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Multi-annual changes in the parasite communities of rabbitfish *Siganus rivulatus* (Siganidae) in the Gulf of Aqaba, Red Sea

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Abstract The parasite communities of the rabbitfish, *Siganus rivulatus*, were used to track multi-annual changes in the northern Gulf of Aqaba, Red Sea, in an environment subjected to ongoing anthropogenic impact. Parasitological data from these fish were collected from 1998 to 2000, with spring and fall samplings at three locations: at a coral reef (OBS), at a sandy beach area (NB) and at a mariculture cage farm (FF). These data were compared with data from 1995–1997 as well as data collected during 1981–1985 at the coral reef sampling site. The data analyses indicate that the ratio between heteroxenous and monoxenous parasite species declined significantly at all sites between 1995–1997 and 1998–2000. During the same period, the species richness of monoxenous parasites increased significantly at all sites. The species richness of heteroxenous parasites decreased significantly at the coral reef site, but remained steady at the other two sites. This coincided with a significant increase in the prevalence of monogeneans at the OBS and FF sites and a significant decrease in the prevalence of digeneans at the FF and NB sites. The decline in the abundance of the latter, specifically of *Opisthogonoporoides* sp. and *Gyliauchen* sp., was even more significant when compared with the 1981–1985 data. The prevalence of other gut helminths, namely the digenean *Hexangium sigani* and the nematodes *Cucullanus sigani* and *Procamallanus elatensis*, however, showed a significant increase over the same period. Analysis of the species richness and diversity indices of the parasite communities did not reveal conspicuous differences. These, however, did become apparent when heteroxenous

and monoxenous members of particular taxa were analyzed separately. Therefore, when using parasite assemblages to detect ecological changes, it is essential to analyze not only at the community level, but also to consider separate components of particular parasitic groups.

Keywords Diversity indices · Species richness · Red Sea · Rabbitfish · *Siganus rivulatus*

Introduction

Fish parasites with complex life histories have been used as a tool for the determination of geographical boundaries (Koie 1983, 1984; Bamber and Henderson 1985), migration routes (MacKenzie 1987; Diamant 1989b) breeding territories (MacKenzie 1987) and predation behavior (Khan and Kiceniuk, 1988). The use of fish parasites as bioindicators and tags has led to their utilization as monitors of perturbations in aquatic ecosystems. While physical, chemical and even biological oceanographic parameters have constraints in time and space, fish parasites integrate and accumulate adverse effects of complex and variable stresses that influence them as well the other components of the ecosystem. Therefore, parasite community indices should arguably present an accurate and comprehensive picture on the state of an ecosystem. Various species of parasites and the parasite community as a whole have been proposed as bioindicators for environmental pollution. Changes in the prevalence and intensity of infection in certain parasite taxa (Khan 1987; Khan and Kiceniuk 1988; Khan and Payne 1997; Margolis and Cone 1997; Yeomans et al. 1997), as well as changes in the composition of parasite communities (D'Amelio and Gerasi 1997; Gelnar et al. 1997; Valtonen et al. 1997; Broeg et al. 1999; Diamant et al. 1999) have been used to estimate anthropogenic impacts on aquatic environments. Relatively few studies have, however, addressed long-term changes in fish parasite communities as bioindicators for ecosystem stress (Kennedy 1985, 1993, 1997a; Koskivaara and Valtonen 1992;

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Kennedy et al. 1994). In addition to indices derived from population structure and diversity, the ratio between heteroxenous and monoxenous parasite species (Sh/Sm) was introduced as a parameter for adverse changes in the state of the environment (D'Amelio and Gerasi 1997; Diamant et al. 1999).

In the last two decades, the northern Gulf of Aqaba has been undergoing accelerated environmental degradation which has been associated with adverse anthropogenic impacts of urban and port development, tourism and mariculture (Fishelson 1995; Schuhmacher et al. 1995). However, this association was mostly circumstantial and not backed with biological studies. The rabbitfish (*Siganus rivulatus*) is one of the most conspicuous and abundant components of the coastal fish communities in the northern Red Sea. This fish was therefore chosen as a model for studies using parasites as bioindicators for monitoring the extent of anthropogenic impact on habitats in the Gulf of Aqaba (Banet 1998; Diamant et al. 1999). Initial studies on the parasites of this fish were carried in the mid 1980s and later, since 1995 (Diamant 1985, 1989a, 1989b; Diamant and Paperna 1986; Banet 1998; Diamant et al. 1999).

In the present communication, we analyze multi-annual changes in the rabbitfish parasite communities, from 1995 to 2000, compare them with those of the mid 1980s and discuss the possible causes for these changes.

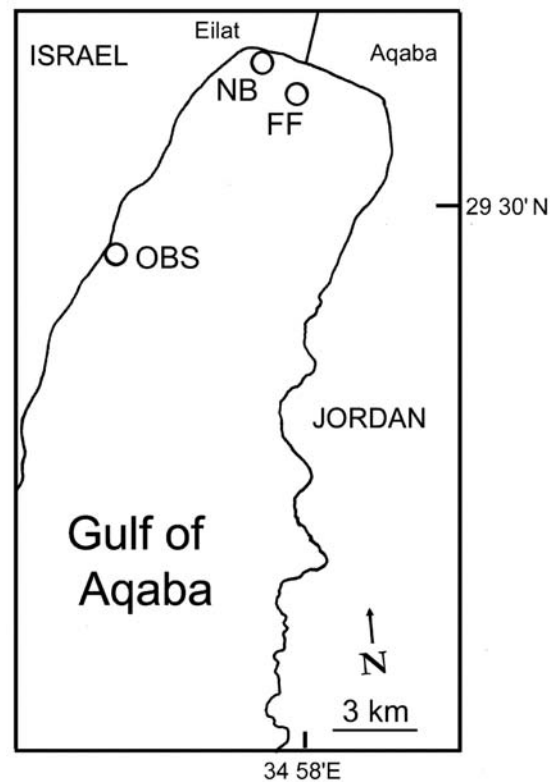


Fig. 1 Map and coordinates of sampling sites at the northern Gulf of Aqaba. NB, North Beach; FF, fish farm; OBS, observatory

Methods

Location and sampling design

Parasitological data on *S. rivulatus* were collected from 1995 to 2000 at three sites along the Israeli northern part of the Gulf of Aqaba, Red Sea: FF, at the periphery of an offshore mariculture cage farm; NB, at the north-western, sandy, tourist beach next to the hotel district of Eilat, and OBS, in the western coral fringed zone (a nature reserve) near the Eilat underwater observatory (Fig. 1). Data from 1981 to 1985 were obtained from fish collected at OBS (Diamant 1985).

Parasitological data processing

Fish were trapped and kept alive in seawater tanks until they were dissected for parasitological examination (within 48 h of capture). Parasites were identified and quantified according to the protocol described in Diamant et al. (1999).

Table 1 General parameters of the rabbitfish (*Siganus rivulatus*) sampled at the Red Sea sampling sites. FF, fish farm; NB, North Beach; OBS, observatory. Results are presented as mean±SE. The

numbers of heteroxenous parasite species are in brackets. Significant differences are marked by superscript (a, b...)

	FF		NB		OBS		OBS
Total No. of parasites species	22 (11)		21 (11)		23 (12)		7***
Sampling period	95-00		95-00		95-00		81-85
	95-97	98-00	95-97	98-00	95-97	98-00	
Total fish Number	248		271		242		143
	123	125	171	100	134	108	
Fish length (mm.)	209.5±1.9 ^b		188.1±1.8 ^c		237.1±1.9 ^a		201.6±6.7 ^a
	218.7±2.7*	200.4±2.7	192.9±2.3 *	180±3	240.4±2.6*	233.1±2.9	
Fish weight (gr.)	162.8±4.5 ^b		104.8±4.3 ^c		191.9±4.6 ^a		226.4±3.6 ^a
	164±6.4	161.7±6.4	112.8±5.4*	90.5±7.1	196.1±6.1	186.65±6.8	
Condition factor	0.16±0.001 ^b		0.14±0.001 ^b		0.14±0.002 ^b		0.26±0.002 ^a
K=100*Weight/Length ³	0.15±0.002	0.17±0.002	0.15±0.001	0.15±0.002	0.14±0.001	0.14±0.002	

Table 2 Relative representation of major parasite taxa within the entire rabbitfish parasite community at each sampling site in 1995–1997 and 1998–2000 at the Red Sea sampling sites. *FF*, Fish farm; *NB*, North Beach; *OBS*, observatory. Results are presented in %. Significant differences ($P < 0.05$) are marked with superscript (a, b)

	OBS		FF		NB	
	1995–1997	1998–2000	1995–1997	1998–2000	1995–1997	1998–2000
Digenea	19.5	19.6	12.5 ^a	6.2 ^b	18.9	19.1
Nematoda	16.4	17.6	2.1	4.5	2.3	2.7
Acanthocephala	20.3	19.3	7.7	6.9	7.8	11.5
Myxosporea	22.9	19.8	48.2	47.3	39.9	34.3
Monogenea	20.9	23.7	29.5	35.1	31.1	32.4

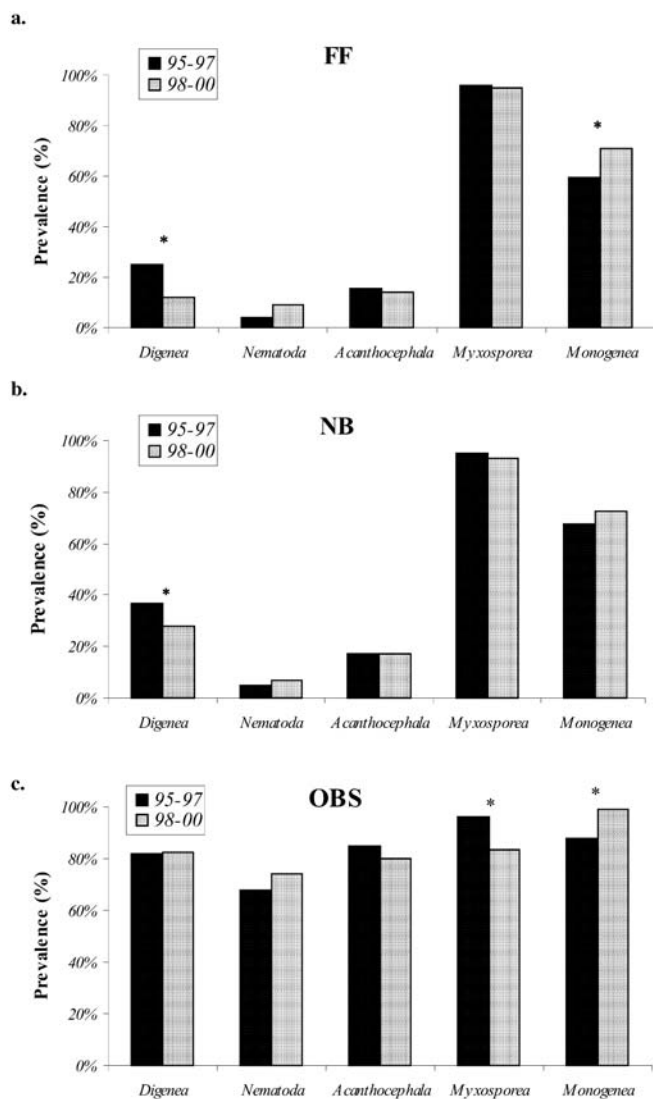


Fig. 2 Prevalence of major parasite taxa in 1995–1997 and 1998–2000 rabbitfish samples at the Red Sea sampling sites. *NB*, North Beach; *FF*, fish farm; *OBS*, observatory

Statistical analysis

Parasite data were entered into a Microsoft Excel database, which was established to include data collected from each fish caught at the three sampling sites, and were used to calculate the parasitological indices (Diamant et al. 1999). “True” species richness was extrapolated by $Y = a(1 - e^{-bx})/b$ (Walter et al 1995) separately for

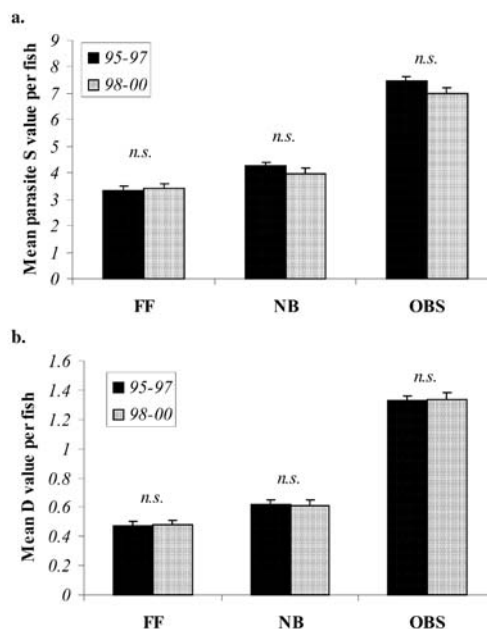


Fig. 3 Comparison of parasite species richness between 1995–1997 and 1998–2000 at the Red Sea sampling sites. *NB*, North Beach; *FF*, fish farm; *OBS*, observatory. *S*=no. of parasite species; *D*=(*S*–1)/ln(*N*) (Margalef 1951) **a** = *S* values, **b** = *D* values

monoxenous and heteroxenous species in order to detect differences in abundance trends between these two parasite groups. The calculation of “true” species richness and its confidence limits were done using SlideWrite Plus software. The effects of all parameters were tested using one-way ANOVA (after verifying a normal distribution). Differences between index values of the three sampling sites were compared using Tukey-Kramer multiple range test. Differences in prevalence and the relative representation of major parasite taxa were tested with χ^2 tests. All statistical procedures were carried out with JMP statistical software with the significance level set at $P < 0.05$.

Results

From 1995 to 2000, a total of 761 fish were sampled. The length of the fish ranged from 150 to 250 mm (all specimens outside of this range were excluded from the analyses). Although the fish sampled from 1995 to 1997 were larger than those sampled from 1998 to 2000, these differences did not exceed 2 cm at any site and they have no implication regarding the fish size/age class. Condition factor values ($K = 100 \times \text{weight}/\text{length}^3$, see Bolger and

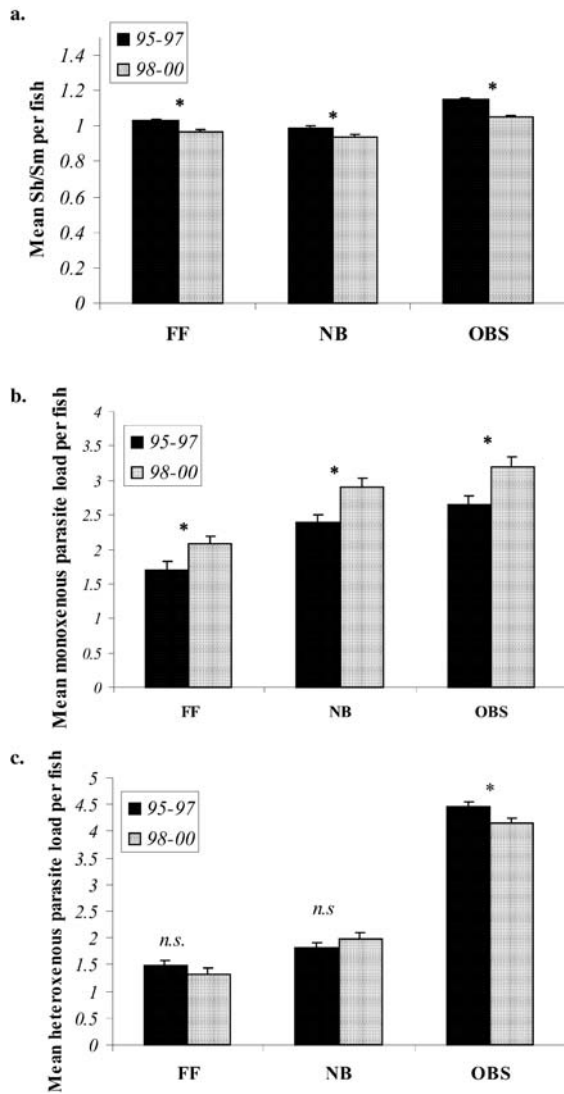


Fig. 4 Comparison of mean Sh/Sm (a), monoxenous (b) and heteroxenous (c) parasite species richness between 1995–1997 and 1998–2000 at the Red Sea sampling sites. NB, North Beach; FF, fish farm; OBS, observatory

Connolly 1989) of the fish samples from OBS from 1981 to 1985 were significantly higher than those sampled from 1995 to 2000 at all sites (Table 1). No change in the number of parasite species was noted between the 1995–1997 and 1998–2000 samples nor between the number of gut helminth species between 1981 and 1985, and between 1995 and 2000. When fish parasite communities from 1995–1997 and 1998–2000 were compared, the prevalence of infection with (heteroxenous) digeneans showed an overall decrease from 1995–1997 to 1998–2000 at FF and NB but no change at OBS. The prevalence of (monoxenous) monogeneans, on the other hand, increased significantly at FF and OBS. At the same time, a significant decrease in the prevalence of myxosporeans at OBS was also noted (Fig. 2).

The relative representation of members of the major parasite taxa, in relation to the whole parasite communi-

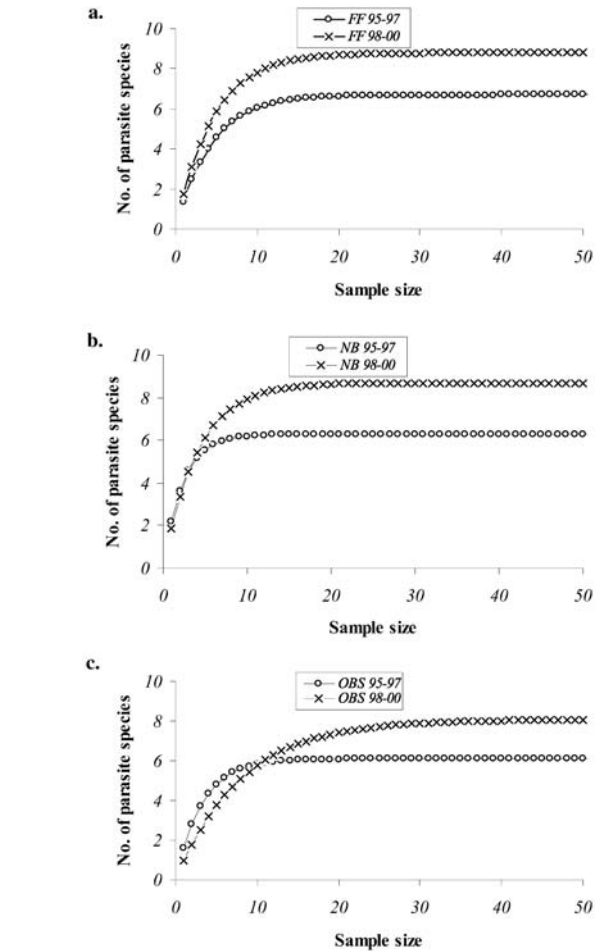


Fig. 5 Cumulative curves of monoxenous parasites “true” species richness extrapolated by $Y=a(1-e^{-bx})/b$ (Walther et al. 1995) between 1995–1997 and 1998–2000 in each of the Red Sea sampling sites. FF, fish farm (a); NB, North Beach (b); OBS, observatory (c). Significant differences in the values of variables a, b at OBS and b at NB

ties, in each sampling site is shown in Table 2. At OBS, the representation of each taxon was relatively uniform and no significant shifts occurred between the years 1995–1997 and 1998–2000. At FF and NB, monogenean and myxosporean infections were the most abundant. There was a significant decrease in the presence of digeneans at FF as well as a trend, though not significant, towards an increase in the abundance of nematode infections at FF and acanthocephalan infections at NB.

The mean values of parasite species richness in the 1998–2000 samples showed no significant changes from those of 1995–1997 at any of the three sites (Fig. 3).

The Sh/Sm ratio decreased significantly at all three sampling sites (Fig. 4a). Species richness, plotted separately for heteroxenous and for monoxenous parasites of 1995–1997 vs. 1998–2000 demonstrated a significant increase in monoxenous parasite species richness at all three sampling sites, with no change in heteroxenous parasites at FF and NB and a significant decrease at OBS (Fig. 4b, c).

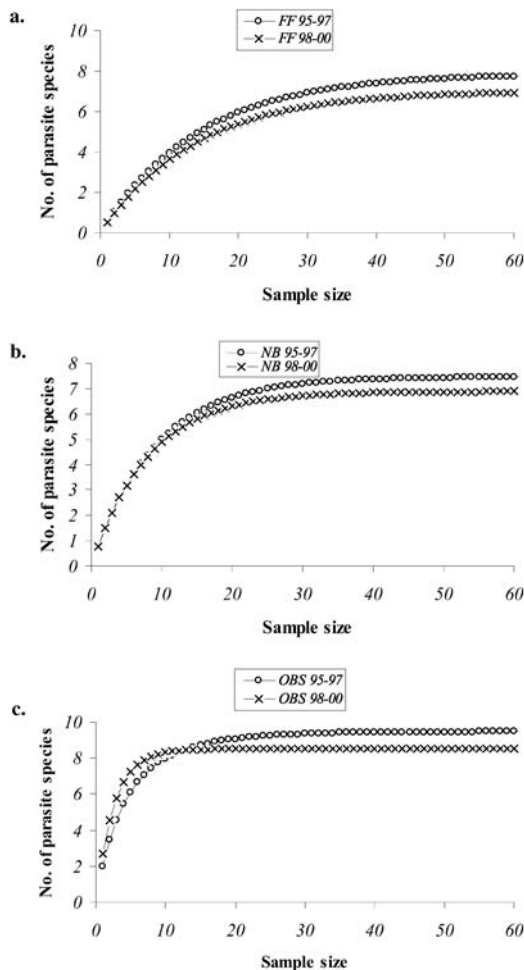


Fig. 6 Comparison of heteroxenous parasite “true” species richness extrapolated by $Y=a(1-e^{-bx})/b$ (Walther et al. 1995) between 1995–1997 and 1998–2000 in each sampling site. *FF*, fish farm (a); *NB*, North Beach (b); *OBS*, observatory (c) with no significant differences in the values of variables a, b

“True” species richness curves for all parasite species did not vary significantly from 1995–1997 to 1998–2000. Monoxenous parasite “true” species richness curves, on the other hand, increased significantly in all three sampling sites (Fig. 5). Similar curves plotted for heteroxenous parasites revealed some decline, but this trend was not statistically significant (Fig. 6).

Shannon-Weiner diversity values increased significantly at *OBS* and the Janion index of evenness increased significantly at *FF* when comparing 1995–1997 with 1998–2000, while values of the same indices from the other sampling sites showed no significant change (Fig. 7).

Fish sampled at *OBS* during 1995–2000 (1990s), when compared with those from 1981–1985 (1980s) showed a significant decline in the abundance of two gut digeneans, *Opisthogonoporoides* sp. and *Gyiliauchen* sp. (Figs. 8, 9). Their mean intensity (number of worms per infected fish) decreased significantly over this period, from 28.1 to 4.6 and 27.2 to 1.5, respectively. When the prevalence of

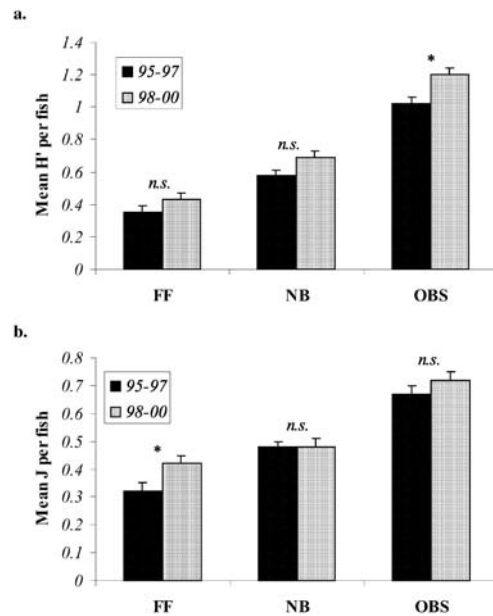


Fig. 7 Comparison of diversity index values between 1995–1997 and 1998–2000 at the Red Sea sampling sites. *NB*, North Beach; *FF*, fish farm; *OBS*, observatory. a: Diversity $H'=-\sum(P_i \ln P_i)$; b: evenness $J=H'/H_{max}$ (where $H_{max}=\ln[S]$)

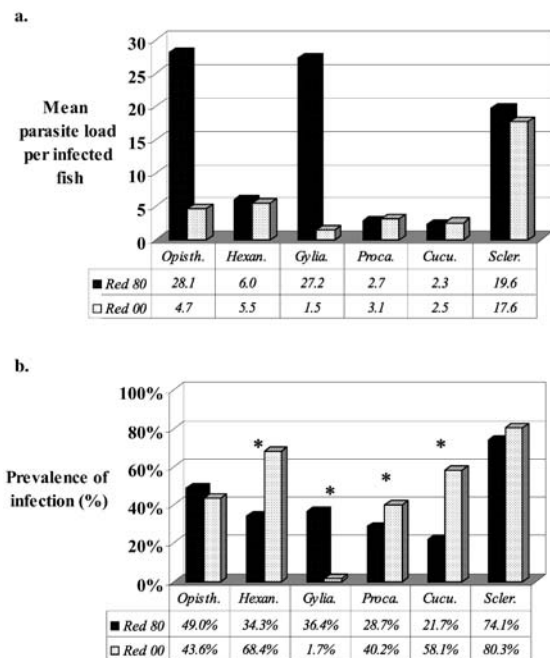


Fig. 8 Comparison of prevalence (a) and intensity (b) of gut helminth infections between the 1990s (1995–2000) and 1980s (1981–1985) at *OBS*. Abbreviations: *Opisth.*, *Opisthogonoporoides* sp.; *Hexan.*, *Hexangium sigani*; *Gylya.*, *Gyiliauchen* sp.; *Procam.*, *Procamallanus elatensis*; *Cucu.*, *Cucullanus sigani*; *Scler.*, *Sclerocollum rubrimaris*

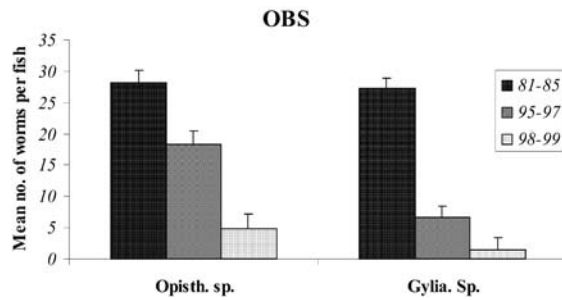


Fig. 9 Changes in the mean intensity of infection of *Opisthogonoporoides* sp. and *Gyiliauchen* sp. over time

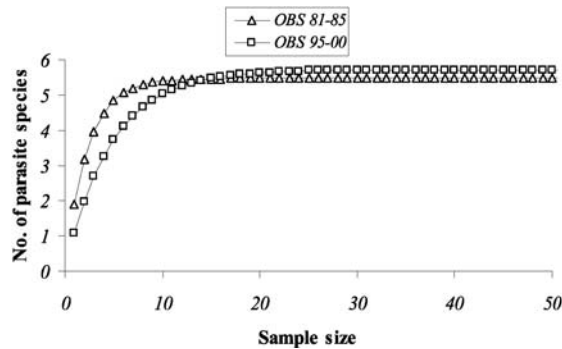


Fig. 10 Comparison of parasites “true” species richness extrapolated by $Y=[a*(1-e^{-bx})]/b$ (Walther et al. 1995) of *S. rivulatus* gut helminths between 1981–1985 and 1995–2000 at OBS (values of variables a and b are significantly different)

these two species was considered, *Opisthogonoporoides* sp. displayed a minor, albeit consistent over time, decrease (49% to 43.5%) while *Gyiliauchen* sp. showed a dramatic decrease, from 36.3% to only 1.7%.

On the other hand, the prevalence of other rabbitfish gut helminths, namely the digenean *Hexangium sigani* and the nematodes *Cucullanus sigani* and *Procamallanus elatensis*, showed a significant increase over the last 15 years (Fig. 8b).

The model demonstrated that the sampling effort required for reaching the “true” species richness of the habitat was considerably lower during the 1980s than in the recent samplings (Fig. 10).

Discussion

The Gulf of Aqaba is fringed by a highly developed, diverse coral reef, which supports a broad variety of marine life. The existence of this delicate, complex ecosystem depends on the stability of the oligotrophic conditions, which have typically prevailed in this water body (Fishelson 1995). The Gulf of Aqaba was a near-pristine area when the city of Eilat was founded in 1949. In recent years, studies have provided evidence of coastal degradation in the region (Fishelson 1995; Schuhmacher et al. 1995). There are three main sources of organic

pollution for the reported increased eutrophication in the gulf: urban and maritime sewage, mariculture cage farms and phosphate loading berths (Silverman et al. 2001). Abelson et al. (1999) demonstrated that particulate matter from point sources at Eilat’s North Beach reached the remote coral reefs to the south, both as bed-load and suspension-load particles. Such shifting of organic matter with currents may promote phytoplankton blooms and a corresponding increase in zooplankton and zoobenthos, such as copepods, polychaetes and other species, all potential intermediate hosts of fish parasites. However, there is no direct quantitative evidence to confirm the above predictions.

The most important change between 1995–1997 and 1998–2000 is probably the significant decline in the heteroxenous/monoxenous parasite species ratios. While monoxenous parasite abundance increased at all sampling stations, heteroxenous species showed a marked decline only at OBS. This trend was unmistakably demonstrated by the cumulative model that extrapolates “true” species richness of the habitat as a function of sample size (Walther et al. 1995). Our basic assumption was that in a disturbed environment heteroxenous parasites are less likely than monoxenous parasites to complete their life cycles, either due to direct adverse effects on their free living stages, or as an indirect consequence of the elimination of their intermediate hosts (Paperna 1997). Previous studies have proposed and applied Sh/S_m as a tool for monitoring the state of an ecosystem (D’Amelio and Gerasi 1997; Diamant et al. 1999). The transition with time in the relationship between heteroxenous and monoxenous parasites, with a more conspicuous increase in monoxenous species abundance than the decline in heteroxenous species, agrees with the reports of Khan and Thulin (1991), Gelnar et al. (1997) and Bagg and Valtonen (1996) which associate an increase in monoxenous species abundance, monogeneans in particular, with increased perturbation of the habitat. Chronic exposure to organic pollution and contaminated sediment has been shown also to enhance monoxenous ciliate infections, in particular trichodinids (Yeomans 1997; Broeg et al. 1999). The suggested indirect effect (physiological or immunosuppressive) on the hosts under such circumstances (Khan and Thulin 1991) still warrants experimental validation.

We found between 1995–1997 and 1998–2000 a consistent decrease in the abundance of digenetic trematodes at FF and NB as well as their relative representation in the community at FF. Similarly, the digeneans *Gyiliauchen* sp. and *Opisthogonoporoides* sp. declined significantly in abundance between the 1980s and 1990s. The abundance of other gut helminths—the digenean *H. sigani* and the nematodes *C. sigani* and *P. elatensis*—increased, while the abundance of the acanthocephalan *Sclerocollum rubrimaris* maintained steady levels. The life cycle details of all these helminths are as yet unknown, and we may consider them only in broad terms, i.e. implicating molluscs as intermediate hosts of digeneans and amphipods, copepods and possibly also

annelids for nematodes and acanthocephalans. *S. rivulatus* apparently becomes infected when ingesting these and other small invertebrates associated with macroalgae or sea-grass (Lundsberg and Lipkin 1979; Diamant 1989a).

The observed decline in the populations of the digeneans *Gyuliauchen* sp. and *Opisthognoporoides* sp. could be linked to the decline in the abundance of molluscs in the reef habitat (see Fishelson 1995). The observed increase in the digenean *H. sigani* and the two nematode species could be attributed to a possible rise in the abundance of benthic or planktonic invertebrates acting as intermediate hosts. In summary, the remarks of Kennedy (1997b) are as relevant as ever and well worth repeating here. At present, we lack knowledge on parasite life histories, on the effect of different sources of pollution on each stage of their life cycle and on the effects on their intermediate hosts. What we can do is record changes and differences in populations and communities; but we are still incapable of unequivocally relating them to any specific causal factors.

Extreme conditions may have different, sometimes even conflicting impacts on diverse taxa of parasites. Under such circumstances, overall community indices may incorporate opposite trends and possibly fail to track transitions that have actually taken place in the composition of the community. Accordingly, species richness and diversity indices of the complete parasite communities demonstrated only some differences between the 1995–1997 and 1998–2000 sampling periods. The present results are in agreement with those of Gelnar et al. (1997), who investigated the diversity of freshwater fish parasites in relation to pollution. These authors showed a succession of species with no change in diversity indices, a significant decrease in the community's species richness. Therefore, we emphasize that in order to pinpoint ecological changes, it is essential to analyze parasite assemblages not only at the community level, but also to carefully consider components of particular parasitic groups.

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