# Marine Mammals and Human Health in the Eastern Bering Sea: Using an Ecosystem-based Food Web Model to Track PCBs

by

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## Abstract

The comprehensive changes that have occurred in the Bering Sea over the last 30 years have prompted a wide range of studies to better understand the ecosystem as a whole. One set of studies has used the Ecopath with Ecosim (EwE) modelling software to synthesise existing biological data and gain insight into how the ecosystem was before and after the system-wide changes. This modelling framework provides a means for tracing contaminants through the ecosystem, and evaluating the role that persistent organic pollutants (POPs) may have played in the changing dynamics of the eastern Bering Sea. Using the EwE software, the likely pathways of PCB flow within the eastern Bering Sea were identified and health implications of contaminant exposure for Steller sea lions, other species of marine mammals, and humans were evaluated. The base EwE model was refined from existing models and validated with traditional stock assessment data. Ecotracer (a component of the EwE software) tracked the bioaccumulation of contaminants moving through the system with biomass. The models estimated contaminant concentrations for species and functional groups that have not previously been measured. Results suggest that PCB concentrations for most species in the eastern Bering Sea have remained below threshold levels associated with negative reproduction and survival effects. However, these concentrations may have subtle effects on adults and more serious effects on foetuses and nursing young, which could inhibit the recovery of Steller sea lions and other species that have declined in the eastern Bering Sea. Although the benefits of traditional foods appear to continue to outweigh the risks posed by contaminants for humans, PCB exposure and dietary intake for many Alaska Natives subsisting on marine mammals is above the USEPA Daily Reference Dose. Results extend the existing eastern Bering Sea models and are important in terms of management alternatives for marine mammals and human health. They also synthesise evidence regarding the presence, extent, and movement of PCBs throughout the system. The refined eastern Bering Sea models are useful tools for exploring different scenarios and hypotheses, to inform resource managers, and to further our understanding of this ecosystem.

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# CHAPTER 1 : GENERAL INTRODUCTION

## 1.1 STUDY OBJECTIVE

An ecosystem refers to a community of "organisms interacting with one another and with the chemical and physical factors making up their environment" (Miller 1990). The bond and extent of these links give the ecosystem its structure and boundaries. Even the most basic and uncomplicated ecosystems are comprised of many interrelationships at all levels of the food chain. Accordingly, chemical contamination within any ecosystem will have profound effects throughout the food webs that it supports.

The Bering Sea is a diverse region of tremendous biological productivity (Loughlin et al. 1999) and is home to a wide variety of marine wildlife. Such a remote area, far away from heavily industrialised areas, would intuitively be thought of as an especially 'pristine' environment, free from outside disturbances. However, because of atmospheric and oceanic transport processes, scientists and regulators are becoming increasingly concerned about the presence and potential hazards of contaminants in this ecosystem.

Over the last 30 years, comprehensive changes have occurred to the biomass and composition of the marine community in the Bering Sea. The effects of these system-wide shifts are particularly apparent, and of great concern, to the communities that rely on the long-term sustainability of all Bering Sea resources (Loughlin et al. 1999). The Bering Sea region is home to many indigenous peoples and the marine resources of the Bering Sea continue to be essential to their survival (Loughlin et al. 1999).

The focus of my study was on the potential impacts of polychlorinated biphenyls (PCBs) on marine mammals and human health in the eastern Bering Sea. The hypothesis was that PCBs have accumulated to high enough concentrations that they threaten marine mammal survival and pose a health threat to Alaska Natives. Using the modelling software Ecopath with Ecosim (Christensen and Pauly 1992), and a new component called Ecotracer (Christensen and Walters 2004), the objectives of this study were to: 1) Simulate the movements of PCBs in the eastern Bering Sea; 2) Assess the implications of PCB contamination for marine mammals and humans, and 3) Identify options to reduce or prevent this potential risk. In order to fulfil the above objectives, it was first necessary to refine previously constructed models of the Bering Sea (Trites et al. 1999a) to reflect a more comprehensive understanding of the ecosystem.

Associated issues that were also addressed included: a) a review of PCB concentrations present in the eastern Bering Sea ecosystem; b) comparison of PCB levels relative to recommended and legislated guidelines, and c) consideration of the extent to which PCB concentrations may adversely affect marine mammals, and also humans who consume marine mammals.

My study constitutes a description and analysis of contaminant flow through the eastern Bering Sea food web. It also provides a tool that will facilitate ecosystem-based conservation and management by resource managers (federal, regional and community), decision-makers, scientists and fisheries personnel in the region. This is an important step in increasing the understanding of the dynamics of the eastern Bering Sea ecosystem and enhancing the ability of resource managers to protect healthy, sustainable fish and wildlife populations in this productive and diverse region.

## 1.2 STUDY AREA/MODEL DOMAIN

> "The name Alaska is probably an abbreviation of Unalaska, derived from the original Aleut word agunalaksh, which means 'the shores where the sea breaks its back.' The war between water and land is never-ending. Waves shatter themselves in spent fury against the rocky bulwarks of the coast; giant tides eat away the sand beaches and alter the entire contour of an island overnight; williwaw winds pour down the side of a volcano like snow sliding off a roof, building to a hundred-mile velocity in a matter of minutes and churning the ocean into a maelstrom where the stoutest vessels founder. Here is the very breeding ground of storms. Cold air blowing off the Siberian land mass strikes the moisture-laden air of the warm Japanese current; the cauldron bubbles and boils, and a succession of lows, like jets of steam from a tea kettle, shoot eastward along the Aleutian chain" (Ford 1966, p. 10).

The Bering Sea is a sub-arctic environment that is separated from the northern Pacific Ocean by the Aleutian Islands and the Alaska Peninsula. The Aleutian Islands archipelago extends south-west from the Alaska Peninsula for more than 1,770 km and consists of more than 50 islands (NRC 1996; Brennan and Tower 1998; Cone 2000). Located west of the North American continental land mass, the Bering Sea/Aleutians region broadly includes all Alaskan waters in the area between longitudes 162° east and 157° west, and between latitudes 52° and 66° north (NRC 1996).

The Bering Sea and waters on the broad subarctic shelf constitute one of the most biologically productive systems in the world (Loughlin et al. 1999). This is due in part to extensive winter ice coverage and turbulence, as well as density inversion bringing nutrient-rich current systems from the bottom up into the water column (Fienup-Riordan 1988; Loughlin et al. 1999). Populations of primary producers expand in the spring as the amount of light and nutrients increase. This is followed in succession by increases in zooplankton and invertebrates. Invertebrates such as copepods, euphausids, and squid are important components in the diet of baleen whales. Pelagic fish such as herring and capelin, as well as toothed whales and other marine mammals, all congregate in the shallow coastal waters for breeding and feeding during the warm summer months. Migratory species such as the great whales (blue, fin, sei, humpback, gray, right, and sperm) are drawn to the area by the abundance of food, spending their summers feeding and building up fat reserves before migrating south to their breeding grounds for the winter (Fienup-Riordan 1988). This abundance of food also attracts and supports some of the world's largest commercial fisheries, exploiting salmon, crabs, and groundfish such as walleye pollock (Loughlin et al. 1999).


**FIGURE 1.1.** The eastern Bering Sea is separated from the northern Pacific Ocean by the Aleutian Islands and the Alaska Peninsula.

The area covered by the eastern Bering Sea models encompasses both the south-central and south-eastern Bering Sea regions, i.e., the region covered by the Alaska Fisheries Science Center's bottom trawl surveys (Figure 1.1). This region can be divided into four parts: the Inner Shelf, the Middle Shelf, the Outer Shelf, and the Slope. Within this area are the Pribilof Islands, which are well known as important rookeries for approximately 70% of the world's population of northern fur seals (*Callorhinus ursinus*) (Trites and Larkin 1989; Trites 1992; Beckmen et al. 2002; Beckmen et al. 2003). St. Matthew Island is also within the modelled boundary, as are the smaller Hall and Pinnacle Islands.

## 1.3 BACKGROUND INFORMATION

### 1.3.1 Alaska Natives

The region is also home to many indigenous peoples. There are two major cultural and linguistic distinctions of arctic peoples – the Aleut and Eskimo, with the Eskimo further divided into the Yup'ik and Iñupiat language groups (Pullar 1999). In 1996 the US National Research Council (NRC) determined that the Bering Sea coast was home to approximately 65,000 Alaska Natives, most notably the Aleut, Iñupiat, and Yup'ik peoples (NRC 1996). By 1999 this number was 62,646 and represented 51% of the 123,118 people living in the 270 rural communities in Alaska (Wolfe 2000). While the term 'Aleut' may be the more widely recognised name used to refer to the Alaska Natives who occupy the Aleutian Islands, the westernmost tip of the Alaska Peninsula, the Pribilof and Shumagin Islands, and the Commander Islands of Russia, the term 'Unangan' (a word meaning "people") is what they typically call themselves (AMAP 1997; Brennan and Tower 1998; Pullar 1999). "Aleut" was the name given to these people by Russian fur traders in the 18th century (Brennan and Tower 1998; Pullar 1999), and may have been derived from the Chukchi word "aliat", which means "island" (Brennan and Tower 1998). The Iñupiat/Inuit of Alaska are located in the north and north-western coastal and tundra regions, north of Norton Sound in the Bering Sea (AMAP 1997; Brennan 1998) and are also dispersed as far as Greenland (Pullar 1999). The Yup'ik in Alaska are part of the Eskimo/Inuit group (Brennan and Schmittroth 1998). The Siberian Yup'ik in Alaska live on St. Lawrence Island in the Bering Sea while the more numerous Central Yup'ik live mainly in small villages along the coast of the Bering Sea and also in the tundra of the Yukon-Kuskokwim Delta. The Central Yup'ik are the most abundant of Alaska's Inuit groups (AMAP 1997). The Alutiiq peoples of south-western and south-central Alaska inhabit Kodiak Island, the Alaska Peninsula, Lower Kenai Peninsula and Prince William Sound, sharing cultural attributes with both the Aleut and the Yup'ik (AMAP 1997; Schmittroth 1998).

These coastal and island dwelling Alaska Natives are primarily a sea-going people and they all depend on the marine resources of the Bering Sea. These resources have contributed to the food, clothing, and cultural heritage of the indigenous peoples in the rural coastal communities of Alaska for centuries, and they continue to be essential to their survival today (Loughlin et al. 1999). Rural Alaskans currently have a mixed subsistence-cash economy, supporting themselves through fishing, hunting, and gathering of wild foods combined with small-scale cash employment (Wolfe 1996). As Alaska Natives consume large quantities of wild food each year, the contamination of these food sources is of growing concern throughout rural Alaska (Wolfe 1996).

### 1.3.2 Timeline: Aleut History

Roughly 13,000 years ago, as the intense cold of the last glaciation turned water into ice, the sea level dropped and what was termed "Beringia" emerged. Beringia, also known as the Bering Land Bridge, was an enormous land mass (~1,600 km wide) that once joined the North American and Asian continents and now lies beneath the Bering Strait (Chance 1990; Pullar 1999). Scholars generally believe that indigenous peoples from Asia migrated to North America in two waves, the first approximately 11,000 years ago across the Bering Land Bridge (Brennan 1998; Brennan and Schmittroth 1998). Another recent theory suggests that the first Americans may have arrived even earlier from a wave of seafaring coastal people, possibly originating from regions other than Asia (Erlandson 2002). However, the evidence on which this is based is still largely circumstantial and the theory is not widely accepted. It may also be that the first Americans migrated by both means (Erlandson 2002). In any case, archaeological and geological evidence indicates that the ancestors of present-day Alaska Natives can be traced back to the second migratory wave around 4,000 B.C. (Brennan 1998; Brennan and Schmittroth 1998; Brennan and Tower 1998). At the time, these migratory peoples spoke a common language known today as 'Eskaleut', with their later geographic separation resulting in separate peoples: Aleut, Yup'ik, and Iñupiat (Chance 1990; Brennan and Tower 1998).

Alaska Native cultures are in the midst of change as they adapt to a wide variety of modern influences. This process dates from the time of first European contact, when the greatest changes began. Prior to the first European encounter with Alaska, the Aleut population was believed to have reached 15,000 people. Since then a number of events and changes have taken place as listed in the following timeline (Spencer et al. 1977; Fienup-Riordan 1988; Chance 1990; McCartney 1995; Jarman et al. 1996a; AMAP 1997; Brennan and Schmittroth 1998; Brennan and Tower 1998; Bacon et al. 1999; Loughlin et al. 1999; Malinowski et al. 1999; Pullar 1999; Wespestad et al. 2000).

**1700s** – Vitus Bering made first European contact with Natives in Alaska.

> **1745** – Inspired by Vitus Bering, Russian fur traders arrived on Attu Island. Tribes united against their Russian 'intruders', often ending in bloody battles. Fifteen Aleuts were killed in an inlet now known as Massacre Bay. Continued resistance toward Russian dominance led to repeated confrontations. Progressively the Russians occupied the Aleutian Islands and, through the Russian-American Company, controlled the fur trade in Alaska.
>
> **1786** – Discovery of northern fur seals by Gerassim Pribylov, commander of the Russian ship St. George, led to the naming of the Pribilof Islands. It also marked the beginning of an almost 200-year-long (until 1984) commercial harvest. Known as the "fur seal period", this harvest was the only commercial activity in the Bering Sea until 1845.
>
> **1790** – Russians lose interest in Alaska. By this time the Aleut population had suffered a reduction of over 50% of their numbers, due to ongoing conflicts and the impact of new diseases.
>
> **18$^{th}$ and 19$^{th}$ centuries** – Merging of Russian and Aleut cultures; intermarriages began to dilute indigenous ancestry.

**19$^{th}$ century** – Aleut life returned to a more traditional pace.

> **1845** – American whaling ships expanded from the Kodiak Island area and south of the Aleutian chain into the Bering Sea to begin harvesting bowhead and right whales. The "whaling period" flourished for nearly 70 years.

**1867** – United States purchased Alaska from the Russian-American Company under the Treaty of Cession. The bowhead whale population had already been reduced to half of its original numbers.

**1870** – Federal legislation sanctioned Aleut harvesting of fur seals on the Pribilof Islands, provided the use was for food and clothing.

**20th century** – The Americanization of Alaska.

**1914** – Whale populations had been dramatically reduced; whaling in the Bering Sea was no longer economical for most commercial whaling companies.

**1942** – World War II, Japanese took control of Attu Island and relocated the Aleut residents to Japan. In response, the US government evacuated the remainder of the Aleutian archipelago and relocated the Aleut peoples to refugee camps in southeast Alaska. The Aleutian Islands became the site of considerable military activity.

**1952** – "Commercial fishing period" in the Bering Sea began with Japanese trawlers. Herring fishing predominated the initial period from the 1950s to the 1960s.

**1958** – Alaska became a state of the U.S.; the US government confiscated 104 million acres of land previously reserved for indigenous peoples.

**1965 to 1971** – The Aleutian Island of Amchitka was the site of 3 underground nuclear experiments.

**1971** – Implementation of the Alaska Native Claims Settlement Act (ANCSA). The initial Act excluded many Aleut peoples by specifying that in order for an Alaskan to qualify as a person of indigenous heritage they must have a minimum of one-half native ancestry. The terms of the Act were later amended to require only one-quarter indigenous ancestry. Settlement resulted in 12% of Alaskan land being reverted back to Native ownership.

**1972** – Marine Mammal Protection Act formally recognized Alaska Natives' use of wildlife for subsistence and enabled coastal Alaska Natives to take marine mammals for this purpose.

**1973 – present** – Walleye pollock comprised 85% of commercially important groundfish species.

**1978** – Alaska Department of Fish and Game establishes the Division of Subsistence and adopts legislation requiring authorization and protection for subsistence uses of fish and wildlife.

**1980s –** Many populations of marine mammal and seabird declined.

**1990s –** Studies indicate only 4,000 Aleut descendants remain.

For centuries, Alaska Natives have lived off the land using wild resources for food, clothing, shelter, and handicrafts. Numerous species also have cultural significance as part of native heritage. Although many things have changed in the lifestyles of Alaska Natives over the last several hundred years, a traditional subsistence way of life remains very important to them. In addition, because employment opportunities in rural Alaska are few and typically low paying, subsistence activities are the more reliable sector of the economy and provide much of rural Alaska with food (Rue 1998; Wolfe 2000).

### 1.3.3 Hunting and Subsistence in Aleut Culture

Over the centuries Alaska Natives have generally maintained a subsistence hunting and gathering lifestyle (Spencer et al. 1977; Brennan and Schmittroth 1998). The economies and cultures of many communities in Alaska rely on subsistence hunting and fishing, which remain central to the customs and traditions of the Aleut, Yup'ik, and Iñupiat. Alaska's subsistence harvesting is regulated by State and Federal governments, and there are overlapping jurisdictions in many areas (Wolfe 2000). Federal and State laws define subsistence as the "customary and traditional uses" of wild resources for food, clothing, fuel, transportation, construction, sharing, and customary trade (Wolfe 2000). Both jurisdictions prohibit subsistence harvesting in non-rural areas, and both give subsistence uses of fish and mammals priority over commercial fishing, recreational fishing, and recreational hunting. Once taken, subsistence wildlife may not be sold as part of an ongoing business, although subsistence hunters are permitted to continue the traditional trade. As a result, most rural residents are able to make use of subsistence foods for at least part of the year.

The entitlement of Alaska Natives to harvest wildlife for subsistence use was further reinforced by the Marine Mammal Protection Act (1972), which permitted Alaska Natives to hunt beluga whales, polar bears, sea otters, walrus, sea lions and five seal species for subsistence purposes without restriction (Chance 1990). With the view that maintaining traditional ways is of vital importance, Dr. Angayuqaq Oscar Kawagley, a Yupiaq educator, interpreted the concept of subsistence for Alaska Native people as follows: "Alaska Native peoples have traditionally tried to live in harmony with the world around them. This has required the construction of an intricate subsistence-based worldview, a complex way of life with specific cultural mandates regarding the ways in which the human being is to relate to other human relatives and the natural and spiritual worlds" (Pullar 1999, p. 625).

Subsistence activities of Alaska Natives are also 'controlled' by customary laws passed through generations (Wolfe 1989). Customary laws are focused on maintaining ecosystems in a sustainable manner so that they will provide subsistence resources for future generations.

Alaska Natives' subsistence activities tend to be part of a mixed 'subsistence & cash' economy, characterized by both paid employment and subsistence food production, although the monetary component has always been relatively small (Wolfe 1989). In these mixed subsistence-cash economies, a large part of cash income is directed towards the technology and supplies needed to continue subsistence activities. Generally, the largest number of subsistence items are harvested by individuals with the greatest cash income (Rue 1998). If families are unable to obtain sufficient subsistence foods, substitutes would need to be purchased. Assuming a replacement cost of $6 to $11 per kilogram, the replacement value of wild food harvests for Alaska Natives would be between \$130 million and \$220 million each year (Wolfe 2000).

Alaska Natives harvest and consume approximately 20 million kilograms of wild foods annually. Based on 1990 population numbers, this averages between 160 and 170 kilograms (useable weight) per person per year (Wolfe 1996; Rue 1998; Wolfe 2000). Although it comprises the majority of the food supply for Alaska Natives, the subsistence harvest represents only 2% of Alaska's total annual wildlife harvest; commercial fisheries harvest approximately 97%, while hunting and sport fishing account for the other 1% (Wolfe 2000).

Those Alaska Natives who consume marine mammals obtain most of their nutrition from 'muktuk', the outer layer of skin and marine mammal blubber, which is very rich in vitamins A, D, and C. Subsistence foods are important sources of vitamins, minerals, and energy for Alaska Natives. A shift away from these traditions would likely be detrimental to their social, spiritual, and physical health (AMAP 1997).

Because the Aleuts inhabit an island chain bordered by the Bering Sea and the Pacific Ocean, a large proportion of their diet has traditionally been derived from the marine environment (AMAP 1997; Brennan and Tower 1998). It is probably no coincidence that Aleut and Alaska Native villages were often in close proximity to known and predictable locations for intercepting local marine mammal populations, and migrating fur seals and whales, particularly bowhead and gray whales (Dumond 1995; Harritt 1995). Marine mammals are an integral part of Aleut culture, livelihood, and diet, and for centuries Aleuts have hunted sea otters, seals, sea lions, walrus, and whales (McCartney 1995). Fish, such as salmon, cod, halibut and herring, as well as mussels and clams, have also been a large part of the Aleut diet. Additionally, caribou have been hunted when available, and ducks, geese, other birds, and bird eggs have been favoured delicacies (AMAP 1997; Brennan and Tower 1998).

For many Aleutian and Pribilof Islands communities, marine mammals have constituted a large portion of the total annual subsistence harvest. However a number of species declined markedly since the mid 1970s. These include harbour seals (Pitcher 1990), northern fur seals (Trites 1992) and Steller sea lions (Trites and Larkin 1996). Numbers of Steller sea lions declined by more than 80% between the 1970s and 1990s (Trites and Larkin 1996). Subsistence take of Steller sea lions has fallen and was estimated to be 432 in 1992 and 164 in 1995 (Wolfe 1997; J. Bereskin, Mayor of Akutan, *pers. comm.*). Reasons for the population declines are uncertain (DeMaster and Atkinson 2002; Trites and Donnelly 2003).

Generally speaking, Alaskan and Arctic people live in close contact with their environment. This is especially true for indigenous people. Any threat or compromise to subsistence resources, such as fish, marine mammals or terrestrial food sources, is therefore a threat to Native cultures and their ability to acquire nutritious foods. Greater awareness regarding the presence of persistent organic pollutants (POPs), and their effect on human health (particularly PCBs), has led many to question whether or not traditional foods are safe to eat (AMAP 1997).

# CHAPTER 2 : REFINING THE EASTERN BERING SEA ECOSYSTEM MODELS

## 2.1 INTRODUCTION

Historically, human activities have negatively affected animals and the environment on which they depend (Jackson et al. 2001). Thus humans need to be thought of as a part of nature. The dependence of people on the life-support function of ecosystems, irrespective of technological advancements and sophistication, posses a challenge to those charged with conserving and managing ecosystems (Berkes et al. 1998). In particular, the direct and indirect interactions of humans and their activities on ecosystem health and functionality need to be understood as fully as possible to best manage and conserve natural resources for the future.

Attempts to examine and understand ecosystems have largely been driven by analysis of single species and their place in an ecosystem. However, the inadequacies of single-species approaches have increasingly become apparent with improved understanding that many interwoven factors contribute to and determine the pattern of functioning within an ecosystem. The systematic study of complex and variable systems is often made easier through the development and use of ecosystem models. As a result, a shift from single-species approaches to broader ecosystem-based approaches has emerged (Pauly et al. 2002).

Comprehensive changes have occurred over the last 30 years to the biomass and composition of the Bering Sea's marine community (NRC 1996). One hypothesis is that the changes were the result of a sudden shift in ocean climate in the late 1970s that favoured the survival of one suite of species over another (Mantua et al. 1997; Benson and Trites 2002). Another proposed explanation is that the commercial removal of large whales cascaded through the food web, affecting the flow of energy and species at other trophic levels (Trites et al. 1999a; Springer et al. 2003). The effects of these system-wide shifts have been apparent and of great concern to the communities that rely on the sustainability of all Bering Sea resources (Loughlin et al. 1999).

The decline of so many top predators may indicate that the ecological integrity of the eastern Bering Sea ecosystem was disrupted. This led to the construction of two ecosystem models to move beyond the assessment of an individual species and evaluate impacts on the structure and function of the entire food web (Trites et al. 1999a). The Ecopath with Ecosim (EwE) software package (Christensen and Pauly 1992; Walters et al. 1997; Christensen and Walters 2004) was therefore used to compare the state of the ecosystem at two very different times (the 1950s and the 1980s), and the effects of different harvesting scenarios on the structure of the system.

Ecosystem models can be used to predict the properties of a system that are often difficult or too costly to measure (Hall and Day 1977; Keen and Spain 1992). In a contaminant context, it would be very time consuming and costly to measure 'everything'. However, models can help to conceptualize complex systems. They are essentially simplified representations of real systems and are dependent on the information and data that are fed into them. The nature of a model is to synthesise just enough detail to produce observable patterns; good models do not seek to reproduce every detail of a system (Levin 1992). A model can also be thought of as a tool to reveal data gaps and identify areas for future research and data collection. Consequently, the eastern Bering Sea models constructed by Trites et al. (1999a) were revisited to incorporate a better understanding of this system.

## 2.2 METHODS

### 2.2.1 Defining the modelled area

The geographic area defined for the modelled ecosystem is located in the eastern Bering Sea, and encompasses the region covered by the Alaska Fisheries Science Center's bottom trawl surveys of the shelf and slope down to 800m and is bounded by the Aleutian Islands to the south (Figure 2.1). The area covers 484,000 $km^2$ and includes 119,000 $km^2$ of Inner Shelf area (<50m), 211,000 $km^2$ of Middle Shelf area (50-100m), 133,000 $km^2$ of Outer Shelf area (100-200m), and 21,000 $km^2$ of Slope (200-800m) (Trites et al. 1999a; Trites et al. 1999b). Although the area defined in the model covers both the shelf and slope regions and includes a wide range of marine habitats and organisms, all are considered to be functionally connected within the framework of the ecosystem. The area is therefore treated as a single homogenous area (Trites et al. 1999b). The northern portion of the Bering Sea and nearshore fauna were not included in the scope of the original model. The boundaries chosen for the modelled area were based on the availability of systematically collected assessment data in relation to fisheries and marine mammals (Trites et al. 1999a).

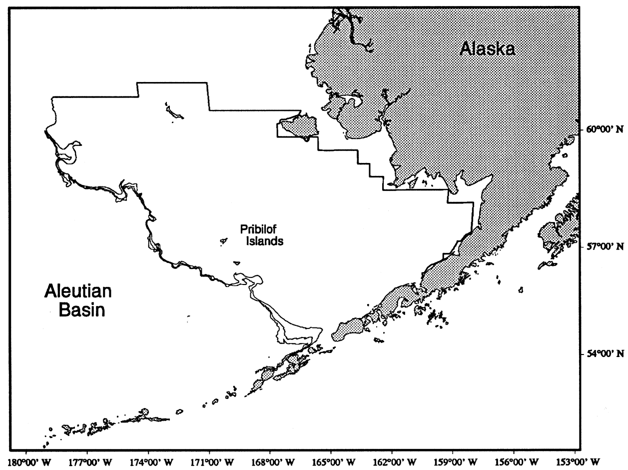


**FIGURE 2.1.** The eastern Bering Sea as defined in the ecosystem model. Total area is 484,000 $km^2$ (Trites et al. 1999a).

The Pribilof Islands are a group of four islands within the study area: Saint Paul, Saint George, Otter and Walrus Islands. They are in the middle of the Bering Sea approximately 322 km north of the Aleutian Island Unalaska (NRC 1996). Of these, St. Paul and St. George are the two largest and populated islands. The topography of St. George is one of hills and ridges with steep cliffs rising up to 274 m, while St. Paul has a rolling plateau with some extinct volcanic peaks (NRC 1996). The islands of St. Matthew, Hall, and Pinnacle, also within the modelled boundary, are approximately 324 km west of mainland Alaska and north of the Pribilof Islands (NRC 1996).

### 2.2.2 Ecosystem modelling using Ecopath

The Ecopath software is a mass-balance approach that was initiated by the work of Polovina (1984) and later modified by Christensen and Pauly (1992). It has since been further developed to include a dynamic routine called Ecosim, capable of simulating ecosystem changes over time (Walters et al. 1997), as well as the spatially explicit Ecospace model, designed to explore the impact and placement of protected areas (Walters et al. 1999). Both of these components are based on the Ecopath mass-balance approach. Ecotracer is a more recent addition; a simulation model running parallel to Ecosim to track the flow of a tracer or contaminant (Christensen and Walters 2004).

The two fundamental master equations in Ecopath are:

1) Production = fisheries catch + predation mortality + biomass accumulation + other mortality + net migration;

2) Consumption = production + respiration + unassimilated food.

The first equation includes only the production of a given functional group and describes how that production is split into components. The second equation ensures energy balance within each functional group; in other words, the energy going into a group is balanced by the energy leaving it.

When creating an Ecopath model, several basic input parameters are required for each functional group included in the ecosystem model. These include biomass (B); production/biomass ratio (P/B); consumption/biomass ratio (Q/B), and ecotrophic efficiency (EE). Of these, Ecopath can estimate one, given the others. Input of diet composition data is also essential as it defines the linkages between functional groups. As an Ecopath model need not be in steady-state, biomass accumulation (BA) can be used as an additional parameter, included where appropriate to represent the accumulation or depletion of biomass in a functional group.

The Ecopath with Ecosim (EwE) suite of software (www.ecopath.org) has been applied worldwide and encapsulates the shift from traditional single-species research and management to ecosystem-based research and management. More detailed descriptions of EwE are contained in Christensen and Walters (2004).

### 2.2.3 Eastern Bering Sea models

Two models of the eastern Bering Sea were constructed by Trites et al. (1999a); one representing the late 1950s and covering the period from 1955 to 1960, while the 1980s model covered the period from 1979 to 1985. The models were constructed in an earlier version of the EwE software, before certain current functions and capabilities had been developed. For this reason, both models were refined here to better represent the ecosystem defined in the models. In the process, a number of changes were made to each of the models, as described below. For detailed information on those aspects of model parameterization not mentioned below, see Trites et al. (1999a).

*2.2.3.1 Refining the 1950s model*

The original 1950s model had nine functional groups with ecotrophic efficiencies above 1.0. This meant that a number of changes had to be made to the input parameters to balance the model. Harvest rates of walrus & bearded seals, sperm whales, and baleen whales were reduced by Trites et al. (1999a) so that they would balance with production. In the refined 1950s model, these landings were set back to the reported harvest levels (Table 2.1).

**TABLE 2.1.** Harvest levels in baleen whales, sperm whales, and walrus & bearded seals were returned to observed 1950s levels. Resulting EE's were greater than 1.0. Assuming EE's of 0.95 for each of the three functional groups, negative biomass accumulation (BA) values were estimated by Ecopath in order to balance the 1950s model.

| Functional Group | Original harvest ($t{\cdot}km^{-2}$) | Refined harvest ($t{\cdot}km^{-2}$) | Resulting EE | BA ($t{\cdot}km^{-2}{\cdot}year^{-1}$) |
|---|---|---|---|---|
| Baleen whales | 0.0130 | 0.0837 | 6.03 | -0.0707 |
| Sperm whales | 0.0088 | 0.0210 | 2.392 | -0.0127 |
| Walrus & bearded seals | 0.0030 | 0.0061 | 1.047 | -0.0006 |

Using the initially reported harvest levels of baleen whales, sperm whales, and walrus & bearded seals resulted in these groups again having ecotrophic efficiencies greater than 1.0 (Table 2.1).

Balancing the model, using these observed harvest levels for the 1950s meant employing negative biomass accumulation (BA) (Table 2.1). Biomass accumulation rates were estimated by the Ecopath modelling software by providing an EE of 0.95 for the three overharvested functional groups. They represent the biomass lost each year due to the high level of harvesting of these groups.

Walrus & bearded seal biomass were underestimated in the original model, which also contributed to the high EE. It was recalculated using the total Bering-Chukchi 1960 walrus population estimate of 80,000 individuals provided in Fay et al. (1997). This changed the biomass estimate from 0.054 to 0.099 $t{\cdot}km^{-2}$ for the walrus & bearded seal functional group. This recalculation also affected the calculated Q/B ratio, changing it from 11.651 to 10.989 $year^{-1}$.

Pacific herring appears to play a key role in the eastern Bering Sea ecosystem and was therefore separated from the pelagics group and considered as its own functional group in the model. Adjustments were made throughout the diet composition matrix to reflect this separation and the key role of Pacific herring (Table A1.1). Diet adjustments indicated the importance of Pacific herring to top level predators, particularly Steller sea lions where over 80% of their diet was apportioned to Pacific herring. The Pacific herring and other pelagics groups were assigned EE's of 0.95 and 0.90 respectively, in order for Ecopath to estimate their biomasses.

To better indicate the importance of individual fishing fleets, fisheries information was divided into 7 separate fleets: Bottom Trawl, Midwater Trawl, Longline, Crab Pots, Pelagic, Whaling, and Seal/Sea lion take. This separation allows more flexibility in Ecosim when simulating various scenarios and assessing their respective impacts on the ecosystem.

Changes were made to the original parameter inputs as follows:

- Steller sea lion landings were reduced from 0.001 to 0.0001 t·km$^{-2}$;
- Deepwater fish landings were reduced from 0.001 to 0.0001 t·km$^{-2}$; and
- Small flatfish biomass was corrected from 8.530 to 8.593 t·km$^{-2}$.

Adult pollock biomass had been increased to 5.5 t·km$^{-2}$ in order to balance the original model. In the refined model, this was amended back to the originally reported value of 4.32 t·km$^{-2}$ as it did not need to be adjusted to balance the model.

The Ecopath parameters describing the refined 1950s eastern Bering Sea ecosystem are shown in Table 2.2.

**TABLE 2.2.** Ecopath parameters describing the 1950s eastern Bering Sea ecosystem with 25 functional groups where P/B is the ratio of production to biomass; Q/B is the ratio of consumption to biomass, and EE is the ecotrophic efficiency. See Trites et al. (1999a) for further details on parameter references. Shaded cells are those that were estimated by Ecopath; dashes indicate a parameter not applicable to a given group.

| Functional Group | Biomass (t·km$^{-2}$) | P/B (year$^{-1}$) | Q/B (year$^{-1}$) | Landings (t·km$^{-2}$) | EE |
|---|---|---|---|---|---|
| 1. Baleen whales | 0.696 | 0.020 | 13.678 | 0.0837 | 0.950 |
| 2. Toothed whales | 0.009 | 0.020 | 13.108 | 0.0000 | 0.655 |
| 3. Sperm whales | 0.439 | 0.020 | 4.553 | 0.0210 | 0.950 |
| 4. Beaked whales | 0.001 | 0.020 | 10.515 | 0.0000 | 0.000 |
| 5. Walrus & bearded seals | 0.099 | 0.060 | 10.989 | 0.0061 | 0.950 |
| 6. Seals | 0.106 | 0.060 | 15.577 | 0.0050 | 0.823 |
| 7. Steller sea lions | 0.029 | 0.060 | 12.703 | 0.0001 | 0.125 |
| 8. Piscivorous birds | 0.006 | 0.800 | 60.000 | 0.0000 | 0.000 |
| 9. Adult Pollock | 4.320 | 0.500 | 2.640 | 0.0140 | 0.262 |
| 10. Juvenile Pollock | 0.942 | 2.500 | 8.333 | 0.0000 | 0.403 |
| 11. Other demersal fish | 8.957 | 0.433 | 2.226 | 0.0010 | 0.338 |
| 12. Large flatfish | 1.169 | 0.400 | 2.444 | 0.0016 | 0.399 |
| 13. Small flatfish | 8.593 | 0.400 | 2.968 | 0.1050 | 0.907 |
| 14. Pacific herring | 16.249 | 1.000 | 3.650 | 0.0206 | 0.950 |
| 15. Other pelagics | 8.005 | 0.800 | 3.650 | 0.0619 | 0.900 |
| 16. Deepwater fish | 1.011 | 0.400 | 2.490 | 0.0001 | 0.534 |
| 17. Jellyfish | 0.048 | 0.875 | 2.000 | 0.0000 | 0.018 |
| 18. Cephalopods | 3.500 | 3.200 | 10.667 | 0.0000 | 0.962 |
| 19. Benthic particulate feeders | 29.000 | 1.480 | 7.690 | 0.0100 | 0.304 |
| 20. Infauna | 75.000 | 1.373 | 12.000 | 0.0000 | 0.983 |
| 21. Epifauna | 8.000 | 1.578 | 5.777 | 0.0000 | 1.000 |
| 22. Large zooplankton | 44.000 | 5.091 | 22.000 | 0.0000 | 0.837 |
| 23. Herbivorous zooplankton | 55.000 | 6.000 | 22.000 | 0.0000 | 0.917 |
| 24. Phytoplankton | 32.000 | 60.000 | - | 0.0000 | 0.971 |
| 25. Detritus | - | - | - | 0.0000 | 0.956 |

*2.2.3.2 Refining the 1980s model*

In the original 1980s model, initial parameter estimates for the 25 functional groups were consistent with one another, and only one group, the walrus & bearded seals, had an ecotrophic efficiency above 1.0 at 2.076. Still, a number of changes were made to the input parameters to balance the model. The harvest of walrus & bearded seals had originally been reduced by half in order to balance the model (Trites et al. 1999a). In the refined 1980s model, walrus landings were returned to observed harvest levels from 0.004 to 0.0091 $t \cdot km^{-2}$.

After basic parameterization, walrus & bearded seals again had an ecotrophic efficiency greater than 1.0. Walrus & bearded seal biomass had been underestimated in the original model and was recalculated using the total Bering-Chukchi 1980 walrus population estimate of 256,900 individuals provided in Fay et al. (1997). Not only did this change the biomass estimate from 0.074 to 0.1953 $t \cdot km^{-2}$ for the walrus & bearded seal functional group, but this recalculation also affected the calculated Q/B ratio, changing it from 11.249 to 10.596 $year^{-1}$. With this biomass estimate corrected, the ecotrophic efficiency was no longer above 1.0 and it was not necessary to employ the negative biomass accumulation required for the refined 1950s model.

As for the 1950s model, Pacific herring was again separated from the Pelagics group in the refined 1980s model. The herring biomass (0.779 $t \cdot km^{-2}$) used in the refined model was taken from the 45-box model initially constructed for the eastern Bering Sea prior to aggregation of the 45 groups into the current 26 functional groups (Trites et al. 1999a). While a biomass estimate exists for Pacific herring, the remaining species, now represented in the other pelagics group, are not well sampled (Trites et al. 1999a). For Ecopath to estimate the biomass of other pelagics, an EE of 0.90 was assigned. Adjustments were also made throughout the diet composition matrix to reflect these changes (Table A1.2).

By-catch data exists for fisheries in the Bering Sea during the 1980s (Alverson and Hughes 1995; Queirolo et al. 1995) and was incorporated into the model. Both catch and discard data were divided into 7 separate fleets: Bottom Trawl, Midwater Trawl, Longline, Crab Pots, Pelagic, Whaling, and Seal/Sea lion take. The original diet composition matrix treated discards as detritus and thus had small proportions of detritus in the diet of several top predators. This was revised to reflect the addition of a discard group and entered as the proportion of discards being consumed by these groups.

An error in original parameter input resulted in Steller sea lion ‘landings’ being corrected from 0.001 to 0.0002 t·km$^{-2}$. The Ecopath parameters describing the refined 1980s eastern Bering Sea ecosystem are shown in Table 2.3.

**TABLE 2.3.** Ecopath parameters describing the 1980s eastern Bering Sea ecosystem with 26 functional groups where P/B is the ratio of production to biomass; Q/B is the ratio of consumption to biomass, and EE the ecotrophic efficiency. See Trites et al. (1999a) for further details on parameter references. Shaded cells are those that were estimated by Ecopath; dashes indicate a parameter not applicable to a given group.

| Functional Group | Biomass (t·km$^{-2}$) | P/B (year$^{-1}$) | Q/B (year$^{-1}$) | Landings (t·km$^{-2}$) | EE |
|---|---|---|---|---|---|
| 1. Baleen whales | 0.394 | 0.020 | 11.383 | 0.0000 | 0.030 |
| 2. Toothed whales | 0.009 | 0.020 | 13.108 | 0.0000 | 0.655 |
| 3. Sperm whales | 0.208 | 0.020 | 4.553 | 0.0000 | 0.000 |
| 4. Beaked whales | 0.001 | 0.020 | 10.515 | 0.0000 | 0.000 |
| 5. Walrus & bearded seals | 0.195 | 0.060 | 10.596 | 0.0091 | 0.787 |
| 6. Seals | 0.066 | 0.060 | 15.926 | 0.0013 | 0.388 |
| 7. Steller sea lions | 0.019 | 0.060 | 12.702 | 0.0002 | 0.175 |
| 8. Piscivorous birds | 0.006 | 0.800 | 60.000 | 0.0000 | 0.021 |
| 9. Adult pollock | 27.451 | 0.500 | 2.640 | 1.8950 | 0.417 |
| 10. Juvenile pollock | 6.000 | 2.500 | 8.333 | 0.0000 | 0.902 |
| 11. Other demersal fish | 3.904 | 0.433 | 2.226 | 0.1285 | 0.733 |
| 12. Large flatfish | 1.900 | 0.400 | 2.444 | 0.0496 | 0.418 |
| 13. Small flatfish | 9.161 | 0.400 | 2.968 | 0.2109 | 0.545 |
| 14. Pacific herring | 0.779 | 1.000 | 3.650 | 0.0550 | 0.872 |
| 15. Other pelagics | 13.772 | 0.800 | 3.650 | 0.1568 | 0.900 |
| 16. Deepwater fish | 0.407 | 0.400 | 2.490 | 0.0069 | 0.995 |
| 17. Jellyfish | 0.048 | 0.875 | 2.000 | 0.0000 | 0.018 |
| 18. Cephalopods | 3.500 | 3.200 | 10.667 | 0.0000 | 0.827 |
| 19. Benthic particulate feeders | 5.800 | 1.480 | 7.690 | 0.1080 | 0.898 |
| 20. Infauna | 46.500 | 1.373 | 12.000 | 0.0000 | 0.698 |
| 21. Epifauna | 5.858 | 1.578 | 5.777 | 0.0000 | 0.619 |
| 22. Large zooplankton | 44.000 | 5.091 | 22.000 | 0.0000 | 0.540 |
| 23. Herbivorous zooplankton | 55.000 | 6.000 | 22.000 | 0.0000 | 0.900 |
| 24. Phytoplankton | 32.000 | 60.000 | - | 0.0000 | 0.971 |
| 25. Discards | 1.000 | - | - | 0.0000 | 0.492 |
| 26. Detritus | 100.000 | - | - | 0.0000 | 0.703 |

### 2.2.4 Ecosystem modelling using Ecosim

Ecosim is a dynamic model that is based on the Ecopath mass-balance approach. Developed through the work of Walters et al. (1997), Ecosim allows for the simulation of biomass changes over time. While various mortality rates are altered on selected functional groups, Ecosim calculates the relative biomass changes and plots them over time. To ensure the model accurately predicts biomass changes and also as a form of validation (Christensen and Walters 2004), Ecosim allows for predicted biomasses to be fit to time series stock assessment data. This validation process often requires repeated simulations and increases the confidence in the outputs of the model.

Ecosim also allows for policy exploration. The user can explore different fishing mortality rates and gain insights into the possible optimum rates that may achieve the sustainability objectives sought by regulators. These estimates should not be expected to be precise or taken at face value, but should be considered to provide a reasonable range of values to guide management decisions for the sustainable use of resources.

## 2.3 RESULTS

### 2.3.1 Ecopath outputs

There are many outputs that can be generated from Ecopath. Among the most informative are the system statistics generated within the basic parameterization routine shown in Table 2.4. Several of these indices can be used to assess the status and maturity of an ecosystem (Odum 1971) and are useful when comparing the ecological characteristics of a system over time; in this case from the 1950s to the 1980s.

Odum (1971) considered the ratio between total primary production and total respiration to be important for the description of the maturity of an ecosystem. System maturity in the eastern Bering Sea, as indexed by this ratio, decreased from the 1950s to the 1980s. In a mature system (i.e., the 1950s), this ratio approaches 1 whereas in an immature system the ratio is greater than 1 (i.e., the 1980s) (Table 2.4). Ecosystem maturity can also be inferred from net system production, i.e., the difference between total primary production and total respiration (Christensen and Walters 2004). The large system production value estimated for the 1980s model (297 $t \cdot km^{-2} \cdot year^{-1}$) indicates an immature system while that for the 1950s (48 $t \cdot km^{-2} \cdot year^{-1}$) approaches zero indicating a more mature system (Table 2.4). One index that contradicts the conclusion regarding the loss of maturity in the eastern Bering Sea is the ratio between total primary production and total biomass (Table 2.4). In an immature system, biomass is expected to accumulate, which will, in turn, influence the ratio. The value for the 1950s (6.46) was expected to exceed the value for the 1980s (7.47).

The eastern Bering Sea supported more biomass in the 1950s, when the system was considered mature, indicating that a disruption in the ecological integrity of the system occurred between then and the 1980s. As the sum of all flows in a system, total system throughput indicates the ‘size’ of a system in terms of its flow. Throughput in the system as described in the 1950s model was greater than in the 1980s.

**TABLE 2.4.** Summary statistics generated by Ecopath within the Basic Parameterization routine for the 1950s and 1980s eastern Bering Sea ecosystem models.

| Parameter | 1950s Value | 1980s Value | Units |
|---|---|---|---|
| Sum of all consumption | 3558.37 | 3078.27 | $t \cdot km^{-2} \cdot year^{-1}$ |
| Sum of all exports | 48.52 | 297.32 | $t \cdot km^{-2} \cdot year^{-1}$ |
| Sum of all respiratory flows | 1871.56 | 1623.15 | $t \cdot km^{-2} \cdot year^{-1}$ |
| Sum of all flows into detritus | 1087.05 | 992.51 | $t \cdot km^{-2} \cdot year^{-1}$ |
| Total system throughput | 6566 | 5991 | $t \cdot km^{-2} \cdot year^{-1}$ |
| Sum of all production | 2677 | 2615 | $t \cdot km^{-2} \cdot year^{-1}$ |
| Mean trophic level of fishery catches | 3.41 | 3.29 | - |
| Gross efficiency (catch/net p.p.) | 0.0002 | 0.0016 | - |
| Calculated total net primary production | 1920 | 1920 | $t \cdot km^{-2} \cdot year^{-1}$ |
| Unaccounted primary production | - | - | - |
| Total primary production/total respiration | 1.03 | 1.18 | - |
| Net system production | 48.44 | 296.85 | $t \cdot km^{-2} \cdot year^{-1}$ |
| Total primary production/total biomass | 6.46 | 7.47 | - |
| Total biomass/total throughput | 0.045 | 0.043 | - |
| Total biomass (excluding detritus) | 297.18 | 256.98 | $t \cdot km^{-2} \cdot year^{-1}$ |
| Total catches | 0.33 | 3.10 | $t \cdot km^{-2} \cdot year^{-1}$ |
| Connectance Index | 0.27 | 0.30 | - |
| System Omnivory Index | 0.173 | 0.148 | - |

Flow/food web diagrams of the trophic interactions and energy flow in the eastern Bering Sea were similar in layout to those in Trites et al. (1999a). The estimated trophic level for each functional group depicted in the flow diagrams is provided in Table 2.5. As in the original models, large numbers of flows originate from a few species at trophic level three (pollock, small flatfish and pelagic fish). However, the flow from the pelagics is now divided between Pacific herring and other pelagics. Top predators such as marine mammals, birds, large flatfish and deepwater fish are the major consumers, and have trophic levels between 4.0 and 4.8.

**TABLE 2.5.** Estimated trophic levels for each functional group in the 1950s and 1980s models. Modified from Trites et al. (1999a) to reflect additional functional groups and changes in trophic level estimation.

| Functional group | 1950s | 1980s | Functional group | 1950s | 1980s |
|---|---|---|---|---|---|
| Sperm whales | 4.75 | 4.68 | Adult pollock | 3.34 | 3.29 |
| Beaked whales | 4.59 | 4.56 | Pacific herring | 3.25 | 3.20 |
| Toothed whales | 4.36 | 4.31 | Other pelagics | 3.11 | 3.09 |
| Steller sea lion | 4.30 | 4.20 | Small flatfish | 3.20 | 3.16 |
| Piscivorous birds | 4.13 | 4.02 | Jellyfish | 3.16 | 3.15 |
| Large flatfish | 4.03 | 4.02 | Juvenile pollock | 3.09 | 3.07 |
| Deepwater fish | 4.01 | 4.04 | Benthic particulate feeders | 2.82 | 2.78 |
| Seals | 4.01 | 3.94 | Epifauna | 2.63 | 2.38 |
| Other demersal fish | 3.80 | 3.85 | Large zooplankton | 2.27 | 2.22 |
| Cephalopods | 3.76 | 3.68 | Infauna | 2.00 | 2.00 |
| Baleen whales | 3.61 | 3.65 | Herbivorous zooplankton | 2.00 | 2.00 |
| Walrus & bearded seals | 3.52 | 3.52 | Phytoplankton | 1.00 | 1.00 |

Flow and biomass pyramids generated within the network analysis routine in Ecopath are useful for inter-system comparison, provided the pyramids are built on the same scale. Pyramids created using the refined Bering Sea models were virtually identical between years and to those in Figure 8 of Trites et al. (1999a) (Figure 2.2). Because the volume of each compartment is proportional to the biomass of that trophic level, they show that apex predators contribute little to the flow of biomass through the system. The top angle of the flow pyramid is inversely proportional to the geometric mean of the transfer efficiencies between trophic levels and indicates that much of the flow in the system occurs within the lower trophic levels. The conclusion remains that, compared to other shelf systems, the eastern Bering Sea is still a mature system.

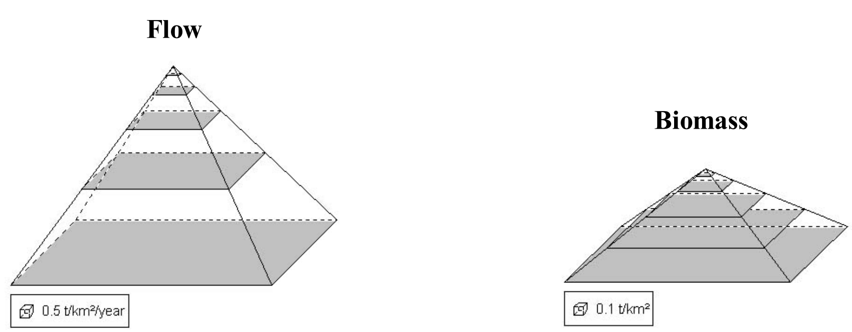


**FIGURE 2.2.** Flow and biomass pyramids illustrating the distribution of biomass and energy flow in the 1950s eastern Bering Sea ecosystem. The volume of each compartment is proportional to the total throughput (sum of all flows) of that trophic level. The top angle of the flow pyramid is inversely proportional to the geometric mean of the transfer efficiencies between trophic levels. An acute angle indicates high efficiency.

Mixed trophic impact diagrams are also generated within the network analysis routine. Using the Leontief Matrix, the relative direct and indirect impacts that a very small increase in the biomass of one group has on the biomass of other groups can be explored (Figure 2.3). The findings from Trites et al. (1999a) indicated that an increase in the biomass of the apex predators of the eastern Bering Sea had little or no impact on the biomass of other groups and that most impacts were associated with biomass increases of lower trophic levels. Although some subtle differences in the graphs from the refined models exist, these findings remain similar. In general, increasing the biomass at low and mid trophic levels has a positive effect on higher trophic levels such as for Steller sea lions and other marine mammals (Trites et al. 1999a). These positive impacts indicate that the dynamics of the eastern Bering Sea are driven in large part by bottom-up processes. The trophic impact graphs from the refined models are shown in Figure 2.3 to again illustrate these findings and also to reflect the adjusted functional groups and separation of fishing fleets.

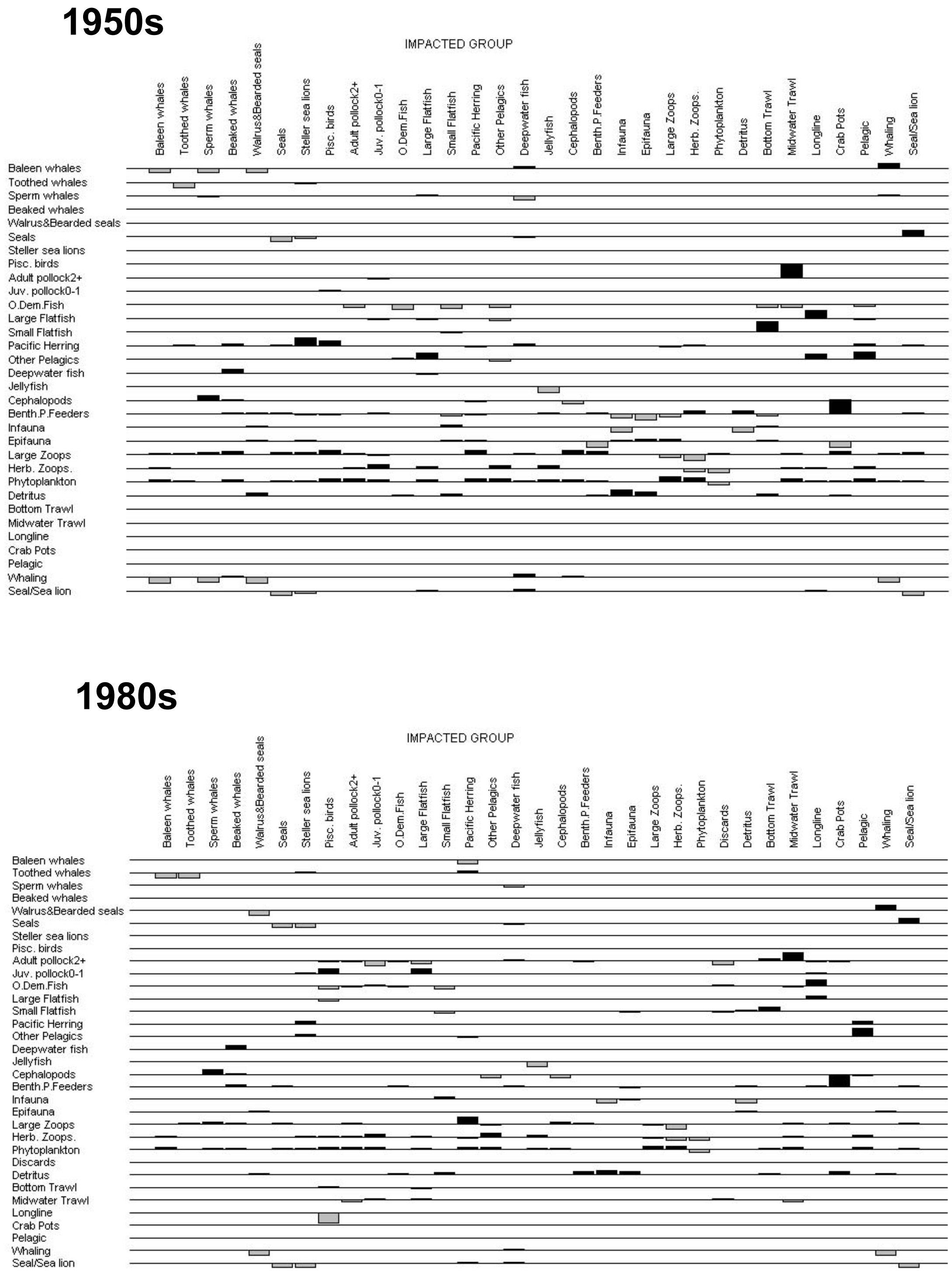


**FIGURE 2.3.** Mixed trophic impacts showing the combined relative direct and indirect trophic impacts in the eastern Bering Sea ecosystem in the 1950s and 1980s models. Increasing the abundance of a group on the Y-axis will either have a positive (black bar pointing upwards), negative (grey bar pointing downwards), or no impact on a group listed on the X-axis. Impacts are relative but comparable between groups. Modified from Trites et al. (1999a) to reflect additional functional groups and the separation of fishing fleets.

The separation of fishing fleets reveals the importance of individual functional groups to a given fleet. As expected, increases in abundances of key target species resulted in considerable positive impacts for their respective fishery, while species having negative impacts on target species also had relative negative impacts on the fishery. For example, while a small increase in adult pollock biomass had a positive impact on the midwater trawl fishery, other demersal fish, which had a negative impact on adult pollock, also had a negative impact on midwater trawlers.

The largest positive impact on Steller sea lions resulted from an increase in Pacific herring biomass in both the 1950s and 1980s models. Pacific herring biomass in the eastern Bering Sea was high in the 1950s, and the model assumed that over 80% of sea lion diet was comprised of this pelagic species. This explains the positive impact an increase in herring biomass would have on Steller sea lions. While comparatively less (32%) of sea lion diet was comprised of herring in the 1980s model, an increase in herring biomass nevertheless resulted in a relative positive impact on Steller sea lions.

It is interesting to note the difference between the two modelled time periods in terms of the impact resulting from increases in biomass. As indicated in the trophic impact diagrams (Figure 2.3), although essentially the same directional trends are observed for the two time periods, the size of potential impacts in the 1950s exceeds those in the 1980s. Although the reasons for this are unclear, it may be at least in part due to commercial harvesting pressures of the time. Perhaps this could be considered in future research.

### 2.3.2 Ecosim outputs – Validating the model

Many changes occurred in the eastern Bering Sea between the 1950s and 1980s. In order to verify that the 1950s model accurately predicted these changes, time series data were incorporated into Ecosim. Time series information included data to account for the changing fishing mortality (F) in a number of commercially exploited groups (Fay et al. 1997; Myers et al. 1995; A. Springer, University of Alaska Fairbanks, *pers. comm.*; C. Walters, University of British Columbia, *pers. comm.*), as well as biomass changes (Fay et al. 1997; Myers et al. 1995; C. Walters, University of British Columbia, *pers. comm.*) known to have occurred for comparison with predicted biomasses from the model.

Potential impacts predicted through a review of the trophic impact diagrams above indicated that most impacts in the system were associated with biomass changes at lower trophic levels. Ecosim was used to search for time series values of annual relative primary productivity in order to estimate the historical productivity 'regime shift' thought to have occurred in the late 1970s (Christensen and Walters 2004). This fluctuating production was expected to produce biomass and mortality dynamics changes in the model and was used to explore ecosystem impacts from the bottom up through the inherent trophic linkages in the system.

Each time Ecosim was run, the model generated a statistical measure of goodness of fit to the time series data as a weighted sum of squared (SS) deviations of log biomasses, i.e., time series data, from log predicted biomasses from the model (Christensen and Walters 2004). Predator/prey vulnerability settings were adjusted from the default value (0.3, which represents a mix of top-down and bottom-up control) to those that gave better 'fits' of Ecosim to the time series data (i.e., lower SS). Note that the lowest vulnerability setting (0.0) represented full bottom-up control, while a vulnerability setting of 1.0 represented full top-down control.

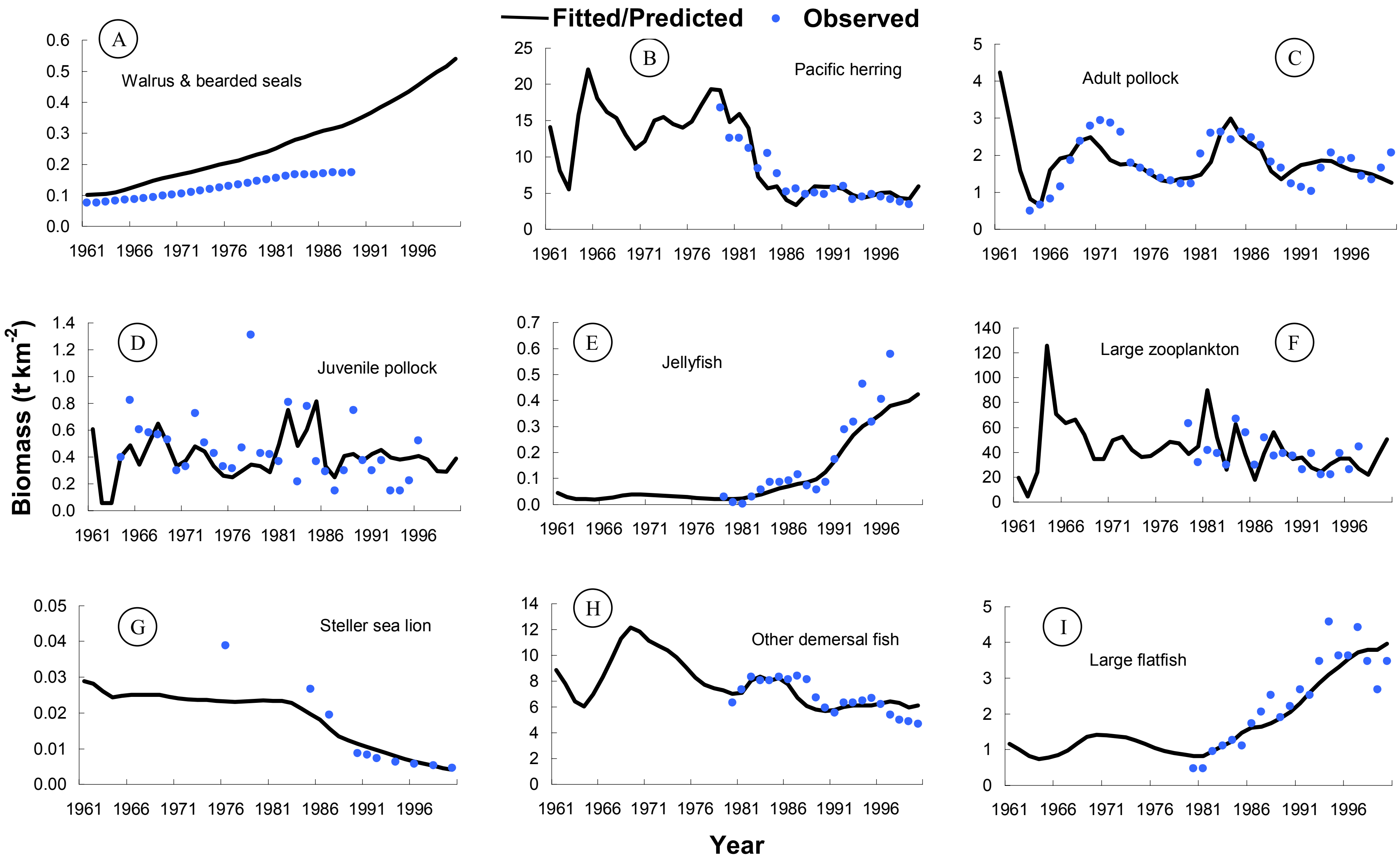


**Figure 2.4.** Observed biomass estimates (dots) for 9 species or species groups in the eastern Bering Sea model compared to EwE fitted/predicted biomass (line). A (Fay et al. 1997); B, E, F, G, H, I (C. Walters, University of British Columbia, *pers. comm.*); C, D (Mito et al. 1990; Myers et al. 1995).

Leaving the vulnerability settings at the default value of 0.3 failed to explain the observed biomass dynamics. Instead, the dynamics were better explained by increasing the prey vulnerabilities to 0.9 for most settings. Such high values indicate a high degree of top-down dynamics. Best fits were obtained for the remaining groups when vulnerabilities for prey of deepwater fish were set to 0.1, and phytoplankton settings were left at the default value of 0.3.

Although Ecosim does not include a formal sensitivity analysis, the developers acknowledge that Ecosim predictions are most sensitive to the prey vulnerability settings (Walters et al. 1997; Christensen et al. 2000; Christensen and Walters 2004). As expected, the biomass changes that developed from this simulation were strongly dependent on prey vulnerability settings. The simulation resulted in a sum of squares of 29.4, compared to an initial value of 66.6. To provide a quantitative measure of uncertainty, 1,000 model scenarios were also run using the Monte Carlo sampling procedure in Ecosim. Monte Carlo simulations indicate that the data from the best fitting trial is that with which the model was parameterized (i.e., the scenario described above).

My model generated temporal dynamics for several of the species groups that had independent time series data (Figure 2.4). The model predicted some of the time series biomasses well (e.g., Pacific herring, Adult pollock, and Large flatfish). However, other dynamics were less well explained (e.g., Walrus & bearded seals and Steller sea lions). Although the model predicted a decline in Steller sea lion biomass, it did not capture the full extent of change observed.

## 2.4 DISCUSSION

Ecosystem models provide a framework to visualize ecosystems and a broad means to understand a region and its function. Such models synthesise a wide array of available data and can be validated with data from more traditional single-species research. Ecosystem models, such as the eastern Bering Sea models, can predict trends in biomass while other parameters are manipulated and also provide useful insights into the workings of the system. Although ecosystem models can be criticized for oversimplifying systems, the shift from traditional single-species research and management to ecosystem-based research and management is particularly important for resource managers seeking to achieve the sustainable use of all resources.

### 2.4.1 Utility of model to resource managers

The status of marine mammal populations and fisheries abundance in Alaska is of growing concern to the general public, the North Pacific fishing industry, and to Native communities. Traditional fisheries management has focused on single-species or stocks. With an increasing knowledge that all components of an ecosystem are linked and need to be managed as a whole rather than separately, resource managers and policy-makers are charged with the task of whole ecosystem management (FAO 1997). As basic management tools, ecosystem models have proven to be very helpful for resource managers in this regard (FAO 1997). Their predictive power in defining probable trends of relative abundance is especially important in guiding managers towards effective management decisions; with the result that more and more managers are turning towards models for guidance (Caddy and Sharp 1986; Brussard et al. 1998).

Predictions based on models are particularly useful because they can be verified with field observations and can shed light on ecosystem interactions. While the simulation conducted in my study (i.e., primary productivity fluctuations together with commercial harvest of Pacific herring) explains a number of the biomass dynamics known to have occurred in the eastern Bering Sea, it is important to remember that this is but one simulation and that the model represents only one configuration of this ecosystem. However, one can gain confidence in the model's predictions through this exercise, and additional hypotheses can be explored with this or other parameterization of the system (e.g., Aydin et al. 2002).

### 2.4.2 Conclusions

Although refining the eastern Bering Sea models led to several changes in the input parameters, the overall findings outlined in Trites et al. (1999a) remain. The two time periods compared (i.e., the 1950s and 1980s) indicate that while the eastern Bering Sea was a more mature system in the 1950s, the system described in the 1980s was still mature (Trites et al. 1999a).

Validating the 1950s model with Ecosim incorporated fluctuating primary production. The subsequent biomass reductions of lower trophic level animals, together with high harvest levels, led to depleted populations of high trophic level predators such as marine mammals and seabirds. This study supports the statement that the effects of environmental fluctuation and heavy commercial pressures on key species resulted in system-wide biomass changes (NRC 1996).

The criticism that the Ecopath with Ecosim approach assumes homogeneous spatial behaviour has been remedied with the development of Ecospace (Walters et al. 1999). Ecospace is as yet an untapped tool for the eastern Bering Sea models. A spatially explicit Ecospace model could be used to track species movements within the system and to explore the potential role of Marine Protected Areas (MPAs). The models could also be further refined to include animal migrations to and from the eastern Bering Sea.

While validation of the refined 1950s model provides one insight into what may have contributed to the decline of Steller sea lions in Western Alaska, there are a number of hypotheses that could still be tested. For instance, the annual ice cover in this region is of considerable importance to phytoplankton production (NRC 1996; Whitledge and Luchin 1999), as well as to the distribution and density of many marine mammals (Braham et al. 1984). Incorporating the effects of this additional variable into the models may prove useful in providing further understanding of the system. Some scientists have also proposed that PCBs and other environmental contaminants could have been a factor in the decline of marine mammals and other species in Western Alaska and the Bering Sea (see Chapter 3).

Although there have been a number of hypotheses suggested for the causes of Steller sea lion decline in Western Alaska, few have proposed ways in which sea lion abundance may be restored. The 1980s eastern Bering Sea model could be used to explore different scenarios in this regard. For instance, Barrett-Lennard et al. (1995) suggested that killer whale predation has the potential to inhibit the recovery of Steller sea lions and may be a contributing factor to further decline (Springer et al. 2003). Such a hypothesis could be examined with the refined 1980s model.

Predator-prey interactions were captured by the vulnerability settings in Ecosim. Studies using Ecosim often manually manipulate vulnerability to fit model predictions to time-series data because simulations are sensitive to changes in vulnerability. It is therefore important to report the values used. Above all, it is important that default settings not be relied upon without considering the possible implications of doing so.

Models of the eastern Bering Sea can be criticized as being oversimplifications of the system. Such criticism usually fails to recognise that trade-offs are inherent in building workable ecosystem models. While models that are too simple are not useful, models that are overly complex often lack detailed supporting data and incorporate increased uncertainty regarding both input parameters and predictions. The challenge is to incorporate sufficient detail to reveal patterns and dynamics, without necessarily attempting to include every detail of the system (Levin 1992). In any case, analyses of uncertainty such as Monte Carlo simulations should be undertaken to provide a quantitative measure of uncertainty when presenting results.

From a scale perspective, it is important to remember that there may be a number of appropriate scales of investigation. Large-scale models should not be seen as replacements for existing smaller-scale research; rather, such models should complement and extend our understanding of the system. As functions and patterns are assessed on larger scales, the determinants and underlying dynamics should still be explored at smaller scales. In a fisheries context, this means that ecosystem advice should accompany traditional stock assessment advice provided to fishery managers, not replace it (Christensen and Pauly 2004).

The primary goal of the ecosystem-based approach to resource management is to maintain ecosystem health and sustainability. Ensuring that ecosystem integrity takes precedence over other management goals is a primary tenet of resource conservation (Grumbine 1994; Brussard et al. 1998). This central goal gives rise to a number of key objectives, including sustainable exploitation of living resources, maintaining characteristic structure and function, ensuring productivity and biodiversity, maintaining genetic diversity, and protecting habitats. Achieving these objectives is a challenge that involves assessing ecosystem status and health, defining system reference states, and managing human actions to maintain the system within these states. To this end and with reference to the eastern Bering Sea, these models are invaluable for illuminating system interactions, and can guide management decisions and future research.

### 2.4.3 Summary

Ecosystem changes in the Bering Sea seem inevitable given the magnitude of commercial fisheries, whaling, climate change and other environmental factors. The comprehensive changes that have occurred over the last 30 years have lead to studies to better understand the ecosystem as a whole. The Ecopath with Ecosim modelling software has been used worldwide and was used to construct two models of the eastern Bering Sea. One model represented the 1950s time period and the other the 1980s, before and after the system-wide changes, respectively. While the original models were constructed in an earlier version of the software, this study used the latest version of the software to focus on refining the models and validating them with traditional stock assessment data. Quantification of ecosystem parameters was improved for modelling and assessment purposes. Although the models indicate that the eastern Bering Sea was essentially a mature system in both time periods, marked differences between them were observed. Results extend the existing eastern Bering Sea models (Trites et al. 1999a) and indicate that system-wide biomass changes have been associated with environmental fluctuation and commercial harvesting pressures on key species. The refined eastern Bering Sea models are useful tools for exploring different scenarios and hypotheses, and to further our understanding of this ecosystem and potential responses to these.

# CHAPTER 3 : ASSESSING IMPACTS OF PCB CONTAMINATION ON MARINE MAMMALS AND HUMAN HEALTH IN THE EASTERN BERING SEA

## 3.1 INTRODUCTION

The Bering Sea is a diverse ecosystem with tremendous biological productivity and a wide variety of marine wildlife (Loughlin et al. 1999). However, comprehensive changes have occurred over the last 30 years to the biomass and composition of its marine community (NRC 1996). The effects of these system-wide shifts are of great concern to the communities that rely on the long-term sustainability of all Bering Sea resources (Loughlin et al. 1999).

Located far from heavily industrialised areas, the remoteness of the Bering Sea intuitively implies that it should be an especially 'pristine' environment, free from outside disturbances. However, this is not the case. Considerable concentrations of contaminants have been transported here by atmospheric and oceanic processes.

Polychlorinated biphenyls (PCBs) and other persistent organic pollutants (POPs) could have been, and may continue to be, a factor in the decline of marine mammals and other species in western Alaska and the Bering Sea. POPs are the most discussed contaminants in Alaska. They tend to resist degradation and accumulate in fish and marine mammal tissues. Marine mammals readily accumulate POPs throughout their lifespan and are particularly susceptible to high levels of POP contamination as a consequence of their high trophic positions.

The focus of this chapter is on the potential impacts of PCBs on marine mammals and human health in the eastern Bering Sea. Using the modelling software Ecopath with Ecosim (Christensen and Pauly 1992) and the newly refined eastern Bering Sea models (see Chapter 2), a new component to the software called Ecotracer (Christensen and Walters 2004) was used to account for the bioaccumulation of PCBs moving through the system with biomass. The objectives were to: 1) Simulate the movements of PCBs in the eastern Bering Sea; 2) Assess the implications of PCB contamination for marine mammals and humans, and 3) Identify options to reduce or prevent this potential risk.

### 3.1.1 Persistent Organic Pollutants (POPs) in the marine environment

Persistent organic pollutants (POPs) are a group of synthetic organic chemicals that share similar properties, are highly toxic, and can directly affect the health of animals and people (Chary 2000). POPs are lipophilic (fat soluble) compounds that are very persistent in the environment (Erickson 1997; Chiu et al. 2000) and can accumulate in the fat stores of all organisms (Parker and Dasher 1999). Accordingly, water quality is not a reliable indicator of the levels of contamination present in an ecosystem – the highest concentrations of these chemicals are typically found in sediments and tissue samples (Connolly 1991; Erickson 1997; Hope et al. 1997).

POPs are widespread, and many, such as polychlorinated biphenyls (PCBs), represent a serious threat to all living organisms within an ecosystem (Fromberg et al. 1999; Chary 2000). The United Nations Environment Programme (UNEP) has identified twelve of the most persistent and bioaccumulative POPs for priority action. Dubbed the 'dirty dozen', these are: aldrin, dieldrin, endrin, chlordane, DDT, heptachlor, mirex, toxaphene, hexachlorobenzene (HCB), polychlorinated biphenyls (PCBs), polychlorinated dibenzodioxins (dioxins), and polychlorinated dibenzofurans (furans) (Chary 2000). All contain chlorine.

POPs can enter marine environments in many ways, including spillage, sewage outfall, leakage from industrial sites and landfills, dumping, run-off, agricultural activities, pesticide applications, and atmospheric deposition. These contaminants can travel great distances from their points of origin to remote regions of the earth through atmospheric and oceanic currents (Ewald et al. 1998; Fromberg et al. 1999), as well as animal migration (Ewald et al. 1998; Krummel et al. 2003). Atmospheric transport and deposition is the dominant pathway for POP input into Arctic and subarctic regions (Barrie et al. 1992; Iwata et al. 1993; Iwata et al. 1994; Muir et al. 1996; Wania and Mackay 1996; Wania et al. 1998; Parker and Dasher 1999; Macdonald et al. 2000). Once released into the environment, POPs are difficult, if not impossible, to remove (Oceana 2003c).

POPs are termed multi-hop contaminants in view of how they can be transported around the globe as gases (Parker and Dasher 1999). Volatile and semi-volatile contaminants vaporise at warm temperatures (Parker and Dasher 1999). In particular, the high temperatures found at tropical and subtropical latitudes (where harmful POPs are often still in use, primarily in developing countries) facilitate volatilization (Fromberg et al. 1999; Parker and Dasher 1999). Following atmospheric transport to the cooler temperatures of higher latitudes, these compounds condense and fall out into colder waters (Iwata et al. 1993; Fiedler and Lau 1998a; Parker and Dasher 1999; Macdonald et al. 2000; Kajiwara et al. 2002). The transport process continues in a series of atmospheric hops until the contaminants reach polar regions. This has been termed the 'grasshopper effect' (Wania and Mackay 1996; AMAP 1997; Parker and Dasher 1999; Macdonald et al. 2000). Re-volatilization and degradation of POPs is reduced at low temperatures (Macdonald and Bewers 1996; Muir et al. 1996). This process of trapping POPs in cold climates is known as the 'global distillation' or 'cold condensation' effect (Macdonald and Bewers 1996; Muir et al. 1996;

Macdonald et al. 2000). As a result of this distillation, the POPs settle in high concentrations and make colder regions a sink for contaminants (Tanabe et al. 1994a; AMAP 1997; Fiedler and Lau 1998a; Parker and Dasher 1999; Chary 2000; Yao et al. 2001). Contaminants are then readily available for uptake into the food chain.

Bioconcentration, bioaccumulation, and biomagnification are three terms widely used to describe the dynamics of contaminants in living systems. In an aquatic environment, *bioconcentration* refers to the uptake of contaminants directly from water while *bioaccumulation* refers to a process that includes bioconcentration as well as dietary uptake. These processes concentrate contaminants in the food chain through *biomagnification*, which is the bioaccumulation of chemicals accumulated by prey at higher trophic levels (Macdonald and Bewers 1996; AMAP 1997; Connell 1998; Fiedler and Lau 1998b; Fromberg et al. 1999; Parker and Dasher 1999; Chary 2000). The primary pathway of contamination in marine mammals and humans is through biomagnification. Contaminants accumulate through successive trophic levels and persist at higher concentrations in the fatty tissue of top predators.

Marine environments are particularly susceptible to high POP contamination compared to other ecosystems (Macdonald and Bewers 1996; AMAP 1997). Marine mammals are apex predators that are characterized by long lives, low fecundity, and high body fat content (Macdonald and Bewers 1996; AMAP 1997). They are also supported by long and complex food chains, which accentuates biomagnification (Barrie et al. 1992; Tanabe et al. 1994b; Macdonald and Bewers 1996; AMAP 1997; Pauly et al. 1998; Berrow et al. 2002). These features make marine mammals particularly vulnerable to accumulating contaminants and manifesting their chronic toxic effects. In turn, the high contaminant burden carried by marine mammals poses a potential threat to the health of humans who subsist upon them (Barrie et al. 1992; AMAP 1997).

### 3.1.2 Polychlorinated Biphenyls (PCBs)

Of the 'dirty dozen', the most environmentally prominent POP is the group of chemicals called polychlorinated biphenyls (PCBs) (Thomas et al. 1992; Oceana 2003c). PCBs are organic chemicals that were first synthesized in 1881, and first manufactured and marketed on a commercial scale in the United States beginning in 1929 (IARC Monographs 1977; Environment Canada 1988, 1997; Erickson 1997).

Known in North America by the trade name 'Aroclor', PCBs are very stable, chemically inert, have low flammability, are virtually insoluble in water, resist heat and degradation, are soluble in organic solvents and oils, and have electrical insulating properties (Environment Canada 1988, 1997; Erickson 1997). These characteristics led to their widespread application in a variety of commercial products including:

dielectric (insulating) fluids in capacitors and transformers; heat transfer fluids; hydraulic fluids; lubricating and cutting oils; and as additives in pesticides, paints, inks, copying paper, carbonless copy paper, adhesives, sealants, and plastics (Environment Canada 1988, 1997; Erickson 1997; Fiedler and Lau 1998a).

There are 209 possible PCB congeners or compounds, each differing by the quantity and position of chlorine atoms in the biphenyl structure (Environment Canada 1997; Erickson 1997; Valoppi et al. 1998). The general structure of the polychlorinated biphenyl molecule is shown in Figure 3.1 and shows the conventional numbering of substitution positions when chlorine atoms replace hydrogen atoms. The chemical formula is expressed as $C_{12}H_{10-n}Cl_n$, where n = 1 to 10 and represents the number of chlorine atoms in the molecule (Environment Canada 1997; Erickson 1997).

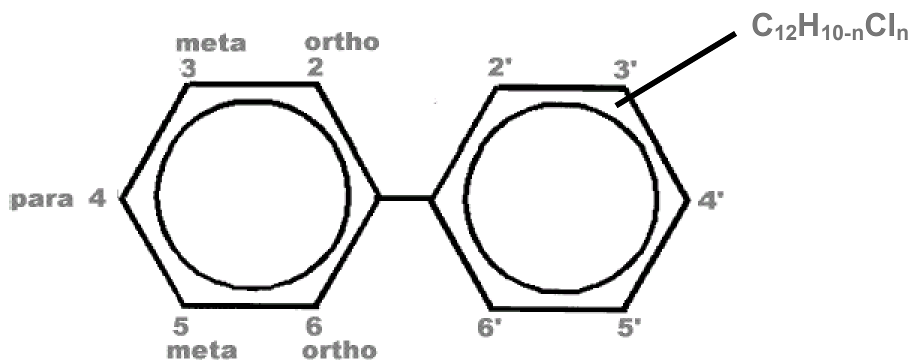


**FIGURE 3.1.** General molecular structure and nomenclature of polychlorinated biphenyls (PCB). The ten numbers indicate possible positions where hydrogen atoms are replaced by chlorine atoms. Modified from Environment Canada (1997).

The system of nomenclature adopted by the International Union of Pure and Applied Chemistry (IUPAC) assigned PCB congeners with numbers from 1 to 209. This allocation is in ascending order and is based on chlorine substitution (Environment Canada 1997). PCBs can also be subdivided into *homologs*, which represent the degree of chlorination from mono-chlorobiphenyl through to deca-chlorobiphenyl. Within the same homolog, compounds have the same number of chlorines (i.e., chemical formula) but at different sites of substitution and are referred to as *isomers* (Erickson 1997; Valoppi et al. 1998). Pure PCB congeners are colourless and odourless compounds, while commercial mixtures are clear thick liquids (Environment Canada 1997; Erickson 1997).

PCBs were first discovered in the environment in the late 1960s (Environment Canada 1988; Kajiwara et al. 2002). As a result of increasing awareness of their environmental hazards and possible detrimental effects on wildlife and humans (Garrett 1985), a ban on the commercial production of PCBs was implemented in North America in 1977 (IARC Monographs 1977; Erickson 1997; Fiedler and Lau

1998a). The use of PCBs was restricted to completely enclosed systems such as electrical capacitors and transformers. Total global PCB production has been estimated in the order of 1.8 million metric tons, and approximately two-thirds of this has been used in dielectric and hydraulic fluids (Simmonds 1991; Barrie et al. 1992). An estimated 30% of total production was released into the global environment prior to the mid-1980s (Tanabe 1988). The remaining 70% has not been released and is still in use in industrialised countries today (Erickson 1997; Fiedler and Lau 1998a).

The 1977 North American ban on PCB production led to an initial reduction of PCB concentrations in wildlife, with a stabilisation of levels since the mid-1980s (Ross et al. 1996; Muir et al. 1999; Parker and Dasher 1999; Ross 2000). This levelling off suggests that ongoing environmental cycling and continued leakage of PCBs is enough to maintain concentrations at a steady state (Tanabe 1988; AMAP 1997; Hope et al. 1997; Borrell and Reijnders 1999; Wilkening et al. 2000; Ross and Troisi 2001; Aguilar et al. 2002). Although closed systems limit the risk, environmental releases due to spills, improper handling or improper disposal still occur (Erickson 1997). PCBs continue to cause concern as old transformers approach the end of their serviceable lives (~30 years) and the potential exists for significant environmental release in the future (Simmonds 1991; Ross and Troisi 2001).

The persistence of PCBs and their earlier use in open applications, such as inks and copying paper, have led to their ubiquitous presence around the globe (Tanabe et al. 1994a; Erickson 1997; Hope et al. 1997). PCBs are now one of four chemicals that dominate organochlorine concentrations in fish, marine mammals, and seabirds (Muir et al. 1992; Norstrom and Muir 1994; Macdonald and Bewers 1996; AMAP 1997; Muir et al. 1999; Becker 2000). Several studies have found that PCBs are often the predominant contaminant among organochlorines in the blubber of marine mammals making them an ecotoxicological concern to these species (Tanabe et al. 1994a; Tanabe et al. 1994b; Muir et al. 1999; Becker 2000; Kajiwara et al. 2002).

PCBs have been associated with several population effects for marine mammals, particularly in the Baltic Sea (Koschinski 2002), the St. Lawrence River (Hickie et al. 2000a), and for European harbour seals (Van Loveren et al. 2000) and European otters (Kannan et al. 2000). Toxic effects on marine mammals include immunosuppression, developmental abnormalities, carcinogenicity, endocrine disruption, reproductive impairment, neurotoxicity, skin disorders, tumours, and lipid degeneration (de Swart et al. 1994; Ross et al. 1996; AMAP 1997; Tjeerdema and Olsen 1998; Kannan et al. 2000). Many of these same effects are also observed in humans (Berner 1999; Rubin and Lanier 1999; Simmonds et al. 2002).

### 3.1.3 PCBs in the Bering Sea

Air masses moving northwards over the Pacific Ocean from heavily industrialised and/or agricultural areas transport contaminants to Arctic and subarctic regions. With their volatile and semi-volatile nature, it is not surprising that PCBs have found their way to the Bering Sea. Although abandoned military stations across the Aleutian Islands are a potential local source of PCB contamination, evidence suggests that PCBs are primarily introduced by long-range transport (Iwata et al. 1994; Parker and Dasher 1999). PCB characteristics that favour this long-range transport include: low water solubility, presence in a gas phase or associated with small particles that have long atmospheric residence times, and high stability (Barrie et al. 1992).

Several studies have confirmed that volatilization and degradation of PCBs is limited by the low temperatures of the Bering Sea region, causing them to settle out and accumulate in the environment and its biota (Tanabe et al. 1994a; Chary 2000; Yao et al. 2001). PCBs also partition strongly onto particles, allowing bottom sediment to act as an important reservoir (Environment Canada 1997; Macdonald et al. 2000). The expansive (up to 500km wide) continental shelf of the Bering Sea ranges from nearshore depths of 50m down to depths of 200m as it nears the shelf break (Hood 1981; Trites et al. 1999a). The shelf is prone to considerable turbulence and tidal mixing that allows contaminated sediment to be resuspended into the water column, particularly during storms (Macdonald et al. 2000). Sediment bound contaminants can also be resuspended by foraging gray whales, walruses, and other biota, as well as by commercial bottom-trawling practices, leaving little or no opportunity for harmful chemicals like PCBs to be buried and prevented from recycling (Dumond 1995; Macdonald et al. 2000).

PCBs are one of the dominant POPs found in marine mammals throughout the Bering Sea (Norstrom and Muir 1994; Macdonald and Bewers 1996; Becker 2000) and may contribute up to 50% of the total organochlorine load in some individuals. Penta- and hexa-chlorobiphenyls (primarily PCBs 153, 138, 149, 118, 101, 99, and 180) dominate PCB patterns in the Bering Sea, accounting for ~70% of total PCBs (Becker et al. 1997; Becker 2000; Minh et al. 2000; Yao et al. 2001).

The hazards for wild animals and humans associated with PCBs make them a major contaminant of concern in the Bering Sea and other regions (AMAP 1997; Ross and Troisi 2001; Tanabe 2002). In the Baltic Sea, for example, environmental contaminants affect the long-term viability of harbour porpoise and may have been a primary cause for the decline of their stocks between the 1940s and 1970s (Koschinski 2002). PCBs and other environmental contaminants could have also been a factor in the decline of marine mammals and other species in western Alaska and the Bering Sea (Braham et al. 1980; Miles et al. 1992; Barron et al. 2003; NRC 2003). These problems are both immediate and long term, creating numerous challenges for current, ongoing and future resource management strategies. Many stakeholders, including marine researchers, conservationists, local Alaska native (i.e., Aleut and Yup'ik) communities, resource managers and the general public, continue to be concerned about the reduced viability and ongoing decline of marine mammal, bird and other species' populations in the eastern Bering Sea. They are also concerned that the traditional foods upon which many Alaska Natives subsist may be contaminated.

## 3.2 METHODS

Ecosystem models can be used to predict the properties of a system that are often too difficult or too costly to measure directly (Hall and Day 1977; Keen and Spain 1992). In the context of contaminants, it would be very time consuming and costly to measure concentrations in every component of a marine ecosystem. Incorporating contaminant data into the eastern Bering Sea ecosystem models enables concentrations to be estimated for species and functional groups that have not previously been measured, as well as providing possible temporal trends of concentrations found or expected in biota.

### 3.2.1 Contaminant modelling using Ecotracer

The Ecopath with Ecosim software includes a routine termed Ecotracer (Christensen and Walters 2004) that is capable of tracing the flow of a single contaminant type, or tracer, moving through a food web with biomass. Ecotracer runs a parallel simulation in Ecosim, calculating contaminant concentrations as the biomass equations are solved and then plotting them over time. The linear dynamic equations required for the flow of a contaminant depend on the biomass flow rates in Ecosim (Christensen and Walters 2004). The Ecotracer routine currently assumes that lethal levels do not occur. A more detailed description of Ecotracer and the dynamic equations behind it are contained in Christensen and Walters (2004).

Within Ecotracer, a contaminant is considered to be either in the environment (water) or in the biota of functional groups at any moment (Christensen and Walters 2004). However, there are a number of components contributing to and reducing the concentration in any given functional group. Three inputs are accounted for in Ecotracer that are considered to contribute to the contaminant concentration in biomass: 1) direct absorption from the environment, 2) concentration in immigrating biomass, and 3) uptake from food (Figure 3.2). There are also four outputs reducing the concentration in biomass: 1) decay or metabolism rate, 2) detritus (death), 3) emigration, and 4) predation (Figure 3.2). Of these seven components, up to four can be entered by the user for each functional group. The other parameters are calculated dynamically by the Ecosim software. The initial data inputs for each functional group may include some or all of the following parameters: initial concentration, direct absorption rate, concentration in immigrating biomass, and decay or metabolism rate.

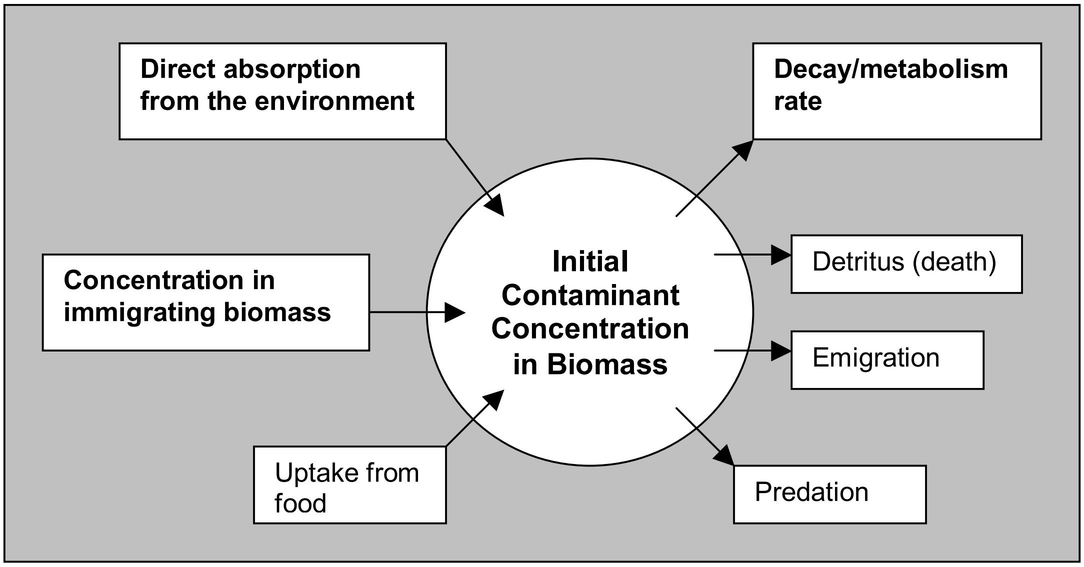


**FIGURE 3.2.** A flowchart illustrating the process that determines the level of contamination in the biomass of a functional group using Ecotracer. There are three inputs considered to contribute to this concentration and four outputs taking away from it. Those in **bold** font represent input required from the user for each functional group defined in the model and are further explained in the text. Modified from Christensen and Walters (2004).

Three additional inputs not represented in Figure 3.2 are 'initial environmental concentration', 'base inflow rate to the environment ($H_20$)', and 'base volume exchange loss from the environment'. While Ecosim runs the model over time, the Ecotracer routine uses all of these inputs to calculate the concentration of a contaminant in the biomass of each functional group.

Ecotracer works in relative units, which means that every concentration is measured relative to the input concentration and depends primarily on relative rather than absolute biomass flows (as well as changes within these). Accordingly, Ecotracer simulations were run at equilibrium in Ecosim.

*3.2.1.1 Ecotracer inputs*

Relevant literature and findings from recent research, long-term studies and monitoring programs were reviewed. This provided PCB concentrations found in biota throughout the eastern Bering Sea, as well as concentrations in important environmental compartments (e.g., water, sediment), for incorporation into the refined models of the region.

Any scaling may be applied in Ecotracer, provided unit consistency is maintained throughout. Prior to entering data into Ecotracer it is necessary to first consider what is sought from it. There are two output options for Ecotracer results: either (i) as concentrations of contaminant per unit area (e.g., tons·$km^{-2}$) or (ii) as concentrations of contaminant per unit biomass (e.g., kg of contaminant/ton of biomass). In the case of PCBs, it is most useful to consider concentrations in terms of concentrations per unit biomass in units of μg·$g^{-1}$ or parts per million (ppm). Accordingly, input concentrations in the present study were set up in units of grams (g) so that the output in concentrations per unit biomass would be g·$ton^{-1}$ or ppm.

Input is required from the user for four parameters (as listed above in Section 3.2.1 and discussed in detail below) for each functional group represented in the ecosystem model. All parameters used in the models are provided in Tables 3.1 and 3.2.

**Initial concentration**

Within Ecotracer, the initial concentration in biomass is the concentration of total PCBs in a particular functional group at the time of the model (i.e., 1950s or 1980s).

*1980s model*

Total PCB concentrations in the biotic and abiotic components of the 1980s eastern Bering Sea ecosystem were collected from published literature. Because PCB concentrations can be reported in a number of different ways, and in order to facilitate comparison, all data collected was standardised into parts per million (ppm) wet weight. As most functional groups in the model represented several species, averages weighted on the abundance of species' in a given group were calculated and used for initial concentrations. Further calculations were made to convert standardised PCB concentrations from the literature into units consistent with the Ecotracer routine (g·$km^{-2}$ for initial concentration). This involved multiplying the weighted mean concentration by the biomass of the functional group. In the few cases where a date of sample or study was not stated, it was assumed to be two years before the date of publication.

In view of the highly lipophilic nature of PCBs, concentrations for marine mammals were taken from blubber samples. Blubber represents the largest lipid reservoir in marine mammals and is where approximately 90% of their PCB load accumulates (Aguilar et al. 1999; Becker 2000). For non-mammals, PCB concentrations in fat tissue were used when available. Muscle tissue is also an important reservoir for PCBs due to its large contribution to body mass (Aguilar et al. 1999). PCB concentrations in this medium were also used where appropriate, particularly for fish.

Data collection for marine mammals was further restricted to consider only those concentrations from adult males. Although age and sex have a great deal to do with contaminant load, there is no age- or sex-structure represented in the eastern Bering Sea models. Given that PCBs are accumulated over time, newborns and juveniles carry a lower contaminant load relative to adults. Furthermore, it is widely recognised that reproductively active females offload a large portion of their contaminant burden to their offspring during gestation (3 – 6%) and particularly during lactation (60 – 95%) (Lee et al. 1996; Aguilar et al. 1999; Ross et al. 2000; Ylitalo et al. 2001; Hoekstra et al. 2002). This maternal transfer of PCBs varies between species and with birth order, but ultimately reduces the contaminant concentration in the adult female (Lee et al. 1996; Aguilar et al. 1999; Ross et al. 2000; Ylitalo et al. 2001). The use of concentrations only found in males does not bias the results to an unusually high worst-case scenario because studies have shown that females begin to re-accumulate PCBs once they reach senescence and reproduction ceases (Ross et al. 2000; Ylitalo et al. 2001; Tilbury et al. 2002; Barron et al. 2003).

PCB concentrations from stranded marine mammals were also not considered. Decreased blubber stores together with volatilization of PCBs from a decaying carcass may cause changes in concentrations. Concentrations in stranded animals tend to be higher than those found in live animals (Ross et al. 1996; Ross and Troisi 2001). Inclusion of these concentrations would skew the data obtained from free-ranging individuals (Ross et al. 1996; Ross and Troisi 2001).

Prior to and during the early 1980s, total PCBs tended to be determined as Aroclor equivalents, which may overestimate PCB levels in marine mammal blubber due to the transformation of congeners (Muir et al. 1992; Barron et al. 2003). More recently, total PCBs have been reported as the sum of individual congeners in a sample analysed using improved and more widely recognised gas chromatography methods. Since these methods are more accurate and sensitive than earlier Aroclor-based methods (Valoppi et al. 1998), studies that reported PCB concentrations in terms of Aroclor equivalents were not used.

Studies conducted in the neighbouring Beaufort Sea, Chukchi Sea and Gulf of Alaska, where species and trophic levels are similar and comparable to those in the Bering Sea, were also a source of data for data-poor species. In the absence of published data, PCB concentrations were estimated based on the diet composition in the model.

*1950s model*

Initial concentrations for biotic functional groups in the 1950s model were left at zero for a number of reasons. The first was because analysis of sediment cores from lakes in Alaska, as well as cores from the Gulf of Alaska (Iwata et al. 1994), indicated that the onset of significant PCB input occurred in sediment layers deposited post-1950 (Muir et al. 1996; AMAP 1997). Secondly, Anas and Wilson (1970) found only trace amounts of PCBs in nursing fur seal pups, which is consistent with post-1950 input of PCBs. Thirdly, because PCB concentrations were often determined as Aroclor equivalents prior to the 1980s, the few data sources available from that time period may have overestimated observed levels. And finally, because contaminant data for this time period was generally lacking for the majority of species in the eastern Bering Sea. Accordingly, predicted PCB concentrations using the 1950s model depended on bottom-up processes.

**Concentration in immigrating biomass**

The concentration in immigrating biomass refers to the contaminant load, measured in parts per million (g PCB/t biomass) that is present in individuals entering or returning to the Bering Sea. Migrating species that feed outside of the modelled area may accumulate PCBs in other regions and bring them into or back into the eastern Bering Sea system. While this input can be captured as the concentration in immigrating biomass, the current models do not incorporate species migrations to and from the Bering Sea. However, this process of incorporating the concentrations found in immigrating biomass may overestimate PCB levels in migrating species, particularly baleen whales that typically do not feed during their migration. Concentrations obtained from migrating baleen whales while at lower latitudes are markedly higher, likely because of their reduced lipid stores. Incorporating such high concentrations would skew the predicted levels above those that are actually observed in Bering Sea biota. Accordingly, this parameter was left at zero for all functional groups in both models.

**Direct absorption**

The direct absorption from the environment (water) is the concentration of PCBs taken up directly from the water. Fish and other gill-breathing species can be expected to accumulate (bioconcentrate) PCBs directly from the water, whereas marine mammals do not because they breathe at the surface. Contaminants are biomagnified by marine mammals through the prey they ingest.

It is very difficult to estimate the bioconcentration of highly lipophilic compounds (such as PCBs) for primary producers, and the dynamics of these processes are still not very well known (Carrer et al. 2000). For instance, increasing lipophilicity renders uptake from the phytoplankton cells even more difficult to predict (Carrer et al. 2000). To ensure that the predictions were correctly scaled for all other functional groups that uptake PCBs via trophic flows from primary production, the direct absorption rate by primary producers was adjusted in Ecotracer to achieve concentrations in ppm that matched the field data. Direct absorption rates were estimated in this manner using the 1980s model, for which better field data existed. These rates, with minor adjustments, were then incorporated into the 1950s model.

**TABLE 3.1.** Ecotracer parameters used in the 1980s eastern Bering Sea model. Dash indicates a parameter not applicable to a given group; while blank cells are those for which no value was entered.

| Base inflow rate ($g \cdot km^{-2} \cdot year^{-1}$) to environmental concentrations: | | | 6.3[1] |
|---|---|---|---|
| Base volume exchange loss ($year^{-1}$) from env. concentrations: | | | 0.0 |
| Functional group | Initial concentration ($g \cdot km^{-2}$) | Direct absorption rate ($g \cdot t^{-1} \cdot g^{-1} \cdot year^{-1}$) | Decay rate ($year^{-1}$) |
| 1. Environment | 65.0000[a] | - | |
| 2. Baleen whales | 0.5071[b] | | 0.150 |
| 3. Toothed whales | 0.1013[c] | | 0.035 |
| 4. Sperm whales | 0.8819[d] | | |
| 5. Beaked whales | 0.0051[e] | | 0.010 |
| 6. Walrus & bearded seals | 0.0138[f] | | 0.400 |
| 7. Seals | 0.0704[g] | | 0.600 |
| 8. Steller sea lions | 0.2390[h] | | 0.070 |
| 9. Piscivorous birds | 0.0051[i] | | 2.000 |
| 10. Adult pollock | 0.4345[j] | | |
| 11. Juvenile pollock | 0.0540[k] | | |
| 12. Other demersal fish | 0.0640[l] | | 9.000 |
| 13. Large flatfish | 0.7980[m] | | |
| 14. Small flatfish | 2.1794[n] | | 0.100 |
| 15. Pacific herring | 0.0069[o] | | 8.000 |
| 16. Other pelagics | 0.2754[p] | | 9.000 |
| 17. Deepwater fish | 0.0176[q] | | 3.000 |
| 18. Jellyfish | 0.0008[r] | | |
| 19. Cephalopods | 0.0578[s] | | 9.000 |
| 20. Benthic particulate feeders | 0.0875[t] | | 5.000 |
| 21. Infauna | 0.1349[u] | | |
| 22. Epifauna | 0.0117[v] | | 9.000 |
| 23. Large zooplankton | 1.4168[w] | 0.002 | |
| 24. Herbivorous zooplankton | 0.9075[x] | 0.001 | |
| 25. Phytoplankton | 0.0960[y] | 0.002 | |
| 26. Discards | | | 0.500 |
| 27. Detritus | 1.0500[z] | | 9.000 |

[1] From Iwata et al. (1994); [a] From Tysban (1999); [b] Weighted average from Chary (2000), Borrell (1993), Aono et al. (1997), Tilbury et al. (2002), and Becker (2000); [c] Weighted average from Muir et al. (1992), Jarman et al. (1996b), Ross et al. (2000), Tanabe et al. (1988), Minh et al. (2000), and Aguilar et al. (2002); [d] From Borrell (1993); [e] From Subramanian et al. (1988); [f] Weighted average from Norstrom and Muir (1994) and Becker (2000); [g] Weighted average from Norstrom and Muir (1994), Miles et al. (1992), Tanabe et al. (1988), and Muir et al. (1992); [h] From Lee et al. (1996); [i] Weighted average from Muir et al. (1999) and Kawano et al. (1986); [j] From Kawano et al. (1986); [k] Based on diet composition; [l] Weighted average from Tanabe et al. (1988) and Bright et al. (1995); [m] From Tanabe et al. (1988); [n] Weighted average from Chary (2000) and Valette-Silver et al. (1999); [o] From Aono et al. (1997); [p] Weighted average from Aono et al. (1997), O'Shea and Brownell (1994), Kawano et al. (1986), and Tanabe et al. (1988); [q] Based on diet composition; [r] Based on diet composition; [s] Average from Kawano et al. (1986) and Tanabe et al. (1988); [t] Based on diet composition; confirmed with observed concentrations in the White Sea (Muir et al. 2003); [u] From Muir et al. (1999); [v] From Livingston (2000); [w] Weighted average from Muir et al. (2003) and Norstrom (1994); [x] From Norstrom (1994); [y] From Rice et al. (1992); [z] From Valette-Silver et al. (1999).

**Decay/metabolism rate**

The decay rate is the rate at which a group can metabolise or depurate PCBs and thereby reduce their contaminant load. This is difficult to assess because the capacity to metabolise contaminants varies between species and even between individuals within a species, although in general marine animals have a low capacity for metabolism of contaminants (Tanabe et al. 1988; Boon et al. 1992; Tanabe et al. 1994a; Boon et al. 1997; Van den Berg et al. 1998; Becker 2000; Miao et al. 2000; Hoekstra et al. 2002). Assessment is further complicated by the fact that most functional groups in the models represent several species (e.g., 7 species of baleen whales; 4 toothed whales; 5 seals etc.). Also, different congeners representing a variation in chlorination are degraded at different rates (Fiedler and Lau 1998b). Generally, the persistence of PCBs increases with increasing chlorination (Erickson 1997; Fiedler and Lau 1998a, b; Van den Berg et al. 1998) and the penta- and hexa-chlorinated biphenyls most commonly found in Bering Sea biota are resistant to biodegradation (Ross et al. 2000; Hoekstra et al. 2003).

For the higher chlorinated congeners, PCB uptake from water is of minor importance for fish, although the gill is the primary site of depuration (Connolly 1991). Given that marine mammals metabolise PCBs and that fish depurate some PCBs via their gills, and in view of the difficulty of calculating this rate, the metabolism rate parameter was estimated using Ecotracer. Applying the 1980s model, metabolism rates were adjusted in Ecotracer so that predicted concentrations matched the field data. These rates were then incorporated into the 1950s model with minor adjustments.

**TABLE 3.2.** Ecotracer parameters used in the 1950s eastern Bering Sea model. Blank cells are those for which no value was entered.

| | | |
|---|---|---|
| Base inflow rate (g·km$^{-2}$·year$^{-1}$) to environmental concentrations: | | 4.7[1] |
| Base volume exchange loss (year$^{-1}$) from env. concentrations: | | 0.0 |
| Initial concentration (g·km$^{-2}$) in the environment: | | 9.2[a] |
| Functional Group | Direct absorption rate (g·t$^{-1}$·g$^{-1}$·year$^{-1}$) | Decay rate (year$^{-1}$) |
| 1. Baleen whales | | 0.150 |
| 2. Toothed whales | | 0.015 |
| 3. Sperm whales | | |
| 4. Beaked whales | | 0.005 |
| 5. Walrus & bearded seals | | 0.370 |
| 6. Seals | | 0.450 |
| 7. Steller sea lions | | |
| 8. Piscivorous birds | | 0.650 |
| 9. Adult pollock | | |
| 10. Juvenile pollock | | |
| 11. Other demersal fish | | 9.000 |
| 12. Large flatfish | 0.001 | |
| 13. Small flatfish | | 0.0001 |
| 14. Pacific herring | | 8.000 |
| 15. Other pelagics | | 9.000 |
| 16. Deepwater fish | | 1.000 |
| 17. Jellyfish | | 1.000 |
| 18. Cephalopods | | 9.000 |
| 19. Benthic particulate feeders | | 6.000 |
| 20. Infauna | | |
| 21. Epifauna | | 9.000 |
| 22. Large zooplankton | 0.0005 | |
| 23. Herbivorous zooplankton | 0.001 | |
| 24. Phytoplankton | 0.002 | |
| 25. Detritus | | 7.000 |

[1] Derived from sediment core data from Iwata et al. (1994);
[a] From Kawano et al. (1986).

In addition to the input parameters required for each functional group discussed above, there were also three required environmental parameters (as listed above). Values used for these parameters are provided in Tables 3.1 and 3.2.

### Initial concentration in the environment

For both the 1950s and 1980s models, total PCB concentrations in the environment of the eastern Bering Sea were collected from published literature. PCB concentrations obtained from the literature were then converted into units consistent with the Ecotracer routine, these being g·km$^{-2}$ for initial concentration in both biotic and abiotic compartments.

## Base inflow

Atmospheric transport is the major pathway for PCBs reaching the Bering Sea and greatly exceeds those from riverine sources of discharge (Macdonald et al. 2000). For this reason, the base inflow rate was considered to be solely from atmospheric deposition.

### *1980s model*

In high latitudes, where the air and water are colder, fluxes indicate a net air to water mass transfer (Iwata et al. 1993). Iwata et al. (1994) reported a net air to water flux for the Bering Sea of 630 $pg \cdot cm^{-2} \cdot year^{-1}$ in 1990. Atmospheric PCB concentrations have remained constant since the 1980s (Iwata et al. 1994), as have concentrations found in biotic compartments (Ross et al. 1996; Muir et al. 1999; Parker and Dasher 1999; Ross 2000). The inflow of PCBs to the Bering Sea was therefore assumed to have remained stable through the 1980s and 1990s. The air to water flux calculated by Iwata et al. (1994) was converted into units consistent with Ecotracer ($g \cdot km^{-2} \cdot year^{-1}$) and used as the single inflow rate in the 1980s model. This inflow rate continued for the duration of the simulation.

### *1950s model*

Modelling the flow of PCBs with Ecotracer also provides a means of examining past levels. In an effort to explain the sometimes sudden and large population fluctuations that occurred in the region between the 1950s and the 1980s, PCB input to the Bering Sea, based on sediment core analyses (Iwata et al. 1994), was incorporated into the 1950s model to estimate PCB concentrations throughout the food web. Time series data read into the model for the base inflow rate incorporated the changing PCB inputs over time, reflecting the increasing rate of global PCB production until restrictions were implemented in the mid-1970s.

Gulf of Alaska sediment core data from Iwata et al. (1994) was used to calculate the corresponding inflow rate to the Bering Sea. Iwata et al. (1994) sampled sediment cores during June-July 1990 in the Gulf of Alaska and reported on the temporal trends of organochlorine accumulation rates into sediments. Gulf of Alaska sediment cores were used in lieu of Bering Sea cores because Bering Sea cores showed no correlation between residues and sediment depth (Iwata et al. 1994). Sediment disturbance in the eastern Bering Sea can be attributed to factors such as commercial trawling activities, foraging gray whales, and natural turbulence and tidal mixing, which may affect PCB accumulation rates (Iwata et al. 1994). The PCB accumulation rates into sediments established for the Gulf of Alaska represents approximately 2.5% of the total atmospheric input (Iwata et al. 1994). Iwata et al. (1994) further reported that the sedimentation rate in the Gulf of Alaska was at least twice as high as that in the Bering Sea. Considering the above, inflow rates for the eastern Bering Sea based on Gulf of Alaska sediment cores were calculated assuming that the PCB accumulation rates represented 1% of the total atmospheric input.

Time trends were incorporated into Ecotracer in the 1950s model to account for the changes in PCB input over time, otherwise the rate entered for the base inflow would have continued for as many years as the simulation was run. This time trend component is represented as the "Forcing function number for environmental inflow rate" on the Ecotracer entry form.

**Base volume exchange loss**

The base volume exchange loss parameter was incorporated into the base inflow rate described above. In both models, the base inflow rate represents the net inflow of PCBs and therefore the loss from the environment was left at zero.

With the parameters entered into the Ecotracer entry form, and the routine enabled, an Ecosim simulation was run for 100 years in each of the two models to predict PCB concentrations.

## 3.3 RESULTS

### 3.3.1 Ecotracer outputs

There are two output options for Ecotracer results: as concentrations of the tracer per unit area (e.g., g of tracer/$km^2$) or as concentrations per biomass (e.g., g of tracer/ton of biomass). Once the simulation in Ecosim is complete, Ecotracer plots the predicted concentration of PCBs over time.

*1980s model*

In the 1980s model, Ecotracer was used to forecast future PCB concentrations in the eastern Bering Sea. Ecotracer simulated the accumulation of PCBs over time based on the inflow rate and initial concentrations of the 1980s. Results from this model indicated that concentrations reached their peak in the mid-1980s, followed by a slow decline to a steady state (Figure 3.3). This trend is similar for all functional groups. For functional groups with available field data, predicted PCB concentrations were compared with weighted averages of observed values in order to validate the model predictions (Figure 3.4). Although comparison to observed values was limited by a lack of available time series data for PCB concentrations in Bering Sea biota, the model yielded reasonable predictions as to the concentrations currently found in functional groups. Model results were intended to fall within a range of observed values, rather than predict exact concentrations in species.

In the event that sources other than atmospheric deposition (e.g., abandoned military sites) become a major input of PCBs to the Bering Sea, the total inflow rate would be expected to increase. Incorporating a time series representing a 1% increase in PCB input each year from 2001 tested the possible effect of increased introduction of PCBs into the marine food web of the Bering Sea. The model clearly indicates that an increase in inflow would also result in an increase in concentrations in animal tissue (Figure 3.5). There is, however, a 5 to 20 year delay before concentration changes are observed in upper trophic level animals, suggesting that a lengthy lag time exists before changes would be noticed.

In contrast, the effect of decreased PCB inflow was tested by incorporating a time series representing a 1% decrease in PCB input each year from 2001. In this case, the results were declining concentrations in animal tissue (Figure 3.6). These results indicated that further worldwide restrictions on the use of PCBs, together with appropriate clean-up measures that result in decreased input levels, would decrease the exposure of Bering Sea biota and humans to these harmful chemicals. The model also suggests that concentrations would continue their declining trend with little or no lag time or delay.

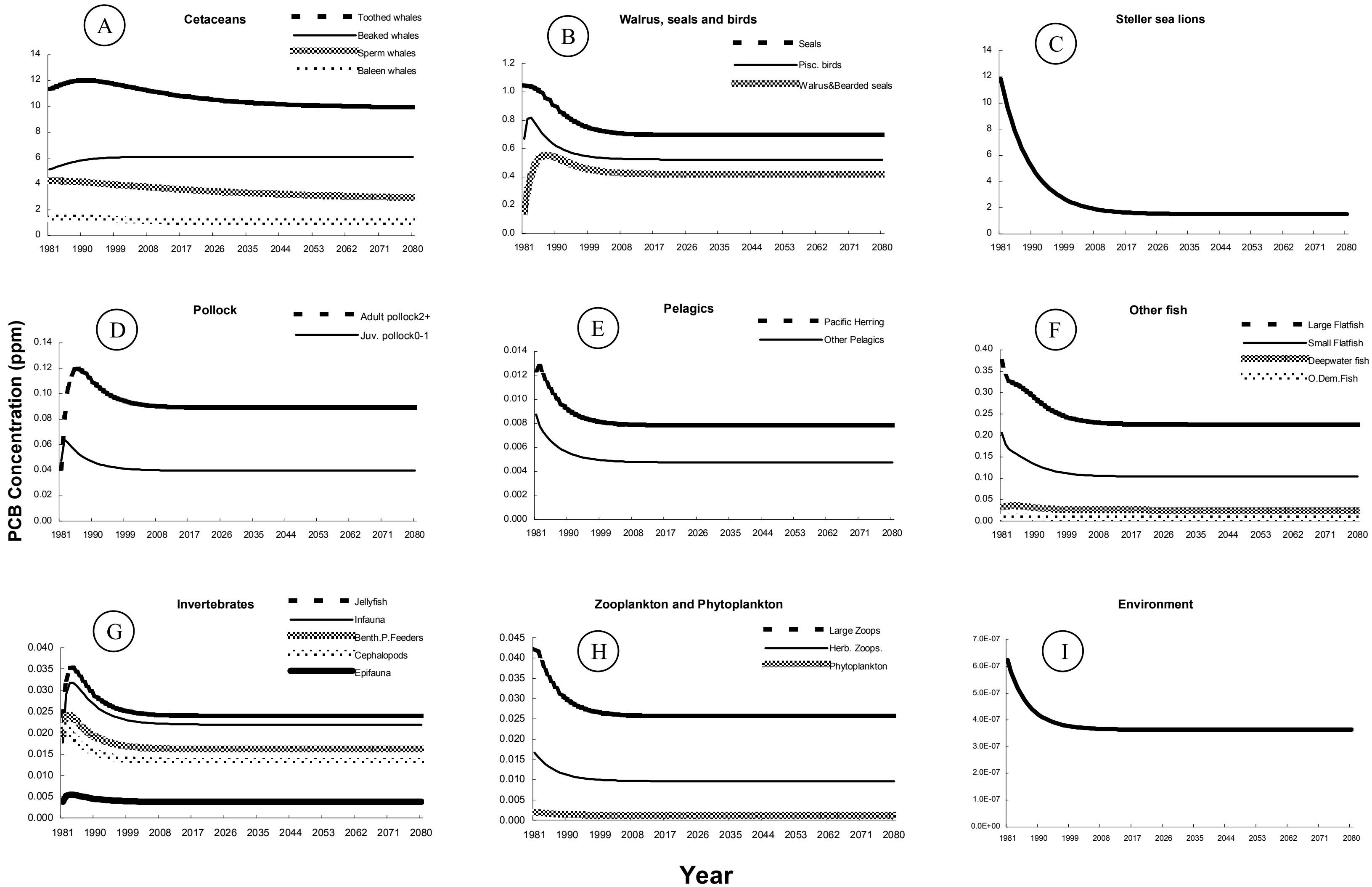


**FIGURE 3.3.** Predicted total PCB concentrations for all functional groups from the 1980s model. The Ecotracer simulation was run for 100 years and the model assumes a constant inflow rate. Note that the Y-axes vary between panels.

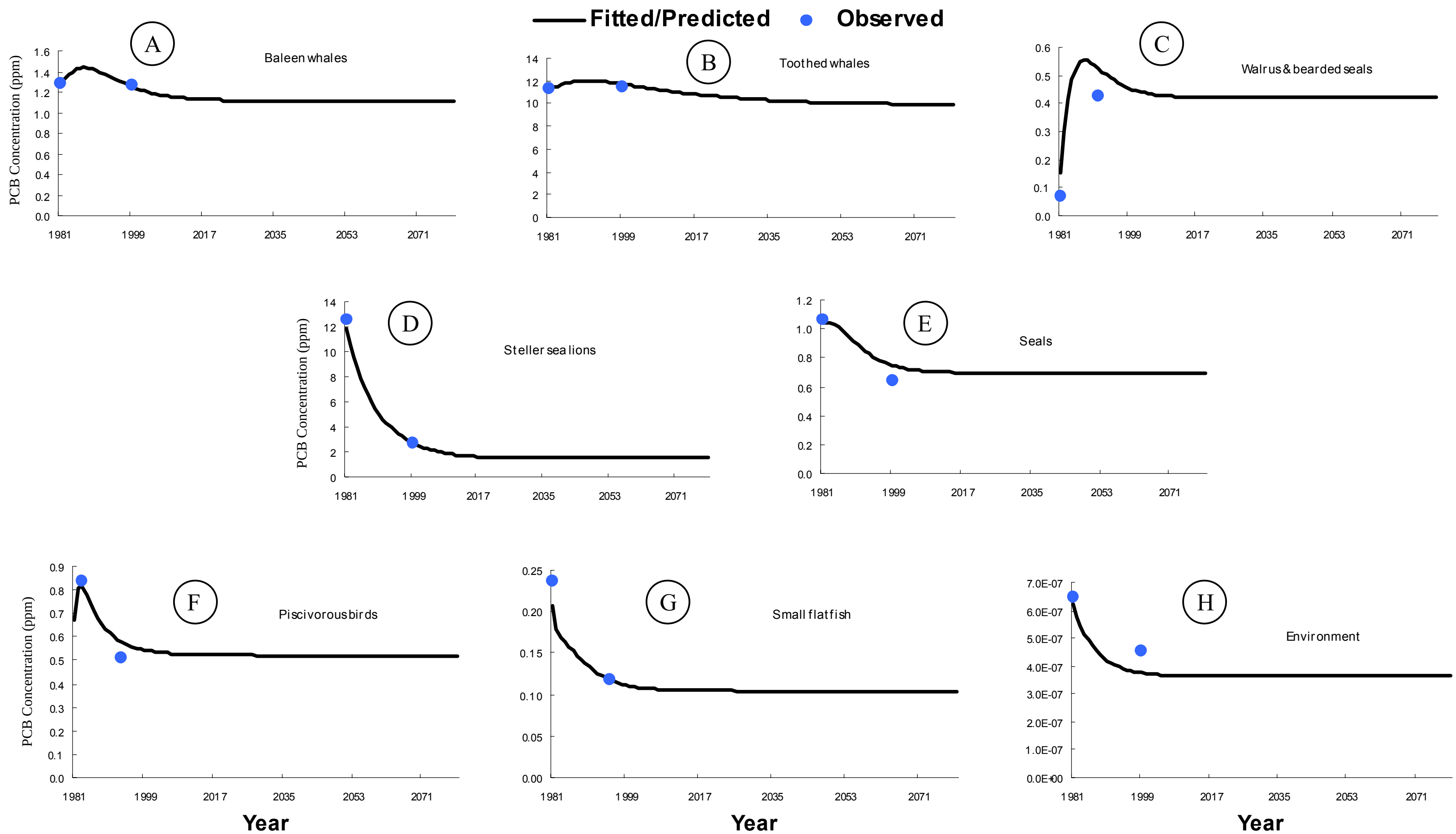


**FIGURE 3.4.** Observed PCB concentrations (dots) for 7 functional groups and the environment compared to Ecotracer fitted/predicted concentrations (line) from the 1980s model. Sources for observed concentrations: A (Borrell 1993; Aono et al. 1997; Verbrugge 1999; Becker 2000; Chary 2000; Tilbury et al. 2002; Hoekstra et al. 2003); B (Tanabe et al. 1988; Jarman et al. 1996b; Chary 2000; Minh et al. 2000; Ross et al. 2000; Ylitalo et al. 2001; Aguilar et al. 2002; Kajiwara et al. 2002); C (Norstrom and Muir 1994; Krahn et al. 1997; Becker 2000; Seagars and Garlich-Miller 2001); D (Lee et al. 1996; Chary 2000); E (Tanabe et al. 1988; Miles et al. 1992; Muir et al. 1992; Norstrom and Muir 1994; Verbrugge 1999; Hoekstra et al. 2003); F (Kawano et al. 1986; Muir et al. 1999); G (Chary 2000); H (Tsyban 1999; Yao et al. 2001).

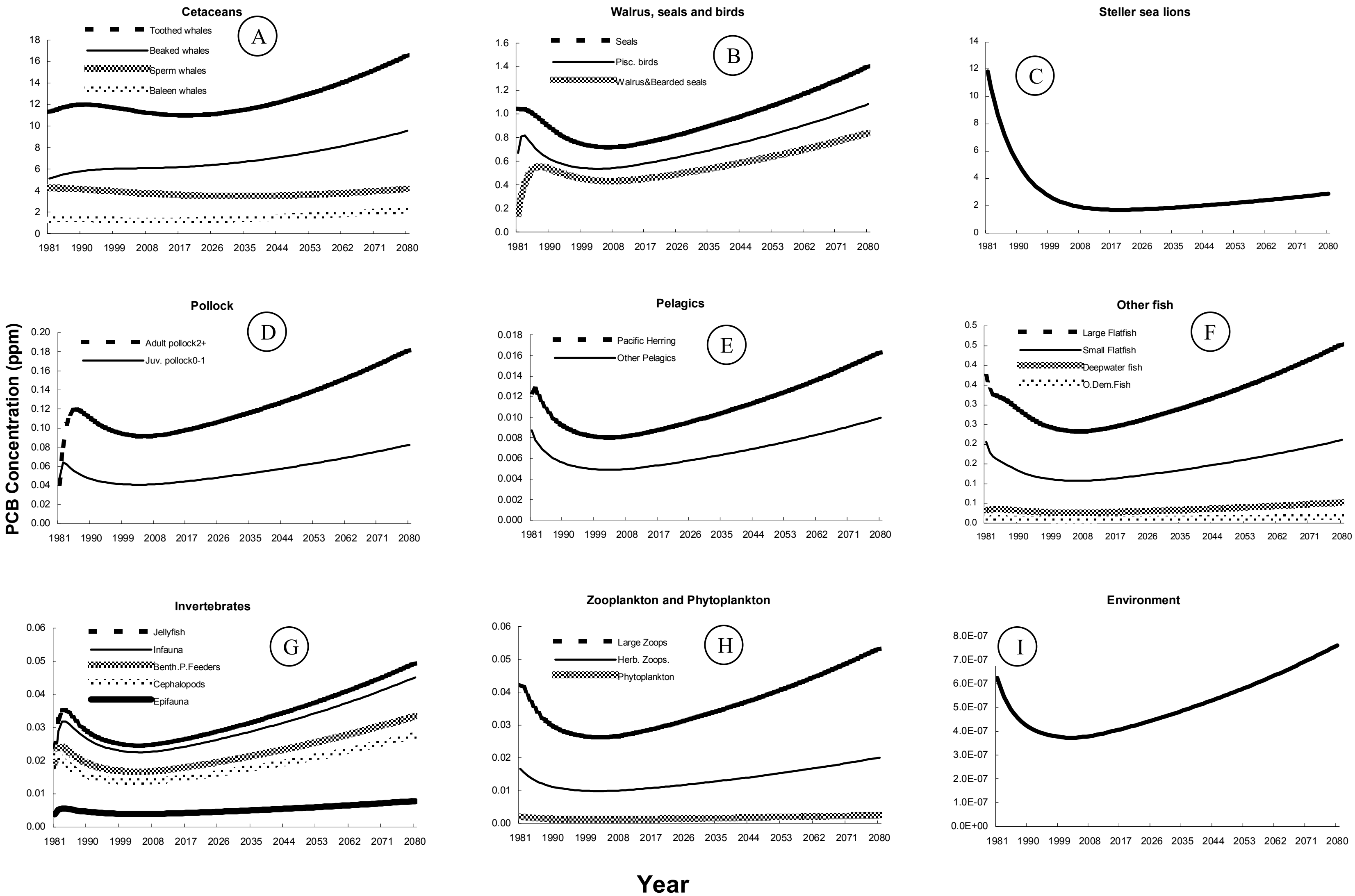


**FIGURE 3.5.** Predicted total PCB concentrations from the 1980s model with increasing inflow to the Bering Sea. The model assumes a 1% increase in inflow each year from the year 2001 onwards.

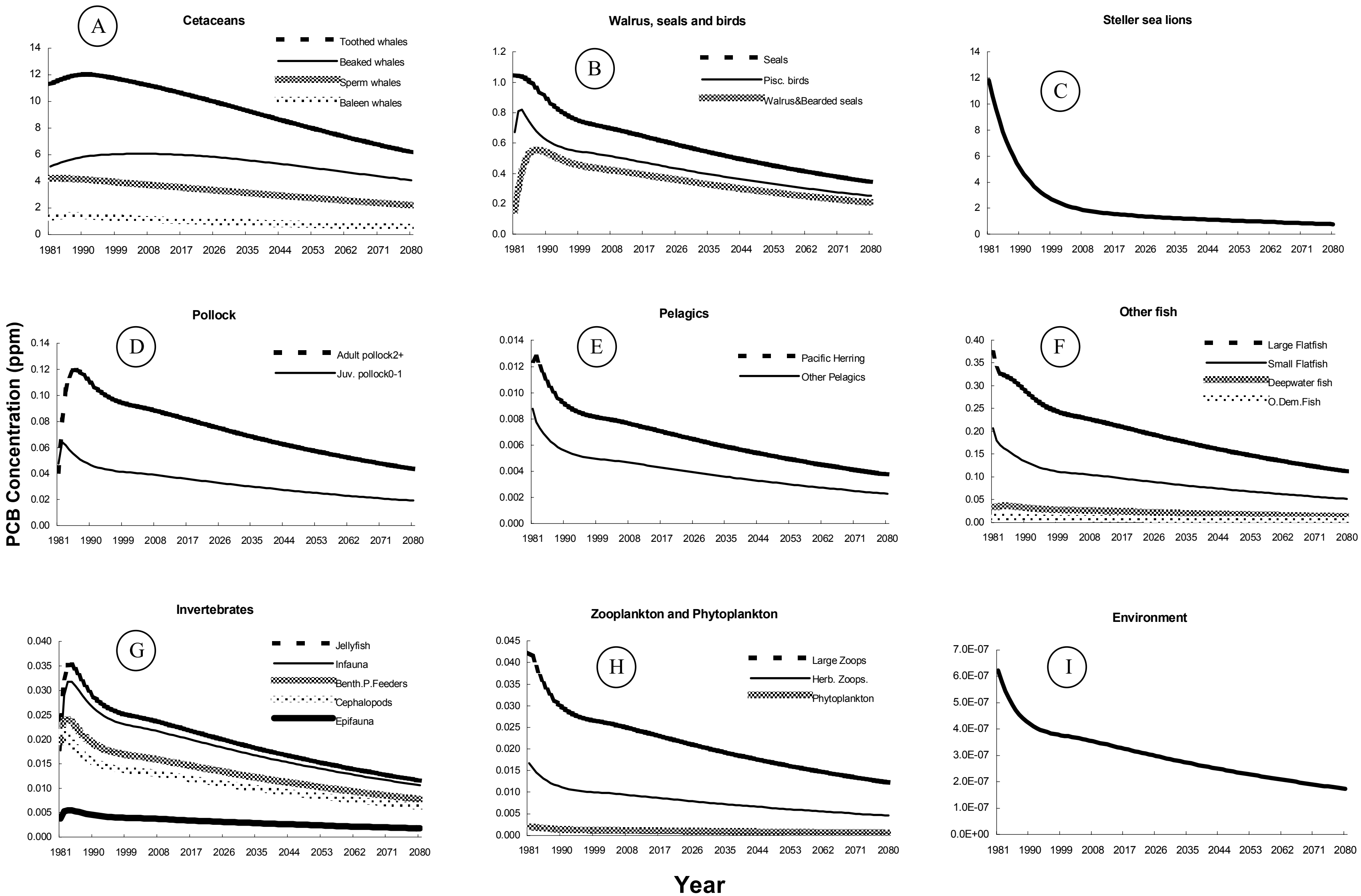


**FIGURE 3.6.** Predicted total PCB concentrations from the 1980s model with decreasing inflow to the Bering Sea. The model assumes a 1% decrease in inflow each year from the year 2001 onwards.

*1950s model*

Once confident that the parameters for the 1980s model were reasonable and that the model accurately predicted PCB concentrations in the eastern Bering Sea food web, equivalent simulations were conducted with the 1950s model. Using the 1950s model, Ecotracer provided historical insights regarding PCB concentrations in the eastern Bering Sea by simulating the accumulation of PCBs over time. Data from sediment core profiles described the increasing rate of the manufacture of PCBs until their ban in the mid-1970s and allowed Ecotracer to account for the corresponding changes in inflow rate. Results from the 1950s model indicated that concentrations reached their peak in the mid-1980s, slowly declined in a delayed response to restrictions, and then reached a steady state (Figure 3.7). This trend is similar for all functional groups. Output from the 1950s model also provided an estimate of PCB concentrations for species and functional groups that have not previously been measured or for which data is sparse.

Predicted PCB concentrations were also compared with available observed values to validate the model's predictions (Figure 3.8). As with the 1980s model, this comparison was limited by a lack of available time series data for PCB concentrations in Bering Sea biota. Predicted concentrations were reasonable in most cases, although the model failed to explain the high concentration observed in Steller sea lions. There are two possible reasons for this: 1) either the observed value in the late-1970s overestimated the actual concentration, or 2) the model failed to capture the dynamics of total PCB concentrations in Steller sea lions. Lee et al. (1996) reported an average total PCB concentration for blubber samples collected from Steller sea lions between 1978 and 1981 of 12.6 ppm wet weight (shown in panel F of Figure 3.8). More recently observed concentrations are ~80% lower than this figure, only 15 to 20 years later. Although PCB concentrations have declined since the mid-1980s, such a drastic and rapid decline is not typical and would not be expected. Given this, and noting that model predictions fit well with observed values in other functional groups, it seems likely that the PCB levels reported by Lee et al. (1996) may overestimate actual concentrations for Steller sea lions.

Evidence suggests that even low-level chronic exposure to PCBs can have sub-lethal effects in marine mammals from the individual to population level (de Swart et al. 1994; Jenssen et al. 1995; Ross et al. 1996; Seagars and Garlich-Miller 2001; Gaydos et al. 2004). Accordingly, a number of authors have established benchmark or threshold concentration levels in the blubber of marine mammals above which health effects would be expected (e.g., Helle et al. 1976a, b; Blomkvist et al. 1992; Ross et al. 1996; Kannan et al. 2000; Ylitalo et al. 2001; Berrow et al. 2002; Barron et al. 2003). Predicted PCB concentrations from the 1950s model were compared to established threshold levels in order to consider the possible impacts on marine mammals occupying the upper trophic levels and to assess whether contamination may have been a factor in the decline of top predators in the region. Given that the

Ecotracer routine currently operates at only a non-lethal level, it is not possible to incorporate threshold levels other than by comparing them to predicted concentrations. The model suggests that, with the exception of toothed whales, concentrations in marine mammals remained from 23% (beaked whales) to 90% (walrus) lower than threshold levels (above which health effects are expected) throughout the time period between the 1950s and the present (Figure 3.9).

Toothed whales was the only functional group that was within the range of threshold levels from 7 to 15 $mg \cdot kg^{-1}$ wet weight established by Kannan et al. (2000). As Kannan's threshold level was originally reported in a lipid weight concentration, this was adjusted to a possible range of wet weight concentrations (using the % lipid in adult males) in order to facilitate comparison (Figure 3.9). The formula used was: $W = L \cdot \frac{\%lipid}{100}$, where W is the wet weight concentration, L is the lipid weight concentration, and % lipid is the percentage lipid in the sample. It should also be noted that the toothed whale functional group includes both transient (mammal-eating) and resident (fish-eating) killer whales. While the PCB concentration in transient whales was likely within or above Kannan’s range of threshold levels, the concentrations in other species within this functional group are probably below this benchmark.

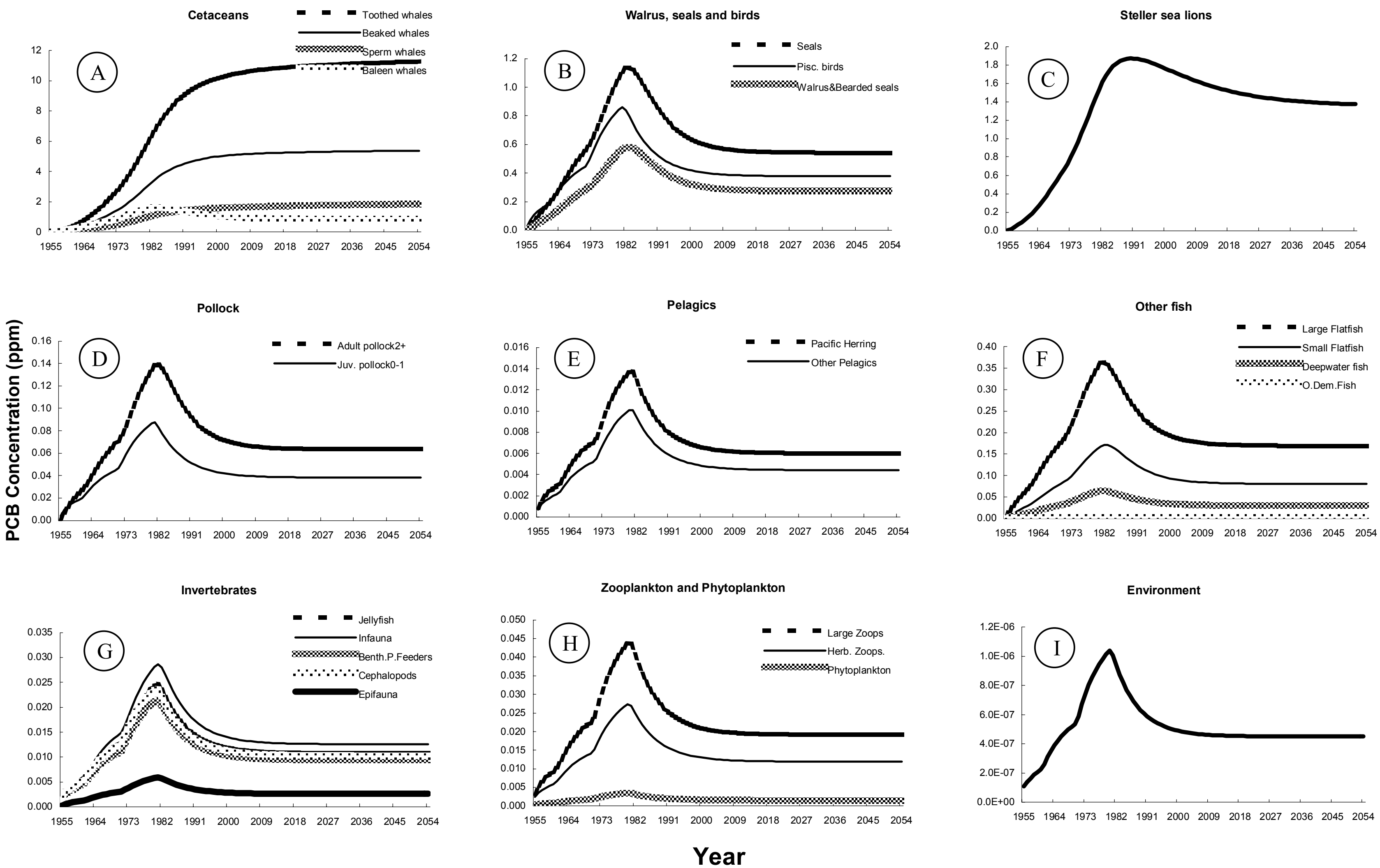


**FIGURE 3.7.** Predicted total PCB concentrations for all functional groups from the 1950s model. The Ecotracer simulation was run for 100 years. The model incorporates the change in inflow rate based on sediment core profiles and assumes a constant inflow rate from 1981 onwards. Note that the Y-axes vary between panels.

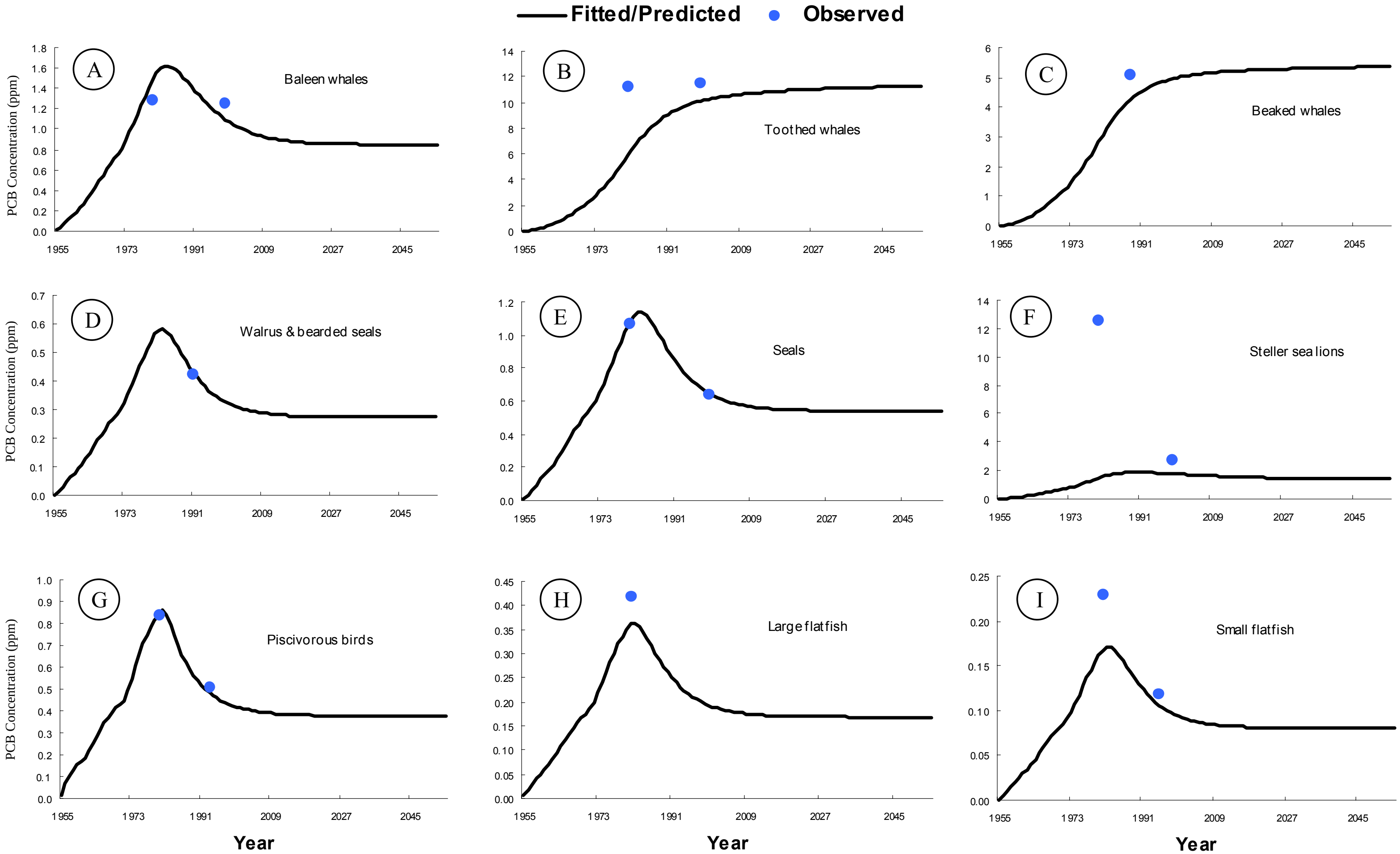


**FIGURE 3.8.** Observed PCB concentrations (dots) for 9 functional groups compared to Ecotracer fitted/predicted concentrations (line) from the 1950s model. Sources for observed concentrations are listed in Figure 3.4.

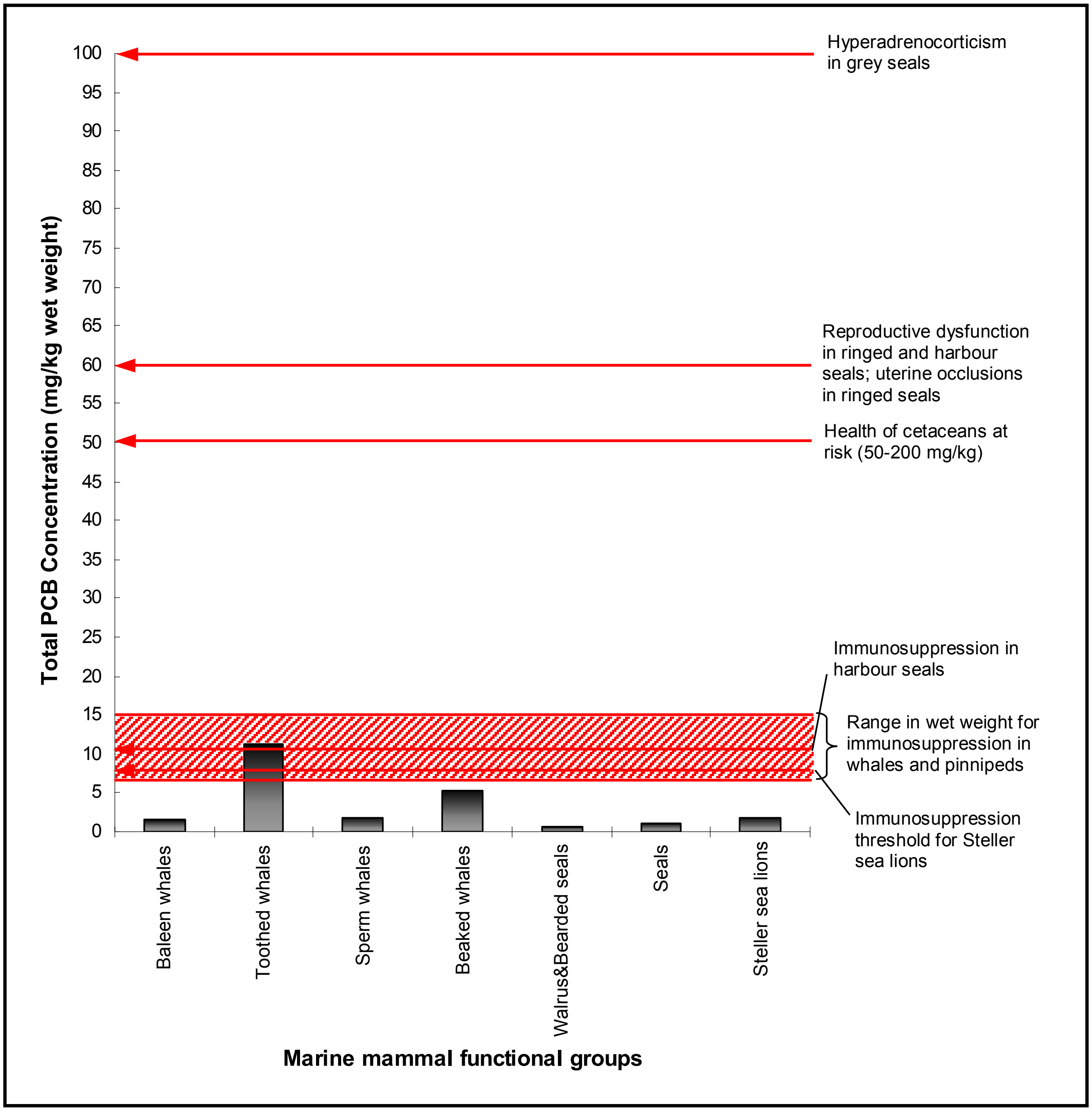


**FIGURE 3.9.** Highest predicted PCB concentrations (with the 1950s model) in marine mammals compared to threshold levels. Where necessary, threshold levels originally reported in a lipid weight concentration were adjusted (using % lipid in adult males) to wet weight concentrations for comparison.

*3.3.1.1 Biomagnification*

Concentrations of the highly chlorinated and hydrophobic PCBs that are routinely found in Bering Sea biota increase with trophic position (Russell et al. 1999). The simulation of PCB accumulation conducted with the 1950s model is consistent with this. The simulation explains the observed concentration distribution in the food web and further verifies that biomagnification is not due to equilibrium partitioning of PCBs between water and lipids. Table 3.3 shows the sum of total PCB concentrations at the end of the 100-year run for each trophic level. Figure 3.10 illustrates the increase in PCB concentration with each step up the food web.

**TABLE 3.3.** Trophic levels (TLs) and corresponding total PCB concentrations show evidence of biomagnification of PCBs within the eastern Bering Sea.

| Trophic Level (TL) | Total PCB Concentration (ppm) | Magnification factor between TLs |
|---|---|---|
| 1 | 0.0015 | - |
| 2 | 0.0556 | 37 |
| 3 | 1.3379 | 24 |
| 4 | 20.978 | 16 |

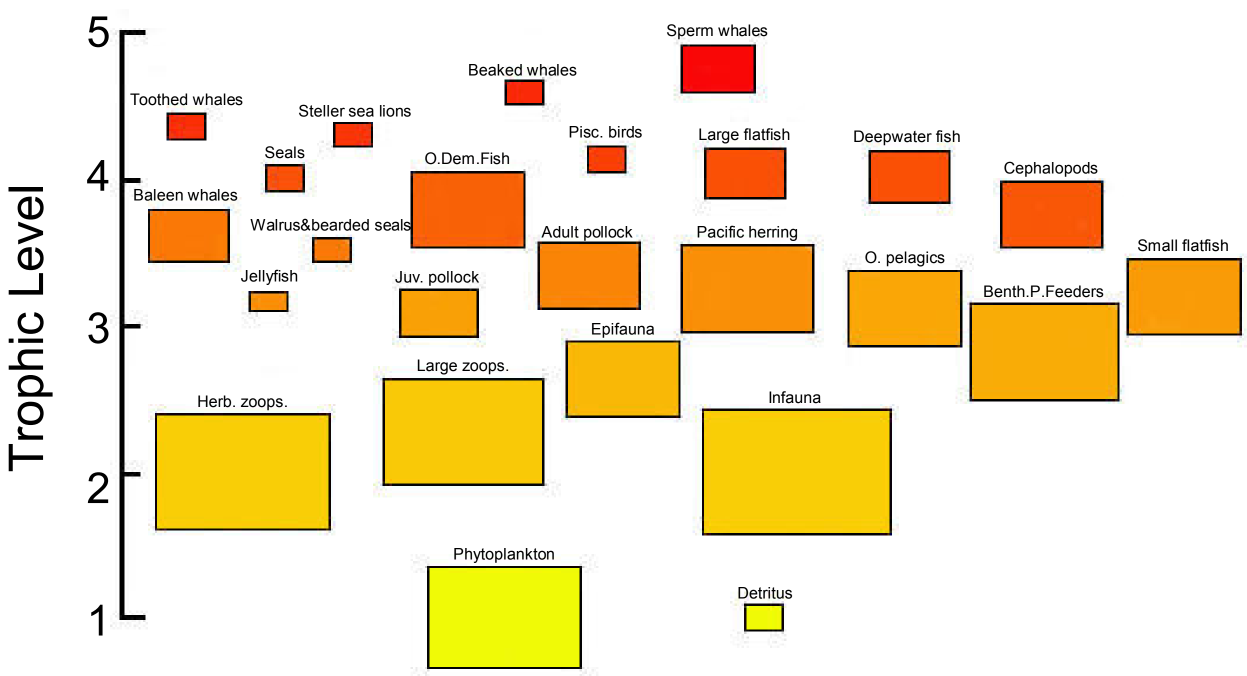


**FIGURE 3.10.** State variables (functional groups) of a model of the eastern Bering Sea in the 1950s. Flows between groups have been omitted for simplicity. For each step up the food web there is a subsequent increase (biomagnification) in PCBs as illustrated by the gradient colour scheme from yellow (TL 1, lowest concentration) to red (TL 4+, highest concentration).

## 3.4 DISCUSSION

The recent addition of Ecotracer to the Ecopath with Ecosim modelling software enables the accumulation of contaminants to be predicted throughout the food web. Incorporating contaminant concentrations extends the refined eastern Bering Sea models and their applications. These models can predict trends in PCB accumulation while other parameters (e.g., inflow rate) are manipulated, as well as provide concentration estimates for species that have not previously been monitored. Modelling the flow of PCBs with Ecotracer also provides a means of examining how past exposure levels, derived from sediment core profiles, contribute to current PCB levels throughout species represented in the food web model (Hickie et al. 2000b).

### 3.4.1 Implications of predicted PCB concentrations

The Bering Sea and Aleutian Islands are remote. Far from heavily industrialised areas, this region would intuitively be thought of as a 'pristine' environment, free from outside disturbances. However, this subarctic ecosystem has undergone comprehensive changes since the 1950s, and is prone to accumulating PCBs both in the environment and in it's biotic compartments. The Bering Sea has historically had lower levels of contamination relative to waters closer to more highly populated and industrialised areas of the world. However, PCB levels in the region rose steadily from the 1950s until the mid-1970s, when restrictions to their production and use were implemented (Livingston and Low 1998). PCB flow into the eastern Bering Sea ecosystem is primarily associated with long-range atmospheric transport from other regions. As a result of the cold climate and the chemical properties of PCBs, the Bering Sea is more sensitive to the threats posed by contaminant input than are comparable warmer regions. The challenge now is to determine what role contaminants, and specifically PCBs, are playing in these observed changes and also how wildlife and Alaska Natives of the region may be affected.

Water quality objectives set by the United States Environmental Protection Agency (USEPA) specify that PCB levels should not exceed 0.03 μg/L (ppb) for marine waters and 0.014 μg/L (ppb) for freshwaters. While Bering Sea seawater concentrations (0.00065 μg/L) are almost two orders of magnitude lower than this guideline, environmental exposure is predominantly via sediment and/or tissue. Accordingly, sediment quality guidelines (SQGs) were developed for use in interpreting chemical concentrations in sediments. SQGs provide estimates of 'safe concentrations' below which effects are unlikely (Effects Range-Low, ERL) as well as concentrations above which adverse effects are more likely (Effects Range-Median, ERM) (NOAA 1999). The sediment quality guideline ERL and ERM values generated for total PCBs were: 22.7 ppb, dry weight and 180 ppb, dry weight, respectively (NOAA 1999). Canadian sediment quality guidelines are similar at 21.5 ppb, dry weight and a probable effects level (PEL) of 189 ppb, dry weight (Canadian Council of Ministers of the Environment 1999). Although observed concentrations in Bering Sea sediments are again two orders of magnitude lower (0.13 ppb, dry weight) than ERL, it is important to note that the guidelines were not derived as toxicity thresholds; they give no assurance that toxicity will not occur below the ERL value (NOAA 1999). Consequently, bioaccumulation through the food web may give rise to serious implications for the ecosystem, top predators, and humans.

The 1977 North American ban on PCB use and production led to an initial reduction in PCB concentrations in wildlife. Given that only an estimated 1% of the total 1.8 million tons of PCBs produced has reached the ocean (Reijnders 1996), this trend is unlikely to continue; degradation will be compensated for by ongoing environmental cycling and new inputs from PCBs currently in non-marine compartments (Tanabe 1988; Barrie et al. 1992; Borrell and Reijnders 1999). Similarly to field studies that have revealed stabilized concentrations since the mid-1980s (Ross et al. 1996; Muir et al. 1999; Parker and Dasher 1999; Ross 2000), the results for the 1980s model also indicate that PCB concentrations have reached a steady state (Figure 3.3). This levelling off suggests that ongoing environmental cycling and continued input of PCBs is enough to maintain concentrations at their current levels (Tanabe 1988; AMAP 1997; Hope et al. 1997; Borrell and Reijnders 1999; Wilkening et al. 2000; Ross and Troisi 2001; Aguilar et al. 2002). On a global scale, noticeable reductions in the exposure of marine mammals to PCBs are not likely to occur for at least several decades (Borrell and Reijnders 1999).

Although PCB concentrations have currently levelled off, the potential for increased input still exists. There are more than 500 military installations, or Formerly Used Defence Sites (FUDS), scattered across Alaska, and many are situated on the coast and along the Aleutian Island chain (Alaska Department of Environmental Conservation 1997; NMFS 2001). PCBs in electrical equipment at these sites are a continuing cause for concern as the equipment reaches the end of it's serviceable life (~30 years) and increases the potential for significant environmental release in the future (Ross and Troisi 2001).

#### *3.4.1.1 Marine Mammal Health*

The breakdown and loss of PCBs is much slower in cold temperature ecosystems such as the Bering Sea. The wildlife of these ecosystems, particularly those at high trophic levels such as marine mammals, tend to have large amounts of insulating fat tissue that readily accumulates and stores PCBs. The body composition and life history patterns of marine mammals make individuals and populations highly susceptible to chemical contamination. Body composition plays a key role in susceptibility because of the lipophilic (fat-soluble) nature of PCBs. PCBs are not easily eliminated once sequestered in the blubber of marine mammals, and species with annual fasting periods run the risk of mobilizing contaminants when they use their fat reserves for energy (e.g., 90% utilisation of lipid reserves in seals) (Aguilar et al. 1999). Cetaceans and pinnipeds are also long-lived animals, which means they accumulate contaminants throughout the 30, 40 or more years of their life.

Seals and cetaceans generally have a low capacity to metabolise PCB compounds and may vary considerably in their ability to do so (Tanabe et al. 1988; Tanabe et al. 1994a; Boon et al. 1997; Becker 2000). Furthermore, the ability to metabolise PCB compounds does not necessarily minimise their effect. In most mammals and birds, PCB compounds are metabolised into non-degradable compounds, which are often more persistent and toxic than their parent compounds (Norstrom and Muir 1994; Fiedler and Lau 1998b; Aguilar et al. 1999).

The toxic effects of PCBs on marine mammals include developmental abnormalities, carcinogenicity, endocrine disruption, reproductive impairment, neurotoxicity, skin disorders, tumours, lipid degeneration, and immunosuppression (de Swart et al. 1994; Ross et al. 1996; AMAP 1997; Tjeerdema and Olsen 1998; Kannan et al. 2000). PCBs are known to cause immunosuppression in a number of marine mammal species (Aguilar and Borrell 1994; de Swart et al. 1994; Luebke et al. 1997). Weakened immune systems may render marine mammals more vulnerable to disease and virus epizootics, as evidenced by morbillivirus in several mass mortality events (Simmonds 1991; Aguilar and Borrell 1994; Van Loveren et al. 2000; Koschinski 2002; Struntz et al. 2004). In species that are long-lived such as marine mammals, infectious diseases that affect reproductive success may play a role in the decline of threatened species and may also affect population recovery (Gaydos et al. 2004).

Threshold levels for health effects in wild species are often based on the results of laboratory studies involving other species (e.g., rats, mice, chickens, mink). Unfortunately it is not clear how transferable they are between species. Some studies have sought to mimic field conditions in controlled settings, such as the captive harbour seal study by Ross et al. (1996). However, it must be remembered that harbour seals are not Steller sea lions, nor are they cetaceans or walrus, and it may well be wrong to assume that each species would react to contamination at the same level and in the same ways. Ideally, threshold levels will be established for each species at risk from contamination; until then, threshold levels can only be inferred from studies with other species (Barron et al. 2003).

In most cases, model predictions suggest that total PCB concentrations in marine mammals remained below threshold levels associated with health effects (Figure 3.9; Table 3.4) throughout the time period between the 1950s and 1980s. The exception to this was the toothed whale functional group. This group represents four species – beluga whales, killer whales (both resident and transient), Dall's porpoise, and harbour porpoise. Transient killer whales feed primarily on other marine mammals, a characteristic incorporated into the diet composition of the model where a proportion of the toothed whale diet is marine mammals. While predicted concentrations are intended to be representative for each species in this group, transient killer whales are known to carry much higher PCB loads than resident killer whales and other toothed whales (Ross et al. 2000). Accordingly, the inclusion of transient killer whales in this group may produce a bias toward higher concentrations.

Marine mammals typically mature later in life and have a slow reproductive rate, producing only one offspring every one to three years on average. Reproductively active females offload a significant amount of their contaminant burden (~80%) to their offspring through gestation and lactation (Lee et al. 1996; Ross et al. 2000; Ylitalo et al. 2001). First-born offspring are at the greatest risk for toxicological effects because the maternal contaminant burden transferred is highest in primiparous females (Covaci et al. 2002). While health effects may not be obvious or readily detectible in adult animals, foetuses and nursing offspring are exposed to these (often high) concentrations at a critical stage of development and when they are particularly sensitive (Beckmen et al. 1999).

Contaminant burdens and effects may also be linked into the nutritional stress or 'junk-food' hypothesis for Steller sea lions (Rosen and Trites 2000). Contaminant burdens may have a greater effect under circumstances where the sea lions are stressed from longer foraging trips and a lack of food energy, particularly if fat reserves are used to compensate and contaminants are mobilized (Burek et al. 2003). Mortality from disease may increase as a result of greater nutritional stress or stress from avoiding predators (Loughlin and York 2000).

On the basis of current inputs, both models indicate that total PCB concentrations have reached a steady state in the eastern Bering Sea ecosystem (Figures 3.3 and 3.7). Whether or not the assumption that current PCB inputs will be maintained is correct does not alter the fact that Steller sea lions and other top predators are at risk from the contaminant loads that they already carry; these loads could ultimately compromise their survival and hinder population recovery (Atkinson et al. 2003; Burek et al. 2003; Gaydos et al. 2004).

**TABLE 3.4.** Threshold levels of PCBs in blubber associated with toxicity in marine mammals (mg/kg wet weight).

| Species | Total PCBs (mg/kg wet weight) | Effects | Source |
|---|---|---|---|
| Marine mammals | 100 | Hyperadrenocorticism[a] | Blomkvist et al. (1992) |
| Cetaceans | 50-200 | Health at risk | Berrow et al. (2002) |
| Whales and pinnipeds | 7-15[b] | Immunosuppression | Kannan et al. (2000) |
| Ringed and harbour seals (*Phoca hispida, P. vitulina*) | 60[c] | Reproductive dysfunction | Ylitalo et al. (2001) |
| Ringed seal (*Phoca hispida*) | 60[d] | Uterine occlusions | Helle et al. (1976a; 1976b) |
| Harbour seal (*Phoca vitulina*) | 11[e] | Immunosuppression | Ross et al. (1996) |
| Steller sea lion (*Eumetopias jubatus*) | 7.8[f] | Immunosuppression | Barron et al. (2003) |

[a] Hyperadrenocorticism refers to a disease complex that includes reproductive failure, bone lesions, adrenal hyperplasia, and lesions in the female reproductive system (Norstrom and Muir 1994; Chiu et al. 2000);
[b] Adjusted from 17 mg/kg lipid weight to wet weight concentrations using minimum (40% in Sperm whales) and maximum (90% in Beluga whales) % lipid reported in blubber samples;
[c] Adjusted from 77 mg/kg lipid weight to wet weight concentration for ringed and harbour seals using an average of 78% lipid in blubber;
[d] Adjusted from 70 mg/kg lipid weight to wet weight concentration for ringed seal using 85% lipid;
[e] Adjusted from 16.5 mg/kg lipid weight to wet weight concentration for harbour seals using 66% lipid;
[f] Barron et al. (2003) adjusted the threshold established for harbour seals (16.5 mg/kg lipid weight) to a wet weight concentration for Steller sea lions using a mean percentage in SSL blubber of 47.3%. Using the percentage lipid in blubber samples of 74% for adult males and 82% for adult females reported by Lee et al. (1996), would result in thresholds of 12.2 and 13.5 mg/kg ww., respectively.

Even though PCBs have been banned in most developed countries since the mid-1970s, the risks they pose persist today and will continue to do so into the future. Because long-lived marine mammals accumulate these contaminants throughout their lifetime and pass them on from one generation to the next (Tanabe et al. 1994a), concentrations are unlikely to decline rapidly. Continued environmental cycling maintains current concentrations while other sources threaten to increase PCB input (O'Shea and Brownell 1994).

Using the 1980s model, a simulation was run with Ecotracer to illustrate the effect of increasing PCB input by 1% every year from 2001 (Figure 3.5). While the model indicates the expected outcome that increased input would lead to increased concentrations, it also suggests that there would be a 5 to 20 year delay before these increased concentrations were noticeable in upper trophic levels. Delayed responses are characteristic of long-lived species (Holmes and York 2003). The results of this simulation are consistent with this and should caution researchers against taking 'snapshots' of PCB concentrations in top predators. The delayed response highlights the need for long-term, ongoing monitoring in order for changes to become apparent. Accordingly, continued monitoring of PCB input into the Bering Sea is essential to ensure the health and long-term viability of top predators, including humans. In conjunction with this, it is necessary to measure PCB levels in marine mammal lipid tissue on a regular basis (preferably at least annually) in order to detect concentration trends.

An equivalent but opposite simulation was run to illustrate the effect of decreasing input by 1% per year from 2001 (Figure 3.6). As expected, the model indicates that this would lead to declining concentrations in animal tissues. Notably, the model also suggests that there would not be a substantial delay before these decreased concentrations became evident. It is therefore reasonable to propose that further restrictions on the use of PCBs worldwide, together with clean-up measures for the numerous abandoned military sites in Alaska, would reduce the overall input of PCBs and decrease the exposure of marine mammals to PCBs in the eastern Bering Sea.

Chemicals occur in complex mixtures in the environment and in biota. As a result, toxicity can rarely be attributed to a single contaminant. Although PCBs are unlikely to be the sole cause of the Steller sea lion decline in the eastern Bering Sea region, they may have contributed along with factors such as an environmental regime shift, overlap with fisheries, and/or increased predation. Limited data exist regarding dose-response effects on marine mammals, even for well-known contaminants like PCBs (Macdonald and Bewers 1996; NRC 2003). Although PCB concentrations have remained below established threshold levels for total PCBs in blubber, these concentrations may have serious effects in foetuses and nursing young. Losing offspring to chemical poisoning greatly compromises the ability to grow or maintain populations and could inhibit the population recovery of Steller sea lions and other species that have declined in the Bering Sea.

#### *3.4.1.2 Human Health*

PCBs are generally not considered to be a threat to human health unless inhaled, absorbed or ingested. Unfortunately, Alaska Natives regularly encounter PCBs via all of these exposure routes. Direct inhalation and absorption exposures are limited and contribute a very small proportion of the total exposure – PCB contamination occurs primarily through the ingestion of contaminated wild foods (World Health Organization 2000). The Aleuts and other Alaska Natives are at greatest risk because they consume the highest amounts of wild foods. Their diets include a wide variety of species and they consume more parts of each animal than do non-Natives (Wolfe 1996; Malinowski et al. 1999).

PCB contamination in humans has been associated with neurological or neurobehavioural and reproductive effects. It can also cause immunosuppression, cancer, endocrine disruption, and abnormal growth (Berner 1999; Verbrugge 1999; World Health Organization 2000; Simmonds et al. 2002).

Similarly to marine mammals, developing human foetuses and nursing infants are at greatest risk for two reasons: 1) they have a small body mass and, 2) they are at a critical stage in their development where cell differentiation and migration are easily disrupted (Berner 1999). Although terrestrial mammals, including humans, metabolise PCBs better than marine mammals (Norstrom and Muir 1994), there is considerable concern as to whether traditional foods are safe for Alaska Natives to eat (AMAP 1997).

As a threshold geared towards protecting humans who consume fish, the U.S. Food and Drug Administration (FDA) has specified 2 ppm wet weight as the maximum amount of PCBs that may be present in any fish intended for human consumption. The U.S. Environmental Protection Agency (USEPA) further recommends that consumers restrict their consumption of fish containing more than 0.05 ppm wet weight PCBs to once a month in order to minimise the risk of cancer, and also to avoid consumption of any fish containing more than 0.097 ppm wet weight PCBs (Environmental Protection Agency 1999). The USEPA Daily Reference Dose for the protection of human health is 0.02 µg PCB·kg body weight$^{-1}$·day$^{-1}$. This is based on the results of a long term feeding study with monkeys, where the lowest dose at which an effect was observed was 5 µg·kg$^{-1}$·day$^{-1}$. Dividing this concentration by a safety factor of 300 provided the reference dose for humans (Verbrugge 1999).

Although it seems plausible that the risks posed by PCB contamination could be reduced by avoiding consumption of contaminated foods (Chiu et al. 2000), this is usually not a viable option for Alaska Natives that have relied on traditional foods for centuries. Furthermore, choices from the wide range of new 'U.S. diet' foods are often of lower quality when compared to the traditional items they are intended to replace (Draper 1978; Becker 2000). Traditional foods provide many nutrients, vitamins, antioxidants (Vitamin E and selenium), essential fatty acids, calories, and protein that contribute to numerous health benefits (Draper 1978; Egeland et al. 1998; Nobmann 1999; Becker 2000). Traditional food choices in the Bering Sea region, as well as preparation methods, were considered below.

#### *3.4.1.3 To Eat or Not to Eat*

To eat or not to eat – that is the question at hand for many Alaska Natives. Given that Alaska Natives consume large quantities of wild food each year, the contamination of these food sources is of concern throughout rural Alaskan communities. Important subsistence animals for Alaska Natives include several species of fish (e.g., salmon, halibut, and herring) and marine mammals (e.g., seals, sea lions, walrus, beluga whales, bowhead whales) (Wolfe 1996; AMAP 1997; Wolfe 2000). On average, subsistence activities provide rural Alaskans with 160-170 kg of food per person each year (Wolfe 1996; Rue 1998; Wolfe 2000), or approximately 0.45 kg per person per day. While the importance of any particular species may vary between communities, marine mammals have always been an integral part of Aleut culture, livelihood, and diet. For many Aleutian and Pribilof Islands communities, marine mammal harvests constitute a large portion of the total annual subsistence harvest, being highest in St. Paul (~50%), St. George (~40%), and Atka (~35%) (Wolfe 1997).

In the eastern Bering Sea ecosystem, animals taken for subsistence are primarily from Trophic Levels 3 and 4. This research indicates that the greatest flow and concentrations of PCBs also occur within these trophic levels, with predicted concentrations in marine mammals and fish ranging from 0.005 to 2.15 ppm. Given that marine mammals and fish constitute a large portion of the traditional Aleut diet, these concentrations, being well above USEPA guidelines, present a considerable contaminant risk to subsistence consumers. Although the Daily Reference Dose guideline is quite conservative (i.e., 50 times lower than the Canadian Tolerable Daily Intake dose of 1 $\mu g \cdot kg^{-1} \cdot day^{-1}$) and it's use as a benchmark is not ideal, it seems clear that measures to reduce human exposure to PCBs should be implemented.

As an example, in the Faroe Islands (Denmark) it was recognised that contaminants such as mercury and PCBs posed a serious risk for people who ate pilot whale products. Accordingly, the Danish government issued the following diet recommendations: 1) adults should only eat blubber and pilot whale meat once or twice a month; 2) girls and women should not eat blubber until they have given birth to all their children; 3) pilot whale meat should not be eaten within three months of planned pregnancy and not eaten at all by pregnant and nursing women; and 4) pilot whale organs (e.g., liver and kidney) should not be eaten at all (Endo et al. 2004). While some of these or equivalent recommendations may not be practical for all Alaska Natives, similar advisories for eastern Bering Sea residents may be warranted.

Perhaps the aspect of greatest concern for Alaska Natives is that PCB concentrations in their traditional foods are no longer declining (Figure 3.3 and 3.7) and that the full extent of this long-term chronic exposure has yet to be realised (Carson 1962). However, despite this concern and increased awareness regarding contamination of wild food sources, the Alaska Division of Public Health (ADPH) continues to recommend unrestricted consumption of wild foods (Egeland et al. 1998). Although the view seems to be that the benefits of traditional foods outweigh the health risks posed by PCB contamination (O'Hara et al. 1999), continued monitoring of contaminant levels in subsistence foods would be very prudent in order to reveal trends and maintain appropriate dietary advice (Egeland et al. 1998; de Bendern 2002).

#### *3.4.1.4 To Cook or Not to Cook ... The Beluga Cookbook*

There is increasing concern about contamination of traditional food sources, and discussion usually focuses on whether traditional foods are safe to eat. Marine mammal blubber is a key storage tissue and typically contains the greatest contaminant concentrations. Therefore, consumption of blubber can be a primary means of accumulating PCBs and other lipophilic contaminants (Becker 2000). While consumption of different tissues varies between villages and individuals, the Aleuts often eat their food raw because drying food for storage is usually impractical in the wet climate of the Aleutian Islands (Brennan and Tower 1998). If it is stored, meat and blubber is frozen and known as 'quaq' (Malinowski et al. 1999). Quaq is relatively easy to eat because ice crystals in the frozen tissue and blood assist chewing. Research has indicated that eating raw animal meat and blubber facilitates absorption of large amounts of vitamins and minerals stored in animal tissue, while the fat from these animals is readily converted into energy for body heat (Malinowski et al. 1999). Accordingly, eating raw animal meat and blubber will lead to accumulation of any contaminants present in the tissue.

In addition to selecting less contaminated food sources, food preparation methods should also be considered. Some studies have shown that different preparation methods for traditional foods can have an effect on their contaminant concentrations (Zabik et al. 1992; Simmonds et al. 2002). Zabik et al. (1992) found that boiling or steaming blue crab tissue reduced PCB content by more than 20%. However, this advantage is lost if the cooking medium (usually water) is used to prepare other foods such as soups. Simmonds et al. (2002) reported that total PCB concentrations were higher in fresh or frozen blubber when compared to cooked blubber, indicating that cooking has an effect on contaminant levels. Heat can cause thermal degradation and evaporation of lower chlorinated PCB congeners and less stable pesticides (Simmonds et al. 2002). Although boiling appears to liberate organochlorines associated with lipids, other non-lipophilic contaminants such as mercury may remain, due to resistance to thermal degradation (Simmonds et al. 2002). In other words, because mercury concentrates in muscle tissue, it cannot be cooked out of wild foods.

While cooking seems to have the desired effect of reducing PCB concentrations in traditional foods, it is possible that unwanted effects in relation to other contaminants might also result. Unfortunately data in this regard is very limited and many questions regarding the fate of the complex mixture of contaminants in traditional foods remain unanswered for the range of preparation methods used (Becker 2000).

### 3.4.2 Ecotracer routine

Ecotracer is a useful tool for predicting PCB concentrations in the eastern Bering Sea, and could be used for other lipophilic compounds (such as DDT, HCH, and HCB), nutrients (Watkinson 2001) or radioactivity (Dalsgaard 1999); in fact, for anything that moves through an ecosystem with biomass. Ecotracer can also be used to verify the diet compositions in a model because what a group eats directly affects the flow of PCBs into that group. For example, the prediction of a higher contaminant concentration for a lower trophic level organism may indicate an error in the diet composition matrix.

*3.4.2.1 Improvements to Ecotracer*

There are limitations to what can be achieved with Ecotracer. Although Ecotracer currently operates at a non-lethal level it doesn't incorporate some important longer term and cumulative factors, such as the reduced growth and fecundity in marine mammals that can result from prolonged exposure to low concentrations of PCBs. It is important to consider both lethal and sub-lethal impacts when assessing the toxicity of chemicals like PCBs. The Ecotracer software could be improved by allowing inclusion of direct lethal effects of PCB toxicity, as well as secondary effects such as sub-lethal impacts on growth and reproduction. This would likely make the role of individual contaminants more evident and improve prediction of the effects of chronic, low-level exposures (Chapman 1997).

Further refinement would also be required to enable the models to fully capture these toxic effects. With particular reference to marine mammals, refinements could include incorporation of age and sex structures to reflect the differences in toxic burden and also to account for maternal transfer to offspring. High contaminant burdens are transferred to developing offspring and can have lethal effects for the first recruited. As the contaminant burden decreases in females with each offspring, this transfer and risk (or effect) is less severe for successive young. Females are also known to resume accumulating contaminants once reproduction ceases. Other refinements could include further separation of functional groups to account for obvious diet differences (e.g., transient and resident killer whales) and to reflect species differences in maternal transfer of contaminants and their metabolic capacity. The addition of a functional group to represent Alaska Natives, the human component of the system, would also be appropriate.

Ecopath with Ecosim is a constantly evolving software package; the developers strive to extend or modify the software where possible to accommodate user requests and requirements. Regrettably, due to the complexity of chemical properties and individual species responses, coupling tissue residue levels with toxicity responses in an ecosystem modelling setting is very difficult. It may therefore be an unrealistic goal for the near future.

*3.4.2.2 Future work with Ecotracer and the eastern Bering Sea models*

In order to assess likely contaminant effects on Native communities and wildlife in the eastern Bering Sea region, it is important to know the nature, sources, flow patterns, and levels of contaminants that are present in the region. An accurate estimate of total PCBs throughout the eastern Bering Sea food web, as undertaken by this study, was necessary to provide an initial description of this contaminant of concern. Full analysis of the toxic effects of PCBs on mammals is complex due to the number of congeners present in the environment and their respective properties and toxicities. While total PCB concentrations provide an approximation of toxicity for higher trophic level species, the risks associated with exposure may be misleading without congener-specific data (Valoppi et al. 1998; Ross and Troisi 2001; Atkinson et al. 2003).

This study provides an estimate of total PCBs in the eastern Bering Sea food web, and relies on literature-based toxicity information, combined with modelled exposure concentrations, to predict hazards to wildlife and humans. Future research could include congener-specific modelling, particularly for the 12 non- and mono-ortho congeners for which toxic equivalency factors (TEFs) exist, in order to assess toxicity in more detail and with greater accuracy (Valoppi et al. 1998; Van den Berg et al. 1998; Neff et al. 1999). Toxicity assessments based on TEFs, together with the corresponding calculation of toxic equivalency quotients (TEQs), would account for the different toxicity values of certain congeners and provide an accurate measure of the toxicity levels contributed by individual congeners.

Other contaminants for which the largest database on contaminant concentrations in Alaska marine mammals exists are DDT (dichlorodiphenyltrichloroethane) and its metabolites (DDD and DDE), HCB (hexachlorobenzene), HCH (hexachlorocyclohexane), dieldrin, endrin, total mercury, and cadmium. These are the most discussed contaminants of concern in Alaska, and further research should include modelling of the flow and accumulation of these contaminants in the eastern Bering Sea. This would provide a comprehensive perspective on contaminant impacts, including consideration of possible additive and/or synergistic effects.

### 3.4.3 Utility of PCB model to resource managers

The Ecotracer routine has been shown to provide satisfactory predictions for beta radioactivity (Dalsgaard 1999) and nitrogen (Watkinson 2001). However, prior to this study the routine had not been tested in relation to organochlorines. One main goal was to demonstrate that the models and Ecotracer software were sufficiently accurate for this type of chemical, using PCBs as an example. This type of modelling has rarely been done for PCBs in complex marine food webs. Most of the literature relates to freshwater food webs (e.g., Gobas 1993; Morrison et al. 1999), and/or food webs that are very simplified (e.g., Connolly 1991). Incorporating PCB information into the refined eastern Bering Sea models extends the models and their applications. Ecosystem modelling of this nature illuminates how contaminant burdens in Bering Sea predators are responding to, or have responded to, the production, use and subsequent restriction of PCBs. Predictions of contaminant concentrations based on models are particularly useful because they can be verified with field observations and provide insights into bioaccumulation processes (Chapman 1997).

This research contributes to a better understanding of the eastern Bering Sea ecosystem and the possible health implications for those who depend on its resources. The PCB concentrations in marine mammals predicted by this study will be of particular interest to Alaska Natives and others who rely on hunting for subsistence, as well as to conservation biologists and toxicologists.

Contemporary resource management would benefit from modifying its primary focus away from single species and toward the roles of every species, including humans, in the total ecosystem. Quantitative single species data alone is inadequate for dealing with all of the factors and linkages at work. Accordingly, resource managers should consider adopting broader ecosystem management practices that incorporate system-wide interrelationships. The refined Bering Sea models can now be used to inform management. Future resource management decisions for the region can be based on improved information and can be consistent with best conservation practices.

### 3.4.4 Reducing or preventing the PCB risk

PCBs are persistent, bioaccumulative, toxic, and a product of human activity; they should also be subject to elimination strategies (Environment Canada 1997). Although it is essentially impossible to remove PCBs once they are established in natural environments, options exist to reduce the health risk for marine mammals and humans. A worldwide ban on PCB manufacture and use would be a valuable foundation. Additionally, formulation and implementation of effective strategies to prevent PCBs from entering the environment from current and potential sources would be very worthwhile. In particular, this could include cleaning-up sites such as abandoned military facilities, as well as destroying PCBs currently in storage.

The mid-1970s North American ban on manufacture and open use of PCBs lead to an initial reduction in concentrations in many Bering Sea species. As indicated by this study, if further comprehensive bans, restrictions, and clean-up strategies are implemented to reduce or eliminate input, as proposed by the Stockholm Convention on Persistent Organic Pollutants (POPs), additional declines in PCB concentrations can be expected. The Convention calls for the complete phasing out of PCBs, including those currently being manufactured and those still in use in closed systems such as electrical equipment. Unfortunately, the Convention requires ratification by 50 countries before it comes into effect. The USA has yet to ratify this Convention.

Although it does not appear that Alaska is the primary source of PCBs in the Bering Sea, PCBs are the most prominent contaminant encountered at the numerous formerly-used defence sites (FUDS) and distant early warning (DEW) stations across Alaska (Thomas et al. 1992; Bacon et al. 1999). These sites need to be cleaned up before they too become a major source of PCBs to the region.

It should also be noted that PCBs are just one of a complex suite of contaminants that are pervasive throughout the world's ecosystems (Carson 1962), and both synergistic and antagonistic relationships between chemicals likely take place (Kinloch et al. 1992; Beckmen et al. 2002). Other POPs could be contributing to toxic effects. Equal targeting for strict regulation and phasing out of these POPs may also be warranted. The Stockholm Convention is the first global agreement seeking to ban a class of chemicals because of their toxic effects on wildlife and humans (Oceana 2003b). Although the Convention initially identifies the 12 most toxic contaminants (the 'dirty dozen') for priority action, it also includes a science-based process to identify other dangerous POPs (Oceana 2003a).

In the long term, international agreements such as the Stockholm Convention are likely to be the instruments that most effectively reduce the contaminant load in biota and thereby reduce the risks posed to marine mammals and humans. However, a reduction in marine mammal PCB levels would not occur overnight, and in the meantime continued monitoring of contaminants in subsistence taken animals would help to provide the best dietary advice to reduce the risks for humans (Simmonds 1991; de Bendern 2002).

### 3.4.5 Conclusions

Ecotracer has proved useful in predicting PCB concentrations in the eastern Bering Sea and other researchers may want to consider using it for contaminant investigations in other regions of the world. The application and development of models that predict contaminant concentrations at an ecosystem level is necessary to understand and anticipate future trends across various ecosystems (Loganathan and Kannan 1991).

Relatively little information is available regarding the temporal trends of PCB levels in eastern Bering Sea biota. Without the background and time trend data made possible by Ecotracer, it was difficult to piece together a complete picture of PCB contamination in the eastern Bering Sea food web. With the development of Ecotracer, realistic predictions can now be made beyond a single trophic level. This research fills the historical-void with predicted concentrations from the 1950s model, and provides insights into the progress of PCB contamination over time.

Although model results are in good agreement with the limited observed data available for some marine mammal and fish species, it is particularly evident that more long-term monitoring efforts are required to generate a data set sufficient for model validation.

PCBs are known to cause immunosuppression in a number of marine mammal species and may render them more vulnerable to disease and virus epizootics. Although predictions from the model suggest that concentrations have remained below threshold levels, young animals are even more susceptible to the effects of contaminants than adults because they are exposed at critical stages in their development. These effects, particularly on younger animals, may play a role in the decline of threatened species and may also affect population recovery in the Bering Sea (Gaydos et al. 2004).

On a cultural rather than purely nutritional basis, the contaminant risks for humans who consume these marine mammals continue to be outweighed by the benefits associated with a traditional diet. Perhaps the risks posed by contaminants in depleting marine mammal populations are of greater concern (as regards maintaining a steady subsistence food source) than direct compromises to human health. This indirect effect of PCBs and other contaminants on Alaska Natives (i.e., depletion of marine mammals) may be a greater threat to their culture, diet, and overall well being than the contaminants themselves.

Assuming that PCB input to the eastern Bering Sea continues at current levels, model predictions also suggest that concentrations have reached a steady state. It is likely that the full extent of this long-term chronic exposure has yet to be manifested in marine mammals and humans (Carson 1962; Tanabe 2002).

As the sensitivity of canaries to carbon monoxide was often used to detect high levels of this gas in coalmines, perhaps pinnipeds are the 'canary in the coalmine' of the Bering Sea; their depleted populations providing early warning signals that contaminant concentrations in the food web are too high (Ross and Troisi 2001).

It is difficult to reach definitive conclusions regarding PCBs as a cause for marine mammal declines in the eastern Bering Sea because field studies detailing concentration levels are lacking for the most critical time periods. Although modelling the flow and predicted PCB concentrations using the 1950s ecosystem model indicates that concentrations remained below threshold levels associated with reproduction and survival effects, this research does not rule out the possibility that PCBs and other contaminants are a contributing factor that should be taken into account when considering the decline of Steller sea lions and other marine mammals in the eastern Bering Sea.

Furthermore, organisms differ and do not experience the same contaminant dose or effect under the same circumstances. Based on the precautionary principle and given continued uncertainty in relation to the dose-response effects of PCB contamination, further restrictions and effective clean-up measures should be implemented to reduce the exposure of marine mammals and humans and to protect the resources of the eastern Bering Sea.

### 3.4.6 Summary

Persistent organic pollutants (POPs) are lipophilic, resist degradation, and readily accumulate in marine mammal and fish tissue. As a consequence of their trophic level and longevity, marine mammals and people are particularly susceptible to high POP contamination. POPs, such as polychlorinated biphenyls (PCBs), are considered possible contributors to marine mammal population declines in the eastern Bering Sea. Using ecosystem modelling software (Ecopath with Ecosim – EwE), this study identified likely pathways of contaminant flow within the eastern Bering Sea and evaluated health implications of contaminant exposure for Steller sea lions, other species of marine mammals, and humans. Ecotracer (a component of the EwE software) tracked the bioaccumulation of contaminants moving through the system with biomass. Data deficiencies were identified and the model estimated contaminant concentrations for species and functional groups that have not previously been measured. A description of the methods and approach used for tracing organic contaminants using the Ecotracer software was also provided.

Although PCBs are unlikely to be the sole cause of Steller sea lion decline in the eastern Bering Sea region, they may be a contributor along with factors such as an environmental regime shift, overlap with fisheries, and killer whale predation. Limited data exist regarding dose-response effects on marine mammals, even for well-known contaminants like PCBs. Results of the model suggest that PCB concentrations for most species in the eastern Bering Sea have remained below threshold levels associated with reproduction and survival effects. However, these concentrations may have subtle effects on adults and more serious effects on foetuses and nursing young, which could inhibit the recovery of Steller sea lions and other species that have declined in the eastern Bering Sea. Although the benefits of traditional foods appear to continue to outweigh the risks posed by contaminants for humans, PCB exposure and dietary intake for many Alaska Natives subsisting on marine mammals is above the USEPA Daily Reference Dose. Traditional food choices and preparation methods were considered. The results are important in terms of management alternatives for marine mammals and human health in the eastern Bering Sea region, and to synthesise evidence regarding the presence, extent and movement of PCBs in this ecosystem. Possible refinements to the methodology of the Ecotracer routine were also identified.

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# Appendices

## APPENDIX 1 – DIET MATRIX TABLES

**TABLE A1.1.** The estimated proportion of prey (nos. 1-25) eaten by predators (nos. 1-23) during the 1950s. Modified from Trites et al. (1999a) to include Pacific herring and to reflect adjusted diet compositions.

| Prey | Predator | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 | 10 | 11 |
| 1. Baleen whales | 0.000 | 0.002 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 2. Toothed whales | 0.000 | 0.001 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 3. Sperm whales | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 4. Beaked whales | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 5. Walrus & bearded seals | 0.000 | 0.001 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 6. Seals | 0.000 | 0.002 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 7. Steller sea lions | 0.000 | 0.001 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 8. Piscivorous birds | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 9. Adult pollock | 0.002 | 0.008 | 0.002 | 0.002 | 0.002 | 0.002 | 0.002 | 0.000 | 0.000 | 0.000 | 0.020 |
| 10. Juvenile pollock | 0.007 | 0.013 | 0.002 | 0.006 | 0.002 | 0.018 | 0.029 | 0.095 | 0.020 | 0.000 | 0.008 |
| 11. Other demersal fish | 0.000 | 0.056 | 0.050 | 0.000 | 0.018 | 0.052 | 0.031 | 0.106 | 0.007 | 0.000 | 0.043 |
| 12. Large flatfish | 0.000 | 0.053 | 0.000 | 0.000 | 0.018 | 0.046 | 0.003 | 0.000 | 0.001 | 0.000 | 0.000 |
| 13. Small flatfish | 0.000 | 0.050 | 0.000 | 0.000 | 0.010 | 0.040 | 0.003 | 0.000 | 0.000 | 0.000 | 0.120 |
| 14. Pacific herring | 0.182 | 0.319 | 0.056 | 0.172 | 0.044 | 0.288 | 0.862 | 0.572 | 0.123 | 0.000 | 0.076 |
| 15. Other pelagics | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.200 |
| 16. Deepwater fish | 0.000 | 0.053 | 0.050 | 0.333 | 0.018 | 0.046 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 17. Jellyfish | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 18. Cephalopods | 0.150 | 0.230 | 0.840 | 0.320 | 0.000 | 0.009 | 0.070 | 0.040 | 0.000 | 0.000 | 0.010 |
| 19. Benthic particulate feeders | 0.009 | 0.165 | 0.000 | 0.167 | 0.280 | 0.269 | 0.000 | 0.000 | 0.031 | 0.000 | 0.173 |
| 20. Infauna | 0.000 | 0.000 | 0.000 | 0.000 | 0.420 | 0.000 | 0.000 | 0.001 | 0.010 | 0.000 | 0.200 |
| 21. Epifauna | 0.015 | 0.000 | 0.000 | 0.000 | 0.178 | 0.000 | 0.000 | 0.000 | 0.001 | 0.000 | 0.072 |
| 22. Large zooplankton | 0.336 | 0.046 | 0.000 | 0.000 | 0.000 | 0.230 | 0.000 | 0.186 | 0.451 | 0.330 | 0.064 |
| 23. Herbivorous zooplankton | 0.299 | 0.000 | 0.000 | 0.000 | 0.010 | 0.000 | 0.000 | 0.000 | 0.356 | 0.670 | 0.014 |
| 24. Phytoplankton | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 25. Detritus | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| Sum | 1.000 | 1.000 | 1.000 | 1.000 | 1.000 | 1.000 | 1.000 | 1.000 | 1.000 | 1.000 | 1.000 |

**TABLE A1.1.** *continued.*

| Prey | Predator | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | 12 | 13 | 14 | 15 | 16 | 17 | 18 | 19 | 20 | 21 | 22 | 23 |
| 1. Baleen whales | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 2. Toothed whales | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 3. Sperm whales | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 4. Beaked whales | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 5. Walrus & bearded seals | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 6. Seals | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 7. Steller sea lions | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 8. Piscivorous birds | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 9. Adult pollock | 0.025 | 0.000 | 0.000 | 0.000 | 0.020 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 10. Juvenile pollock | 0.084 | 0.003 | 0.001 | 0.000 | 0.014 | 0.002 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 11. Other demersal fish | 0.009 | 0.003 | 0.000 | 0.000 | 0.003 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 12. Large flatfish | 0.000 | 0.001 | 0.000 | 0.000 | 0.018 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 13. Small flatfish | 0.010 | 0.010 | 0.000 | 0.000 | 0.100 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 14. Pacific herring | 0.103 | 0.021 | 0.000 | 0.000 | 0.356 | 0.008 | 0.210 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 15. Other pelagics | 0.600 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 16. Deepwater fish | 0.002 | 0.000 | 0.000 | 0.000 | 0.002 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 17. Jellyfish | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.008 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 18. Cephalopods | 0.004 | 0.000 | 0.000 | 0.000 | 0.110 | 0.000 | 0.190 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 19. Benthic particulate feeders | 0.047 | 0.054 | 0.000 | 0.000 | 0.146 | 0.128 | 0.000 | 0.000 | 0.000 | 0.140 | 0.000 | 0.000 |
| 20. Infauna | 0.002 | 0.600 | 0.005 | 0.000 | 0.102 | 0.000 | 0.000 | 0.300 | 0.000 | 0.300 | 0.000 | 0.000 |
| 21. Epifauna | 0.009 | 0.084 | 0.000 | 0.000 | 0.020 | 0.000 | 0.000 | 0.037 | 0.000 | 0.008 | 0.000 | 0.000 |
| 22. Large zooplankton | 0.105 | 0.191 | 0.904 | 0.400 | 0.109 | 0.139 | 0.600 | 0.344 | 0.000 | 0.044 | 0.003 | 0.000 |
| 23. Herbivorous zooplankton | 0.000 | 0.033 | 0.090 | 0.600 | 0.000 | 0.715 | 0.000 | 0.017 | 0.000 | 0.008 | 0.271 | 0.000 |
| 24. Phytoplankton | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.676 | 1.000 |
| 25. Detritus | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.302 | 1.000 | 0.500 | 0.050 | 0.000 |
| Sum | 1.000 | 1.000 | 1.000 | 1.000 | 1.000 | 1.000 | 1.000 | 1.000 | 1.000 | 1.000 | 1.000 | 1.000 |

**TABLE A1.2.** The estimated proportion of prey (nos. 1-26) eaten by predators (nos. 1-23) during the 1980s. Modified from Trites et al. (1999a) to include Pacific herring and discards and to reflect adjusted diet compositions.

| Prey | Predator | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 | 10 | 11 |
| 1. Baleen whales | 0.000 | 0.002 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 2. Toothed whales | 0.000 | 0.001 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 3. Sperm whales | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 4. Beaked whales | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 5. Walrus & bearded seals | 0.000 | 0.001 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 6. Seals | 0.000 | 0.002 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 7. Steller sea lions | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 8. Piscivorous birds | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 9. Adult pollock | 0.064 | 0.081 | 0.012 | 0.041 | 0.020 | 0.037 | 0.063 | 0.000 | 0.000 | 0.000 | 0.229 |
| 10. Juvenile pollock | 0.065 | 0.081 | 0.013 | 0.042 | 0.020 | 0.115 | 0.185 | 0.599 | 0.130 | 0.000 | 0.048 |
| 11. Other demersal fish | 0.000 | 0.053 | 0.050 | 0.000 | 0.021 | 0.046 | 0.031 | 0.106 | 0.007 | 0.000 | 0.035 |
| 12. Large flatfish | 0.000 | 0.053 | 0.000 | 0.000 | 0.021 | 0.046 | 0.003 | 0.000 | 0.001 | 0.000 | 0.000 |
| 13. Small flatfish | 0.000 | 0.053 | 0.000 | 0.000 | 0.020 | 0.046 | 0.003 | 0.000 | 0.000 | 0.000 | 0.128 |
| 14. Pacific herring | 0.065 | 0.081 | 0.013 | 0.042 | 0.019 | 0.077 | 0.320 | 0.000 | 0.000 | 0.000 | 0.010 |
| 15. Other pelagics | 0.065 | 0.081 | 0.013 | 0.042 | 0.021 | 0.077 | 0.314 | 0.051 | 0.014 | 0.000 | 0.020 |
| 16. Deepwater fish | 0.000 | 0.053 | 0.050 | 0.333 | 0.018 | 0.046 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 17. Jellyfish | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 18. Cephalopods | 0.143 | 0.246 | 0.849 | 0.333 | 0.000 | 0.007 | 0.080 | 0.053 | 0.000 | 0.000 | 0.014 |
| 19. Benthic particulate feeders | 0.049 | 0.165 | 0.000 | 0.167 | 0.250 | 0.270 | 0.000 | 0.000 | 0.031 | 0.000 | 0.244 |
| 20. Infauna | 0.000 | 0.000 | 0.000 | 0.000 | 0.339 | 0.000 | 0.000 | 0.001 | 0.016 | 0.000 | 0.134 |
| 21. Epifauna | 0.054 | 0.000 | 0.000 | 0.000 | 0.250 | 0.000 | 0.000 | 0.000 | 0.001 | 0.000 | 0.072 |
| 22. Large zooplankton | 0.253 | 0.046 | 0.000 | 0.000 | 0.000 | 0.232 | 0.000 | 0.188 | 0.441 | 0.330 | 0.064 |
| 23. Herbivorous zooplankton | 0.243 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.357 | 0.670 | 0.000 |
| 24. Phytoplankton | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 25. Discards | 0.000 | 0.001 | 0.001 | 0.000 | 0.001 | 0.001 | 0.001 | 0.002 | 0.002 | 0.000 | 0.002 |
| 26. Detritus | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| Sum | 1.000 | 1.000 | 1.000 | 1.000 | 1.000 | 1.000 | 1.000 | 1.000 | 1.000 | 1.000 | 1.000 |

**TABLE A1.2.** *continued.*

| Prey | Predator | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | 12 | 13 | 14 | 15 | 16 | 17 | 18 | 19 | 20 | 21 | 22 | 23 |
| 1. Baleen whales | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 2. Toothed whales | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 3. Sperm whales | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 4. Beaked whales | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 5. Walrus & bearded seals | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 6. Seals | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 7. Steller sea lions | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 8. Piscivorous birds | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 9. Adult pollock | 0.225 | 0.000 | 0.000 | 0.000 | 0.196 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 10. Juvenile pollock | 0.523 | 0.016 | 0.001 | 0.000 | 0.087 | 0.010 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 11. Other demersal fish | 0.006 | 0.002 | 0.000 | 0.000 | 0.003 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 12. Large flatfish | 0.000 | 0.001 | 0.000 | 0.000 | 0.018 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 13. Small latfish | 0.013 | 0.011 | 0.000 | 0.000 | 0.100 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 14. Pacific herring | 0.004 | 0.000 | 0.000 | 0.000 | 0.010 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 15. Other pelagics | 0.057 | 0.008 | 0.000 | 0.000 | 0.090 | 0.000 | 0.200 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 16. Deepwater fish | 0.002 | 0.000 | 0.000 | 0.000 | 0.002 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 17. Jellyfish | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.008 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 18. Cephalopods | 0.004 | 0.000 | 0.000 | 0.000 | 0.123 | 0.000 | 0.200 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 19. Benthic particulate feeders | 0.048 | 0.055 | 0.000 | 0.000 | 0.236 | 0.128 | 0.000 | 0.000 | 0.000 | 0.004 | 0.000 | 0.000 |
| 20. Infauna | 0.002 | 0.623 | 0.005 | 0.000 | 0.002 | 0.000 | 0.000 | 0.317 | 0.000 | 0.308 | 0.000 | 0.000 |
| 21. Epifauna | 0.009 | 0.084 | 0.000 | 0.000 | 0.020 | 0.000 | 0.000 | 0.037 | 0.000 | 0.008 | 0.000 | 0.000 |
| 22. Large zooplankton | 0.105 | 0.198 | 0.904 | 0.400 | 0.111 | 0.139 | 0.600 | 0.338 | 0.000 | 0.044 | 0.003 | 0.000 |
| 23. Herbivorous zooplankton | 0.000 | 0.000 | 0.090 | 0.600 | 0.000 | 0.715 | 0.000 | 0.000 | 0.000 | 0.000 | 0.213 | 0.000 |
| 24. Phytoplankton | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.676 | 1.000 |
| 25. Discards | 0.002 | 0.002 | 0.000 | 0.000 | 0.002 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 26. Detritus | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.308 | 1.000 | 0.636 | 0.108 | 0.000 |
| Sum | 1.000 | 1.000 | 1.000 | 1.000 | 1.000 | 1.000 | 1.000 | 1.000 | 1.000 | 1.000 | 1.000 | 1.000 |