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The effects of boat propeller scarring intensity on nekton abundance in subtropical seagrass meadows

Received: 30 December 2004 / Accepted: 21 September 2005 / Published online: 29 November 2005
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Abstract Seagrasses play a critical role in the function and structure of coastal ecosystems, and they are an important habitat for a variety of marine organisms. Damage to seagrass beds caused by boat propeller scarring is significant in many areas. This study was designed to assess the impact of varying scarring intensities on nekton density. We selected ten replicate (10 m×25 m quadrats) sites representing four distinct scarring intensities: reference (0%), low (5% or less), moderate (5–15%), and severe (>15%). Sites were sampled in 2003–2004 for nekton during four seasons (summer, fall, winter, and spring) using epibenthic sleds. There were eight taxa numerically dominant in all seasons and an additional four seasonally dominant species. We were unable to detect a significant effect of propeller scarring on nekton density at any scarring level. Additionally, regression analysis indicated no relationship between scarring intensity and nekton density. These results suggest that propeller scarring intensities of up to ca. 27% may not impact nekton densities. However, seagrass loss, higher scarring intensity, and scale may play a critical role in determining the impact of propeller scarring on nekton.

habitat types (Quammen and Onuf 1993; Short and Wyllie-Echeverria 1996; Levin et al. 1997) and typically support a great abundance of fish and invertebrates than other habitat types (Beck et al. 2001). Seagrasses function as a nursery habitat for economically and recreationally important fishery species (Heck and Thoman 1981; Short and Wyllie-Echeverria 1996; Levin et al. 1997; Minello 1999; Beck et al. 2001; Heck et al. 2003). SAV is a structurally complex habitat (Attrill et al. 2000; Heck et al. 2003) providing protection from predation (Rooker et al. 1998; Stunz and Minello 2001) and increased growth rates for associated fauna (Holt et al. 1983; Heck and Thoman 1984; Orth et al. 1984; Rozas and Odum 1988; Stunz et al. 2002b; Heck et al. 2003). Despite the importance of seagrass, this marine habitat type has gone through worldwide (Short and Wyllie-Echeverria 1996) and local (Quammen and Onuf 1993) decline in recent decades. Seagrass decline results from several anthropogenic disturbances including dredging (Quammen and Onuf 1993; Onuf 1994), nutrient enrichment (Tomasko and Lapointe 1991; Short et al. 1995), and mechanical damage (e.g., propeller scarring) (Zieman 1976; Sargent et al. 1995; Dawes et al. 1997; Bell et al. 2002; Dunton and Schonberg 2002; Uhrin and Holmquist 2003).

As boating activity in seagrass meadows has increased, damage from boat propellers has become a significant problem (Dunton and Schonberg 2002). A propeller scar is created when a boat propeller tears through the rhizomal mat of a seagrass bed (Zieman 1976; Dawes et al. 1997). This may cause erosion of the surrounding area (Eleuterius 1987), potentially leading to deterioration of seagrass bed integrity and coverage that may affect the function of the entire seagrass community (Zieman 1976). Propeller scarring is prevalent in the shallow seagrass flats along the coast of south Texas (Dunton and Schonberg 2002). They occur for a number of reasons including: shortcuts at channel junctions and access to shallow grass beds from blind channels (Zieman 1976; Sargent et al. 1995; Dawes et al. 1997; Bell et al. 2002; Dunton and Schonberg 2002; Uhrin and

Introduction

Seagrasses, or submerged aquatic vegetation (SAV), play a critical role in the function and structure of coastal ecosystems (Hemminga and Duarte 2000). They are one of the most productive and valuable marine

Communicated by P.W. Sammarco, Chauvin

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Holmquist 2003). In addition, the intensity of propeller scarring increases with population density (Sargent et al. 1995; Dunton and Schonberg 2002). It is estimated that seagrasses along the Texas coast are worth 2.1–6.6 billion dollars, with the per acre value of seagrass on the Gulf Coast between \$9,000 and \$28,000 (Lipton et al. 1995). Using the per acre value of seagrass (Lipton et al. 1995) and data of scarring intensity (Dunton and Schonberg 2002), the loss of seagrass from propeller scarring in two popular boating areas (Estes Flats and Redfish Bay) alone can be estimated at \$800,000 to 6.7 million dollars in lost recreational and commercial value.

Seagrass can recover and regrow in propeller scars, but the process is slow, species dependant, and affected by prevailing wave and current regimes. Single scars can regrow in 0.9–4.6 years in *Halodule wrightii* (Sargent et al. 1995) and 1.7–10 years in *Thalassia testudinum* (Dawes et al. 1997). Due to this slow growth rate seagrass may show long-term damage from propeller scarring (Dawes et al. 1997). To date, studies examining scar regrowth have focused on the recovery of a single scar. However, areas along channel edges and at channel junctions are susceptible to repeated scarring (Sargent et al. 1995; Dunton and Schonberg 2002), and it is unknown how long, if ever, it will take for these areas to recover.

Typically, areas with greater invertebrate or fish densities are considered as better nursery habitats (Minello 1999; Beck et al. 2001). Invertebrates and fish show differential selection to habitat types (Minello 1999; Stunz et al. 2002a); therefore, examining habitat-specific density patterns is useful in determining the relative habitat value. For example, several studies have shown greater nekton densities in vegetated versus unvegetated habitats (Zimmerman and Minello 1984; Minello and Webb 1997; Rozas and Minello 1998; Castellanos and Rozas 2001; Stunz et al. 2002a), with densities 2–25 times greater in seagrass (see SCPT 1999). Furthermore, a variety of commercially important estuarine taxa show a positive relationship between seagrass coverage and production (Heck et al. 2001).

There have been few published studies examining the faunal effects of propeller scarring (see Bell et al. 2002; Uhrin and Holmquist 2003). Uhrin and Holmquist (2003) took a small-scale approach to examine the effects of propeller scarring by measuring faunal densities within the scar and at varying distances from the scar. They found a significant decrease in shrimp and mollusks up to 5 m from the scar. Bell et al. (2002) took a landscape approach to examine propeller scarring, and they did not find a difference between scarred (6–31%) and unscarred sites. Clearly, further study is needed to determine if there is a relationship between nekton and scarring intensity.

Propeller scarring removes seagrass creating an overall decline in the amount of structured habitat. Several studies have shown that more structured habitats increase survival of juvenile organisms. For

example, Rooker et al. (1998) and Stunz and Minello (2001) have shown that more structurally complex habitats often increase the survival of juvenile fish. Alternately, some species use seagrass beds for shelter and forage over adjacent unvegetated habitats (Summerson and Peterson 1984). Seagrass-associated fauna may exhibit different density patterns and behavior based on their use of the edge habitat (Bell et al. 2002). It has also been shown that the abundance of organisms is higher in patchy than homogeneous seagrass meadows (Holt et al. 1983). It is believed that an increased edge habitat gives a greater area for the organisms to forage for food (Holt et al. 1983). The degree to which habitat fragmentation alters animal dispersal depends upon an organism's mobility and the scale of fragmentation under investigation (Doak et al. 1992). It is unknown if a single scar can affect the faunal distribution (Uhrin and Holmquist 2003). To date, there is no clear understanding of the exact effects of habitat fragmentation from propeller scarring: at what level, if any, the fragmentation is beneficial, and at what point this causes degradation to the functionality of the community. This research is the first study to examine nekton responses to varying levels of fragmentation. Specifically, we examined three distinct scarring intensities to assess the effect of propeller scarring on nekton density.

Materials and methods

Study site

Redfish Bay (27°54'27"N, 97°06'45"W) is a secondary bay in the 447-km² Aransas Bay complex (Fig. 1), located along the coast of south Texas (Britton and Morton 1997). It is a barrier-built, positive estuary with freshwater inflows from the Mission and Aransas Rivers (Britton and Morton 1997), and one open connection to the Gulf of Mexico. All five species of seagrass (*H. wrightii*, *T. testudinum*, *Syringodium filiforme*, *Halophila engelmannii*, and *Ruppia maritima*) found in Texas occur in Redfish Bay region; however, *H. wrightii* is dominant in this system (SCPT 1999). The tides are mixed, primarily diurnal, with a mean daily range of 0.12 m (Rockport, Aransas Bay, National Ocean Service, NOAA). Aerial surveys (Dunton and Schonberg 2002) confirmed widespread presence of propeller scarring within our study area in Redfish Bay.

Delineation quadrats for scarring intensity

Maps of scarring intensity by Dunton and Schonberg (2002), aerial surveys, and intensive ground truthing were used to locate sites. The majority of propeller scarring occurs in waters <1 m (Zieman 1976). Sites were selected with a mean water depth of 0.5 m making them highly susceptible to propeller scarring.



Fig. 1 Redfish Bay ($27^{\circ}54'27''\text{N}$, $97^{\circ}06'45''\text{W}$), a secondary bay in the 447-km² Aransas Bay complex, located along the coast of south Texas. Sampling blocks are indicated with *plus sign*

In ten separate sites in Redfish Bay (Fig. 1) we selected replicate 10 m \times 25 m quadrats (Fig. 2) representing four distinct scarring intensities: low (1–5% scarring), moderate (5–15% scarring), severe (>15% scarring), and reference (0%) sites (see Sargent et al. 1995). To characterize the quadrat, the length and width of each scar was measured every 5 m, averaged, and used to calculate the percentage of scarring within each quadrat. Reference sites were in areas without propeller scarring and within ca. 100 m of scarred sites.

Environmental characterization

Water depth, salinity, dissolved oxygen, and temperature were measured at each site at each sampling date to assess variability among sampling locations. Temperature and dissolved oxygen were measured using a YSI model DO 200, and water depth was recorded as the average of four random depths taken in each quadrat.

To compare seagrass characteristics at each site, samples were taken in monotypic stands of *H. wrightii*

and seagrass; shoot density and above and below ground biomass were measured. Shoot density and above and below ground biomass were measured between 18 April 2004 and 6 May 2004. Three 10.05-cm diameter cores were taken at each site and averaged to calculate the mean shoot density and biomass per site. Cores were taken haphazardly within areas of solid seagrass within the quadrat, and prop scars, when present, were avoided. Biomass samples were processed by separating above and below ground material, and placing samples in aluminum trays in an oven for 120 h at 90°C. Desiccated samples were weighed to the nearest 0.001 g and converted to g seagrass m⁻¹. Shoot density, and above and below ground biomass were analyzed by scarring intensity with analysis of variance (ANOVA).

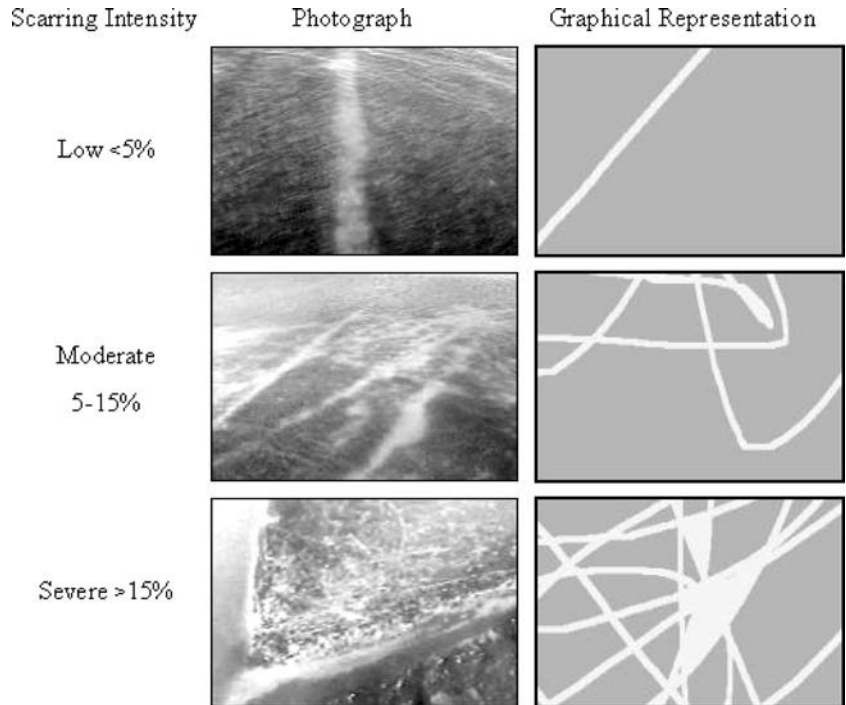
Nekton collection

We sampled for nekton during four seasons: summer (30 July–10 August 2003), fall (18–24 October 2003), winter (7–8 January 2004) and spring (17–18 March 2004) using epibenthic sleds (see Stunz et al. 2002a). Briefly, an epibenthic sled is a 1-mm mesh conical plankton net attached to a fixed metal frame 0.6 m (length) \times 0.75 m (height) used in estimating small-scale density patterns in seagrass. We sampled each site by simultaneously placing two sleds at the top of each quadrat and walking a semicircular route around the sampling area to avoid disturbance. Sleds were pulled at the same time with one person towing each sled. Sleds were pulled by hand, the length of the rope (16.7 m) to cover 10 m² of bottom. Organisms were sorted from seagrass and detritus and preserved in 70% ethanol. Density from duplicate sled tows were averaged between tows. Organisms were identified to species or the lowest possible taxon.

Nekton analysis

We selected the eight most numerically abundant taxa to analyze individually by season. Pinfish (*Lagodon rhomboides*), pipefish (*Syngnathus* spp.), code goby (*Gobionellus robustum*), darter goby (*Gobionellus boleosoma*), killifish (Fundulidae), blue crab (*Callinectes sapidus*), Atlantic mud crab (*Panopeus herbstii*), and grass shrimp (*Palaemonetes* spp.) were dominant in all seasons. We selected an additional four species of commercial importance: red drum (*Sciaenops ocellatus*), spot (*Leiostomus xanthurus*), brown shrimp (*Farfantepenaeus aztecus*), and white shrimp (*Litopenaeus setiferus*) that were not dominant in all seasons, but were analyzed in the season(s) when they were most abundant. For the purpose of analysis all species of killifish (Fundulidae) were grouped together by family, and pipefish (*Syngnathus* spp.) and grass shrimp (*Palaemonetes* spp.) were grouped together by genus. All organisms collected were included in the overall density analysis.

Fig. 2 Sites were classified into three distinct scarring intensities (modified from Sargent et al. 1995): low (1–5%), moderate (5–15%), and severe (>15%). This figure shows pictures taken in Redfish Bay, Texas of these scarring intensities and graphical representation. Note that large bare patches in the severely scarred areas are created by multiple scars and erosion



We converted the number of organisms collected in each quadrat to density ($\#/m^2$) and a log ($x+1$) transformation was used to minimize heteroscedasticity. We used analysis of variance (ANOVA) to assess the differences in abundance of nekton among sites and at different levels of scarring intensity. We compared the percentage of scarring in each site with the density of organisms present using a simple linear regression. We used a significance level of $\alpha=0.05$ for all statistical tests.

SE=0.192 (low), 8.86%, SE=0.975 (moderate), and 20.27%, SE=1.209 (severe). Scarring intensities were significantly different in each scarring level ($F=105.841$, $df=10$, $P<0.001$). At each site, dissolved oxygen, salinity, temperature, and depth were similar within each season (Table 1). We measured seagrass shoot density (Fig. 4), and seagrass above and below ground biomass (Fig. 5) at each site. We found no difference among sites.

Results

Environmental characterization

All scars within sites were measured to calculate scarring intensity (Fig. 3). Mean scarring intensities were: 1.93%,

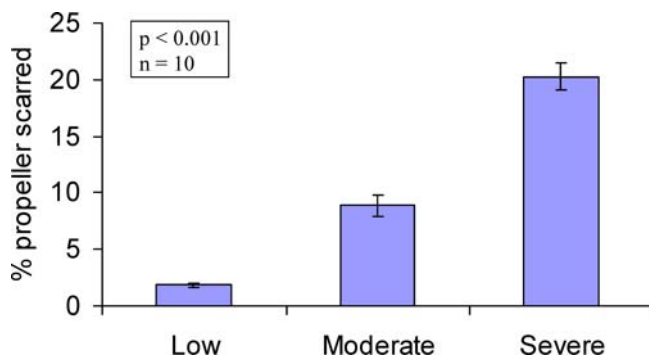


Fig. 3 Mean measured scarring intensities at sites in Redfish Bay, Texas were: 1.93%, SE=0.192 (low), 8.86%, SE=0.975 (moderate), and 20.27%, SE=1.209 (severe). The P -value is from an ANOVA ($\alpha=0.05$) comparing measured scarring intensities within each scarring level

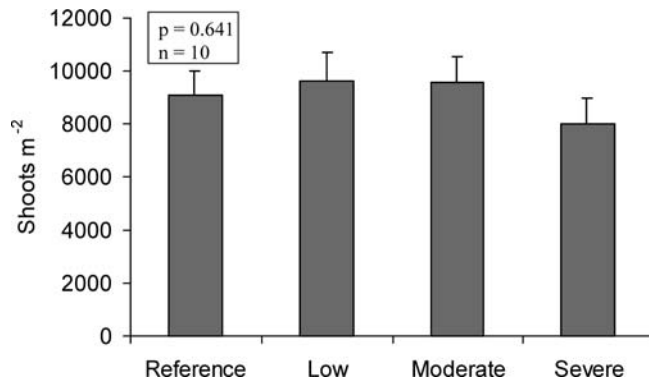
Nekton density

We collected a total of 24 species of fish and 6 species of crustaceans in the summer, 23 species of fish and 6 species of crustaceans in the fall, 20 species of fish and 6 species of crustaceans in the winter, and 18 species of fish and 4 species of crustaceans in the spring (Table 2). Crustaceans numerically dominated the catch and accounted for 84% in summer, 92% in fall, 87% in winter, and 77% in spring of the total fauna. As would be expected, we found differences in nekton densities and composition in relation to season (data not shown); therefore, seasons were analyzed separately. There were eight taxa numerically dominant in all seasons: pinfish, pipefish, code goby, darter goby, killifish, blue crab, Atlantic mud crab, and grass shrimp (Table 2). We found no significant differences in overall nekton density (all species included) across varying levels of scarring intensity in any season (Fig. 6, Table 3).

In summer, the eight dominant taxa accounted for 94% of the total catch. Brown shrimp accounted for an additional 2% of the total catch in summer. Densities of the nine most abundant taxa in spring were not significantly different in relation to scarring intensity

Table 1 Seasonal mean water depth (cm), salinity (ppt), dissolved oxygen (DO; mg/l), and temperature measurements (C) for all sites (\pm SE) in Redfish Bay, Texas

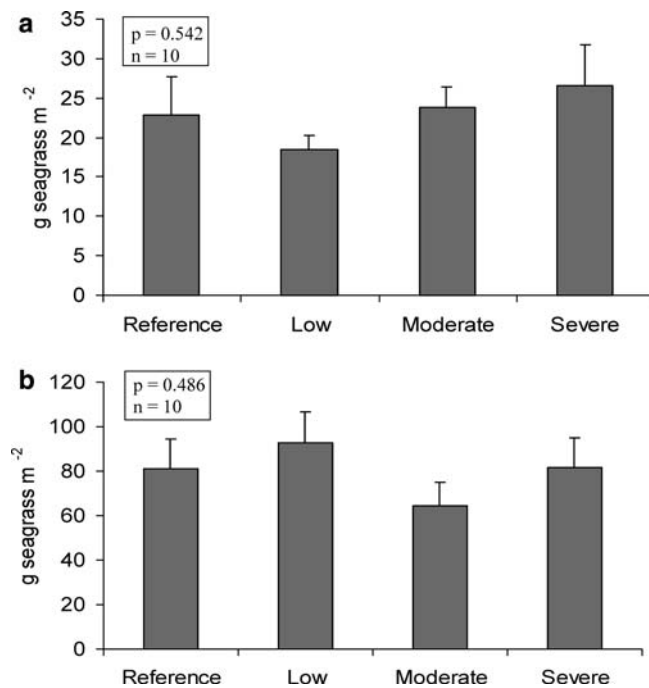
	Depth	SE	Salinity	SE	DO	SE	Temperature	SE
Summer	27.82	1.38	33.33	0.46	6.91	0.34	30.21	0.17
Fall	49.46	1.63	15.89	0.10	7.90	0.10	24.17	0.14
Winter	25.49	2.36	25.27	0.15	9.20	0.11	11.88	0.07
Spring	25.38	1.59	20.87	0.34	7.21	0.21	22.41	0.18

**Fig. 4** Shoot density of *H. wrightii* (\pm SE) measured in spring of 2004, Redfish Bay, Texas. The *P*-value is from an ANOVA ($\alpha=0.05$) comparing shoot density within each scarring level

(Table 3). In fall, the eight dominant taxa again accounted for 94% of the total catch. Brown shrimp and red drum accounted for 5 and 0.32% total catch, respectively. Densities of the ten most abundant taxa in

fall were not significantly different in relation to scarring intensity (Table 3). In winter, the eight most abundant taxa accounted for 97% of the total catch. As evident in Fig. 6, winter had the lowest organism density in relation to the other seasons. However, there were no differences in the density of the eight most abundant taxa in relation to scarring intensity (Table 3). In spring, the eight most abundant taxa accounted for 71% of the total catch. White shrimp, spot, and bay whiff accounted for an additional 24, 2, and 1% of the total catch, respectively. The greatest numbers of organisms were collected in the spring. However, there were no significant differences in the density of the 11 most abundant species in relation to scarring intensity (Table 3).

Each site was measured for total scarring intensity, and a regression of scarring intensity versus nekton density (all species included; Fig. 7) showed no relationship (Table 4). This analysis was run each season for the dominant species of pinfish, code goby, darter goby, red drum, bay whiff, spot, blue crab, Atlantic mud crab, penaeid shrimp, white shrimp, brown shrimp, and grass shrimp. Results were not significant for any species for any season (Table 4).

**Fig. 5** Above (a) and below (b) ground biomass of *H. wrightii* (\pm SE) measured in spring of 2004, Redfish Bay, Texas. The *P*-value is from an ANOVA ($\alpha=0.05$) comparing aboveground biomass within each scarring level

Discussion

Density patterns of juvenile organisms can serve as an indicator of habitat quality since it reflects recruitment, mortality, and emigration (Minello 1999), and this study evaluated the effects of propeller scarring on nekton abundance. It has been suggested that propeller scarring decreases the faunal densities (Zieman 1976). However, this study indicates that different levels of propeller scarring, up to 27%, do not affect faunal densities in any season. Nekton densities measured in this study are comparable to other studies in the same region and vegetation type. Furthermore, there does not appear to be a relationship between nekton density and the percent scarring in a site for scarring intensities between 1 and 27%.

We found similarity across all scarring intensities and no relationship between scarring intensity and nekton density. Since there is no difference in density, it may imply that organisms must aggregate in greater densities in the vegetation in scarred areas versus unscarred areas (Bell et al. 2002). This may be the result of the nearby presence of large unscarred areas where fauna can either move to find resources or from where new individuals can immigrate (Bell et al. 2002). When there is a

Table 2 Mean density (\pm SE) of eight numerically dominant taxa and four seasonally abundant taxa collected in different scarring intensities

	Reference		Low		Moderate		Severe	
	Mean	SE	Mean	SE	Mean	SE	Mean	SE
Summer								
<i>Lagodon rhomboides</i> pinfish	0.17	0.04	0.17	0.04	0.24	0.08	0.24	0.08
<i>Gobionellus robustum</i> code goby	0.25	0.07	0.44	0.15	0.46	0.21	0.50	0.26
<i>Gobionellus boleosoma</i> darter goby	2.05	0.55	0.91	0.30	1.32	0.45	0.86	0.37
<i>Syngnathus</i> spp. pipefish	0.52	0.27	0.25	0.13	0.39	0.19	0.40	0.22
Fundulidae killifish	0.33	0.27	0.20	0.10	0.88	0.42	0.04	0.02
<i>Callinectes sapidus</i> blue crab	0.58	0.12	0.63	0.18	0.57	0.08	0.54	0.15
<i>Panopeus herbstii</i> Atlantic mud crab	0.16	0.07	0.07	0.03	0.24	0.18	0.20	0.07
<i>Farfantepenaeus aztecus</i> brown shrimp	0.68	0.31	0.66	0.19	0.73	0.24	0.34	0.13
<i>Palaemonetes</i> spp. grass shrimp	14.73	3.08	12.45	2.71	9.99	2.30	10.78	3.66
Fall								
<i>Lagodon rhomboides</i> pinfish	0.61	0.29	0.35	0.11	0.39	0.16	0.45	0.31
<i>Gobionellus robustum</i> code goby	0.54	0.20	0.72	0.28	0.71	0.30	0.83	0.40
<i>Gobionellus boleosoma</i> darter goby	1.00	0.36	1.11	0.30	1.30	0.53	1.27	0.54
<i>Syngnathus</i> spp. pipefish	0.97	0.50	0.64	0.13	1.18	0.55	0.95	0.44
Fundulidae killifish	0.12	0.10	0.03	0.02	0.03	0.02	0.21	0.21
<i>Sciaenops ocellatus</i> red drum	0.10	0.05	0.12	0.06	0.11	0.05	0.11	0.05
<i>Callinectes sapidus</i> blue crab	0.95	0.22	1.09	0.22	0.87	0.18	0.57	0.17
<i>Panopeus herbstii</i> Atlantic mud crab	0.43	0.36	0.05	0.02	0.07	0.03	0.06	0.03
<i>Farfantepenaeus aztecus</i> brown shrimp	1.34	0.26	1.54	0.35	1.83	0.22	1.67	0.30
<i>Palaemonetes</i> spp. grass shrimp	27.17	6.64	27.42	8.14	34.31	7.13	23.73	5.67
Winter								
<i>Lagodon rhomboides</i> pinfish	0.08	0.03	0.07	0.02	0.09	0.02	0.03	0.02
<i>Gobionellus robustum</i> code goby	0.13	0.09	0.12	0.06	0.04	0.03	0.05	0.03
<i>Gobionellus boleosoma</i> darter goby	0.37	0.13	0.58	0.29	0.40	0.23	0.25	0.11
<i>Syngnathus</i> spp. pipefish	0.16	0.09	0.04	0.02	0.12	0.07	0.02	0.01
Fundulidae killifish	0.04	0.02	0.04	0.04	0.02	0.01	0.05	0.03
<i>Callinectes sapidus</i> blue crab	0.91	0.26	0.90	0.34	1.02	0.54	1.16	0.53
<i>Panopeus herbstii</i> Atlantic mud crab	0.06	0.03	0.06	0.03	0.02	0.01	0.02	0.01
<i>Palaemonetes</i> spp. grass shrimp	11.78	6.20	5.25	2.44	6.58	3.65	5.90	2.80
Spring								
<i>Lagodon rhomboides</i> pinfish	1.56	0.91	2.00	0.98	2.46	1.12	2.95	1.27
<i>Gobionellus robustum</i> code goby	0.54	0.20	0.72	0.28	0.71	0.30	0.83	0.40
<i>Gobionellus boleosoma</i> darter goby	1.37	0.77	1.41	0.64	2.36	0.78	1.55	0.45
<i>Syngnathus</i> spp. pipefish	0.19	0.09	0.19	0.10	0.26	0.13	0.38	0.19
Fundulidae killifish	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01
<i>Leiostomus xanthurus</i> spot	0.34	0.16	0.30	0.09	0.81	0.41	0.78	0.28
<i>Callinectes sapidus</i> blue crab	2.61	0.63	2.21	0.61	4.11	0.90	2.30	0.47
<i>Panopeus herbstii</i> Atlantic mud crab	0.20	0.11	0.07	0.03	0.32	0.14	0.34	0.17
<i>Litopenaeus setiferus</i> white shrimp	2.80	0.85	4.88	1.86	7.82	2.98	6.61	2.58
<i>Palaemonetes</i> spp. grass shrimp	9.65	4.34	7.18	2.75	8.90	2.84	8.08	2.30

Samples were collected in summer and fall 2003, and winter and spring 2004 in Redfish bay, Texas, USA using an epibenthic sled

high-scarring intensity there is proportionately less seagrass coverage and more edge habitat (Uhrin and Holmquist 2003). This small-scale patchiness or habitat heterogeneity may be beneficial, because it provides protection from predation and unvegetated areas to forage for food. Holt et al. (1983) found that habitat patchiness was the greatest landscape factor in affecting density and found more red drum at the seagrass sand ecotone than in homogeneous seagrass. They also found high densities of red drum in patchy seagrass meadows, suggesting movement between unvegetated feeding areas and predation refuge in the seagrass.

Examining higher scarring intensities, different spatial scales, and modeling may serve as a useful tool in estimating a threshold point in propeller scarring for

both faunal responses and limits of seagrass bed stability. Though patchy habitats can temporarily support higher faunal densities, they have decreased ability to endure physical disturbance (Holt et al. 1983) and are highly variable based on wind-generated waves and tidal currents (Robbins and Bell 1994). The highest scarring intensities examined in this study were ca. 27%, and areas greater than 15% were uncommon. However, areas of high-scarring intensity may cause a loss in seagrass bed stability, and it would be useful to focus future research on finding limits to bed stability (Bell et al. 2002). Scarring intensities of 50% would be a good way to examine the effects of propeller scarring (Fonseca and Bell 1998) and would be useful in addressing the possibility of a threshold point. Looking at areas of 50%

Fig. 6 Mean nekton density (\pm SE; all species combined) in each season: summer $P=0.628$, fall $P=0.702$, winter $P=0.952$, spring $P=0.699$. Differences in means were analyzed with an ANOVA ($\alpha=0.05$)

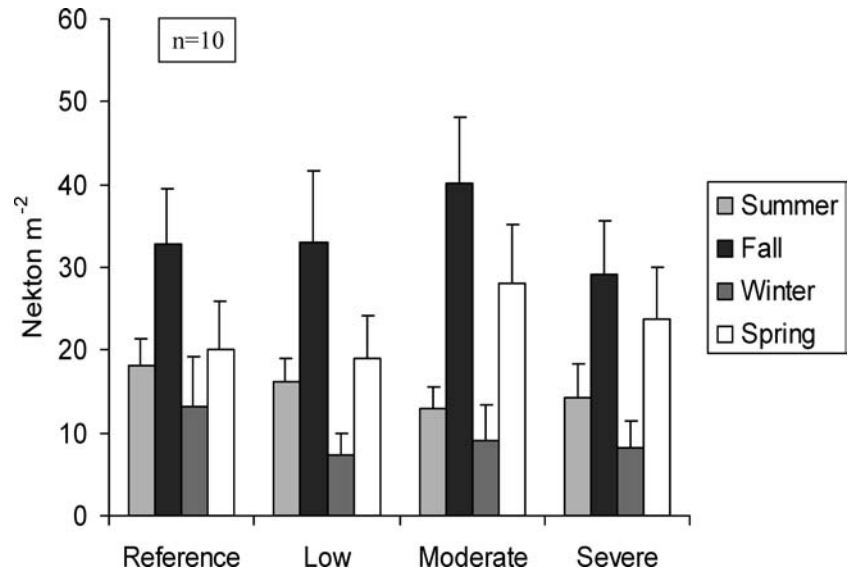


Table 3 Analysis of variance table for nekton density patterns in Redfish Bay, Texas

Species	Summer				Fall			
	<i>n</i>	SS	<i>F</i>	<i>P</i>	<i>n</i>	SS	<i>F</i>	<i>P</i>
<i>Lagodon rhomboides</i> pinfish	10	0.004	0.260	0.854	10	0.003	1.419	0.253
<i>Gobionellus robustum</i> code goby	10	0.022	0.345	0.793	10	0.008	0.065	0.978
<i>Gobionellus boleosoma</i> darter goby	10	0.207	1.119	0.355	10	0.006	0.038	0.990
<i>Syngnathus</i> spp. pipefish	10	0.016	0.212	0.888	10	0.016	0.114	0.951
Fundulidae killifish	10	0.025	0.726	0.543	10	0.011	0.437	0.728
<i>Sciaenops ocellatus</i> red drum					10	0.000	0.010	0.999
<i>Callinectes sapidus</i> blue crab	10	0.008	0.223	0.880	10	0.078	1.351	0.273
<i>Panopeus herbstii</i> Atlantic mud crab	10	0.011	0.443	0.723	10	0.033	0.963	0.421
<i>Farfantepenaeus aztecus</i> brown shrimp	10	0.055	0.790	0.508	10	0.053	0.628	0.602
<i>Palaemonetes</i> spp. grass shrimp	10	0.332	0.813	0.496	10	0.230	0.459	0.712
	Winter				Spring			
	<i>n</i>	SS	<i>F</i>	<i>P</i>	<i>n</i>	SS	<i>F</i>	<i>P</i>
<i>Lagodon rhomboides</i> pinfish	10	0.013	0.153	0.927	10	0.124	0.284	0.836
<i>Gobionellus robustum</i> code goby	10	0.005	0.467	0.707	10	0.009	0.103	0.958
<i>Gobionellus boleosoma</i> darter goby	10	0.016	0.235	0.872	10	0.184	0.928	0.437
<i>Syngnathus</i> spp. pipefish	10	0.005	0.249	0.862	10	0.013	0.258	0.855
Fundulidae killifish	10	0.002	0.493	0.689	10	0.000	0.133	0.940
<i>Leiostomus xanthurus</i> spot					10	0.094	1.083	0.369
<i>Callinectes sapidus</i> blue crab	10	0.001	0.007	0.999	10	0.273	1.905	0.146
<i>Panopeus herbstii</i> Atlantic mud crab	10	0.002	0.917	0.443	10	0.039	1.064	0.377
<i>Litopenaeus setiferus</i> white shrimp					10	0.224	0.334	0.801
<i>Palaemonetes</i> spp. grass shrimp	10	0.238	0.312	0.817	10	0.100	0.151	0.928

or greater propeller scarring was considered in this study; however, it was not possible to find enough areas < 50% scarred to properly replicate a treatment at that level.

This study took a large-scale approach to examine propeller scarring by looking at a large area of scarring and did not determine a difference between scarred and unscarred sites at any scarring intensity. Bell et al. (2002) took a similar approach and also did not find any relationships between scarring and nekton density. Others,

Uhrin and Holmquist (2003) took a small-scale approach by examining a single scar. They found that shrimp and mollusk abundances were lower in the scars and up to 5 m from the scars. However, it is unknown how, if at all, these effects will scale up to a system (Uhrin and Holmquist 2003). The results of our study suggest that several spatial and temporal scales need to be considered when studying fragmentation and these parameters may vary by species, location, and habitat type (Haila 2002). Scale is important in examining distribution and abundance

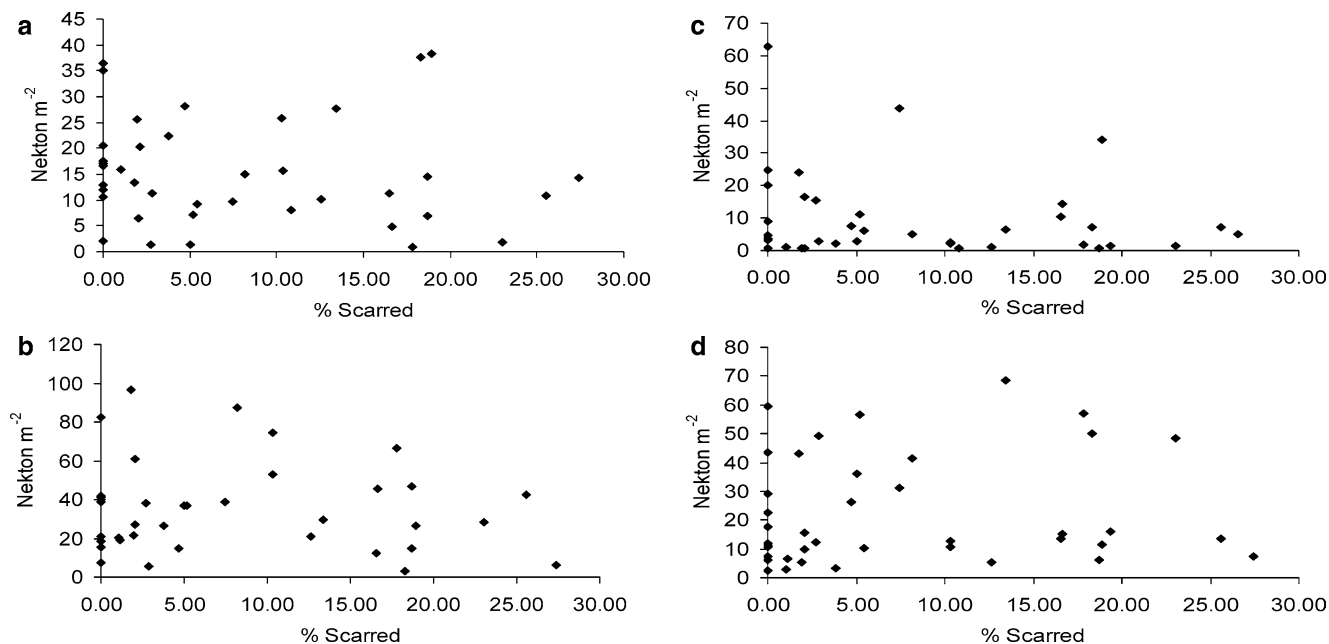


Fig. 7 Linear regression of scarring intensity versus nekton density (all species included). Graph **a** (summer); graph **b** (fall); graph **c** (winter); graph **d** (spring)

Table 4 Summary of linear regressions to determine the relationship between scarring intensity and organism density

Species	Summer				Fall			
	<i>n</i>	<i>F</i>	<i>P</i>	<i>r</i> ²	<i>n</i>	<i>F</i>	<i>P</i>	<i>r</i> ²
<i>Lagodon rhomboides</i> pinfish	40	0.899	0.349	0.024	40	0.015	0.902	0.000
<i>Gobionellus robustum</i> code goby	40	0.822	0.370	0.022	40	0.533	0.470	0.014
<i>Gobionellus boleosoma</i> darter goby	40	0.492	0.487	0.013	40	1.233	0.274	0.031
<i>Syngnathus</i> spp. pipefish	40	0.070	0.793	0.002	40	0.082	0.776	0.002
<i>Sciaenops ocellatus</i> red drum	40				40	0.096	0.759	0.003
<i>Callinectes sapidus</i> blue crab	40	0.130	0.720	0.004	40	1.111	0.298	0.028
<i>Panopeus herbstii</i> Atlantic mud crab	40	0.456	0.504	0.012	40	0.698	0.409	0.018
<i>Farfantepenaeus aztecus</i> brown shrimp	40	0.745	0.394	0.021	40	0.055	0.817	0.002
<i>Palaemonetes</i> spp. grass shrimp	40	0.847	0.364	0.024	40	0.045	0.833	0.001
	Winter				Spring			
	<i>n</i>	<i>F</i>	<i>P</i>	<i>r</i> ²	<i>n</i>	<i>F</i>	<i>P</i>	<i>r</i> ²
<i>Lagodon rhomboides</i> pinfish	40	5.180	0.029	0.120	40	0.478	0.494	0.012
<i>Gobionellus robustum</i> code goby	40	1.130	0.295	0.030	40	0.345	0.560	0.009
<i>Gobionellus boleosoma</i> darter goby	40	0.807	0.375	0.021	40	0.008	0.928	0.000
<i>Syngnathus</i> spp. pipefish	40	0.431	0.516	0.012	40	0.001	0.974	0.000
<i>Leiostomus xanthurus</i> spot	40				40	2.462	0.125	0.061
<i>Callinectes sapidus</i> blue crab	40	0.327	0.571	0.009	40	0.142	0.708	0.004
<i>Panopeus herbstii</i> Atlantic mud crab	40	0.603	0.442	0.016	40	2.721	0.107	0.067
<i>Litopenaeus setiferus</i> white shrimp	40				40	1.061	0.310	0.030
<i>Palaemonetes</i> spp. grass shrimp	40	0.325	0.572	0.009	40	0.073	0.789	0.002

patterns and may greatly affect the interpretation of results (see review by Eggleston et al. 1999). It is difficult to determine the scale at which habitat structure affects survival and abundance of organisms (Hovel and Lipcius 2002). The appropriate scale to examine a population depends upon the size and dispersal capability of the organism (Fahrig and Merriam 1994). In this study we

examined highly mobile species, and therefore it may be more appropriate (Robbins and Bell 1994) to look at higher scarring intensities of propeller scarring over a large area (bay) rather than small regions within the system (Bell et al. 2002).

Though there is inherently a habitat loss associated with habitat fragmentation they are separate entities

(McGarigal and Cushman 2002); consequently, it may be more important to focus on the overall loss versus the underlying mosaic. Single scar regrowth can take 0.9–4.6 years in *H. wrightii* (Sargent et al. 1995) and 1.7–10 years in *T. testudinum* (Dawes et al. 1997). Due to this slow growth rate seagrass may show long-term damage from propeller scarring (Dawes et al. 1997). Additionally, propeller scars can fill with sediment creating an environment that inhibits rhizome growth (Zieman 1976). To date, studies examining scar regrowth have focused on the recovery of a single scar. However, areas along channel edges and at channel junctions are susceptible to repeated scarring (Sargent et al. 1995; Dunton and Schonberg 2002), and there have not been any studies looking at the effects of scarring aggregation. The physical disturbance of seagrass through propeller scarring often creates a clear habitat loss (SCPT 1999) resulting in a cumulative reduction of productive habitat. This landscape fragmentation can generate changes in the physical forces across the landscape, which may have important effects on the remaining vegetation (Saunders et al. 1991). Specifically, wave energy in propeller scars could lead to erosion (Zieman 1976) and deepening of the disturbed area (Eleuterius 1987).

Habitat fragmentation has become a key theme, when examining anthropogenic degradation of the environment (Haila 2002), and it is particularly important to consider the structure of the landscape (Fahrig and Merriam 1994). More information is needed to characterize the effects of propeller scarring on both the seagrass and associated fauna. Future research looking at higher scarring levels and different spatial scales as they relate to both seagrass bed stability and faunal impact would aid in understanding the net impact of propeller scarring, because it may be difficult to detect a change until the bed has become badly degraded. Until the impacts of propeller scarring are fully understood, it is important to protect the remaining seagrass habitat from further degradation.

Acknowledgements We would like to thank the Members of the Fisheries Ecology Lab at Texas A&M University-Corpus Christi for their help on this project. Special thanks to Brooke Stanford, Ryan Fikes, Megan Reese, Amanda Bushon, and Annette Cardona for their hours of assistance in the field and laboratory, and to Steve Moore for producing Fig. 1. We gratefully acknowledge the comments and input by Chris Onuf. We would like to thank Ken Dunton of UTMSI and Richard Watson for flights over the study area. We would also like to thank anonymous reviewers for comments that greatly improved the earlier drafts of the manuscript.

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