Fish Sampling Bias Associated with Stream Access

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Abstract: We investigated the effect of sampling site access on estimates of fish abundance in 2 eastern Oklahoma streams. Centrarchid species were sampled by electrofishing at public and remote access sites on Baron Fork Creek in northeastern Oklahoma and Glover River in southeastern Oklahoma. We verified differences in recreational use and habitat between access types in both streams. Recreational use was generally higher at public than remote access areas in each stream. Public areas in Glover River had higher fish densities, were deeper, and had more instream cover than remote areas. However, mean density of fish, mean depth, and frequency of cover types at public and remote areas in Baron Fork Creek were not significantly different. Although we did not observe a consistent trend in fish abundance between streams at public and remote access sites, our findings indicate that accessibility sampling from public access areas may yield biased estimates of population size. Therefore, we urge caution when making inferences about populations based on samples taken solely from these areas.

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Sampling for fish frequently occurs where it is convenient, logistically feasible, and least threatening to those conducting the work (Johnson 1983). Streams present a particular sampling challenge because they often course through remote areas where roads are sparse and access is difficult. When accessible, the shallow nature of streams can preclude the movement of sampling equipment in water to remote sites. Furthermore, stream access is often privately owned and permission is needed for entrance. Consequently, easily accessible sites, where roadways intersect streams at bridge and low-water crossings that are deep enough (e.g., pools) for gear maneuverability are

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typically sampled in stream fisheries surveys. This type of non-probability sampling is termed accessibility sampling (Krebs 1989).

When accessibility sampling is used to estimate abundance of stream fishes, the resulting data may be biased. Human activity at public access areas, such as angling and swimming, can directly remove or indirectly displace fish. For example, public access areas (e.g., bridge crossings) often have higher angling pressure and harvest rates than remote access areas, the latter of which usually have restricted shoreline access and thus lower angling pressure. Intense exploitation associated with public access areas can reduce standing crop (Fajen 1975) and production (Reed and Rabeni 1989), which may result in bias toward underestimating fish population size. In addition, bridge structures and other channel modifications that alter stream habitat at public access areas may influence fish distribution and abundance.

Besides anthropogenic influences, biological populations are naturally distributed in a complex manner such that abundance estimates based on sampling at highly accessible areas may or may not be representative depending on local conditions (Krebs 1989). For example, fishes are not uniformly distributed longitudinally in a stream, but instead are associated with various habitats (Paragamian 1981, McClendon and Rabeni 1987, Lyons 1991). However, even if suitable habitat is available in some form throughout a stream, not all stream locations are equally used by fishes (Funk 1975, Todd and Rabeni 1989). Thus, longitudinal distribution patterns and habitat-based aggregations of stream fish makes site-to-site comparisons of population estimates difficult (Hendricks et al. 1980, Waters and Erman 1990) and expanded abundance estimates potentially misleading (Hankin and Reeves 1988, Hawkins et al. 1993).

We investigated the effect of sampling site access on estimates of fish abundance. We compared the abundance of centrarchids in 2 eastern Oklahoma streams at easily accessible public use areas, which had high angling pressure and recreational activities, to that at remote areas where public access was limited and angling pressure was low.

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Methods

Baron Fork Creek and Glover River are scenic and unregulated waterways in eastern Oklahoma. Baron Fork Creek is located in the Ozark Plateau region of east central Oklahoma (Fig. 1) and is characteristic of streams in this region with clear and cool water, stable base flow maintained by springs, and a gravel-dominated substrate.

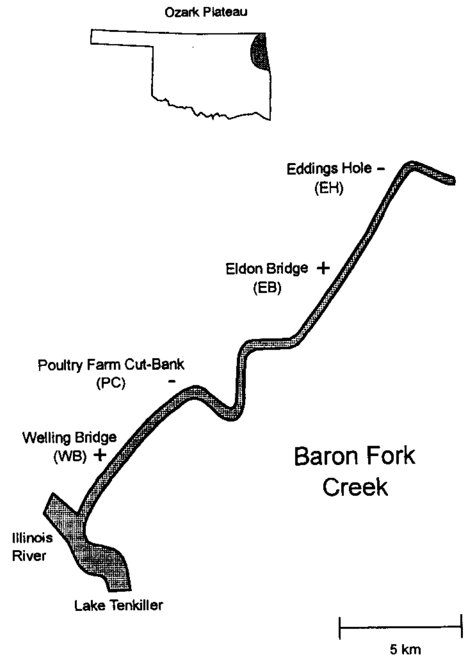

Figure 1. Study area on Baron Fork Creek in the Ozark Plateau region of eastern Oklahoma. Plus signs denote public access and minus signs denote remote access sampling sites.

Baron Fork Creek originates in Arkansas and flows westward for 56.9 km (Okla. Water Resour. Board 1990) through Adair and Cherokee counties, Oklahoma, draining 936 km^2 (Storm et al. 1996) before it joins the Illinois River above Lake Tenkiller. Major land use in the basin includes cattle grazing, forestry (Okla. State Environ. Inst. 1996), and numerous poultry operations (Nolan et al. 1989). Glover River drains the rugged and remote Ouachita Mountains of southeastern Oklahoma (Fig. 2) and is typical of streams in this region with a substrate dominated by emergent bedrock and flow primarily from runoff. Glover River begins in Pushmataha and Leflore counties, Oklahoma and flows southerly for 54.2 km (Okla. Water Resour. Board 1990) through McCurtain County, Oklahoma, draining 876 km^2 (Orth and Maughan 1984) before joining the Little River. The basin is heavily forested and supports intensive silviculture activities, including forest clear-cutting and associated road building (Rutherford et al. 1992), and cattle grazing.

Baron Fork Creek and Glover River differ in the type and ease of access to the stream for recreational activities and fish sampling. The land bordering Baron Fork Creek is mostly private, and public use within our study area was restricted to 3 public access areas, 2 highway bridge crossings, and a confluence access area at the Illinois River. In Baron Fork Creek, favorable flow and substrate conditions and permission to enter private lands allowed us to transport sampling gear to remote areas

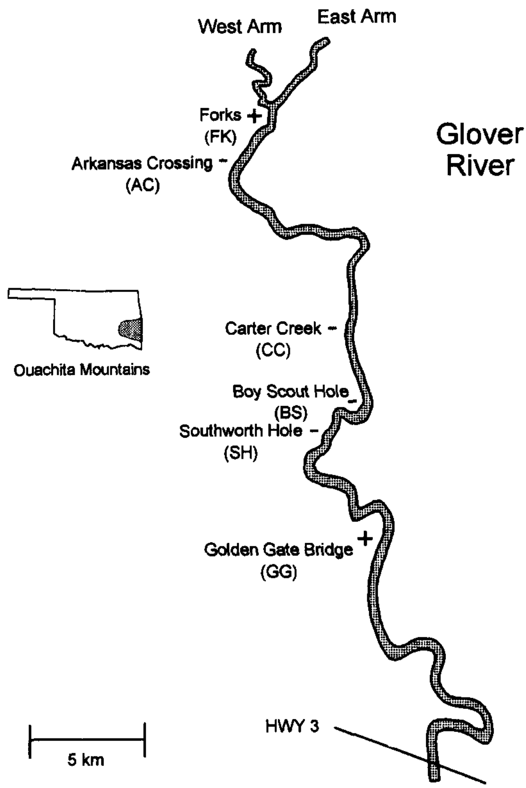

Figure 2. Study area on Glover River in the Ouachita mountains of southeastern Oklahoma. Plus signs denote public access and minus signs denote remote access sampling sites.

between public access sites. Conversely, the Weyerhaeuser Company grants unlimited public access to their land bordering the Glover River study area. However, the remote nature of this stream limits physical access to low-water bridge crossings (public access areas) and some unimproved logging roads (remote access areas) that end at the stream. Additionally, low summer flows and exposed bedrock substrate prohibited movement of sampling equipment up or downstream from public to remote access sites. We entered remote access areas in Glover River through logging roads until we were close enough to carry sampling equipment to the stream.

Stream study reaches were defined, and potential sampling sites were chosen along a 16.7-km section of Baron Fork Creek (Fig. 1) and a 38.6-km section of Glover River (Fig. 2). Study sites were randomly selected in each stream from a pool of potential sampling sites including both public and remote access areas. Sampling was conducted at 2 public access sites (Eldon bridge and Welling bridge) and 2 remote access sites (Eddings hole and poultry farm cut-bank) on Baron Fork Creek (Fig. 1) and 2 public access sites (forks and Golden Gate bridge) and 4 remote sites

(Arkansas crossing, Boy Scout hole, Carter Creek, and Southworth hole) on Glover River (Fig. 2). Centrarchid species (black basses *Micropterus* spp., sunfishes *Lepomis* spp., and rock bass *Ambloplites rupestris*) were collected and stream habitat was measured at these sites during October 1993 and August–October 1994 and 1995 when the streams were at or near base flow. Sampling site area ranged from 0.12 to 1.17 ha (Table 1), and there was no change in stream order between upstream and downstream sites in both streams.

Fish were collected by electrofishing with pulsed direct current from a boat equipped with a Smith-Root 2.5 GPP electrofisher. Before sampling, each site was block-netted at the upstream and downstream end with block nets (55 m × 1.8 m with 12.7-mm^2 mesh) to prevent emigration or immigration. The area of each site was calculated by multiplying mean stream width between the block nets by site length. The enclosed areas were thoroughly sampled twice to mark and recapture marked fish. After each sampling run, captured fish were held in an instream pen for subsequent processing. Following the first electrofishing (marking) run, captured fish were marked (partial caudal fin clip) and released back into the enclosed area and left undisturbed for about 2 hours to allow them to disperse. After the second

Table 1. Population size and 95% confidence intervals and total density (*N*/ha) estimates for centrarchids (*Micropterus* spp., *Lepomis* spp., and *Ambloplites rupestris*) collected at public and remote access sampling sites in Baron Fork Creek and Glover River, Oklahoma, 1993–1995. Abbreviated site names correspond with those in Figures 1 and 2.

Site and year	Access type	Sample area (ha)	Population estimate	Confidence interval	Density
		Baron Fork Creek			
EB 1993	public	0.17	169	86–836	994
EB 1994	public	0.40	257	178–441	643
EB 1995	public	0.41	373	319–455	910
WB 1994	public	0.45	534	364–928	1,187
WB 1995	public	0.30	288	174–693	960
EH 1994	remote	0.46	543	376–929	1,180
EH 1995	remote	0.50	713	495–1,207	1,426
PC 1994	remote	0.32	142	89–315	444
PC 1995	remote	0.43	479	347–749	1,114
		Glover River			
FK 1994	public	0.80	478	335–787	598
FK 1995	public	0.12	901	785–1,070	7,508
GG 1993	public	0.24	1,194	825–2,009	4,975
GG 1994	public	0.24	2,093	1,311–3,450	8,721
GG 1995	public	0.21	1,532	1,287–1,939	7,295
AC 1994	remote	0.40	348	253–542	870
AC 1995	remote	0.52	1,231	736–2,337	2,367
BS 1994	remote	0.42	820	463–1,724	1,952
BS 1995	remote	0.55	1,617	1,105–2,608	3,038
CC 1993	remote	0.72	1,464	891–2,476	2,033
SH 1993	remote	1.17	554	216–1,228	474

electrofishing (recapture) run, collected fish were examined for marks and released. Total number of fish captured and number of marked fish was recorded for estimates of population size (Ricker 1975).

Total population estimates at each site were calculated using Chapman's (1951) modification of the Petersen method (Ricker 1975). Approximate confidence limits, based on either Poisson, binomial, or normal distributions, were calculated using the guidelines of Seber (1982). Density (*N*/ha) was estimated by dividing estimated population size by area sampled at each site. We analyzed fish density estimates from 1993, 1994, and 1995 for each stream and access type with *t*-tests to examine the null hypothesis that mean density did not differ between public and remote access areas. For this comparison we made the following assumptions: (1) angler harvest and/or recreational activity was greater at public than at remote access areas; (2) habitat was similar between public and remote access areas; and (3) fish abundance for each access type was similar among years.

The assumption of greater angler activity at public vs. remote access sites was verified with findings from a concurrent creel study by Martin (1995). Martin also observed considerable non-fishing recreational activity at public access sites in Baron Fork Creek.

To test the assumption that habitat was similar between public and remote access areas, stream depth and instream cover were measured at each site with the transect method described by Todd and Rabeni (1989). Measurements were taken along 3 or more transects across the stream beginning at the upstream end and proceeding to the downstream end of each site. Frequency of transect locations was based on relative homogeneity of habitat (i.e., more transects were used in pools with heterogeneous habitat, fewer in those with homogeneous habitat). Depth and cover were measured at 4 equally-spaced points along each transect. Individual depth measurements were scaled to the maximum depth to account for variation caused by extremely deep or shallow sites. Principal instream cover features at each station were classified into 1 of the following categories: type 1—open water areas with small grain or bedrock substrates that were free of obstructions such as submerged or floating woody debris; type 2—open water areas with course grain substrates and/or uneven channel contours (bedrock ledges); and type 3—areas with inundated or exposed brush piles, rootwads, or log jams along the channel margins. Six other cover types occurred at such low frequencies (< 8 observations) that they were excluded from further analyses. For each stream, we tested for differences in standard depth with a *t*-test and for differences in cover with a chi-square test. We tested the assumption that mean fish abundance for each access type was similar between 1994 and 1995 for each stream with a *t*-test. This assumption could not be verified for 1993 because < 2 samples were collected from each access type.

Results

A total of 4,283 fish representing 9 centrarchid species including largemouth bass (*M. salmoides*), smallmouth bass (*M. dolomieu*), spotted bass (*M. punctulatus*), rock bass, green sunfish (*L. cyanellus*), warmouth (*L. gulosus*), orangespotted sunfish (*L. humilis*), bluegill (*L. macrochirus*), longear sunfish (*L. megalotis*), and redear sunfish (*L. microlophus*) were captured in 20 samples from the 2 streams. Smallmouth bass dominated the black bass catch in both public and remote access areas in each stream, followed in order by largemouth bass and spotted bass in Baron Fork Creek, and spotted bass and largemouth bass in Glover River. In Baron Fork Creek, longear sunfish dominated the sunfish catch in public access areas followed in order by rock bass and bluegill, but remote access areas were dominated by bluegill, followed by longear sunfish and rock bass. Sunfish in public and remote access areas in Glover River were dominated by longear sunfish, followed in order by green sunfish and bluegill.

Fish densities at the 10 sites ranged widely (444–8,721 fish/ha; Table 1), but were not significantly different at public and remote access areas in 1994 and 1995 in Baron Fork Creek ($t = -0.0732$, $P = 0.9483$; and $t = -1.1459$, $P = 0.3705$, respectively) or Glover River ($t = -0.6749$, $P = 0.5693$; and $t = -2.0288$, $P = 0.1796$, respectively). Confidence intervals around the population estimates were large due to small sample sizes and low recapture rates of marked individuals. Mean fish densities at public and remote access areas in Baron Fork Creek were not significantly different ($t = -0.488$, $P = 0.641$; Fig. 3). Contrastingly, fish were significantly more abundant at public than at remote access areas in Glover River ($t = 2.703$, $P = 0.047$; Fig. 3). The high variability in these estimates is indicated by the large standard errors around the means for both access types in both streams.

We found habitat to be similar between access types in Baron Fork Creek but different in Glover River. In Baron Fork Creek, mean standard depth was similar between access types ($t = 0.918$, $P = 0.359$, $N = 476$). However, public access areas in Glover River were significantly deeper than remote access areas ($t = 4.471$, $P = 0.001$; $N = 244$). We found 3 major cover habitats in Baron Fork Creek: areas with no available cover (type 1; $N = 334$), areas with instream large rocks or bedrock ledges (type 2; $N = 80$), and areas with shoreline woody debris (type 3; $N = 23$). Glover River also had appreciable amounts of type 1 ($N = 79$) and type 2 ($N = 220$) cover. In Baron Fork Creek, types 1, 2, and 3 did not differ between public and remote access areas ($X^2 = 4.796$, $df = 2$, $P = 0.091$). Conversely, in Glover River, remote access areas were disproportionately dominated by type 2 cover habitat compared to public access ($X^2 = 7.698$, $df = 1$, $P < 0.006$).

Discussion

Our estimates of fish abundance showed bias associated with sampling at public vs. remote access areas in Glover River, but not in Baron Fork Creek. However, data supporting this conclusion were highly variable, which is partially attributable to low sample sizes and low recapture rates of marked fish. With more samples and/or higher recapture rates, the variances around the means of these abundance estimates would likely have decreased, and statistical results might have been more conclusive. Abundance estimates gathered from a larger data set (multiple streams or years) would be

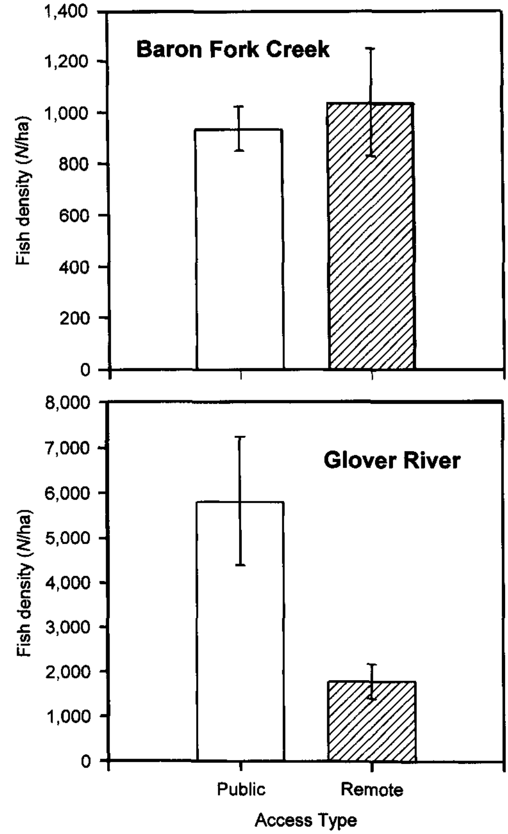

Figure 3. Comparison of mean fish density ±1 SE between public and remote access areas in Baron Fork Creek and Glover River, Oklahoma.

needed to achieve greater confidence in the statistical results for this type of analysis.

The differences we observed in fish abundance between public and remote access areas in Glover River may have been attributable to habitat effects. It is well documented that abundance and distribution of stream fish, and centrarchids in particular, is strongly correlated with instream habitat characteristics (Paragamian 1981, McClendon and Rabeni 1987, Todd and Rabeni 1989, and Lyons 1991). Rankin (1986) reported that smallmouth bass generally preferred habitats with coarse substrate in deep areas. Todd and Rabeni (1989) noted smallmouth bass were commonly associated with rocky substrates, and Rabeni and Jacobson (1993) observed most adult smallmouth bass in deep pools with variable substrates. Remote areas in Glover River had a higher availability of coarse substrates and bedrock ledges, and were significantly shallower than public access areas. Low water bridges at most public access areas in Glover River impounded water above and increased channel scouring and removal of fine-grain substrate particles below these structures.

We did not detect a habitat effect in Baron Fork Creek. Bridges over this stream are suspended above the water and do not obstruct flow; consequently, we found no differences in depth and cover between access types. Study reaches without instream channel modifications should provide better sampling sites than those with modifications (e.g., low-water bridges) because habitat is not significantly altered by the influence of structures on stream hydraulics.

The assumption that angler effort, harvest, and other human activities that could displace sport fish were greater at public access areas than remote access areas was verified by Martin (1995). He found that 90% of anglers in Glover River used highly accessible (public) areas, but that angler preference for these sites was not as apparent in Baron Fork Creek. However, highly accessible areas in Baron Fork Creek were heavily used by non-angling recreationalists. Similarly, we observed considerable human activity (fishing and swimming) in the 2 streams, and most of it occurred at public access sites. While it is known that these activities can significantly influence standing crop and production at public access areas, it is not known how quickly fish communities recover from these perturbations. Stream fish densities in remote areas may be high enough that fish from these areas disperse into voids left in public access areas depleted by harvesting fish (Funk 1975). Other effects of recreational activities, such as swimmers dislodging stream benthos, may actually serve to temporarily increase fish densities at public access sites. Experimental testing for differences in angler effort, harvest, and human activities and the associated fish response between public vs. remote access areas is needed to improve our understanding of stream sampling biases.

Although we were not able to detect a consistent trend between streams in sampling bias related to stream access, our findings from Glover River indicate that accessibility sampling from public access areas may yield biased estimates of fish population size. Therefore, caution should be used when making inferences about populations based on samples taken only from accessible areas. Fisheries biologists can minimize bias during surveys and improve estimates of population size by optimizing their sampling design with probability sampling (Waters and Erman 1990, Wilde and Fisher 1996). For example, simple random, stratified random, systematic, or cluster sampling can help ensure that data have not been biased by observer preferences (Johnson 1983, Thompson 1992, Wilde and Fisher 1996) and that estimates can be applied to the entire population, not just the locations sampled (Waters and Erman 1990, Wilde and Fisher 1996). Using these sampling designs in complex natural ecosystems, such as streams, will minimize bias in population estimates (Krebs 1989, Thompson 1992).

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