

GUIDELINES FOR DEVELOPING A
POTENTIAL BIOLOGICAL REMOVAL (PBR)
FRAMEWORK FOR MANAGING SEA TURTLE
BYCATCH IN THE PAMLICO SOUND
FLOUNDER GILLNET FISHERY

by

Nancy Young

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Abstract

Bycatch of sea turtles is a serious management problem in the Pamlico Sound flounder gillnet fishery. Three species of sea turtles are caught in this fishery and are protected under the Endangered Species Act. In this paper, I argue that the current scheme for managing sea turtle bycatch under the Endangered Species Act is flawed, and I suggest adapting a mechanism used for marine mammals, the potential biological removal (PBR) approach, for use in sea turtles.

PBR is a useful approach for calculating allowable incidental take levels. It is a simple, data-driven formula requiring a minimum amount of data. It is conservative in that it uses minimum population estimates and a recovery factor based on the population status, addresses data uncertainty in a straightforward way, and follows the precautionary principle with more conservative management when data are less precise. PBR is also comprehensive because it calculates total take per stock. Finally, it was developed as the result of extensive modeling that evaluated the performance of the decision rule based on explicit risk thresholds.

Because PBR was developed for marine mammals, the approach and equation must be modified before it can be used for sea turtles. I describe these modifications, and list methods and sources for obtaining the necessary data.

There are no fundamental data limitations to calculating PBR for sea turtles, but political resistance may prove a more difficult challenge. Without clear incentives for implementing a PBR framework, it will be difficult to move beyond the status quo. But by providing an example of how PBR would work, as I have done here for the Pamlico Sound flounder gillnet fishery, this approach may gain support. As Biological Opinions and jeopardy decisions are increasingly challenged, managers and other stakeholders may see the advantages of PBR and provide the pressure necessary to change the current sea turtle bycatch management scheme.

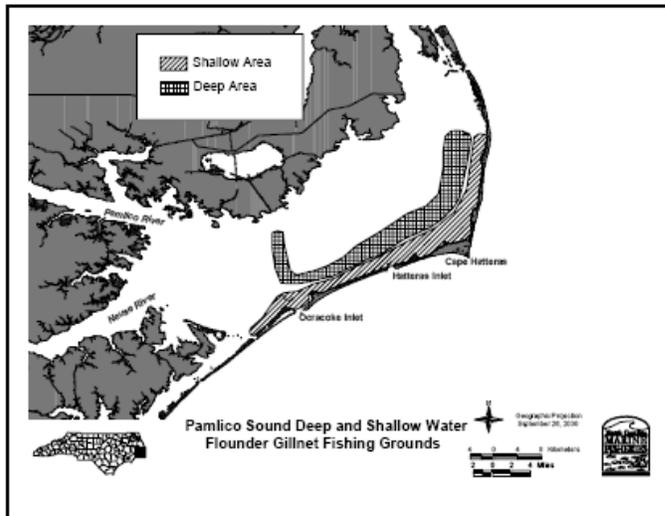
Introduction

North Carolina's Pamlico Sound is an important nursery ground for the commercially valuable southern flounder (*Paralichthys lethostigma*). In this area a commercial gillnet fishery exploits this resource, but non-target species, including sea turtles, are often caught as well. The North Carolina Division of Marine Fisheries (NCDMF), in consultation with the National Marine Fisheries Service (NMFS), manages the fishery with the goal of minimizing interactions (takes) with sea turtles. In this paper, I argue that the current scheme for managing sea turtle bycatch is flawed, and I suggest adapting a mechanism used for marine mammals, the potential biological removal (PBR) approach, for use in sea turtles.

Description of the fishery

Flounder fisheries peak from September through December as the flounder migrate out of the sounds and estuaries into the ocean to spawn (NCDMF 2005). From 1994-2003, more than fifty-five percent of all inshore landings of flounder in North Carolina were caught with large mesh gillnets (Pate 2005). In recent years, two large-mesh (>5" stretched mesh) gillnet flounder

Figure 1 Flounder gillnet fishing areas in the southeastern Pamlico Sound, North Carolina (NCDMF 2005).



fisheries have operated in Pamlico Sound (Figure 1) (Gearhart 2002). One is a shallow water fishery that occurs along the sound-side of the Outer Banks. Fishermen in small open skiffs set 500-2,000 yards of 5.5"-7.0" (stretched) mesh in water less than 3 feet deep, and the nets soak overnight. The second is a deep water fishery that occurs further from shore in the main basin of Pamlico Sound. The shallow

water fishery is more traditional, but the deep water fishery developed approximately 10 years ago and expanded steadily until its closure in 2001. In the deep water fishery, fishermen use larger boats to set 2,000-5,000 yards of 5.5"-6.5" (stretched) mesh in 10-20 feet of water. These nets can be left to soak for up to three days.

Several other gillnet fisheries operate in the area. A large mesh flounder fishery operates along the shorelines of mainland Hyde and Pamlico counties. Fishermen set 500-2,000 yards of 5.5"-7.0" mesh in the shallow waters (>3 feet) within 200 yards of shore, and soak them overnight. Another is the small mesh (<5" stretched) gillnet fishery which targets other species. Fishermen operate both "runaround" and "set" gillnets to target striped mullet, spotted seatrout, weakfish, and bluefish (Pate 2005).

The southern flounder is North Carolina's most economically valuable finfish (Bianchi 2003; Burgess and Bianchi 2004). Flounder comprised nearly 30% of the almost \$39 million ex-vessel value of North Carolina's finfish landings in 2004 (NCDMF 2004). A preliminary analysis of the 2004 Pamlico Sound fall gillnet fishery shows that approximately 410,000 pounds of flounder was landed, worth nearly \$700,000 wholesale and \$3 million retail (Pate 2005; NCDMF 2004). There are over 1,000 participants in flounder gillnet fisheries in the Albemarle, Pamlico, Rivers, and Southern areas, and between 132 and 209 permits have been issued for the managed areas in southeastern Pamlico Sound, of which 70% are actively fished each year (NCDMF 2005; NMFS 2005). Thus, any closure of this fishery will cause hardship for gillnet fishermen and the economies of coastal North Carolina (Pate 2005; Santora 2003).

Fishery interactions with turtles

Bycatch of sea turtles is a serious management problem in the Pamlico Sound flounder gillnet fishery. Entanglement restricts the turtle's ability to swim and feed, and constriction of appendages can cause deep flesh wounds. Depending on the depth and configuration of the gillnet, turtles can be prevented from returning to the surface to breathe, which results in drowning or potentially lethal physiological stress reactions (NMFS 2005).

Three species of sea turtles are caught in this fishery; all are protected under the Endangered Species Act. The sea turtle species involved are: the threatened loggerhead turtle (*Caretta caretta*), and the endangered green (*Chelonia mydas*) and Kemp's ridley (*Lepidochelys kempii*). The most common turtles in Pamlico Sound are loggerheads (80%), followed by greens (15%) and Kemp's ridley (5%) (Epperly *et al* 1995b). Endangered hawksbill (*Eretmochelys imbricata*) and leatherback (*Dermochelys coriacea*) turtles are rarely found in Pamlico Sound, so interactions between these species and flounder gillnets are unlikely (Pate 2005).

Fisheries interactions with sea turtles are called “takes.” Under the Endangered Species Act, the take of endangered and threatened species is prohibited, where "take" is defined as “to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture, or collect, or to attempt to engage in any such conduct” (16 USC 1532(19)). “Harm” is further defined in the regulations to mean "an act which actually kills or injures wildlife. Such acts may include significant habitat modification or degradation where it actually kills or injures wildlife by significantly impairing essential behavioral patterns, including breeding, feeding or sheltering" (50 CFR 17.3 (FWS) and 222.102 (NMFS)).

NMFS shares responsibility for the protection of sea turtles with the Fish and Wildlife Service (FWS). NMFS has jurisdiction when the turtles are in estuarine or marine environments. NMFS may grant exemptions to the takings prohibition for activities such as fisheries that take listed species incidentally to their operations. The issuance of a federal fishery management plan requires formal consultation with NMFS in which the fishery is analyzed for its potential to jeopardize the protected populations. State fisheries such as the Pamlico Sound flounder gillnet fishery require federal Endangered Species Act Section 10 permits, the issuance of which also require formal consultation with NMFS.

History of the management of the fishery’s turtle bycatch

In November and December 1999, an unusually large number (97) of sea turtles, mostly Kemp’s ridleys and loggerheads, stranded in southeastern Pamlico Sound. From aerial surveys and on-board observations, NCDMF and NMFS determined that deep-water large-mesh gillnet fisheries operating in the area were the likely cause of the turtle deaths (NMFS 2005). As a result, NMFS issued an emergency ruling that closed the Sound to gillnets with a mesh size of >5” (64 FR 70,196, December 16, 1999).

To avoid a repeat of the 1999 closure, NCDMF applied for an Endangered Species Act Section 10 permit on June 21, 2000 for the fall large-mesh gillnet fishery. Section 10(a)(1)(B) of the Endangered Species Act authorizes NMFS to permit the otherwise prohibited taking of fish and wildlife if the taking is "incidental to, and not the purpose of carrying out otherwise lawful activities." NMFS regulations (50 CFR 217-222) allow non-federal parties such as NCDMF to apply for a Section 10 incidental take permit to incidentally take threatened or endangered sea turtles. To receive the permit, the applicant must also develop and implement a conservation plan

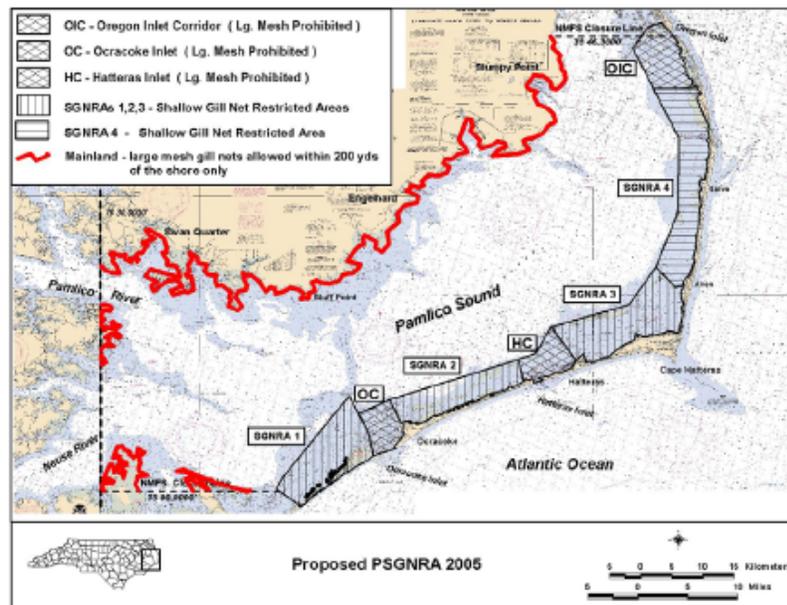
that includes actions that minimize impacts to the species, identifies sources of funding for mitigation efforts, demonstrates that the survival of the species will not be appreciably reduced, and guarantees full implementation and enforcement of the plan. NCDMF's conservation plan contains measures for permitted entry into the fishery, restricted areas, gillnet yardage limits, observer coverage, weekly logbook reporting, reporting takes, and requirements for turtle resuscitation, handling, and tagging. There are also penalties for violations of the requirements and a provision for immediate closure of the fishery if certain take levels are reached (STAC 2006).

The first Section 10 incidental take permit (#1259) was issued in 2000 with the goal of reducing turtle mortality by 50% from 1999 levels. The permit established the Pamlico Sound Gillnet Restricted Area (PSGNRA) for management of the fishery, and required permitted entry, gillnet restrictions of <3,000 yards per set, and observer coverage of 5% or more (Santora 2003). If take limits were approached, NCDMF would adopt additional area closures, gear restrictions, decreased soak times, required attendance of nets, or prohibition of tie-downs (Santora 2003), and close the fishery if authorized takes were exceeded. Despite these measures, the number of observed lethal takes of green turtles exceeded the permitted level in the fifth week of the fishing season, and NCDMF closed the PSGNRA to >5" stretched mesh gillnets on October 27, 2000. More turtle strandings followed the closure, probably due to the use of slightly smaller gear or perhaps a shift in fishing effort to just outside the closed area (NMFS 2005).

NCDMF received a second Section 10 incidental take permit (#1348) on September 27, 2001. In addition to the designation of a new PSGNRA for management of the fishery, NMFS closed the rest of Pamlico Sound to stretched net greater than 4.25" from September 28-December 15 of each year (66 FR 50350, October 3, 2001). To fish within the PSGNRA, gillnet fishermen were required to hold a NCDMF permit, fish a maximum of 2,000 yards of net, report weekly, and report any turtle interactions. Additionally, a small deep-water large-mesh fishery was permitted as a gear-testing experiment (NMFS 2005). Incidental take of sea turtles during the fishing season was below the authorized take levels (Gearhart 2002). A third Section 10 incidental take permit, #1398, was issued August 30, 2002 for the 2002-2004 seasons. This permit was similar to #1348, but added management of shallow water along the mainland coast. Sea turtle takes remained below authorized levels for in three fishing seasons.

Most recently, NCDMF received Section 10 permit #1528 for the 2005-2010 fishing seasons, with management from September 1 to December 1 of each year. The managed area consists of seven PSGNRAs (4 shallow areas and 3 inlet corridors) with time and area restrictions (Figure 2).

Figure 2 Proposed Pamlico Sound Gillnet Restricted Area for managing the fall flounder gillnet fishery 2005-2010 (NMFS 2005).



NCDMF's management of the fishery under this permit is similar to that of previous years, but there are several changes, including reduced observer coverage in areas where past observer data showed few sea turtle interactions, no reporting requirement for in-active fishermen, and no net length limit in shallow areas near the mainland where only one past interaction had been documented. The permit authorizes takes for a period of six years, but requires yearly evaluation and re-analysis of the data and take levels. NCDMF applied for the same number of takes as provided for in the 2002-2004 permits, with the addition of permitted takes of hawksbill and leatherback turtles. However, the authorized take was revised based on the highest year's upper 95% confidence interval. Additionally, the permit increased the authorized take of Kemp's ridley by 11% based on the predicted population growth of 9-13% (NMFS 2005).

Problems with setting sea turtle take limit

NCDMF's management of the fall flounder gillnet fishery has been successful in keeping sea turtle takes below permitted levels from 2001-2004 (Price 2004, 2005; Gearhart 2002, 2003). With the newly issued incidental take permit (ITP), NCDMF continues to use adaptive management to improve the effectiveness of its efforts. Further, NCDMF has now gathered a large database of information on sea turtles in the area, including the location of takes. However, these efforts may provide inadequate protection for the threatened and endangered sea turtles if

the number of authorized takes is not sufficiently conservative. To highlight this problem, the advocacy group Oceana has raised formal objections to the general process of setting sea turtle take limits and the analysis of the effects of anticipated take on sea turtle populations (70 FR 52984, September 6, 2005).

Applications for incidental take of threatened and endangered species must include a level of expected take. NCDMF calculates this number using a worst-case scenario (upper 95% confidence limit of the highest year) to account for inter-annual variability in turtle takes by the fishery (70 FR 52984, September 6, 2005). This is a conservative approach in that NMFS considers the impacts to the species at a take level that is higher than that likely to occur in any given year. For its first Section 10 permit for the fishery, NCDMF lacked the data to estimate the expected level of turtle takes, but predicted that management measures would reduce take to 50% of the 1999 stranding level.

The number of sea turtle strandings is not an appropriate metric for estimating the level of sea turtle takes. The North Carolina Sea Turtle Stranding and Salvage Network has variable coverage of the state's coastline and waterways, which is heavily biased towards ocean-facing beaches (STAC 2006). Turtles taken within the Sound are likely to strand in areas with low coverage, leading to an underestimation of the number of turtles taken. Additionally, very few stranded turtles can be positively linked to a mortality source, with 81% of stranded turtles classified as "unknown cause of death" (STAC 2006). Finally, the number of observed stranded turtles represents a small but unknown percentage of the total number of turtles taken. This percentage may be as low as 7%, but at best strandings represent 25% of actual nearshore mortalities (Epperly *et al* 1996; Murphy and Hopkins-Murphy 1989). Thus, the number of sea turtle takes by the Pamlico Sound flounder gillnet fishery cannot be accurately estimated from stranding data. Anticipated take levels are now calculated from observer and effort data (70 FR 52984, September 6, 2005).

Applications for incidental take permits are reviewed and analyzed by NMFS's Office of Protected Resources through formal Section 7 consultations. NMFS must ensure that any action authorized, funded, or carried out is not likely to "jeopardize the continued existence of any endangered species or threatened species or result in the destruction or adverse modification of critical habitat" (16 USC 1536). The result of formal consultation is a Biological Opinion that describes how the agency action will affect the species and its critical habitat and provides

jeopardy/no jeopardy and adverse modification decisions. Biological Opinions may also contain an incidental take statement authorizing otherwise-prohibited take, expressed as "the number of individuals reasonably likely to be taken or the extent of habitat likely to be destroyed or disturbed" (USFWS "Consultations with Federal Agencies").

These analyses are difficult and often controversial. I will now describe how jeopardy analysis should work, and then how it works in practice.

Jeopardy analysis: how it should work

The Endangered Species Act and implementing regulations provide NMFS and the Fish and Wildlife Service (FWS) with guidelines for conducting jeopardy analyses. In a joint regulation of NMFS and FWS, "jeopardize the continued existence of" is defined as engaging in an action that "reasonably would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of a listed species in the wild by reducing the reproduction, numbers, or distribution of that species" (50 CFR 402.02). Further clarification is provided in a joint handbook of NMFS and FWS for section 7 consultations, which describes a three-step approach to jeopardy analyses in Biological Opinions:

1. Identify probable effects of the action on the physical, chemical, and biotic environment;
2. Determine the reasonableness of expecting reductions in reproduction, numbers, or distribution; and
3. Determine if those reductions can be expected to appreciably reduce the likelihood of the species surviving and recovering in the wild (USFWS & NMFS 1998).

To follow these steps, NMFS must first describe the current status of the species, and consider the "environmental baseline." The environmental baseline consists of "past and ongoing human and natural factors leading to the current status of the species, their habitat, and ecosystem, within the action area," and provides "a snapshot of the species' health at a specified point in time and includes state, tribal, local and private actions already affecting the species, or that will occur contemporaneously with the consultation in progress" (NMFS 2005). The agency then considers the direct and indirect effects of the proposed action on threatened and endangered species and their habitat. The next step is to examine the *cumulative effects* of human activities on the species, including future actions that may likely occur in the action area. NMFS then integrates and synthesizes the information included in the status of the species,

environmental baseline, cumulative effects, and effects of the action sections to determine the total effects of the requested incidental take on the species. Finally, the agency decides whether this additional take will jeopardize the species. Given a no jeopardy opinion, NMFS then issues the permit for incidental take.

Jeopardy analysis: how it actually works

Despite consultation guidelines and handbooks, there is a lack of clear definitions and standards, which allows for inconsistency in jeopardy analyses. Terms such as “reasonableness” and “appreciably reduce,” are vague. For example, there is no clear distinction between appreciable and non-appreciable reduction, and because of this, decisions are necessarily subjective. Additionally, the statute and regulations provide no quantitative standard for determining jeopardy. Such a standard could be based on population viability, which might be stated as “not to exceed x% probability of extinction within y years” (Goodman 2005). A lack of clearly articulated quantitative standards for decision-making has been cited as a major point of conflict within the National Oceanic and Atmospheric Administration (NOAA), with frequent disagreement between the offices of Sustainable Fisheries and Protected Resources, headquarters and regional offices, NMFS and regional fishery management councils, and the agency and its research science centers (Angliss and DeMaster 2003). Without explicit standards, there is room for interpretation that invites conflict between the goals of fisheries management and protected species conservation.

Secondly, while NMFS “considers” the environmental baseline, it lists but does not integrate already-permitted takes. According to a FWS study guide on the interpretation of environmental baselines,

The [e]nvironmental baseline is an analysis of the factors that have, are, or will continue to affect the listed resources; not merely a recitation of the actions that have occurred or are occurring in the action area. We need to articulate how the other actions are specifically affecting the base conditions of the listed resources within the action area. Simply reciting a list of the actions that have or are occurring within the action area without explaining how these actions are impacting the listed resources is insufficient. The courts have affirmed this in *D[e]fenders O[f] W[ildlife] v Babbitt* (2001) and *G[reen]P[eace] v NMFS* (2000) (USFWS 2005).

It is impossible to analyze the effect of multiple activities on sea turtle populations if the activities are examined separately. In the Biological Opinion for NCDMF’s flounder gillnet

fishery permit, the various activities are described and their authorized takes are listed, but they are not integrated.

The lack of integration continues in the analysis of cumulative effects. Analysis of the effects is not actually cumulative because the agency is not adding the anticipated takes of the proposed action to the total number of already-authorized takes. This may be a result of the fact that under the Endangered Species Act, Biological Opinions are written action-by-action on a per-consultation basis. Instead of managing the potential threats to and *total* anticipated take of individual species, Biological Opinions deal with the problem in a piecemeal way, and thus, NMFS may be “[n]ickel and diming species toward extinction” (Rohlf 2001). This system could be an effective form of management, but only if the documents address the cumulative effects of all actions on the species and determine how that *total* number of takes affects the status of the species. In the Biological Opinion for NCDMF’s permit, NMFS examined only the effects of the fishery’s anticipated takes on the whole sea turtle population, and did not put them in the context of additional takes on top of all already-authorized takes.

With action-by-action Biological Opinions, NMFS must identify the action that, when analyzed in the context of all previously-authorized actions, reaches the threshold of jeopardizing the populations survival and recovery. However, as described above, there is no quantitative threshold for jeopardy, and biological thresholds for species are likely unknown, so accurate predictions of risk are difficult. A National Research Council document exploring the role of science in the Endangered Species Act concluded that “[w]hile we should assume that each additional human activity has an incremental effect and additional risk, it is hard to know if that relationship is linear (more activity means more risk), or if there’s a critical level of activity above which the risk of extinction increases dramatically” (NRC 1995). A NMFS discussion paper on Section 7 jeopardy standards describes this situation: “We assume many threatened and endangered species can tolerate some reductions [in reproduction, numbers, or distribution] without also experiencing reductions in their likelihood of survival and recovery in the wild. As we confront this task, our challenge is to distinguish between reductions that can be expected to affect a species’ likelihood of survival and recovery in the wild and those that do not” (Johnson 2000).

Identifying the activity which, combined with all previously authorized activities and takes, exceeds the jeopardy threshold is a difficult task:

On its face, this analytical process by its nature makes a jeopardy finding for a particular project extremely unlikely. FWS of NMFS must find that the very project undergoing consultation will be the ‘straw that breaks the camel’s back’ and thus put the entire species into a jeopardy situation. This methodology demands biological distinctions that are virtually impossible, particularly given the paucity of data and even scientific knowledge concerning most species (Rohlf 2001).

As a result, “no jeopardy” findings are made and incidental take permits are often issued even when the species may be truly at risk.

Biological Opinions involving incidental take of sea turtles are particularly problematic. It is difficult to define a sustainable take level because sea turtles have complex life histories and poorly understood population structure. Additionally, a key parameter for determining status and the potential effects of incidental take on the population is a measure of abundance. Currently, NMFS uses nesting beach surveys as a proxy for turtle abundance, but it is impossible to extrapolate from the number of nesting females to an accurate estimate of population size because the age structure of the turtle populations is unknown. Additionally, the long time lag between sea turtle generations prevents NMFS from observing changes in the number of nesting females resulting from current management choices for decades. In other words, the agency uses a long-term indicator to set short-term take limits. The “multiple assumptions [that] must be made to translate nests into adult females, adult males, and juveniles of both sexes” are further weakened by a “lack of corroborating empirical estimates” (TEWG 2000).

The 2005 Biological Opinion for NCDMF’s flounder gillnet fishery permit suffers from all of the shortcomings described above. The document provides a lengthy recitation of the sea turtle species’ status and threats, but at the critical point of jeopardy analysis, the conclusions and reasoning are brief, superficial, and not quantitative. Additionally, the effect of the anticipated incidental take is described using nesting beach trends, despite the shortcomings described above. For example, NMFS concludes that “NMFS anticipates that the annual removal of juvenile Kemp’s ridleys as a result of this fishery is being sustained as indicated by the increase in nesting since 1990,” and for green turtles, “[b]ased on the increases in nesting activity and the increase in CPUE documented at limited in-water study sites, NMFS anticipates that the annual loss of 28 juvenile greens to the breeding population over the permit duration would not have a significant effect on the distribution and reproduction of the population” (NMFS 2005).

It is clear that NMFS needs a transparent and fair method for determining biologically sustainable levels of take for all threatened and endangered species. Biological Opinions do not explicitly determine whether anticipated takes are sustainable in the context of all other authorized takes that affect sea turtle populations. A more logical management scheme would estimate the total number of removals that sea turtle populations can withstand, and then allocate that to all activities that have incidental take of sea turtles. This is the premise of current marine mammal management in the United States.

Marine mammals and PBR

Under the Marine Mammal Protection Act, marine mammals are managed on a population basis in units called stocks. Stocks are groups that either have low rates of genetic exchange or are defined because increased risk requires separate management (Taylor 1997). Potential biological removal is the maximum number of animals, not including natural mortality, that can be removed without preventing the stock from reaching or maintaining its optimum sustainable population (OSP) level (16 USC 1362(20)). OSP is defined as “the number of animals which will result in the maximum productivity of the population or the species, keeping in mind the optimum carrying capacity of the habitat and the health of the ecosystem of which they form a constituent element” (16 USC 1362). This level is further defined in NMFS regulations as falling between the carrying capacity and the maximum net productivity level.

Potential biological removal is calculated according to the following equation:

$$PBR = N_{\min} \times 0.5 (R_{\max}) \times F_R$$

Where

- N_{\min} = minimum estimate of population size
- R_{\max} = maximum growth rate of population
- F_R = “recovery” factor between 0.1 and 1.0.

This decision rule for determining the allowable incidental take of marine mammals was developed to fit the following three performance standards:

1. 95% probability a population at MNPL will not be below MNPL 20 years later
2. 95% probability a population at 30% of K will not be below MNPL 100 years later
3. 95% probability the time to recovery of a depleted population is not delayed by more than 10% (Wade 1998).

Although PBR satisfies these standards, only the decision rule (PBR), and not the standards, is written into the statute. By using performance criteria, however, both the level of uncertainty and the acceptable quantitative levels of risk are made explicit (Taylor *et al* 2000).

Unlike the Endangered Species Act, in which anticipated take is permitted on a per case basis, the Marine Mammal Protection Act uses PBR to estimate a total allowable take level for all fisheries that interact with the stock. If estimated take levels exceed PBR for a stock, the stock is designated "strategic." Strategic stocks are also those that are threatened or endangered under the Endangered Species Act, or designated depleted under the Marine Mammal Protection Act. For strategic stocks, certain management actions are mandated, including the formation of a Take Reduction Team (TRT). Take Reduction Teams are organized by NMFS's Office of Protected Resources and include a variety of stakeholders. Their goal is to develop a consensus Take Reduction Plan to reduce incidental take to below PBR within six months, with the ultimate goal of a zero mortality and serious injury rate.

Potential biological removal is a useful approach for calculating allowable incidental take levels. It is a simple, data-driven formula requiring a minimum amount of data. It is conservative in that it uses minimum population estimates and a recovery factor based on the population status (threatened, endangered, or healthy), addresses data uncertainty in a straightforward way, and follows the precautionary principle with more conservative management when data are less precise (Holt and Talbot 1978). Potential biological removal is also comprehensive because it calculates total take per stock. Finally, it was developed as the result of extensive modeling that evaluated the performance of the decision rule based on explicit risk thresholds. Potential biological removal meets established performance standards, and its use in marine mammal management ensures that stocks are conserved in accordance with the statute's overall goals.

PBR for sea turtles?

Because sea turtles share many life history characteristics with marine mammals, including low fecundity, slow development, delayed maturation, and long lifespans, researchers and managers have proposed adapting PBR for sea turtles (TEWG 2000). One such researcher, Dr. Selina Heppell, explored quantitative tools for evaluating jeopardy for sea turtles in a contract report to the Southeast Fisheries Science Center. In this report, she describes a PBR-like approach. Although the goals of marine mammals and sea turtle management as defined in the

Marine Mammal Protection Act and the Endangered Species Act are different (e.g. PBR was designed to allow stocks to be maintained within OSP, whereas sea turtles should be “recovered”), PBR is a useful tool for calculating allowable take and could be adapted for turtles by using different performance standards to reflect sea turtle recovery goals (Heppell 2005).

Dr. Heppell suggests using nesting beach surveys in defining the parameters for the PBR equation because they are the most complete datasets currently available. Minimum population size (N_{\min}) should be calculated as the lower confidence interval of the total adult population size, using the estimated adult sex ratio to extrapolate from adult female abundances. The maximum intrinsic population growth rate (R_{\max}) should be based on observed trends in nesting beach data, with nesting beach totals pooled and standardized by an index beach trend whenever possible. The recovery factor should be the same as the defaults for marine mammals.

However, because PBR was developed for marine mammals, the approach and equation must be modified before it can be used for sea turtles. In particular, because of differences in the reproductive value of specific life history stages of sea turtles, a separate PBR could be calculated for several life history stages (juvenile, sub-adult, adult) (TEWG 2000). Because of the impracticality of having several PBRs for each stock of sea turtle, Dr. Heppell suggests using “adult equivalents” based on reproductive value to account for takes of juveniles (Heppell 2005).

The PBR approach is simple and can be used for all sea turtle species. It has been tested in marine mammal management, where it has been successful (Heppell 2005). The approach satisfies five of six criteria that Dr. Heppell identifies as critical for successful sea turtle management: it is based on available data, or at least obtainable data; it is precautionary, in that where there is less information available, a more conservative decision is made, but also “reasonable”; it considers all sources of anthropogenic mortality; it is based on estimates of mortality; and it is simple enough to explain to stakeholders and other interested parties. Potential biological removal does not satisfy the criteria for more conservative management for declining populations; however, it could be modified to do so (Heppell 2005).

Despite its strengths, the approach has several weaknesses. Because it is so simple, the equation does not take into account the complexity of population dynamics and contains little biology. Additionally, although very little data is required to calculate PBR, those data are not easily obtained. Estimates of population size are particularly problematic because they rely on nesting beach surveys and require multiple assumptions, each of which carries uncertainty, to

extrapolate to a total population figure. The use of a maximum potential growth rate may not be conservative enough for stocks that are declining, but the approach may be too conservative for stocks that are experiencing rapid increases due to nesting beach protections. Finally, PBR requires that all sources of anthropogenic mortality be accounted for, which may be difficult without better bycatch assessments and coordination among international, federal, and state agencies.

Even so, potential biological removal has already been used by NOAA to estimate the level of mortality that sea turtle populations can withstand (Gerrodette 1996). In response to concerns over the level of incidental take of sea turtles in the Hawaii-based longline fishery, NMFS researchers calculated PBR for North Pacific loggerhead and leatherback populations. PBR has also been calculated for adult loggerheads that interact with fisheries along the southeast coast of the US (TEWG 2000). Both of these calculations relied on extrapolated nesting beach data to obtain abundances of adult turtles, marine mammal default recovery factors, and maximum growth rates from nesting beach trends of recovering populations. No management measures resulted from these calculations, but the fact that NMFS has performed the calculations is an indication that they consider it a valuable exercise.

Potential biological removal is an appealing management mechanism because it provides defensible numbers and a biologically-relevant total take limit. Once the total allowable take per stock is defined, the take can be allocated amongst fisheries and other activities that interact with the turtles. It also allows for more flexibility in management, because fisheries can be managed in whatever way that will keep turtle takes below the authorized level, without continued consultations with NMFS whenever changes are made.

How would PBR work for the Pamlico Sound flounder gillnet fishery?

Once all necessary data are collected, a separate PBR would have to be calculated for each stock of each sea turtle species that interacts with the Pamlico Sound flounder gillnet fishery. This fishery interacts with stocks of loggerhead, green, and Kemp's ridley sea turtles. Ideally, a separate PBR should be calculated for each life history stage, but as described below, adult equivalents for juveniles could be used to properly address the relative reproductive value of turtles at different life stages.

Once total allowable take is calculated, the take must be allocated among fisheries and other activities. Sea turtles are highly migratory, so potential takes that occur in stocks that spend time in waters outside of U.S. jurisdiction must be addressed. For migratory species that cross national boundaries, marine mammal guidelines advise that PBR should be divided based on the proportion of time spent in each area of jurisdiction (Wade and Angliss 1997). This proportion could be estimated from satellite tracking studies that follow the movements of individual animals. The U.S. fraction of the PBR can then be allocated.

The allocation of take is likely to be highly contentious. A fair allocation system would be both transparent and participatory. I recommend mediated negotiations among multiple stakeholder groups to work out the details of allocation. These negotiations would be modeled on the Take Reduction Team (TRT) process used for marine mammal management. As specified by the Marine Mammal Protection Act, TRTs include representatives of federal agencies, coastal states, Regional Fishery Management Councils, interstate fisheries commissions, academic and scientific organizations, environmental groups, all commercial and recreational fishing groups and gear types which incidentally take the species or stocks, and others as deemed appropriate by the Secretary of Commerce, with the goal of balancing both resource user and nonuser interests (16 U.S.C. 1387(f)(6)(C)). The list of stakeholders is lengthy, but the most successful TRTs have been small in order to facilitate better discussion and to involve all members more fully (Young 2001).

In general, participants report a favorable impression of the TRT process. It is particularly valuable as an alternative to the traditional top-down approach to management and for bringing disparate stakeholder groups to consensus (Young 2001). Team members are forced to work together, but if consensus cannot be reached, the Marine Mammal Protection Act requires the Secretary of Commerce to develop his own plan. This provides additional motivation for Team members to move beyond their differences and work toward mutually acceptable bycatch mitigation strategies (Read 2000). A similar incentive could be used for sea turtle take allocation teams, such that if team members could not reach a consensus, the Secretary of Commerce would design the allocation scheme.

The purpose of Take Reduction Teams is not to allocate take. However, there is a precedent for dividing take among different groups. The Gulf of Maine Harbor Porpoise Take Reduction Team dealt with the transboundary Gulf of Maine/Bay of Fundy harbor porpoise

stock, which inhabits both U.S. and Canadian waters. Both nations have fisheries that interact with the stock. Canadian representatives provided the TRT with estimates of Canadian bycatch of the Bay of Fundy stock. Canadian and U.S. representatives then negotiated and allocated 100 animals from the PBR level to the Canadian fisheries (63 FR 48670, Sept. 11, 1998). That same year, Canada's Department of Fisheries and Oceans developed a Harbor Porpoise Conservation Strategy and set a hard cap of 110 porpoise takes per year for the Bay of Fundy (Trippel and Shepherd 2004). In the case of a sea turtle PBR, the allocation would be made among domestic fisheries and activities instead of between nations, but the harbor porpoise example is still applicable.

It is important to emphasize that PBR is a method for calculating allowable take. Once that number is determined and a system of allocation has been implemented, the responsible agencies are free to manage fisheries and activities using whichever management tools they consider best suitable for maintaining sea turtle bycatch below authorized levels.

I will now describe the parameters necessary for calculating PBR and the methods for obtaining this information, and discuss their feasibility and potential data sources.

Parameters

Abundance (N_{\min})

Sea turtles would be managed by stock and not as a species as a whole, so the minimum abundance figure must be estimated for each stock. First, however, the stocks must be defined.

Defining stocks

Stocks are management units that ideally represent distinct populations. Gene flow among sea turtle populations is low due to nesting beach fidelity, so dispersal from other areas will not compensate for the overexploitation of individual populations, and human activities may lead to the loss of ecologically significant units (NMFS 2005). Properly defining sea turtle stocks as separate management units is critical. On one hand, inappropriate pooling of stocks ("conservation error") can lead to overestimation of the abundance of animals available for "harvest" (PBR is too high), which can then lead to the depletion of the stock experiencing high mortality (Taylor 1997; Wade and Angliss 1997). On the other hand, incorrect splitting of stocks ("economic error") may lead to a PBR that is too low, which would unnecessarily restrict fishing and other human activities (Taylor 1997). Barlow *et al* (1995) suggest minimizing conservation

error by defining stocks by their smallest known groupings, and pooling them only when tagging, genetic, or morphological studies provide strong evidence that they are one unit.

Stock structure may be defined using various types of data, including differences in geographical distribution and movements, population dynamics and trends, morphology, genetics, contaminant and natural isotope loads, parasites, and oceanographic habitat (Wade and Angliss 1997). For example, genetic data is used to assess the degree of differentiation among populations by testing the hypothesis that the differences in individuals between populations are greater than for individuals within a population (Taylor 1997).

Distinct sea turtle populations have been identified, so stocks should be defined based on these already-established units. Below, I summarize information on the populations of sea turtles that may interact with the Pamlico Sound flounder gillnet fishery.

Kemp's ridley

Kemp's ridleys nest almost exclusively at Rancho Nuevo, Tamaulipas, Mexico and nearby beaches. With only one large nesting aggregation, there is only one genetic population.

Loggerhead

There are at least five loggerhead populations in the Western Atlantic; members of all five have been found in the Pamlico Sound area (TEWG 2000; Bass *et al* 2004; Bowen *et al* 2004). The following populations have been identified:

1. Northern Nesting Population, extending south from North Carolina to northeastern Florida at ~29° N;
2. South Florida Nesting Population, from 27° N on Florida's east coast to Sarasota on the west coast;
3. Florida Panhandle Nesting Population, at Eglin Air Force Base and the beaches around Panama City, Florida;
4. Yucatan Nesting Population, on the eastern Yucatan peninsula, Mexico; and
5. Dry Tortugas Nesting Population in the Florida Keys, Florida (TEWG 2000; NMFS 2001).

Green

Green turtles have regional populations with distinct mitochondrial DNA haplotypes associated with each nesting rookery (Bowen *et al* 1992). There are two populations in the eastern Atlantic: Equatorial Guinea (Bioko Island) and Guinea-Bissau (Bijagos Archipelago).

There is one in the central Atlantic, Ascension Island, and five in the western Atlantic: Suriname, Venezuela (Aves Island), Costa Rica (Tortuguero), Mexico (Yucatan Peninsula), and the United States (Florida) (Marine Turtle Specialist Group 2004).

Despite the growing body of knowledge on population definition in sea turtles, there is still considerable uncertainty in the identity and degree of mixing of sea turtles in Pamlico Sound. However, the recovery factor in the PBR equation may provide a “margin of safety” that offsets this uncertainty (Taylor 1997).

Accounting for differential reproductive value of juvenile and adult turtles

Reproductive value is a measure of a turtle’s current and expected future contribution to the population through reproduction (Heppell 2005). It is based on the turtle’s current age, its probability of surviving to sexual maturity, and its expected lifespan (Heppell 2005). Sea turtles in various life stages have different reproductive values, so anthropogenic mortality of these turtles has differential effects on the population. Because of this, ideally there should be a separate PBR for adults and juveniles (Gerrodette 1996). However, because this would involve two PBRs per stock per turtle species, it would be impractical. An alternative is to have one PBR per stock, but to use the “adult equivalency” of juvenile turtles to account for differential reproductive value.

“Adult equivalency” is a calculation of the number of immature sea turtles mortalities that would equal the effect on population dynamics of the death of a reproducing adult (Gerrodette 1996). It is based on the reproductive value of turtle age classes, as calculated by a deterministic age-structured matrix model (Heppell 2005). To use this concept in the PBR calculation, the abundance estimate of juvenile and sub-adult sea turtles would be “converted” to adults by multiplying by the calculated adult equivalency. This number can then be added to the estimate of adult abundance to get a total population abundance estimate.

Estimating the reproductive value of sea turtles would require a simulation model analysis. This is difficult because reproductive value is a volatile parameter, and small changes in variables in the analysis such as the turtles’ remigration time have large effects on its value (S. Heppell, pers. comm.). However, it is important to acknowledge the different impact on population dynamics of removing juveniles and sub-adults versus adults by using adult equivalents in the PBR calculation.

Calculating abundance

Calculation of PBR requires an estimate of abundance for each stock. The best datasets available for sea turtle abundance are nesting beach surveys, which provide a reasonably accurate estimate of the number of nesting females. This number is then extrapolated to a total adult population number using estimates of the average number of nests per year, the remigration time, and the sex ratio in the adult population. This extrapolation does not include juvenile and sub-adult turtles. It is extremely difficult to get an estimate of the non-adult segment of the population. Turtles spend only a very small portion of their lifetime on beaches and very little is known about turtles once they head into the pelagic ocean because of the difficulties presented by the oceanic environment. In particular, juveniles are pelagic, widely distributed, and hard to detect (Gerrodette and Taylor 1999). Consequently, there are large gaps in the data for sea turtles, including a lack of accurate abundance estimates and of movement patterns.

Because of the difficulty in obtaining absolute abundances of sea turtles, managers often use indices of abundance, or relative abundance (Gerrodette and Taylor 1999). If the ratio of relative and absolute abundance is unknown, relative abundance cannot be translated to total population abundance. Even though abundance indices are important for detecting trends over time, determining whether mortality levels are unsustainable at the population level requires a measure of absolute abundance (Gerrodette and Taylor 1999).

The PBR equation uses a minimum (20th percentile) estimate of population abundance to address uncertainty due to imprecision with which abundance can be estimated (Wade 1998). Surveys with more uncertainty have wider confidence intervals, so the minimum population estimate and PBR are lower. Not only does this allow for more conservative management when stocks which are less well-known, it encourages improvement in the precision of abundance estimates, because narrower confidence intervals (lower coefficients of variation) result in higher PBR levels (Taylor 1997; Taylor *et al* 2000; Wade 1998). This incentive would promote sea turtle research and improved abundance estimates.

Methods for gathering abundance data and current data sources

As described above, nesting beach surveys are the most readily available data source for sea turtle abundances. However, because of the numerous assumptions that must be made in extrapolating the number of nests or number of nesting females to a total population figure, other methods should be used to get a better estimate of population abundance. These include in-water surveys, aerial surveys, and mark-recapture studies.

In-water Surveys

Several studies have used in-water surveys to estimate abundance of sea turtles in the southeastern U.S. In a four-year study by the South Carolina Department of Natural Resources, researchers surveyed the waters from Winyah Bay, SC to St. Augustine, FL during the summer months (May through August) from 2000-2003 (Maier *et al* 2004). The study had both a fishery-independent trawl component and fishery-dependent component using commercial trawlers. The researchers were able to create a statistically valid regional index of abundance (catch per unit effort), which was then used to estimate the number of turtles in the sampling area. In another study, researchers surveyed pound nets in the Pamlico-Albemarle estuarine complex from September through December in 1995-1997 to estimate sea turtle abundance from the number of turtle incidentally caught in the nets (Epperly and Braun 1998).

The draw-backs to in-water surveys are that they expensive and can only be performed on a limited spatial scale. However, they can be used to estimate abundances of turtles in oceanic and neritic life stages, which have previously been unknown, to within an order of magnitude (S. Heppell, pers. comm.). In-water surveys could be complemented by the use of aerial surveys.

Aerial surveys

Aerial surveys are a good way to estimate relative distribution and abundance, particularly for large spatial areas (McDaniel *et al* 2000). However, there are weaknesses to this approach. Importantly, observers cannot distinguish among sea turtle species, with the exception of leatherbacks. Observers also cannot obtain any biological or demographic information from the turtles. As described above, these surveys would be useful in conjunction with in-water surveys, particularly if the aerial surveys locate “hot spots” of turtle aggregations. Additionally, factors such as habitat, tide, weather, time of day, observer variations, and turtle size and behavior may influence estimates of turtle abundance (Bayliss 1986; Buckland *et al* 1993). Even so, aerial surveys are an important tool for gathering abundance data.

To estimate absolute abundance (rather than relative abundance), researchers must correct for submerged turtles by factoring in the amount of time that turtles spend on the surface versus underwater. This information can be determined through radio, sonic, and satellite telemetry studies (Henwood and Epperly 1999). Sightings per unit of surveyed area data can be extrapolated to total surface abundance in the study area, and then the densities can be corrected to include submerged turtles. This technique is used routinely for marine mammals, even deep-diving species, and there is a rich literature on the subject.

Joanne McNeill and other researchers at the NOAA laboratory in Beaufort, NC are working with the Cherry Point Marine Corps Air Station to conduct aerial surveys of Pamlico and Core Sounds. They are gathering data on the relative abundance and seasonal distribution of sea turtles and marine mammals within restricted airspace in those waters, especially in the vicinity of the two bombing targets. The surveys began in July 2004 and will be completed in the spring of 2006 (J. McNeill, pers. comm.). The areas surveyed include all of Core and Pamlico Sounds north to Hatteras Island, and approximately 1 mile offshore of this area. Data from this and other aerial studies (e.g. Shoop and Kenny 1992; Musick *et al* 1994; Epperly *et al* 1995a) should be compared and possibly combined to aid in determining total population abundance estimates.

Mark-Recapture studies

Mark-recapture studies are another method that could be used to estimate sea turtle abundance. Estimates of abundance from these studies are based on the following five assumptions: no births, deaths, immigration, or emigration during the study period (although this can be relaxed for open population models); all animals have the same probability of being tagged; tagging does not affect an animal's probability of being recaptured; tags are not lost, and tags that are present are always detected; and recaptured animals represent a random sample of the population (Gerrodette and Taylor 1999).

Given that these requirements are fulfilled, mark-capture studies can be performed at a wide variety of scales. For example, Smith *et al* (1999) conducted an ocean-basin-wide study of the North Atlantic humpback whale to estimate population size using photographic and skin-biopsy sampling. On a smaller scale, mark-recapture studies have been used to estimate trends in

the abundance of green and loggerhead turtles resident in the waters of the southern Great Barrier Reef (Chaloupka and Limpus 2001).

A limitation of mark-recapture studies is that the probability of recapturing marked individuals is low in many populations. This may be due to long-distance movements, unknown migratory patterns, high natural mortality of hatchling and juveniles, and high anthropogenic mortality in sea turtles (Bjorndal *et al* 2001). Even so, there is encouraging news: a study of juvenile loggerheads in Core Sound, North Carolina has shown that the turtles exhibit site fidelity to foraging areas despite long migrations, and a high percentage of individuals are recaptured each year (Avens *et al* 2003). Despite potential difficulties of mark-recapture research, any additional information on sea turtle abundance that could be obtained from these studies should be utilized.

Recommendation

Because these surveys are often done in mixed stock foraging or breeding areas, researchers must be able to identify the stocks to get an estimate of abundance per stock. Probability analyses using information on the contribution of each stock to the total number of turtles in these areas, as summarized below in the Mortality section (“*Assigning takes to a stock’s PBR*”), should be used to differentiate stocks in surveys where identification is not feasible (e.g. aerial surveys). If abundance data are truly deficient, nesting beach data should be used, which would result in low PBRs because of high variance in the data. In this case, the low PBR would spur better studies and more reliable estimates of abundance, as has been seen for marine mammals.

Growth rate

The maximum intrinsic growth rate of sea turtle populations is not known for many sea turtle species. However, an estimate may be calculated from the maximum population growth rate observed in nesting beach surveys. This rate should be based on combined nesting beach totals and standardized by an index beach trend, where data permit (Heppell 2005). Nesting surveys indicate that Kemp’s ridleys have increased at a rate of 12% from 1985-1998, and loggerhead nests in South Florida have increased by 4% from 1989-1998 (TEWG 2000). Estimates for other sea turtle species could easily be obtained in this manner.

These figures may not represent the maximum growth rate because incidental take and other anthropogenic threats are preventing the populations from rebuilding as quickly as they can (TEWG 2000). However, because many populations are declining, the use of a maximum growth rate may not be appropriate for calculating total allowable take. Dr. Selina Heppell has proposed adding a factor to the PBR equation, a value that would be multiplied by the maximum growth rate based on the ratio of the current population growth rate to the rate that would be required to meet the turtles' conservation goal (2005). This factor would be greater than 1 for increasing populations and less than 1 for declining populations. This would add current biological information and trends to the PBR equation and make the total allowable take calculation more accurately reflect the current status of sea turtles. The PBR level would also be lower for declining populations, which is a more conservative management scheme.

Recovery factor

The PBR guidelines for marine mammals set default recovery factor values for endangered populations ($F_R = 0.1$), threatened or depleted populations ($F_R = 0.5$), and populations of unknown status ($F_R = 0.5$) (Barlow *et al* 1995). These values adjust the level of allowable take for biases in estimates, such as incorrectly defining stocks and grouping two populations into one stock (Taylor 1997). The recovery factor is perhaps the most controversial piece of the PBR equation because it is difficult to set the value objectively when biases are unknown (Taylor *et al* 2000). However, modeling efforts show that with the default recovery factor, even large biases and errors in estimating abundance still allow populations to remain within their optimum sustainable population level (Taylor 1997). Additionally, using a recovery factor reduces the allowable take level such that more of the population's net productivity goes toward rebuilding the population, and the time to recovery is not delayed by permitted incidental takes (Wade 1998). As Wade explains, "[e]nsuring that the time to recovery is not substantially delayed is a way of ensuring that the population growth rate is not substantially reduced, thus promoting recovery (Wade 1998).

There is a great deal of uncertainty in our understanding of sea turtle status. The management model should seek to avoid the unnecessary economic harm from closing fisheries because PBR is set too low, but the level of allowable take should be conservative. Extensive modeling and simulations of PBR under different recovery factors has shown the defaults to be

robust to bias (Taylor 1997). Therefore, the default recovery factors used for marine mammals could be used in calculating PBR for sea turtles.

Currently, threatened and endangered designations are given at the species level for sea turtles, and populations are not protected separately. Until further classification below the species level is made, I recommend using a recovery F_R value of 0.1 for all populations of the endangered green and Kemp's ridley turtles. However, I recommend splitting loggerheads by population and using the more conservative value of 0.1 for the declining Northern population, and 0.5 for the other populations, despite the threatened status for the species as a whole.

Mortality

Mortality is not a parameter in the PBR calculation, but it is necessary to know the number of turtles that have been injured or killed in order to know when PBR has been exceeded. There are two issues regarding sea turtle mortality that need to be addressed: accounting for all sources of mortality, and assigning turtle takes to an individual stock's PBR given mixed foraging grounds.

Accounting for all sources of mortality

To know the effect of takes on the turtle stocks, all sources of anthropogenic mortality throughout the turtles' ranges must be accounted for. In its Biological Opinion for the Pamlico Sound flounder gillnet fishery, NMFS identified federal permitted actions that take sea turtles, which include oil and gas exploration and activities, vessel operations of the Navy and the Coast Guard, additional military activities (e.g. bombing, sonar, dredging), and the following fisheries: American lobster, monkfish, dogfish, southeastern shrimp trawl fishery, Northeast multispecies, Atlantic pelagic swordfish/tuna/shark, and summer flounder/scup/black sea bass (NMFS 2005). Other actions include electric plants, research, private and commercial vessels, and state-regulated trawl, purse seine, hook and line, gillnet, pound net, longline, and trap fisheries (NMFS 2005). Additional threats to sea turtles, such as beach erosion, beach armoring, artificial lighting, and marine debris and pollution have indirect effects, and they cannot feasibly be included in the tally of anthropogenic mortality.

The U.S. does not have the authority to regulate activities that occur outside of its jurisdiction, such as legal directed harvest of green turtles in several Caribbean countries (Fleming 2001). However, these take can be accounted for in the PBR framework as described

above (“*How would PBR work for the Pamlico Sound flounder gillnet fishery?*”). The calculated PBR should be multiplied by the proportion of turtles that are in U.S. waters, or if the whole stock spends time outside of U.S. jurisdiction, multiplied by the proportion of time the turtles spend in U.S. waters.

Observer coverage is critical to estimating mortality. NMFS has established the National Working Group on Bycatch to develop a system of standardized bycatch reporting and monitoring programs. This group identified several sources of data for estimating bycatch, including fishery-independent surveys, at-sea observations (including observers), self-reporting (logbooks, trip reports, etc.), and stranding networks (NMFS 2004). To collect mortality data, all bycatch and observer data for the fisheries must be pooled, and observed turtle takes can be extrapolated based on effort to estimate the total number of takes.

While observer coverage is adequate in the small Pamlico Sound gillnet fishery (~10%), it is not currently sufficient to get an accurate estimate of all sea turtle takes. NMFS is in the process of expanding and modernizing observer programs for federal commercial fisheries, but because of budget constraints and logistical difficulties, observing sea turtle takes in state and recreational fisheries will be more challenging (70 FR 72099, December 1, 2005).

Another way to track mortality is through strandings. However, as noted above, the number of observed strandings represents only a small and unknown proportion of the actual number of sea turtles that are injured or killed. This value is variable and depends on factors such as oceanographic conditions. Epperly *et al* (1996) found that strandings may only represent 7-13% of actual nearshore mortality in the mid-Atlantic area, while another study estimated the proportion as 18% (WRC unpublished data, from STAC 2006). At best, strandings represent 25% of nearshore mortalities (Murphy and Hopkins-Murphy 1989). The cause of death cannot be determined for the majority of stranded turtles, but strandings should continue to be monitored.

Assigning takes to a stock's PBR

Sea turtle stocks are often defined by distinct mitochondrial haplotypes, or genetic “tags” that emerge as a result of site fidelity in nesting females (Bowen 2003). However, sea turtles are highly migratory, and while turtles return to their natal beaches to nest, different stocks form mixed groups on foraging grounds. The relative contribution of nesting populations to the mixed stock groups varies from location to location due to the proximity of rookeries, the availability of

feeding habitat, physical ocean conditions, and migration routes (Bass 1999). This makes the stock identification of bycaught turtles difficult. It is certainly infeasible to assign takes to different stocks by testing the genetic stock identity of every turtle taken (Gerrodette 1996). A better option is to use mixed stock analysis to determine the representativeness of each stock in a geographical region. Once the approximate proportion of turtles on foraging grounds originating from each population is known, managers can use probability to separate bycaught turtles by source population (and by stock) and assign the takes to each stock's PBR.

Mixed stock analysis was developed for salmon management in the US as a tool that would allow researchers to estimate the probable source population for groups of individual fish by examining the frequencies of genetic or morphological characters (Pella and Milner 1987). The analyses were based on a maximum likelihood framework, but Bayesian based algorithms are now used (Bass *et al* 2004). As described above, mixed stock analysis can be used to estimate the proportions of the individuals in a mixed stock population that come from various source populations by comparing genetic data from nesting beaches to foraging areas (Bolker *et al* 2003). It can also be used to understand changes in the composition of stocks at foraging grounds based on temporal changes in mitochondrial DNA haplotype frequencies (Bass *et al* 2004).

If distinct haplotypes cannot be identified, assigning individuals to stocks is difficult. When there are overlapping haplotypes, researchers must use probability analysis, which requires large sample sizes (Heppell 2005; Maier *et al* 2004). However, new research by Okuyama and Bolker improves mixed stock analysis by incorporating ecological covariates such as nesting population size and location within major ocean currents (2005). Using a hierarchical Bayesian model allows researchers to draw stronger conclusions from existing data and to accurately estimate the turtles' stocks even when there is a high degree of genetic overlap among the rookeries.

Below, I summarize information on the representativeness of stocks on foraging grounds for each species.

Kemp's ridleys

All turtles come from the single identified stock.

Loggerheads

As described above, turtles from five stocks of loggerheads may be found in the foraging habitat of Pamlico Sound. To understand the relative proportion of these stocks in mixed foraging grounds, Bowen *et al* surveyed loggerheads from ten juvenile feeding habitats across the eastern US and compared their mitochondrial DNA to potential source populations in the Atlantic and the Mediterranean (2004). Mixed stock analysis indicated that juveniles preferentially remain in the vicinity of their natal nesting beaches. Turtles are found with increased frequency near their natal rookery and do not form a completely mixed stock. This is confirmed by data from loggerheads that stranded in North Carolina, which suggest that the proportion of turtles from the Northern nesting population, around 28-32%, is over-representative of their total numbers in the overall Atlantic loggerhead population (NMFS 2005). A trawl survey of offshore foraging grounds in the waters from Winyah Bay, South Carolina to St. Augustine, Florida also found an over-representation of loggerheads from the Northern nesting population, but the percentage was lower, 19.1%, compared with the population's 9% of total Atlantic nesting activity (Roberts *et al* 2005).

More specific genetic analyses of loggerheads in the Pamlico-Albemarle estuarine complex show a different distribution. Sampling from pound nets from 1995-1997, researchers found that 80% of the turtles originated from the south Florida nesting population, 12% from the Northern population, 6% from Yucatan population, and 2% from other rookeries (Bass *et al* 2004). This supports the finding by Bass *et al* (2004) that the size of the regional nesting populations influences the relative contribution to mixed stock foraging grounds. Because these proportions are specific to the area of this fishery, I suggest that they be used in assigning takes to stocks' PBRs.

Greens

According to North Carolina's Sea Turtle Advisory Committee, there is little tag information and no genetic information available on green turtles that have stranded in North Carolina (STAC 2006). However, there have been mixed stock analyses of this species in other areas, such as green turtle aggregations on Hutchinson Island, Florida (Bass and Witzell 2000). This type of analysis could feasibly be performed for turtles in the Pamlico Sound. In fact, the NMFS lab in Beaufort, NC has a paper in review on the genetics on green turtles caught in

Pamlico and Core Sounds. Their study provides mitochondrial DNA analysis and mixed stock analysis to show the proportion of the feeding aggregation of that comes from each rookery (C. McClellan, pers. comm.).

Recommendation

The research on the relative representativeness of each stock of sea turtles within the geographic region of each fishery and permitted activity's operation should be compiled. Sea turtle takes by these fisheries and activities can then be assigned, based on probability, to each stock's PBR.

Conclusion

A modified potential biological removal mechanism could be a useful management tool for calculating a total allowable mortality level for sea turtles in a transparent and biologically-meaningful way. As I have shown, there is no fundamental obstacle to calculating PBR in terms of data. The information needed, if not already available, could be acquired with continued efforts. However, there may be political implications involved in gathering the data that prevent its collection. For example, there are currently no published estimates of total sea turtle abundance, so their true status is unknown. If efforts are made to calculate abundance, it may be much higher than expected. If so, PBR would be much higher, and there could be a call to de-list the species from fisheries advocates. This might be worrisome to sea turtle managers and advocates.

The goal of managing threatened and endangered species is recovery and de-listing. Even so, de-listing can be politically controversial. For example, the eastern Pacific population of gray whales was removed from the Endangered Species List in 1994 because it had reached recovery goals, but in 2001 a group of conservation organizations petitioned NMFS to re-list the gray whale as threatened or endangered. Their argument was that despite the whales' current recovered status, continued protection was essential in the context of increasing threats to the species and their environment (global warming, El Niño, bottom trawling and commercial fishing, and offshore oil and gas development) (66 FR 32305, June 14, 2001). There is a legitimate concern that removing protections will ultimately harm the species. Additionally, while scientists, managers, and conservation organizations work toward improving the status of

the species, they also have a great deal invested (time, money, resources, jobs) in a continued conservation struggle, and conservation “victories” may not be seen as actual successes.

Political resistance is not only an issue in the calculation of PBR, but also in the implementation. Much of the challenge revolves around uncertainty. Because PBR has never been calculated for sea turtles, it is not known whether that number will be greater or less than the level of currently permitted takes. As a result, stakeholders do not know how the PBR framework would affect them. A brief analysis of the attitudes of major stakeholders, commercial fishermen, environmental NGOs, and government managers, helps to demonstrate the effects of this uncertainty.

Commercial fishermen would benefit from an increase in authorized takes because their operations would be less restricted. If PBR is lower than the current level of take, fishermen would be harmed, particularly if PBR has already been exceeded and fishery closures and more restrictions are required. In either case, though, a PBR framework would force fishermen to negotiate the allocation of takes with other fishermen and fisheries. The fisheries would be in direct competition for a “share” of the sea turtle takes, and negotiations would likely be heated. Fishermen have already adapted to the current management scheme and regulations, and would probably prefer the status quo.

Second, environmental NGOs and conservation organizations might be equivocal about a PBR framework. A lower level of authorized take would be considered a conservation victory for sea turtles, but an increased level of take would be unpalatable. NGOs and conservation organizations would appreciate the transparency of PBR, but as with the fishermen, because of the uncertainty in the level of PBR, they may be content with the status quo management scheme and continue to sue NMFS over Biological Opinions.

Finally, managers would also benefit from more transparent process, but they also assume risk in implementing a PBR framework. Government sea turtle scientists and managers would be doing work that they have not done before and they face a great deal of uncertainty, both in terms of the outcome of the calculation (the level of authorized takes) and the level of public support for the program. Government managers also face a perpetual lack of resources and an ever-increasing list of responsibilities, both of which would limit the full implementation of the PBR framework.

The bottom line is that no stakeholder group has a clear incentive for pushing for a PBR framework for sea turtles. Without strong support, the force of bureaucratic inertia will remain an important obstacle to moving beyond the status quo. A potential way to gain support would be to set up a test case, as I have done in this paper. By focusing on a small scale version of how PBR would work, as with the Pamlico Sound flounder gillnet fishery, the calculation of allowable removal levels could be made and compared to the current level of authorized take. Stakeholders could then decide how this might affect them, and with uncertainty removed, there might be more supporters. I believe that gathering the necessary data and calculating PBR would be a useful exercise, and as Biological Opinions and jeopardy decisions are increasingly challenged, managers and other stakeholders will see the advantages of PBR. These supporters could then provide the pressure necessary to change the current sea turtle bycatch management scheme and implement a PBR framework.

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