

## Research Article

## Monitoring invasive quagga mussels, *Dreissena rostriformis bugensis* (Bivalvia: Dreissenidae), and other benthic organisms in a western US aqueduct

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### Abstract

A quagga mussel (*Dreissena rostriformis bugensis* Andrusov, 1897) invasion of an aqueduct in Arizona was monitored from 2007 – 2011 using colonization substrates. As numbers increased, a filtering-collector caddisfly (*Smicridea fasciatella* McLachlan, 1871) declined significantly in abundance. After two years of colonization, freshwater sponges were detected and associated with a decline in *D. r. bugensis* numbers. Periphyton biomass increased considerably on substrates; perhaps partially, the result of decreased turbidity. Aqueduct biofoulers could have major impacts on costs associated with aqueduct maintenance. From an operations viewpoint, mussels are undesirable due to flow restriction associated with increased friction. Augmented sponge and periphyton biomass may also influence aqueduct operations and efficiencies.

**Key words:** aqueduct; canal; *Dreissena rostriformis bugensis*; filtering-collector; freshwater sponge; quagga mussel; *Smicridea fasciatella*

### Introduction

The Central Arizona Project (CAP) aqueduct (canal) system delivers 85 cms (m<sup>3</sup>/s) or approximately 1.6 million acre feet of water annually from Lake Havasu on the Colorado River (and lesser amounts from the Agua Fria River) to central and southern Arizona near Tucson. CAP water comes from two basic sources: the Colorado River and Lake Pleasant. Lake Pleasant primarily stores water from the Colorado River via CAP and also receives water from the Agua Fria River basin. Generally, water is stored in Lake Pleasant during low demand periods such as the winter and released during high demand summer periods. Municipal, industrial, agricultural, and Native American entities use this critical water resource. The recent discovery of the exotic invasive quagga mussel (*Dreissena rostriformis bugensis* Andrusov, 1897) in the region (Stokstad 2007; Wittman et al. 2010) has focused attention on potential

alterations to CAP water delivery. This bio-fouler has had significant impacts on water conveyance infrastructure in the area (Holdren and Turner 2011). Aqueduct ecology is also of concern, with quagga mussel invasion potentially resulting in altered aquatic invertebrate communities. Introduction of *D. r. bugensis* into the aqueduct could affect, in some indefinite manner, populations of the caddisfly (*Smicridea fasciatella* McLachlan, 1871), a long-term pest species in the system. Indeed, the present investigation was initiated in December of 2006 as part of a study on this abundant nuisance caddisfly.

Descriptions of interactions between invasive bivalves and other benthic organisms are needed to understand the former's effects on aquatic invertebrate assemblages and energy flow in western US freshwater ecosystems. *D. r. bugensis* may affect benthic communities through several mechanisms including habitat alteration by the presence of shells, increased benthic food from production of pseudofeces,

and decreased food for other filter feeders in direct competition with them (Ward and Ricciardi 2007). Changes brought about by *D. r. bugensis* may impact higher organisms, including endangered fish species, through food resource changes or alterations in ability to consume prey items because of substrate changes. Some invertebrate assemblages may be resistant to bivalve invasion (e.g., Thompson and Starzomski 2006). In terms of scale, ecosystem alteration by mussels may be local, such as substrate composition, or system wide, as in the case of sestonic organic matter (Karatayev et al. 2012).

The original introduction of quagga mussels to the western US is suspected to be from transport of veligers in bilge water transported via boat from the Great Lakes region (Wong and Gerstenberger 2011). The spread of *D. r. bugensis* in this region is considered to be especially rapid when compared to the eastern US (Wong and Gerstenberger 2011) with apparent transfer between widely separated reservoirs occurring within a single year.

This study provides information on the invasion of *D. r. bugensis* in a western US aqueduct and the response of native macroinvertebrates to this invasion. There is little published information on invasion impacts to lotic benthic invertebrates in this part of the US. Karatayev et al. (2012) also point to the near absence of information on quagga mussel impacts in the absence of zebra mussels (*D. polymorpha* Pallas, 1771) and the lack of information on ecological invasion effects by *D. r. bugensis*.

## Methods

### *Site information*

The CAP aqueduct extends 541 km from Lake Havasu City on the Colorado River eastward to Tucson, Arizona. The aqueduct averages 24 m across the top and 7 m across the bottom and is concrete lined. Sideslopes are 1.5:1 (run: rise), and the concrete lining is 9-cm thick. Average water depth is 5 m. Maximum design water velocity is approximately 1 m/s. Three sites along the CAP in the area of Scottsdale, AZ were arrayed with substrates for study of the quagga mussel invasion and aquatic macroinvertebrate response. Site 2 was 6.6 km downstream from Site 1, while Site 3 was 9.6 km downstream from Site 1 (Figure 1).

Quagga mussels were first detected within reservoirs on the Colorado River in January of 2007 with veligers (the larval, free floating stage of the mussel) found in the CAP aqueduct early in 2007; adult quagga mussels were confirmed at Lake Pleasant in December 2007 (Stokstad 2007).

### *Substrates/macroinvertebrate collections*

Concrete pavers (30×30×8 cm) were used as colonization substrates in the CAP aqueduct and were considered replicate experimental units. Eye bolts were attached to pavers with epoxy and then a length of chain attached from the eye bolt to a stake on the canal bank. Chain was cut at 2.4, 4.9, 7.3 and 9.8-m lengths. This corresponded roughly to 0.9, 2.4, 4.3, and 5.1-m depths, dependent upon water levels in the aqueduct. Three substrates were used for each position at each site (total of 12 substrates per site). A single substrate from each position at each site was sampled on each visit with other substrates left undisturbed until the next sampling occasion. This design allowed for extensive invertebrate colonization opportunities on any given substrate. A colonization period of two months has been found adequate to represent invertebrate communities in other studies (Shaw and Minshall 1980), and is similar to the time needed for substrates to accrue periphyton biomass to natural levels (Biggs 1988). Minimum colonization time for this study was approximately three months. Regionally, new quagga mussel settlers occur on substrates placed in Lake Mead after only two weeks, even during the less productive winter season (Chen et al. 2011). Dreissenid mussels spread rapidly, aided by downstream drift of pelagic larvae and translocation of adult mussels (e.g., Griffiths et al. 1991). CAP samples were collected on multiple occasions each year of the study (2007 – 2011), with substrates initially placed in the aqueduct in December of 2006.

A brush, in conjunction with an alcohol rinse, was used to harvest macroinvertebrates along with periphyton and sediment from the entire substrate surface. Material was retained on a 600-micron mesh sieve and preserved in 70% propanol.

To avoid confounding mussel impacts with natural, temporal changes in macroinvertebrate assemblage variability over time, a regional area without mussels (reference site) was also sampled (e.g., Smith 2002). It was assumed that



**Figure 1.** Sampling sites along the CAP near Phoenix, Arizona, USA.

temporal variability would be similar between CAP and the regional reference site. Regional macroinvertebrate reference samples were collected from Las Vegas Wash which flows into Lake Mead near Las Vegas, Nevada. The Las Vegas Wash community (e.g., Nelson 2011) is similar to that of the aqueduct and the caddisfly *S. fasciatella* is a species abundant in both areas. These samples were collected on an approximately quarterly basis from three sites along the Wash with a D-frame net during a 1-minute period while kicking up stream-bottom substrate. Substrates in the form of bricks were also placed in the Wash to further aid in detection of adult mussels. At the time of sample collection, no dreissenid mussels had been observed at this site. Reference samples were used to determine whether there might be a regional change in macroinvertebrate richness or abundance that was linked to something other than a mussel invasion, such as a regional change in weather patterns.

In the laboratory, samples were washed in a 600- $\mu\text{m}$  mesh sieve to remove alcohol, invertebrates were picked from the sample with

the aid of an illuminated 10 $\times$ -magnifier, and then enumerated and identified under a binocular dissecting scope. Colonial sponge (Porifera) presence was noted. Biomass of coarse particulate organic matter (CPOM) and plant matter related to autotrophic production (periphyton) were measured for each sample. Periphyton also contained sponge material when this organism became common on substrates. Sediment from substrates was retained on a 63- $\mu\text{m}$  sieve during sample processing. These samples were dried at 60°C for 48 hrs and weighed to the nearest hundredth of a gram.

#### *Mussel veliger collections*

A 65- $\mu\text{m}$  Nitex® nylon zooplankton tow net (30-cm diameter opening) with a 64- $\mu\text{m}$  dolphin bucket was used in 1-minute timed sampling for veligers on occasion in the main flow of the aqueduct. A single tow sample was also collected from Las Vegas Wash. Samples were preserved in ethanol and processed as described by Hosler (2011) where the whole bulk sample is settled out, and veligers reported as the concentrated count in the sample.

### *Water quality*

Surface water quality [dissolved oxygen (DO), conductivity, pH, turbidity, and temperature] and velocity were measured at each of the three CAP sites. A 250-ml water sample was collected for fine particulate organic matter (FPOM) and then filtered onto glass-fiber filters. Ash-free-dry-mass was determined using standard methods (Eaton et al. 1995). Filters were dried for 48 hr at 105°C, weighed using an analytical balance, ashed at 500°C for 1 hr, and mass of the residue (ash weight) determined. Data were reported as g FPOM/L.

### *Data analysis*

Ordination techniques were used to examine patterns in macroinvertebrate data and to identify physical and biological variables (including quagga mussel abundance) most closely associated with invertebrate distributions. Initial analyses of the macroinvertebrate data sets used detrended correspondence analysis (DCA), and revealed a data gradient length  $< 3$ , suggesting that linear models were appropriate for analysis. Therefore, redundancy analysis (RDA) was used for direct gradient analyses. Faunal data were transformed (square root transformation) before analysis. Forward selection of environmental variables and Monte Carlo permutations were used to determine which and to what extent environmental variables exerted a significant ( $P \leq 0.05$ ) effect on invertebrate distributions. If environmental variables were strongly correlated (Pearson correlation,  $r \geq 0.6$ ), only a single variable was selected for use in RDA to avoid problems with multicollinearity. Environmental variables were normalized [( $\ln(X+1)$ ) or arcsine square root transformation for percentage data] if the Shapiro-Wilks Test indicated non-normality. In the ordination diagram, taxa and sites are represented by points and environmental variables by arrows. Arrows roughly orient in the direction of maximum variation of the given variable.

Kruskal-Wallis was used to test for differences in biotic variables associated with substrates between years and was followed by the All-Pairwise Comparisons Test if a significant difference ( $P \leq 0.05$ ) was detected. Different sites, depths, and substrates were all sources of variability between years. This allowed for realistic testing for differences between years for this section of aqueduct.

Wilcoxon Rank Sum Test was used to compare taxa richness and invertebrate abundance between substrates with and without quagga mussels. Spearman-rank was used to examine correlations between quagga mussel abundance and some taxa that appeared to respond to mussel abundance in the RDA. Non-parametric tests were used because of the strong deviation from normality (tested with Shapiro-Wilk) that occurred with some variables. Taxa richness and abundance were also examined for correlation with substrate mussel abundance.

## **Results**

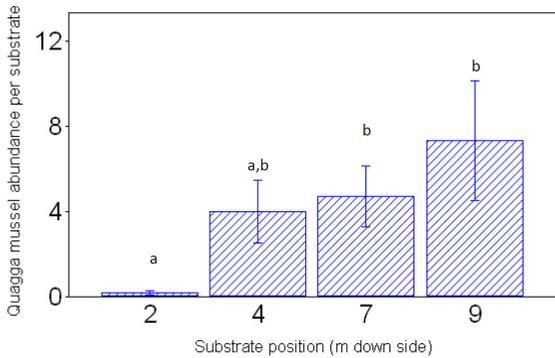
Substrate collections varied among years but were most intensive during the first year (2007) with dates in March, April, June, August, and October. Sampling only occurred in January and December of 2008, July and December of 2009, March and July of 2010, and January and July of 2011.

### *Mussel abundance*

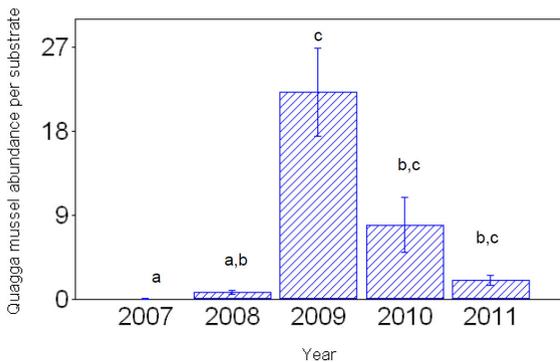
Mussel density was lowest at the most shallow substrates along the side of the aqueduct (Kruskal-Wallis Statistic = 18.5076, P-value = 0.0003) (Figure 2). The most upslope substrates located 2.4 m down from the bank (approximately 0.9 m depth) were sometimes exposed to the atmosphere as water levels varied in the aqueduct, making conditions inhospitable to some aquatic organisms.

A single attached mussel was detected in the initial sampling year, 2007. Mean quagga mussel abundance differed between years (Kruskal-Wallis Statistic = 66.9382, P-value  $< 0.0001$ ) with the highest recorded numbers in 2009 (Figure 3). After 2009, there was a decrease in mussel abundance in 2010 and 2011. In 2007 – 2008 only 11 of 57 substrates (top substrates omitted) contained quagga mussels. From 2009 to 2011, 45 of 54 of the deeper positioned substrates contained quagga mussels. Mean mussel density (with shallow substrates omitted) after 2008 was 85 mussels/m<sup>2</sup> with a maximum density of 457/m<sup>2</sup>.

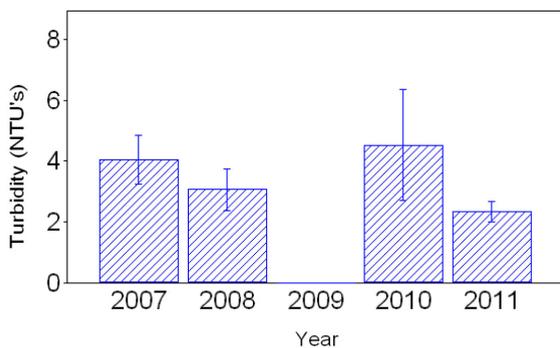
Intermittent collection of mussel veligers from the aqueduct indicated presence in 2007 (5.9 veligers/ml) and the continued presence in 2009 (16.0 veligers/ml), 2010 (3.4–13.6 veligers/ml), and 2011 (98.0 veligers/ml).



**Figure 2.** Quagga mussel abundance at different distances down the side of the Central Arizona Project aqueduct. Bars with the same lower-case letters indicate no significant difference (Kruskal-Wallis All-Pairwise test,  $P > 0.05$ ).



**Figure 3.** Quagga mussel abundance on artificial substrates in the Central Arizona Project aqueduct from the years 2007 to 2011. Bars with the same lower-case letters indicate no significant difference (Kruskal-Wallis All-Pairwise test,  $P > 0.05$ ). Variation is represented by standard error.



**Figure 4.** Mean turbidity values from samples taken along the Central Arizona Project aqueduct from 2007 through 2011. Turbidity in 2009 differed significantly from other years ( $P = 0.0039$ ). Variation is represented by standard error.

### Water quality

Mean water quality variables for the 5 years ( $n = 36$ ) were  $16.1^{\circ}\text{C}$  (range  $10.6 - 21.1$ ) for water temperature,  $973\mu\text{S}/\text{cm}$  (range  $771 - 1082$ ) for conductivity,  $9.0\text{ mg}/\text{L}$  (range  $6.6 - 11.6$ ) for DO,  $8.1$  (range  $7.2 - 8.6$ ) for pH, and  $0.43\text{ m}/\text{s}$  (range  $0.02 - 1.0$ ) for flow velocity. It is unlikely that any of these values would have a negative impact on biota associated with the aqueduct. Turbidity averaged  $3.0 \pm 0.2$  NTU's over the course of the study, however, values measured in 2009 were all zero (Figure 4) and values differed significantly between years (Kruskal-Wallis Statistic =  $15.4123$ ,  $P$ -value =  $0.0039$ ).

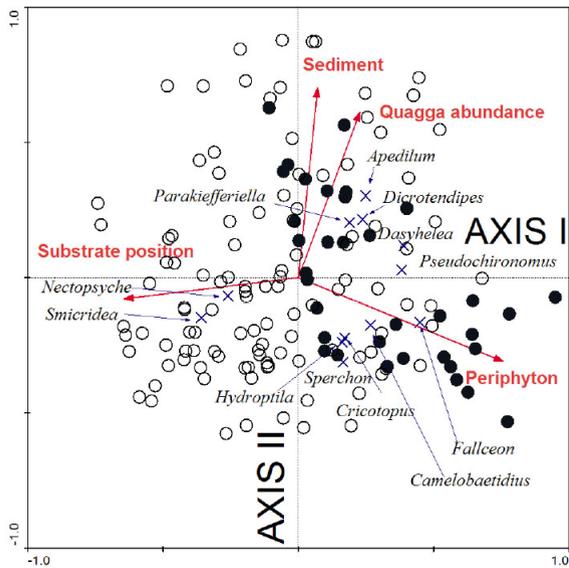
### Ordination of aqueduct invertebrate communities

The abundance of quagga mussels on substrates, amount of sediment, CPOM mass, substrate position along the side of the canal, and periphyton biomass were the initial variables used in the RDA model. We only used variables specific to individual substrates. Water quality variables were not used because they were collected in a more general manner and unlikely to be limiting.

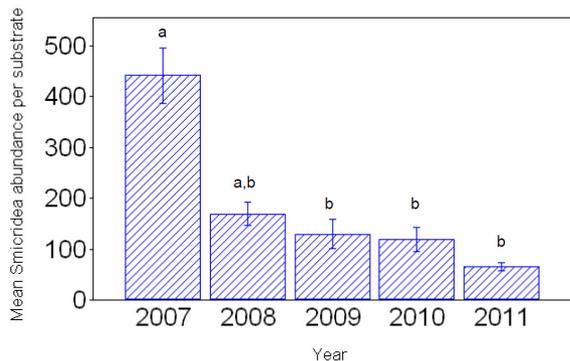
Of the initial variables, only CPOM was not significant ( $P < 0.05$ ) in the model. Periphyton biomass and substrate position were important along the first axis (Figure 5) while sediment and mussel abundance influenced invertebrate communities along Axis II. Eigenvalues for Axis I were  $0.067$  and for Axis II,  $0.038$ . Cumulative percentage of explained variance in the species-environment relationship for the first two axes was  $94.3\%$ . The amount of sediment on a substrate was significantly higher (Wilcoxon Rank Sum Test, Two-tailed  $P$ -value =  $0.0376$ ) on substrates that contained quagga mussels.

Invertebrate community groupings appeared to be roughly divided into shallow substrate communities, deep substrate communities, and a post quagga mussel invasion (2009 – 2011) community (Figure 5). The shallowest substrates were located in the positive portion of Axis I of Figure 5. Mayflies such as *Fallceon* (Waltz and McCafferty 1987) and *Camelobaetidius* (Demoulin, 1966) tended to be most abundant on these shallow substrates that often contained high amounts of periphyton.

The caddisfly *S. fasciatella* (along with less abundant *Nectopsyche* (Mueller, 1879)) was found on deeper substrates and opposite of increased quagga mussel abundance (Figure 5).

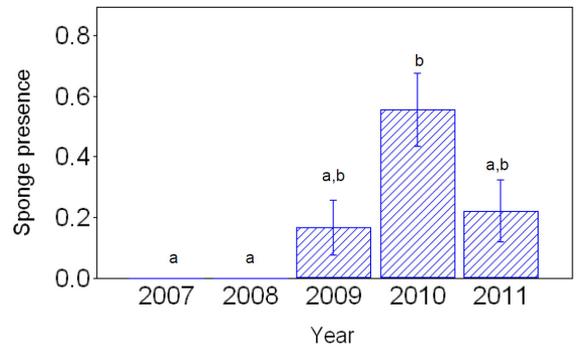


**Figure 5.** Redundancy Analysis (RDA) of macroinvertebrate communities on artificial substrates in the Central Arizona Project aqueduct. Environmental variables were related to community attributes as shown by arrows. Substrate samples are represented by circles as shown in the legend. Filled circles represent shallow substrates and open circles represent substrates further down the canal sides.

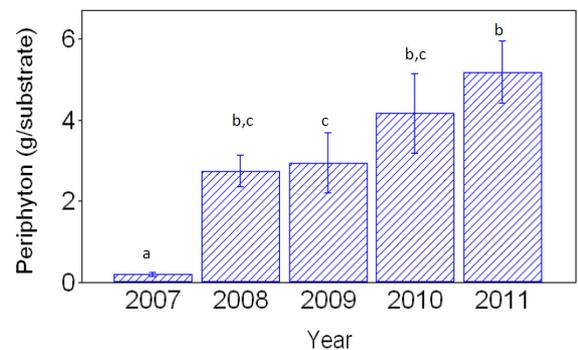


**Figure 6.** Abundance of the caddisfly *S. fasciatella* on artificial substrates in the Central Arizona Project from 2007 to 2011. Bars with the same lower-case letters indicate no significant difference (Kruskal-Wallis All-Pairwise test,  $P > 0.05$ ). Variation is represented by standard error.

*S. fasciatella* abundance was significantly and negatively correlated with quagga mussel abundance ( $n = 111$  total substrates with shallow substrates omitted, Spearman rank,  $r = -0.3582$ ,  $P = 0.0001$ ). Initially this caddisfly made up 68% of invertebrate abundance on substrates, but values decreased by 2010 (11%) and 2011 (39%). The density of *S. fasciatella* was significantly reduced from 2009 through 2011,



**Figure 7.** Sponge presence on artificial substrates in the Central Arizona Project aqueduct during the years 2007 to 2011. Presence increased after 2008. Bars with the same lower-case letters indicate no significant difference (Kruskal-Wallis All-Pairwise test,  $P > 0.05$ ). Variation is represented by standard error.



**Figure 8.** Periphyton biomass on artificial substrates in the Central Arizona Project aqueduct from 2007 to 2011. There were significant differences in mass between years. Bars with the same lower-case letters indicate no significant difference (Kruskal-Wallis All-Pairwise test,  $P > 0.05$ ). Variation is represented by standard error.

compared to 2007 through 2008 (Kruskal-Wallis Statistic = 25.5775,  $P$ -value  $< 0.0001$ ) (Figure 6) in the CAP. It appeared that some taxa increased in abundance in association with quagga mussel increases. Chironomid larvae like *Apeditum* (Townes, 1945), *Dicotendipes* (Kieffer, 1913), and *Parakiefferiella* (Thienemann, 1936) were especially common in 2010. When grouped together as chironomids, there was a significant difference (Kruskal-Wallis Statistic = 47.5113,  $P$ -Value  $< 0.0001$ ) in chironomid abundance between years, with abundance especially high in 2010. Sponge associates, *Xenochironomus* (Kieffer, 1921) and *Climacia chapini* (Parfin and Gurney, 1956), also became more common. Sponges were not detected in the aqueduct until

2009 after which sponge presence per substrate remained insignificantly different from 2009 through 2011 (Figure 7) (Kruskal-Wallis Statistic = 33.1675, P-Value < 0.0001).

#### *Non-dreissenid taxon richness and abundance in relation to quagga mussels*

Because mussels were nearly absent from top substrates, these substrates were omitted from analyses. There was a significant difference (Wilcoxon Rank Sum Test, Two-tailed P-value = 0.0468) in invertebrate abundance between substrates with quagga mussels (n=56) and those without mussels (n = 55). Mean abundance was higher on substrates not inhabited by mussels ( $\bar{X} = 619 \pm 80$  individuals / substrate) compared to those where mussels were detected ( $\bar{X} = 484 \pm 87$  individuals/substrate). There was also a significant difference (Wilcoxon Rank Sum Test, Two-tailed P-value = 0.0127) in taxa richness between substrates with and without mussels: Mean richness was higher on substrates with mussels ( $11.9 \pm 0.5$  taxa/substrate) than those without mussels ( $9.9 \pm 0.3$  taxa/substrate).

#### *Macroinvertebrate food resources/relationship with mussel abundance*

CPOM did not differ from year to year (Kruskal-Wallis Statistic = 6.0120, P-value = 0.1983) nor did FPOM (Kruskal-Wallis Statistic = 5.6215, P-value 0.2293). Periphyton biomass, however, did differ significantly (Kruskal-Wallis Statistic = 61.2999, P-value < 0.0001) between years (Figure 8). While CPOM was not significantly correlated to quagga mussel abundance (Spearman rank, P = 0.2735), both periphyton biomass ( $r = 0.3200$ , P = 0.0007) and FPOM ( $r = 0.2731$ , P = 0.0038) were correlated with mussel abundance.

#### *Reference macroinvertebrate communities*

Unlike *S. fasciatella* abundance in the CAP, no significant difference (Kruskal-Wallis Statistic = 2.5543, P-value = 0.6349) was detected in abundance between the years 2007 – 2011 at Las Vegas Wash sites ( $\bar{X} = 20.5 \pm 4.5$  *S. fasciatella*/kick-net). Chironomid abundance also did not differ significantly (Kruskal-Wallis Statistic = 8.1881, P-value = 0.0791) between years at Las Vegas Wash sites ( $\bar{X} = 98.0 \pm 23.7$  midges/kick-net). No adult mussels were detected during

benthic sampling, and veligers were not detected in a tow sample collected in April of 2010.

## **Discussion**

Based on observational data, *D. r. bugensis* invasion appeared to impact the relatively simple aquatic community associated with the CAP. The different colonization times, despite other studies indicating that minimum times were met for colonization of substrates (Shaw and Minshall 1980; Biggs 1988; Chen et al. 2011), is a study weakness and these observations should be viewed with some caution.

Strong declines in abundance of a collector-filterer caddisfly (*S. fasciatella*) were observed in conjunction with the mussel invasion, while numbers of some chironomid larvae (mostly deposit feeders) increased. Data from a temporal reference site collected over the same period did not demonstrate any year-to-year differences in abundance of these groups. The decline in one filtering-collector taxa found in the CAP was followed by the increased presence of a different filterer, the freshwater sponge.

Freshwater sponges may have, in turn, impacted the aqueduct community and reduced the presence of mussels. Ricciardi and co-workers (1995) have hypothesized that sponges secrete inhibitory toxins that prevent colonization by other organisms; according to the hypothesis, quagga mussels placed on sponge colonies would fail to lay down byssal threads. Sponge growth may be enhanced by the presence of mussel filtration activities because of increased light intensity and grazing pressure that may cause shifts in FPOM towards smaller cell sizes that are selectively utilized by sponges (e.g., Ricciardi et al. 1995). Ricciardi and co-workers (1995) further suggest that sponges could control dreissenids on hard substrates like canal walls. If this inhibitory effect of sponges operates similarly on other species, an increased spread of sponges could prevent increased abundance of *S. fasciatella* in the future despite potential declines in mussels. We did not observe statistically significant differences in FPOM between years; however, size categories could have been altered to favor sponges. Our infrequent sampling likely did not capture the true variability of measures like FPOM and turbidity. The decreased turbidity found in our study is, however, typical of dreissenid invasions (Higgins and Vander Zanden 2010). Decreased

turbidity is likely a system-wide symptom of mussel invasion that may have occurred, not only in the aqueduct, but in reservoirs that provide water to the CAP.

Large increases in CAP periphyton biomass were observed after 2007. Potential causes for the observed increases in periphyton mass include lower turbidity, which may have increased autotrophic production (e.g., Lloyd et al. 1987), and the increased biomass of freshwater sponges. The increased sediment that we found in association with quagga mussels could be from biodeposits (e.g., Sardiña et al. 2011) and these deposits may be high in nutrients that can lead to increased periphyton biomass (Grenz et al. 1990). Lower numbers of *S. fasciatella* may have also allowed for increased periphyton biomass. While generally operating as a filtering-collector, *Smicridea* may be omnivorous (de Carvalho and Uieda 2009) and perhaps consume some periphyton directly. Higgins and Vander Zanden (2010) in a meta-analysis study of dreissenid mussel impacts found algal biomass increased 170% following invasion, indicating periphyton increase in the CAP is an expected response.

Increased taxa richness was observed with quagga mussel colonization in the CAP and has been observed in other areas where dreissenids have established (Ozersky et al. 2011). However, at least two of the newly detected taxa (*Xenochironomus* and *Climacia chapini*) were those that have relationships with sponges (Roback 1962, Brown 1952). It seems possible that the increased presence of sponges was an indirect effect of the mussel invasion.

Abundance of non-dreissenid invertebrates often increase in areas of mussel colonization (Ozersky et al. 2011), although such increases are dependent upon the makeup of substrate (Ward and Ricciardi 2007). Ward and Ricciardi (2007) also concluded from a meta-analysis of dreissenid invasions that increased benthic macroinvertebrate density and taxonomic richness are effects generally associated with these invasions. In the CAP, increased taxa richness occurred on substrates with mussels; however, benthic abundance was significantly reduced. The relatively simple CAP benthic community was numerically dominated by the caddisfly *S. fasciatella*, which declined precipitously following quagga mussel invasion and caused decreases in overall invertebrate abundance. This finding of a decrease in filtering-collectors is also suggested by the dreissenid invasion analysis of

Ward and Ricciardi (2007). Decreased numbers of net-spinning caddisflies were noted after mussel establishment in the St. Lawrence River in Canada (Ricciardi et al. 1997). Conversely, the observed increase in deposit-feeders, such as the chironomids, *Apedilum* and *Parakiefferiella*, has been seen in other studies where they dominate mussel beds (Sardiña et al. 2011).

*S. fasciatella* appeared to experience a large decline in abundance as mussels invaded and this may have been a desirable result for neighbourhoods that were in close proximity to large numbers of emerging nuisance adult *S. fasciatella*. In a more natural ecosystem, this decreased terrestrial biomass could have impacts on riparian biota that are dependent upon these subsidies.

It is unclear what trajectory the aqueduct macroinvertebrate community will take after 2011. The presence of quagga mussels appeared to quickly alter the community and may have destabilized it, allowing for continued community changes. What was a relatively simple community dominated by a single species of caddisfly may have been especially susceptible to invasion. Ecological communities differ in vulnerability to invasion, with some support for species-rich systems having a greater ability to exclude invading species than species-poor systems (e.g., Thompson and Starzomski 2006). Post-invasion these simple communities may undergo dramatic changes in community structure.

Overall, however, the ecosystem response in the CAP is similar to that described in the meta-analysis studies of Ward and Ricciardi (2007) and Higgins and Vander Zanden (2010) which lends a weight-of-evidence that our observations were the result of quagga mussel invasion. Decreased turbidity and increased periphyton biomass along with increased benthic macroinvertebrate taxa richness are all predicted by these studies. Within the macroinvertebrate community, the decreased abundance of a filtering-collector caddisfly and increased abundance of some deposit feeders are also expected results. In this particular study, the spread of freshwater sponges may have long-term impacts to the canal community in terms of preventing further mussel colonization and perhaps in maintaining low numbers of *S. fasciatella*. The lower abundance of *S. fasciatella* may be a desirable impact resulting from aqueduct invasion by the quagga mussel. However, the increased presence of colonial sponges and periphyton may lead to

decreased flow in the canal through increased friction loss, inhibiting water deliveries and resulting in need for more frequent canal cleaning (e.g., Bryant 1974).

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