Age and Growth of Red Snapper, *Lutjanus campechanus*, in the Northwestern Gulf of Mexico: Implications to the Unit Stock Hypothesis

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ABSTRACT

Red snapper, *Lutjanus campechanus*, in the northwestern Gulf of Mexico are being examined for differences in age and growth parameters among populations east and west of the Mississippi River. In the first year of this three-year study, nearly 2,100 specimens from the recreational fisheries of Alabama, Louisiana, and Texas were sampled for morphometric data and otoliths. Red snapper ranged in age from 1 to 34 years (199 - 916 mm FL), from 1 to 37 years (345 - 913 mm FL), and from 2 to 45 years (315 - 846 mm FL) for Alabama, Louisiana, and Texas, respectively. Regression analyses of fork length (FL in mm) – total weight (TW in kg) relationships did not differ significantly between sexes but did differ between states (TW unavailable from Texas): TW = 2.57 X 10^{-8} FL^{2.94} (r^2 = 0.96) for Alabama and TW = 1.54 X 10^{-8} FL^{3.03} (r^2 = 0.96) for Louisiana. Von Bertalanffy growth models derived from FL at age were $L_\infty = 884 \left(1 - e^{-0.22(t-0.12)}\right)$ for Alabama, $L_\infty = 873 \left(1 - e^{-0.24(t-0.26)}\right)$ for Louisiana, and $L_\infty = 1,017 \left(1 - e^{-0.07(t+3.39)}\right)$ for Texas. Elevated $L_\infty$ and $t_0$ values for Texas are due to a lack of larger individuals in the sample population. Regression analysis on the first ten years of life indicated that red snapper in Texas waters grow at a slower rate than those from Alabama and Louisiana.

KEY WORDS: Growth model, otolith, red snapper

INTRODUCTION

The red snapper, *Lutjanus campechanus* (Family Lutjanidae), inhabits the continental shelves of the Atlantic coast of the United States as far north as Massachusetts and the Gulf of Mexico from Florida to the Yucatan Peninsula. It is also found in the waters off Bermuda, the Bahamas, and off Northern Cuba but is absent in the Caribbean Sea (Rivas 1966, Robins and Ray 1986, Hoese and Moore 1998).

Red snapper support an important recreational fishery in the Gulf of Mexico and the most important commercial fishery in the snapper/grouper complex from Florida to southern Texas (Goodyear 1995a). Both the commercial and recreational red snapper fisheries went essentially unregulated prior to 1990 resulting in a decrease in landings from historic highs of about 6,389 metric tons (mt) in 1965 to 1,015 mt in 1991 and from 4,734 mt in 1979 to 581 mt in 1990 for the commercial and
recreational fisheries, respectively (Schirripa and Legault 1999). These declines prompted the Gulf of Mexico Fishery Management Council (GMFMC) to institute harvest quotas, minimum size limits, trip quotas for commercial fishers, creel limits for recreational fishers, and moratoria on issuing commercial reef fish permits in 1991. However, pressure on the fishery persists (Goodyear 1995a, Schirripa and Legault 1999).

An underlying critical assumption to any fisheries management strategy is that the fish being managed belong to a single unit stock. Currently red snapper in the Gulf of Mexico are managed under that assumption (Camper et al. 1993, Gold and Richardson 1994, Goodyear 1995a, Gold et al. 1997). Herein, we present our preliminary interpretations of data from the first year of a three-year, multi-institutional study investigating the stock structure of red snapper in the northern Gulf of Mexico and examining whether their management as a single unit stock is justified. We determine the ages of fish sampled from recreational sources in the northwestern Gulf of Mexico and examine these data for differences in age distributions east and west of the Mississippi River. We also use size at age information to determine and compare growth rates of red snapper from across the Gulf of Mexico.

METHODS AND MATERIALS

Red snapper from recreational catches were sampled in Alabama, Louisiana, and Texas from April through September 1999. Morphometric measurements (fork length (FL) in mm, total weight (TW) in kg, and eviscerated body weight (BW) in kg), both sagittal otoliths, and gonads were removed (and sex determined).

All undamaged sagittal otoliths were weighed to the nearest 0.1mg. The left otolith from each individual was thin sectioned with the Hillquist model 800 thin sectioning machine equipped with a diamond embedded wafering blade and precision grinder (Cowan et al. 1995). In those instances where the left otolith was damaged or unavailable, the right otolith was used. Examinations of otolith sections were made with a dissecting microscope with transmitted light and polarized light filter.

Counts of annuli (opaque zones) were made on the medial surface of the transverse section along the ventral side of the sulcus groove (Figure 1). Annulus counts were done by two independent readers without knowledge of date of capture or morphometric data of the fish. The appearance of the otolith margin, or edge condition, was coded as opaque or translucent (Beckman et al. 1989). Sections were recounted by both readers when initial counts disagreed. When a consensus could not be reached on the second reading, annulus counts of the more experienced reader were used. Ages of red snapper were estimated from opaque annulus count and adjusted for edge condition when necessary. A uniform hatching date of 1 July was assigned based on previous studies of red snapper reproduction (Render 1995; Collins et al 1996).
Figure 1. Photomicrograph of a transverse section at the core of a red snapper sagittal otolith. White squares indicate annuli counted for age estimation.

Length – weight regressions were fit with linear regression (SAS,1985) to the model $\text{TL} = \text{a} \cdot \text{TW}^b$ with log$_{10}$ transformed data for Alabama and Louisiana specimens (weight data not available for Texas specimens). Analysis of covariance was used to compare sexes and sample sources (states). Von Bertalanffy growth models were fit for FL with nonlinear regression (SAS 1985) in the form: $L_t = L_\infty(1 - e^{k(t-t_0)})$ where $L_t$ is estimated fork length at age $t$, $L_\infty$ is the theoretical maximum fork length, $k$ is the growth coefficient, and $t_0$ is a hypothetical age when length is zero. Growth of red snapper in the first ten years of life appears to be linear (Szedlmayer and Shipp 1994, Patterson 1999), therefore, growth was also evaluated by performing a linear regression of FL at age with a general linear model (GLM) (SAS 1985). Significance level for statistical analyses was 0.05.

RESULTS

During the first year of this three-year project, 2,098 red snapper were sampled from recreational sources including charter boats and dive and fishing tournaments (AL = 786, LA = 737, TX = 575). Among the 1,037 males and 1,014 females sampled from the three states, males ranged from 240 - 868mm, 345 - 913mm, and 315 - 844mm FL for Alabama, Louisiana, and Texas, respectively (Figure 2). Females ranged from 254 - 916mm, 365 - 910mm, and 315 - 795mm FL for Alabama, Louisiana, and Texas, respectively. Komolgorov-Smirnov two sample
tests (Tate and Clelland 1957) indicated no significant differences in length distributions between sexes or among sample sources.

Figure 2. Length frequency distribution of red snapper sampled from recreational catches in A) Alabama (199 - 916mm FL), B) Louisiana (345 - 913mm FL), and C) Texas (315 - 846mm FL).
Regression analyses of FL–TW relationships did not differ significantly between sexes. Neither the slopes (df = 1, 1524; F = 2.76; P < 0.096) nor the intercepts (df = 1,1524; F = 2.18; P < 0.1397) were found to be significantly different; thus data for the two sexes were combined. Regression analysis between states did differ, however, for slopes (df = 1,1524; F = 77725.47; P < 0.001) and for intercepts (df = 1,1524; F = 153.11; P < 0.001), so predictive models were generated for each state:

\[ \text{TW} = 2.57 \times 10^{-8} \text{FL}^{2.94} \ (r^2 = 0.96) \text{ for Alabama} \]
\[ \text{TW} = 1.54 \times 10^{-8} \text{FL}^{3.03} \ (r^2 = 0.96) \text{ for Louisiana} \]

(TW unavailable from Texas).

Two thousand and fifty-nine otoliths were available and sectioned for age determination. Of those, nine were deemed unreadable by both readers 1 and 2 and were excluded from further analysis. Of the remaining 2050 sections, consensus was reached in the initial count on 1,841 (89.8%) individuals. A second reading produced consensus on 2,045 (99.8%) individuals. Red snapper in this study ranged in age from 1 to 45 years (Figure 3). Alabama fish ranged in age from 1 to 34 years; only 18 of the 772 fish (2.3%) aged from Alabama were over 15 years old. Louisiana red snapper ranged from 2 to 37 years with only 27 out of 712 fish (3.8%) over 15 years. Texas fish ranged from 2 to 45 years with only 4 out of 567 (0.7%) aged individuals over 15 years. Komolgorov–Smirnov two sample tests indicated no significant differences in age distributions between sexes or among sources.

Von Bertalanffy growth models to describe red snapper size at age were fitted for each state (Figure 4). The resultant models for FL at age are:

\[ \text{AL: } \text{FL(mm)} = 884[1 - e^{-0.21(t-0.6)}] \ (r^2 = 0.99) \]
\[ \text{LA: } \text{FL(mm)} = 873[1 - e^{-0.24(t-0.73)}] \ (r^2 = 0.99) \]
\[ \text{TX: } \text{FL(mm)} = 1023[1 - e^{-0.07(t+2.92)}] \ (r^2 = 0.98) \]

Predicted FL at age for red snapper from the three states generated with each of the models above illustrate rapid growth to an age of approximately 10 years after which an asymptote is approached and growth is in length is negligible.

Regression analysis of FL at age to evaluate growth of red snapper in the first ten years of life indicated significant differences among states (Figure 5). Alabama and Louisiana differed significantly for slopes (df = 1,1378; F = 2898.54; P < 0.0001) and intercepts (df = 1,1378; F = 30.29; P < 0.0001). Alabama and Texas also exhibited significant differences in both slopes (df = 1,1282; F = 1993.79; P < 0.0001) and intercepts (df = 1,1282; F = 72.01; P < 0.0001). Not surprisingly, Louisiana and Texas showed significant differences in slopes (df = 1,1188; F = 1711.34; P < 0.0001) and intercepts as well (df = 1,1188; F = 164.4; P < 0.0001).
Figure 3. Age frequency distribution for red snapper sampled from recreational sources in April to September 1999 in A) Alabama, B) Louisiana, and C) Texas.
Figure 4. von Bertalanffy growth models fit for red snapper sampled from recreational catches in April – September 1999 for A) Alabama, B) Louisiana, and C) Texas.
Figure 5. Linear regression of Log_{10} transformed fork length on age examining growth rates in the first ten years of life of red snapper sampled from Alabama (thin, solid black line), Louisiana (dotted black line), and Texas (thick gray line) in April through September 1999.

CONCLUSIONS

Several studies of red snapper age and growth mentioned herein have validated that otoliths do accrete one annulus per year during the winter and spring and can therefore be accurately used to determine age (Wilson et al. 1994, 1997, Render 1995, Manooch and Potts 1997, Patterson 1999). The pattern of winter-spring annulus formation is consistent with that reported for other long-lived fish in the northern Gulf of Mexico (Beckman et al. 1989, Beckman et al. 1990, Beckman et al. 1991). In addition, Baker (1999) used radiometric age estimates to validate the continued use of otolith sections as the best method to estimate age for red snapper.

The age distributions from this study are not representative of the red snapper population in the Gulf of Mexico. Due to minimum size limits on the recreational fishery, age 0 and age 1 snapper are not represented adequately in our sample population. The majority of fish sampled in all three states fall between the ages of 2 – 5 years (76% of aged fish) perhaps reflecting migratory aspects of red snapper life history. After migration from shallow waters, these fish reside around structures such as oil and gas platforms to seek refuge from large predators (Render 1995). Because these platforms harbor large populations of red snapper as well as other species (Stanley and Wilson 1996, 1998), they are preferred destinations of recreational and commercial fishermen. In addition, few old red snapper were sampled. Though large adults were targeted at fishing and dive tournaments, only
49 fish (2.9%) were found to be over 15 years old. The small numbers of older adults could be due to natural or fishing mortality, or due to emigration away from the platforms where they are less susceptible to capture.

The growth models in this study produced results similar to those in previous red snapper studies (Nelson and Manooch 1982, Szedlmayer and Shipp 1994, Patterson 1999). All show a rapid growth in the first ten years of life followed by a leveling of growth rate. The models’ fit for Alabama and Louisiana differ, however, from the Texas model in that they produced lower \( L_0 \) and higher \( K \) values. This may be an artifact of sampling. Both Alabama and Louisiana personnel sampled fishing tournaments targeting larger, older individuals. These fish pull the von Bertalanffy curve down producing a smaller maximum theoretical size. Goodyear (1995b) warned that non-random, or selective sampling such as fishing tournaments could introduce significant bias into growth functions. The lack of larger individuals in the sample population resulted in elevated \( L_\infty \) and \( t_0 \) values for Texas. Szedlmayer and Shipp (1994) reported an \( L_\infty \) of 1025. This value is likely due to a small sample size (\( n = 409 \)) in which only 11 fish were aged over 10 years old. The range in values of parameters derived from these various growth models are more a reflection of the size and age distribution of the sample population of each study than true growth effects of red snapper from different regions.

The regression analysis of FL at age indicated a difference in growth rates of red snapper among Alabama, Louisiana, and Texas in the first ten years of life. Although this difference was significant between all three states, red snapper from Texas clearly show a slower rate of growth than those from Alabama and Louisiana. Texas fish also display a smaller maximum size at 846 mm FL compared to 916 mm FL for Alabama and 913 mm FL for Louisiana red snapper. This slower rate of growth does not, however, have an affect on longevity as the oldest fish in this study (aged at 45 years) came from Texas waters. These differences in growth rates could be an indication of possible biologically meaningful management stocks within the fishery and could have profound affects on how the fishery is managed in the future. Should separate stocks exist, fishery units could be assessed and managed on a sub-regional basis, providing the opportunity to adjust regulations to the unique needs of sub-regional populations and resource users.

LITERATURE CITED


Análisis de la Pesquería de Huachinango (*Lutjanus campechanus*) en el Banco de Campeche

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RESUMEN

La pesquería de huachinango es una de las más importantes del estado de Yucatán, México. Se realizó una evaluación del estado actual de la pesquería y se simularon estrategias de manejo. Se utilizó un modelo de biomasa dinámico en tiempos discretos. Se hicieron proyecciones a partir de 1999 y se establecieron dos Punto de Referencia 1) contrastar la biomasa actual y futura con la biomasa inicial (riesgo del stock), para que la biomasa actual se mantenga por arriba del 50% de la biomasa inicial (Bo/2); y 2) mantener el Máximo Rendimiento Sostenible (MRS) (riesgo de la pesquería), en donde la mortalidad por pesca no sea mayor a $u_{MRS}$ (0.08/año). Los resultados indican una disminución de la biomasa a través del tiempo. Se estimó una biomasa inicial (1984) de 32,957 t y para 1999 una biomasa de 16,877 t, en los últimos 16 años la biomasa presentó un decremento del 51.2%. De acuerdo a los resultados del modelo se estimó el RMS en 1,271 t año$^{-1}$ y una taza de explotación de 0.04/año hasta alcanzar un valor de 0.15/año en 1992, año en que se registró una captura de 3,083 t. De esta, la flota yucateca capturó 1,334 t y la flota del estado de Campeche 1,749 t. Esto significa que ambas flotas capturaron en 1992, 1,850 t por arriba del MRS. En los últimos 5 años se estimó un valor promedio de la tasa de explotación de 0.08/año, con una captura promedio de 1,384 t. La serie histórica de la captura (1984 a 1999), siempre excedió el MRS y por consecuencia la biomasa presenta un fuerte decremento. Además, se aplicó un análisis de Monte Carlo para evaluar la probabilidad de alcanzar los puntos de referencia ante diferentes cuotas de captura.

PALABRAS CLAVES: Huachinango, *Lutjanus campechanus*, Campeche, Mexico

ABSTRACT

The red snapper fishery is one of the most important fisheries in Yucatan, Mexico. The current state of the fishery was evaluated with a biomass dynamic model in discreet times to assess the stock size and the yield. Management strategies were simulated, starting from 1999, and two options were proposed:

i) Contrast the current and future stock biomass with the initial biomass (risk of the stock), it must be 50% of the initial biomass (Bo/2), and

ii) Maintain the Maximum Sustainable Yield (risk of the fishery) using the exploitation rate $U_{MSY}$. 
The results indicate a biomass decline from an initial biomass of 32,957 tons (t) in 1984 to 16,877 t in 1999, 51.2% in the last 16 years. The model estimates that the exploitation rate increased from 0.04/year in 1984 to 0.15/year in 1992, when the yield was 3,083 t. The Yucatan fleet catch was 1,334 t, and the Campeche fleet catch was 1,749 t. Both fleets catches exceeded the MSY in those years. In the last five years the estimated exploitation rate averaged 0.08/year, or an average catch of 1,384 tons. The historical catch series show from 1984 to 1999, always exceeded the MSY and like consequence the biomass presents a decrease. Monte Carlo analysis was applied to evaluate the risk of reaching the points of reference with alternative quotas scenarios.

INTRODUCCION

En el estado de Yucatán la pesca de especies de escama (Serránidos, Lutjánidos, Spáridos, Haemulídos, etc.) ha sido la más importante, por su volumen de captura y el número de empleos que genera. El huachinango *Lutjanus campechanus* es una de las especies principales de esta pesquería, la cual por la calidad de su carne la mayor parte de su captura es de exportación (Estados Unidos), siendo generadora de divisas para el estado.

El Banco de Campeche una de las áreas de mayor abundancia de *Lutjanus campechanus*, ha sido objeto de una intensa pesquería comercial; en ésta han participado las flotas de Estados Unidos, Cuba y México (Camber 1955). Es un recurso que en México se captura todo el año. Actualmente participan en la pesca la flota mayor del estado de Yucatán, la flota cubana, la flota menor del estado de Campeche y la flota camaronera que capture juveniles de esta especie en forma incidental.

En la administración de los recursos pesqueros, los modelos matemáticos son una herramienta importante para evaluar el status y productividad de una población. Uno de los métodos más usados son los modelos de biomasa dinámicos. En su forma más simple, solo requieren de una serie de captura y un índice de la abundancia relativa (CPUE) para estimar los parámetros del modelo. De estas estimaciones pueden ser derivados dos parámetros de manejo: el máximo rendimiento sostenible (MRS) y el esfuerzo de pesca en el cual se alcanza el MRS ($F_{MSY}$). Estos modelos son ampliamente usados y más confiables que los modelos estructurados por edades cuando la información sobre la estructura por edades de la población es pobre o no existe, además de que pueden ser aplicados en situaciones donde solo existan datos de captura y un índice de la abundancia relativa (Punt y Hilborn 1996).

MODELO DINAMICO DE BIOMASA

Para evaluar el estado actual de la pesquería de huachinango (*Lutjanus campechanus*) en el Banco de Campeche y simular algunas estrategias de manejo,
se utilizó un modelo de biomasa dinámico, propuesto por Punt y Hilborn (1996). La información con la que se cuenta es una serie de capturas y un índice de la abundancia relativa (captura por unidad de esfuerzo, CPUE) entre 1984 a 1999.

El supuesto fundamental del modelo es que los efectos de los factores de crecimiento, mortalidad natural y reproducción, pueden ser incorporados en una sola función y ésta proporciona un solo valor del tamaño del stock. De esta forma la función de biomasa dinámica, determina el efecto neto en la combinación de estos factores a un tamaño particular del stock. En el cual, se asume que el cambio en el tamaño de la población de un año a otro, es la diferencia entre la biomasa dinámica y la captura generada por la pesquería. El planteamiento del modelo, propuesto por Punt y Hilborn (1996) es de la siguiente forma:

\[ B_{t+1} = B_t + g(B_t) - C_t \]  \hspace{1cm} (1)

\[ I_t = q B_t e^{r t} \]  \hspace{1cm} (2)

Donde \( B_t \) es la biomasa explotable al empezar el tiempo \( t \), \( g(B) \) es la función de la biomasa dinámica o del crecimiento poblacional, que toma la forma de Schaefer (1954): \( g(B) = rB (1-B/K) \), \( r \) es la tasa intrínseca de crecimiento de la población, \( K \) es la capacidad de carga (biomasa promedio antes de ser explotar), \( q \) es el coeficiente de capturabilidad, \( C_t \) es la captura en el tiempo \( t \), \( I_t \) es el índice de abundancia relativa en el tiempo \( t \) (CPUE) y \( \eta_t \) es el error de observación en el tiempo \( t \).

Para el ajuste del modelo, se consideró un estimador de error de observación (Ludwig y Walters 1985, Polacheck et al. 1993). Que asume que el modelo dinámico de la población, es determinístico y que el error ocurre entre la relación de la biomasa del stock y el índice de abundancia. La serie de tiempo de la biomasa del stock, es estimada proyectando la biomasa al iniciar la serie de captura, empezando con la biomasa inicial (\( B_0 \)) y continuando con toda la serie histórica de la captura. El error en el modelo observación se considera que es multiplicativo, con una distribución log-normal (v.g., \( I_t = q B_t e^{\eta_t} \), \( \eta_t \sim N(0; \sigma^2) \)). La estimación de los parámetros del modelo (\( B_0, r, \) y \( K \)) se obtuvieron minimizando una función de verosimilitud y se consideró que la biomasa inicial era igual a la capacidad de carga (\( B_0 = K \)):

\[ L(Bo,r,K/datos) = \prod \exp \left[-\frac{U_t^2}{2\sigma^2} \right] \ast \left[ \frac{1}{\sqrt{2\pi\sigma^2}} \right] \]  \hspace{1cm} (3)
Donde el producto es sobre todos los años para los cuales existan datos disponibles de la CPUE.

\[ \hat{u}_t = \log (\text{CPUE}_{\text{obs}, t}) - \log(\text{CPUE}_{\text{est}, t}) \]  

(4)

\[ \hat{\sigma}_u^2 = \frac{\sum \hat{u}_t^2}{n} \]  

(5)

\[ \text{CPUE}_{\text{est}} = q \hat{B}_t \]  

(6)

\[ q = \exp \left[ \frac{1}{n} \sum \log \frac{\text{CPUE}_{\text{obs}, t}}{\hat{B}_t} \right] \]  

(7)

Donde \( n \) es el número de datos, CPUE es la captura por unidad de esfuerzo, \( q \) es el coeficiente de capturabilidad que se obtiene minimizando la ecuación (3) con la siguiente ecuación.

RESULTADOS DE LA EVALUACION

La estimaciones de los parámetros de la población y de manejo de la pesquería, obtenidos de la evaluación se presentan en la Tabla 1. Los resultados indican una tendencia decreciente de la biomasa a través del tiempo, con una biomasa en 1999 alrededor de las de 16,877 t. Lo cual significa, que en los últimos 16 años la biomasa presentó un decremento alrededor del 50%. Esta disminución también se presentó en el índice de abundancia (CPUE), con un CPUE promedio entre 1984 a 1987 de 1,183 kg de huachinango / viaje, mientras que entre 1988 a 1999 fue de 713 kg de huachinango / viaje (Figura 1). Esto es un reflejo de que las capturas registradas entre 1984 a 1999, siempre excedieron el MRS.

La tasa de explotación se incrementó de 0.04/año hasta alcanzar un valor de 0.15/año en 1992, año en que se registró una captura de 3,083 t. De ésta, la flota yucateca capturó 1,334 t. y la flota del estado de Campeche 1,749 t. Esto significa que ambas flotas capturaron en 1992, 1,850 t por arriba del MRS. En los últimos cinco años se registró una captura promedio de 1,384 t con un promedio de 150 t por arriba del MRS y una tasa de explotación de 0.08/año. A partir de 1999 se hicieron proyecciones analizando tres diferentes cuotas de captura.
A partir de 1999 se hicieron proyecciones del comportamiento dinámico de la biomasa y la tasa de explotación al aplicar tres medidas de regulación; mantener la captura promedio de los últimos cinco años y disminuir e incrementar un 20% esta captura promedio (Tabla 2). Para lo cual se consideró un intervalo de simulación de 12 años, corto (2002), mediano (2005) y largo plazo (2011) (Figuras 2 y 3).

**Tabla 1.** Estimación de los parámetros de la población y de manejo pesquero, obtenidos con el modelo de biomasa dinámica de Schaefer (1954).

<table>
<thead>
<tr>
<th>Parámetros</th>
<th>Valor estimado</th>
</tr>
</thead>
<tbody>
<tr>
<td>Biomasa inicial (Bo)</td>
<td>33,048 (t)</td>
</tr>
<tr>
<td>Capacidad de carga (k)</td>
<td>33,048 (t)</td>
</tr>
<tr>
<td>Tasa de crecimiento poblacional (r)</td>
<td>0.16</td>
</tr>
<tr>
<td>Coeficiente de proporcionalidad (q)</td>
<td>3.6E-05</td>
</tr>
<tr>
<td>Máximo Rendimiento Sostenible (MRS)</td>
<td>1,271 t año⁻¹</td>
</tr>
<tr>
<td>Esfuerzo en el Máximo Rendimiento Sostenible (Eₘₛₛ)</td>
<td>2,039 / viajes año</td>
</tr>
</tbody>
</table>

**Figura 1.** Comportamiento dinámico de la biomasa del stock de huachinango (*Lutjanus campechanus*), en el Banco de Campeche e índice de la abundancia relativa (CPUE) observados y estimados por el modelo.

<table>
<thead>
<tr>
<th>Cuotas de captura</th>
<th>Probabilidad PRL ≤ Bo/2 (16,870 t)</th>
<th>Probabilidad PRL ≤ Bo/2 (16,870 t)</th>
<th>Probabilidad PRL ≤ Bo/2 (16,870 t)</th>
</tr>
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<tbody>
<tr>
<td>-2002</td>
<td>-2005</td>
<td>-2011</td>
<td>0.02</td>
</tr>
<tr>
<td>1,117</td>
<td>0.36</td>
<td>0.14</td>
<td>0.88</td>
</tr>
<tr>
<td>1,384</td>
<td>0.86</td>
<td>0.86</td>
<td>0.88</td>
</tr>
<tr>
<td>1,661</td>
<td>0.93</td>
<td>1.00</td>
<td>1</td>
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</table>

<table>
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<tr>
<th>Probabilidad PRL &gt; ( u_{\text{meta}} ) 0.08 año(^{-1} )</th>
<th>Probabilidad PRL &gt; ( u_{\text{meta}} ) 0.08 año(^{-1} )</th>
<th>Probabilidad PRL &gt; ( u_{\text{meta}} ) 0.08 año(^{-1} )</th>
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</thead>
<tbody>
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<td>1,117</td>
<td>0.00</td>
<td>0.00</td>
</tr>
<tr>
<td>1,384</td>
<td>0.96</td>
<td>1.00</td>
</tr>
<tr>
<td>1,661</td>
<td>1.00</td>
<td>1.00</td>
</tr>
</tbody>
</table>

Figura 2. Comportamiento y proyecciones de la dinámica de la biomasa del stock de huachinango (\textit{Lutjanus campechanus}), en el Banco de Campeche, ante diferentes cuotas de captura.
ANÁLISIS DE RIESGO

Se realizó un análisis de riesgo para evaluar el impacto de la pesquería al aplicar las cuotas de captura. Para lo cual se definieron dos Puntos de Referencia Limite, a) contrastar la biomasa actual y futura del stock con la biomasa inicial (riesgo del stock) de manera que la biomasa se mantenga por arriba del nivel de Bo/2 (16,870 t) y; b) mantener el Rendimiento Máximo Sostenible (riesgo de la pesquería), en donde la mortalidad por pesca no sea mayor a $u_{MRS}$ (0.08/año). Se aplicó un análisis de Monte Carlo, para poder expresar el riesgo en términos probabilísticos y se consideró como fuente de incertidumbre la biomasa inicial (Bo) con una distribución normal como función de densidad probabilística, se realizaron 500 corridas de simulación.

Los resultados indican que disminuir un 20% la captura promedio de los últimos 5 años (1,117 t/año) representa la menor probabilidad de alcanzar el PRL de una biomasa igual o menor a las 16,870 t (riesgo del stock), tanto en el corto, mediano y largo plazo. Las otras estrategias de manejo presentaron mayor riesgo de alcanzar el PRL, con mayor probabilidad la cuota de captura 1,661 t/año (Tabla 2).
En cuanto a la probabilidad de alcanzar el PRL, de que la tasa de explotación sea mayor a $u_{MR5}=0.08$ anual. Solo disminuyendo la captura registrada en la pesquería permitiría mantener una tasa de explotación menor o igual a 0.08 ($u_{MR5}$) y por lo tanto la cuota de captura de 1,117 t/año no presenta ninguna probabilidad de alcanzar este PRL. Así mismo, una cuota de captura de 1,661 t presenta mayor probabilidad de estar por arriba de este PRL (Tabla 2).

DISCUSION

Los modelos de simulación y el análisis de riesgo son una valiosa herramienta en la difícil tarea de evaluación de los recursos. Esto proporciona a los administradores mayor elementos en la toma de decisiones, que les permite evaluar las consecuencias de implementar una u otra estrategias de manejo. El modelo de biomasa dinámico utilizado para la pesquería de huachinango *Lutjanus campechanus* describió satisfactoriamente el sistema. Los resultados muestran una disminución en la biomasa vulnerable del stock y del índice de abundancia relativa de la población (CPUE). Esto puede ser el reflejo de que las capturas (1984 - 1999) siempre excedieron el RMS; a partir de 1993 se registró una caída paulatina en el rendimiento de la pesquería, tanto de las flotas de Yucatán como de Campeche.

Hay que mencionar que la captura de huachinango por la flota yucateca se realiza en zonas donde se distribuyen los organismos adultos. Esta flota se mantiene activa por la combinación de la captura de mero, huachinango y otras especies como pargos, coronado, mojarrones, cazones y pulpo que presentan un margen de ganancias. Sin embargo, es obvio que en el estado existe una sobrecapilazación; en 1999 de las 598 embarcaciones con permiso para pesca de escama, realizaron en promedio 308 viajes mensuales, que equivale a que sólo el 52 % de la flota estuvo activa. Lo que indica que los factores económicos están reajustándose y que se esperan tiempos difíciles para la actividad pesquera.

Respecto a la captura de las flotas de Campeche y principalmente el de la flota camaronería que es considerada una pesquería que captura una gran cantidad de juveniles de huachinango, ya que estos se distribuyen en las mismas zonas donde habita el crustáceo, se ha propuesto introducir excluidores de peces que tiene como fin la reducción de la captura de la fauna de acompañamiento. Sin embargo, la captura en el estado de Campeche ha disminuido a partir de los problemas que afronta la pesquería de camarón que ha obligado a establecer vedas a partir de 1993 y cada año son más prolongadas. Por otro lado, el esfuerzo de la flota menor se ha mantenido debido a la lejanía de la zona de pesca incrementa sus costos variables y requieren de instrumentos de navegación y ecodetección.

En México y en particular dentro de la Sonda de Campeche es probable que en la medida que el esfuerzo ejercido por la flota camaronería se contraiga, además de la salida de los barcos arrastreros que trabajaban dentro de la plataforma, se permitirá que los juveniles lleguen a tallas mayores, beneficiándose la flota yucateca a mediano plazo y probablemente se presente una recuperación en la población.
Para la pesquería de huachinango en el Banco de Campeche no existen medidas de regulación, los pescadores mexicanos solo deben de contar con un permiso oficial de la Secretaría del Medio Ambiente Recursos Naturales y Pesca (SEMARNAP). Sin embargo, dentro de un contexto de manejo precautorio de los recursos pesqueros, se sugiere que debido a que la pesquería de huachinango en el Banco de Campeche se encuentra ligeramente por arriba de su máximo rendimiento sostenible, es necesario establecer todas las medidas pertinentes para que el stock y la pesquería se mantengan en niveles que no sobrepasen los PRL, ya que de continuar pescando en estos niveles, se ocasionará un deterioro de la Biomasa, que puede llevar a una sobreexplotación del recurso. Tomando en cuenta la situación actual de la pesquería y los resultados obtenidos en el plan de manejo de esta especie en los Estados Unidos (Goodyear 1995, Baker et al. 1998), se hace necesario instrumentada una estrategia de manejo donde se integren diferentes medidas de regulación como: cuotas de captura, talla mínima y reducción de la captura incidental de la flota camaronera que opera en el Banco de Campeche.

LITERATURA CITADA


A Method for the Estimation of the von Bertalanffy Growth Rate Parameter by Direct Examination of Otolith Microstructure

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ABSTRACT
A method is introduced which uses otolith growth rate to provide a direct estimate of \(K\), the von Bertalanffy growth rate parameter. We obtained estimates of \(K\) of \(1.078 \pm 0.687 \ \text{y}^{-1}\) and \(0.621 \pm 0.076 \ \text{y}^{-1}\) for the queen snapper, *Etelis oculatus*, by two variations of the method. There is no significant difference between the two estimates of \(K\), or with an earlier value estimated with the use of ELEFAN 1. We suggest that this method may be used where otolith microstructure analysis does not allow for ageing the fish either by conventional counting of rings or even by analysing the width of growth increments. We acknowledge that from a statistical perspective, with one of the variations we may have violated the assumption regarding the independence of the observations used in the linear regression. Notwithstanding this, we propose that estimates of \(K\) by this general method can, at least, be a starting point or seed value for the use in other length-based methods.

KEY WORDS: *Etelis oculatus*, Growth rate parameter, otolith analysis

INTRODUCTION
Changes in otolith size, with few exceptions, are thought to closely reflect the somatic growth rate of fish, an observation which is of critical importance for studies of fish age and growth at the annual and daily levels of precision (Campana and Neilson 1985). Recently, workers have attempted to use the property of isometric growth with respect to somatic growth to develop approaches for investigations of age and growth which do not rely on conventional techniques for otolith microstructure examination. The latter techniques often involve complete enumeration of all increments along a standard axis of the otolith, a task which has proven difficult in older fish and in otoliths which have complex growth patterns.

Ralston and Miyamoto (1983) and Ralston (1984) have suggested that one can estimate the average width of daily increments at various points in the otolith, and
an estimate of this rate and total length of the otolith would allow one to estimate
the age of the specimen by dividing the otolith length by its rate of growth. It is
suggested (Ralston 1984) that this procedure works well if one takes into account
otolith and fish size. Ralston and Miyamoto (1983), while admitting that some of
the precision theoretically possible with daily increments is sacrificed in using this
method, have suggested that it provides a reasonably reliable estimate of age.

This essay suggests a method for directly calculating von Bertalanffy growth
rate parameter, K, based on a study of otolith growth rate for *Etelis oculatus* Val.
landed at Vieux Fort in the south of St. Lucia during 1987. Consistent with the
Ralston and Miyamoto (1983) method, we wished to develop an approach which did
not entail complete enumeration of growth increments along a given radius of an
otolith. Our suggested technique differs from that of Ralston and Miyamoto in that
a direct estimate of K is derived rather than an estimate of age. In the latter case,
further computations would be required to arrive at an estimate of K. The
occasional inability to detect all growth increments using a light microscope
(Morales-Nin 1988) means that otolith microstructure may not always give results
that can be used directly for obtaining age-at-length. Thus, if the researcher is only
able to easily access light microscopy, as may often be the case in developing
countries, a reasonable estimate of the growth rate parameter can nonetheless be
obtained.

METHODS

The Gulland and Holt plot (Gulland and Holt 1959) is a length-based method
of von Bertalanffy growth parameter estimation which provides estimates of L∞ and
K through the feature that the difference between successive lengths of a fish, when
divided by the difference in the corresponding ages can be plotted as a straight line
against the mean of the successive lengths, with the modulus of he slope of the line
being equal to K, L∞ being the ratio between the y-intercept and the modulus of the
slope. The method finds use when continuous growth lines cannot be traced, or
when only unequal time intervals (for example, those that may be obtained from
tagging and recapture data) are available (Pauly 1983).

Measurements taken for the Ralston and Miyamoto method on otolith thin
sections (Figure 1) are used in a "quasi-Gulland and Holt" plot (after Murray 1989)
wherein the regression of the otolith growth rate, assuming increments to be daily,
for a given segment of the otolith against the distance of the mid-point of that
segment to the focus of the otolith is calculated. This is done for all otoliths where
at least two segments had been measured and where it could be seen that the otolith
growth rate decreased with increasing distance from the nucleus. The absolute value
of the slope of this curve is considered to be equal to K for that fish (*ibid*). Initially,
the estimates of K were averaged over all the individual otolith determinations. For
comparison, the regression was also done with the segments from all the otoliths (in
other words, all nine fish) pooled into one regression. The two estimates of K thus
obtained were in days⁻¹, and converted to years⁻¹ by multiplying by 365.25.
RESULTS

Table 1 is a summary of the otolith growth rate analysis for *Etelis oculatus* (after Murray, 1989). When data from the otolith thin sections of *E. oculatus* were used individually and the mean value of K, at the 95% confidence level calculated, the value obtained was 1.078 ± 0.687/year. The value obtained when the segments from all the otoliths were pooled (Figure 2) was 0.621 ± 0.076/year (dL/dt = 5.738 - 0.0017 L; n = 34; K = 0.0017 x 365.25). There is no significant difference (t_{calc} = 1.627; t_{0.05(2,36)} = 2.447) between the two estimates of K. The mean value of K estimated from the individual regressions is also not significantly different (t_{calc} = 1.310; t_{0.05(2,36)} = 2.447) from the average of 0.71/year obtained (Murray, 1989) with the use of ELEFAN I (Gayanilo et al. 1988).

DISCUSSION

The "quasi-Gulland and Holt" plot may be a way of estimating K where otolith microstructure analysis does not allow for either conventional counting of rings or when the growth marks are not seen clearly enough for use of the Ralston and Miyamoto method (1983). Such a circumstance may arise where, having attempted one of the "usual" methods it is found there is, for example, unevenness in the incremental plane such that increments cannot be observed clearly all along the otolith radius being used. This method can also be used where the data available for input into one of the length-based methods such as the ELEFAN I, Shepherd (1987), Ebert (1987), or Damm (1987) methods are such that there would be difficulty deciding among (say) multiple maxima, or was otherwise not of a quality that would allow them to be used without some type of "seed" value with which to start. If a few otoliths are available, our suggested method could then provide an estimate of K to serve as such a starting point or seed value.

Like other techniques of otolith microstructure study, our method may be considered to be limited by the assumption as to the periodicity of increment deposition. Additionally, if growth fluctuates on a seasonal basis, results may be biased if increments were selected from one part of the seasonal cycle only. It would seem likely that the period of faster growth, and hence more noticeable increments, could be chosen, thus leading to an overestimation of the growth rate parameter. To avoid this, it would be advisable to choose the segments at random.
Figure 1. Photomicrograph of sagitta of *Etelis oculatus*.

Figure 2. Graph showing otoliths segment growth rate versus nucleus to mid-segment distance.
### Table 1. Summary of growth rate analysis for *E. oculatus*. Individual estimates of growth rate parameter are shown.

<table>
<thead>
<tr>
<th>Table</th>
<th>Length (mm TL)</th>
<th>Otolith length (µm)</th>
<th>Otolith width (µm)</th>
<th>Otolith growth rate (µm day⁻¹)</th>
<th>Nucleus to mid-segment distance (µm)</th>
<th>Diameter of otolith (µm)</th>
<th>Ring diameter (µm)</th>
<th>Ring number</th>
<th>Ring growth rate (µm day⁻¹)</th>
<th>Diameter of ring (µm)</th>
</tr>
</thead>
<tbody>
<tr>
<td>0.546</td>
<td>577</td>
<td>2390</td>
<td>3216</td>
<td>763</td>
<td>1.714</td>
<td>5.333</td>
<td>48</td>
<td>9</td>
<td>0.539</td>
<td>1.674</td>
</tr>
<tr>
<td>0.745</td>
<td>577</td>
<td>2390</td>
<td>3216</td>
<td>763</td>
<td>1.714</td>
<td>5.333</td>
<td>48</td>
<td>9</td>
<td>0.539</td>
<td>1.674</td>
</tr>
<tr>
<td>8.867</td>
<td>572</td>
<td>192</td>
<td>432</td>
<td>763</td>
<td>1.714</td>
<td>5.333</td>
<td>48</td>
<td>9</td>
<td>0.539</td>
<td>1.674</td>
</tr>
<tr>
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<td>1854</td>
<td>3564</td>
<td>542</td>
<td>1.723</td>
<td>5.600</td>
<td>10</td>
<td>6</td>
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<td>8.000</td>
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*Note: The table continues with similar data entries for different lengths and parameters.*
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</tr>
<tr>
<td>43 180 4.186 1.432 1824</td>
</tr>
<tr>
<td>45 98 2.133 0.758 1896</td>
</tr>
<tr>
<td>30 48 1.600 0.470 1920</td>
</tr>
<tr>
<td>635 1475</td>
</tr>
<tr>
<td>20 72 3.600 1.281 840 0.537</td>
</tr>
<tr>
<td>14 60 4.285 1.455 900</td>
</tr>
<tr>
<td>23 120 5.217 1.852 972</td>
</tr>
<tr>
<td>19 48 2.526 0.927 1224</td>
</tr>
<tr>
<td>9 24 2.666 0.981 1392</td>
</tr>
<tr>
<td>54 132 2.444 0.894 1920</td>
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<tr>
<td>20 24 1.200 0.182 2040</td>
</tr>
<tr>
<td>17 80 3.529 1.261 2136</td>
</tr>
<tr>
<td>632 2145</td>
</tr>
<tr>
<td>33 156 4.727 1.552 156 0.489</td>
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<tr>
<td>25 84 3.360 1.212 720</td>
</tr>
<tr>
<td>20 80 3.000 1.099 792</td>
</tr>
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<td>Table 1 continued:</td>
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<tr>
<td>-------------------</td>
</tr>
<tr>
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</tr>
<tr>
<td>20  60  3.000  1.099  1688</td>
</tr>
<tr>
<td>36  60  1.667  0.511  1660</td>
</tr>
<tr>
<td>783  2090  14  60  2.609  1.455  1500  0.555</td>
</tr>
<tr>
<td>17  36  2.118  0.750  1824</td>
</tr>
<tr>
<td>783  2755  11  36  3.272  1.186  1615  0.416</td>
</tr>
<tr>
<td>52  120  2.308  0.836  1710</td>
</tr>
<tr>
<td>58  168  2.897  1.604  1957</td>
</tr>
<tr>
<td>17  42  2.471  0.805  2052</td>
</tr>
<tr>
<td>35  72  2.057  0.721  2090</td>
</tr>
<tr>
<td>130  231  1.777  0.575  2660</td>
</tr>
</tbody>
</table>

Key:  - used in pooled version of plot, but not for calculation of individual K values
(-) not used in calculations of mean K value from individual calculations, neither used to calculate "pooled"
K value: obvious outlier
It should be noted that Ralston and Miyamoto (1983) attempt to derive an absolute—the age of the individual fish. In such circumstances, precise and accurate knowledge of increment deposition rates are more crucial than for the estimation of a population parameter that is itself a mean value having a finite variability. Further, in this method the value used to estimate K is a slope and hence it is only if the time represented by one increment varies within any given otolith, that one would expect significant changes in the value. Our method has analogies to length-based approaches; in fact, the very name "quasi-Gulland and Holt plot" is suggestive of that fact. This implies that the theory derived for length-based methods could apply to this method. Thus, we note Isaac's (1990) suggestion that at least three length-based methods give accurate estimates of K when individual variability of this growth parameter is small (<20%). Applying her observation to this method, the smaller variability, and therefore greater precision, of the growth rate parameter estimate derived by our pooled-segment determination makes it the preferable form of the method. We suggest that in this form, the method also provides a population estimate for K.

Given the possibility of underestimating the periodicity of increment deposition, a corresponding overestimate of K would not be surprising. The estimate of K obtained with our proposed pooled segment approach technique are comparable with Murray's (1989), but are among the highest recorded by Manooch (1987) in his comprehensive review of age and growth studies of tropical snappers and groupers. Manooch's (ibid.) review included values of K for snappers (but not including any estimate for Etelis sp.) ranging from 0.090 - 0.370/year. A more recent determination of K = 0.40/year for queen snapper has been made by Murray and Moore (1992).

The large, though not statistically significant, difference between our two estimates is noteworthy. This may in part be related to the high variance associated with the estimate obtained when using the mean of K values calculated from the nine otoliths. We acknowledge that from a statistical perspective we violate an assumption of linear regression with the approach using all individual segments, since they are not truly independent observations. The approach of using one datum from each fish may be better in this regard.

Consideration of the strengths and weaknesses of the two variations on the "quasi-Gulland and Holt" approach also begs the question of whether, to obtain an estimate of K that is representative over the life of the fish, the data collection should be length-stratified in some manner, or whether length segments should be collected at consistent points along the standard radius.

There is also now a need to confirm our estimates of K for queen snapper and to test the pooled segment form of our proposed approach with a species where K is well known from other approaches.
ACKNOWLEDGMENTS

The authors would like to thank Mr. Boris Fabres, previously Senior Biologist at the CARICOM Fisheries Resources Assessment and Management Program's Pelagic, Reef, and Slope Fishes Resource Assessment Unit, and Mr. Raymond Ryan of the Fisheries Division, Ministry of Agriculture and Labour, both in St. Vincent and the Grenadines, for their valuable comments on an earlier version of this contribution.

LITERATURE CITED


Indirect Estimation of Red Snapper (*Lutjanus campechanus*) and Gray Triggerfish (*Balistes capriscus*) Release Mortality

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ABSTRACT

Release mortality was estimated indirectly for red snapper and gray triggerfish in the northern Gulf of Mexico from condition of tagged fish. Condition at release was assessed for tagged red snapper (n = 2,932) and gray triggerfish (n = 842) based upon their swimming behavior. Condition-1 was assigned to fish that oriented toward the bottom and swam down vigorously. Fish in condition-2 oriented toward the bottom but swam erratically. Fish in condition-3 swam very erratically and remained at the surface, while fish in condition-4 were unresponsive and presumed dead. Overall, 86.5% of red snapper and 99.2% of gray triggerfish were released in condition-1. Depth of capture, fish size, and transporting fish prior to release significantly affected red snapper release condition (*p* < 0.01). Of 550 recaptured red snapper, 98.4% were released in condition-1, while 100% of 160 recaptured gray triggerfish were released in condition-1. Release condition significantly affected recapture rate for red snapper (*p* < 0.001). Comparisons with reported estimates of release mortality for red snapper indicate that the cumulative percentage of fish released in conditions other than condition-1 may serve as a proxy for acute release mortality. Further research is required to validate this approach; however, it may prove to be a practical method to evaluate release mortality in recreational and commercial fisheries through observer programs.

KEY WORDS: Reef fish, release mortality, tagging

INTRODUCTION

Reef fishes support valuable commercial and recreational fisheries in the U.S. south Atlantic and Gulf of Mexico (GOM), but many economically and ecologically valuable species in the snapper/grouper complex currently are estimated to be overfished (Anon. 1998, Coleman et al. 1999, 2000, Collins et al. 1999). Management of these reef fisheries is complicated by fish life history characteristics that include slow growth, late maturity, high site fidelity, and protogyny (Coleman et al. 1999, 2000, Huntsman and Waters 1987, Parish 1987). Furthermore, reef fisheries in a given geographic area typically target a variety of species, thus, traditional single-species management is often difficult or untenable (Ault et al. 1997;

Red snapper (*Lutjanus campechanus*) support the most economically valuable fishery in the GOM snapper/grouper complex, and are among the most targeted fishes by GOM commercial and recreational fishers (Goodyear 1995, Minton and Heath 1998, Stanley and Wilson 1990). This species is a long-lived apex predator that ranges from North Carolina to Florida in the U.S. south Atlantic, and from Florida to Yucatan in the GOM (Hoese and Moore 1998). The GOM red snapper stock has been subjected to heavy fishing pressure since the 1800s and is currently estimated to be severely overfished (Goodyear 1995, Schirripa and Legault 1999). Management measures first were implemented in the late 1980s, and have evolved since then, to increase red snapper spawning stock biomass (GMFMC 2000) (Table 1). Despite overfished status, combined commercial and recreational landings of GOM red snapper averaged (± SE) 3.26 (± 0.35) x 10^3 metric tons from 1990 to 1998 (Figure 1). Also during this time period, estimates of recreational discards rose sharply (Figure 1), likely in response to increasing management regulations (Table 1).

Table 1. History of management regulations for recreational and commercial red snapper fisheries in the GOM since 1990. Recreational size limit in 1999 changed to 457 mm TL from June through August to extend the recreational season. Length of 2000 commercial season is projected.

<table>
<thead>
<tr>
<th>Year</th>
<th>Recreational</th>
<th>Commercial</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Size Limit mm TL</td>
<td>Bag Limit</td>
</tr>
<tr>
<td>1990</td>
<td>330</td>
<td>none</td>
</tr>
<tr>
<td>1991</td>
<td>330</td>
<td>7</td>
</tr>
<tr>
<td>1992</td>
<td>330</td>
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<td>1993</td>
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<tr>
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<td>1999</td>
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<td>4</td>
</tr>
<tr>
<td>2000</td>
<td>406</td>
<td>4</td>
</tr>
</tbody>
</table>
Figure 1. Reported recreational and commercial landings of red snapper in the U.S. GOM. Estimated numbers of live recreational discards are also given (Source of data: Schirripa and Legault (1999)).

Declining abundances of, as well as increasing management regulations on, red snapper and other overfished GOM reef fishes, such as gag grouper (Mycteroperca microlepis) and red grouper (Epinephelus morio), appear to have caused a shift in fishing effort to previously less exploited species (Ault et al. 1997, Schirripa 1996). Gray triggerfish (Balistes capriscus) experienced an increase in landings in the late 1980s and early 1990s, followed by a steady decline (Figure 2). Similar trends were observed for GOM vermilion snapper (Rhomboptilus aurorubens) (Schirripa 1996), which is now considered to be approaching an overfished condition (Anon. 1998). The status of the GOM gray triggerfish stock is unknown, but concern over trends in landings of gray triggerfish and other reef fishes prompted the Gulf of Mexico Fishery Management Council (GMFMC) in 1995 to create a 20-fish aggregate recreational bag limit for reef fishes not covered by species-specific bag limits (GMFMC 1995). Most recently, the GMFMC introduced a 305 mm total length (TL) size limit for gray triggerfish (GMFMC 1999).

Traditionally, fisheries management measures such as size limits, fishing quotas, and seasonal closures have been used by managers in the southeastern U.S. and GOM to limit fishing mortality on reef fish resources; however, protected size ranges or species may incur incidental mortality due to hooking and release while fishers target legal-sized fish or other species (Collins et al. 1999, Gitschlag and Renaud 1994, Schirripa and Goodyear 1994). High rates of incidental catch coupled with high release mortality rates can negate potential gains of conservation measures (Goodyear 1995, Murphy et al. 1995, Schirripa and Goodyear 1994). As part of tagging studies conducted in the northern GOM off the coast of Alabama, we
indirectly estimated release mortality of tagged red snapper and gray triggerfish based upon visual assessment of their condition at the surface following release. The need for further validation of our approach is discussed, as well as its potential use for estimating release mortality through fishery observer programs.

![Graph showing Rec Landings and Comm Landings](image)

Figure 2. Reported U.S. GOM recreational and commercial landings of gray triggerfish from 1986 to 1996. (Source of data: Skip Lazauski, Alabama Department of Conservation, Marine Resources Division, Gulf Shores, AL 36524)

METHODS

Red snapper and gray triggerfish were tagged and released over reef sites off the coast of Alabama during two separate tagging studies. Red snapper were tagged over nine artificial reef sites on 28 tagging trips from March 1995 to July 1998 (Figure 3). Three sites each were located at water depths of 21, 27, and 32 m. Gray triggerfish were tagged over two natural hardbottom sites on 24 tagging trips from July 1997 to March 2000 (Figure 3). Gray triggerfish tagging sites were located at water depths of 24 and 35 m.

Tagging trips were made aboard a chartered fishing boat and methods were the same for each study except where noted below. Fish were captured over reef sites with rod and reel fished with straight-shank barbed hooks (3/0 size for red snapper and 1/0 size for gray triggerfish). Once hooked, fish were retrieved slowly from the bottom (approximate rate = 0.5 m/s) and placed in holding tanks with running seawater. Individuals were measured (TL and fork length (FL)) and then tagged by inserting an internal anchor tag through a small (<5mm) incision made with a scalpel in the abdominal cavity. Tags were yellow Flory FM-89 anchor tags and each was marked with a tag number, the word “reward,” and a phone number for fishers to
report tag recoveries. In each study, a $5 reward was offered for tag returns, with a chance to win $500 in a drawing of tag returners.

Once tagged, red snapper were either released immediately overboard or transported (up to 0.5 hours) in holding tanks to other tagging sites for release. All gray triggerfish were released over their site of capture. Condition of tagged fish was assessed visually based upon their swimming behavior at the surface (Table 2). The effect of fish size, season of capture (winter, spring, summer, fall), depth of capture, and transportation prior to release on red snapper release condition was tested with logistic regression (SAS, Inc. 1996). In the model, fish released in condition-1 were coded as a one and fish released in other conditions were coded as zeroes.

Figure 3. Map of region off the coast of Alabama where tagging studies were conducted. Red snapper tagging sites (labeled 1-9) and gray triggerfish tagging sites (labeled G1 and G2) are shown.
Tagged fish from both studies were recaptured at tagging sites on subsequent tagging trips and were reported by recreational and commercial fishers (through October 2000). The effect of release condition on red snapper recapture rate was tested with contingency table analysis (SAS, Inc. 1996).

Table 2. Release conditions for tagged red snapper and gray triggerfish.

<table>
<thead>
<tr>
<th>Condition</th>
<th>Characteristics Displayed by Fish</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Fish oriented toward the bottom and swam down vigorously.</td>
</tr>
<tr>
<td>2</td>
<td>Fish appeared disoriented upon entering the water, oriented toward the bottom, but swam erratically.</td>
</tr>
<tr>
<td>3</td>
<td>Fish appeared very disoriented upon entering the water and remained at the surface.</td>
</tr>
<tr>
<td>4</td>
<td>Fish was either dead or unresponsive upon entering the water.</td>
</tr>
</tbody>
</table>

RESULTS

Two thousand two hundred thirty-two red snapper were tagged; 879 were transported prior to release and 2,053 were not. Mean TL (± SE) of tagged individuals was 335.1 (± 1.34) mm. Overall, 86.5% of tagged red snapper were released in condition-1 (Table 3). The logistic regression computed on release condition was significant ($\chi^2_{df=4} = 109.9, p < 0.001$). Depth of capture ($\chi^2_{df=1} = 15.95, p < 0.001$), fish size ($\chi^2_{df=1} = 8.52, p = 0.004$), and transportation prior to release ($\chi^2_{df=1} = 53.39, p < 0.001$) all significantly affected whether tagged fish were released in condition-1; season was not significant ($\chi^2_{df=1} = 1.20, p = 0.273$). The probability that fish were released in condition-1 significantly decreased with increasing depth, transportation prior to release, and decreasing fish size. Five hundred ninety-four recaptures were made of 550 tagged red snapper (Table 4); 42 fish were recaptured twice and one fish was recaptured three times. Two hundred thirty-five recaptures were made on tagging trips and 359 recoveries were reported by fishers. Of the 550 recaptured individuals, 98.4% were released in condition-1 at tagging. Release condition significantly affected red snapper recapture rate (contingency table analysis: $\chi^2_{df=1} = 74.3, p < 0.001$).

Eight hundred forty-two gray triggerfish were tagged; 589 fish were tagged at the 24 m depth site (G1) and 253 were tagged at the 35 m depth site (G2). Mean FL (± SE) of tagged individuals was 297.1 (± 1.39) mm. Overall, 99.2% of gray triggerfish were released in condition-1. One fish in condition-2 and two fish in condition-3 were released at site G1, and one fish in condition-2 and three fish in
condition-3 were released at site G2.

**Table 3. Number and percentage of tagged red snapper released in condition-1**

<table>
<thead>
<tr>
<th>Transport</th>
<th>Depth of Capture Site</th>
<th>Overall in Condition-1</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>21 m</td>
<td>27 m</td>
</tr>
<tr>
<td>Fish Transported</td>
<td>91.1%</td>
<td>79.8%</td>
</tr>
<tr>
<td></td>
<td>(113 of 124)</td>
<td>(280 of 351)</td>
</tr>
<tr>
<td>Fish not Transported</td>
<td>91.1%</td>
<td>91.1%</td>
</tr>
<tr>
<td></td>
<td>(856 of 940)</td>
<td>(460 of 505)</td>
</tr>
<tr>
<td>Mean in Condition-1</td>
<td>91.1%</td>
<td>86.4%</td>
</tr>
<tr>
<td></td>
<td>(968 of 1,064)</td>
<td>(740 of 856)</td>
</tr>
</tbody>
</table>

**Table 4. Number and percentage of tagged and recaptured red snapper released in each condition. Percentages are of total sample size (n = 2,932).**

<table>
<thead>
<tr>
<th>Red Snapper Release Condition</th>
<th>Tagged Red Snapper not Recaptured</th>
<th>Recaptured Red Snapper</th>
<th>Total</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>1,998</td>
<td>541</td>
<td>2,537</td>
</tr>
<tr>
<td></td>
<td>88.1%</td>
<td>18.5%</td>
<td>86.6%</td>
</tr>
<tr>
<td>2</td>
<td>152</td>
<td>5</td>
<td>157</td>
</tr>
<tr>
<td></td>
<td>5.2%</td>
<td>0.17%</td>
<td>5.4%</td>
</tr>
<tr>
<td>3</td>
<td>134</td>
<td>3</td>
<td>137</td>
</tr>
<tr>
<td></td>
<td>4.6%</td>
<td>0.10%</td>
<td>4.7%</td>
</tr>
<tr>
<td>4</td>
<td>100</td>
<td>1</td>
<td>101</td>
</tr>
<tr>
<td></td>
<td>3.4%</td>
<td>0.03%</td>
<td>3.3%</td>
</tr>
<tr>
<td>Total</td>
<td>2,382</td>
<td>550</td>
<td>2,932</td>
</tr>
<tr>
<td></td>
<td>81.3%</td>
<td>18.7%</td>
<td>100.00%</td>
</tr>
</tbody>
</table>

One hundred ninety-six recaptures were made of 160 gray triggerfish. One hundred seventy-four recaptures were made on tagging trips and 22 recoveries were reported by fishers. Nineteen fish were recaptured twice, seven fish were recaptured three times, and one fish was recaptured four times. All recaptures were released.
at tagging in condition-1.

DISCUSSION

Release condition of gray triggerfish generally was assessed to be excellent, as most tagged fish disappeared from sight immediately upon entering the water. No statistical tests were performed on gray triggerfish condition because nearly all fish were released in condition-1, and 100% of recaptured individuals were released in condition-1. The high return rate at tagging sites infers that gray triggerfish display high site fidelity. Moreover, multiple recaptures of some individuals provide further evidence that capture and release, as well as tagging, did not significantly affect the apparent health of gray triggerfish.

For red snapper, condition at release was significantly lower for fish transported prior to release. While it is unlikely that commercial and recreational fishers would transport fish prior to releasing them, this result illustrates that increased handling of fish decreased their condition. That release condition was significantly lower for smaller fish has important implications for size limits that are designed to protect young fish from fishing mortality. This result suggests that small fish may be more vulnerable to release mortality than larger fish; however, Gitschlag and Renaud (1994) reported there was no significant difference in red snapper release mortality estimates between fish smaller and larger than 30 cm TL. Lastly, decreasing release condition with increasing depth reflects trends in release mortality reported by Gitschlag and Renaud (1994) for red snapper and by Wilson and Burns (1996) for red grouper and scamp (Myceteroperca phenax).

Gitschlag and Renaud (1994) reported that estimates of red snapper release mortality rates inferred from the condition of released fish at the surface were similar to mortality estimates from in situ caging experiments conducted at similar depths. They and Render and Wilson (1994) reported that most of the mortality suffered by red snapper held in cages or laboratory tanks occurred soon after capture. Render and Wilson (1994) also reported there was no significant difference in mortality between tagged and untagged individuals. Szedlmayer and Shipp (1994) held 30 tagged red snapper in laboratory tanks for six months with no mortality or signs of infection, implying individuals that did not suffer acute tagging mortality (i.e., fish released in condition-1) likely would not suffer chronic injuries leading to death. Patterson et al. (in review) reported that fish tagged in the present study grew at similar rates as otolith-aged individuals, which also implies no chronic effects of tagging.

Based on these studies, and the fact that nearly all recaptures of tagged red snapper in the present study were of fish released in condition-1, we hypothesize that the cumulative percentage of tagged fish released in conditions other condition-1 may serve as a conservative (indirect) estimate of release mortality. Following this logic, estimated release mortality for gray triggerfish would be less than 1%. Overall, estimated release mortality for tagged red snapper would be 13.5%, which is within the range of release mortality estimates reported from previous studies.
Render and Wilson (1994) reported an overall release mortality rate of 20% for red snapper caught off Louisiana. Gitschlag and Renaud (1994) presented data from several studies of red snapper release mortality, and mean mortality rate (± SE) was 14% (± 4.5) for studies (n = 7) that were conducted at depths similar to the present study.

Our results, as well as those of Gitschlag and Renaud (1994), indicate that release condition of fish at the surface may serve as proxy for release mortality. Controlled experiments are needed, however, to validate that release condition of individual fish correlates to release mortality, especially over greater depth and fish size ranges than those of the present study. In particular, the likelihood of survival for fish that appear in poor condition at release and the likelihood of mortality for fish that appear healthy need to be examined.

Accurate estimation of discard rates and release morality rates is critical to fishery managers that must consider the effect of release mortality on stock-specific production while attempting to maximize yield per recruit (Goodyear 1995, Schirripa and Goodyear 1994). If it can be shown that release condition accurately approximates survival potential of individual fish, then the approach presented here may have broader applications for indirectly estimating release mortality. For example, observer programs could be established to estimate discard rates and release mortality of fish caught incidentally and released by recreational and/or commercial fishers.

ACKNOWLEDGMENTS

Funding for this study was provided by the United States’ National Oceanographic and Atmospheric Administration through MaRFIN grants NA57FF0054 and NA77FF0547 to R.L.S. The authors thank Mike and Ann Thierry for deploying artificial reefs used in this study and for allowing us to tag fish onboard their charter fishing vessel Lady Ann. The authors acknowledge a great debt to all volunteer fishers who helped in the tagging process and to all commercial and recreational fishers who reported tag recoveries.

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