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Please direct correspondence to:
Armand Gonzales
Editor-in-Chief
California Fish and Game
Armand.Gonzales@wildlife.ca.gov

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Notes from the Editor

In this issue of the California Fish and Game Journal, two articles report the negative effects, or potential negative effects of non-native invasive species on native species. There is yet another article reporting a novel appearance of a species not previously known to occur in Monterey Bay. As species move in response to changing habitat conditions, or as other factors such as climate create conditions that benefit non-native species, we are likely going to see more such events. As habitats change, species that depend on them, directly or indirectly, will need to adapt or relocate to areas that are more favorable. Hall et al., in this issue presents the results of a multi-year study which documents the decline in benthic organism in a residential stream, a condition which could in turn affect the numbers and types of birds, or fish that have food chains linked to the benthic community. The decline was largely due to an altered flow regime. The study focused on just one stream and its tributaries, although similar conditions can be found in many urban settings. Compounding factors such as extended drought exacerbate already degraded conditions. If extrapolated across the state where urban streams are being impacted it is possible we might see environmental or ecological conditions that have not previously existed or been described. Species will continue to move, however it may become increasingly difficult to distinguish between a non-native species extending its range and a native species becoming an invasive.

I would like to welcome Ms. Lorna Bernard, as our new production coordinator. Lorna worked for about 20 years with the Department of Fish and Wildlife and is now working as a retired annuitant. I would also like to thank Ms. Carol Singleton who has moved on, returning to her roots in education at the Department of Education. We wish the best for Carol in her new position. I would like to also welcome Paul Reilly and James Ray from the Department's Marine Region and David Wright from the North Central Region to the Editorial Board as new Associate Editors. Our last issue welcomed Kevin Flanders to the Editorial Board but we should have welcomed Kevin Flemings from our Habitat Conservation Planning Branch. Welcome Kevin.

Armand Gonzales
Editor-in-Chief
California Fish and Game

About the Covers

Front.—The leopard grouper (*Mycteroperca rosacea*) has a narrow range of occurrence and is listed as vulnerable on the IUCN Red-list. Its status on the red-list is due to over-exploitation in the form of intense subsistence, artisanal, and recreational fishing throughout the range of the species. Photograph by Dr. Paddy Ryan.

Rear.— In 2011 a gray wolf designated OR7 by the Oregon Department of Fish and Wildlife (ODFW) dispersed from his home pack in Oregon and entered California in December, 2011. He returned to Oregon in 2013, mated, and sired pups in 2014. His pack in Oregon is designated as the Rogue Pack.

In May and July, 2015 images were captured on a trail camera in Siskiyou County of a single adult, black wolf. Additional cameras were placed in the vicinity and in August, 2015 images of two separate adult black wolves, and five pups were captured. The California Department of Fish and Wildlife (CDFW) designated these animals the Shasta Pack. These are the only wolves known to occur in California at this time.

California Fish and Game 102(2):37-40; 2016

Northward range extension of the crowned sea urchin (*Centrostephanus coronatus*) to Monterey Bay, California

Jan Freiwald*, Colleen J. Wisniewski, Dan Abbott

Reef Check Foundation, 13723 Fiji Way, B-2 Marina del Rey, CA 90292 USA (JF, CJW, DA)

Long Marine Laboratory, Institute of Marine Sciences, UC Santa Cruz, 100 Shaffer Road, Santa Cruz, CA 95060 (JF)

**Correspondent: jfreiwald@reefcheck.org*

Keywords: *Centrostephanus coronatus*, Monterey Bay, range extension, El Niño, central California

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A crowned sea urchin, (*Centrostephanus coronatus)* was seen and documented on 12 April 2016 on rocky reef substrate in the kelp forest near the Monterey breakwater in Monterey Bay, California (36° 36.63' N, 121° 53.66' W) by the first author during a scuba dive. This observation represents a northward range extension of about 330 kilometers from the previously reported northernmost occurrence of this species at the northern Channel Islands, California (Engle and Richards 2001; Pearse 1972). This subtropical species has a reported distribution from the Galapagos Islands (Ecuador) in the south to the northern Channel Islands (California, USA) in the north (Gotshall 2005). The specimen was photographed and its test diameter was estimated to be about 4 centimeters (Figure 1). Its identification as *Centrostephanus coronatus* was confirmed by the co-authors based on photographs. There is no other known, almost black species of sea urchin in California waters with the characteristic blue around the base of the spines (R. Mooi, California Academy of Sciences, personal communication).

In southern California, this species is present in much lower densities than other common sea urchin species. Nevertheless, *C. coronatus* has increased in abundance over the last seven years in the southern Californian region. Reef Check California, a program that uses citizen scientists to survey California's rocky reefs using standardized 60 m^2 belt transects, has surveyed the densities of this species since 2006 (Freiwald et al. 2015; Gillett et al. 2012). These surveys document the spatial and temporal variation of crowned urchin recruitment and increases in population densities in southern California and show how, especially in 2015, its abundance has increased in the northern Channel Islands (Figure 2).

Adult crowned sea urchins occupy holes and crevices on shallow reefs and do not move far. After nightly feeding excursions of a few meters at most, they return to their home crevice before sunrise (Nelson and Vance 1979). Limited adult movement in combination with the body size of the observed individual suggests that it settled at least a year or two ago and has survived and grown on the reef in Monterey Bay. The northward range expansion

Figure 1.—Crowned urchin (arrow) photographed on 12 April 2016 near the Monterey breakwater in Monterey Bay, California. Specimen was found next to purple urchins in a crevice on rocky reef substrate at a depth of about 4 m.

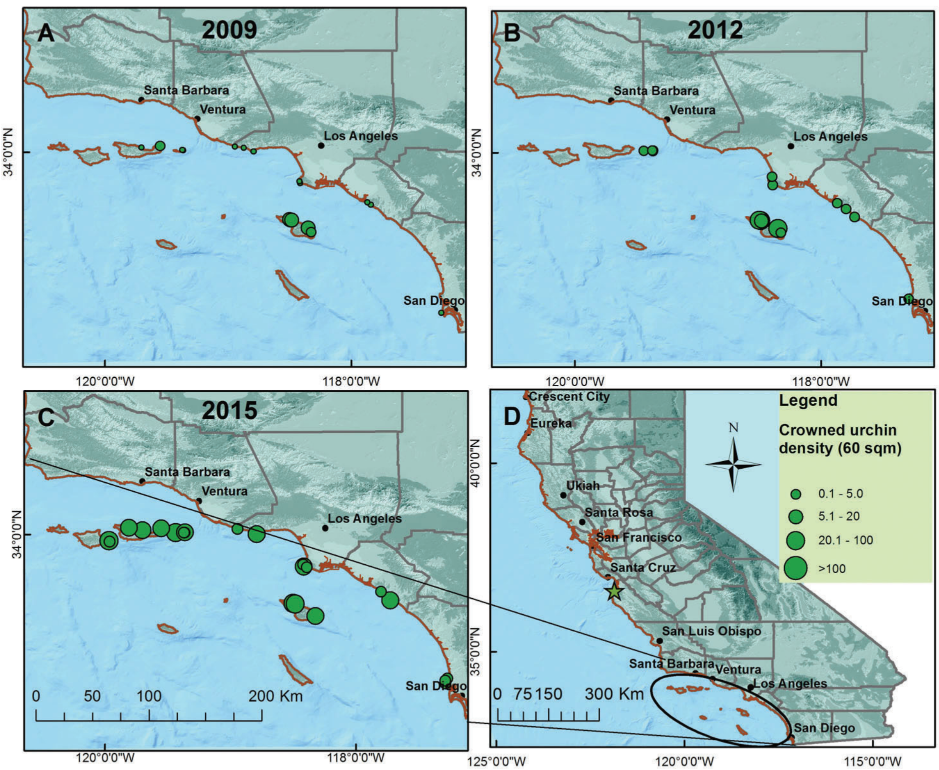

Figure 2.—Distribution of crowned urchins (*Centrostephanus coronatus*) in southern California as documented during Reef Check California surveys from 2009 to 2015 (panels A-C) and location of a single crowned urchin observed and photographed on 12 April 2016 (panel D, star). Oval in panel D indicates historic range of crowned urchins in California and dashed lines show section of coast shown in panels A-C.

of the crowned urchin is likely due to the recent marine coastal warming that has persisted along the California coast since 2013, and is referred to as the 'warm blob' (Leising et al. 2015; Peterson et al. 2015). Similar recruitment of this species was observed in association with a warm water event during the 1997/98 El Niño around the northern Channel Islands where it had been rare before that (Engle and Richards 2001). In the southern part of the species' range, increases in population density associated with warm water events have also been documented during an El Niño in 2009/10 when populations increased by about 70% at a site in southern Mexico (López-Pérez *et al.* 2016).

The observation of this subtropical species of sea urchin in central California is not the only case of a southern species being observed in this region during the recent warming along central California. In 2015, two species of urchins, *Lytechinus pictus* and *Arbacia stellata*, were found on a reef about one km upcoast of the site at which the crowned urchin was observed (Lonhart 2015). Along with recent sightings of several species of fish typically found in southern California (J. Freiwald, personal communication), these observations support the conclusion that this range extension of *C. coronatus* is related to the unusually warm water and El Niño conditions that have persisted over several years in central California since late 2013 (i.e. warm blob 2013-15 and El Niño in 2015-16).

Acknowledgments

We would like to thank the many Reef Check California volunteers that have helped collect the data used in this paper. We are truly grateful to those who have generously donated their time and effort to monitor California's kelp forest ecosystems. We would like to thank M. Carr for comments that helped improve this paper. Over many years, major supporters of the Reef Check California program have been the Resources Legacy Fund Foundation, the Keith Campbell Foundation for the Environment, the Lisa and Douglas Goldman Foundation, the California State Coastal Conservancy and California Ocean Protection Council (California Sea Grant) and the Annenberg Foundation.

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Incidence of juvenile Chinook salmon (*Oncorhynchus tshawytscha)* predation by green sunfish (*Lepomis cyanellus*)

Shaun T. Root,* Zachary Sutphin, and Donald E. Portz

Bureau of Reclamation, Fisheries and Wildlife Resources Group, P.O. Box 25007, Denver CO 80225, USA

**Correspondent: sroot@usbr.gov*

Key words: California, diet, *Lepomis cyanellus*, green sunfish, *Oncorhynchus tshawytscha*, Chinook salmon, San Joaquin River, San Joaquin Restoration Program

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Adverse effects of invasive piscivores have been cited as contributing to the declining abundances of inland California fishes, particularly in the Central Valley (Moyle and Marchetti 2006). Habitat alterations are also commonly cited, and the San Joaquin River Restoration Program has been established to promote both environmental and biological restoration within the San Joaquin River drainage. However, biological restoration efforts are often hampered due to an abundance of invasive piscivores that impede re-establishment or recruitment of native species (Portz and Tyus 2004). As a result, it is important to understand system-specific predator-prey relationships in order to set management priorities regarding invasive species control or eradication efforts. This understanding is of particular importance when invasive species are new to a system, or when interactions between native and non-native species are not well understood (Mack et al. 2000).

Green sunfish (*Lepomis cyanellus*) are native to North America east of the Continental Divide and west of the Appalachians, from the Great Lakes region south to the gulf coast states and into northeastern Mexico (Sublette et al. 1990), and were likely first introduced to California at Lake Cuyamaca, San Diego County in 1891 (Dill and Cordone 1997). Since introduction, they have become well established throughout the state in most suitable habitats (Moyle 2002).

Green sunfish are omnivorous foragers, and are equipped with a characteristically larger mouth when compared with other fishes within the Genus *Lepomis*. Dietary items include a variety of insects, mollusks, crustaceans, vascular plants, and are known opportunistic picivores. In Utah, fish comprise approximately 10% of total dietary intake (Wydoski and Whitney 2003), and are speculated to suppress native fish populations in North Carolina (Lemly 1985), Arizona (Dudly and Matter 2000), Colorado (Lohr and Fausch 1996), New Mexico (Paroz et al. 2010), and California (Moyle and Nichols 1974; Smith 1982). Effects of green sunfish predation on salmonid populations is elusive (Moyle 2002; Bonar et al. 2004; Grossman et al. 2013) and remains largely undocumented.

Juvenile Chinook salmon and green sunfish populations were historically geographically isolated from each other. Within the San Joaquin River they occur sympatrically and are likely restricted to similar habitat types defined by warm shallow water, low flow, with complex structures (ex. vegetation, woody debris, and rock weirs) as the San Joaquin River currently lacks continuous Chinook salmon (*Oncorhynchus tshawytscha*), habitat (Hallock and Fry 1967). The river has been channelized and is subject to major anthropogenic disturbances within the study area (eg., mining pits and diversion dams) that reduce flow and may cause an overlapping of habitat between the two species. In addition to anthropogenic disturbances, in 2015 the river experienced critically low water, reducing the volume and flow of water released from Friant Dam, which may have further reduced the ability of juvenile Chinook salmon to access suitable habitat and elude predators.

While conducting 2015 spring juvenile Chinook salmon trap and haul efforts (Portz et al., 2015) a series of V-shaped fence mesh weirs with tandem catch boxes were used to capture juvenile Chinook salmon. Potential juvenile Chinook salmon predators found in catch boxes were opportunistically gastrointestinally examined. Of green sunfish examined with a total length ranging from 74 to 210 mm (*n* = 36), four were found to contain juvenile Chinook salmon. One individual at Highway 99 (119º 56' 00" E, 36º 50' 31" N) and three individuals at Milburn (119º 52' 40" E, 36º 51' 20" N; Figure 1).

We are unable to discern if the predation events occurred in the catch boxes or within the river, and as such, it is unknown if the predation events are a valid reflection of free ranging predator-prey interactions within the system or the result of anthropogenic artifact as influenced by the weir.

To our knowledge, this is the first report of a Chinook salmon predation by green sunfish. Future efforts to understand factors associated with predator-prey relationships that drive successful reintroduction or successful recruitment in native salmonids should not exclude green sunfish.

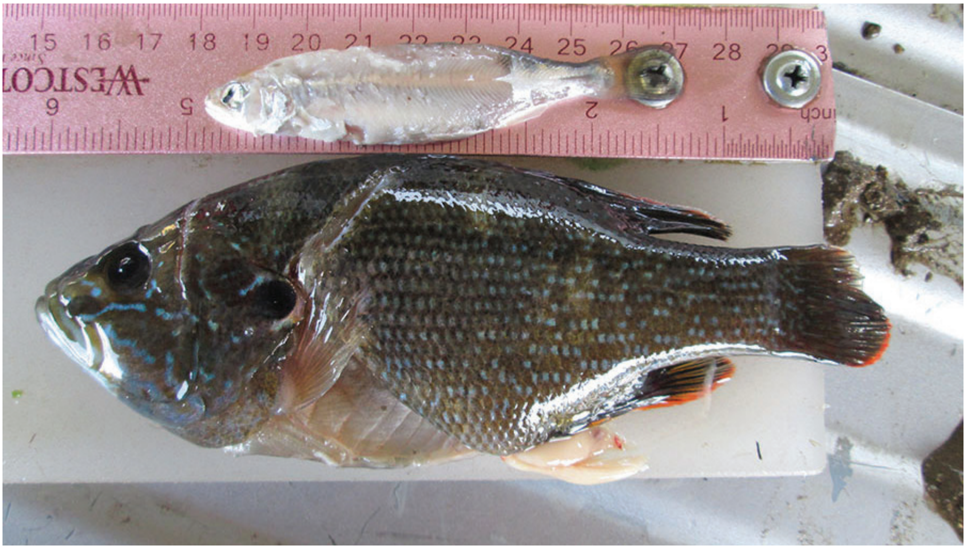

FIGURE 1.—Green sunfish (*Lepomis cyanellus*) captured in the San Joaquin River, Milburn unit, Fresno, California, USA, on 21 April 2015. Fork length was 147 mm Individual was gastrointestinally examined to determine dietary items. Items recovered included six sunfish (Lepomis spp.) and one Chinook salmon *(Oncorhynchus tshawytscha,* pictured*)*. Photograph by authors.

Acknowledgments

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Range expansion of the Shimofuri goby (*Tridentiger bifasciatus*) in southern California, with emphasis on the Santa Clara River

Steve R. Howard and Michael T. Booth*

United Water Conservation District, 106 N 8th St, Santa Paula, CA 93060 (SRH, MTB)

**Correspondent:* mikeb@unitedwater.org

Key words: Shimofuri goby, tidewater goby, California State Water Project, invasive species

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Shimofuri gobies (*Tridentiger bifasciatus*) are native to Asian estuaries from Hokkaido, Japan to Hong Kong, China and were introduced to the San Francisco Bay most likely via ballast water exchange from cargo ships shortly prior to 1985 (Moyle 2002). Shimofuri gobies were first discovered in the Suisun Marsh, part of the upper San Francisco Estuary. Gobies then spread to the Sacramento-San Joaquin Delta (Delta) (Matern 2001). Shimofuri gobies have spread from the Delta more than 500 km through aqueducts of the California State Water Project (CSWP), over the Transverse Ranges, and into coastal drainages of southern California (Matern and Fleming 1995). Shimofuri gobies have invaded reservoirs associated with the East and West Branches of the CSWP in Southern California (Figure 1). These include Pyramid Lake (Swift et al. 1993), Castaic Lake (D. Black, California Department of Fish and Wildlife (CDFW), personal communication), Silverwood Lake, Lake Matthews, and Skinner Reservoir (Q. Granfors, CDFW, personal communication). Shimofuri gobies are likely also present in Lake Perris and Diamond Valley Lake because of connection to the CSWP, but have not been detected as of June 2016 (Q. Granfors, CDFW, personal communication). Gobies were collected in Lower Otay Reservoir in San Diego in June, 2016 (D. Black, CDFW, personal communication). Shimofuri gobies were first discovered in the Santa Clara River watershed in 1990 at Pyramid Lake and were later found downstream of Pyramid Dam in middle Piru Creek in 1992 (Swift et al. 1993). Swift (1993) described these fish as chameleon gobies (*Tridentiger trigonocephalus*), but later identified them as Shimofuri gobies (C. Swift, Natural History Museum of Los Angeles, personal communication). Piru Creek is connected to the CSWP via the California Department of Water Resources (DWR) Pyramid Lake. Pyramid Lake functions in conjunction with Castaic Lake (which also contains Shimofuri gobies, date of first observation unknown), also located in the Santa Clara River watershed, but the outflow of Castaic Lake, Castaic Creek, typically has zero flow except in exceptional rainfall years or during water rights releases.

Lake Piru, formed by Santa Felicia Dam, is located on Piru Creek approximately 29 km downstream of Pyramid Lake (Figure 2). United Water Conservation District, the operator of Santa Felicia Dam, has performed occasional snorkel surveys of 500 m of the creek just below the dam, typically focused on population monitoring for Southern California steelhead/ coastal rainbow trout (*Oncorhynchus mykiss*). Fishes observed during surveys conducted

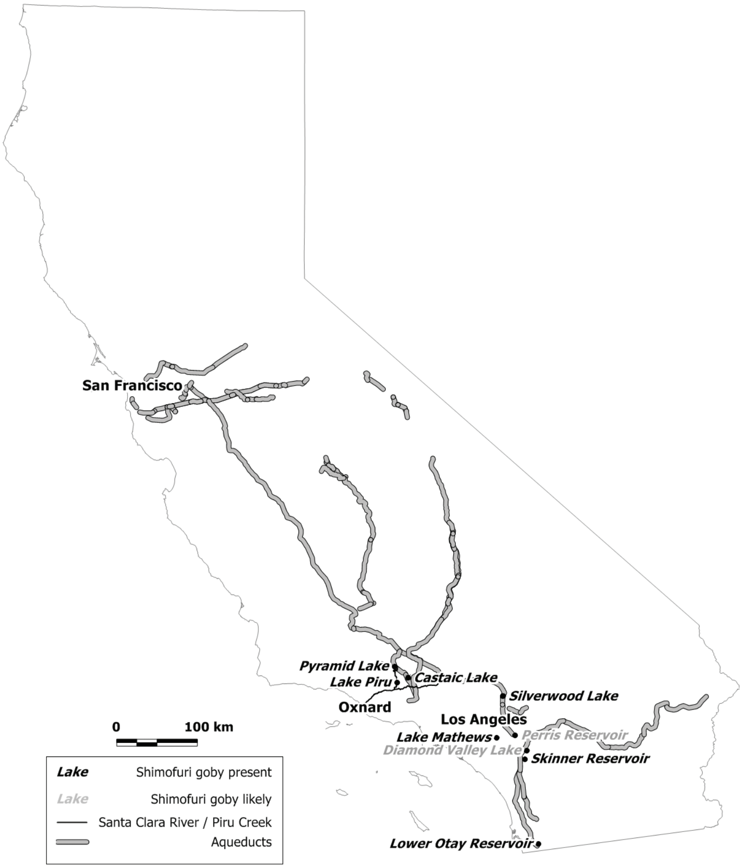

FIGURE 1.—Map of major canal and aqueduct systems in California, including the California State Water Project, highlighting lakes where Shimofuri gobies have been observed or are likely present in southern California.

before 2013 were partially armored threespine stickleback (*Gasterosteus aculeatus*), arroyo chub (*Gila orcuttii*), Santa Ana sucker (*Catostomus santaanae*), Southern California steelhead/coastal rainbow trout, prickly sculpin (*Cottus asper*), fathead minnow (*Pimephales promelas*), green sunfish (*Lepomis cyanellus*), and largemouth bass (*Micropterus salmoides*). During snorkel surveys in April, May, and August 2013, Shimofuri gobies were observed only during the August 2013 survey, but were abundant throughout the reach. On 20 August 2013, 17-voucher specimens were collected by seine directly below the Santa Felicia Dam outlet works. These specimens ranged in size from 36 mm standard length (SL) to 68 mm SL, and were submitted to the Natural History Museum of Los Angeles County ichthyology collection (LACM 58175-1) on 4 September 2013. A scuba dive survey on 7 July 2015, in Lake Piru showed Shimofuri gobies at high abundance near the intake for the penstock of the dam as well as other areas of the lake, but gobies were not observed in Piru Creek during a snorkel survey on 4 August 2015. Gobies were present in low numbers during seining

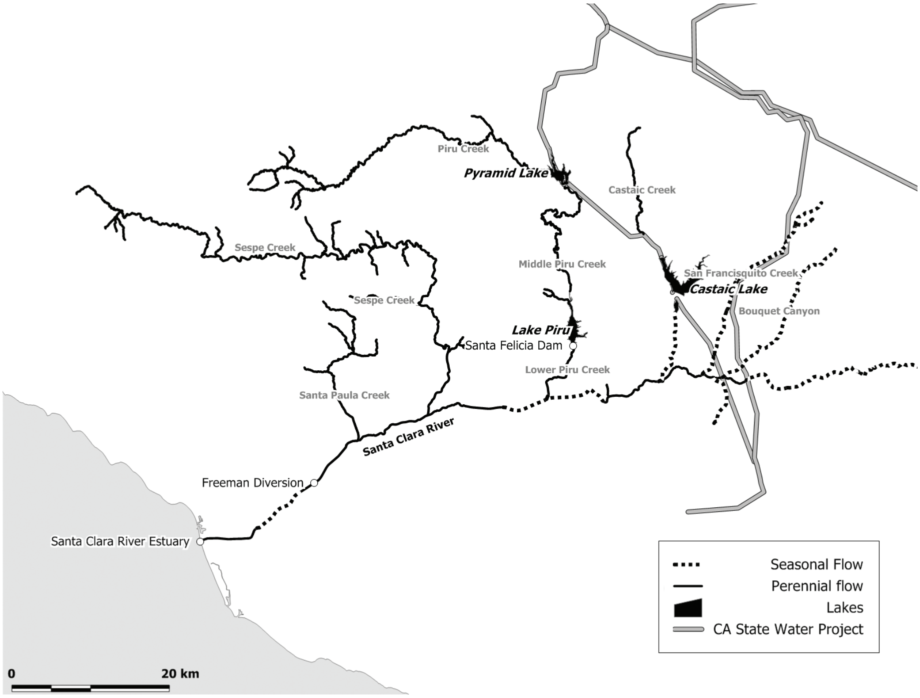

FIGURE 2.—Map of the Santa Clara River drainage and major tributaries, highlighting the California State Water Project aqueduct and lakes with Shimofuri gobies, the seasonally intermittent connection of the Santa Clara River and Piru Creek, and the Santa Clara River estuary.

surveys on 20 April 2016, below the Santa Felicia Dam outlet works. Gobies observed during snorkel and seine surveys in Piru Creek ranged from 40 mm total length (TL) to 80 mm TL.

Because regular monitoring did not occur on lower Piru Creek prior to 2013, it is not possible to definitively identify the timing of the invasion. However, based on the closely spaced surveys in 2013, and absence of the fish during the April and May 2013 surveys, it is possible that the invasion into lower Piru Creek occurred during a maintenance operation at Santa Felicia Dam on 30 July 2013. Typical water releases from Santa Felicia Dam route water through a series of pipes and valves that constrict the flow and release the water in a high velocity, turbulent spray into lower Piru Creek. While high flow releases (e.g., 5.66 m^3/s) may suction fish out of the lake or scour them out of the penstock due to pipe velocities that exceed 3.1 m/s, all typical flow release conditions likely kill or injure fish passing through the valves due to excessive velocity and turbulence at the pipe exit orifice. Unlike typical releases, during the maintenance draining operation the penstock pipe slowly dewaters with substantially slower and less turbulent flows, and allows fish to pass through alive and undamaged.

To test whether the draining of the penstock was a likely pathway for goby introduction to lower Piru Creek, on 9 May, 2016, all the water draining from the penstock during a maintenance event (approximately 450 m^3) was passed through a seine or dipnet to capture any fish released through the valves. Both Shimofuri gobies (155 individuals) and prickly sculpin (38 individuals) were captured during the draining event. All but one fish exited the drain near the end of the event, when the penstock was nearly empty and the velocity and volume of flow was minimal. Fish in the penstock appeared to maintain their position until dewatering was imminent, resulting in the transfer of live fish from Lake Piru into Piru Creek. Gobies collected from the penstock drain ranged from 25 mm TL to 90 mm TL.

The invasion of lower Piru Creek provides a direct, if intermittent, pathway for gobies to disperse to the Santa Clara River estuary. Typical flow releases from Santa Felicia Dam (0.20 m^3/s) to lower Piru Creek typically percolate subsurface at or near the confluence with the Santa Clara River 9.6 km downstream. The Santa Clara River at the confluence of Piru Creek is typically dry, and falls in the middle of an 11 km "dry gap" of the Santa Clara River, which is generally only wetted during significant storm events (Figure 2). Flow from Piru Creek to the Santa Clara River is contiguous when elevated flows are released from Santa Felicia Dam for groundwater recharge and diversion, steelhead migration, or when the dam is above storage capacity and spills. Between 2012 and 2015, only one high flow release occurred in Piru Creek, from 1 March to 7 March 2014. The high flow event consisted of a 5.66 m^3/s release for two days followed by a five day gradual ramp-down to baseflow of 0.20 m^3/s. This was the only flow event during this period where Shimofuri gobies had the opportunity to disperse from Piru Creek into the Santa Clara River. During this period, Castaic Lake (Figure 2) did not release water into Castaic Creek and was unlikely to be a source contributing gobies into the Santa Clara River.

United Water Conservation District typically operated a steelhead and Pacific lamprey (*Entosphenus tridentatus*) downstream migrant trap at the Freeman Diversion daily from January to June, 1994 to 2014, when streamflow was present. The diversion is located on the Santa Clara River approximately 43.5 km downstream from the base of Santa Felicia Dam (Figure 2). Shimofuri gobies were not detected at the Freeman Diversion prior to 2014. Due to extended drought and low flows in the Santa Clara River in 2014, the downstream migrant trap was only operated from 4 March to 30 April 2014, when a large storm increased river flow. On 5 March 2014, five Shimofuri gobies were collected in the downstream migrant trap. At a minimum, these fish traveled 43 km downstream in five days, assuming dispersal started at the beginning of the release on 1 March, a dispersal rate of approximately 8.6 km/day. Additional Shimofuri gobies were collected on 6 March (one individual), 8 (two individuals), and 11 (one individual), 2014, but were not observed for the remainder of the monitoring season.

This is the first known dispersal of this species within a natural river system in southern California. These observations indicate that a functional dispersal pathway exists to the Santa Clara River estuary 59 km below Santa Felicia Dam and 16 km below the Freeman Diversion (Figure 2), home to the federally threatened tidewater goby (*Eucyclogobius newberryi*). However, no Shimofuri gobies were collected during a survey of the estuary on 19 August, 2014 (Cardno-ENTRIX 2014) or observed in subsequent surveys between 2014 and 2016 (E. Bell and K. Jarrett, Stillwater Sciences, personal communication).

The invasion of alien gobies, including the Shimofuri goby, may negatively affect the native tidewater goby. Tidewater gobies are present in the Santa Clara River estuary, but the population appears to be in decline. Shimofuri gobies, which have established populations in the San Francisco Bay region (Moyle 2002), compete with and prey upon smaller tidewater gobies (R. Swenson, Environmental Science Associates (ESA), personal communication).

Initial experiments indicated that Shimofuri gobies aggressively intimidate, outcompete, and prey upon tidewater gobies in the laboratory (R. Swenson, ESA, personal communication). Tidewater goby and Shimofuri goby diets overlap, including benthic macroinvertebrates like oligochaetes, polychaetes, ostracods, copepods, and isopods (Swenson and McCray 1996), and may result in direct competition for food resources. To date, the possible effects of interactions in the wild between exotic goby species and tidewater gobies are largely conjectural (U.S. Fish and Wildlife Service 2005), and effects may be less than those observed in the laboratory because Shimofuri gobies prefer hard substrates, whereas tidewater gobies prefer sandy substrates. Further monitoring in the Santa Clara estuary may provide insight into real world interactions between these goby species and potential consequences for tidewater gobies in the future.

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Weight-length relationship and condition factor of leopard grouper *Mycteroperca rosacea* (Perciformes: Serranidae) from the Gulf of California

M.S. Irigoyen-Arredondo, E. Marín-Enríquez, X.G. Moreno-Sánchez*, L.A. Abitia-Cárdenas and J. S. Rámirez-Pérez

Instituto Politécnico Nacional (CICIMAR-IPN). Departamento de Pesquerías y Biología Marina. Av. IPN s/n, Col. Playa Palo de Santa Rita, La Paz, Baja California Sur. C.P. 23096, México. (MSIA, XGMS, LAAC)

Centro de Investigaciones Biológicas del Noroeste, La Paz, Baja California Sur, C.P. 23201, México. (EME)

Universidad Autónoma de Sinaloa- Facultad de Ciencias del Mar, Paseo Claussen S/N. Col. Los Pinos, Mazatlán, Sinaloa. C.P. 82000, México. (JSRP)

**Correspondent: xchel.moreno@gmail.com*

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The weight-length (*W-L*) relationship and condition factor (*K*) are descriptors that provide fundamental information to evaluate the condition of fish and to determine possible differences among populations (Froese 2006). Fulton´s condition factor has been widely used as a proxy for the health status of organisms, relating size and weight of recently caught individuals (Nash et al. 2006). These relationships (*W-L* and *K*) are biologically relevant and provide fundamental information for the management of populations with high ecological and economic importance.

The leopard grouper (*Mycteroperca rosacea)* is a serranid of importance within the Gulf of California (GC). This species is endemic to the northwestern Mexican Pacific (Thomson et al. 2000), and is included in the Red List of Threatened Species as vulnerable (Craig and Sadovy 2008). However, despite its ecological importance, there are no studies on the weight-length relationship and condition factor for this species, and this study was carried out to provide information on this topic.

Monthly sampling was carried out from March, 2014 to May, 2015 by free diving using a polespear at depths of 6 to 10 m in the mining port of Santa Rosalía, Baja California Sur, in the central region of the GC (27 °20.353 'N, 112 °15.797' W). Total length (*TL*, ±0.5 cm precision) was measured using an ichtyometer, and total fresh weight was measured using an Explorer Pro electronic analytical balance (*W*, ±1g precision). Sex was determined by the direct observation of gonads.

Data were analyzed by sex (male or female). No juveniles were caught; sizes were therefore categorized as small (<360 mm), medium (>360 mm and <510 mm), or large

(>510 mm). Seasonality was defined according to temperature records obtained during the study period from MODIS-AQUA satellite images with 1.1 km resolution. Anomalies were calculated from the annual average of 23ºC. The months with positive anomalies were assigned to the warm season (June, July, August, September, October and November, 2014) and months with negative anomalies were assigned to the cold season (March, April, May, December 2014 and January, February, March, April and May, 2015).

The weight-length relationship was calculated using the equation (Froese, 2006), where *W* is the fish´s total weight (g), *L* is total length (mm), *a* is the point of intersection with the y axis, and *b* is the slope of the curve. The degree of association of the *W-L* relationship was calculated using the coefficient of determination (r^2). The 95% confidence interval of parameters *a* and *b* was also estimated.

The condition factor was obtained from Fulton´s equation (Ricker, 1975): ,where *W* is total weight (g) and *L* is length (cm) cubed, multiplied by 100 to represent values as percentages. The program R (R Core Team, 2015) was used for statistical analyses. Wald´s test was applied to all combinations using the wald.test function from the analysis of overdispersed data (aod) library (Lesnoff and Lancelot 2012) to verify the isometric growth hypothesis (H_o: b=3, H_1: b≠3). A chi squared test was used (Sokal and Rohlf 1981) to test whether there was a 1:1 sexual proportion.

A total of 332 leopard grouper organisms was analyzed. The *W-L* parameters and their different combinations are presented in Table 1. Of all analyzed specimens, 210 were female (63.3%) and 122 were male (36.7%). The sexual proportion was 1.72F:1M, differing from the expected 1F:1M proportion ($X^2_{(1,332)} = 23.32$ $P < 0.05$). The *W-L* relationship fit a power function, and the coefficients of determination were high in all cases (r^2 = 0.90-0.99). Growth was allometric for most combinations ($b \neq 3$; $p<0.05$), except during the warm season ($b=3$; $p=0.20$). The categories of male, female, cold season, medium, and large-size fish had positive allometric growth (b>3), whereas small-size fish had negative allometric growth (b<3). The condition factor resulted in values over 1 (*K*<1) for all combinations (Table 1).

The sexual proportion obtained in the present study (1.72F:1M) was different from the expected 1F:1M proportion. This was reported for this species by Estrada-Godínez et al. (2011), who recorded an even higher proportion of females (3.6F:1M). It has also been reported for other members of the Serranidae family, for species such as red grouper (*Epinephelus morio)* (4.2F:1M), red hind (*E. guttatus)* (1.5F:1M), gag grouper (*Mycteroperca microlepis)* (3.3F:1M*)*, and tiger grouper *(M. tigris)* (1.3F:1M) (Renán et al. 2015). It has been noted that although the 1:1 sexual proportion is commonly found in nature, there can be great variability in a population, depending on environmental, physiological, genetic, and ethological factors, as well as on the effect of fisheries on commercially important species, due to size selectivity (Estrada-Godínez et al. 2011).

The *b* values obtained for most combinations (sex, season and size) were high (*b*>3), which is also similar to what has been reported for other species within the Serranidae family such as black grouper (*M. bonaci),* gag grouper (*M. microlepis)*, and red grouper (*Epinephelus morio)* (Renán et al. 2015); however, small specimens had *b* values <3. This difference could be due to differential growth rates, such that small-sized individuals grew more in size than in weight (negative allometric growth).

Growth is not constant and is indeterminate in fish during their first developmental stages (Froese 2006). It is therefore suggested that data should be worked independently to establish *W-L* relationships, separating sexes, sizes, and maturity stages, because these relationships could be under or overestimated and not result in real data. The type of growth

Table 1. Statistics on Length-weight (L-W) relationships and Fulton´s condition factor *K* for *Mycteroperca rosacea* from the mining port of Santa Rosalia, Baja California Sur, México sampled monthly from March, 2014 to May, 2015.

Categories	Length (mm)		Weight (g)	Parameters of *W-L*					Fulton´s condition factor (*K*)	
	n	Range (min-max)	Range (min-max)	a	b	a95%CL	b95%CL	R^2	Range (min-max)	Mean value
General	332	210-700	120-5554	$3.08x10^{-6}$	3.23	$1.94x10^{-6}$- $4.88x10^{-6}$	3.16-3.30	0.9543	0.94-2.17	1.27
Male	122	240-610	155-3205.2	$2.07x10^{-6}$	3.30	$8.57x10^{-7}$- $4.69x10^{-6}$	3.17-3.44	0.9508	0.96-2.17	1.27
Female	210	210-700	120-5554	$1.65x10^{-6}$	3.33	$9.01x10^{-7}$- $2.99x10^{-6}$	3.24-3.43	0.9595	0.94-1.97	1.26
Hermaphrodites	9	300-685	340-3775	$2.98x10^{-5}$	2.85	$7.75x10^{-6}$- $1.01x10^{-4}$	2.66-3.06	0.9953	1.14-1.48	1.27
Cold season	274	210-700	120-5554	$3.89x10^{-6}$	3.20	$2.35x10^{-6}$- $6.39x10^{-6}$	3.12-3.28	0.9585	0.94-2.17	1.29
Cold season *Male	97	235-610	155-3205	$2.92 x10^{-6}$	3.24	$1.06x10^{-6}$- $6.67x10^{-6}$	3.11-3.41	0.9541	0.94-2.17	1.26
Cold season * Female	138	210-700	120-5554	$1.88 x10^{-6}$	3.31	$1.05x10^{-6}$- $3.87x10^{-6}$	3.23-3.41	0.9652	0.94-2.17	1.27
Warm season	67	296-556	243-2582	$1.15x10^{-5}$	3.00	$4.51x10^{-6}$- $3.03x10^{-5}$	2.84-3.15	0.9603	1-1.66	1.21
Warm season *Male	25	275-590	290-2582	$2x10^{-7}$	3.67	$1.98x10^{-7}$- $2.01x10^{-7}$	3.43-3.90	0.9251	0.94-2.17	1.26
Warm season *Female	72	258-556	243-2060	$1.65 x10^{-5}$	2.94	$4.26x10^{-6}$- $3.61x10^{-5}$	2.81-3.16	0.9639	0.94-2.17	1.25
Small length	161	210-360	120-610	$1.80x10^{-4}$	2.53	$5.85x10^{-5}$ - $5.41x10^{-4}$	2.34-2.73	0.8441	0.94-2.17	1.26
Medium length	148	363-505	495-1890	$1.65x10^{-6}$	3.33	$4.15x10^{-7}$ - $6.95x10^{-6}$	3.10-3.57	0.8416	1-1.78	1.25
Big length	32	515-700	1403.1-5554	$1.63x10^{-6}$	3.33	$1.63X10^{-6}$ - $2.99X10^{-6}$	2.86-3.33	0.7862	0.96-1.78	1.35

n is sample size; min and max are the minimum and maximum length (mm) and weight (g); *a* and *b* are the parameters of the weight–length relationship, and $CL_{95\%}$ are 95% confidence limits; r^2 is the coefficient of determination of 332 analyzed specimens. *combinations by seasons per sex.

found in the present study for leopard grouper was positive allometric for most combinations, except for the warm season (isometric), and small sizes (negative allometric). This coincided with what was reported by Díaz-Uribe et al. (2001), who reported allometric growth (b = 2.6). According to the estimated age, most individuals caught by those authors were adult fish; they concluded that this is a slow-growing serranid. It should be noted, however, that the authors did not take into account sex or size when calculating the *W-L* relationship and their results should therefore be taken with caution.

According to Nash et al. (2006) environmental factors, feeding, and reproduction affect the condition of organisms. In this study the leopard grouper was found to be in good condition. This was more evident in males during the cold season (K=1.28). Big length individuals (K=1.35) were in better condition than small (K=1.26) and medium-sized individuals (K=1.25). This was attributed to the physiological processes through which individuals pass during their first stages of development. Low condition factor values in females during the warm season (K=1.25) coincide with what was reported by Estrada-Godínez et al. (2011), who stated that leopard groupers start reproducing during the warm season. The condition factor was therefore low because females use all their energy reserves for gonad maturation; once that process ends a good condition is re-established. This is the first study to provide data on the weight-length relationship and condition factor of leopard groupers in the Gulf of California.

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Spatiotemporal trends analysis of benthic communities and physical habitat during non-severe drought and severe-drought years in a residential creek in California

Lenwood W. Hall, Jr*, Ronald D. Anderson, and William D. Killen

University of Maryland, Wye Research and Education Center, P. O. Box 169 Queenstown, Maryland 21658, USA

* *Correspondent: lwhall@umd.edu*

Both temporal and spatial trends analysis was conducted to determine if the condition of the benthic community of a residential California stream represented by metrics measuring richness, composition, tolerance/intolerance and trophic composition has changed from 2006 to 2015. A secondary objective was to determine if 10 habitat metrics and total habitat scores measured concurrently with the benthic assessments have also changed temporally or spatially over the same 10 year time period. The 2006 to 2015 time period for this study included both non- severe drought (2006-2008), and severe drought (2013-2015), years. The most significant results from this study were that the conditions of the benthic communities as well as the physical habitat conditions have declined in this water body from 2006 to 2015. Both non-chemical (habitat and water quality parameters) and potential chemical stressors (metals and pyrethroids) were evaluated to determine their possible role in the decline in benthic community conditions. The factors most likely contributing to the decline in benthic communities during the 10 year study period are declining physical habitat conditions measured by declining metrics such as velocity/depth/diversity, channel flow status and riparian areas and possibly low dissolved oxygen concentrations below effects thresholds. Both declining physical habitat and low dissolved oxygen concentrations are likely related to the severe drought conditions reported during 2013-2015.

Key words: residential stream, benthic communities, physical habitat, drought, metals, pyrethroids

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Environmental stressors that may impact aquatic biota in freshwater streams include but are not limited to floods, fires, droughts, hurricanes, volcanic activity, climate change, land use change, introduction of exotic species, physical change of attributes such as temperature, substrate, or hydrology, and chemical changes such as pollution or nutrient enrichment. In order to determine the possible impact of these various environmental stressors on resident aquatic life, biological endpoints are often used (Karr and Chu 1999). Of the various biological indicators, benthic macroinvertebrates are often used to monitor the effects of both chemical and non-chemical stressors in the aquatic environment (Cairns and Pratt 1993). Rosenberg and Resh (1993), have reported the following advantages for using benthic macroinvertebrates in biomonitoring: (1) they are ubiquitous and can be affected by environmental perturbations in many types of aquatic systems; (2) they encompass a large number of different species and therefore offer a spectrum of responses to environmental stressors; (3) their basic sedentary nature allows for effective spatial scale analysis of pollutant or disturbance effects; (4) they have long life cycles compared to other taxonomic groups which allow for elucidation of temporal changes caused by perturbations; and (5) they act as continuous monitors of the water they inhabit thus enabling long term analysis of both regular and intermittent discharges, variable concentrations of single and multiple pollutants, and even synergistic or antagonistic effects. Benthic macroinvertebrate sampling can also be conducted using simple inexpensive equipment and the taxonomy of many groups is well known with keys to identification available (Hellawell 1986). In addition, many methods of data analysis, including biotic diversity indices have been developed and are widely used for benthic macroinvertebrates (Ohio Environmental Protection Agency 2015).

Bioassessments, formally defined as a quantitative survey of physical habitat and biological communities (benthic macroinvertebrates) of a water body to determine ecological condition have been used in California's Central Valley for a number of years (Bacey 2005, Brown and May 2004, Hall et al. 2009, Hall et al. 2013). One stream in California where bioassessments with benthic macroinvertebrates have been conducted since 2006, is Pleasant Grove Creek (Hall et al. 2015). Pleasant Grove Creek is a typical residential stream in California's Central Valley located in Roseville, California. This stream is on the California 303d list (impaired water body) based on the presence of pyrethroids, low dissolved oxygen and sediment toxicity (California Water Boards, 2010).

Bioassessment multiple stressor studies have been conducted in Pleasant Grove Creek spanning 10 years since 2006, with an extensive spatial scale (21 sites) given the size of the stream (Hall et al. 2015). Therefore, this database provides a unique opportunity to conduct benthic community trends analysis for key benthic metrics from data sets where both sampling and benthic identifications have been consistent. There was also the opportunity with this data set, spanning from 2006 to 2015, to determine the possible influence of the severe drought conditions that occurred during 2013 to 2015, on benthic macroinvertebrate communities (Howitt et al. 2015). Various investigators have reported that drought conditions can have adverse effects on resident benthic communities (Beche et al. 2009, Lake 2003). However, consistent long term studies that assess the impact of drought conditions on benthic communities along with consistent measurements of habitat, water quality parameters and potential chemical stressors (for example, metals and pyrethroids) have not been conducted.

The primary objective of this study was to determine if nine benthic metrics representing measures of richness, composition, tolerance/intolerance and trophic composition have changed temporally or spatially (increased, remained stable or decreased) from 2006 to 2015 (excluding 2009-2012), in Pleasant Grove Creek. The following benthic metrics were used in this analysis: percent dominant taxa; percent tolerant taxa; Shannon diversity index; EPT (*Ephemeroptera*, *Plecoptera*, and *Tricoptera* Index) taxa; percent collector/gatherers; percent collector/filterer; percent intolerant taxa; taxa richness; and the number of *Hyalella* collected. A secondary objective was to determine if 10 habitat metrics and total habitat scores measured concurrently with the benthic sampling have changed temporally or spatially over the 2006 to 2015, time period in Pleasant Grove Creek.

Materials and Methods

Study area.—A total of 21 sites was sampled in Pleasant Grove Creek and its tributaries (South Branch and Kaseberg Creek) in the spring of 2006, 2007, 2008, 2013, 2014 and 2015 (Figure 1). Sites are referenced as PGC 1 through PGC 22 (no site PGC 13) throughout the text. Sites that are located near storm drains are highlighted in Figure 1. Pleasant Grove Creek, located in Roseville, California, USA, is characterized by numerous contiguous subdivisions of single family homes less than 10 years old. There is no industry in the area and also sparse commercial development and agriculture. The distance from the upstream to downstream site was approximately 19 km in the mainstem of Pleasant Grove Creek. The distance from the upstream to downstream site in South Branch was approximately 8 km while the distance from the upstream to downstream site in Kaseberg Creek was approximately 8 km.

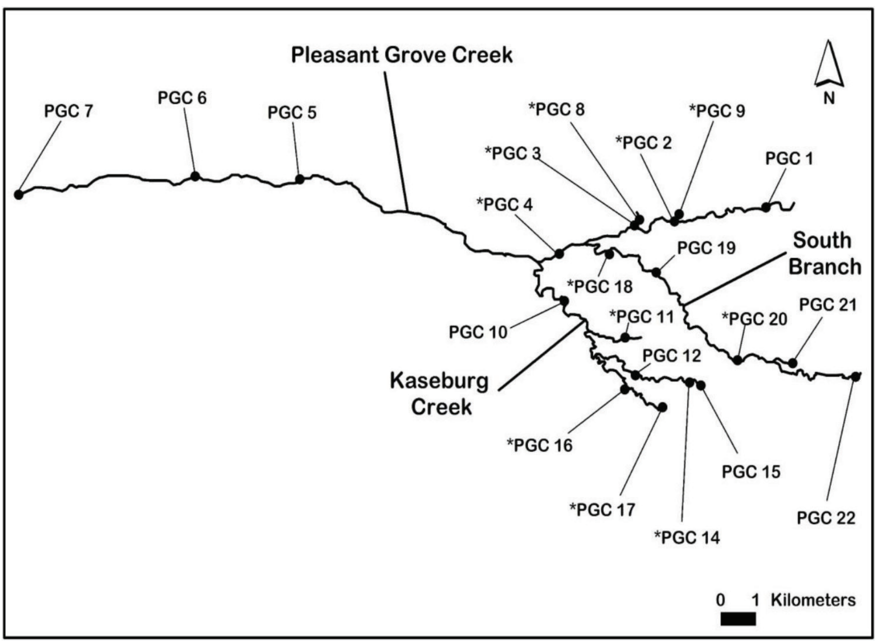

Figure 1.— Pleasant Grove Creek (PGC) core sites in California. Sites near storm drains are designated with an *.

Benthic macroinvertebrate sampling.—Benthic macroinvertebrates were collected in the spring of 2006, 2007, 2008, 2013, 2014 and 2015 from three replicate samples at all 21 sample sites. The sampling procedures were conducted in accordance with methods described in Harrington and Born (2000). Within each of these sample reaches, a riffle was located (if possible) for the collection of benthic macroinvertebrates. A tape measure was placed along the riffle and potential sampling transects were located at each meter interval of the tape. Using a random numbers table, three transects were randomly selected for sampling from among those available within the riffle. Benthic samples were taken using a standard D-net with 0.5 mm mesh starting with the most downstream portion of the riffle. A 30.5 x 61 cm section of the riffle immediately upstream of the net was disturbed to a depth of 10.2 to 15.2 cm to dislodge benthic macroinvertebrates for collection. Large rocks and woody debris were scrubbed and leaves were examined to dislodge organisms clinging to these substrates. Within each of the randomly chosen transects, three replicate samples were collected to reflect the structure and complexity of the habitat within the transect. If habitat complexity was lacking, samples were taken near the side margins and thalweg (deepest path) of the transect and the procedures described above were followed. All samples were preserved in 95% ethanol.

Due to the physical nature of this residential stream, it was often difficult to locate a substantial number of riffles to sample. Therefore, alternative sampling methods for non-riffle areas were used for all sites when needed as outlined in Harrington and Born (2000). This involved sampling the best available 30.5 x 61 cm sections of habitat throughout the reach using the same procedures described above. Nine 30.5 x 61 cm sections were randomly selected for sampling (i.e., stratified random sampling). Groups of three 30.5 x 61 cm sections were composited for each replicate for a total of three replicates per site.

Taxonomy of benthic macroinvertebrates and metric development.—The goal of the current study was to identify all benthic samples to the species level if possible. For taxa such as oligochaetes and chironomids, family and genus level, respectively, were often the lowest level of identification possible. Benthic macroinvertebrate subsampling (resulting in a maximum of 300 individuals) and identifications were conducted by the California Department of Fish and Wildlife (CDFW) in Rancho Cordova, California, USA. The benthic macroinvertebrate samples were subsampled and sorted by personnel at the CDFW Laboratory located at Chico State University, Chico, California, USA. Level 3 identifications (species level identifications) followed protocols outlined in Harrington and Born (2000). Slide preparations and mounting for species such as midges and oligochaetes followed protocols from the United States Geological Survey National Quality Control Laboratory described in Moulton et al. (2000).

Taxonomic information was used to develop benthic metrics. Benthic metrics for wadeable streams in California were developed by California Department of Fish and Wildlife (Harrington and Born 2000). Metrics were selected to represent different categories of ecological information (i.e., richness, composition, tolerances and trophic measures). The various metrics were selected to maximize the effectiveness of detecting degradation in concert with communicating meaningful ecological information. The following benthic metrics (along with the expected response to impairment) were used in the analysis: percent dominant taxa (increase); percent tolerant taxa (increase); Shannon Diversity (decrease); EPT taxa (decrease); percent collectors/gatherers (increase); percent collectors/filterers (increase); percent intolerant taxa (decrease); taxa richness (decrease); and number of

Hyalella (increase). The *Hyalella* metric is not a typical metric used in this type of analysis but was used in this study because it is a commonly used toxicity test species in California. *Hyalella* are considered tolerant of general environmental stressors with a tolerance value of 8 on a scale of 0-10, with 10 as the most tolerant (U. S. Environmental Protection Agency 1999). However, *Hyalella* are very sensitive to pyrethroids (Giddings and Wirtz 2012).

Physical habitat assessments.—Physical habitat was evaluated at each site concurrently with benthic collections. The physical habitat evaluation methods followed protocols described in Harrington and Born (2000). The physical habitat metrics used for this study were based on nationally standardized protocols described in Barbour et al. (1999). The following 10 continuous metrics scored on a scale of 0-20 (0=very poor to 20=optimal) were evaluated: epifaunal substrate; embeddedness; velocity/depth/diversity; sediment deposition; channel flow status; channel alteration; frequency of bends/riffles; bank stability; vegetation protection; and riparian zone; and given a total score (maximum score=200).

Statistical analysis.—In advance of the statistical trends analysis, it was determined whether assumptions of normality and equal variance were met with the benthic and habitat data sets. If assumptions of normality and equal variance were met, as was the case for most of the data, regression analysis (a parametric test) was used to determine both temporal and spatial trends as recommended by other investigators (Hirsch et al. 1982). Trends were considered statistically significant if p values were less than 0.10 and r^2 values were greater than 0.25 (Hall and Anderson 2012). If assumptions of normality and equal variance were not met then a Spearman rank order correlation (a non-parametric test) was used to determine significant trends.

Results

Overview of the six year benthic community dataset.—The number of different benthic taxa collected by year in Pleasant Grove Creek were: 2006 (142); 2007 (145); 2008 (153); 2013 (153); 2014 (143) and 2015 (145). These results show that the range of number of different benthic taxa collected for six years of sampling (142 to 153) was similar. A total of 273 different benthic taxa were collected over the six year sampling period.

The number of individual benthic taxa collected by year were: 2006 (18,334); 2007 (17,994); 2008 (21,291); 2013 (15,993); 2014 (17,550) and 2015 (18,116). The number of individual benthic taxa collected by year ranged from 15,993 in 2013 during the severe drought period to 21,291 in 2008 during the non-severe drought period. In summary, a total of 109,278 individual benthic taxa were collected during the six years of sampling in Pleasant Grove Creek.

The five most dominant benthic taxa collected during six years of sampling in Pleasant Grove Creek and percent of the total samples were: immature tubificidae (oligochaetes)–9.3%; *Physa* (snails)–9.2%; *Hyalella* (amphipods)–7.2%; *Paratanytarsus* (chironomids)–6.5% and *Dugesia tigrina* (flatworms)–4.9%. All of these taxa are considered tolerant to moderately tolerant of environmental stressors (Harrington and Born 2000).

Temporal analysis of benthic metrics.—A temporal trends analysis of selected benthic metrics using standard linear regression showed a statistically significant increase in percent tolerant taxa from 2006 to 2015 (Table 1; Figure 2b). Since percent tolerant taxa increase in stressed environments, this significant increase in percent tolerant taxa suggests an increase in impairment in this stream over time. A statistically significant decline in EPT taxa–taxa that are sensitive to stress–was also observed in Table 1, and Figure 2d. This result would also suggest an increase in impairment. There were no statistically significant changes for the other seven benthic metrics presented in Table 1. However, increasing slopes for stress tolerant metrics such as percent dominant taxa and number of *Hyalella* and declining slopes for stress sensitive metrics such as Shannon Diversity Index and taxa richness would suggest a decline in benthic community condition.

Table 1.—Linear regression of Pleasant Grove Creek benthic metrics trends for 2006, 2007, 2008, 2013, 2014 and 2015. Significant trends are in bold.

Benthic Metric	Linear Regression			Significant Trend Over Time?[a]
	r^2	P	Slope	
% Dominant Taxa	0.028	0.752	Increase	No
% Tolerant Taxa	**0.915**	**0.003**	**Increase**	**Yes**
Shannon Diversity Index	0.267	0.294	Decline	No
EPT Taxa	**0.615**	**0.065**	**Decline**	**Yes**
% Collector/Gatherers	0.177	0.406	Decline	No
% Collector/Filterers	<0.001	0.990	Decline	No
% Intolerant Taxa	0.157	0.437	Increase	No
Taxa Richness	0.412	0.169	Decline	No
# of Hyalella	0.156	0.439	Increase	No

[a] Statistically significant at $p < 0.10$.

Spatial analysis of benthic metrics.—Spatial analysis of benthic metrics generally confirmed the results from the temporal analysis presented above (Figure 3). For example, a greater number of sites were reported to have a significant increase in stress-tolerant metrics such as percent dominant taxa, percent tolerant taxa and percent collector-filterers. The exceptions were the stress tolerant metrics number of *Hyalella* and percent collectors-gatherers where a greater number of sites did not show increases for these metrics. In addition, stress sensitive metrics such as EPT taxa and taxa richness were also reported to significantly decrease at a higher number of sites as illustrated in Figure 3. The above results from spatial analysis suggest that benthic communities have declined in Pleasant Grove Creek during the 2006 to 2015 time period.

Temporal analysis of habitat metrics.—Temporal trends analysis of individual habitat metrics and total score in Table 2, and Figure 4, showed a significant decline from 2006 to 2015, for velocity depth diversity, channel flow status and riparian zone. A decline in these three metrics would demonstrate a decrease in habitat quality in Pleasant Grove Creek for the 10 year time span. It is also noteworthy that declining slopes (although not statistically significant) were also reported for epifaunal substrate, frequency of bends/riffles, bank stabilization, and total habitat scores. The declining slopes for these habitat

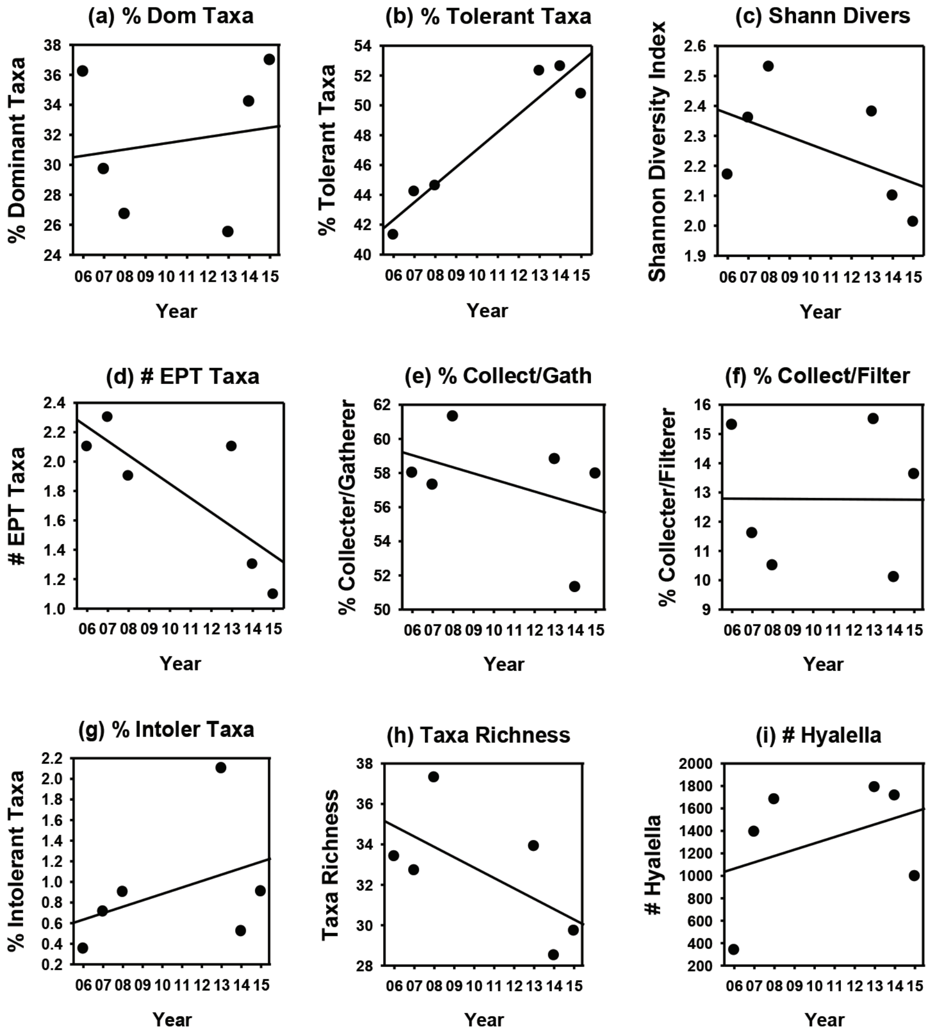

Figure 2.— Linear regression plots of nine Pleasant Grove Creek benthic metric trends for 2006, 2007, 2008, 2013, 2014 and 2015.

metrics and total score would also suggest a deterioration of physical habitat in this stream over time. Based on the above data, it appears that the overall habitat, that is consistently marginal based on annual sampling, has demonstrated a decline during the 2006 to 2015, time period in Pleasant Grove Creek.

Spatial analysis of habitat metrics.—Spatial analysis of habitat metrics and total score provided additional support for the declining habitat conditions in this stream as reported above based on temporal analysis (Figure 5). A greater number of sites showed a significant decrease for the following metrics: epifaunal substrate; velocity/depth/diversity; channel flow status; frequency bends/riffles; bank stability; and riparian zone. Spatial analysis of total habitat scores in Figure 5, also showed that five sites (PGC3, PGC9, PGC11, PGC14, and PGC21) had significant declining total habitat scores. In contrast, only one site (PGC10) had an increase in total habitat scores over the 2006 to 2015, time period.

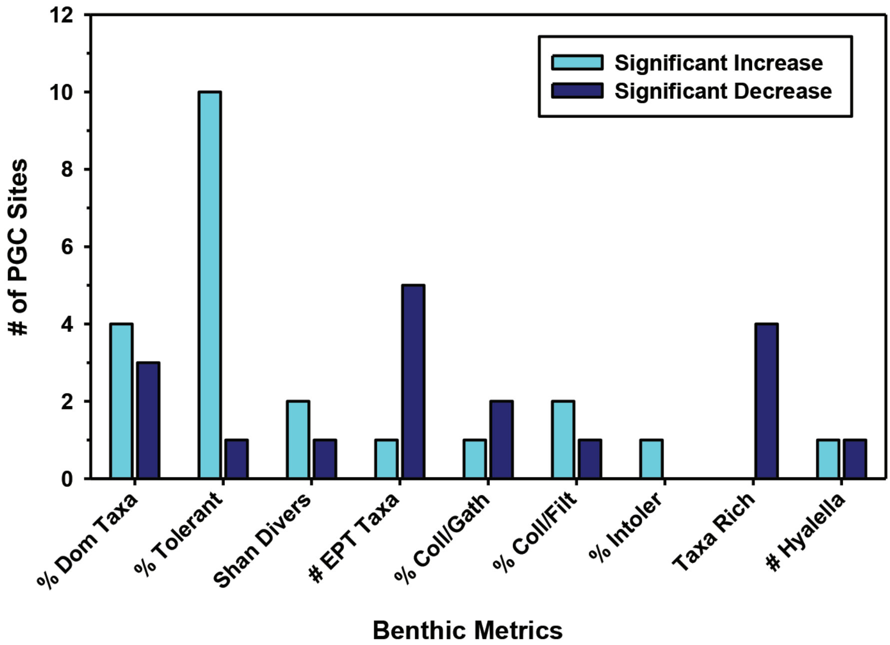

Figure 3.— Number of Pleasant Grove Creek sites with significant increasing or decreasing temporal trends for the various benthic metrics from 2006-2015.

Table 2.—Pleasant Grove Creek habitat metric trends data for 2006, 2007, 2008, 2013, 2014 and 2015. Significant trends are in bold. No data transformations were necessary to improve the overall fit of the data for statistical analyses.

Habitat Metric	Linear Regression			Significant Trend Over
	r^2	P	Slope	Time?[a]
Epifaunal Substrate	0.169	0.417	Decline	No
Embeddedness	0.016	0.808	Increase	No
Velocity Depth Diversity	**0.584**	**0.077**	**Decline**	**Yes**
Sediment Deposition	0.084	0.577	Increase	No
Channel Flow Status	**0.639**	**0.056**	**Decline**	**Yes**
Channel Alteration	0.066	0.622	Increase	No
Frequency of Bends/Riffles	0.503	0.115	Decline	No
Bank Stabilization	0.179	0.404	Decline	No
Vegetative Protection	0.187	0.391	Increase	No
Riparian Zone	**0.755**	**0.025**	**Decline**	**Yes**
Total Score	0.420	0.164	Decline	No

[a]Statistically significant at $p = 0.10$.

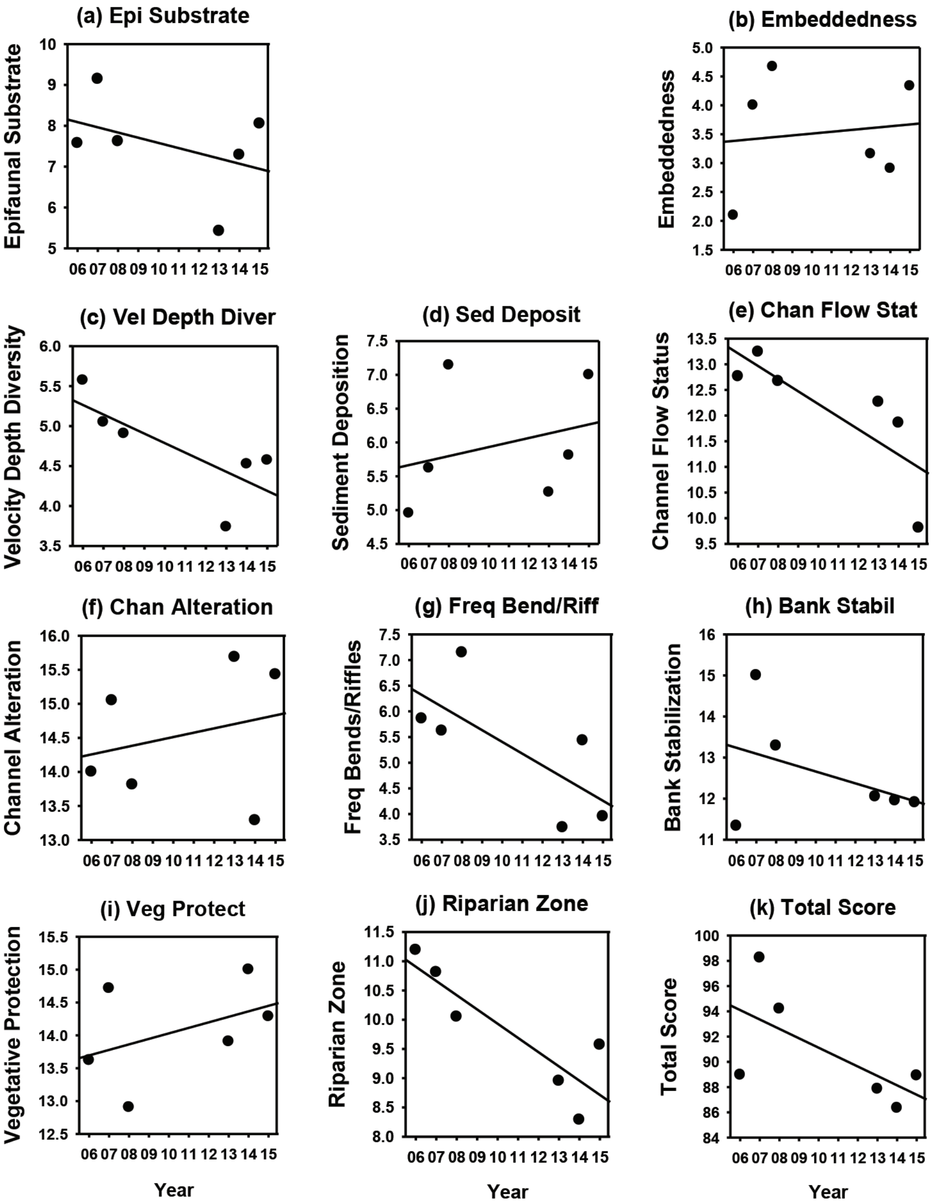

Figure 4.— Linear regression plots of 11 Pleasant Grove Creek habitat metrics trends for 2006, 2007, 2008, 2013, 2014 and 2015.

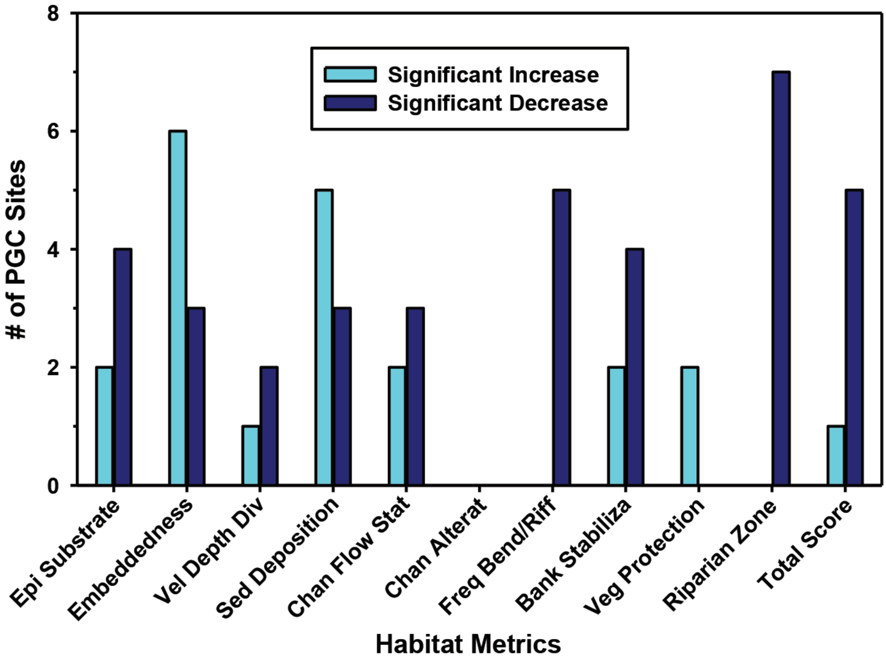

Figure 5.— Number of Pleasant Grove Creek sites with significant increasing or decreasing temporal trends for the various habitat metrics from 2006 -2015.

Discussion

A significant finding from this study is that benthic community conditions have declined in Pleasant Grove Creek from 2006 to 2015. Lotic aquatic systems, such as Pleasant Grove Creek, are very complex and are subject to multiple stressors, so it is often difficult to determine factors responsible for the decline in the condition of benthic communities. Various possible factors that may be responsible for the decline in benthic community condition in Pleasant Grove Creek are physical habitat decline, severe drought (also correlated with habitat decline), water quality stressors (e.g., reduced dissolved oxygen), and chemical contaminants (metals and pyrethroids). The possible role of each factor as a contributor to benthic community decline in Pleasant Grove Creek is discussed below.

Impaired physical habitat (including sediment loading) has been identified as a major stressor to aquatic life in California streams (Anderson et al. 2003; Hall et al. 2007). Altered physical habitat structure is also considered one of the major stressors of aquatic systems throughout the United States resulting in extinctions, local expirations and population reductions of aquatic fauna (Karr et al. 1986, Rankin 1995). Identifying degraded physical habitat in streams is particularly critical for biological monitoring as failure to do so can sometimes hinder investigations on the effects of toxic chemicals or other water quality related stressors. Rankin (1995) has reported that there is a small but still significant risk of reporting a water quality related impact when one does not exist (i.e., a false positive) when habitat assessments are insufficient or absent. Physical habitat evaluations are not intended to replace biological assessments but rather to add an additional line of evidence about the status of lotic systems when conducted in concert with biological assessments.

The physical habitat metric temporal trends reported in the current study shows that habitat has declined in Pleasant Grove Creek from 2006 to 2015, based on a significant decline in key metrics such as velocity/depth/diversity, channel flow status, and riparian zone. The decline in physical habitat occurs concurrently with the decline in benthic community condition during this 10 year time period. Previous bioassessment multiple stressor studies in Pleasant Grove Creek spanning 2006 to 2015, have shown that physical habitat, and not chemical stressors such as metals or pyrethroids, have a stronger correlation with benthic community metrics (Hall et al. 2015). Therefore, the results from historical data analysis of the bioassessment multiple stressor studies and the declining habitat trends data in the current study would support the decline in habitat conditions as a likely reason for the decline in benthic community condition in Pleasant Grove Creek from 2006 to 2015.

Severe drought conditions during 2013 to 2015, as reported by Howitt et al. 2015, is suspected in reducing stormwater runoff, overall stream flow and wetted stream bed area as habitat for benthic communities in Pleasant Grove Creek when compared with the earlier years of sampling during non-severe drought years in 2006-2008. The best evidence to support this severe drought period (2013-2015) vs the non-severe drought period (2006-2008) would be direct documentation of continuous flow from gauging stations in Pleasant Grove Creek during these two three year windows of time. Unfortunately, continuous flow data were not available for Pleasant Grove Creek for all years of sampling to demonstrate the extremely dry conditions during 2013- 2015. However, flow data were available for another waterbody in the general area (Arcade Creek) that can serve as a suitable surrogate (U.S. Geological Survey 2016). A t-test comparison of Arcade Creek mean flow data for 2006-2008, with 2013-2015, showed significantly higher flow five days prior to Pleasant Grove Creek sampling in 2006-2008 (mean discharge of approximately 2.8 ft^3 /sec) when compared with 2013-2015 (mean discharge of approximately 0.2 ft^3/sec). Therefore, the available data would support lower flow and reduced wetted stream area in Pleasant Grove Creek in 2013-2015, when compared with 2006-2008.

There are a number of studies that have been designed to determine the influence of drought conditions on benthic macroinvertebrates (Suren and Jowett 2006, Miller and Golladay 1996, Rose et al., 2008, Boulton 2003, Acuna et al. 2005, Chessman 2015, Love et al. 2008, Beche et al. 2009, Lake 2003, Bogan and Lytle 2011, Sponseller et al. 2010, Stanley et al. 1994, Wood and Petts 1999, McElravy et al. 1989, Resh 1992, and Beche and Resh 2007, among others.) For example, Rose et al. (2008) reported that stream assessments during drought only give an indication of river health at sites where water still persists, which may only be a small proportion of the remaining stream network. Stanley et al. 1994 have reported that the greatest changes in macroinvertebrate composition occur during drought when sites become isolated from upstream reaches. Beche et al. (2009), have reported that droughts have equivocal effects on the abundance and richness of invertebrate communities in two California streams as drought may reduce habitat suitability, particularly in already water stressed temporary habitats. Other investigators have also reported that severe drought periods greater than one year are more likely to result in persistent habitat changes, reduce the sources of nearby benthic colonists characteristic of higher flows, and to cause some local benthic populations to disappear due to lack of suitable habitat (Lake 2003).

The severe drought conditions experienced from 2013 to 2015, in the current study is likely related to the significant decline in the velocity/depth/diversity metric and channel flow status metric as previously discussed. Both of these metrics are important for benthic communities and are dependent on flow conditions. Although continuous flow data were not available for Pleasant Grove Creek for all years of sampling as previously discussed, individual flow measurements conducted at each of the 21 sites during the six year period showed that flow could only be measured at less than half the sites during 2013-2015 (presence of pools or dry stream bed), in contrast to 2006-2008, where flow was measured at more than half the sites. Velocity/depth/diversity is a measure of various velocity depth regimes present (slow/deep; slow/ shallow; fast/deep; and fast/shallow) so lower flow conditions experienced during a severe drought period could cause a reduction in this metric. The channel flow status metric is a measure of the percent of the channel that is filled by water so this metric is also highly dependent on the flow and the amount of water in the stream. Since the drought conditions are likely responsible for the decline in these two habitat metrics and these metrics are important for the condition of benthic communities, it is logical to assume that the severe drought conditions contributed to the decline in resident benthic community condition.

Standard water quality measurements of temperature, dissolved oxygen, pH, conductivity, salinity and turbidity were conducted at each sample site during each year of the six year sampling effort. The one water quality parameter that does appear to decline to stressful conditions during the six year study is dissolved oxygen. Oxygen availability in aquatic environments is widely recognized as a factor influencing the composition of freshwater benthic communities because it critically affects the distribution of many aquatic species (Hynes 1960). Lee and Lee (2002), have reported a toxic threshold of 5.0 mg/L for dissolved oxygen for aquatic species. In the current study, annual mean dissolved oxygen concentrations were greater than 6 mg/L for all years except 2015. For 2015, the annual mean dissolved oxygen concentration was 4.2 mg/L and dissolved oxygen values less than 5.0 mg/L were reported at 12 of 21 sites, thus suggesting stressful low dissolved oxygen conditions for benthic communities. Rose et al. 2008 have reported that reduced dissolved oxygen concentrations are a consistent response to drought conditions in water bodies. Therefore, the low and potentially stressful dissolved oxygen concentrations reported during one of the three severe drought years (2015) may have adversely impacted resident benthic communities.

In addition to the non-chemical stressors discussed above, both metals and pyrethroids are also potential chemical stressors to resident benthic communities that have been consistently measured in sediment at all sample sites during the six years of bioassessment sampling in Pleasant Grove Creek (Hall et al. 2014, Hall et al. 2015). Temporal trends analysis of the Pleasant Grove Creek metals data (arsenic, cadmium, chromium, copper, lead, mercury, nickel and zinc) from 2006 to 2015, showed no statistically significant change in concentrations for the eight metals and total metals (Hall et al. 2015). Therefore, the metals trends data demonstrating no significant temporal change over the 10 year time span do not suggest a relationship with metals and declining benthic community condition.

The following pyrethroids have also been measured concurrently in sediment with benthic communities in this waterbody: bifenthrin, cypermethrin, cyfluthrin, deltamethrin, esfenvalerate, fenpropathrin, lambda-cyhalothrin, and permethrin (Hall et al. 2014, Hall et al. 2015). Temporal trends analysis of these eight pyrethroids in sediment showed a significant decline in concentrations for six of the eight pyrethroids measured from 2006 to 2015 (Hall et al. 2016). The pyrethroids trends data, showing a decline over time, would suggest that there is no correlation with benthic community decline because if pyrethroids were a key stressor one would expect an increase in benthic community condition with a decline in pyrethroids. However, this was not the case. A summary of six years of bioassessment multiple stressor data in Pleasant Grove Creek also indicated that pyrethroids were not a significant stressor to benthic community metrics when evaluated using multivariate analysis that included both habitat metrics and metals (Hall et al. 2015). Therefore, the available data demonstrate that there is no relationship between pyrethroid concentrations in Pleasant Grove Creek and the decline in benthic community condition.

In summary, the likely factors contributing to the decline in benthic community conditions in Pleasant Grove Creek from 2006 to 2015, are declining physical habitat conditions and low dissolved oxygen concentrations. Both of these factors are likely associated with the severe drought conditions present during 2013 to 2015. This result is not surprising as numerous studies addressing the impact of drought on benthic communities in freshwater lotic systems previously discussed above provide support for this finding. Continued benthic sampling in Pleasant Grove Creek in 2016, with the predicted El Nino (Northern California Water Association Newsletter 2015) and associated predicted intense rainfall could provide insight on the possible impact of drought conditions on benthic communities. If increased rainfall resulted in increased stream flow and wetted stream area, then continued benthic sampling would provide useful information to determine if benthic communities can recover to pre-drought conditions in Pleasant Grove Creek. However, it may take more than one year of non-drought conditions for benthic communities to recover in this water body.

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