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Water Quality and Macroinvertebrate Community Structure Associated with a Sportfish Hatchery Outfall

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Abstract.—Water quality and benthic macroinvertebrate communities near the A. E. Wood Fish Hatchery effluent outfall in the San Marcos River, Texas, were studied from October 1996 to July 1998. The upper San Marcos River is a hydrologically stable, spring-fed system that provides habitat for several endemic and endangered species. The hatchery effluent generally had little discernable effect on water quality at downstream locations, although total suspended solids and chlorophyll-*a* levels were elevated compared with levels in upstream areas. Benthic macroinvertebrates likewise showed no appreciable differences between upstream and downstream sampling locations, and upstream and downstream communities overlapped considerably in terms of species composition, species richness, and functional feeding group composition. We conclude that hatchery effluent did not substantially affect water quality or benthic macroinvertebrate community structure during the course of this study.

Aquaculture effluents have come under increasing scrutiny in the USA since the passage of the Clean Water Act in 1972. That act led to the creation of the National Pollution Discharge Elimination System (NPDES) program in 1973. Fish hatcheries defined as concentrated animal feeding operations (CAFOs) under the NPDES criteria are required to apply for a wastewater discharge permit from either the U.S. Environmental Protection Agency (USEPA) or a state agency delegated the appropriate regulatory authority. The criteria that define a CAFO differ for warmwater and coldwater hatcheries and are based on production, feeding, and discharge patterns (for details, see Code of Federal Regulations, title 40, section 122.24, appendix C). Under the CAFO criteria, which include warmwater aquatic animal production of 45,400 kg or more, most warmwater sportfish hatcheries were exempt from applying for permits. Prior to October 1999, when the USEPA delegated the NPDES program to the Texas Natural Resource Conservation Commission (TNRCC), most aquaculture operations in Texas, including those operated by the Texas Parks and Wildlife Department (TPWD), were either exempt from the permit requirement or authorized under a general aquaculture permit. In 1994, the TPWD applied for Investigational New Animal Drug exemptions from the U.S. Food and Drug Administration for legal use of unlabeled medications in aquaculture. To satisfy environmental concerns related to the use of therapeutics, the TPWD was required to apply for NPDES permits for all its fish hatcheries.

The TPWD operates the A. E. Wood Fish Hatchery (AEWH) on the San Marcos River in San Marcos, Hays County, Texas. The San Marcos River was designated exceptional for aquatic life and contact recreation by the TNRCC. Historically, San Marcos Springs, which forms the headwaters of the San Marcos River, has exhibited the greatest flow dependability and environmental stability of any spring in the southwestern United States (Brune 1981). This hydrological stability allowed development of a diverse and ecologically unique ecosystem with a high level of endemism (Bowles and Arsuffi 1993). Among the endemic species, five are listed by the U.S. Fish and Wildlife Service (USFWS) as threatened or endangered under the U.S. Endangered Species Act (ESA) of 1973, including the fountain darter *Etheostoma fonticola* and Texas wild-rice *Zizania texana* (USFWS 1995). The hatchery, located approximately 3.7 km downstream from San Marcos Springs, is situated on 47 ha of land with about 970 m of river frontage. Hatchery operations include withdrawal and subsequent discharge of water into the San

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Marcos River in an area that includes critical habitats for the fountain darter and Texas wild-rice.

Because endangered species and their critical habitats could be affected by AEWH operations, the USEPA consulted with the USFWS on the NPDES permit application for AEWH. In July 1994, the USFWS raised concerns that changes in water quality of the San Marcos River associated with AEWH effluent could alter the macroinvertebrate community and, in turn, disrupt the food base of the fountain darter. Subsequently, the USEPA requested that the TPWD complete a biological assessment as part of an ESA Section 7 consultation. One component of the biological assessment was to estimate the effects of AEWH effluent on the benthic macroinvertebrate communities in the San Marcos River near the hatchery outfall.

Typically, a biological assessment characterizes the status of natural resources and monitors biological community changes associated with anthropogenic perturbation (e.g., Rosenberg and Resh 1993). Barbour (1996) reported that biological assessments are particularly useful because they reflect the condition of the resident biota in response to cumulative effects of various impacts. Such assessments generally use species richness, composition, and abundance to detect and assess human impacts. Because many macroinvertebrate species are sessile or have limited migration patterns, macroinvertebrate assemblages are well suited for indicating local stream conditions, such as in site-specific impact evaluations involving upstream–downstream comparisons (Barbour et al. 1999). Biological assessments based on aquatic macroinvertebrates can identify water quality problems related to pollution or other perturbations in a shorter and more cost-effective manner than other quantitative methods. Due to these advantages, evaluation of benthic macroinvertebrate communities is among the most widely used means for assessing anthropogenic effects on the condition of biological communities in streams (Resh et al. 1995; Barbour et al. 1999).

The effects of aquaculture effluents on receiving waters, including the consequences for benthic macroinvertebrate community structure, have recently received greater attention. However, most reports evaluate the effects of salmonid culture effluents (e.g., Kendra 1991; Camargo 1992; Kelly 1993; Loch et al. 1996; Selong and Helfrich 1998), whereas there are no reports on the effects of warmwater aquaculture effluents in spring ecosystems. In this paper, we summarize portions of the biological assessment concerning water quality and benthic macroinvertebrate community structure at the AEWH effluent outfall and at locations upstream and downstream of the outfall.

Study Area

The upper San Marcos River is a spring-fed system that derives its flow from the Edwards Aquifer. San Marcos Springs, which form the river's headwaters, are the second largest spring system in Texas (Brune 1981). The mean annual flow of the spring system from 1957 to 1993 was 4.8 m^3/s (Gandara et al. 1993). The upper portion of the river, which includes the AEWH site, is a rapidly flowing, thermally constant, clearwater stream up to 12 m wide and generally less than 1 m deep.

Original construction of AEWH was completed in 1949, and a major renovation was completed in December 1988. The renovation included a rebuilding of the ponds and water supply system, as well as construction of indoor intensive rearing facilities. Production facilities at AEWH include a variety of plastic-lined ponds totaling 18.8 ha, indoor raceways, troughs, circular tanks, and McDonald jar racks. Although fish production numbers and species vary, a typical production year yields over 4 million largemouth bass *Micropterus salmoides* fingerlings, 1 million channel catfish *Ictalurus punctatus* fingerlings, and 12,000 kg of live forage for captive broodstock. Additionally, the facility holds 67,000 rainbow trout *Oncorhynchus mykiss* during winter months (December–February). Maximum daily water diversion by AEWH is about 16.5 million L, with about 3.8×10^9 L, or approximately 2.6% of mean annual river flow, diverted each year. The hatchery is considered nonconsumptive because an estimated 98.6% of the water diverted to the hatchery is eventually returned to the river.

In the early 1900s, the San Marcos River was diverted to form a millrace. This detour created Thompson's Island, which divides the river into the natural channel and a millrace located from approximately 0.8 km upstream of the AEWH effluent outfall to about 92 m downstream (Figure 1). Effluent from AEWH is discharged at the right bank of the natural river channel. Effluent leaving the hatchery typically continues along the river's right bank, where it mixes with water in the natural channel, and then mixes further downstream with water exiting the millrace.

Methods

We sampled water quality at four sites: approximately 90 m upstream of the effluent outfall, the

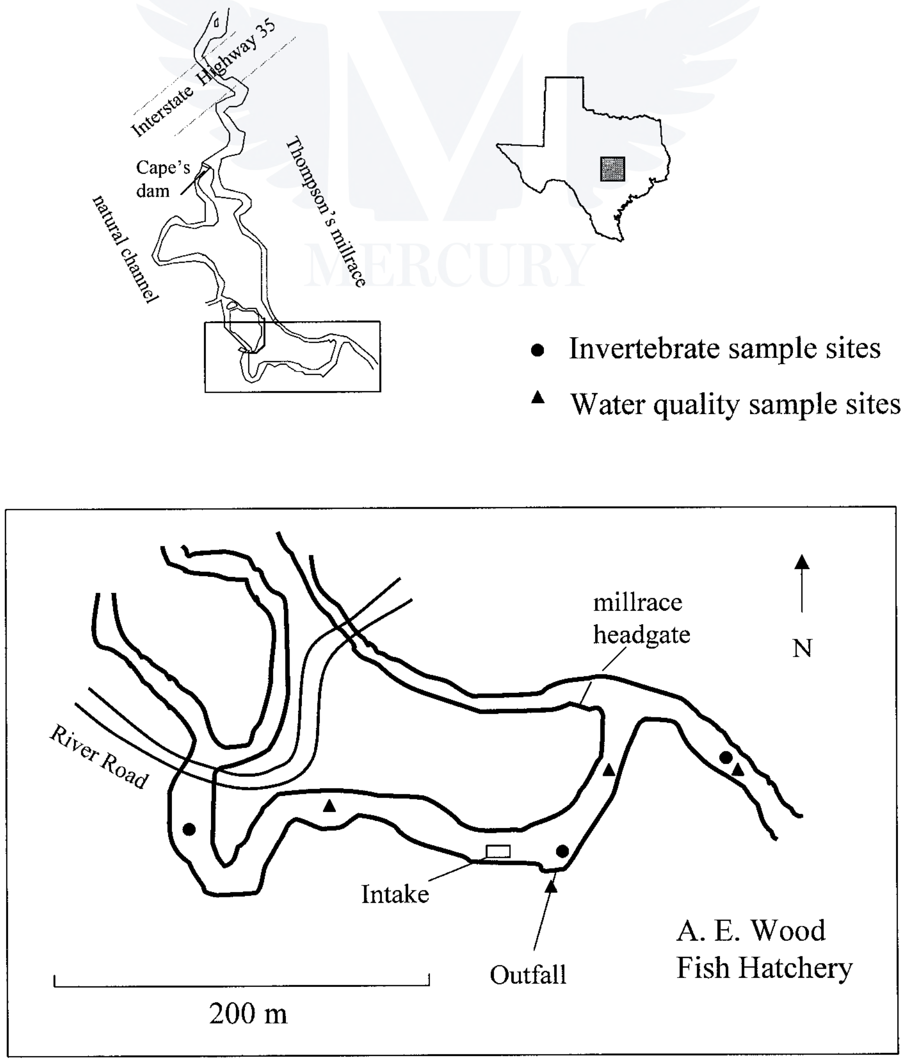

FIGURE 1.—Map of the upper San Marcos River, Texas, near the A. E. Wood Fish Hatchery, showing sites where water quality and benthic macroinvertebrates were sampled from October 1996 to July 1998.

effluent channel itself, the mixing zone 60 m downstream of the outfall, and 175 m downstream of the outfall (Figure 1). Water quality measurements included temperature, pH, dissolved oxygen, total and un-ionized ammonia nitrogen, phosphorus, inorganic suspended solids, total suspended solids (total filterable residue), chlorophyll *a*, and 5-d carbonaceous biological oxygen demand (CBOD). Water was sampled approximately three times per week from October 1996 to July 1998, with the following exceptions: inorganic suspended solids were measured starting in January 1998, and CBOD was measured approximately once per week. All samples were collected in 2-L grabs from the surface to about 30 cm depth. Temperature, pH, and dissolved oxygen data were measured with a calibrated Yellow Springs Instruments/Grant model 3800 water quality data logger. Ammonia concentrations were determined with the USEPA-approved Nessler method, and phosphorus was measured as orthophosphate with the USEPA-approved ascorbic acid method (Hach Co. 1992). Total suspended solids and inorganic suspended solid concentrations were determined with standard methods (APHA et al. 1989). Chlorophyll *a* was determined with the chloroform-methanol extraction method (Wood 1985). Water drained from ponds was untreated during the course of the study.

Water quality data collected during the study period were compared among sampling dates and sites. Because of inherent problems associated with pseudoreplication among the sites and the subsequent lack of independence among samples (Hurlbert 1984), we analyzed water quality data with Friedman's two-way analysis of variance (ANOVA; $\alpha = 0.05$) and used a multiple comparison procedure to determine which pairs of sampling sites differed significantly (Daniel 1978).

Benthic samples were collected from riffle and glide habitats once per month during October 1996, March and October 1997, and January, April, and July 1998. Benthic sampling methodology generally followed that of Barbour et al. (1999). Samples were collected from the same general sites as water samples, with the exception that benthic samples were taken from the effluent outfall zone rather than from the effluent channel (Figure 1). Also, no benthic samples were taken in the mixing zone downstream of the outfall. On each sampling date, we collected six benthic samples across the channel from bank to bank at each location with a D-frame aquatic net (30×25 cm; 150-μm mesh). Samples were collected so as to represent available habitats. Benthic sample collections involved kicking for 20 s to disturb the substrate in a 0.3-m^2 area approximately 1 m in front of the net. Samples were collected sequentially from downstream to upstream to avoid disturbing the benthic communities prior to collection efforts. Individual samples were preserved in the field with 95% isopropyl alcohol. Samples were sorted in the laboratory, and all invertebrates were removed and identified to the lowest practical taxonomic level, then stored permanently in 70% isopropyl alcohol.

Metrics calculated for the macroinvertebrate community at each sample location were species richness, mean benthic density (number/m^2), the percentage Elmidae (riffle beetles; Coleoptera), Jaccard's similarity coefficient, and the Shannon–Weiner index (Barbour et al. 1999). We also calculated the percentage of Ephemeroptera and Trichoptera (percentage E–T). This metric was modified from the percentage Ephemeroptera–Plecoptera–Trichoptera metric (Barbour et al. 1999) because plecopterans do not occur in the upper San Marcos River. We also included the percentage native species as a metric because exotic species are well represented in the upper San Marcos River (Bowles and Bowles 2001). Genera were assigned to functional feeding groups (FFGs) according to Merritt and Cummins (1996) and Pennak (1989). However, due to difficulty in generic identification of chironomids, the family was not included in FFG analysis; however, most FFGs are represented within the Chironomidae. Though many different metrics have been proposed for rapid bioassessment of stream communities (Resh et al. 1995; Barbour et al. 1999), we restricted our analysis to the set of metrics we believed were sufficient to characterize the benthic communities at our sample sites. Further, there is no consistent use of metrics in similar studies of aquaculture effluents and receiving streams (e.g., Camargo 1992; Loch et al. 1996; Selong and Helfrich 1998).

The sampling site upstream of the hatchery effluent outfall served as the reference site and the basis of comparison for macroinvertebrate community metrics. We made no attempt to compare our calculated metrics with those of other stream benthic studies. Rather, our intent was to demonstrate the relative magnitude of differences in metrics between the collection sites and the hatchery outfall site. We subjected benthic invertebrate data to chi-square goodness-of-fit tests ($\alpha = 0.05$) when appropriate (Conover 1980). Representative specimens of all taxa collected were deposited in a reference collection maintained by the TPWD River Studies Program, San Marcos, Texas.

Results and Discussion

Water Quality

Our water quality data (Table 1) were similar to those reported by Groeger et al. (1997) and Poole and Bowles (1999). Hatchery effluent differed from water in the San Marcos River in several ways. Hatchery effluent generally had higher levels of water quality measurements (Friedman's ANOVA, $P < 0.01$), except for dissolved oxygen and temperature, which did not differ among sampling sites. However, the range of water temperature in the effluent was greater than that of the river, with warmer hatchery water releases in the summer and cooler releases in the winter. Although taxa-specific tolerances for invertebrates collected in this study are unknown, freshwater benthic invertebrates may have at least moderate tolerance to temperatures up to 30°C (Bush and Mar 1974). Phosphorus was greatest at the upstream site, which agrees with a previous study (Groeger et al. 1997) indicating a higher phosphorus concentration at the headwaters of the San Marcos River than at the sampling site near the hatchery. In general, our data indicate that once the two sections of river merge below Thompson's Island,

TABLE 1.—Summary analyses of water quality data collected in the San Marcos River, Texas, near the A. E. Wood Fish Hatchery outfall, October 1996–July 1998. Values in the first row for each variable are means (SD); those in the second row are ranges. Within rows, means followed by the same letter are not significantly different (multiple comparison procedure; Conover 1980). Data are expressed in mg/L unless specified otherwise.

Variable	*N*	Upstream	Effluent	Mixing zone	Downstream	Friedman's two-way ANOVA[a]
pH (standard units)	215	7.9 (0.3) z 7.1–8.8	8.4 (0.4) y 7.1–10.3	8.0 (0.3) z 7.1–9.1	8.0 (0.3) z 7.1–8.9	375.0 (<0.01)*
Dissolved oxygen	215	8.9 (0.8) 6.5–11.8	8.8 (1.1) 6.0–11.8	8.9 (0.8) 5.4–12.0	8.9 (0.7) 5.1–11.5	3.34 (0.34)
Temperature (°C)	215	21.4 (1.6) 16.8–24.5	21.4 (5.8) 5.8–31.0	21.3 (1.9) 15.4–24.8	21.3 (1.6) 16.2–24.1	2.19 (0.53)
Total ammonia nitrogen	210	0.16 (0.18) z 0.001–1.61	0.29 (0.15) y 0.020–0.92	0.17 (0.13) z 0.010–1.11	0.15 (0.12) z 0.001–1.09	252.32 (<0.01)*
Un-ionized ammonia	210	0.01 (0.01) z <0.01–0.06	0.06 (0.08) y <0.01–0.57	0.01 (0.02) z <0.01–0.25	0.01 (0.01) z <0.01–0.06	327.93 (<0.01)*
Phosphorus	207	0.012 (0.021) z <0.01–0.177	0.011 (0.015) z <0.01–0.120	0.009 (0.014) y <0.01–0.093	0.009 (0.015) y <0.01–0.136	32.62 (<0.01)*
Total suspended solids	213	5.5 (6.3) z 0.4–64.8	16.9 (14.5) y 2.0–97.2	7.4 (12.1) z 0.0–151.7	6.1 (6.6) z 0.4–62.4	265.45 (<0.01)*
Inorganic suspended solids	76	4.2 (3.5) z 0.8–27.6	7.3 (7.4) y <0.01–46.5	5.1 (4.4) z <0.01-27.6	4.8 (5.0) z <0.01–36.8	14.41 (<0.01)*
Chlorophyll *a* (μg/L)	210	2.2 (2.4) z <0.1–23.8	33.8 (35.6) y <0.1–304.6	4.0 (5.1) z <0.1–53.3	2.9 (2.9) z <0.1–22.0	352.85 (<0.01)*
Carbonaceous biochemical oxygen demand	73	2.1 (0.2) z 2.0–3.6	5.3 (2.1) y 2.1–9.9	2.1 (0.4) z 2.0–4.8	2.1 (0.6) z 2.0–5.4	129.16 (<0.01)*

[a] Asterisks denote significant differences among sampling sites (α = 0.05). Friedman's statistics are chi-square values, with probabilities in parentheses.

hatchery effluent is diluted such that changes in water quality are minimal. In addition, we contend that some of the observed differences at the downstream-most location may not be biologically significant (e.g., pH, total suspended solids), and others (e.g., chlorophyll *a*) were within the ranges of error for the instrumentation used to conduct the tests. We also suspect that some water quality parameters, particularly total suspended solids downstream of the outfall, were exacerbated by activity at a popular swimming site located immediately below the millrace dam.

Although most water quality variables measured downstream from the hatchery effluent were similar to those recorded upstream of the hatchery, some components of the effluent are of potential concern to aquatic life in the mixing zone. For instance, the hatchery effluent usually had higher levels of ammonia nitrogen than the river, although the highest values were recorded at the upstream reference site. Un-ionized ammonia nitrogen, generally considered the most toxic nitrogen metabolite produced in fish culture systems, was found at levels lower than those known to produce acute toxicity in fish (Ruffier et al. 1981). Indeed, median un-ionized ammonia concentrations at the two downstream sites below the effluent were 0.006 and 0.005 mg/L, which are considerably lower than the established maximum water quality standard for fish culture of 0.02 mg/L (USEPA 1979–1980). By comparison, un-ionized ammonia toxicity to aquatic invertebrates has received much less attention than toxicity to fish. Williams et al. (1986) found that the safe level of exposure to un-ionized ammonia was 0.03 mg/L for 11 species of aquatic invertebrates. However, Colt and Armstrong (1981) reported that un-ionized ammonia nitrogen levels of 0.05–0.20 mg/L could lead to significant growth reduction in most aquatic animals. Although some of our data fell within the ranges

TABLE 2.—Summary analyses of macroinvertebrate samples taken from the San Marcos River, Texas, near the A. E. Wood Fish Hatchery outfall, October 1996–July 1998. A list of the macroinvertebrates collected during the study is available from D. E. Bowles.

Date	Site	Species richness	Mean density (number/m^2)	Native species (%)	E–T species (%)[a]	Elmidae (%)	Dominant taxa (%)[b]	Chironomidae (%)	Shannon-Weiner index
Oct 1996	Upstream	30	380	12.4	7.0	0.8	73.6	1.4	1.28
	Outfall	32	152	38.8	12.7	3.7	58.7	0.3	1.93
	Downstream	34	256	47.2	13.7	7.4	41.0	12.7	2.29
Mar 1997	Upstream	32	115	31.4	12.5	3.4	62.3	2.6	1.57
	Outfall	29	121	76.9	23.9	10.3	21.5	15.7	2.61
	Downstream	22	115	70.4	23.5	17.8	26.9	10.4	2.41
Oct 1997	Upstream	32	176	57.8	30.3	3.4	36.3	5.7	2.45
	Outfall	31	109	83.5	36.4	9.7	15.2	5.1	2.79
	Downstream	26	87	67.4	43.0	2.9	27.9	5.2	2.17
Jan 1998	Upstream	22	140	31.8	12.4	1.8	50.7	6.4	1.79
	Outfall	24	77	66.2	25.5	11.1	23.5	15.0	2.46
	Downstream	22	77	34.0	17.0	5.9	37.9	3.9	2.03
Apr 1998	Upstream	23	68	71.0	20.4	4.2	16.4	18.5	2.49
	Outfall	26	98	79.5	28.2	6.2	11.8	8.7	2.88
	Downstream	25	162	68.4	31.0	4.3	22.9	3.4	2.41
Jul 1998	Upstream	29	167	38.6	14.4	0.6	53.6	4.2	1.95
	Outfall	25	25	76.3	24.1	3.2	18.9	2.8	2.56
	Downstream	33	121	70.1	35.2	4.6	24.9	4.2	2.67

[a] E–T species = ephemeropterans (mayflies) and trichopterans (caddisflies).
[b] Quilted melania *Tarebia granifera* dominated, except in the April 1998 (Oligochaeta) and July 1998 (Hirudinea) samples.

reported by Colt and Armstrong (1981) and Williams et al. (1986), we collected discrete grab samples rather than collecting composite samples or conducting continuous monitoring, and therefore it was not possible to establish continuous exposure levels. Although specific toxicity studies have not been conducted with hatchery effluent, we conclude that the ammonia concentrations in the effluent were sublethal at worst and generally safe at best. Furthermore, the ammonia concentration of the downstream-most sampling site did not differ significantly from that of the upstream reference site (Friedman's ANOVA, $P > 0.01$).

At the time of this study, AEWH employed a single-pass, flow-through water system, and water residence time varied from less than 1 h to several months. Ponds were typically drained about 141 d/year, with a mean water flow from the ponds of about 5.7×10^6 L/d. Ponds generally were not drained when water quality variables were measured at extreme values of their ranges, with the exception of suspended solids. Solids likely were highest at the onset of pond draining, when kettles were cleaned, and near the end of draining, when fish were crowded into smaller volumes of water and bottom sediments were disturbed (Schwartz and Boyd 1994).

Benthic Community Structure

Species diversity was relatively consistent among collection sites, although some appreciable variation was observed among sampling dates (Table 2). However, none of the differences were significant (chi-square goodness-of-fit tests, $T \leq 2.28$). In general, species diversity in benthic samples is positively correlated with ecosystem health (Barbour et al. 1999). Shannon–Weiner diversity indices ranged from 1.28 (October 1996; upstream) to 2.88 (April 1998; outfall), and the median for all sampling dates was 2.41. Shannon–Weiner values at the upstream location generally were lower than for the outfall and downstream sites for most sampling dates, but these differences are not significant (chi-square goodness-of-fit tests, $T \leq 0.28$). The similarity of Shannon–Weiner values reported here indicates that both the number of species and the distribution of individuals among species (i.e., evenness) did not substantially differ among the three sampling sites. Jaccard's similarity coefficients also were largely consistent among both sampling dates and locations (Table 3). Jaccard's similarity coefficients typically were around 0.4, indicating moderate similarity among the benthic communities from upstream to downstream and from outfall to downstream sites. Most of the observed variation among collection sites in the benthic communities was due to the occurrence of rare species. The importance of rare species in biological assessments, including this one, is further emphasized by the consideration that most macroinvertebrate species

TABLE 3.—Jaccard's similarity coefficients for benthic macroinvertebrate communities collected from the San Marcos River, Texas, near the A. E. Wood Fish Hatchery outfall, October 1996–July 1998.

Sampling date	Upstream–outfall	Upstream–downstream	Outfall–downstream
Oct 1996	0.45	0.53	0.45
Mar 1997	0.44	0.71	0.49
Oct 1997	0.48	0.35	0.36
Jan 1998	0.44	0.46	0.39
Apr 1998	0.76	0.50	0.55
Jul 1998	0.46	0.47	0.46

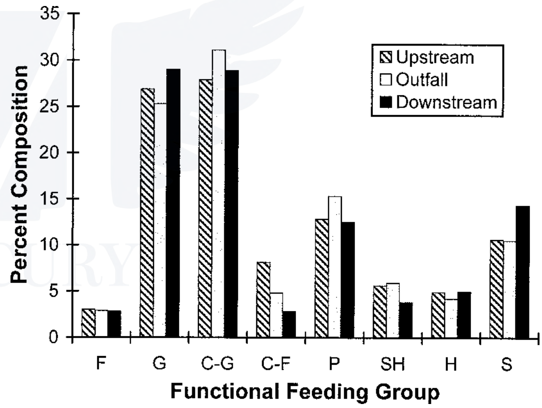

FIGURE 2.—Mean percentage composition of benthic macroinvertebrate functional feeding groups collected at three sites near the A. E. Wood Fish Hatchery effluent outfall (upstream, at the outfall, and downstream) in the San Marcos River from October 1996 to July 1998. Functional feeding group codes are as follows: F = filterers, G = grazers, C–G = collector–gatherers, C–F = collector–filterers, P = predators, SH = shredders, H = herbivores, and S = scrapers. Functional feeding group composition for Chironomidae is not included.

in natural biological communities are rare (Cao et al. 2001). The majority of common species were collected at all sampling sites and dates.

Mean benthic macroinvertebrate densities varied considerably among both sampling dates and locations (Table 2). Variation among stream benthic communities is not unusual because benthic macroinvertebrate species normally have patchy or aggregated distributions that are further confounded by temporal variability (Canton and Chadwick 1988; Resh and Rosenberg 1989; Merritt et al. 1996; Linke et al. 1999). Such variation can be attributed to a wide variety of factors (Cummins and Merritt 1996). Lack of control over natural variability in stream biological data hinders accurate assessments, as does the inability to objectively verify the accuracy of macroinvertebrate assessments in the field (Resh and Jackson 1993). Due to these constraints, we did not attempt statistical analysis of benthic macroinvertebrate densities.

Other metrics estimated in our study generally did not show consistent patterns with respect to sampling location. The lack of consistency probably arises from the highly variable benthic densities recorded among collection dates and locations. The percentage of native species was higher at the outfall and downstream sampling locations in all months except April 1998. The numerically dominant taxon in most instances was an exotic snail, the quilted melania *Tarebia granifera* (Lamarck), except at the outfall sampling location in April and July 1998, when aquatic worms (Oligochaeta) and leeches (Hirudinea), respectively, were the dominant taxa collected. In contrast, for all sampling dates, riffle beetles (Elmidae) were more abundant at the outfall and downstream locations than at the upstream sampling site. A similar pattern was observed for the percentage E–T. On three sampling dates, the percentage Chironomidae was greater at the upstream sampling location than at the outfall and downstream sites. However, the dominant chironomids in all samples belonged to the relatively pollution-intolerant subfamily Orthocladiinae.

Functional feeding group composition of the benthos ultimately reflects the energy pathways in an ecosystem, and stresses to the stream will be expressed in the functional roles of the resident benthos. To evaluate potential impacts to the benthic community, we combined FFG data across all sampling dates (Figure 2) because observed variation was relatively small among both sampling dates and locations and did not appear to follow any particular pattern. When viewed across all sampling dates, collector–gatherer and grazer feeding groups predominated in all instances, with each contributing over 25% of the FFG composition at each sampling site. Predator (12.5%) and scraper (10.5%) FFGs were the next most commonly represented groups in benthic samples. Other FFGs (filterers, collector–filterers, shredders, and herbivores) each contributed 8% or less to the benthic faunal composition. Collector–gatherers were represented more at the outfall and downstream sampling locations than at the upstream site, although the difference was not significant (chi-square goodness-of-fit test, T = 0.18). Poff and Matthews (1986) studied benthic macroinvertebrate community structure and FFG responses to thermal enhancement and found that moderate thermal increases in their study stream resulted in increased percentages of collector–gatherer, collector–filterer, and scraper FFGs as compared with an unaffected site. However, the thermal increase was relatively more constant than the intermittent thermal alterations associated with the hatchery outfall in our study. By comparison, Barbour et al. (1999) suggested that the percentages of grazers and scrapers in the benthic community should generally decrease in response to increasing perturbation. However, our data showed that the percentage compositions of grazers and scrapers at the outfall and at downstream sites were equal to or greater than that of the upstream location, but the differences were not significant (chi-square goodness-of-fit test, $T = 1.09$). Interestingly, the collector–filterers were best represented at the upstream sampling location, contrary to expectations that a higher percentage would be found at the outfall and downstream locations due to the fine particulate material in the hatchery effluent (Table 1).

The minor differences that we observed in benthic macroinvertebrate community structure and corresponding metrics likely are a function of the random, clumped distribution of the macroinvertebrates. Moreover, spring-fed ecosystems like the upper San Marcos River tend toward uniformity in physical and hydrological functions. Correspondingly, this stability has a pronounced influence on macroinvertebrate community structure. In our study, the benthic community structure was highly consistent among all sampling locations. Indeed, the species collected and their respective life history stages (e.g., larval instars) were found at all sampling sites and in all seasons. Linke et al. (1999) noted that, in examinations of benthic communities for bioassessment purposes, temporal variation can influence judgment as to whether or not a site is degraded. However, in large, spring-fed ecosystems, phenological and temporal variations are largely masked.

Aquatic insect distribution and community structure are ultimately determined by the physical and chemical tolerances of individuals in the population (Cummins and Merritt 1996). Based on the present data, we conclude that the hatchery effluent did not substantially affect downstream water quality and benthic communities, despite the relatively high total suspended solids and chlorophyll-*a* levels in the effluent.

Our data show that sportfish hatchery operations can have negligible effects on receiving waters, even in environmentally sensitive systems. This finding is significant in light of the growing scrutiny of the aquaculture industry from regulatory authorities responsible for implementing the Clean Water Act and the NPDES permitting program. Under federal statutes associated with these programs, the majority of warmwater sportfish hatcheries have been exempted from having to obtain discharge permits unless annual production exceeded 45,454 kg (e.g., Code of Federal Regulations, title 40, section 122.24, appendix C). However, regulatory authorities have considerable discretion and may require NPDES permits on a case-by-case basis following an on-site inspection. Notably, the concentrations of measured effluent characteristics from AEWH were generally lower than those reported from commercial catfish pond effluents (e.g., Schwartz and Boyd 1994; Seok et al. 1995; Tucker et al. 1996). Our data suggest that practices recommended for catfish pond effluent management, such as removal of fish by seining and use of a settling period (Schwartz and Boyd 1994; Seok et al. 1995) or rainwater storage (Cathcart et al. 1999) prior to final draining, may be unnecessary for sportfish fingerling production. Shireman and Cichra (1994) also noted considerable differences between sportfish hatcheries and those producing food fish. Water quality in sportfish ponds generally was better than that of food fish ponds due to lower densities, shorter growout periods, and lower feeding rates.

In 2000, the USEPA announced a study of the aquaculture industry (Federal Register: September 14, 2000, Volume 65, Number 179). One outcome of the study may be increased regulation of the aquaculture industry. Aquaculturists may therefore need to increase their understanding of biological monitoring of receiving waters as well as changing some routine practices. A recent draft NPDES permit issued for net-pen culture of Atlantic salmon *Salmo salar* required routine monitoring of the benthic fauna immediately downstream of the enclosures and in the mixing zone in addition to the traditional monitoring of certain physicochemical characteristics. If a biological monitoring requirement becomes commonplace in NPDES permits, aquaculture facility operators likely will have to acquire additional expertise.

It has been suggested that the aquaculture community demonstrate a proactive approach to managing effluent discharge (Cathcart et al. 1999). Effluents from commercial channel catfish facilities are fairly well described, and a variety of water management practices have been developed (e.g., Schwartz and Boyd 1994; Seok et al. 1995; Tucker et al. 1996; Cathcart et al. 1999), but data on sportfish fingerling production are limited. Management practices employed at AEWH include manual removal of pond and kettle sludges, and the minimization of runoff by collection of rainwater in ponds. Although our data showed no significant environmental perturbations to the San Marcos River below the hatchery outfall, the TPWD opted for a proactive approach and completed construction of a wastewater treatment plant in late 2000. The treatment plant consists of three rotating drum filters, two sand filters, and two evaporation ponds. The treatment plant also increases water reuse capabilities. Installation of the treatment plant demonstrates the TPWD's commitment to promoting resource stewardship while operating a highly productive fish hatchery in an environmentally sensitive area. Such an investment may not be warranted for all sportfish hatcheries, but proactive and innovative pond management practices should nevertheless be adopted.

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