



Fish assemblage structure of Koycegiz Lagoon–Estuary, Turkey: Spatial and temporal distribution patterns in relation to environmental variation

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Received 20 September 2004; accepted 18 March 2005

Available online 21 June 2005

Abstract

Spatial and temporal variation in fish assemblage structure of Koycegiz Lagoon–Estuarine System (KLES), located on the northwestern Turkish coast of Mediterranean, was investigated along an estuarine gradient where salinity ranged from 5 in upper reaches to 40 in lower reaches during October 1993–September 1994. Throughout the study, 42 species, consisting of marine (25), marine–estuarine-dependent (12), freshwater (3), catadromous (1), and estuarine resident (1) forms, were collected in trammel nets. Although species richness of marine species was greater than that of other groups, numerical contribution by marine species to the total catch was only 16%. *Tilapia* spp., the most abundant species mostly during summer and early spring at upper reaches, contributed 17% of the total samples. Among the seven species of Mugilidae, which contributed 42% of the total catch, *Mugil cephalus*, *Liza aurata*, and *Liza salines* contributed 10, 13, and 10% of the total catch, respectively. Consistent with findings from other studies, species richness and abundance were highest during late spring and summer and the lowest during winter and early spring. Samples from sites at or near the sea had more marine species. Samples from upper reaches had more freshwater and marine–estuarine-dependent species. Canonical correspondence analysis (CCA) indicated that salinity and turbidity were the most important environmental parameters affecting fishes. Sites near the sea were associated with high salinity and low turbidity, and sites in upper reaches had low salinity and high turbidity. Thus, the pattern observed in fish assemblage structure appears to be strongly influenced by species' responses to dominant salinity and turbidity gradients.

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Keywords: canonical correspondence analysis; turbidity; salinity; fish assemblage; lagoon–estuarine system; Mediterranean; Turkey

1. Introduction

Estuaries are transition zones between seas and freshwater that are occupied by a combination of freshwater and marine species including many juveniles (Claridge et al., 1986). Fish assemblage structure of estuaries is characterized by low diversity but high

abundance, especially for juveniles (Whitfield, 1999). Examination of the ecological factors important in defining habitats for fishes has been the main focus of many previous studies (Able, 1999; Martino and Able, 2003). Most estuaries are characterized by high biological productivity associated with relatively extreme and varying environmental conditions (Day et al., 1989; Kennish, 1990; Whitfield, 1999). The fact that estuaries serve as nurseries for many fishes and macrocrustaceans, including many important fishery species (Shenker and

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Dean, 1979; Weinstein, 1979; Rakocinski et al., 1996; Blaber, 2000; Elliott and Hemingway, 2002; Akin et al., 2003) is another important reason for examining factors that shape fish assemblage structure.

Fish distributions within biologically and physically complex estuarine systems may be influenced by many mechanisms. Several estuarine ecologists have pointed out that biotic processes, such as competition and predation, may be important in driving the occurrence of spatial and temporal patterns of fishes in estuaries (Holbrook and Schmitt, 1989; Ogburn-Matthews and Allen, 1993; Lankford and Targett, 1994; Barry et al., 1996). The consistency of temporal occurrence of fishes within the estuaries implies the importance of species-specific reproductive biology (Potter et al., 1986, 2001; Drake and Arias, 1991; Thiel and Potter, 2001; Hagan and Able, 2003). In addition to biological factors, abiotic factors affect occurrences of fishes within estuaries. These factors include salinity (Gunter, 1961; Weinstein et al., 1980; Peterson and Ross, 1991; Rakocinski et al., 1992; Szedlmayer and Able, 1996; Wagner and Austin, 1999; Hagan and Able, 2003; Jaureguizar et al., 2003; Martino and Able, 2003), temperature (Peterson and Ross, 1991; Rakocinski et al., 1992; Szedlmayer and Able, 1996; Marshall and Elliott, 1998; Araújo et al., 1999), turbidity (Peterson and Ross, 1991; Cyrus and Blaber, 1992; Hagan and Able, 2003), dissolved oxygen (DO) (Blaber and Blaber, 1980; Rakocinski et al., 1992; Fraser, 1997; Maes et al., 1998; Whitfield, 1999), freshwater inflow (Rogers et al., 1984; Fraser, 1997; Whitfield, 1999), structural attributes of habitat (Weinstein et al., 1980; Thorman, 1986; Sogard and Able, 1991; Everett and Ruiz, 1993; Szedlmayer and Able, 1996; Wagner and Austin, 1999), depth (Zimmerman and Minello, 1984; Rakocinski et al., 1992), geographic distance from the estuary mouth (Martino and Able, 2003), and hydrography (Cowen et al., 1993).

Remmert (1983) proposed that large-scale (kilometers) patterns of fish distribution are the results of species response to their physical environment. Abiotic factors (salinity, temperature, turbidity, DO, etc.) operating over large spatial scale are believed to determine coarse community structure, whereas biotic interactions refine species abundance and distribution patterns within that structure (Sanders, 1968; Menge and Olson, 1990). Here we report findings from an investigation of associations between environmental factors and fish distribution patterns in a relatively deep estuary on the southern coast of the Mediterranean Sea, the Koycegiz Lagoon—Estuarine System (KLES).

Fish assemblage structure in European estuaries has been well studied (Wheeler, 1969; Drake and Arias, 1991; Elliott and Dewailly, 1995; Marshall and Elliott, 1998; Araújo et al., 1999; Thiel and Potter, 2001). In spite of this large number of studies in western European estuaries, we are not aware of any published accounts of assemblage

structure in estuaries of Eastern Europe. Although there are quite large numbers of studies investigating different aspects of the KLES (Geldiay, 1977; Ozhan, 1988; Yerli, 1991; Kazanci et al., 1992; Buhan, pers. comm.), none of them examined relationships between fish assemblage structure and environmental variables. The current study was designed to fill this gap. The purpose of this study was to investigate the effects of physical water quality parameters on spatial and seasonal variation in fish assemblage of KLES in Turkey.

2. Material and methods

2.1. Study area

KLES, located on the northwestern coasts of the Mediterranean Sea in Turkey (Fig. 1), can be divided into two main basins; Lake Koycegiz (the largest basin) in the north and Lake Sultaniye (the smallest basin) in the south. Freshwater inflow to the KLES is supplied by series of streams and both sulfuric and freshwater springs on its shore and the bottom. The Dalyan River, outflow channel of the estuary to the sea, follows a meandering bed that widens into a labyrinth-like channel system discharging into the Mediterranean Sea at Dalyanagzi (Kazanci et al., 1992). Lake Alagol, located at the mouth of the channel near Lake Sultaniye, and Lakes Sulungur and Suluklu, located near Lake Iztuzu, are three major mesohaline lakes. Lake Iztuzu is a small lagoon with varying salinity close to the Mediterranean Sea shoreline (Kazanci et al., 1992).

2.2. Sampling protocol

Samples were collected every month between October 1993 and September 1994 at five sites located along the longitudinal salinity gradient. The first three sites (1, 2, and 3) were located in the two main lake basins (Koycegiz and Sultaniye). Site 3 was located at the mouth of the channel opening to Lake Sultaniye. Site 4 was within the mesohaline Lake Sulungur, and site 5 was located in the sea near to the mouth of the Dalyan River (Fig. 1).

Fishes were collected using two trammel nets with inner nets consisting of five 100 m long panels of 17, 20, 25, 28, and 32-mm mesh. The nets of inner panel were sandwiched by two 500-m nets having 110-mm mesh. Trammel nets are selective for certain fish and are particularly effective in capturing relatively large, mobile species (Bronte and Johnson, 1984) but are not effective in capturing small fishes, which are sampled effectively by seines and throw traps in estuaries and lagoons (Rozas and Minello, 1997). Thus, patterns obtained in this study reflect the distributions and abundances of relatively larger species, and not the entire fish

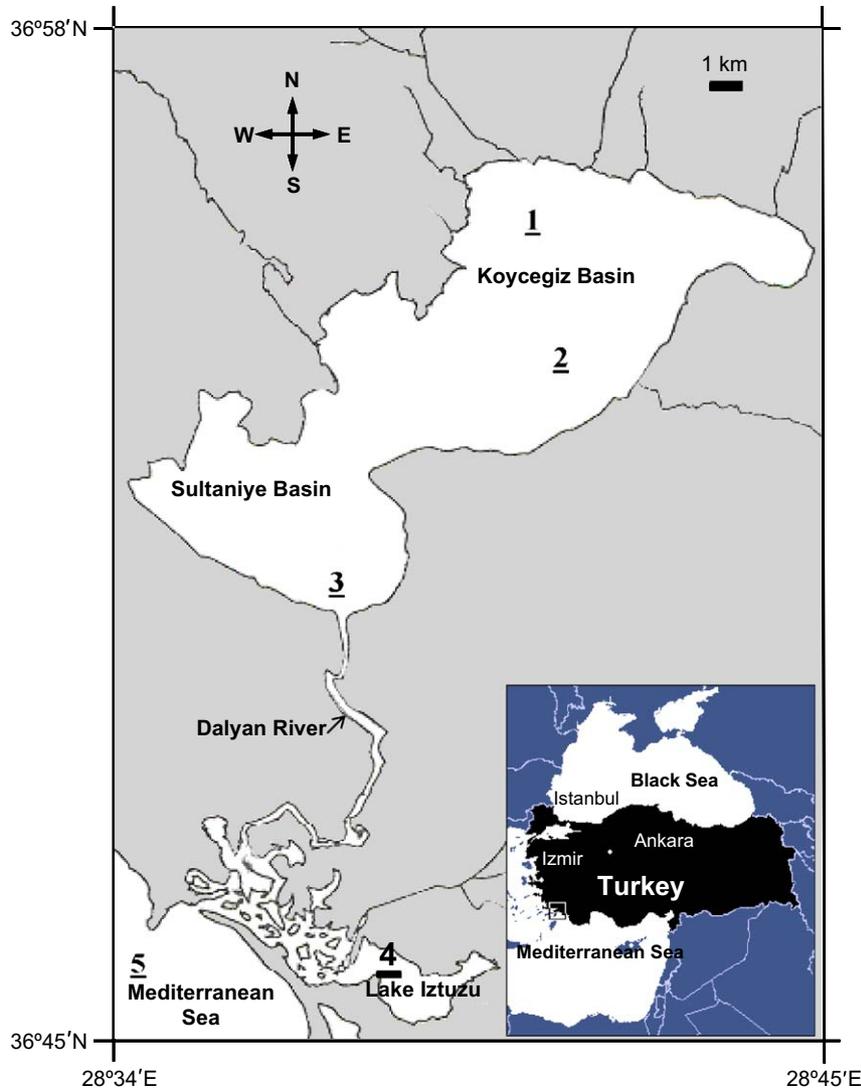


Fig. 1. Map of the Koycegiz Lagoon–Estuarine System with sampling sites.

assemblage. Two trammel nets were deployed at sites over three consecutive days, and remained in place at a given site from 20:00 to 08:00 hour. The duration of each trammel net set was recorded, and catch data were recorded as number of individuals captured per hour. Captured fishes were anesthetized in MS-222 then fixed in 4% formalin in the field. In the laboratory, samples were sorted, identified to species, and counted. *Atherina* and *Tilapia* were reported at genus level due to difficulties in identifying individuals to species level since both *Atherina* and *Tilapia* had three different species in KLES (*Atherina boyeri*, *Atherina heptesus*, *Atherina lacunosus*, *Tilapia zilli*, *Oreochromis aurea*, *Oreochromis nilotica*). Among these species, *A. boyeri* and *T. zilli*, however, are the most abundant species in KLES (E. Buhan, Personal Observation). Reporting of *Atherina* and *Tilapia* at genus level probably did not affect the interpretation of results, because these congeneric species at KLES have been shown to have

similar ecological response to environmental variables (E. Buhan, Personal Observation).

Prior to trammel netting, water quality parameters were measured at each survey site. Temperature ($^{\circ}\text{C}$), salinity (practical salinity scale) and conductivity were measured with a YSI-33 SCT meter. When the YSI-33 was not available, a mercury thermometer and reflectometer were used to measure temperature and salinity, respectively. A Schott Gerate CG 817 model pH meter was used to measure total alkalinity of the water (pH) and oxygen concentration was determined by using either a YSI 5514 oxygen meter or Winkler Method in the laboratory.

2.3. Data analysis

Taxon numerical abundance for trammel net data was standardized to CPUE (Catch Per Unit Effort) as

abundance per trammel net hour. Species richness (S) was recorded as the total number of species occurring at a given site. A two-way ANOVA was used to test for significant differences in environmental variables, species richness, and abundance among sites and months. Prior to analysis of variance, all variables were tested for normality (Kolmogorov–Smirnov test) and homogeneity of variances (Cochran tests). Temperature, salinity, and Secchi depth data did not meet the assumptions of normality and homogeneity of variances even though performing diverse data transformations. These variables, thus, were tested with Kruskal–Wallis test. Since the data were not replicated at a given site, in addition to checking the data for normality and homogeneity of variances, possible interactions between site and month were tested using Tukey's 1 degree-of-freedom test (Sokal and Rolf, 1998). Significant interaction between month and site was detected for abundance data only. When a significant ($P < 0.05$) difference for main effects was detected, Student Newman–Keul (SNK) multiple comparison test was used to test for significant mean differences. Tukey multiple comparison tests were performed to test for significant mean differences of non-normal distributed variables (i.e., temperature, salinity, and Secchi depth) using WINKS statistical software. Spearman's rank correlation (partial correlation) was used to examine simple relationship between environmental variables and CPUE.

Fishes inhabiting in the KLES were categorized as marine, marine–estuarine-dependent, freshwater, estuarine resident, and catadromous, based on their life histories. According to Whitfield's (1999) life cycle terminology, we defined: (a) marine–estuarine-dependent (Potter et al., 1990), also called marine migrants (Whitfield, 1999), as those species extensively use estuaries during juvenile and/or adult life stages; (b) marine species, which are also named as occasional marine visitors (Day et al., 1989) or marine stragglers (Potter et al., 1990; Whitfield, 1999), as those only a small proportion of the overall population use estuaries (Whitfield, 1999); (c) freshwater species (Day et al., 1989), as those restricted to rivers but sometimes enter the estuaries when conditions are favorable; (d) estuarine resident (Whitfield, 1999), as species of marine origin that reside in estuaries and can complete their life cycle within these systems; and (e) catadromous (Whitfield, 1999), as those spawn at the sea but use freshwater catchment areas during the juvenile and sub-adult life stages.

Associations between species CPUE ($\log(\text{CPUE} + 1)$), and log-transformed environmental variables were examined with the canonical correspondence analysis (CCA) using CANOCO. CCA is a weighted averaging method that directly relates community data to environmental variables by constraining species ordination to a pattern that correlates maximally with environmental variables. To reduce the effects of rare species, only species

having CPUE = 1% of the total based on all species and samples were included in CCA. Inter-set correlations between environmental variables (salinity, temperature, DO, pH, and Secchi depth) and CCA axes were used to assess each variable's contribution. Monte Carlo permutation analysis simulation and the forward selection option within the CANOCO package were used to test the significance ($P = 0.05$) of each variable's contribution to each CCA axis.

3. Results

3.1. Environmental variation

Water temperature, ranged from 8.8 °C (January '94 at site 3) to 30.3 °C (July '94 at site 4) (mean = 19.7 °C; S.D. = 6.82 °C), was significantly higher during summer months (June, July, and August) than those of winter months (December, January, and February) ($H = 55.75$; $P < 0.0001$) (Fig. 2). Although mean water temperature tended to increase from site 1 (19.85 °C) to site 5 (20.15 °C), this increase was not statistically significant ($H = 0.28$; $P = 0.991$) (Fig. 2).

Salinity values ranged from 3.9 (April '94 at site 1) to 40 (all months at site 5) (mean = 14.97; S.D. = 14.07) (Fig. 2). The highest and lowest mean salinity values were measured for summer and winter months, respectively; however, this pattern of variation was not statistically significant ($H = 5.40$; $P = 0.910$). Salinity, on the other hand, showed a spatial gradient along the length of KLES. Mean salinity values of sites 4 (18.54) and 5 (40) were significantly higher than those of upper most reaches of the estuary (i.e., 1, 2, 3) ($H = 46.55$; $P < 0.0001$) (Fig. 2).

Oxygen concentration (mean = 9.3 mg l⁻¹; S.D. = 1.25) attained maxima in January (12.8 mg l⁻¹ at site 1) and minima in August (6.4 mg l⁻¹ at site 4) with significant differences in mean monthly mean values ($F_{11, 43} = 3.01$; $P = 0.0045$). February and May had significantly higher mean values than September. On the contrary, in spite of relatively higher values at upper most sites than those at the lower, mean oxygen concentration did not indicate significant differences among sites ($F_{4, 43} = 2.08$; $P = 0.1001$) (Fig. 2).

The pattern of variation in Secchi depth was approximately similar to that obtained in salinity (Fig. 2). Secchi depth varied from 17.2 m (July '94 at site 5) to 1.2 m (May '94 at site 4) (mean = 4.71; S.D. = 3.74). In general, summer and winter months had highest and lowest levels of Secchi depth, respectively, however, the differences in monthly mean values were not significant ($H = 15.82$; $P = 0.152$). Secchi depth, on the other hand, exhibited a strong spatial gradient. Mean value at site 5 (10.12 m) was significantly higher than that at site 4 (2.21 m) ($H = 26.1$; $P < 0.0001$) (Fig. 2).

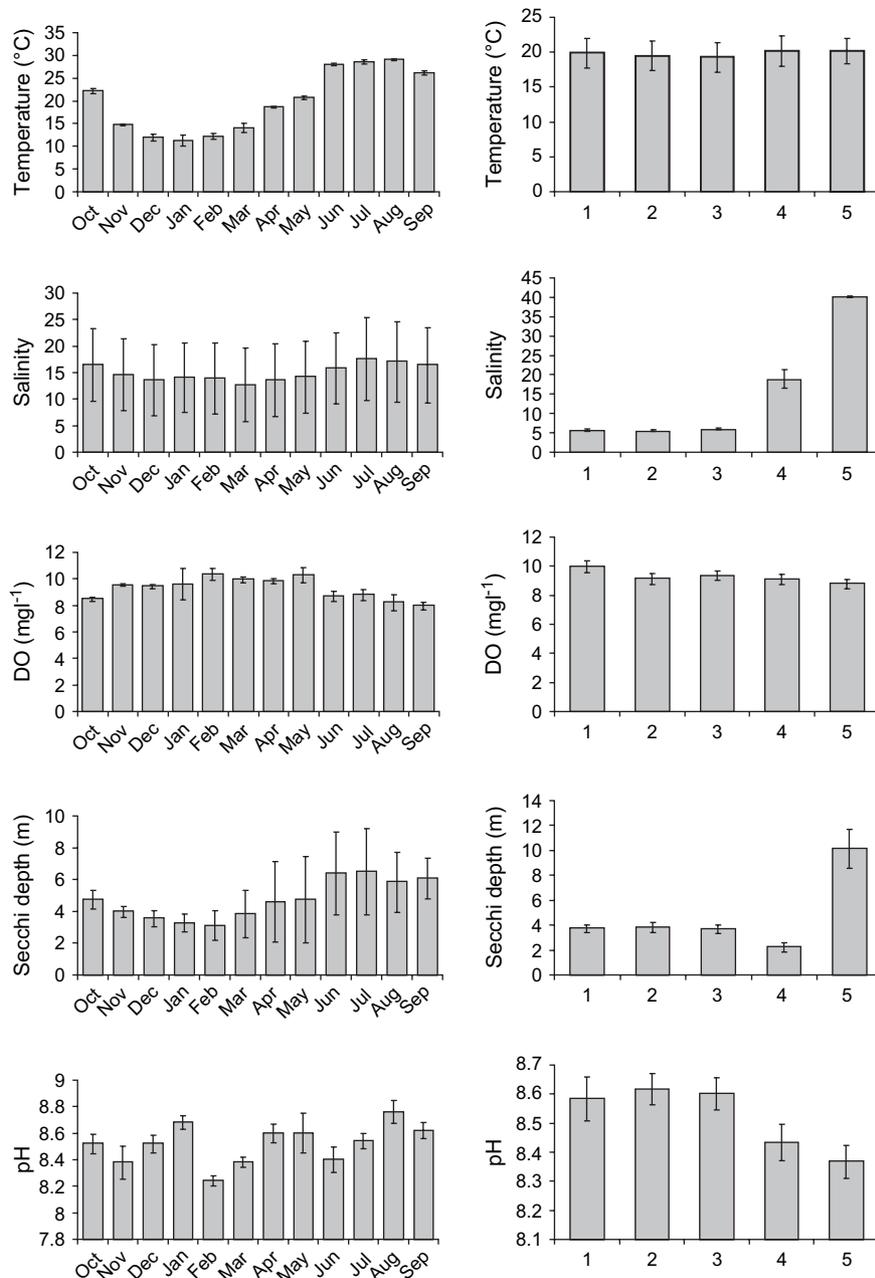


Fig. 2. Temporal and spatial variations in mean temperature, salinity, DO, Secchi depth, and pH at Koycegiz Lagoon–Estuarine System (bars \pm 1SE).

pH ranged from 8.1 to 8.9 (mean = 8.52; S.D. = 0.22) and varied significantly among months, attaining the highest and lowest mean values in August (8.76) and February (8.24), respectively ($F_{11, 43} = 4.59$; $P < 0.0001$) (Fig. 2). In general, pH followed a decreasing trend towards the lower reaches of the estuary. Lower sites (i.e., 4, and 5) had significantly lower values than the upper sites (1, 2, and 3) ($F_{4, 43} = 6.46$; $P = 0.0004$) (Fig. 2).

3.2. Fish community composition

A total of 42 fish species, representatives of 29 families, were collected during the period of the study

(Table 1). Most of these species were within the groups of marine and marine–estuarine-dependent, represented by 25 and 12 species, respectively. On the other hand, the number of species belonging to freshwater, estuarine residents, and catadromous groups was low, represented by only 3, 1, and 1 species, respectively (Table 1). In spite of being represented by highest number of species, numerical abundance of the marine species ($\#h^{-1}$) was lower than the abundances of other groups. Marine species contribution to the total abundance of the fishes was 13% only, a percentage lower than the contribution made by 3 freshwater species (27%) and 12 marine–estuarine-dependent species (52%). The numerical

Table 1

Frequency, CPUE rank, mean, minimum, and maximum values of species abundance ($\#h^{-1}$) at Koycegiz Lagoon–Estuarine System. MED: marine–estuarine-dependent; M: marine; F: freshwater; CA: catadromous; ES: estuarine resident

Family	Scientific name	Frequency	CPUE rank	Mean CPUE	Min. CPUE	Max. CPUE
Mugilidae	<i>Mugil cephalus</i> (MED)	73	4	0.96	0.18	4.55
	<i>Liza aurata</i> (MED)	70	2	1.36	0.18	6.00
	<i>Liza ramada</i> (MED)	67	7	0.66	0.18	3.00
	<i>Liza saliens</i> (MED)	58	3	1.23	0.18	5.27
	<i>Liza carinata</i> (MED)	2	41	0.18	0.18	0.18
	<i>Odeachilus labeo</i> (M)	3	39	0.27	0.27	0.27
	<i>Chelon labrosus</i> (MED)	18	10	0.91	0.18	2.45
Cyprinidae	<i>C. Capoeta bergamea</i> (FW)	33	5	1.78	0.18	8.00
Anguillidae	<i>Anguilla anguilla</i> (CA)	33	13	0.39	0.18	1.00
Cyprinodontidae	<i>Cyprinus carpio</i> (FW)	25	11	0.64	0.18	1.00
Cichlidae	<i>Tilapia</i> spp. (FW)	30	1	4.02	0.18	17.00
Atherinidae	<i>Atherina</i> spp. (ES)	75	6	0.61	0.18	2.73
Sparidae	<i>Sparus aurata</i> (MED)	22	12	0.64	0.18	1.73
	<i>Diplodus annularis</i> (MED)	7	29	0.48	0.27	1.00
	<i>Diplodus sargus</i> (MED)	7	20	0.80	0.18	2.00
	<i>Lithognathus mormyrus</i> (MED)	22	9	0.91	0.18	3.00
	<i>Sarpa salpa</i> (M)	3	31	0.68	0.36	1.00
	<i>Pagellus acarne</i> (M)	10	17	0.71	0.18	1.00
	<i>Boops boops</i> (M)	2	38	1.00	1.00	1.00
	<i>Diplodus vulgaris</i> (M)	3	33	0.59	0.18	1.00
Moronidae	<i>Dicentrarchus labrax</i> (MED)	3	28	1.00	1.00	1.00
Sphraenidae	<i>Sphraena sphraena</i> (M)	5	21	1.00	1.00	1.00
Synodontidae	<i>Synodus saurus</i> (M)	5	30	0.45	0.18	1.00
Scombridae	<i>Scomber scombrus</i> (M)	2	36	1.00	1.00	1.00
Scaridae	<i>Sparisoma cretense</i> (M)	3	25	1.00	1.00	1.00
Centracanthidae	<i>Spicara smaris</i> (M)	2	34	1.00	1.00	1.00
Clupeidae	<i>Sardinella aurita</i> (M)	7	19	0.86	0.45	1.00
Engraulidae	<i>Engraulis encrasicolus</i> (MED)	27	8	1.35	0.27	4.73
Scorpaenidae	<i>Scorpanea scrofa</i> (M)	2	35	1.00	1.00	1.00
Lichidae	<i>Lichia amia</i> (M)	2	38	1.00	1.00	1.00
Mullidae	<i>Mullus barbatus</i> (M)	12	14	0.88	0.18	1.00
Triglidae	<i>Trigla lyra</i> (M)	8	23	0.53	0.18	1.00
Trichiidae	<i>Trachinus areneus</i> (M)	7	24	0.59	0.18	1.00
Uranoscopidae	<i>Uranoscopus scaber</i> (M)	12	18	0.51	0.18	1.00
Phycidae	<i>Physic physic</i> (M)	3	27	1.00	1.00	1.00
Labridae	<i>Xyrichtys novacula</i> (M)	3	32	0.59	0.18	1.00
Echeneidae	<i>Remora remora</i> (M)	3	26	1.00	1.00	1.00
Bothidae	<i>Bothus podas</i> (M)	10	16	0.86	0.18	1.27
Dasyatidae	<i>Dasyatis pastinaca</i> (M)	12	22	0.40	0.18	1.00
Sepiidae	<i>Sepia officinalis</i> (M)	18	15	0.53	0.18	1.00
Loliginidae	<i>Loligo vulgaris</i> (M)	2	37	1.00	1.00	1.00
Octopodidae	<i>Octopus vulgaris</i> (M)	2	40	0.18	0.18	0.18

abundance of the other fish group, i.e. estuarine resident and catadromous species solely represented by Atherinidae and Anguillidae families, was 6 and 2% of the total catch, respectively. Mugilidae, represented by 7 species, contributed 41% of numerical abundance of the total catch, and mostly consisted of *Liza aurata* (13.08%), *Liza saliens* (9.82%), and *Mugil cephalus* (9.63%). Although Sparidae was represented by 7 species, the contribution of these species to the numerical abundance of samples was quite low (7.5%). *Tilapia* spp. was the most abundant freshwater species comprising 17% of the numerical abundance. The other common species were the sole representatives of the

Atherinidae and Engraulidae: silversides (*Atherina* spp.) and anchovy (*Engraulis encrasicolus*), which comprised 6.3% and 5% of the total catch, respectively. The remaining species were mostly collected at site 5 and each contributed almost 1% of the total catch (Table 2).

3.3. Spatial and temporal variation in fish abundance

Fish abundance (CPUE) was the highest between sites 1 and 3 during September–December and July–August periods (Fig. 3), with a significant site \times time interaction ($F_{1,43} = 11.97$; $P < 0.0001$). This interaction was probably

Table 2
Fish abundance ($\#h^{-1}$) and percentage of contribution to the total catch of each site based on data aggregated across all seasons

Family	Scientific name	Sites									
		1		2		3		4		5	
		Num.	%	Num.	%	Num.	%	Num.	%	Num.	%
Mugilidae	<i>Mugil cephalus</i> (MED)	8.91	9.94	10.45	13.34	14.18	10.50	6.64	9.76	2.00	2.99
	<i>Liza aurata</i> (MED)	11.73	13.08	9.09	11.60	29.73	22.01	6.00	8.82	0.73	1.09
	<i>Liza ramada</i> (MED)	4.64	5.17	3.36	4.29	10.00	7.40	5.18	7.62	3.36	5.03
	<i>Liza saliens</i> (MED)	7.00	7.81	14.00	17.87	8.36	6.19	12.18	17.91	1.45	2.17
	<i>Liza carinata</i> (MED)	–	–	–	–	–	–	–	–	0.18	0.27
	<i>Odeachilus labeo</i> (M)	–	–	–	–	0.27	0.20	0.27	0.40	–	–
	<i>Chelon labrosus</i> (MED)	–	–	2.18	2.78	3.55	2.62	4.27	6.28	–	–
Cyprinidae	<i>C. Capoeta bergamea</i> (FW)	14.64	16.33	4.36	5.57	16.64	12.32	–	–	–	–
Anguillidae	<i>Anguilla anguilla</i> (CA)	2.36	2.64	3.36	4.29	2.09	1.55	–	–	–	–
Cyprinodontidae	<i>Cyprinus carpio</i> (FW)	4.18	4.67	1.18	1.51	4.27	3.16	–	–	–	–
Cichlidae	<i>Tilapia</i> spp. (FW)	25.09	27.99	17.64	22.51	29.36	21.74	0.18	0.27	–	–
Atherinidae	<i>Atherina</i> spp. (ES)	7.27	8.11	7.00	8.93	7.36	5.45	5.91	8.69	–	–
Sparidae	<i>Sparus aurata</i> (MED)	–	–	–	–	1.00	0.74	7.36	10.83	–	–
	<i>Diplodus annularis</i> (MED)	–	–	–	–	–	–	–	–	1.91	2.85
	<i>Diplodus sargus</i> (MED)	–	–	–	–	–	–	2.55	3.74	0.64	0.95
	<i>Lithognathus mormyrus</i> (MED)	–	–	–	–	–	–	6.64	9.76	5.18	7.74
	<i>Sarpa salpa</i> (M)	–	–	–	–	–	–	–	–	1.36	2.04
	<i>Pagellus acarne</i> (M)	–	–	–	–	–	–	–	–	4.27	6.39
	<i>Boops boops</i> (M)	–	–	–	–	–	–	–	–	1.00	1.49
	<i>Diplodus vulgaris</i> (M)	–	–	–	–	–	–	–	–	1.18	1.77
Moronidae	<i>Dicentrarchus labrax</i> (MED)	–	–	–	–	2.00	1.48	–	–	–	–
Sphyraenidae	<i>Sphyraena sphyraena</i> (M)	–	–	–	–	–	–	–	–	3.00	4.48
Synodontidae	<i>Synodus saurus</i> (M)	–	–	–	–	–	–	–	–	1.36	2.04
Scombridae	<i>Scomber scombrus</i> (M)	–	–	–	–	–	–	–	–	1.00	1.49
Scaridae	<i>Sparisoma cretense</i> (M)	–	–	–	–	–	–	–	–	2.00	2.99
Centracanthidae	<i>Spicara smaris</i> (M)	–	–	–	–	–	–	–	–	1.00	1.49
Clupeidae	<i>Sardinella aurita</i> (M)	–	–	–	–	–	–	3.45	5.08	–	–
Engraulidae	<i>Engraulis encrasicolus</i> (MED)	3.82	4.26	5.73	7.31	6.27	4.64	5.36	7.89	0.36	0.54
Scorpaenidae	<i>Scorpaena scrofa</i> (M)	–	–	–	–	–	–	–	–	1.00	1.49
Lichiiidae	<i>Lichia amia</i> (M)	–	–	–	–	–	–	–	–	1.00	1.49
Mullidae	<i>Mullus barbatus</i> (M)	–	–	–	–	–	–	2.00	2.94	4.18	6.25
Triglidae	<i>Trigla lyra</i> (M)	–	–	–	–	–	–	–	–	2.64	3.94
Trichiidae	<i>Trachinus areneus</i> (M)	–	–	–	–	–	–	–	–	2.36	3.53
Uranoscopidae	<i>Uranoscopus scaber</i> (M)	–	–	–	–	–	–	–	–	3.55	5.30
Phycidae	<i>Physic physic</i> (M)	–	–	–	–	–	–	–	–	2.00	2.99
Labridae	<i>Xyriichthys novacula</i> (M)	–	–	–	–	–	–	–	–	1.18	1.77
Echeneidae	<i>Remora remora</i> (M)	–	–	–	–	–	–	–	–	2.00	2.99
Bothidae	<i>Bothus podas</i> (M)	–	–	–	–	–	–	–	–	5.18	7.74
Dasyatidae	<i>Dasyatis pastinaca</i> (M)	–	–	–	–	–	–	–	–	2.82	4.21
Sepiidae	<i>Sepia officinalis</i> (M)	–	–	–	–	–	–	–	–	5.82	8.70
Loliginidae	<i>Loligo vulgaris</i> (M)	–	–	–	–	–	–	–	–	1.00	1.49
Octopodidae	<i>Octopus vulgaris</i> (M)	–	–	–	–	–	–	–	–	0.18	0.27

due to solely greater numbers of individuals captured during July (16 $\#h^{-1}$), August (33 $\#h^{-1}$), September (20 $\#h^{-1}$) and November (18 $\#h^{-1}$) at site 3. In these months different common species achieved their peak abundance. Abundance peaks observed during October–December and July–August periods were due to high densities of *Tilapia* spp., *Liza aurata*, *Mugil cephalus*, *Liza saliens*, thick-lipped grey mullet (*Chelon labrosus*), and *Capoeta capoeta bergamea*, *Engraulis encrasicolus*, and *Atherina* spp. *Tilapia* spp. and *L. aurata* were largely responsible for an abundance peak in October, whereas a peak in November was largely due to these two species plus *L. saliens* and *C. capoeta bergamea*. Five species

(*L. saliens*, *C. labrosus*, *C. capoeta bergamea*, *M. cephalus*, *L. aurata*, and *Liza ramada*) almost equally (11%) contributed to the peak in December. An abundance peak in July was due to an influx of *C. capoeta bergamea*, *E. encrasicolus*, and *M. cephalus*, and each species comprised almost 20% of the July sample. During August, *Tilapia* spp., *M. cephalus*, and *Atherina* spp. were the most abundant species contributing almost 50% of the catch. However, the peak during September was largely influenced by *Tilapia* spp. that comprised almost 65% of the sample.

The highest and lowest mean abundance were obtained at sites 3 and 5, respectively (Fig. 3). Site 5 had lower fish

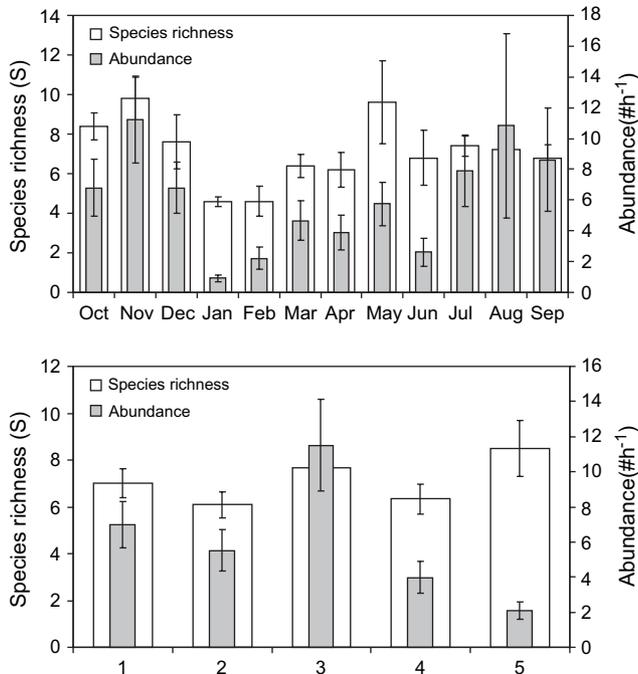


Fig. 3. The temporal and spatial variations in mean species richness and abundance of fishes at Koycegiz Lagoon–Estuarine System (bars \pm 1SE).

abundance (mean = 2 #h⁻¹) than other sites. The large number of fishes collected at site 3 might be due to its location. Located at the opening of the lake to the Dalyan River, site 3 was a transition zone between freshwater and seawater. High fish abundance recorded at site 3 was primarily influenced by *Tilapia* spp., especially during summer and early spring, and four species of Mugilidae (e.g., *Mugil cephalus*, *Liza aurata*, *Liza ramada*, and *Liza saliens*). Overall, fish abundance was inversely correlated with species richness ($r = -0.68$, $P < 0.0001$), salinity ($r = -0.51$, $P < 0.0001$), pH ($r = -0.32$, $P < 0.05$), and time ($r = -0.47$, $P < 0.0001$).

3.4. Spatial and temporal variation in species richness

Species richness ranged from 2 to 18 species (February '94 site 1 and May site 5) (Fig. 3). Monthly mean values of species richness differed significantly ($F_{11, 43} = 2.79$; $P = 0.0079$). In general, species richness was higher during fall and spring–summer months than winter months. November (9.8) and May (9.6) samples yielded significant higher mean number of species than January and February samples, both of which had mean values of 4.6. Although mean species richness did not reveal a particular pattern among sites, sites 3 (7.7) and 5 (8.5) had relatively higher number of species than other sites. But these differences among sites were not statistically significant ($F_{4, 43} = 2.99$; $P = 0.0910$). On the other

hand, the total number (not the averages) of species obtained at each site throughout the study exhibited a strong spatial variation (Table 2). Sites 1 and 2 yielded 10 species, whereas 14 species were captured at sites 3 and 4. Site 5, on the other hand, yielded 32 species mostly consisting of marine species (Table 2). Across all samples, species richness was correlated with, pH ($r = 0.36$; $P = 0.005$), Secchi depth ($r = -0.44$; $P < 0.0001$), salinity ($r = 0.301$; $P = 0.020$), and distance ($r = -0.255$; $P = 0.049$).

3.5. Canonical correspondence analysis

CCA eigenvalues of the first four multivariate axes were 0.52 (CCA1), 0.18 (CCA2), 0.12 (CCA3), and 0.03 (CCA4). Species–environment correlation coefficients for the first four axes were 0.91, 0.74, 0.75, and 0.49, respectively. Cumulative percentage variance of species for the first four axes (CCA 1–4) was 29.6. The first and second axes modeled 18.2% and 6.2% of species data, respectively, and they cumulatively accounted for 81.7% of variance of species–environment relationship modeled by CCA. Therefore, results obtained from the first two axes were plotted (Fig. 4).

The length of vector of a given variable on the CCA plots indicates the importance of that variable. Salinity (0.84), which had the longest vector along the first axis, was significantly correlated with the first axis which explained most of the variation (59.1%) in the species data. The second axis, which explained 19.8% of the variation, was only significantly (-0.65) associated with

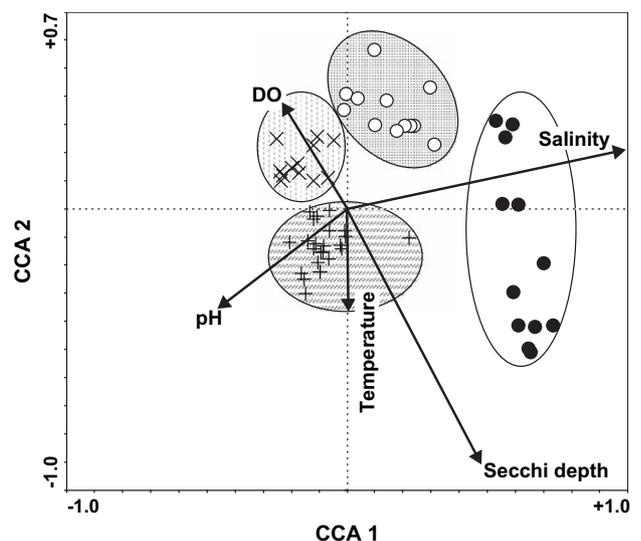


Fig. 4. Plot of site scores on the first two axes with five environmental variables from CCA. Sites 1, 2, and 3 of summer and early fall months (▨), sites 1, 2, and 3 of winter and early spring months (□), site 4 of all months (▨), and site 5 of all month (□).

Secchi depth (Fig. 4). Secchi depth was also significantly associated with the first axis (0.42), given the assumption that any variable having an inter-set correlation coefficient = $|0.4|$, which is the correlation coefficient between site scores (derived from the species scores) and the environmental variables, was biologically important (Rakocinski et al., 1996). The first axis was highly correlated with salinity and Secchi depth and distinguished upper estuarine sites (1, 2, 3) from sites 4 and 5. Secchi depth varied along both spatial and temporal gradients. Secchi depth, like salinity, separated the highly turbid sites of upper reaches from low turbidity sites located at the sea (site 5). Secchi depth separated site 4 during all months and sites 1, 2, 3 and 5 during late fall and winter months from samples obtained during late spring and summer months, the latter having low turbidity.

Associations between environmental parameters and the most abundant species were demonstrated in Fig. 5. Species plotted closer to the vector, or other species have stronger relationships with them. Species located near the origin either do not show a strong relationship to any of the variables or are found at average values of environmental variable (Marshall and Elliott, 1998). Thus, most species in the estuary had average values in relation to environmental variables. Only a few marine species such as wide-eyed flounder (*Bothas podas*), red mullet (*Mullus barbatus*), and sea bream (*Lithognathus mormyrus*) indicated a strong response to the longitudinal salinity gradient (Fig. 5).

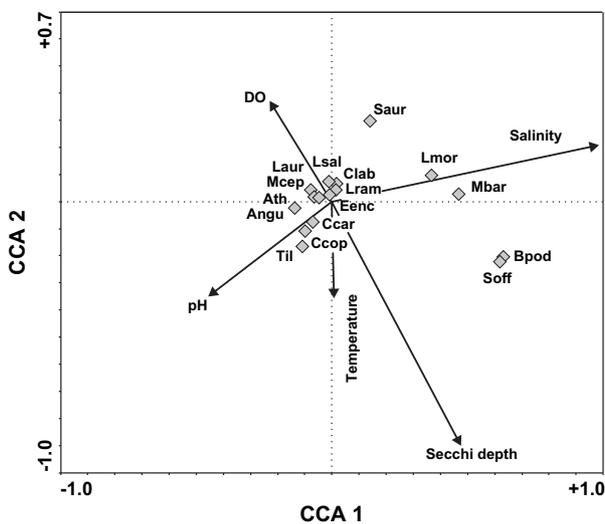


Fig. 5. Plot of species scores with 5 environmental variables in the first two CCA axes. **Angu**, *Anguilla anguilla*; **Ath**, *Atherina* spp; **Bpod**, *Bothus podas*; **Ccar**, *Cyprinus carpio*; **Ccop**, *C. Copoeta bergamea*; **Clab**, *Chelon labrosus*; **Eenc**, *Engraulis encrasicolus*; **Laur**, *Liza aurata*; **Lsal**, *Liza saliens*; **Lram**, *Liza ramada*; **Lmor**, *Lithognathus mormyrus*; **Mbar**, *Mullus barbatus*; **Mcep**, *Mugil cephalus*; **Saur**, *Sparus aurata*; **Soff**, *Sepia officinalis*; **Til**, *Tilapia* spp.

4. Discussion

4.1. Species composition

Relatively low species richness but high abundance of some species, many of which had a broad range of tolerance limits to fluctuating abiotic conditions, was a general fish assemblage structure of KLES. Fish assemblage structure of KLES consisted of approximately the same ecological groups that are common in European estuaries (Drake and Arias, 1991; Maes et al., 1998; Marshall and Elliott, 1998; Araújo et al., 1999; Laffaille et al., 2000; Thiel and Potter, 2001). Marine species in KLES, mostly restricted on the lower reaches of the estuary, consisted of more species than estuarine-dependent species, however, marine–estuarine-dependent species were numerically more abundant than the marine ones, a general pattern of estuarine systems (Gunter, 1938; Rogers et al., 1984; Loneragan et al., 1987; Rozas and Minello, 1998; Whitfield, 1999; Akin et al., 2003). Comparisons of species richness between estuaries are tenuous at best due to differences in sampling methods (i.e., gear, the length of the survey, location of the samples taken) and geographic variation (Kneib, 1997). KLES supported fewer species than most of the European estuaries including the Thames Estuary (62 species) (Araújo et al., 1999), Forth Estuary (43), Scotland (Costa and Elliott, 1991), Targus Estuary (45), Portugal, (Costa and Elliott, 1991), Scheldt Estuary (60), Belgium (Maes et al., 1998), Elbe Estuary (58) (Thiel and Potter, 2001). The number of species captured in KLES, however, was higher than those of Mont-Saint Michel Bay Macrodital Salt Marsh (31), France (Laffaille et al., 2000) and those of Cadiz Bay (39), Spain (Drake and Arias, 1991) even though we captured fishes with trammel nets only. The total number of species in KLES is undoubtedly higher, and could be documented using other collecting methods.

Following hatching at the sea, marine–estuarine-dependent species enter highly productive and vegetated shallow upper reaches of estuaries for growth and survival (Gunter, 1938; McErlean et al., 1973; Whitfield, 1999). The orientation of these marine–estuarine-dependent species to the upper reaches of estuaries has been hypothesized to be facilitated by the catchments olfactory cues transported by freshwater inflow (Whitfield, 1999). Collection of many estuarine-dependent species in upper reaches of estuaries during periods of high freshwater inflow in many estuarine studies (Rogers et al., 1984; Grimes and Kingsford, 1996; and many others) has supported this view. Even though upper reaches of KLES had relatively low salinity (5), only a few species of abundant marine–estuarine-dependent took advantage of these productive habitats. Most of these species (members of Mugilidae) are known to withstand a broad range of salinity concentrations (Thomson, 1966; Hotos

and Vlahos, 1998). The other estuarine–marine dependent, such as gilthead bream (*Sparus aurata*), annular seabream (*Diplodus annularis*), white seabream (*Diplodus sargus*), striped sea bream (*Lithognathus mormyrus*), and sea bass (*Dicentrarchus labrax*), on the other hand, had distributions limited to the lower reaches where salinity concentration was close to that of seawater (35). In other estuarine studies, however, these species have been reported at sites with lower salinity. For example, in Scheldt Estuary, Belgium, Maes et al. (1998) recorded *D. labrax* at sites where salinity ranged from 0.6 to 13.4. The different spatial distribution patterns obtained in our study and that of Maes et al. (1998) were probably due to use of different sampling effort. With trammel nets, we captured mainly larger fishes that maybe more sensitive to variation in salinity compared to juveniles or larvae (Akin et al., 2003).

4.2. Spatial and temporal variation in species richness and abundance

Assemblage structure in estuaries is shaped by both abiotic and biotic environmental components (Blaber and Blaber, 1980; Weinstein et al., 1980; Rozas and Hackney, 1984; Rakocinski et al., 1996; Maes et al., 1998; Marshall and Elliott, 1998; Araújo et al., 1999; Whitfield, 1999; Garcia et al., 2001; Gelwick et al., 2001; Akin et al., 2003, and many others). Salinity and temperature have been postulated to be important determinants of spatial and temporal assemblage structure (Grioche et al., 1999; Akin et al., 2003; Hagan and Able, 2003). Salinity was the one of two main factors affecting fish assemblage structure in KLES. As revealed by CCA, salinity contributed a strong spatial gradient separating upper reaches of the estuarine sites from the lower reaches. Upper reaches were dominated by mugilids, freshwater, anadromous, and resident species. Lower reaches, on the other hand, were dominated by marine species, such as such as seabream, striped seabream, red mullet, flounder, and common cuttle fish. Salinity requirements of juvenile and adult of the same species can differ. For example, studying the effects of salinity on habitat selection and growth performance of *Mugil cephalus*, Cardona (2000) indicated that juvenile mullet preferred fresh or oligohaline waters to polyhaline and euhaline waters. In contrast, adults always avoided freshwaters and preferred euhaline waters.

Juvenile or larvae fishes use estuaries not only for feeding but also for predator avoidance. High turbidity has been cited as factors that enhance survival of estuarine fishes (Abrahams and Kattenfeld, 1997; Whitfield, 1999). Turbidity gradients between the sea and adjacent estuaries provide orientation cues for juveniles migrating into estuaries and lagoons (Blaber, 1997). As revealed by CCA, turbidity associated with the first and second axes was significantly associated with spatial and

temporal variation in fish assemblage structure. Upper reaches were less turbid during summer and early fall, and had high turbidity during winter and spring months. An increase in turbidity during spring and winter was probably a result of heavy rainfall (E. Buhan Personal Observation). Even though turbidity varied seasonally, it did not seem to affect temporal variation in fish assemblage structure. Regardless of season, assemblage structure at upper sites consisted of mullets, estuarine residents, and freshwater and catadromous species. Thus, spatial variation seen in fish assemblage structure was more pronounced than the spatial variation created by salinity and turbidity gradients.

The turbidity preference of fishes is a species-specific response (Cyrus and Blaber, 1987a,b) and variable with the age of the fish (Blaber and Cyrus, 1983). For example, a study by Blaber and Cyrus (1983) demonstrated *Caranx sexfasciatus* primarily occupied turbid areas as juveniles and more transparent water as adults. These ontogenetic habitat shifts may reduce competition (Blaber and Cyrus, 1983). In addition to reducing intraspecific competition, turbidity could influence habitat selection based on costs and benefits of the feeding. For example, species that inhabit turbid estuarine waters were marine–estuarine-dependent (mostly members of Mugilidae), resident, freshwater, and anadromous species; which were relatively smaller in size (E. Buhan Personal Observation) with many species (especially mullets) serving as important prey of marine piscivores (Akin, pers. comm). Even though upper reaches had highly abundant resources for other marine estuarine species (E. Buhan Personal Observation), these areas of the estuary apparently were not preferred by marine migrants. Time and energy spent locating and capturing prey in turbid waters is greater than in clear waters (Whitfield et al., 1994), which might be a factor, together with salinity contributing to spatial distributions of fishes in KLES.

Estuaries tend to have more species at lower reaches than those of upper reaches (Whitfield, 1999; Akin et al., 2003; Martino and Able, 2003). Greater numbers of species in lower reaches have been linked to prevalent marine conditions (Maes et al., 1998; Martino and Able, 2003; Vega-Candejas and de Santillana, 2004). In KLES, most of the species occurring at lower reaches were of marine origin. The high number of species occurring in lower reaches has been related to the factors other than salinity and turbidity. For example, Yoklavich et al. (1991) argued that occupation of lower reaches of the estuary and absence from upper reaches by many marine species is due to fact that lower reaches are strongly influenced by marine processes, and warm temperature at upper sites excludes thermally intolerant marine species. They further argued that habitat complexity in lower reaches supports many fish species. Another factor that raised by Loneragan et al. (1987) and Martino and Able (2003) is the size of the estuarine systems or geographic

distance. They found a reverse relationship between number of species and distance from the estuary mouth. We also found an inverse relationship between geographic distance and number of species in KLES. Spatial covariation among salinity, turbidity, and geographic distance makes it difficult to ascertain whether the spatial assemblage patterns were driven by salinity, turbidity, or geographic distance. Larger fish can easily travel to upper reaches of KLES which should eliminate the geographic distance as a factor. Occurrence of mullets, which are highly tolerant to fluctuating environmental conditions including salinity (Thomson, 1966) and turbidity, at both upper and lower reaches and occurrence of marine species, such as *Mullus barbatus* and *Bothus podas*, which are commonly found in clear and deep waters with salinity ranging from 18 to 40 in Turkey (B. Cihangir, pers. comm.), in lower reaches suggested that salinity and turbidity together might influence assemblage structure of KLES.

In addition to salinity tolerances and preferences, the spatial pattern described above could be related to fish feeding habits. For example, Gordo and Cabral (2001) reported that microalgae and detritus feeders, such as mugilids, are more common in the muddy inland areas with higher abundance of detritus and algae. They further reported that these areas were exploited by species that fed on infauna. Conversely, lower reaches were associated with zooplanktivores and macroinvertebrate feeders. Occurrence of mugilids mostly in upper reaches, and occurrence of striped seabream, red mullet, and wide-eyed flounder (*Bothus podas*), all of which mainly feed on macrofaunal food items such polychaetes, gastropods, echinoderm, decapods, bivalves (Stergiou and Karpouzi, 2002) in lower reaches suggests that spatial assemblage structure at KLES is partially explained by feeding preferences of fishes in addition to salinity and turbidity gradients.

Variation in species distribution explained by the first four axes of CCA was 29.6%, which indicates that some other physical and biological factors, not measured in this study, could account some of the unexplained variation. These factors could be aquatic vegetation (Zimmerman and Minello, 1984; Killgore et al., 1989; West and King, 1996; Rozas and Minello, 1998; Akin et al., 2003), food availability (Rozas and Hackney, 1984; Barry et al., 1996; Kneib, 1997), sediment type (Marchand, 1993), status of the estuarine mouth (open or intermittently open) (Young et al., 1997; Bell et al., 2001; Griffiths, 2001), and biological interrelationships (i.e., group behavior, competition, and predation Martino and Able, 2003) have been cited as being associated with fish assemblage structure in other estuarine systems. Although these variables were not examined here, the five variables measured in this study explained species distributions well compared with most other estuarine studies. For example, Marshall and Elliott (1998) found that five

environmental variables accounted for 18.4% of the total species variation even though they included bottom, mid, and surface values of each variable in CCA. Studying the ichthyoplankton assemblage structure in Mississippi Sound, Rakocinski et al. (1996) used 11 environmental variables that together explained only 21.9% of the total species variation in CCA. On the other hand, Martino and Able (2003) explained 29.9% of the total species variation in Mullica River Estuary, New Jersey, using five environmental variables that included salinity and geographic distance. Similar to our investigation, their study provided a large-scale perspective on fish assemblage structure across an ocean–estuarine ecotone, with similar ranges of salinity [0.1–32 in Martino and Able (2003); 5–40 in our study]. Martino and Able (2003) concluded that large-scale patterns in the structure of estuarine fish assemblages were primarily a result of individual species responses to a dominant environmental gradient, whereas smaller-scale patterns seemed to be the result of biotic habitats such as predator avoidance, competition, and habitat selection.

4.3. Abundant species

Mullets, the numerically most abundant fishes at KLES, are economically important fishes of estuaries and lagoons of the Mediterranean Sea (Kapetsky and Lasserre, 1984; Yerli, 1991; Buhan, pers. comm.). Of the 7 species of Mugilidae at KLES, *Liza aurata*, *Mugil cephalus*, *Liza ramada*, and *Liza saliens* numerically comprised 94% of the mullet species sampled. High abundance and high percentage of occurrence of these species within the estuary indicated that they are tolerant to fluctuating environmental conditions in this system.

According to the Schoener (1974), resource partitioning may occur by segregation along the one of three main resource axes: food, space, and time. In coastal lagoons and estuaries, time segregation among juveniles of migratory fishes occurs from differential recruitment derived from differences in spawning periods (Drake and Arias, 1991; Harmelin-Vivien et al., 1995). In addition to being inversely associated with salinity, *Mugil cephalus* also was negatively correlated with the abundance of *Liza saliens*. *Mugil cephalus* were more abundant during summer whereas *L. saliens* were more abundant during the late fall. Studying the resource partitioning between Mediterranean grey mullets, Cardona (2001) reported that *L. saliens* and *M. cephalus* had higher trophic overlap during fall than other seasons, which suggests that temporal segregation of these two species in KLES could reduce competition. In contrast, *L. saliens* and *Liza ramada*, both of which have similar diets consisting of detritus, diatoms, and other benthic microalgae (Cardona, 2001), co-occurred over most of the year, which may increase the potential for competition.

5. Conclusions

High species richness at lower reaches of the estuary, high abundance of estuarine-dependent species, and high numbers of species during spring, summer-early autumn were dominant features of KLES. The spatial and temporal distributions of fishes were negatively associated with Secchi depth and positively associated with salinity. Less turbid waters at freshwater–seawater interface zone were occupied by mostly marine species that apparently avoided low salinity and high turbidity. In contrast, mullets and several other fishes known to tolerate a broad range of salinity concentrations were broadly distributed. Thus, patterns in fish assemblage structure of this large-scale lagoon–estuarine system seemed to be influenced primarily by species-specific responses to dominant environmental gradients. The results of this study together with those of Martino and Able (2003) are consistent with the theoretical assumptions of community structure which imply that physiological tolerances of organisms to the dominant gradient determine the frame of the structure, while biotic interactions determine the species distribution within this frame (Menge and Olson, 1990). Biotic interactions may account some of the unexplained variation in fish assemblage structure of KLES, however, the quantity of this variation needs to be determined by further investigations of fish assemblage structure in KLES.

Acknowledgments

We thank Yavuz Morkan, Murat Topgul, and Zafer Akpınar for assistance with data and specimen collection in the field. Funding was provided by Turkish Ministry of Agriculture, Bodrum Fisheries Research Institute. The comments of Professors Whitfield, Cyrus and Elliott were greatly appreciated.

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