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A random utility model of trip demand for steelhead fishing is constructed by using a logit regression technique and data from the 1990 Oregon Angler Survey, conducted by the Oregon Department of Fish and Wildlife. The model is formulated to estimate demand when only the regional destination is known. Results from the regression are used to calculate expected consumer's surplus for an additional steelhead caught by anglers in Oregon. Regional marginal values for an additional steelhead caught in Oregon range from $10.07 to $15.63. The value obtained for an additional steelhead in the Oregon Department of Fish and Wildlife southwest management zone is used to assess the value of a change in fish production related to management alternatives for tributaries of the Elk River in southwestern Oregon. There was an increase in fish production between the Forest Plan (USDA 1992) and Option 9 in FEMAT (1993), resulting in a present net value of $177,414 for the increase in fish production between the alternatives, under given assumptions of fish population growth and stability.

Applying Random Utility Modeling to Recreational Fishing in Oregon: Effects of Forest Management Alternatives on Steelhead Production in the Elk River Watershed

by

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1. Introduction

Increasing attention is focusing on the declining populations of anadromous salmonid fishes, which include salmon, steelhead, and sea-run cutthroat. Anadromous fish travel through a variety of habitats in their life cycle, and they are influenced by virtually everything humans do. Forest management activities, particularly timber harvest, road construction, and burning, deposit sediments into spawning streams. Agricultural activities, including use of herbicides and pesticides, annual crop harvest, and burning affect streams and rivers in which fish spend part of their life cycle. Dams create reservoirs that contribute to increased water temperatures. Smolts are believed to be destroyed when they pass through turbines used to generate electricity. Fish are harvested in streams and rivers by recreational anglers, and by commercial fishermen and recreational anglers in the sea. In addition to human influences, fish populations are subject to natural conditions such as predation. They must contend with weather patterns that cause variations in stream flow from year to year, and stochastic ocean conditions in which temperatures fluctuate as ocean currents change direction and depth. Fish must deal with inter- and intra-species competition for food and space. All these pieces of the life-cycle puzzle of anadromous fish have been studied to various degrees, but the pieces have not been put together, and most of the linkages and causes are poorly understood.

So who is responsible for the decline in fish populations, and who should pay? How much should the players modify how they do business, and how much should one segment of the community pay as opposed to another to stop or even reverse the decline in anadromous fish populations? Society is currently struggling with these questions; the quality of the answers depends heavily on the amount and quality of

information available about the life cycle of fish populations, the relationships between human activities and anadromous fish, and information on the costs and benefits of alternative policies to increase salmonid populations.

Policymakers need information about wildlife population responses to land management activities, and they need information about prices and benefits to assess tradeoffs. Information about many aspects of fish populations and the response of those populations to natural events or human activities is sparse or nonexistent. Even so, researchers are being asked questions about population response to changes in habitat. One of the ways managers have tried to compare alternatives is by comparing the worth of the outputs. Many people have calculated values for fish, individually or as populations. Fish have been valued in a variety of ways, from economic impact studies to the use of nonmarket valuation techniques, such as travel cost and contingent valuation.

The purpose of this study is to calculate marginal values for steelhead in four regions in Oregon by using a random utility model, and to use the value in the southwest region to compare alternative forest management activities on the Elk River watershed in southwestern Oregon through the effect the activities have on fish populations. Information about anglers in Oregon is collected about every decade. The most recent survey has incomplete trip destination information, in that people were asked in a very general sense where they went to fish, but were not asked about specific sites. Random utility modeling is similar to the travel cost method, but can more easily adapted for use with incomplete trip destination information. Both random utility and travel cost are used to derive measures of recreation value. Recreation value is one part of the value fish have to society, and forest streams and rivers are believed to be important in part of their life cycles.

The theoretical process involved in assigning economic values to the difference in fish produced under different management regimes can be viewed as a vertically integrated production function (figure 1).

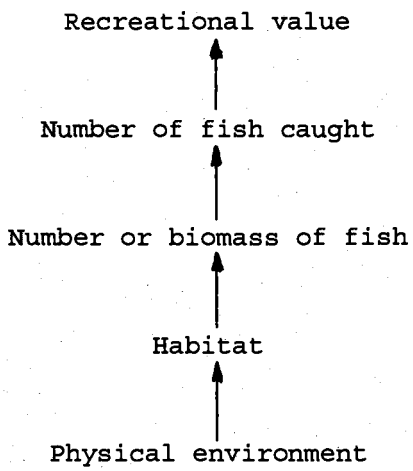

Figure 1. A vertically integrated production function for fish.

The physical environment defines the habitat available for fish. The type of habitat determines, in part, the number of fish that can live in the stream. A portion of the fish population is caught by recreational anglers, who place a value on their fishing experience and on the fish. That value can be ascertained by an analysis of the cost of the fishing trip, the number of fish caught, the distance traveled, and other factors.

Assessing the physical and economic effects of forest management on fish populations may be looked at as a series of processes and submodels (figure 2). Forest management affects the physical environment; the effects can be quantified by using measurable habitat characteristics. There are static habitat models that use habitat characteristics to predict fish biomass or population. Dynamic habitat models predict changes in fish biomass or population due to changes in measurable habitat characteristics. Nonmarket valuation methods use observations of behavior to determine a value for recreation sites and characteristics of recreation sites. Fish catch is often modeled as a

site characteristic. One component of the economic effects of forest management alternatives can be measured by placing a value on the difference in fish produced by one forest management alternative as opposed to another.

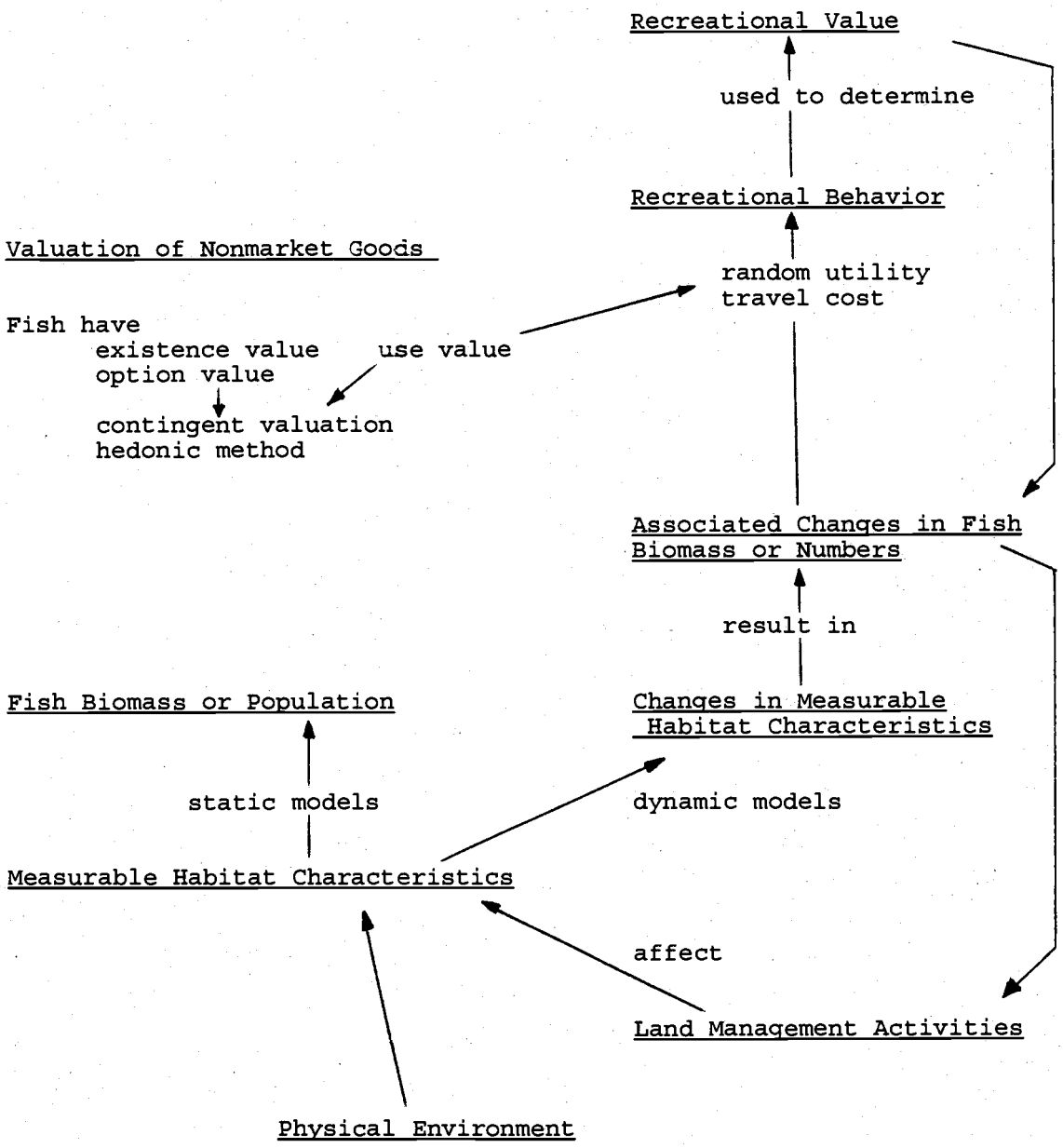

Figure 2. Land management affects fish populations, and can be assessed by changes in measurable habitat characteristics. Economic values assigned to fish are used to value the differences in land management alternatives.

Land management affects the physical environment in which fish live. The physical environment of fish includes the geology and soils of the area, the interactions of plants and animals, water quality and hydrology, climate, and natural events such as disease, wildfire, and sedimentation. There are measurable habitat characteristics in the physical environment that can be used in modeling habitat, fish biomass, and fish populations. The scope and desired precision of the analysis determines the scale of the habitat model to be used; if the analysis focuses on a creek, the scale of the model will be small. The scale of the model determines which habitat characteristics are appropriate. If the scale of the model is small, such as a pool or riffle, appropriate habitat characteristics might include substrate size or pool width and depth. At large scales, such as watersheds or regions, large characteristics must be assessed, such as slope, and soil and vegetation types. Small-scale models are more precise than large scale models, as would be expected (Fausch, Hawkes, and Parsons 1988). Many habitat models predict the number or biomass of fish per unit length or area of stream from habitat variables (Fausch, Hawkes, and Parsons 1988), and numerous studies discuss the influence of land management on fish resources (such as Chapman 1962; Chamberlin 1982; Everest and others 1987; Sullivan and others 1987; Botkin and others 1994).

Land management activities often change riparian habitat. Land management activities (figure 2) affect measurable habitat characteristics, such as number of pools per mile or substrate size. Changes in measurable habitat characteristics can result in associated changes in fish biomass or numbers. This linkage has been studied, and many of the effects of changes in riparian habitat on anadromous fish have been documented. Many of the studies are generalizations about populations decreasing or shifting in species mix. Few models predict the response of fish populations to land management activities. One such model is the southeast Alaska multiresource model (SAMM) (Fight,

Garrett, and Weyermann 1990). The SAMM model was developed to assess timber availability and growth, and the interaction of other resources with timber harvest through interactive models of hydrology, fish population, and deer population. A model was developed to predict salmonid response to sediment yields resulting from forest management in Idaho (Stowell and others 1983). A regional model integrating forage, wildlife, water, and fish projections was developed for the southeastern United States by Joyce and others (1990). Loomis (1988) used a fish habitat index developed by Heller, Maxwell, and Parsons (1983) for the Siuslaw National Forest in western Oregon to quantify the effects of timber production on anadromous fish. Loomis assigned values to the fish produced by different management regimes, and used the differences in fish values to compare forest management alternatives.

Calculating the value of fish produced in different management regimes requires a method of valuing the fish resource. Fish caught by recreational anglers are valued by using nonmarket valuation techniques. Two intertwined definitional issues in nonmarket analysis are 1) the nature of the goods in question, and 2) the types of values you can assign to them. A particular good (like a fish) can be either nonexclusive (one person can't prevent another person from consuming the good) or exclusive, or have characteristics of both. Fish have exclusive characteristics; an example of exclusiveness is an angler catching a fish can keep it, preventing others from catching it. Fish also have nonexclusive characteristics. Goods that are nonexclusive can have existence value, which is the value in simply knowing something exists. The value a person derives from direct contact, or use of a resource, is often referred to as use value. Existence value is often referred to as nonuse value. Different methods of valuing fish incorporate these different characteristics of goods; most methods are defined by whether they measure use, nonuse, or both types of values.

Economists measure value of nonmarket goods in several ways, all of which incorporate the concept of consumer's surplus. Marshall (1930) defined consumer's surplus as the difference between the value of a bundle of goods to a consumer and the amount the consumer actually paid for it. Both existence value and use value of fish are measured in terms of consumer's surplus (figure 2). The contingent valuation method uses surveys of people's willingness-to-pay, which is one method of estimating consumer's surplus. Contingent valuation can measure aspects of either existence value, use value, or both. The hedonic method models demand for generic attributes, and measures aspects of both nonuse and use value. Random utility and travel cost methods measure use values through recreational behavior. The distance people travel to recreational sites, the reason they visit one site as opposed to another, and what they do at the site are used to determine the value they place on the site or the activity, or the site attributes, such as number of fish. The consumer's surplus value obtained by assessing recreational behavior can then be applied to the changes in fish numbers and used to assess the value of the change in fish population under different land management alternatives.

When the value of a nonmarket good is calculated, the distinction between average and marginal values must be kept in mind. Average values are useful in assessing nonmarginal changes in quantity which affect price, and marginal prices are used to assess price in the event of a small change in quantity which does not affect price. Different methods of calculating value will result in prices that are considered marginal or average. The characteristics and applicability of marginal versus average values must be kept in mind when applying these measures of value to policy analysis. Assessing whether the change in the resource being valued is a marginal or nonmarginal change is important. Contingent valuation is useful in calculating either average or marginal values, as are random utility and travel cost methods.

Many attempts have been made to translate logging practices into economic losses for recreational salmon and steelhead fishing (for example, Kunkel and Janik 1976, USDA 1990). Many of these attempts have been criticized because they fail to provide values consistent with consumer theory. As Grobey (1985) pointed out, the value per fish caught is often calculated by value per day for recreational fishing times days required to catch a fish. The value of the entire trip is attributed to the fish, and the fish do not have diminishing marginal value. Fish values are sometimes calculated as average values, not marginal values, but the analysis proceeds as if the values were marginal.

In this study, the most recent Oregon Angler Survey (ODFW 1991a, b) data were assessed by using a random utility model to derive marginal consumer's surplus values for anadromous fish. In general, a survey such as the Oregon Angler Survey is one way to record recreational behavior, and can be used to derive the associated value the angler places on a recreational site, activity, or the fish. The data are the most recent and most comprehensive information available about recreational fishing in Oregon. The specific questions in the Oregon Angler Survey are difficult to use for nonmarket valuation analysis, however. Anglers were asked what management zone they fished in, but were not asked specifically where they went. They were also not asked where they were from, other than by zip code. There are many zip codes in the large management zones used by the Oregon Department of Fish and Wildlife (ODFW). Many people fished in the same management zones in which they live; for them, distance travelled cannot be derived. Some people may have traveled five minutes, others two hours. Anglers were asked how much they spent on each trip they took, so trip cost was treated as self-reported. Anglers were also asked about primary target species; the purpose of the trip is known. This data set could be assessed using either some variant of travel cost, such as zonal or

individual choice methods, or with random utility. Both travel cost and random utility are reasonable choices. Travel cost is relatively more straightforward and well-known than random utility methods. Random utility, however, incorporates substitution between alternatives more easily than do travel cost methods. In addition, discrete choice theory is designed for alternatives so large that most consumers choose only one or two (Small and Rosen 1981). There is work currently being done to estimate prices for fish using travel cost methods with this data set; it is useful to bring another approach to bear on the question of valuation. For these reasons, random utility was used in this study to derive values for steelhead in Oregon. The demand model used in this study demonstrates a method useful for assessing data that are not entirely suited for nonmarket demand analysis.

As an example of how values for fish could be useful in policy analysis, the values derived for steelhead were used to compare two management alternatives in the Elk River watershed, in the Siskiyou National Forest in southwestern Oregon. To use values derived for fish in assessing the comparative worth of fish in two different forest management alternatives, the effects of changes in habitat on fish populations must be determined. A sediment model and fish habitat index developed by Ricks and Chen (1990) were used to illustrate how values for anadromous fish can be applied to a production function for fish, and the results were used to assess two management alternatives. The results were expected to illustrate the need for changes in data collection, fish population assessment, and valuation analysis for nontimber forest resources.

Several hypotheses can be made about the results of this study. Land management alternatives were traced to changes in fish populations and resultant changes in value of fish in the Elk River watershed. As the amount of disturbance decreases, as measured by the landslide sediment delivery index (LSDI) (Ricks and Chen 1990), the population of

anadromous fish was expected to increase. The difference between fish production under the Forest Plan and an alternative that approximates option 9 (FEMAT 1993) was compared. The US Forest Service Forest Plan was developed in response to the requirements set out under the National Forest Management Act (NFMA) of 1976, and in legislation directed from Congress to the Secretary of Agriculture. In 1974 and 1978, Congress called upon the Secretary of Agriculture to establish land management plans for national forests (Findley and Farber 1983). The alternatives developed under the FEMAT process, on the other hand, were developed in response to the Endangered Species Act of 1973 (amended in 1978). The Forest Service, as a land management agency, developed conservation strategies for endangered species, from which Option 9 was selected. On the Elk River, the Forest Plan resulted in more land disturbance than the alternative that approximates FEMAT. The Forest Plan was expected to result in higher LSDI values, fewer fish, and lower economic values than FEMAT.

There are some specific results that would be expected for the coefficient on travel cost, which is an integral part of the results of the random utility model. As part of the random utility model, the coefficient on travel cost, which Morey, Shaw, and Rowe (1991) refer to as the marginal utility of money (β_0), and the probability of fishing at a particular site and water type was estimated. The marginal utility of money should be positive, and less than one. Morey, Shaw, and Rowe (1991) obtained values ranging from 0.0179 to 0.0185 depending on the flexibility of the random utility model used. They were measuring β_0 for recreational ocean fishing for anglers in coastal Oregon counties in the late 1980s. This coefficient is used to develop the estimates of consumer's surplus for anglers in a given site or region.

Several estimates of value have been calculated for salmon and steelhead in Oregon rivers in the past few years. Loomis (1989) used a travel cost model to calculate a marginal value per fish caught by

recreational anglers for steelhead in various rivers in Oregon. Values ranged from $18 per fish on the Coos River to $333 per fish on the Willamette River. His point was that fish values vary considerably from one site to another because of differences in distance and population, and probably other factors, such as site quality. Johnson and Adams (1988) used contingent valuation to come up with a value of $6.65 per steelhead on the John Day River. Olsen, Richards, and Scott (1991) valued steelhead and salmon on the Columbia River from $14.81 to $54.84 per fish. Walsh, Johnson, and Mckean (1990) surveyed the literature and reported estimates for the value of salmon ranging from $2 to $55 per angler day. These are per-trip measures. Most valuation studies count recreationally caught fish. To convert values for fish caught to values for populations of fish, the ratio of catch to escapement is needed for that species in that area. The State's Department of Fish and Wildlife is an excellent source of such information.

The values obtained in this study could be in the low end of this range of values for several reasons. The most important difference is that the way value is measured in this study is different from how value is measured in many studies that use travel cost or contingent valuation methods. In this study, nonparticipation is included in the choice set, and substitution across sites is included in the model. Dealing with substitution directly is an important aspect of this model. Nonparticipation is incorporated by including utility measures of alternatives not chosen out of the choice set in each period. Both nonparticipation and substitution act to lower estimates of consumer's surplus. Many studies use on-site sampling, and do not correct for oversampling avid anglers, which biases estimates of value upwards (Morey, Shaw, and Rowe 1991). This study used a stratified random sample of anglers.

The issues outlined here are discussed in more detail in the chapters that follow. Chapter two is a discussion and literature review

on forest management and its effects on riparian areas. Models of forest management and its effects on other resources are summarized, and methods of valuing recreational resources are outlined. The methods discussed include travel cost and random utility models, hedonic valuation, and contingent valuation.

Chapter three outlines the sediment and fish habitat models developed for the Elk River watershed. The data from the angler survey are discussed. A simple repeated random utility model is presented that will be used to calculate recreational fish values with the data collected in the Oregon Angler Survey between September 1988 and August 1989. The expected consumer's surplus measures are summarized.

In chapter four, results from the interaction of the Elk River sediment model and the fish habitat model are summarized. Results of the random utility demand equation are presented and discussed, and the values obtained from the consumer's surplus analysis are summarized. The value for the southwest region is used to compare two management options for the Elk River watershed: an approximation to FEMAT, and the Forest Plan.

In chapter five, the results presented in chapter four are discussed in terms of the original proposals and objectives, and the conclusion of the study presented. Inferences that might be useful in the current debates about forest-fish interactions and nonmarket methods of valuing nontimber resources will be discussed. Implications for areas in need of more study will be presented, particularly focusing on weak points in this and other similar analyses and where more and better information would increase the quality of the results. It is hoped that this study will shed some light on what has become a hotly debated topic, about which insufficient quantitative evidence exists to firmly support a stable conclusion.

2. Theoretical Background and Literature Review

The specific effects of forest management, hydroelectric power generation, irrigation, agriculture, and commercial and recreational fishing on fish populations are incompletely understood. The interaction of forest management and fisheries has become a topic of debate as forest planners have begun to realize that their activities have more significant effects on nontimber resources than they had thought. Some preliminary work has been done on riparian area resources, as have some assessments of the economics of forest management-fish interactions. This chapter will summarize studies that span one or more of the steps outlined in figure 2.

Many studies assess the physical environment of fish and the measurable habitat characteristics that can be used to model habitat in terms of indices, or models that predict fish numbers or biomass. Other studies have concentrated on the many effects land management activities have on fish habitat, biomass, and species diversity. Recreational valuation studies exist that use different methods to value salmon, steelhead, trout, and other species. A few studies tie land management into effects on fish numbers, and even fewer attempt to put a value on fish stocks.

2.1 Forest Management and Its Effects on Fish Habitat

Land management activities affect the physical environment in which fish spend parts of their life cycle. The effects of forest management on fish habitat are numerous and diverse. Timber harvest can cause sedimentation, deposition or removal of all sizes of debris, from fine to large woody debris; changes in stream temperature; modification or removal of stream-side vegetation; and changes in channel morphology, such as a reduction in pool number and size (Berkman and Rabeni 1987; Chapman 1962; Chamberlin 1982; Everest and others 1987; and others).

The literature on this topic is summarized in Hicks and others (1991), who point out that intact stream-side management zones have many benefits in retaining cover and bank integrity, protecting and providing for future large wood, reducing sedimentation, and preventing substantial temperature changes. The effects of roads and hill-slope timber harvest are also important, particularly from the standpoint of sedimentation and debris torrents. Water quality problems appear to be directly related to how much timber is harvested in a basin. Small but critical zones in a watershed can have tremendous effects on water quality. Several authors conclude that direct influences on stream habitat can be minimized with buffer strips and careful logging and removal, depending on the characteristics of the site. Habitat degradation is considered a causal or contributing factor in more than 75 percent of fish species extinctions (Miller, Williams, and Williams 1989) and more than 90 percent of threatened and endangered populations (Williams and others 1989). Recovery programs for those fishes that have been federally listed have mostly proven unsuccessful (Miller, Williams, and Williams 1989), indicating that habitat degradation and population declines are often irreversible (Frissell 1990). Impacts to habitat are pervasive and persistent because they accumulate from widely separate sites and diverse activities in a watershed across long spans of time (Regier and others 1990).

Frissell (1990) mapped the number of fish species by drainage in western Washington, Oregon, and Northern California that are considered endangered, threatened, or extinct. The highest extinction risk centers tend to lie in regions of steep, forested uplands with highly erosive soils and high precipitation, suggesting that logging of steep, unstable slopes may be a significant causal factor. In regions of comparatively low relief, stable slopes, and relatively low erosion rates, extensive logging does not appear to coincide with extinction or endangerment of fish. That forest management is just one of many threats to aquatic

biodiversity and fish must be stressed. Miller, Williams, and Williams (1989) found that 82 percent of fish extinctions involved more than one causal factor.

Some effects of forest management on fish populations have been studied. Such studies assess measurable habitat characteristics and relate them to fish populations, either qualitatively or in quantitative measures of fish biomass or numbers. Chapman and Knudsen (1980) studied streams in the Puget Sound area in Washington. They found that activities reducing winter habitat and cover damage salmonid populations, regardless of summer habitat quality. Activities increasing light to streams without siltation increase salmonid populations, indicating their study areas are light-limited. Recently, Reeves, Everest, and Sedell (1993) studied 14 basins in coastal Oregon with various degrees of area harvested. They discovered streams with high rates of harvest had less-diverse juvenile communities and were more dominated by a single species than were communities in basins with low harvest. Total density of fish did not differ. Some species of fish are becoming threatened or endangered (Williams and others 1989), and land managers are concerned with how management affects fish and wildlife populations and habitat.

Sullivan and others (1987) state that despite widespread concerns about the effects of logging on channel morphology, surprisingly few well-documented studies have been reported. Even fewer studies link changes in channel morphology with changes in habitat and fish populations. There are a number of reasons for the lack of studies in this area. The broad spatial and temporal scales over which most channels change require both small- and large-scale monitoring or evaluation to determine the type and extent of channel responses. In addition, historical records or ongoing studies are needed to document time trends. Assessing channel changes is time consuming and expensive.

Such studies often do not yield results for many years, and results have limited geographic scope.

Relating changing fish populations to changes in habitat has also proven difficult because factors other than channel changes associated with forest management have contributed to decreasing fish populations. Fishing pressure, construction of hydroelectric dams and diversions, and urban and agricultural land use practices are generally believed to have contributed to the general decline of fish runs observed over the past three decades in the Pacific Northwest. Thus, even though habitat and population changes can be demonstrated within forest stream reaches, the relation between fish populations, habitat conditions, and land use in the basins has not been adequately documented.

The focus in many studies has been on roads as contributors to sediment load. Riparian zones are coming under increased scrutiny for sediment control and supply of large woody debris. Results of an analysis of four basins done by Sullivan and others (1987) reveals that two factors stand out as having an important effect on the rate of stream recovery; the particle size of the increased supply of sediment (determined by geological factors), and the condition of the riparian zone.

2.2 Modeling Forest Management and Its Effects

A diverse collection of models that predict standing crop of stream fish (number or biomass per unit length or area of stream) from measurable characteristics of the environment have been developed since 1970. Fisheries biologists have searched for variables closely linked to fish abundance since at least 1950 (Allen 1951; McKernan and others 1950). Fausch, Hawkes, and Parsons in 1988 summarized all the models they could find and organized them by the types of habitat variables found significant, the mathematical structure, the size of the data sets used to develop the models, and how well the model fit the data. They

found that physical habitat characteristics, particularly the many forms of cover, are most closely related to salmonid abundance.

Fausch, Hawkes, and Parsons also concerned themselves with precision and generality. They concluded that most precise models lack generality. Levins (1966) outlined three desirable attributes of models: realism, precision, and generality. He believed that, at most, two of the three attributes can be attained in biological models. Precision does not necessarily imply realism or accuracy. A model that is precise and realistic will not necessarily be general; a small scale precise and realistic model might not be easily generalized to include other streams or conditions. On the other hand, a realistic and general model may be useful, but imprecise.

Many studies tie sedimentation to habitat quality for aquatic life. Sedimentation has been shown to decrease fish productivity and diversity. Berkman and Rabeni (1987) studied the effect of siltation on stream fish in northern Missouri. They show that siltation reduced benthic insectivores and herbivores, and fish that require a clean gravel substrate for spawning. Fish may be more affected by changes in breeding and spawning conditions than by alterations in their food supply (Balon 1975). Siltation may degrade spawning grounds and cause behavioral changes in spawning fish. It increases egg mortality, decreases larval growth and affects development rate and survival of larval fish (Langlois 1941; Cordone and Kelly 1961; Smith 1971; Muncy and others 1979). Everest and others (1987) make some generalizations about sediment as a limiting factor for salmonid populations. The effect of sediment on salmon species depends on life history, abundance, behavior, and habitat preferences. Salmonids that rear in fresh water for long periods have less risk of being limited by sediment than those that move directly to the sea after emergence. Species whose young spend little time in fresh water, such as pink and chum salmon, are more likely to have their populations limited by poor spawning success

(McNeil 1966), and so they are vulnerable to sedimentation in streambed gravels.

Habitat characteristics, such as sediment and cover, and fish behavior interact to influence fish populations in streams. Species with long freshwater residence times are less likely to be limited by spawning success but are vulnerable to large volumes of sedimentation that cause changes in the channel morphology habitat diversity. Species such as coho, chinook, sockeye salmon, and trout are usually limited by available rearing habitat, except when population densities are extremely low. Because these species are territorial and partition their rearing habitats, each stream has a maximum carrying capacity. The bottleneck in production for these species is often habitat diversity or food availability in summer and shelter in winter (Koski and others 1984). Sediment may be limiting or it may cause a shift in the dominant species if it changes or reduces food, cover, or preferred habitat for rearing juveniles. Sedimentation from road surfaces, for example, probably presents a greater hazard to pink and chum salmon populations than to coho salmon or steelhead trout populations, because roads produce a continual supply of fine-grained sediments. Accelerated mass erosion in systems that are already rich in sediments could have negative effects on all species because both spawning and rearing habitats would be affected. Swanson and others (1987) state that two major links between forestry and fisheries management are sediment delivery to streams by a variety of erosion processes, and the dramatic effects of debris slides and flows on the valley floor geomorphography and stream ecosystems. Ricks and Chen (1990) developed the landslide sediment delivery index (LSDI), an index of debris slides and flows from roads and timber-harvested areas in the Elk River watershed in the Siskiyou National Forest. The LSDI measures sediment volume per decade delivered into stream channels for specific areas, and includes natural, road-related, and harvest-related landslides. Roads and harvested areas

contribute the largest proportion of sediment to streams from land management activities. Nonpoint sources have generally not been monitored; they cannot be detected by using aerial photos, and so are not included as sources of sedimentation in the LSDI.

According to Hueth, Strong, and Fight (1988), efforts to develop models that estimate logging-stream quality relations have been hampered by the complexity of the ecosystems and the difficulty in generalizing from a specific area to a more regional scale. Despite these problems, modeling efforts have been aimed specifically at the forest-fish interaction question. Such efforts attempt to determine the effects of land management activities on measurable habitat characteristics, and some of the efforts tie changes in habitat characteristics to changes in fish biomass or numbers. Joyce and others (1990) examined the effects of shifts in timber management and land use by using models for timber, forage, wildlife, fish, and water resources in the South. Model outputs were aggregated to provide multiresource projections from the same land base. They examined the effect of timber management and land-use shifts (Hoekstra and Joyce 1988). Stowell and others (1983) developed models linking sediment yield data to fish habitat and population responses in Idaho batholith watersheds. A fish habitat index was developed by Heller, Maxwell, and Parsons (1983) for the Siuslaw National Forest in Oregon. Stream flow, pool-riffle ratio, stream surface shading, pool quality, and riffle quality were used to calculate a habitat condition score (HCS) for each reach. Reach scores were weighted by length and used to develop a total HCS for the stream. An adjustment score was provided for recent debris torrents, erosion or deposition, or for high salmonid populations. The goal was to develop a model that would predict the HCS values. Stowell and others (1983) and Leathe and Enk (1985) linked sediment yield models of Cline and others (1981) with models predicting stream fish from habitat to predict changes in

standing crops under different timber harvesting practices in each watershed.

The LSDI and the HCI were developed for the Elk River drainage basin in southwestern Oregon, in the Siskiyou National Forest (Ricks and Chen 1990). The Elk River model is one of the most recent biological models of forest management-fish production interactions. The habitat condition index (HCI) (Ricks and Chen 1990; Chen 1992) is a specific example of a habitat model developed to predict fish populations in given habitat types. The HCI model was developed for the Elk River in southwestern Oregon using data on fish populations gathered by Reeves[1] and others. The HCI is unusual in that it can be used to predict changes in fish populations from management activities, when used in conjunction with the LSDI. The HCI uses measurable habitat characteristics to predict numbers of steelhead; fish numbers were then converted to an index. The data on the Elk River fish populations and habitat conditions are extensive. Chen (1992) used habitat data from pools on the lower 2 miles of the North Fork Elk, an area with minimal management effects, to develop the HCI model. The model was then used to determine an HCI value for other tributary sub-basins. All values were converted to a scale of 0 to 1, with the North Fork given a 1. Pool depth, wood, and wider valley floor areas turned out to be key elements for yearling steelhead in all Elk River tributaries analyzed. The primary problem of the HCI is that habitat potential is assumed in all basins to be as high as the North Fork, an undisturbed tributary to the Elk. The susceptibility of any other watershed to sedimentation and the ability of that watershed to be as good as the North Fork may be correlated, but the model does not account for this possibility. The potential quality of the habitat may be correlated with the sensitivity

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[1]Gordon Reeves. Research scientist, fisheries. Pacific Northwest Research Station, Corvallis, Oregon.

of the watershed to sedimentation from management activities.[2] In addition, the data set is limited, in that the model was developed by using data for one stream for two years, assessing one life stage of one species.[3] One of the findings of the draft report to the State legislature by The Center for the Study of the Environment (Botkin and others 1994) is that the year-to-year return of adult salmonids to rivers for the past 50 years has been variable. The data set for fish counts on the Elk has since been expanded, with fish counts from several years, leaving open the possibility that the model could be improved.

Ricks and Chen (1990) combined the LSDI and the HCI models in an effort to predict the effects of land management activities on fish. They developed a preliminary production function to predict changes in the HCI based on changes in the LSDI. They used the model to predict changes in the HCI given assumed changes in sediment loading.

Use of the effects analysis in management decisions was discontinued as a result of the peer review of the Progress Report.[4] Linking the LSDI and the HCI models, and using them to predict future conditions, was not viewed as yielding results appropriate for small-scale management decisions, such as timber sales, as the model was developed for watershed analysis (Luce and Ricks, unpublished). These concerns are less relevant to the more aggregated analysis in this thesis.

A simple sensitivity analysis of the sediment model shows that basins with high HCI values (i.e., the North Fork or Panther Creek) are more sensitive to errors in sediment production than are basins with low HCI values (i.e., the South Fork). If one more landslide occurs than is

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[2]Cynthia Ricks, personal communication, October 1993. Geologist, Siskiyou National Forest, Gold Beach, Oregon.

[3]Glen Chen, personal communication, November 1993. Biologist, Department of Fish and Wildlife, Utah State University, Logan, Utah.

[4]Cynthia Ricks, personal communication, October 1993. Geologist, Siskiyou National Forest, Gold Beach, Oregon.

predicted, the effect would be 15 percent more than predicted on the North Fork and no change on the South Fork. This result assumes the slope of the function is correct at the point of analysis, which may not be true.[5]

2.3 Methods for Valuing Nonmarket Resources

Several steps are needed to calculate the difference in value between forest management alternatives in terms of fish production (figure 2). The land management activities must be defined in terms of their effect on measurable habitat characteristics. The changes in the habitat characteristics can be modelled, and an approximation of the associated changes in fish biomass or numbers can be calculated. This step in the process is the one about which the least is known.

A value for the fish must be derived, so the changes in fish numbers and the differences between management alternatives can be assessed by using monetary measures derived from consumer's surplus analysis. A wide body of literature has developed that assesses nonmarket values in recreation. Sorg and Loomis (1986) point out that distinguishing between economic impact values and economic efficiency values is important. Each set of values addresses different economic issues and they cannot be used interchangeably. Economic impact values are measured by recreationists' expenditures in a local area. Economic efficiency values identify net gains to society resulting from management alternatives. In recreation, economic efficiency values are derived from various measures of consumer's surplus.

Consumer's surplus was first defined as the difference between the value of a bundle of goods to a consumer and the amount the consumer actually paid for it (Marshall 1930). Hicks (1943) proposed four measures of consumer's surplus: compensating variation, compensating

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[5]Cynthia Ricks, personal communication, February 1994. Geologist, Siskiyou National Forest, Gold Beach, Oregon.

surplus, equivalent variation, and equivalent surplus. Following the conventions in the literature, the equivalent measures are defined as the amount of compensation paid or received that would bring the consumer to the new welfare level if the change did not take place. The compensating measures are defined as the amount of compensation paid or received that would keep the consumer at the initial welfare level after the change had taken place.

Three basic methods can be used to obtain measures of consumer's surplus for recreational goods. The first are models of demand, derived from behavior, for the services of recreational sites. Travel cost models and random utility models are both based on the use of and demand for a site's services. The second method is to model demand for generic attributes, which is done with hedonic models. The hedonic method is also a behavioral approach which must rely on recreational use for measurement of site or site characteristic values. The third method, contingent or hypothetical valuation, values access or quality changes directly from consumer's responses to questions. The values derived from travel cost and random utility methods are considered use values; the consumer's surplus measures derived from hedonic and contingent value methods can have elements of both use and non-use values embedded in them (Bockstael, McConnell, and Strand 1991).

2.3.1 Travel cost and random utility methods

In recreation analysis, one of the most prominent of the three basic methods are those based on the demand for a site's services. The underlying assumption with the travel cost and random utility methods is that a person must visit a site to consume its services. Travel costs are used as proxies for price. Variation in price causes variation in consumption. Observations on variations in price, consumption, and sometimes quality characteristics are primary components in the

derivation of demand functions and the estimation of welfare measures (Bockstael, McConnell, and Strand 1991).

The first step of the simple zonal travel cost method is to divide the area around the recreation site into zones of origin. All individuals in a given zone are assumed to incur the same travel costs to the site. A per capita equation is developed to predict visitation rates based on travel costs and socioeconomic information. The total number of observed visits for a given price is one point on the trip demand curve. Additional points on the demand curve are obtained by adding hypothetical fees to the travel costs from each origin, then estimating visits from each origin at each fee using the per capita equation (Sorg and others 1984). The travel cost method works best when visitors travel a wide range of distances to the site (Dwyer, Kelly, and Bowes 1977). Travel cost models work best when the trip is to one site for one purpose (such as Seller, Stoll, and Chavas 1985; Caulkins, Bishop, and Bouwes Sr 1986). Some travel cost models are unable to account for substitution that may subsequently occur; they may overestimate benefits when changes in site quality are assessed. Several studies have included the effects of substitution with the use of a system of demand equations among sites (Burt and Brewer 1971; Cicchetti, Fisher, and Smith 1976). Other studies have used indices to represent site substitution (Cesario and Knetsch 1976; Loomis 1989).

Both travel cost and random utility methods use observations on site visitation, and both use travel cost as price. The primary difference between the two methods is the assumption about the recreationist's planning horizon. The travel cost method models demand over a period of time. The random utility method describes how people choose among sites in their choice set each time they decide to go to a recreational site (Bockstael, McConnell, and Strand 1991). Random utility modeling is regarded as part of probabilistic choice theory. A choice can be viewed as an outcome of a sequential decision-making

process. First, the choice problem is defined, and then alternatives are created. The characteristics of the alternatives are assessed, the choice is made, and finally the choice is implemented. A specific theory of choice defines a decision maker, the alternatives, the attributes of the alternatives, and a decision rule.

A random utility approach to probabilistic choice theory treats utilities as random because of observational errors that result from unobserved attributes, unobserved taste variations, and measurement errors. The decision maker is always assumed to choose the utility-maximizing alternative. The choice probabilities can be interpreted as the analyst's statement of the probability that for any decision maker, the utility of an alternative will be greater than the utilities of all other alternatives considered (Ben-Akiva and Lerman 1985).

The random utility method uses a utility decision rule. A utility decision rule assumes the attributes of alternatives are commensurable. The objective function expresses the desirability of an alternative in terms of its characteristics. The index of attractiveness is utility, and the decision maker tries to maximize utility through his or her choice. The assumption of a single index is based on the notion of tradeoffs that a decision-maker is using, explicitly or implicitly, in comparing different attributes (Ben-Akiva and Lerman 1985).

Kling (1986) compared several models that have been used to value quality changes in an experiment where true surpluses were known. When substitution was frequent between sites, the discrete choice or random utility model estimates were most accurate. Discrete choice models are good at explaining substitution among alternatives, but not changes in total demand (Kling 1988), because discrete choice methods are point estimates along discontinuous demand functions.

The random utility method explicitly models the choice among substitute alternatives on a given choice occasion. In Morey, Shaw, and Rowe's (1991) random utility model of recreational participation, site

choice, and activity valuation. the individual chooses where to fish (site), and whether to fish from shore, charter boat, bank, or other mode. Also, the person may target a particular species. None of these choices is independent of the others. The fishing mode chosen will determine to some extent the species targeted, and the area of fishing may be accessible only with certain modes. The alternative set may be different for different alternatives, and mode and target species may differ by region.

One approach to handling this problem in random utility modeling is to treat it as one estimation problem by including all possible site, mode, and target species combinations in the alternative set (Bockstael, McConnell, and Strand 1989). Then, the individual would chose among the combinations based on the indirect utility function

$$V_{ims} = v_{ims}(z_{ims}) + \epsilon_{ims}, \qquad (1)$$

where v_{ims} is the indirect utility associated with choosing site i, mode m, and target species s. The vector z_{ims} includes variables that affect utility and which could vary with any or all of the three dimensions of the alternatives. One of the components of random utility theory is that the random term ϵ_{ims} is assumed to be known to the recreationist but unknown to the observer.

Bockstael, McConnell, and Strand (1989) are concerned that estimating the problem in one stage may not realistically represent the angler's decision-making process, and it may violate the independence of irrelevant alternatives assumption. The independence of irrelevant alternatives is the third property for Arrow's social choice rule (Kreps 1990). It states that the only thing that ought to matter in social choice rules are the relative rankings of alternatives. No intrapersonal comparisons of intensity of ranking occur. Morey, Shaw, and Rowe (1991) state that an important advantage of their model is that it does not impose the IIA restriction when the comparison of alternatives includes nonparticipation; the relative odds of choosing a

particular fishing site versus not participating depends on the number of other sites available, their characteristics, and their costs. The standard logit model restricts this ratio to be constant no matter what happens to the costs and characteristics at other sites. Bockstael, McConnell, and Strand (1989) believe that the structure of the decision process is critical to random utility modeling but has received little attention. They estimated a model in which the angler first chooses a mode and target species combination, and then, depending on the first choice, the person chooses a site.

Sometimes data sets are too large to work with without using aggregation. Parsons and Kealy (1992) studied lake recreation in Wisconsin, and estimated a random utility model by using randomly drawn opportunity sets from large choice sets. For estimation, they used each person's chosen site plus a random draw of as few as 11 others, when hundreds were available. They were trying an alternative to aggregation. Aggregation reduces the number of alternatives in the choice set and simplifies estimation, but at the expense of aggregation bias.

Numerous studies compare one method of estimating consumer surplus to another. Caulkins, Bishop, and Bouwes Sr. (1986) compared random utility to travel cost model estimates of site quality and substitution effects in lake recreation. They used a two-equation multinomial logit specification for the discrete choice model and a single equation travel cost model. Both models were used to estimate the change in visits for a change in water quality. The multinomial logit model predicted a smaller increase in visits than did the travel cost model. The authors state that the multinomial logit model results are a "more likely outcome" based on Freeman (1979). The travel cost model overestimates the increase in the number of visits to the improved water quality site primarily because of the inability of the travel cost method to model how people would reallocate visits among the sites in their choice sets.

2.3.2 Hedonic models

Hedonic modeling has become common in housing studies, but it is not used as much in recreation economics as travel cost or contingent valuation. In hedonic modeling, bundles of attributes are assumed to be purchased, and an attribute in one bundle is the same as that attribute in another bundle. A common example is bedrooms; a bedroom is assumed to be the same from one house to another. The composition of the bundles in terms of attributes determines the preference for one bundle over another. In the housing market, the market assures that attributes to be valued are priced at the margin (Bockstael, McConnell, and Strand 1991).

Brown and Mendelsohn (1984) developed a hedonic travel cost model. In the first stage, travel costs are regressed on measures of quality characteristics at the site. The marginal cost of each quality characteristic is calculated from the estimated equation, and used in estimating the demand for each characteristic.

Bockstael, McConnell, and Strand (1991) summarize criticisms of the use of hedonics for modeling recreation demand. No market exists between buyers and sellers in recreation as it does in the housing market. Relations estimated using hedonic modeling techniques must be considered carefully. The relation between travel costs and the quality characteristics that can be measured in the recreation problem will be determined by nature, not in the market. The arrangement and number of sites, and their quality characteristics, along with traditional population centers, will influence whether a person can obtain positive marginal costs for the characteristics of interest. In addition, people's relative preferences for different characteristics will affect the set of chosen sites.

The hedonic method and its use in recreation valuation have been further criticized. Smith (1991) believes the opportunity costs of

travel time should be included as a cost, and that time costs vary among people, which suggests that everyone faces a different hedonic price function. Smith and Kaoru (1987) found another problem with valuing sites that have different amounts of a quality characteristic. The demand equation estimated in the second stage of the procedure relates the "price" of acquiring the quality characteristic with its desirability. For example, people will pay more for individual camping sites that are larger and thus less congested. If the good is purchased once, as in housing, this treatment of quality results in positive marginal prices. In recreation, however, several trips are chosen. Serious empirical problems can result, such as the negative marginal prices obtained by Bockstael, Hanemann, and Kling (1987) and Smith and Kaoru (1987).

Bockstael, McConnell, and Strand (1991) believe that the hedonic model can only be used to answer policy questions about generic characteristics. Most policy questions are about specific resources. The hedonic model produces a function specific to only one origin, and has as its elements the sociodemographic characteristics and the particular combination of sites and quality characteristics relevant to that population. The values obtained are marginal, and will not hold for discrete changes in the characteristics that would change the cost function (Bockstael, McConnell, and Strand 1991).

2.3.3 Contingent or hypothetical valuation

Contingent valuation techniques are often referred to as bidding techniques. Bidding game scenarios require individuals to respond to discrete changes in the quality or quantity of a nonmarket good. The contingent valuation method requires asking people what they would be willing to pay for some hypothetical change. As Sorg and Loomis (1986) note, the contingent valuation method depends on a concise questionnaire format. Hypothetical valuation was introduced in a 1963 study of

recreation in the Maine woods by Robert Davis. It was used next when Randall, Ives, and Eastman published a study about the esthetics of recreation in the Southwest in 1974. Since 1974, a growing body of work uses contingent valuation to value nonmarket goods. Contingent valuation is particularly appropriate when the effect of a characteristic is difficult to model. One such characteristic is congestion. Contingent valuation methods are flexible, and have been used to value a wide variety of goods and services, and characteristics of those goods and services. Many studies value fish caught and game harvested. Many also calculate the consumer surplus that results from new management or environmental policies.

The most compelling reason to use the contingent valuation method is that in some settings, no behavioral methods can value the services of interest. The best example is existence value, which by definition does not include behavior. Contingent valuation can also be used as a substitute for random utility or travel cost methods. Bockstael, McConnell, and Strand (1991) find no evidence that contingent valuation gives answers that are dramatically different from or more random than behavioral methods. This result reflects as much on the assumptions required in using behavioral models as much as it does the accuracy of contingent valuation. In addition, they say, no evidence exists that strategic considerations are as important as early critics of contingent valuation expected. The strongest arguments in favor of contingent valuation are due as much to the strengths of hypothetical valuation as they are to the weaknesses inherent in recreation demand modeling. Some researchers are still skeptical about contingent valuation. Cummings, Brookshire, and Schulze (1986) suggest that better results will be obtained with contingent valuation when the person is familiar with the resource being valued. They believe, however, that contingent valuation is most needed where the respondent has little experience.

The main advantage of contingent valuation over the travel cost method is the ability of contingent valuation to measure the value of multidimensional and multipurpose trips. The basic assumption of the travel cost method is that the primary reason for taking the trip was to visit a specific site or participate in a specific activity. The travel cost method would overestimate the value of fishing at a site, for example, if some other reason was the primary one for visiting the site. With contingent valuation, people can be presented with hypothetical situations and asked how their visitation would change or their willingness-to-pay would be altered (Sorg and Loomis 1986). Seller, Stoll, and Chavas (1985) point out that the travel cost method provides estimates of Marshallian consumer's surplus, but contingent valuation is a Hicksian measure of welfare change. When the income effect is small, the difference should be small (Willig 1976; Randall and Stoll 1980). The income effect will be small if the recreational good or service being measured takes a small percentage of the person's total income.

In several studies, the travel cost method and contingent valuation results have been calculated and compared for the same data set. Bishop and Heberlein (1980) compare the travel cost method to contingent valuation and simulated markets. They find the values derived from travel cost and contingent valuation are lower than those derived from simulated markets. Thayer (1981) compares contingent valuation and a site substitution approach based on travel cost. The results from the two methods are similar. He concludes that hypothetical bias, starting point bias, and information bias are not a problem. Seller, Stoll, and Chavas (1985) compare an open-ended form of contingent valuation, a close-ended form, and the travel cost method for recreation on three lakes. The open-ended form of contingent valuation, in which respondents were asked to put a value on their recreational experience with no suggested guidelines, provides very low estimates of consumer's surplus, with some negative values. The close-ended form of

contingent valuation, in which respondents are given a range of values to choose from, and the travel cost method provide comparable estimates of value for the three lakes in the study. Huppert (1989) compares contingent valuation and travel cost values for central California anadromous fish. The values are not the same. Sometimes the contingent values are lower, suggesting the trips were multipurpose. Travel cost values will be high if the value of the entire trip is attributed to the good. Huppert points out that errors may have occurred in specifying the travel cost model.

Some authors believe that the similar values obtained in comparative studies validate contingent valuation, but the limitations of all nonmarket valuation techniques must be kept in mind. For contingent valuation, questionnaires must be carefully devised; the wording and range of values must be well thought out. In random utility modeling, the structure of the decision process must be considered while formulating a model of recreation demand. When either random utility or travel cost methods are used, multipurpose trips and trips to multiple sites can be a problem. When travel cost is used, the importance of substitutes must be carefully dealt with, and the value of time must be estimated. When the method is chosen, the structure of the data set, the good or service to be valued, and the nature of the recreation visit itself must be taken into consideration.

2.4 Fish Valuation in Oregon

The question of salmon and sport fisheries resource valuation has been addressed by several authors (such as Sorg and Loomis 1986; Loomis 1988; Huppert and Fight 1991). Two major categories of value are associated with recreational angling and fish stocks; use value and existence value. Use value is the value associated with catching, consuming, or otherwise directly using the fish. Existence value is the

value people place on simply knowing that the fish exist or are preserved for future generations.

The objective of the nonmarket valuation part of this dissertation is to estimate recreational fishing values for Oregon Department of Fish and Wildlife management zones in Oregon. Several studies have recently estimated recreational fishing values in Oregon. Morey, Shaw, and Rowe (1991) report yearly compensating variations (CV's) for the elimination of ocean salmon fishing in Clatsop county for people from various counties, in 1981 dollars. The values range from $0.31 in Douglas county to $2.76 in Clatsop county. Morey, Shaw, and Rowe interpret the values as meaning that people from Douglas county would pay $0.31 for the opportunity to fish for salmon in Clatsop county, and people from Clatsop county would pay $2.76. They do not report per-trip values for salmon. Morey, Shaw, and Rowe state that the estimated CV's for salmon fishing are small because most anglers substitute easily across sites and species. Loomis (1989) used a pooled travel cost model to estimate marginal values for steelhead for 21 rivers in Oregon. The values range from $18 per fish on the Coos River to $333 per fish on the Willamette. Loomis used 1977 angler survey data and 1984 vehicle cost data; he does not report what year's dollars the values are in. His point is that marginal values per fish vary systematically from one site to another, and the use of one value for a state or multi-state region results in misallocation of resources. Johnson and Adams (1988) used contingent valuation to calculate the benefits of increased stream flow in the John Day River. They estimate the value of an additional recreationally caught steelhead to be $6.65 in 1986 dollars. Olsen, Richards, and Scott (1991) estimate the total value of each fish in the Columbia River basin at $68.49 (including existence value, option value, and use value) if the Columbia River basin fish runs were doubled. The marginal value per fishing trip ranges from $23.55 to $54.31, depending on the location and species of fish caught (steelhead or salmon). The marginal value

per fish ranges from $14.81 to $54.84. All their estimates are in 1989 dollars.

2.5 Linking Models to Assess Management Effects

The values obtained for anadromous fish in this study were applied to the production function for steelhead developed by Ricks and Chen (1990). The difference between the Forest Plan and Option 9 management alternatives on the Elk River watershed was assessed. Other studies have done similar analyses. Loomis (1988) presented a bioeconomic model of the response of recreational and commercial salmon and steelhead fisheries to alternative timber harvest rates associated with two different landtype associations in Oregon. He calculates net economic values of recreational fishing by using a demand equation developed from a multi-site travel cost model. Loomis used the Fish Habitat Index developed by Heller, Maxwell, and Parsons (1983) for the Siuslaw National Forest. Substantial losses in fish benefits resulted from logging on 32 percent of the Siuslaw, but the value is far outweighed by the timber value. The study demonstrates that the travel cost method can be used to measure marginal losses in fish catch resulting from timber harvests. The Southeast Alaska Multiresource Model (SAMM) (Fight, Garrett, and Weyermann 1990) is a multiresource forest management model with a fisheries submodel. The fisheries submodel calculates production and harvest of pink, chum, and coho salmon. Inputs for fish value are user days of recreational fishing; eventual output is an economic evaluation of timber management activities. Other inputs for value include user days of recreational hunting and jobs produced by harvesting timber and building roads. Fausch, Hawkes, and Parsons (1988) caution that linking several models, each with associated prediction errors, can result in significant statistical error. The results should be interpreted with care.

In the next chapter, the data, the production function for the anadromous fish of the Elk River watershed in southwestern Oregon, and the random utility model will be described and discussed. The angler survey has little information on cost and includes angler's destinations only in broad terms, and so is difficult to assess by using traditional travel cost methods. Random utility modeling allows more information to be included in anglers' decision-making processes by including substitution across sites, and by including the utility of not fishing in sites not chosen in each decision period. For these reasons, a non-nested random utility model was used to calculate fish catch values by using the most recent Oregon Angler Survey data. This study develops and uses a vertically integrated production function, parts of which are not very reliable. Until better recreational data become available, and better production functions for nonmarket resources are available, keeping the price analysis as simple as possible is reasonable.

3. Production Model, Survey Data, and the Random Utility Model

In this chapter, the data, the production function for the anadromous fish of the Elk River watershed in southwestern Oregon, and the random utility model are described and discussed. A random utility model is used to calculate fish catch values by using the most recent Oregon Angler Survey data. Consumer surplus measures are outlined and discussed in the last section of the chapter.

3.1 Elk River Landslide Sediment Delivery Index and Habitat-Condition Index

Information about fish populations has been collected on the Elk River watershed in southwestern Oregon by Reeves[6] and others. Historic information about mass failures and aerial photos of the area that date back several decades also exist. Ricks and Chen (1990) used this accumulated information to develop two models. One is a landslide sediment delivery index which relates area of ground harvested, or number of miles of road, and historic information about mass failures to predict the volume of sediment expected to be delivered to tributaries in the Elk River watershed. The other is a habitat index which relates woody debris, stream size, and stream topography to predict number of steelhead. The two models were then assessed together and a production function for fish developed based on changes in the landslide sediment delivery index. The authors both caution that the habitat model is a preliminary one, and needs further development. Luce[7] noted that the experimental design of the HCI did not control for history of sediment delivery and stream channel morphology before harvest began in the

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[6]Gordon Reeves, Research Scientist, Fisheries. Pacific Northwest Research Station, Corvallis, Oregon.

[7]Charles Luce, personal communication, February 1993, July 1994. Research Hydrologist, Intermountain Research Station, Moscow, Idaho.

basin, and it did not control for geologic setting. For example, the model compares the "critical reach" on Purple Mountain Creek, a high gradient creek with small floodplains, to the "critical reach" on the North Fork of the Elk, a low gradient creek with comparatively large floodplains, on an equal basis.

The LSDI and HCI models will be used in conjunction with consumer's surplus values for the random utility model of recreational fishing demand to compare two land management plans for the Elk River in southwest Oregon: the Siskiyou Forest Plan (alternative A) and an alternative that approximates Option 9 in FEMAT (1993) (alternative B). The results are more appropriately assessed in terms of degree of difference rather than as precise quantitative measures of dollars and cents differences.

3.1.1 The landslide sediment delivery index

The LSDI is composed of two equations, as follows:

For individual harvest units:

$I_h = f(A, R_h, V_h, D_h, T_h)$, where (2)

- I = volume of sediment expected to be delivered;
- A = area of ground harvested in high sensitivity areas;
- R = number of landslides per acre;
- V = volume per slide;
- D = percentage of material delivered to streams;
- T = susceptibility (a probability measure); and
- h = parameter calibrated for harvested units.

A and T are variables; R, V and D are parameters.

For individual road segments:

$I_r = f(M, R_r, V_r, D_r, T_r)$, where (3)

- M = number of road miles in high and moderately sensitive areas;
- R = number of slides per mile of road constructed in moderate and high sensitivity areas; and

r = parameters calibrated for road segments.

I_h and I_r are summed over a watershed to give an estimate of landslide sediment delivery to streams over time, as a result of forest management. Uncertainty in values for R and V contribute most to uncertainty in I (Luce and Ricks, unpublished). The prediction for two areas in the basin was substantially different from the actual sediment volume. Luce and Ricks (unpublished) offer two explanations. One is that the model may contain bias because of the volume distribution, and the other is that some process may not be well represented by the model, such as synergistic effects of either multiple roads close to each other on a slope or harvest areas next to roads. Explanations for differences between predicted and actual values for different periods are based on weather trends, changes in management practices, and inaccuracies in the landslide susceptibility curve. The actual volume of slides for the whole basin between 1952 and 1986 was greater than the predicted volume, but the index value falls within the 90 percent confidence interval for the estimate.

3.1.2 The habitat condition index

The HCI equation (Chen 1992) is as follows:

$$\begin{array}{lllll} \text{HCI} = & -4.501 & + 3.396(\text{constraint}) & + 12.752(\text{wood} * \text{depth}). & (4) \\ \text{SE} & & 1.273 & 1.999 & \end{array}$$

$F_{2,18} = 62.74$ corresponding to a $p < 0.05$; $n = 20$.

HCI = number of 1+ steelhead;

index for wood = $\Sigma \dfrac{\text{(size class*number of pieces in that size class)}}{\text{total number of pieces in all size classes}}$;

constraint = an index; $\dfrac{\text{valley floor length}}{\text{active channel width}}$; and

depth = mean pool depth.

Chen used STATGRAPHICS to develop the model and stepwise regression included in the package to select variables.

3.1.3 Linking the landslide sediment delivery index and the habitat condition index

Ricks and Chen (1990) combined the LSDI and the HCI models in an effort to predict the effects of land management activities on fish. They developed the following preliminary production function:

$$HCI = 3.76 - 2.91 * (LSDI^{.0625}). \quad (5)$$

They used the model to predict possible changes in fish habitat given assumed changes in sediment loading.

Linking the LSDI and the HCI models, and using them to predict future conditions, will not yield results appropriate for small-scale management decisions. The LSDI model is not highly precise. The LSDI model is applicable for cumulative effect analyses where the objective is to predict long-term effects of landslides over large basins, roughly 5000 acres or more (Luce and Ricks, unpublished).

3.1.4 Converting the habitat condition index to fish numbers

The HCI model was developed using data from 1986 and 1987 for yearling steelhead. Actual and calculated numbers of fish in Elk River tributaries,and the HCI for each tributary, are reported in Table 1. The HCI was converted to fish numbers by using the relation: conversion factor = average fish numbers divided by the HCI for each tributary listed in Table 1.

Location	Average fish numbers 1986-87	Existing HCI (USDA 1992)	Conversion factor
North Fork	1336	1	1336.00
Panther	680	0.87	781.61
Blackberry	228	0.71	321.13
Bald Mt.	4426	0.64	6915.63
Butler	590	0.55	1072.73
South Fork	628	0.48	1308.33

Table 1. Yearling steelhead in Elk River tributaries.

Six streams were used in the analysis, all of which are tributaries of the Elk River, but the main stem of the Elk is not included. Data are available for fish counts in the main stem of the Elk River, but LSDI calculations are not available for that part of the river. Landslide sediment delivery index numbers are available for areas or tributaries other than those reported in Table 1, but those areas include tributaries with no anadromous fish in them (for example, Purple Mountain Creek), or no information about anadromous fish.

Numbers of fish for 1986 and 1987 were obtained from fish counts made by Reeves and others.[8] The fish count data was expanded by number of units of habitat types by multiplying the mean number of fish per habitat type (the sample) by the sum of the individual habitat types,[9] a method developed by Hankin and Reeves (1988). The existing HCI is reported in the Progress Report (Ricks and Chen 1990). When the HCI model was developed, the North Fork of the Elk was divided into upper and lower reaches. In subsequent analyses, the two reaches were combined. As a result, the existing HCI for the combined reaches is

[8]Gordon Reeves, Research Scientist, Fisheries. Pacific Northwest Research Station, Corvallis, Oregon.

[9]Final fish counts provided by Cynthia Rugger, Statistician/Biologist. Pacific Northwest Research Station, Corvallis, Oregon.

reported as 1 as the upper North Fork has a barrier and anadromous fish are not generally found there.[10]

3.1.5 A policy application

Two alternatives for managing the Elk River basin were compared by using results from the random utility model developed in this study. One is the Siskiyou National Forest Land Resource Management Plan (alternative A) outlined in detail in the Draft Environmental Impact Statement Elk Wild and Scenic River Management Plan (USDA 1992). Alternative A includes 27,500 acres suitable for harvest within the Elk River basin. The other alternative (alternative B) (USDA 1992) has no acres suitable for harvest within the basin. Arterial roads are maintained, many secondary roads closed, and sites with erosion problems rehabilitated. The basin is recommended for withdrawal from new mineral entry, and development of recreational facilities is not planned. This alternative is analogous to Option 9, the President's Plan (FEMAT 1993), which is currently being implemented.[11] This study compared the effect of these two alternatives on anadromous fish in the six tributaries listed in Table 1.

3.2 Oregon Angler Survey Data

The data to be used in estimating recreational fishing demand and consumer surplus values for anadromous fish come from the most recent Oregon Department of Fish and Wildlife (ODFW) angler survey which was conducted from September 1988 to August 1989 (ODFW 1991a, b). A stratified random sample of license holders was selected, based on

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[10]Gordon Reeves, personal communication, September 1994. Research Scientist, Fisheries. Pacific Northwest Research Station, Corvallis, Oregon.

[11]Cynthia Ricks, personal communication. August 1994. Geologist, Siskiyou National Forest, Gold Beach, Oregon.

expected angler effort, a method used for the 1977 and previous Oregon angler surveys (ODFW 1991a). Three data sets were collected. The first contains license information, including address, gender, and age. A random sample of people in the first data set (the address file) were sent a short mail survey. Questionnaires were sent in two-month waves to a stratified random sample of people who purchased Oregon fishing licenses during the survey period. The mail survey, which included 5259 individuals, asked anglers about up to eight fishing trips they had taken in the past two months. The mail survey data has information about catch rates, some cost information specific to each trip, trip dates and lengths in days, and ODFW management zone destinations.

The third data set had 1867 respondents, and was obtained with a telephone interview of a subsample of the mail survey. The telephone survey was designed to ask specific information about one of the trips in the mail survey. Detail the telephone survey has which is not in the mail survey includes distance to the fishing site in miles, time to site, income, income loss from time off work, and detailed expenditure information. The destination of the trip in both the telephone survey and the mail survey was reported by zone; this data set does not include information about specific rivers, lakes, streams, and reservoirs.

The data were stratified according to a number of criteria, including license date and type, residency, and urban versus rural population, so sampling was intended to be proportional to expected angler effort. According to Ben-Akiva and Lerman (1985), the first step in drawing a stratified random sample is to partition the population into G strata, each defined in terms of choices and attributes. Then sampling fractions H_1, H_2,..., H_G are selected as the fractions of the sample to be drawn from the G strata. The total sample size N_s is selected, and then $N_{sg} = H_g N_s$ observations are drawn at random from stratum g for all g = 1, 2,..., G. The fraction of the population that consists of members of stratum g can be defined as w_g. When the

sampling fractions H_g are chosen to be equal to the population share w_g, maximum likelihood estimation (MLE) for simple random samples is appropriate. The sampling fractions in the 1991 Oregon Angler Survey were chosen to be equal to the population share, so MLE is appropriate when using Survey data.

The telephone survey includes hypothetical questions intended for contingent value analysis. The percentage of fish at the site were increased, and the person was asked how many more fish they thought they would catch. The cost of the trip was then increased, and the person was asked if they would still go to the same site, or go to another site. The data were assessed by using frequency tables (Department of Agricultural and Resource Economics, Oregon State University 1993); fish numbers were increased in increments of 10 percent up to a 100 percent increase, and cost was also increased in 10 percent increments up to 100 percent. The cumulative proportion of people still willing to take the trip despite a 100 percent cost increase was 84 percent. This high a proportion of the respondents saying yes to the highest price variable means there is insufficient variability in the answer to use contingent value analysis with this data set.[12]

Data for trips to each management zone to fish for steelhead in rivers and streams were analyzed using SAS (SAS Institute Inc. 1988). An example of the programming used for each site is included in appendix A. Observations with obvious errors or with missing values in critical variables were dropped. SAS was also used to combine the information about trips to each site by individual. The programming that merges site information is included in appendix A. Data were used from all three data sets in the Oregon Angler's Survey--the address file, the mail survey, and the telephone survey. The primary source of data was the mail survey, as it had information about up to eight trips the

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[12]Robert Berrens, personal communication, June 1994. Assistant professor, Department of Economics, University of New Mexico, Albuquerque, New Mexico.

person had taken in a three month period. Data in the Angler's Survey were used to develop variables for the number of times an individual visited a particular management zone and water type (y_{jwi}); the average amount spent per individual on trips to site j water type w (p_{jwi}); the average catch rate per individual for species k in site j water type w (a_{jwk}); the total number of trips an individual took to all sites, in the water types and for the species being examined (K_i^s); and the gender of the individual (c_i). Number of visits to a particular site and to all sites are the dependent variables, which have been moved to the right hand side. Average catch rate is an indicator of site quality; catching fish is one determinant of demand for a site. The amount spent by people taking fishing trips is the proxy for price. In this case cost is self-reported. Distance cannot be determined for much of the sample.

Socioeconomic variables are included in behavioral demand studies because they are demand shifters. They help explain both the demand at a site and the demand from one site to another. Examples of socioeconomic variables include income, gender, age, race, and demographic origin. Socioeconomic variables that were available for this study were gender and age. Most of the anglers were male, but some were female. The percentage of women in the samples from the four regions varied from 7.5 to 23.1 percent. There was not, however, much variability in the average age from one site to another; the average was between 48 and 51. In addition, the model would not converge when age was included early in the model-building process, nor when it was included in the final model. Model building is a tightrope walk; the power of an explanatory variable must be weighed against the loss of degrees of freedom each additional variable brings. For these reasons, gender is the only socioeconomic variable included in the model.

3.3 Participation and Site Choice

This study uses a repeated discrete-choice random utility model to estimate demand for recreational fishing. The method models both participation and site choice. The data set cannot be analyzed by using CVM because contingent valuation questions in the survey do not have sufficient variability in the price variable. Hedonic modeling was not used because the policy question in this study is about a specific resource, and the hedonic model is used to answer questions about generic characteristics. A zonal travel cost model assumes homogeneous tastes and preferences, and requires that people from the same area visit the same site. All individuals in one area do not visit the same site. Both travel cost and random utility are valid methods that could be used to value steelhead in the ODFW management zones. Because individuals often visit more than one zone, substitution is an important consideration. There is some work being done to estimate a travel cost model using this data set. For these reasons, a random utility model was chosen as the method to model site choice and recreational values. Morey, Shaw and Rowe's (1991) model is modified to accommodate a stratified random sample with full information on each trip. The mail and address survey data are used, with some information brought in from the telephone survey.

The fishing season consists of T periods, where T is an estimated parameter. One alternative in the set of choices is the decision not to fish. If J fishing sites are available, then the individual chooses one of (J+1) alternatives in each period. The model has as its arguments individual characteristics, characteristics of the alternatives, and the reported trip costs for the four alternatives.

In section 3.3.1, the theoretical derivation of the random utility model developed by Morey, Shaw, and Rowe (1991) is outlined. They developed a model to deal with data in which not all the trip destinations are known. Section 3.3.2 compares the model developed by

Morey, Shaw, and Rowe to other random utility models. In section 3.3.3, the model used in this study is developed, using the theoretical basis outlined in section 3.3.1. The model in this study is different from that of Morey, Shaw, and Rowe because, in this study, destination is known for all trips. In addition, Morey, Shaw, and Rowe used an intercept survey, whereas this study uses a stratified random sample.

3.3.1 Morey, Shaw, and Rowe's (1991) random utility model for incomplete data sets

The survey sweep period of two months is divided into T decision periods, $t = 1, 2, \ldots, T$, such that no one can take more than one fishing trip per decision period. Each participant was assumed to know T but it cannot be observed; T is estimated in the model.

The choice set facing a person is characterized by J fishing sites with W water types at each site. The fishing sites are derived from the Oregon Department of Fish and Wildlife management zones. Oregon has seven management zones; they are the Columbia River, the northwest coastal zone, the southwest zone, the Willamette valley, central Oregon, northeast Oregon, and southeast Oregon. A map of Oregon with the ODFW management zones is included in appendix C. Data for the Columbia River zone and the northwest coastal zone were combined because relatively few observations were obtained from the Columbia River zone; respondents were probably confused about its boundaries and may have overlapped it with the northwest and the Willamette zones.[13] The central and the northeast zones were combined because each had relatively few observations. No observations for steelhead fishing were from the southeast zone. The sites used in this model are as follows:

j_1 = southwest zone;

j_2 = Willamette River zone;

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[13]Chris Carter, personal communication, March 1994. Economist, Oregon Department of Fish and Wildlife, Portland, Oregon.

j_3 = Columbia River and northwest coastal zones; and

j_4 = central and northeast zones.

These four sites are the sites people fished in; their destinations. People who went to these four sites were residents and non-residents of Oregon. The purpose of the random utility model was to derive marginal values for steelhead for steelhead anglers visiting these four sites in Oregon; there are four destination sites and one origin.

The water types included in the data set are river/stream, lake/reservoir, bay/estuary, and Pacific Ocean. A few observations for steelhead angling in bays and estuaries were from the southwest zone. In all other zones, steelhead angling observations were only from the river/stream water type. The only water type included in this study is rivers and streams. Rivers and streams could be considered to be distinct substitutes that enable anglers to switch depending on rainfall, mode, and so on. The Oregon Department of Fish and Wildlife, however, combines them into one water type. The w subscript is included with the model to demonstrate how the model can be generalized.

The individual has a choice of (JxW) fishing alternatives. For each individual, we observe $\{y_i, Q_i\}$ where

Q_i = the number of trips the individual takes, not including the trip where we observe distance traveled (so total number of trips = Q+1), and

y_i = $[y_{jwi}]$ where y_{jwi} = 1 if individual i visited site j water type w on the trip where we observe the distance, and y_{jwi} = 0 otherwise.

The individual is assumed to have consistent and transitive preferences over the alternatives that result in a unique preference ranking. The utility individual i receives during period t, if he chooses to fish at site j water type w, is

$$U_{jwti} = U(B_{ti}, p_{jwi}, a_{jw1}, a_{jw2}, \ldots, a_{jw6}) + \epsilon_{jwti}, \qquad (6)$$

where

B_{ti} = person i's budget for period t;

p_{jwi} = cost of a trip to site j water type w for person i; and

a_{jwk} = average per-trip per-person catch rate for species k at site j water type w.

ϵ_{jwti} = a random component known to person i when the decision is made, in period t. ϵ_{jwti} varies across individuals, sites/water types, and from period to period.

In period t, if individual i chooses not to fish, his utility for that period is assumed to be

$$U_{00ti} = \alpha(c_i) + \beta_0 B_{ti}, \tag{7}$$

where c_i is a vector of characteristics specific to individual i that affects the fish/nonfish decision but not the choice of site/mode. Under the assumption that c_i does not change, the physical characteristics of goods consumed when a person is not fishing are included in the constant term in α;

$$\alpha(c_i) = \alpha + \alpha_1 c_{1i} . \tag{8}$$

The only individual characteristic included in $\alpha(c_i)$ in this study is gender, so

$$\alpha(c_i) = \alpha + \alpha_1 sex_i. \tag{8a}$$

The term β_0 is the constant marginal utility of money. The assumption of a constant marginal utility of income is common in discrete-choice models even when income data are available because, without it, the expected consumer's surplus measures (the compensating and equivalent variation) will not have closed-form solutions (Morey, Shaw, and Rowe 1991).

Assuming utility maximizing behavior, at the beginning of each period, each individual chooses the alternative (JxW + 1 = sites * water types + not fishing) that provides the greatest utility in that period.

Given (6) and (7), the probability that individual i will choose site j water type w on a given trip is

$$\pi^{f}_{jwi} = \text{Prob}[U_{jwti} > U_{lhti},\ \forall\ l,h], \tag{9}$$

where l = other sites and h = other water types; jw ≠ lh. The probability that the individual will fish during period t (π_{fi}) is

$\pi_{fi} = 1 - \pi_{nfi}$ where π_{nfi} = probability of not fishing during period t,

$$= 1 - \text{Prob}[\alpha(c_i) + \beta_0 B_{ti} > U_{jwti},\ \forall\ j,\ w]. \tag{10}$$

Next, Morey, Shaw, and Rowe (1991) specifically assume that

$$U_{jwti} = u_{jwti} - \varepsilon_{jwti} = \beta_0 (B_{ti} - p_{jwi}) + \sum_{k=1}^{K} \beta_k a_{jwk} + \varepsilon_{jwti}. \tag{6a}$$

The choice of site/water type is independent of B_{ti}, which is implied by the assumption of a constant marginal utility of income, β_0.

The sample error ϵ_{jwti} is assumed to be randomly drawn from a Type I Extreme Value distribution. The joint cdf for a Type I Extreme Value distribution is

$$F_{\varepsilon}(\varepsilon_{ti}) = \exp\left[-\sum_{j=1}^{J} \sum_{w=1}^{W} e^{-\varepsilon_{jwti}} \right]. \tag{11}$$

Most discrete choice models assume that each ϵ_{jwti} is independently drawn from a Type I Extreme Value distribution. This distribution results in a "logit" equation for the probability of choosing site j water type w, given the decision to fish (equation 12 below). The Type I Extreme Value distribution simplifies the derivation of expected consumer's surplus measures. This follows from the fact that the maximum element of a vector of variables that are each independently drawn from a Type I Extreme Value distribution also has a Type I Extreme Value distribution (Ben-Akiva and Lerman 1985).

Given that the ϵ_{ti} is randomly drawn from a Type I Extreme Value distribution, it can be shown that the probability of fishing at a

particular site and water type is a function of the coefficient on cost, the specific trip cost, the individual's matrix of trip costs, the catch rate at site j water type w, and the matrix of site and water type catch rates for the individual, as follows:

$$\pi^{f}_{jwi} = \pi(\beta,\ p_{jwi},\ P_i,\ a_{jw},\ A)$$

$$= 1/\sum_{l=1}^{J}\sum_{h=1}^{W} \exp[\ -\beta_0(p_{lhi}-p_{jwi}) + \sum_{k=1}^{K}\beta_k(a_{lhk}-a_{jwk})]\ . \qquad (12)$$

With an incomplete data set, information about price variables available for only one or a few trips are included as elements in p_{jwi}. The probability of not fishing also follows a Type I Extreme Value distribution:

$$\pi_{nfi} = \pi_{nf}(\alpha(c_i),\ \beta,\ P_i,\ A)$$

$$= \exp\{-\sum_{j=1}^{J}\sum_{w=1}^{W}\exp[\alpha(c_i)-\beta_0 p_{jwi}+\sum_{k=1}^{K}\beta_k a_{jwk}\ ]\}. \qquad (13)$$

Given a sample size of N, the likelihood function formulated by Morey, Shaw, and Rowe (1991) for incomplete data sets for α, β, and X, given y_i and Q_i for each individual, is

$$L = \prod_{i=1}^{N} f(y_i, Q_i) = \prod_{i=1}^{N} f_1(y_i)\, f_2(Q_i)\ , \qquad (14)$$

where

$f(y_i,\ Q_i)$ = the joint probability distribution function (pdf) for y_i and Q_i,

$f_1(y_i)$ = the marginal distribution of y_i,

$f_2(Q_i)$ = the marginal distribution of Q_i,

X = T-1 = the number of periods not including the period where distance is observed, where X is restricted to be an integer $\geq$ the largest Q in the sample.

As was mentioned earlier,

Q_i = the number of trips the individual takes, not including the trip where distance traveled is observed (so total number of trips = Q+1), and

$y_i \equiv [y_{jwi}]$ where y_{jwi} = 1 if individual i visited site j water type w on the trip where the distance is observed, and y_{jwi} = 0 otherwise.

The pdf for y_i is the multinomial

$$f_1(y_i) = \prod_{j=1}^{J} \prod_{w=1}^{W} (\pi_{jwi}^{f})^{y_{jwi}} \tag{15}$$

and the pdf for Q_i is the binomial distribution:

$$f_2(Q_i) = [\frac{X!}{Q_i!\,(X-Q_i)!}] (\pi_{nfi})^{(X-Q_i)} (1-\pi_{nfi})^{Q_i}. \tag{16}$$

So the likelihood function is as follows:

$$L = \prod_{i=1}^{N} [[\frac{X!}{Q_i!\,(X-Q_i)!}] (\pi_{nfi})^{(X-Q_i)} (1-\pi_{nfi})^{Q_i}] [\prod_{j=1}^{J} \prod_{w=1}^{W} (\pi_{jwi}^{f})^{y_{jwi}}]], \tag{17}$$

and the log of the likelihood function is

$$\begin{aligned} \ln L(\alpha,\beta,X) = \sum_{i=1}^{N} \{ & \ln(X!) - \ln(Q_i!) - \ln(X-Q_i)! + (X-Q_i)\ln\pi_{nfi} \\ & - Q_i \ln(1-\pi_{nfi}) + \sum_{j=1}^{J}\sum_{w=1}^{W} y_{jwi} \ln(\pi_{jwi}^{f}) \}. \end{aligned} \tag{18}$$

This equation is the one that should be used with incomplete data from a stratified random sample. Morey, Shaw, and Rowe present a likelihood function to assess data obtained in intercept surveys, and include a correction for oversampling bias.

3.3.2 Comparison to other random utility model formulations

Morey, Shaw, and Rowe (1991) note that the participation decision depends solely on the form and stochastic specification of $\{\max (U_{jwti}) - U_{00ti}\}$, and not on the form or stochastic specification of any of the

specific conditional indirect utility functions. The variable $\{\max(U_{jwti}) - U_{00ti}\}$ has a Type I Extreme Value distribution, whereas the choice of a specific site/water type depends on the random variables $(U_{jwti} - U_{lhti})$, $jw \neq lh$, which are logistically distributed. Joint decisions as to whether to participate and, if so, where can also be modeled in either a repeated standard logit framework or a repeated nested logit framework. This model is not a special case of either of these, nor does it include either one as a special case.

The random utility model in this study is different than a repeated standard logit model or a repeated nested model. The model in this study has some advantages over both the repeated standard model and the repeated nested model. If equation 7, $U_{00ti} = \alpha(c_i) + \beta_0 B_{ti}$, is replaced with $U_{00ti} = \alpha(c_i) + \beta_0 B_{ti} + \epsilon_{00ti}$, where the vector $\epsilon_{ti} \equiv [\epsilon_{00ti}, \epsilon_{jwti}]$ is randomly drawn from a Type I Extreme Value distribution, the resulting model is a standard logit model (Morey, Shaw, and Rowe 1991). In a standard logit model, the random component of the participation decision has the same logistic distribution as the random component of the site/water type choice. The model in this study allows the random error terms in the participation decision and the site/water mode choice to have different distributions. In addition, the model in this study does not impose the independence of irrelevant alternatives assumption when the choice set includes nonparticipation. The relative odds of not participating versus choosing a particular fishing site will depend on the number of other sites available, their characteristics, and their costs. In a standard logit model, this ratio is restricted to be constant no matter what happens to the costs and characteristics at the other sites. Both the model in this study and the standard logit model impose the independence of irrelevant alternatives restriction across pairs of site/ water type alternatives.

A repeated nested logit model results if equations 6a and 7 are replaced by

$$U_{jwti} = \beta_0 (B_{ti} - p_{jwi}) - \frac{K}{k-1} \beta_k a_{jwk} + \varepsilon_{jwti} + \mu_{1ti} \tag{19}$$

and

$$U_{00ti} = \alpha(c_i) + \beta_0 B_{ti} + \epsilon_{00ti} + \mu_{1ti}, \tag{20}$$

where $\epsilon^0_{ti} \equiv [\epsilon_{00ti}, \epsilon_{jwti}]$ is randomly drawn from a Type I Extreme Value distribution, and μ_i is distributed so that both U_{00ti} and $\max_{jw} U_{jwti}$ are distributed Type I Extreme Value (Morey, Shaw, and Rowe 1991).

The repeated nested logit model is more general than the model used in this study, but it is also more complex. Because nested logit models are complex, they are harder to interpret, and adjusting for limited data sets is more difficult. The resulting consumer's surplus measures often will not have closed-form solutions.

The way a random utility model is formulated is important because the formulation reflects the way the person is assumed to make decisions about recreational options (Bockstael, McConnell, and Strand 1989). Modeling all the choice variables simultaneously assumes all the choices a person makes about where to fish, how to do it, and what to fish for are made simultaneously. Nested models allow the researcher to assume decisions are made sequentially; the person may decide first what to fish for and then where to go. When precise data about destinations, travel cost, distance, and site characteristics are available, nested models may be the best method to use. In this study, the quality of the conclusions about the value of additional fish produced by one land management alternative as opposed to another are affected by several levels of uncertainty. The link between management activities and changes in fish populations may have a large confidence interval, and the cost information included in the data from the mail survey is from one survey question, and is subject to the respondent's interpretation. A non-nested discrete choice model that models angler's choices simultaneously makes the assumption that anglers make decisions about where to fish, what to fish for, and how to do it, all at the same time.

The model does not impose any particular order on the decision process. In this sense, the non-nested model could be considered the least constraining.

3.3.3 Random utility model with full information

In their appendix, Morey, Shaw, and Rowe (1991) outline how their model can be generalized when destination information is available for a few or for all trips. The Oregon Angler Survey has destination for all trips. The model in this study was formulated using the suggestions presented by Morey, Shaw, and Rowe in their appendix. To expand the model, they define the following terms:

K^s_i = the number of trips individual i takes, where we have full information. Note the number of trips with full information can vary across individuals.

Q^s_i = the number of trips individual i takes in the year where we do not have full information.

y^s_i = $[y^s_{jwi}]$, where y^s_{jwi} is the number of trips for which we have information about site j water type w. Note that

$$\sum_{j=1}^{J}\sum_{w=1}^{W} y^{s}_{jwi} = K^{S}_{i}. \tag{21}$$

The marginal distribution of Q^s_i is a binomial. The term $T-K^s_i$ is the number of periods and the marginal distribution of y_i is a K^s_i trial multinomial. Given a random sample of N independent participants, the likelihood function for α, β, and T, given y^s_i and Q^s_i, is

$$L_k(\alpha,\beta,T) = \prod_{i=1}^{N} \{[\frac{(T-K^S_i)!}{Q^S_i!\,(T-K^S_i-Q^S_i)!}]\,(\pi_{nfi})^{(T-K^S_i-Q^S_i)}\,(1-\pi_{nfi})^{Q^S_i} \\ [\frac{K^S_i!}{\prod_{j=1}^{J}\prod_{w=1}^{W} y^{s}_{jwi}!}]\prod_{j=1}^{J}\prod_{w=1}^{W}(\pi^{f}_{jwi})^{y^{s}_{jwi}}\}. \tag{22}$$

If information about each trip is complete, $Q^s_i = 0$ and the likelihood function becomes

$$L_k(\alpha,\beta,T) = \prod_{i=1}^{N} \{(\pi_{nfi})^{(T-K_i^S)} [\frac{K_i^S!}{\prod_{j=1}^{J}\prod_{w=1}^{W} y_{jwi}^{s}!}] \prod_{j=1}^{J}\prod_{w=1}^{W} (\pi_{jwi}^{f})^{y_{jwi}^{s}}\} \tag{23}$$

and the log of the likelihood function with complete information on each trip is

$$\ln L_k(\alpha,\beta,T) = \sum_{i=1}^{N} \{(T-K_i^S)\ln(\pi_{nfi}) - \ln(K_i^S!) - \sum_{j=1}^{J}\sum_{w=1}^{W} \ln(y_{jwi}^{s}!) + \sum_{j=1}^{J}\sum_{w=1}^{W} (y_{jwi}^{s})\ln(\pi_{jwi}^{f})\}. \tag{24}$$

Data from the Oregon Angler Survey were assessed by using equation 24 to determine demand functions for fishing trips. These demand functions were used to determine values for mature steelhead in the seven management zones in Oregon. This model was modified from that used by Morey, Shaw, and Rowe (1991) because the model used in this study has been formulated to use data from a stratified random sample and to be used with full information on all trips.

The likelihood function representing demand for steelhead fishing trips was estimated using the *minimize* command in LIMDEP (Greene 1992). The likelihood function must be multiplied by negative one to be estimated using *minimize*. The following regression equation represents the general form of the negative of the log-likelihood function for steelhead in rivers and streams in the four sites in Oregon.

$$-LnL_k(\alpha,\beta,T)=\sum_{i=1}^{N}\left((T-K_i^s)\left(\sum_{j=1}^{4}\exp(\alpha_{j0}+\alpha_{j1}sex-\beta_j p_{ji}+\sigma_j a_j)\right)-\right.$$
$$\ln(k_i^s!)-\sum_{j=1}^{4}\ln(y_{ji}!)+ \tag{25}$$
$$\left.\sum_{j=1}^{4}(y_{ji})\ln\left(\sum_{l=1}^{3}\exp(-\beta_{lj}(p_{li}-p_{ji})+\sigma_{lj}(a_l-a_j))\right)\right)\Bigg),$$

where T = number of periods;

K_i^s = the total number of trips individual i takes to rivers and streams for steelhead in all sites;

$\alpha(c_i)$ = individual information (gender in this study);

β_j = marginal utility of money for site j;

σ_j = coefficient on the catch rate variable in site j;

p_{ji} = cost for individual i of going to site j;

a_j = average per-person catch rate for steelhead in site j;

y_{ji} = number of trips taken by individual i to site j to fish for steelhead in rivers and streams;

β_{lj} = substitution coefficient on price, across sites l,j; $l \neq j$; and

σ_{lj} = substitution coefficient on catch rate, across sites l,j; $l \neq j$.

The four sites in order are the southwest zone, the Willamette zone, the combined Columbia River and the northwest coastal zone, and the combined central and northeast Oregon zone (see appendix c for a map). These sites are the destination; there is one origin, as described in chapter 3.

The number of periods, T, was derived by estimating several models that differed only in that in each model, T was restricted to a different integer value so $T \geq K_i^s$. The log of the likelihood function was maximized when T = 8. The parameter values were obtained by using the Davidon, Fletcher, Powell (DFP) algorithm in LIMDEP. The DFP algorithm is a gradient method with good convergence properties for the function, even for poorly behaved problems. It is a quasi-Newton

method, and is outlined in Greene (1990). The algorithm forms $H_{t+1} = H_t + E_t$, where H_t and E_t are positive definite matrices. After a sufficient number of iterations, H_{t+1} will be an approximation to $-G^{-1}$, where

$$G = \frac{\partial^2 F(\theta)}{\partial\theta\partial\theta'}. \tag{26}$$

Although H_t accumulates an estimate of G, the negative inverse of the second derivatives matrix, in maximum likelihood problems H_t rarely converges to a very good estimate of the estimator. Greene cautions that the variance matrix should not be relied upon to assess the significance of each coefficient. The t-values of the coefficients are included in the LIMDEP output in appendix B. Greene says that with linear models, a low R^2 or some other diagnostic test may suggest a problem, such that the model and data are mismatched. As long as the full rank condition is met by the regressor matrix, however, a linear regression can always be computed. Nonlinear models are not so forgiving. Using different starting points resulted in the same coefficients, which suggests that the maximum likelihood estimates were found.

The likelihood function is highly nonlinear and was difficult to estimate. Some manipulation of the data was necessary to make estimation of the function possible. The cost information was divided by 100 so the cost data would be of the same relative scale as the data on gender and catch rate. Gender was manipulated so that it became a dummy variable; women were zero and men were one. In addition, the convergence criteria were changed; the proportional change in the function value was set to less than 0.001.

3.4 Expected Consumer's Surplus Measures

Site-specific recreational activities are often considered separable from all other commodities in the consumer's choice set.

Assuming separability is appropriate, a partial demand system can be estimated, based on prices and attributes of the goods, and the individual's budget and perhaps social and demographic information. A partial expenditure function can be derived, which can then be used to estimate a partial compensating or equivalent variation (CV or EV) associated with a change in the attributes of the goods being assessed. Care must be taken in using these partial measures of CV and EV because the relation between partial measures of consumer's surplus and the conventional CV and EV is not well understood. Hanemann and Morey (1992), however, provide a proof that partial CV is less than or equal to CV as estimated from a complete demand system (conventional CV). They state that, intuitively, partial CV provides a lower bound on conventional CV because conventional CV includes budget adjustments a person will make when conditions change, but partial CV does not.

If recreation fishing is assumed to be separable from all other activities in the utility function, the partial indirect utility function can be defined as

$$U = \vartheta(P_i,\ A,\ C_i,\ B_{ti}), \tag{27}$$

where P_i = the matrix of costs for individual i;

A = site\water type catch rates;

C_i = characteristics of the individual; and

B_{ti} = the budget for individual i in period t.

The indirect utility function can be used to define monetary measures of the welfare effects of a proposed change.

Let $P_i^\circ = [p^\circ_{jwi}]$ be the initial matrix of costs for person i;

$P_i' = [p'_{jwi}]$ be the new matrix of costs for person i;

$A^\circ = [a^\circ_{jwk}]$ be the initial matrix of site/water type catch rates; and

$A' = [a'_{jwk}]$ be the new matrix of site/water type catch rates.

Individual i's expected per-period compensating variation (PPCV) (and equivalent variation (PPEV)) associated with a change from (P^o, A^o) to (P', A') is

$$PPCV_i = PPEV_i = (1/\beta_o)[\vartheta(P_i', A', C_i, B_{ti}) - \vartheta(P_i^o, A^o, C_i, B_{ti})], \tag{28}$$

where $\vartheta(P_i', A', C_i, B_{ti})$ is person i's expected maximum utility in the new state and $\vartheta(P_i^o, A^o, C_i, B_{ti})$ is the person's expected maximum utility in the initial state. Multiplying by the negative of the inverse of the coefficient on travel costs, $(-1/\beta_o)$, converts the expected utility into a money metric of the expected change. The negative is used because integration occurs under a demand curve with a negative slope. Formulas for calculating expected consumer's surplus are reported in several studies (such as Hellerstein 1992; Bockstael and Strand 1987; Adamowicz, Fletcher and Graham-Tomasi 1989). The CV_i equals the EV_i because there are no income effects in the model. The $PPCV_i$ is the compensation (or payment) that would make the expected maximum utility after the change the same as it was before the change (Morey, Shaw, and Rowe 1991).

Given that ϵ_{ti} in the conditional utility function, equation 6a, has a Type 1 Extreme Value distribution, Morey, Shaw, and Rowe (1991) show that

$$\begin{aligned}\vartheta(P_i, A, C_i, B_{ti}) &= E\max[U_{00ti}, U_{11ti}, U_{12ti}, \ldots, U_{1wti}, \ldots, U_{jwti}] \\ &= U_{00ti}F_z(U_{00ti}) - \int_{U_{00ti}}^{\infty} z_{ti} f_z(z_{ti})\, dz_{ti},\end{aligned} \tag{29}$$

where $F_z(z_{ti})$ and $f_z(z_{ti})$ are, respectively, the cdf and the pdf associated with the random variable z_{ti}. Note that

$$F_z(z_{ti}) = \exp[-e^{-(z_{ti}-K_{ti})}], \tag{30}$$

where

$$k_{ti} = \ln \sum_{j=1}^{J} \sum_{w=1}^{W} e^{U_{jwti}}. \tag{31}$$

The first term in equation 29 is the probability of not fishing (equation 13) multiplied by the utility from not fishing. If equation 31 is substituted into equation 30 and evaluated at $z_{ti} = U_{00ti}$, the result is π_{nfi}, equation 13. The second term in equation 29 is the contribution to expected maximum utility from having the fishing alternative available.

To calculate $PPCV_i$ for a given change, $\vartheta(P_i, A, C_i, B_{ti})$ has to be evaluated in both states. Note that $\vartheta(P_i, A, C_i, B_{ti})$ is a function of B_{ti}, but that $[\vartheta(P_i', A', C_i, B_{ti}) - \vartheta(P_i^o, A^o, C_i, B_{ti})]$ is not. The second term in equation 29 does not have a closed form, but it can be accurately approximated using numerical techniques. Morey, Shaw, and Rowe (1991) point out that

$$\int_{U_{00ti}}^{\infty} z_{ti} f_z(z_{ti})\, dz_{ti} = e^{(K_{ti} - U_{00ti})} \int_0^{\infty} e^{-x} m(x)\, dx, \tag{32}$$

where

$$m(x) = (x + U_{00ti})\ e^{-e^{-(x + U_{00ti})}}. \tag{33}$$

The expression $\int e^{-x}\, m(x)\, dx$ is a Laguerre integral and can be approximated by using formulas in math handbooks, such as Abramowitz and Stegun (1966).

Given the parameter estimates, $PPCV_i$ can be calculated for any individual as a function of (P^o, A^o) and (P', A'). The first step is to increase the level of the fish catch variable from its current value, a_{jwk}, to a'_{jwk} in equations 12 and 13. Next, the log-likelihood function is estimated for the initial and the new states. Then $\vartheta(P_i, A, C_i, B_{ti})$ is calculated for each individual visiting the site for each state. The $PPCV_i$ is calculated using equation 28. This calculation gives the per-trip value of a steelhead in the site in question, and is the per-fish value that will be used to value the difference in fish production from alternative A to alternative B.

Equation 25 was used to estimate the log-likelihood demand function for the initial conditions. One fish was added to everyone's catch rate, and the log-likelihood function for the new state was estimated. This was done twice, to calculate the value of the first additional fish caught, and then the value of another additional fish. The second additional fish should have diminishing marginal value. Equation 29 was then estimated for the initial conditions and the new state, for each change. Some specific assumptions about how to estimate equation 29 were made in accordance with economic theory and the fact that the goal was to estimate coefficients and consumer's surplus for each zone, while allowing for substitution across zones.

One assumption made was the manner in which utilities were calculated. Equation 7 is the general equation for the utility of not fishing, U_{00ti}. For this study, equation 7 was estimated by

$$U_{01ti} = \alpha_{21} + \alpha_{11}(\text{sex}) + \beta_1 * \text{budget}_{ti}. \qquad (34)$$

Equation 34 was estimated for each site not fished in every choice period for each individual. The utility of fishing, U_{jwti}, was calculated by using equation 6a for each site fished.

Budget was estimated by using census information and some data from the telephone survey. Income information was available for a few individuals because some people from the mail survey were interviewed in the telephone survey. For the rest of the sample, average household income by zip code was used for income. Six sample waves of 2 months each, and T=8 in each sample wave, results in 48 annual periods. Income was divided by 48 for the budget in period T.

The probability of fishing was calculated using equation 13, and is based on catch rates and expenditures in all sites the individual visited. In all calculations of utility and probability, β_j was scaled by dividing by 100 because cost was scaled in the log-likelihood function. When consumer's surplus was estimated, the values were

unscaled. The values for average per-period CV for steelhead in rivers and streams were calculated by using equation 28.

4. Results from the Production Model and the Random Utility Model

4.1 Fish Production in Elk River Tributaries

Values for the landslide sediment delivery index (LSDI) model were calculated for the North Fork, Panther Creek, Bald Mountain Creek, Blackberry Creek, Butler Creek, and the South Fork of the Elk River for the 1990 Forest Plan (alternative A) and the alternative that best approximates FEMAT (alternative B).[14] The existing and the predicted LSDI values and existing habitat condition index (HCI) values are reported in Table 2. The predicted LSDI values for each tributary were used to calculate a predicted HCI value for each alternative and each tributary by using equation 5. The HCI values were converted to fish numbers using the conversion factors listed in Table 1.

Data for the average fish count for 1986 and 1987 were obtained from fish counts on the Elk.[15] The fish count data were expanded by number of units of habitat types by multiplying the mean number of fish per habitat type (the sample) by the sum of the individual habitat types,[16] a method developed by Hankin and Reeves (1988). Streams are stratified by natural habitat units, such as pools and riffles, and independent samples are drawn from strata constructed on the basis of habitat type and location. The method was developed after Hankin (1984) showed that errors of estimation of total fish abundance in small streams within selected strata are likely to be small compared with errors resulting from variation in fish numbers between strata.

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[14]LSDI numbers provided by Cynthia Ricks, August 1994. Geologist, Siskiyou National Forest, Gold Beach, Oregon.

[15]Gordon Reeves, Research Scientist. Pacific Northwest Research Station, Corvallis, Oregon.

[16]Calculations by Cindy Rugger, September 1994. Statistician/Biologist, Pacific Northwest Research Station, Corvallis, Oregon.

Creek		North Fork	Panther	Bald Mt.	Black berry	Butler	South Fork	Total
LSDI[a]	Exist -ing[c]	0.65[d]	1.06	1.91	2.05	4.5	6.11	--
	Alt. A	0.595	0.728	1.171	0.648	0.393	0.453	--
	Alt. B	0.243	0.397	0.829	0.170	0.213	0.172	--
HCI	Exist -ing	1[d]	0.87	0.821	0.710	0.55	0.48	--
	Alt. A	0.943	0.907	0.640	0.928	1.015	0.99	--
	Alt. B	1.096	1.013	0.884	1.155	1.118	1.153	--
Fish[b]	Avg. '86-'87	1336	680	4426	228	590	628	7888
	Alt. A	1260	709	4426	298	1089	1295	9077
	Alt. B	1464	792	6113	371	1199	1508	11447

Table 2. Existing and predicted LSDI, HCI, and number of mature steelhead per year for Elk River tributaries under two forest management alternatives.
a: Decade sediment delivery in cubic yards, per acre.
b: Number of fish; annual adult steelhead production by the end of the first decade.
c: Ricks and Chen 1990.
d: Value for lower North Fork from Ricks and Chen 1990.

Note that predicted HCIs are clustered around 1.00 for most of the creeks under the two alternatives. The lower North Fork was indexed at 1.00 in the existing HCI calculations, and the other tributaries in the Elk River watershed had values as low as 0.46. The predicted HCI for alternatives A and B was calculated by using equation 5. The limitation

of the HCI model, as mentioned by Ricks and Luce[17] is that it assumes all other tributaries can have an HCI as high as the lower North Fork.

There are other possible sources of error resulting from the use of the HCI as a proxy for a fish production function. A simple assumption was made about the conversion of the fish index to fish numbers. As mentioned in chapter three (table 1), the HCI was converted to fish numbers by dividing the average fish count in each stream by the HCI for the stream. A simple sensitivity analysis of the LSDI model (Ricks and Chen 1990) showed that the error in the LSDI prediction with one more landslide was 15 percent more effect than was predicted on the North Fork, and no change in the South Fork. Some of the most important tributaries in terms of fish production were the most sensitive to errors in prediction. Variability in the HCI model can be assumed to add to the variability in predictions when linked to the LSDI model.

The predicted difference in numbers of yearling steelhead produced in all tributaries of the Elk River between alternatives A and B is 2370 fish, an increase of 26 percent. Although the difference in predictions for number of fish between alternatives A and B was not tested statistically, no significant difference may exist in numbers of fish produced by each management alternative after variability in each model is taken into account. This variability is a function of the structure and interaction of the models. Focusing on the trends, the interaction of the LSDI and the HCI models suggested the production of adult steelhead would increase in these tributaries, both from current conditions and from the Forest Plan proposal (alternative A), if the Elk River watershed is managed as directed under FEMAT (1993) (alternative B).

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[17]Cynthia Ricks, personal communication, 1993-94. Geologist, Siskiyou National Forest, Gold Beach, Oregon. Charles Luce, personal communication, 1993-94. Research Hydrologist, Intermountain Research Station, Moscow, Idaho.

4.2 The Likelihood Function

4.2.1 Results

The maximum likelihood estimates of the parameters are summarized in Table 3.

Coefficient	j = 1 southwest	j = 2 Willamette	j = 3 Col. R.-northwest	j = 4 central-northeast
α_{0j} intercept	-2.3508	-2.6325	-2.7750	-2.4375
α_{1j} sex	-2.1435	-2.4704	-2.6481	-2.2239
β_j cost	0.38497	0.28539	0.32457	0.22073
σ_j catch	-1.2092	-1.2030	-1.3046	-0.75441
β_{1j} subst. cost j-SW	NA	-0.47429	-0.28526	-0.35654
β_{2j} subst. cost j-Wil	-0.36988	NA	-0.42460	-0.37667
β_{3j} subst. cost j-CNW	-0.47177	-0.45229	NA	-0.35572
β_{4j} subst. cost j-CNE	-0.46647	-0.44373	-0.41267	NA
σ_{1j} subst. catch j-SE	NA	2.1643	1.8529	1.2476
σ_{2j} subst. catch j-Wil	2.4752	NA	1.8164	1.2442
σ_{3j} subst. catch j-CNW	2.4387	2.2024	NA	1.2477
σ_{4j} subst. catch j-CNE	2.4397	2.1757	1.8398	NA
log-likelihood function value: 48.26945				

Table 3. Maximum Likelihood Estimates.

The t-values for the coefficients are reported in appendix B. The t-values for the intercept, sex, cost and catch are all insignificant; the coefficient on catch is the least significant of the four for most

regions. T-values for many of the substitution coefficients on cost and catch rate are significant. The gradient matrix is also reported in the appendix. The optimization algorithm approximates the hessian by calculating numeric approximations for the derivatives, which in this case violate the second-order conditions for a maximum, as 16 of the gradient values are negative. As Greene (1990) cautions, t-values in this case should be interpreted with care.

4.2.2 Interpretation and likelihood function sensitivity analysis

The intercept and the coefficient on gender are negative for every region. Because of interaction between coefficients and the embedded structure of the model, saying unequivocally what each parameter means is difficult. One thing that can be examined is the relation between gender of anglers and number of trips, and whether the way the gender variable was defined influences the results. The average number of trips taken by women to fish for steelhead is higher than the average number of trips taken by men in the Columbia River-northwest region, and also higher in the southwest region. Because sex = 0 for women, it may have influenced the sign on the coefficient. When women and men were reversed, however, so that women = 1 and men = 0, the likelihood value changed only slightly, and none of the signs on the coefficients changed. Some coefficients changed values in the fourth significant place. Far more men than women were in the sample. The results are fairly stable regardless of how the gender variable is defined.

The coefficient on catch rate is negative in all regions. This implies that as catch rates go up, the number of trips taken declines. An examination of the relationship between catch rate and number of trips taken to a region will show one possible reason as to why the coefficient is negative. The average catch rate per trip, by region and by number of trips taken, is summarized in Table 4. The number of people taking certain numbers of trips in each region is also included.

Number of trips	Average catch rate and number of observations, by region			
	southwest	Willamette	Columbia River - northwest	central-northeast
1	1.18 n=48	0.29 n=51	0.77 n=62	0.43 n=23
2	0.68 n=25	0.21 n=54	0.41 n=37	1.00 n=16
3	0.49 n=16	0.23 n=32	0.35 n=26	1.60 n=14
4	1.29 n=10	0.62 n=14	0.27 n=14	0.60 n=7
5	0.81 n=9	0.44 n=14	0.15 n=5	0.10 n=3
6	0.79 n=8	0.48 n=7	0.69 n=6	0.30 n=1
7	0.80 n=1	0.06 n=5	0.19 n=3	1.86 n=1
8	1.15 n=4	0.20 n=5	1.00 n=1	0.30 n=1

Table 4. Average catch rate per trip and number of observations, by region and by number of trips taken.

Each person in the survey could have taken as many as eight trips. Most took fewer. A simple linear correlation was run in SAS (1988) between trips and average catch rates. The relationship is negative for the southwest and the Columbia River-northwest regions. More than half the users in the sample (276 out of 475) fished in these regions.

The coefficient on catch rate is probably negative because the model is estimating demand for number of trips based on, among other things, catch rates that generally decline as more trips are taken. Why this happens could be endlessly conjectured about, as could why people are willing to take numerous trips for which the primary purpose is to fish for steelhead when the average catch rate hovers close to zero. Probably the more experienced anglers have higher catch rates and people fish for anadromous fish more for the experience than in the expectation that they will catch their limit of fish. Such hypotheses cannot be tested with this data set. The fact remains that as number of trips increases, average catch rate declines, giving the coefficient on catch

rate a negative sign. The implication is that people take trips to fish for steelhead for many reasons other than expectation of a consistent catch rate. Other reasons for a negative coefficient could be recall bias, in that people taking more trips could have a hard time remembering how many fish they caught on each trip, or strategizing, in that people who take more trips may under-report the number of fish they caught. Anglers sometimes say they caught fewer fish than they actually did when talking to others, in an effort to make a site look less appealing to others.

The coefficient on price was positive, as would be theoretically expected. In the equation, price was subtracted. As price increased, the demand for trips would decrease, decreasing the value of the log-likelihood function.

The substitution coefficient on price was consistently negative. In the function, this coefficient had a minus sign before it. As the price of a trip elsewhere increases, the demand for trips to the site in question increases, increasing the value of the log-likelihood function. The substitution coefficient on catch rate was consistently positive. The implication here was that as catch rate went up elsewhere, the demand for trips to the site being examined increased. This change seems odd, but keep in mind that in most regions, as the number of trips goes up, the catch rate declines. This pattern is another indication that catch rate, as defined, is not the only reason people go fishing for steelhead.

To test whether the sequential order of the regions in the likelihood function influenced the values, the order was changed (number 4 (the central-northeast zone), became number one, and number one (the southwest zone) became number 4) and equation 25 was re-estimated. The likelihood function value changed slightly, and some coefficients changed values in the fourth significant place. The order in which the regions appear in the likelihood function do not have much effect on the

values obtained for the likelihood function coefficients; the results are not subject to path dependency.

4.2.3 Comparison of results to Morey, Shaw, and Rowe's (1991) random utility model

The results of the estimation in this study were quite different from the results reported by Morey, Shaw, and Rowe (1991). They reported values people placed on fishing in general, not on trips or fish. In addition, this model differed from theirs in several respects. As mentioned, this model was formulated to use a stratified random sample, and data that included some information for all trips. In addition, in this study, coefficients were estimated for each site, rather than the state as a whole. Morey, Shaw, and Rowe based their estimation on an average individual from each of 36 counties. In this study, individual data were retained and used to estimate separate information for each of four regions. Morey, Shaw, and Rowe developed estimates based on people's origins, whereas this study developed estimates of fish value for sites visited by people both from within and outside the state. Morey, Shaw, and Rowe also developed estimates of how much people from various sites valued fishing in Clatsop County. In this study, estimates were developed of how much, on average, people from one large origin valued an additional steelhead caught in four regions in Oregon.

Morey, Shaw, and Rowe (1991) included information about travel time and distance in their study of angler behavior. In this study, the only specific trip cost data available was the cost information reported by individuals taking the survey. How far they traveled or how long it took was unknown. Detailed information on trip expenditures was not gathered in the mail survey. A smaller data set was available, the telephone survey, that includes detailed information, and perhaps it could be used with the model that uses incomplete trip data. If the

telephone survey data were to be assessed with the model presented in this study, several species and water types would have to be assessed because the total number of observations was relatively small. To retain individual information and estimate coefficients at each site, such a model would have 40 additional coefficients for each additional water type, and 16 *(total number of water types) additional coefficients for each species added. About 1800 observations were in the more detailed telephone survey. To use the data from the telephone survey, both the model presented by Morey, Shaw, and Rowe (1991) and the model presented in this study would probably need to be reformulated with averaged data, resulting in statewide coefficients.

4.3 Estimated Expected Consumer's Surplus

The marginal value of an additional steelhead was calculated using equation 28. The probability of fishing used in equation 28 was calculated using catch rates and expenditures in all sites the person visited, using equation 35. The per-period compensating variation (PPCV) values for an additional steelhead caught are reported by region in Table 5. The PPCV values for a second additional fish are also included. These are per-trip measures of value.

Region	PPCV of the first additional fish	PPCV of the second additional fish
southwest	10.07	2.42
Willamette	13.53	3.33
Columbia River-northwest	11.93	2.83
central-northeast	15.63	3.58

Table 5. Per-trip value of an additional steelhead, 1989 dollars.

The value of the second additional fish is considerably less than that of the first additional fish, in all regions. Steelhead have diminishing marginal value. The limit on steelhead is two fish per day, but even if an angler catches just one, they are quite satisfied. The Oregon Angler Survey does not distinguish between steelhead caught and released and steelhead caught and kept, so the low value of a second additional fish applies to both.

The number of observations, average number of trips, average catch rate in the initial state, average cost and the PPCV for an additional steelhead caught for each region are shown in Table 6. Each of these factors plays a part in determining recreational steelhead values.

Region	Number of observa -tions	Average number of trips	Average catch rate per trip	Average cost per trip	fish value
southwest	121	2.628	0.9398	73.54	10.07
Willamette	182	2.736	0.293	40.20	13.53
Col. River - northwest	154	2.357	0.535	47.96	11.93
central - northeast	66	2.400	0.839	73.59	15.63

Table 6. Fish value (PPCV), number of observations, average trips, average catch rate, and average cost for each region.

The average number of trips taken in each region is virtually the same from one region to another. Since the average of trips taken is similar, number of trips taken would not be expected to cause differences in fish values. Number of trips taken is, however, an important determinant of fish values. The difference in PPCV for the first additional steelhead in the Willamette and the combined Columbia River-northwest zones is the smallest for any pair of regions. Part of

the similarity in value can be attributed to similarity in cost, which results from their adjacent locations. In addition, substitution between sites is an important determinant of price differentials. The more substitution that occurs, the more similar prices will be. The rate of substitution between the Willamette and the Columbia River-northwest regions was higher than between any other pair of sites. Twenty-two people surveyed took trips to both the Willamette and the Columbia River-northwest regions, more than twice the rate reported for any other pair of sites. Nineteen percent of people who visited the Willamette region visited at least one other site. Eighteen percent of people who visited the Columbia River-northwest region reported trips to at least one other site.

The least amount of visitation to other sites occurred for people who reported trips to the central-northeast region. Twelve percent of the people who visited the central-northeast region reported a trip to one other site. No one took trips to both the central-northeast and the southwest regions. The low rate of substitution between the central-northeast region and all other regions is one factor causing the price for an additional steelhead in the central-northeast region to be the highest in all regions.

The second lowest rate of substitution occurred between the southwest region and all other pairs of sites; slightly more than 12 percent of the people who took trips to the southwest visited another site. Most people who visited another site went to the Willamette region. The PPCV for an additional steelhead in the southwest region is the lowest in all regions.

The average number of trips, the average catch rate and the average cost per trip were quite similar between the southwest and the central-northeast regions, but the PPCV of the first additional fish is one-third higher in the central-northeast region. The contribution of utilities of places not visited appears most striking in this difference

in values between the southwest and the central-northeast zones. Every individual had a utility measure both for places visited and for places not visited. The interaction of catch rate, cost, substitution between sites, and utility measures determine values obtained in this study. In addition, as Loomis (1989) points out, other factors affect values for fish, including quality and preference variables; those factors influence the error term and thereby the answers.

To relate the value of fish to anglers to the difference in number of fish produced by a change in management, catch rates must be related to population. The Oregon Department of Fish and Wildlife has unpublished data relating catch rates to populations of adult fish, in terms of catch-to-escapement ratios, in the 1977 ODFW Manual for Fish Management. The ratio of catch to escapement listed for winter steelhead on the south coast of Oregon is 0.6 to 1.[18] Loomis (1988) used conversions from Kunkel and Janik (1976), in which a catch-to-escapement ratio of 0.6 to 1 is listed for the south coast. The 1977 ODFW Manual for Fish Management also cites Kunkel and Janik as a source of catch-to-escapement ratios for some species. A catch-to-escapement ratio of 0.6 to 1 was used for this study.

The increase in fish production between alternatives A and B in the North Fork, Panther Creek, Bald Mountain Creek, Blackberry Creek, Butler Creek, and the South Fork is 2370 steelhead per year at the end of the planning period. The planning period is one decade. For this study, the LSDI per acre value was determined for planned activities in the 1990s. Predicted HCI values predict the habitat condition at the end of the decade. With a catch-to-escapement ratio of 0.6 to 1, 889 steelhead of the 2370 additional steelhead produced could be caught by recreational anglers. Assuming the PPCV of $10.07 is an accurate

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[18] The term "catch to escapement ratio" is not a division-type ratio. In this case, it means that 0.6 fish are caught out of a total of 1.6 fish produced, equivalent to 37.5 percent of the total population being caught. Chris Carter, personal communication, October 1994. Economist, Oregon Department of Fish and Wildlife, Portland, Oregon.

estimate of the value of an additional steelhead for the southwest region, the increased production of fish in these tributaries at the end of the decade would be worth about $8,952 annually, in 1989 dollars.

Two basic assumptions must be made to calculate a present net value (PNV) for the increased production of fish. The first regards the function that best represents the increase of the steelhead population to the end-of-the-decade level of 2370 additional fish. The second has to do with the assumed interest rate used to discount future values of fish.

The LSDI and the HCI models indicate that 2370 additional fish will be produced in the year 2000, when the watershed is managed under FEMAT guidelines. No additional fish are assumed to be produced until the fourth year of the decade, and the increase thereafter is linear. This pattern approximates a log-linear increase in fish population for the decade. After the year 2000, three possibilities are presented. The population could remain stable at end-of-decade size, it could drop by one-third, or it could increase by one-third. The PNV of these three options are represented in table 7. A discount rate of 4 percent is used in the analysis; values are for the difference in fish produced under the FEMAT alternative as compared to the Forest Plan, in 1989 dollars.

Population projection	Present net value
stable after 2000	$177,414
drops by 1/3 after 2000	$127,073
increases by 1/3 after 2000	$227,756

Table 7. Present net value of the difference in steelhead population projections between the Forest Plan and FEMAT on Elk River tributaries.

The chinook and trout populations would probably also increase under FEMAT, adding their share of recreational consumer's surplus to this analysis. Even if all three fish populations increased, however, forgone timber values would probably still exceed their value.

In summary, the random utility model served satisfactorily to estimate recreational steelhead values in Oregon. Results from the likelihood function were stable, regardless of the definition of the gender variable and the order of the regions. Steelhead in all the regions have steeply diminishing marginal value. Substitution, catch rates, and trip costs affect fish values, in addition to utility measures both of places visited and places not visited in each decision period. The value of recreationally caught steelhead in the Elk River is probably not as high as the market value of the timber in the watersheds of each tributary in the analysis, but recreational use value of steelhead are only part of the value of the fish. Steelhead have recreational use value and they have nonuse values. People place a value on simply knowing the fish exist. In the next chapter, the random utility model will be discussed and put into context in terms of how it fits into the vertically integrated production function for steelhead production on the Elk (figure 1) and where it is used in modeling the effects of land management alternatives on different resources (figure 2).

5. Conclusions

This study focused on the use of a random utility model to assess consumer's surplus values for an additional steelhead caught by individual anglers in four regions in Oregon, and the application of one of the values to a production function for steelhead on the Elk River in southwest Oregon. In this section, the random utility method will be assessed, and then the implications of its use in assessing management alternatives will be discussed. Finally, needs for further research will be summarized.

5.1 The Random Utility Model

Random utility modeling assumes people choose recreation sites from a set of choices, each time they go. The randomness comes from the assumption that when recreationists' behavior is modeled, the individuals' utility functions are treated as though they are random to the observer. The recreationist is always assumed to choose the utility-maximizing alternative. The objective function expresses the relative desirability of an alternative in terms of its characteristics. Both random utility and travel cost methods model the demand for services at a site. They use the same type of information in model formulation, such as travel cost, distance, individual characteristics, and site characteristics.

The random utility model in this study is computationally complex, but with the rapid advances in computer technology and software, such models are becoming easier to estimate. The model is nonlinear, and it reaches across sites for each individual to derive values for demand for angling in each zone, whether the individual visited that site or not. Many recreation demand studies do not account for substitution, but it occurs. If site substitution is thought to occur, it should be dealt

with when site demand and the values of site characteristics are calculated.

Economic analysis must commonly use data not entirely suited to the desired analysis because there is usually no opportunity to generate ideal data. This study showed that imperfect data can be used to develop estimates of value that would otherwise be difficult to derive using traditional travel cost methods. The Oregon Angler Survey reports destinations for regions, not recreational sites. The survey does report the primary purpose of individual trips, so many people who caught no fish were included in the analysis. Knowing the purpose of the trip gives a better indication of the true willingness-to-pay of anglers for a specific species. Including trips in which no fish were caught makes the value of fish higher than if only trips in which fish were actually caught were included in the analysis.

Several kinds of information are not included in the random utility model in this study. Although substitution across pairs of sites is accounted for, substitution across all sites simultaneously is not modeled. In addition, substitution across species is not included. Salmon are probably an important substitute for steelhead. Trout may be a substitute too, to a lesser extent. Because of the nature of the data set, estimates of travel cost are based only on people's reported trip cost. Distance and time traveled are not specifically included, although people may have included them in their estimates of trip cost. Assuming most people did not include the value of time in their estimate of trip cost, the values in this study are lower than if time and distance were fully accounted for.

The values in this study are measures of an average individual angler's willingness-to-pay for an additional steelhead caught, in particular zones in Oregon. The value applies to anglers who went on trips with a primary purpose of fishing for steelhead. It includes both residents and nonresidents surveyed in the Oregon Angler Survey. Loomis

(1989) shows that two important determinants of steelhead values are average price (i.e., trip cost) and population in the market area. Marginal values per fish are sensitive to price. He points out that variables other than distance and population influence the marginal values per fish. Loomis is talking about the population of the site of origin of anglers. The Oregon Angler Survey was stratified based on population of origin, so more people were selected from heavily populated areas than from sparsely populated areas. The estimated marginal values for an additional steelhead are based on trips to the region in question.

Several recent studies have assessed marginal values for steelhead in the Pacific Northwest. They are not directly comparable because each of them measures the value of an additional fish by using different methods, different assumptions, and different data. Values will differ by the valuation method used for a variety of reasons. Some studies account for substitution effects; others do not. Because the values derived in this study include substitution across sites, the values should theoretically be lower than values calculated for the same areas using a model without substitution. Different methods use different ways to measure price, by observing behavior or by asking people what they think something is worth. Welfare measures in many studies are calculated by using Marshallian demand as an approximation of Hicksian demand. The expected value of the maximum of the indirect utility function is used to measure welfare in the random utility method. Values will also differ from one study to another based on the area considered to be a recreation site, the data used, and the way the data are manipulated to formulate variables used in the model. Individual data versus averaged data can yield different answers. The estimates nevertheless give an idea of the range of values and the trends in differences in values from one site to another.

Some recent studies that derive values for steelhead and salmon are compared in table 8.

Study	Morey, Shaw, and Rowe 1991	Olsen, Richards, and Scott 1991	Loomis 1989	Johnson and Adams 1988	This study
method	random utility	contingent value	pooled travel cost	contingent value	random utility
substi-tution	yes	no	indirect-ly	no	yes
use value	yes	yes	yes	yes	yes
nonuse value	no	yes	no	yes	no
scale of area	county	regional	rivers	John Day River	regional
good valued	salmon fishery	steelhead caught	steelhead caught	steelhead caught	steelhead caught
value report-ed (year dollars are in)	$0.31 to $2.76 per person (1981)	$25 to $55 per fish (1989)	$18 to $333 per fish (1984?)	$6.65 per fish (1986)	$10.07 to $15.63 per fish (1989)

Table 8. Comparison of recent studies reporting values for salmon and steelhead in the Pacific Northwest.

The economic model in this study adapts a random utility model developed by Morey, Shaw, and Rowe (1991). Morey, Shaw, and Rowe did not know the targeted species for trips. They modeled all sites and species simultaneously, and used an average individual from each county for each mode and species. They mention that their values for anadromous salmonid fishing are probably biased downward because salmon anglers are generally considered willing to pay more than are anglers for other species. Their model accounts for substitution across sites. The values reported by Morey, Shaw, and Rowe (1991) are the amounts people from various counties would pay to be able to go fishing in

Clatsop County throughout the season. The reported values are not marginal values for fish.

Olsen, Richards, and Scott (1991) used a contingent valuation survey to estimate marginal values for steelhead. They asked a random sample of people to state their willingness-to-pay in either higher electric bills or license fees for doubling the size of salmon and steelhead runs in the Columbia River basin by the year 2000. Their estimated marginal sport value for steelhead in Washington and Oregon coastal rivers is $24.96, and $54.84 in the Columbia River basin, in 1989 dollars. Nonuse and use values are included in the measure because of the way the contingent valuation survey question was worded. Olsen, Richards, and Scott (1991) did not specifically account for substitution across sites. Loomis (1989) estimated marginal values for steelhead in 21 rivers in Oregon by using a multiple site pooled travel cost model. Values for an individual steelhead range from $18 on the Coos River to $333 on the Willamette. Loomis used 1977 Oregon Angler Survey data and 1984 vehicle cost data. His point is that the values for site characteristics, such as fish, are site specific and can differ considerably from one site to another. Substitute sites are included as an explanatory variable in his travel cost model. Johnson and Adams (1988) used contingent valuation to estimate the marginal value of a recreationally caught steelhead on the John Day River. They asked anglers what they would be willing to pay, in the form of a steelhead stamp, for increased runs. Johnson and Adams report a value of $6.65 per additional steelhead caught, in 1986 dollars. The goal of the study was to estimate the benefits of increased streamflow in the John Day River.

The value reported in this study is the value of an additional steelhead caught by anglers in each region. The estimate does not include the value of a steelhead to other people who either buy licenses and do not fish for steelhead, or people who did not buy a fishing

license at all. To get use plus nonuse values, a contingent valuation study could be used with a representative sample of all users and nonusers. Many questions must be addressed before implementing such a survey. They include problems of scale, i.e., the size of the area for which an estimate is to be obtained. In addition, who to survey, and how much to discount their values must be decided. Should values be discounted by how far away the people live from the site? A nonuse value does not necessarily depend on distance. Should different groups have different discount values? Are the weights taken care of by assuming different utility functions? Several studies address some of these issues (i.e., Olsen, Richards, and Scott 1991; Gum and Martin 1975). Many of the questions are still being debated. While the search for improved methods of estimation continues, we must also concentrate on appropriate interpretation of the estimates in studies of resource allocation. The model presented in this study provides an estimate of recreational use value of one forest resource among many. The model also is useful for assessing incomplete recreational data.

5.2 Applying Valuation to Resource Allocation Decisions: Problems and Research Needs

If we are to concentrate on interpreting nonmarket estimates of resource values in studies of resource allocation, we need to better understand the interactive effects of decisions about resource extraction. The specific targets of research should center on those aspects about which the least is known, or where variation is most problematic. We know generally what fish are worth to recreational uses, and have ideas of what they are worth to nonusers. Numerous studies show the effects of land management activities on water and fish habitat. What analysts lack is reliable quantitative information about the long-term effects of forest and agricultural practices on plant and animal communities. To understand the consequences of management

alternatives, habitat models that use numbers of affected animals (such as steelhead, elk, and frogs) as the dependent variable are necessary (figure 1). Those habitat models must be used interactively with models of environmental quality as affected by management practices. Existing habitat models can be adapted, or new ones developed, for use in population effects modeling. This study used the habitat condition index (HCI) and the landslide sediment delivery index (LSDI) to trace the effects of the Forest Plan and an alternative that represents Option 9 in FEMAT (1993) on the production of steelhead in Elk River tributaries in southwestern Oregon (figure 3).

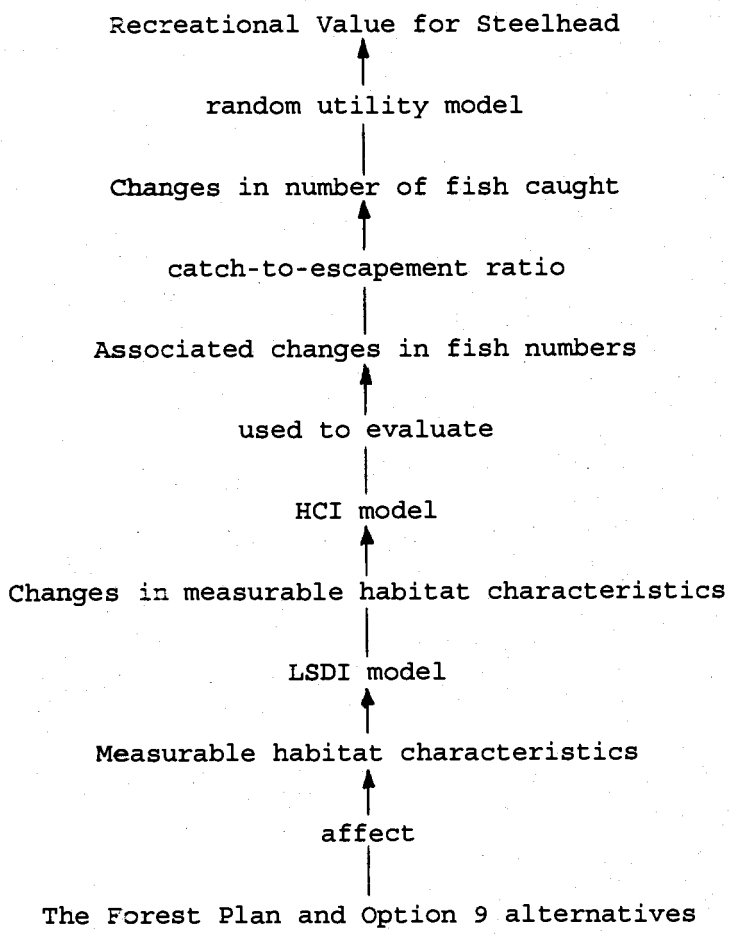

Figure 3. Land management effects on fish populations were quantified using the HCI and the LSDI for the Elk River in southwest Oregon. The difference in fish production was valued and used to compare the Forest Plan and Option 9 (FEMAT 1993).

Land management alternatives affect measurable habitat characteristics, in this case measured by sediment loads. The relation between sediment loads and fish population was modeled through the interaction of the LSDI and the HCI. The HCI was converted to fish numbers produced in Elk River tributaries, and the number of fish produced that would be caught were calculated using a catch-to-escapement ratio of 0.6 to 1. The focus of this study was the development of a random utility model that could be used with incomplete trip data to derive values for recreational goods. The random utility model was used to value additional fish caught. The recreational value of steelhead caught in the Elk River was used to compare the Forest Plan and an alternative that represents Option 9 (FEMAT 1993). The weakest point in the linkage from management activities to catchable fish is the modeled changes in fish populations associated with changes in sediment loads.

This study linked three models to measure and value the change in fish population between management alternatives; a random utility model to determine recreational use value for steelhead; the LSDI, a sediment model; and the HCI, a habitat model. The results of the random utility analysis are consistent with other studies of the recreational value of anadromous fish. The landslide sediment delivery index is based on several decades of data, and has had some sensitivity analysis performed. The significant limiting factor in the accuracy of assessing the value of additional fish produced under alternative B as compared to alternative A is the quality of the biophysical production function. The habitat condition index, based on two years of data or one life cycle of a steelhead, has been criticized for its assumptions about the capacity of all tributaries to reach unrealistically high quality. The conversion of the index to fish numbers was a simple algebraic transformation. The projection of steelhead population increases is

very likely overstated. The trend in steelhead populations will presumably increase as Option 9 from FEMAT is implemented.

The HCI-LSDI interaction is not a complete biophysical production function for steelhead because it does not account for all the other habitats the fish inhabit through their life cycle. This study makes the *ceteris paribus* assumption that only habitat in the tributaries of the Elk changes, and all else remains constant over the decade, i.e., whatever factors affect steelhead production outside the study area continue to affect the population in the same way. The HCI-LSDI interaction is viewed as a proxy for a fish yield function under different land management alternatives.

Fausch, Hawkes, and Parsons (1988) and Levins (1966) discussed realism, precision, and generality in models; Levins believes that biological models can attain, at best, two of the three. Generality is how well the model can be generalized to other locations and other situations. In statistics, accuracy is used instead of realism. Accuracy is defined as the nearness of a measurement to the actual variable being measured. Precision refers to the closeness to each other of repeated measurements of the variable being measured (Zar 1974).

Predictions from the landslide sediment delivery index model are not very precise (Luce and Ricks, unpublished), but the model has been subject to some validation work. Predictions from the model have been shown to be relatively accurate when compared to actual data from the 1970's and before. The model is specific to the Elk River watershed, and so cannot be generalized to other locations.

The habitat condition index (Chen 1992) is based on two years of steelhead count data, and is biased upward; it is not very accurate. In this study, however, the difference between two measurements of fish population are used. Precision is more important than accuracy, as long as the bias is consistent. The precision of the HCI is unknown because

it is not known how the population of steelhead for the late 1980s compares to historical averages. The difference in fish production between the two alternatives should be taken as an accurate, but not very precise, estimate until shown otherwise. The HCI is specific to the Elk River watershed, and so is not a general model.

The random utility model is a general model that can use data on catch rates and trip costs from any combination of alternative recreation areas. The precision and accuracy of the model depend heavily on the cost data used to develop estimates of value. The cost data in the Oregon Angler Survey is self-reported; it could be assumed that if the same people were asked the same questions about their trip costs, they would report similar costs. In that sense, the predictions of value would be precise. Accuracy of the predictions is subject to interpretation, as the "true" value of nonmarket goods is constantly being sought. Current literature in behavioral models (for example, see Morey, Shaw, and Rowe 1991; Loomis 1989) includes distance costs and often time costs in the trip cost estimate. People are not likely to include all of their milage and time costs when they report how much a fishing trip costs. For this data set, the estimates of value are most likely lower than if time and milage costs were consistently included.

The estimate of steelhead value in this study may be accurate for the regions as they are defined. It is important to note, however, that the estimates of value are not necessarily accurate when used to value the fish in one small area in that larger region. The southwest management zone includes rivers other than the Elk popular to steelhead and salmon anglers, including the Smith River, the Umpqua, the Coos, the Coquille, and the Rogue. Loomis (1989) has made a point of the variation in site-specific values of fish and recreation. This study had shown how managers could use the most recent data on angler behavior and on the effects of management on fish populations to evaluate management activities on watersheds. The answers are useful in that

they give information about trends and general values, and point out weak spots in such an analysis. Additional research would be most useful in obtaining better angler cost data, site-specific destination information, and better estimates of the effects of land management on fish populations. Botkin and others (1994) said that models currently used to set harvest quotas are not adequate for realistic or accurate predictions of population trends, nor for estimating the effects of human action on fish abundance. They recommend that a set of realistic, pragmatic models be developed. Such models are essential for the conservation and management of salmon.

Other studies have attempted to develop vertically integrated demand and supply models for timber versus other resources (e.g., Loomis 1988; Fight, Garrett, and Weyermann 1990). They have had similar problems predicting production of nontimber resources under different land management alternatives. Loomis (1988) converted a fish habitat index developed by Heller, Maxwell, and Parsons (1983) for the Siuslaw National Forest to smolt production using smolt counts developed by ODFW. The conversion from an index to fish numbers used the best information available, but it was still an approximation. The fish production submodel in the Southeast Alaska Multiresource Model (SAMM) (Fight, Garrett, and Weyermann 1990) accounted for habitat, population density, ocean conditions, predation, and recreational and commercial catch. Some parts of the model were based on published literature, some on unpublished data, and some parts were the result of professional opinion. As quantitative information on fish populations accumulates, our understanding of the effects of specific habitat changes on fish numbers will improve. With a better understanding of the effects of land management on nontimber resources, resource allocation decisions can be made with a clearer idea of the consequences of a given decision.

The FEMAT (1993) process has replaced forest planning based on the National Forest Management Act of 1976 with a new process. Option 9 in

FEMAT (1993) was chosen from a set of options developed by a team of scientists for managing west-side forests. Option 9 is a matrix-reserve system. It sets aside areas for the northern spotted owl, as required by the Endangered Species Act (ESA). Riparian zones were added to provide connectivity and to protect key watersheds and salmonids. Some timber harvest is included because community stability and effects on rural economies were taken into consideration. As part of the implementation of Option 9, Forest Service Districts are supplied with suggested guidelines for watershed management. A District may find, for example, that more of the timber in a given watershed can be harvested than the guidelines suggest. Reasons might include stable soils, favorable aspects, and low topography. Option 9 also calls for restoring watersheds that have been degraded. Watershed restoration analysis includes evaluating the benefits derived from restoration. Part of the value of watershed restoration might be an increase in fish populations. This study assessed the value of fish produced in an area that will be restored. Another part of the value of restoration will be the value of an increase in site quality.

As Option 9 is implemented on Districts, marginal analysis could be useful for evaluating the benefits of restoration and as a basis for departures from the suggested guidelines. The random utility model presented in this study is a useful tool for deriving use values for recreational goods. The model can use data with incomplete trip information and can derive regional values for recreational goods. Using a recreational use value for steelhead, however, to compare management alternatives tells only part of the story. If economic value were a decision criterion for choosing among land management alternatives, it would need to include use and nonuse value for all the resources being compared. This study used recreational values to put a value on harvestable surpluses of fish produced in a watershed. Conservation strategies were developed in the FEMAT process for

endangered species, which has to do with nonuse values and extinction issues. Comparing forest management alternatives in terms of the recreational values they produce is useful, but the scope of such an analysis must be kept in mind.

The model presented in this study has a direct application to recreation management decisions. Decisions about recreation management are aided by studies that assign values to the goods and activities being managed. Examples might include decisions about where to locate a boat ramp, or where to concentrate funding for site enhancement. Random utility modeling results in use values for sites and site characteristics.

Forest management decisions on public lands are driven by legal mandate. The legislation under which public policy makers operate includes requirements to consider economic impacts, but there are many other goals that are often in conflict with each other and with economic efficiency criteria. Much of the struggle over public land management has been the result of conflicting interpretations of the National Environmental Policy Act of 1969, the National Forest Management Act of 1976, and the Endangered Species Act of 1973.

Some form of land management will occur in public forests; the debate is about what society's goals are in terms of effects on all resources. We need to understand the implications of our actions, or lack thereof, on all the resources of the forest. Steelhead are not the only nontimber resource affected by forest management; others include plants and fungi used for food, medicine, cosmetics, and the floral market. Forests produce both nongame and game animals and birds that provide recreational benefits in hunting and wildlife viewing. Amphibians, reptiles, and fish species other than steelhead use all habitats and seral stages of forests, in highly varied ways. Assessing one good versus another good produced in forests gives one part of the picture, but not all. As we develop more production functions for goods

produced in forests, and more work is done on pricing nontimber resources, a more complete picture of the tradeoffs will emerge. Perhaps a better understanding of the scope of these tradeoffs will help people better balance conflicting demands for forest products.

Literature Cited

Abramowitz, M., and I.A. Stegun. 1966. Handbook of Mathematical Functions with Formulas, Graphs and Mathematical Tables. U.S. Department of Commerce, U.S. Government Printing Office, Washington D.C. 1046 p.

Adamowicz, W.L., J.J. Fletcher, and T. Graham-Tomasi. 1989. Functional form and the statistical properties of welfare measures. American Journal of Agricultural Economics. 71(2):414-421.

Allen, K.R. 1951. The Horokiwi stream: a study of trout population. Fisheries Bulletin 10. Wellington, New Zealand: New Zealand Marine Department. 237 p.

Balon, E.K. 1975. Reproductive guilds of fishes: a proposal and definition. Journal of the Fisheries Research Board of Canada. 32:821-864.

Ben-Akiva, M.E., and S.R. Lerman. 1985. Discrete Choice Analysis: Theory and Application to Travel Demand. MIT Press, Cambridge, Mass. 390 p.

Berkman, H.E. and C.F. Rabeni. 1987. Effect of siltation on stream fish communities. Environmental Biology of Fishes. 18(4): 285-294.

Bishop, R.C., and T.A. Heberlein. 1980. Simulated markets, hypothetical markets, and travel cost analysis: Alternative methods of estimating outdoor recreation demand. Department of Agricultural Economics Staff Paper 87, University of Wisconsin, Madison.

Bockstael, N.E., W.M. Hanemann, and C.L. Kling. 1987. Modeling recreational demand in a multiple site framework. Water Resources Research. 23:951-960.

Bockstael, N.E., K.E. McConnell and I.E. Strand. 1989. A random utility model for sportfishing: Some preliminary results from Florida. Marine Resource Economics. 6:245-260.

Bockstael, N., K.E. McConnell, and I. Strand. 1991. Recreation. pp. 227-270 in Measuring the Demand for Environmental Quality, John B. Braden and Charles D. Kolstad, eds. Elsevier Science Publishing Co. Inc. New York. 370 p.

Bockstael, N.E., and I.E. Strand, Jr. 1987. The effect of common sources of regression error on benefit estimates. Land Economics. 63(1):11-20.

Botkin, D., K. Cummins, T. Dunne, H. Regier, M. Sobel, and L. Talbot. 1994. Status and future of salmon of western Oregon and northern California: findings and options. Draft. The Center for the Study of the Environment, Santa Barbara, California. 265 p.

Brown, G., Jr., and R. Mendelsohn. 1984. The hedonic travel cost method. Review of Economics and Statistics. 66:427-433.

Burt, O., and D. Brewer. 1971. Evaluation of net social benefits from outdoor recreation. Econometrica. 39:813-827.

Caulkins, P.P., R.C. Bishop, and N.W. Bouwes, Sr. 1986. The travel cost model for lake recreation: A comparison of two methods for

incorporating site quality and substitution effects. American Journal of Agricultural Economics. May 1986, p. 291-297.

Cesario, F., and J. Knetsch. 1970. Time bias in recreation benefit estimates. Water Resources Research. 6(3):700-704.

Chamberlin, T.W. 1982. Influence of forest and rangeland management on anadromous fish habitat in North America. 30 p. In: Influence of Forest and Rangeland Management on Anadromous Fish Habitat in Western North America. Meehan, William R., tech. ed. USDA Forest Service GTR PNW-136.

Chapman, D.W. 1962. Effects of logging upon fish resources of the West Coast. Journal of Forestry. 60(8):533-537.

Chapman, D.W., and Eric Knudsen. 1980. Channelization and livestock impacts on salmonid habitat and biomass in western Washington. Transactions of the American Fisheries Society. 109:357-363.

Chen, G.K. 1992. Use of basin survey data in habitat modeling and cumulative watershed effects analysis. FHR Currents. R-5 Fish Habitat Relationship Technical Bulletin 8. June 1992. USDA Forest Service, Pacific Southwest Region. 11 p.

Cicchetti, C., J. Fisher, and K. Smith. 1976. An economic evaluation of a generalized consumer surplus measure: The Mineral King controversy. Econometrica. 44:1259-1276.

Cline, R., G. Cole, W. Megahan, R. Patten, and J. Potyondy. 1981. Guide for predicting sediment yields from forested watersheds. Forest Service, Northern and Intermountain Regions, USDA, Ogden, Utah. 48 p. and appendices.

Cordone, A.J., and D.W. Kelly. 1961. The influences of organic sediment on the aquatic life in streams. California Fish and Game. 47:189-28.

Cummings, R.G., D.S. Brookshire, and W.D. Schulze. 1986. Valuing environmental goods: a state of the arts assessment of the contingent valuation method. Rowland and Allanheld Publishers. Totawa, N.J.

Davis, R.K. 1963. The value of outdoor recreation: an economic study of the Maine woods. Ph.D. dissertation, Harvard University, Cambridge, Mass.

Department of Agricultural and Resource Economics, Oregon State University. 1993. Summary Statistics of the 1989 Oregon Angler Survey. Compiled by Kerri Haworth. Corvallis, Ore. 72 p.

Dwyer, J.F., J.R. Kelly, and M.D. Bowes. 1977. Improved procedure for valuation of the contribution of recreation to national recreation development. Research Report 128. Water Resources Center, University of Illinois.

Everest, F.H., R.L.Beschta, J.C. Scrivener, K.V. Koski, J.R. Sedell and J.C. Cederholm. 1987. Fine sediment and salmonid production: A paradox. p. 98- 142 In: Ernest O. Salo and Terrance W. Cundy, Eds. Streamside Management: Forestry and Fishery Interactions. Contribution 57. College of Forest Resources, University of Washington, Seattle. 471 p.

Fausch, K.D., C.L. Hawkes, and M.G. Parsons. 1988. Models that predict standing crop of stream fish from habitat variables: 1950-85. Gen. Tech. Rep. PNW-GTR-213. Forest Service, Pacific Northwest Research Station. Portland, Ore. 52 p.

[FEMAT] Forest Ecosystem Management Assessment Team. 1993. Forest ecosystem management: an ecological, economic, and social assessment. U.S. Department of Agriculture; U.S. Department of the Interior [and others]. Portland, Ore. [Irregular pagination].

Fight, R.D., L.D. Garrett and D.L. Weyermann. 1990. SAMM: A prototype southeast Alaska multiresource model. General Technical Report PNW-GTR-255. USDA Forest Service, Pacific Northwest Research Station. Portland, Ore. 109 p.

Findley, R.W. and D.A. Farber. 1983. Environmental Law in a Nutshell. West Publishing Co., St. Paul, Minn. 343 p.

Freeman, A.M. III. 1979. The benefits of environmental improvements: Theory and practice. Resources for the Future. Washington DC. 272 p.

Frissell, C.A. 1990. Water quality, fisheries, and aquatic biodiversity under two alternative forest management scenarios for the west-side federal lands of Washington, Oregon, and northern California. A report prepared for The Wilderness Society. Review draft 11/20/90. [Copy on file at Pacific Northwest Research Lab, Corvallis, Ore.]

Greene, W.H. 1990. Econometric Analysis. Macmillan Publishing Co. New York. 783 p.

Greene, W.H. 1992. LIMDEP Econometric Software. Econometric Software Inc., 43 Maple Ave., Bellport, N.Y.

Grobey, J.H. 1985. Politics versus bioeconomics: salmon fishery and forestry values in conflict. p. 169-200, In Forestlands Public and Private, ed. by Robert T. Deacon and M. Bruce Johnson. Ballinger Publishing Co., Cambridge, Mass. 332 pages.

Gum, R.L., and W.E. Martin. 1975. Problems and solutions in estimating the demand for and value of rural outdoor recreation. American Journal of Agricultural Economics. 57(4):558-566.

Hanemann, M., and E. Morey. 1992. Separability, partial demand systems, and consumers surplus measures. Journal of Environmental Economics and Management. 22:241-258.

Hankin, D.G. 1984. Multistage sampling designs in fisheries research: applications in small streams. Canadian Journal of Fisheries and Aquatic Sciences. 41:1575-1591.

Hankin, D.G., and G.H. Reeves. 1988. Estimating total fish abundance and total habitat area in small streams based on visual estimation methods. Canadian Journal of Fisheries and Aquatic Sciences. 45:834-844.

Heller, D., J. Maxwell and M. Parsons. 1983. (Revised in 1990) Modelling the Effects of Forest Management on Salmonid Habitat. Siuslaw National Forest, U.S. Forest Service, Corvallis, OR.

Hellerstein, D. 1992. Estimating consumer surplus in the censored linear model. Land Economics. 68(1):83-92.

Hicks, J.R. 1943. The four consumer's surpluses. Review of Economic Studies. Winter:31-41.

Hicks, B.J., J.D. Hall, P.A. Bisson and J.R. Sedell. 1991. Responses of salmonids to habitat changes. p. 483- 518 In Influences of forest and Rangeland Management on Salmonid Fishes and Their Habitats, Ed. by William R. Meehan. American Fisheries Society Special Publication 19, Bethesda, Maryland. 751 p.

Hoekstra, T.W., and L.A. Joyce. 1988. An overview of the southern United States multiresource modeling. In: Perspectives on Land Modeling. R. Gelinas, D. Bond, and B. Smit, eds. Workshop proceedings, Toronto, Ontario. Nov. 17-20, 1986. Polyscience Publications, Inc. Montreal, Canada.

Hueth, D.L., E.J. Strong and R.D. Fight, tech. eds. 1988. Sport fishing: A comparison of three indirect methods for estimating benefits. Research Paper PNW-RP-395. USDA Forest Service, Pacific Northwest Research Station, Portland Or.

Huppert, D.D. 1989. Measuring the value of fish to anglers: Application to central California anadromous species. Marine Resource Economics, 6:89-107.

Huppert, D.D., and R.D. Fight. 1991. Economic considerations in managing salmonid habitat. In: Influences of Forest and Rangeland Management on Salmonid Fishes and Their Habitats. William R. Meehan, Ed. American Fisheries Society Special Publication 19. Bethesda, Maryland.

Johnson, N., and R. Adams. 1988. Benefits of increased streamflow: The case of the John Day River steelhead fishery. Water Resources Research. 24(11):1839-1846.

Joyce, L.A., C.H. Flather, P.A. Flebbe, T.W. Hoekstra and S.J. Ursic. 1990. Integrating forage, wildlife, water and fish projections with timber projections at the regional level: a case study in the southern United States. Environmental Management. 14(4): 489-500.

Kling, C.L. 1986. Measuring the recreational benefits of environmental amenities using multiple site models: An evaluation of techniques. Ph.D. dissertation, University of Maryland, College Park.

Kling, C.L. 1988. Comparing welfare estimates of environmental quality changes from recreation demand models. Journal of Environmental Economics and Management. 15:331-340.

Koski, K.V., J. Heifetz, S. Johnson, M. Murphy and J. Thedinga. 1984. Evaluation of buffer strips for protection of salmonid habitat and implications for enhancement. p. 138-155. In Proceedings: Pacific Northwest Stream Habitat Management Workshop, T.J. Hassler, ed. Humboldt State University, Arcata, California.

Kreps, D.M. 1990. A course in microeconomic theory. Princeton University Press, Princeton, New Jersey. 850 p.

Kunkel, C., and P. Janik. 1976. An economic evaluation of salmonid fisheries attributable to Siuslaw National Forest. Siuslaw National Forest, USDA Forest Service, Corvallis OR.

Langlois, T.H. 1941. Two processes operating for the reduction in abundance or elimination of fish species from certain types of water

areas. Transactions of the North American Wildlife Conference. 6:189-201.

Leathe, S.A., and M.D. Enk. 1985. Cumulative effects of micro-hydro development on fisheries of the Swan River drainage, Montana. Vol. 1: Summary Report. U.S. Department of Energy, Bonneville Power Administration; final report; project 82-19. Portland, Ore. 114 p. and appendices.

Levins, R. 1966. The strategy of model building in population biology. American Scientist. 54(4): 421-431.

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

Loomis, J.B. 1989. Estimation of and variation in site specific marginal values for recreational fisheries. Journal of Environmental Management. 29:183-191.

Luce, C.H., and C.L. Ricks. [unpublished.] Estimating volumes of landslide sediment production and delivery from forested watersheds. Manuscript in preparation.

Marshall, A. 1930. Principles of Economics. Royal Economic Society, London. 8th ed. 871 p.

McKernan, D.L., D.R. Johnson, and J.I. Hodges. 1950. Some factors influencing the trends of salmon populations in Oregon. Transactions of the North American Wildlife Conference. 15:427-49.

McNeil, W.J. 1966. Effect of the spawning bed environment on reproduction of pink and chum salmon. U.S. Fish and Wildlife Service Fish Bulletin. 65(2):495-523.

Miller, R.R., J.D. Williams and J.E. Williams. 1989. Extinctions of North American fishes during the past century. Fisheries. 14(6):22-38.

Morey, E.R., W.D. Shaw, and R.D. Rowe. 1991. A discrete-choice model of recreation participation, site choice, and activity valuation when complete trip data are not available. Journal of Environmental Economics and Management. 20:181-201.

Muncy, R.J., G.J. Atchinson, R.V. Bulkley, B.W. Menzel, L.G. Perry, and R.C. Summerfelt. 1979. Effects of suspended solids and sediment on reproduction and early life of warm-water fishes: a review. EPA-600/3-79-0. Corvallis Environmental Research Laboratory, Corvallis, Oregon. 100 p.

[ODFW] Oregon Department of Fish and Wildlife. 1991a. Oregon Angler Survey and Economic Study, Final Report. Prepared by the Research Group, Corvallis, Oregon, for the Oregon Department of Fish and Wildlife. 41 p.

[ODFW] Oregon Department of Fish and Wildlife. 1991b. Oregon Angler Survey and Economic Study, Technical Supplement. Prepared by the Research Group, Corvallis, Oregon, for the Oregon Department of Fish and Wildlife. [irregular pagination]

Olsen, D., J. Richards, and R. Scott. 1991. Existence and sports values for doubling the size of the Columbia River basin salmon and steelhead runs. River. 2(1):44-56.

Parsons, G.R., and M.J. Kealy. 1992. Randomly drawn opportunity sets in a random utility model of lake recreation. Land Economics. 68(1):93-106.

Randall, A., B.C. Ives, and C. Eastman. 1974. Bidding games for valuation of aesthetic environmental improvements. Journal of Environmental Economics and Management. 1:132-149.

Randall A., and J.R. Stoll. 1980. Consumer's surplus in commodity space. American Economic Review. 70(June): 449-455.

Reeves, G.H., F.H. Everest, and J.R. Sedell. 1993. Diversity of juvenile anadromous salmonid communities in basins in coastal Oregon, USA, with different rates of timber harvest. Transactions of the American Fisheries Society. 122:

Regier, H.A., R.L. Welcomme, R.J. Steedman, and H.F. Henderson. 1990. Rehabilitation of degraded river systems. p. 86-97 In Proceedings of the International Large River Symposium, D.P. Dodge, ed. Canadian Special Publications in Fisheries and Aquatic Sciences 106.

Ricks, C., and G. Chen. 1990. August 1990 Progress Report, Elk River Drainage Basin, Analysis of Cumulative Watershed Effects. USDA Forest Service, Powers Ranger District, Siskiyou National Forest. Gold Beach, OR.

SAS Institute Inc, 1988. SAS/STAT User's Guide, release 6.03 edition. SAS Institute Inc. Cary, NC. 1028 p.

Seller, C., J.R. Stoll and J-P Chavas. 1985. Validation of empirical measures of welfare change: A comparison of nonmarket techniques. Land Economics. 61(2):156-175.

Small, K.A., and H.S. Rosen. 1981. Applied welfare economics with discrete choice models. Econometrica. 49(1):105-130.

Smith, P.W. 1971. Illinois streams: a classification based on their fishes and an analysis of factors responsible for disappearance of native species. Illinois Natural History Survey Biological Notes 76. 14 p.

Smith, V.K., and Y. Kaoru. 1987. The hedonic travel cost model: a view from the trenches. Land Economics. 63:179-192.

Smith, V.K. 1991. Household production functions and environmental benefit estimation. p. 41-120 In Measuring the demand for environmental quality, John B. Braden and Charles D. Kolstad, eds. Elsevier Science Publishing Co. Inc. New York. 370 p.

Sorg, C.F., and J.B. Loomis. 1986. Economic value of Idaho sport fisheries with an update on valuation techniques. North American Journal of Fisheries Management. 6:494-503.

Sorg, C.F., D.M. Donnelly, J.B. Loomis and G.L. Peterson. 1984. Implications of economics as applied to wild trout fisheries management in Idaho. Paper presented at the Wild Trout III Symposium, Mammoth, Wyoming. Yellowstone National Park, Sept. 24-25.

Stowell, R., A. Espinosa, T.C. Bjornn, W.S. Platts, D.C. Burns and J.S. Irving. 1983. Guide for predicting salmonid response to sediment yields in Idaho batholith watersheds. USDA Forest Service, Northern Region, Missoula Montana, and Intermountain Region, Ogden, Utah.

Sullivan, K., T.E. Lisle, C.A. Dollof, G.E. Grant and L.M. Reid. 1987. Stream channels: The link between forests and fishes. p. 39-97. In: Ernest O. Salo and Terrance W. Cundy, eds. Streamside Management: Forestry and Fishery Interactions. Contribution 57. College of Forest Resources, University of Washington, Seattle, Wash. 471 p.

Swanson, F.J., L.E. Benda, S.H. Duncan, G.E. Grant, W.F. Megahan, L.M. Reid and R.R. Ziemer. 1987. Mass failures and other processes of sediment production in Pacific Northwest forest landscapes. p. 9-38. In: Streamside Management: Forestry and Fishery Interactions. Ernest O Salo, and Terrance W. Cundy, Eds. Contribution 57. College of Forest Resources, University of Washington, Seattle, Wash. 471 p.

Thayer, M.A. 1981. Contingent valuation techniques for assessing environmental impacts: Further evidence. Journal of Environmental Economics and Management. 8:27-44.

USDA Forest Service. 1990. Resource pricing and valuation procedures for the recommended 1990 RPA program. Washington, DC: Resources Program and Assessment Staff. 33 p.

USDA Forest Service. 1992. Draft Environmental Impact Statement, Elk Wild and Scenic River Management Plan. Siskiyou National Forest, State Parks and Recreation Dept. Grants Pass, Ore.

Walsh, R.B., D. Johnson, and J. Mckean. 1990. Review of outdoor recreation demand studies with non-market benefit estimates, 1968-1988. In Advances in Applied Microeconomics: Vol. 5: Recent Developments in Modeling of Technical Change; and Modeling Demand for and Valuation of Recreation Resources. Link and Smith, eds. JAI Press, Greenwich, CT. Cited in Morey, Shaw, and Rowe 1991.

Williams, J.E., J.E. Johnson, D.A. Hendrickson, S. Contreras-Balderas, J.D. Williams, M. Navarro-Mendoza, D.E. McAllister and J.E. Deacon. 1989. Fishes of North America endangered, threatened, or of special concern: 1989. Fisheries. 14(6):2-20.

Willig, R.D. 1976. Consumer's surplus without apology. American Economic Review. 66:589-597.

Zar, J.H. 1974. Biostatistical analysis. Prentice-Hall, Inc. Englewood Cliffs, N.J. 620 p.

Appendices

Appendix A Data Manipulation Programming

A1. Basic data extraction by using SAS

This section contains the programming in SAS to extract data for cost and number of trips by species and water type, for one region.

```
LIBNAME C 'C:\DATA';

DATA ADDR; SET C.NDATASET;
KEEP LICENSE AGE SEX;
PROC SORT; BY LICENSE;

DATA MAIL; SET C.MAIL3;
KEEP LICENSE BUY1-BUY8 PRIMSIT1-PRIMSIT8 WATCOD1-WATCOD8
     NUMTRGT1-NUMTRGT8 TRGSPEC1-TRGSPEC8;
PROC SORT; BY LICENSE;


DATA TRIP1; SET MAIL;
DROP PRIMSIT2-PRIMSIT8 TRGSPEC2-TRGSPEC8 WATCOD2-WATCOD8
     BUY2-BUY8 NUMTRGT2-NUMTRGT8;
IF PRIMSIT1 = 2; RUN;

DATA TRIP2; SET MAIL;
DROP PRIMSIT1 PRIMSIT3-PRIMSIT8 TRGSPEC1 TRGSPEC3-TRGSPEC8
     WATCOD1 WATCOD3-WATCOD8 BUY1 BUY3-BUY8
     NUMTRGT1 NUMTRGT3-NUMTRGT8;
IF PRIMSIT2 = 2; RUN;

DATA TRIP3; SET MAIL;
DROP PRIMSIT1-PRIMSIT2 PRIMSIT4-PRIMSIT8
    TRGSPEC1-TRGSPEC2 TRGSPEC4-TRGSPEC8
    WATCOD1-WATCOD2 WATCOD4-WATCOD8
    BUY1-BUY2 BUY4-BUY8
    NUMTRGT1-NUMTRGT2 NUMTRGT4-NUMTRGT8;
IF PRIMSIT3 = 2; RUN;

DATA TRIP4; SET MAIL;
DROP PRIMSIT1-PRIMSIT3 PRIMSIT5-PRIMSIT8
    TRGSPEC1-TRGSPEC3 TRGSPEC5-TRGSPEC8
    WATCOD1-WATCOD3 WATCOD5-WATCOD8
    BUY1-BUY3 BUY5-BUY8
    NUMTRGT1-NUMTRGT3 NUMTRGT5-NUMTRGT8;
IF PRIMSIT4 = 2; RUN;

DATA TRIP5; SET MAIL;
DROP PRIMSIT1-PRIMSIT4 PRIMSIT6-PRIMSIT8
    TRGSPEC1-TRGSPEC4 TRGSPEC6-TRGSPEC8
    WATCOD1-WATCOD4 WATCOD6-WATCOD8
    BUY1-BUY4 BUY6-BUY8
    NUMTRGT1-NUMTRGT4 NUMTRGT6-NUMTRGT8;
IF PRIMSIT5 = 2; RUN;

DATA TRIP6; SET MAIL;
DROP PRIMSIT1-PRIMSIT5 PRIMSIT7-PRIMSIT8
    TRGSPEC1-TRGSPEC5 TRGSPEC7-TRGSPEC8
    WATCOD1-WATCOD5 WATCOD7-WATCOD8
```

```
    BUY1-BUY5 BUY7-BUY8
    NUMTRGT1-NUMTRGT5 NUMTRGT7-NUMTRGT8;
IF PRIMSIT6 = 2; RUN;

DATA TRIP7; SET MAIL;
DROP PRIMSIT1-PRIMSIT6 PRIMSIT8
     TRGSPEC1-TRGSPEC6 TRGSPEC8
    WATCOD1-WATCOD6 WATCOD8
    BUY1-BUY6 BUY8  NUMTRGT1-NUMTRGT6 NUMTRGT8;
IF PRIMSIT7 = 2; RUN;

DATA TRIP8; SET MAIL;
DROP PRIMSIT1-PRIMSIT7  TRGSPEC1-TRGSPEC7  WATCOD1-WATCOD7
     BUY1-BUY7  NUMTRGT1-NUMTRGT7;
IF PRIMSIT8 = 2; RUN;

DATA SWRIV; SET TRIP1 TRIP2 TRIP3 TRIP4 TRIP5 TRIP6 TRIP7 TRIP8;
IF WATCOD1 = 1 THEN CODE = 1;
IF WATCOD2 = 1 THEN CODE = 2;
IF WATCOD3 = 1 THEN CODE = 3;
IF WATCOD4 = 1 THEN CODE = 4;
IF WATCOD5 = 1 THEN CODE = 5;
IF WATCOD6 = 1 THEN CODE = 6;
IF WATCOD7 = 1 THEN CODE = 7;
IF WATCOD8 = 1 THEN CODE = 8;
IF CODE >.;

VIS = (SUM (OF PRIMSIT1-PRIMSIT8))/2;
TOTCOS = SUM (OF BUY1-BUY8);
DATA SWCST; SET SWRIV;
PROC SORT; BY LICENSE;
PROC MEANS SUM NOPRINT; VAR VIS TOTCOS;
BY LICENSE; OUTPUT OUT = TEMP1 SUM = NWRVIS TCOST;
RUN;
DATA TEMP11; SET TEMP1;
NWAVCST = TCOST/NWRVIS;
DATA TEMP3; SET TEMP11;
KEEP LICENSE NWAVCST NWRVIS;
PROC SORT; BY LICENSE;
RUN;

DATA SWSP; SET SWRIV;
IF TRGSPEC1 = 3 THEN FISH = 1;
IF TRGSPEC2 = 3 THEN FISH = 2;
IF TRGSPEC3 = 3 THEN FISH = 3;
IF TRGSPEC4 = 3 THEN FISH = 4;
IF TRGSPEC5 = 3 THEN FISH = 5;
IF TRGSPEC6 = 3 THEN FISH = 6;
IF TRGSPEC7 = 3 THEN FISH = 7;
IF TRGSPEC8 = 3 THEN FISH = 8;
IF FISH >.;

VISI = (SUM (OF PRIMSIT1-PRIMSIT8))/2;
TOTCATCH = SUM (OF NUMTRGT1-NUMTRGT8);
DATA SWCAT; SET SWSP;
PROC SORT; BY LICENSE;
PROC MEANS SUM NOPRINT; VAR VISI TOTCATCH;
BY LICENSE; OUTPUT OUT = TEMP1B SUM = VISITS TCATCH;
RUN;

DATA TEMP11B; SET TEMP1B;
NWAVCTST = TCATCH/VISITS;
```

```
PROC MEANS SUM; VAR VISITS; RUN;
DATA TEMP4; SET TEMP11B;
PROC SORT; BY LICENSE;
RUN;

DATA ALLOT; MERGE TEMP3 TEMP4 ADDR;
BY LICENSE;
IF NWAVCTST >.;
DATA C.NWRST; SET ALLOT;
RUN;
```

A2. Combining data for two zones

This section outlines the programming in SAS to combine and transform data for two zones, in this case, the central and the northeast zones.

```
LIBNAME C 'C:\DATA';
DATA ONE; SET C.CENTRAL;
PROC SORT; BY LICENSE;
DATA TWO; SET C.NEAST;
PROC SORT; BY LICENSE;
DATA TRE; MERGE ONE TWO;
BY LICENSE;
IF LVIS = . THEN LVIS = 0;
IF LCST = . THEN LCST = 0;
IF VISITS = . THEN VISITS = 0;
IF TCATCH = . THEN TCATCH = 0;
IF LCTST = . THEN LCTST = 0;
IF EVIS = . THEN EVIS = 0;
IF ECST = . THEN ECST = 0;
IF EVISITS = . THEN EVISITS = 0;
IF ETCATCH = . THEN ETCATCH = 0;
IF ECTST = . THEN ECTST = 0;
DATA BUNCH; SET TRE;
LEVIS = LVIS + EVIS;
LTCOST = LCST * LVIS;
ETCOST = ECST * EVIS;
LETCOST = LTCOST + ETCOST;
LECST = LETCOST/LEVIS;
FCATCH = TCATCH + ETCATCH;
FVIS = VISITS + EVISITS;
LECTST = FCATCH/FVIS;
IF SEX = . THEN SEX = 1;
IF AGE = . THEN AGE = 52;
DATA CNE; SET BUNCH;
KEEP LEVIS LECST LECTST AGE SEX LICENSE;
PROC SORT; BY LICENSE;
RUN;
PROC PRINT;
RUN;
DATA ZOO; SET CNE;
LECTST = LECTST + 1;
PROC PRINT;
RUN;
DATA C.CLNE; SET ZOO;
RUN;
```

Appendix B Likelihood Function and Consumer's Surplus Estimation

B1 Estimation of the likelihood function, by using LIMDEP

This section is the programming in LIMDEP used to estimate likelihood fuction values for Oregon, for the initial state. It includes some data transformations, and the likelihood function as programmed.

```
OPEN; OUTPUT = C:\DATA\STATOc.OUT $
READ; NOBS = 475; NVAR = 15;
      NAMES = LICENSE, swrvis, swavcst, swavctst, SEX, AGE, wrvis,
wavcst, wavctst, levis, lecst, lectst, cnwrvis, cnwavcst, cnwctst;
      FILE = C:\DATA\STATo.DAT $

CREATE; KSS = SWRVIS + WRVIS + CNWRVIS + LEVIS
; LFSWRVIS = LGM(SWRVIS + 1)
; LFWRVIS = LGM(WRVIS + 1)
; LFCNWVIS = LGM(CNWRVIS + 1)
; LFLEVIS = LGM(LEVIS + 1)
; KSF = LGM(KSS+1)
; SWRSC = SWAVCST/100
; WRSC = WAVCST/100
; CNWSC = CNWAVCST/100
; LESC = LECST/100
; IF (SEX = 2) SEX = 0; (ELSE) SEX = 1
; LVIS = LFSWRVIS + LFWRVIS + LFCNWVIS + LFLEVIS
; TKS = 8-KSS
; WMSWC = WRSC - SWRSC
; NWMSWC = CNWSC - SWRSC
; LMSWC = LESC - SWRSC
; WMSWCT = WAVCTST - SWAVCTST
; NWMSWCT = CNWCTST - SWAVCTST
; LMSWCT = LECTST - SWAVCTST
; SWMWC = SWRSC - WRSC
; NWMWC = CNWSC - WRSC
; LMWC = LESC - WRSC
; SWMWCT = SWAVCTST - WAVCTST
; NWMWCT = CNWCTST - WAVCTST
; LMWCT = LECTST - WAVCTST
; SWMCNWC = SWRSC - CNWSC
; WMCNWC = WRSC - CNWSC
; LMCC = LESC - CNWSC
; SWMCNWCT = SWAVCTST - CNWCTST
; WMCNWCT = WAVCTST - CNWCTST
; LMCCT = LECTST - CNWCTST
; SWMLC = SWRSC - LESC
; SWMLCT = SWAVCTST - LECTST
; WMLC = WRSC - LESC
; WMLCT = WAVCTST - LECTST
; CMLC = CNWSC - LESC
; CMLCT = CNWCTST - LECTST$

MINIMIZE; LABELS = α0, α1, B0, σ0, α2, α3, B1, σ1, α4, α5, B2, σ2, α6,
α7,
B3, σ3, B4, σ4, B5, σ5, B6, σ6, B7, σ7, B8, σ8, B9, σ9, B10, σ10, B11,
```

```
σ11, B12, σ12, B13, σ13, B14, σ14, B15, σ15;
START = -2, -2, 0.35, -1.2, -2.4, -2.4, 0.26, -1.2, -2.6, -2.6, 0.31,
-1.3, -2.1, -2.1, 0.21, -0.75, -0.12, 2, -0.28, 2, -0.27, 2, -0.24, 2,
-0.23, 2, -0.22, 2, -0.14, 1.6, -0.23, 1.6, -0.21, 1.6, -0.18, 0.99,
-0.2, 0.98, -0.18, 0.99;
TLF = .001;

FCN = TKS*(EXP(α0 + α1*SEX - B0*SWRSC + σ0*SWAVCTST) +
 EXP(α2 + α3*SEX - B1*WRSC + σ1*WAVCTST) +
 EXP(α4 + α5*SEX - B2*CNWSC + σ2*CNWCTST) +
 EXP(α6 + α7*SEX - B3*LESC + σ3*LECTST)) - KSF + LVIS +
SWRVIS* LOG(EXP(-B4*WMSWC + σ4*WMSWCT) + EXP(-B5*NWMSWC + σ5*NWMSWCT) +
EXP(-B6*LMSWC + σ6*LMSWCT)) +
WRVIS* LOG(EXP(-B7* SWMWC + σ7*SWMWCT) + EXP(-B8*NWMWC + σ8*NWMWCT) +
EXP(-B9*LMWC + σ9*LMWCT)) +
CNWRVIS* LOG(EXP(-B10*SWMCNWC + σ10*SWMCNWCT) +
EXP(-B11*WMCNWC + σ11*WMCNWCT) + EXP(-B12*LMCC + σ12*LMCCT)) +
LEVIS* LOG(EXP(-B13*SWMLC + σ13*SWMLCT) + EXP(-B14*WMLC + σ14*WMLCT) +
EXP(-B15*CMLC + σ15*CMLCT))$
```

B2 LIMDEP output for the likelihood function

This section contains the output from LIMDEP, which shows the coefficient and gradient estimates; these are statewide liklihood function results, for T=8, in the initial state.

```
       Reading file : C:\DATA\STATO.DAT
Current sample      :      475 observs.
Total variables now :  15
Missing values read :                 0
```

```
MODEL COMMAND:
MINIMIZE;LABELS=α0,α1,B0,σ0,α2,α3,B1,σ1,α4,α5,B2,σ2,α6,α7,B3
,σ3,B4,σ4,B5,σ5,B6,σ6,B7,σ7,B8,σ8,B9,σ9,B10,σ10,B11,σ11,B12,
σ12,B13,σ13,B14,σ14,B15,σ15;START=-2,-2,0.35,-1.2,-2.4,-2.4,
0.26,-1.2,-2.6,-2.6,0.31,-1.3,-2.1,-2.1,0.21,-0.75,-0.12,2,-
0.28,2,-0.27,2,-0.24,2,-0.23,2,-0.22,2,-0.14,1.6,-0.23,1.6,-
0.21,1.6,-0.18,0.99,-0.2,0.98,-0.18,0.99;TLF=.001;FCN=TKS*(E
XP(α0+α1*SEX-B0*SWRSC+σ0*SWAVCTST)+EXP(α2+α3*SEX-B1*WRSC+σ1*
WAVCTST)+EXP(α4+α5*SEX-B2*CNWSC+σ2*CNWCTST)+EXP(α6+α7*SEX-B3
*LESC+σ3*LECTST))-KSF+LVIS+SWRVIS*LOG(EXP(-B4*WMSWC+σ4*WMSWC
T)+EXP(-B5*NWMSWC+σ5*NWMSWCT)+EXP(-B6*LMSWC+σ6*LMSWCT))+WRVI
S*LOG(EXP(-B7*SWMWC+σ7*SWMWCT)+EXP(-B8*NWMWC+σ8*NWMWCT)+EXP(
-B9*LMWC+σ9*LMWCT))+CNWRVIS*LOG(EXP(-B10*SWMCNWC+σ10*SWMCNWC
T)+EXP(-B11*WMCNWC+σ11*WMCNWCT)+EXP(-B12*LMCC+σ12*LMCCT))+LE
VIS*LOG(EXP(-B13*SWMLC+σ13*SWMLCT)+EXP(-B14*WMLC+σ14*WMLCT)+
EXP(-B15*CMLC+σ15*CMLCT))$
```

```
**********************************************************************

                        Method=D/F/P ; Maximum iterations= 50
   Convergence criteria: Gradient=   0.1000000E-03
                         Function  =   0.1000000E-02
                         Parameters=   0.1000000E-03
   Starting values:   -2.000      -2.000      0.3500      -1.200
-2.400
                      -2.400      0.2600      -1.200      -2.600
-2.600
                      0.3100      -1.300      -2.100      -2.100
0.2100
                     -0.7500     -0.1200       2.000     -0.2800
2.000
                     -0.2700       2.000     -0.2400       2.000
-0.2300
                       2.000     -0.2200       2.000     -0.1400
1.600
                     -0.2300       1.600     -0.2100       1.600
-0.1800
                      0.9900     -0.2000      0.9800     -0.1800
0.9900
```

```
 Iteration:   6 Fn=   48.26945
 Param   -2.35     -2.14     0.385     -1.21     -2.63     -2.47
0.285
         -1.20     -2.78     -2.65     0.325     -1.30     -2.44
-2.22
         0.221    -0.754    -0.370      2.48    -0.472      2.44
-0.466
          2.44    -0.474      2.16    -0.452      2.20    -0.444
2.18
        -0.285      1.85    -0.425      1.82    -0.413      1.84
-0.357
          1.25    -0.377      1.24    -0.356      1.25
 Gradnt   53.7      21.0     -5.33      1.42      38.5      11.4
-4.13
         0.491      30.0      8.09     -2.48     0.783      52.4
18.4
         -1.63     0.663      44.2     -86.3      37.6     -85.0
38.5
         -84.9      44.0     -31.3      41.4     -37.9      41.8
-33.1
          27.7     -47.3      36.8     -41.7      37.6     -44.5
33.4
         -48.9      33.5     -49.6      33.2     -48.9
** Function has converged.

*************************************************************************

User defined minimization
Maximum Likelihood Estimates
Log-Likelihood..............   -48.26945
N[0,1] used for significance levels.
Variable  Coefficient  Std. Error   t-ratio Prob|t|≥x
-------------------------------------------------------------------------
α0        -2.3508        3.598       -0.653  0.51349
α1        -2.1435        4.346       -0.493  0.62186
B0        0.38497        2.214        0.174  0.86194
σ0        -1.2092        8.897       -0.136  0.89189
α2        -2.6325        7.352       -0.358  0.72030
α3        -2.4704        8.430       -0.293  0.76948
B1        0.28539        3.280        0.087  0.93067
σ1        -1.2030        18.72       -0.064  0.94875
α4        -2.7750        2.190       -1.267  0.20520
α5        -2.6481        9.693       -0.273  0.78470
B2        0.32457        2.100        0.155  0.87716
σ2        -1.3046        6.899       -0.189  0.85001
α6        -2.4375        4.931       -0.494  0.62105
α7        -2.2239        5.971       -0.372  0.70958
B3        0.22073        3.244        0.068  0.94575
σ3       -0.75441        10.04       -0.075  0.94009
B4       -0.36988       0.2182       -1.695  0.09002
σ4         2.4752       0.3837        6.451  0.00000
B5       -0.47177       0.4789       -0.985  0.32461
σ5         2.4387        9.845        0.248  0.80435
B6       -0.46647       0.6257       -0.746  0.45594
σ6         2.4397        10.06        0.242  0.80841
B7       -0.47429       0.2668       -1.778  0.07545
σ7         2.1643       0.2994        7.228  0.00000
B8       -0.45229       0.3924       -1.153  0.24903
σ8         2.2024       0.2736        8.048  0.00000
B9       -0.44373       0.4410       -1.006  0.31435
σ9         2.1757       0.4154        5.238  0.00000
B10      -0.28526       0.1138       -2.507  0.01218
```

σ10	1.8529	0.4743	3.906	0.00009
B11	-0.42460	1.328	-0.320	0.74919
σ11	1.8164	0.3394	5.353	0.00000
B12	-0.41267	1.325	-0.311	0.75551
σ12	1.8398	0.5966	3.084	0.00204
B13	-0.35654	21.76	-0.016	0.98693
σ13	1.2476	160.6	0.008	0.99380
B14	-0.37667	2.195	-0.172	0.86374
σ14	1.2442	1.064	1.169	0.24230
B15	-0.35572	20.89	-0.017	0.98641
σ15	1.2477	160.4	0.008	0.99379

End cmnd. entry from editor.

B3 Programming in SAS to calculate PPCV

This section contains the programming in SAS used to calculate values used in estimating PPCV for an additional steelhead in each region.

```
libname c 'c:\data';
data one; set c.incom;
proc sort; by license;
run;
data two; set c.swrstf;
keep license swrvis swavcst swavctst sex;
proc sort; by license;
run;
data tre; merge one two;
by license;
if swrvis > .;
if newinc = . then newinc = 30046;
if sex = 2 then sex = 0;
bud = newinc/48;
pnfi1 = exp(-2.3508 - 2.1435*sex - .0038497*swavcst - 1.2092*swavctst);
unot1 = 0;
unot2 = -2.6325 - 2.4704*sex + .0028539*bud;
unot3 = -2.7750 - 2.6481*sex + .0032457*bud;
unot4 = -2.4375 - 2.2239*sex + .0022073*bud;
budsw = bud - swavcst;
if budsw < 0 then budsw = 0;
ufish1 = .0038497*(budsw) - 1.2092*swavctst;
eufish1 = exp(ufish1);
run;

data four; set c.wrstf;
keep license wrvis wavcst wavctst sex;
proc sort; by license;
run;
data fiv; merge one four;
by license;
if wrvis > .;
if newinc = . then newinc = 30046;
if sex = 2 then sex = 0;
bud = newinc/48;
pnfi2 = exp(-2.6325 - 2.4704*sex - .0028539*wavcst - 1.2030*wavctst);
unot5 = -2.3508 - 2.1435*sex + .0038497*bud;
unot6 = 0;
unot7 = -2.7750 - 2.6481*sex + .0032457*bud;
unot8 = -2.4375 - 2.2239*sex + .0022073*bud;
budw = bud - wavcst;
if budw < 0 then budw = 0;
ufish2 = .0028539*(budw) - 1.2030*wavctst;
eufish2 = exp(ufish2);
run;

data six; set c.cnwst;
drop age;
proc sort; by license;
run;
data sev; merge one six;
by license;
if cnwrvis > .;
if newinc = . then newinc = 28456;
```

```
if sex = 2 then sex = 0;
bud = newinc/48;
pnfi3 = exp(-2.7750 - 2.6481*sex - .0032457*cnwavcst - 1.3046*cnwctst);
unot9 = -2.3508 - 2.1435*sex + .0038497*bud;
unot10 = -2.6325 - 2.4704*sex + .0028539*bud;
unot11 = 0;
unot12 = -2.4375 - 2.2239*sex + .0022073*bud;
budcnw = bud - cnwavcst;
if budcnw < 0 THEN BUDCNW = 0;
ufish3 = .0032457*(budcnw) - 1.3046*cnwctst;
eufish3 = exp(ufish3);
run;

data ate; set c.clne;
drop age;
lectst = lectst - 1;
proc sort; by license;
run;
data nin; merge one ate;
by license;
if levis > .;
if newinc = . then newinc = 31148;
if sex = 2 then sex = 0;
bud = newinc/48;
pnfi4 = exp(-2.4375 - 2.2239*sex - .0022073*lecst - .75441*lectst);
unot13 = -2.3508 - 2.1435*sex + .0038497*bud;
unot14 = -2.6325 - 2.4704*sex + .0028539*BUD;
unot15 = -2.7750 - 2.6481*sex + .0032457*bud;
unot16 = 0;
budcne = bud - lecst;
if budcne < 0 then budcne = 0;
ufish4 = .0022073*(budcne) - .75441*lectst;
eufish4 = exp(ufish4);
run;

data all; merge tre fiv sev nin;
by license;
keep eufish1-eufish4  pnfi1-pnfi4 unot1-unot16 license;
if eufish1 = . then eufish1 = 0;
if eufish2 = . then eufish2 = 0;
if eufish3 = . then eufish3 = 0;
if eufish4 = . then eufish4 = 0;
if pnfi1 = . then pnfi1 = 0;
if pnfi2 = . then pnfi2 = 0;
if pnfi3 = . then pnfi3 = 0;
if pnfi4 = . then pnfi4 = 0;
if unot1 = . then unot1 = 0;
if unot2 = . then unot2 = 0;
if unot3 = . then unot3 = 0;
if unot4 = . then unot4 = 0;
if unot5 = . then unot5 = 0;
if unot6 = . then unot6 = 0;
if unot7 = . then unot7 = 0;
if unot8 = . then unot8 = 0;
if unot9 = . then unot9 = 0;
if unot10 = . then unot10 = 0;
if unot11 = . then unot11 = 0;
if unot12 = . then unot12 = 0;
if unot13 = . then unot13 = 0;
if unot14 = . then unot14 = 0;
if unot15 = . then unot15 = 0;
if unot16 = . then unot16 = 0;
```

```
data alla; set all;
kti = log(eufish1 + eufish2 + eufish3 + eufish4);
pnf = exp(- (pnfi1 + pnfi2 + pnfi3 + pnfi4));

unots = unot1 + unot5 + unot9 + unot13;
unotw = unot2 + unot6 + unot10 + unot14;
unotc = unot3 + unot7 + unot11 + unot15;
unote = unot4 + unot8 + unot12 + unot16;
ekus = exp(kti - unots);
ekuw = exp(kti - unotw);
ekuc = exp(kti - unotc);
ekue = exp(kti - unote);
lagurs = (.369188589342 *(.170279632305 + unots)*
exp(-exp(-.170279632305 - unots))) +
(.418786780814 *(.9037017767 + unots) *exp(-exp(-.9037017767 - unots)))
+
(.175794986637 *(2.2510866298 + unots) *exp(-exp(-2.2510866298 -
unots))) +
(.0333434922612 *(4.2667001702 + unots) *exp(-exp(-4.2667001702 -
unots))) +
(.0027945362352 *(7.0459054023 + unots) *exp(-exp(-7.0459054023 -
unots))) +
(.0000907650877 *(10.7585160101 + unots) *exp(-exp(-10.7585160101 -
unots))) +
(.00000084857467 *(15.7406786412 + unots) *exp(-exp(-15.7406786412 -
unots))) +
(.000000001048 *(22.8631317368 + unots) *exp(-exp(-22.8631317368 -
unots)));
lagurw = (.369188589342 *(.170279632305 + unotw)*
exp(-exp(-.170279632305 - unotw))) +
(.418786780814 *(.9037017767 + unotw) *exp(-exp(-.9037017767 - unotw)))
+
(.175794986637 *(2.2510866298 + unotw) *exp(-exp(-2.2510866298 -
unotw))) +
(.0333434922612 *(4.2667001702 + unotw) *exp(-exp(-4.2667001702 -
unotw))) +
(.0027945362352 *(7.0459054023 + unotw) *exp(-exp(-7.0459054023 -
unotw))) +
(.0000907650877 *(10.7585160101 + unotw) *exp(-exp(-10.7585160101 -
unotw))) +
(.00000084857467 *(15.7406786412 + unotw) *exp(-exp(-15.7406786412 -
unotw))) +
(.000000001048 *(22.8631317368 + unotw) *exp(-exp(-22.8631317368 -
unotw)));
lagurc = (.369188589342 *(.170279632305 + unotc)*
exp(-exp(-.170279632305 - unotc))) +
(.418786780814 *(.9037017767 + unotc) *exp(-exp(-.9037017767 - unotc)))
+
(.175794986637 *(2.2510866298 + unotc) *exp(-exp(-2.2510866298 -
unotc))) +
(.0333434922612 *(4.2667001702 + unotc) *exp(-exp(-4.2667001702 -
unotc))) +
(.0027945362352 *(7.0459054023 + unotc) *exp(-exp(-7.0459054023 -
unotc))) +
(.0000907650877 *(10.7585160101 + unotc) *exp(-exp(-10.7585160101 -
unotc))) +
(.00000084857467 *(15.7406786412 + unotc) *exp(-exp(-15.7406786412 -
unotc))) +
(.000000001048 *(22.8631317368 + unotc) *exp(-exp(-22.8631317368 -
unotc)));
lagure = (.369188589342 *(.170279632305 + unote)*
exp(-exp(-.170279632305 - unote))) +
```

```
(.418786780814 *(.9037017767 + unote) *exp(-exp(-.9037017767 - unote)))
+
(.175794986637 *(2.2510866298 + unote) *exp(-exp(-2.2510866298 -
unote))) +
(.0333434922612 *(4.2667001702 + unote) *exp(-exp(-4.2667001702 -
unote))) +
(.0027945362352 *(7.0459054023 + unote) *exp(-exp(-7.0459054023 -
unote))) +
(.0000907650877 *(10.7585160101 + unote) *exp(-exp(-10.7585160101 -
unote))) +
(.00000084857467 *(15.7406786412 + unote) *exp(-exp(-15.7406786412 -
unote))) +
(.000000001048 *(22.8631317368 + unote) *exp(-exp(-22.8631317368 -
unote)));

consumso = unots * pnf + ekus * lagurs;
consumwo = unotw * pnf + ekuw * lagurw;
consumco = unotc * pnf + ekuc * lagurc;
consumeo = unote * pnf + ekue * lagure;
pnfo = pnf;

data c.conzero; set alla;
keep pnfo consumso consumwo consumco consumeo LICENSE;
run;

data one; set c.cononea;
proc sort; by license;
data two; set c.contwo;
proc sort; by license;
data tre; merge one two; by license;
ppcv1 = -(1/0.38497)*(consums2 - consums);
ppcv2 = -(1/0.28539)*(consumw2 - consumw);
ppcv3 = -(1/0.32457)*(consumc2 - consumc);
ppcv4 = -(1/0.22073)*(consume2 - consume);

if ppcv1 < 0 then ppcv1 = 0;
if ppcv2 < 0 then ppcv2 = 0;
if ppcv3 < 0 then ppcv3 = 0;
if ppcv4 < 0 then ppcv4 = 0;

run;
proc means mean min max; var ppcv1 ppcv2 ppcv3 ppcv4;
run;
```

Appendix C Oregon Department of Fish and Wildlife Management Zones

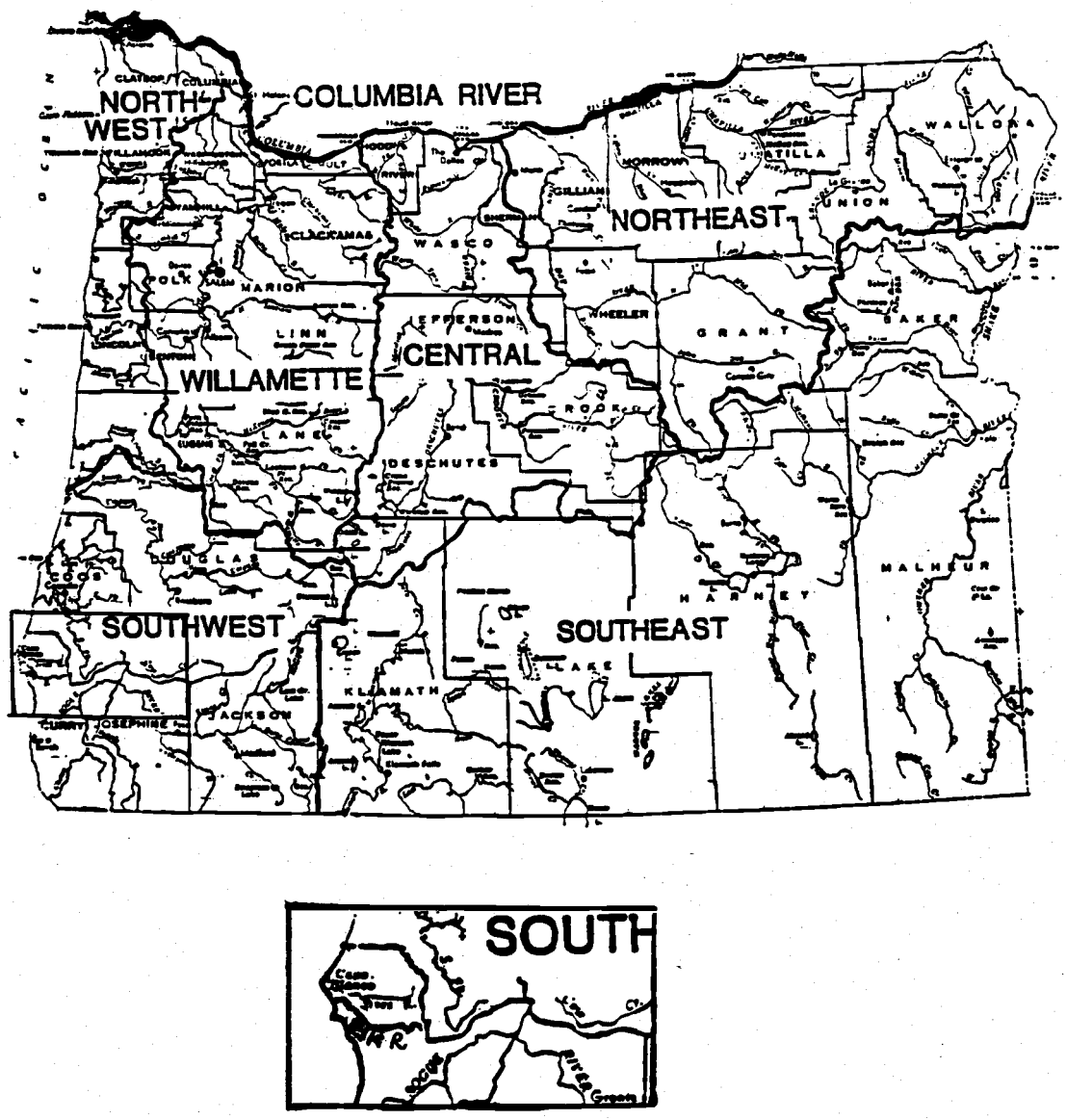

Figure C1. Oregon Department of Fish and Wildlife management zones, adapted from figure 1.1 in ODFW 1991a, showing an enlargement of the Elk River area.