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Of reef fishes, overfishing and *in situ* observations of fish traps in St. John, U. S. Virgin Islands

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Abstract: Fishing with a variety of methods and gears, including traps, is allowed within the waters of Virgin Islands National Park (St. John, U.S. Virgin Islands). Randall's 1961 observation of the effects of overfishing in nearshore waters off St. John has been followed by three and a half decades of reports documenting the declining reef fish catch in the Virgin Islands and much of the Caribbean. To assess the state of the trap fishery in St. John waters, traps set by fishers were visually censused *in situ* in 1992, 1993 and 1994 both inside and outside park waters. Fifty-nine species of fishes representing 23 families and 1340 individuals were identified from 285 traps set in five habitat types (coral reef, octocoral hard-bottom, seagrass beds, algal plains and non-living substrate). The greatest number of observed traps were in algal plain (31%) and gorgonian habitat (27%), pointing to greater exploitation of deeper, non-coral habitats. Coral habitat accounted for the most species trapped (41), whereas the mean number of fishes per trap was highest in algal plain (5.7, se=0.6). Six species made up 51% of all fish observed in traps. The Acanthuridae was the most abundant family. Species composition and number of fishes per trap were similar inside and outside park waters. Scarids and serranids were more frequently observed in traps inside the park. Between 1992 and 1994, patterns in the data emerged: smaller numbers of fish per trap; shifts to smaller size classes; fewer serranids, lutjanids, sparids, and balistids, and all feeding guilds except herbivores per trap; more acanthurids per trap. Compared with other trap data from the Virgin Islands and the Caribbean - Florida region, the mean number of fish and biomass per St. John trap are low, serranid numbers are low, and acanthurid and herbivore numbers are high. The reef-associated fishes of St. John appear to be overexploited.

Keywords: Caribbean, herbivores, reef-associated fishes, serranids, trap fishery, Virgin Islands.

The Virgin Islands fishery dates back at least 2800 years, when pre-Taino and Taino inhabitants gathered a variety of reef fishes for consumption (K. Wild *et al.* unpublished), using many of the same techniques fishers employ today (Price 1966). The multigear, small-scale fishery harvests over 100 species of reef-associated and open water fishes, with fish traps the most commonly used gear since at least the 1930s (Fiedler and Jarvis 1932, Randall 1961). The gear is non-selective, efficient and requires little capital investment or expertise to use (Fiedler and Jarvis 1932, Idyll and Randall 1959, Swingle *et al.* 1970).

Although fishery surveys were conducted in the Caribbean and southwest north Atlantic in the late 1800s and early 1900s (Bayer 1968), the first assessment in the Virgin Islands was carried out in 1931 (Fiedler and Jarvis 1932). Since then, a considerable number of reports and publications have documented the status and changes in the St. Thomas and St. Croix fisheries (eg. Dammann 1969, Olsen and LaPlace 1978, deGraf and Moore 1987, Beets 1997) but none addressed St. John specifically until 1959, when the Marine Laboratory of the University of Miami initiated a series of studies (eg. Idyll and Randall 1959, Randall 1961, 1963, 1965, 1967, 1968) for the National Park Service (NPS).

Historically, Virgin Islands fish traps contained primarily serranids and lutjanids (deGraf & Moore 1987), but by the mid 1980s, herbivorous scarids and acanthurids dominated catch (de Graf & Moore 1987). Beginning in the late 1960s and through the 1980s, Virgin Islands Department of Fish & Wildlife reports documented disturbing signs: decreases in the average size of fishes trapped; a high proportion of juveniles in the catch; increased trapping effort without a proportionate increase in landings (Dammann 1969); the near extirpation of the most valuable commercial species, *Epinephelus striatus* (Olsen and LaPlace 1978). These reports of decreasing finfish stocks throughout the Virgin Islands and concern over the adequacy of the protection provided by territorial, federal and Virgin Islands National Park (VINP) regulations in St. John waters, prompted the initiation of a series of fisheries projects in 1992. The purpose of the studies was to assess the status of and trends in the St. John reef fish assemblages, fishery resources and the effects of the trap fishery on the reef fishes both inside and outside VINP park waters. Were the reef fish assemblages inside the park (which is also an International Biosphere Reserve) being altered by fishing? As a corollary, were park fishing regulations and their enforcement protecting the resources that the park is mandated to "protect for the enjoyment of future generations"? The primary objectives of this study were to: quantify the species, number and size (biomass) of all fishes observed in traps in St. John waters; record locations and habitats where fishes were trapped; document species composition in traps over the study period; and determine any differences in species composition and biomass between park and non-park waters.

Study site: St. John, U.S. Virgin Islands lies near the northeastern edge of the Caribbean Plate and the eastern extremity of the Greater Antilles. The prevailing ocean currents are driven by the trade winds and flow to the west-northwest, with the Antillean current to the north of the island and the Caribbean current on the south. Seawater temperatures vary annually from 24.4 to 30.9° C [NPS/Biological Resources Division, USGS, Department of Interior (BRD) unpublished data]. Water quality closely approaches Sargasso Sea values for most parameters [NPS/BRD unpublished data], with the exception of a few bays with sluggish circulation, altered shorelines (removal of mangroves and importation of sand) and developed watersheds. Tropical storms and hurricanes periodically damage the fringing coral reefs (Rogers *et al.* 1991, Rogers 1993) and seagrass beds (Muehlstein 1997), and sometimes temporarily alter the composition of reef fish assemblages (Beets, unpublished). St. John rises from a shelf with average depths of 20-30 m a few hundred meters from shore. Approximately 11 km to the south, the shelf drops abruptly from 45 m to the abyssal depths of the Anegada Trough (National Ocean Survey Chart No. 25641) and extends nearly 25 km to the north before sloping into the Puerto Rican Trench.

St. John is the site of Virgin Islands National Park (VINP) and International Biosphere Reserve. Established in 1956, VINP includes 56% (2816 ha or 25.9 km^2) of the island and 2 287 ha of the adjacent marine waters (added in 1962). Fishing is allowed in park waters; specifically, the park's enabling legislation allows for the "customary uses of or access" to park waters for fishing, including the use of traps of "conventional" Virgin Islands design. VINP regulations prohibit spearfishing, commercial fishing and the use of cast nets with mesh >2.5 cm (1.0 in) or over 6.1 m (20 ft) in length within park waters (Code of Federal Regulations, Title 36, Section 7.74).

Territorial fishing regulations also apply in park waters (eg. fish trap permits, operational biodegradable panel, minimum mesh size, buoy colors).

Five habitats were identified where fishers set traps around St. John: scleractinian coral reef, gorgonian hard-bottom, algal plain, seagrass beds and nonliving substrate. Coral reefs occur primarily nearshore as fringing or patch reefs with scleractinian cover less than 30% (Rogers *et al* 1997). At least 19 species of gorgonians dominate the gorgonian hard-bottom where coral heads and sponges also occur (Boulon 1986a and b, Gladfelter, unpublished). The rugosity of the hard substrate is low. However, some gorgonian species provide vertical relief of up to a meter or more. The seagrass beds are small, not well developed and have been subject to cumulative damage from hurricanes and boat anchors in the last decade, especially nearshore. *Thalassia testudinum, Syringodium filiforme,* and *Halodule wrightii* occur. The algal plains are large areas of very low relief, deeper (over 20 m) than the other major habitats and dominated by one or more species of alga(e), often in association with sponges and carbonate nodules from coralline algae (Olsen *et al.* 1981, unpublished). Fifty-two species of algae and 25 species of sponges were reported by Olsen *et al.* (unpublished). Non-living substrate habitats include areas of sand, rubble and/or pavement. Spatial relief is very low.

MATERIALS AND METHODS

This is the first study in which *in situ* observations have been made to visually census the contents of fisher-set traps; the distinct advantages of this method are knowing exactly where the fishes are being harvested, from what habitat, with what type of traps and the species composition and numbers.

Visual censuses: For seven consecutive days in November 1992, July 1993, and July 1994, all buoyed fish traps observed within 2 km of St. John were visually censused by divers using SCUBA. The extent of the area censused to the south was dependent on weather (visibility of buoys) and depth (for SCUBA) at which traps were set. The following information was recorded: location, depth, substrate/habitat, trap type (Antillean chevron, rectangular or square), trap materials, mesh type (hexagonal, square, rectangular), minimum mesh diameter (cm), number of traps connected in a "string", number of buoys, buoy colors, presence/absence of a functional biodegradable panel, presence of anodes. For each trap, fishes were enumerated by species and fork lengths (mm) were estimated for each individual. Trap dimensions (cm) and types of bait, if any, were recorded in July 1993 and 1994.

"Soak time" was unknown and only observed buoyed traps were sampled. Unbuoyed traps were not censused and represented an unknown percentage of all traps. In the Virgin Islands the average soak time is reported to be 3-4 and 7-8 days (Dammann 1969, Olsen *et al.* unpublished, Wolff *et al.* in press). Based on our knowledge of the trapping schedules of many of the fishers, we believe the censused traps had been set for 1 to 8 days. On several occasions, traps were seen being hauled within an hour of being censused.

Because data were not recorded when the traps were hauled, exact catch by the fisher is unknown. Nonetheless, we believe the *in situ* census data very closely approximate the species, numbers and sizes of fishes extracted.

Data analysis: Species were assigned to feeding groups based on Randall (1963, 1967, 1968) and the authors' observations. The groups are: herbivores; omnivores; sessile invertebrate feeders; mobile invertebrate feeders; mobile invertebrate feeders/piscivores; piscivores. Species were considered mobile invertebrate feeders/piscivores if both prey groups contributed substantially to the diet.

All data were entered and verified in QuattroPro and Excel and analyzed using Paradox, Systat6 and SigmaStat. *Calamus bajonado, C. calamus, C. penna* and *C. pennat-*

TABLE 1

Summary of visually censused fish traps

	Bouyed traps	Illegal traps (%)	No bio-panel (%)	Mesh <1.25 inch (%)	Live substrate (%)
In park waters					
Nov 1992	25	64	64	8	84
July 1993	44	23	23	0	84
July 1994	58	76	59	0	90
Out of park waters					
Nov 1992	27	74	74	19	78
July 1993	104	45	45	3	75
July 1994	37	78	70	6	81
St. John Totals					
Nov 1992	52	69	69	13	81
July 1993	148	39	39	2	78
July 1994	95	77	63	2	86
1992-1994	295	56	52	4	81

Percent of traps are shown for each of the following categories: "Illegal traps" - all traps which do not meet territorial or federal requirements (eg. no biodegradable panel, not buoyed, mesh size too small); "No bio-Panel" - traps with no functioning biodegradable panel (required); "Mesh < 1.2 in" - minimum diameter mesh less than 1.25 in (minimum mesh diameter); "Live substrate" - traps set on living substrate (coral, gorgonian, seagrass, etc).

ula were pooled into *Calamus spp.* due to difficulties in differentiating those species in crowded traps. Because all parameters were not consistently recorded for every trap, sample size of analyzed categories varied. Complete data exist for 285 of the 295 traps censused.

Relative abundance of taxa is a useful measure which can mask or enhance changes in populations/assemblages; therefore, we present both number of individuals per trap and relative abundance data.

A Biomass Index was calculated for each species (Bohnsack and Harper 1988 and unpublished data).

{log(gm)= log(a) + [b*log(mm)]}, where a and b are constants for a species

When the length/weight relationship for a species was not known, the constants for a morphologically similar species in the same genus were used (Bohnsack and Harper 1988). To assign a length to each of *n* fish in a size class delineated by a maximum and minimum length, one individual was assumed to be the minimum length, one the maximum length and each of the remaining n-2 individuals were assumed to be (min+max/2) (Bohnsack and Harper 1988). The biomass index for each individual was then calculated based on the individual's estimated length.

Although the observational nature of the study, short time period and large number of traps with no fish prevented us from analyzing for statistically significant temporal changes in number of fishes per trap, relative abundance and species composition, the data are presented.

Frequency data for families and feeding guilds in relation to location inside or outside park waters were tested for independence, based on an asymptotic model. The null hypothesis:

H_0: no difference in the frequency of a family or feeding guild in traps inside and outside park waters.

Pearson's Chi Square statistics and associated p values are reported. The large variance in the number of fish per trap and abundance data (approximately five times the mean) and high proportion (25%) of empty traps precluded further statistical analysis. A much larger sample size would be needed to satisfy the assumptions of the tests.

TABLE 2

Trap summary by habitat type in which they were set (standard errors in parentheses).

	All habitats	Algal plains	Coral reef	Gorgonian	Seagrass	Non- living
Number of species	59	39	41	36	22	28
Number of fish	1340	517	224	321	74	204
Percent of total	100	38.6	16.7	24	5.5	15.2
Number of traps	295*	91	47	79*	22	56
Mean number fish/trap	4.7 (0.3)	5.7 (0.6)	4.8 (1.0)	4.7 (1.0)	3.4 (1.0)	3.6 (0.7)
Mean depth of traps (m)		23.4	17.3	17.4	21.5	20.8

* Fish in 10 traps were not censused.

RESULTS

Traps: A total of 295 traps were censused (Table 1). Traps were Antillean chevron (arrowhead) (50%) or rectangular (49%). Most were reinforced with a steel rebar frame (58%) and covered with 3.1-5.0 cm mesh (94%). Hexagonal mesh was most commonly used (61%). One third were set as single traps. The remaining were set in strings of 2 to at least 23 traps connected by polypropylene line (mean 4.8 traps/string). Fifty-two percent of all traps had no biodegradable panel or one that was not functional (wired closed). Traps were set with and without bait. Types of bait included bread, cactus, lobster exoskeletons/carapace, bait fish in a plastic bag, and whelk shells.

Censused traps were set between 3 - 38 m (mean 19.6m), with 68% deeper than 15.1 m (50 ft) and containing 1,045 (80%) of the fish. Eighty-one percent of traps were set on live substrate. Over half of the traps were set in algal plain or gorgonian hard-bottom habitats and contained 63% of all censused fish (Table 2).

Fish Abundance: A total of 1 340 fish (59 species and 23 families) were identified and their lengths estimated in 285 visually censused traps of which 211 (74%; range 68-80% per census period) contained fish (Table 3). Traps averaged 4.7 individuals/trap (se=0.3, range 0-31) (Table 4). The effect of mesh size was not tested for due to the low sample size of traps with mesh <31 or >50 mm and other confounding factors (habitat, depth, size of funnel openings, trap design and materials, etc.). More fish were observed in traps constructed of both wood and steel (mean of 7.0 fish/trap, se=0.9) than in those of steel (4.6 fish/trap, se=0.5) or wooden frames (2.9 fish/trap, se=0.4). A total of 779 fish were in 147 Antillean chevron traps and 541 were in 145 rectangular traps. Algal plain habitat produced the greatest number of fish per trap (mean of 5.7, se=0.6) and seagrass habitat caught the least (mean of 3.4, se=1.0), although this may reflect the low number of traps (22) in seagrass (Table 2).

Length/Biomass: The mean fork length of pooled individuals was 250 mm (se=20) . Mean biomass per trap was 2.0 kg (se=0.17) and somewhat greater inside the park (Table 4).

Species: The mean number of species per trap with fish was 3.1 (se=0.1; range 1 to 13). Six of the 59 species trapped accounted for 51.4% of individuals. Only nine species were trapped every census period, both inside and outside park waters: *Haemulon plumieri, H. sciurus, Calamus spp., Pomacanthus arcuatus, Sparisoma aurofrenatum, Acanthurus bahianus, A. coeruleus, Lactophrys quadricornis*, and *L. triqueter*. The three most abundant species were also the most frequent occupants of traps: *A. coeruleus*, pooled *Calamus spp.* and *P. arcuatus* (Table 5).

Families: Acanthurids and ostraciids were the most frequently trapped families (Fig. 1). The acanthuridae was numerically the most abundant family in traps; ostraciids were second

TABLE 3

Summary of species identified in visual censusing of St. John, USVI traps: number of individuals; relative abundance; mean number of fish per trap; frequency in traps; mean, minimum, maximum length; biomass index (percent of biomass contributed by a species) and the trophic guild.

Fish species	No. indiv.	% abund.	Fish/ trap	Freq.	Mean length mm	Std. error	Min. length mm	Max. length mm	Biomass index (%)	Mean Ind. bio.	Std. error	Trophic guild
Ginglymostoma cirratum	4	0.3	0.01	0.003	813	278	550	1200	3.1	4.416	4.399	P
Holocentrus adscensionis	39	2.9	0.14	0.058	244	48	160	350	2.3	0.332	0.178	MI
Scorpaena sp. *	1	0.1	0.00	0.003	200	—	200	200	0.0	0.132	—	MI/P
Cephalopholis cruentatus	1	0.1	0.00	0.003	220	—	220	220	0.0	0.169	—	MI/P
C. fulvus	7	0.5	0.02	0.017	261	23	230	300	0.4	0.280	0.075	MI/P
Epinephelus adscensionis	1	0.1	0.00	0.003	300	—	300	300	0.0	0.438	—	MI/P
E. guttatus	39	2.9	0.14	0.081	296	49	190	400	3.1	0.460	0.228	MI/P
E. morio	2	0.1	0.00	0.007	500	71	450	550	0.6	1.812	0.757	MI/P
E. striatus	6	0.4	0.02	0.020	488	115	350	650	2.3	2.155	1.515	MI/P
Mycteroperca tigris	1	0.1	0.00	0.003	500	—	500	500	0.3	1.830	—	P
Echeneis sp. *	1	0.1	0.00	0.003	830	—	830	830	0.3	1.437	—	O
Caranx crysos	1	0.1	0.00	0.003	250	—	250	250	0.0	0.302	—	P
C. ruber	17	1.3	0.06	0.027	330	43	250	400	1.9	0.642	0.262	P
Lutjanus analis	9	0.7	0.03	0.017	437	186	100	730	3.3	2.064	2.043	MI/P
L. apodus *	4	0.3	0.01	0.014	295	71	250	400	0.4	0.525	0.414	MI/P
L. griseus	5	0.4	0.02	0.014	360	65	300	450	0.7	0.757	0.394	MI/P
L. jocu	1	0.1	0.00	0.003	650	—	650	650	0.8	4.670	—	MI/P
L. synagris	31	2.3	0.11	0.068	293	56	190	360	2.4	0.432	0.201	MI/P
Ocyurus chrysurus	60	4.5	0.21	0.054	282	89	150	500	4.5	0.437	0.358	MI/P
Gerres cinereus	2	0.1	0.00	0.003	210	0	210	210	0.0	0.220	0.000	MI
Haemulon aurolineatum	3	0.2	0.01	0.007	167	29	150	200	0.0	0.089	0.052	MI
H. flavolineatum	4	0.3	0.01	0.007	230	26	200	260	0.2	0.269	0.093	MI
H. parra	2	0.1	0.00	0.003	325	35	300	350	0.2	0.678	0.217	MI
H. plumieri	60	4.5	0.21	0.105	298	55	190	450	6.5	0.619	0.352	MI
H. sciurus	16	1.2	0.06	0.044	299	46	230	380	1.6	0.555	0.251	MI
Calamus spp.	157	11.7	0.55	0.190	278	55	180	500	13.6	0.493	0.319	MI
Equetus punctatus *	2	0.1	0.00	0.007	310	13	220	400	0.2	0.673	0.707	MI
Mulloidichthys martinicus	4	0.3	0.01	0.010	338	25	300	350	0.6	0.855	0.207	MI
Pseudupeneus maculatus	2	0.1	0.00	0.003	275	35	250	300	0.1	0.370	0.141	MI
Chaetodon capistratus *	12	0.9	0.04	0.031	112	21	80	120	0.1	0.051	0.026	SI
C. sedentarius *	8	0.6	0.03	0.020	118	17	100	140	0.0	0.052	0.023	SI
C. striatus	20	1.5	0.07	0.037	119	17	70	150	0.2	0.055	0.023	SI
Holacanthus ciliaris	20	1.5	0.07	0.044	218	46	150	300	1.0	0.287	0.164	SI
H. tricolor	11	0.8	0.04	0.027	179	38	100	250	0.3	0.180	0.103	SI
Pomacanthus arcuatus	126	9.4	0.44	0.197	240	68	80	450	11.8	0.534	0.456	SI
P. paru	2	0.1	0.00	0.007	225	64	180	270	0.1	0.389	0.308	SI
Abudefduf saxatilis *	1	0.1	0.00	0.003	100	—	100	100	0.0	0.032	—	O

Halichoeres radiatus	1	0.1	0.00	0.003	400	—	400	400	0.2	0.963	—	MI
Scarus croicensis	20	1.5	0.07	0.007	182	42	150	250	0.4	0.121	0.083	H
S. taeniopterus	13	1.0	0.05	0.031	288	75	150	400	0.8	0.344	0.201	H
S. vetula	1	0.1	0.00	0.003	280	—	280	280	0.0	0.395	—	H
Sparisoma aurofrenatum	37	2.8	0.13	0.061	233	50	140	400	1.8	0.278	0.297	H
S. chrysopterum	18	1.3	0.06	0.020	278	59	200	380	1.4	0.434	0.292	H
S. rubripinne	6	0.4	0.02	0.010	318	23	280	350	0.7	0.638	0.137	H
S. viride	8	0.6	0.03	0.020	293	48	220	350	0.7	0.513	0.229	H
Acanthurus bahianus	47	3.5	0.16	0.071	205	39	140	290	1.7	0.209	0.114	H
A. chirurgus	20	1.5	0.07	0.044	242	44	140	340	1.2	0.354	0.217	H
A. coeruleus	206	15.4	0.72	0.210	200	44	100	500	8.3	0.229	0.212	H
Bothus lunatus	1	0.1	0.00	0.003	250	—	250	250	0.0	0.282	—	P
Aluterus scriptus *	6	0.4	0.02	0.014	408	20	400	450	0.7	0.688	0.644	O
Balistes vetula	59	4.4	0.21	0.105	298	72	150	420	8.3	0.802	0.514	MI
Cantherhines macrocerus	9	0.7	0.03	0.020	314	100	160	450	1.0	0.633	0.470	MI
C. pullus	5	0.4	0.02	0.014	184	9	180	200	0.1	0.120	0.016	O
Lactophrys bicaudalis	25	1.9	0.09	0.061	232	69	150	400	1.6	0.360	0.239	MI
L. polygonia	42	3.1	0.15	0.075	248	43	150	350	2.0	0.268	0.164	MI
L. quadricornis	79	5.9	0.28	0.132	245	50	150	350	3.6	0.259	0.113	MI
L. triqueter	50	3.7	0.18	0.112	186	38	120	270	1.9	0.222	0.101	MI
Chilomycterus antennatus *	3	0.2	0.01	0.010	167	47	130	220	0.0	0.184	0.162	MI
Diodon holocanthus *	2	0.1	0.00	0.003	290	57	250	330	0.2	0.529	0.240	MI

(H=herbivore; MI=mobile invertebrate feeder; MI/P=mobile invertebrate feeder/piscivore, O=omnivore, P = piscivore and S = sessile invertebrate feeder).
* species not eaten.

TABLE 4

Summarized data from visually censused traps by year and location inside or outside park waters.

	Inside park	Outside park	1992	1993	1994
Number of species	55	43	45	46	38
Number of fish	692	648	315	658	367
Number of traps	127	168*	52	148*	95
Mean number fish/trap	5.4 (0.6)	4.1 (0.4)	6.1 (1.1)	4.8 (0.5)	3.9 (0.5)
Total kg of fish	296.4	273.9	144.6	298.6	127.5
Mean kg/trap	2.3 (0.3)	1.7 (0.2)	2.8 (0.4)	2.2 (0.3)	1.3 (0.2)
Mean kg/fish	0.428 (0.02)	0.423 (0.02)	0.459 (0.02)	0.453 (0.02)	0.347 (0.03)
Mean length (mm)	250 (4)	251 (3.1)	262 (5)	254 (4)	233 (4)

Standard error of the mean in parentheses. * indicates contents of 10 traps were not recorded.

and pomacanthids third, overall and in the park (Fig. 2a). Scarids (Chi Sq= 10.2, p=0.05) and serranids (Chi Sq= 9.3, p= 0.05) occurred significantly more frequently in traps inside the park than outside (Fig. 2b). All other families occurred equally in traps inside and outside the park.

Feeding guilds: The mobile invertebrate feeder guild exceeded other feeding guilds in frequency trapped (Fig. 3a) and relative abundance (Fig. 3b). Piscivorous fish numbers remained very low (2%). All species feeding on fishes (both piscivores and mobile invertebrate feeders/piscivores) accounted for 14% of indi-

TABLE 5

Relative abundance by year and location inside/outside park waters of the nine most abundant species in visually censused traps in St. John waters.

Species	Pooled (%)	1992 (%)	1993 (%)	1994 (%)	In park (%)	Outside (%)
Acanthurus coeruleus	15.4 (1)	6.0	11.3	30.8	16.9	13.8
Calamus spp.	11.7 (2)	9.2	15.0	8.2	6.7	17.2
Pomacanthus arcuatus	9.4 (3)	9.8	11.6	5.2	10.4	8.3
Lactophrys quadricornis	5.9 (4)	6.7	6.8	3.5	7.2	4.5
Haemulon plumieri	4.5 (5)	5.7	5.2	2.2	4.6	4.3
Ocyurus chrysurus	4.5 (6)	6.3	5.9	4.6	5.9	2.9
Balistes vetula	4.4 (7)	3.2	5.8	3.0	1.9	7.1*
Lactophrys triqueter	3.7 (8)	3.8	4.3	2.7	2.3	5.3
Acanthurus bahianus	3.5 (9)	2.2	4.9	2.2	3.9	3.1

TABLE 6

The number of individual fishes (and biomass) observed in fish traps in St. John waters by year and location inside or outside park waters extrapolated to the minimum number of fishes extracted annually.

	1992	1993	1994
Total number of individuals/week	315	658	367
- in park	189	199	304
Total number of individuals/year	16 380	34 216	19 084
- in park	9 829	10 348	15 808
Total estimated kg/week	145	298	128
- in park	91	101	105
Total estimated kg/year	7 519	15 506	6 630
- in park	4 711	5 273	5 450

viduals. Herbivorous fishes (Chi Sq=9.3, p=0.05) and mobile invertebrate feeders/piscivores (Chi Sq=5.0, p=0.05) occurred more frequently in traps inside the park (Fig. 3a).

Extracted individuals and biomass: During the three years of this study, an estimated 36 000 fish weighing 15 400 kg were extracted from the waters of Virgin Islands National Park by fish trap, while a minimum of 69 700 individuals weighing 29 500 kg were taken from St. John waters (Table 6). Annual extraction estimates were calculated by multiplying the total number of fishes (and estimated biomasses) censused during each of the 7 day census periods by 52 weeks per year. The annual estimates were then summed. These estimates are very conservative as they do not include traps set without marker buoys ("blind"), uncensused buoyed trap, or fishes taken by "ghost traps" or other gears (hook and line, nets, spearguns).

Temporal patterns: Although changes in the relative abundance and number of fish per trap by year primarily reflect temporal variation, annual data are presented to document the variation, whatever the underlying causes.

From 1992 through 1994, several patterns emerged from the trap data:

- mean number of fish per trap fell from 6.1 (se=1.1, n_{trap}=52) in 1992 to 3.9 (se=0.5, n_{trap}=95) in 1994 (Fig. 4).
- mean biomass per trap dropped from 2.8 kg (se=0.4) in 1992 to 1.3 kg (se=0.2) in 1994 (Fig. 4).
- mean fork length changed from 262 mm (se=5, n_{fish}=315) in 1992 to 233 mm (se=4, n_{fish}=367) in 1994 (Table 4).
- size frequency distributions shifted to smaller size classes for pooled data (Fig. 5) and *Epinephelus guttatus* (Fig. 6)
- *Acanthurus coeruleus* was five times as abundant in 1994 (Table 5).
- in 1994, traps contained a greater number of

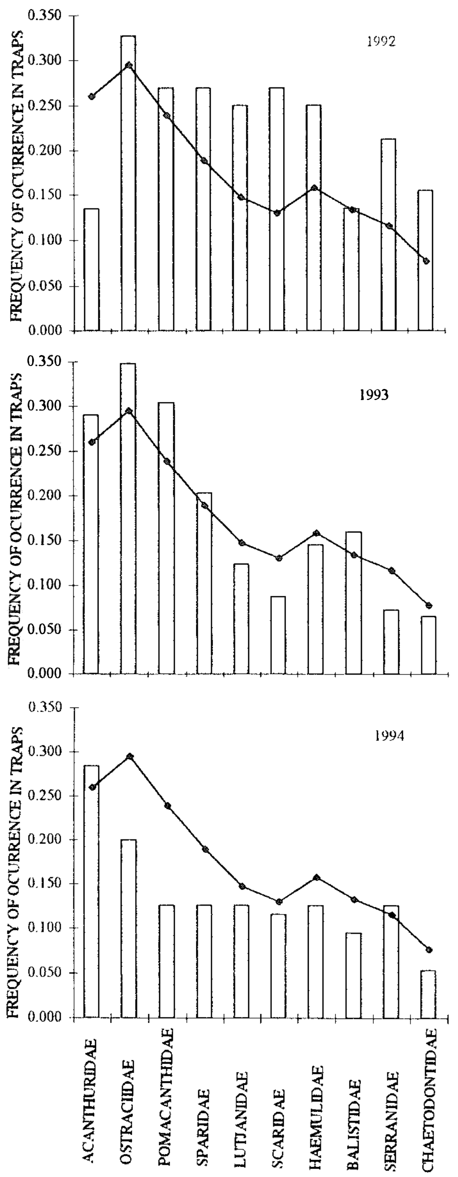

Fig. 1 Frequency of occurrence of families in traps by year (bars) compared to all years pooled (line).

acanthurids and fewer haemulids, lutjanids, ostraciids, scarids, pomacanthids and balistids. The number of acanthurids per trap rose by nearly a factor of three (Fig. 7). relative abundance of herbivores doubled between 1992 and 1994.

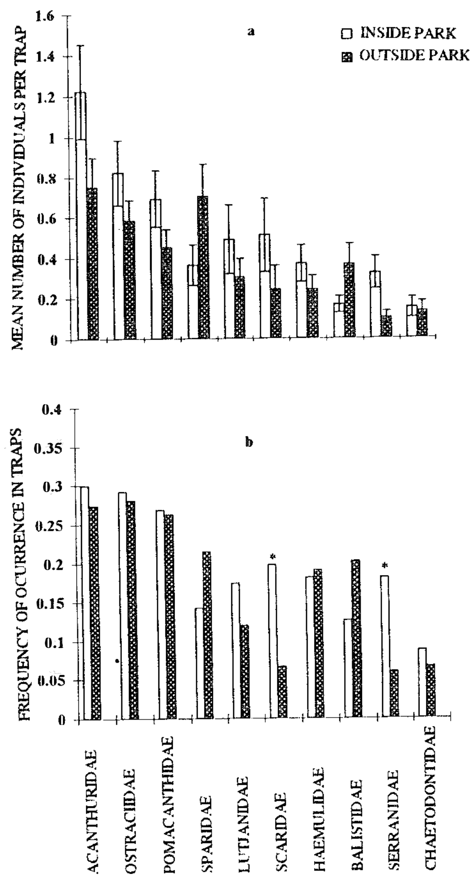

Fig. 2a Mean number of individuals per trap by family, inside and outside the park. Error bars represent standard error of the mean. 2b. Frequency of occurrence by family, inside and outside VINP waters. Asterisks indicate families which occurred in a significantly greater proportion of traps ($p<0.05$) inside or outside park waters.

DISCUSSION

In 1961, Randall observed that fishing around St. John was primarily by trap and concentrated on the narrow fringing reef in the nearshore waters. He noted that the effects of overfishing were already evident, particularly in the low numbers of large serranids and lutjanids (Randall 1963). Of the 295 traps visually censused in 1992-1994, 68% were set below 15.1 m (50 ft) but caught 80% of all fishes; only 16.5% were set in coral reef habitat and accounted for a proportionate total of individuals. These data show an apparent shift to deep-

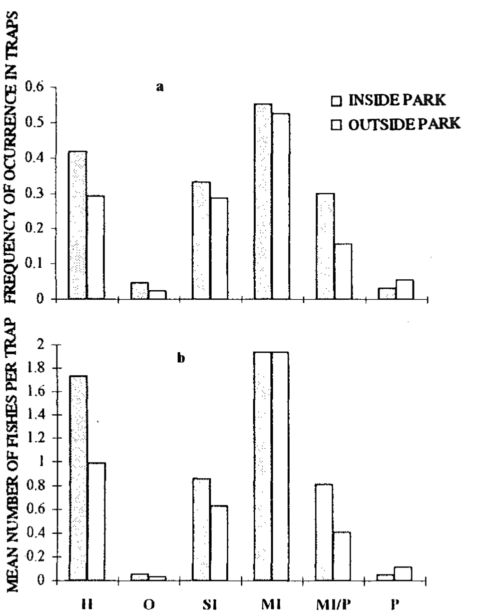

Fig. 3a Frequency of occurrence of feeding guilds in traps inside and outside the park. 3b. number of fish per trap by feeding guild by location inside and outside park waters.

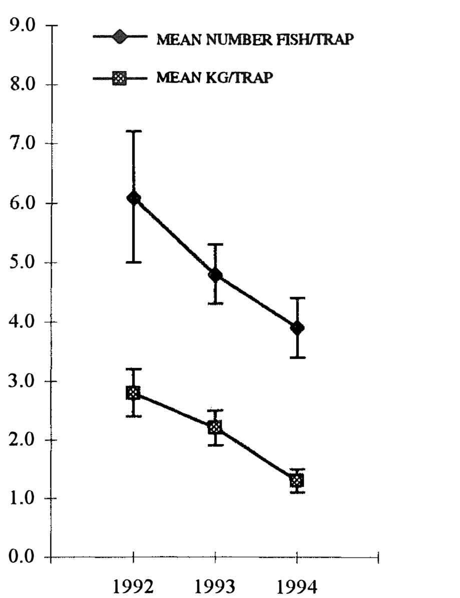

Fig.4 Mean number of individuals and biomass per censused trap by year. Error bars indicate standard error of the mean.

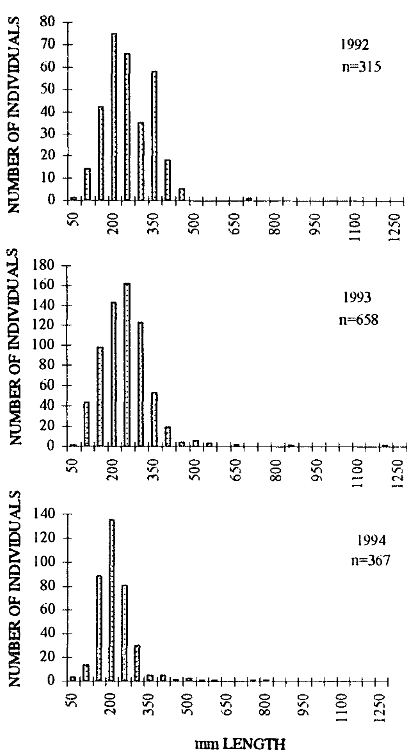

Fig. 5 Length (mm) frequency graphs of all fish censused by year.

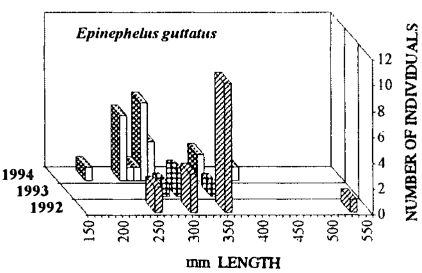

Fig. 6 Size frequency distribution of *Epinephelus guttatus* in 1992 (n=16, median 350 mm), 1993 (n=4, median 300 mm) and 1994 (n=20, median 270 mm).

er trap sets and increased exploitation of algal plain and gorgonian-dominated habitats since Randall's observation in 1961 (Randall 1963). This move to deeper and/or offshore areas may be the result of a combination of factors: overfishing in the nearshore, shallow reef habitat; an increase in the number of traps to offset a decreasing catch; the replacement of traditional

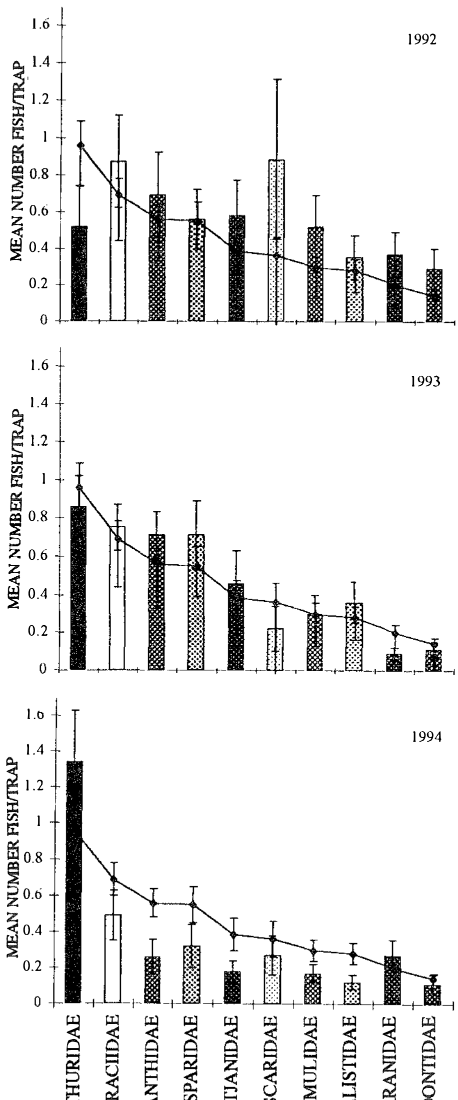

Fig. 7 Mean number of individuals per trap by family are shown in columns by year; pooled 1992-1994 mean individuals per trap by family are shown for comparison, as points connected by a line. Error bars represent standard error of the mean.

propulsion and fishing gear with more modern materials and technology (eg. engines, electric winches); higher market value increasing the incentive to invest more time and capital in fishing; attempts to decrease the allegedly high incidence of theft of traps; the greater abundance of targeted species in deeper, previously less exploited habitats; or higher yield in deeper areas due to lower habitat complexity. Numerous publications have documented declines in the abundance and size of a number of species in Caribbean reef fisheries (Olsen and LaPlace 1978, Bohnsack *et al.* 1986, Munro 1983, Appeldoorn and Lindeman 1985, Koslow *et al.* 1988, Recksiek *et al.* 1991, Appeldoorn *et al.* 1992, Beets and Friedlander 1992, Butler *et a.l* 1993, Beets *et al.* 1994, Hughes 1994, Koslow *et al.* 1994, Sadovy 1994, Rakitin and Kramer 1996, Beets 1997). Data from this and previous studies corroborate the growing evidence that many reef-associated species of finfish in the Caribbean are decreasing in abundance and size.

Comparisons of our data from visually censused traps with historical St. John and Virgin Islands data show the following changes in the St. John resource:

1. fewer fish per trap.
2. lower biomass per trap.
3. shift to smaller size classes.

4a. fewer individuals per trap and lower relative abundance of serranids, lutjanids, sparids, balistids, and all feeding guilds except herbivores.

4b. greater number of acanthurids and herbivores per trap.

5. exploitation of deeper non-coral habitats, not traditionally exploited.

These are the patterns of an overfished resource (Russ 1985, 1991, Koslow *et al.* 1988, Russ and Alcala 1989, G. Sedberry and J. McGovern, unpublished, Jennings and Polunin 1996 and references).

Changes in Virgin Islands trap contents

Changes in length and size: Mean length of fishes in traps can be affected by changes in species composition and other factors including intensive fishing. Average length (or weight) can be an indicator of the state of and/or changes in the fishery. The change in mean length of observed fish from 262 mm in 1992 to 233 mm in 1994 is reflected by a shift in the

TABLE 7

Comparison of St. John and St. Thomas trap catch in number of individuals and biomass

Year	St. John mean kg/trap	St. John mean fish/trap	St. Thomas mean kg/trap	St. Thomas mean fish/trap	St. Thomas mean kg/yr/trap	Reference
1917					1 000	Census of the VI 1917
1967	4.6		4.0			Swingle *et al.* 1970
1968					980	Dammann 1969
1970		20-25 1-11				Munro 1983
mid 1970s				6.6		Boulon 1986
1975-76			2.3			Olsen & McCrain unpublished
1978-79					19	Mudre, unpublished
1988-90			1.6			Beets, unpublished
1988-92		5.2-5.7				Beets, unpublished
1993	6.8*					Beets, unpublished
1992-94	2.0	4.7				This study

* Catch in experimental trap sets off St. John in 1993 (Beets, unpublished) appears high because traps without fish were not included in the data and a small mesh diameter (1 in) was used, retaining smaller and therefore greater numbers of fish.

length-frequency histograms to smaller sizes with time. Although the number of smaller fishes reaching exploitable size each year appears fairly steady, fewer and fewer fishes are over 250 mm in length. The extraction of individuals from all exploitable size classes over a number of years could shift the length-frequency curve to smaller individuals, yet retain similar catches of individuals in the first exploitable size classes. A relatively large pool of smaller individuals growing to a size retained by traps could replenish the smallest exploitable size classes. Also, natural fluctuations in species composition and abundance could account for the observed shift.

The observed changes in maximum and mean length of *E. guttatus* and the shift to smaller sizes could be due to removal of larger fishes through trapping, natural mortality, or variations in recruitment. However, the shift could be an artifact of the small sample size (n=39).

Changes in the Virgin Islands catch size and biomass over the last two decades are evident from a comparison of data from previous Virgin Islands fishery studies and this observational study (Table 7).

The catch in experimental trap sets off St. John in 1993 (J. Beets unpublished) appears high because a smaller mesh diameter (2.5 cm) was used, retaining smaller and therefore greater numbers of fishes, and traps without fish were not included in the analysis. Declines in average annual catch also appear in Virgin Islands port sampling data. Port sampling between August 1986 and July 1987 documented a 20% decrease in the average length of landed fish (deGraf and Moore 1987). Olsen (unpublished) suggested that landings in the Virgin Islands in 1979 probably exceeded maximum sustainable yield. In 1979 the average annual catch was 159 kg (350 lb) per trap and dropped to 45.4 kg (100 lb) per trap in 1989 (Appeldoorn *et al.* 1992).

Changes in family and species composition and abundance: Serranids, lutjanids and balistids dominated trap catch in the Virgin Islands (Dammann 1969, Brownell and Rainey 1971, Olsen and LaPlace 1978, Olsen *et al.* 1978, Boulon and Clavijo 1986) until the early 1980s. *Epinephelus striatus*, *E. adscensionis*, *E. guttatus*, and *Cephalopholis fulvus* were all reported to be common components of shallow water trap catch in the Virgin Islands in 1970 (Brownell and Rainey 1971). *E. guttatus*, a highly sought food fish, was the most abundant species in trap landings on St. John in 1984

(Boulon and Clavijo 1986), fourth in abundance in traps in a 1994 experimental trapping study in coral reef and gorgonian habitats (Wolff *et al.* in press) but the eleventh most abundant in this study (Table 3). Not only has the relative abundance of many targeted species declined, but so has the number of fish per trap. In Dammann's comprehensive analysis of the VI fishery (1969), catch data for 12 species were detailed from an experimental trapping study off of St. Thomas in 1968. A comparison with this study shows lower numbers of *C. fulvus*, *E. guttatus*, and *B. vetula* (commercially targeted species) and greater numbers of *A. coeruleus*, *Lactophrys spp.*, and *Calamus spp.* per St. John trap (Table 8). Comparing by family, traps from this study contained fewer serranids and balistids and more acanthurids and lutjanids (Table 9). This apparent rise in numbers of lutjanids is most likely due to the presence of *O. chrysurus* and not the larger lutjanids. Evidence from three other Virgin Island experimental trapping studies corroborate our findings of acanthurid dominated trap contents and a low relative abundance (2-5%) of serranids (Beets 1997, Wolff *et al.* in press).

Beets (1997) documented declines in the average length and number of species of lutjanids and serranids trapped from a St. John reef between 1982 and 1993; he also found declines in serranid and lutjanid densities (using belt transects) between 1994 and 1996 on four St. John reefs (unpublished). The abundance and lengths of two small serranid species, *E. fulvus* and *E. guttatus* declined in the Virgin Islands (Olsen *et al.* unpublished, Beets and Friedlander 1992; Beets *et al.* 1994). *Epinephelus 'striatus* was common in trap catches in 1971 (Brownell and Rainey 1971) and was frequently caught by hand-line in the Virgin Islands until a large spawning aggregation was fished to extirpation in the late 1970s (Olsen and LaPlace 1978). The rarity of *E. striatus* and *Lutjanus analis* in 1985 landings in the Virgin Islands and Puerto Rico indicated they were "essentially extinct for fisheries purposes" (Bohnsack *et al.* 1986). During experimental trapping in 1994, 41 *E. striatus* (0.7% of catch) were trapped in 451 hauls (Wolff *et al.* in press). Only six individuals (0.4%) were observed in traps during this study; the species is rarely seen today (Beets and Friedlander 1992; Beets 1997; Wolff *et al.* in press).

TABLE 8

A comparison of mean number of individuals per trap from [1]St. Thomas (Dammann 1969) and [2]St. John (this study) (standard error of the mean in parentheses)

Species	Fish/trap 1968[1]	Fish/trap 1992-94[2]
Epinephelus guttatus.	0.52	0.14 (0.03)
Balistes vetula	0.48	0.21 (0.05)
Lactophrys spp.	0.44	0.69 (0.09)
Holocentrus spp.	0.31	0.18 (0.04)
Acanthurus coeruleus	0.24	0.72 (0.11)
Calamus spp.	0.22	0.55 (0.10)
A. bahianus	0.16	0.16 (0.05)
Haemulon plumieri	0.15	0.21 (0.05)
Ocyurus chrysurus	0.13	0.21 (0.08)
H. sciurus	0.13	0.06 (0.02)
Sparisoma chrysopterum	0.13	0.06 (0.03)
Cephalopholis fulvus	0.31	0.02 (0.01)

TABLE 9

Mean number of fish/trap by family for St. Thomas in 1969 (Dammann 1969) and St. John, this study (standard errors in parentheses)

	1969	1992-4	1992	1993	1994
Serranidae	0.6	0.2 (0.04)	0.4 (0.12)	0.1 (0.03)	0.3 (0.09)
Balistidae	0.5	0.3 (0.06)	0.3 (0.12)	0.4 (0.11)	0.1 (0.04)
Acanthuridae	0.4	1.0 (0.13)	0.5 (0.22)	0.9 (0.16)	1.3 (0.29)
Ostraciidae	0.4	0.7 (0.09)	0.9 (0.25)	0.8 (0.12)	0.5 (0.14)
Haemulidae	0.3	0.3 (0.06)	0.5 (0.17)	0.3 (0.10)	0.2 (0.05)
Sparidae	0.2	0.6 (0.10)	0.6 (0.16)	0.7 (0.18)	0.3 (0.12)
Lutjanidae	0.1	0.4 (0.09)	0.6 (0.19)	0.5 (0.17)	0.2 (0.06)

Comparison with other tropical reef fisheries: How does the St. John fishery compare to other tropical reef fisheries? Comparing the trap data from this observational study to previous trap studies, St. John traps contain fewer fishes and lower biomass than most other reported values (Fig. 8) and appear closest to Jamaican shelf data from the early 1970s (Munro 1974) and Virgin Islands data from 1988-1992 (J. Beets, unpublished). In this study, only six species accounted for over 50% of the fishes in traps. Other studies have reported a greater number of species comprising the majority of the catch (Juhl 1969, Munro *et al* 1971, Munro 1974, Craig 1976, Olsen *et al* 1978, Stevenson and Stuart-Sharkey 1980, Taylor and McMichael 1983, Bohnsack *et al* 1994). The indication is that the St. John fishery has been under heavy fishing pressure.

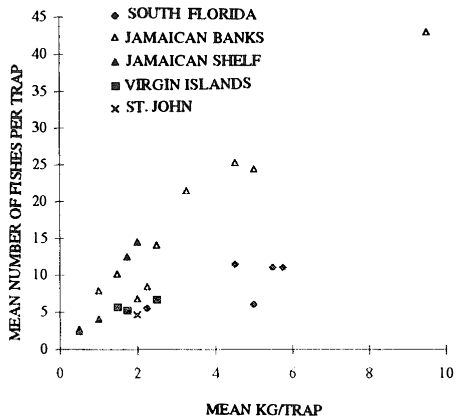

Fig. 8 Trap data showing mean number of fish per trap and respective mean biomass (kg) per trap from Caribbean studies: South Florida (Sutherland & Harper 1983, Taylor & McMichael 1983, Bohnsack et al 1989); Jamaican Banks (Hartsjuiker & Nicholson 1981); Jamaican shelf (Munro 1974, 1983); Virgin Islands (Dammann 1969, Sylvester & Dammann 1972); St. John (this study).

Serranids: Serranids grow slowly and are relatively old at first sexual maturity; they are known to be vulnerable to most fishing gears, aggressive (High and Beardsley 1970), relatively large and highly sought after for food. Because of these factors, their abundance and mean size have been suggested to be the best measure of the state of a fishery (Stevenson and Stuart-Sharkey 1980; Munro and Smith 1984; Munro and Williams 1985; Russ 1985, 1991). Declines in these predators have been documented in Barbados (Rakitin and Kramer 1996), Bermuda (Butler *et al.* 1993), Jamaica (Koslow *et al.* 1988; Koslow *et al.* 1994), the Philippines (Russ and Alcala 1989) and Florida (Bohnsack *et al.* 1994). Ferry and Kohler (1987), however, reported a higher serranid catch rate on the more heavily exploited of two Haitian reefs.

Compared with other published data, the number of serranids per trap in this study exceeds only that from the Port Royal Cays and SE Pedro Bank, Jamaica (Koslow *et al.* 1988) (Table 10). The relative abundance of serranids (4%) in this study was much lower than recorded in any other study. Because preferred food-fish species represent a greater proportion of the reported or marketed catch than of the total catch, the relative abundance of serranids and other targeted fishes from this observational study and other studies using total catch would be expected to appear slightly deflated compared to studies using port sampling or fisher reports. Koslow *et al* (1994) suggested that in

TABLE 10

Number of serranids per trap reported in Caribbean studies

Date	Location	Serranids/trap	Study
1983	South Florida	6.0	Taylor & McMichael 1983
1986	SW Pedro Bank, Jamaica	1.8	Koslow *et al.* 1988
1969	Virgin Islands	0.6	Dammann 1969
1984	Haiti	0.5	Ferry & Kohler 1987
1992-4	St. John, USVI	0.2	this study
1986	SE Pedro Bank, Jamaica	0.14	Koslow *et al.* 1988
1986	Port Royal Cays, Jamaica	0.01	Koslow *et al.* 1988

a light to moderately fished area large piscivores would account for 5-10 % of the catch, while in a heavily fished area, they might contribute 1-2%. This would indicate the St. John resource is moderately fished. However, Koslow developed this yardstick with hook-and-line and trap data from Belize and Jamaica. The proportion of serranids and lutjanids in the catch from both gears would be expected to be higher than in trap catch alone, due to the higher catchability of both families with hook-and-line.

Factors affecting the St. John Fishery

Overfishing: Certain patterns associated with overfishing (Koslow *et al.* 1988, G. Sedberry and J. McGovern , unpublished, Jennings and Polunin 1996 and references) have been identified in this study. The observed dominance by acanthurids in this study fit Jennings' and Polunin's (1996) prediction that a small, fast-growing species from a lower trophic level would eventually dominate as a result of intense fishing pressure. Change in species composition would result from fishers simply targeting or keeping species previously considered "trash" fish or "bycatch". However, as Jennings *et al.* (1995) point out, it is difficult to determine whether changes in a fishery are due to changes in recruitment, differences in growth, and/or degrees of fishing intensity.

Mesh size: Mesh size may have contributed to the smaller sizes and numbers of fish observed in St. John traps. Some of the most commonly (and legally) used mesh sizes, in St. John and the Caribbean region in general, retain sexually immature fishes and individuals smaller than the legal minimum size for harvesting (Munro 1974, 1983, Thompson and Munro 1974, Olsen *et al.* 1978, Stevenson 1978, Taylor and McMichael 1983, Bohnsack *et al.* 1989). The legal minimum mesh diameter in the Virgin Islands between 1992-1994 was 32 mm (1.25 in), smaller than meshes known to retain sexually immature individuals of commercially important species. The observed change in maximum length of scarids (400 to 330mm) and serranids (525 to 400mm) and shift to smaller sizes (Fig. 6) may be the result of "growth overfishing" (Russ 1991), due to long-term use of small mesh traps.

Variations in recruitment: The observed changes in abundance of fishes in St. John traps may result from temporal or spatial variations in recruitment (Munro and Williams 1985, Doherty and Williams 1988, Doherty 1991). Shifts in the length-frequency distribution by year were associated with a fairly constant number of individuals in the smallest exploited size classes over time. A series of lean recruitment years for a large number of species from several families and trophic guilds is unlikely, especially across habitats, depths and at the scale of kilometers. However, it is possible that a decrease in recruitment of species A could be balanced out in our pooled data by a large increase in species B, with the net result of no observed change in the smallest exploited size classes. Only after a period of time would the differences in the growth rates and size at maturity show up in the length-frequency distributions. If recruitment of smaller species such as acanthurids was high and of large scarids was low, within a few years, the frequency of, for example, individuals under 250 mm would increase and the number larger than that would have decreased. If removal of individuals from the system increased, the number of larger fish would plummet. Overfishing of the sources of recruitment to St. John, be they hundreds of kilometers upcurrent or local, may already be a factor.

We suggest that the high relative abundance and number of herbivores per trap, and acanthurids in particular, is most probably the result of a number of factors: the removal of predators through fishing (Olsen and LaPlace 1978, Beets 1997); an abundance of food (Rogers *et al* 1991, Rogers *et al* 1997; unpublished NPS/BRD data); and good recruitment years (Williams 1980, 1991, Munro and Williams 1985).

Habitat degradation: A combination of natural processes (hurricanes and diseases) and human activities (land clearing, boat groundings, anchoring, fishing, snorkeling and diving) are responsible for a decrease in the extent and density of seagrass beds, a decrease in coral and an increase in macroalgal cover on some reefs in St. John waters. Since the late 1980s, there has been no recovery on two damaged scleractinian coral reefs and little recovery in seagrass beds in St. John (Muehlstein 1997, Rogers *et al.* 1997). Damage to these habitats which are used as nurseries, shelter for adults and sources of food, undoubtedly has far-reaching effects on the abundance and species composition of reef fishes (Hughes 1994, Jennings and Polunin 1996). With the rapid increase in development, the island's population, tourism, and marine recreational activities, habitat degradation will undoubtedly continue, if not accelerate.

Inside *versus* outside park waters: The primary mission of the National Park Service is to preserve and protect the natural and historical resources of national parks for the enjoyment of future generations. Theoretically, Virgin Islands National Park fishing regulations protect the species diversity and age-class structure of the finfishes within park waters, providing greater protection to reef fishes within than outside the park. Were that the case and assuming the same habitats and depths inside and outside the park, we would expect the number of fishes per trap to be greater, fishes to be larger and commercially targeted taxa more abundant in the park.

Data from this observational study, a general trapping study around St. John and visual point censuses (J. Beets, unpublished) all have documented similar species, family and feeding guild composition, abundance and number of fishes per trap inside and outside the park. However, the number of commercially sought sparids and balistids per trap appeared higher outside park waters while more acanthurids and scarids were in traps within park waters. Serranids and scarids occurred significantly more frequently in traps inside park waters. For serranids, this is probably a sampling artifact due to the small sample size and low number of serranids in traps. Of the few serranids in traps (n=49), 39 were the small *E. guttatus* and not the larger, highly sought piscivorous serranids. Habitat and depth preference do not appear to be confounding factors, as similar habitats and depths (over 90% deeper than 15m) were censused inside and outside the park.

From these data, it appears that either the park's present fishing regulations are insufficient to protect the reef-associated fishes, enforcement of existing regulations (mesh size, functioning biodegradable panel, non-commercial fishing only) is lacking, or a combination of both.

This 1992-1994 study of the St. John trap fishery documented disturbing patterns. Algal plain, gorgonian hard-bottom and non-living substrates contributed 78% of the fish, pointing to increased pressure on the non-coral habitats and deeper waters. The low number of individuals per trap, low relative abundance of serranids, rise in relative abundance and number of herbivores per trap, change in species composition, shift to smaller size classes, and the dwindling numbers of most commercially targeted species are all indications of an overfished resource (Russ 1985, Koslow *et al* 1988, Sedberry and McGovern, unpublished, Jennings and Polunin 1996 and references therein). Unless Virgin Islands National Park and the Territorial government of the U.S. Virgin Islands provide increased protection for the reef-associated fishes, the trap fishery may go the way of the Jamaican shelf (Hughes 1994).

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