Managing the Ecosystem by Leverage Points: A Model for a Multispecies Fishery

Nicholas Bax, Alan Williams, Stevie Davenport, and Cathy Bulman
CSIRO (Commonwealth Scientific and Industrial Research Organisation)
Marine Research, Hobart, Tasmania, Australia

Abstract
Ecosystem management has been popularly adopted as a goal of fisheries management, but what does it really mean? In stressed ecosystems, ecosystem functions may remain unchanged while changes in species composition—particularly dominance—and the health of individuals can change dramatically (Schindler et al. 1985). The species are more sensitive indicators of stress than the system. We suggest, therefore, that there is little value in trying to manage a marine ecosystem as a whole. Instead one should manage people's interaction with the particular ecosystem components that influence the quantity and quality of valued products. We call these particular interactions "leverage points" and use a 5-year study of the southeastern Australian continental shelf fishery ecosystem to show how leverage points can be found. Neither nearshore production nor predation on commercial fish species has a major influence on fisheries production; they therefore have little leverage potential. However, the interaction of benthic habitat with fish and fishers is a potential leverage point. Benthic habitat directly influences the fish community, and specific benthic habitats are vulnerable to fishing. Their vulnerability is increasing because of technological advances in accurate positioning (Global Positioning System) and position recording (trackplotters), fishing vessel power, and fishing gear. Accurate positioning can also be used to manage the location of fishing effort. For this leverage point to be successful, the spatial management of fishing effort would need to be developed in collaboration with the fishing industry and to be targeted specifically at the vulnerable habitats. We suggest that new management instruments such as the transferable ecological stock rights are needed to link fisheries management directly to the ecosystem.
Introduction

Ecosystem management is a commonly stated goal of marine fisheries management in Australia and overseas, but what does it mean? Do ecosystems even exist? In 1953, Odum defined “any entity or natural unit that includes living and non-living parts interacting to produce a stable system in which the exchange of materials between the living and nonliving parts follows circular paths is an ecological system or ecosystem.” Forty years later, Golley (1993) reviewing the history of the ecosystem concept in ecology, questions whether the ecosystem concept is reasonable—philosophers require that wholes have genuine properties, where a genuine property is one that is unique to the whole and not reducible to the properties of its components. As a whole, ecosystem properties are not studied in relation to marine fisheries (though see Caddy 1993). Extension of traditional marine fisheries management has frequently been through amalgamating single-species models linked by predation into multispecies models, rather than looking at system properties (e.g., Laevastu and Larkins 1981, Gislason and Helgason 1985). Even in these multispecies models, few studies considered levels higher than individual species (though see May et al. 1979). Given the high public and political profile of ecosystem management, why is this so?

The Experimental Lakes project in Canada (Schindler et al. 1985) provides one answer, and answers Golley’s (1993) question. Systematic acidification of the lakes changed the abundance, composition, and dominance of species, but did not affect genuine ecosystem properties such as productivity and nutrient cycling. Rare species became common and common species of interest to fishery managers, such as trout, became rare. Biological redundancy in the lakes meant that the abundance and activities of individual species changed in response to changing environmental conditions, but the higher-level system properties of the lake, which are a function of the watershed and the atmosphere as well as the biota, were more robust and varied much less in response to environmental change. The authors concluded that higher-level system properties (including species diversity, numbers, and abundance) were not sensitive indicators of ecosystem stress. The more sensitive indicators are species dominance, and species’ physical condition (including disease prevalence). A similar conclusion can be drawn from changes in the continental shelf ecosystem of the U.S. Northeast, where large-scale shifts in species composition resulted from fishing (Sissenwine 1986), while similar changes in system-level processes have not been reported.

Until recently, ecosystem approaches in marine fisheries have concentrated on species interactions. This has happened (1) because the multispecies models were a logical adaptation of familiar single-species models; (2) because the models have an extensive theoretical background; and (3) because they use the sorts of data (species, abundance, diets, growth, and natural mortality rates) that are the fodder of fisheries science
(they can be collected from fishing vessels or fish markets). These models have been used to correct misconceptions of processes at the single-species level and to provide advice on managing multispecies communities (e.g., Gislason and Helgason 1985).

But species interactions, and associated energy flows, are only one facet of ecosystem functioning. Recent technological advances in remote sensing and geographic positioning systems are changing the ways in which we can study the marine environment and therefore the ways in which we can monitor and manage ecosystem processes. We are now no longer absolutely limited by technology in our choice of which aspect of the marine ecosystem to study, but can now choose aspects of marine ecosystems that are likely to benefit most from management intervention.

CSIRO Marine Research is concluding a 5-year study of the southeast Australian continental shelf ecosystem to determine which new management measures would usefully supplement the current single-species management of the South East Fishery. The South East Fishery is Australia’s largest fishery for domestic scalefish markets. Trawling started on the continental shelf in the early 1900s and expanded to the slope in the 1970s and to deepwater pinnacles in the late 1980s. A non-trawl sector uses dropline, demersal longline/setline, gill nets, and traps to target many of the same species as the trawl fishery. Until recently, management of this multispecies fishery (there are about 90 retained species) centered on minimum lengths for some species, minimum cod-end mesh sizes, and protection of estuarine and nearshore waters. In 1985, management tried, but failed, to cap effort through restricting vessel numbers and size. In 1989, total allowable catches and individual transferable quotas were introduced for the 16 main targeted species. However, effort is still increasing—reported trawling in the study area increased 25% in the last 10 years (Unpublished data, N.J. Bax)—and effort is increasingly targeted on specific bottom features. Our study of this system was based on the premise that management interventions aimed at units greater than single species could improve sustainability of this fishery and ecosystem. We present an overview of our results here.

**Methods**

One approach to managing complex systems is to begin by determining where the “leverage” is greatest (Senge 1990). Leverage is based on the notion that small, well-focused actions can produce enduring improvements if they are directed at sensitive system components. We used the notion of leverage to direct our research.

At the outset of the study, a conceptual model of the factors that could affect productivity of the fish community was developed (Fig. 1a) and refined (Fig. 1b). The sampling program focused on key factors and their potential as leverage points for management of fish resources. Four potential leverage points were identified:
1. Primary production from coastal seagrass beds.

2. Predation on commercial fish species.

3. Effects of fishing on commercial fish species.

4. Effects of fishing on benthic habitat.

Results that address leverage points 1, 2, and 4 are presented here.

There were four surveys: July 1993, August 1994, April 1996, and November 1996. Each survey consisted of a broad-scale survey examining the seasonal distribution of biota and physical oceanography of the region, followed by intensive sampling of selected habitats. Collected samples were used to determine the relationships between biological species and the habitat they occupy by analyzing production sources, trophic position, diet, and morphological adaptation.

**Study Area**

The study area was the Australian continental shelf between the latitudes of 36º and 39ºS—the southeastern point of the continental margin where east and south coasts meet (Fig. 2). The shelf, which is defined as the area from the coast out to water depths of ~170-200, is narrow on the east coast (~25 km) and wide on the south coast (~175 km). Several small rivers flow onto the shelf, but their discharge is low. The area has a complex oceanography that is variable between the south and east coasts. Eddy fields from the seasonally variable southward-flowing East Australian Current bring intrusions of continental slope water onto the shelf, particularly in spring and summer (Church and Craig 1998). An underlying northward countercurrent at the shelf break also transports cool slope water onto the shelf (Cresswell 1994). Northerly winds sometimes enhance these intrusions by bringing nutrient-rich water to the surface (Cresswell 1994). Associated with northeasterly winds, intermittent upwellings off the east coast bring cool, nutrient-rich water to the surface (Edwards 1990). An eastward outflow of water from Bass Strait in winter, driven by strong prevailing westerly winds, cascades down the slope to the east of Bass Strait and can be detected over 11,000 km northward along the slope (Tomczak 1985). Average waves in the area are 1-3 m in height with periods of 5-6 seconds, and penetrate to depths 60 m or more (Morrow and Jones 1988). The southeast Australian continental shelf is, therefore, a moderate- to high-energy, wave-dominated environment.

The sediments on the shelf are autochthonous. Continental basement rocks that underlie the sediments outcrop on the inner shelf near the coast. Fossiliferous limestone reefs, consisting largely of bivalve and bryozoan clasts, occur across the shelf. The coarse-grained quartzite sandstones that occur on the inner shelf of the south coast could have originated as elongate sand bodies formed in a high-energy coastal plain environment parallel to palaeo-shorelines.
Figure 1. Conceptual models of factors influencing the southeast Australia continental shelf fish community (a) before sampling began, (b) after the first survey, and (c) at the end of the study.

Figure 2. Location of the study area in southeastern Australia.
Sampling-Site Description

We developed sound working relationships with key fishers of our study area. Once a level of trust had been established, the fishers generously provided us with details on fishing grounds and fish-habitat associations collected over many years of fishing. A map of this information enabled us to identify eight mesohabitats (sensu Greene et al. 1994) with significant topographic heterogeneity to survey (Fig. 3).

Each mesohabitat was divided into two or three macrohabitats—“soft,” “hard,” and “rough”—discriminated from echo returns at 12, 38, and 120 kHz. Discrimination was based on the length and intensity of the tail of the first echo and the intensity of the first and second echoes (Bax et al., In press).

The geomorphology of each macrohabitat was described from sediment and rock samples, and from video and still photographs collected from a towed camera system developed for the study (Barker et al., In press). Current flows around selected topographic features were interpreted from CTD and hydrographic profiles, supplemented in one instance with moored current arrays.
Sources of Production and Trophic Position

A broad collection of biological material was analyzed for stable isotopes to describe production sources and overall system structure. Biological material included hand-collected estuarine seagrasses and marine algae; particulate organic matter and phytoplankton collected on fiberglass filters from water sampled in niskin bottles at the surface and below; zooplankton collected with oblique bongo net (500 µm mesh) tows and a 56-cm drop net (100 µm mesh); sediments and macroalgae collected with benthic grabs and sleds; invertebrates collected with an infauna/epifauna benthic sled; fish collected with demersal and midwater trawls (40 mm cod-end liner), traps, and variable-mesh gillnet; and marine mammals and seabirds collected adventitiously from animals that died of natural causes.

Ratios of $^{13}\text{C}/^{12}\text{C}$ and $^{15}\text{N}/^{14}\text{N}$ were expressed as the relative per mil ($\‰$) difference between the sample and conventional standards (Pee Dee Belemnite carbonate and N₂ in air). The formula used to express these values is:

$$\delta X = \left\{ \frac{R_{\text{sample}}}{R_{\text{standard}}} - 1 \right\} \times 1,000 \‰$$

where $X = ^{13}\text{C}$ or $^{15}\text{N}$; and $R = ^{13}\text{C}/^{12}\text{C}$ or $^{15}\text{N}/^{14}\text{N}$.

Diet

Fish stomachs were collected throughout each cruise to obtain samples of (ideally) each species caught at a range of depths, times, geographical locations, and sizes of fish. Up to 10 stomachs per species per tow and 50 stomachs per species per cruise were taken. Large stomachs were frozen at –20ºC and small preserved in 10% formalin. The length, weight, and sex of donor fish were recorded. Stomachs were assessed for fullness in the laboratory and then dissected. Prey items were identified to the lowest possible taxon, counted, blotted on absorbent paper to remove excess moisture, and weighed (to 0.001 g in the case of very small items). Fish digested beyond recognition were identified from an otolith guide created for this study. Diets were described in terms of wet weight of prey. For the purposes of the present paper all prey items were classified into broad trophic categories.

Fish Communities and Habitat

Following Hudson et al. (1992), we define habitat as “simply the place where an organism lives.” The 19 macrohabitat sites were grouped into habitat types by analyzing fish community structure. Three fishing gears—trawl, gillnet, and trap—were needed to effectively sample the range of bottom types (Bax et al., In press).
Fish caught by all gears were sorted to species, weighed, and counted; all species represented by more than five individuals were measured, and a broad selection of species was retained for morphometric analysis in the laboratory. Abundance data were standardized and communities determined from non-metric multi-dimensional scaling (MDS) and hierarchical agglomerative clustering based on between-sample, Bray-Curtis similarities (Clarke 1993). Data were double-square-root transformed and analyzed with the PRIMER statistical package. A similarities analysis (SIMPER) identified species making the greatest contributions to grouping of macrohabitats by community. “Typifying” species were those contributing most to within-community similarity, while “discriminating” species were those contributing most to between-community dissimilarity. In the figures in this paper we show the species making up 80% biomass (ranked abundance) caught by all gears (“abundant”), less abundant species with typifying or discriminating power (“distinctive”), and species restricted to one habitat type (“indicators”).

**Results**

**Sampling-Site Description**

The sea floor may be visualized as a series of extensive sediment flats (“soft grounds”) with interspersed outcrops of consolidated material (“hard grounds”). Physical and photographic sampling showed soft grounds are composed of particulate material, primarily sands, with areas of mud and gravel, whereas hard grounds include cemented sediments, reefs, and bedrock. These geological features are primary attributes of seafloor habitat in a demersal fishery and determine the distribution and abundance of fishes, and therefore fishing effort.

Geological properties also determine the vulnerability of hard grounds to modification or permanent damage by fishing gear. The key attributes of vulnerability are hardness, relief, and patch size. Fossiliferous limestones, probably often in conjunction with sandstones, comprise most of the hard grounds in the study area, including most large tracts of reef; numerous scattered small outcrops throughout the study area; components of the elongate inner shelf south coast reefs; and patchy, low-relief hard grounds in the deep-shelf “Flower Patch” (Fig. 3). Devonian granite bedrock (probably lateral submarine extensions of adjacent rocky headlands) also outcrops from soft sediments on the inner-shelf south coast.

Composition, relief, and spatial extent of hard grounds are among the factors determining their vulnerability to damage by fishing gears. The extensive, high-relief and heavily cemented mid- to deep-shelf limestone reefs such as “Gabo Reef” are relatively resilient to trawling, as are the localized, high-relief granite outcrops such as “Point Hicks Reef” (Fig. 3). Patchy, low-relief, inner- to mid-shelf limestone reefs that were weathered during previous sea-level regressions, such as “Broken Reef,” are relatively
vulnerable to progressive erosion or removal by trawling. Fishers report that “Broken Reef” is gradually being opened up to trawling as prominent features are mapped and obstructions removed. The patchy hard grounds include relatively small “bryozoan” reefs in mobile substrates towards the shelf edge that support stands of stalked crinoids as in the “Flower Patch” (Fig. 3). Fishers report that some hard ground areas that supported local aggregations of important commercial species have now disappeared, together with the fish.

**Sources of Production**

Stable nitrogen and stable carbon isotopes were analyzed in 1,214 fish (teleost and elasmobranch) samples representing 87 species; 153 invertebrate (benthic and pelagic) samples from eight phyla; 10 species of marine mammal; 1 seabird; 9 species of algae; 91 samples of water column particulates from four surveys and 103 samples of sediment from three surveys (Fig. 4).

Stable carbon ratios for seagrasses are higher ($\delta^{13}C$ less negative) than those of the benthos, fish and higher predators, and therefore cannot have contributed significantly to their trophic pathway. Stable carbon ratios for phytoplankton and zooplankton (and their derivatives—particulate organic matter [POM] and sediments) are consistent with marine phytoplankton being the primary source of production. Therefore the foundations of the ecosystem in the study region are marine phytoplankton. Benthic macroalgae (red and brown) may also contribute to a lesser degree, but pigment analysis of POM and sediments (Unpublished data, S. Davenport) indicate their input to be limited to water depths of 25 m or less. No primary production of terrestrial origins was detected by isotope or pigment analysis (Unpublished data, S. Davenport).

**Predation**

Quota species and 16 other species were examined because of their commercial value or potential importance as predators. Their diets were analyzed and summarized by major prey group: 10 were markedly piscivorous (Fig. 5a). Piscivorous fish occurred in all habitats: for example tiger flathead (*Neoplatycephalus richardsoni*) is a dorsoventrally flattened fish associated with sediment, while striped trumpeter (*Latris lineata*) is an obligate reef dweller. Of the 28 species, the velvet leatherjacket (*Parika scaber*) had no fish in its stomach, while spotted warehou (*Seriolella punctata*), blue warehou (*S. brama*), butterfly perch (*Caesioperca lepidoptera*), sparsely-spotted stingaree (*Urolophus paucimaculatus*), stinkfish (*Synchiropus calauropomus*), and blue morwong (*Nemadactylus douglasii*) had unidentifiable fish in their stomachs. In the remaining 21 piscivorous fishes, we subdivided the fish component of the diet into commercial and non-commercial species. The proportions of commercial fish were small (Fig. 5b). The species with the largest proportion—the striped trumpeter—
Figure 4. Mean stable isotope ($\delta^{13}$C and $\delta^{15}$N) values (± 1SD) of primary producers to apex predators. Cetacean samples integrate isotope values from several ecosystems. Mobile benthos includes: asteroids, bivalves, crustaceans, echinoids, gastropods, octopus, and ophiuroids. Sessile benthos includes: anemones, ascidians, bryozoans, crinoids, soft corals, and sponges.

is a highly valued fish, but is seldom caught in this area so can be considered to be of low abundance. No major predator of commercial fish species was found.

Piscivores other than fish in this area include seals (Arctocephalus pusillus doriferus), dolphins (Delphinus delphis, Tursiops truncatus), shearwaters (Puffinus tenuirostris), and penguins (Eudyptula minor). These animals have a high trophic position (Fig. 4), and are locally (some seasonally) abundant. They eat mainly small pelagic fish (e.g., anchovies, pilchards, mackerel), and cephalopods (e.g., arrow squid), while shearwaters also eat crustaceans. Juveniles of important commercial species in this area are not a large part of their diet.

**Fish Communities and Habitat**

Seven habitat types were defined by the groups of macrohabitats formed from the analysis of fish abundance data (Unpublished data, A. Williams). Habitats were also clearly differentiated by physical features—depth,
Figure 5. Dietary analysis of 28 key South East Fishery fishes: (a) proportion of diet due to pelagic invertebrates, fish, and benthos; and (b) proportion of commercial species in diet.
location, and acoustic bottom type—with only the bottom types with greatest contrast (“soft” and “rough”) grouped separately. Thus four habitats were identified on the east coast—reef and sediment flats on both the inner and outer shelf—and three on the south coast—inner shelf reef and sediment flats between Point Hicks and Gabo Island, and the outer shelf “Horseshoe.” The differences in community composition of the habitats, and therefore the ways in which habitats are used by different suites of fishes, is illustrated here by reference to two contrasting habitats: the Point Hicks-Gabo Island sediment flats and the east coast inner-shelf reef.

The Point Hicks-Gabo Island sediment-flat community (Fig. 6a) was composed of 76 species, and the inner-reef community (Fig. 6b) 55 species. The former was sampled by trawl; both were sampled by trap and gill net. Only three of the abundant species overlapped: jack mackerel (Trachurus declivis) and draughtboard shark (Cephaloscyllium laticeps), which were ubiquitous in this region; and velvet leatherjacket (Parika scaber), which was abundant and broadly distributed across many habitats.

A high proportion of elasmobranchs and a large suite of indicator species distinguished the remainder of the Point Hicks-Gabo Island community. Five of the abundant species—jack mackerel, draughtboard shark, sparsely-spotted stingaree (Urolophus paucimaculatus), piked spurdog (Squalus megalops), and southern eagle ray (Myliobatis australis)—were highly typical of this habitat. The indicator species, which were primarily trawl-caught, comprised four species of flathead (Platycephalidae), three

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**Figure 6.** (Facing page.) “Abundant,” “typical,” and “indicator” groups from fish communities at (a) Point Hicks–Gabo Island sediment flats, and (b) inshore reef habitat. Species, ordered phylogenetically within the groups and with commercially important species in bold, are:

1. Cephaloscyllium laticeps
2. Squalus megalops
3. Urolophus paucimaculatus
4. Urolophus cruciatus
5. Narcine tasmaniensis
6. Cyttus australis
7. Trachurus declivis
8. Synchiropus calaupomus
9. Parika scaber
10. Thyrsites atun
11. Diodon nictemerus
12. Callorhynchus milii
13. Myliobatis australis
14. Galeorhinus galeus
15. Squatina australis
16. Trygonorrhina fasciata
17. Neoplatycephalus richardsoni
18. Platypocephalus arenarius
19. Neoplatycephalus aurimaculatus
20. Platypocephalus longispinis
21. Kathetostoma laeve
22. Ammotretis rostratus
23. Helicolenus percoideus
24. Pseudophycis bouchus
25. Latridopsis forsteri
26. Nemadactylus douglasii
27. Nemadactylus macropterus
28. Seriola brama
29. Caesioperca lepidoptera
30. Hypoplectrodes annulatus
31. Hypoplectrodes maculillochi
32. Callanthis australis
33. Pempheris multiradiata
34. Scorpius lineolatus
35. Atypichthys strigatus
36. Seriola lalandi
37. Ophthalmolepis lineolatus
38. Notolabrus tetricus
39. Bodianus unimaculatus
40. Trygonorrhina fasciata
41. Cheirolepontus spectabilis
42. Ophthalmocephalus lineolatus
43. Neoplatycephalus aurimaculatus
44. Bodianus unimaculatus
rays (Rajiidae), Australian angel shark (*Squatina australis*), school shark (*Galeorhinus galeus*), common stargazer (*Kathetostoma laeve*), and longnose flounder (*Ammotretis rostratus*). Among discriminating species, three provided reliable contrasts with other habitats: school shark, which was moderately abundant and restricted to this site; piked spurdog, which was highly abundant and otherwise restricted mainly to the outer shelf; and elephant fish (*Callorhynchus milii*), which was moderately abundant but found elsewhere only at an adjacent granite reef habitat.

A high proportion of reef specialists, including a large suite of indicator species, distinguished the inner shelf reef community (Fig. 6b). Among the abundant fishes, blue warehou (*Seriolella brama*), bastard trumpeter (*Latridopsis forsteri*), and blue morwong (*Nemadactylus douglasii*) were typical and differentiated this habitat from other inner shelf habitats. The two ubiquitous species—draughtboard shark and jack mackerel—were also typical. A further 12 species occurred only in this habitat type: three wrasses (Labridae), three perches (Serranidae), two sweeps (Scorpidae) and yellow-tail kingfish (*Seriola lalandi*), snapper (*Pagrus auratus*), common bullseye (*Pempheris multiradiata*), and banded morwong (*Cheilodactylus spectabilis*).

**Discussion**

It is the existence of observers who notice what is going on that imparts reality to the origin of everything. When we choose to experiment for one aspect, we lose our ability to see any others. Every act of measurement loses more information that it obtains, closing the box irretrievably and forever on other possibilities. (Wheatley 1992)

Ecosystem management requires a model of system structure and processes. The model may be a precise mathematical model or a less well defined set of beliefs, but in choosing it many other potential models will be omitted. No model will fully represent ecosystem structure and the processes, so it is important to be aware of the serious omissions. For example, there has been much debate on the relative merits of “bottom-up” or “top-down” trophic models of aquatic systems, but relatively little on whether trophodynamics itself is the appropriate focus. Models are not only a “marvelous crutch to the imagination” (Larkin 1978); when taken too literally they limit imagination and the growth of understanding. A good model is a disposable model.

Our first model of the southeast Australian shelf ecosystem was that the demersal trawl fishery caught demersal fish and that benthic habitat was essential to these fish communities (Fig. 1a). On our first demersal trawl survey we caught a high proportion of pelagic and benthopelagic fish—for example, the very abundant carangid, jack mackerel. It was clear that our conceptual model was wrong or incomplete. We therefore extended the model to coarsely represent production sources as well as extractive processes (Fig. 1b), but we left the link between benthic habitat
and fish communities unspecified. Given the broadened scope, it was clear that we did not have sufficient resources to study all aspects of system structure, so we concentrated on those we thought had leverage potential. For our purposes we defined potential leverage points as system structures or processes to which our chosen output measure (fisheries production) was sensitive and, as importantly, structures or processes that were amenable to management intervention.

The first potential leverage point that we identified was the input of primary production from seagrass. Estuarine and terrestrial sources of primary production, including seagrasses, have been identified as contributing to production over the entire continental shelf for 110 km off northeast Australia (Risk et al. 1994), and seagrass is important in the trophic ecology of juvenile blue grenadier off southeast Australia (Thresher et al. 1992). Thus it seemed plausible that seagrass production was an important source of primary production for the southeast shelf. Seagrass conservation also provided an attractive management option, because seagrass acreage in Australia has been greatly reduced (Poiner and Peterken 1995), seagrass coverage is easily monitored, and seagrass conservation could involve fishers in ecosystem management without affecting their own livelihoods. However, stable isotope analyses (backed up by analysis of photoreactive pigments; Unpublished data, S. Davenport) could detect no contribution of seagrass or terrestrial production to the continental shelf food webs. Shallow water red and brown algae may contribute to local primary production, but sources are local and not amenable to management intervention. The primary source of production for the shelf ecosystem is pelagic phytoplankton in the open ocean. This production source is also not amenable to management intervention at the local scale.

Our second potential leverage point was predation on fish, a well-studied aspect of ecosystem interactions (e.g., Bax, In press). It has been suggested that the abundance of desirable fish species could be increased by removing their predators (Gulland 1982, Harwood and Greenwood 1985). Marine mammals and birds in the area are strongly piscivorous, as indicated by their enriched δ15N ratios (Fig. 4). Diet studies, however, show that they eat mainly surface and midwater pelagic species, including jack mackerel and Australian pilchard (*Sardinops neopilchardus*). These species are part of the midwater prey community, sustained by euphausiids and lanternfish, and exploited by many taxa including tuna and pelagic sharks (e.g., Young et al. 1997). Some of the fish species caught with demersal nets were piscivorous, but ate few commercial species. If the more abundant piscivorous species, such as jack mackerel, eat commercial fish (even occasionally), they could have a marked impact. However, the abundant piscivores had essentially no commercial species in their diets, although unidentified fish in stomach contents could have hidden predation on the larvae of commercial species. Many taxa feed on the midwater food and there may be competitive interactions among them, but in practice it would be very difficult to demonstrate that resources were limiting to the
extent that competition was occurring. Monitoring and managing competitive interactions would prove even harder.

The third potential leverage point was the direct impacts of fishing on fish populations; indirect impacts, for example, fish feeding on discards, has yet to be addressed for this system. Direct impacts are well covered in annual assessment reports (summarized in Caton et al. 1997) and in focused discarding studies (Liggins 1996). Discarding of commercial species can be high, either because they are too small for the market or because market prices are temporarily too low to cover transport costs. Discarding of juvenile redfish (*Centroberyx affinis*), for example, can exceed 90% of catch in some ports (Liggins 1996). There is an ontogenetic change in habitat with movement to greater depth for many commercial species on the southeast Australian continental shelf (Unpublished data, A. Williams). Therefore, most discards of many commercial species are caught in shallow waters, typically either when sea conditions prevent vessels from fishing offshore, or when they are targeting marketable commercial species whose adults occur in shallow waters. This is an obvious leverage point. Modifications to gear and fishing practices have the potential to reduce discarding (Bax 1997) and thereby affect fish populations, but the implications for fishers’ activities and financial return have not been determined.

The link between the fish community and habitat was one potential leverage point we identified. The impacts of demersal trawling on benthic organisms, habitat, and fish communities have been well documented (e.g., Jones 1992, Schwinghammer et al. 1996, Sainsbury et al. 1997). Comparisons of the diets of fish species caught in different habitats did not indicate any particular trophic link with habitat (Unpublished data, C. Bulman). However, multispecies abundances clearly delineated fish communities associated with distinct habitats. Individual species were mostly either obligate or facultative users of particular habitat types, and rarely ubiquitous. Analysis of the shape and morphology of obligate and facultative habitat users suggested that the relationship between habitat and fish might be mediated through fish seeking refuge from prevailing currents. Fish found in current-swept sediment flat habitats were frequently dorso-ventrally flattened for low drag, or were burrowers or sustained swimmers (Fig. 6a). Fishes found in topographically complex reef areas were mostly deep-bodied, with specializations such as fin shape and positioning that would confer good maneuverability (Fig. 6b). Although we cannot determine the full scope of relationship between benthic habitat and fish community, the distribution of morphotypes together with measurements of water chemistry and currents around reefs, indicates that habitat topography has a role through changing current flow. It may not be necessary to define the link between benthic habitat and fish populations precisely because the association of many taxa with structural habitat implies an increase in individual fitness that would be lost if the structural features were lost (Auster and Malatesta 1995). In addition, even if
particular benthic habitat conferred no increase in individual fitness, the role of particular habitat types in aggregating particular species would increase fishers' effectiveness. Because major commercial species in the South East Fishery are managed by quota, measures that increase fishers' effectiveness without increasing habitat impacts may aid conservation by reducing effort.

Fishers target very specific habitats on the southeast Australian shelf. They report that some key habitats have been, and are being, impacted by fishing. For example, low-relief limestone reefs that traditionally yielded good catches of high-value fishes such as snapper are being eroded or removed. Patchy mosaics of low-relief reef are particularly vulnerable to being “opened-up” as vessels become more powerful and use thicker warps and heavier bottom gear on trawls. The gear development that has made precise targeting possible is the combination of GPS and electronic trackplotters, which enable skippers to plot obstacles precisely and to either avoid or remove them.

The links between fish communities and benthic habitat suggest that habitat preservation could be a strong leverage point. Some fishers have spoken out on the need to preserve habitat, but may be reluctant to diminish their own catching efficiency unless other fishers also avoid—and are seen to avoid—the sensitive habitat. For fishers to agree to limitations on their fishing practices they must see the potential benefits clearly and also accept that any restrictions are not excessive. For example, although some topographically complex habitats are vulnerable to fishing impacts, other complex habitats (for example, those based on granite or large contiguous areas of fossiliferous limestone) are less vulnerable. At the moment they are considered untrawlable. However, trawlable areas close to these complex habitats are prime fishing grounds. Restricting fishing on all complex habitat, regardless of its vulnerability to fishing, would unnecessarily impede fishing on these prime grounds. Other topographically complex habitats are vulnerable to fishing and it is these that should be targeted by habitat-based management. Habitat-based management need not require that habitats be closed to all fishing, so long as management objectives are clearly specified and outcomes monitored. Satellite-linked vessel monitoring systems, as used to manage effort in the Australian orange roughy fishery, provide one means of monitoring.

An alternative to closing particular habitats is to limit their use through economic means. Fishers in the South East Fishery pay an annual levy for fishery management based on the estimated market value of their individual transferable quota (ITQ) holdings. No account is taken of the biological or environmental impacts of their fishing practices, although managing biological impacts is the goal of single-species management, and managing broader environmental impacts is one goal of ecologically sustainable development (ESD)—a legislative requirement for the Australian Fisheries Management Authority. As Alain Laurec of the European Union said in reference to sustainable fisheries: “Limiting catches is a symptom
of the disease rather than the cure" (Senior 1996). One proposed alternative to ITQs is transferable dynamic stock rights based on a fraction of a year class rather than a set tonnage, enabling a fisher to profit from catching his/her fraction of the year class at an appropriate biological (or economic) age (Townsend 1995). Future stock rights could also be dependent on the opportunity a fisher's year-class fraction has had to contribute to future generations before being caught.

Transferable dynamic stock rights have attractions, but because they require monitoring of catch and discarded catch to be effective, they would be cumbersome to monitor and enforce in most fisheries. We propose a modification of these rights: transferable ecological stock rights. In this instance a fisher would be given the right to harvest a certain fraction of a year class subject to the perceived ecological damage associated with harvesting. Monitoring (satellite-derived positions for fishing vessels) and enforcement would be based on the distribution of fishing effort in relation to habitat as a proxy for the likelihood of catching (and discarding) immature fish or causing ecological damage. If fishing in shallow waters where smaller fish reside would be expected to lead to higher discarding, then landed catch would count more against stock rights than a similar tonnage landed in deeper waters. In a similar fashion, fish caught from fishing in sensitive areas or with gear that damages benthic habitat would attract a higher deduction from that year's stock rights. Transferable ecological stock rights would provide managers an instrument more clearly linked with the goals of ecosystem management and ESD than ITQs are—and would treat the problem, not the symptoms.

Improved remote sensing and satellite-tracking technology has enabled scientists to cost-effectively research new features of marine ecosystems. The same technology has enabled fishers to target particular habitats more precisely, increasing their impact on particular productive habitats. Limiting landed catch no longer meets the requirements of managers attempting to satisfy goals of ecosystem management and ecologically sustainable development. Management of marine ecosystems requires more than management of landed catches. "Fisheries management is environmental management" (Martin Cabot, head, Newfoundland Inshore Fishermen's Association, quoted in Griffin 1993). If fisheries managers are to become environmental managers, then fisheries (environmental?) scientists must provide them with the appropriate concepts, tools, and information. In a complex system it will be essential to understand where the leverage points are. We have identified one such point for the continental shelf off southeast Australia, but it remains for managers and fishers, supported by scientists to determine how this particular leverage point can be used profitably.
Acknowledgments

We gratefully acknowledge our many colleagues at CSIRO who have contributed to this project. Of special note are the biological technicians who have processed and analyzed data, the crew of the Southern Surveyor who helped collect the data, and the technical and workshop staff who designed and fabricated items of equipment for us. We also thank the commercial fishers who gave us the benefit of their accumulated knowledge of the area, which was critical to planning and interpreting our comparatively brief scientific surveys. Drs. Tony Smith and Xi He and our editor Vivienne Mawson improved the manuscript with their reviews. This study was supported by research grant 94/040 from the Fisheries Research and Development Corporation.

References


Sustainability: Empirical Examples and Management Implications

National Marine Fisheries Service, National Marine Mammal Laboratory, Seattle, Washington

Abstract

Management in regard to ecosystems must meet a number of criteria. As indicated in Table 1, such management must: (1) be consistent with management in respect to other biological systems; (2) account for reality, including uncertainty; (3) result in components of each level of biological organization falling within normal ranges of natural variation; (4) exercise precaution and consider risk in achieving sustainability; (5) be information-based and interdisciplinary; (6) include monitoring and assessment; (7) have clear objectives; (8) recognize that our ability to control is limited to managing human activity; and (9) consider humans as part of ecosystems.

Management to meet these requirements can be guided by empirical examples of sustainability demonstrated by other species. An example is consumption of resources within ecosystems by predators. Limits to sustainability are demonstrated by frequency distributions of consumption rates observed among predators. These empirical examples are the result of a variety of influences and constraints, including processes of natural selection resulting from exposure to the suite of factors we wish to account for in management. In this paper, the consumption by marine vertebrates is presented to empirically exemplify sustainable resource consumption from both ecosystems and individual species.

The implications of empirical examples of sustainability for management are emphasized by the central tendencies in frequency distributions among species. The lack of consumption rates beyond the tails of observed distributions indicates that consumption rates in these two extremes are not sustainable over long time scales.
Introduction

Much has been published on the topic of management involving ecosystems (e.g., Christensen et al. 1996; Mangel et al. 1996; Fowler 1999, In press). The requisites for successful management are numerous but can be condensed to the nine essential elements or criteria presented in Table 1, all of which need to be adhered to simultaneously (Fowler 1999). While conceptually rich, the available literature merely describes reasonable management methods and serves well to eliminate alternatives but does not specify an acceptable approach.

Existing forms of management routinely fail to conform to one or more of the criteria presented in Table 1, exposing the difficulty of finding a practical form of management that meets all nine. In this paper, we suggest a way forward in the form of management that follows the example of other species, treating them as empirical models of sustainability (Fowler 1999; Fowler, In press). We exemplify this approach by considering management of the harvest (take, utilization, or consumption) of biomass from both ecosystems and individual resource species as two levels of biological organization.

Management at the Ecosystem Level

What level of biomass consumption in an ecosystem achieves the greatest sustainability? To answer this question, we begin by examining relevant empirical information provided by other species, then turn to management based on this information.

Figure 1 shows frequency distributions for estimated rates of consumption among 13 species of marine mammals found in the Bering Sea. Information for each species is represented through its contribution to the height of the bar corresponding to the biomass it is estimated to consume (log10 metric tons; Sobolevsky and Mathisen 1996). Figure 2 shows a similar distribution representing biomass consumption by 24 species of marine mammals and seabirds from the Georges Bank ecosystem off the east coast of the United States. These nonhuman species of the Bering Sea and Georges Bank are empirical examples of varying degrees of sustainability within their ecosystems having, in part, survived the risks of extinction to which they are exposed. Thus, Figs. 1 and 2 illustrate the types of information that can be used to answer the question raised at the outset of this section.

How can we base management on such information? Generally, management would restrict consumption by humans to within the bounds of the normal range of natural variation in the sustainability exhibited by other species. Specifically, in this case, catches by commercial fisheries would be confined to within the range shown in Figs. 1 and 2 as one step toward assuring that humans consume resources sustainably.
Table 1. A list of criteria that must be met, or principles to be adhered to, by any form of management, particularly any that applies to the human use of natural resources (Christensen et al. 1996, Mangel et al. 1996, Fowler 1999).

1. Management of the harvest of biomass from individual resource species cannot be in conflict with management of the harvest of biomass from the ecosystems in which the harvested species occur. Similarly, biomass consumption by humans from the biosphere must be guided by principles that are not in conflict with those guiding the harvest of biomass from either an individual resource species or any particular ecosystem. Any form of management must apply simultaneously at the various levels of biological organization and it must do so consistently.

2. Management action must be based on an approach that accounts for reality in its complexity over the various scales of time, space, and biological organization. The context of environmental factors (e.g., ecological complexity) must be accounted for along with the elements of stochasticity and the diversity of processes, mechanics, and dynamics. The complexes of chemical and physical substances and processes as well as energetic dynamics must be taken into account. Furthermore, we must be able to deal with uncertainty, including what we cannot know, or may never know.

3. A core principle of management is that of undertaking actions that ensure that individuals, species, and ecosystems are within (or will return to) their respective normal ranges of natural variation as components of the more aggregated levels of biological organization (Rapport et al. 1981, 1985; Christensen et al. 1996; Holling and Meffe 1996; Mangel et al. 1996). Any form of management must apply this principle.

4. Management must be risk-averse and exercise precaution in achieving sustainability. Sustainability is, by definition, not achieved by any form of management that generates risk rather than minimizing it.

5. Guidance must be available to management in the form of useful information that enables managers to develop meaningful, measurable, and reasonable goals and objectives. This information must be based on interdisciplinary approaches to adhere to the principle behind criterion 2.

6. Management must include science (scientific methods and principles) in research, monitoring and assessment, not only to produce the information that is used for guidance (criterion 5), but also for evaluation of progress in achieving established goals and objectives (criterion 7).

7. There must be clearly defined goals and objectives that are measurable (quantifiable) to provide quantitative evaluation of problems to be solved and gauge progress in solving them.

8. It must be recognized that control over other species and ecosystems is impossible. The only option for control is the control of human action (Christensen et al. 1996, Holling and Meffe 1996, Mangel et al. 1996). For example, we can control fishing effort but not the resource population or ecosystem in which it occurs. We can influence the resource population and its ecosystem, but we cannot control them to avoid indirect changes, side effects, or secondary reactions brought about by our influence. The guidance (criterion 7) we need for management is guidance regarding the level of influence (e.g., harvest rate) that meets the other criteria of this list.

9. Humans must have the option of being components of at least some ecosystems to avoid the unrealistic option of precluding human existence.
Figure 1. Two frequency distributions representing consumption rates ($\log_{10}$ metric tons consumed annually) by 13 cetacean (whale) species from the Bering Sea showing variability and its limits among these species. This figure compares distributions for the mid-1900s (a) and the late 1980s and early 1990s (b). From Sobolevsky and Mathisen (1996).

Figure 2. A frequency distribution representing the Georges Bank ecosystem showing variability among 24 species of marine mammals and birds, as distributed according to estimated annual biomass consumption for each species ($\log_{10}$ metric tons) within this region. From Backus and Bourne (1986).
The concept of using other species as empirical examples of sustainability may seem simple on the surface. In practice, however, implementing this form of management is extremely difficult. This is true at the ecosystem level as introduced above as well as at the individual species level discussed in our next example.

**Management at the Single-Species Level**

Criterion 1 (Table 1) requires that management be consistent in its application at both the single-species level and the ecosystem level. One of the many questions facing managers regarding individual species is: “What level of harvest (e.g., biomass consumption) is the most sustainable?” To address such questions we must consider more than the population dynamics of the resource species. It is important to account for evolution and coevolution, including anthropogenic impacts (i.e., the genetic influence of harvesting; Law et al. 1993). This helps account for systemic complexity (criterion 2, Table 1).

Frequency distributions of estimated rates of consumption among fishes, birds, and marine mammals as predators on specific prey species are shown in Figs. 3 and 4. Similar to the ecosystem approach outlined above, a sustainable level of harvest from each individual species would be achieved by management to restrict commercial harvests to levels within the normal range of natural variation exhibited by the respective set of consuming species exemplified in Figs. 3 and 4. Again, implementation of such a strategy is complicated. Yet if such an approach were employed, the risks that preclude species from occurring beyond the range of such distributions would be avoided. These risks include those posed by long-term ecosystem change stimulated by the influence of species that even briefly occur in the upper extreme of the normal range of natural variation. This approach leads to what can be called ecologically sustainable yield rates (ESY for the yield, ESYR for the rate; Fowler 1999) which correspond to the central tendencies (e.g., mean) of frequency distributions similar to those illustrated in Figs. 3 (yields) and 4 (rates).

**Meeting Management Standards**

How does the management described above meet the criteria presented in Table 1? Fishing practices in our examples would be constrained based on information like that exhibited in Figs. 1-4, but not prohibited. Thus, humans would be constrained (criterion 8) to fall within the normal range of natural variation (criterion 3), rather than being excluded (criterion 9). The approach is risk-aversive and precautionary (criterion 4) because commercial harvests confined to levels near the central tendencies of such distributions avoid the collective risks and constraints that have prevented the accumulation of species in the tails of such distributions. Thus, sustainability (criterion 4) is more likely in the central regions of species
Figure 3. The frequency distribution of nonhuman vertebrate species that consume hake (Merluccius bilinearis; N = 11), herring (Clupea harengus; N = 12), mackerel (Scomber scombrus; N = 16), and sand eel (Ammodytes americanus; N = 13) measured as the log$_{10}$ (metric tons) of biomass consumed in an area (ecosystem) of the northwest Atlantic Ocean (Overholtz et al. 1991; pers. comm., S.A. Murawski and W.J. Overholtz, National Marine Fisheries Service, Northeast Fishery Science Center, Woods Hole, MA 02543).

Figure 4. The frequency distribution of 21 nonhuman vertebrate species that consume walleye pollock (Theragra chalcogramma) in the Bering Sea and North Pacific, according to the log$_{10}$ of the fraction of the standing stock biomass of pollock they consume (Livingston 1993 and pers. comm. 1994).
frequency distributions than it is in the tails. As such, the central tendencies of frequency distributions provide specific measures that can be used to define goals or objectives for management (criteria 5 and 7).

The various temporal and spatial scales and elements of complexity are accounted for in this approach (criterion 2), although the available data are limited in this regard. Long time scales and broad spatial scales are accounted for, in part, by the evolutionary dynamics that influence where species fall in the range represented by species frequency distributions. Natural selection affects these distributions both through classical individual selection and the dynamics of selective extinction and speciation (Lewontin 1970; Slatkin 1981; Arnold and Fristrup 1982; Fowler and MacMahon 1982; Levinton 1988; Eldredge 1989; Williams 1992; Fowler, In press). Extinction is one of the risks that prevent the accumulation of species in the tails of such distributions. All forms of selection are influenced by the variety of factors making up the reality (e.g., ecological conditions, criterion 2) to which species are exposed (and simultaneously a part).

Applied at both the single-species and ecosystem levels (as well as other levels of biological organization, criterion 1 of Table 1), this approach meets all of the requirements of management presented in Table 1. Criterion 6 is met when science is used to produce data for species frequency distributions and to monitor change.

**Limitations to Existing Data**

Existing data are clearly imperfect. The information in species frequency distributions would be more reliable if integrated (e.g., averaged) over longer periods of time. Patterns (e.g., correlations) across environmental factors such as mean temperature and annual variance in temperature should be taken into account. The same holds for spatial patterns. Scientists are faced with an immense challenge in providing such information.

It is important to remove the effects of factors that may introduce bias. The species frequency distributions in Figs. 1 and 2 are probably biased. Ideally, we want data that represent ecosystems for which human influence is either absent or historically within the normal ranges of natural variation. The information in Fig. 1a cannot be considered free of such influence, but may better reflect the natural state of the ecosystem than that presented in Fig. 1b. Different levels of human influence may explain part of the differences between Figs. 1a and 1b. These include the cumulative effects of considerable sealing, whaling, and fishing, plus a number of other factors such as atmospheric transport of pesticides and their degradation products (National Research Council 1996). Interdisciplinary efforts will be necessary to produce adequate data by taking such factors into account.

Similarly, the data summarized in Fig. 2 may be influenced by commercial fishing, especially as harvest rates have often been outside the
normal range of natural variation for the Georges Bank ecosystem. Changes introduced to such distributions by human influence could include increased variation, biased mean, or modified shape.

Temporal dynamics and measurement bias also likely affect species frequency distributions. Figures 1 and 2 represent little more than a snapshot of a dynamic system. Beyond process-related variance, there is error and bias attributable to measurement and estimation procedures that influence observed distributions.

Thus, the quality of existing data is one problem. Our choice and interpretation of data can result in others. It is important to avoid being misled. It is important to account for factors that are involved in correlative relationships as would be the case in our examples where trophic level and total consumption are presumably interrelated. Tiny invertebrates might be inferior as examples of sustainable biomass consumption compared to small cetaceans that consume at the same trophic level as humans. Care, and often much more research, are needed to find the subsets of data that provide adequate and relevant information regarding the normal ranges of natural variation to guide decision-making.

**Discussion**

In this paper, we have formulated management by using nonhuman species as natural empirical examples of sustainability. Humans cannot claim to be operating sustainably when we are found at the extremes of the range of natural variation. Species in the tails of frequency distributions like Figs. 1-4 do not serve as good examples, especially if they have been there only briefly.

We must now deal with the burden of proof (Mangel et al. 1996, Dayton 1998). Consumption rates by nonhuman predators empirically exemplify sustainability and are clearly more proven alternatives for human fisheries management than harvest rates based on population models (e.g., maximum sustainable yield [MSY] rates). For a variety of reasons it would seem on the surface that sustainability can be maximized for harvest levels in or beyond the upper tails of unbiased frequency distributions (e.g., through consideration of population dynamics and resulting estimates of MSY). There certainly is no lack of desire for greater harvest rates (e.g., considering economic, historical, or cultural factors). But now we must prove that harvest rates larger than the central tendencies of unbiased frequency distributions achieve greater sustainability if we are going to use them. To do otherwise is not precautionary.

We recommend consideration of the approach described herein by using the preliminary information in cases represented by frequency distributions such as those of Figs. 1-4 (Fowler 1999; Fowler, In press), in spite of their limitations. Management based on such information satisfies all of the nine criteria laid out in Table 1 and therefore satisfies the
need for management that applies at the ecosystem level. Doing so leads to application at other levels of biological organization and to other ways of measuring species, thereby further adhering to criteria 1 and 2 of Table 1.

However, the acceptance and implementation of the management approach that we propose are extremely difficult. This difficulty emphasizes the complexity with which we are dealing. Existing fisheries harvest at rates that are often more than ten-fold greater than the mean consumption rates for nonhuman predators. Making the changes required to implement management as we describe it here is even more of a challenge than the research needed for more reliable information. Involved are institutional, social, economic, political, and behavioral changes, many of which are beyond comprehension. The magnitude of these challenges, however, is more a measure of the size of the problems to solve than justification for avoiding the work required.

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References


Incorporating Ecosystem Considerations into Management of Bering Sea Groundfish Fisheries

David Witherell
North Pacific Fishery Management Council, Anchorage, Alaska

Abstract
The North Pacific Fishery Management Council has made significant progress toward ecosystem-based management of the Bering Sea and Aleutian Islands area. First and foremost, the council has taken a precautionary approach to extraction of fish resources. As a result, all groundfish stocks covered under the fishery management plan are considered healthy and not overfished. Potential impacts of fishing on other ecosystem components such as marine mammals and seabirds are also considered. The council’s approach involves public participation, reliance on scientific research and advice, conservative catch quotas, comprehensive monitoring and enforcement, bycatch controls, habitat conservation areas, and other biological and socioeconomic considerations.

Introduction
Ecosystem-based management strategies are being adopted throughout the United States in response to biodiversity concerns. As defined by the Ecological Society of America, ecosystem management is “management driven by explicit goals, executed by policies, protocols, and practices, and made adaptable by monitoring and research based on our best understanding of the ecological interactions and processes necessary to sustain ecosystem composition, structure, and function” (Christensen et al. 1996). In more general terms, ecosystem-based management is shorthand for a more holistic approach, which focuses more on maintaining system integrity than on maximizing extraction of certain resources.

The precautionary principle has developed over the past 10 years as a policy measure to address sustainability of natural resources in the face
of uncertainty. The principle is becoming widely adopted throughout the world in both national and international environmental policies. The central element of the precautionary principle is that precise impacts caused by human activity cannot be known with certainty, so that a more cautious approach is required (Dovers and Handmer 1995). This is particularly true when there is a high level of uncertainty and there are large (potentially irreversible) costs if a mistake is made (Garcia 1995). Fisheries management around the world had traditionally been based on a preventative and trial-and-error approach, yet the collapse of some fisheries indicates that a more precautionary approach should have been applied.

Although the precautionary principle and ecosystem-based management are relatively new concepts being debated in the scientific literature, the North Pacific Fishery Management Council has used these approaches for management of North Pacific groundfish fisheries since 1976. The council is a regional organization established by the Magnuson-Stevens Act in 1976 when the United States extended its fisheries jurisdiction out to 200 miles. The council, together with the National Marine Fisheries Service, has primary responsibility for groundfish management in the Gulf of Alaska, Bering Sea, and Aleutian Islands area, encompassing about 900,000 square nautical miles. Conservative management policies were implemented right from the start, with adoption of the first fishery management plans (FMPs). Since 1984, the council’s number one comprehensive management goal has been to conserve fishery resources, maintain habitats, and give full consideration for interactions with other elements of the ecosystem. The council has also had a comprehensive policy on habitat since 1988. The objectives of this policy are to maintain the current quantity and productive capacity of habitats, and restore and rehabilitate any habitats previously degraded.

The primary goal of the Bering Sea and Aleutian Islands (BSAI) groundfish FMP is to promote conservation while providing optimum yield from marine resources. To accomplish this goal, the FMP specifies six objectives for conservative management (Table 1), all of which are consistent with a precautionary approach and ecosystem-based management. Clearly, these objectives have been met successfully, as all BSAI groundfish stocks are considered healthy after 20 years of sustained annual harvests of nearly 2 million t. No fish stocks were deemed overfished in a recent evaluation of the status of U.S. fisheries (National Marine Fisheries Service 1997). When revised overfishing definitions are implemented in 1999, only one fish stock in the region (Bering Sea Chionoecetes bairdi crab) is expected to be below its minimum stock size threshold and declared overfished.

Concerns about the impacts of fish removals on other components of the ecosystem have motivated the council to continue development of a more ecosystem-based management strategy for the Bering Sea. This development has progressed at all levels, from science to policy making. Since 1995, the Groundfish Plan Teams have prepared an Ecosystem Considerations section to supplement the annual Stock Assessment and Fish-
The chapter provides an annual assessment of the ecosystem, a review of recent ecosystem-based management literature, updates of ongoing ecosystem research, and new information on the status of seabirds, marine mammals, habitat, and other components of the North Pacific ecosystem.

In 1996, the council established an Ecosystem Committee to discuss possible approaches to incorporating ecosystem concerns into the fishery management process. The committee has held workshops on ecosystem research, held several meetings to discuss essential fish habitat, and has hosted numerous informal discussions on ecosystem-based management and habitat concerns. A major role of this committee has been to provide the council and stakeholders information on ecosystem-based management in the North Pacific. The committee identified primary principles and elements of ecosystem management from scientific literature (e.g., Grumbine 1994, Mangel et al. 1996, Christensen et al. 1996). The concept of ecosystem-based management includes the elements of sustainability, goals, ecological models and understanding, complexity, dynamic character, context and scale, adaptability, and humans as ecosystem components. The committee’s draft policy for ecosystem-based management of North Pacific fisheries is shown in Table 2.

Fisheries can impact ecosystems in numerous ways, including the selectivity, magnitude, timing, location, and methods of fish removals. Fisheries can also impact ecosystems by gear interactions, vessel disturbance, nutrient cycling, introduction of exotic species, pollution, and habitat alteration. Managers attempt to minimize potential impacts while at the same time allowing the extraction of fish resources at sustainable levels. A review of ecosystem-based management actions taken by the council is provided below.
### Table 2. Draft ecosystem-based management policy of the North Pacific Fishery Management Council.

<table>
<thead>
<tr>
<th>Definition</th>
<th>Ecosystem-based management, as defined by the NPFMC, is a strategy to regulate human activity towards maintaining long-term system sustainability (within the range of natural variability as we understand it) of the North Pacific, covering the Gulf of Alaska, the Eastern and Western Bering Sea, and the Aleutian Islands region.</th>
</tr>
</thead>
<tbody>
<tr>
<td>Objective</td>
<td>Provide future generations the opportunities and resources we enjoy today.</td>
</tr>
</tbody>
</table>
| Goals           | 1. Maintain biodiversity consistent with natural evolutionary and ecological processes, including dynamic change and variability.  
                    2. Maintain and restore habitats essential for fish and their prey.  
                    3. Maintain system sustainability and sustainable yields of resources for human consumption and non-extractive uses.  
                    4. Maintain the concept that humans are components of the ecosystem. |
| Guidelines      | 1. Integrate ecosystem-based management through interactive partnerships with other agencies, stakeholders, and public.  
                    2. Utilize sound ecological models as an aid in understanding the structure, function, and dynamics of the ecosystem.  
                    3. Utilize research and monitoring to test ecosystem approaches.  
                    4. Use precaution when faced with uncertainties to minimize risk; management decisions should err on the side of resource conservation. |
| Understanding   | 1. Uncontrolled human population growth and consequent demand for resources is inconsistent with resource sustainability.  
                    2. Ecosystem-based management requires time scales that transcend human lifetimes.  
                    3. Ecosystems are open, interconnected, complex, and dynamic; they transcend management boundaries. |
Management Actions

Conservative Catch Limits

Total removals of groundfish are controlled by annual catch limits established for each stock. For target species, three harvest levels are set, corresponding to the overfishing level (OFL), the acceptable biological catch (ABC), and total allowable catch (TAC). TACs are essentially annual catch limits for the fishery, and are established at or below the ABC. ABCs generally define acceptable harvest levels from a biological perspective, and OFL defines the unacceptable harvest level. Specification of harvest limits is done in a precautionary manner, due to a number of reasons as explained below.

Harvest rates specifications are more conservative when less information is available (Thompson 1997). The maximum allowable rates are prescribed through a set of six tiers which are listed below in descending order of preference, corresponding to descending order of information availability (Table 3). For most tiers, ABC is based on $F_{40\%}$, which is the fishing mortality rate associated with an equilibrium level of spawning per recruit (SPR) equal to 40% of the equilibrium level of spawning per recruit in the absence of any fishing. The $F_{40\%}$ rate is considered to be a very conservative harvest rate for most fish stocks (Clark 1993, Rosenberg and Restrepo 1995). To further minimize the possibility of catches jeopardizing a stock's long term productivity, there is a buffer established between ABC and OFL. The OFL definition was recently increased from $F_{30\%}$ to $F_{35\%}$ for stocks having tiers 2-4 information.

Harvest rates used to establish ABCs are reduced at low stock size levels, thereby allowing rebuilding of depleted stocks. If the biomass of any stock falls below $B_{msy}$ or $B_{40\%}$ (the long-term average biomass that would be expected under average recruitment and $F = F_{40\%}$), the fishing mortality is reduced relative to stock status. This serves as an implicit rebuilding plan should a stock fall below a reasonable abundance level. The council has a record of rebuilding groundfish stocks that were depleted prior to implementation of the Magnuson-Stevens Act. For example, conservative harvest policies adopted in the 1980s had the effect of restoring depleted stocks such as yellowfin sole, *Pleuronectes asper*, and sablefish, *Anoplopoma fimbria* (Megrey and Wespestad 1990).

As a result of these definitions, specified harvest rates for groundfish stocks are very low. Actual harvest rates are significantly lower for many species, as the TAC may be set lower than ABC, and harvests may be less than TAC due to regulatory closures. All fish caught in any fishery (including bycatch), whether landed or discarded, accrue towards the TAC for that stock. Although 100% mortality for all discards is assumed, some fish likely survive, so actual removals are lower than published catch numbers indicate.

Additional precaution is incorporated into the BSAI groundfish catch specification. Since 1981, the total annual allowable catch for BSAI ground-
Table 3. Tiers used to determine catch specifications for BSAI groundfish stocks as approved under Amendment 56.

1. Information available: Reliable point estimates of $B$ and $B_{MSY}$ and reliable pdf of $F_{MSY}$.
   1(a) Stock status: $B/B_{MSY} > 1$
   \[ F_{OFL} = m_A, \] the arithmetic mean of the pdf
   \[ F_{ABC} \leq m_H, \] the harmonic mean of the pdf
   1(b) Stock status: $a < B/B_{MSY} \leq 1$
   \[ F_{OFL} = m_A \times (B/B_{MSY} - a)/(1 - a) \]
   \[ F_{ABC} \leq m_H \times (B/B_{MSY} - a)/(1 - a) \]
   1(c) Stock status: $B/B_{MSY} \leq a$
   \[ F_{OFL} = 0 \]
   \[ F_{ABC} = 0 \]

2. Information available: Reliable point estimates of $B$, $B_{MSY}$, $F_{MSY}$, $F_{30\%}$, and $F_{40\%}$.
   2(a) Stock status: $B/B_{MSY} > 1$
   \[ F_{OFL} = F_{MSY} \]
   \[ F_{ABC} \leq F_{MSY} \times (F_{40\%}/F_{35\%}) \]
   2(b) Stock status: $a < B/B_{MSY} \leq 1$
   \[ F_{OFL} = F_{MSY} \times (B/B_{MSY} - a)/(1 - a) \]
   \[ F_{ABC} \leq F_{MSY} \times (F_{40\%}/F_{35\%}) \times (B/B_{MSY} - a)/(1 - a) \]
   2(c) Stock status: $B/B_{MSY} \leq a$
   \[ F_{OFL} = 0 \]
   \[ F_{ABC} = 0 \]

3. Information available: Reliable point estimates of $B$, $B_{40\%}$, $F_{30\%}$, and $F_{40\%}$.
   3(a) Stock status: $B/B_{40\%} > 1$
   \[ F_{OFL} = F_{35\%} \]
   \[ F_{ABC} \leq F_{40\%} \]
   3(b) Stock status: $a < B/B_{40\%} \leq 1$
   \[ F_{OFL} = F_{35\%} \times (B/B_{40\%} - a)/(1 - a) \]
   \[ F_{ABC} \leq F_{40\%} \times (B/B_{40\%} - a)/(1 - a) \]
   3(c) Stock status: $B/B_{40\%} \leq a$
   \[ F_{OFL} = 0 \]
   \[ F_{ABC} = 0 \]

4. Information available: Reliable point estimates of $B$, $F_{30\%}$, and $F_{40\%}$.
   \[ F_{OFL} = F_{35\%} \]
   \[ F_{ABC} \leq F_{40\%} \]

5. Information available: Reliable point estimates of $B$ and natural mortality rate $M$.
   \[ F_{OFL} = M \]
   \[ F_{ABC} \leq 0.75 \times M \]

   \[ OFL = \text{the average catch from 1978 through 1995, unless an alternative value is established by the SSC on the basis of the best available scientific information} \]
   \[ ABC \leq 0.75 \times OFL \]

$ABC$ is acceptable biological catch, $OFL$ is the overfishing level, $F$ is instantaneous fishing mortality rate, $B$ is exploitable biomass, “pdf” is probability density function, and $MSY$ is maximum sustainable yield.
fish complex must fall within an optimum yield range of 1.4-2.0 million t. This has limited the sum of TACs for all species to 2 million t per year, which has been considerably less than the sum of all ABCs (averaging about 2.8 million t per year). As a result, most groundfish stocks, particularly flatfish stocks, are being exploited well below sustainable levels (Witherell 1995). Based on observer data and reports provided by the fleet, the National Marine Fisheries Service closes directed fisheries for each species or complex prior to when the TAC is taken. As such, inseason managers have been effective at maintaining catches of groundfish within biologically acceptable levels (Fig. 1).

**Bycatch Controls**

Bycatch management measures implemented for groundfish fisheries of the eastern Bering Sea have focused on reducing the incidental capture and injury of species traditionally harvested by other fisheries. These species include king crab, *Paralithodes* and *Lithodes* spp.; Tanner crab, *Chionoecetes* spp.; Pacific herring, *Clupea pallasi*; Pacific halibut, *Hippoglossus stenolepis*; and Pacific salmon and steelhead trout, *Oncorhynchus* spp. Collectively, these species are called "prohibited species," as they cannot be retained as bycatch in groundfish fisheries and must be discarded with a minimum of injury.

Bycatch controls were instituted on foreign groundfish fisheries prior to passage of the Magnuson-Stevens Act in 1976 and have become more
restrictive in recent years (Witherell and Pautzke 1998). Bycatch limits for 1998 BSAI groundfish fisheries included 3,775 t of halibut mortality, 1,697 t of herring, 100,000 red king crabs, 2,850,000 C. bairdi crab, 4,654,000 C. opilio crab, 48,000 chinook salmon, and 42,000 other salmon. Bycatch limits for herring and crab are based on biomass of those stocks, and therefore fluctuate from year to year. The bycatch limits are apportioned to specific groundfish target fisheries and may also be seasonally apportioned. Attainment of any apportionment closes that groundfish target fishery for the remainder of the season.

To address Magnuson-Stevens Act mandates to reduce bycatch, the council recently adopted an amendment to prohibit the use of non-pelagic trawl gear for vessels targeting pollock in the BSAI. Only pelagic trawl gear as defined in regulations (together with the performance-based bycatch standard of 20 crabs) will be allowed in the directed pollock fishery. Although this action could have been taken annually as part of the BSAI TAC specification process, the plan amendment will make this prohibition a permanent regulation. Total bycatch limits of prohibited species will be reduced to reflect this gear prohibition. Prohibited species bycatch will be reduced by 100 t of halibut mortality, 3,000 red king crab, 50,000 C. bairdi crab, and 150,000 C. opilio crab.

**Limits on Discards and Waste**

The issue of discarding and waste of fish resources stems from social, economic, and conservation concerns. Fish are discarded for two reasons: either they are required to be thrown back due to regulations, or they are unwanted by that fishing vessel. Discards of unwanted fish (so-called economic discards) result when fishermen do not have markets, sufficient equipment, time, or return to retain and process the catch. In the 1997 BSAI fisheries, a total of 258,000 t of groundfish were discarded, equating to about 15% of the total groundfish catch. Although this discard rate is much lower than most of the world’s fisheries (Alverson et al. 1994), the sheer volume of the discards is troublesome to many people who consider economic discards as waste of food and an impact to the ecosystem.

To reduce discarding, the council adopted an improved retention and utilization program for all Bering Sea and Aleutian Islands groundfish fisheries. Beginning in 1998, 100% retention of pollock (Theragra chalcogramma) and Pacific cod (Gadus macrocephalus) was required. Rock sole (Lepidopsetta bilineata) and yellowfin sole (Limanda aspera) retention will be required beginning in 2003, with the delay time necessary to allow for development of markets and gear technological responses by the vessels engaged in these fisheries. These retention requirements are expected to reduce overall discard rates from 15% to less than 5%. The council addressed the utilization side of the program not by mandating specific product forms, but instead by requiring a minimum required product recovery rate of 15%.
Marine Protected Areas

The council and the Alaska Board of Fisheries have established several marine protected areas to protect habitat for fish, crabs, and marine mammals. Three large areas of the Bering Sea have been closed to groundfish trawling and scallop dredging to reduce potential adverse impacts on the habitat for crab and other resources (Fig. 2). The Pribilof Islands Conservation Area was closed to protect blue king crab habitat (primarily shell hash). The Red King Crab Savings Area was established to protect adult red king crab and their habitat, and as a precautionary measure to reduce potential unobserved mortality. To protect juvenile red king crab and critical rearing habitat (stalked ascidians and other living substrate), Bristol Bay was declared another marine protected area. Specifically, the area east of 162°W longitude is closed to trawling and dredging, with the exception of a small area that opens for a brief period each spring to accommodate a yellowfin sole trawl fishery. A limited amount of longlining for Pacific cod and halibut, as well as pot fishing for Pacific cod and crabs occurs within all three of these marine protected areas.

These marine protected areas comprise a relatively large portion of the continental shelf, and in many respects, serve as marine reserves. In total, these three area closures encompass about 30,000 square nautical miles. To put this in perspective, this is an area more than twice the size of Georges Bank off the New England coast. Lauck et al. (1998) recently suggested that marine reserves should be at least 20% of available habitat in order to be effective. The Bering Sea marine protection areas exceed this threshold by encompassing about 25% of the shelf where commercial quantities of groundfish can be taken with bottom trawl gear, based on
interpolation of data from Fritz et al. (1998). Marine protection areas of the Bering Sea include essential fish habitat (EFH) for crab species and fish species such as walleye pollock, Pacific cod, yellowfin sole, rock sole, and other flatfish species (North Pacific Fishery Management Council 1998).

The Sustainable Fisheries Act of 1996 amended the Magnuson-Stevens Fishery Conservation and Management Act to require the description and identification of essential fish habitat (EFH) in FMPs, adverse impacts on EFH, and actions to conserve and enhance EFH. EFH assessments for fish species covered under Alaska’s five FMPs (BSAI groundfish, GOA groundfish, BSAI crab, Alaska scallops, Alaska salmon) were adopted by the council in June 1998. EFH assessment reports will form the basis for management actions taken to conserve and enhance essential fish habitat in the Alaska region. Additional marine protected areas may be required to minimize fishing impacts on EFH, particularly in sensitive, rare, and vulnerable habitats.

**Marine Mammal and Seabird Considerations**

In addition to setting maximum harvest levels, fisheries have been both seasonally and spatially allocated to reduce potential impacts of localized depletion of prey. For example, the Bering Sea pollock TAC is split among a winter fishery (A-season) and a late summer fishery (B-season). Seasonal and regional apportionment is also done for Atka mackerel (*Pleurogrammus monopterygius*) in the Aleutian Islands. The council recently adopted a regulation to reduce fishing for Atka mackerel near rookeries to reduce potential for localized depletion of Atka mackerel and competition with Steller sea lions (*Eumetopias jubatus*), an endangered species. Because Atka mackerel and pollock are important prey for higher trophic levels, these measures reduce the impacts of harvesting on the ecosystem.

Area closures have also been implemented to prevent disrupting marine mammals at rookeries and haulouts, and to reduce competition from fisheries. To protect Pacific walrus (*Odobenus rosmarus*), fishing vessels are prohibited in that part of the Bering Sea within 12 miles of Round Island, the Twins and Cape Peirce in northern Bristol Bay during the period April 1 through September 30. To protect Steller sea lions, no trawling is allowed year round in the BSAI within 10 nautical miles of 27 Steller sea lion rookeries (Fig. 3). In addition, six of these rookeries have 20 nautical mile trawl closures during the winter pollock season. There are additional rookery closures in the Gulf of Alaska as well.

In 1997, the council adopted an amendment that prohibits directed fishing for forage fish, which are prey for groundfish, seabirds, and marine mammals. Under this amendment, protection is provided for forage fish species such as capelin (*Mallotus villosus*) and a host of other forage species such as euphausiids (krill). The council took this proactive approach by preventing fisheries for these important species from expanding or developing.
Concern for the incidental bycatch of seabirds (the endangered short-tailed albatross, Diomedea albatrus, in particular) led to regulations requiring deterrent devices to be employed on groundfish longline vessels beginning in 1997. Approximately 9,600 seabirds (including about 1 albatross) are incidentally killed in Alaska groundfish fisheries each year (Wohl et al. 1995). It is hoped that these deterrent devices, which are actively being developed and improved upon by fishermen, will significantly reduce incidental mortality.

**Discussion**

Fisheries management of Bering Sea groundfish provides an excellent example of sustainable fisheries using a precautionary approach. Bering Sea fisheries have produced annual landings of nearly 2 million t of groundfish over the past 20 years, yet groundfish stocks in the Bering Sea have remained healthy to this day. The council and the National Marine Fisheries Service have used a precautionary approach by relying on scientific research and advice, conservative catch quotas, comprehensive monitoring and enforcement, bycatch controls, habitat conservation areas, and additional ecosystem considerations. By incorporating ecosystem considerations into management of groundfish fisheries, human impacts can be lessened, while at the same time providing sustained yields of fishery resources.

It is widely recognized that the Bering Sea ecosystem is driven by climate-induced variability and is subject to regime shifts, which have
occurred in the past (McGowan et al. 1998). Populations of some marine mammals (e.g., Steller sea lions), seabirds (e.g., red legged kittiwakes at St. Paul Island), crab (e.g., Tanner crab), and prey species (such as capelin) are in relatively low abundance compared to historical records. Other populations appear to be in relatively high abundance (e.g., orcas, northern fulmars, snow crab, rock sole). It has been hypothesized that large-scale removals of whales from 1848 through 1976 caused major cascading effects (National Research Council 1996). From 1950 to 1976, estimated total catches of whales in the BSAI and Gulf of Alaska exceeded 5,700 blue whales, 26,000 fin whales, 74,000 sei whales, 30,000 humpback whales, and 210,000 sperm whales (National Marine Fisheries Service 1991). It is likely that the Bering Sea ecosystem, including groundfish populations, is still responding to this massive perturbation. Future changes to the ecosystem should be expected.

References


Integrated Operational Rule Curves for Montana Reservoirs and Application for Other Columbia River Storage Projects

Brian L. Marotz  
Montana Fish, Wildlife and Parks, Kalispell, Montana

Daniel Gustafson  
Montana State University, Department of Biology, Bozeman, Montana

Craig Althen and Bill Lonon  
Montana Fish, Wildlife and Parks, Kalispell, Montana

Abstract

Reservoir operation guidelines were developed to balance resident fisheries concerns with anadromous species recovery actions in the lower Columbia River. Fisheries requirements were integrated with power production and flood control to reduce the economic impact of basin-wide fisheries recovery actions. These Integrated Rule Curves (IRCs) were developed simultaneously in the Columbia Basin System Operation Review (SOR 1995), the fish and wildlife program of the Northwest Power Planning Council (NPPC), and recovery actions for endangered fish species (Marotz et.al. 1996). IRCs were adopted into the council’s fish and wildlife program (NPPC 1994). However, the IRC operations were supplanted by the Biological Opinion (BiOp) of the National Marine Fisheries Service (NMFS 1995). BiOp operations are designed to enhance the downstream migrational success of populations of juvenile chinook and sockeye salmon from the Snake River listed under the Endangered Species Act. The BiOp calls for summer releases of water from storage projects, including Hungry Horse and Libby reservoirs. BiOp operations cause drawdown of these reservoirs during the summer and unnatural flow fluctuations downstream causing impacts to the aquatic ecosystem. The decision to release water from headwater dams to augment summer flows downstream pivots on the potential benefits to anadromous fish relative to the potential impacts to resident fish.
Introduction

The construction and operation of dams in the Columbia River basin in western North America negatively impacted aquatic and riparian environments. Dams alter stream hydrology, isolate fish spawning migrations, and directly kill fish. Snake River chinook salmon (Oncorhynchus tshawytscha), sockeye salmon (O. nerka), several steelhead stocks (O. mykiss), Kootenai River white sturgeon (Acipenser transmontanus), and bull trout (Salvelinus confluentus) have been listed as threatened or endangered under the Endangered Species Act. Recently, dam operations at many federally operated dams were altered to restore these dwindling fish populations (USFWS 1994; NMFS 1995, 1998). As restoration actions are taken, system operations should be designed to benefit all native species in the watershed.

Prior to dam construction, the hydraulic cycle in Columbia River tributaries included a high flow event during spring snowmelt (late May through early June) and a stabilized low flow period throughout the remainder of the year (Parret and Hull 1985). Hydropower operations reversed this discharge pattern by storing water during the runoff and releasing water during the fall and winter, when flows were naturally at their lowest level. Loss of the spring freshet is believed to be a primary factor in the decline of anadromous and resident fish populations in the Columbia River basin (ISG 1996).

Montana has resident fish at risk in the Flathead and Kootenai river systems (Fig. 1), including the westslope cutthroat trout (O. clarki lewisi), interior redband trout (O. mykiss), the bull trout, and the Kootenai river white sturgeon. The Kootenai River white sturgeon is listed as endangered under the Endangered Species Act (59 Fed. Reg. 45989; 1994), the bull trout as threatened (63 Fed. Reg. 31647; 1998), and the westslope cutthroat trout and interior redband trout are candidate species of special concern (Williams et al. 1989). These fish were adversely affected by the Montana dams.

The Kootenai River white sturgeon is a genetically distinct population that occurs from Kootenai Falls below Libby Dam downstream to Kootenay Lake in British Columbia. There has been no significant recruitment of juveniles into this population for 25 years since the construction of Libby Dam. Less than 2,000 individuals remain. The aging fish in the population may not recruit young in sufficient numbers to avoid extinction. Conservation stocking has been initiated to avoid extinction while actions are taken to enhance natural reproduction. Evidence suggests that flow and temperature are related to reproduction and survival (Apperson and Anders 1991, Apperson 1992). The operation of Libby Dam is critical to white sturgeon recovery.

The bull trout is the largest native trout in Montana. Bull trout numbers have been reduced by habitat degradation and negative interactions with non-native species (Montana Bull Trout Scientific Group 1998). Some of the strongest meta-populations exist in the Canadian headwaters of
Figure 1. Flathead and Kootenai systems of Montana.
Libby Reservoir and in the South Fork Flathead River upstream of Hungry Horse Dam. Fortunately, Hungry Horse contains a native species assemblage and much of their habitat is in wilderness and national forest that retains many natural attributes. The headwaters of Libby Reservoir also contain sufficient habitat suitable for the persistence of bull trout. Dam operation directly affects reservoir and riverine habitat, insect production for juveniles, and the availability of fish prey for adults.

The westslope cutthroat trout has been reduced to less than 10% (estimated approximately 5%) of the species’ historic range. Major threats to this species are habitat loss, genetic introgression, and competition with introduced species (Van Eimeren 1996). The most secure meta-population of the westslope cutthroat trout exists in Hungry Horse Reservoir and the South Fork Flathead River upstream. Libby Reservoir headwaters also contain genetically pure stocks. After westslope cutthroat emigrate from their natal tributaries to the reservoir or river, the availability of insects, zooplankton, and habitat is controlled by dam operation.

The interior redband trout is the only rainbow trout native to Montana (Allendorf et al. 1980). Redband trout have been reduced to a fraction of their historical range (Williams et al. 1989, Behnke 1992). In Montana, redband occur only in five Kootenai River tributaries. Only the Callahan Creek population is known to migrate to the Kootenai River where the availability of food and habitat is directly related to dam operation. Redband trout were also documented in the Yaak River, a Kootenai River tributary. Interior redband are a category 2 subspecies; listing may be warranted but precluded due to lack of biological information in their present range.

Recent changes in dam operation to aid in the recovery of salmon and steelhead in the lower Columbia River basin have caused undesirable reservoir and river operations in headwater storage projects. The operating plan described herein reduces the impacts to native fish species in the Columbia River headwaters while maintaining features essential to the National Marine Fisheries Service’s operational strategy (NMFS 1995, 1998). Our Integrated Rule Curves (IRCs) for dam operation are a series of drawdown and refill targets for operating the dams that incorporate incremental adjustments for uncertainties in water availability (Marotz et al. 1996). The IRCs limit the frequency of deep drawdowns and reservoir refill failure and produce a more natural discharge hydrograph (Fig. 2). Actual operations will vary due to inflow forecasting error. This flexibility allows operators to market power.

For Montana reservoirs, the intent is to reduce reservoir drawdown and increase refill probability to protect the most biologically productive period from mid to late summer through fall (May et al. 1988, Chisholm et al. 1989). At full pool, the reservoirs contain the maximum volume of optimal-temperature water for fish growth, and a large surface area for the deposition of terrestrial insects from the surrounding landscape. Reduced drawdown protects aquatic food production in the reservoirs, assuring an ample springtime food supply for fish.
Figure 2. Integrated Rule Curves for Hungry Horse (top) and Libby Dam (bottom) operation. Reservoir elevations are selected on monthly inflow forecasts beginning in January. Elevational targets are interpolated when the inflow forecast is intermediate between volumes described by the curves shown above.
The IRCs create a naturalized annual discharge hydrograph. In a natural river environment, the nearshore habitat is productive and critical to fish. Riparian vegetation reestablishes seasonally, providing secure habitat along river margins and reducing erosion of silt into the river. Spring flushing flows sort river gravel and define the channels creating a healthy environment for fish and their prey organisms. Conversely, rapid flow fluctuations caused by power operations intermittently flood and desiccate shoreline habitats. Aquatic insects, fish, and fish eggs occupying the zone of water level fluctuation, or varial zone, may become stranded on the dry banks (Hauer et al. 1994, 1997). The varial zone is biologically unproductive. Intermittent high discharges disrupt the natural revegetation process. Vegetation that would normally provide secure habitat and stabilize soils cannot fully reestablish. The IRCs cause discharges from the dam to decline gradually, reducing biological impacts.

Methods

Hydraulic Modeling

This operating plan was based on over 16 years of field and laboratory research. We modeled the hydropower system from the headwaters downstream (Fraley et al. 1989, Marotz et al. 1996). Forecasting error was applied to each dam to minimize error propagation. The models interpolate elevational targets in the reservoir based on inflow volume. This technique mimics the decision process used by dam operators. Dam operators receive their first annual inflow forecast in early January. As forecasts are updated monthly, the operator adjusts the elevation target to the expected inflow volume, allowing operational flexibility as runoff forecasts vary over time. The IRCs were smoothed using Microsoft Excel and multiple iterations of the reservoir models HRMOD and LRMOD (Marotz et al. 1996).

The IRCs incorporate a new strategy for system flood control. The operation currently being implemented by the U.S. Army Corps of Engineers (ACOE) attempts to store as much of the spring runoff as possible. This requires a large reservoir drawdown to evacuate sufficient storage to contain the spring runoff. As the reservoir refills with snowmelt, discharge is typically held to the minimum allowable flow. Conversely, the variable discharge strategy (VARQ) embodied in the IRCs releases a naturally shaped spring freshet and stores only the amount of water that would exceed flood stage downstream (ACOE 1997). Flood control criteria at downstream locations (e.g., the Flathead River at Columbia Falls; and Kootenai River at Bonners Ferry, Idaho, and Kootenay Lake, British Columbia (IJC 1938)), further limit the maximum allowable flow.

During 1996 and 1997, the ACOE Hydraulics Branch reevaluated flood control requirements in the Columbia River basin. The study was conducted in accordance with NMFS (1995) and USFWS (1994), and the fish and wildlife program of the Northwest Power Planning Council (NPPC 1994). The ACOE critically compared the IRCs and VARQ and determined that the strat-
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Strategies were similar (ACOE 1997). However, in years of less-than-average water availability, VARQ required less drafting for flood control and reservoir elevations were higher than the IRCs. Operational flexibility above the IRCs can be used to save more water during dry years to augment spring flows. It should be noted, however, that in this analysis the model was configured to select the lower of the two targets (IRC as opposed to VARQ). Therefore, possible additional benefits to anadromous recovery by implementing VARQ during years of less-than-average water were not presented. For years of average or high water, VARQ and IRCs were modeled identically.

**Thermal Modeling**

Thermal effects downstream of Hungry Horse and Libby dams are controllable by regulating the depth of water withdrawal at the dams (Christenson et al. 1996, Marotz et al. 1996). For modeling comparisons, withdrawal depths were standardized across operational alternatives using an automated selective withdrawal (thermal control) model component. This resulted in identical discharge temperature under differing operational strategies.

**Comparison with the NMFS 1995 Biological Opinion Operations**

Comparisons of the IRC with the NMFS operation plan were conducted using the reservoir models (LRMOD and HRMOD). The two operational plans differed from each other in nearly parallel ways at Hungry Horse and Libby dams. For sake of brevity, only information for Libby Dam is presented. The operations specified by the NMFS 1995 Biological Opinion (BiOp) were provided by the Dittmar Control Center of the Bonneville Power Administration (BPA), Study 98C-01.OPERB (NMFS 1995; Pers. comm., Roger Schiewe, BPA, and Michael Newsom, NMFS, Portland, OR).

Plots of reservoir elevations and dam discharge schedules were overlaid for visual comparison of the two alternatives during low (inflow of 6.068 million acre feet [MAF]), average (8.088 MAF), and high (> 10.110 MAF) water availability. A representative NMFS operation for years of low, average, and high water availability was constructed by selecting five or more years with inflows approximately equal to the specified annual inflow volumes (±0.5 SD). We then calculated the mean elevation for each of the 14 periods. This was necessary to mask the effect of differences in water availability in the main-stem Columbia, relative to the Kootenai subbasin (i.e., water availability in the Kootenai system varies somewhat independently from water availability in the lower Columbia River). Water years included in the composite operations are shown in Table 1.

The NMFS operation assumed that storage reservoirs would be drafted to 20 feet below full pool in August only if the seasonal target (July 1-August 31) of 200 thousand cubic feet per second (kcf/s) at McNary Dam was not met. This resulted in varying degrees of reservoir drafting (0-20
Table 1. Water years included in the NMFS composite operation.

<table>
<thead>
<tr>
<th>Water availability</th>
<th>Water year&lt;sup&gt;a&lt;/sup&gt;</th>
<th>Annual inflow (MAF)</th>
</tr>
</thead>
<tbody>
<tr>
<td>High</td>
<td>1956</td>
<td>10.863</td>
</tr>
<tr>
<td></td>
<td>1934</td>
<td>10.658</td>
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<tr>
<td></td>
<td>1959</td>
<td>10.496</td>
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<td></td>
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<td>10.068</td>
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<td></td>
<td>1976</td>
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<td>Medium</td>
<td>1963</td>
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<td></td>
<td>1953</td>
<td>8.088</td>
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<td></td>
<td>1935</td>
<td>8.046</td>
</tr>
<tr>
<td></td>
<td>1932</td>
<td>8.017</td>
</tr>
<tr>
<td>Low</td>
<td>1929</td>
<td>6.259</td>
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<td></td>
<td>1936</td>
<td>5.974</td>
</tr>
<tr>
<td></td>
<td>1945</td>
<td>5.904</td>
</tr>
</tbody>
</table>

<sup>a</sup> A water year extends from October 1 through September 30.

...feet from full pool during August) throughout the 50-year record, and caused the composite data to underestimate the effect of summer flow augmentation.

**Biological Modeling**

Comparisons of trophic responses resulting from the alternative dam operational strategies were examined using the empirically calibrated biological reservoir models HRMOD and LRMOD (Fraley et al. 1989, Marotz et al. 1996). Model simulations were configured for annual, as opposed to continuous runs.

**Fish Entrainment**

Fish entrainment through Libby Dam was qualitatively assessed using the empirically calibrated entrainment model developed by Skaar et al. (1996). Since the two operational alternatives compared herein are hypothetical, comparative field data were not possible. Nonetheless, trends in fish density and vertical distribution can be extrapolated from sampling conducted by Skaar et al. (1996). Discharges during spring and summer can be accurately estimated through computer modeling. We assumed that seasonal trends in vertical fish distributions were constant to qualitatively assess entrainment under the operational alternatives as correlated to discharge volume.
Results and Discussion

Libby Reservoir Conditions

Alternative 1: IRC/VARQ

The reduced summer drawdown and improved refill probability resulting from the IRC/VARQ operation (Figs. 3, 4, and 5) protects aquatic and benthic food production in the reservoirs during the warm months, late May through early September. Overall, this operation sustains roughly 70% of the optimum reservoir productivity (Table 2).

Entrainment of fish through Libby Dam is proportional to discharge volume. During spring, fish are concentrated near the surface associated with warmer water in the top 20 m (Skaar et al. 1996). Fish densities are highest during spring. As a result of the tiered flow approach, highest entrainment rates would occur in years of above-average water availability. Lowest entrainment rates would occur in years of below-average water, proportional to lower discharge volumes. Fish entrainment during spring would be similar to the NMFS 95 BiOp Alternative 2 in wetter years and less than the NMFS 95 BiOp during average or drier years. During summer, areal fish densities are lower than in May and June, although densities are typically higher in August than in late fall and winter. Entrainment during August resulting from the IRC/VARQ alternative would be the less as compared to the Alternative 2.

Alternative 2: NMFS Biological Opinion

Computer simulations performed at BPA Dittmar Control Center show that the NMFS BiOp, in attempting to meet an August flow target of 200 kcf/s at McNary Dam in the lower Columbia River, reduces reservoir refill probability (Wright et al. 1996). In some years, Libby Reservoir fails to refill by 20 feet or more, thus affecting the sustainability of the operation.

Failure to refill the reservoirs by July and draw down during the summer reduce biological production in the reservoirs (Table 2). The food web supporting fish is most productive in the shallower and warmer littoral or nearshore zones during the summer months. Frequent dewatering reduces the biomass of larval insects that are killed as water recedes. A brief deep drawdown which exposes a large percentage of the reservoir bottom requires at least 2 years for aquatic insect populations to rebound. Also, the contribution of terrestrial insects as a food source for fish is reduced as the reservoir surface recedes from shoreline vegetation. Terrestrial insects are most abundant near the shore from June through September when the NMFS operation is lower than the IRCs. Zooplankton, an important food for juvenile trout and adults during winter, is washed out of the reservoirs through dam turbines as the reservoirs shrink. Thus excessive reservoir drawdown and refill failure negatively impact fish food availability and, therefore, fish growth.

The effects of the 1995 BiOp were recently assessed by the Independent Scientific Advisory Board (ISAB 1997a). The NMFS asked whether the
Figure 3. Comparison of Libby Reservoir elevations (top) and dam discharge (middle) and Kootenai River discharge at Bonners Ferry (bottom) under low water conditions.
Figure 4. Comparison of Libby Reservoir elevations (top) and dam discharge (middle) and Kootenai River discharge at Bonners Ferry (bottom) under average water conditions.
Figure 5. Comparison of Libby Reservoir elevations (top) and dam discharge (middle) and Kootenai River discharge at Bonners Ferry (bottom) under high water conditions.
Table 2. Trophic responses in Libby Reservoir calculated using the reservoir model LRMOD (Marotz et al. 1996).*

<table>
<thead>
<tr>
<th>Water avail.</th>
<th>Name</th>
<th>Primary production (t)</th>
<th>Secondary production (t)</th>
<th>Terrestrial insect deposition by insect order (% maximum)</th>
<th>Fish growth, kokanee TL (mm)</th>
<th>Weight (g)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Carbon fixed</td>
<td>Wash-out</td>
<td>Zoop prod</td>
<td>Bent³</td>
<td>Col</td>
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<tr>
<td>Low</td>
<td>NMFS</td>
<td>11,836</td>
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<td>1,354</td>
<td>382.1</td>
<td>74.1</td>
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<tr>
<td></td>
<td>KIRC</td>
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<td>1,489</td>
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<td>79.0</td>
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<tr>
<td>Avg.</td>
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<td>1,265</td>
<td>337.3</td>
<td>62.0</td>
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<td>1,236</td>
<td>229.7</td>
<td>56.6</td>
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<tr>
<td></td>
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<td>11,680</td>
<td>45</td>
<td>1,335</td>
<td>301.5</td>
<td>62.7</td>
</tr>
</tbody>
</table>

* Results represent phytoplankton production (metric tons of carbon fixed) calibrated by Cl⁴ liquid scintillation. Phytoplankton washout through the dam (metric tons) calibrated by chlor a vertical distribution and entrainment sampling. Total zooplankton production (metric tons) calibrated on phytoplankton production and seasonal measures of carbon transfer efficiencies. Benthic production (metric tons of emergent insects) calibrated on depth distribution of insect larvae and emergence captures. Terrestrial insect deposition (percentage of maximum) by insect order (Col = Coleoptera, Hem = Hemiptera, Hom = Homoptera, and Hym = Hymenoptera), calibrated on nearshore (< 100 m) and offshore surface insect tows. Fish growth (end-of-year kokanee size) in total length (TL) and weight (grams) calculated through multivariate analysis on water temperature structure and food availability.

³ Benthic insect production is artificially enhanced by reservoir refill failure. This single year event is caused when the warm epilimnetic water settles over substrate containing high larval densities (in the infrequently dewatered zone), thus enhancing larval production and emergence. A single deep drawdown event or reservoir refill failure can impact benthic insect production for 2 years or longer.

⁴ KIRC = Kootenai Integrated Rule Curve. The KIRC was modified by the Kootenai River white sturgeon recovery team to balance the needs of the endangered white sturgeon with the reservoir fishery and recovery actions for anadromous species in the lower Columbia River.
resident fish populations are at risk of extinction in Libby and Hungry Horse reservoirs due to flow augmentation strategies. The ISAB (1997a) found that the biological effects of summer drafting will not likely drive resident fish populations to extirpation in Montana, however, late summer reservoir drawdown adversely effects resident fishes in the reservoirs and downstream through flow fluctuation in the streams and lakes below the two reservoirs.

Under the NMFS 95 BiOp alternative, fish entrainment through Libby Dam (proportional to dam discharge) would be higher during spring in average and dry years, and higher during August, compared to the IRC/VARQ alternative.

**Kootenai River Conditions Downstream of Libby Dam**

**Alternative 1: IRC/VARQ**

A comparison of Columbia River flows during spring performed by Wright et al. (1996) revealed that flows resulting from the IRCs were nearly the same as the NMFS BiOp. Spring discharges in the Kootenai River, resulting from the IRC/VARQ operation are nearly consistent with the NMFS BiOp during years of average to high water. During years of low water, the tiered flow approach incorporated into the IRC/VARQs releases less water than called for by the NMFS BiOp (Fig. 3).

The VARQ flood control strategy, if used, would allow greater operational flexibility during drier years than average years. Reservoir elevations higher than the IRCs could be achieved. Thus, operators could store more water that can be released to augment stream flows for listed resident and anadromous stocks. A naturalized spring freshet (greater than provided by the IRC strategy) could be created, even in dry years, without compromising reservoir refill. In average or wetter years, VARQ and Integrated Rule Curves were identical. This improves on historical operations because reservoir elevations remain higher prior to the spring runoff, so that a larger percentage of the runoff volume can be shaped to create a normalized spring freshet.

The IRC alternative was designed to gradually ramp down from the spring peak to reduce flow fluctuations. During dry years, the maximum drawdown of the reservoir was reduced consistent with the NMFS 95 BiOp and VARQ to increase the volume of pass-through flows during spring runoff. In wetter years, the discharge was smoothed to further extend the descending limb of the hydrograph.

Maximum flows regulated by storage dams are less than would occur prior to dam construction. Maximum allowable dam discharge is dictated by the physical capacity of the turbines and acceptable spill levels. Libby Dam presently contains five turbines that can release a maximum of 27 kcfs, collectively. The spillway entrains atmospheric gas during operation, so only a small percentage of the total flow can be spilled before
Montana water quality laws pertaining to dissolved gas are violated (e.g., not to exceed 110% gas saturation). The maximum flow is typically limited by turbine capacity plus additional flows from unregulated tributaries that enter downstream from the dam. Flood control criteria at Bonners Ferry, Idaho, and Kootenay Lake, British Columbia (IJC 1938), further limit the maximum allowable flow.


Clean gravels with subsurface and groundwater inflow are sought by nest-building salmonids and broadcast spawners (Peters 1962). Gravels consisting of less than 30% fines (< 0.65 cm) provide suitable oxygenation for egg incubation and sack fry, enhancing early-life survival (Weaver and Fraley 1993). Preferred spawning substrate for white sturgeon, which are broadcast spawners, consists of gravel, cobbles, and boulders (Parsley et al. 1993, Hildebrand and McKenzie 1994).

Fine clays, silts, sands, and organic materials deposited in low-velocity areas (e.g., high on the streambanks) become dry as spring flows gradually recede. If stream flows stabilize, this rich soil becomes bound by rooted terrestrial vegetation. Erosion and subsequent siltation of the streambed is reduced. Fine materials in the stream support aquatic plants which provide habitat for aquatic and terrestrial organisms. The IRCs gradually ramp down from the spring runoff peak and restore these favorable biological conditions.

The benefit to anadromous salmon would be greater flows in the spring, a gradual rampdown from the spring freshet that enhances summer flows, and improved reservoir levels during drought cycles so that the operation is sustainable from year to year.

**Alternative 2: NMFS Biological Opinion**

The August releases called for by the NMFS BiOp are designed to aid the migration of juvenile Snake River salmon in the lower Columbia River. The NMFS BiOp calls for maximum dam discharge during August until reservoirs are drafted to 20 feet from full pool. Water from headwater storage projects is released to augment the natural flows in the Columbia River to meet a summer flow target of 200,000 cfs at McNary Dam. The goal is to increase water velocities in the pools upstream from dams in the lower Columbia to speed the juvenile salmon migration toward the ocean.

The NMFS BiOp creates an augmented spring freshet followed by a trough, then a second flow peak in August (Figs. 3, 4, and 5). The rapid

Fluctuating or abnormally high discharges also disrupt the natural revegetation, insect, and larval fish recolonization process. Aquatic and terrestrial vegetation that would normally provide secure habitat and stabilize soils cannot fully reestablish, so fine materials are more easily eroded into the channel. These discharge fluctuations could be moderated by delaying the date of reservoir refill or by extending the period of flow augmentation.

Hyporheic interactions are also altered by intermittent, abnormally high flows. Augmented summer flows may increase the river stage by up to 4 feet. Fluctuating flows alternately saturate and dewater the streambanks, which can weaken the riverbanks and cause bank failure and increased sedimentation.

**Kootenay Lake Conditions (British Columbia)**

Releases from Libby Dam affect water retention time, and thus biological productivity in Kootenay Lake, British Columbia. The warm, sunlit epilimnion contains the highest density of phytoplankton and zooplankton. As inflow to the lake increases, more water must flow through the outlet or be stored in the pool. If the pool elevation is stable or declining, inflowing waters displace a commensurate volume through Corra Linn Dam, British Columbia. The physical configuration of Kootenay Lake results in an epilimnetic release of water from the lake, which results in greater downstream loss (entrainment) of organisms through the turbines. High summer discharges from Libby Dam exacerbate this effect during the summer when thermal stratification in Kootenay Lake is well established. Concerns over nutrient levels in the lake are evident by past investigations of nutrient loading (Daley et al. 1981) and ongoing lake fertilization experiments (Ashley and Thompson 1996).

**Alternative 1: IRC/VARQ**

Dam releases under this alternative were designed to create a gradual rampdown from the spring runoff toward basal flows. Water retention time in the epilimnion of Kootenay Lake would therefore be greater than Alternative 2 during the warm summer months because Libby Dam discharge is less.

**Alternative 2: NMFS Biological Opinion**

The late summer water releases from Libby Dam would cause a higher rate of water exchange in Kootenay Lake's epilimnion. Downstream loss of
the most productive surface layer of Kootenay Lake would reduce food availability for lake-dwelling species, including white sturgeon. This would likely affect survival of fish and may jeopardize the success of the fertilization program (Ashley and Thompson 1996).

The Kootenay River downstream of Kootenay Lake passes through numerous small (and old) hydro dams. This water must be passed relatively quickly and will likely result in increased levels of dissolved gas supersaturation (Pers. comm., Jay Hammond, B.C. Ministry of Environment).

**Effects on White Sturgeon**

**Alternative 1: IRC/VARQ**

Maximum flows are regulated by maximum allowable flood stage (approximately 60,000 cfs) at Bonners Ferry, which eliminates the high flows necessary to completely resor the river substrate. The tiered flow approach in the IRC/VARQ alternative reestablishes a more natural spring runoff period (Fig. 6). Model simulations estimate that combined flows in excess of 50,000 cfs can be achieved at Bonners Ferry in approximately 4 out of every 10 years (Marotz et al. 1996). Approximating the bankfull flow on this frequency is expected to reduce imbeddedness and clean interstitial spaces in riffle areas. Flows during dry years are less under the tiered flow approach than those specified by the NMFS 95 BiOp.

Historically, white sturgeon incubation, hatching, and early fry stage coincided with gradually declining flows, immediately after the spring runoff. The gradual flow reduction after the spring peak may reduce predation mortality in larval sturgeon by increasing the area of submerged riverbed, thus increasing security habitat (Pers. comm., Carl Walters, University of British Columbia, Vancouver, B.C.).

**Alternative 2: NMFS Biological Opinion**

The spring release called for by the NMFS 1995 BiOp is similar to Alternative 1 in that it would mimic the natural spring runoff. The 1995 BiOp differs in dry years, when dam discharge would be greater than the sturgeon tiered flow strategy. Bankfull flows could be achieved on the same frequency as the IRC/VARQ.

However, the August release is inconsistent with the restorative flows recommended in IRCs. The varial zone is flooded, then dewatered twice during the period crucial to sub-yearling sturgeon development. White sturgeon can be directly affected (through stranding of juveniles) or indirectly affected (through food web dynamics) by summertime flow augmentation (Stanford et al. 1996, Hauer et al. 1997). Unseasonably high water velocities during August could displace juvenile sturgeon that evolved under stable low flows during the critical early life stage. Summer releases dictated by the NMFS BiOp, therefore, likely impact postlarval survival.
Flood Control

A new strategy for system flood control (VARQ) is required to balance the needs of resident and anadromous species in the Columbia system. The IRC/VARQ strategy releases a naturally shaped spring freshet and stores only the amount of water that would exceed flood stage downstream. Less reservoir drafting is required, which benefits reservoir biology and improves reservoir refill probability. Thus, a naturalized spring freshet can be created, even in dry years, without compromising reservoir refill.

ACOE modelers established that the IRCs were nearly identical to a new flood control strategy being developed by the ACOE (1997). The IRCs are consistent with VARQ in years of average and higher water. Differences between VARQ and IRCs during years of lower water are a result of integrating power constraints. A combination of IRCs and VARQ is being explored for Hungry Horse and Libby operation (ACOE 1997). VARQ is a critical tool to simultaneously balance the needs of resident and anadro-
mous fish recovery by providing greater operational flexibility in dry years to help salmon and steelhead without harming native resident fish species.

**Economic Effects**

Wright et al. (1996) reported that the enhanced reservoir operation (IRC concept) was the least expensive of the alternatives analyzed, saving the power system an annual incremental average of $27 million as compared to the NMFS BiOp. Furthermore, Wright et al. (1996) stated, “the mathematical decision process for establishing reservoir elevations and flow targets, based on updated inflow forecasts, is amenable to power and flood control planning.”

**Applicability to Other Storage Projects**

Integrated Rule Curves are a tool for examining trade-offs among the environmental needs of anadromous fish downstream and resident fish in the Columbia River headwaters. By implementing IRCs at other storage projects, sub-basins experiencing wet conditions can supply spring and summer flow augmentation for anadromous fish whereas dry sub-basins provide less flow, protecting important reservoir and riverine stocks. Preliminary IRCs for other projects can be based on the hydraulic balance in the watershed and physical capabilities of the dam, then modified to site-specific biological concerns as data become available. As flows descend through the system, water can be temporarily delayed en route, then released as flows decline to protract the pulse of water. The result is a sustainable operation that can function within the vagaries of annual water availability.

A number of advisory groups have provided scientific and policy review of the Columbia River salmon recovery efforts and the IRCs. The Independent Scientific Group (ISG 1996) endorsed the IRC concept for other reservoirs in the Federal Columbia River Power System. The Independent Scientific Advisory Board recognized that the IRCs were designed to minimize the impacts of drawdowns on reservoir food webs (ISAB 1997a) and that the Hungry Horse and Libby IRCs “provide seasonality of flow in downstream reaches as called for under our normative river concept.” (ISG 1996, ISAB 1997b).

**Columbia River Flow Augmentation**

Flow augmentation for anadromous species in the lower Columbia is based on a relationship between water velocity and smolt (juvenile salmon or steelhead) migration speed. Evidence suggests that fish generally travel faster at higher water velocities and that more rapid travel results in better survival. The release of water from Montana reservoirs during August is intended to speed, thereby improving the survival of smolts. Water velocity profiles measured by the U.S. Geological Survey (USGS) in McNary reservoir were used to describe the influence of Montana dams on water velocity (Fig. 7). Since most smolts travel in the top 10 m of the reservoirs
in the lower Columbia, we can focus on the average velocity in that zone. (Note: USGS data do not include the low-velocity nearshore portion of the channel cross-section.) An exponential regression of average water velocity under varying discharge volumes shows that the average velocity increases 1.6 cm per second when the discharge is augmented by 20 kcf/s (roughly the equivalent of the total release from Montana), increasing the flow at McNary from 180 kcf/s to 200 kcf/s (the August flow target). Thus, if no Montana water was released, the velocity would be reduced by an average of 1.6 cm per second. Since John Day Reservoir, immediately downstream, has a larger cross-section and volume than McNary, the change in water velocity and smolt travel time would be even less.

**Conclusion**

It our conclusion that the Endangered Species Act and the action agencies operating the dams in the Columbia River basin must consider effects on the ecosystem, including listed and nonlisted stocks of native species to avoid additional listings. The current operational strategy results in unnatural flow fluctuations below Libby and Hungry Horse dams that are harmful to the riverine ecosystem that supports the endangered Kootenai...
white sturgeon, threatened bull trout, westslope cutthroat trout, and interior redband trout.

Water released for anadromous fish recovery as called for by NMFS disrupts the desired balance between resident fish needs, storage reservoir operation, and river flows. Reservoir refill failures during dry years are expected under the IRC/VARQ operation, but less frequently than would occur by implementing the NMFS operation. Extreme reservoir refill failure (more than 20 feet) negatively affects biological production in the reservoirs, entrains more fish through Libby Dam, and impacts fishing and recreation in the Flathead and Kootenai rivers downstream. Reservoir refill failure also complicates the system’s ability to store water for release during the following spring. The best conditions for Columbia Basin fish resources can be achieved by implementing the IRC/VARQ in Montana and other storage projects (e.g., Mica, Arrow, Dworshak). Sub-basins experiencing wet conditions can supply salmon flow augmentation, whereas dry sub-basins would provide less flow, protecting important resident fish stocks. Combined flows from the headwater sub-basins could then be shaped to achieve the greatest benefit for salmon and other anadromous stocks while protecting fish populations in the dry sub-basins. A gradual rampdown from the spring runoff in the sub-basins can be used to normalize the river hydrograph below headwater projects.

References


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Utilizing Ecosystem Concepts in Fisheries Management Strategies

Kristine D. Lynch and William W. Taylor  
*Michigan State University, Department of Fisheries and Wildlife, East Lansing, Michigan*

John M. Robertson  
*Michigan Department of Natural Resources, Forest Management Division, Lansing, Michigan*

Kelley D. Smith  
*Michigan Department of Natural Resources, Fisheries Division, Lansing, Michigan*

**Abstract**

Fisheries ecosystem management programs and policies must incorporate holistic, integrative, and multidisciplinary collaborative strategies. Those who manage fisheries are often administratively, procedurally, and disciplinarily separated from those who manage other related ecosystem elements, such as forests, wildlife, water quality, and land use. This organizational and philosophical division within agencies can render ecosystem management programs incomplete, inefficient, and ineffective. Since management processes influence how managers conceptualize and respond to resource issues, it is important that management systems be congruent with the dynamics of ecosystems. The Michigan Department of Natural Resources (MDNR) is redesigning their management processes to enable more efficient and effective ecosystem management. Boundaries for management units within all divisions (including Fisheries, Wildlife, Forest Management, and Parks and Recreation) are being redrawn along watershed and eco-region lines. Instead of each division developing their management plans in isolation, the design and implementation of all management plans will require input and contributions from managers in all relevant divisions. Furthermore, each management plan will be based on and measured by locally specific criteria and indicators that are most diagnostic for each ecosystem. New management plans will thereby be more
comprehensive and ecosystem-specific, although they will require a great deal of coordination, collaboration, and communication among agency personnel. By redesigning their management planning and implementation processes to incorporate ecosystem priorities and activities, the MDNR should experience long-term improvements in the sustainability of fisheries (and other) ecosystems.

Introduction

Regional landscape inputs are integrated in freshwater ecosystems, holistically reflecting complex physical, chemical, biological, and social processes, interrelationships, and dynamics. Therefore, fisheries ecosystem management programs and policies must also incorporate holistic, integrative, and multidisciplinary collaborative strategies. For many natural resource management agencies, these ecosystem management requirements pose a significant challenge to conventional methods of planning and implementing management activities. By reviewing ecosystem management concepts and examining one agency’s strategy for ecosystem management, we can better understand the potential successes and pitfalls surrounding the implementation of agency-based ecosystem management. Through learning about one agency’s experiences, other agencies can better anticipate and possibly avoid many of the potential difficulties in order to reap the benefits of ecosystem management.

Ecosystem Concepts

Ecosystem scientists have long demonstrated that aquatic ecosystems function as integrators of regional watershed and airshed dynamics (Hynes 1975). The quality and quantity of water in aquatic ecosystems, the basis for fish production, is therefore directly affected by physical, chemical, and biological inputs which are a result of the watershed-specific environmental and social processes (Taylor et al., In press). These include inputs from processes such as erosion and sedimentation, landscape runoff, point and nonpoint source pollution, and atmospheric deposition. Diverse human activities, such as urbanization, deforestation, industrial manufacturing, energy production and consumption, and intensive agriculture, drive these processes and therefore affect aquatic ecosystem productivity and fisheries sustainability. Such land use and development activities are a product of diverse and fragmented planning and policy-making bodies, which allow development to proceed with little or no coordination or regard for the structure and function of ecological systems (Caldwell 1994, Ferreri et al. 1998).

For aquatic ecosystems and their resources to be effectively managed, they must be understood in context of their function as holistic integrators of regional social and environmental processes (Knight and Meffe 1997). This integration is a guiding principle of ecosystem management, a
management philosophy that emphasizes: (1) the integration of the physical, chemical, biological, and social components of the resource, (2) the interactions of these four components, and (3) how these components relate to ecosystem productivity and resource sustainability (Ferreri et al. 1998). Management based on these ecosystem principles will help maintain and improve the sustainability of these systems.

While the concept of ecosystem management is widely accepted by managers, scientists, and stakeholders as an appropriate management paradigm, it has undergone intense scrutiny and refinement throughout its evolution (Yaffee et al. 1996). The definitions of ecosystem management are as varied as ecosystems. Recognizing that this kind of management needs a strong conceptual foundation to be useful, the Ecological Society of America (ESA) assembled an ad hoc Committee on Ecosystem Management to address the concept of ecosystem management and its scientific foundation. Their 1995 report established that every ecosystem management effort, regardless of its specific definition, should include eight principles: “(1) long-term sustainability as fundamental value; (2) clear, operational goals; (3) sound ecological models and understanding; (4) understanding complexity and interconnectedness; (5) recognition of the dynamic character of ecosystems; (6) attention to context and scale; (7) acknowledgment of humans as ecosystem components; (8) commitment to adaptability and accountability” (ESA 1995:1).

Adapting Agency Management to Ecosystem Concepts

Due to the diverse and dynamic nature of regional social and environmental processes, holistic management of aquatic ecosystems requires a high level of coordination and collaboration among those in the watershed whose activities affect water quality and quantity (Ferreri et al. 1998). Those who manage fisheries are very often administratively, procedurally, and disciplinarily separated from those who manage other related ecosystem elements, such as forests, wildlife, water quality, and land use (Knight and Meffe 1997). These organizational and philosophical divisions within and between management agencies can render ecosystem management programs incomplete, inefficient, and ineffective (Caldwell 1994). Agency management activities are also usually separated from other watershed activities that influence aquatic ecosystems and fishery production; for example, activities by private landowners, non-governmental organizations, organized citizen groups, and commercial industries. These groups engage in a wide range of water- and land-based activities that directly and indirectly affect various components of aquatic ecosystems, such as nutrient cycling, temperature, and flow rates (Decker and Krueger 1993).

Agency-based management of fisheries and aquatic ecosystems must often be transformed from a “closed system” of short-term fish production and habitat manipulation to a more open, dynamic, and integrative system based on long-term collaboration with other ecosystem scientists, managers, and stakeholders (Ferreri et al. 1998). Implementation of ecosystem
management will require natural resource agencies to reevaluate their management philosophy, which will enable them to change their management structures and in ways that are consistent with ecosystem concepts (Knight and Meffe 1997). Since management structures and processes influence how managers conceptualize and respond to resource issues, it is essential that agency management systems be congruent with the dynamics of ecosystems.

Case Study: The Michigan Department of Natural Resources

Some natural resource management agencies are redesigning their management structures and processes in order to enable more efficient and effective ecosystem management (Yaffee et al. 1996). The Michigan Department of Natural Resources (MDNR) is one such agency. Recognizing that their traditional management system has been incompatible with many ecosystem concepts, the MDNR is undergoing a comprehensive transformation to make their management activities more congruent with ecosystem structure and function. Following this transformation, the MDNR intends to produce ecosystem management plans that are more comprehensive and ecosystem-specific, thereby producing more effective and efficient management tools. A review of the overall MDNR ecosystem management strategy and a closer look at one division’s role in the transformation reveal that the MDNR approach to ecosystem management has potential for success, but the MDNR must overcome many challenges.

Michigan Department of Natural Resources Joint Venture Strategy

For decades, the MDNR has been managing natural resources through the fragmented efforts of narrowly focused divisions, such as Fisheries, Wildlife, Forest Management, and Parks and Recreation. As a result, managers from different divisions often had little involvement with each other, whether they were working for common or conflicting ecosystem objectives. For example, the Fisheries Division’s stream habitat management activities are strongly influenced by upstream logging activities, but the Forest Management Division may not be aware of the Fisheries Division fish habitat objectives in that stream. The divisions’ planning and management activities were therefore often inefficient and ineffective.

According to its mission statement, the MDNR is committed to the conservation, protection, management, use, and enjoyment of the state’s natural resources for current and future generations. In order to satisfy the current multiple values of natural resources and manage on an intergenerational time scale, the MDNR recognizes that the principles of ecosystem management are highly appropriate. The MDNR conceptualizes ecosystem planning and management as a process that integrates
physical, chemical, biological, and ecological principles, along with economic and social factors, into a comprehensive strategy aimed at protecting and enhancing sustainability, diversity, and productivity of a system. This definition of ecosystem management emphasizes the need for an integrative and comprehensive perspective, which is necessary to maintain the long-term multiple values of Michigan’s natural resources.

The MDNR’s traditional management structure and processes were neither very integrative nor comprehensive. Each division’s management activities have traditionally been administratively and scientifically separated from the management activities of other divisions, despite the fact that the resource management divisions both influence and rely upon various aspects of common ecosystems. For example, landscape activities such as canopy removal through logging may increase water temperature in rivers, thereby altering aquatic habitat. Other changes to aquatic habitats can shift the distribution of wildlife, such as waterfowl, thereby altering ecosystem structure and the availability of recreational opportunities. Realizing the close interdependence between their management objectives and activities, each division has been motivated to seek a more collaborative and ecologically appropriate management style. Creating a more ecologically compatible management agency forced the MDNR to critically examine how management procedures and administration should be organized. Through the extensive joint efforts of division leaders and numerous subcommittees charged with designing ecosystem management processes, a new joint venture (JV) strategy for ecosystem management has emerged.

Each division, including Fisheries, Wildlife, Forest Management, and Parks and Recreation, is undergoing a structural and procedural transformation as part of the JV initiative. This JV strategy preserves the MDNR’s divisions, although the organization of each division’s management units and processes will be realigned according to appropriate ecosystem concepts. For example, since the State of Michigan contains four distinct eco-regions and four distinct Great Lakes basins, these areas will become the fundamental ‘eco-units’ for coordinating regional management activities. While divisions are committed to the eco-unit concept, divisions will not all adopt common boundaries for their eco-units. The ecosystem-based boundaries that are appropriate for fisheries (i.e., watersheds and basins) are not necessarily the most appropriate unit for planning forest or wildlife management, where management units based on eco-regions or habitat type are more appropriate. Management activities will be implemented at the management area level, a new geographic subunit of the eco-units. Furthermore, each division will establish eco-unit teams, consisting of regional division staff and an eco-unit coordinator, to coordinate and facilitate the management activities for their respective eco-units. Coordinators will also serve on a department-wide ecosystem management team, responsible for overseeing, linking, and coordinating statewide ecosystem planning and assessment based on appropriate criteria and indicators. The concept for the revised MDNR structure is illustrated in Fig. 1.
Ecosystem management requires more than structural revisions. Divisions are also re-designing their management processes to allow for more comprehensive integration of all available information about ecosystems. Instead of independently developing fragmented management plans in isolation, each division will be required to develop management plans at their eco-unit level in cooperation with resource managers from other divisions, agencies, and public groups. This broader base of input and contributions will allow each division to plan on a more holistic eco-regional perspective, and it will enable more appropriate allocation of budget and staffing resources in the implementation of ecosystem projects. Division-based eco-unit teams will be responsible for adapting their management planning processes to incorporate the relevant groups. They will also be fundamentally held accountable for their eco-unit’s management objectives and accomplishments. Furthermore, as eco-unit teams develop management plans, they will also cooperatively develop the criteria and indicators, customized for each eco-unit’s unique biological, physical, and social conditions, which will be used to gauge management success.
**Fisheries Division Joint Venture Strategy**

The MDNR Fisheries Division, responsible for managing the fisheries and other aquatic resources in Michigan’s waters of the Great Lakes as well as inland aquatic ecosystems, is one of the divisions undergoing a structural and procedural transformation as part of the JV initiative. In the fall of 1997, the Fisheries Division began planning and designing new administrative structures and management processes. This work was undertaken by a special committee, consisting of representatives from Fisheries Division research, management, fish production, and program services sections, charged with identifying a strategy for managing aquatic resources based on ecologically based management units and processes.

The committee began by changing the division’s management planning and implementation units within Michigan. Traditional management units were based on county lines for convenience in statistical reporting, not according to watershed or other eco-region boundaries. Recognizing that the structure and function of aquatic ecosystems are influenced by watershed-level processes, the committee recommended establishing watershed-based boundaries for both management areas (smaller local watersheds) and eco-units (regional groupings of adjacent watersheds). Eco-units will be based on the watersheds for the four Great Lakes that border Michigan: Lakes Superior, Michigan, Huron, and Erie (Fig. 2); within the Fisheries Division these are termed “basin units.” These four basin units will encompass one (Lakes Erie and Superior), two (Lake Huron), or three (Lake Michigan) management areas.

The fisheries management roles, responsibilities, and processes for basin units and management areas will be distinct, yet interdependent. Fisheries in each of the four basin units are to be primarily managed by a basin team (comparable to other divisions’ eco-unit teams), responsible for both Great Lakes and inland waters. Basin team responsibilities will include both long- and short-term management planning. Long-term planning includes goal setting; determining staffing, research, and fish production needs; and providing input for division strategic planning. Basin teams will also use their collaborative, basin-wide perspective to oversee the short-term planning of management area activities, such as stocking fish and setting work priorities. To plan these management activities, the basin teams will consult with other divisions in the MDNR and the Department of Environmental Quality, as well as with other state and federal agencies, universities, consultants, and public groups. Furthermore, basin teams will be responsible for allocating budgets to management areas according to basin-wide work priorities, and they will be responsible for monitoring budget accountability work performance. Management areas, in contrast, will implement management plans approved by the basin team and report on the status of management activities. Due to the collaborative and holistic requirements of ecosystem management, these work plans
may involve contributing to the management activities of other divisions, such as Forest Management or Wildlife, developed in cooperation with Fisheries Division personnel.

Basin-wide management activities will be coordinated by basin teams, since the team approach enables broad-scale collaboration and encourages the exchange of perspectives and knowledge across program areas. Members of the interdisciplinary basin teams will come from all management areas in each respective basin and will consist of area managers, fisheries management biologists, technician supervisors, inland researchers, Great Lakes researchers, fish production specialists, or other staff specialists as needed. Specific basin team composition and the decision-making structure (e.g., voting, consensus) will vary at different times, depending on the specific issues or problems that need to be addressed.
The activities of each basin unit will be facilitated and represented by a basin coordinator. Each basin coordinator will be responsible for: (1) determining the statewide needs for the basin and incorporating these into area work planning; (2) providing the Fisheries Division management team with annual basin priorities set by the basin team; (3) working with management areas on budget allocation issues; and (4) drafting an annual report on the basin unit’s accomplishments. The basin coordinators will also represent their basin units on the department-wide ecosystem management team, with the public, and on other ad hoc ecosystem management initiatives. The four basin coordinators will hold this position on a full-time basis.

At the upper end of the management hierarchy, the Fisheries Division will maintain a single division-wide management team. This team is responsible for long-term oversight, program development, and program review. It will consist of the division chief, section leaders (e.g., fish production, research, program services), basin coordinators, and other division specialists as needed. The management team will ultimately be responsible for setting division priorities and allocating budgets among basins. They will review the accomplishments of the four basin teams to ensure adherence to priorities and implementation of projects. The concept for the proposed Fisheries Division organizational structure is illustrated in Fig. 3.

**Evaluation of the Joint Venture Initiative**

By developing the JV strategy, the MDNR believes it is adopting and internalizing the concept of ecosystem management. By comparing features of the JV strategy to the eight guiding principles developed by the ESA (1995), it may be possible to determine the extent to which the JV strategy has embraced ecosystem management standards. This may also indicate potential areas of JV success or problems.

1. **Long-term sustainability as fundamental value:** The MDNR definition of ecosystem management emphasizes the need to protect and enhance the sustainability of natural resource systems. Instead of promoting multiple *uses* of ecosystems, the MDNR acknowledges that sustainability is one of the multiple *values* that must be maintained.

2. **Clear, operational goals:** At this point, the goals of the MDNR’s JV initiative are largely at the conceptual stage; for example, JV goals include: (a) efficient and effective management and supervision, (b) holistic resource management, (c) communication and education, (d) organizational culture, (e) employee development and training, and (f) strategic management. While these goal areas are explained in the JV document, they do not yet consist of measurable goals that specify the outcomes of ecosystem management activities. These will be determined as ecosystem management activities are designed and implemented.
(3) **Sound ecological models and understanding**: The MDNR will continue to conduct research and monitoring at multiple levels of ecological organization. Divisions will continue research on various ecosystem components, and a common GIS (geographic information system) database will be available for use among all divisions. Their ability to develop ecological models and understanding will rely upon their ability to integrate their research activities into a common ecological framework.

(4) **Understanding complexity and interconnectedness**: By requiring cross-divisional collaboration in the management planning and implementation processes, the MDNR understands that ecosystems are far more complex and interconnected than any one division can understand. While the JV initiative requires interdepartmental cooperation and collaboration with other non-MDNR partners in the watersheds and landscapes, the MDNR must still demonstrate how these collaboration requirements will be for-
mally integrated into management processes. Like principle (2) above, these will be determined in the future.

(5) Recognition of the dynamic character of ecosystems: Instead of managing for only one set of ecosystem conditions, the MDNR recognizes that ecosystems are dynamic and require adaptive management approaches so that managers can learn and adjust their management plans accordingly. The departmental ecosystem management team and the division eco-unit teams will develop appropriate criteria and indicators that are most diagnostic for different ecosystem conditions and goals; these will be used to gauge the direction of ecosystem trends, not static conditions, for determining management success. Like principles (2) and (4), these criteria and indicators will be developed as ecosystem management evolves.

(6) Attention to context and scale: Divisions recognize that there are multiple scales at which ecosystem components function; therefore, each division is independently designing their most appropriate management scales. They recognize that the “ecosystem” for a fish or a tree may encompass different social and environmental processes, and they have established appropriate spatial scales for management. While they also acknowledge the need for a long-term temporal scale, this too will likely be customized for each ecosystem component and demonstrated as the management planning process evolves.

(7) Acknowledgment of humans as ecosystem components: The MDNR definition of ecosystem management acknowledges that economic and social factors must be integrated with other ecosystem processes. Divisions know that their management activities must meet a wide range of human needs and values, and they know that various stakeholder groups must be incorporated into management processes. Again, the eco-unit teams must establish appropriate mechanisms for ensuring effective involvement of multiple stakeholders.

(8) Commitment to adaptability and accountability: The JV documentation repeatedly stresses the importance of adaptive management that allows for evaluation, learning, and management modification as ecosystems change. The MDNR also intends to hold eco-units accountable for developing and implementing appropriate management activities. Once ecosystem management process are implemented, the extent of the eco-units’ adaptability and accountability must be determined.

According to the standards developed by the ESA (1995), the MDNR JV strategy appears to be on the right path for establishing an ecosystem-based approach to management. The extent to which these standards are developed and implemented, however, has yet to be determined. Many of the above standards must be manifested in the specific ecosystem management plans developed by eco-unit teams, which should be developed by mid-1999. At that time, and periodically thereafter, these eight standards should again be reviewed by those on eco-unit teams.
Other Challenges to the Joint Venture Initiative

While the JV initiative demonstrates a solid commitment to ecosystem management, the divisions are experiencing certain problems that may impede their ability to establish successful management. Many of these problems are related to the MDNR's administrative structure and communication strategy, and they may experience other potential problems related to team performance.

As the divisions proceed with developing ecosystem management processes, they will have to reconcile division-specific eco-unit structure with the need for common administrative structures regarding budgets, work locations, and staffing issues. In other words, although the eco-units will be based on different boundaries in different divisions, there is still a need to share facilities, office space, and support staff for managers in different eco-units; this will require coordination of administrative structure across different divisions and management boundaries. This kind of coordination will likely present challenges as their ecosystem management strategy is implemented.

The MDNR will also experience challenges as they communicate the JV strategy to their personnel and other stakeholders. The concept of ecosystem management has been embraced and pursued by MDNR leadership, although the majority of MDNR personnel are not involved in the process of adapting the agency's structure and processes to ecosystem requirements. As a result, they often lack complete understanding of this process; yet this misunderstanding is expected since the process of adopting ecosystem management has gone through several revisions since its initiation. Leaders in the MDNR are aware they must develop a strategy for communicating the new structures and processes to their personnel to ensure staff support, which may require extensive continuing education and training programs on ecosystem management philosophy as well as the new administrative procedures. The MDNR is also aware they must develop a strategy for informing other stakeholders about their new structures and processes, since many stakeholders regularly deal with MDNR staff through both formal partnerships and informal inquiries. They need to understand that the services provided by the MDNR (resource information, visitor centers, etc.) may be provided differently in the future.

The MDNR should anticipate other potential problems surrounding the roles and responsibilities of various teams and personnel in the JV initiative; this may help them avoid any shortcomings in team or job performance. The key to successful ecosystem management planning rests on the eco-unit teams' ability to identify and incorporate the appropriate ecosystem groups and information. Will they comprehensively incorporate other ecosystem stakeholders in their planning and become involved in other divisions' planning, potentially at a cost of their own division's
budget, time, and personnel? Successful ecosystem management requires successful coordination and collaboration, which, in turn, requires active participation from eco-units, inclusion of other partners, and diligent oversight from the ecosystem management team. If these teams fail to function appropriately, the ecosystem management process will break down. The JV initiative must develop safeguards to ensure certain standards of team and personnel performance so that managers do not resort to traditional fragmented management systems.

Conclusions about Implementing Ecosystem Management

In embracing the concept of ecosystem management, the MDNR has undertaken a significant challenge, yet the JV approach appears to have a solid foundation in ecosystem management principles. Their challenge is far from over, however, as there are many areas in which the initiative could break down, and there are many important ecosystem principles that need to be further addressed through specific management plans. While it is too early to predict the outcomes of this ecosystem management initiative, the MDNR has the potential to experience success through appropriate development and implementation of management plans. By redesigning their administrative structure and management processes to incorporate holistic ecosystem priorities and activities, the MDNR may experience long-term improvements in the health and sustainability of fisheries (and other) ecosystems.

The MDNR’s experience with ecosystem management, while far from being fully implemented, reveals many issues and challenges that natural resource agencies must address. The MDNR experience should be compared with the experiences of other natural resource agencies to better establish a framework for designing and implementing agency-based ecosystem management. All natural resource managers, scientists, and stakeholders have a great deal to learn about this approach to management, as the future of natural resource use and enjoyment depends on our ability to successfully manage ecosystems and their resources. Successful ecosystem management will provide Michigan, and the Great Lakes region, with long-term benefits by sustaining multiple uses and values of natural resources.

References


A Solution to the Conflict Between Maximizing Groundfish Yield and Maintaining Biodiversity

Albert V. Tyler
University of Alaska Fairbanks, School of Fisheries and Ocean Sciences, Fairbanks, Alaska

Abstract
There is a practical conflict between the goals of biodiversity maintenance and maximization of long-term fishery yields. It is not possible to maximize both simultaneously. On one hand, how to achieve long-term maximum yields while accounting for species interactions cannot be formulated because of variable relationships among species. On the other hand, it is known that catch rates tuned to the most productive species will bring about decreases in biomasses of less productive species taken as bycatch. This means diversity will decrease. Yet the long-term yields of economically valuable assemblages are likely dependent on the very diversity that is threatened by maximizing yields. A comparison of groundfish assemblages of continental shelves from the North Atlantic Ocean demonstrates that whole assemblages have declined—not just a few economically valuable species. An assemblage maintenance approach may be the only method of achieving multispecies persistence. A program of assemblage maintenance is suggested that reduces destructive exploitation risk while achieving economically viable levels of fisheries yields. The proposal involves setting up areas of contrasting fishing effort, with areas based on species stock structure and assemblage maps. It might be possible to hold part of a region with present effort levels, with increases encouraged in another part. The establishment of management areas would be followed by monitoring commercial catches of selected species, and by a program of research surveys that included assemblage analysis. Criteria are presented for adjustments of catch rates in response to assemblage changes related to fishing effort.
**Introduction**

My contention is that it is impossible to maximize both the fishery yield of major species and to maintain the biodiversity (Pielou 1975) of an exploited system. High rates of exploitation inevitably lead to changes in species composition and decreases in diversity that in turn feed back to the productivity of the economically important stocks and interfere with that productivity. I have come to conclude that without the original, natural diversity of the production system, the high productivity of the original target species cannot be maintained.

In this paper I will review some concerns about existing changes in a number of major groundfish production systems in regions of North America. I will then show reasons why the present paradigm of maximizing the yields of the main economic species in these systems is not only inadequate for maintaining total yield, but will lead to unlooked-for changes in the total production system that will decrease the value of the system to fisheries. Some of these changes will likely be irreversible in terms of the composition of fish and invertebrate fauna. Yet there is an approach to stock assessment that could be used as an adjunct to the yield maximization paradigm that would likely prevent these damaging changes. I will describe this alternative, empirically based suggestion that is similar to an earlier, but less complete, attempt (Tyler et al. 1982).

**Catch Rate and the Mixed-Stock Management Problem**

The mixed stock management problem is well known to biologists in the North Pacific as it applies to salmon. Paulik et al. (1967) showed that stocks of a single species of salmon often have differing productivity rates, and that when these stocks are caught together in an offshore fishery at a high catch rate, the least productive of the stocks will decline in spawning biomass. He showed how to calculate the rates of decline in relation to offshore catch rate and how to arrive at the rate that would sustain all of the stocks. He pointed out that maintaining the least productive stock means giving up catch potential of the most productive stock. The catch rate that maximizes the take for the most productive stock will cause the least productive stock to disappear.

When hatchery programs and ocean ranching became big along the Pacific coast, the mixed stock problem took on new dimensions. Hatcheries and ocean ranching operations could put so many young fish in the ocean that the catch rates for fisheries on returning runs could be very high, leading to increased catches of the hatched species. This catch is often taken in areas just offshore from the hatchery along with fish from wild stocks. The result is that the nearby wild stocks decline because they do not have the rate of productivity of the hatchery. The fishery takes wild and hatchery fish at the same catch rate. "Similarly the establishment of a
new and highly productive hatchery stock might be responsible for an increase in the rate of exploitation that resulted in elimination of less productive existing stocks harvested in a common fishery" (Paulik et al. 1967). In terms of the portion of the stock taken, small wild stocks often have a greater portion removed than would ever be contemplated if they were the only stocks being managed.

It is the simultaneous fishing of stocks with differing productivities at a high rate suitable for the most productive stock that causes the problem. The result can only be that either the catch rate is diminished so that all stocks survive, or that the diversity of stocks decreases. The principle of the mixed stock management problem applies to other mixed stock situations as well as to salmonids; for example, to mixed species of ocean flatfish.

**Multispecies Fishery Models**

The Paulik model is a multistock model that is usable as a general multispecies model. The heart of the model is a series of stock-recruitment functions. Either stock or species somatic growth rates can be added and suitable changes can be made so that the stock being modeled is more like a cod as in the model POPSIM (Walters 1969) than a salmon. Several stocks can then be modeled with the Baranov function using catch rates from competing fleets (Ricker 1975); e.g., a longline fleet and a trawler fleet, both fishing the same stock of cod. These multispecies models can be put into a trophic dynamic setting where the target stocks compete with one another for a food resource base of varying complexity (Daan 1987). Statistically determined abiotic relationships can be added so that the rates of natural mortality, reproduction, and somatic growth respond to the biotic and abiotic environment (Pope and Knight 1982, Hilborn and Walters 1992).

I have heard some say that they dislike the single species orientation of the current stock assessment models and that analysts should develop more realistic multispecies models. On the other hand, analysts often respond by saying that they do have multispecies models, and they are developing better ones—that data are not available to either parameterize or drive the models. The limitation is that the statistical nature of species interaction data is uncertain. Often the common wisdom is that “predation has got to be important” or “food supply must be limiting.” However, it is usually not possible to show the statistically significant relationships that are the best evidence for these relationships. As a result, multispecies models are generally used for summarizing an understanding of a complex dynamic situation and in gaining insight rather than for making quota estimates.

In our reductionist way, we hope that we will be able to develop natural production system models in computers that will account for species interactions so that we can extend the yield maximization paradigm to the
context of a total production system. Though this thesis is attractive, I believe it will be an impossible objective to achieve for many years. The reason that elaborate computer models are not used to calculate potential yields of a species, for example, walleye pollock, is that the information needed is grossly incomplete. Many biologists are working on making this more complete. As scientists gain information and insight on how to apply the information to fishery problems, production system models will be more valuable.

Our partial understanding of variable species interactions and the influence of the physical-chemical world still prevents us from developing predictive models of multispecies yields in a multifactor setting that are predictive, ecosystem-dynamics models. Consequently, there is wisdom in not using those models to make yield estimates, but instead refining the single species models as analysts have done, changing the parameters and driving variables of the models as new information comes on-line.

**Large-Scale Changes in Assemblages of Groundfish**

It is important to look at the evidence for multispecies system change in case histories. There is evidence of change that goes beyond the reductions in the biomass of the target species that have been overfished. I will review assemblage findings from the Grand Banks of Newfoundland, Canada (western North Atlantic), and for United States fisheries: Georges Bank off New England (western Atlantic), the continental shelf off Washington and Oregon (eastern North Pacific), and the eastern Bering Sea off Alaska.

In terms of species richness, the Grand Banks is the simplest of the four areas. Thirty-five fish species occurred consistently enough in the regional data set to be included in a statistical analysis (Gomes et al. 1995). The dominant species were Atlantic cod (*Gadus morhua*), American plaice (*Hippoglossoides platessoides*), thorny skate (*Raja radiata*), redfish (*Sebastes marinus* and *S. mentella*), Greenland turbot (*Reinhardtius hippoglossoides*), and witch flounder (*Glyptocephalus cynoglossus*). Eleven assemblage groupings of species were identified that had affinities to particular areas (Fig. 1). Species were usually members of more than one group, and the groups were characterized by particular compositions of species. It was particularly interesting that the geographical boundaries of the assemblage groups were consistent over a 15-year period. As Newfoundland’s northern cod fishery crisis emerged during the 1980s, it was not only the biomass of cod that changed, the abundances of nontarget species changed as well (Fig. 2). American plaice and broadhead wolffish (*Anarhichus denticulatus*) diminished, while witch flounder became scarce in the more shallow regions but more abundant in deeper assemblages. The multispecies biomass became dominated by thorny skate, apparently resisting the high intensity of trawling.
Five assemblages were identified on Georges Bank (Fig. 3) by using statistical methods similar to the Grand Banks analysis (Overholtz and Tyler 1985, Gabriel 1992). Thirty-six species were included in the assemblage analysis. Dominant species originally included Atlantic cod, haddock (*Melanogrammus aeglefinus*), yellowtail flounder (*Limanda ferrugina*), winter flounder (*Pseudopleuronectes americanus*), and winter skate (*Raja ocellata*). As was the case off Newfoundland, the assemblage boundaries remained fairly constant geographically, but strong distortions occurred in the relative abundance of the component species’ abundances concurrent with high fishing rates (Fig. 4). The multispecies biomass became dominated by skates.

On the continental shelf off Washington and Oregon, 53 species were included in the first analysis of assemblages (Gabriel and Tyler 1980), while 33 species were included in a more recent analysis (Jay 1996). The recent assemblage analysis indicates that many of the assemblages have retained their identity to date since the first research survey in 1977, while varying in their geographic distributions. The most abundant species,
Figure 2. Abundance changes for Grand Banks assemblages (Gomes et al. 1995). The width of the species band, or the species group band, shows the relative abundance. The bands are stacked. The order of the names at the right matches the order of the bands. The first band, marked "Others," is a miscellaneous species category.
Pacific hake (whiting) (*Merluccius productus*), has remained at high levels through the entire period (Pacific Fishery Management Council 1998). English sole (*Pleuronectes vetulus*) has shown a strong increase in its abundance, while petrale sole (*Eopsetta jordani*) has remained fairly constant. The stock assessments for the region indicate, however, that many of the commercially targeted stocks have declined during the past 15 years (Fig. 5), including lingcod (*Ophiodon elongatus*), widow rockfish (*Sebastes entomelus*), yellowtail rockfish (*S. flavidus*), sablefish (*Anoplopoma fimbria*), shortspine thornyhead (*Sebastolobus alascanus*), and longspine thornyhead (*S. altivelis*). Dover sole (*Microstomus pacificus*) showed a decline ending in 1994, and has increased since then.

Five fish assemblages (Fig. 6) have been identified for the continental shelf of the eastern Bering Sea (Walters and McPhail 1982). Regrettably, assemblage analysis has not been carried out on an annual basis, so it is not possible to look at species composition changes by assemblage. The region is by far the most species-rich of the four areas in this comparison. Seventy-eight species of fish were used for the assemblage analysis. In a later publication, Bakkala (1993) reported that there were over 150 species of fish recorded in National Marine Fisheries Service (NMFS) surveys. The number is an estimate because some of the species were identified only to genus. Species biomass levels for the entire area of the annual survey conducted since 1979 by the Alaska Fishery Science Center, NMFS, were available from the computer database (Pers. comm., G. Walters). Striking changes have occurred for all of the dominant species since the first survey (Fig. 7). Many of the dominants have increased over the past decade. Pacific cod (*Gadus macrocephalus*), rock sole (*Pleuronectes bilineata*), and Greenland turbot showed a long-term decline followed by a recent increase in biomass. Note that this last species is also abundant in the Grand Banks fauna. Yellowfin sole (*Pleuronectes asper*) has shown a fairly steady biomass level. The dominant walleye pollock (*Theragra chalcogramma*) has shown increases and decreases, and is now near historical average levels. One high-biomass species that has shown a long-term decline and is still declining is sablefish. The assessments of the 12 main economically important species show that the stocks are in good shape and above the biomasses needed for maximum sustainable yield (MSY), except for sablefish (North Pacific Fisheries Management Council 1997). The stock trends evidenced in the NMFS survey data indicate that the mix of biomasses of less abundant species is also at historical levels with no evidence of deterioration due to fishing.

**Adaptive Assemblage Maintenance Programs**

What is the alternative to ecosystems dynamics models? I propose an empirically based assemblage maintenance program with extensive, standardized research surveys and fishery observer programs with regulation by area, with two or more levels of fishing effort, one in each of several
Figure 3. Assemblage map of Georges Bank, New England, U.S. (Gabriel 1992).

Figure 4. Abundance changes for Georges Bank assemblages (Overholtz and Tyler 1985). The width of the species band, or the species group band, shows the relative abundance. The bands are stacked. The order of the names at the right matches the order of the bands.
Figure 5. Decline of multispecies landings except for Pacific hake of the Pacific region of the United States (Pacific Fishery Management Council 1998).

Figure 6. Map of eastern Bering Sea assemblages from Walters and McPhail (1982).
Figure 7. Abundance changes for eastern Bering Sea assemblages from the NMFS database (courtesy of G. Walters, NMFS, Seattle).
specific areas in an experimental or test fishery manner. This is an adaptive approach (Walters and Hilborn 1976) in that it depends on the response of fishery management to trials among test areas. The data would consist of multispecies biomass trends resulting from the contrasting levels of fishing effort spaced over large areas of sea bottom. I believe that such a method should be carefully considered by assessment analysts and that it could be applied to many fishery regions.

My example, given below, is of a possible deployment of an assemblage maintenance program to Hecate Strait, the large shelf area between the Pacific mainland of Canada and the Queen Charlotte Islands (Fig. 8). A series of research cruises has been conducted in this area since 1984. These were begun with the Hecate Strait Project of the Pacific Biological Station, Research Branch, Canada Department of Fisheries and Oceans (Westrheim et al. 1984, Tyler 1989). Hecate Strait is bounded to the north by the deep (>200 m) Dixon Entrance and the Alaska Archipelago, and to the south by deep gullies of Queen Charlotte Sound. Consequently the continental shelf fauna within Hecate Strait tends to be separated ecologically from its counterparts elsewhere. Analysis of the research cruises of the 1980s showed that there were four assemblages of fishes in the strait (Fig. 8). Fifty species of fish were used in the statistical analysis of assemblage structure (Fargo and Tyler 1991). The deepwater assemblage in the gullies to the south is quite different from the other Hecate Strait assemblages and will not be discussed in this paper. The three assemblages occupying the 20-100 fathom range (40-200 m) are the subject of this discussion. They are dominated by rock sole, halibut (*Hippoglossus stenolepus*), rex sole (*Glyptocephalus zachirus*), Pacific sanddab (*Citharichthys sordidus*), English sole, Pacific cod, spiny dogfish (*Squalus acathias*), and big skate (*Raja binoculata*). Note that some of the dominant species of the eastern Bering Sea are also members of this fauna. There is evidence that some of the species, for example rock sole, form substocks within Hecate Strait that do not mix between the north and south areas of the strait (Fargo 1990). The differences between the north and south areas have been recognized for years by the traditional fishery there of small trawlers (65-95 ft; 22-32 m). As a result, biologists at the Pacific Biological Station divided the north and south into two fishery statistical areas: 5D (north) and 5C (south) (Fargo and Tyler 1991). This fleet remained the chief fishing effort on groundfish from the 1950s through the 1980s.

A natural fishery experiment, quite apart from fishery management, occurred in Hecate Strait during the 1980s, with the trawler fleet putting two to three time as much fishing effort in the north compared to the south (Fig. 9). The reason for this difference in effort was that the north was closer to the port of Prince Rupert and its processing plants. This effort differential gave the biologists a chance to determine whether the extra fishing effort was causing differences in relative abundance of species between north and south in the assemblages detected by the surveys. They found that catch rates from the research surveys for the groups of
Figure 8. Assemblage map of Hecate Strait, British Columbia (Fargo and Tyler 1991).
species were unrelated to commercial effort (Fargo and Tyler 1991), and that differences in species diversity, species richness, and evenness were also unrelated. It was clear that fishing practices in Hecate Strait did not bring about the kind of species abundance distortions that were witnessed in the Grand Banks or Georges Bank groundfish assemblages. Skates did not become dominant as they did on Georges Bank. An experiment in fishery management was proposed (Fargo and Tyler 1991), in which the effort level of the south area (5C) would be held at 1988 levels, while levels in the north would be encouraged to double again by way of testing in an adaptive sense (Walters and Hilborn 1976, Sainsbury 1988, Hilborn and Walters 1992) for the level of effort that would just bring about undesirable change. Though this management measure has not been carried out, the analysis can serve as a model for future adaptive management programs and could yet be implemented. The increases could have been implemented through stepwise quota increases every 5 years on key target species such as Pacific cod and English sole. The process of stepwise quota increase could then have been managed over a period of many years.
How to Design an Assemblage Maintenance Program

In looking over the various assemblage responses to heavy fishing effort in the ocean around North America, it is possible to say that there are the specific kinds of multispecies changes that one would expect from an adaptive assemblage maintenance program. I suggest a series of five assemblage-change hypotheses that are testable and based partly on Atlantic experience.

1. The least productive species would become scarce on the basis of the mixed stock model of Paulik et al. (1967).

2. The biomasses of the economically valuable target species would decline by too much if effort was too high.

3. Some of the productive, nontarget species would increase because they were not targeted.

4. Some skate species would become more abundant, perhaps becoming dominant in the mixed-species biomass. The reason for this change is unclear. Perhaps most skates are discarded and have high survival rates.

5. Species diversity indices would be lower because of the many biomass declines without other species replacing them.

If those are the assemblage-level responses one might expect, what would be the management program that would set up a decision-making framework? Change in biomass must be detected that is due to catch rate and distinguished from responses to physical environmental change.

a. For this decision to be made, an adaptive design must include geographic regions with applied fishing rates that are sharply differing—one area having two or three times the level of effort as another area. It would be better if there were more than two areas and levels of effort. The areas would have to be very large, large enough to include subpopulations of several species that do not mix among the areas. Only in this way will differing effects of the effort levels be independently detectable and not smeared through the areas.

b. Fishing effort itself is known to be generally difficult to manage as compared to annual catch quotas. The management of fishing level must be done via annual quotas on key species that would produce approximately a given level of effort.

c. The total allowable catch in any one area would have to be increased in a stepwise manner in 4- or 5-year intervals in order to allow analysis of assemblage change. If no assemblage or individual species re-
response was detected that could be attributed to fishing effort, then either the level would remain in place or an increase could be made.

d. A low catch rate would always be maintained in one area as a control and as a way to sample the assemblage. If similar changes were detected on both the control and the high fishing rate area, it would be assumed that it was not fishing-related but due to an external factor acting on both areas.

e. Specific biological reference points would have to be developed in relation to possible assemblage changes. These reference points would be quantitative thresholds for management action taken from the biological changes described as five hypotheses (above). The reference points would be indicators that changes related to the effects of fishing had progressed too far. When a combination of thresholds was reached, fishing effort would be reduced as a management measure in the high fishing rate area.

f. The program would be carried out as an adjunct to the standard stock assessment programs being conducted on main economic species. If serious changes at the assemblage level were detected, though no overfishing occurred according to individual stock assessments, the level of fishing effort would be cut back to maintain the assemblage. This would be done by reducing the annual quota on one or more of the target species. The assemblage maintenance program would in this way override the individual stock assessments. The operating principle is that assemblages as a whole cannot withstand the high levels of effort appropriate for the most productive species.

g. Assemblage maintenance programs would not apply to very migratory species and to species like walleye pollock, Pacific halibut, and sablefish whose single stocks occupy such vast areas as the continental shelf and slope of the Gulf of Alaska. The program would be applied to groups of stocks that have more restricted distributions and that can be divided into substocks that do not mix rapidly throughout their species distribution. Typically such substocks are not genetically distinct, but have differing somatic growth rates and other vital parameters.

What Next?

Of the cases reviewed here, where could the assemblage maintenance programs be developed in Atlantic or Pacific North America? The strong remedial action that either the Grand Banks or Georges Bank require would mitigate against setting up adaptive programs there. There is a good possibility an area of low effort should be set up with respect to the flatfish fisheries in the Pacific region of either Canada or the United States. A low
effort region would be arranged for the flatfish-dominated assemblages described in the published analyses (Fargo and Tyler 1991, Jay 1996). Off Washington and Oregon these lie along the depth contours and tend to be narrow areas running for many miles north and south. The species in these areas would have to have at least two substocks that could be fished with low and high effort levels. These areas would be quasi-replicates in the adaptive management plan. The assemblage areas described by Jay (1996) tend to be very limited, and adjacent assemblages may have to be combined so that the area would be large enough for management. It is well known that it is next to impossible to enforce quota levels on small areas, and so it may be that the low-effort area would have to be closed entirely except for an arranged test fishery that would keep track of abundance changes.

It is likely that an assemblage management program could be developed for the eastern Bering Sea. There are already extensive closures in the southern part put into place to help restore crab fisheries and to prevent trawling in areas frequented by feeding Steller sea lions (Fritz et al. 1995). In addition, some of the on-bottom trawling is limited by the bycatch limitations on catch of red king crabs, Tanner crabs, and Pacific halibut. Because of these restrictions, the yield maximization paradigm has not been fully implemented in the eastern Bering Sea. Perhaps that is why those assemblages seem to be in good condition. A plan might be to set up a specific area for increased fishing rate while simultaneously outlining an area for maintaining the present fishery effort levels consciously into the future. There is evidence from flatfish studies in the continental shelf of the eastern Bering Sea (Wilderbuer et al. 1992) that there are substocks of yellowfin sole that would possibly be a basis for dividing the area into two adaptive management zones (Fig. 10). One might be a high effort zone, the other a zone where present levels would be maintained. The substock structure of other species should be examined to make decisions regarding the boundaries of the areas for assemblages.

The present political climate in fisheries makes it more likely than in previous years that an assemblage maintenance program be established. Never before have so many North American fisheries been closed, or nearly closed, due to overfishing. It is clear to most stakeholders that ocean stocks can be overfished. Fifteen to twenty years ago many were still arguing that overfishing was unlikely. Others will argue that the yield maximization paradigm is sufficient, if only it is applied correctly. I believe the approach is not sustainable and have given arguments here in that direction. In the present way of conducting fisheries, the only proof that it is not working is a failed fishery. The onus of proof must be on the advocates of this intensive approach to resource use, not the other way around (Dayton 1998). If assemblage maintenance programs were instituted, much of the biomass would be fished at levels below those recommended by the yield maximization paradigm. These substocks would be available for
Figure 10. Substock distributions of yellowfin sole in the eastern Bering Sea (Wilderbuer et al. 1992).
reestablishing future fisheries. Perhaps more important, the assemblage maintenance programs would keep a portion of the fishery alive while the rest of the resource is rebuilding.

**References**


Ecosystem Approaches for Fisheries Management


