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# Chicago's fish assemblage over ~30 years – more fish and more native species

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**Abstract**

How fish assemblages change over time in highly-modified urban rivers, where physical and chemical properties rarely mimic non-urban systems, has sparsely been documented. Data have been collected on fishes within the boundaries of the Chicago Metropolitan area routinely since the mid-1980's. Representing fish assemblages in one of the largest cities of America, this dataset offers the ability to investigate and track changes in assemblage composition in an urbanized river. To this end, multivariate modelling, as well as various visualization techniques, were used to assess and describe compositional changes in the fish assemblage of Chicago's waterways. In general, there were gradual enhancements in the fish assemblages of Chicago's waterways throughout the years studied, which are characterized by more fish, of which more are native species. Small-bodied native fishes (Cyprinidae), game fishes (Centrarchidae), as well as catfish (Ictaluridae) have increased in relative abundance, whereas several invasive fish species exhibited declines. Exponential growth of Banded Killifish (*Fundulus diaphanous*) relative abundance appears to continue from previously noted range expansions. As Chicago and other cities move towards supporting fishable waterways, interest may lay in investigating population vital rates and habitat or water quality factors affecting them in heavily urbanized settings.

**Keywords** Chicago · Fish assemblage · Community · Urban River · Electrofishing

## Introduction

When studied over long periods of time, the composition of species at a location may be classified as continually shifting, changing in sudden steps, returning to a state prior to some disturbance, or resisting change (Holling 1973; Connell and Sousa 1983). Fish assemblages within stream systems experiencing natural disturbances (i.e., droughts or floods) are thought to return to some pre-disturbance state in the absence of human disturbance (Matthews et al. 2013). Whereas in human disturbed riverine systems fish assemblages appear to be temporally dynamic, often reflecting directional change rather than a return to some previous composition (Pyron et al. 2006; Whitten et al. 2018; DeBoer et al. 2019).

However, many rivers upon which urban centers have developed have experienced landscape level physical and chemical alterations to the point that they may be considered novel ecosystems, rather than something disturbed from a previous natural state (Francis 2014). Modifications such as channelization, dams, and levees alter flow regimes as well as reduce physical habitat variability in order to reduce flooding and increase navigability (Gurnell et al. 2007). Such physical alterations compound upon effects of urban and industrial effluents affecting water quality, together leading to reduced ecosystem function and biodiversity (Walsh et al. 2005; Booth et al. 2016). Consequently, urban rivers often simultaneously represent highly valuable resources and highly degraded ecosystems compared to non-urban systems (Vörösmarty et al. 2010; Everard and Moggridge 2012). While regulations like the Clean Water Act seek to abate water quality issues, the level of engineering (e.g., sheet pile walls), maintenance (e.g., removal of debris), and barge traffic often preclude restoration or rehabilitation efforts (Francis 2009). The hard structure construction of urban waterways, combined with water quality issues make the ecosystem un-like typically studied

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riverine systems and their associated communities. Considering highly-modified urban rivers as novel ecosystems would suggest enhancement of biotic communities is to be expected in the years and decades following the Clean Water Act, rather than restoration or rehabilitation, however published works are lacking and thus temporal patterns in fish assemblages remain difficult to predict.

The Chicago area waterway system (CAWS) is composed of 78 miles of modified waterways, 75% of which is man-made, and the remaining 25% have been dredged, reconfigured, or rerouted (Greenberg 2004; Solzman 2006; LimnoTech 2010; Hill 2019). Charged with managing Chicago's wastewater treatment and stormwater management, the Metropolitan Water Reclamation District of Greater Chicago (MWRD) has monitored biotic and abiotic aspects of the CAWS since early on in its history and developed a more comprehensive program in the 1970's, coinciding with the enactment of the Clean Water Act. On the Illinois River, downstream of the CAWS, improvements in both water quality and in-turn fish assemblages have been noted since the enactment of the Clean Water Act (Pegg and McClelland 2004; McClelland et al. 2012; Parker et al. 2016; Gibson-Reinemer et al. 2017b). It is unknown whether fish assemblages in heavily-modified urbanized river systems such as the CAWS have exhibited similar positive changes since passage of the Clean Water Act, but this information is needed to assess its ecological impacts.

Fish communities of the CAWS have been monitored at various locations since the late 1970's by MWRD, several locations have been monitored nearly every year since the mid-1980s. Such monitoring efforts, done by electrofishing both sides of the waterway, identify and enumerate fishes found to help MWRD assess the general condition of the CAWS and its inhabitants. Although data is publicly available (http://geohub.mwrd.org/), changes in the fish assemblage throughout the history of the dataset, and the CAWS, have yet to be assessed and described outside of internal reports available from MWRD. To this end, goals were to test for changes in the fish assemblage since 1985 as well as provide visualizations of such changes. It was hypothesized that fish communities would exhibit gradual directional changes, going from a degraded community to one of improvement, realized through increases in species richness as well as changes in native species relative abundances (CPUEs).

## Methods

### Study area description

The CAWS consists of two rivers (Chicago River and Calumet River) whose flows have been reversed, connected by two man-made canals (Chicago Sanitary and Ship Canal and the Cal-Sag Channel), which all eventually flow south to the Illinois River. An additional constructed canal, the North Shore Channel, supplies water, via discretionary flow, from Lake Michigan aiding the flow of water through the channel through the northern Chicago communities towards downtown (Illinois Coastal Management Program 2011). Other inlets from Lake Michigan are located on the Chicago River (Chicago River Control Works) and the Calumet River (T.J. O'Brien Lock and Dam) help to maintain adequate flows and water levels in the CAWS. The CAWS includes outflows from four wastewater treatment plants (Terrence J. O'Brien, Stickney, Lemont, and Calumet), one of them the world's largest (MWRD 2019), as well as a combined sewer overflow system which releases untreated wastewater into the surface waterways when the tunnel and reservoir sewer system is overwhelmed by heavy rainstorms. The system is generally described as slow flowing, non-wadable, lotic waters with vertical steel walls or near vertical banks, soft fine sediment, and little submerged habitat (LimnoTech 2010). Overall the system stands as an engineering marvel and its history is well summarized well by L. Hill (2019) and Olson and Morton (2017).

Numerous actions have been taken by MWRD over the years to improve water quality of the system, including the building of five Sidestream Elevated Pool Aeration (SEPA) stations in the Cal-Sag Channel (3), The Little Calumet River (1), and Calumet River (1) (Butts et al. 1999), an instream aeration station in each of the North Shore Channel and the North Branch Chicago River (Melching 2018), improved nitrification and disinfection practices, as well as continued construction of the Tunnel And Reservoir Plan (known as: TARP) which greatly reduces combine sewer overflow events. Such actions were, and continue to be, taken toward reaching the general goal of having fishable and swimmable waters, a mandate by the Clean Water Act (Karr and Dudley 1981).

### Program description

Electrofishing sampling has been carried out by MWRD personnel in the Chicago Area Waterways since 1974, although a fully expanded, comprehensive program did not begin until 1985. Several locations were chosen to collect water quality data as part of an Ambient Water Quality Monitoring (AWQM) program, and to coincide with such data, electrofishing has been conducted to collect fish community information at the same locations. Initially, Alternating Current (AC) electrofishing was used until the agency transferred to pulsed-Direct Current (DC; generally, 120 pulses $sec^{-1}$ targeting 12–14 amps) electrofishing at the start of 2001.

At each location, each river bank (where applicable) was sampled for 400 m separately, and effort (timed in seconds) was recorded. Fish were netted, identified, length and weight were recorded, and fish were returned to the waterway. When both banks were sampled, each "haul" was kept separate and treated as a separate sample for the location, after both "hauls" were completed fish were returned to the waterway to avoid recaptures. Catch per unit effort (CPUE) was standardized to the number fish caught per 30 min of sampling for each "haul" at each site. Sampling of fish communities occurred in July, August and September, sometimes more than once in a year, whereas data were not available for other years (e.g., 2008, 2014, and 2015).

In total, the data set included 58 species, including one group for hybridized sunfish (*Lepomis* spp.), which summarized an additional 6 "species", since whether these represented F1, or some other generation, was unknown (Table 1). Species were classified into tolerance levels 1 being tolerant, 2 being neutral, and 3 being intolerant in accordance with Grabarkiewicz and Davis (2008) and Poff and Allan (1995), some species were not located in either source and thus remain as "NA" herein. Species are also categorized by being native or invasive (including simply non-native species) in accordance with Laird and Page (2011) and IL-DNR (2018).

## Data selection and treatment

Nine AWQM sites were chosen for this analysis, ones which were most comprehensively sampled throughout the 34-yr period of interest. Sites in close proximity were lumped together leading to six waterways of interest (Fig. 1). Of note, AWQM station 37 is just downstream of the confluence of the North Shore Channel and the North Branch of the Chicago River although it is lumped under the general waterway "North Shore Channel" herein, along with two other locations which were 3.2 km apart. Two sites on the Chicago Sanitary and Ship Canal (shortened to "Sanitary-Ship Canal" in figures and tables) were 5 km apart and were used together to represent this waterway. The Calumet River AWQM station (#55) is on the Lake Michigan side of the O'Brien lock and dam (at 130th street bridge), meaning it is heavily influence by both Lake Calumet and Lake Michigan compared to other locations included herein which are all "inside" the lock and dam system of the CAWS. One sampling location was used to represent each of the Calumet Saganashkee Channel (shortened to "Cal-Sag Channel" throughout manuscript), Little Calumet River, and Lockport Lock and Dam (shortened to "Lockport Dam" in figures and tables) (Fig. 1).

Data were separated by the two types of electrofishing, leading to a 1985–2000 AC dataset and a 2001–2018 DC dataset. This separation was done to remove the effect different electrofishing currents may have on fish community data (McClelland et al. 2013).

## Data analysis

All analyses were conducted using R (v3.6.3) programming (R Development Core Team 2020). We used generalized linear models (GLMs) to test the effects of year, waterway, and their interactions on richness and abundance of native and invasive species. We used negative binomial distributions in models to account for mean-variance relationships and overdispersion. We also included sampling effort (measured in the seconds; log transformed) as an offset to account for variation in effort across samples. Separate models were constructed for richness and abundance of native and invasive species, for the 1985–2000 AC dataset and a 2001–2018 DC dataset. We used $\chi^2$ test statistics to assess significance and differences among waterways were assessed using Tukey-Kramer *post hoc* tests.

We used a multivariate GLM with a negative binomial distribution to assess the effects of year and waterway on the fish assemblage using the *manyglm* function ("mvabund" package; Wang et al. 2012, 2020). The predictor variables are fit to each species as GLMs and a resampling procedure (PIT-trap resampling; Warton et al. 2017) used to assess the significance of assemblage-level effects, as well as adjusted *P*-values for effects of each species. We used multivariate Wald $\chi^2$ test statistic to evaluate compositional differences and species level differences, with 9999 sampling iterations. We used the Tukey-Kramer method to compare expected marginal means and trends (i.e., interactions) among waterways for each species with the emmeans and emtrends functions ("emmeans" package; Lenth et al. 2018) .

We used non-metric multidimensional scaling (nMDS) to visualize community-level changes across years and among waterways. Data were transformed into catch per unit effort (CPUE) and standardized to 30-min of effort. We performed nMDS on the 1985–2000 AC and 2001–2018 DC datasets separately using the *metaMDS* function ("vegan" package; Oksanen et al. 2019; Oksanen 2013, 2019), which automatically transforms raw data using a square-root transformation followed by Wisconsin standardization prior to calculating Bray-Curtis distances between samples. We averaged nMDS scores to yearly means for each waterway to illustrate differences in assemblage level changes over years among waterways. Faceting was used to separate each waterway, avoiding overlap in their points.

# Results

The dataset consisted of a total of 456 sampling events in which a total of 53,917 fish were caught across the 58 species (hybrid Lepomids counted as 1 species total; Table 1). The species captured the most during all of the sampling events included Gizzard Shad *Dorosoma cepedianum* (total catch 16,845), Bluntnose Minnow *Pimephales notatus* (5294), Common Carp *Cyprinus carpio* (4740), Goldfish *Carassius auratus* (3665), Pumpkinseed *Lepomis gibbosus* (3204), Fathead Minnow *Pimephales promelas* (3156), Bluegill *Lepomis macrochirus* (2543), and Largemouth Bass

**Table 1** Species found within the CAWS dataset used in this analysis of compositional changes from 1985 through 2018

| Species | Tolerance | Native/ Invasive | Years Found | AC | DC |
|---|---|---|---|---|---|
| Alewife (*Alosa pseudoharengus*) | NA | I | 15 | 122 | 41 |
| Banded Killifish (*Fundulus diaphanus*) | NA | N | 4 | – | 728 |
| Black Buffalo (*Ictiobus niger*) | NA | N | 6 | – | 28 |
| Black Bullhead (*Ameiurus melas*) | 1 | N | 21 | 136 | 34 |
| Black Crappie (*Pomoxis nigromaculatus*) | 2 | N | 22 | 31 | 40 |
| Blackstripe Topminnow (*Fundulus notatus*) | NA | N | 5 | – | 13 |
| Bluegill (*Lepomis macrochirus*) | 1 | N | 30 | 481 | 2062 |
| Bluntnose Minnow (*Pimephales notatus*) | 1 | N | 30 | 2625 | 2669 |
| Brook Silverside (*Labidesthes sicculus*) | 3 | N | 6 | – | 54 |
| Brook Stickleback (*Culaea inconstans*) | 3 | N | 6 | 45 | – |
| Brown Bullhead (*Ameiurus nebulosus*) | 1 | N | 3 | – | 4 |
| Common Carp (*Cyprinus carpio*) | 1 | I | 31 | 2141 | 2599 |
| Carp x Goldfish (*C. carpio* X *C. auratus*) | NA | I | 25 | 373 | 17 |
| Central Mudminnow (*Umbra limi*) | 1 | N | 9 | 6 | 7 |
| Channel Catfish (*Ictalurus punctatus*) | 2 | N | 18 | 5 | 92 |
| Chinook Salmon (*Oncorhynchus tshawytscha*) | NA | I | 1 | – | 1 |
| Creek Chub (*Semotilus atromaculatus*) | 1 | N | 7 | 2 | 9 |
| Emerald Shiner (*Notropis atherinoides*) | 2 | N | 25 | 821 | 1056 |
| Fathead Minnow (*Pimephales promelas*) | 1 | N | 24 | 3084 | 81 |
| Flathead catfish (*Pylodictis olivaris*) | 2 | N | 1 | – | 1 |
| Freshwater Drum (*Aplodinotus grunniens*) | 2 | N | 17 | 4 | 51 |
| Gizzard Shad (*Dorosoma cepedianum*) | 1 | N | 31 | 4992 | 11,853 |
| Golden Shiner (*Notemigonus crysoleucas*) | 1 | N | 30 | 966 | 1112 |
| Goldfish (*Carassius auratus*) | NA | I | 30 | 3286 | 379 |
| Grass Carp (*Ctenopharyngodon idella*) | NA | I | 1 | – | 1 |
| Grass Pickerel (*Esox americanus vermiculatus*) | 2 | N | 4 | 2 | 3 |
| Green Sunfish (*Lepomis cyanellus*) | 1 | N | 31 | 474 | 781 |
| Hybrid Lepomids | NA | N | 18 | 18 | 11 |
| Largemouth Bass (*Micropterus salmoides*) | 2 | N | 31 | 617 | 1667 |
| Longnose Dace (*Rhinichthys cataractae*) | 3 | N | 1 | 1 | – |
| Mimic Shiner (*Notropis volucellus*) | 3 | N | 2 | – | 4 |
| Mosquitofish (*Gambusia affinis*) | NA | I | 12 | – | 1768 |
| Nile Tilapia (*Oreochromis niloticus*) | NA | I | 1 | 5 | – |
| Ninespine Stickleback (*Pungitius pungitius*) | NA | N | 1 | 1 | – |
| Northern Pike (*Esox lucius*) | 3 | N | 3 | – | 3 |
| Orangespotted Sunfish (*Lepomis humilis*) | 1 | N | 12 | 31 | 4 |
| Oriental Weatherfish (*Misgurnus anguillicaudatus*) | NA | I | 10 | 5 | 78 |
| Pumpkinseed (*Lepomis gibbosus*) | 2 | N | 29 | 200 | 3004 |
| Quillback (*Carpiodes cyprinus*) | 2 | N | 5 | 1 | 5 |
| Rainbow Smelt (*Osmerus mordax*) | NA | I | 3 | 8 | – |
| Rainbow Trout (*Oncorhynchus mykiss*) | 3 | I | 1 | 1 | – |
| Rock Bass (*Ambloplites rupestris*) | 3 | N | 18 | 24 | 295 |
| Round Goby (*Neogobius melanostomus*) | NA | I | 14 | 4 | 116 |
| Sand Shiner (*Notropis stramineus*) | 2 | N | 3 | – | 9 |
| Skipjack Herring (*Alosa chrysochloris*) | NA | N | 2 | – | 4 |
| Smallmouth Bass (*Micropterus dolomieu*) | 3 | N | 17 | 34 | 293 |
| Smallmouth Buffalo (*Ictiobus bubalus*) | 2 | N | 3 | – | 4 |
| Spotfin Shiner (*Cyprinella spiloptera*) | 1 | N | 15 | – | 378 |

**Table 1** (continued)

| Species | Tolerance | Native/ Invasive | Years Found | AC | DC |
|---|---|---|---|---|---|
| Spottail Shiner (*Notropis hudsonius*) | 3 | N | 16 | 156 | 16 |
| Tadpole Madtom (*Noturus gyrinus*) | 2 | N | 1 | – | 1 |
| Threadfin Shad (*Dorosoma petenense*) | NA | N | 1 | – | 1 |
| White Bass (*Morone chrysops*) | 2 | N | 2 | – | 3 |
| White Crappie (*Pomoxis annularis*) | 1 | N | 5 | – | 7 |
| White Perch (*Morone americana*) | NA | I | 17 | 135 | 107 |
| White Sucker (*Catostomus commersoni*) | 1 | N | 24 | 51 | 298 |
| Yellow Bass (*Morone mississippiensis*) | NA | N | 11 | 3 | 43 |
| Yellow Bullhead (*Ameiurus natalis*) | 2 | N | 18 | 5 | 375 |
| Yellow Perch (*Perca flavescens*) | 2 | N | 15 | 708 | 103 |

Tolerance values reported by Poff and Allan (1995) and Grabarkiewicz and Davis (2008), or if unavailable ("NA"), as well as native or invasive status (Laird and Page 2011; IL-DNR 2018), and the number of individuals caught in the years AC fishing was used (1985–2000) and DC fishing was used (2001–2018)

*Micropterus salmoides* (2284) (Table 1). By occurrence in sampling events, the top six species were Common Carp (present in 401 surveys), Gizzard Shad (354), Largemouth Bass (252), Goldfish (241), Bluegill (241), and Green Sunfish *Lepomis cyanellus* (237).

Native species richness increased across years in both 1985–2000 and 2001–2018 periods ($\chi^2 = 13.10$, $P < 0.001$; $\chi^2 = 37.22$, $P < 0.001$). Native species richness was also different among waterways ($\chi^2 = 110.67$, $P < 0.001$; $\chi^2 = 65.04$, $P < 0.001$), there were no significant interaction effects ($\chi^2 = 7.90$, $P = 0.16$; $\chi^2 = 8.78$, $P = 0.12$). For the 1985–2000 period, native species richness was significantly higher in the Calumet River than other waterways, the North Shore Channel was second highest compared to others, and all other waterways were not significantly different from each other (Fig. 2). For the 2001–2018 period, differences in native species richness among waterways were less distinct with a gradient in differences richness, with Little Calumet River and Calumet River having similar but higher richness than others and Lockport Lock and Dam had the lowest (Fig. 2).

Invasive species richness exhibited no significant change across years nor among waterways in the 1985–2001 dataset ($\chi^2 = 0.0008$, $P = 0.98$; $\chi^2 = 3.62$, $P = 0.60$ respectively). Similarly, there were not changes in invasive species richness across years in the 2001–2018 dataset ($\chi^2 = 0.073$, $P = 0.79$) but there were differences among waterways ($\chi^2 = 24.927$, $P < 0.001$), primarily with there being fewer invasive species at Lockport Lock and Dam, compared to the Little Calumet River, Chicago Sanitary and Ship Canal, and Cal-Sag channel (Fig. 2).

Interactions between years and waterways existed for native species abundance (CPUEs) in both 1985–2000 and 2001–2018 periods ($\chi^2 = 10.02$, $P = 0.075$; $\chi^2 = 14.23$, $P = 0.014$; respectively, Fig. 3). Between 1985 and 2001 a significant positive trend was noted in the Chicago Sanitary and Ship Canal as well as a negative trend in the Calumet River, whereas the other waterways were not significantly different from either of these trends (Fig. 3). In the 2001–2018 period, a significant positive trend was noted at the Lockport Lock and Dam. No trends were noted for the North Shore Channel and the Chicago Sanitary and Ship Canal, significantly different from Lockport Lock and Dam, whereas the other waterways were not significantly different from either of these two groups (Fig. 3).

For both the 1985–2000 and 2001–2018 periods, invasive species abundance (CPUEs) was affected by differences among waterways ($\chi^2 = 88.46$, $P < 0.001$; $\chi^2 = 112.35$, $P < 0.001$; respectively), more so than trends in years ($\chi^2 = 29.79$, $P < 0.001$; $\chi^2 = 3.74$, $P = 0.053$; respectively), there was no evidence of significant interactions ($\chi^2 = 7.79$, $P = 0.17$; $\chi^2 = 7.38$, $P = 0.19$; respectively). Higher CPUEs were found in the Chicago Sanitary and Ship Canal compared to others and lower CPUEs in the Calumet River, Cal-Sag Channel and at Lockport Lock and Dam in the 1985–2000 period. In the 2001–2018 period, invasive species CPUE was higher in the Chicago Sanitary and Ship Canal and Little Calumet River compared to others, which were not different from each other.

Multivariate GLMs indicated that how assemblages changed over years was different among waterways for each 1985–2000 and 2001–2018 period (interactions; $\chi^2 = 17.72$, $P < 0.001$; $\chi^2 = 19.74$, $P < 0.001$). Effects of year ($\chi^2 = 20.57$, $P < 0.001$; $\chi^2 = 21.16$, $P < 0.001$) and waterway ($\chi^2 = 34.14$, $P = 0.002$; $\chi^2 = 39.35$, $P = 0.003$) were also significant for both datasets. Species specific outputs indicated multiple species exhibited trends across years (Table 2), different CPUES among waterways (Tables 2 and 3), as well as different trends across years

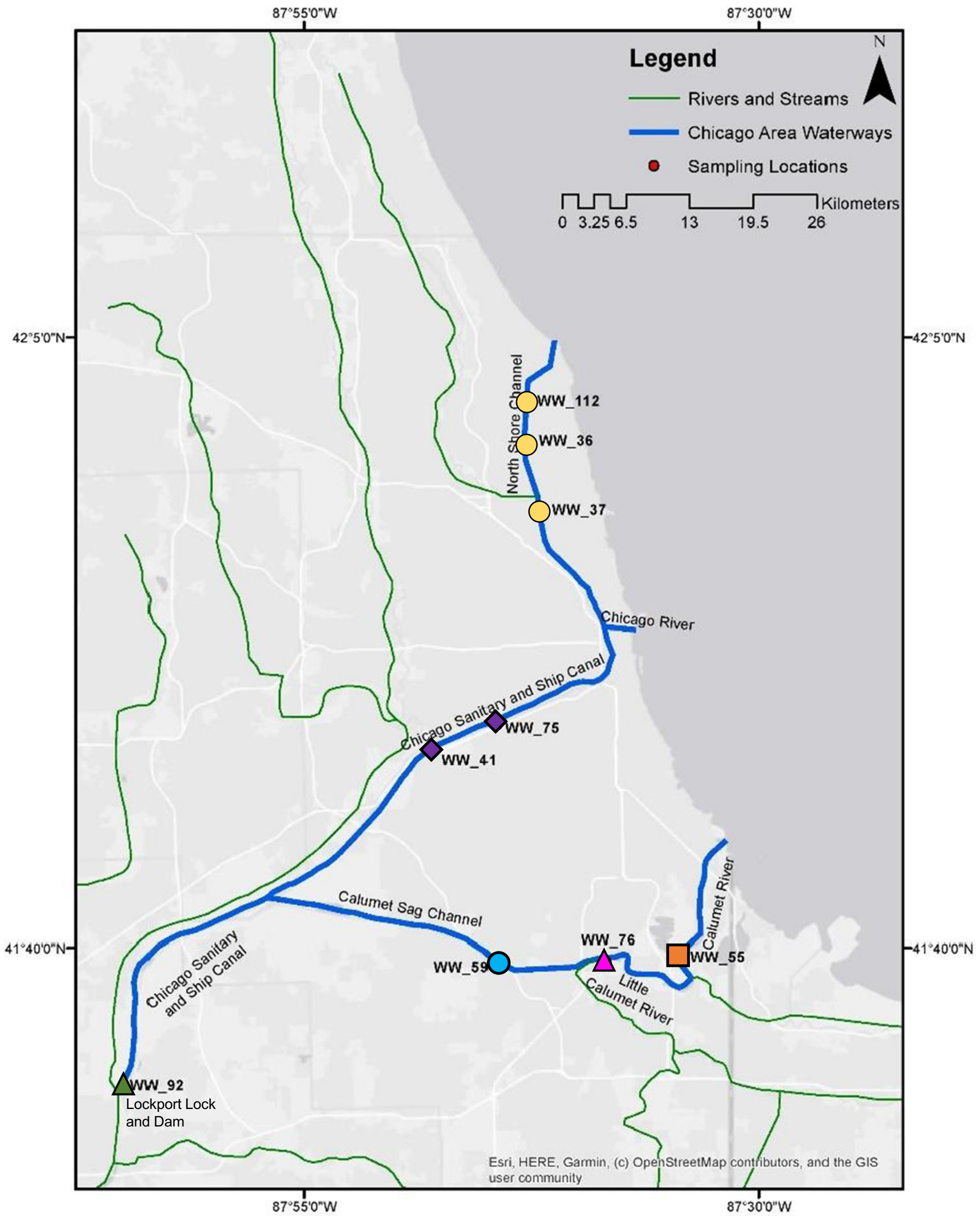


**Fig. 1** A subset of the locations in the Chicago Area Waterways sampled by MWRD since the mid-1980's, from which fish community data is summarized herein. Specific sampling sites in close proximity were grouped into the main waterway of interest, signified by sharing symbols and colors

among waterways (Tables 2 and 4), leading to the assemblage level interaction. Faceting nMDS scores by waterways illustrated how changes across years differed among waterways (Figs. 4 and 5). For both nMDS plots, data scores for samples from the Calumet River were located more positively on nMDS1, and for the 2001–2018 period more positively on nMDS2, compared to other waterways. This is likely due to larger CPUEs of several species in Calumet River compared to other locations (Table 3). All waterways exhibited a trend across 1985–2001 from lower to higher values on nMDS2 (Fig. 4), which can be attributed to the loss of Brook Stickleback *Culaea inconstans*, Black Bullhead *Ameiurus melas*, Common Carp-Goldfish hybrids, Goldfish, and Fathead Minnows concomitant with increases in Gizzard Shad and Largemouth Bass (Tables 1 and 4). Changes in the fish community across 2001–2018 were visible as diagonal trend in nMDS scores from the top left to the bottom right of the nMDS plot (Fig. 5). For the 2001–2018 period, changes in the fish community in the Cal-Sag Channel were not as evident compared to Lockport Lock and Dam and the Chicago Sanitary and Ship Canal.

General trends for 10 species of interest are shown in Fig. 6, selected for their importance as sport fish (Bluegill, Green Sunfish, Largemouth Bass, and Pumpkinseed), being an invasive species (Common Carp), having high abundance (Bluntnose Minnow and Gizzard Shad), or a species with large changes in CPUE (Banded Killifish *Fundulus diaphanus*, Golden Shiner *Notemigonus crysoleucas*, and Yellow Bullhead *Ameiurus natalis*), as well as generally all having significant trend outputs in Table 2.

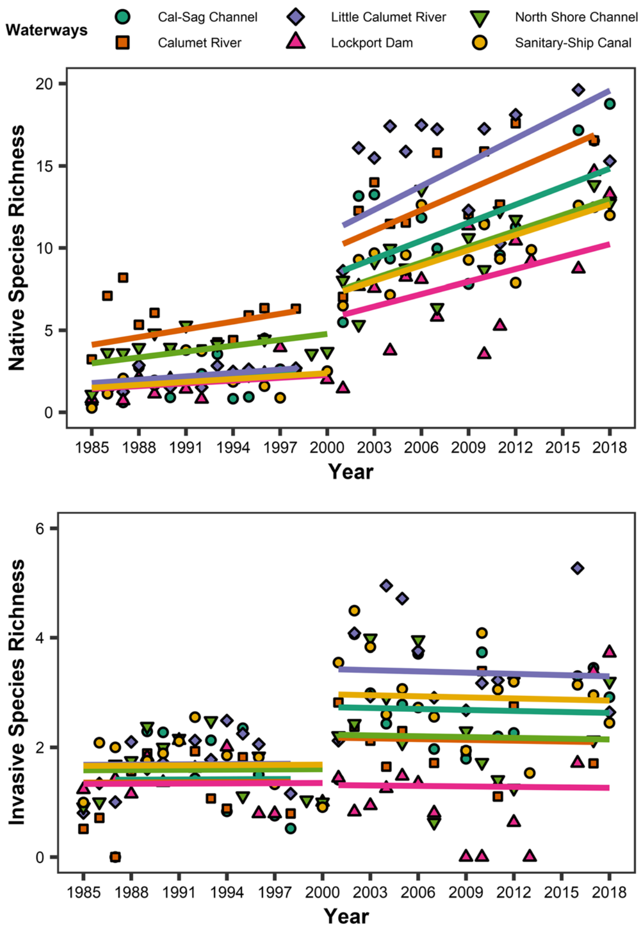


Fig. 2 Changes in native and invasive species richness in six waterways within the Chicago Area Waterways from 1985 to 2018. Negative binomial regression used separately for the period of 1985–2000 and 2001–2018 due to different fishing gears being used, and included sampling time as an offset. To reduce clutter, yearly averages are plotted for each waterway. We refer readers to the text for discussion of significant differences among waterways

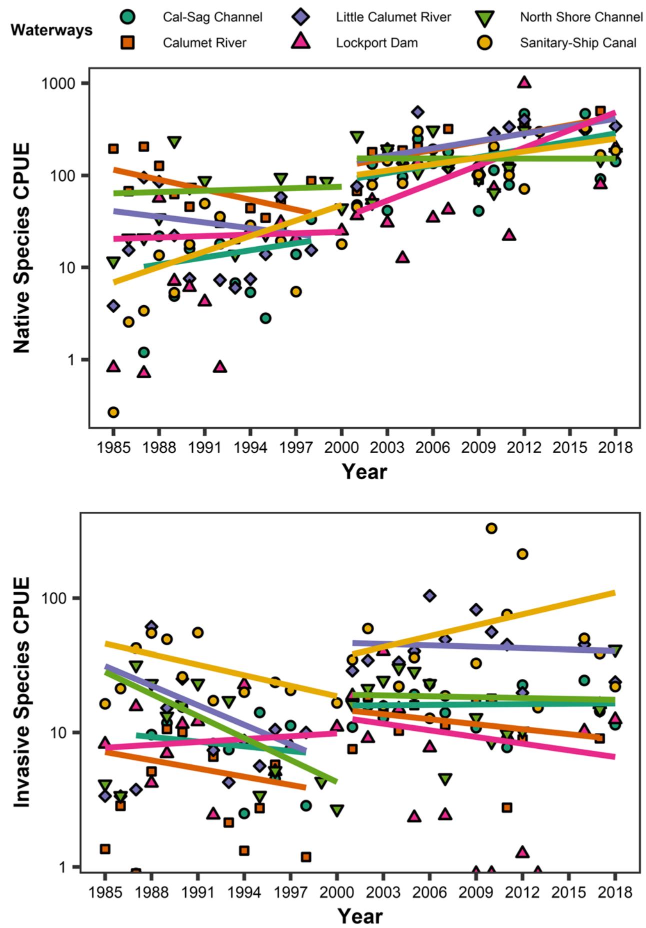


Fig. 3 Changes in catches (log scale) of native and invasive fishes in six waterways within the Chicago Area Waterways from 1985 to 2018. Negative binomial regression used separately for the period of 1985–2000 and 2001–2018 due to different fishing gears being used and included sampling time as an offset. To reduce clutter, yearly averages are plotted for each waterway. We refer readers to the text for discussion of significant differences among waterways

## Discussion

The fish assemblage of the Chicago Area Waterway System presently has increased species richness and abundance compared to the 1980's, and those changes are due to native species. On average, between 13 and 16 native species were found during a 30-min electrofishing trip on the CAWS in 2018, compared to < 3 species in 1985. A total of 19 new species were found since 2001, of which only Mosquitofish *Gambusia affinis* represented an invasive species with a sizable population (> 10 individuals). Conversely, six species were captured prior to 2001 but not in recent years, the largest decline being native Brook Stickleback. Several native and sportfish species exhibited increases in CPUEs, suggesting improvements to the ecosystem of the CAWS from previous years. Ordination plots (nMDS) illustrated significant directional changes in fish assemblages over the entire period of data collection, suggesting gradual enhancements rather than stepwise changes or reversion to previous states.

Changes in species richness seem to be influenced by new occurrences of native species, rather than newly invasive populations. Invasive fishes such as Common Carp, Goldfish, and Oriental Weatherfish *Misgurnus anguillicaudatus* have been found throughout the 34 years of the data series. Round Goby *Neogobius melanostomus* were first documented in 1994 and continue to be routinely found, and there were single instances of Nile Tilapia *Oreochromis niloticus* (1999) and Grass Carp *Ctenopharyngodon idella* (2001) being found in the CAWS. Instead, evidence is provided that native fishes such as Mimic Shiner *Notropis volucellus*, Sand Shiner *Notropis stramineus*, Smallmouth Buffalo *Ictiobus bubalus*, Tadpole Madtom *Noturus gyrinus*, Threadfin Shad *Dorosoma petenense*, and White Crappie *Pomoxis annularis* have returned to the system, although abundances appear to remain low and could be

**Table 2** Outputs of regressions testing for changes in species catches over the years for time periods of 1985–2000 and 2001–2018, where AC and DC electrofishing were used respectively

| | 1985–2000 AC | | | | 2001–2018 DC | | | | | 1985–2000 AC | | | | 2001–2018 DC | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Species | Yr | Wwy | Inter. | Tr. | Yr | Wwy | Inter. | Tr. | Species | Yr | Wwy | Inter. | Tr. | Yr | Wwy | Inter. | Tr. |
| **Alewife** | NS | ** | NS | | NS | NS | NS | | Longnose Dace | NS | NS | NS | | NS | NS | NS | |
| **Banded Killifish** | NS | NS | NS | | *** | *** | NS | + | Mimic Shiner | NS | NS | NS | | NS | NS | NS | |
| Black Buffalo | NS | NS | NS | | NS | NS | NS | | **Mosquitofish** | NS | NS | NS | | ** | ** | NS | + |
| **Black Bullhead** | NS | *** | NS | | NS | NS | NS | | Nile Tilapia | NS | NS | NS | | NS | NS | NS | |
| Black Crappie | NS | NS | NS | | NS | NS | NS | | Ninespine Stickleback | NS | NS | NS | | NS | NS | NS | |
| Blackstripe Topminnow | NS | NS | NS | | NS | NS | NS | | Northern Pike | NS | NS | NS | | NS | NS | NS | |
| **Bluegill** | *** | *** | *** | + | *** | *** | * | + | **Orangespotted Sunfish** | NS | NS | ** | – | NS | NS | NS | |
| **Bluntnose Minnow** | ** | *** | ** | + | NS | *** | NS | | **Oriental Weatherfish** | NS | NS | NS | | *** | NS | NS | + |
| **Brook Silverside** | NS | NS | NS | | * | ** | NS | + | **Pumpkinseed** | NS | *** | * | – | *** | *** | ** | + |
| **Brook Stickleback** | ** | NS | NS | – | NS | NS | NS | | Quillback | NS | NS | NS | | NS | NS | NS | |
| Brown Bullhead | NS | NS | NS | | NS | NS | NS | | Rainbow Smelt | NS | NS | NS | | NS | NS | NS | |
| **Common Carp** | NS | *** | NS | | NS | *** | *** | – | Rainbow Trout | NS | NS | NS | | NS | NS | NS | |
| **Carp X Goldfish** | ** | *** | NS | – | NS | NS | NS | | **Rock Bass** | ** | NS | NS | + | NS | *** | NS | |
| Central Mudminnow | NS | NS | NS | | NS | NS | NS | | **Round Goby** | NS | NS | NS | | NS | *** | NS | |
| **Channel Catfish** | NS | NS | NS | | *** | NS | * | + | Sand Shiner | NS | NS | NS | | NS | NS | NS | |
| Chinook Salmon | NS | NS | NS | | NS | NS | NS | | Skipjack Herring | NS | NS | NS | | NS | NS | NS | |
| Creek Chub | NS | NS | NS | | NS | NS | NS | | **Smallmouth Bass** | *** | NS | NS | + | NS | *** | NS | |
| **Emerald Shiner** | NS | NS | NS | | NS | *** | NS | | Smallmouth Buffalo | NS | NS | NS | | NS | NS | NS | |
| **Fathead Minnow** | *** | *** | *** | – | NS | NS | NS | | **Spotfin Shiner** | NS | NS | NS | | ** | *** | NS | + |
| Flathead Catfish | NS | NS | NS | | NS | NS | NS | | **Spottail Shiner** | ** | NS | NS | – | NS | NS | NS | |
| Freshwater Drum | NS | NS | NS | | NS | NS | NS | | Tadpole Madtom | NS | NS | NS | | NS | NS | NS | |
| **Gizzard Shad** | * | *** | *** | + | NS | NS | ** | – | Threadfin Shad | NS | NS | NS | | NS | NS | NS | |
| **Golden Shiner** | *** | *** | NS | – | ** | *** | NS | + | White Bass | NS | NS | NS | | NS | NS | NS | |
| **Goldfish** | *** | *** | ** | – | NS | *** | NS | | White Crappie | NS | NS | NS | | NS | NS | NS | |
| Grass Carp | NS | NS | NS | | NS | NS | NS | | **White Perch** | NS | NS | NS | | NS | *** | NS | |
| Grass Pickerel | NS | NS | NS | | NS | NS | NS | | **White Sucker** | NS | NS | NS | | NS | *** | * | + |
| **Green Sunfish** | NS | *** | NS | | *** | *** | *** | + | Yellow Bass | NS | NS | NS | | NS | NS | NS | |
| Hybrid Sunfish | NS | NS | NS | | NS | NS | NS | | **Yellow Bullhead** | NS | NS | NS | | ** | *** | NS | + |
| **Largemouth Bass** | *** | *** | *** | + | NS | *** | NS | | **Yellow Perch** | *** | NS | NS | – | ** | NS | NS | + |

Negative binomial regressions included year (Yr), waterway (Wwy), and their interaction effects (Inter.) with sampling time as an offset. Significance denoted with: NS = non-significant, * = < 0.1, ** = < 0.05, and *** = < 0.01. The general trend (Tr.) of the effect for year (i.e, the general direction of change) is denoted using "+" for increases and "-" for decreases in catches. Species with significant results are bolded

considered rare. Similarly, increases in species richness since the mid-1970's were shown to be due to increases in sportfish and native richness indices downstream of Chicago on the Illinois River (Gibson-Reinemer et al. 2017b) as well as several other midwestern riverine systems (Mapes et al. 2015; Holloway et al. 2018; Pyron et al. 2019). Parker et al. (2016) provide evidence that the improvements in fish assemblages downstream of Chicago were associated with reductions in sewage-related variables (e.g. ammonia and fecal-coliform bacteria). The results of studies downstream (e.g., Parker et al. 2016, 2018; Gibson-Reinemer et al. 2017b), together with ours suggest that reduced sewage effluents within the Chicago area and improvements in wastewater treatment have had impacts on the fish community locally (within Chicago) as well as further downstream.

Changes in species-specific relative abundances (as judged by CPUE) do not offer strong inferences on changes in water quality. For example, many of the species that increased in abundance (e.g., Banded Killifish, Bluegill, Green Sunfish, Pumpkinseed, Yellow Bullhead, etc.) are considered either neutral to or tolerant of pollution (Poff and Allan 1995; Grabarkiewicz and Davis 2008). In seeming contrast, a report on the more natural (e.g., not fully channelized, riparian areas remain, not manually cleared of debris) Des Plaines River,

**Table 3** Expected marginal mean CPUEs (unit of effort = 30 min) for each waterway for species indicated to have significantly different CPUEs in Table 2

| Species | Cal-Sag Channel | Calumet River | Little Calumet River | Lockport Dam | North Shore Channel | Sanitary-Ship Canal |
|---|---|---|---|---|---|---|
| | 1985–2000 AC | | | | | |
| Alewife | 0.02(ab) | 0.02(b) | 0.01(ab) | NA | 0.96(a) | 0.10(b) |
| Black Bullhead | 0.01(ab) | NA | 0.07(b) | NA | 0.84(a) | 0.17(b) |
| Bluegill | 0.41(b) | 1.62(a) | 0.38(b) | 0.18(bc) | 1.89(a) | 0.05(c) |
| Bluntnose Minnow | 1.28(b) | 14.1o(a) | 0.77(b) | 0.80(b) | 7.00(a) | 8.53(a) |
| Common Carp | 5.48(b) | 2.12(c) | 6.02(b) | 5.56(b) | 2.34(c) | 14.70(a) |
| Carp X Goldfish | 0.43(bc) | 0.07(c) | 0.49(b) | 0.38(bc) | 1.75(a) | 0.78(b) |
| Fathead Minnow | <0.01(ab) | <0.01 (ab) | 0.24(b) | 0.02(ab) | 7.88(a) | 0.66(b) |
| Gizzard Shad | 9.42(abc) | 27.0(a) | 20.90(a) | 16.50(ab) | 4.49(bc) | 3.39(c) |
| Golden Shiner | <0.01(ab) | 0.21(b) | 1.17(b) | 0.14(b) | 5.97(a) | 0.35(b) |
| Goldfish | 1.77(c) | 0.19(d) | 6.75(ab) | 2.45(bc) | 6.88(ab) | 9.32(a) |
| Largemouth Bass | 0.28(bc) | 6.39(a) | 0.12(c) | 0.72(bc) | 0.67(b) | 0.46(bc) |
| Green Sunfish | 0.25(cd) | 1.48(ab) | 0.20(cd) | 0.69(bc) | 2.61(a) | 0.07(d) |
| Pumpkinseed | 0.03(bc) | 2.30(a) | 0.13(bc) | NA | 0.43(b) | 0.09(c) |
| | 2001–2018 DC | | | | | |
| Banded Killifish | <0.01(ab) | 0.38(a) | <0.01 (ab) | <0.01(ab) | <0.01 (ab) | <0.01 (b) |
| Bluegill | 2.00(c) | 7.65(ab) | 20.30(a) | 2.15(c) | 10.00(ab) | 7.93(b) |
| Bluntnose Minnow | 15.00(bc) | 61.20(a) | 5.65(cd) | 1.47(d) | 2.48(d) | 20.30(b) |
| Brook Silverside | 0.07(b) | 1.10(a) | <0.01 (ab) | NA | NA | NA |
| Common Carp | 13.70(bc) | 7.95(c) | 25.30(ab) | 2.56(d) | 15.40(abc) | 24.10(a) |
| Emerald Shiner | 13.60(a) | 16.80(a) | 8.35(ab) | 1.93(abc) | 0.17(c) | 1.50(bc) |
| Golden Shiner | 2.05(b) | 0.17(c) | 8.91(a) | 0.50(bc) | 13.80(a) | 6.62(a) |
| Goldfish | 0.51(bc) | 0.042(bc) | 12.40(a) | 0.01(abc) | 0.06(c) | 1.48(b) |
| Green Sunfish | 4.86(a) | 2.19(abc) | 1.76(bc) | 5.95(a) | 1.00(c) | 3.85(ab) |
| Largemouth Bass | 11.50(b) | 30.10(a) | 26.80(a) | 1.63(c) | 9.25(b) | 0.89(c) |
| Mosquitofish | NA | NA | NA | 0.45(b) | NA | 26.60(a) |
| Pumpkinseed | 0.83(c) | 6.46(b) | 30.30(a) | 4.84(b) | 7.53(b) | 30.60(a) |
| Rock Bass | NA | 7.95(a) | NA | NA | 0.19(b) | NA |
| Round Goby | 0.35(bc) | 2.73(a) | 1.42(ab) | <0.01(abc) | 0.05(c) | 0.07(c) |
| Smallmouth Bass | 0.05(b) | 15.70(a) | 0.04(b) | <0.01(ab) | NA | NA |
| Spotfin Shiner | 0.85(bc) | 0.82(bc) | 0.30(c) | 0.10(c) | 5.96(a) | 2.53(ab) |
| White Perch | 0.53(ab) | 0.33(b) | 3.82(a) | NA | 0.02(b) | 0.15(b) |
| White Sucker | NA | 2.22(b) | 7.74(a) | NA | 1.93(b) | <0.01(b) |
| Yellow Bullhead | 1.05(b) | <0.01 (ab) | 4.82(a) | 0.64(b) | 0.43(b) | 3.76(a) |

Those sharing letters are not significantly different (Tukey-Kramer). NA represents that this species was not found in this waterway and thus its CPUE cannot be estimated

which flanks the CAWS, indicated moderate decreases in proportions of tolerant species, increases in neutral species, and relatively minor increases in intolerant species over the same time period (Pescitelli and Widloe 2018). Increases in intolerant species linked to water quality improvements have been documented in in rivers in Illinois (Illinois River; Parker et al. 2018), Indiana (West Fork of the Wabash; Holloway et al. 2018), Ohio (the Scioto and Cuyahoga Rivers; Rahel 2010), and Texas (Trinity River; Perkin and Bonner 2016). In a watershed in Kansas, Whitney et al. (2019) found that species intolerant to pollution did not change in abundance and in fact species considered tolerant increased in abundance after water quality improvements. Similarly, although water quality downstream of the CAWS has improved over recent decades (Parker et al. 2016, 2018), there were no notably large increases in intolerant species within the CAWS. It is important to note that water quality is not a factor in isolation, and that fish populations respond to habitat factors beyond just water

**Table 4** Expected marginal trends across years (i.e., multiplicative rate of change in CPUE) for each waterway for species indicated to have significantly different CPUEs in Table 2

| | 1985–2000 AC | | | | | |
|---|---|---|---|---|---|---|
| Species | Cal-Sag Channel | Calumet River | Little Calumet River | Lockport Dam | North Shore Channel | Sanitary-Ship Canal |
| Bluegill | 0.993(ab) | 0.899(b) | 0.853(b) | 1.01(ab) | 1.22(a) | 1.02(ab) |
| Bluntnose Minnow | 1.1(ab) | 0.813(b) | 1.06(ab) | 1.04(ab) | 0.898(b) | 1.11(a) |
| Fathead Minnow | 2.81(ab) | 0.0974(ab) | 0.781(a) | 0.634(ab) | 0.425(b) | 1.19(a) |
| Gizzard Shad | 1.02(b) | 0.939(b) | 0.983(b) | 1.03(b) | 1.44(a) | 1.22(ab) |
| Goldfish | 0.724(ab) | 1.03(a) | 0.754(ab) | 0.939(a) | 0.807(ab) | 0.69(b) |
| Largemouth Bass | 1.3(ab) | 1.03(b) | 1.68(a) | 1.38(a) | 1.39(a) | 1.36(a) |
| Orangespotted Sunfish | 0.729(ab) | 0.476(b) | NA | NA | 1.09(a) | NA |
| Pumpkinseed | 0.993(ab) | 0.848(b) | 1.26(ab) | NA | 1.15(a) | 1.33(a) |
| | 2001–2018 DC | | | | | |
| Bluegill | 1.17(abc) | 1.11(bc) | 1.11(c) | 1.39(ab) | 1.17(abc) | 1.32(a) |
| Common Carp | 0.99(a) | 0.947(a) | 0.962(a) | 0.731(b) | 0.975(a) | 0.989(a) |
| Channel Catfish | 1.54(a) | 201(abc) | NA | 0.844(c) | 1.07(bc) | 1.12(ab) |
| Gizzard Shad | 1.11(a) | 0.871(b) | 1.05(ab) | 1.13(a) | 0.925(b) | 0.935(b) |
| Green Sunfish | 1.03(b) | 1.2(ab) | 1.07(b) | 1.37(a) | 1.03(b) | 1.08(b) |
| Pumpkinseed | 1.15(ab) | 1.26(a) | 1.15(a) | 1.26(a) | 0.975(b) | 1.09(ab) |
| White Sucker | NA | 0.915(b) | 0.998(b) | NA | 1.28(a) | 1.83(ab) |

Values <1 indicate decreases in abundance, whereas values >1 indicate increases in CPUEs. Those sharing letters are not significantly different (Tukey-Kramer). NA represents that this species was not found in this waterway and thus its CPUE cannot be estimated

quality (i.e., canopy cover, riparian areas, local land use, overwinter habitat, etc.; Raibley et al. 1997; Sawyer et al. 2004). To this point, a study by LimnoTech (2010) indicated that in the early 2000's, habitat quality was more important to predicting fish community indices within the CAWS than dissolved oxygen (a single proxy of many for water quality). Our evidence of few to no populations of pollution-intolerant species increasing in abundance (except Smallmouth Bass *Micropterus dolomieu* and Rock Bass *Ambloplites rupestris*) compared to other species, combined with the habitat studies mentioned suggests that both water quality as well as habitat remain as limiting factors in the CAWS.

The seemingly exponential increase in Banded Killifish relative abundance is of interest to regional biologists as the species is listed as state-threatened in Illinois (IESPB 2015, 2019). Although classified by MWRD as one species, both Western (*F. diaphanus menona*) and Eastern (*F. diaphanus diaphanus*) subspecies of Banded Killifish occur in the area and appear to readily hybridize (Willink et al. 2018). It has been suggested that Eastern Banded Killifish populations have migrated from Lake Michigan into the CAWS system as several Banded Killifish of unknown subspecies appeared in seine surveys of Illinois beaches in 2001 (Willink et al. 2018). Banded Killifish first appeared in the CAWS in 2010 at SEPA station 1, near our Calumet River location, at which they did not appear until 2012, and were later found in other parts of the CAWS. Causative agents for the exponential population growth of the Banded Killifish in the CAWS remains uncertain. It has been suggested that the Eastern subspecies is more tolerant to pollution and has less strict vegetative demands for spawning (Trautman 1981), and thus the Eastern subspecies (or hybrids) may be able to take advantage of degraded habitats the Western subspecies once utilized.

Decreases in Common Carp have been shown throughout the Illinois River as well as along the upper Mississippi (Gibson-Reinemer et al. 2017a). Such declines, believed to have begun in the 1960's, were thought to be due to disease as other possibilities (i.e., water quality, predation, competition) had little or contradicting support. We note that Common Carp populations in the CAWS declined in 2001–2018, especially at the Lockport Lock and Dam sampling site compared to other waterways included herein. Declines were not noted between 1985 and 2000 and populations throughout the CAWS appeared steady during this period. Whether a similar lack of recruitment is experienced by this population as those described by Gibson-Reinemer et al. (2017a) is unclear and may be of further research interest given the spatial differences in noted declines in the CAWS compared to across the Mississippi River drainage.

Switching from AC to DC electrofishing changes which species are susceptible to electrofishing (McClelland et al. 2013). Some species in the CAWS were caught only with one type of electrical current and within that gear type did not have significant changes in abundance suggesting that

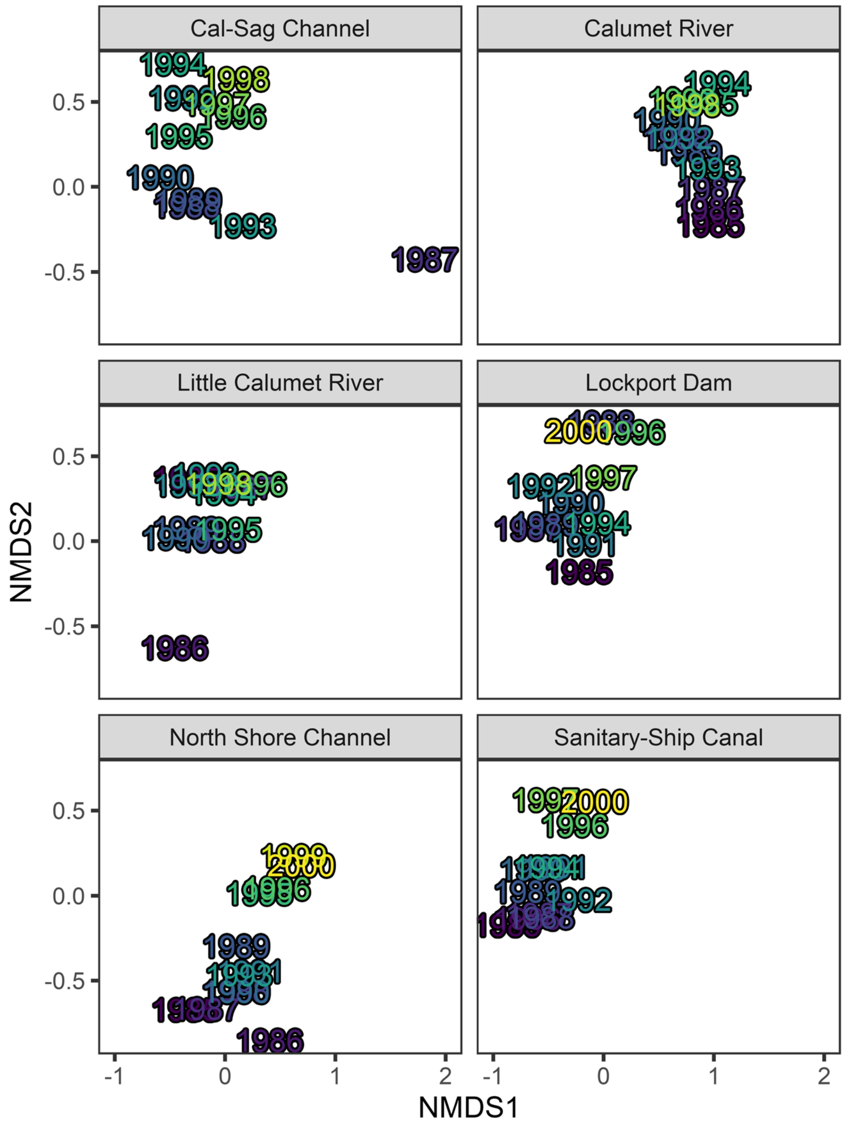


**Fig. 4** Changes in fish communities of six Chicago waterways between 1985 and 2000 visualized using nMDS, faceted for each waterway to reduce overlap. Points represent yearly averaged nMDS scores for each waterway. nMDS scores based on Bray-Curtis similarities among square-root transformed and Wisconsin double standardized fish CPUE data

switching gears did affect which species were susceptible to the gear. Although not formally tested, Brook Stickleback and Spotfin Shiner *Cyprinella spiloptera* each occurred for multiple years under a single electrofishing regime (6 AC-years and 15 DC-years respectively). A comparative study conducted on the Illinois River suggests that biomass captured as well as species richness are increased with DC electrofishing over AC electrofishing (McClelland et al. 2013). As gear-related differences in species richness were expected, we chose to separate data by the time periods the different electrofishing types were used. Within the recent DC (2001–2018) period of sampling there were larger increases in species richness than in the AC (1985–2000) period, offering evidence that the CAWS is home to a greater diversity of fishes now than any period included in the study.

The uniqueness of the Calumet River sampling location was evident in both the 1985–2000 and 2001–2018 nMDS ordinations. This sampling location is proximal to Lake Calumet, which hosts a fish community routinely targeted by tournament anglers (anglers-dream.com). Lake Calumet is a heavily modified lake that is largely shallow but has several man-made slips that are deep enough for barge traffic. Some slips within the CAWS have been shown to host more diverse fish communities than the mainstem of the waterways (Gallagher and Wasik 2018), and slips potentially act as refuges for fishes during poor water quality events (Gaulke et al. 2015). Slips have also been known to have higher abundances of Largemouth Bass than encountered in the CAWS proper (Gallagher and Wasik 2018). This sampling location routinely received higher index of biotic integrity scores, than others included herein, when assessed by MWRD biologists (Gallagher et al. 2014). When compared to other waterways herein, the diverse fish community at this location, as well as increases in sportfish here, offers evidence that habitat availability may be a larger limiting factor within other areas of the CAWS at this point in time.

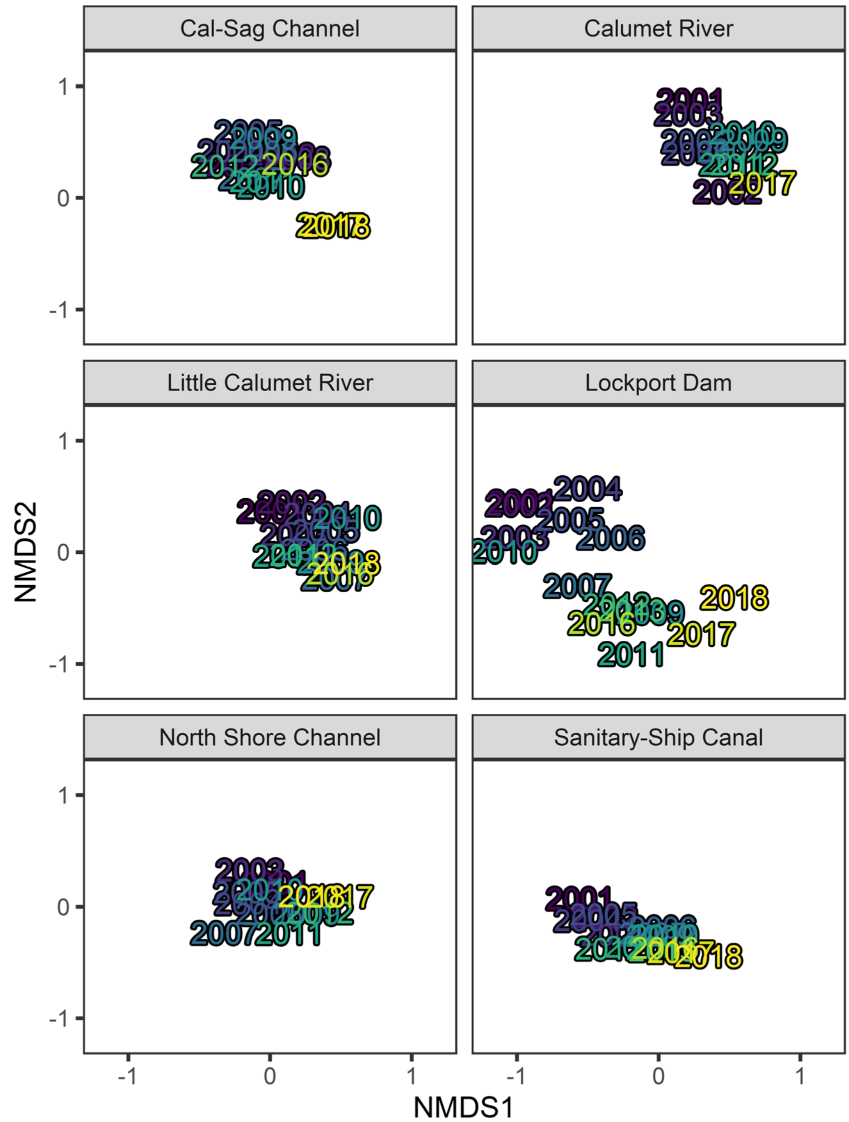


**Fig. 5** Changes in fish communities of six Chicago waterways between 2001 and 2018 visualized using nMDS, faceted for each waterway to reduce overlap. Points represent yearly averaged nMDS scores for each waterway. nMDS scores based on Bray-Curtis similarities among square-root transformed and Wisconsin double standardized fish CPUE data

Given the level of engineering performed to construct and maintain CAWS, and other working rivers, future fish-habitat enhancements likely need to be creative compared to traditional restoration activities (Francis 2009). Any in-river habitat improvements must preserve the functionality of the river as a shipping conduit, and thus also be able to withstand the hydrodynamics caused by barge traffic. Large woody debris is well known for a number of benefits in riverine systems (Larson et al. 2001; Gurnell et al. 2005), however to maintain navigability, would need to be attached to and flush with existing artificial structures. For example, timber fenders added to Deptford Creek in London has supported new plant communities on sheet pile, typically void of vegetation (Francis and Hoggart 2008). Riprap (i.e., mixed composition rock piles) used to stabilized channelized banks has been shown to increase abundances of fish species over bare mud banks (White et al. 2010). Protection from current and wave action has been trialed by creating planted wetlands along the banks behind existing sheet pile walls, allowing a diversity of aquatic and riparian plants to grow (Weber et al. 2012), which may potentially act as nursery areas for fishes. Floating enhancements such as docks and pontoons or less anthropocentric methods like artificial floating wetlands provide shelter, shade, and spawning possibilities (Schanze et al. 2004). Despite reports and presentations of such creative enhancements, published outputs of their effectiveness in enhancing fish communities remain sparse and more evidence is needed to support such tools in urban river management.

In conclusion, directional improvements have occurred in the fish assemblages of the Chicago Area Waterway System. Although a pristine riverine community for comparison is unavailable, nor data from prior to Chicago's development, management and policy actions have likely elicited the noted enhancements toward including more native species, typically

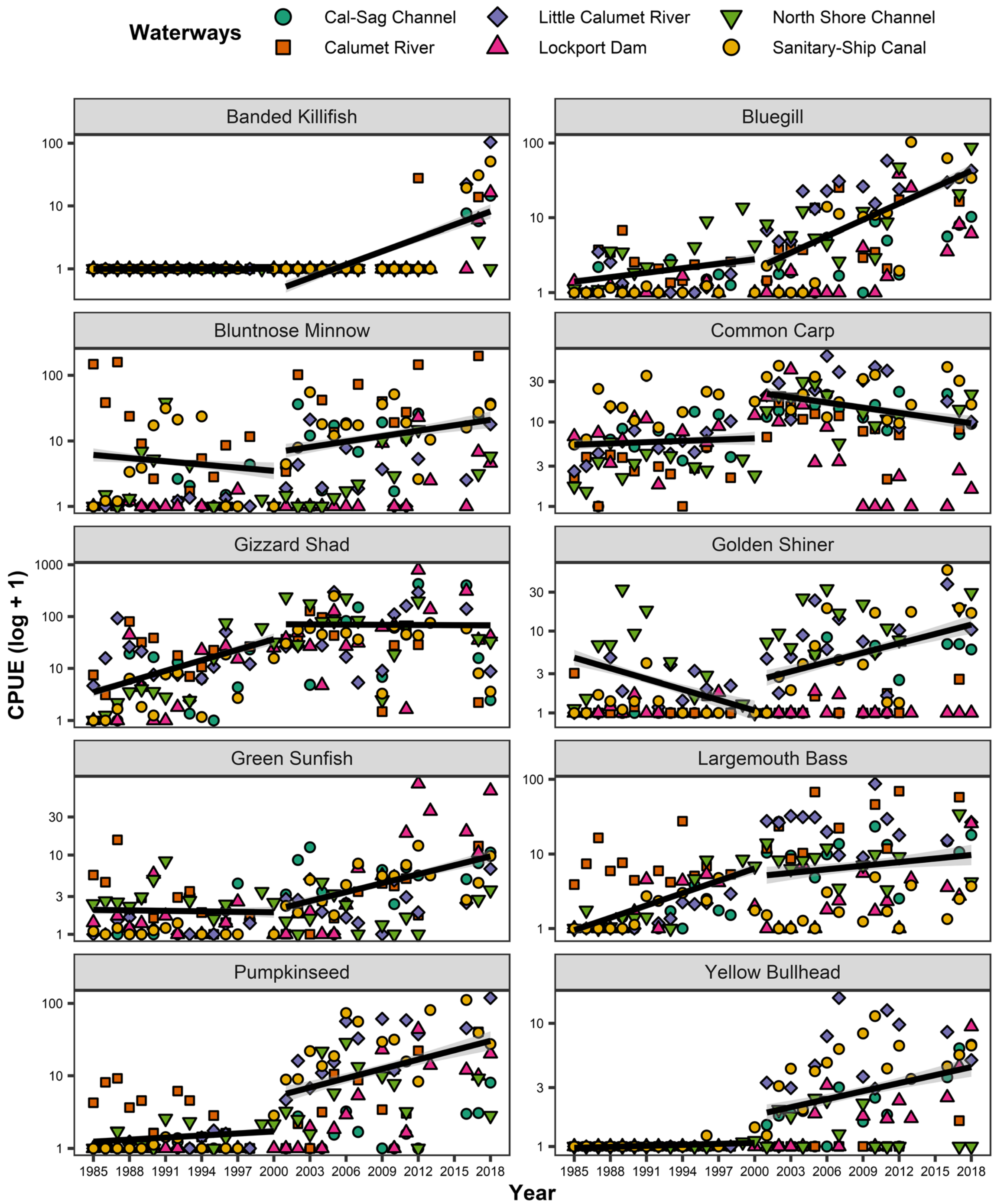


**Fig. 6** Changes in selected species' CPUEs (30 min electrofishing; log+1 transformed) in six waterways within the Chicago Area Waterways from 1985 to 2018. Negative binomial regressions used separately for the periods of 1985–2000 and 2001–2018 due to different fishing gears being used, and included sampling time as an offset. To reduce clutter, yearly mean CPUEs for each waterway are displayed as points instead of all of the data

thought of as signals of less degraded systems. Continued improvements to water infrastructure and increased attention to submerged or in-water habitat quality is likely needed to further aide fish assemblages in similar highly-modified urban rivers. Beyond the assessment of assemblage-level changes, our understanding of urban rivers as their own subset of ecosystem is sparse (Francis 2012). As Chicago and other urban centers strive to support fishable waterways (a general condition of The Clean Water Act; Karr and Dudley 1981), interests may lay in investigating water quality and habitat variables affecting population vital rates, food web structures, and life history characteristic variations within such urban ecosystems.

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