

Potential population and assemblage influences of non-native trout on native nongame fish in Nebraska headwater streams

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Accepted for publication September 17, 2014

Abstract – Non-native trout are currently stocked to support recreational fisheries in headwater streams throughout Nebraska. The influence of non-native trout introductions on native fish populations and their role in structuring fish assemblages in these systems is unknown. The objectives of this study were to determine (i) if the size structure or relative abundance of native fish differs in the presence and absence of non-native trout, (ii) if native fish-assemblage structure differs in the presence and absence of non-native trout and (iii) if native fish-assemblage structure differs across a gradient in abundances of non-native trout. Longnose dace *Rhinichthys cataractae* were larger in the presence of brown trout *Salmo trutta* and smaller in the presence of rainbow trout *Oncorhynchus mykiss* compared to sites without trout. There was also a greater proportion of larger white suckers *Catostomus commersonii* in the presence of brown trout. Creek chub *Semotilus atromaculatus* and fathead minnow *Pimephales promelas* size structures were similar in the presence and absence of trout. Relative abundances of longnose dace, white sucker, creek chub and fathead minnow were similar in the presence and absence of trout, but there was greater distinction in native fish-assemblage structure between sites with trout compared to sites without trout as trout abundances increased. These results suggest increased risk to native fish assemblages in sites with high abundances of trout. However, more research is needed to determine the role of non-native trout in structuring native fish assemblages in streams, and the mechanisms through which introduced trout may influence native fish populations.

Key words: non-native trout; longnose dace; assemblage structure; size structure; white sucker

Introduction

Stream fish assemblages are formed as a result of abiotic (*e.g.*, temperature and flow regime) and biotic pressures (*e.g.*, predation and competition; Grossman & Sabo 2010). The relative importance of these pressures in structuring fish assemblages is frequently debated (Ross 1991; Grossman & Sabo 2010), and pressures often change through time due to anthropogenic activities such as stream impoundment, habitat alteration and fish stockings. Non-native species in particular can substantially alter fish assemblages (Ross 1991) and are considered one of the major

causes of widespread declines in freshwater biodiversity (Dudgeon et al. 2006).

Salmonids are currently one of the most widespread and frequently stocked non-native families in the world (Krueger & May 1991; Welcomme 1992). Introduced salmonids have directly and indirectly influenced native populations (*e.g.*, abundance, distribution, life history and size structure) and communities (*e.g.*, composition and structure) in several systems via competition, predation, hybridisation, and disease and pathogen transfer (Turek et al. 2013). However, most studies examining the influences of non-native salmonids on native species have

focused on salmonid influences on native game fish and their supporting communities. This is unfortunate because an understanding of the ecological risk to native nongame fish is equally important given that many native threatened and endangered species fall into the nongame realm. Anticipating potential interactions between introduced non-native salmonids and native species will help ensure conservation of these native species.

Non-native trout are currently stocked in several headwater streams throughout Nebraska to support recreational fisheries. Although the majority of non-native trout found in Nebraska streams are stocked fish, there is also a fair amount of natural reproduction in some streams. Recent reductions in abundance and distribution of some native species, along with numerous examples of negative predatory and competitive interactions between non-native trout and native species in other systems (*e.g.*, Turek et al. 2013), have prompted resource managers to prohibit stocking of trout in streams containing at-risk native fishes. However, the influence of non-native trout introductions on native fish populations and their role in structuring fish assemblages in these systems has not been examined. The relatively simple fish-assemblage structure of headwater streams, along with the lack of a widespread native predatory fish species, may exacerbate the impact of non-native trout on native species in these systems. Identifying what factors may influence native fish assemblages will assist in efforts to conserve at-risk native species.

Several studies have attributed declines in abundance of native fishes to the introduction of

salmonids (*e.g.*, Townsend & Crowl 1991; Rinne & Alexander 1995; Arismendi et al. 2009) suggesting that introductions of non-native trout into Nebraska may result in population-level changes to native fishes. Therefore, the first objective of this study was to determine whether the size structure or relative abundance of native species differs in the presence and absence of non-native trout. Changes in the size structure and relative abundance of native populations may also result in shifts in native fish assemblage. Therefore, the second objective of this study was to determine whether native fish-assemblage structure differs in the presence and absence of non-native trout. Further, influences of non-native trout on native fish assemblages are likely dependent on non-native trout abundance. Therefore, the third objective of this study was to determine whether native fish-assemblage structure differs across a gradient in abundances of non-native trout in headwater streams throughout Nebraska.

Materials and methods

Study area

Fish assemblages and habitat attributes were surveyed at 56 headwater stream sites throughout northern and western Nebraska from 29 June 2011 to 17 August 2011 (Fig. 1). These low gradient, headwater streams are typically groundwater fed with sandy substrate, herbaceous riparian vegetation and little in-stream habitat complexity. Sites were located mainly within three EPA defined Level III Ecoregions: Western

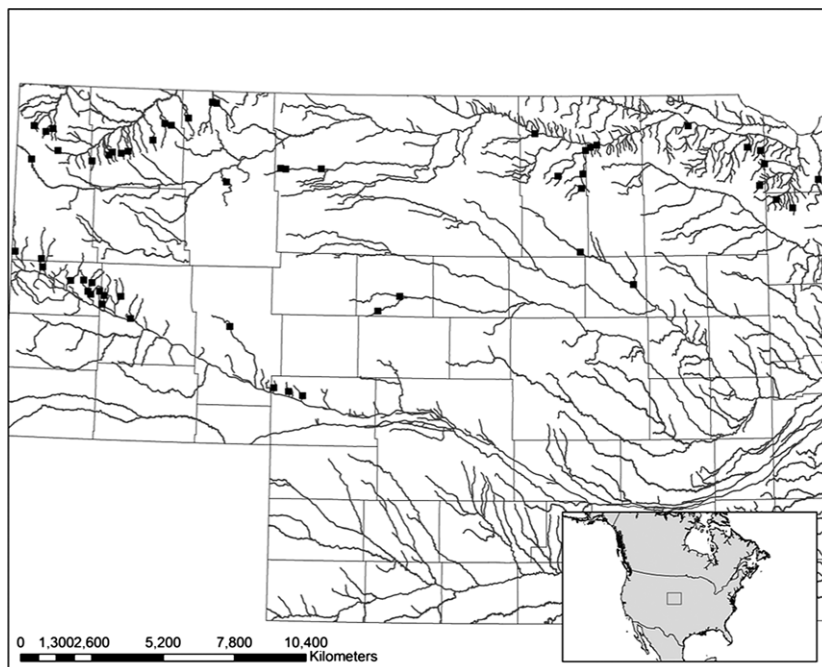


Fig. 1. Location of study sites sampled in 2011.

High Plains, Nebraska Sandhills and North-western Glaciated Plains (Chapman et al. 2001). All three Ecoregions are characterised by semi-arid to arid climates, and short-grass or mixed-grass prairie. Dominant land uses include cropland agriculture and rangeland (Chapman et al. 2001). Native fish assemblages include catostomids (*e.g.*, *Catostomus* spp. and *Moxostoma* spp.), centrarchids (*e.g.*, *Lepomis* spp. and *Micropterus* spp.), cyprinids (*e.g.*, *Rhinichthys* spp., *Semotilus* spp., and *Notropis* spp.), ictalurids (*e.g.*, *Ameiurus* spp. and *Noturus* spp.) and percids (*e.g.*, *Etheostoma* spp.). Non-native species include brook trout *Salvelinus fontinalis*, brown trout *Salmo trutta* and rainbow trout *Oncorhynchus mykiss*.

Sampling methods

Fish were collected using a pulsed DC, backpack electrofisher with one dip netter following Nebraska Game and Parks Commission stream survey methods. Output voltage settings ranged from 150 to 350 V and frequency settings from 40 to 80 Hz. Sites were sampled for a minimum of two 600 s intervals of effort. Additional 600 s intervals were sampled until no new species were collected in an interval for that site. All fishes were identified to species. Total lengths (mm) of a random subsample (20 fish) of each species were recorded, and all identified fishes were released. Unidentified specimens were kept, preserved in 10% formalin, and later identified.

Habitat (including substrate, fish cover and riparian cover) and human-use characteristics were also measured and recorded because these characteristics likely alter the magnitude and nature of non-native and native species interactions. Habitat and human-use characteristics were measured according to Nebraska Game and Parks Commission stream survey methods (adapted from EPA Wadeable Streams Assessment Protocol; Steve Schainost, NGPC). Dominant substrate (>50% of substrate) was classified based on particle size (*e.g.*, sand = 0.062–0.25 mm, coarse sand = 0.25–2.0 mm, fine gravel = 2.0–8.0 mm; Wentworth Scale). Fish cover categories (*e.g.*, filamentous algae, macrophytes, and woody debris) were classified categorically based on type and per cent cover [0 – none; 1 – sparse (1–10%); 2 – moderate (11–40%); 3 – heavy (41–75%); 4 – very heavy (>75%)]. Riparian cover (*e.g.*, trees, grasses, herbs and forbs) was estimated for a 10 m × 10 m area on each bank and was classified based on type and per cent cover. Presence of human use (*e.g.*, parks, landfill, and mining activity) within the riparian zone or in close proximity to the riparian zone (Kaufman & Robinson 1998) was categorised based on type and proximity to stream (0 – none; 1 – on bank; 2 – within 10 m; 3 – >10 m).

Native fish populations in the presence and absence of non-native trout

Size structures, relative abundance, and habitat and human-use characteristics were examined between (i) sites where any trout species or a combination of trout species were present and sites where no trout species was present, (ii) sites where brown trout was the only trout species present and sites where no trout species was present, (iii) sites where rainbow trout was the only trout species present and sites where no trout species was present and (iv) sites where brook trout was the only trout species present and sites where no trout species was present. Size structures of native species were compared between sites with and without trout using Kolmogorov–Smirnov two-sample tests (PROC NPARIWAY, SAS v.9.2, Cary, NC, USA) after pooling length data across all sites with trout (within each category listed above) and all sites without trout. Lengths were pooled across sites because sample size within individual sites was not sufficient to evaluate differences in size structure (only 20 fish were measured at each site). Relative abundances of native species (number of fish per second) were compared between sites with and without trout using linear models (PROC GLIMMIX, SAS v.9.2). Relative abundances were assessed using electrofishing time as the measure of effort, which admittedly may have contributed to variation in the area of sample sites. However, all surveys were conducted by the same crew, streams were similar in size and habitat complexity, and qualitatively similar-sized areas were sampled at each site. Differences in habitat and human-use characteristics between sites with and without trout were compared using one-way analysis of similarity (similarity matrix based on Euclidean distance, ANOSIM procedure, Primer-E v.6, Plymouth, UK). Emphasis was placed on Global *R* values to determine differences for all ANOSIM, as suggested by Clarke & Gorley (2006), instead of *P*-values because sample sizes are known to bias *P*-values. Global *R* values close to zero indicate minimal differences among groups, whereas values close to one indicate complete separation among groups. In general, it is accepted that groups are similar if Global *R* values are <0.20 (Clarke & Gorley 2006), and are different if Global *R* values are >0.40 (Clarke & Warwick 2001).

Native fish-assemblage structure in the presence and absence of non-native trout

Fish-assemblage assessments were made using only native species collected during stream surveys (*i.e.*, excluding non-native trout). Non-native trout were removed from the analysis because the addition of

non-native trout may inherently change the fish-assemblage structure in sites where trout are introduced compared to sites where trout are not introduced (*i.e.*, the addition of a new species will, in itself, change the structure). If the addition of non-native trout is having a real effect on the structure of the native fish assemblage, removing non-native trout from the analyses should result in a change in the remaining native fish assemblage between sites where trout were present and sites where trout were not present (*i.e.*, the change in assemblage will not be due solely to the addition of new species).

Differences in native fish-assemblage structure (relative abundance, fourth-root transformed, Bray–Curtis similarity matrix) between sites with and without trout were compared using one-way analysis of similarity (ANOSIM procedure, Primer-E v.6). Relative abundances were fourth-root transformed to down-weight highly abundant species and allow more rare species to exert some influence on the calculation of similarity (Clarke & Warwick 2001). Fish-assemblage structure between sites with and without trout was also visually assessed using NMDS (MDS procedure, Primer-E v.6). Differences in fish-assemblage structure between sites with and without trout nested within basins and Ecoregions were also compared independently using two-way nested analysis of similarity (ANOSIM procedure, Primer-E v.6) to account for more variation in assemblages between these geographic regions. The extent to which habitat and human-use characteristics (similarity matrix based on Euclidean distance) explained fish-assemblage structure was analysed using Mantel tests (BEST procedure, Primer-E v.6).

Native fish-assemblage structure at differing abundances of non-native trout

The number of trout collected at each site was expected to vary widely, and the assumption that widely varying abundances have the same influence on native fish assemblages is not realistic. To test the effect of trout abundance on native fish-assemblage structure, sites with trout were stratified into five subsets with similar trout abundances. These subsets were based on the following percentiles when ranked by abundance: ≤ 20 th percentile (CPUE ≤ 0.25 trout·min⁻¹), 21–40th percentiles (CPUE = 0.31–0.48 trout·min⁻¹), 41–60th percentiles (CPUE = 0.50–0.98 trout·min⁻¹), 61–80th percentiles (CPUE = 1.22–1.73 trout·min⁻¹), and ≥ 81 st percentile (CPUE ≥ 1.97 trout·min⁻¹). These subsets were chosen to allow for a balance between number of subsets ($N = 5$) and number of sites within each subset ($N = 7$). Using a bootstrapping approach (1000 iterations, with replacement), native fish-assemblage

structure (relative abundance, fourth-root transformed, Bray–Curtis similarity matrix) was compared between sites with trout (each subset) and a randomly chosen set of sites without trout ($N = 7$) using one-way analyses of similarity (ANOSIM function, vegan library, R v.2.15.1). Global R values (mean and 95% CI) from the analyses of similarities (1000 iterations) were then compared across the 5 subsets of trout CPUEs to determine whether greater abundances of trout had a greater influence on native fish-assemblage structure.

Habitat differences between subsets of sites with trout may result in differences in native fish-assemblage structure between subsets regardless of trout abundances. Therefore, differences in habitat and human-use characteristics (similarity matrix based on Euclidean distance) between the five subsets of sites with trout were compared using one-way analysis of similarity (ANOSIM procedure, Primer-E v.6). Differences in size structure of non-native trout may also result in differences in native fish-assemblage structure between subsets. Therefore, differences in size structure of non-native trout were compared visually using relative length-frequency histograms.

Results

Native fish populations in the presence and absence of non-native trout

Of the 56 sites surveyed during 2011, no fish were collected in two sites and non-native trout were collected in 35 sites. Additionally, there were 19 sites where native fish were collected but non-native trout were not collected. Only four native species (*i.e.*, creek chub *Semotilus atromaculatus*, fathead minnow *Pimephales promelas*, longnose dace *Rhinichthys cataractae* and white sucker *Catostomus commersonii*) were collected in sufficient numbers ($N \geq 30$ in sites with and without trout) to assess potential differences in size structures to trout presence. When sample size was sufficient to make comparisons, the size structures of creek chub and fathead minnow were similar between sites with and without trout (Table 1).

A total of 2082 longnose dace was collected at 33 sites in 10 watersheds during 2011. Longnose dace were larger in sites where brown trout was the only trout species present, and sites where any trout species or a combination of trout species were present compared to sites where trout were not present (Table 1). Longnose dace were smaller in sites where rainbow trout was the only trout species present compared to sites where trout were not present (Table 1). The size structure of longnose dace was similar in sites where brook trout was the only trout species

Table 1. Tests of differences in size structures of longnose dace (LND), white sucker (CWS), creek chub (CCH) and fathead minnow (FHM) between sites where any trout species or a combination of trout species were present (All Trout), sites where brown trout was the only trout species present (BNT only), and sites where rainbow trout was the only trout species present (RBT only) compared to sites where no trout were present (No Trout; N_Y = sample size in sites with trout, N_N = sample size in sites without trout, \bar{X}_Y = mean length in sites with trout, \bar{X}_N = mean length in sites without trout, $\bar{X}_{diff} = \bar{X}_Y - \bar{X}_N$, Med_Y = median length in sites with trout, Med_N = median length in sites without trout, $Med_{diff} = Med_Y - Med_N$, KS_a = asymptotic Kolmogorov–Smirnov statistic, P -value = asymptotic P -value of KS_a).

Comparison	Species	N_Y	N_N	\bar{X}_Y	\bar{X}_N	\bar{X}_{diff}	Med_Y	Med_N	Med_{diff}	KS_a	P -value
All Trout vs. No Trout	LND	222	233	71	67	4	73	66	7	1.54	0.02
	CWS	179	185	208	149	59	200	142	58	2.82	<0.01
	CCH	122	197	99	98	1	93	96	-3	1.07	0.20
	FHM	43	184	52	51	1	54	52	2	0.97	0.30
BNT only vs. No Trout	LND	117	233	74	67	7	74	66	8	1.68	0.01
	CWS	51	185	230	149	81	265	142	123	2.88	<0.01
	CCH	69	197	96	98	-2	90	96	-6	0.95	0.32
RBT only vs. No Trout	LND	40	233	55	67	-12	55	66	-11	1.83	<0.01
	CWS	37	185	124	149	-25	112	142	-30	1.02	0.25

present compared to sites where trout were not present ($KS_a = 0.92$, $P = 0.36$).

A total of 682 white suckers was collected at 34 sites in 10 watersheds during 2011. There was a greater proportion of larger white suckers in sites where brown trout was the only trout species present, and sites where any trout species or a combination of trout species were present compared to sites where trout were not present (Table 1). The size structure of white suckers was similar between sites where rainbow trout was the only trout species present and sites where trout were not present (Table 1). No white suckers were found in the presence of brook trout.

Relative abundances of longnose dace, white suckers, creek chub and fathead minnow did not differ in the presence or absence of trout (Table 2). Habitat and human-use characteristics did not differ between

sites where brown trout was the only trout species present (Global R : 0.103, $P = 0.003$), sites where any trout species or a combination of trout species were present (Global R : 0.083, $P = 0.052$), and sites where rainbow trout was the only trout species present (Global R : 0.010, $P = 0.405$) compared to sites where trout were not present. Sites exhibited low variation overall in habitat and human-use characteristics (most likely due to the broad categorical nature of our classifications) and the majority had sandy substrate, herbaceous riparian vegetation and little in-stream habitat complexity.

Native fish-assemblage structure in the presence and absence of non-native trout

Native fish-assemblage structure did not differ between sites with and without trout (Global R :

Table 2. Tests of differences in relative abundance (PROC GLIMMIX, SAS v.9.2) of select native species in sites where any trout species or a combination of trout species were present (ATP), sites where brook trout was the only trout species present (BKT), sites where brown trout was the only trout species present (BNT), and sites where rainbow trout was the only trout species present (RBT) compared to sites where no trout species was present ($CPUE \pm SE_n$ = mean catch per unit effort (number per second) \pm standard error in sites without trout, $CPUE SE_y$ = mean catch per unit effort (number per second) \pm standard error in sites with trout, Num d.f. = numerator degrees of freedom, Den d.f. = denominator degrees of freedom).

Native species	Trout species	$CPUE$ (number per second) $\pm SE_n$	$CPUE$ (number per second) $\pm SE_y$	Num d.f.	Den d.f.	F -value	P -value
Creek chub	ATP	0.030 \pm 0.021	0.002 \pm 0.008	1	51	1.92	0.17
	BKT	0.020 \pm 0.006	0.012 \pm 0.022	1	51	0.17	0.68
	BNT	0.015 \pm 0.010	0.017 \pm 0.018	1	51	0.02	0.88
	RBT	0.019 \pm 0.009	0.012 \pm 0.018	1	51	0.24	0.62
Fathead minnow	ATP	0.056 \pm 0.053	0.007 \pm 0.019	1	51	0.94	0.34
	BKT	0.021 \pm 0.015	0.042 \pm 0.054	1	51	0.19	0.66
	BNT	0.024 \pm 0.024	0.038 \pm 0.046	1	51	0.13	0.72
	RBT	0.031 \pm 0.023	0.032 \pm 0.044	1	51	0.00	0.98
Longnose dace	ATP	0.029 \pm 0.046	0.006 \pm 0.016	1	51	0.28	0.60
	BKT	0.022 \pm 0.013	0.013 \pm 0.046	1	51	0.05	0.82
	BNT	0.021 \pm 0.021	0.014 \pm 0.039	1	51	0.04	0.84
	RBT	0.024 \pm 0.020	0.011 \pm 0.038	1	51	0.21	0.65
White sucker	ATP	0.003 \pm 0.007	0.003 \pm 0.003	1	51	0.00	0.94
	BKT	0.007 \pm 0.002	0.001 \pm 0.008	1	51	1.33	0.25
	BNT	0.005 \pm 0.003	0.001 \pm 0.006	1	51	0.52	0.47
	RBT	0.003 \pm 0.003	0.003 \pm 0.006	1	51	0.02	0.90

0.015, $P = 0.343$), between sites with and without trout nested within basins (Global R : 0.315, $P = 0.171$) or between sites with and without trout nested within Ecoregions (Global R : -0.031 , $P = 0.600$). Habitat and human-use characteristics had weak correlations to native fish-assemblage structure (highest correlation: 0.241).

The NMDS plot of native fish-assemblage structure in the presence and absence of non-native trout indicated that six sites were considerably different than the remaining 48 sites (Fig. 2, note that because of their similarity, 48 of these sites cannot be distinguished and appear as one point near the centre of the figure). Further examination of the fish assemblage in these sites showed that although non-native trout were collected, no other fish were present. These six sites were temporarily removed to examine what influence trout may have on native fish assemblages in sites with native fish present at the time of sampling ($N = 48$). Native fish-assemblage structure still did not differ between sites with and without trout when excluding sites without any native fish (Global R : 0.115, $P = 0.008$, Fig. 2).

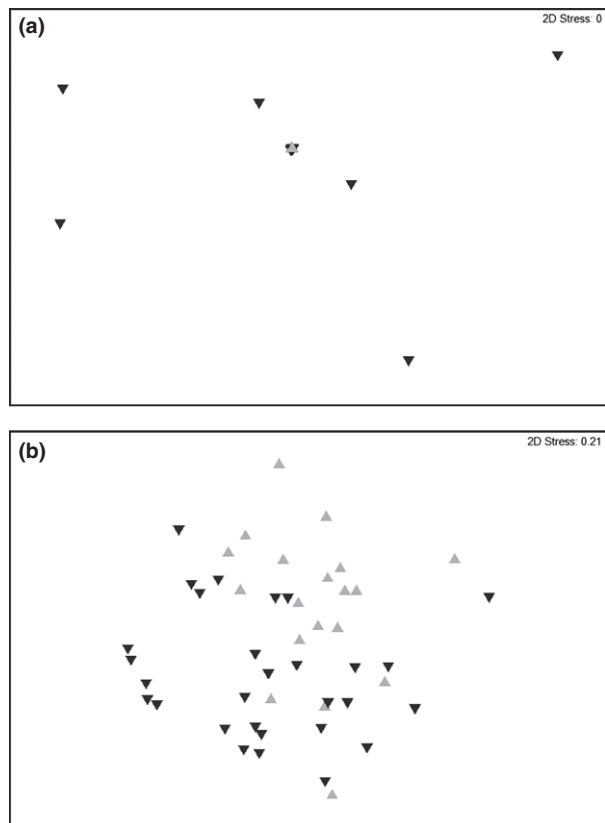


Fig. 2. Nonmetric multidimensional scaling plot of native fish community structure (relative abundance, fourth-root transformed, Bray–Curtis similarity matrix) in sites where any trout species or a combination of trout species were present (black) and sites where no trout species were present (grey) for (a) all sites ($N = 54$) and (b) excluding 6 sites with no native fish ($N = 48$).

Native fish-assemblage structure at differing abundances of non-native trout

Trout CPUE varied widely across sites surveyed during 2011 allowing us to test the relation between trout abundance and native assemblage structure. Greater trout abundances resulted in greater separation in native fish-assemblage structure between sites with and without trout (Fig. 3). Assemblage structure was similar or differed only slightly between sites with and without trout for the three subsets of data with the lowest trout abundances (Global R values < 0.40), but clear separation in native assemblage structure occurred in sites with trout CPUEs > 1.22 trout·min $^{-1}$ (Global R values > 0.40 ; Fig. 3). However, species-specific contributions to dissimilarity for each subset could not be determined because of the bootstrapping methodology used (*i.e.*, each iteration would have different species-specific contributions). Habitat and human-use characteristics did not differ across the subsets of sites with trout present (Global R : -0.052 , $P = 0.863$) suggesting no relationship between habitat and separation in native fish-assemblage structure. Additionally, the size structure of non-native trout did not appear to differ across subsets of sites with trout (Fig. 4).

There were six sites in which trout were present, but no native fish were collected. Five of these six sites fell into the top two subsets of trout abundances (*i.e.*, those subsets where there was clear

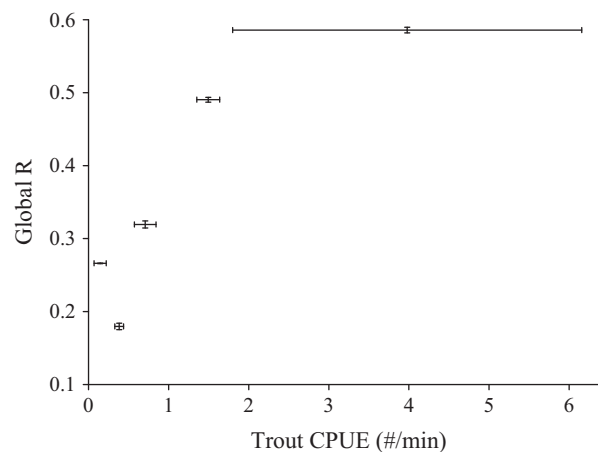


Fig. 3. Mean Global R values \pm 95% CI from analysis of similarities (1000 iterations) comparing native fish-assemblage structure (fourth-root transformed, Bray–Curtis similarity matrix; Primer-E v.6) between sites with and without trout with increased relative abundance of trout (CPUE, trout·min $^{-1}$). Subsets were based on the following percentiles when sites were ranked by trout abundance: ≤ 20 th percentile (CPUE ≤ 0.25 trout·min $^{-1}$), 21–40th percentiles (CPUE 0.31–0.48 trout·min $^{-1}$), 41–60th percentiles (CPUE 0.50–0.98 trout·min $^{-1}$), 61–80th percentiles (CPUE 1.22–1.73 trout·min $^{-1}$), and ≥ 81 st percentile (CPUE ≥ 1.97 trout·min $^{-1}$). Intervals are plotted at the mean trout CPUE \pm 95% CI for each subset.

Potential influences of non-native trout

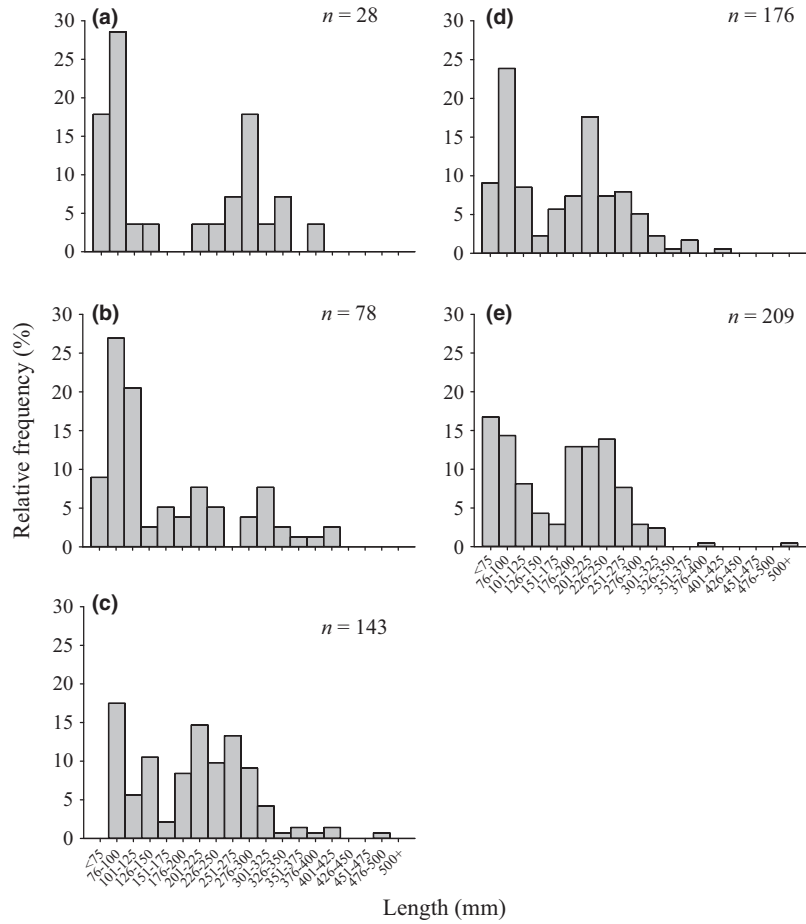


Fig. 4. Relative length frequency distribution of all non-native trout in each subset of sites with trout examined: (a) ≤ 20 th percentile (CPUE ≤ 0.25 trout \cdot min $^{-1}$), (b) 21–40th percentiles (CPUE 0.31–0.48 trout \cdot min $^{-1}$), (c) 41–60th percentiles (CPUE 0.50–0.98 trout \cdot min $^{-1}$), (d) 61–80th percentiles (CPUE 1.22–1.73 trout \cdot min $^{-1}$), and (e) ≥ 81 st percentile (CPUE ≥ 1.97 trout \cdot min $^{-1}$).

separation in native assemblages between sites with and without trout). The lack of any native species in these six sites may have caused the increased distinction in native fish-assemblage structures as trout abundances increased. Therefore, we re-ran the analyses excluding the six sites without native fish to determine what, if any, influence these sites had on the relation between trout abundance and native fish-assemblage structure. Removal of the six sites without native fish did not change the overall relation between trout abundance and separation in native fish-assemblage structure between sites with and without trout.

Discussion

Creek chub and fathead minnow relative abundance and size structures were similar in the presence and absence of non-native trout. Similarly, longnose dace and white sucker relative abundances were similar in the presence or absence of non-native trout. Trout may therefore have little influence on the relative abundance of these species at the reach scale. Alternatively, only widespread, common native species were examined, potentially biasing our results. Those

species prevalent enough to test differences may be those most capable of coexisting with trout.

Although there were no changes in relative abundance of the native species we examined, there were some differences in the size structures of longnose dace and white sucker. Longnose dace were larger in the presence of brown trout and smaller in the presence of rainbow trout compared to sites without non-native trout, and there was a greater proportion of larger white suckers in the presence of brown trout. Differences in size structure could be the result of differences in growth, recruitment or mortality among these populations. Additionally, brown trout and rainbow trout are known to differ in feeding ecology and habitat use suggesting the mechanisms through which they may influence native species size structures may differ. Data (*e.g.*, age structure of populations) were not available to fully evaluate the biological implications of differences in size structure for each native species, or to determine what mechanism is responsible for these differences. However, we suggest that the size structure differences we observed could potentially be biologically important based on what is known about the growth and reproduction of these species.

The longnose dace is a short-lived species with a maximum age of 4–5 years (maximum size of approximately 125 mm; Scott & Crossman 1973a). Longnose dace mature at age 1–2 (Scott & Crossman 1973a; and Roberts & Grossman 2001). They spawn during the spring, but can spawn multiple times within a season (Roberts & Grossman 2001). Females lay between 200 and 1200 eggs in a spawning event (Scott & Crossman 1973a) and potential fecundity in a given year is thought to range from approximately 1100 to 2500 eggs (Roberts & Grossman 2001). Longnose dace growth is slow relative to other species. Reed & Moulton (1973) summarised average lengths at ages of longnose dace from four different populations (including populations from Reed 1959 and Kuehn 1949). From this summary, longnose dace growth appears to average between 11 mm and 15 mm in a given year but ranges from 6 mm to 33 mm depending on the population, age and sex of longnose dace (Reed & Moulton 1973). We observed a 7- to 11-mm difference in longnose dace median length in the presence of non-native trout. Based on what is known about age and growth of longnose dace, a 7- to 11-mm difference in size structure could be the difference in an entire year class of longnose dace (either due to changes in recruitment or mortality) or could be due to changes in individual growth rates. Further, potential fecundity and clutch size of longnose dace are positively correlated with standard length and somatic mass (Roberts & Grossman 2001) suggesting that differences in size structure could potentially result in changes in fecundity. For such a short-lived species, these potential changes could have an influence on the overall fitness of an individual during its lifespan either due to changes in reproductive potential in a given year or changes in the total number of spawning events throughout its life.

The white sucker is a longer-lived species than longnose dace with a maximum age of approximately 17 years (Scott & Crossman 1973b). They reach sexually maturity between 3 and 8 years of age (Beamish 1973; Scott & Crossman 1973b), spawn in the spring, are known to spawn multiple times a year, and produce >20,000 eggs (Scott & Crossman 1973b). Growth of white suckers is variable (Beamish 1973), but much faster than longnose dace within the first year of life. Growth is thought to slow to between 10 and 20 mm per year after age 1 and likely ceases after sexual maturity (Scott & Crossman 1973b). We observed a 58- to 123-mm difference in white sucker median length in the presence of non-native trout. This could mean extreme changes in grow rates or changes in multiple year classes of white sucker. As in the longnose dace, these changes could have reproductive consequences. For example,

early growth in length of white sucker has been correlated with timing of maturity, in that faster early growth results in earlier maturity (Chen & Harvey 1994). Changes in the timing of maturation could then influence the overall reproductive potential of white sucker.

Aside from the reproductive consequences, changes in the size of longnose dace and white sucker may also alter other predatory and competitive interactions which could result in community-wide responses to trout presence. For example, an increase in size of white sucker (without a change in relative abundance) likely requires increased resource use, which could result in depletion of resources for other species in these systems. This, in turn, could result in changes in native fish-assemblage structure in the presence of non-native trout.

Although native fish-assemblage structure did not differ in the presence of non-native trout across all sites surveyed during 2011 (all abundances of non-native trout), native fish-assemblage structure did differ at high abundances of non-native trout. This suggests that current trout *presence* alone may not influence native fish assemblages. Similarly, non-game fish species richness and diversity did not differ in southern Appalachian Mountain streams that had been stocked with trout relative to paired, unstocked streams (although unstocked streams contained trout as well; Weaver & Kwak 2013). Native fish-assemblage structure also did not differ in an Oklahoma Ozark stream before and after rainbow trout introduction in riffles and glides (Walsh & Winkelman 2004). Assemblage structure did, however, differ in pool habitats due to declines in abundances of seven species poststocking (Walsh & Winkelman 2004).

Fish-assemblage structure in headwater streams in Nebraska may also be the result of historical trout presence (*e.g.*, Thorp 1986). Emphasis in site selection was on historical trout presence, and therefore, all streams historically contained trout. The remnants of historical trout presence may still be evident, making it difficult to differentiate fish communities based on current trout presence (*e.g.*, trout may have previously homogenised stream fish communities prior to their disappearance in sites that currently have no trout). Therefore, monitoring streams at multiple points in time may provide more insight into the potential influences of non-native trout on native fish assemblages. Additionally, examining a wider variety of streams (*i.e.*, streams with no historical trout presence) may lend further insight into the mechanisms structuring fish communities in these systems.

Although there were no differences in native fish-assemblage structure in the *presence* of non-native trout, there appears to be increased separation (or

more dissimilarity) in fish-assemblage structure (Fig. 3) between sites with and without trout as trout abundances increase. There also may be some threshold of trout density (around 0.5–1.5 trout·min⁻¹) above which non-native trout have strong influences on native fish assemblages, but below which non-native trout have weak or no influences on native fish assemblages. Therefore, stocking trout at high abundances may alter native fish-assemblage structure in Nebraska headwater streams. Stocking trout at low abundances, however, may not have direct adverse effects on native fish-assemblage structure. Future research should focus on identifying the threshold in trout density, if any, that will alter native fish assemblages, as it may be possible to simultaneously manage for a low-density trout fishery and for native species conservation if a threshold exists.

Conclusion

Non-native trout may have both population and assemblage influences on native species in headwater streams in Nebraska. Longnose dace were larger in the presence of brown trout and smaller in the presence of rainbow trout compared to sites without non-native trout, and there was a greater proportion of larger white suckers in the presence of brown trout. Based on what we know about these species from other systems, we suggest that the differences in size structures we observed for longnose dace and white sucker in the presence and absence of non-native trout could potentially have biological significance. Therefore, caution is warranted when stocking non-native trout into streams with these species even if there appears to be no initial changes in relative abundance of native species. These differences could also result in community-wide responses to non-native trout, and there appears to be increased risk to native communities in sites with high abundances of trout.

Acknowledgements

The authors thank two anonymous reviewers and Dr. Thomas Kwak for reviewing earlier drafts of this manuscript, Dr. Christopher Chizinski for assistance with data analyses in R and technicians for field assistance. This project was funded by State Wildlife Grants Program project T-79-R, which was administered by the Nebraska Game and Parks Commission. Any use of trade, firm, or product names is for descriptive purposes only and does not imply endorsement by the U.S. Government. The Nebraska Cooperative Fish and Wildlife Research Unit is jointly supported by a cooperative agreement among the U.S. Geological Survey, the Nebraska Game and Parks Commission, the University of Nebraska, the U.S. Fish and Wildlife Service and the Wildlife Management Institute.

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