Zooplankton Monitoring 2019

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Introduction

Zooplankton are a vital trophic link between aquatic primary producers and higher-level consumers of the San Francisco Estuary (SFE). As primary consumers of phytoplankton, zooplankton facilitate the flow of carbon into a large and complex food web, historically supporting abundant fisheries in the region (Schroeter et al. 2015; Kimmerer et al. 2018). Many fishes, including Striped Bass (*Morone saxatilis*) and Chinook Salmon (*Oncorhynchus tshawytscha*) feed on zooplankton while rearing in the estuary as larvae and juveniles (Goertler et al. 2018; Heubach et al. 1963), while others like Tule Perch (*Hysterocarpus traski*) and Prickly Sculpin (*Cottus asper*) feed on zooplankton throughout their lifetimes (Kimmerer 2006Feyrer et al. 2003)). Zooplankton in the SFE are also a key food source for several endangered and threatened species, notably the Delta Smelt (*Hypomesus transpacificus*) and Longfin Smelt (*Spirinchus thaleichthys*) (Hobbs et al. 2006; Slater and Baxter 2014).

This importance of zooplankton prompted the implementation of the Zooplankton Study in 1972 to assess fish food resources in the upper SFE. Mandated by the State Water Resources Control Board's Water Right Decision D-1641, the study is conducted jointly by the California Department of Fish and Wildlife and the California Department of Water Resources under the guidance and management of the Interagency Ecological Program. For nearly 5 decades, this study has monitored the zooplankton community in the region, tracking abundance trends and distributional patterns, detecting and monitoring introduced species, and documenting the dramatic shifts in the community's composition. Changes in zooplankton abundance and composition have since been linked to major declines of the pelagic fishes in the upper estuary (Sommer et al. 2007; Winder and Jassby 2011). This report presents zooplankton annual and seasonal abundance indices and distribution trends from 1974 through 2019 for the most common copepods, cladocera, rotifers, and mysids of the upper estuary.

Methods

Zooplankton sampling has been conducted since 1974 at least once a month March-November through 1995, monthly from 1995 on. at 20 fixed stations in the upper SFE (Figure 1). Three gear types are used for each sampling event: a pump with a 43-micron mesh net for micro-zooplankton (rotifers, nauplii, and small cyclopoid copepods); a Clarke-Bumpus (CB) net with a 160-micron mesh for sampling meso-zooplankton (cladocera and most juvenile and adult calanoid copepods); and a mysid net with a 505-micron mesh for sampling mysid shrimp and other macro-zooplankton. Both the mysid and CB nets are attached to a sled and towed obliquely from the bottom through the surface for a 10-minute tow. Volume is measured using a General Oceanics flowmeter placed in the mouth of each net so that: $V = (end\ meter - start\ meter) * k * a$; where *V* is the volume of water sampled, *k* is a flowmeter correction value, and *a* is the area of the mouth of the net. The Teel Marine 12V utility pump is also used at each station to sample approximately 19.8 gallons from the entire water column, which is filtered through a 43-micron mesh net to concentrate the pump sample. Samples are preserved in 10% formalin with Rose Bengal dye before being processed in the laboratory for identification and enumeration of organisms using a dissecting microscope. More information about the sampling and processing methods can be found in the metadata at ftp://ftp.wildlife.ca.gov/IEP_Zooplankton/.

Abundance indices are calculated for each organism based on the gear type most effective at its capture and reported as the mean catch-per-unit-effort (CPUE). CPUE is calculated as the number of each organism collected per cubic meter of water sampled, so that: $CPUE = s * V^{-1}$; where s is the estimated count of the target organism in the sample. Copepod abundance indices reported here only include adults, as juveniles were not always identified to species. Annual and seasonal abundance indices were calculated using 14 fixed stations sampled consistently since 1974 (Figure 1) and 2 non-fixed stations sampled where bottom specific conductance was roughly 2 and 6 millisiemens per centimeter (approximately 1 and 3 psu).

To analyze long-term trends (1974 to present), annual abundance indices were calculated as the mean CPUE for samples collected from March through November, as winter sampling was inconsistent before 1995. Seasonal abundance indices were calculated as the mean CPUE for samples collected during each season: winter (previous December to February), spring (March to May), summer (June to August), and fall (September to November). Long-term seasonal trends for winter are only shown for 1995 to present. Spatial distribution indices for organisms is described as seasonal mean CPUE for by region. Estuary regions were defined as San Pablo Bay (stations D41 and D41A), Suisun Bay (stations D6, 28, 54, and 48), Suisun Marsh (stations 32 and S42), West Delta (stations 60, 64, and 74), Central Delta (stations D16, 86, and D28), and the East Delta (92 and M10).

Results and Discussion

Since the implementation of the Zooplankton Study in 1974, a significant decrease in the overall abundance of zooplankton has been detected in the estuary (Figure 2). Only the abundance of cyclopoid copepods increased in the region during this period, driven by the invasion and spread of *Limnoithona tetraspina*. The overall decrease in zooplankton abundance in the estuary can be attributed to a series of invasions into the estuary, most notably that of the Asian clam *Potamocorbula amurensis* in the mid-1980s (Kimmerer, Gartside, and Orsi 1994; Carlton et al. 1990). The spread of *P. amurensis* throughout SFE has had significant impacts on planktonic abundance in the upper estuary due to its high filtration feeding rates on phytoplankton and copepod nauplii. Not only has abundance decreased for most of the zooplankton groups, but dramatic shifts in the composition of these communities have been detected during the study period. These changes have been driven by the introduction and spread of non-native zooplankton species throughout the estuary, compounded with changes to the abiotic and biotic environments.

Calanoid copepods

While overall calanoid copepod abundance has declined slightly over the study period, community composition has shifted dramatically (Figure 2A). When the study began in the early 1970s the copepods *Eurytemora affinis* and *Acartia* spp. dominated the calanoid community. The non-native *E. affinis* was once the primary prey item of the endangered Delta Smelt, but its abundance has declined to a fraction of what it once was, forcing fish to prey switch to more recently introduced calanoids like *Pseudodiaptomus forbesi* (Moyle et al. 1992; Slater and Baxter 2014).

One of the first recorded introduced calanoid copepods was *Sinocalanus doerrii*, a freshwater species native to China that invaded the estuary in 1978 and became the most dominant calanoid species in the estuary for a decade (Orsi et al. 1983). Then in 1987, after the invasion of *P. amurensis*, the calanoid *Pseudodiaptomus forbesi* was introduced and first detected in the region, competing with the abundant *E. affinis* (Orsi and Walter 1991). *P. forbesi* quickly became the numerically dominant calanoid in the upper estuary and remains the most abundant to this day. Another introduction occurred in 1993, when the predatory calanoid copepod *Acartiella sinensis* quickly became the second most abundant calanoid in the upper estuary, dominating the low-salinity zone in Suisun and the West Delta (Orsi and Ohtsuka 1999). This invasion was hypothesized to have narrowed the range of *P. forbesi* towards the freshwater zone of the estuary, as predation on *P. forbesi* nauplii by *A. sinensis* has been recorded (Kayfetz and Kimmerer 2017).

In general, calanoid copepod abundance is highest in the estuary during the summer and fall months, with lowest abundance during the winter (Figure 3A). While calanoid copepod abundance peaked in the summer of 2017 at a nearly 20 year high, 2018 and 2019 abundance returned to levels comparable to the previous two decades. In 2019 the distribution of calanoids throughout the estuary was similar to 2018, with *P. forbesi* the most abundant in summer and fall in most of the upper estuary (Figure 4A). The predatory *Acartiella sinensis* was seen in highest densities in the summer and fall mostly in the Suisun Bay and West Delta regions, similar to 2018. In fall 2019, *A. sinensis* was the most abundant calanoid in Suisun, where it co-occurred with high densities of one of its primary prey items *Limnoithona tetraspina* (Figure 4B), while *P. forbesi* was found to the east of Suisun Bay (Hennessy 2018). *Acartia* spp*.* was the only native calanoid copepod commonly found in 2019, but it was restricted to Suisun Bay and San Pablo Bay in the lower reaches of the estuary. *Eurytemora affinis* was the most dominant calanoid throughout the upper estuary in the spring of 2019, similar to seasonal distribution patterns of the last two decades.

Cyclopoid copepods

While overall zooplankton abundance has declined over the study period, the abundance of cyclopoid copepods exploded (Figure 2B). The native *Oithona* spp*.* and *Acanthocyclops* copepods were at low abundances when the study began, but with the introduction of *Limnoithona sinensis* in the early 1980s, and the later identification of the invasive *Limnoithona tetraspina* in 1993, cyclopoid indices have increased dramatically(Ferrari and Orsi 1984; Orsi and Ohtsuka 1999) Abundance indices for the two species of *Limnoithona* were reported together from 1980 through 2006 as *Limnoithona* spp., then separately since 2007 when they were identified and enumerated as *L. sinensis* and *L. tetraspina*. In 2019, *L. tetraspina* abundance was the highest observed for all copepods (Figures 2A and 2B).

Much smaller than calanoid copepods collected in the CB net, the *Limnoithona* cyclopoids are best retained in pump samples, which use a smaller mesh. Since the early 1990s, *Limnoithona* spp*.* abundance has been higher than calanoid copepod abundance, and the small *L. tetraspina* has become the most common copepod in the upper estuary. This increase in *L. tetraspina* abundance is likely due to a decline of Northern Anchovy in the upper SFE and subsequent decreased predation (Kimmerer 2006), as well as the cyclopoid’s small size, high growth rate, and motionless behavior, making it very difficult for visual feeders to capture (Bouley and Kimmerer 2006; Greene et al. 2011). These characteristics may increase its ability to escape predation in a region where visual predation is most dominant among fish (Kimmerer 2006). The introduction of *L. tetraspina* is also linked to the reduction of the range of *P. forbesi* out of the low-salinity zone of the estuary, as high *L. tetraspina* densities may have fed and sustained larger populations of the predatory *A. sinensis*, which also fed on *P. forbesi* nauplii (Kayfetz and Kimmerer 2017).

Seasonally, *Limnoithona tetraspina* peaks in summer and fall (Figure 3B), with lower abundance in winter and spring. As in prior years, this cyclopoid was most abundant in the low-salinity zone of the estuary in Suisun Bay and the West Delta (Figure 4B). *Oithona davisae*, a native cyclopoid, was the most abundant cyclopoid in the higher-salinity San Pablo Bay in summer and fall (Figure 4B).

Cladocera

The cladoceran community of the upper estuary is composed of *Bosmina*, *Daphnia*, *Ceriodaphnia*, and *Diaphanosoma* species, whose populations have also substantially declined since the onset of the study (Figure 2C). These cladocera tend to be herbivorous, feeding primarily on phytoplankton, and were likely hard hit by the invasion of *P. amurensis* (Baxter et al. 2008; Kratina and Winder 2015). Cladocerans make up a significant portion in the diets of Delta Smelt, juvenile Chinook Salmon, and young-of-the-year Striped Bass throughout the upper estuary (Heubach et al. 1963; Slater and Baxter 2014; Goertler et al. 2018). The invasion and increase of available copepod prey such as *P. forbesi* and the decline in cladocera has created a shift in the nutritional content of the plankton community available for fish, with yet to be determined consequences (Kratina and Winder 2015).

While Cladocera abundance has declined overall, recent years summer abundance has been increased and in 2018, summer cladocera abundance was the highest observed since the *P. amurensis* invasion (Figure 3C). However, in 2019, abundances dropped back down to the lowest seen in almost a decade, with summer abundance slightly higher than other seasons. In the high outflow year of 2017 some cladocera, namely *Bosmina*, were found downstream in Suisun and the West Delta, while in 2019 the highest densities of cladocera were found in the East Delta, with trace concentrations found in other regions of the estuary (Figure 4C).

Rotifer

While they are the most abundant zooplankton in the estuary, long-term sampling of rotifers using the pump system shows a dramatic decrease in their annual abundance in the estuary since the beginning of this study (Figure 2D). Several species of rotifers make up the community: most abundant being the *Polyarthra*, *Synchaeta*, and *Keratella* genera. Interestingly, the decline of rotifer abundance beginning in the late 1970s preceded the invasion of *P. amurensis* in the estuary (Cloern and Jassby 2012).

The distribution and abundance of rotifers were similar between 2018 and 2019, with abundances similar to those found since the invasion of *P. amurensis*. Rotifers were the most abundant zooplankton sampled during 2019 (Figure 2) and were found throughout the estuary (Figure 4D). Overall rotifer abundance peaked in the summer and spring (Figure 3D). *Keratella* and *Polyarthra* tend to be most abundant in the freshwater and low-salinity zone of the estuary, while *Synchaeta* species occur most in the higher-salinity areas of San Pablo Bay and Suisun (Figures 3D and 4D)(Winder and Jassby 2011). A spatial and temporal split was discernable between *Synchaeta* and other rotifers, with *Synchaeta* having highest densities in the low-salinity zone Bay during the spring, and other rotifers being most abundant in the East Delta in summer.

Mysida

Not only have mysid abundances declined significantly since the 1970s, but the community has also shifted from being composed almost entirely by the native *Neomysis mercedis*, to being dominated by the non-native *Hyperacanthomysis longirostris* (formerly *Acanthomysis bowmani*) (Figure 2E). The first significant decline in *N. mercedis* occurred during the 1976-1977 drought, likely caused by food limitation from an absence of diatoms due to very low river discharges (Siegfried et al. 1979; Cloern et al. 1983). The populations of *N. mercedis* were able to rebound after the years of drought and stayed at high densities in the Suisun Bay region of the upper estuary until the introduction of *P. amurensis* in the mid-1980s, after which their numbers crashed.

In 1993 the introduced *H. longirostris* was first detected by this study, shortly after the decline of *N. mercedis*, and it quickly became the most common mysid in the system. However, overall mysid abundances have not returned to their pre-clam invasion levels (Modlin and Orsi 1997, Figure 2E). Mysids have always peaked in the spring and summer months, fluctuating with the higher productivity in the estuary during those seasons (Figure 3E). Historically mysids have been of critical importance in the diets of many fish species in the SFE including Delta Smelt, Longfin Smelt, Striped Bass, and Chinook Salmon (Moyle et al. 1992; Feyrer et al. 2003; CDFG 2009; Goertler et al. 2018). However, the decline of mysids in the upper estuary has resulted in a significant decrease in their presence in the diets of fishes of the region (Feyrer et al. 2003).

This general decline in abundance continued in 2019, even though 2016 and 2017 had a modest peak in mysid abundances (Figure 2E), and the distribution and timing of peaks stayed similar over the last two decades (Hennessy 2018). *Hyperacanthomysis longirostris* was again the most common mysid in the estuary during all seasons, while the once common and native *Neomysis mercedis* continued to be almost imperceptible in the region. This has been the overall trend in the estuary's mysid communities since 1994. As in prior years, mysids in 2019 were most abundant during the summer and fall (Figure 3E), and highest concentrations occurred in the low-salinity zone of Suisun Bay and Marsh (Figure 4E).

2010 – 2019 A Decade in Review

2019 marks the end of the fifth decade of operations for the Zooplankton Study. The previous 10 years have been a period of extremes in the San Francisco Estuary, with the historic drought of 2012 to 2016, immediately followed by record levels of precipitation in the winter and spring of 2017 (Figure 5). We briefly analyzed some of the trends in abundance and distribution of key species during these periods of extreme climate events to better understand their impacts on zooplankton in the estuary. Year hydrologic classification was based on the California Department of Water Resources indices for the San Joaquin Valley at (https://cdec.water.ca.gov/reportapp/javareports?name=WSIHIST) (Figure 5).

Abundance and distribution analysis were limited to stations in the lower estuary, the southern Delta, and the San Joaquin River.

We selected five of the most abundant taxa in the estuary to focus analysis on: *Pseudodiaptomus forbesi*, *Limnoithona tetraspina*, *Diaphanosoma* spp., *Synchaeta* spp., and *Hypercanthomysis longirostris*. Abundance and distribution analysis were limited to the dry years (2012-2016) and the wet years (2011, 2017, and 2019) and the summer months when abundance is highest for most taxa. Due to nonparametric distributions of abundance estimates a Kruskal-Wallis test was performed on the monthly mean CPUE for the summer months to compare the CPUE and the centers of distribution between drought and flood years (Figure 5). The center of distribution for each month was plotted for each year and taxa so that:

$$D_{t,i} = \frac{\sum CPUE_{t,s} * km_s}{\sum CPUE_{t,s}}$$

where $D_{t,i}$ is the center of distribution for taxa *t* for month *i* and km_s is the distance of station *s* in km from the Golden Gate Bridge.

When examining the variation in abundances between dry and wet years both *P. forbesi* and *Diaphanosoma* spp. saw significant increases in their abundances during wet years, while *L. tetraspina* saw a significant decrease (Figure 6). Interestingly only *Diaphanosoma* spp. and *H. longirostris* had significant shifts of their distribution seaward in wet years compared to dry years. These trends witnessed over the past decade of the study suggest that outflow can have an influence on the abundance and distribution of different zooplankton taxa in the estuary. This supports prior research showing the correlation between summer outflows and zooplankton abundances or distribution amongst mysid species before the invasion of *P. amurensis* (Siegfried et al. 1979; Cloern et al. 1983). Analyzing how outflow affects zooplankton abundance and location in the estuary will require more research but will be important to understanding the spatial and temporal relationships between zooplankton and their fish predators.

Conclusion

In 2019 the Zooplankton Study recorded the abundances of calanoids, cladocera, rotifers, and mysids at lower densities comparable to other recent years and consistent with the downward historic trends in the estuary. Calanoid and cyclopoid copepod abundance peaked in fall, whereas cladocera, rotifers, and mysids peaked in summer. The small, abundant *Limnoithona tetraspina* was again the most abundant copepod in the upper estuary. This multi-decade zooplankton study has enabled researchers and managers to track the shifts in zooplankton abundances and community composition across the estuary for nearly 5 decades. The Zooplankton Study has documented the introduction and dominance of *Pseudodiaptomus forbesi*, *Limnoithona tetraspina*, and *Hypercanthomysis longirostris*, as well as the community's response to the invasive clam *Potamocorbula amurensis*. Understanding these dynamics and how they have fundamentally changed trophic interactions is critical to assessing food resources for fish and conservation strategies in the San Francisco Estuary.

References

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Figures

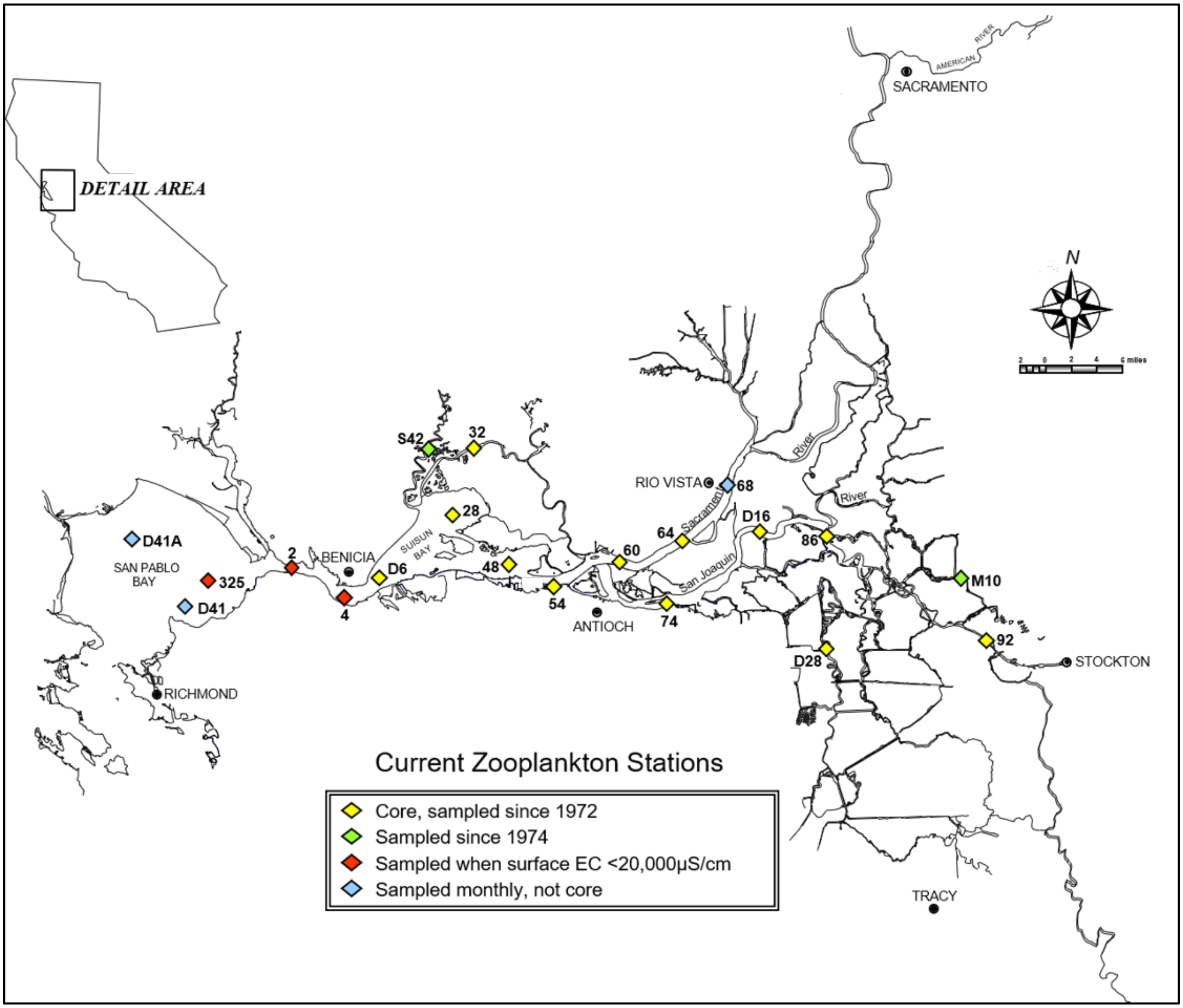

Figure 1. Map of fixed Zooplankton Study stations in the San Francisco Estuary.

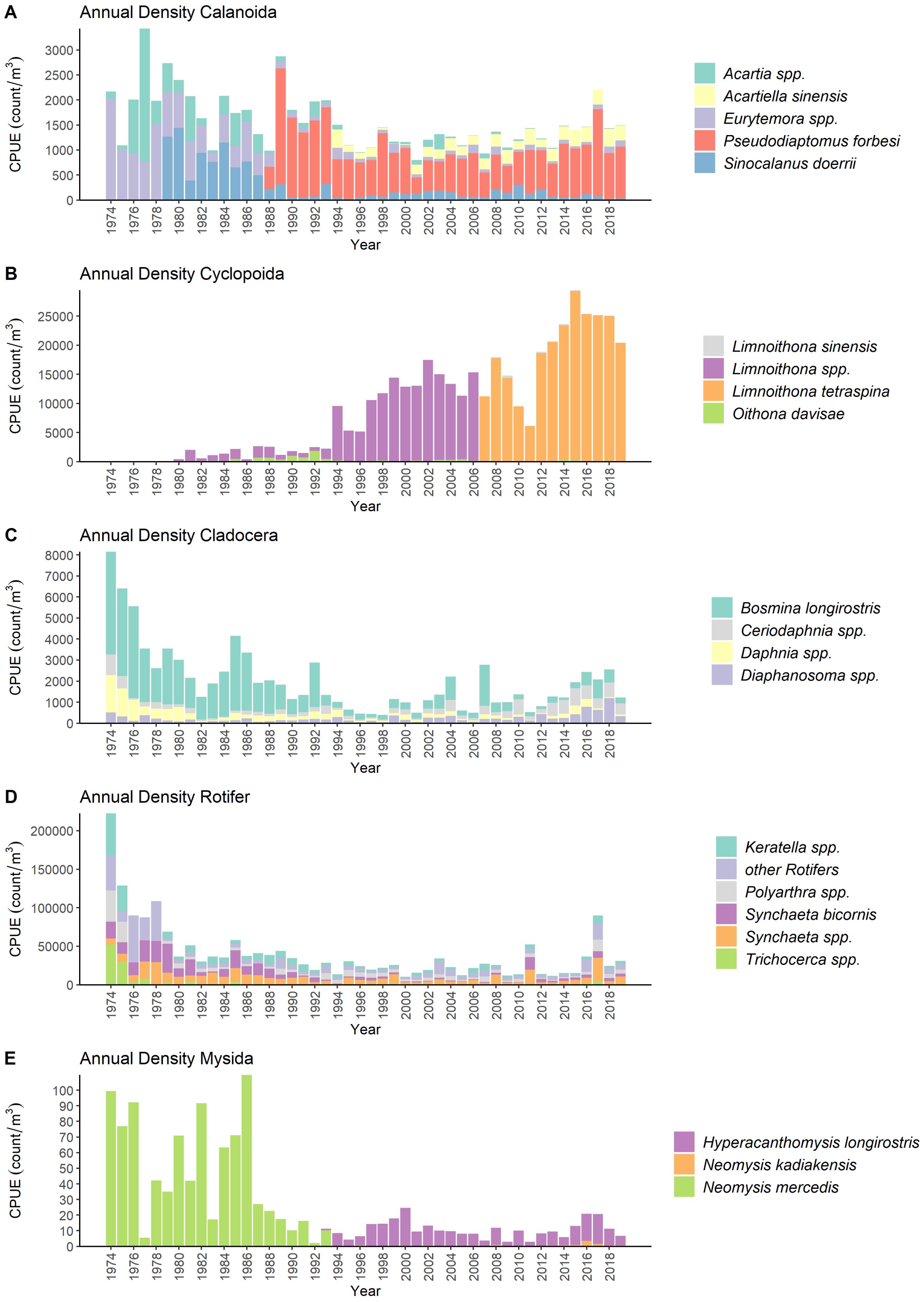

Figure 2. Annual (Mar-Nov) mean zooplankton CPUE for A) Calanoid CPUE in the CB net, B) Cyclopoida CPUE in pump samples, C) Cladocera CPUE in the CB net, D) Rotifer CPUE in pump samples, and E) Mysid CPUE in the mysid net.

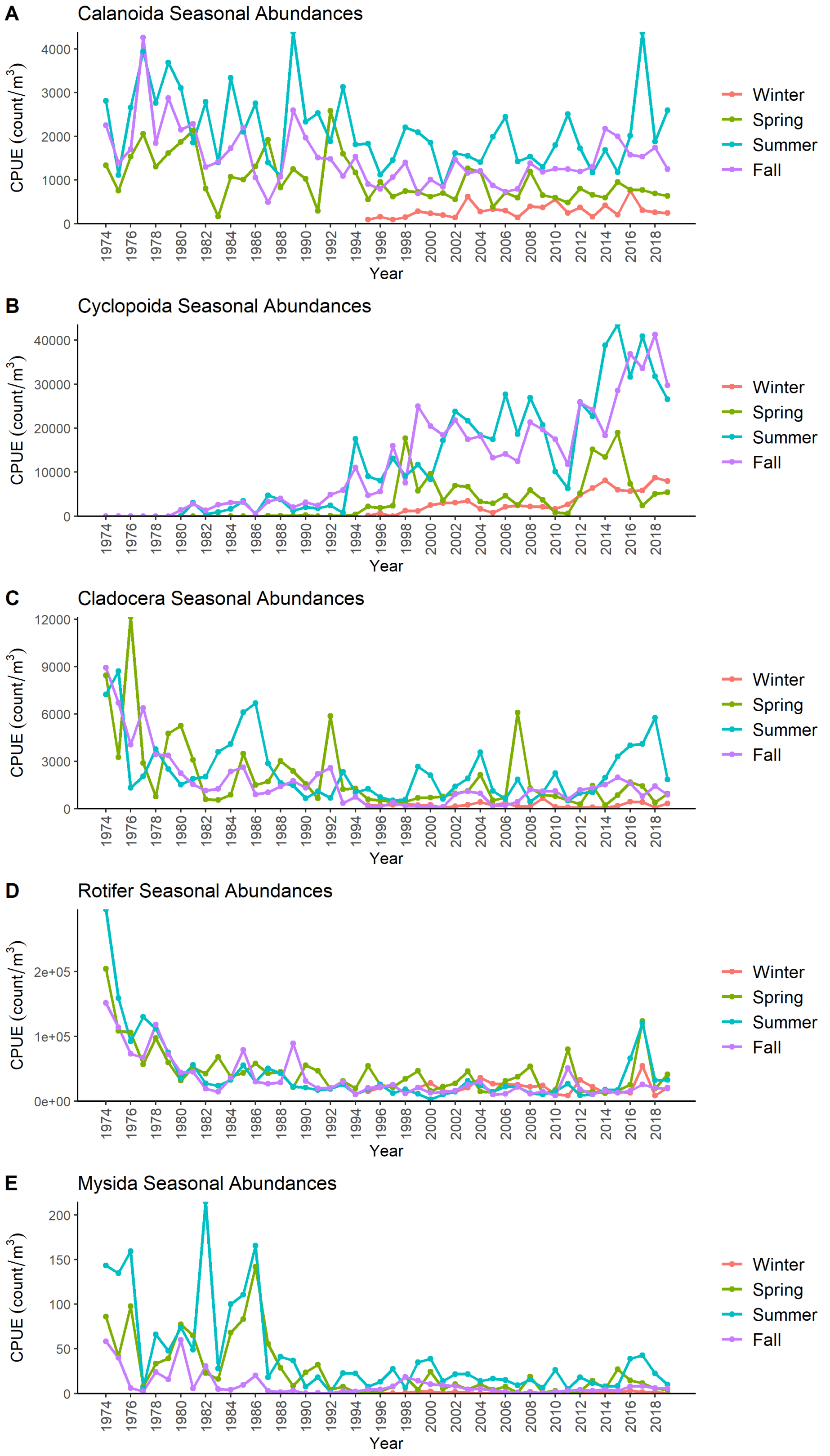

Figure 3. Seasonal mean zooplankton CPUE. Spring, summer, and fall are reported for 1974-2018, winter is reported for 1995-2019. A) Calanoid CPUE in the CB net. B) Cyclopoida CPUE in pump samples. C) Cladocera CPUE in the CB net. D) Rotifer CPUE in pump samples.

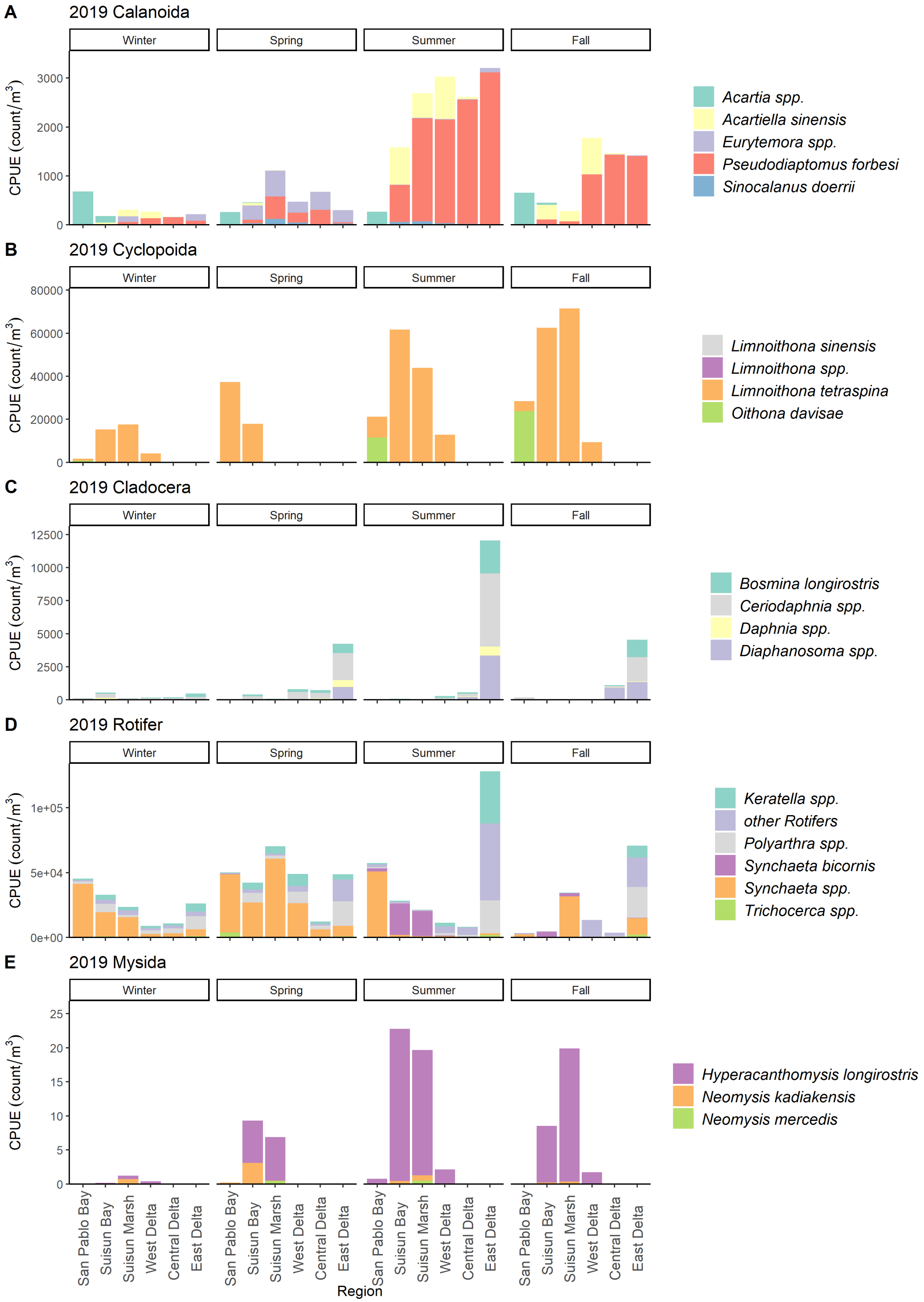

Figure 4. Seasonal mean zooplankton CPUE for 2019 by region for A) Calanoid CPUE in the CB net, B) Cyclopoida CPUE in pump samples, C) Cladocera CPUE in the CB net, D) Rotifer CPUE in pump samples, and E) Mysid CPUE in the mysid net.

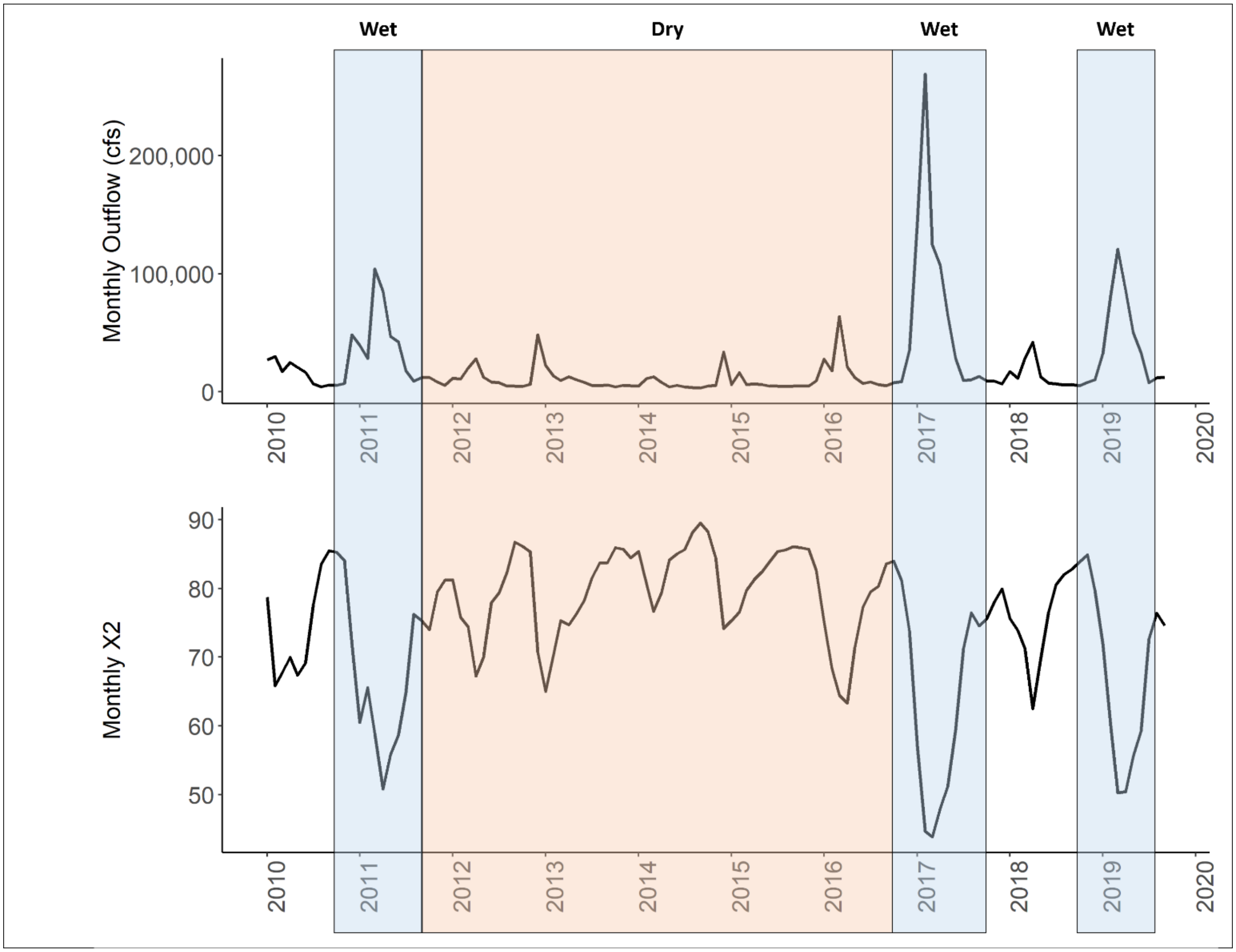

Figure 5. Average monthly outflow (cfs) and average monthly X2 position for 2010 – 2019. Dayflow data from Department of Water Resources https://water.ca.gov/Programs/Environmental-Services/Compliance-Monitoring-And-Assessment/Dayflow-Data.

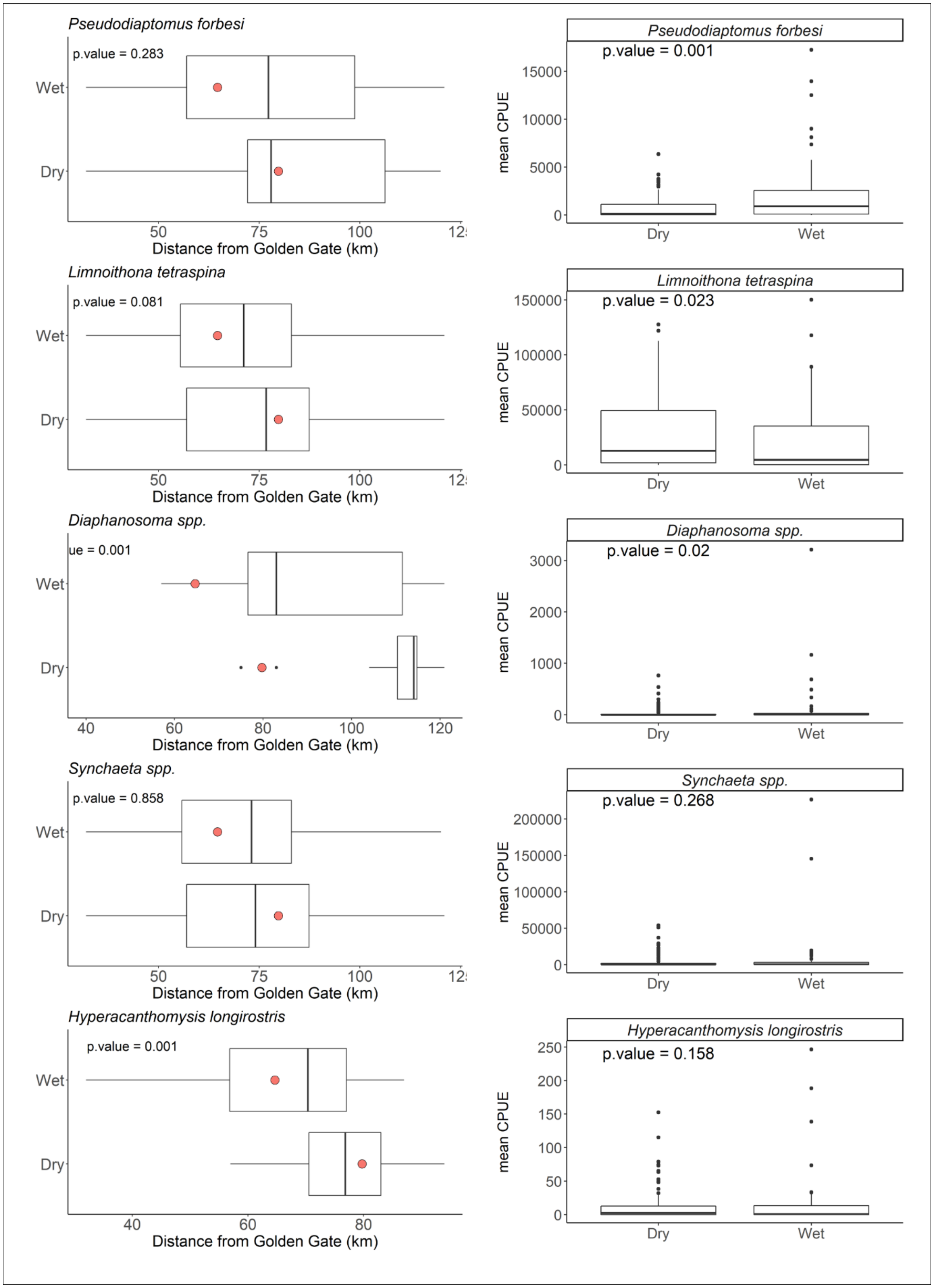

Figure 6. Average summer month CPUE and center of distribution for select taxa in drought (2012-2016) and flood years (2011, 2017, and 2019). Red points in distribution charts represent mean X2.