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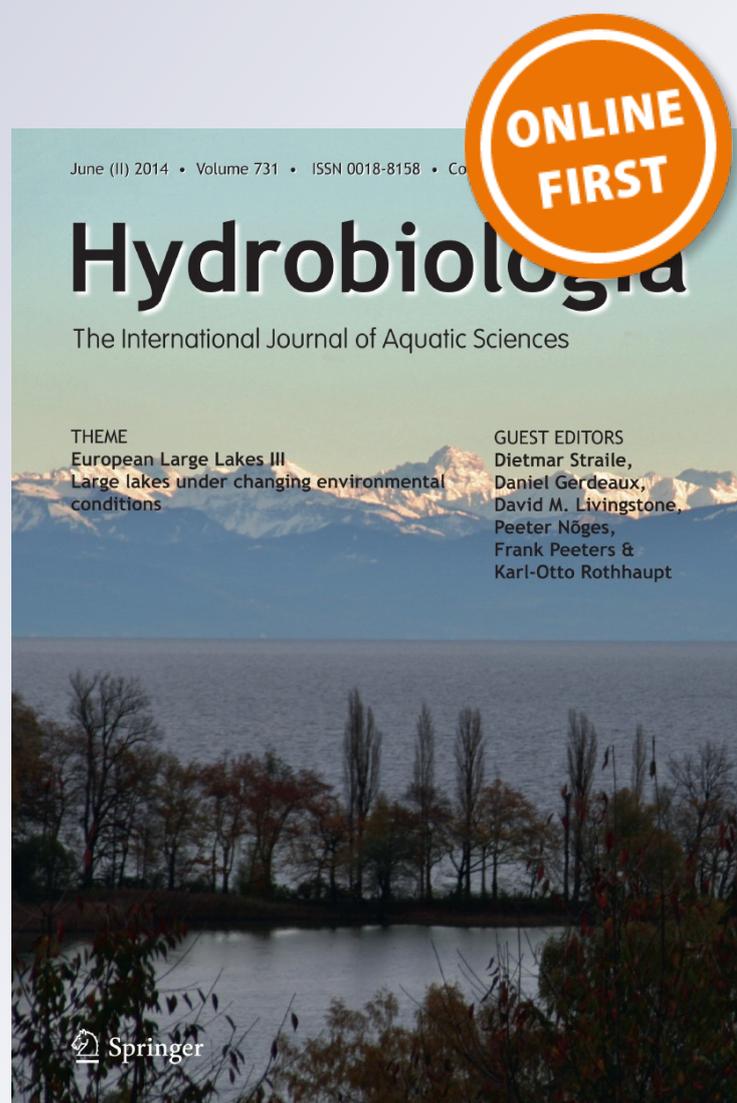
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# Effects of invasive European bird cherry (*Prunus padus*) on leaf litter processing by aquatic invertebrate shredder communities in urban Alaskan streams

David A. Roon · Mark S. Wipfli · Tricia L. Wurtz

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**Abstract** European bird cherry (*Prunus padus*) (EBC) is an invasive ornamental tree that is spreading rapidly in riparian forests of urban Alaska. To determine how the spread of EBC affects leaf litter processing by aquatic invertebrate shredders, we conducted complementary leaf pack experiments in two streams located in Anchorage, Alaska. The first experiment contrasted invasive EBC with three native tree species—thin-leaf alder (*Alnus tenuifolia*), paper birch (*Betula neoalaskana*), and black cottonwood (*Populus trichocarpa*)—in one reach of Chester Creek; finding that EBC leaf litter broke down significantly faster than birch and cottonwood, but at a similar rate to alder. The second experiment contrasted EBC with alder in four reaches of Campbell

and Chester creeks; finding that while EBC leaf litter broke down significantly faster than alder in Chester Creek, EBC broke down at a similar rate to alder in Campbell Creek. Although EBC sometimes supported fewer shredders by both count and mass, shredder communities did not differ significantly between EBC and native plants. Collectively, these data suggest that invasive EBC is not currently exhibiting strong negative impacts on leaf litter processing in these streams, but could if it continues to spread and further displaces native species over time.

**Keywords** Invasive species · Riparian forests · Leaf litter processing · Aquatic invertebrate shredders · *Prunus padus* · Alaska

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## Introduction

The spread of non-native plants is a concern worldwide because they can displace native species, reduce biodiversity, and disrupt ecological processes (Blossey, 1999; Mack et al., 2000; Dukes & Mooney, 2004). Riparian zones often support a higher percentage of non-native plants than upland habitats due to the combined effects of flood events that disturb riparian soils and landscape connectivity that links populations along riparian corridors (Planty-Tabacchi et al., 1996; Hood & Naiman, 2000). As a result, non-native plants in riparian forests are currently spreading in many locations, with notable invasions of Russian olive

(*Elaeagnus angustifolia*) and tamarisk (*Tamarix* spp.) in the western United States (Royer et al., 1999; Sher et al., 2002; Friedman et al., 2005; Mineau et al., 2012), knotweeds (*Fallopia* spp.) in the United States and Europe (Braatne et al., 2007; Lecerf et al., 2007; Claeson et al., 2013), and willows (*Salix* spp.) in Australia (Read & Barmuta, 1999). Because riparian zones link terrestrial and aquatic ecosystems, the spread of non-native plants in riparian forests could affect stream organisms, food webs, and ecosystem function (Gregory et al., 1991; Hood & Naiman, 2000; Gessner & Chauvet, 2002).

One pathway through which non-native riparian plants can affect stream ecosystems is by changing the species composition of leaf litter inputs into streams. Small streams often rely on these allochthonous leaf litter inputs as basal sources of organic matter (Vannote et al., 1980; Webster & Benfield, 1986; Wallace et al., 1997). Aquatic invertebrate shredders feed on conditioned leaf litter and process it into smaller fragments that serve as food for other functional feeding groups of aquatic invertebrates (Vannote et al., 1980; Cummins et al., 1989; Wallace et al., 1997). Collectively, these aquatic invertebrate communities support upper trophic level consumers like stream fishes (Wallace et al., 1997; Baxter et al., 2005).

The spread of non-native plants can alter the timing, quantity, and quality of leaf litter inputs into streams (Abelho & Graça, 1996; Lecerf et al., 2007; Mineau et al., 2012). Non-native plants can differ in quality from native species, for example, through lower nutrient content, increased leaf toughness, or presence of secondary plant compounds (Irons et al., 1988; Cummins et al., 1989; Abelho & Graça, 1996; Motomori et al., 2001). Previous studies have documented that differences in leaf litter quality associated with some non-native species can affect shredder feeding activity and leaf litter processing (Bailey et al., 2001; Dangles et al., 2003; Lecerf et al., 2007; Going & Dudley, 2008; Moline & Poff, 2008). If the spread of a non-native riparian plant negatively affects shredder communities, then effects could be expressed through multiple trophic levels, and potentially disrupt ecological processes and ecosystem function of streams (Cummins et al., 1989; Gessner & Chauvet, 2002; Baxter et al., 2005).

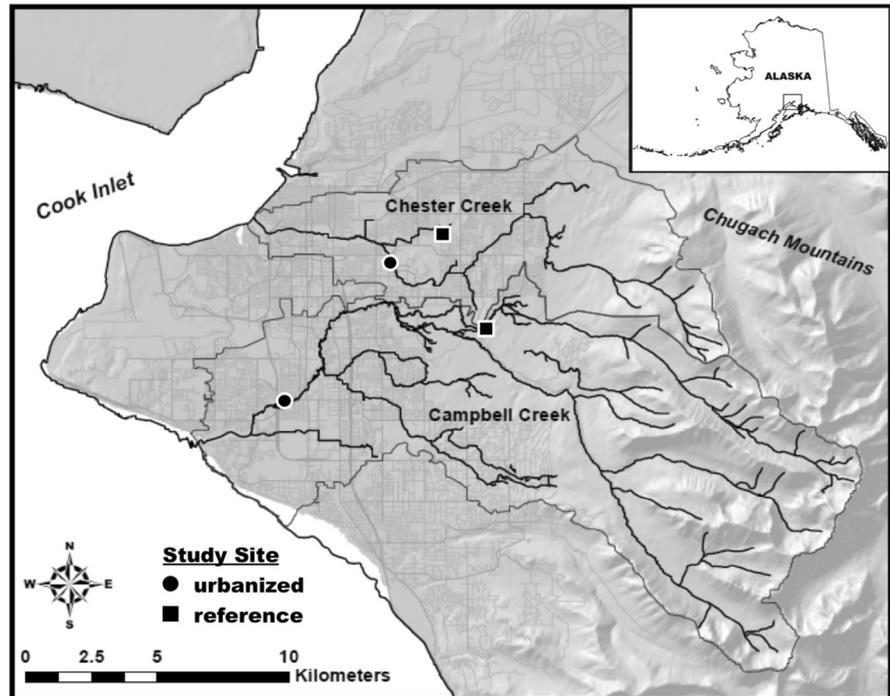
European bird cherry (*Prunus padus* L.) (Leather, 1996) (EBC) is a non-native ornamental tree

frequently planted in Alaska because it is one of the few flowering trees able to survive through the winter (Alaska Natural Heritage Program, 2006). Birds feed on the cherries and disperse the seeds from source trees to adjacent natural areas, creating wild populations over time (Alaska Natural Heritage Program, 2006). Able to tolerate a wide range of climates and habitat types in its native Europe (Leather, 1996), EBC thrives in cold climates and wet soils, making it well suited for riparian zones in Alaska. EBC has spread rapidly in the last 10 years, and appears to be displacing native trees in riparian forests along streams in parts of urban Alaska (Roon, 2011). EBC is able to form dense, monotypic stands, and among invasive plant species in Alaska it is ranked as “highly invasive” (Carlson et al., 2008).

At the same time, the dominant riparian tree species, thin-leaf alder (*Alnus tenuifolia* Nutt.), appears to be decreasing across southcentral Alaska due to the combined effects of a native canker and invasive insect pests that defoliate it (Ruess et al., 2009; Kruse et al., 2010). Due to a symbiotic relationship with nitrogen-fixing bacteria, alder is an important source of nitrogen for riparian and stream ecosystems (Helfield & Naiman, 2002) and is a desirable food source for aquatic invertebrate shredders (Irons et al., 1988; Motomori et al., 2001). Because EBC contains a much lower concentration of foliar nitrogen (Goldman, 1961; Leather, 1993), this shift in riparian vegetation from alder to EBC could affect stream food webs by changing the composition and quality of leaf litter inputs entering streams where this tree has colonized.

To determine if EBC affects leaf litter processing by aquatic invertebrate shredders, this study compared leaf litter breakdown rates and aquatic invertebrate shredder communities associated with EBC and native plant species in selected urban Alaskan streams. Based on previous studies characterizing *Prunus* leaf litter as a fast decomposer due to its thin leaves, we hypothesized that EBC would break down faster than native species (Collen, 1994; Grubbs & Cummins, 1994). We also hypothesized that because EBC is a novel resource in these streams and may differ in leaf litter quality, EBC would support fewer aquatic invertebrate shredders than native plants. This study will help us understand how EBC leaf litter is processed in streams relative to native plant species and to what extent aquatic invertebrate shredders utilize EBC leaf litter as

**Fig. 1** Map of leaf pack study sites in Campbell and Chester creeks, Anchorage, Alaska



a food source. The results from this study will provide insight into whether riparian EBC is disrupting key ecological processes in stream ecosystems of Alaska.

## Materials and methods

### Study area

This study was conducted in two streams in Anchorage, Alaska where EBC is currently spreading and displacing native species—Campbell and Chester creeks (Roon, 2011). Located in the south-central part of the state, Anchorage is characterized by a coastal climate considered to be a transitional zone between boreal and temperate rainforest biomes (Milner & Oswood, 2000). Originating in the Chugach Mountains, these streams flow west through the urbanized lowlands of the Anchorage bowl before ultimately reaching the saltwater of Cook Inlet. Greenbelt corridors of municipal parkland parallel the lower portions of both streams, keeping the riparian zones largely intact as they flow through the city (Fig. 1) (Ourso & Frenzel, 2003). Chester Creek is a second-order stream draining 78 km<sup>2</sup>, and Campbell Creek is a fourth-order

stream draining 158 km<sup>2</sup>. Riparian vegetation is comprised of mixed coniferous and deciduous forests dominated by spruce (*Picea* spp.), alder (*Alnus* spp.), black cottonwood (*Populus trichocarpa*), paper birch (*Betula neoalaskana*), and willow (*Salix* spp.). EBC is widespread along the lower, urbanized portions of these watersheds, forming monotypic stands in places (Roon, 2011).

### Study site characteristics

We measured selected physical habitat and stream chemistry parameters to characterize the study reaches. Total canopy cover and canopy composition were measured with a handheld densiometer. Discharge was measured monthly and calculated by multiplying stream channel width and depth with flow velocity measurements taken at each study site with a flow meter (Flo-Mate 2000, Marsh-McBirney Inc., Hach, CO). Stream pH and conductivity were taken monthly with a Hanna combination meter<sup>®</sup> (Hanna Instruments, Woonsocket, RI). Stream temperature was measured hourly with StowAway Tidbit Temperature data loggers<sup>®</sup> (Onset Computer Corporation, Pocasset, MA) to calculate mean temperature and total degree days accumulated throughout the experiments.

## Leaf pack experiments

We set up two complementary leaf pack experiments with slightly different designs, following the procedure outlined by Benfield (1996). The first experiment in 2009 compared EBC to three common native trees including thin-leaf alder (*A. tenuifolia* Nutt.), paper birch (*B. neoalaskana* Sarg.), and black cottonwood (*P. trichocarpa* Torr. & A.Gray) in a single urbanized stream reach in Chester Creek where EBC dominated the riparian forest canopy. Five replicate leaf packs of all four species were removed at four dates (after 2, 22, 43, and 63 days) for a total of 80 leaf packs.

The second experiment in 2010 compared EBC to native thin-leaf alder in four total stream reaches in Campbell and Chester creeks. Leaf packs were deployed in several stream reaches in the second experiment to determine if patterns observed in the first experiment were repeated at multiple locations subject to different local habitat conditions, such as possible differences in thermal and flow regimes, aquatic invertebrate communities, and levels of urbanization. We specifically compared EBC to thin-leaf alder because alder is the dominant riparian plant lining these streams (Ruess et al., 2009; Kruse et al., 2010) contributing the bulk of leaf litter inputs, and the native species most likely to be displaced by spreading EBC. Three replicate packs of each species were removed at four dates (after 2, 12, 27, 54 days) and four sites for a total of 96 packs.

For both experiments, senesced leaves were collected after abscission the previous autumn, and air-dried and stored in paper bags at room temperature. All leaf packs were constructed from 10-mm mesh (20 × 20 cm) filled with 4 g of senesced leaf litter from a single species. All leaf packs were deployed randomly in a riffle habitat of stream and tethered to the streambed. At each removal date, we removed random sets of leaf packs, stored them individually in Ziploc® bags, and refrigerated them until processed in the lab. We constructed an extra set ( $n = 5$ ) of “handling loss” packs for each species that were submerged in the water and then immediately removed to quantify leaf mass lost due to handling.

Once removed from the stream, leaf packs were processed immediately in the lab. Leaves were gently rinsed with water over a 250- $\mu$ m sieve to remove sediment and aquatic invertebrates. Leaves were then oven-dried for 24 h at 50°C and ashed for 2 h at 550°C

to calculate ash free dry mass (AFDM). AFDM was subtracted from the AFDM of the handling loss packs to calculate percent mass loss over time. All sediment and aquatic invertebrates collected in the 250- $\mu$ m sieve were stored in 90% ethanol in Whirlpaks® (Nasco, Fort Atkinson, WI) until later processed in the lab. Aquatic invertebrates were identified to family, enumerated and their length measured to estimate dry mass using published length-weight regressions (Benke et al., 1999). While other functional feeding groups of aquatic invertebrates colonized the leaf packs, we focused exclusively on shredding taxa (Merritt & Cummins, 1996).

## Statistical analysis

To calculate leaf litter breakdown rates of EBC and native riparian plants, we used an exponential decay model:  $W_t = W_0 e^{-kt}$ , where  $W_0$  is the initial AFDM (g),  $W_t$  is the final AFDM (g) at time  $t$ , and  $k$  is the decay coefficient (Peterson & Cummins, 1974; Benfield, 1996). We then solved for the breakdown rate ( $k$ ) where  $k$  is equal to the slope coefficient of the line when comparing the natural log of % AFDM remaining over time. To compare the breakdown rates between EBC and native plants, we used an ANCOVA model where leaf species was the categorical variable and time or degree day was the continuous variable used to predict breakdown rate ( $k$ ). In 2009, we compared EBC to three native species at one site, and in 2010, we compared EBC to thin-leaf alder at four sites. Because, the experiment only took place at single site in 2009, we used time as our continuous variable. However, to account for differences in stream temperature between sites in 2010, we compared breakdown rates by degree day (accumulated degrees above 0°C during the incubation period). Post-hoc  $t$  tests with a Bonferroni corrected  $P$  values ( $\alpha = 0.015$ ) were used in 2009 to compare individual native species to EBC. In 2010, because we only compared EBC to alder no post hoc tests were necessary ( $\alpha = 0.05$ ).

To compare differences in aquatic invertebrate shredders associated with EBC and native riparian plants in 2009, we used a 2-way ANOVA where leaf species and time were used to predict shredder abundance (number) and dry mass. Post-hoc  $t$ -tests with a Bonferroni corrected  $P$  values ( $\alpha = 0.015$ ) were used in 2009 to compare individual native

**Table 1** Site characteristics for study reaches in Campbell and Chester creeks in Anchorage, Alaska

Parameter	Stream and site				
	2009	2010			
	Chester Creek (urbanized)	Chester Creek (urbanized)	(reference)	Campbell Creek	
			(urbanized)	(reference)	
Elevation (m)	60	60	71	12	98
Canopy composition	B > EBC > A > S	B > EBC > A > S	S > A > B > EBC	A > B > EBC > S	A > B > C > S
Canopy cover (%)	92.5 (±2.6)	91.0 (±4.2)	89.6 (±8.0)	37.8 (±20.2)	74.0 (±12.4)
Stream width (m)	3.6 (±0.3)	3.6 (±0.3)	2.8 (±0.2)	11.5 (±1.1)	5.7 (±0.6)
Flow velocity (m <sup>2</sup> s <sup>-1</sup> )	0.75 (±0.07)	0.77 (±0.06)	0.37 (±0.05)	0.88 (±0.06)	0.61 (±0.05)
Discharge (m <sup>3</sup> s <sup>-1</sup> )	0.55 (±0.08)	0.51 (±0.05)	0.11 (±0.01)	2.43 (±0.32)	0.72 (±0.10)
pH	7.9 (±0.3)	7.7 (±0.3)	7.5 (±0.3)	8.0 (±0.2)	7.6 (±0.1)
Conductivity (µS cm <sup>-1</sup> )	236 (±27)	242 (±33)	273 (±21)	116 (±7)	148 (±13)
Temperature (°C)	13.3 (±0.1)	12.4 (±0.1)	6.2 (±0.0)	10.1 (±0.2)	8.7 (±0.2)
Degree days	854	672	333	545	470

Stream flow velocity, pH, and conductivity were measured at the initial deployment of leaf packs and each removal date ( $n = 5$ ) of the experiment. Stream temperature measurements were calculated hourly throughout the experiment. Degree days were calculated by accumulated temperature above 0°C through the experiment. All mean values are presented with (±1 SE)

Canopy composition species include A Alder, B Birch, C Cottonwood, EBC European bird cherry, S Spruce

species to EBC. In 2010, to account for differences between sites and streams, we modified our 2-way ANOVA model with a nested design, where leaf packs were nested within sites, and sites were nested within streams. In 2010, because we only compared EBC to alder no post hoc tests were necessary ( $\alpha = 0.05$ ). Equal variance and normality were checked with residual plots and boxplots. Shredder abundance and dry mass data initially violated the assumptions of equal variance and normality, but these data were able to meet these assumptions after applying a  $\log(x + 1)$  transformation. All data were analyzed with the statistical program R (R Development Core Team, version 2.10.1).

## Results

### Study site characteristics

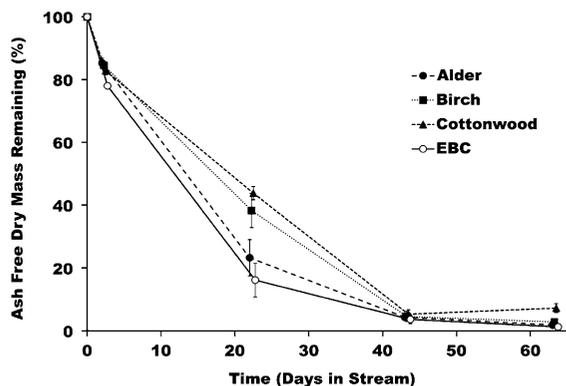
All sites with the exception of the Campbell Creek urbanized site were small streams less than 6 m in width, with less than 1 m<sup>3</sup> s<sup>-1</sup> discharge and greater than 50% canopy cover. EBC was present in the canopy at three of the four study sites, and was most abundant at the two urbanized sites along Campbell

Creek and Chester Creek (Table 1). Study sites varied substantially by stream temperature, with the downstream, urbanized sites averaging a higher temperature and accumulating more degree days than the reference sites further upstream in both Campbell and Chester creeks (Table 1). The Chester Creek urbanized site accumulated more degree days and averaged 1°C higher in 2009 than in 2010 (Table 1).

### Leaf litter breakdown

In 2009, all four species broke down quickly in the Chester Creek urbanized site, losing over 50% of their initial mass within the first 23 days and 90% of their mass after 43 days of being placed in the stream (Fig. 2). There was a significant effect of leaf species and time when comparing breakdown rates ( $k$ ) (ANCOVA; leaf species:  $P < 0.001$ ; time:  $P < 0.001$ ). EBC leaf litter broke down significantly faster than native birch and cottonwood (Table 2). EBC also broke down faster than alder, but not significantly so (Table 2).

In 2010, EBC broke down significantly faster than alder in two of the four sites. Breakdown rates ranged between 0.0036 and 0.0081  $k$  degree day<sup>-1</sup> for EBC and between 0.0027 and 0.0044  $k$  degree day<sup>-1</sup> for



**Fig. 2** Breakdown of EBC and native leaf litter in Chester Creek in 2009. Percent of ash free dry mass that was remaining over time for EBC and native alder birch and cottonwood leaf packs deployed in the Chester Creek urbanized site in 2009. Error bars represent  $\pm 1$  SE

alder. EBC broke down significantly faster than alder in the Chester Creek urbanized site (ANCOVA; degree day:  $P = 0.014$ ) and the Chester Creek reference site (ANCOVA; degree day:  $P = 0.009$ ) (Table 2). While EBC also broke down faster than alder in both sites in Campbell Creek, the differences were not significant in the urbanized site (ANCOVA; degree day:  $P = 0.106$ ) or the reference site (ANCOVA; degree day:  $P = 0.114$ ) (Table 2). EBC

leaf litter also broke down faster than alder leaf litter in urbanized sites in Chester and Campbell Creeks than in reference sites (Table 2).

While the Chester Creek urbanized site accumulated more degree days and averaged  $1^{\circ}\text{C}$  higher over the experiment in 2009 than in 2010, alder and EBC did not break down consistently faster. Alder leaf litter broke down at  $0.0048$   $k$  degree day $^{-1}$  in 2009 and at  $0.0042$   $k$  degree day $^{-1}$  in 2010. EBC leaf litter broke down at  $0.0052$  and  $0.0058$   $k$  degree day $^{-1}$  in 2010 (Fig. 3).

#### Aquatic invertebrate shredders

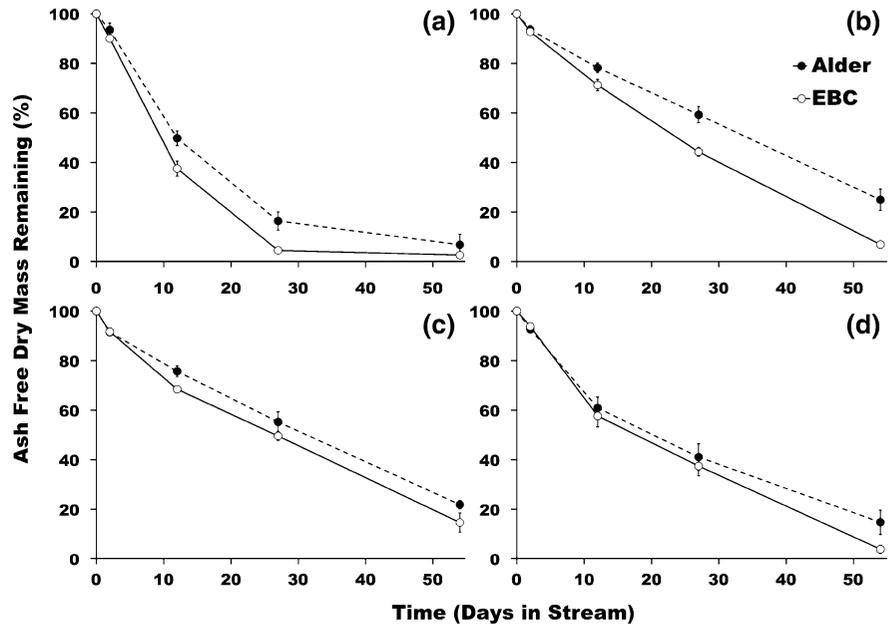
Aquatic invertebrate shredders were present, and community compositions similar, in both Campbell and Chester creeks, colonizing EBC and native leaf litter. In 2009, limnephilid caddisflies (which included a combination of *Ecclisocosmoecus* spp., *Ecclisomyia* spp., *Hesperophylax* spp., *Onocosmoecus* spp., and *Psychoglypha* spp.) were the most abundant taxa accounting for more than 66% of shredder abundance and 90% of shredder mass for all litter types except cottonwood (Table 3). Nemourid stoneflies (*Zapada* spp.) were the most abundant shredder by number associated with cottonwood leaf litter, accounting for 63% while only comprising between 23 and 32% of

**Table 2** Breakdown rates ( $k$ ) of EBC and native leaf litter in Chester and Campbell creeks, Alaska

Site	Leaf species	$k$ (degree day) $\pm 1$ SE	$r^2$	$P$ value vs. EBC
2009				
Chester Creek <i>Urbanized</i>	Alder	0.0048 ( $\pm 0.0002$ )	0.99	0.044
	Birch	0.0045 ( $\pm 0.0005$ )	0.96	<b>0.005</b>
	Cottonwood	0.0035 ( $\pm 0.0007$ )	0.87	<b>&lt;0.001</b>
	EBC	0.0052 ( $\pm 0.0001$ )	0.99	
2010				
Chester Creek <i>Urbanized</i>	Alder	0.0042 ( $\pm 0.0004$ )	0.97	<b>0.014</b>
	EBC	0.0059 ( $\pm 0.0011$ )	0.90	
Chester Creek <i>Reference</i>	Alder	0.0041 ( $\pm 0.0003$ )	0.98	<b>0.009</b>
	EBC	0.0080 ( $\pm 0.0010$ )	0.95	
Campbell Creek <i>Urbanized</i>	Alder	0.0027 ( $\pm 0.0004$ )	0.97	0.106
	EBC	0.0034 ( $\pm 0.0006$ )	0.96	
Campbell Creek <i>Reference</i>	Alder	0.0040 ( $\pm 0.0002$ )	0.99	0.114
	EBC	0.0067 ( $\pm 0.0009$ )	0.94	

Breakdown rates were calculated by degree day for both leaf pack experiments in Chester and Campbell creeks. Numbers in bold indicate that the differences between native species and EBC are statistically significant (2009: ANCOVA with Bonferroni correction,  $\alpha = 0.015$ , 2010: ANCOVA,  $\alpha = 0.05$ )

**Fig. 3** Breakdown of EBC and alder leaf litter in Campbell and Chester creeks in 2010. Percent of ash free dry mass that was remaining over time for EBC and native alder leaf packs deployed in **a** Chester Creek urbanized site, **b** Chester Creek reference site, **c** Campbell Creek urbanized site, and **d** Campbell Creek reference site in 2010. Error bars represent  $\pm 1$  SE



other litter types by number and less than 5% by mass (Table 3). Tipulid craneflies (*Tipula* spp.) were also present but only occurred on alder and birch litter (Table 3). In 2010, limnephilid caddisflies again were the dominant shredder taxa by number and mass for alder and EBC at all sites except the Campbell Creek reference site where nemourid stoneflies dominated (Table 3). Tipulid craneflies colonized both EBC and alder leaf litter in 2010 but did not account for much mass or numbers (Table 3).

In 2009, alder and birch leaf litter supported the highest numbers of shredders followed by cottonwood and EBC. While shredder abundance varied significantly by time, there was no significant difference between EBC and native species or species  $\times$  time interaction (2-way ANOVA; leaf species:  $P = 0.439$ ; time:  $P < 0.001$ ; species  $\times$  time interaction:  $P = 0.597$ ) (Fig. 4). Shredder mass did not differ significantly between EBC and native species (2-way ANOVA; species:  $P = 0.641$ ) or by time ( $P = 0.179$ ) (Fig. 4). Alder leaves supported the highest overall mass of shredders throughout the experiment, but this was not significantly greater than EBC ( $P = 0.731$ ). Nor did shredder mass differ significantly between EBC and birch ( $P = 0.932$ ) or cottonwood ( $P = 0.638$ ) (Fig. 4).

In 2010, overall shredder abundance varied significantly by stream, time, and sites nested within stream, but not by leaf species nested within site (nested 2-way

ANOVA: stream:  $P < 0.001$ ; time:  $P < 0.001$ ; sites nested within stream:  $P < 0.001$ , leaf species nested within sites:  $P = 0.064$  (Fig. 5). Shredder abundance on alder leaf litter deployed in the Chester Creek urbanized site was higher and marginally significant relative to EBC ( $P = 0.057$ ) (Fig. 5). Otherwise, shredder abundance did not differ significantly by leaf species at the remaining sites (Chester Creek reference  $P = 0.522$ ; Campbell Creek urbanized  $P = 0.176$ ; Campbell Creek reference  $P = 0.272$ ) (Fig. 5). Shredder mass varied significantly by time, sites nested within stream, but not by stream or leaf species nested within site (nested 2-way ANOVA: time:  $P < 0.001$ ; sites nested within stream:  $P < 0.001$ ; stream:  $P = 0.567$ ; or leaf species nested within site:  $P = 0.299$ ) (Fig. 5). Shredder mass did not differ significantly between leaf species at sites in Campbell Creek (urbanized site  $P = 0.296$ ; reference site  $P = 0.141$ ) or in Chester Creek (urbanized site  $P = 0.459$ ; reference site  $P = 0.342$ ) (Fig. 5).

## Discussion

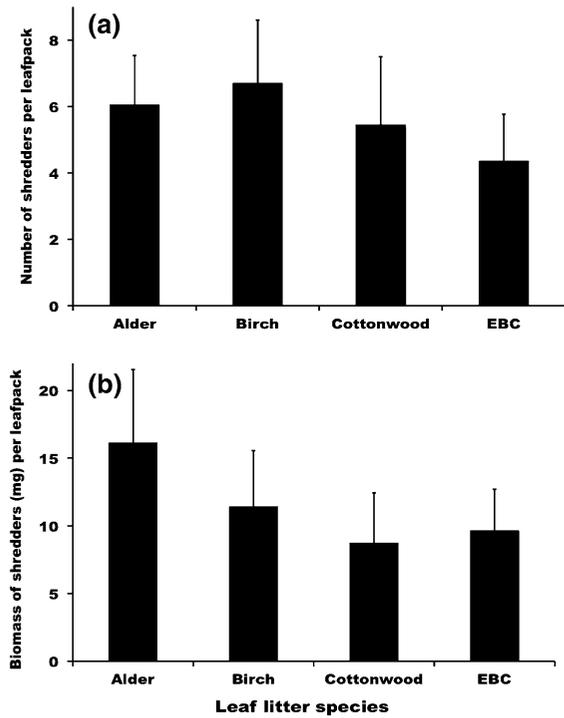
The spread of non-native plants in riparian forests can potentially affect stream ecosystems by changing the composition of leaf litter inputs (Hood & Naiman, 2000; Lecerf et al., 2007; Mineau et al., 2011). If leaf

**Table 3** Shredder community composition associated with EBC and native leaf litter in Campbell and Chester creeks

Stream	Leaf Species	Shredder taxa						Total	
		Limnephilidae		Nemouridae		Tipulidae		Dry mass	Number
		Number	Dry mass	Number	Dry mass	Number	Dry mass		
2009									
Chester	Alder	4.0 (±1.2)	15.8 (±5.4)	1.9 (±0.6)	0.2 (±0.1)	0.1 (±0.1)	0.2 (±0.2)	6.1 (±1.5)	16.1 (±5.4)
<i>Urbanized</i>	Birch	4.6 (±1.6)	10.5 (±4.2)	1.9 (±0.7)	0.2 (±0.1)	0.1 (±0.1)	0.7 (±0.8)	6.7 (±1.9)	11.4 (±4.1)
	Cottonwood	2.0 (±0.7)	8.4 (±3.7)	3.4 (±1.9)	0.3 (±0.2)	0	0	5.5 (±2.1)	8.7 (±3.7)
	EBC	3.3 (±1.2)	9.5 (±3.1)	1.0 (±0.4)	0.1 (±0.1)	0	0	4.4 (±1.4)	9.6 (±3.1)
2010									
Chester	Alder	10.9 (±4.2)	18.4 (±6.4)	3.3 (±0.8)	0.5 (±0.2)	0.1 (±0.1)	0.1 (±0.1)	<b>14.3 (±4.2)</b>	18.9 (±6.4)
<i>Urbanized</i>	EBC	5.4 (±1.6)	17.3 (±9.7)	1.7 (±0.6)	0.5 (±0.2)	0.3 (±0.2)	0.4 (±0.3)	<b>7.4 (±1.9)</b>	18.1 (±9.7)
Chester	Alder	6.2 (±0.9)	24.8 (±4.8)	0.1 (±0.1)	0.1 (±0.1)	0.2 (±0.1)	0.5 (±0.3)	6.6 (±0.9)	25.3 (±4.9)
<i>Reference</i>	EBC	6.0 (±1.0)	19.7 (±4.7)	0	0	0.1 (±0.1)	0.1 (±0.1)	6.1 (±1.1)	19.7 (±4.7)
Campbell	Alder	14.2 (±3.1)	30.1 (±5.9)	4.8 (±1.6)	0.8 (±0.4)	0.7 (±0.3)	0.8 (±0.4)	19.7 (±2.7)	31.7 (±5.8)
<i>Urbanized</i>	EBC	12.5 (±4.6)	29.6 (±8.9)	3.2 (±0.8)	0.8 (±0.3)	3.0 (±0.9)	3.1 (±1.1)	18.7 (±5.0)	33.4 (±8.9)
Campbell	Alder	1.5 (±0.5)	6.4 (±2.3)	10.4 (±2.8)	3.6 (±0.8)	0.2 (±0.2)	0.1 (±0.1)	12.2 (±3.1)	10.2 (±2.8)
<i>Reference</i>	EBC	0.8 (±0.3)	2.3 (±1.4)	9.2 (±3.1)	3.5 (±1.2)	0.2 (±0.1)	0.1 (±0.1)	10.2 (±3.2)	5.9 (±2.1)

Mean abundance (number) and dry mass (mg) ± 1 SE of shredder taxa present in leaf packs in 2009 and 2010

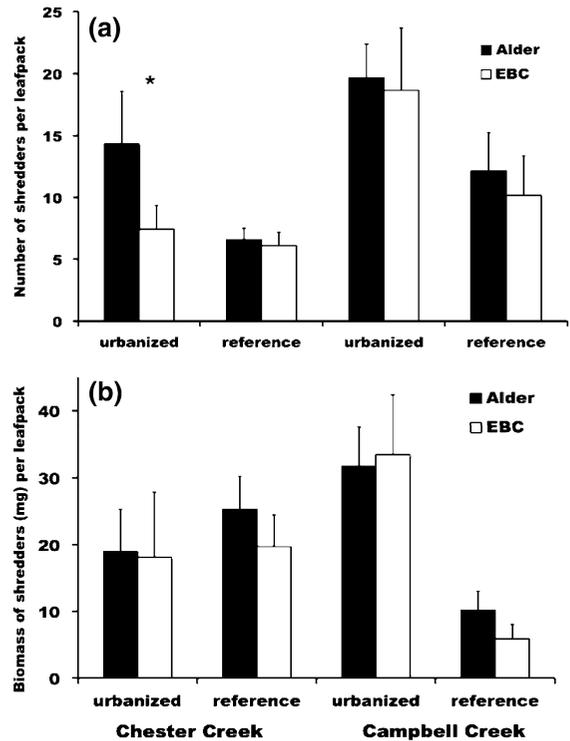
Numbers in bold indicate that the differences between native species and EBC are marginally significant (nested 2-way ANOVA:  $P = 0.057$ )



**Fig. 4** Abundance and biomass of aquatic invertebrate shredders in 2009. Mean abundance (a) and dry mass (b) of shredders that were associated with EBC and native alder, birch, and cottonwood leaf packs in the Chester Creek urbanized site in 2009. Error bars represent + 1 SE

litter processing by aquatic invertebrate shredders is disrupted by a non-native plant species, then impacts could be expressed through multiple trophic levels, ultimately impacting ecosystem function of streams (Hood & Naiman, 2000; Gessner & Chauvet, 2002; Baxter et al., 2005). In our leaf pack experiments conducted in urban Alaskan streams, we found that invasive European bird cherry leaf litter supported similar aquatic invertebrate shredder communities to the leaf litter of native species despite variable breakdown rates between study sites.

EBC leaf litter broke down at different rates compared to the leaf litter of native species between experiments and between our Anchorage study streams. In 2009, EBC leaf litter broke down significantly faster than native birch and cottonwood and a similar rate to native alder. In 2010, EBC leaf litter broke down significantly faster than native alder in both Chester Creek study sites. However, while EBC leaf litter continued to break down at a faster rate than alder in the Campbell Creek sites, the differences were



**Fig. 5** Abundance and biomass of aquatic invertebrate shredders in 2010. Mean abundance (a) and dry mass (b) of shredders that were associated with EBC and native alder leaf packs in Campbell and Chester creek in reference and urbanized sites in 2010. Error bars represent +1SE. \*Differences between native species and EBC are marginally significant (nested 2-way ANOVA:  $P = 0.057$ )

not significant. These results partially support our hypothesis that EBC leaf litter would break down faster than the leaf litter of native species.

It is unclear why EBC leaf litter broke down at variable rates between study streams, but considering that leaf litter breakdown is the result of both physical and biological processes; it is likely due to a combination of factors. First, EBC leaves were thinner than all other native species tested, suggesting it would break down consistently faster than native species. Leaf toughness has been documented as an important physical characteristic that influences breakdown rates (Irons et al., 1988; Motomori et al., 2001). Our results are supported by other studies finding that *Prunus* leaf litter breaks down quickly in streams. Cummins et al. (1989) and Grubbs and Cummins (1994) documented *Prunus* leaf litter to break down quickly in streams and categorized *Prunus* species as “fast” decomposers. A

study by Collen (1994) found EBC leaf litter broke down faster than alder and birch leaves in Scottish streams.

Stream temperature is an often-highlighted factor in leaf litter studies that can influence breakdown rates (Peterson & Cummins, 1974; Webster & Benfield, 1986; Cummins et al., 1989). Although thermal regimes varied considerably between study sites, we observed no consistent effect of temperature on leaf litter breakdown in our experiments. EBC leaf litter broke down significantly faster than native species in both Chester Creek sites, which supported the warmest (Chester Creek urbanized site, 672 degree days) and the coolest (Chester Creek reference site, 333 degree days) study sites (Table 1). Stream temperature in the Campbell Creek sites were in between those of the Chester Creek sites, yet supported breakdown rates inconsistent with the patterns observed in Chester Creek.

Another interesting pattern emerged when comparing the influence of stream temperature on the breakdown rates of EBC and native alder leaf litter deployed in the Chester Creek urbanized site between 2009 and 2010. In 2009, stream temperatures were warmer with a total of 854 degree days accumulated over the duration of the experiment, while in 2010 stream temperatures were cooler with a total of 672 degree days accumulated (Table 2). If temperature was the only factor that influenced leaf litter breakdown rates, then one would expect that leaf litter breakdown rates would be consistently faster in 2009 than in 2010 for both invasive EBC and native alder. However, while leaf litter breakdown rates were faster for both species in 2009, the difference in rates between years was more pronounced for alder than invasive EBC (Table 2). This suggests that other factors may have influenced the faster breakdown rate of alder leaf litter at warmer temperatures such as increased microbial colonization (Peterson & Cummins, 1974; Webster & Benfield, 1986; Irons et al., 1994). Irons et al. (1994) and Rowe et al. (1996) support this, finding that stream temperature alone was not enough to predict leaf litter breakdown rates and suggest other factors may be responsible.

Another factor that could help explain why EBC leaf litter breakdown varied between study systems could be due to the increased flow in Campbell Creek, a fourth-order stream. In theory, one would expect physical fragmentation to increase with higher stream

flow, resulting in faster leaf litter breakdown rates (Webster & Benfield, 1986). However, we observed the opposite in our leaf pack experiments, where the site with the highest stream flow (Campbell Creek urbanized site, mean discharge = 2.43 m<sup>3</sup>/s) actually supported the slowest breakdown rates for both EBC and alder. One explanation could be that while the increased flow may have heightened the physical abrasion that the leaf litter of both species was subjected to, it also muted the biological factors responsible for the differences in breakdown rates observed in Chester Creek (Peterson & Cummins, 1974; Webster & Benfield, 1986; Ferreira et al., 2006; Hoover et al., 2006). Our results are supported by Baldy et al. (1995) who found that leaf litter breakdown decreased in main-stem habitats with higher stream flow than in headwater habitats due to decreased biological activity of aquatic bacteria and fungi responsible for conditioning leaf litter.

Other studies examining the behavior of non-native leaf litter in streams often found non-native leaf litter broke down at a slower rate compared to native species (Sampaio et al., 2001; Kennedy & Hobbie, 2004; Braatne et al., 2007; Lecerf et al., 2007; Going & Dudley, 2008; Mineau et al., 2012). This is sometimes due to the presence of recalcitrant compounds (i.e., lignin and tannins) that can slow both physical fragmentation and leaf litter conditioning by aquatic bacteria and fungi (Webster & Benfield, 1986; Graça, 2001; Lecerf et al., 2007; Mineau et al., 2012). Although it is likely that EBC differed in leaf litter chemistry and quality compared to native species, especially a nitrogen-rich species like alder, we observed no decrease in leaf litter breakdown rate. Instead, the fact that EBC broke down at a similar or faster rate than native species suggests that EBC leaf litter is a readily available source of organic matter in these Anchorage streams.

Aquatic invertebrate shredders utilized EBC leaf litter in both experiments, colonizing leaf packs deployed in Campbell and Chester creeks. EBC leaf litter supported similar shredder communities relative to native species, not supporting our hypothesis. We observed that shredders in EBC leaf packs did not differ significantly by abundance or mass compared to the leaf packs of native species of leaf litter. Collectively, these data indicate that aquatic invertebrate shredders utilized EBC leaf litter as a food source equally to native species in these streams.

While our results do not support our hypothesis that shredders would be less likely to utilize EBC leaf litter inputs than native species, they are supported by previous studies examining *Prunus* leaf litter. Grubbs and Cummins (1994) found that shredders in Midwestern streams were abundant in *P. serotina* leaf packs. Another study found shredders colonized *P. padus* leaf litter at similar rates to alder and birch leaf litter in Scottish streams (Collen, 1994). A limiting factor of these studies is that they consider shredder utilization of *Prunus* leaf litter in the context of its native habitat (Thompson & Townsend, 2003). However, our results are also supported by a follow up field experiment conducted in Anchorage streams finding no significant differences in survival or growth of *Psychoglypha* caddisflies when fed EBC leaf litter compared to mixed native/EBC leaf litter and exclusively native (alder and birch) leaf litter (Merrigan, 2011). These data complement our findings suggesting that shredders can survive and grow on EBC leaf litter at similar levels to the leaf litter of native species. It is still unknown whether, if given a choice, shredders would feed preferentially on EBC leaf litter.

Many studies have cited concerns that differences in leaf chemistry of non-native plant species can negatively affect leaf litter processing by shredders (Abelho & Graça, 1996; Sampaio et al., 2001; Braatne et al., 2007; Lecerf et al., 2007; Going & Dudley, 2008). While we did not analyze leaf chemistry in our study, we can nevertheless assess the suitability of EBC as a food source by looking at the functional response of shredders. In contrast to these studies, our data showed shredders fed on EBC leaf litter at similar levels to native species suggesting that any potential difference in leaf chemistry did not appear to have any significant effect on shredder communities. It is possible that the rapid breakdown rate of EBC resulted in leaf litter that conditioned quickly in streams and provided a readily available food source for stream shredders. Our results align with the observations from Cummins et al. (1989) that shredders were less concerned with individual species of leaf litter than the state of leaf litter conditioning (Graça, 2001; Sampaio et al., 2001; Braatne et al., 2007; Lacan et al., 2010).

Although we detected no substantial ecological effects of EBC on leaf litter processing in these streams, the spread of EBC in riparian forests could have other ecological consequences we did not evaluate. The shift from native trees to EBC in

riparian forests may alter the timing of leaf litter inputs entering streams. We observed EBC held onto its leaves later than other deciduous trees, delaying when leaf litter inputs enter the stream channel (Roon, personal observation). In addition to the delayed timing of inputs, the rapid breakdown of EBC leaf litter entering the stream at a single time could result in a pulse of organic matter that disappears quickly. Moline & Poff (2008) highlighted leaf litter retention as a concern with tamarisk in Colorado streams. If the phenology of leaf litter inputs does not match shredder life histories, then a “trophic mismatch” could result that reduces shredder abundance and affect litter processing in streams lined by riparian EBC (Strayer, 2010).

As EBC continues to spread in Alaska’s riparian forests, it may also homogenize the diversity of leaf litter inputs available to shredders. Especially in streams lined by mixed deciduous forests, such as those of southcentral Alaska, shredders receive a wide range of leaf litter inputs that enter the stream channel at different times and break down at different rates, providing shredders with what Peterson & Cummins (1974) referred to as a “processing continuum” of resources. This “processing continuum” of leaf litter inputs for shredders can be altered by the homogenization of these inputs (Peterson & Cummins, 1974). A study by Jonsson et al. (2001) found that shredder diversity in boreal streams was important for leaf litter processing. This supports the hypothesis made by Dangles et al. (2003) that invasive plants (in this case Japanese knotweed) can reduce the diversity of shredders and the shredders’ ability to process leaf litter in streams. Because shredders support other functional feeding groups of aquatic invertebrates, which collectively act as major prey sources for upper trophic level consumers, including stream fishes, the loss in shredder diversity could have detrimental effects on the ecosystem function of streams (Cummins et al., 1989; Gessner & Chauvet, 2002; Baxter et al., 2005).

The shift in riparian vegetation from alder to EBC could affect stream food webs. Alder is an important source of nitrogen for stream and riparian ecosystems (Helfield & Naiman, 2002). Much research in Alaska has documented the value of alder for stream productivity, nutrient cycling, and as a preferred food source for shredders and other aquatic invertebrates (Irons et al., 1988; Piccolo & Wipfli, 2002; Orlikowska et al.,

2004). Thus, the shift from alder to EBC in riparian forests could have perpetuating direct and indirect ecological effects in streams by decreasing the amount of nitrogen available for nutrient cycling or by changing the composition of leaf litter inputs entering Alaskan streams (Lecerf et al., 2005; Strayer, 2010).

As invasive species like EBC continue to spread across landscapes, it is important to understand the ecological interactions and effects on native species in their new environs (Strayer, 2010). This is especially true in riparian forests where the spread of invasive plants has the potential to affect both terrestrial and aquatic ecosystems, as well as the exchange of resources between the two (Baxter et al., 2005). It is difficult to predict which invasive species will be problematic, as previous studies have found varying and even conflicting results sometimes within the same species (Abelho & Graça, 1996; Lacan et al., 2010). Our study found that while EBC leaf litter breakdown rates varied between watersheds, EBC leaf litter supported similar aquatic invertebrate shredder communities to native species. Collectively, these data suggest that the spread of EBC in riparian forests is not currently exhibiting strong negative impacts on these ecological processes in urban Alaskan streams, but could if it continues to spread and further displaces native species over time.

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