

Joint Determination to Assess the Economic Importance of Ecosystem Services *

David Finnoff
finnoff@uwyo.edu

Min Gong
mgong@uwyo.edu

John Tschirhart
jtsch@uwyo.edu

Department of Economics and Finance
Department 3985 • 162 Ross Hall
University of Wyoming
Laramie, WY 82071

Although ecosystems provide myriad services to economies, only one service is considered in most renewable-resource models. The bioeconomic model introduced here admits a second service, and more importantly it accounts for how the two services are impacted by interactions within a thirteen species ecosystem and interactions within a regional economy. Endangered Steller sea lion recovery measures via alternative pollock quotas are shown to cause altered levels of all ecosystem populations, and altered levels of all economic variables. While non-use values associated with the ecosystem (e.g., existence values) are not considered, all species matter for the economy because they are all used indirectly as support for ecosystem services. Regional welfare losses from reduced quotas are mediated by increased provision of ecosystem services.

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1 Introduction

Human economies are not self contained, they are embedded in ecological systems. These ecosystems provide services that "sustain and fulfill human life" (Daily, 1997, p. 3); in fact, all economic activity ultimately depends on ecosystem services (Dasgupta, 1993). The extent to which economies can draw upon ecosystem services has limits, and by some accounts the limits have been reached and depleted ecosystems are having negative impacts on living standards (Arrow, et al., 1995). Because most analyses that measure the welfare implications of economic activity omit ecosystems, assessing the welfare consequences of policies may be biased. The objective of this work is to add a necessary amount of underlying ecology to an economic model so that the welfare consequences of economic activity can be better assessed.

Economists have demonstrated a growing awareness that ecosystems services are important inputs into economic activity. There are two themes that recur in the literature: 1) ecosystems and economies are jointly determined, and 2) both systems are general equilibrium in nature. Regarding 1), joint determination was emphasized early on by Daly (1968), and more recently Crocker and Tschirhart (1992), Nordhaus and Kokkelenberg (1999) and Settle and Shogren (2002) among others have pointed to joint determination as a key ingredient in introducing biological issues into economics. Some authors have admitted considerably more ecological detail to capture the interplay between the systems (e.g., see Brown and Roughgarden, 1995, Carpenter et al., 1999, Brock and Xepapadeas, 2003, or Tilman, et al., 2005).

Regarding 2), general equilibrium (GE) theory has been referred to as the most important development in economics in the twentieth century (Sandler, 2001). We assert that GE can be applied to ecology because ecosystems are subject to the same reality that economies are: "They are,..., highly nonlinear complex adaptive systems with extensive interconnections among components" (Arrow, et al., 2000). Including ecological variables in economic analyses without recognizing the general equilibrium nature of ecosystems can introduce errors for the same reasons that partial equilibrium compared to GE economic analyses can introduce errors in measuring welfare (Kokoski and Smith, 1987).

In this paper we address both themes by jointly determining an economic computable general equilibrium (CGE) model with a dynamic general equilibrium ecosystem model (GEEM). Variables from the CGE model depend on variables from the GEEM model, and vice versa. The method is unique among extant methods¹ in that

¹ A large literature introduces environmental issues into the CGE framework. Studies assessing the costs of environmental protection (Jorgenson and Wilcoxon 1990, 1993) have been augmented by those that include contributions of environmental resources to consumer welfare in a separable (Perroni and Wigle, 1994) and non-separable (Espinosa and Smith, 1995, 2002) fashion. A body of literature has also investigated the consequences of optimal environmental taxation in the presence of preexisting distortions (Bovenberg and Goulder, 1996, Goulder, Parry, Williams and Butraw, 1999, and Fullerton and Metcalf, 2001). There have been many notable contributions concerning global warming and the expected impacts of climate change. Early literature focused on the costs of proposed CO₂ abatement strategies (including Manne and Richels 1992, Nordhaus and Yang 1996, and Bohringer and Rutherford 1997) while recent examples expanding this to look at the benefits and costs of mitigating climate change (Goulder and Schneider, 1999). In this work environmental quality is viewed as an aggregate stock degraded by economic pollution. The natural system underlying environmental quality is neglected, ignoring the mediating behavior of the economic and ecological systems. Amir (1979) and Crocker and Tschirhart (1992) present linked economic/ecological analytical models while Eichner and Pethig (2003) develop the integration further. Of the few analyses that attempt empirical economic/ecological linkages, Watts et al. (2001) use a CGE model to investigate the economic impacts of preserving several endangered fish species, and Seung et al. (2000) use a dynamic CGE model in conjunction with a recreation demand model to assess the impacts of water reallocation policies. Neither of these papers has a separate ecosystem model. Jin, Hoagland, and Dalton (2003) merge a static economic input-output model of New England with a static ecological input-output model of a marine foodweb, but this approach omits dynamic and behavioral considerations. The work presented herein is differentiated from all of the above because the ecosystem is represented by a stand-alone, behaviorally based, dynamic, general equilibrium model. Just as the economic system consists of agents exhibiting behavior, the ecosystem also consists of agents exhibiting behavior, although the agents are plants and animals.

both the linked economic and ecological systems are general equilibrium aggregations of individual behavior.

CGE/GEEM is applied to the Alaskan economy to evaluate welfare consequences of endangered Steller sea lion recovery measures via alternative pollock quotas (a primary prey species). The economic model is linked through two ecosystem services, to a thirteen species marine ecosystem. While there are many ways to link economies and ecosystems, we make ecosystem services the linkages to show how the provision of the services is determined by both intra-ecosystem functions and economic activities. Commercial fishing and tourism are the two ecosystem services considered. Walleye pollock in the marine ecosystem support one of the world's largest fisheries, but pollock are also an important prey species for Steller sea lions. The sea lions, along with other charismatic marine mammals support a tourist industry. Steller sea lions are an endangered species and we analyze the welfare consequences of alternative pollock fishery quotas that are set to restore sea lion populations. Including a renewable resource in CGE is novel and requires significant modifications to standard renewable resource models.

Our use of CGE/GEEM to analyze this policy is purposeful and has some important consequences. Similar to the decision to use a general equilibrium perspective for the economy, deciding the boundaries of the linked models is critical. In the economy there are questions of what sectors to disaggregate and what detail to add while in the ecosystem there are questions of what species to include. And as with CGE, there is an issue of whether or not enough data exist to test the predictions of the model. But because economies and ecosystems are heavily interconnected and ecosystem services are numerous, we believe that linking CGE with GEEM can be useful for identifying which interconnections are important and where boundaries can be judiciously drawn.

The point is that when economic activity draws upon one ecosystem service, it inevitably affects other ecosystem services, and to understand the effects requires including enough of the ecosystem that supplies the services. Including ecological variables in economic analyses without recognizing the general equilibrium nature of ecosystems can introduce errors for the same reasons that partial equilibrium compared to GE economic analyses can introduce errors in measuring welfare (Kokoski and Smith, 1987).

The results of the model are necessarily numerical as it is analytically intractable. Although these quantitative results are reported in detail, it is the qualitative implications of these results that are important and demonstrate the power of the method. We show how alternative endangered Steller sea lion recovery measures that translate into alternative pollock quotas change all ecosystem populations and all economic variables. While existence values for individual species are not considered, all species matter for the economy because they are all used indirectly as support for ecosystem services. This is consistent with Worm et al. (2006) who show how greater species richness, including species that are not economically exploited, leads to more stable populations of commercially important species.

Regional welfare falls with reductions in Pollock quotas, but its decline is attenuated by a growth in the other ecosystem service. Welfare declines because capital and labor move from the regulated open access fishery sector to other sectors where they earn less on an annual basis. The welfare decline is mitigated because the tourism industry grows owing to increased numbers of marine mammals. Regional welfare

changes from reduced quotas are thus influenced not only by economic factor reallocation, but also by the economy relying less on resource extraction and more on non-extraction. Finally, we provide an estimate of the non-consumptive use value of marine mammals, demonstrating the potential of the method for non-market valuation.

CGE and GEEM are described in the next two sections. Economic and ecological data used in the simulations precede the policy analysis, followed by a brief conclusion.

2 The Economy Model

The linked CGE/GEEM is applied here to the state of Alaska and a marine ecosystem comprising Alaska's Aleutian Islands (AI) and the Eastern Bering Sea (EBS). A simplified view of the linked economy and ecosystem are represented in Figure 1. The economy consists of Alaskan households and producing sectors, linked to one another and the rest of the world through commodity and factor markets. The rest of the world includes both domestic trade (with the lower 48 states) and foreign trade. All species in the food web are linked together through predator-prey relationships and several species provide inputs to economic production. The prominent groundfish of the system, pollock, support a very large fishery, and charismatic marine mammals, Steller sea lions (an endangered species), killer whales, blue whales, sperm whales, northern fur seals and sea otter, provide important non-use inputs to the state's recreation sector.

Within the economy, all agents are assumed to be perfectly myopic (consistent with the fishery regulation), and all intertemporal consequences of savings and investment behavior omitted such that factor stocks remain constant over time. Federal and State governments are included in the model through taxation and redistribution, and through fishery regulation.

The simplified Alaskan economy is modeled as having four production sectors: fish harvesting FH , fish processing FP , recreation and tourism R , and composite goods C .² Fish harvesting (not shown in Figure 1) consists of differentiated pollock harvesting FHP (dependent on the modeled ecosystem) and harvesting of all other species FHH (not dependent on the modeled ecosystem). Recreation and tourism represents the Census Bureau's classification of Wildlife Related Recreation, and composite goods are a catch all for the residual private industries in Alaska (including of course the large energy sector of the state). Profit-maximizing, price-taking firms employ harvests of pollock in the fishery, non-consumptive use of marine mammals (Steller sea lions, killer whales, blue whales, sperm whales, northern fur seal and sea otter) in recreation, and capital and labor in all sectors, to produce their outputs in a continuous, nonreversible, and bounded process. Outputs from fish harvests, fish processing, recreation, and composite goods are used as intermediate goods in production by other sectors, sold in regional markets and exported out of the region, while regional production is differentiated from aggregate imports following Armington (1969). Capital K and labor L are homogeneous, perfectly mobile within the region, and defined in service units per period. Sector i factor employment levels are given by K_i and L_i ($i = FH, FP, R, C$). Firms in sector i employ factors of production and intermediate goods to produce their output, which is sold in regional markets and exported out of the region to either domestic markets, or foreign markets. Substitution between regional supply and aggregate exports is given by constant elasticity of transformation (CET) functions as are the substitution possibilities between exports to domestic markets and exports to

² The sector and regional profiles follow the Steller Sea Lion Supplemental Environmental Impact Statement (SEIS, U.S. Dept. of Com., 1991).

foreign markets. Firms smoothly substitute over primary factors through constant elasticity of substitution (CES) functions, but employ intermediates in fixed proportions through Leontif functions.

Households are differentiated by their income category and demand composites of regionally produced goods and imports, where imperfect substitution possibilities are given by nested CES functions. Household demands are governed by CES functions over composite goods, where the price consumers' face are indices of aggregate import and domestic prices, with domestic and foreign import prices taken exogenously. Substitution possibilities in demand between foreign and domestic imports are also governed by CES functions. Household incomes are derived through a two-stage process. Households are endowed with varying amounts of labor and capital. These factors are exchanged in factor markets, and through production generate value added. Value added expenditures flow first to the factor "institutions", and are then redistributed to households. Total factor payments to households are net of factor taxes, depreciation allowances, rents attributable to the factor, and labor payments out-of-region.

Government behavior is admitted to the model at both Federal and State levels and through fishery regulation. Both governments operate according to a balanced budget, produce and consume goods and tax trade related activity. Government revenues are from taxes (indirect business taxes, primary factor taxes, and income taxes all taken fixed as proportions of output), sales of governmentally produced commodities, and government borrowing and transfers. These revenues are then redistributed in lump sum to both consumers and producers.

Equilibrium conditions follow Ballard et. al (1985) and de Melo and Tarr (1992). Model closure follows Waters, Holland and Weber (1997) and Coupal and Holland (2002) where exogenous endowments are specified to balance the regional current account and the regional investment savings balance.

Welfare measures follow Ballard et al. Annual equivalent variations EV_t measure welfare changes for any single period across policy scenarios. Cumulative aggregate welfare measures are found using discounted³ summations of EV_t given by P_{EV} , which provide a comparative measure as they based upon a common baseline price vector.

The novel linkages between the ecosystem, fish harvesting, recreation, and the factor market implications are presented in detail. The remainder of the model specification follows standard methods as given by Ballard et al (1985) and de Melo and Tarr (1992) and is presented in brief.

Fish Harvesting Incorporating a fishery into a CGE framework raises issues that require two modifications to the standard fishery models. First, where most of the fishery literature employs effort as the single human factor of production,⁴ capital and labor must be included in CGE so that the fishery interacts with other sectors. Second, the non fishery sectors hire capital and labor in service units per time period, in this case one year, but in the fishery factors are employed considerably less than one year and may earn more per unit than in other sectors. This is due to restricted season lengths and potential unemployment during the off season. The divergence in fishery factor prices are termed compensating differentials in the fashion of the unemployment risk differentials of Averett, Bodenhorn, and Staisiunas (2003) and similar to the factor price

³ By consumers' rate of time preference.

⁴ Exceptions include Boyce (2004) and Weninger and McConnell (2000), who present an analysis that deliberately separates fixed and variable inputs in a partial equilibrium model of a fishery regulated through TAC's and a limited entry program.

differentials of Kwon and Paik (1995).

Expressions (2.1) – (2.4) summarize production in the fishery sector:⁵

$$TAC = a + bN_4 \quad (2.1) \quad H_{FH} = d_F T^{a_F} N_4 \quad (2.2)$$

$$\text{minimize } \hat{w}L_{FH} + \hat{r}K_{FH} \quad \text{subject to } T = d_F^m L_{FH}^{a_F^m} K_{FH}^{(1-a_F^m)} \quad (2.3)$$

where time subscripts are omitted for brevity. Equation (2.1) introduces government into the model in the form of a fishery manager. Homans and Wilen (HW, 1997) developed a model of a regulated open-access fishery to reflect that fishery managers set total allowable catch, TAC , and fishing season length, T . The heavily-regulated Alaskan pollock fishery fits this institutional arrangement. To mesh an HW type model with the CGE framework, the fishery manager chooses period t 's TAC according to (2.1) where N_4 is the population of pollock. No harvests are allowed whenever the actual biomass is less than the minimum level set by the manager. For given TAC and technology, the season length is determined from the aggregate harvest function in (2.2), where a_F and d_F are parameters and H_{FH} is aggregate harvest. The industry is assumed to harvest up to their limit so that $H_{FH} = TAC$.

The season length is the time needed to land the TAC (and thus a measure of effort) given the fish stock and is increasing in TAC (HW, 1997). Following the fishery manager's choices for TAC and T , the industry is assumed to minimize the cost of harvesting according to (2.3) by employing capital and labor to work time T . The production function exhibits constant returns to scale, a_F^m , and d_F^m are parameters, and \hat{w} and \hat{r} are the fishery wage and rental rate of capital that may diverge from the market wage and rental rate in other sectors. The associated cost function is linearly homogenous in time, allowing the total costs of harvesting to be written as $C(\hat{w}, \hat{r})T$.

The divergence of fishery factor prices from market factor prices in the other sectors is endogenous to the system and arises from the restricted season length. Entry is assumed to dissipate all rents in open access models. But these are partial equilibrium models and factors are either not defined over time or if they are defined, they are instantaneous rates or daily rates as in Clark (1976). What these factors are doing off season is not an issue, because there is no off season in the models. In the CGE setting where all other sectors are operating year round, the fishery experiences an off season during which factors are either unemployed or employed elsewhere, often outside the region. In reality, unemployment is common and it may be either voluntary, or involuntary owing to factor immobility between seasons.⁶ In either case, rational factors may demand higher than market payments in season in anticipation of being unemployed off season. If they do, seasonal factor payments will not be driven down to market levels in season, leaving positive seasonal factor price differentials.⁷ One might argue that these above market payments are not really compensating differentials, because they are merely covering the opportunity costs of factors in the off season. This is certainly not true for voluntary unemployment because the factors are enjoying leisure. But even for involuntarily unemployment, rational factors will anticipate some transition time before reemployment, and will enjoy factor price differentials if the

⁵ We are indebted to Robert Deacon for his invaluable input in the development of this section.

⁶ The Alaskan Department of Labor and Workforce Development provides information about fishing jobs in Alaska on various websites (e.g., http://www.labor.state.ak.us/esd_alaska_jobs/careerstreams.htm). The job descriptions suggest that workers can save money, and pay can be substantial if the fishing is good. College students are encouraged to apply and then return to college in the off season. Boyce (2004) examines rents in fisheries and assumes that fishing inputs cannot be redeployed during the off season.

⁷ Chen and Edin (2002) investigate inter-industry wage differentials for time wages in comparison to piece rates and make the argument that differentials are more likely to exist for time wages, consistent with the definition here.

transition time is equal to or less than what they anticipate.

Let W and R be the market determined factor prices for labor and capital in other sectors. Because labor and capital are defined in service units per year, W and R are annual payments. Let $\beta \in (0,1)$ be the percent of the year the fishery is active so that market factor prices in the fishery are βW and βR . If there are intra-seasonal differentials in the fishery, they must be reflected in factor prices that deviate from market prices such that $C(W\beta, R\beta) < C(\hat{w}, \hat{r})$. Assuming any differentials impact labor and capital uniformly and linearly, let δ be a factor price divergence term so that the factor prices in the fishery are:

$$\hat{w} = \beta\delta W \quad \text{and} \quad \hat{r} = \beta\delta R \quad (2.4)$$

where $\delta = 1 \Rightarrow$ no differentials and $\delta > 1 \Rightarrow$ positive differentials.⁸

In developing the simulation model the available data provides estimates for β and δ . But the data is inadequate to determine whether factors were voluntarily or involuntarily unemployed or whether they were reemployed during the off season. Therefore, without strong evidence one way or another and for demonstration purposes the assumption made here is that labor is voluntarily unemployed, i.e., enjoying leisure, and capital is idle or employed outside the Alaskan economy. Labor's leisure time in the off season is not admitted to the welfare measures and remains a focus of future work.

Equilibrium for the industry is given by a pseudo zero-profit condition that allows for intra-season factor price differentials:

$$\pi_{FH} = PD_{FH} H_{FH} - C(\beta\delta W, \beta\delta R)T = 0 \quad (2.5)$$

where PD_{FH} is the unit price of regionally produced goods. In this representation, total factor payments over the season equal total revenue divided by season length, or average revenue per time. An exogenous increase in TAC or H_{FH} increases season length for a given fish stock and δ falls to maintain equality in (2.5). Intuitively, the longer season implies less off-season time for the factors, and they require a smaller factor price differential in season to get through the off season. To summarize, after the fishery manager sets TAC by (2.1) and T by (2.2), the factor demands and the factor price divergence δ are endogenous and determined by (2.3) and (2.5).

The fishery has a significant impact on welfare changes from its influence on factor payments. To illustrate, consider labor payments. Let \bar{L} be the annual labor stock and $L_{FH}(T, \hat{w})$ the labor factor demand function from the fishery cost minimization in (2.3). Labor in other sectors is the residual $(\bar{L} - L_F)$. Annual household labor payments are:

$$\text{labor payments} = \hat{w}L_{FH}(T, \hat{w}) + W[\bar{L} - L_{FH}(T, \hat{w})] \quad (2.6)$$

where $\hat{w} = W\beta(T)\delta(T)$. Of course, W also depends on season length, but for simplicity in this analytical treatment, assume that the change in the market wage for other sectors is small and can be ignored (this is not the case in the simulations). The first term in (2.6) is the payment to fishery labor, and the second term is the payment to non-fishery labor.

Figure 2 (also employed in the results) demonstrates the relationship between

⁸ Factor price distortions commonly enter the CGE literature in the form of pre-existing distortions and taxes (Shoven and Whalley, 1992, Ballard et al., 1985, and Bovenberg and Goulder 1996). The divergences here are not distortions in the usual sense and correspond more to the factor price differentials of Kwon and Paik (1995): β is merely an accounting adjustment to correct for a shorter work year, and a $\delta > 1$ may be welfare enhancing since some positive differentials are desirable.

fishery factor payments, market factor payments, and δ . In a steady state, let T^0 be the season length and the average revenue per time from (2.5) is downward sloping as shown by the solid line. At T^0 factor payments are $C(\hat{w}^0, \hat{r}^0)$ which exceeds market-based factor payments $C(W, R)$ owing to the compensating differentials. For a discrete reduction in TAC , because the fish stock has not changed, the season length falls to T^1 . With the decrease in season length, ΔT , the change in fishery labor is:

$$\frac{\Delta L_{FH}}{\Delta T} = \frac{\Delta L}{\Delta T} + \frac{\Delta L}{\Delta \hat{w}} W \frac{\Delta \delta}{\Delta T} > 0 \quad (2.7)$$

The shorter season means less labor and capital in the fishery, but these remaining factors enjoy a higher differential ($C(\hat{w}^1, \hat{r}^1) - C(W, R)$) per time employed as δ adjusts upward. Thus in (2.7) when T goes down less labor is needed to produce fish (first term), and less labor is demanded because the wage differential increases and labor is more expensive (second term). From (2.6), the change in labor payments is:

$$\frac{\Delta \left(\begin{matrix} \text{labor} \\ \text{payments} \end{matrix} \right)}{\Delta T} = \left[W \delta(T) \beta(T) \frac{\Delta L_{FH}}{\Delta T} + L_{FH}(\cdot) W \left[\delta \frac{\Delta \delta}{\Delta T} + \beta \frac{\Delta \beta}{\Delta T} \right] - W \frac{\Delta L_{FH}}{\Delta T} \right] \quad (2.8)$$

with signs given below terms. For a reduction in TAC and a shorter season, the first term is the decrease in fishery payments owing to labor exiting the fishery. The second term is the change in payments to the remaining fishery labor. Its sign is ambiguous because it shows that while payments per labor unit are higher owing to a greater wage differential, they are made over a shorter season. The third term is the increased payments in other sectors as exiting fishery labor joins these sectors. The results depend on the relative magnitudes of the terms, and are in general ambiguous (although calculated in the numerics).

Recreation and Tourism The importance of marine mammals to the state's tourism sector is reflected through linkages between their populations and the sector's cost function. While mammals are non-use inputs to the sector, the more abundant the populations the cheaper it is for the sector to provide each unit of tourism, which equivalently increases the productivity of factors employed in the sector. For tourism and all sectors other than pollock harvesting, each industry's primary factor cost PVC_j function can be found as:

$$PVC_j = (\phi_j)^{-1} QD_j \left[\delta_j^{\sigma_j} W^{1-\sigma_j} + (1-\delta_j)^{\sigma_j} R^{1-\sigma_j} \right]^{1/(1-\sigma_j)} \quad (2.9)$$

where δ_j are primary factor value added distribution parameters, σ_j are partial elasticities of substitution in value added and ϕ_j are industry specific value added efficiency parameters. Changes in marine mammal populations change recreation's cost function through the specification of the efficiency parameter $(\phi_R)^{-1} = \Delta^{MM} (\phi_R)^{-1}$ where Δ^{MM} is the percentage change in recreation costs induced by the population changes and ϕ_R is the sector's efficiency parameter in the absence of any cost impacts. Recreation costs are influenced by marine mammal population changes (in aggregate given by Δ^N) according to an elasticity ε_{MM}^R such that $\Delta^{MM} = 1 + \varepsilon_{MM}^R \Delta^N$. Annual marine mammal population changes are given by the average percentage change in species populations of killer

whales, blue whales, sea lions, sea otters, northern fur seals and sperm whales from their benchmark levels:

$$\Delta^N = \left(\sum_{\substack{\text{marine} \\ \text{mammals}}} \text{Annual \% change in population} \right) / \# \text{marine mammals} \quad (2.10)$$

As the annual % changes are calculated from the benchmark population level, increases in populations lower recreation unit costs, and decreases in populations increase unit costs, depending on the magnitude of ε_{MM}^R .

3 The Ecology Model

All energy in the ecosystem, shown in Figure 1, originates from the sun and is turned into biomass through plant photosynthesis. Photosynthesis is carried out in the AI by individuals of various species of algae, or kelp, and in the EBS by individuals of various species of phytoplankton. All individual animals in the system depend either directly or indirectly on the kelp and phytoplankton plant species. The ecosystem is an extension of the food-web in Finnoff and Tschirhart (2003) with five added species as well as multiple predator-prey interactions. The choice of additional species and design of the energy flows are based on findings from across the ecological literature, and include such sources as Lowry et al (1982), Perez and McAlister (1993), and Williams et al (2004).

GEEM combines two disparate ecological modeling approaches: optimum foraging models and dynamic population models. The former approach has been likened to consumer theory (Stephens and Krebs, 1986) and describes how individual predators search for, attack and handle prey to maximize net energy intake per unit time. Optimum foraging models do not account for multiple species in complex food webs and do not track species population changes. Dynamic population models track population changes by using a difference or differential equation for each species. The familiar logistic-growth model used extensively in the economics literature is the simplest example, although extensions include resource competition models (e.g., Gurney and Nisbet 1998) and the Lotka-Volterra predator/prey model and its variations. However, the parameters in the dynamic equations represent species-level aggregate behavior: optimization by individual plants or animals or by the species is absent. Alternatively, GEEM employs optimization at the individual level as in foraging models, and uses the results of the optimization to develop difference equations that track population changes.

In GEEM, demand and supplies are developed somewhat similarly to CGE. Species are analogous to industries, and individual plants and animals are analogous to firms. Plants and animals are assumed to behave as if they maximize their fitness net energy flows. Where perfectly competitive firms sell outputs and buy inputs taking market-determined prices as signals, plants and animals transfer biomass from prey to predators taking 'energy prices' as signals. (Plants can be thought of as preying on the sun.) An energy price is the energy a predator loses to the atmosphere when searching for and capturing prey. A key difference between economic markets and ecological transfers, however, is that in the latter the prey does not receive this energy price. Therefore, the biomass transfer is not a market because there is no exchange. Nevertheless, predators' demands and preys' supplies are functions of the energy prices.

The application of GEEM here follows Tschirhart (2002, 2003, and 2004) and Finnoff and Tschirhart (2003). The three basic equations that comprise GEEM are

given by (3.1) – (3.3). The first equation is a general expression for the fitness net energy flow through a representative animal from species i .

$$R_i = \sum_{j=1}^{i-1} [e_j - e_{ij}] x_{ij} - \sum_{k=i+1}^m e_i [1 + t_i e_{ki}] y_{ik} - f^i \left(\sum_{j=1}^{i-1} x_{ij} \right) - \beta_i \quad (3.1)$$

$$N_i x_{ij}(\mathbf{e}) = N_j y_{ji}(\mathbf{x}(\mathbf{e})) \quad (3.2)$$

$$N_i^{t+1} = N_i^t + N_i^t \left[\frac{1}{s_i} \right] \left[\frac{(R_i(\cdot) + v_i)}{v_i^{ss}} - 1 \right] \quad (3.3)$$

R_i is in power units (e.g., Watts or kilocalories/time).⁹ The species in (3.1) are arranged so that members of species i prey on organisms in lower numbered species and are preyed on by members of higher numbered species. The first term on the right side is the inflow of energy from members of prey species (including plants) to the representative individual of species i . The choice variables or demands, x_{ij} , are the biomasses (in kilograms/time) transferred from the member of species j to the member of species i ; e_j are the energies embodied in a unit of biomass (e.g., in kilocalories/kilogram) from a member of species j ; and e_{ij} are the energies the member of species i must spend to locate, capture and handle units of biomass of species j . These latter energies are the energy prices. There is one price for each biomass transfer between a predator and prey species. As in economic CGE models, the prices play a central role in each individual's maximization problem, because an individual's choice of prey will depend on the relative energy prices it pays. Individuals are assumed to be price takers: they have no control over the energy price paid to capture prey, because each is only one among many individuals in a predator species capturing one of many individuals in a prey species.

The second term is the outflow of energy to animals of species k that prey on i . The e_i is the embodied energy in a unit of biomass from the representative individual of species i , and y_{ik} is the biomass supplied by i to k . The term in brackets is the energy the individual uses in attempts to avoid being preyed upon. It is assumed to be a linear function of the energy its predators use in capture attempts: the more energy predators expend, the more energy the individual expends escaping. t_i is a tax on the individual because it loses energy above what it loses owing to being captured. The third and fourth terms in (3.1) represent respiration energy lost to the atmosphere which is divided into a variable component, $f^i(\cdot)$, that depends on energy intake and includes feces, reproduction, defending territory, etc., and a fixed component, β_i , that is basal metabolism.

Time in the Alaskan ecosystem is divided into yearly reproductive periods. Each year a general equilibrium is determined wherein the populations of all species are constant, each plant and animal is maximizing its net energy (using the derivatives of (3.1) for first-order conditions), and aggregate demand equals aggregate supply between each predator and prey species. For each price that equates a demand and supply transfer there is an equilibrium equation given by (3.2). Each plant and animal is assumed to be representative individuals from its species; therefore, the demand and supply sums are obtained by multiplying the representative individual's demands and

⁹ According to Herendeen (1991) energy is the most frequently chosen maximand in ecological maximization models, and energy per time maximization as adopted here originates with Hannon (1973), and expanded in Crocker and Tschirhart (1992) and to the individual level in Tschirhart (2000). Energy per time is also the individual's objective in the extensive optimum foraging literature (e.g., Stephens and Krebs 1986).

supplies by the species populations given by the N terms.

A representative plant or animal and its species may have positive, zero or negative fitness net energy in equilibrium. Positive (zero, negative) net energy is associated with greater (constant, lesser) fitness and an increasing (constant, decreasing) population between periods. (The analogy in a competitive economy is the number of firms in an industry changing according to the sign of profits.) Fitness net energies, therefore, are the source of dynamic adjustments. If the period-by-period adjustments drive the net energies to zero, the system is moving to stable populations and a steady state. The predator/prey responses to changing energy prices tend to move the system to steady state.

The adjustment equation for the i^{th} species (a top predator in this case) is given by (3.3) where $R_i(\cdot) = R_i(x_i; \mathbf{N})$ is the optimum fitness net energy obtained by substituting the optimum demands and supplies as functions of energy prices into objective function (3.1). \mathbf{N}^t is a vector of all species' populations and it appears in $R_i(\cdot)$ to indicate that net energies in time period t depend on all populations in time period t . In the steady state, $R_i(\cdot) = 0$. Also, s_i is the lifespan of the representative individual, v_i is the variable respiration, v_i^s is the steady-state variable respiration, and N_i^s is the species steady-state population. The first and second terms in brackets in (3.9) are the birth and death rates. Expression (3.9) reduces to the steady state if $R_i(\cdot) = 0$ (in which case $v_i = v_i^s$ and $N_i^t = N_i^s$). Because the biomass demands depend on the period t populations of all species, the population adjustment for species i depends on the populations of all other species. In addition, out of steady state $R_i(\cdot)$ and v_i change across periods. These changes distinguish the GEEM approach from most all ecological dynamic population models, because the latter rely on fixed parameters in the adjustment equations that do not respond to changing ecosystem conditions.

4 Model Specification

Economic Data and Specification The economic specification is based on a benchmark of 2001. The benchmark dataset is taken from an IMPLAN (MIG, 2004) derived social accounting matrix SAM and is available in the Appendix on request. All parameters estimated and calibrated as detailed below are also available in the Appendix upon request.

Though deficient in many areas, IMPLAN data is the industry standard, it is comprehensive and provides a decent building block on which to add. The industry sectors were aggregated down from IMPLAN's 509 sectors to 5 sectors. These sectors consisted of pollock harvesting, (all other) fish harvesting, fish processing, miscellaneous (a catch all for all other productive sectors in the state), and recreation and tourism. IMPLAN differentiates households according to income class, and this classification was maintained. The federal government's interactions with the state were kept distinct while city, county, state and governments were also aggregated into a single institution labeled state and local government. Given the importance of trade flows into and out of Alaska from both the rest of the United States and foreign countries, foreign and domestic trade were also differentiated.

The base IMPLAN dataset necessarily underwent substantial modification. First, the factors of labor and capital were disaggregated. IMPLAN's employee compensation account was used to construct the labor account. Capital was similarly found as the summation of proprietary income and other property income.

Second, pollock harvesting was differentiated from all other fish harvesting by dividing all Alaskan fish harvesting into that attributable to BSAI pollock harvests and all else. BSAI pollock harvests in 2001 accounted for 29.77% of the value of all landings in Alaska (Hiatt et al, 2004).

Third, the recreation and tourism sector was added using a methodology similar to Colt (2001). The recreation and tourism industry in Alaska is assumed to be the production within Alaska of outdoor recreation services that are consumed by Alaskan residents and nonresidents (tourists or visitors traveling for pleasure). Nonresident recreation and tourism activities are defined to be nonconsumptive and include sightseeing, hiking, camping, rafting, kayaking, (etc.) and related shopping, while resident recreation is based on resident expenditures for wildlife viewing trips.

Colt found each visitor to Alaska spending \$782 per trip for vacation and or pleasure (Table 11, page 23). Given 1.2 million total summer visitors to Alaska in 2001¹⁰ and applying the percentage (76.2%) of Colt's total visitors who were in Alaska for vacation and or pleasure to this total, the total expenditures by non-residents for tourism and recreation were estimated to be \$715.53 million.

For resident expenditures on Wildlife viewing, Colt found 14% of the Alaskan population took 84,000 primary wildlife-viewing trips and 143,000 secondary wildlife viewing trips, or 1.65 primary trips per trip taker and 2.815 secondary trips per trip taker. Each primary trip taker spent \$226 on miscellaneous expenses and an additional \$591 per primary trip (Table 10, page 18), while secondary trip takers spent \$410 per trip. To calculate a lower bound estimate of the total expenditures by residents on wildlife viewing, Colt's expenditure data was implemented by using the number of Alaska households from IMPLAN (262,995) as the relevant population. Estimating that 14% of this population (36,819.3) took wildlife viewing trips in 2001, each of these trip takers was then assumed to have taken 1.65 primary trips and 2.815 secondary trips. Applying the trip related expenses as discussed above the total resident wildlife view expenditures were found to be \$92.13 million. Taking the non-resident and resident expenditures together, the recreation and tourism sectors output was found to be \$807.66m. All intermediate demands, factor demands and indirect business taxes were found as the same proportion as those in the miscellaneous sector.

The final modifications to the IMPLAN SAM involved manipulations of the implausible fish processing capital account. Capital payments in this sector were found as the net of the Bureau of Economic Analysis (BEA's) estimate of the gross state product for food manufacturing in Alaska for 2001 (\$328 million) from the value of labor in the industry (\$301.5 million) as calculated by Chang Seung (AFSC) from BEA data.

The fishery is parameterized through a mix of estimation and calibration. Parameter estimates for the quota and harvest functions are estimated using time series of data on quotas, biomass, harvests, and inputs. Calibration is employed for remaining parameters.

Unlike prices in the non-fishery sectors (which are set to one) prices in the fishery are set according to (2.4). β is obtained by noting that in 2001 the season was 240 days. If the total potential season length for any given year is taken as 365 days, $\beta = 240/365 = 0.6575$. Labor data is used to obtain δ , and capital is assumed to

¹⁰ Leisure & Hospitality , Alaska Economic Trends, January 2004. <http://www.labor.state.ak.us/research/trends/jan04ing.pdf>

exhibit the same divergence. The BSAI pollock fishery employed 2094 full-time equivalent workers (FTE) in 2000 (the closest year available, Northern Economics Inc. and EDAW, 2001) with total labor bill of \$210,612,500, yielding \$100,579 per FTE. The annual wage per FTE in all Alaskan sectors for 2001 was \$39,168.¹¹ The differential is therefore $\delta = \$100,579/39,168 \approx 2.57$. Finally, the factor payments for labor and capital are $\beta\delta W = \beta\delta R = 1.69$ indicating that fishery factor payments relative to the payments in other sectors are offset downward owing to the shorter season, but overwhelmingly offset upward owing to the factor price differential.

We parameterize the quota function in a similar fashion to Finnoff and Tschirhart (2003). Using biomass estimates, total allowable catches and actual catches from Witherell (2000) we estimated the quota function (with additive errors assumed iid) by ordinary least squares using LIMDEP software. Estimating the quota function presented difficulty due to fairly constant *TAC*'s since the fisheries inception (with relatively stable, large populations). In this case the safe minimum population is very low (unrealistically) just because of the historical records having fairly uniform harvests for high population levels. Normalizing the data around its "lower left hand value" (ie the lowest biomass harvest combination) shifts both axes to this point in the data. The quota function was then estimated for this "normalized range". We corrected for first order autocorrelation found in the quota function, and obtained significant parameter estimates employed in the reported results. This provides a quota function for populations greater than the normalization point. For populations less than the normalization point, 1000000 tons was arbitrarily chosen as reasonable. Given this, a linear function was employed between this point and the normalization point. We corrected for first order autocorrelation found in the quota function, and obtained significant parameter estimates.

The harvest function is estimated using a linear transformation of (2.1) and a time series of harvests, aggregate effort, and biomass estimates gathered from Northern Economics Inc. and EDAW (2001), Northern Economics (2001) and Witherell (2000). OLS regressions directly yield a_F , and allowed d_F to be recovered from the estimate of the intercept¹².

Most other parameters (apart from elasticities of substitution) are found through calibration as in Ballard et al. and De Melo and Tarr (1992). The calibration routine sets benchmark input and output prices equal to unity (by constant returns to scale and the units of the initial data being in value terms). Using all first-order conditions from profit maximization, cost minimization, and utility maximization; and the benchmark data and prices, most parameters apart from the elasticities of substitution are found.

Calibration equations for household behavior and all sectors other than pollock harvesting are exactly the same as Ballard et al. (1985) and De Melo and Tarr (1992) and omitted. Calibrated CES and CET parameters are found given the benchmark data and estimates of elasticities of substitution and transformation taken from de Melo and Tarr (1992). All households are assumed to have an elasticity of substitution between consumption goods of 0.9.

Ecological Data and Specification In applying GEEM to the Alaskan ecosystem,

¹¹ Institute of Social and Economic Research, University of Alaska Anchorage, 2003.
http://citizensguide.uaa.alaska.edu/4.COMPARISON_TO_OTHER_PLACES/4.5.Public_Employees_and_Wages.htm
http://citizensguide.uaa.alaska.edu/4.COMPARISON_TO_OTHER_PLACES/4.5.PublicEmployeesandWages.htm

¹² Only catcher processor *CP* results were significant, providing an estimate of a_F of 0.8997 with standard error 0.2324 (p-value of 0.0061) and an adjusted R^2 of 0.636. We assume both harvesting sectors have a homogeneous technology.

ecological studies of the Alaskan and other ecosystems were used. A reasonable time series of pollock biomass estimates exists for the period 1966 through 1997, and the rest of the data are from 1966 or interpolated to that date. Data were obtained for plant and animal populations, benchmark plant biomasses and animal biomass demands, and parameters that include embodied energies, basal metabolisms, and plant and animal weights and lifespans. Benchmark data and details of data sources are available in the Appendix.

The benchmark dataset is assumed to be an observed equilibrium and it is employed in calibration to determine parameter estimates in the plant and animal respiration and supply functions (Finnoff and Tschirhart, 2003). Calibration consists of simultaneously solving for each individual the net energy expressions set to zero, first-order conditions or the derivatives of the net energy expressions set to zero, and the demand/supply clearing conditions. In this a critical assumption is made, namely that the benchmark data (generally from around 1980) represents an equilibrium in the absence of harvesting. Given the ecosystem calibration (in the absence of pollock harvests), harvesting according to the quota function was then initiated and the ecosystem run to a steady state. This steady state (with human intervention) provides the benchmark ecosystem data for the jointly determined model with a base year of 2001.

5 Policy Analysis

The NMFS in 2001 issued a Supplemental Environmental Impact Statement (SEIS) containing alternative management strategies that specify various pollock catch limits and no fishing zones to protect both the sea lions and the fishery. Using the jointly determined CGE models, the effects of the management strategies on economic welfare are examined, and extended to investigate the value of marine mammals to recreation.

Rules for catch limits are given by the regulator's choice of b in the quota function (2.1). Holding N_4^{\min} constant, b is reduced to 50% of its 2001 harvest levels. (Numerous other harvest levels were examined but not reported. The 50% results are indicative of all runs below the benchmark harvest.) All general equilibrium calculations and population updates were made with the nonlinear programming software package GAMS.

The benchmark sequence is initiated using the 2001 benchmark data, then simulated for 50 years.¹³ In the benchmark sequence, all quantities evolve at a constant rate, but the rate may vary over sectors owing to the reliance of the fishery and recreation sectors on biological natural resource inputs. Further, given heterogeneous growth of the natural resources, benchmark relative prices do not remain constant.

Economic Impacts The direct impacts of the management strategies are on the pollock fishery sector and pollock population, although indirect impacts reverberate throughout both systems. In understanding the economic consequences it is useful to first quickly view the underlying ecological impacts. Percentage changes in ecosystem populations from their benchmark levels are shown for the fifty years of the policy in Figure 3. Reduced pollock harvests result in long-term increases in pollock, stellar sea lions, killer whales, Pacific cod, sperm whales, fur seals, blue whales, phytoplankton and sea urchins. But a policy of reduced pollock harvests also finds long-term decreases in zooplankton, herring, kelp, and sea otters. The recreation sector benefits from more sea lions, killer whales, sperm whales, fur seals, and blue whales but is hurt by fewer sea otter.

¹³ Sequence length and a discount rate of 3% were chosen as representative for Federal projects and the prevailing inflation rate.

Welfare changes (from the benchmark sequence) presented in Tables 1 and 2 are the present value of the cumulative sum of equivalent variations P_{EV} over the planning horizon for all Alaskan households (Table 1) and by household income class (Table 2). Across a range of recreation ecosystem input elasticities ε_{MM}^R , decreasing the quota always results in cumulative welfare losses for Alaskan households in aggregate. The more elastic unit costs of recreation are to changes in ecosystem inputs (Δ^N the average percentage change from their benchmark levels in 2001 of killer whales, blue whales, sea lions, sea otters, northern fur seals and sperm whales populations) then the lower the welfare loss associated with reducing the pollock quota rule by 50%. This reduction in welfare loss is by almost \$120 million if $\varepsilon_{MM}^R = -1$.

By household income class (Table 2) and across ε_{MM}^R 's, lower income households experience welfare gains from the policy, while higher income classes (\$25K and up) are the source of the aggregate welfare losses. These household specific impacts are due to price and income effects differentially impacting households. A larger proportion of income of lower income households comes from labor, while a larger proportion of the income of higher income households comes from capital. Reducing the pollock quota causes labor and capital to be released from pollock harvesting and reemployed in the miscellaneous and recreation sectors. But, given the factor price differential in the pollock fishery, while factors released are now reemployed in sectors for the entire year, they forgo the fishery factor price differential, lowering total payments to labor and capital. Average annual percentage changes in key economic variables from the benchmark are available upon request in the Appendix.

Coupled with declining factor payments and given the factor proportions in the fishery, relatively more capital is released than labor. This creates a larger relative surplus of capital such that the price of capital declines by more than the decline in the wage rate. Thus lower incomes do not decline by as much as higher incomes. Couple these income effects with all prices other than that for pollock falling, and lower income households are able to increase their consumption of most goods (other than pollock). Higher income households on the other hand not only have their incomes reduced but they are also forced to reduce their consumption of all goods. Regardless of household income, more elastic ε_{MM}^R 's attenuate the welfare changes (good or bad).

Figure 2 again helps shed light on the fishery's contribution to the welfare changes. If T^0 is the season length in the first period of the benchmark, under the 50% pollock quota rule reduction the fishery manager lowers harvests and because the fish stock has not changed, the season length falls to T^1 . The shorter season means less labor and capital in the fishery, but these remaining factors again enjoy a higher differential as δ adjusts upward. Shorter seasons result in fewer fishery workers who enjoy higher differentials per time worked, while the workers who leave the fishery are employed at market wages in other sectors for the full year.

In the second period following the reduced harvest strategy the fish population is greater and the price of fish is higher. Both changes cause the average revenue curve to shift upward. The fishery manager sets a greater TAC by (2.1) because of the greater fish population, and the season length increases to T^2 although it is less than the initial season length. δ adjusts downward and the differential falls to $C(\hat{w}^2, \hat{r}^2) - C(W, R)$. Some workers now return to the fishery from the other sectors, leaving their full-year market wages for higher part-year wages. In addition, because the fish population is

greater, the fishery factors are more productive. Over the remainder of the planning horizon, the season lengths remain between T^0 and T^1 and the factor price differential remains between the initial low value and the second period high value.

The lower TAC results in a substantial increase in the regional price of pollock. Imports of pollock fall with the increased composite price, but not by as much as the decline in regional production given the relatively inexpensive import price. Although total pollock output falls, the proportion supplied for domestic consumption increases with high regional price, and the proportion exported declines by more than the decline in regional production.

Regional prices of (other) fish harvests and fish processing decline with the cheaper factor prices, but not by as much as the decline of prices in the miscellaneous and recreation sectors. This attenuation of the price decline for harvests and processing is due to their large dependence on intermediate flows of pollock (which is of course higher for processing, evidenced in its price decline being the slightest). These price effects cause regional demands for (other) fish harvests and processing to decline substantially.

The repercussions of the policy and factor market reallocations cause regional prices of recreation and miscellaneous goods to fall. Miscellaneous production increases as it becomes relatively inexpensive. Domestic demand falls as do imports, while exports increase given lower regional prices.

Recreation production also increases with the $\varepsilon_{MM}^R = 0$ case. But, for the non zero ε_{MM}^R cases, recreation production is aided by smaller fish harvests that yield greater populations of marine mammals (increasingly so for more elastic ε_{MM}^R 's). These productivity improvements lower unit costs and allows the price of recreation to fall further. As relatively less expensive, domestic recreation demand rises for all non-zero ε_{MM}^R cases, but by less than the rise in exports (which have a higher price).

Under the policy of reduced harvests, with state and federal government revenues fall. To offset these declines, both federal and state governments have to borrow funds to cover the deficits. More elastic ε_{MM}^R 's though work to attenuate the tax revenue declines lowering the amounts both state and federal governments have to borrow.

Ecosystem Impacts Figure 3 shows that all populations move to new steady states, most very quickly but the large mammals taking far longer (blue whales taking 50 years to approach a steady state).¹⁴ To appreciate the general equilibrium nature of the population changes, the immediate affect of the higher harvest is to increase the pollock population. In the subsequent period the higher population lowers the energy prices for all of the pollock predators. Lower energy prices for Pacific cod, stellar sea lions, northern fur seals, and killer whales prompt higher demands, net energies and eventually populations. These changes work their way up the food web, and the further up the food web from pollock, the less pronounced the impact.

The power of the ecosystem model is clearly shown by inspecting herring populations. Herring populations fall with the reduced pollock harvests for a combination of reasons. First, as pollock populations rise, while per capita pollock

¹⁴ Average lifespan enters into the population update equation, (3.3), similar to the way the less tangible species growth rates enters into the often-used but simplistic logistic update equation; thus, the lifespans are important in determining whether population oscillations occur and how quickly populations will converge to steady state.

demand for zooplankton falls with increased intra-species competition, as there are more pollock the inter-species competition forces the price herring pay for zooplankton to rise. This causes the demand by herring for zooplankton to fall. These higher prices and lower demands for herring imply lower net energies and declining herring populations. But, there are also the effects of predation on herring to account for. Namely, per capita biomass demands for herring by northern fur seals, stellar sea lions, and killer whales all fall. But, only energy prices paid by stellar sea lions and northern fur seals rise (as each of these populations rises). The killer whale energy price for herring on the other hand falls because the relative populations of all other killer whale prey rise in relation to the declining herring population. Although this price falls, killer whale demand actually also falls because the prices of all its other prey decline by more, making herring relatively more costly to killer whales in relation to other prey.

These predation effects would appear to imply that herring populations should rise. But when the population changes of predators are taken into account as well as the increased competition for zooplankton herring face, pressures on herring overwhelm positive impacts and the population declines.

Down the food web from pollock, zooplankton experience more predation risk owing to more pollock and other predators. Although they demand more phytoplankton per capita, in aggregate total demand falls with a smaller population. General equilibrium style feedbacks mitigate to some degree the reduced harvesting-induced rise in the pollock population. That is, larger predator populations and smaller zooplankton populations mean respectively more predation on, and less prey for, the pollock.

The impacts of harvesting are also felt on the other side of the food web. As shown in Figure 3 panels (c) and (d). The sea otter population rose initially but then shows a long-term decrease. The changes in the otter population are small, but the signs of the change are interesting. When the energy price killer whales pay for sea lions decreases, the killer whales demand more sea lions and fewer of the substitute sea otter prey. The change in relative energy prices for prey causes, in ecological terms, *switching* behavior by the killer whales, and the killer whales consume less sea otter and more sea lions.¹⁵ Additionally, more sea lions and other BS prey consumed leads to higher killer whale net energies and higher populations, while the fewer otter consumed leads to lower killer whale net energies and lower populations. The changes in BS populations dominate sea otter population changes so that the net change in the killer whale population is positive. Individual killer whale's behavioral response to changing prey energy prices causes a short-term rise in sea otter, but increased killer whale populations cause a long-term decline in sea otter.

Marine Mammal Valuation To demonstrate the usefulness of linking the economy to GEEM, we quantify those portions of welfare changes of the reduced pollock harvesting attributable to marine mammals. The simulation with recreation marine mammal elasticity $\varepsilon_{MM}^R = 0$ provides a baseline with which to compare those with $\varepsilon_{MM}^R \neq 0$ as in it marine mammal population changes Δ^N are of no consequence for the recreation sector (although of course in the ecosystem they are changing and the pollock population is also dependent on those changes; thus by $\varepsilon_{MM}^R = 0$ the economic influence of these population changes are simply not recognized in this sector). The portion of welfare change solely attributable to changes in marine mammals can then be inferred as the

¹⁵ Killer whale switching behavior has been documented by Estes et al. (1998).

difference in periodic equivalent variations between the simulations with $\varepsilon_{MM}^R \neq 0$ and without $\varepsilon_{MM}^R = 0$ the impacts fishing has on the recreation industry for each specified ε_{MM}^R .¹⁶ This measure removes all components of the welfare changes attributable to the structure of the fishery (which is identical in both linked and non-linked scenarios).

Mean annual marine mammal valuations for alternative ε_{MM}^R are displayed in Table 12. Under decreased pollock harvesting the mean annual welfare changes are greater by \$0.97 million to \$3.4 million than if the marine mammal changes are omitted. Thus, the welfare losses of decreased pollock harvesting will be overstated if the non-consumptive use value of marine mammals were not tied to the fishery as it is in the jointly determined general equilibrium approach.

On a pseudo-marginal basis, each one percent annual change in marine mammals in relation to the baseline is worth from \$0.47 million to \$1.6 million depending on ε_{MM}^R (Table 12). While per percentage changes in marine mammals are small, it is important to remember the specification of marine mammal population changes Δ^N as the simple average across all marine mammals. Further, the total values may be understated because non-use values associated with the ecosystem (e.g., existence values) are not considered. Turcin and Giraud (2001) conducted a willingness to pay survey that asked how much households were willing to pay for continuing the Federal Steller Sea Lion Recovery Program. They found Alaskan households willing to pay in total \$25 million, and extrapolating to U.S. households the figure is \$8 billion. Interestingly, households in the area of Alaska that contains critical habitat for the sea lions were willing to pay considerably less and in some cases negative amounts. These results do not indicate the existence value for changes in the sea lion populations, but they do suggest that the value may be substantial.

6 Conclusion

Although ecosystems provide myriad services to economies, only one service is considered in most renewable-resource models. The bioeconomic model introduced here admits a second service, and more importantly it accounts for how the two services are impacted by interactions within a thirteen species ecosystem. Steller sea lion recovery measures via alternative pollock quotas are shown to cause altered levels of all ecosystem populations, economic factor reallocation, and changes in all regional prices, incomes, demands, outputs, imports and exports.

Of the thirteen species modeled, seven are used directly in the economy either as consumption goods (pollock) or non consumption goods (marine mammals). While non-use values associated with the ecosystem (e.g., existence values) are not considered, all species matter for the economy because the other seven species are used indirectly as support for ecosystem services. A portion of the regional welfare loss from reduced quotas is mitigated by an economy relying less on resource extraction and more on non-extraction. This result is consistent with a report from the Panel on Integrated Environmental and Economic Accounting which states: “economic research indicates that many renewable resources, especially in the public domain, are today more valuable as sources of environmental service flows than as sources of marketed commodities.” (Nordhaus and Kokkelenberg, 1999, p. 177).

The jointly determined CGE/GEEM modeling approach can be applied to

¹⁶ Values attributable to ecosystem inputs were found as $EV^{MM} - EV^{NMM}$, where MM refers to marine mammals being accounted for, and NMM not accounted for.

numerous other conflicts that arise when economic development and environmental conservation appear at odds. For example, in a terrestrial economy/ecosystem there may be the same consumptive and non-consumptive links used here in addition to pollution that can interfere with species respiration patterns, habitat loss that would reduce space available for plants and, therefore, food for animals, and introduced exotic species that compete with native species.

CGE models are useful in judging alternative economic policies for their effects on resource allocation and on the distribution of net benefits. The objective of jointly determining GEEM with CGE is to account for resource allocation in ecosystems as well so that the scope of policies that can be judged is broadened. To carry out the objective we have responded to two popular themes in economics and ecology: 1) that the systems are jointly determined, and 2) that both systems are general equilibrium in nature. While the economic and ecological underpinnings of this approach can be extended and improved in many ways, CGE/GEEM is a step toward integrating disciplines with common structures and goals.

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Table 1 Discounted Cumulative Welfare Impacts for all Alaskan Households
(\$ millions)

Discount Rate	\mathcal{E}_{MM}^R					
	0	-0.01	-0.25	-0.5	-0.75	-1
0	-2096.00	-2094.82	-2066.47	-2036.54	-2006.19	-1975.43
3%	-1111.39	-1110.83	-1097.16	-1082.75	-1068.15	-1053.36

Table 2 Discounted Cumulative Welfare Impacts by Alaskan Households
(\$ millions)

Discount Rate = 0		\mathcal{E}_{MM}^R				
Household Income	0	-0.01	-0.25	-0.5	-0.75	-1
< \$10K	87.27	87.24	86.43	85.58	84.71	83.82
\$10-15K	37.38	37.37	37.11	36.82	36.53	36.23
\$15-25K	21.84	21.84	21.89	21.94	21.98	22.02
\$25-35K	-12.15	-12.13	-11.63	-11.10	-10.57	-10.04
\$35-50K	-178.57	-178.46	-175.81	-173.02	-170.19	-167.33
\$50-75K	-582.53	-582.22	-574.80	-566.96	-559.00	-550.93
\$75-100K	-476.58	-476.32	-470.20	-463.73	-457.18	-450.52
\$100-150K	-589.42	-589.11	-581.75	-573.96	-566.06	-558.04
\$150K+	-319.45	-319.28	-315.30	-311.09	-306.82	-302.49

Discount Rate = 3%		\mathcal{E}_{MM}^R				
Household Income	0	-0.01	-0.25	-0.5	-0.75	-1
< \$10K	47.32	47.30	46.91	46.49	46.06	45.62
\$10-15K	20.27	20.26	20.13	19.99	19.85	19.70
\$15-25K	11.83	11.83	11.86	11.88	11.90	11.92
\$25-35K	-6.61	-6.60	-6.35	-6.09	-5.82	-5.56
\$35-50K	-96.86	-96.81	-95.50	-94.13	-92.73	-91.32
\$50-75K	-315.94	-315.79	-312.13	-308.26	-304.34	-300.36
\$75-100K	-258.48	-258.36	-255.33	-252.14	-248.91	-245.63
\$100-150K	-319.68	-319.53	-315.89	-312.05	-308.15	-304.20
\$150K+	-173.25	-173.17	-171.21	-169.13	-167.03	-164.89

Table 3 Mean Annual Welfare Changes Attributable to Marine Mammals

	\mathcal{E}_{MM}^R					
	0	-0.01	-0.25	-0.5	-0.75	-1
Mean annual welfare change from marine mammals	N/A	\$971,316	\$1,538,405	\$2,137,058	\$2,743,963	\$3,359,283
Mean annual welfare change per percentage change in marine mammals	N/A	\$468,965	\$734,559	\$1,014,829	\$1,298,852	\$1,586,701

Figure 1 Economy Ecosystem Interaction

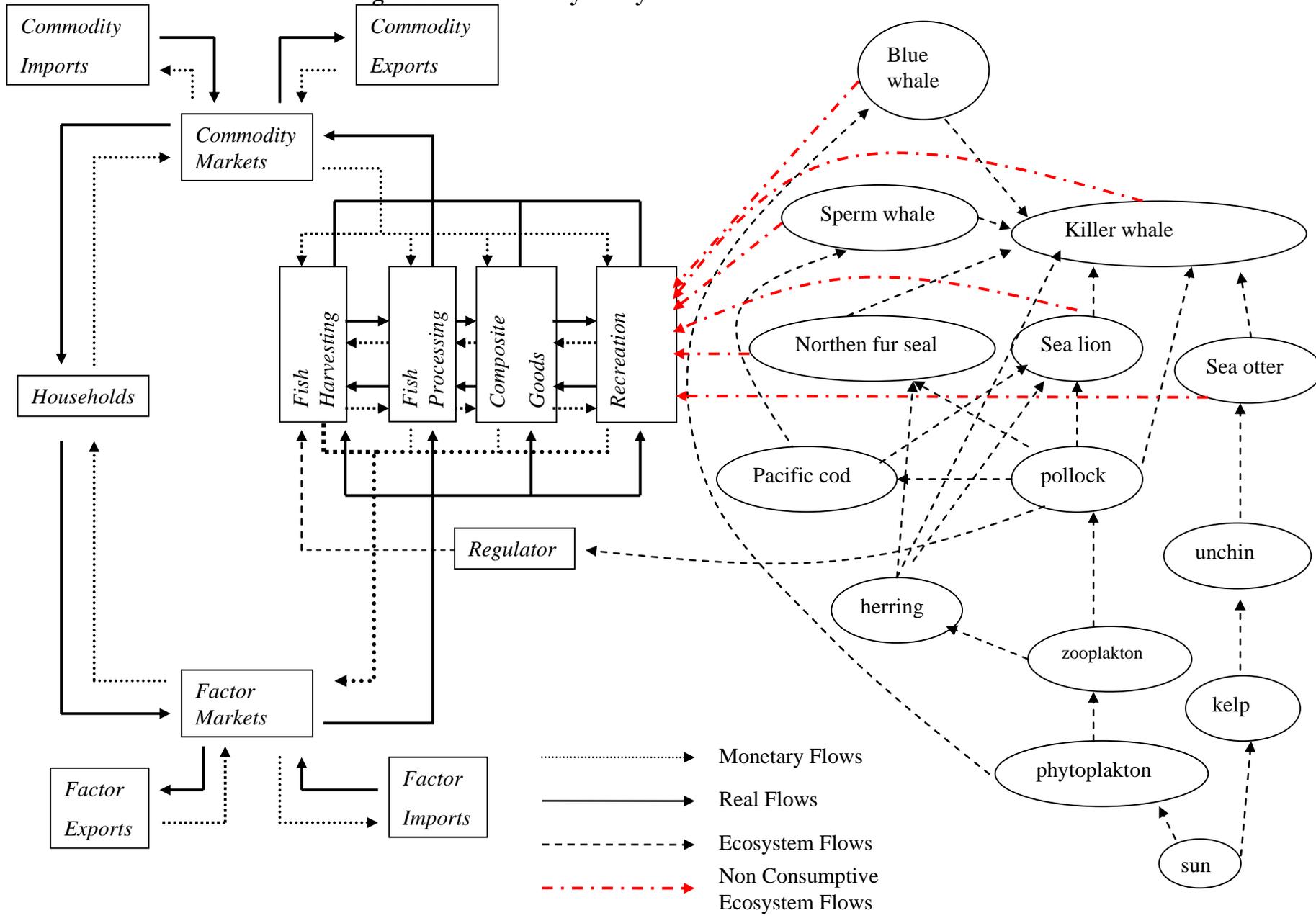


Figure 2 Fishery Intra-season Factor price differentials

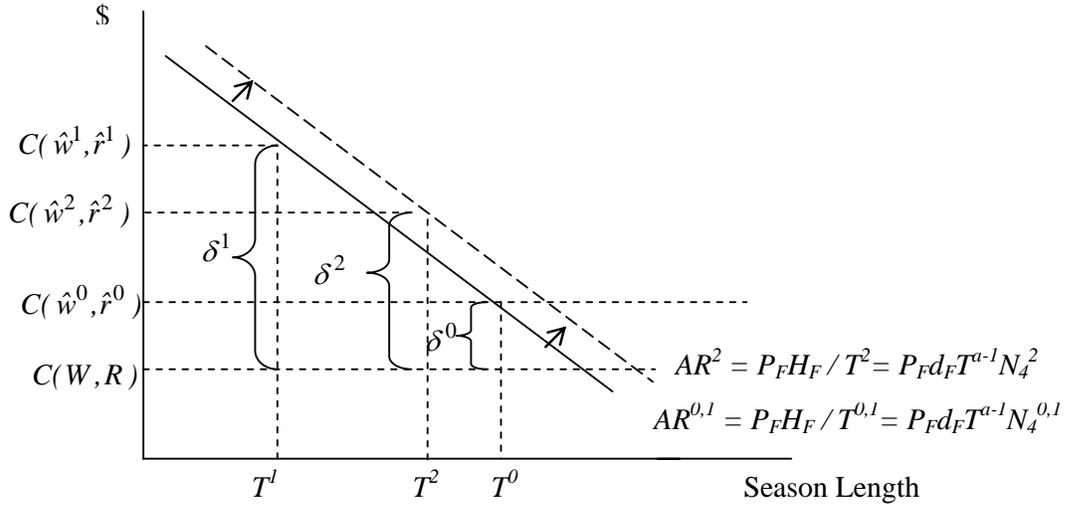
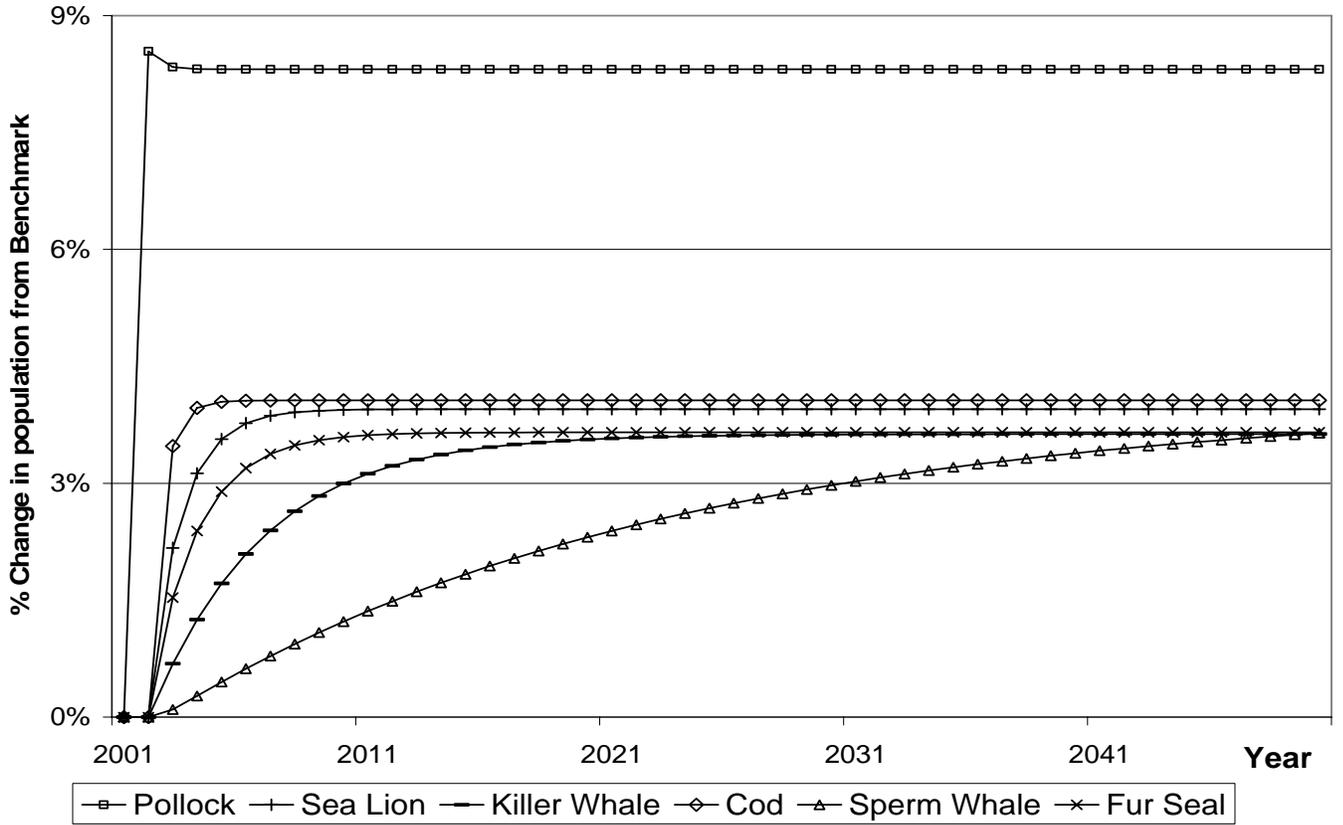
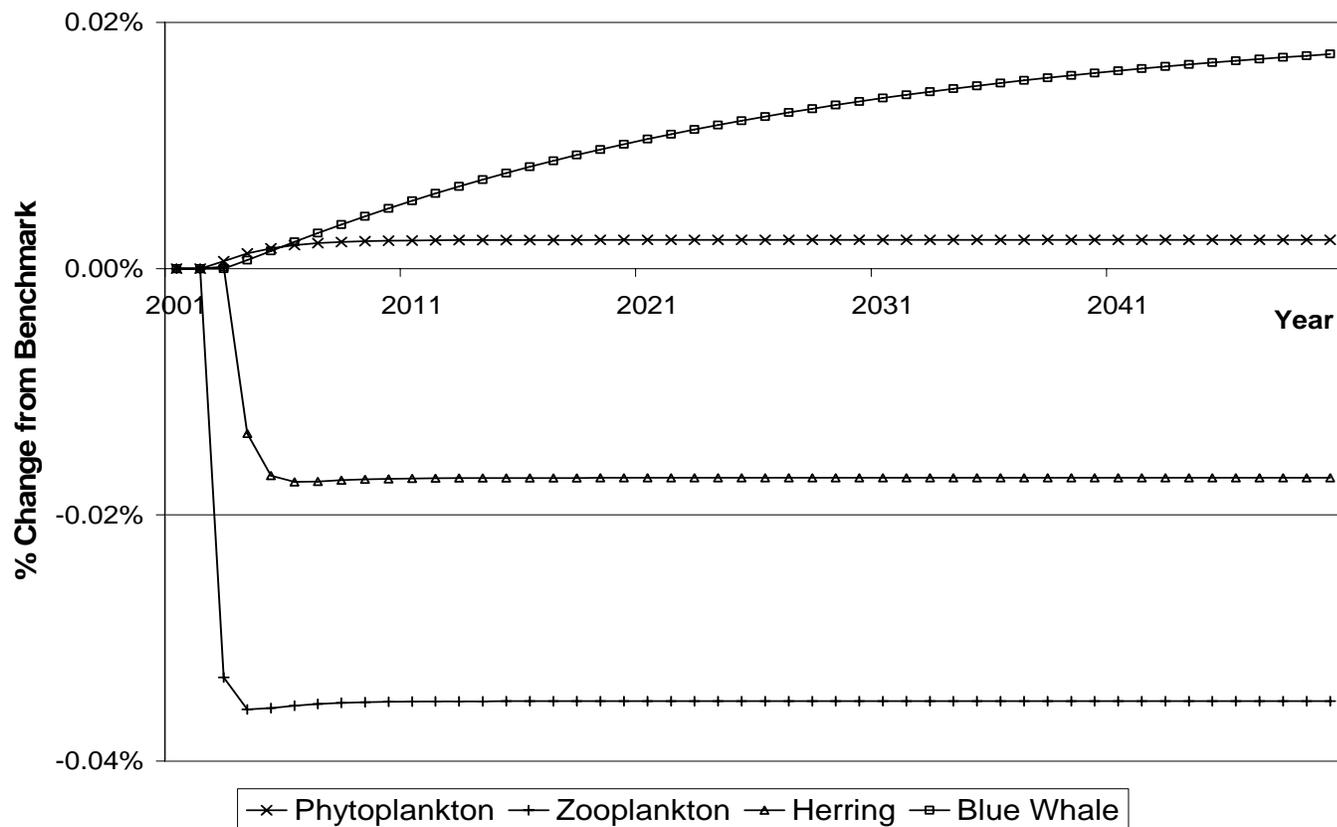


Figure 3 Percentage Changes in Ecosystem Populations from Reducing Pollock Quota

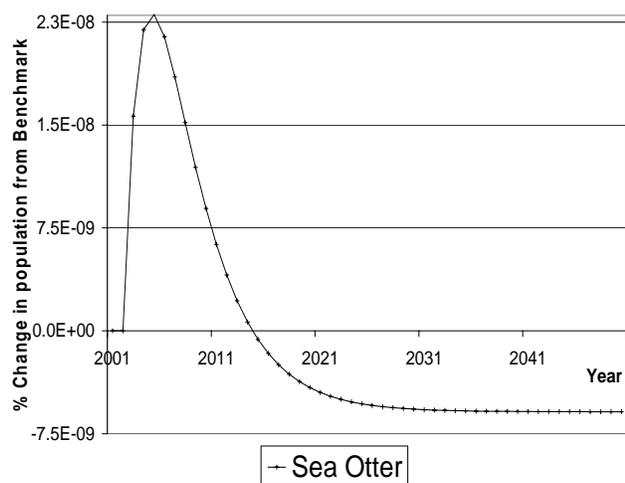


(a)

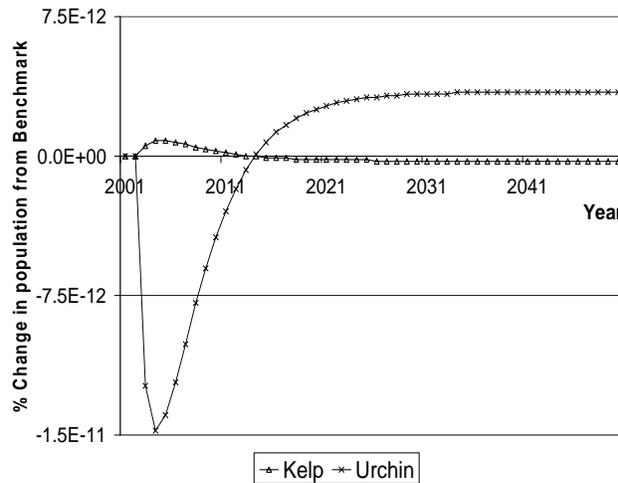
Figure 3 Percentage Changes in Ecosystem Populations From Reducing Pollock Quota (continued)



(b)



(c)



(d)