CANADIAN PARKS COUNCIL
CONSEIL CANADIEN DES PARCS

Valuing the Quality of Freshwater Salmon Habitat – A Pilot Project

Final Report

By
Duncan Knowler
Brice MacGregor
Mike Bradford
and
Randall Peterman

January 2001

Economic Framework Project
Report 557

The Canadian Parks Council is an organization consisting of senior managers representing Canada's national, provincial and territorial parks agencies. It provides a Canada-wide forum for inter-governmental information sharing and action on parks and protected areas that:

promotes excellence in parks and protected areas planning and management;
advocates parks and protected areas values and interests;
encourages cooperation and provides support to member agencies on parks and protected areas issues and initiatives.

The Canadian Parks Council (CPC), known until 2002 as the Federal Provincial Parks Council (FPPC), was established in 1961, as a result of the *Resources for Tomorrow* conference.
Current strategic directions of the Council include:

1. ADVANCING THE PROTECTION EFFORTS OF MEMBER AGENCIES;
2. PROMOTING HERITAGE APPRECIATION AND COMMUNICATIONS ABOUT PARKS AND PROTECTED AREAS;
3. SUPPORTING AGENCIES' EFFORTS TO ADDRESS ABORIGINAL PEOPLES' INTERESTS IN PARKS AND PROTECTED AREAS;
4. SUPPORTING ECOLOGICALLY SUSTAINABLE TOURISM RELATED TO PARKS AND PROTECTED AREAS;
5. FACILITATING EFFECTIVE MANAGEMENT OF PARKS AND PROTECTED AREAS.

For further information about the CPC or the Economic Framework Project contact:

John Good, National Coordinator
Canadian Parks Council
937 Webster Road, RR#1, Warsaw ON K0L 3A0

Phone: (705)652-9752
Fax: (705)652-8496
Email: jgood@parks-parcs.ca
Web: www.parks-parcs.ca

National Provincial and Territorial Parks Agencies: Working Together
Travaillons ensemble: Les organismes des parcs nationaux, provinciaux et territoriaux

Economic Framework Project

In 1998 the Federal Provincial Parks Council called for the development of a common framework for measuring the economic value of protected areas. The purpose of the framework was to help FPPC members speak with one voice when talking about the economic benefits of protected areas within their jurisdictions. It was proposed that the framework should include not only traditional economic impact measurement (e.g., tourism spending, spending on capital development), but also direct user benefits (e.g., consumer surplus, existence benefits) and societal benefits (e.g., benefits from biodiversity, water production, scientific and educational benefits).

Because knowledge and measurement techniques are not equally developed in each of these areas, it was proposed that the work of developing a framework be done in three separate phases, which could be pursued concurrently or sequentially as resources allowed. The three phases are:

I. A user-friendly computerized model for estimating economic impact at the provincial level.
II. A handbook of user benefits showing how the FPPC members could undertake such studies in their own jurisdictions.
III. A series of up to 10 exploratory pilot studies undertaken with the help of academics, to establish a body of case studies on societal benefits

The work was carried out by a project task force, made up of representatives from Ontario Parks, BC Parks, Quebec Parks, NWT Parks and Parks Saskatchewan, and chaired by Dick Stanley, Director, Strategic Research and Analysis, Department of Canadian Heritage (as representative of Parks Canada. The publications in this series are the results of the work of this task force.

THE PAPERS:

315-e	Conservation of Ecological Areas: The Economic Bottom Line
315-f	La Conservation des aires écologiques: les résultats économiques
251-e	Benefits of Protected Areas
251-f	Les avantages des aires protégées
510-e	Economic Benefits of Protected Areas: Guide for Estimating Personal Benefits
519-f	Les avantages économiques des aires protégées: un guide pour estimer les avantages personnels
	Provincial Economic Impact Model: Instruction Manual
	Modèle d'impact économique provincial: manuel d'instruction
556	A Benefit-Cost Framework for Optimal Ecosystem Preservation
549	Carbon Sequestration in Protected Areas of Canada: An Economic Valuation
557	Valuing the Quality of Freshwater Salmon Habitat – A Pilot Project

---

These papers are available on the Canadian Parks Council web site www.parks-parcs.ca

Valuing the Quality of Freshwater Salmon Habitat – A Pilot Project

Final Report

Prepared by:
Duncan Knowler , Brice MacGregor, Mike Bradford, and Randall Peterman
(and with the research assistance of Jesse Wood)

School of Resource and Environmental Management
Simon Fraser University
Burnaby, B.C.
Canada V5H 1S6
(contact: D. Knowler at djk@sfu.ca or 604/291-3421)

JANUARY 11, 2001

Economic Framework Project, Report 557

Published by:
Strategic Research and Analysis
Department of Canadian Heritage

For copies of this report, contact us at:
sradoc_docras@pch.gc.ca
or fax : (819) 997-6765
Reference number: FPPC-557

Valuing the Quality of Salmon Habitat – A Pilot Project

Table of Contents

List of Tables

List of Figures

1. Introduction

As part of an on-going effort to determine the benefits of protected areas in Canada, the Federal Provincial Parks Council (FPPC) and Canadian Heritage are supporting a series of pilot studies. These studies are concerned with the 'societal benefits' of protected areas, one element in a broader valuation framework that also includes user values and economic impacts associated with park areas. One of the societal benefits of interest is the support for ecosystem functions provided by protected areas, including the maintenance of habitat used by wild species of commercial importance.[1] This study is concerned with valuing the benefits from protecting Pacific salmon spawning and rearing habitat on Canada's west coast, using coho salmon (*Oncorhynchus kisutch*) habitat in the Thompson River watershed as a case study. The Thompson River is the largest tributary of the Fraser River, draining 54,600 km^2 of the southern interior of British Columbia, but in portions of its catchment area coho salmon stocks have declined by as much as 90% in the last decade. An initial analysis linking coho population dynamics and land use change in the Thompson River watershed, one of several key factors in the coho salmon's decline, was available to the study's principal investigators (Bradford and Irvine, 2000). This analysis provided key data for estimating the economic benefits of salmon habitat within protected and pristine portions of the case study area (e.g. Lower Adams River Provincial Park).

As a pilot study, the main objective of the research was to test a methodology for estimating economic values associated with the protection of critical fish spawning and rearing habitat. The methodology is a practical and manageable approach for obtaining properly specified economic benefit estimates and has the potential to be extended to other habitat situations. The key innovative elements of the research are:

- its use of a bioeconomic model to derive results that assume the fishery is managed optimally, in the sense of maximizing its net social benefits;
- an attempt to incorporate non-harvest benefits associated with commercial fish stocks;
- its incorporation of the influence of habitat quality at the population aggregate level (e.g. Strait of Georgia);
- the estimation of a general coho stock-recruitment relationship based upon representative data for a cross-section of small watersheds and limited time series data;[2] and,
- the use of geographic information system (GIS) data on land use to measure habitat quality.

To help illustrate the approach, consider two different methods applied to the same habitat valuation problem. In the first case, imagine that habitat deteriorates at a specific site so that there is a financial revenue loss incurred by the fishery that draws on the particular sub-population using this site. Typically, this revenue loss is estimated by first multiplying a measure of the decline in habitat by the recruitment success per unit of habitat, then adjusting this figure for survival and catch rates. This exercise yields a very rough estimate of the reduction in catch.

[1] Ecosystem functions are sometimes categorized as indirect use values, since they support economic activity but in an indirect way. See Chapters 2 and 8 for more discussion of environmental valuation concepts.

[2] Such data, consisting of both time series and cross-sectional data, is referred to as panel data.

Finally, a value is attributed to each incremental fish now lost to the commercial or recreational fishery. If the fish stock is in poor condition then the values derived for its habitat will be commensurately low. If the stock is well-managed then the corresponding habitat values will be higher.

In this study we take a second approach, which is more theoretically defensible (see Freeman, 1993). We begin by assessing the net social benefits available from the fishery assuming it is being managed in some optimal and sustainable fashion, however this might be defined, using a bioeconomic model for this purpose (Clark, 1990). Historically, such optimal management has not been the case in Canada nor in any other nations, so that values associated with the fish resource and ultimately with its habitat have been underestimated.[3]

Optimal values derived from the procedure described above are predicated on some fixed level of habitat quality. If the level of environmental quality is allowed to vary, either infinitesimally or by some larger prescribed amount, the bioeconomic model can be re-solved given this change in one of its 'parameters' and new values for management of the fishery emerge. Once the net social benefit of the fishery under the changed level of environmental quality is calculated, it can be compared to the net benefits estimated for the situation prior to the change. The difference between these two values constitutes the correct welfare measure of the social gain or loss associated with the variation in habitat quality. What this method captures that is not captured in the simpler multiplicative approach is the adjustment that takes place within the fishery to the new situation. For example, if the fishery declines in productivity due to habitat degradation, fewer vessels will fish and those engaged will have lower productivity (or a higher cost per fish caught), thereby reducing the net economic value of the fishery. It is this measure of loss we wish to estimate.

It is a simple step to relate the value calculated above to a park area where the pre-change level of habitat quality might reflect a pristine park and the changed conditions might mimic the level of disturbance characterizing lands surrounding the park. By carrying out the procedure described above and averaging the resulting value over the park area used in the comparison, we can estimate a notional 'habitat value per hectare' stemming from maintaining pristine park conditions. This is the approach taken in this study.

A few further points related to the study methodology can be noted here. First, the approach described above and adopted for the study takes into account site-specific habitat changes but then assesses their impact at the aggregate population level. This basin-wide approach recognizes that while numerous sub-populations reproduce in distinct areas, the fishery operates on a much larger scale and does not differentiate among these smaller sub-populations as a result of the mixing of stocks. Thus, optimal management decisions must reflect the state of the full population aggregate and not simply trace the fate of a single sub-population of fish or a single individual. For example, the loss of an entire spawning stream expresses itself in valuation terms as perhaps a small, marginal change in the basin-wide exploitable stock.

[3] By way of comparison, imagine valuing a farm tractor or other similar financial asset. When assessing its value, the farmer is sure to think of this in terms of what the tractor can perform when used correctly and efficiently. We are simply pointing out that the same should apply when valuing a natural capital asset such as a fish stock or its habitat.

Second, the measure of benefits from the fishery is an important dimension of the valuation problem. Ideally, the analysis would take into consideration the entire set of benefits associated with the West Coast salmon fisheries, beginning with catches allocated to commercial, recreational and native fisheries but extending to the ecosystem benefits supplied by salmon stocks (e.g. support of predator species, nutrient cycling, etc.). Such an all-inclusive analysis was not possible because of the complexity and scale of the task, although certain aspects of this larger perspective have been retained. For example, the presence of contradictory results in previous studies of the B.C. coho recreational fishery created difficulties for the analysis of this fishery (see Chapter 3). Similarly, the early stage of research on ocean survival issues and ecosystem benefits stemming from salmon further limited the scope of research. Instead, we concentrate on a hypothetical commercial coho fishery in British Columbia's Strait of Georgia and integrate one type of ecosystem benefit (e.g. nutrient cycling).

Finally, the difficulties inherent in valuing coho stocks in their presently depleted state, combined with a need to economize on the use of primary data to make new parameter estimates, suggested that a more historical perspective might be appropriate for the research. Thus, we used historical values for a number of estimation procedures and model parameters, not all of which originate from the same time period. For example, we used:

- salmon price and vessel cost data from the mid-1990s (all prices are in 1994 dollars);
- a salmon 'catchability' parameter (measuring the physical catching capability of fishing vessels) which was estimated from 1967-76 data;
- representative land use and coho population data for 16 South Thompson streams originating from the 1980s;
- coho population data for the Strait of Georgia to calibrate the model from 1988-98; and,
- an ocean survival rate estimate averaged from data for 10 to 15 years in the 1970s and 80s;

Ultimately, the valuation estimates contained in this study partly reflect the biological and habitat characteristics of the Strait of Georgia coho stock in its pre-crisis era; that is, before the significant decline in ocean survival and collapse in stocks. By using more recent economic data we are implying that the current value of freshwater habitat should be based on the coho stock under improved conditions, as reflected in their population dynamics several decades ago and perhaps again in the future.[4] This implicit assumption in our work is certainly open to debate, but we feel it is the theoretically correct way to measure the value of habitat.

While the pilot study innovates on the level of incorporating habitat influences in a bioeconomic model of a Canadian salmon stock, new primary data were not collected, nor were aboriginal or recreational fish values treated in any substantial way. As a final caveat, since the research is in the nature of a pilot study, it is understood that the results are indicative only and that they should not be used for planning purposes at the sample sites, let alone at other locations.

[4] Under current low marine survival rates, a fishery for coho is not sustainable. This does not imply that survival rates will remain low in the future. Many fisheries people believe that current low marine survival is a natural occurrence caused by decadal scale shifts in ocean productivity. However, some also believe that we may be approaching another highly productive regime, in which coho marine survival will increase to old levels.

The study is structured in the following manner. Chapter 2 contains an overview and critical evaluation of previous research in the field of habitat valuation relating to marine and anadromous fish stocks with an emphasis on commercial fisheries. Chapter 3 provides background on the case study coho fishery in B.C., including previous valuation estimates relating to this fishery. Chapter 4 presents the bioeconomic modeling approach used in the study and derives the required valuation measures in theoretical terms. Chapter 5 summarizes the biological modeling component of the study, estimating the crucial fish recruitment-environmental quality relationship based on population and habitat quality data for 16 South Thompson streams. Chapter 6 specifies the economic component of the empirical bioeconomic model and derives optimal management results for the Strait of Georgia commercial coho fishery. In Chapter 7, the empirical bioeconomic model results from the previous chapter are used to produce a set of valuation estimates for modifications in salmon habitat in the South Thompson drainage area. The final chapter summarizes these results and discusses policy implications and areas for further research stemming from the pilot study.

2. A Review of Fish Habitat Valuation Studies

Relatively few studies have examined the commercial fishing benefits from improved habitat conditions. One of the first attempts to incorporate a habitat variable into an empirical model of a fishery was by Bell (1972), but in this case the influence was water temperature and not a variable subject to human control. More recent efforts can be grouped according to the type of influence analysed. Several studies consider the outright loss of habitat, particularly coastal mangrove wetlands which support spawning and rearing of juvenile life stages of fish and shellfish. Such studies typically concentrate on valuing the non-market benefits of wetlands serving as fish habitat, perhaps integrated into a broader analysis of the economics of land use conversion. In contrast, the other major group of studies examines modifications in habitat quality brought about by pollution externalities (e.g. nutrient and suspended sediments), as well as altered in-stream flow regimes and salinity levels. We will consider one case from the former group and several from the latter.

In order to provide a framework for understanding the types of benefits valued in the studies cited below, as well as in the study's empirical work reported in later chapters, we adopt the convention based on *total economic value* (TEV).[5] Simply put, TEV makes a fundamental distinction between *use* values and *non-use* values. Use values are grouped according to whether they are *direct* or *indirect* values.[6] Direct uses refer to those uses which are most familiar, such as the harvesting of timber and fish. Ecosystems also serve ecological functions which support economic activity. These services are referred to as indirect use values since it is not the functions themselves but their contribution to economic production which is valued. For example, forested ecosystems may support fish populations, prevent soil erosion and regulate floods. If the forest is lost, damages to fish, agricultural or other production downstream results.

[5] See Pearce and Turner (1990), Barbier, Acreman and Knowler (1997) or IUCN (1998). Also see Chapter 8 and especially Table 8-1 for examples.

[6] In earlier versions of the TEV concept an additional category, *option value*, was included but this is now usually subsumed under the various types of use values once an allowance is made for uncertainty.

Measuring these damages provides an indication of the indirect use value. In contrast, non-use value is now thought of as coinciding with the concept of *existence* value. That is, individuals may be concerned about the continued existence of some environmental resource, such as a tropical forest or endangered wildlife species, even though they have no plans to visit or view it. Non-use values are typically not commercially expressed since they are unrelated to use.

Case Studies

Case Study 1: Logging and a Salmon Fishery in Oregon

Several studies consider the external effects of environmental change, such as timber harvesting, on downstream fisheries. Of interest here is Loomis (1988), who analyses the external costs of logging with respect to commercial and recreational fishing harvests in his study of the Siuslaw National Forest in Oregon. Logging is postulated to affect the downstream salmon and steelhead fisheries via increases in suspended sediments, escalating temperatures, and changes in the debris content of flows. In order to quantify these effects on the salmon available to the fishery Loomis models the productivity of the affected system as a function of a habitat quality index, while incorporating a measure of carrying capacity and stock dependant recruitment.

Loomis constructs a natural Fish Habitat Index (*FHI*) which reflects pristine conditions differentiated on the basis of watershed characteristics and land area. To incorporate the effects of upstream logging, he introduces a Watershed Condition Index (*WCI*), which is a function of the various environmental disturbances described in the previous paragraph. Combining the smolt production function with parameters for smolt survival, returning adults and sport and commercial catch (by fishery), the subsequent harvest of adult salmon and steelhead could be determined as a function of the habitat quality. The physical outputs of fish and timber associated with the four management scenarios considered and natural conditions are presented in Table 1, along with the benefits for the commercial and recreational fisheries.

Table 2-1
Trade-offs Between Timber Harvests and Fish Production over 30 years, Siuslaw National Forest, Oregon

Management Alternative	Timber Production (MMCF)	Timber Area (Acres)	Catchable Salmon (nos.)	Recreational Salmon (US$ millions)	Commercial Salmon (US$ millions)
Current direction	15.52	84,000	8760	0.919	1.147
Timber emphasis	15.16	86,700	9501	0.975	1.130
Fish emphasis	7.88	45,150	11.092	1.381	1.461
Minimum management	0	0	12,323	1.569	1.693
Natural conditions	-	-	15,125	1.654	1.773

Source: Loomis (1988)

For the commercial salmon fishery, the impacts of different practices are valued in gross revenue terms. The approach is justified on the basis that the changes in catches would be marginal and a state of excess capacity existed in the commercial fleet. The impacts on the recreational fishery are valued using a multi-site travel cost model to assign marginal values at specific sites. Since Loomis does not have individual survey data he uses an aggregate model of total site visitation. He uses origin-site specific characteristics and costs to develop the following second stage demand equation for each site:

$$\ln\left(\frac{T_{ij}}{Pop_i}\right) = B_0 - B_1(\ln Dist_{ij}) - B_2(\ln FWSP_j) - B_3(\ln SubS_{jk}) + B_4(\ln INC_i) \qquad (2\text{-}1)$$

where T_{ij} is the number of trips between origin *i* to site *j*, $FWSP_j$ is the catchable salmon at *j*, Pop_i is the population of origin *I*, $SubS_{ik}$ is the *k*th substitute for site *j* in index form, $Dist_{ij}$ is the distance between origin *i* and site *j*, and INC_i is the household income of *i*.

The site demand curves relate total site visitation to travel costs, population, average income, substitute sites and the number of available fish. Travel cost is valued as the cost of transport and the opportunity cost of travel time. The area under this demand curve is a measure of total benefit. Loomis constructed curves for three river and eight ocean port angling sites using the number of harvestable fish associated with the four management scenarios presented in Table 2-1, and angler catch distribution information, provided by the Oregon Department of Fish and Game and the Washington Department of Fish. At each site the difference between original and new demand curves is a measure of benefits loss. Marginal values per fish were attained by dividing the change in consumer surplus by the change in the number of fish (see Table 2-2).

Table 2-2
Site Specific Marginal Values per Salmon

Site	**Marginal Value** (US$ 1981-83)
Neah Bay	23.83
Westport	35.74
Illwaco	21.43
Columbia	27.82
Tillamook	64.61
Newport	39.40
Florence	21.78
Coos	26.21
Alsea River	7.48

Source: Loomis (1988)

Although the logged area under consideration in the analysis represents only 21% of the land within the national forest, the commercial and recreational fishing losses associated with various timber harvesting options amount to over US$ 1 million over a 30 year period (average 1981-1983 prices). Loomis points out that these external costs are relatively low in comparison to the timber values involved, so that managing the forest exclusively for fisheries would not be warranted.

Case Study 2: Mangrove Habitat and a Blue Crab Fishery on Florida's Gulf Coast

One of the best known habitat valuation studies is an analysis of the Florida Gulf Coast blue crab fishery, which relies on threatened coastal mangrove forests for habitat support services. The initial estimations were undertaken by Lynne *et al.* (1981), but these are discussed and extended by Ellis and Fisher (1987) and related research is used by Costanza *et al.* (1989) to estimate the total economic value of coastal wetlands in Louisiana. In these cases, the environmental input takes the form of a coastal marsh or mangrove forest area which supports crab breeding. Time series data relating declining marsh area with fishing effort and crab catch for the period 1952-74 were used by Lynne *et al.* for the initial estimation.

Lynne *et al.* derive a simple static equilibrium model for the blue crab fishery as described in the previous chapter, but define the maximum potential crab stock or habitat carrying capacity as a linear function of the natural logarithm of the mangrove area. Making this substitution, the equation used for estimating the link between catch and mangrove area in reduced form is,

$$C_t = \beta_0 + \beta_1 (\ln M_{t-1})(E_t) - \beta_2 (\ln M_{t-1})(E_t)^2 + \beta_3 C_{t-1} + \varepsilon_t \qquad (2\text{-}2)$$

where C is catch, M is mangrove area, E is effort (as measured by the number of crab traps set) and ε is an error term. Subscripts refer to time periods and help in distinguishing lagged mangrove area and catch, which are retained in the final equation. The results of the estimation are presented in Table 2-3.

Table 2-3

Annual Blue Crab Catch as a Function of Mangrove Area and Effort

Variable	Coefficient	*t* Statistics	Prob value (P^b)
Constant	-6594.2176	-1.43	0.17
$\ln M_{t-1} E$	48.2453	2.03	0.06
$\ln M_{t-1} E^2$	-0.4844	-1.69	0.11
C_{t-1}	0.4060	2.17	0.04

Source: Lynne *et al.* (1981)

Note: additional statistical data is as follows: R^2 = 0.78, DW statistic = 2.05, sample size = 22 (1952-74)

An implicit value for the marsh area is derived by taking the relevant partial derivative for catch with respect to marsh area and then multiplying this by the dockside price of crab. Ellis and

Fisher (1987) point out the need to extend this approach to the use of appropriate welfare measures and make indicative calculations of consumers' and producers' surplus terms instead. Freeman (1991) points out that both previous studies had considered the sole owner situation, whereas the reality of the crab fishery was closer to open access. He then recalculates Ellis and Fisher's values consistent with this property rights regime. Interestingly, he finds that when demand for crabs is inelastic, the net economic benefits attributable to the wetlands input are higher under open access (consumers' surplus only) than under sole ownership (consumers' and producers' surplus).

Case Study 3: Submerged Aquatic Vegetation and Striped Bass in Chesapeake Bay.

In studies of habitat quality modification, the emphasis is on establishing the impact of changes in habitat quality on fish harvests either directly or as a result of some intermediary process such as eutrophication, the loss of submerged vegetation, altered freshwater inflows or suspended sediments. For example, a number of researchers have considered the impact of nutrient flows and toxic pollution on the fish resources of Chesapeake Bay and surrounding marine environments on the eastern U.S. coast. Relatively good data allowed the researchers to quantify the physical effects of environmental degradation on stock growth and ultimately fish and shellfish catches. Kahn and Kemp (1985) assessed the economic losses from the destruction of submerged aquatic vegetation (SAV) due to excessive nutrient and suspended sediment loads. They model the impact on the open access striped bass commercial fishery using a theoretical framework that requires the estimation of an equilibrium catch equation and industry supply and demand curves.

To estimate an equilibrium catch relationship, carrying capacity is modelled as a function of the availability of SAV. The following relationships are used for equilibrium catch (C^e)and carrying capacity (F_C), with these variables expressed in relative terms and the choice of the parameter in the equilibrium catch equation reflecting assumptions about compensatory mortality,

$$C^e = F - \frac{0.85}{F_C} F^2 \qquad (2\text{-}3)$$

$$F_C = 1.36 - e^{-1.004\, V^{0.975}}$$

where F is the fish stock or some index measuring its size and V is the relative level of *SAV*. Taking these two expressions together, relative equilibrium catch can be expressed as a function of *SAV*, similarly to the equations presented earlier relating mangrove area to shrimp catch.

The supply curve is estimated as a log-linear function of the relative prices of alternative target species, the cost of effort, the striped bass stock and a time trend. Striped bass demand is modelled as a function of the price of striped bass, the prices of substitute consumer goods, regional socio-economic variables and a time trend. The hypothesized relationships are shown below,

$$C_c^s = A^s P_1^{\alpha_1} P_2^{\alpha_2} P_3^{\alpha_3} P_E^{\alpha_4} F^{\alpha_5} T^{\alpha_6} \qquad (2\text{-}4)$$

$$C_c^d = A^d P_{SB}^{\beta_1} P_{sub}^{\beta_2} MEN^{\beta_3} PCI^{\beta_4} T^{\beta_5}$$

where C_C designates the commercial catch of striped bass and the superscripts *s* and *d* indicate the supply and demand relationships, respectively. In addition, *A* is a constant term, P_1 is the relative price of striped bass to oysters, P_2 is the relative price of striped bass to clams, P_3 is the relative price of striped bass to crabs (later dropped), P_E is the price of effort, *T* is the time trend, P_{SB} is the striped bass price, P_{sub} is the price of consumer substitutes for striped bass, *MEN* is the regional population and *PCI* is the regional per capita income. The estimated coefficients for the demand and supply functions are presented in Table 2-4.

Table 2-4
Estimated Demand and Supply Relationships for the Commercial Striped Bass Fishery in Chesapeake Bay, U.S.

Variable	Supply Function (C^s)		Demand Function (C^d)	
	coefficient	*t* statistic	coefficient	*t* statistic
Constant	59	1.624	-1.964	-0.021
ln P_1	1.203	1.249		
ln P_2	-1.138	-1.773		
ln P_E	-4.481	-2.214		
ln P_{SB}			-1.281	-2.518
ln P_{sub}			0.8109	0.9664
ln *MEN*			2.969	0.1545
ln *PCI*			4.216	1.561
ln *F*	0.7583	1.775		
ln *T*	-13.28	-1.671	-10.8	-0.6715

Source: Kahn and Kemp (1985)
Note: additional statistics for the supply curve are: *F* statistic (5,9) = 19.69, DW = 2.711, and for the demand curve: *F* statistic (5,9) = 31.11, DW = 2.499

Although the significance of the estimated coefficients is relatively low, Kahn and Kemp use the three estimated relationships to derive the welfare losses from declining *SAV*. By holding all the system variables constant except the catch, the price of striped bass and the fish stock, the three equations can be solved for the equilibrium values of these three variables. This locus of biological, demand and supply curves is referred to as the bioeconomic equilibrium of the system. As the level of *SAV* is then varied, the change in this bioeconomic equilibrium is assessed and changes in consumers' and producers' surpluses can be estimated. Using this procedure, and adding in sport fishing losses as well, a marginal damage function is derived showing the economic losses associated with relative levels of *SAV*.

Case 4: Estuarine Salinity and a Shrimp Fishery in North Carolina.

While the Chesapeake Bay study assesses the effects of nutrients on fish stocks with submerged aquatic vegetation as the intermediary, Swallow (1994) considers the modifications to estuarine salinity in the Pamlico Sound area of North Carolina and its effects on local shrimp catches. Changes in salinity are believed to result from enhanced freshwater flows into the estuary, which in turn arise as nearby wetlands, which normally inhibit freshwater flows, are converted to alternative uses. As salinity in the estuary declines, this is thought to reduce juvenile shrimp survival, culminating in reduced adult stocks and, by extension, falling commercial catches. Thus, the Swallow study concerns downstream effects on habitat quality resulting from land use changes and not habitat loss *per se*. Swallow's empirical work is meant to operationalize a much more complex theoretical model, although empirical realities lead to a number of simplifying assumptions.

Making use of a static partial equilibrium analysis, Swallow establishes the complex linkages between changes in land use and shrimp catch by invoking a simple causal model.[7] Concentrating on a particular type of coastal-freshwater wetland referred to as *pocosin* wetlands, he develops the following set of empirical relationships to describe the shrimp-wetland system,

$$\pi_i = p_i\, k_i\, XFB^{\varphi}\, L_i^{\beta} - w_s\, L_i$$

$$XFB = g\,(SAL) \qquad (2\text{-}5)$$

$$SAL = h\,(POC)$$

where π is profits from shrimp harvesting, p is the shrimp price, k is a parameter reflecting exogenous impacts on shrimp stocks, XFB is an index of the shrimp stock which proxies stock size X, L is labour measured in craft-days, φ and β are parameters in a Cobb-Douglas production function with $\varphi + \beta = 1$ (constant returns to scale), w is the marginal opportunity cost of labour, SAL is estuarine salinity in shrimp nursery areas, POC is the stock of undeveloped *pocosin* wetlands and i is an index representing weekly observations.

Several points concerning Swallow's model warrant explanation. Shrimp stock growth is found to be independent of stock size and rely instead on environmental factors. Thus, expressing the stock index XFB as a function of salinity and ignoring explicit modelling of recruitment seems justifiable. Swallow also captures the interaction between the two production inputs XFB and L, which implies that the value of the marginal product of the shrimp stock VMP_X depends upon the quantity of labour employed as well.[8] Finally, the consideration of two zones of wetlands of

[7] To accomplish this, Swallow draws on the theoretical modelling approach suggested by Kahn (1987).

[8] Proper valuation of the contribution that the shrimp stock makes to profits requires determining the optimal input of labour for each stock level. This is accomplished by taking the partial derivative for catch with respect to L and then equating this to zero. By rearranging, an expression giving the optimal labour input for each value of the stock index is derived.

differing quality and two alternative land uses after conversion added to the analysis by making the resulting value estimates contingent on these attributes.

To determine the marginal impact of wetlands conversion on shrimp catch, VMP_E, requires a complex chain of estimations described in the following disaggregated statement for the problem,

$$VMP_E = \frac{\partial R^*}{\partial X}\frac{\partial X}{\partial E} = \frac{\partial \pi}{\partial XFB}\frac{\partial XFB}{\partial SAL}\frac{\partial SAL}{\partial POC} \tag{2-6}$$

where R^* represents optimized resource rent, proxied by π, and E is the stock of undeveloped wetlands, again proxied by *POC*. Thus, the problem reduces to the estimation of a series of partial derivatives which when multiplied together yield the value of the marginal product of *pocosin* wetlands, VMP_E, expressed as the change in shrimp harvest profits.

Using a variety of empirical procedures the required partial derivatives are estimated for each type of wetland and subsequent land use. The resulting annual estimates of VMP_E range from US$0.28 per acre for lower quality wetlands converted to agriculture to US$3.37 per acre for higher quality areas, again converted to agriculture (1986 prices). For higher quality wetlands converted to a forestry use, the annual impact on shrimp harvesting profits is lower, at US$1.85 per acre. No estimate for lower quality wetlands converted to forestry was made. These values can be interpreted as crude measures of the external costs imposed on the shrimp fishery by land use changes involving nearby wetlands.

Case 5: Altered Stream Flows and a Salmon Fishery in California's Central Valley.

As in the Swallow analysis, Fisher *et al.* (1991) are concerned with the influence of freshwater flows on fish harvests in their analysis of the commercial salmon fishery in California's Central Valley. Initially focussing on water quality influences, they found no relationship with salmon stocks and turned instead to the freshwater flow regime into and out of the San Francisco Bay-Delta system. One of the interesting aspects of the study is its integration of natural and hatchery-supported populations into a single model. It also incorporates a measure of stochastic variability due to exogenous weather conditions by estimating separate relationships for each of three different rainfall regimes.

Of added interest is the authors' use of a highly complex fisheries simulation model, comprising thousands of equations, for their estimations. Since such a complex model is not used easily for the types of policy analysis desired, the authors instead construct a simpler model which tracks the more complex simulation model but at a much more aggregated level. The variables incorporated into this set of simplified regression equations are therefore based upon simulated data, and the resulting model is more readily applied to policy analysis. Not surprisingly, the use of such 'pseudo data' results in very high R^2 values, reflecting the minimal amount of random error and noise (Dowlatabadi *et al*., 1994).

The regressions describing spawning stocks and smolt production demonstrate density dependence because of their quadratic specification, but reflect additional assumptions about mortality and escapement. For example, smolt production (recruitment) is modelled as a function of natural stream flows, water exports (diversions) and spawners. Spawners, in contrast, are determined by target escapement and hatchery production. The authors emphasize the disequilibrium nature of the analysis, arguing that it is this variability which drives the system. They maintain that carrying capacity limits are not relevant because of the managed nature of the stocks: this is a result of the presence of juvenile inputs from hatcheries. Thus, the model does not involve traditional population dynamics.

Preliminary results indicate that the production and harvest of natural salmon is dramatically affected by decreased natural flows in dry years and, therefore, benefits stemming from augmented flows are large. In contrast, augmenting flows in normal or wet years seems to provide few benefits. Given that releases to support fish have an opportunity cost, quantifying these policy options is useful to decision-makers. Undertaking a full cost-benefit analysis of such policies is hampered by the complexities of measuring these opportunity costs and in capturing any other direct or indirect use values associated with augmented releases. The study is best viewed as an assessment of the external effects of disturbed natural flows; these may be positive where a deliberate attempt is made to improve fish habitat or negative where water diversions to serve other users reduce the quality of downstream habitat.

Lessons from the Case Studies for the Pilot Study

This chapter reviewed a selection of empirical studies that integrate habitat modification into models of marine and anadromous fisheries. There are a number of lessons from these applied studies for the empirical analysis to be carried out in this pilot study:

- it is clear that much of the empirical research greatly simplifies the ecological and economic characteristics of the systems investigated. For example, in order to estimate the parameters of interest, numerous studies begin with highly aggregated or even no modelling of population dynamics and a dubious assumption of static bioeconomic equilibrium.

- the measurement of economic gains or losses often does not adhere to proper welfare measurement as economists conceive of it, and in many instances simple changes in gross revenues are inappropriately used for this purpose.

- the multi-functional nature of ecosystems is rarely accommodated; in some cases, optimisation techniques are inappropriately applied in models where important ecosystem values have not been included. Far better are the efforts to either incorporate these values or to approach the problem from the perspective of assessing the welfare effects of a substantial or ‘non-marginal’ change in some environmental influence.

The empirical modelling described in later chapters seeks to integrate these concerns into the analysis using Thompson River coho salmon and the modification of its habitat to illustrate the approach.

3. The Commercial and Recreational Coho Fisheries in B.C.

To gain an understanding of British Columbia's commercial and recreational coho fisheries or any other exploited fishery it is useful to see it as consisting of the biological production of coho and a market for salmon products, with these two components bridged by the action of harvesters. This overview of the exploitation of the Straits of Georgia coho stocks in British Columbia consists of three elements. The first is a review of the population dynamics of coho salmon; the second is an identification of the different coho fisheries (e.g. commercial, recreational, etc.) and their interaction with coho population dynamics; and the third is an investigation of the economic value of these different fisheries.

Coho Population Dynamics

Pacific salmon are *anadromous*, as they are born in freshwater but migrate to the Pacific Ocean to mature before returning to their natal waters to spawn. Coho salmon tend to spawn in small coastal streams or tributaries less than 1 meter wide and 20cm deep (PFRCC, 1999). Spawners return in the fall to lay their eggs before dying. In the following spring, fry emerge from the spawning bed. During their first year fry disperse throughout their natal watershed to rear and grow before migrating to the ocean the next spring as smolts. Most coho make this migration after over wintering their first year. However, in less productive streams, primarily in northern British Columbia, some smolts spend an extra year in fresh water before undergoing this migration (PFRCC, 1999). Coho spend approximately 18 months in the ocean before returning to their natal watersheds as three-year-old spawners in the fall of their second ocean year (Argue *et al*., 1983). Owing to the length of their freshwater residency, the quality and quantity of freshwater habitat play a large role in the population dynamics of coho salmon. Land-use practices can have significant effects on the dynamics of coho by influencing the environment in which eggs incubate and in which juveniles rear prior to migrating after becoming smolts (Meehan, 1991).

During their three year life cycle coho are vulnerable to human harvest at three distinct times in three distinct fisheries. By June of their first ocean summer a few coho are able to recruit to minimum regulated size requirements and so become vulnerable to commercial and recreational harvest (Argue *et al*., 1983). From 1965 to the mid-1990s minimum harvest size and weight for coho were set at 30.5 cm and 1.4 kg, respectively. The winter recreational fishery is much less utilized than the summer fishery, so most coho are not vulnerable to recreational harvest until their second ocean summer. By this time nearly all coho have attained legal size and so this is the period when stocks are most vulnerable to open ocean harvest by both the commercial and recreational fisheries (Argue *et al*., 1983). Finally the freshwater recreational and native fisheries pursue them as they migrate into freshwater to spawn during the fall of their second ocean year.

The Coho Fishery

Coho in Georgia Strait are caught in commercial net and troll fisheries and sport fisheries. Table 3-1 summarizes data on catch by each of these components of the Georgia Strait fishery for the periods 1972-76, 1980-89, and 1990-1998. A more complete data set used in some of the empirical modeling of later chapters is presented as Appendix 1. During the 1990s, conservation concerns, particularly for Thompson River coho, led to fishing restrictions and a declining catch of coho in Georgia Strait. Recent declines in the abundance of coho salmon have been attributed to high exploitation rates in conjunction with low marine survival rates (Bradford, 1998), although degradation of freshwater habitat is also likely to have contributed to the problem (Bradford and Irvine, 2000).

Table 3-1
Average Annual Coho Catch by Canadian fisheries in Georgia Strait for Periods 1972-76, 1980-89, and 1990-98

Georgia Strait Coho Fishery	Troll (1000s)	Troll (%)	Nets (1000s)	Nets (%)	Sport (1000s)	Sport (%)	Total Catch
1972-76	98	16	44	7	470	77	612
1980-89	139	21	13	2	506	77	658
1990-98	71	24	3	1	227	75	301

Source: adapted from Argue *et. al.* (1983) and Simpson, *et al.* (1999)

The Commercial Coho Fishery

The commercial fishery for coho has two components: a troll fishery and a net fishery. The troll fishery is a hook and line fishery using lures. Troll vessels vary from 8 to 18m in length and can employ from two to five crew. Trollers drag several lines with numerous lures slowly through the water at depths down to 110m. The troll fishery primarily targets offshore Chinook and coho salmon for the high value fresh and frozen markets. The net fishery is composed of vessels using both gillnets and seine nets. Netting vessels range between 8 and 24m in length, depending on the gear used, and employ from one to five crew. Gillnetting involves nets that are suspended in the water, when fish swim into them they are caught in its webbing. Seining or purse seining involves a net that can be closed, or pursed, to trap a school of fish (Argue *et al*., 1983). The net fishery operates closer to shore and targets pink, sockeye, and chum salmon primarily for the canned market. Chinook and coho catches brought in by the net fishery are incidental, and are a result of the indiscriminant nature of mixed stock fishing (ARA, 1996).

The troll fishery has been the major commercial harvester of coho salmon in British Columbia. In fact it is the only commercial fishery to target coho salmon. The troll fishery is managed under the Canada-US Pacific Salmon Treaty to operate within a specific catch ceiling and/or harvest rate (ARA, 1996). Commercial trollers were restricted to a 3-month season on coho salmon beginning July 1st until 1995, when the Department of Fisheries and Oceans (DFO) closed the commercial coho fisheries to meet conservation objectives (Argue *et al*., 1983; ARA, 1996). In 1970 there were approximately 1000 commercial trollers and combination net trollers operating in Georgia Strait, of which about 500 depended on Georgia Strait stocks for the majority of their catch and the remaining 500 fished the Georgia Strait on a part-time basis. The normal practice for these vessels was to fish in Georgia Strait for the first 4 weeks of the chinook or coho season, and then to move to the outside coast fisheries for the remainder of the season. During 1972-76 trollers took an annual average of 98,000 coho in Georgia Strait, or 16% of the catch for Georgia Strait (Argue *et al*., 1983).

The net fishery is a secondary harvester of both chinook and coho salmon. DFO regulates the net fishery according to escapement objectives for sockeye, pink and chum salmon (ARA, 1996). The life cycles and abundance of these three species also generally determine fishing effort, location, and timing. Generally, netting vessels target fish close to their natal rivers on their return spawning migration (Argue *et al*., 1983). The use of nets allows for selection of fish size but not species and, therefore, some coho are caught by gillnetters and seiners as incidental catch. In 1973, there were approximately 400 seiners, and 3,000 gill-netters[9], that operated for at least one day within Georgia Strait (Argue *et al*., 1983). In the period from 1972 to 1976 netting vessels took an annual average of 44,000 coho in Georgia Strait, or 7% of the catch for the area (Argue *et al*., 1983).

The Recreational Coho Fishery

The recreational fishery is different from the commercial fishery in that individuals participating in the fishery, either independently or through organized operators, catch salmon for their own personal use and not for sale. Recreational anglers fall into three rough categories: independent anglers (who may be boat or shore anglers), charter anglers (who hire boats and guides) and lodge anglers (who purchase all-inclusive fishing packages). Unlike the commercial fishery, entry into the tidal recreational fishery is not limited. Any one willing to pay the license fee can purchase a fishing license from DFO. Catch and possession limits are also regulated by DFO: in 1983 the combined chinook-coho catch limit for recreational anglers was four fish per day with a possession limit twice the catch limit (Argue *et al*., 1983). In 1995, the catch limit for both chinook and coho was changed to two fish per day with a combined limit of four fish per day for all salmon species. As well the minimum length for Georgia Strait coho was increased to 41cm from 31 cm, ensuring that fish are not vulnerable to harvest until their second ocean summer (ARA,1996).

During the period 1972-76, the annual average coho catch by the recreational fishery in Georgia Strait was 470,000 fish, while the number of license holders in B.C. was 360,000 (Argue *et al*., 1983). Georgia Strait continues to be the most utilized area by recreational anglers. In 1972, it was estimated that 96% of all British Columbia angler days were expended in the Strait, estimates from the mid-1990s put that level at approximately 60% (Argue *et al*., 1983; ARA, 1996).

[9] This number includes trolling–gillnet combination vessels also accounted for in the description of the trolling fishery

Previous Attempts to Value the B.C. Coho Fishery

In this section we review two previous attempts to place values on coho fisheries in B.C. One study estimates the full set of economic benefits associated with the B.C. coho fishery. It provides useful background information on the economic and valuation aspects of the coho fishery for later use in the empirical research. The other study cited describes an attempt to use contingent valuation to obtain values for recreationally caught chinook and coho and difficulties encountered in this study provide the rationale for not including a recreational component in our empirical work. In these valuation studies, the net economic value of the British Columbia coho fishery is the difference between the total benefits and the total costs to society of operating the fishery. The costs and benefits associated with the commercial fishery are all quantifiable market values. Moreover, it is characterized by an internationally-determined price, so we would not expect any consumer surplus. In contrast, the recreational fishery derives its value by producing angling experiences for individual anglers. As well as quantifiable expenditures and revenues, its net economic value is influenced by consumer surplus. In the first study reviewed below, this surplus is referred to as 'angler surplus' and it is the value anglers attribute to a fishing experience in excess of it costs. Angler surplus is affected by many different variables, not just fish availability, and it is usually elicited through contingent valuation methods.

Estimating the Economic Surplus in the B.C. Coho Fishery

In 1996, ARA Consulting (1996) conducted a study of the value of British Columbia's chinook and coho fisheries, covering the commercial and recreational components. In order to derive the benefit of the commercial fishery, they first estimate the gross economic value of the fishery as the difference between expenditures and revenues and then adjust this value to estimate the net economic value or surplus attributable to the fishery. Expenditures are incurred in association with the landed value, processor margin, and retailer margin. Landed value to vessel owners is the price received for the catch less vessel expenditures. Processor margin is the difference between total processor revenues and the cost of purchasing fish from harvesters. Processors add value by producing fresh, frozen and canned products. Three quarters of trolled coho are frozen and one half of net-caught coho are canned. Retailer margin is the difference between retailer revenues and wholesale costs. For fish sold in Canada this is approximately 40% of the wholesale price. However, 65% of the net catch and 85% of the troll catch is exported, so the contribution of retailer margin to the value of commercial coho catches is quite small. The price paid by consumers is the sum of landed value, processor margin and retail margin. The difference between the purchase price and the consumer's willingness-to-pay (WTP) is the consumers' surplus. The sum of all of these values gives the gross economic value of commercial coho catches reported in Table 3-2.

Table 3-2
Economic Valuation of British Columbia's Commercial and Recreational Coho Fisheries in 1994

Commercial Coho Fishery		Recreational Coho and Chinook Fishery	
Total Catch (nos. in thousands)	2,555.6	**Total Catch (nos. in thousands) with Days Fished (thousands) in brackets**	507 (2700)
Troll	2,099.6		
Net	456.00	Lodge	56 (200)
		Charter	36 (150)
Weight ('000 lbs)	17,002	Independent	423 (2350)
Gross Value ($millions)		**Gross Value ($millions)**	
Landed Value	22.8	Expenditure on Boats/Gear	254.1
Processor Margin	20.6	Expenditure on Lodge/Charter	123.5
Retail Margin	6.9	Direct Expenditure	233.7
Consumer Surplus	2.7	Angler Surplus	130.1
Total Gross Value	52.9	Total Gross Value	741.4
Net Value ($millions)		**Net Value ($millions)**	
Consumer Surplus	2.7	Angler Surplus	130.1
Business Surplus	5.8	Lodge Surplus	4.5
Worker Surplus	9.0	Worker Surplus	0.0
Gov't Adjustments	[1.5]	Gov't Adjustments	82.2
Total Net Value	16.0	Total Net Value	216.8
Average Net Value ($ per fish)	6.25	**Average Net Value ($ per day)**	80.30
		Lodge	142.00
Short Term Marginal Value ($ per fish)	12.90	Charter	85.90
Long Term Marginal Value ($ per fish)	7.98	Independent	74.70
		Short Term Marginal Value ($ per fish)	14-27
		Long Term Marginal Value ($ per fish)	11-22

Source: ARA (1996)
Note: Chinook values are incorporated in recreational data as this is a mixed fishery

Consumer surplus, business surplus, worker surplus and government adjustments are used to determine net economic benefit. Consumer surplus is calculated in the same way as before. Business surplus consists of above normal returns to vessel owners, processing firms, and retailing firms.[10] Worker surplus is the wage received by workers in the fishing, processing and retail industries, less their opportunity costs. Barriers to the entry of workers into an industry may inflate wages beyond that which could be realized in other sectors of the economy. Finally government adjustments take account of taxes, special transfers (such as industry specific employment insurance payments) and licensing fees, all of which are transfers between the industry and government and so are not included in expenditures. Table 3-2 also shows the net economic benefit calculations from the ARA study, as well as estimates of the marginal value of an additional fish supplied to the commercial fishery.

[10] Above normal returns are returns higher than could be expected if the capital was invested in the next best alternative. This is usually calculated by comparing the return per dollar invested versus average returns for the economy as a whole.

Economic valuation of the recreational fishery follows the same framework as commercial fisheries. In the determination of gross value angler expenditures on boats and equipment, on lodge and charter services, and on direct expenses like fuel, food, and repairs, take the place of landed values, processor margin, and retail margin. In determination of net value, business surpluses are replaced by lodge and charter surpluses, and the worker surpluses refer to workers in the lodge and charter industry rather than the harvesting and processing industries. Angler surplus is a measure of the benefits received by anglers above the costs they pay for their fishing experience. Determination of the angler surplus, for use in the gross and net value calculations, is complicated because it is a non-market value and is dependant on the value an angler places on an angling day. This value is affected by not only the number of fish available but also by diverse non-fish variables like the setting. Marginal economic values are even more difficult to derive because they depend on the response of anglers to changes in fish availability and the response of the value of an angler day to change in fish availability.

Valuation of Recreational Coho and Chinook Catches

Many studies have attempted to determine the value of an additional, or marginal, fish to a recreational fishery (Freeman, 1995). One study of specific interest is concerned with anglers participating in British Columbia's recreational chinook and coho fishery (Cameron and James, 1987a). Cameron and James use the closed-ended contingent valuation method (CVM) to discern the affect different explanatory variables have on the valuation of an angler's fishing experience.[11] The study uses contingent valuation data collected from an in-person survey of recreational fishermen between July and early December 1984 that resulted in 4161 responses. After establishing the cost of a current day's fishing, respondents from the Victoria, Port Albernie, Campbell River, and Sechelt areas in British Columbia, were asked whether they would still have gone fishing that day if the cost of the day's trip had been higher by some randomly chosen value. Information on variables such as the number of fish taken, size, species, weather, residence and other characteristics was also collected. This information was then used to explain the underlying willingness to pay (WTP) for a fishing day and to estimate the value of an extra unit of each of the attributes of the angling experience.

As Table 3-3 indicates, Cameron and James were able to determine the effect of different variables on WTP. The most important category of variables was found to be catch characteristics. On average an extra chinook salmon adds $14.47 to WTP. These are anglers' preferred sports fish. Additional coho catches appear to reduce WTP by $1.21 or $7.90 depending on whether the angler caught any chinook that day. This appears counter intuitive, but anglers face a per day combined catch limit for the two fish. By taking a coho the angler reduces the number of chinook, the preferred species, that may be taken. Interestingly, if the largest fish caught is a coho the weight of this fish increases WTP by an average of $4.35. Therefore it appears that the typically larger chinook species is preferred, while coho are valued

[11] Closed-ended CV surveys allow respondents to state whether they would accept or reject a theoretical threshold cost that is randomly chosen. In comparison, open-ended CV surveys allow a respondent to name a value, and sequential bid CV surveys ask about a specified sum and then repeat the question using a higher or lower amount depending on the initial response. Closed-ended CV surveys have the advantage of presenting a real market-type scenario without creating the starting point type bias of the sequential approach.

less but the gap is smaller if the coho caught is relatively large. Secondary angler variables are also interesting. It was found that that WTP is substantially higher for guided trips, weekend trips and for non-B.C. residents. These findings probably reflect reduced accessibility for non-resident and employed individuals. For them, opportunities to fish are fewer, and so are likely to be more highly valued. Environmental variables are also shown to affect valuation. Increased levels of sun, temperature, and rain all appeared to lower WTP. This is probably related to angler comfort and to the common belief that fish are harder to catch when it is sunny.

Table 3-3
Coefficients and Per Unit Values for Explanatory Variables ($C 1984)

Variable	Coefficient, MLE	Value (WTP) of a One Unit Change in Variable using Weighted Variable Means
NKCN	0.291	14.47
NKCO	(0.15)	(7.94)
LGCN	0.013	0.65
LGCO	0.087	4.35
GUID	0.435	21.71
RESO	0.463	23.04
SOLO	(0.09)	(3.82)
WKN	0.1186	5.92
SUN	(0.02)	(0.97)
TEMP	(0.03)	(1.34)
PERC	(0.02)	(0.75)

Source: Cameron and James, (1987a)
Legend: NKCN = number of kept Chinook , NKCO = number of kept Coho, LGCN = largest fish caught was a Chinook, LGCO = largest fish caught was a Coho, GUID = fishing trip was guided, RESO = non-Canadian resident, SOLO = angler fished alone, WKN = trip was on weekend, SUN = Hours of direct sunlight experienced, TEMP = mean daily temperature, PERC = total precipitation

Cameron and James also determine the increase in WTP from an additional unit of any of the explanatory variables. Their results tend to indicate that coho salmon are intrinsically valuable but that existing regulations treating coho and chinook as one group, reduced the desire of anglers to catch them. Since a caught coho reduces the number of chinook that can be kept anglers are forced to compare the relative value of each of these species. Instead of allowing additional fish from each species to contribute independently to WTP anglers are placed in an artificial trade-off situation where their greater desire to catch chinook actually places a negative value on marginal coho. In a related paper, Cameron and James (1987b) show how their results can be used to value a salmon habitat enhancement program. However, as noted above, this would produce negative results in the case of coho. For this reason, this study was not used in the subsequent empirical analysis.

4. A Conceptual Model for Valuing Salmon Habitat

In this study we use a bioeconomic modelling approach to obtain values for salmon habitat. Bioeconomic models are defined as models that seek to maximize some measure of economic value, subject to resource dynamics (Conrad, 1995). The standard bioeconomic model takes no account of changing environmental or habitat conditions, since these are implicitly assumed to remain constant. In this study we are concerned with the situation where this assumption is relaxed to reflect the more realistic case where habitat conditions change. One of the most important aspects of the problem is the way in which the habitat support function provided by the environment is modelled. In the fisheries case, the approach selected can depend on a number of factors, such as whether environmental change involves physical destruction or conversion of fish habitat or the modification of habitat quality. Models of the former type typically incorporate some measure of habitat quantity as an input into the natural production function (population dynamics) of the harvested species (Hall, 1977). In simpler models, this input is regarded as costless, a free gift of nature, whereas in more sophisticated models there is an opportunity cost associated with its use, sometimes in the form of lost development benefits. Valuation of changes in the supply of the environmental input can be derived relatively easily using standard techniques of production economics and welfare analysis (Freeman, 1993).

In contrast, models involving modifications in the quality of habitat, especially where this results from the introduction of wastes or other deleterious materials into the aquatic environment, are similar to standard pollution models. Here the objective is not the maximization of harvesting profits under a biological constraint but one of the dual problems of maximizing renewable resource benefits less pollution control costs or minimizing joint pollution damages and control costs. These damages involve the degradation of habitat, leading to less productive natural populations with smaller harvestable surpluses. Both of the approaches described above have relevance for the analysis of salmon habitat in the Thompson River Basin but the first approach is used in this study. In the sections below we first outline a general theoretical methodology to value salmon habitat where only a harvest benefit (direct use value) is included and then discuss an extension that incorporates an ecosystem benefit (indirect use value).

A Dynamic Bioeconomic Modelling Approach with Habitat Influence

When dealing with natural populations it is generally more satisfactory to use a dynamic specification. Static models take no account of the process of adjustment by which an optimal stock size is attained, for instance, and ignore the discount rate.[12] To analyse a bioeconomic model in dynamic terms requires the use of dynamic optimisation techniques, whereby variables such as harvest, fishing effort and fish stock are calculated as functions of time. In formulating the Georgia Strait coho fishery problem we take a discrete time approach and focus on harvest (h) as the choice variable in either a recreational or commercial fishery (but not both together).

[12] A general formulation of the dynamic problem in continuous or discrete time, allowing for linear and non-linear functional forms and assuming downward sloping demand, is presented in Clark (1990).

The expression for net social benefits from the fishery is:

$$W(X_t, h_t) = \int_0^h p(h_t)dz - C(X_t, h_t) \qquad (4\text{-}1)$$

where *W(X,h)* is the net social benefits from the coho stock in period *t*, including harvest *h* and other values; *X* is the aggregate recruitment to the coho stock, measured in numbers of exploitable fish; *p(h)* is the inverse demand function for coho in either the recreational or commercial fishery; *h* is the coho harvest in year *t*, measured in numbers of fish; *z* is a coefficient of integration; and *C(X,h)* is a cost function for either the recreational or commercial fishery;

The population dynamics for coho are presented as a transition equation showing the recruitment of young coho to the exploitable stock after a three-year lag between spawning and recruitment. Drawing on the delay recruitment model suggested by Clark (1976), it is possible to incorporate this lag without resorting to a full age-structured model. To accomplish this, we express coho recruitment in year *t*+1 as the following function of spawner escapement (*X* - *h*) and habitat quality (*Q*) in year *t-2*:

$$X_{t+1} = R(X_{t-2} - h_{t-2}, Q_{t-2}) \qquad (4\text{-}2)$$

where *R(X-h, Q)* is the coho recruitment function governing the entire population aggregate, and *Q* is habitat quality, measured as an index of average land disturbance throughout the entire spawning and rearing area used by the exploitable fish stock.

If we substitute *B*(*h*) for the first term on the right in (4-1), the planner's problem can be expressed as a standard constrained dynamic optimization problem in discrete time as follows:

$$\max \sum_{t=0}^{\infty} \rho^t \{B(h_t) - C(X_t, h_t)\}$$

$$s.t. \quad X_{t+1} = R(X_{t-2} - h_{t-2}, Q_{t-2}) \qquad (4\text{-}3)$$

where ρ is the discount term, defined as $1/(1+\delta)^t$, with δ denoting the appropriate social discount rate. To solve this problem, we form the following Lagrangean expression:
where λ is the Lagrangean multiplier measuring the shadow value of an additional unit of exploitable fish biomass at time *t*+1, discounted to time *t*.

$$L = \sum_{t=0}^{\infty} \rho^t \{B(h_t) - C(X_t, h_t) - \rho \lambda_{t+1} [X_{t+1} - R(X_{t-2} - h_{t-2}, Q_{t-2})]\} \qquad (4\text{-}4)$$

The first order conditions for this problem are:

$$\frac{\partial L}{\partial h_t} = \rho^t \left[p(h_t) - C_h - \rho^3 \lambda_{t+3} R_{X-h}\right] = 0$$
$$\frac{\partial L}{\partial X_t} = \rho^t \left[- C_X - \lambda_t + \rho^3 \lambda_{t+3} R_{X-h}\right] = 0 \qquad (4\text{-}5)$$
$$\frac{\partial L}{\partial \rho \lambda_{t+1}} = \rho^t \left[X_{t+1} - R(X_{t-2} - h_{t-2}, Q_t)\right] = 0$$

where partial derivatives are indicated using subscripts. The above optimality conditions can be rearranged and interpreted along conventional bioeconomic analysis lines.

Setting all variables in the above system equal to their long run equilibrium values, ie. $X_{t+1} = X_t = X$ and $h_{t+1} = h_t = h$, and assuming habitat quality Q is constant at Q (bar), we can derive the expressions governing the coho system at the long run equilibrium:

$$X - R(X - h, \overline{Q}) = 0, \quad \text{for } X_{t+1} = X_t = X$$
$$R_{X-h} \frac{p(h) - C_h - C_x}{p(h) - C_h} = \rho^{-3}, \quad \text{for } h_{t+1} = h_t = h \qquad (4\text{-}6a, 6b)$$

Expression (4-6a) governs the long-run equilibrium population dynamics. Expression (4-6b) is the well-known Golden Rule expression with the discount rate appearing on the right-hand side (Conrad 1995, Clark 1990). Optimal stock (X^*) and harvest (h^*) are uniquely determined by (4-6a) and (4-6b) for a given level of habitat quality Q (bar) and any other parameters.

Note that we can calculate the maximized net social benefits under optimal management for a given level of habitat quality as the following expression:

$$W(X^*, h^*; \overline{Q}) = B(h^*) - C[X^*(h^*, \overline{Q}), h^*] \qquad (4\text{-}7)$$

where $W(X^*,h^*;Q\ bar)$ is maximized net social benefits contingent on a fixed level of habitat quality Q (bar). Here, we can think of the habitat quality variable as a fixed parameter.

To determine the welfare effect stemming from a small or 'marginal' change in habitat quality under these optimal conditions, we measure the change in $W(X^*,h^*;Q\ bar)$ as Q (bar) is varied slightly in the current period. To derive this result, we take the partial derivative of the full Lagrangean expression with respect to Q, since we are dealing with a *maximized* expression in (6).[13] Taking into account that altering Q (bar) now aff3ects recruitment three periods hence, this procedure yields:

$$\frac{\partial W(X^*,h^*;\overline{Q})}{\partial \overline{Q}} = \frac{\partial L}{\partial \overline{Q}} = \rho^3 \lambda_{t+3} R_Q \tag{4-8}$$

Since λ is unobservable, we can eliminate it by substituting from (4-5). Assuming we are at the long run equilibrium situation, λ can be derived from (4-5) as:

$$\lambda = p(h) - C_h - C_X \tag{4-9}$$

This gives the desired partial derivative as:

$$\frac{\partial W(X^*,h^*;\overline{Q})}{\partial \overline{Q}} = \frac{\partial L}{\partial \overline{Q}} = \rho^3 \left[p(h) - C_h - C_X\right] R_Q \tag{4-10}$$

Using (4-10), it is possible to estimate the value of preserving salmon habitat, rather than allowing it to become disturbed by agricultural and other activities not permitted within a protected area. In this case, it is not a marginal change in habitat conditions that is of interest but a non-marginal change, involving a comparison of a given quantity of protected and, therefore, more-or-less pristine habitat, with disturbed and unprotected habitat. First, we re-specify Q as distinct habitat quality levels with and without protected area status, where Q^A signifies the former and Q^B the latter, and $Q^A > Q^B$. This characterisation of the problem yields the following statement for the welfare change from habitat protection within a national park:

$$\Delta W(X^*,h^*;Q) = \int_{Q^B}^{Q^A} \frac{\partial W(X^*,h^*;Q)}{\partial Q} dz \tag{4-11}$$

where the integral is performed along a path where harvest is continuously set at its optimal value. Expression (4-11) can be expressed more simply as:

$$\Delta W(X^*,h^*;Q) = W(X^*,h_A^*;Q^A) \text{ - } W(X^*,h_B^*;Q^B) \tag{4-12}$$

Adding an Ecosystem Service (Indirect Use Value)

As well as providing direct benefits to the commercial, recreational, and native fisheries, coho salmon also provide indirect societal benefits through their function as ecosystem services. Ecosystem services describe environmental attributes that are fundamental to maintaining ecosystem function and resilience, and thereby indirectly meet human demands (Holmlund and

[13] See Silberberg (1978) for a description of this application of the envelope theorem.

Hammer, 1999).[14] Pacific coho salmon provide many fundamental and human-valued ecosystem services, including aesthetic values, ecological information, food chain regulation and nutrient cycling (Holmlund and Hammer,1999). Salmon have been identified as a keystone species in vertebrate communities (Wilson and Halupka, 1995). As an anadromous fish, salmon unite marine, aquatic, and terrestrial ecosystems, by migrating between them during their life cycle. Smolts entering the ocean from fresh water streams transfer energy and nutrients to the marine system. On their return as adult spawners, food energy and nutrients are transferred back to terrestrial and fresh water systems through excretion, gamete release, and mortality. Because West Coast ecosystems are generally nutrient poor, the net transfer of nutrients and energy is from marine to terrestrial ecosystems (Cederholm *et al.*, 1999). This raises the question of the effect that reduced numbers of spawners could have on the nutrient transfer process and the ecosystems they enrich.

Evidence indicates that the marine to terrestrial transfer of nutrients and energy, facilitated by salmon, is very important to the functioning of coastal food chains, and to the maintenance of salmon populations themselves. Studies show that the reproductive cycles and seasonal distributions of many animals that use salmon as a food source, such as eagles, bears, merganser, and mink, are directly linked to the migration of the fish between ecosystems (McClelland *et al*., 1982; Stalmaster and Gessamen, 1984; Wood, 1987; Hilderbrand *et al*., 1996; Ben-David, 1997). Certain bear populations rely on salmon for most of their energy during hibernation (Wilson and Halupka, 1995), and the use of stable isotope analysis has shown that coastal Alaskan brown bears obtain nearly all of their carbon and nitrogen from salmon (Hilderbrand *et al.,* 1996). Stable isotope analysis has also shown that the distribution of salmon nutrients by animals through defecation and carcass transportation is of critical importance to the productivity of riparian vegetation (Cederholm *et al*., 1999). Salmon-facilitated nutrient enrichment is also extremely important within freshwater ecosystems because it enhances primary productivity, resulting in higher juvenile survival rates (Cederholm *et al*., 1999). For example,the nutrient influx from pink salmon has been shown to influence recruit-per-spawner, smolt size and number, and ocean survival of rearing salmonids, while seeding streams with hatchery carcasses has been shown to increase age 0+ coho density (Michael, 1995; Bilby *et al*. 1998). Accordingly, Cederholm (1999) offers the following opinion:

> *"Ecosystem health will presumably benefit from having the largest number of spawners possible, which in turn would produce a large number of carcasses. Escapement goals should be designed to build 'nutrient capital' within watersheds that will help support the next generation of fish."*

We can show an ecosystem service performed by returning salmon as an additional benefit and describe this with the benefit function $V(X–h)$. In effect, we postulate that there is a stock benefit associated with coho escapement X-h. Care is required in assessing this benefit since at present any spawning coho carcasses already contribute to the enhancement of smolt survival and

[14] For example the extraction and retention of atmospheric carbon through photosynthesis reduces greenhouse gas levels while playing an important role in ecosystem maintenance through primary production and oxygen replenishment. Likewise the retention of vegetated areas that provide these fundamental services also provides value to humans through aesthetic appeal and recreational opportunities.

presumably this is captured in the historical relationship between spawning stock size and recruitment. Thus, we are here referring to ecosystem benefits stemming from carcass retention other than this one, so as to avoid the possibility of double counting.

Reformulating the original harvest-only optimisation problem shown in (4-3) to include this consideration, we obtain a new version of the planner's problem:

$$\begin{aligned} &\max\ W(h) = \sum_{t=0}^{\infty} \rho^t \left\{ B(h_t) + V(X_T - h_t) - C(X_t, h_t) \right\} \\ &s.t.\ X_{t+1} = R\,(X_{t-2} - h_{t-2}, Q_t) \end{aligned} \tag{4-13}$$

After solving for the first order conditions and rearranging as previously, the long run equilibrium of the system is determined by the following system equivalent to (4-6):

$$\begin{aligned} &X - R(X - h, \overline{Q}) = 0, \quad \text{for } X_{t+1} = X_t = X \\ &\frac{V' + \rho^3 \left[p(h) - C_h - C_x \right] R_{X-h}}{p(h) - C_h} - 1 = 0, \quad \text{for } h_{t+1} = h_t = h \end{aligned} \tag{4-14}$$

Interestingly, the addition of a salmon carcass value to the problem does not change the statement for salmon habitat value, shown in (4-10) for the marginal case and (4-11) for the non-marginal case. However, the values derived from these expression will be different. Since the long-run equilibrium values for system variables are no longer the same, the calculation of habitat values from these two expressions will differ as well.

5. Estimating Coho Recruitment with Habitat Quality

In our analysis, we apply the bioeconomic valuation framework described in the previous chapter to freshwater habitat utilized by coho salmon. In this chapter, we develop a coho stock-recruitment relationship that incorporates habitat quality at 16 streams in the South Thompson River catchment. In the first stage of our biological analysis, we estimated an empirical relationship between trends in recruitment and an index of habitat quality for the 16 streams in our data set. We then specified a stock-recruitment model for coho salmon that incorporates the effects of freshwater habitat quality. Parameters for the model were calibrated by comparing projections of population abundance using the model to the observed empirical relationship between trends in recruitment and measures of habitat quality. Finally, we estimated the total freshwater capacity (expressed in terms of maximum smolt production) for Georgia Strait coho. This was then used to derive a stock-recruitment relationship for all wild coho in the Georgia Strait fishery. This relationship is later used in our bioeconomic model.

Methods

Empirical Relationship Between Trends In Abundance And Habitat Quality

We evaluated the effects of land use practices on 16 streams in the South Thompson drainage basin by fitting a relationship between trends in recruitment of adult coho and a semi-quantitative measure of 'habitat concerns'. These streams comprise 503.2 km of habitat (measured in terms of stream length) accessible to coho salmon and drain an area of approximately 7130 km^2. Abundance of recruits was not directly observable for each stream, so we estimated recruitment for each stream and year by using data on spawner escapement and a time series of exploitation rates for Thompson River coho salmon using the following relationships:

$$X_t = \frac{SP_t}{1-u_t}$$

$$SP_t = X_t - h_t \tag{5-1}$$

where X and h are recruitment measured as the exploitable stock and harvest, respectively; SP is the escapement or spawning population and u is the exploitation rate. Exploitation rate estimates were taken from coded-wire tag recoveries from Thompson hatchery smolt releases (Irvine *et al.,* 1999b) supplemented by DNA sampling of catches in fisheries during 1998 (Irvine *et al.*, 1999a). Data on spawner abundance were taken from the data set used by Bradford and Irvine (2000). Following the approach used by Bradford and Irvine (2000), we fit the following relationship to recruitment time series for the 16 streams in our data set for the period of 1988 to 1998:

$$\ln(X_t + 1) = \alpha + \beta t + e \tag{5-2}$$

where α and β are parameters, t is the year, and e is an error term. We refer to the slope of this equation (β) as the instantaneous average annual change in recruitment.

Estimates of spawner abundance for each stream were based on visual estimates of escapement from 1988 to 1998. However, recent changes in sampling effort and methods used for estimating abundance indicate that older visual escapement estimates may be biased downwards. To take account of this bias, we multiplied abundance estimates prior to 1988 by an inflation factor of 2.09, which was calculated by taking the mean ratio of abundance estimates using new and old sampling/estimation methods in 1998 and 1999 in the streams in our data set.

To evaluate the effects of freshwater habitat quality on recruitment of coho salmon, we compared the instantaneous average annual change in recruitment (β) for each stream with a semi-quantitative index of habitat concerns (Bradford and Irvine, 2000), which we refer to as the "habitat concerns index". This index has 10 major categories (forestry, agriculture, urbanization, recreation, mining, industrial development, linear development, hydro development, cumulative impacts, and special biophysical concerns) with 1-6 sub-categories in each group. Each issue was rated as either "low" or "high". We used the sum of "high" scores as an index of habitat concerns. Values for this index ranged from 1 to 19, with a weighted average of 9.6 (Figure 5-1).

We estimated a linear relationship between the instantaneous average annual change in recruitment for each stream (β) and the "habitat concerns index" (HCI),

$$\beta_i = \phi + \gamma HCI_i + v \tag{5-3}$$

where β_i is the slope estimate from equation (5-2) for stream i and HCI_i is the value of the "habitat concerns index" for stream i, $\acute{\phi}$ and γ are parameters, and v is an error term.

Stock-Recruitment Model For Coho Salmon

We used a modified Beverton-Holt stock-recruitment function to model the production of coho smolts (SM) as a function of spawner escapement (X_{t-2} - h_{t-2}) and habitat quality (Q):

$$SM(X_{t-2} - h_{t-2}, Q) = \frac{Qa(X_{t-2} - h_{t-2})}{1 + \frac{a}{b}(X_{t-2} - h_{t-2})} \tag{5-4}$$

In a standard Beverton-Holt relationship, a is the productivity parameter, which determines the number of smolts per spawner as spawner abundance approaches zero (in the absence of density-dependence), and b is the capacity parameter, which determines the maximum number of smolts produced by a stock. Q is an additional term, which we call the "habitat quality factor", which scales the entire relationship to reflect changes in quality of freshwater habitat. For freshwater habitat in pristine condition, Q=1, while Q<1 if habitat has been degraded.

With the addition of the "habitat quality factor" in the Beverton-Holt relationship, the effective productivity and capacity of a stock are given by the following equations:

$$\text{Productivity:} \quad \lim_{X_{t-2} - h_{t-2} \to 0} \frac{SM(X_{t-2} - h_{t-2}, Q)}{X_{t-2} - h_{t-2}} = aQ \tag{5-5}$$

Capacity: $$\lim_{X_{t-2}-h_{t-2}\to\infty} SM(X_{t-2}-h_{t-2},Q)=bQ \quad (5\text{-}6)$$

Hence, when Q=1, the effective productivity and capacity of a stock are given by *a* and *b*, respectively. Therefore, these parameters can be interpreted as the productivity and capacity of a stock (in terms of smolt abundance) with freshwater habitat in pristine condition.

Recruitment (R) is simply smolt abundance multiplied by the marine survival rate for coho salmon (m). We assumed that the marine survival rate is density-independent, hence, the stock-recruitment can be expressed as:

$$R(X_{t-3}-h_{t-3},Q) = \frac{Qma(X_{t-3}-h_{t-3})}{1+\frac{a}{b}(X_{t-3}-h_{t-3})} \quad (5\text{-}7)$$

Calibration of the Stock-Recruitment Model for South Thompson streams

We determined the productivity parameter (*a*) in the smolt production function (equation 5-4) from an empirical review of freshwater productivity of coho salmon (Bradford *et al.* 2000). Values for the *b* and *Q* parameters were determined by simulating the dynamics of the 16 streams in our data set and calibrating the parameter values to the empirical relationship estimated between trends in abundance and the "habitat concerns index" (equation 5-3).

a) Estimation of "a" parameter for the smolt production function

Bradford *et al.* (2000) estimated freshwater productivity of coho populations from 14 coastal streams from Oregon to British Columbia. Estimates of productivity averaged 85 smolts per female spawner. Productivity across streams was also quite variable, with a standard deviation of 35 smolts per female. We assumed a value of 40 smolts per spawner (80 smolts per female assuming a 1:1 sex ratio) for the *a* parameter in the stock-recruitment model for the South Thompson.

b) Simulation of South Thompson coho and calibration of "b" and "Q" parameters in the smolt production function

The steps in the simulation of South Thompson coho populations dynamics and calibration of the model to the empirical relationship between trends in abundance and the "habitat concerns index" are summarized in Figure 5-2. Each step is described in more detail below:

Step A: We calculated the value of the instantaneous average annual change in recruitment from the predicted empirical relationship for different values of the "habitat concerns index" (equation 5-3). We calibrated the model parameters for values of the "habitat concerns index" of 0, 1, 5, 10, 15 and 20 to represent the range of conditions for freshwater habitat in the South Thompson (Figure 5-1).

<u>Step B</u>: We first calibrated the habitat capacity (*b*) of the 16 streams in our data set under pristine conditions (i.e. Q=1). After calibrating *b*, we calculated values for the "habitat quality factor" corresponding to values of the "habitat concerns index" given in Step A.

<u>Step C</u>: We used historical values (1985 to 1988) of aggregate spawner abundance in the 16 streams in our data set for spawner abundance for the first three years of the simulation. For subsequent years, we used the following equation to calculate spawner abundance:

$$X_t - h_t = R(X_{t-3} - h_{t-3}, Q)(1 - u_t) \tag{5-8}$$

where $X_t - h_t$ is spawner escapement, $R(..)$ is recruitment, and u_t is the exploitation rate. We used historical value of exploitation rates for Thompson River hatchery smolts (Irvine et al 1999a, 1999b). Smolt abundance and recruitment were calculated using equations (5-4) and (5-7). We derived a historical time series of smolt-to-adult survival rates for South Thompson coho by taking the average smolt-to-adult survival for wild coho in the Salmon River and Black Creek (Simpson *et al.*, 1999).

<u>Step D</u>: After simulating the population dynamics of South Thompson coho from 1988 to 1998, we estimated the instantaneous average annual change in recruitment (β) from the time series of recruits generated by the simulation. The simulation model continued to loop over different values of the parameter being calibrated (*b* or Q) until the instantaneous average annual change in recruitment (β) from the simulation equaled the value predicted by the empirical relationship (equation 5-3) given the value of the "habitat concerns index".

Historical Abundance and Freshwater Habitat Capacity for Coho Salmon in Georgia Strait

The previous analyses described in this section provided a stock-recruitment relationship for coho in the 16 South Thompson streams in our data set as a function of habitat quality. However, coho from these streams are harvested along with many other stocks in the Georgia Strait fishery. In our bioeconomic model, we assume that coho stocks contributing to the Georgia Strait fishery are managed as a single unit. To derive a stock-recruitment relationship for the broader set of coho stocks supporting Canadian fisheries in Georgia Strait, we had to scale up the habitat capacity parameter in equation (5-7).

We derived estimates of habitat capacity for coho caught in Georgia Strait by analyzing data on commercial and sport catches of coho salmon by Canadian fisheries, releases of coho juveniles from hatcheries, and data on survival and exploitation rates. With this data, we reconstructed average historical abundance of wild coho salmon smolts contributing to Canadian fisheries in Georgia Strait for the 1977 to 1990 brood years. We chose 1977 as the first year in our reconstruction due to availability of sufficient data on survival rates for wild coho salmon. Our reconstruction did not include any brood years after 1990 due to precipitous declines in escapement for some coho stocks since that time. Inclusion of data past this date could negatively bias estimates of habitat capacity if stocks are no longer close to their freshwater capacity.

Sport and commercial catch data for southern British Columbia and Georgia Strait were taken from Simpson (1999). We estimated hatchery contributions to *southern B.C.* catch using data on juvenile releases and survival rates from hatcheries in British Columbia and Washington (Coronado and Hilborn, 1998), exploitation rates from B.C. hatcheries (Simpson *et al.*, 1999), and the geographic distribution of catches from hatcheries in B.C. and Washington (Weitkamp *et al.*, 1995). It should be noted that we could not directly estimate the contribution of hatcheries to *Georgia Strait* catch due insufficient spatial resolution of catch distribution data.

An estimate of coho catch of wild origin in southern B.C. was derived by subtracting hatchery contributions to catch from wild catch. We estimated the abundance of wild smolts contributing to southern B.C. fisheries by dividing estimated wild catch by average smolt-to-adult survival rates and exploitation rates. Survival rate data for the period of the reconstruction were available for one wild B.C. stock (Baillie *et al.*, 1999) and three wild Washington stocks (Anonymous, 1998). Exploitation rates were derived from two hatcheries in southern B.C. (Simpson, 1999). Abundance of wild smolts contributing to catch in Georgia Strait was estimated by multiplying wild smolts contributing to southern B.C. catch by the ratio of catches in Georgia Strait and southern B.C. from the 1977 to 1990 brood years.

We used our estimate of wild smolt abundance for Georgia Strait as our estimate of the *current* capacity of freshwater habitat contributing to this fishery. However, historical abundance reflects the state of freshwater habitat. To derive estimates for the capacity of Georgia Strait coho under pristine conditions, assumptions about the state of habitat conditions for the period of our catch decomposition are required. Unfortunately, no comparable measures of habitat quality were available for other coho streams in the Georgia basin. We derived estimates of pristine habitat capacity for Georgia Strait assuming that habitat is less degraded than our 16 South Thompson streams (HCI=5), similarly degraded (HCI=10), and more degraded (HCI=15).

Results

Stock recruitment model for 16 South Thompson Streams

Figure 5-3 summarizes the empirical relationship between the instantaneous average annual change in recruitment (β) from 1988 to 1998 and the "habitat concerns index" for the 16 streams in our data set. The negative slope of this relationship indicates that streams with more degraded habitat (i.e. a higher value for the "habitat concerns index") experienced faster declines in abundance of adult coho than streams with less degraded habitat. Parameter values from calibration of the stock-recruitment model for the 16 South Thompson streams are summarized in Table 5-1. The weighted average of the "habitat concerns index" for these streams was 9.6 indicating that the South Thompson was producing approximately 47 percent of the smolts that it would have under pristine conditions during the years for our simulations.

Table 5-1
Parameter Values for Stock-Recruitment Function for coho in 16 South Thompson streams as a Function Of Habitat Quality

Parameter Description	Parameter Value
Habitat productivity under pristine habitat conditions (*a*)	40.0
Habitat capacity under pristine habitat conditions (*b*)	170,909
Habitat quality factor (*Q*) as a function of the "habitat concerns index" (HCI)	
HCI = 0	1.00
HCI = 1	0.91
HCI = 5	0.67
HCI = 10	0.47
HCI = 15	0.34
HCI = 20	0.27

Average Smolt Abundance and Freshwater Habitat Capacity for Georgia Strait Coho

Catch of coho in sport and commercial fisheries in southern British Columbia averaged 2,954,036 per year for 1977-1990 brood years (Simpson *et al.*,1999). B.C. and Washington hatcheries released 14,421,429 and 27,357,143 juveniles per year, respectively, with average smolt-to-adult survival rates of 5.80 percent for coho from southern B.C. hatcheries and 4.96 percent for coho from Washington hatcheries (Coronado and Hilborn, 1998). Based on hatchery release and survival rate data, the average number of recruits generated by hatcheries in southern B.C. was 14,421,429 x 0.058 = 837,011 adult coho per year, while Washington hatcheries produced 27,357,143 x 0.0496 = 1,356,964 adult coho per year.

We used average exploitation rates for coho from Quinsam and Big Qualicum hatcheries (east coast of Vancouver Island) to calculate the total catch of coho from southern B.C. hatcheries. From the 1977-1990 brood years, exploitation rates for these two hatchery stocks averaged 74.7 percent (Simpson *et al.*, 1999). Greater than 90 percent of coho released from hatcheries in southern B.C. are caught in B.C. fisheries (Weitkamp *et al.*, 1995), so we assumed that 95 percent of southern B.C. coho releases are caught in southern B.C. fisheries. Some of these fish are likely caught in fisheries in northern British Columbia, but data were not available to take this into account. Based on these assumptions, hatcheries in southern B.C. contributed an average of 0.747 x 0.95 x 837,011 = 594,102 fish per year to catches in this region.

Time series of exploitation rates for Washington state hatcheries were not available, so we assumed a total exploitation rate of 80 percent for Washington hatchery coho. The geographic distribution of catch from Washington hatcheries varies considerably by hatchery location with 40 to 60 percent of releases being recovered in B.C. fisheries for most hatcheries along Puget

Sound, Hood Canal, the Straits of Juan de Fuca and along the Washington coast. However, fewer than 20 percent of coho from Columbia river hatcheries were recovered in B.C. fisheries (Weitkamp *et al.*,1995). The average proportion of Washington hatchery coho caught in B.C. fisheries was approximately 40 percent, hence we used this to calculate the contribution of Washington state hatcheries to southern B.C. catch. Based on these assumptions, Washington state hatcheries contributed an average of 0.8 x 0.4 x 1,356,964= 434,229 fish per year to southern B.C. fisheries.

We calculated the average catch of wild salmon in southern B.C. fisheries by deducting estimated catches of hatchery coho from total catch, which yielded an estimate of 2,954,036 - 594,102 - 434,229 = 1,925,705 wild coho caught annually in southern B.C. fisheries for the 1977-1990 brood years. We estimated the abundance of wild smolts contributing to southern B.C. fisheries by dividing our estimate of wild catch by exploitation rates estimated for B.C. hatchery stocks and average smolt-to-adult survival rates for four wild stocks. Survival rates for Carnation Creek, B.C. averaged 10.7 percent (Baillie *et al.*,1999) while survival rates from the Big Beef Creek, Deschutes, and Skykomish stocks in Washington averaged 18.7, 18.8, and 14.7 percent, respectively (Anonymous, 1998). The overall mean survival of coho through this period was 15.7 percent. We assumed wild stocks were subject to the same exploitation rate as B.C. hatchery stocks. Under these assumptions, average abundance of wild smolts feeding southern B.C. fisheries was 1,925,705 / (0.152 x 0.747) = 16,993,577 for the 1977-1990 brood years.

To estimate the number of smolts contributing to Georgia Strait fisheries, we multiplied wild smolt abundance for southern B.C. fisheries by the percent of total catch attributable to Georgia Strait fisheries. From 1980-1998, Georgia Strait fisheries accounted for 20.3 percent of total catch in southern B.C. (Simpson *et al.*,1999), yielding an estimate of 16,993,577 x 0.203 = 3,444,451 for wild smolts contributing to Georgia Strait fisheries. We used this an our estimate of the *current* capacity of freshwater habitat for Georgia Strait coho salmon.

Pristine Habitat Capacity for Georgia Strait Coho

Assuming that habitat in the Georgia basin is similar in quality to the South Thompson average (i.e. HCI = 10, Q = 0.47), the habitat capacity for Georgia Strait coho under pristine conditions (i.e. the 'b' parameter in the smolt production function) is equal to 3,444,451 / 0.47 = 7,257,527 smolts. If we assume that Georgia Strait stocks had an HCI = 5 and Q = 0.0.67 (i.e. they were less impacted than the South Thompson), this yields an estimated pristine habitat capacity of 5,132,059 smolts. Conversely, assuming that, on average, Georgia Strait stocks were more heavily impacted than the South Thompson (e.g. HCI = 15, Q = 0.34), pristine habitat capacity is equal to 9,998,558 smolts.

6. A Bioeconomic Model of a Commercial Coho Fishery

The next step in empirically testing the model presented in Chapter 4 is to specify and parameterize the functional forms for the general functions used in the model other than the coho stock-recruitment relationship estimated in the previous chapter. Once the model is fully specified we can solve the optimisation problem described in Chapter 4 and obtain optimal solution values for harvest, coho stock recruitment and fishing effort. For this study, we make the assumption that the Georgia Strait coho fishery is managed as a commercial troll fishery only, selling salmon into an international market where its price is exogenously determined. This situation does not reflect the current management emphasis on recreational fishing but is adopted here for several reasons. First, the focus is on developing and applying a methodology for valuing habitat and not on deriving management recommendations for the coho fishery. Use of a commercial fishery greatly simplifies the application, particularly in light of the difficulties experienced in previous modelling of the recreational coho fishery.[15] Second, the poor state of the current B.C. coast coho fishery suggests that valuation estimates based on present conditions would give unsatisfactory results that are not reflective of habitat values under better circumstances, such as those existing several decades ago. Thus, the following analysis is based on historical biological conditions (before much of the habitat was altered). Moreover, the solution of the model makes the further simplifying assumption that habitat is fully intact or pristine. The next chapter examines the case where habitat is allowed to degrade.

Functional Forms

The functions to be specified were presented in Chapter 4 and include gross harvesting benefits *B(h)*, a cost function *C(X,h)* and a function describing ecosystem services provided by the escaping stock *V(X-h)*, as an extension. In addition, we employ the recruitment function *R(X-h,Q)* estimated in the previous chapter. The gross benefits of salmon harvests under an assumption of commercial fisheries use only and an internationally determined price are simply:

$$B(h_t) = \int_0^h p(h_t)dz = ph_t \tag{6-1}$$

where p is the ex-vessel price of salmon less the skipper and crew shares expressed as a percentage of gross revenues. To derive the cost function, we begin with the following catch function:

$$h(X_t, E_t) = X_t(1 - e^{-qE_t}) \tag{6-2}$$

where q is the catchability coefficient and E is the total fishing effort expended over the fishing season, expressed in vessel days. Inverting (2) gives the following expression for the level of fishing effort that would produce a catch h when the fish stock is X.

$$E(X_t, h_t) = \frac{1}{q}\left[\ln X_t - \ln(X_t - h_t)\right] \tag{6-3}$$

[15] See the discussion of the Cameron and James (1987a) analysis of the chinook and coho fisheries in Chapter 3, where the authors found a *negative* value associated with increments in the exploitable coho stock. While the regulatory regime at the time likely accounts for this result, we cannot use this result nor in good conscience ignore it and assume a positive willingness to pay.

Pre-multiplying this expression by *c*, the unit variable cost of fishing effort, results in the desired cost function:

$$C(X_t, h_t) = cE_t = \frac{c}{q}\left[\ln X_t - \ln(X_t - h_t)\right] \tag{6-4}$$

A more complex specification was required for the recruitment function *R(X-h,Q)*, the details of which are provided in the previous chapter. The following expression for coho recruitment with an environmental influence (*Q*) was derived, based on the Beverton-Holt recruitment relationship:

$$R(X_{t-3} - h_{t-3}, Q) = \frac{Qma(X_{t-3} - h_{t-3})}{1 + \frac{a}{b}(X_{t-3} - h_{t-3})} \tag{6-5}$$

Inserting the above functions into the solution expressions for the long-run equilibrium or steady-state conditions that were derived earlier as (4-6), we get new expressions for the steady state. Multiplying the second expression derived from (4-6) by *p/p*, we get:

$$X - \frac{Qma(X-h)}{1 + \frac{a}{b}(X-h)} = 0$$

$$\frac{Qma(X-h)}{\left\{1 + \frac{a}{b}(X-h)\right\}^2} \frac{1 + \frac{c}{p}\frac{1}{qX}}{1 - \frac{c}{p}\frac{1}{q(X-h)}} = (1+\delta)^3 \tag{6-6}$$

These expressions constitute a two equation system in the variables *X* and *h* and can be solved to give the long run equilibrium values for coho harvest and exploitable stock, *h** and *X**, respectively.

Parameter Assumptions

To complete the analysis estimates are required for the parameters *Q, m, a, b, c, p* (or *c/p*)*, q* and *δ*. Since the biological (*a, b, m*) and habitat quality (*Q*) parameters were discussed in the previous section their derivation is not discussed here. We assume here that fully pristine conditions exist for habitat throughout the coho range feeding the Georgia Strait fishery; thus, we set $Q = 1.0$. To derive the appropriate value for the density dependent parameter of the recruitment function (*b*), we assume that current habitat throughout the range is degraded equivalently to the study sites within the South Thompson River drainage area where $Q = 0.47$, yielding $b = 7{,}257{,}527$. Finally, we assume an ocean survival rate of 20%, which is consistent with Department of Fisheries and Oceans (DFO) research on the level of natural mortality several decades ago in wild indicator stocks of coho (Simpson *et al*., 1999). Current survival is far lower (2% to 5%) and is unlikely to be sufficient for a viable commercial fishery under the economic and biological assumptions used in the present modelling exercise. This section discusses the economic assumptions of the model, summarized by the cost-price ratio *c/p*, catchability *q* and the social discount rate δ. All parameter estimates for the basic model are presented in Table 6-1.

The key economic input to the model is the price of coho and the harvest cost assuming a commercial troll fishery. Fortunately, there is a recent study that has examined the costs and returns for the commercial coho fishery based on 1994 results and we make extensive use of these data (ARA, 1996). For the price of salmon, the ARA study cites a weighted 1994 value of $2.66 per lb, which we assume is an ex-vessel price. The study also indicates an average whole fish (Rd) weight of 6.56 lb, yielding a price of $17.45 per fish (Rd). Since the skipper and crew are typically paid a share of gross revenues, we simply deduct this from gross price per fish. The ARA study cites an average share of about 40% in the coho troll fishery, leaving a 'net' price of approximately $10.50 per fish.

Average annual costs for the trolling fleet are also provided in the ARA study. Based on 1138 vessels in 1994 fishing an average of 10 weeks (70 days) each, the average daily cost per vessel can be calculated. To accurately capture the welfare benefits of the salmon catch (that we later use to measure the value of salmon habitat), we include only those variable costs associated with daily decisions to enter the fishery. We ignore 'sunk' costs such as insurance, off-season repairs, interest, depreciation, etc. We also leave out license fees as these represent a share of the economic rent appropriated by the resource owners. On a daily per vessel basis, we are left with the following cost items, again in 1994 dollars:

▪ fuel	$47.20
▪ food/trip expenses	$38.66
▪ gear repairs	$ 6.40
▪ other goods and services	$16.70

▪ total (per boat-day)	$108.96

Based on the above cost and price assumptions, the base case modelling uses a cost-price ratio of 10.4.

Estimates of the catchability coefficient (*q*) are available from an earlier management study of the Georgia Strait chinook and coho by Argue *et al.* (1983). The authors model the coho stock and estimate catchability for the troll fleet, allowing this to vary over the fishing season. Their estimates of *q* were from about 0.00001 to 0.00003. Since the latter value produced more meaningful results in our model, we set *q* equal to this value in the calculations and assume it remains constant. Finally, we assume a social discount rate of 5%. The debate over the appropriate rate to use in analysing public investments in Canada has raged since at least Burgess (1981) and most researchers employing a mixed opportunity cost of capital and social time preference approach accept a value between 5% and 10%. Since we are exclusively interested in a social time preference rate here, we select the lower value for the base case calculations and consider some alternatives in the sensitivity analysis.

Table 6-1
Parameters Used for the Base Case in the Empirical Analysis ($ 1994)

Parameter	Units	Value	Source/Comment
Habitat factor in 16 S. Thompson streams, Q	n.a.	0.47	Bradford and Irvine 2000
Habitat factor for calculations, Q	n.a.	1.0	
Ocean survival, m	n.a.	0.20	Bradford and Irvine 2000
Density independent recruitment parameter, a	n.a.	40	this study
Density dependent recruitment parameter, b	n.a.	7,257,527	this study
Unit variable cost of effort, c	$/boat-day	109	ARA 1996
Salmon price, ex-vessel, p	$/lb Rd	2.66	ARA 1996
Crew share of gross revenue	%	40	ARA 1996
Average salmon biomass, b	lb Rd/fish	6.56	ARA 1996
Cost-price ratio, c/p	n.a.	10.4	
Commercial catchability, q	n.a.	0.00003	Argue *et al.* 1983
Social discount rate, δ	%	5.0	

Results for an Optimally Managed Commercial Troll Fishery

Solving the fully specified bioeconomic model presented in the previous section requires the insertion of the parameters contained in Table 6-1 into the two equation system in (6-6). To solve the equation system we used the computer software program Maple 6.0. Two cases are presented below: one considers a direct use or harvest value only, while the second adds an ecosystem service or indirect use value.

Case 1: Optimal Values with No Ecosystem Services

Not surprisingly, the empirical model is characterized by multiple solutions for recruitment and harvest, X and h. Of the three system solutions produced, one contains negative values for recruitment and harvest, and another yields a negative optimal escapement level, which is not permissible. Thus, we are left with a single permissible solution, with the solution values shown in Table 6-2. The optimal nature of a solution value is indicated with an (*).

Table 6-2
Optimal Solution Values for the Strait of Georgia Coho Fishery Model
No Ecosystem Services ($ 1994)

Variable	Optimal Value	Historical Average (1953-77)
Recruitment (stock), X^*	1,071,000 fish	714,000 fish
Catch, h^*		
- commercial	560,000 fish	227,000 fish
- other	0 fish	370,000 fish
Escapement, $X^* - h^*$	521,000 fish	117,000 fish
Fishing Effort, E^*	24,665 boat-days	25,000 boat-days
Economic rent		
- annual, $ph^* - cE^*$	$3.2 million/year	n.a.
- *PV* {ph* - cE*}	$63.8 million	n.a.

Note: present value (*PV*) calculated in perpetuity with a 5% discount rate; historical commercial catch is troll only – total catch including other fisheries averaged 597,000 fish; fishing effort is commercial troll only; source for historical data is Argue *et al.* (1983).

Table 6-2 indicates that under the assumptions made, optimal escapement would be 521,000 fish and the exploitation rate would be 52%. This compares with the historical data in Table 6-2 from Argue *et al*. (1983) which indicate a lower historical value for the stock and a higher total catch, consistent with a non-optimally managed fishery from an economic rent maximization perspective. By way of comparison, the catch per boat-day (CPUE) under historical conditions was just over 9 fish while the optimal management results yield an average CPUE of about 23 fish per boat-day. Of course, this analysis concentrates on rent earned in the fishery and ignores the many other objectives of management that no doubt played a role in the management of the fishery. We also exclude any consideration of the native fishery catch, which ranged from 4000 to 25,000 fish/year over the period 1965-77. In the next section the model is extended to include an ecosystem service performed by spawning salmon.

Case 2: Optimal Values with an Ecosystem Service

In this extended model we incorporate a preliminary assessment of one ecosystem service performed by spawning salmon: nutrient cycling between marine and terrestrial aquatic ecosystems. In this study we value this indirect use benefit by determining the financial price one would have to pay to replace the nutrients no longer being transferred from the marine environment by returning salmon. This 'replacement cost' approach is suggested by a continuing study being conducted by the British Columbia Ministry of Fisheries.[16] As part of a restoration effort, fertilizer pellets have been applied to the Keogh River in northern British Columbia to mimic the effect of rotting salmon carcasses with very positive affects on the number of surviving juvenile salmon and returning spawners (Bruce Ward, personal communication).

The pellets have been specially designed to mimic the release of nutrients from a decomposing salmon carcass. Each pellet weighs about 8 grams, and is equivalent to the phosphorous content in approximately 450 grams wet weight of salmon carcass. One metric ton of pellets costs $2200 or $0.02 per pellet, resulting in a cost of $0.13 to replace the nutrients contained in a 6.5 lb carcass. The number of spawning salmon is then multiplied by this cost to derive a value for the nutrient cycling facilitated by the returning salmon as follows:

$$V(X_t - h_t) = v(X_t - h_t) \tag{6-7}$$

where v is the unit value per fish of nutrient released from salmon carcasses expressed in pellet-equivalent terms. As indicated above, we assume $0.13 per fish as a proxy for this benefit.[17]

Chapter 4 presented the general model reformulated to include the ecosystem benefit and here we simply use the same specifications for functional forms as in the previous section with the addition of (6-7) above. Solving the extended model provides new estimates for the optimal solutions presented in Table 6-2; both sets of estimates are presented in Table 6-3.

As can be noted from Table 6-3, the addition of an indirect use value has several impacts. First, it increases the optimal escapement level since escaping fish are now slightly more valuable (about $0.13 each). This change is associated with a higher optimal recruitment figure and a lower optimal harvest. However, since this harvest now can be taken more easily (i.e. we are taking fewer fish from a larger exploitable stock) the optimal fishing effort decreases significantly. As would be expected if an additional benefit is added to the optimisation problem, the net economic value of the fishery is higher, by about $60,000 per year or $1.3 million in present value terms. The next chapter uses the results presented here and in the previous section to derive the values associated with salmon habitat quality using a portion of the South Thompson drainage basin as a case study.

[16] Under the replacement cost method of non-market valuation, the analyst estimates the value of a natural asset by calculating the cost of replacing its services, often with a human-produced substitute (Knowler and Lovett 1996).

[17] Note that we have not adjusted this value for the portion of the benefit accruing to young salmon who experience better feeding conditions because of the nutrient cycling benefit, which would avoid the double counting referred to in Chapter 4. Since we have approximated a benefit that in reality is far more complex and likely larger than what we have assumed here (e.g. wildlife support values, etc.), we chose to keep the calculations simple.

Table 6-3
Optimal Solution Values for the Strait of Georgia Coho Fishery Model Including Carcass Value as an Ecosystem Service ($ 1994)

Variable	Optimal Value (No Ecosystem Service)	Optimal Value (With Ecosystem Service)
Recruitment (stock), X^*	1,071,000 fish	1,105,000 fish
Catch, h^*	560,000 fish	526,600 fish
Escapement, $X^* - h^*$	521,000 fish	578,400 fish
Fishing effort, E^*	24,665 boat-days	21,580 boat-days
Economic rent		
- annual, $ph^* - cE^*$	$3.19 million/year	$3.25 million/year
- PV {ph* - cE*}	$63.8 million	$65.1 million

Note: present value (*PV*) calculated in perpetuity with a 5% discount rate

7. Valuing Coho Habitat: Results and Discussion

The previous chapter used the biological results from Chapter 5 and specified the other functional forms and parameters of the bioeconomic model to obtain optimal solution values for a hypothetical Strait of Georgia commercial coho fishery. This chapter uses these empirical results to value changes in habitat quality, as expressed through the welfare impact of these changes in a hypothetical commercial coho fishery.

The Marginal Value of Salmon Habitat

We begin by calculating the marginal value of habitat quality. In effect, we ask the question: what is the value associated with a small change in the basin-wide habitat factor of, say, one percent? Once the system's optimal solution (equilibrium) values are determined via the procedure used in the previous chapter, they can be inserted into the following statement for the marginal value of habitat, corresponding to (4-10):

$$\frac{\partial W(X^*,h^*;\overline{Q})}{\partial \overline{Q}} = \frac{0.01}{(1+\delta)^3}\left[p + \frac{c}{qX}\right]\frac{ma(X_t - h_t)}{1 + \frac{a}{b}(X_t - h_t)} \tag{7-1}$$

where *W(X*, h*; Q)* measures welfare in economic rent terms as a function of the optimal recruitment (stock), harvest and a fixed level of environmental quality governing the entire basin supplying recruits to the Strait of Georgia exploitable coho stock. This change could result from a very minor amount of degradation spread evenly throughout the basin or a small isolated disturbance that substantially degrades a small area within the basin. Inserting the base case solution values for X^* and h^* from Table 6-2, we find that a basin-wide change in the habitat factor Q of 0.01 (recall that Q is an index value between 0 and 1.0) leads to a welfare change in the commercial coho fishery of $128,550 per year.

This figure captures the impact on economic rent in the fishery from habitat changes where there is mixing of stocks from various sources within the basin. However, it assumes no adjustment in fishing effort as habitat changes, to accommodate a new sustainable level of catch. Thus, the resulting marginal valuation estimate is different than the result obtained if this adjustment is incorporated, as occurs with the non-marginal valuation estimate in the next section (which is comparatively lower, since habitat degrades).

The Value of a Non-Marginal Change in Salmon Habitat

The comparative static effects of a non-marginal change in environmental quality in the dynamic model can be analysed readily as well. The social returns from optimal harvesting over a designated time period are determined for the situation before a change in environmental quality and then compared with the social returns after the change. The difference represents the welfare effect of the change. Altering habitat has implications for the optimal harvest rate and stock level in equilibrium, so that these might be expected to differ in the before and after scenarios. Drawing on Chapter 4, the proper formulation for estimating the relevant welfare effect in the dynamic model, assuming an infinite time frame and optimal management is:

$$\Delta W(X^*, h^*; Q) = W(X_A^*, h_A^*; Q^A) - W(X_B^*, h_B^*; Q^B) \qquad (7\text{-}2)$$

For the non-marginal analysis in the coho model, the system was solved using (6-6), with the two alternate values of Q, Q^A and Q^B, inserted. We then employed expression (7-2), inserting the optimal system values obtained from the 'with' and 'without' protected status situations.

To establish the required values for Q^A and Q^B, we assumed the entire pristine area of the South Thompson is to be affected by a disturbance, and that the average pre-disturbance value of Q for the entire basin is 1.0. This latter value is Q^A. To derive Q^B, measuring habitat quality after the change, we need the share that the South Thompson contributes to total Georgia Strait smolt production, which we estimate at 2.3%. Furthermore, we assume that disturbance reduces the habitat quality within the 16 study streams in the South Thompson drainage area from 1.0 to 0.47, which represents the habitat factor for this portion of the South Thompson at the present time. Thus, the value for Q^B will be: 0.977 X 1.0 + 0.023 X 0.47 = 0.9878. We then re-solve the model using this value in place of Q = 1.0. Solution values resulting from this procedure are shown in the third column of Table 7-1.

Table 7-1
Optimal Solution Values for the Strait of Georgia Coho Fishery Model
Base Case Situation ($ 1994)

Variable	Optimal Values (Pristine S. Thompson, *Q* = 1.0)	Optimal Values (Degraded S. Thompson, *Q* = 0.9878)	Difference Attributable to Habitat Change
Recruitment (stock), X*	1,071,000 fish	1,056,500 fish	14,500 fish
Catch, h*	560,000 fish	548,500 fish	11,500 fish
Escapement, X* - h*	521,000 fish	508,000 fish	13,000 fish
Fishing effort, E*	24,665 boat-days	24,407 boat-days	258 boat-days
Economic rent			
- annual, ph* - cE*	$3.19 million/year	$3.10 million/year	$90,000/year
- PV {ph* - cE*}	$63.8 million	$62.0 million	$1.8 million

Note: present value (*PV*) calculated in perpetuity with a 5% discount rate

Table 7-1 also shows the difference between the optimal values calculated for each of the two habitat quality levels. Of particular interest are the welfare losses arising from the degradation of habitat. The model indicates that the modification in a portion of the South Thompson drainage area would lead to about a 1.2% decline in the habitat factor (from 1.0 to 0.9878) and that this would result in a loss of $90,000 per year in annual economic rent or $1.8 million in present value terms. To place this on a more useful basis, we can note that the 16 South Thompson streams studied constitute a drainage area of about 7130 km^2 (713,000 ha). Distributing the loss associated with habitat change over this entire basin area leads to a value of about $2.52 per ha attributable to the salmon habitat service provided by pristine parkland. Since the study portion of the South Thompson contains 503.2 km of spawning stream length, the level of degradation of this habitat assumed above amounts to $3580 per km.

Including an Ecosystem Service

As an extension, we considered the case where spawning coho salmon contribute nutrients to the aquatic ecosystem as an additional social benefit of the salmon run. The impact of habitat modification can be analysed under this set of assumptions as well. Table 7-2 shows the results of this calculation. Clearly, the presence of an ecosystem service performed by spawning salmon does not have a pronounced impact on the estimated habitat values when a comparison is made with the values in Table 7-1 (subject to rounding error). However, as noted above, there are marked differences in the optimal values with the ecosystem service included and this is evident in the second and third columns of Table 7-2.

Table 7-2
Optimal Solution Values for the Strait of Georgia Coho Fishery Model Including Carcass Value as an Ecosystem Service ($ 1994)

Variable	Optimal Values (Pristine S. Thompson, *Q* = 1.0)	Optimal Values (Degraded S. Thompson, *Q* = 0.9878)	Difference Attributable to Habitat Change
Recruitment (stock), X*	1,105,000 fish	1,090,000 fish	49,500 fish
Catch, h*	526,600 fish	514,600 fish	37,500 fish
Escapement, X* - h*	578,400 fish	575,400 fish	12,000 fish
Fishing effort, E*	21,580 boat-days	21,295 boat-days	1242 boat-days
Economic rent			
- annual, ph* - cE*	$3.25 million/year	$3.16 million/year	$95,000/year
- PV {ph* - cE*}	$65.1 million	$63.1 million	$1.9 million

Note: present value (*PV*) calculated in perpetuity with a 5% discount rate; figures are subject to rounding error

This chapter has produced a set of values attributable to the quality of habitat used by the Strait of Georgia coho salmon stocks. Initially, we estimated the value associated with a very small or marginal change in basin-wide habitat, which provides an overall benchmark for valuing habitat as an input into smolt production. We estimate that a 1 percent change in the habitat factor (*H*) results in a welfare loss in our hypothetical coho commercial troll fishery of $128,550 per year, under the assumption the fishery is managed so as to maximize economic rent. For a more substantial or non-marginal change in habitat, we considered a pristine South Thompson watershed and allowed this to be degraded to approximately its current average state. This assumption resulted in a non-market loss of ecosystem services in the range of $2 to $3 per ha of drainage basin or about $3580 per km of spawning stream length. Results are slightly higher when the value of the ecosystem service provided by salmon carcasses is included: these are now $2.66 per ha of watershed and $3776 per km of spawning stream.

Sensitivity Analysis

Table 7-3 presents a sensitivity analysis of the study results. The approach taken involves varying a number of key parameters used in the estimation procedure. Base case values used for these parameters were provided in Table 6-1. The emphasis here is on maintaining the key assumptions about habitat degradation, so that habitat is allowed to degrade in the same way as in base case calculations. Thus, the habitat factor *Q* declines from 1.0 in the South Thompson drainage to 0.47 and this reduces *Q* for the entire Strait of Georgia basin from 1.0 to 0.9878. Instead, the sensitivity analysis concerns such parameters as the social discount rate, prices, recruitment parameters, etc. all values are expressed as present values in 1994 prices in perpetuity using a 5% discount rate, except where the latter parameter is varied.

As indicated in Table 7-3, the impact of raising the discount rate from 5% to 10% reduces the habitat value per ha by about 50%, while reducing the catchability coefficient from 0.00003 to

0.00002 has only a relatively small effect. Variations in the salmon price from the base case value of $10.50 per fish have pronounced impacts on habitat value, especially an increase to $15 per fish which raises the habitat value per ha to over $4. As might be expected a halving of the ocean survival rate from 20% leads to a dramatic reduction in habitat value: if fewer salmon survive the rearing stage to eventually contribute to commercial harvests, the value of the spawning and rearing habitat is lower. While variations in the 'a' and 'b' recruitment parameters lead to changes in the habitat value (especially for the latter), the relatively small changes given the size of the sensitivity variations gives some comfort. Thus, despite some inevitable uncertainty over the salmon population dynamics inherent in the recruitment estimation procedure, the results remain within a fairly narrow range.

Table 7-3
Sensitivity Analysis Results for the Strait of Georgia Coho Fishery Model
Base Case Situation
(all values are NPV in perpetuity at 5%; $ 1994)

Sensitivity Cases	Optimal Values (Pristine, Q = 1.0) ($)	Optimal Values (Degraded, Q = 0.9878) ($)	Difference due to Habitat Change ($)	Difference on per ha basis ($/ha)
1. Social discount rate				
(δ = 10%)	31,776,301	30,841,888	934,413	1.31
2. Commercial catchability				
(q = 0.00002)	41,018,524	39,450,222	1,568,302	2.20
3. Salmon price				
(p = $7.50, c/p = 14.5)	32,234,129	31,067,669	1,166,460	1.64
(p = $15, c/p = 7.25)	114,671,980	111,748,082	2,923,898	4.10
3. Ocean survival				
(m = 0.10)	4,729,165	4,359,413	369,752	0.52
4. Density indep. param.				
(a = 50)	72,294,877	70,286,141	2,008,736	2.82
(a = 30)	52,034,316	50,352,551	1,681,765	2.36
8. Density depend. param.				
(b = 9,000,000)	92,028,703	89,563,060	2,465,643	3.46
(b = 5,500,000)	36,841,990	35,565,880	1,276,110	1.79

8. Policy Implications and Areas for Further Research

This section concludes the study with a brief overview of the policy implications of the results for habitat and fisheries management, with an emphasis on the limitations of the study estimates. This is followed by a discussion of the directions for future research stemming from the study. In particular, it is argued that inter-disciplinary collaboration is essential and that there are valuation techniques available for assessing a wide range of park or natural area values. Of particular interest are such techniques as the production function approach (used in this study) and choice modelling.

Policy Implications

Although the analysis reported in this study is of a preliminary nature, refinements are unlikely to change the results we obtained very dramatically. Thus, the value of restoring coho salmon spawning and rearing habitat in the South Thompson drainage area to pristine condition (i.e. park-like) is several dollars per hectare of watershed area. This conclusion is fairly robust over a range of sensitivity variations in key parameters, given the base case description of watershed degradation maintained throughout the study. Nonetheless, these conclusions rely on a few important assumptions that are worth restating:

- We assumed that the quality of the habitat throughout the basin contributing to the exploited aggregate coho stock in the Strait of Georgia was pristine, except for the portion within the South Thompson drainage area that was the subject of study. Clearly, this is not the case in reality as much of the basin has been modified. Incorporating this additional realism would not be difficult but is unlikely to change the results substantially.

- We considered only a commercial coho fishery and ignored the recreational coho fishery, which is at least as important. However, difficulties in using an earlier study of the value of the recreational coho fishery prevented us from including this element. As a result, our results are likely to be a lower bound estimate of the fishery since a commercial fishery places a lower value on each fish harvested.

- Since pacific salmon stocks have been subject to longer term shifts in ocean survival rates which are not fully understood, we were required to make assumptions about these in the future. It is clear that without some recovery in these survival rates, no commercial fishery will be viable so that we assumed for our analysis that these would eventually be restored to historic levels.

- In concentrating only on coho salmon we have ignored the possibility that other species of salmon (or other fish) may rely on the same habitat and thereby increase its value as fish habitat. In a related sense, if habitat destruction leads to closure of the coho fishery, this may result in a loss that is greater than the foregone coho catch. If protecting the coho stock requires the closure of associated fisheries that co-exist with that of coho, then these additional losses should be debited to the initial loss of coho habitat. We have not accounted for this aspect in our results.

Despite these limitations, it is clear that conserving freshwater fish habitat within protected areas has a clearly identifiable production benefit. Along with the other benefits of protected areas, this benefit can be stacked up against the benefits of alternative uses of park land. Such valuation estimates serve to strengthen the case for protecting valuable ecosystems not so much as an end in itself, but as a means of supporting economic activity. These benefits are in addition to the inherent values of pristine lands that cannot be so easily quantified or valued.

Directions for Future Research

Future research could usefully concentrate on a number of issues. Improving the estimates made in this study would not be difficult and would rely on better physical and economic data. In this instance, the researchers were fortunate to have access to land use data that could be used within a bioeconomic modelling framework quite easily. Such data is relatively rare and helps explain why so few such studies have been undertaken previously. In addition, inter-disciplinary collaboration between economists and scientists is critical to successful valuation work. While economists usually derive the final estimates, these are only as good as the physical data on which these estimates are based. Good collaboration means that economists are involved early in the development of physical data collection and research protocols and that natural scientists are involved in refining economic models and developing parameter estimates.

Moreover, this study concentrated on a single indirect use value associated with park lands: this was freshwater habitat used by fish. The concept of *total economic value* (TEV) was discussed in Chapter 2 and it provides a useful framework for classifying environmental values (see Table 8-1). Under the TEV framework, park areas support a vast number of values associated with direct uses (e.g. recreation, tourism, selective timber harvests, hunting, etc.), as well as providing many ecosystem services that yield indirect use values to humans (e.g. watershed protection, carbon sequestration, etc.). On top of these are non-use values associated with pristine wilderness areas that stem from individuals' willingness to pay for the mere existence of a park site, even though they never plan to visit or benefit directly. Future research should take the full range of parkland values into account and select non-market valuation techniques that are appropriate to the case study in question. While we cannot go into detail here, Table 8-2 suggests valuation techniques that can be used to capture the various use and non-use values.

For example, this study made use of the *production function* technique to derive values for salmon habitat. This technique is especially useful for estimating indirect use values that contribute to some marketed or commercial economic activity. Thus, environmental quality influences on commercial recreational or tourism operations are highly amenable to the application of production function techniques. Other techniques that are being recognized as increasingly important in assessing park values include *choice modelling (stated choice)*, which allows for the comparison of different park use scenarios and then derives estimates of the underlying values associated with selected attributes of the park area by asking individuals to make tradeoffs between these options.

Table 8-1
Classification of Total Economic Value for Forested Parkland

USE VALUES a/		NON-USE VALUES (Existence Value)
Direct Use Values	**Indirect Use Values**	
- timber products	- nutrient cycling	- biodiversity b/
- fruits, vegetables, fungi	- hydrological regulation	- culture, heritage
- game animals, fish	- control of soil erosion	
- flowers, fodder	- amelioration of climate	
- medicinal plants	- weather damage protection	
- recreation and tourism	- groundwater recharge	
- education and research	- greenhouse gas sink	
- human habitat	- ecosystem stability b/	

Source: adapted from Barbier (1991), Panayotou and Ashton (1992), Myers (1992) and Pearce and Warford (1993)

a/ Use values are now usually considered as encompassing option and quasi-option values.
b/ Biodiversity is essentially an attribute of the forest; hence, it may also serve important direct and indirect use values. For example, the diversity found in a forest may have direct use value for scientific research, education and as a source of genetic material. Similarly, biological diversity may have an indirect use value in assisting the ecological stability of the entire tropical forest system.

Table 8-2
Selected Valuation Techniques for Assessing Values of Forested Parkland

USE VALUES		NON-USE VALUES (Existence Value)
Direct Use Value	**Indirect Use Value**	
- market or shadow prices	- changes in productivity	- contingent valuation
- changes in productivity	- production function	
- hedonic price method	- damage costs avoided	
- travel cost method	- preventive expenditures	
- production function	- relocation costs	
- direct/indirect substitutes	- replacement costs	
- contingent valuation		
- stated choice		
- indirect opportunity cost		
- replacement costs		

Source: adapted from IIED (1994)

In some cases, parks planners may be able to use value estimates from other sites and transfer these to the site under study. This approach is referred to as *benefits transfer*. Benefits transfer studies are often the only recourse where data is poor or funds are not sufficient for a full-scale valuation study. Whether this practice is advisable depends on a number of factors, not least of which is the similarity of the sites. For recreation, there can be difficulty in using benefits transfer, since values tend to be highly reliant upon site and sample population characteristics. Where visual attributes are at stake, there is liable to be even more problem with the use of benefits transfer. This problem is diminished when the benefits transfer site is located within the same region. For example, as more valuation studies are completed for British Columbia, this will create opportunities for benefits transfer within the province. Where demand or value functions are reported in original studies, these should be used along with variable observations for the site or population under study (benefits function transfer), rather than using simple average unit values from the source study. A decision about whether to use benefits transfer or to proceed with original data gathering, to estimate some parkland value, must weigh the costs of collecting primary data against the disadvantages of not having such information.

Additionally, there is considerable interest in applying spatial analysis to ecosystem valuation problems. This is particularly relevant for the valuation of natural areas where linkages among spatially separated locations affect the viability of some components of natural ecosystems. For example, wildlife populations often require certain habitat connections for dispersion among sites or to aid re-colonizers, if certain sites become less productive due to human or natural disturbances. GIS-based analysis can also be integrated into ecosystem valuation where there is a need to assess values on a spatially disaggregated basis (e.g. marine protected areas).

References

Anonymous. 1998. 'Speaking for the Salmon' workshop record. June 18, 1998. Simon Fraser University. British Columbia. Canada.

ARA Consulting. 1996. *The Economic Value of Salmon - Chinook and Coho in British Columbia*. Prepared for DFO, B.C. Ministry of Agriculture, Fisheries & Food, Ministry of Environment, Lands & Parks and Ministry of Small Business, Tourism & Culture, Vancouver.

Argue, A., Hilborn, R., Peterman, R.M., Staley, M., and Walters, C. (1983). *Strait of Georgia Chinook and Coho Fishery*. Ottawa, Canada: Department of Fisheries and Oceans.

Baillie, S., Patten, B., Till, J., Simpson, K., Luedke, W., and Tschaplinski, P. (1999). *Assessment of Coho Stocks on the West Coast Of Vancouver Island*. CSAS Research Document 99/166. Ottawa, Canada: Department of Fisheries and Oceans.

Barbier, E. (1991). *The Economic Value of Ecosystems: 2 - Tropical Forests*. Gatekeeper Series GK 91-01. London, England: London Environmental Economics Centre.

Barbier, E. Acreman, M., and Knowler, D. (1997*). Economic Valuation of Wetlands - A Guide for Policymakers and Planners*. Gland, Switzerland: Ramsar Convention Bureau/IUCN.

Bell, F. (1972). Technological externalities and common property resources: an empirical study of the U.S. Northern lobster fishery. *Journal of Political Economy*, 80, 148-158.

Ben-David, M. (1997). Timing of reproduction in wild mink: the influence of spawning Pacific salmon. *Canadian Journal of Zoology*, 75, 376-382.

Bilby, R., Fransen, B., Bisson, P., and Walter J. (1998). Response of juvenile coho salmon and (*Oncorhynchus kistusch*) and steelhead (*Oncorhynchus mykiss*) to the addition of salmon carcasses to two streams in southwestern Washington, U.S.A. *Canadian Journal of Fisheries and Aquatic Sciences*, 55, 1909 – 1918.

Bradford, M. (1998). *A Risk Assessment for Thompson River Coho Salmon.* Canadian Stock Assessment Secretariat Research Document 98/92.

Bradford, M. (1999). Temporal and spatial trends in the abundance of coho salmon smolts from western North America. *Transactions of the American Fisheries Society*, 128, 840-846.

Bradford, M. and Irvine, J. (2000). Land-use, fishing, climate change and the decline of Thompson River, British Columbia, Coho Salmon. *Canadian Journal of Fisheries and Aquatic Science*, 57, 13-16.

Bradford, M., Myers, R., and Irvine, J. (2000). Reference points for coho salmon (*Oncorhynchus kisutch*) harvest rates and escapement goals based on freshwater production. *Canadian Journal of Fisheries and Aquatic Sciences*, 57, 677-686.

Cameron, T. (1988). A new paradigm for valuing non-market goods using referendum data: maximum likelihood estimation by censored logistic regression. *Journal of Environmental Economics and Management*, 15, 355-379.

Cameron, T. and James, M. (1987a). Efficient estimation methods for 'close-ended' contingent valuation surveys. *Review of Economics and Statistics,* 69, 269-276.

Cameron, T. and James, M. (1987b). *The Determinants of Value for a Recreational Fishing Day: Estimates from a Contingent Valuation Survey*, Canadian Technical Report of Fisheries and Aquatic Sciences No. 1503. Vancouver, Canada: Regional Planning and Economics Branch, Department of Fisheries and Oceans.

Cederholm, C., Kunze, M., Murota, T., and Sibatani, A. (1999). Pacific salmon carcasses: essential contributions of nutrients and energy for aquatic and terrestrial ecosystems. *Fisheries*, 24, 6-15.

Clark, C. (1976). A delayed-recruitment model of population dynamics, with an application to Baleen Whale populations. *Journal of Mathematical Biology*, 3, 381-391.

Clark, C. (1990). *Mathematical Bioeconomics,* Second Edition. New York, NY: John Wiley and Sons.

Conrad, J. (1995). Bioeconomic models of the fishery. In Bromley, D. (Ed). *Handbook of Environmental Economics*. Oxford and Cambridge, England: Blackwells.

Coronado, C. and Hilborn, R. (1998). Spatial and temporal patterns affecting survival of coho salmon (*Oncorhynchus kisutch*) in the Pacific Northwest. *Canadian Journal of Fisheries and Aquatic Sciences*, 55, 2067.

Costanza, R., Farber, S., and Maxwell, J. (1989). Valuation and management of wetland ecosystems. *Ecological Economics*, 1, 335-361.

DFO. (1998a). *Salmon Watershed Planning Profiles for the South Thompson-Shuswap Habitat Management Area*. Vancouver, Canada: Department of Fisheries and Oceans, Government of Canada.

DFO. (1998b). *Salmon Watershed Planning Profiles for the Thompson-Nicola Habitat Management Area*. Vancouver, Canada: Department of Fisheries and Oceans, Government of Canada.

Dowlatabadi, H., Goulder, L., and Kopp, R. (1994). *Integrated Economic and Ecological Modeling for Public Policy Decision Making*. Discussion Paper 94-37. Washington, D.C.: Resources for the Future.

DPA Group (1988). *British Columbia Salmon Fleet Financial Performance 1981-1985*. Vancouver, Canada: Department of Fisheries and Oceans Canada.

Ellis, G. and Fisher, A. (1987). Valuing the environment as input. *Journal of Environmental Management*, 25, 149-156.

Fisher, A, Hanemann, M. and Keeler, A. (1991). Integrating fishery and water resource management: a biological model of a California salmon fishery. *Journal of Environmental Economics and Management*, 20, 234-261.

Freeman, A.M. (1991). Valuing environmental resources under alternative management regimes. *Ecological Economics,* 3, 247-56.

Freeman, A. (1993). *The Measurement of Environmental and Resource Values*. Baltimore, MA: Johns Hopkins University Press.

Freeman, A. (1995). The benefits of water quality improvements for marine recreation: A review of the empirical evidence. *Marine Resource Economics*, 10, 385-406.

Hall, D. (1977). A note on natural production functions. *Journal of Environmental Economics and Management*, 4, 258-264.

Hildebrand, G., Farley, S., Robbins, C., Hanely, T., Titus, K., and Servheen, C. (1996). Use of stable isotopes to determine diets of living and extinct bears. *Canadian Journal of Zoology*, 74, 2080-2088.

Holmlund, C., and Hammer, M. (1999). Ecosystem services generated by fish populations. *Ecological Economics*, 253-268.

IIED. (1994). *Economic Evaluation of Tropical Forest Land Use Options (Draft)*. London, England: International Institute for Environment and Development, Environmental Economics Programme.

Irvine, J., Bailey, R., Bradford, M., Kadowaki, R. and Shaw, W. (1999a*). 1999 Assessment of Thompson River/Upper Fraser River Coho Salmon*. CSAS Res. Doc. 99/128. Ottawa, Canada: Canadian Stock Assessment Secretariat.

Irvine, J., Wilson, K., Rosenberger, B. and Cook, R. (1999b*). Stock Assessment of Thompson River/Upper Fraser River Coho Salmon*. CSAS Res. Doc. 99/28. Ottawa, Canada: Canadian Stock Assessment Secretariat.

IUCN. (1998). *Economic Values of Protected Areas: Guidelines for Protected Area Managers*. Gland, Switzerland: World Commission on Protected Areas /International Union for the Conservation of Nature.

Kahn, J. (1987). Measuring the economic damages associated with terrestrial pollution of marine ecosystems. *Marine Resource Economics*, 4, 193-209.

Kahn, J. and Kemp, W. (1985). Economic losses associated with the degradation of an ecosystem: the case of submerged aquatic vegetation in Chesapeake Bay. *Journal of Environmental Economics and Management*, 12, 246-263.

Knowler, D. (forthcoming). Bioeconomic modeling with environmental influences. In J. Lovett (Ed.), *A Handbook of Environmental Management*. Cheltenham, U.K.: Edward Elgar.

Knowler, D., and Lovett, J. (1996). *Training Manual for Environmental Assessment in Forestry*. FAO Field Document 8/1996. Bangkok, Thailand.

Knowler, D. Barbier, E. and Strand, I. (submitted). An open access model of fisheries and nutrient enrichment in the Black Sea. *Marine Resource Economics*.

Loomis, J. (1988). The bioeconomic effects of timber harvesting on recreational and commercial salmon and steelhead fishing: a case study of Siuslaw National Forest. *Marine Resource Economics*, 5, 43-60.

Lynne, G., Conroy, P., and Prochaska, F. (1981). Economic valuation of marsh areas for marine production processes. *Journal of Environmental Economics*, 8, 175-186.

McClelland, B.,Young, L., Shea, D., McClelland, P., Allen, H., and Spettigue, E. (1982). The bald eagle concentration in Glacier National Park: origin, growth, and variation in numbers. *The Living Bird*,19, 133-135.

Meehan, W.R. (Ed.). (1991). *Influences of Forest and Rangeland Management on Salmonid Fishes and Their Habitats.* American Fisheries Symposium Special Publication 19.

Michael, J., Jr. (1995). Enhancement affects of spawning pink salmon on stream rearing juvenile coho salmon: managing one resource to benefit another. *New World Scientist*, 69, 228-233.

Myers, N. 1992. *The Primary Source.* New York, NY: W.W. Norton.

Panayotou, T., and Ashton, P. (1992). *Not by Timber Alone: Economics and Ecology for Sustaining Tropical Forests*. Washington, D.C.: Island Press.

Pearce, D. and Turner, R. (1990). *Economics of Natural Resources and the Environment*. Baltimore, MA: Johns Hopkins Press.

Pearce, D. and Warford, J. (1993). *World Without End.* Oxford, England: Oxford University Press.

PFRCC. Pacific Fisheries Resource Conservation Council. (1999). *1998-1999 Annual Report: Background papers.* Vancouver, Canada: Pacific Fisheries Resource Conservation Council.

Rowe, R., Morey, E., Ross, A., and Shaw, W. (1985). *Valuing Marine Recreation Fishing on the Pacific Coast*. Boulder, CO: Energy and Resource Consultants.

Sandercock, F. (1991). Life history of coho salmon (*Oncorhynchus kisutch*). In C. Groot and L. Margolis (Eds.), *Pacific Salmon Life Histories*. Vancouver, Canada: UBC Press.

Simpson, K., Semple, R., Dobson, D., Irvine, J., Lehmann, S., and Baillie, S. (1999). *Status in 1999 of Coho Stocks Adjacent to the Strait Of Georgia*. Canadian Stock Assessment Secretariat Research Document 99/133.

Stalmaster, M. and Gessmen, J. (1984). Ecological energetics and foraging behavior of bald eagles. *Ecological Monographs*, 54, 407-428.

Swallow, S. (1994). Renewable and non-renewable resource theory applied to coastal agriculture, forest, wetland and fishery linkages. *Marine Resource Economics*, 9, 291-310.

Ward, B. personal communication. Government of B.C., July 13, 2000 (by email).

Weitkamp, L., Wainwright, T., Bryant, G., Milner, G., Teel, D., Kope, T., and Waples, R. (1995). *Status Review of Coho Salmon from Washington, Oregon, and California*. NOAA Technical Memorandum NMFS-NWFSC-24.

Wilson, F., and Halupka, K. (1995). Anadromous fish as keystone species in vertebrate communities. *Conservation Biology*, 9, 489-497.

Wood, C. (1987). Predation of juvenile Pacific salmon by the common merganser (*Mergus merganser*) on eastern Vancouver island. I: Predation during seaward migration. *Canadian Journal of Fisheries and Aquatic Sciences*, 44, 941-949.

.

Figures

Figure 5-1
Frequency distribution of the "habitat concerns index"
16 South Thompson streams

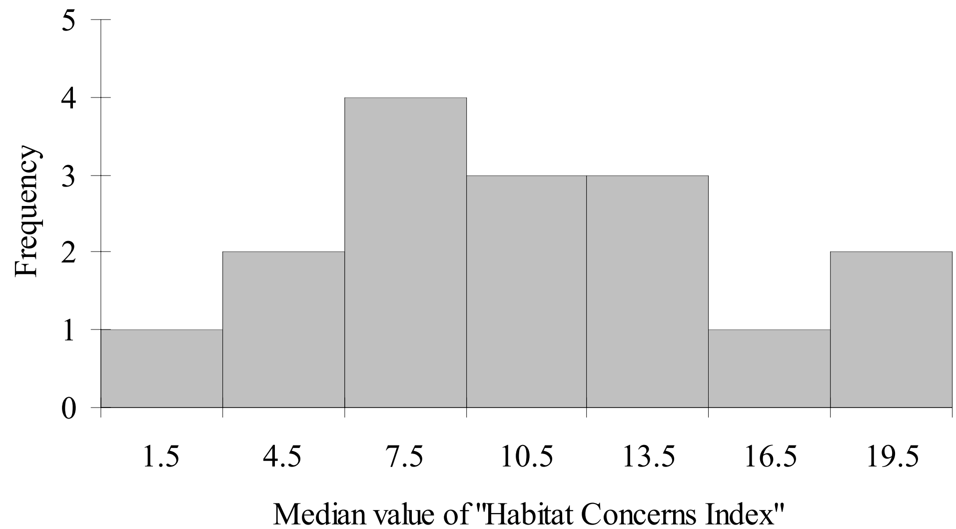


Figure 5-2
Flow chart for simulation and calibration of parameters for the stock-recruitment model incorporating the effects of freshwater habitat quality

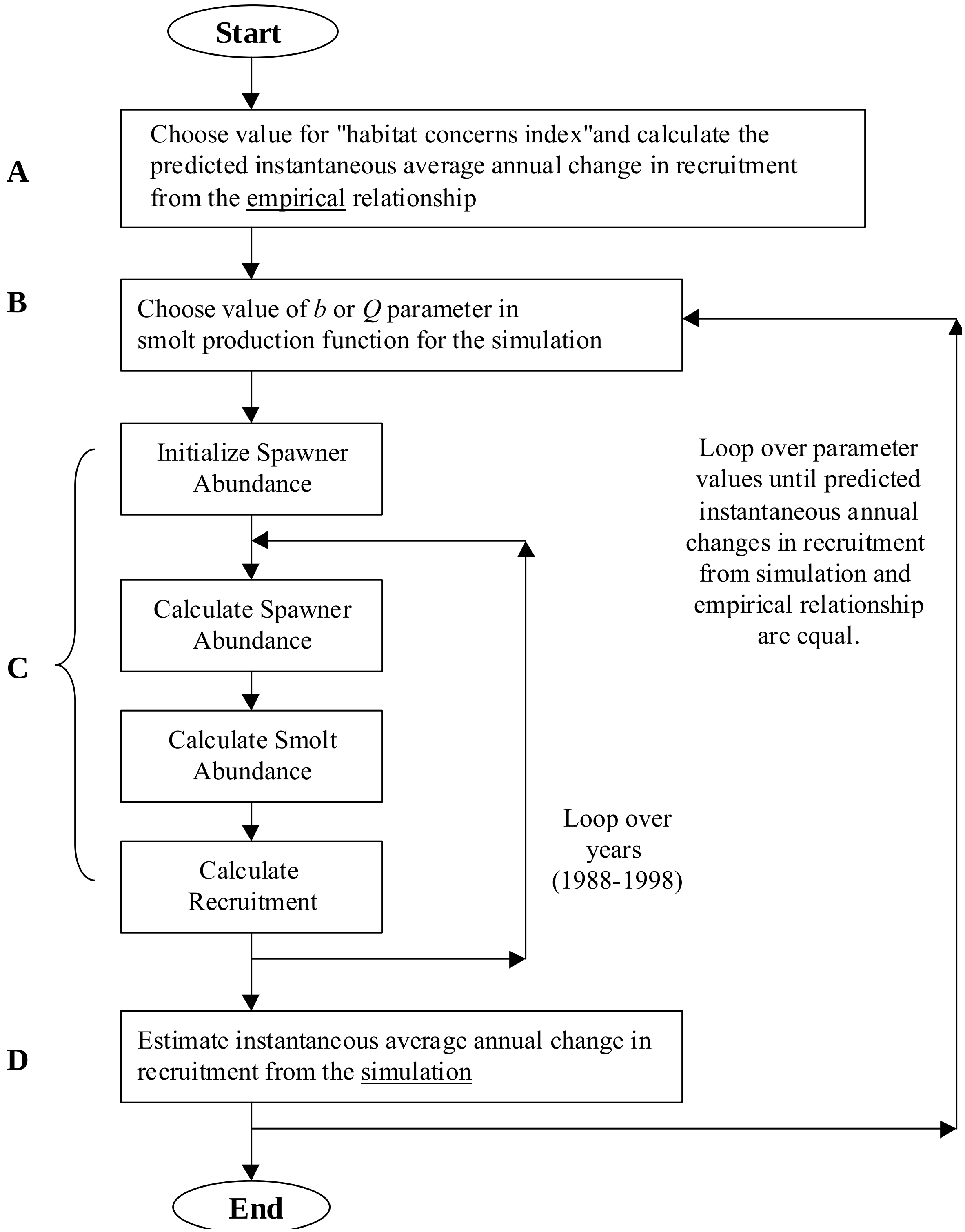
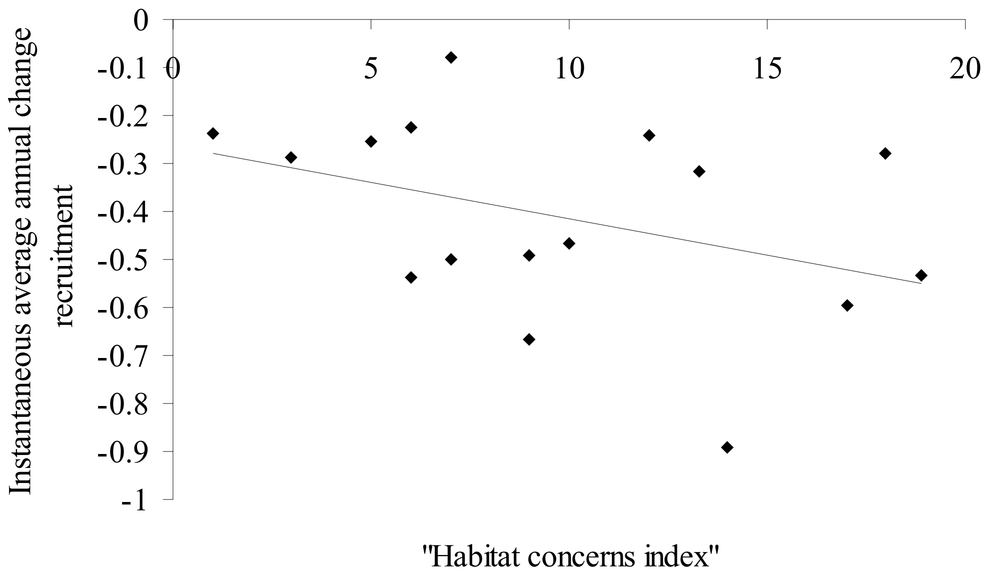


Figure 5-3
Relationship between the instantaneous average annual change in recruitment (*β*) and the "habitat concerns index" (HCI) for 16 South Thompson streams (1988-1998)



Appendices

Appendix 1

Strait of Georgia Coho Data

(all data in '000 fish unless indicated)

Year	Troll Catch	Sport Catch	Gillnet Catch	Total Catch	Exploitable Stock	Escapement	Troll Effort ('000 days)
1953	440	124	73	637	746	109	31
1954	324	130	132	586	725	139	28
1955	459	182	73	714	802	88	29
1956	193	191	125	509	580	71	25
1957	300	241	49	590	760	170	29
1958	361	253	76	690	807	117	35
1959	257	230	59	546	649	103	30
1960	492	238	57	787	870	83	37
1961	302	152	32	486	619	133	33
1962	269	167	68	504	617	113	32
1963	126	199	36	361	464	103	26
1964	284	182	133	599	809	210	28
1965	213	175	42	430	574	144	21
1966	315	249	37	601	751	150	22
1967	143	200	34	377	466	89	22
1968	118	250	82	450	558	108	18
1969	41	200	23	264	310	46	15
1970	162	500	100	762	906	144	21
1971	239	800	71	1110	1261	151	27
1972	62	335	81	478	538	60	19
1973	93	373	54	520	606	86	17
1974	148	772	27	947	1166	219	21
1975	112	454	44	610	739	129	17
1976	73	415	14	502	567	65	20
1977	143	682	42	867	961	94	23
Average	227	308	63	597	714	117	25

Source: Argue *et al.* (1983)

Appendix 2

Maple Program for Base Case Calculations

```
#Analytical Model
> restart;
> R:=(0.2*40*S)/(1+(40/7257527)*S);
> plot (%,S=0..3000000);
> restart;
> f:=X-((Q*m*a*(X-h))/(1+(a/b)*(X-h)))=0;
> g:=(Q*m*a)/((1+(a/b)*(X-h)))^2*(1-(c/p/(q*X)))/(1-(c/p/(q*(X-h))))=(1+r)^3;

> ?Basic Model with 'b' based on current Q = 0.47, m = 0.2, assume Basin Q = 1
> restart;
> f:=X-(1.0*0.2*40*(X-h))/(1+(40/7257527)*(X-h))=0;
> g:=(1.0*0.2*40)/((1+(40/7257527)*(X-h)))^2*(1-(10.4/(0.00003*X)))/(1-(10.4/(0.00003*(X-
h))))=(1+0.05)^3;
> solve({f,g},{X,h});
> restart;
> X:=1071000;h:=560000;
> evalf((ln(X)-ln(X-h))/0.00003);
> 10.5*h-109*%;
> %*20;

> ?S. Thomson disturbed - Basin Q = 0.9878
> restart;
> f:=X-(0.9878*0.2*40*(X-h))/(1+(40/7257527)*(X-h))=0;
> g:=(0.9878*0.2*40)/((1+(40/7257527)*(X-h)))^2*(1-(10.4/(0.00003*X)))/(1-
(10.4/(0.00003*(X-h))))=(1+0.05)^3;
> solve({f,g},{X,h});
> restart;
> X:=1056500;h:=548500;
> evalf((ln(X)-ln(X-h))/0.00003);
> 10.5*h-109*%;
> %*20;

> ?Marginal Value Calculation
> restart;
> 0.01/(1+r)^3*(p+(c/(q*X)))*(m*a*(X-h))/(1+(a/b)*(X-h));
> X:=1071000;h:=560000;
> 0.01/(1+0.05)^3*(10.5+(109/(0.00003*X)))*(0.2*40*(X-h))/(1+(40/7257527)*(X-h));
```