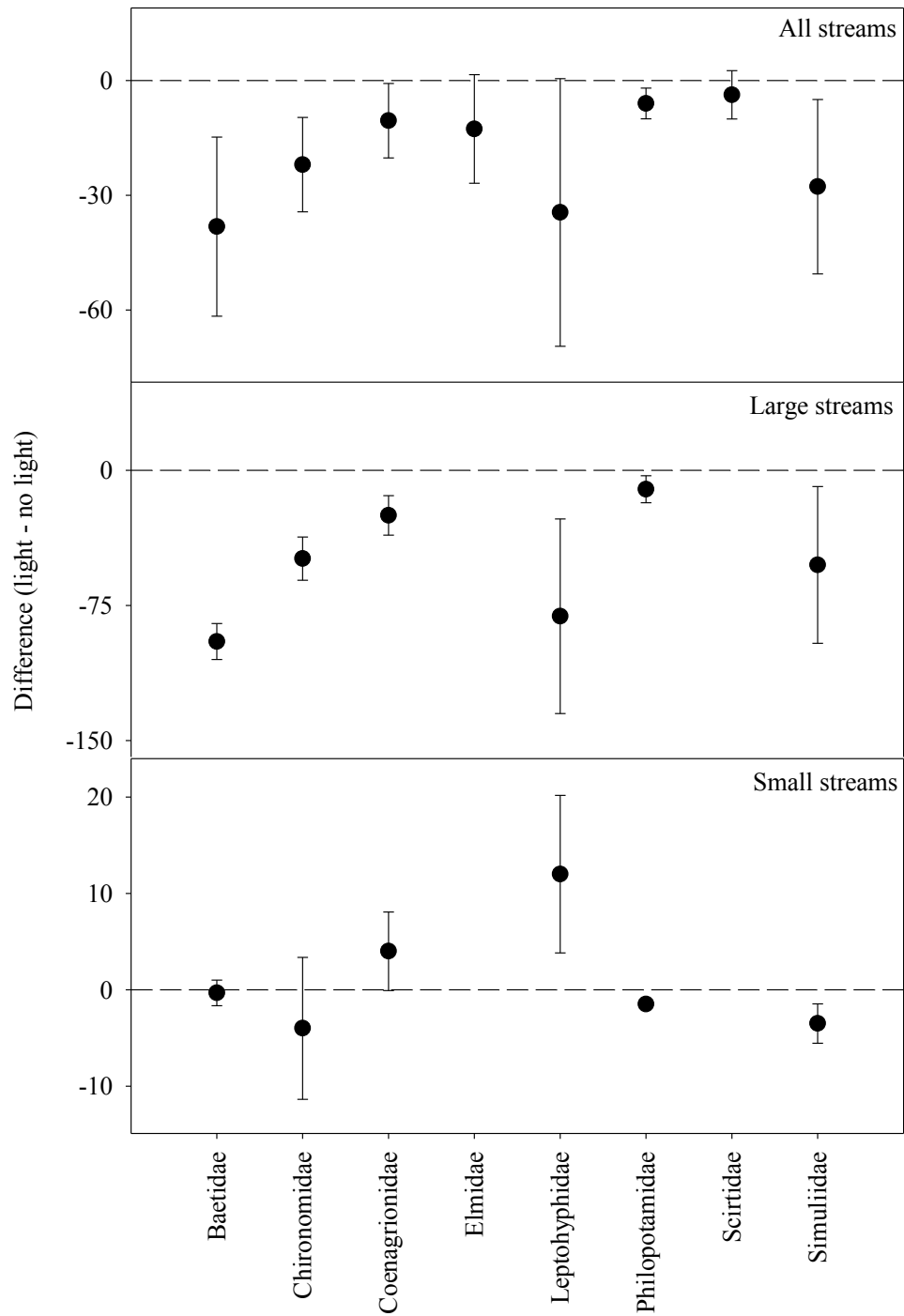


Figure 3. Mean abundance \pm 1SE for each family of aquatic macroinvertebrates across all streams and within large and small streams from November 2012 – December 2012.

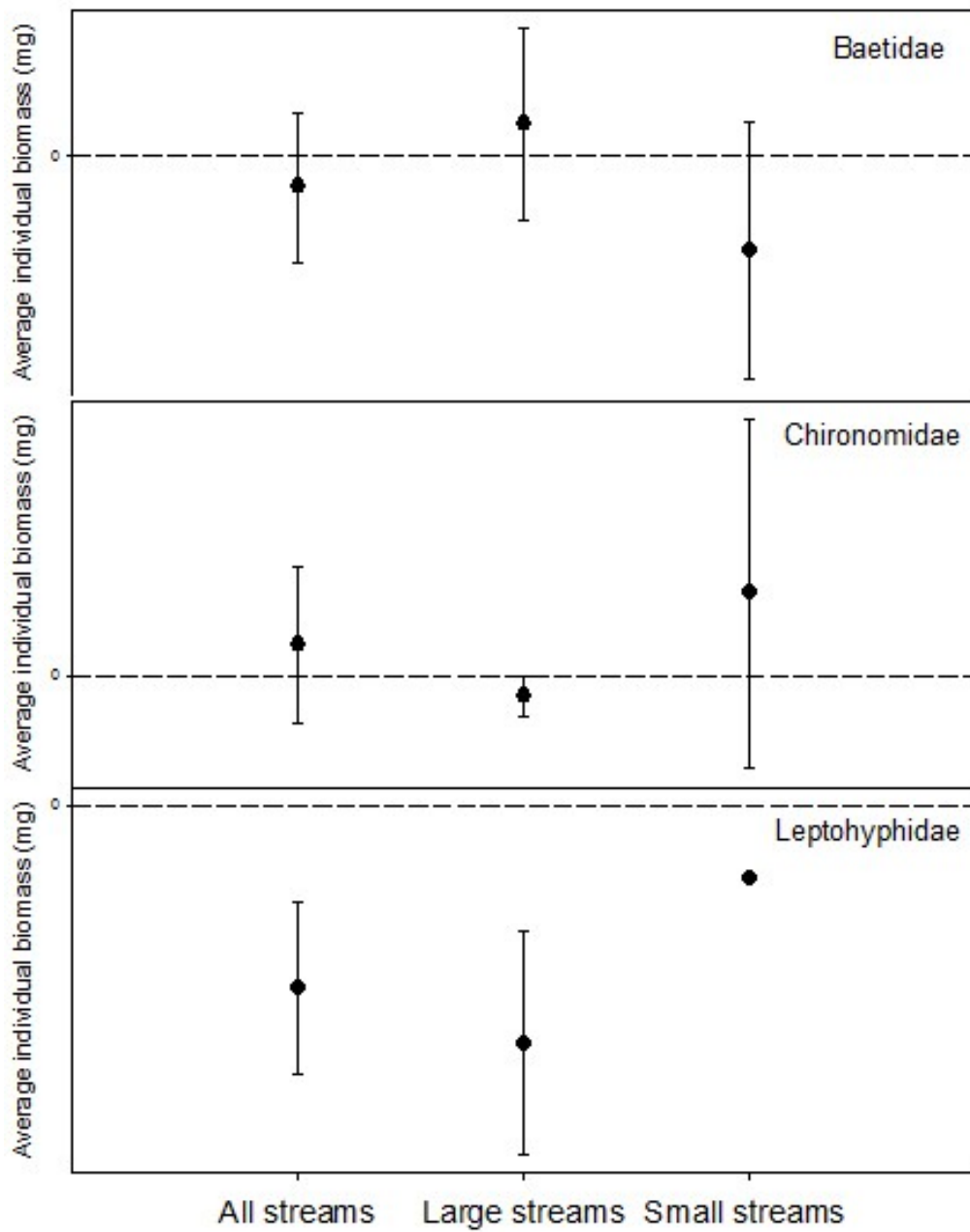


Figure 4. Average individual biomass (mg) \pm 1SE for Baetidae, Chironomidae, and Leptohypidae.

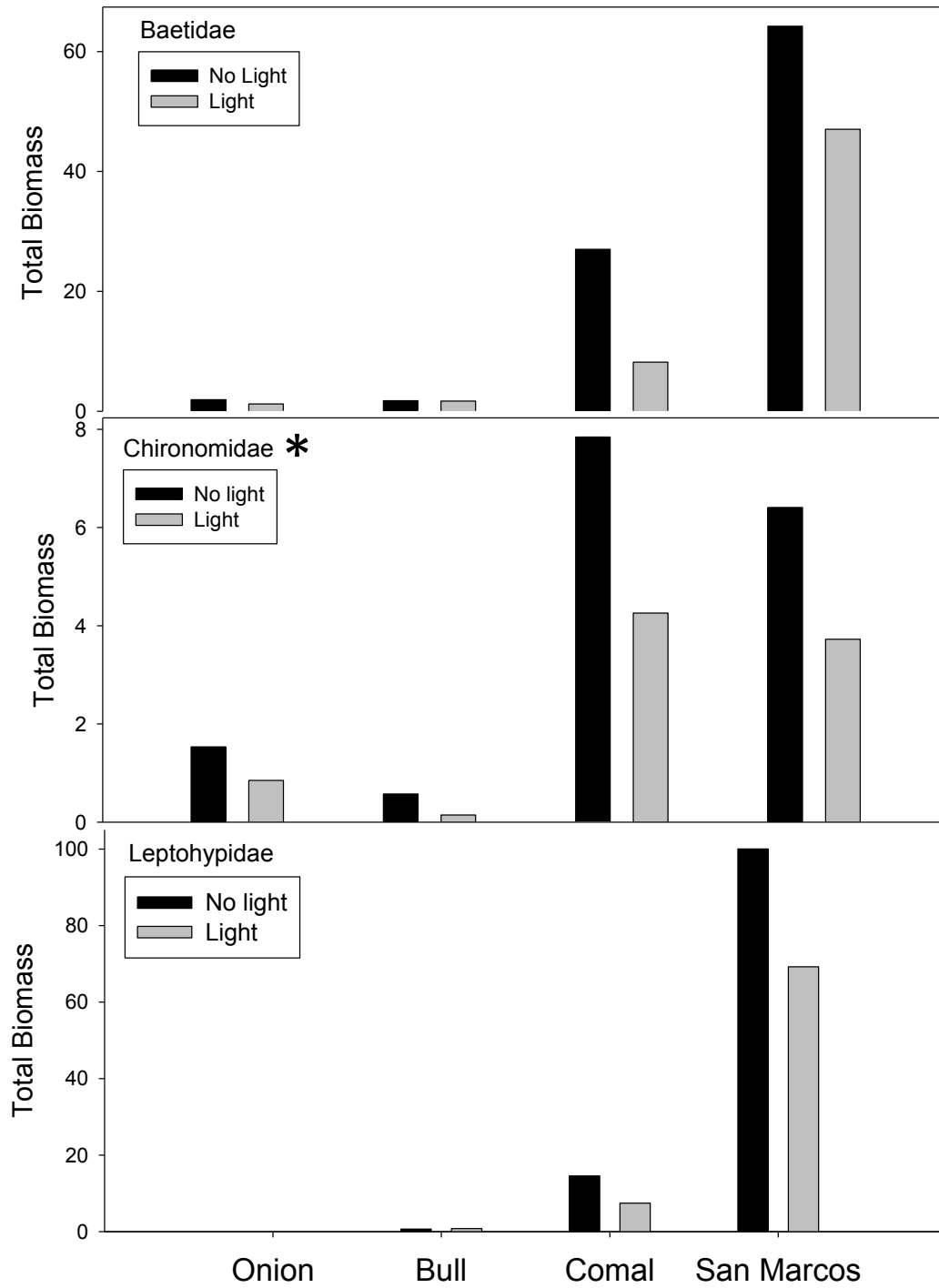


Figure 5. Total biomass (mg) for Baetidae, Chironomidae, and Leptohypidae. * denotes a significant difference based on a paired t-test.

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