AN ABSTRACT OF THE DISSERTATION OF

Michio Watanabe for the degree of Doctor of Philosophy in Agricultural and Resource Economics presented on July 18, 2003.

Title: A Spatially Explicit Model for Allocating Conservation Efforts: The Grande Ronde River Basin, Oregon

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Richard M. Adams

The spatial and dynamic pattern of landscape changes has a profound effect on the supply of environmental services, including the provision of habitat for fish and wildlife. Spatial heterogeneity is a common feature of landscapes in the Pacific Northwest, most notably in areas important to the production of salmonid fish species. This heterogeneity complicates attempts to design and implement policies to conserve the stocks of such species. To date, millions of dollars have been spent to improve habitat for salmonids, with mixed success.

This research examines the spatial implications of habitat restoration activities for the benefit of endangered salmonid fish species. A theoretical model defining an economically efficient allocation of restoration practices is developed for a hypothetical stream with a range of hydrological and spatial characteristics. An integrated hydrological, biological and economic modeling approach is then developed, and an empirical analysis is applied to the upper Grande Ronde River basin in northeastern Oregon.

Results of these analyses indicate that the heterogeneous nature of riparian conditions and stream morphology has a substantial effect on the efficacy of restoration activities. The

minimum cost allocation of restoration activities for small temperature reductions is to apply restoration efforts to nearby upstream reaches, while cumulative effects become important as the magnitude of desired temperature reduction increases. However, as the magnitude of desired temperature reductions increases, temperature reduction per dollar of restoration efforts decreases rapidly. In terms of specific riparian restoration efforts, passive restoration is preferred to active restoration as the magnitude of desired temperature reductions decreases and / or as the time frame considered is increased. It is also less costly in general to implement restoration activities in tributaries if the objective is to maximize stream length where water temperatures decrease by a specific amount. When two targeting options are compared (a fish abundance goal vs. a temperature reduction goal), this study found that temperature targeting is inefficient in the sense that it is possible to produce a larger salmonid population with the same budget, and that the levels of temperature targets have significant impacts on fishery benefits.

A Spatially Explicit Model for Allocating Conservation Efforts:

The Grande Ronde River Basin, Oregon

by

Michio Watanabe

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I understand that my dissertation will become part of the permanent collection of Oregon State University libraries. My signature below authorizes release of my dissertation to any reader upon request.

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Michio Watanabe, Author

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A Spatially Explicit Model for Allocating Conservation Efforts: The Grande Ronde River Basin, Oregon

Chapter 1

Introduction

1.1 Introduction

The Pacific Northwest (PNW) is one of the few places in the world where relatively large natural runs of salmon coexist with millions of people with a high standard of living. Salmon have played an important commercial, cultural and religious role in the development of the PNW. However, salmon populations have been decreasing in the Columbia River basin and other stream systems in the PNW for more than a century. For example, the number of salmon returning to the Columbia River prior to European settlement has been estimated to be between 10 and 16 million. However, the present annual return to the river is approximately 1 million fish, the majority of which are produced artificially in hatcheries (Northwest Power Planning Council, 2000). Certain stocks of salmon and steelhead in the Columbia and Snake River basins have been reduced substantially, and to date six salmonid species in Oregon have been listed as threatened or endangered under provision of the Endangered Species Act (ESA).

The causes for the decline in salmon populations are complex but include the construction of dams on the mainstem of the Columbia River as well as the Snake River, degradation in fish habitat, deterioration in water quality, and excessive ocean and freshwater harvest. To reverse the decline in salmon runs in the Columbia River basin, it is estimated that more than 3 billion dollars have been spent (Lichatowich, 1999). To date, the results of these massive expenditures have been disappointing.

The Grande Ronde River, a tributary of the Snake River, is typical of fishery problems in the PNW. The Grande Ronde River basin contains productive forests and agricultural lands and supports diverse salmonid populations. The basin offers important spawning and rearing habitat for spring and fall chinook salmon (Oncorhynchus tchawytscha), steelhead (Oncorhynchus mykiss) and bull trout (Salvelinus confluentus). The salmon populations, however, have been in decline in the Grande Ronde River basin for over fifty years. For example, chinook salmon escapement dropped from an estimated 12,000 in 1957 to 400 in 1992 (West and Zakel, 1993, cited by GRMWP, 1994).

Degradation of water quality as well as other habitat conditions has been cited as the causes for the decline in salmonid populations (*i.e.*, Anderson *et al.*, 1993; ODEQ, 2000). The loss of riparian vegetation caused by grazing, logging, and road construction has decreased bank stability, increased channel erosion, decreased sediment interception by vegetation, and reduced inputs of large woody debris, and they have severely degraded salmonid habitat conditions (Anderson *et al.*, 1993). In addition, "water quality impairments in tributaries and mainstem reaches of the Grande Ronde and other Upper Columbia River tributaries have reduced the extent of spawning and rearing habitat for these species" (ODEQ, 2000).

High water temperature is one of the primary problems associated with water quality. The U.S. Environmental Protection Agency (2003) states that "water temperatures significantly affect the distribution, health, and survival of native salmonids in the Pacific Northwest." Drake (1999) found that seasonal maximum temperatures and variables related to it explained the distribution and abundance of trout in the Upper Grande Ronde River basin, and argued that management and restoration activities should focus on reducing stream temperatures. Ebersole (2001) also found that maximum water temperatures are one of the significant variables for chinook salmon and rainbow trout densities in the Grande Ronde River basin. In order to

improve water quality problems, ODEQ has developed a Total Maximum Daily Load (TMDL) requirement of the Upper Grande Ronde (UGR) basin, and the focus of this TMDL is to decrease water temperatures.

The primary policy instruments under consideration to attain TMDL targets are watershed conservation programs such as the Conservation Reserve Program (CRP) and the Environmental Quality Incentive Program (EQIP). These programs have proven successful in other settings in the U.S., and the sum of the rental rate and cost share expenditure under the CRP amounted to approximately USD 1.8 billion in fiscal year 2002.[1] Some, however, argue that these conservation funds have not been used efficiently, particularly with respect to riparian improvements (i.e. Ribaudo, 1986; Reichelderfer and Boggess, 1988; Wu and Boggess, 1999; and Wu *et al.*, 2000). Given the mixed success with other salmonid improvement practices in the Columbia basin, it is important that these conservation practices be implemented in an efficient manner to minimize the social cost of restoration.

1.2 Objective of the dissertation

The overall objective of this dissertation is to explore the spatial configuration of conservation practices to decrease water temperatures and to increase salmon and trout populations in the Upper Grande Ronde River basin, Oregon. As discussed above, the Upper Grande Ronde River violates water quality standards for several pollutants. In this research, the primary focus is on stream temperature and riparian conditions. There are three supporting objectives that guide this study in meeting the overall objective. These are:

[1] http://www.fsa.usda.gov/dafp/cepd/stats/FY2002.pdf (Cited 7/2003)

1) Develop a theoretical framework that explores the economically optimal allocation of conservation practices;
2) Examine efficient allocations of restoration efforts in the Upper Grande Ronde to attain certain temperature targets; and
3) Examine how the allocation of restoration efforts under a temperature goal differs from one focused on fish abundance.

The first specific objective is to develop a theoretical model on the allocation of conservation practices that maximizes net benefits derived from water quality improvements, where net benefits are defined as the difference between total benefits and total costs. Total benefits derive from an increase in fish abundance, and it is assumed that fish abundance is the function of water temperature, riparian conditions, and discharge. Control variables are riparian restoration investments and instream flow augmentation.

The second specific objective is to gain insight on the spatial configuration of restoration alternatives. For example, should restoration activities focus on the mainstem or the tributaries if the goal is to decrease temperatures in the mainstem? Likewise, should restoration efforts be concentrated near the point where temperatures are to be reduced or should they be spread along the upstream reaches and tributaries? In other words, which is more effective in reducing temperatures, the local effect or the longitudinal (cumulative) effect?

The third question addresses the spatial distribution of restoration efforts under two different targeting scenarios: one based on physical criteria (such as temperature targeting) and the other based on the value of environmental services (such as fish abundance). While decreases in stream temperature are expected to bring about a variety of fish and wildlife benefits, in this study, the primary focus is on salmonid abundance because such cold-water fish

species are the most temperature-sensitive use in the basin (ODEQ, 2000; McGowan *et al.*, 2001). Previous economic research indicates that conservation efforts are generally not implemented efficiently if they are allocated based on physical criterion (see *e.g.,* Wu *et al.*, 2000). In this third component of the study, the spatial configuration of restoration practices under these two different targeting scenarios is compared, and the direction and the magnitude of any differences in efficiency resulting from physical criteria targeting are examined.

1.3 Background - the Upper Grande Ronde River basin

The Grande Ronde River is a tributary of the Snake River, which is a tributary of the Columbia River. The river originates in northeastern Oregon and joins the Snake River at the southeastern corner of the state of Washington. The entire basin is approximately 4,130 square miles.

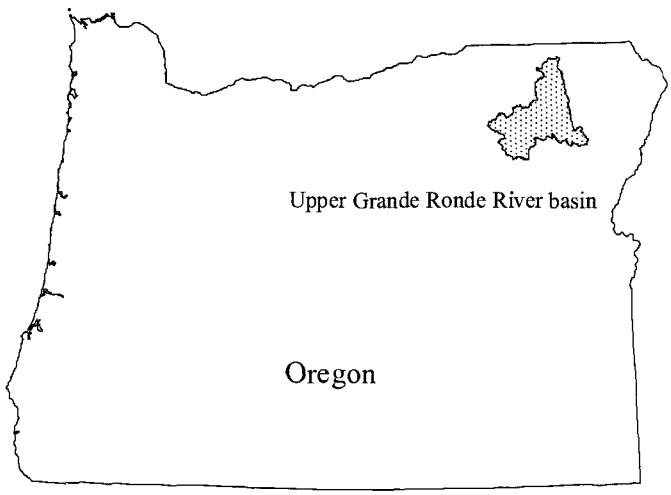

Figure 1.1: Location of the Upper Grande Ronde River Basin

The focus area of this study is the Upper Grande Ronde (UGR) basin located in the northeast corner of Oregon. The study basin is approximately 660 square miles. The elevations vary from 2900 feet to 5800 feet. Lower elevations generally receive 12 to 25 inches of rainfall equivalent precipitation annually. Higher elevations commonly receive up to 50 inches of rainfall equivalent precipitation annually, most of which is received as snowfall. Highest flows are associated with rain on snowpack events, while low flows are generally associated with a long summertime drought and complete melt of the snowpack (ODEQ, 2000). Thus, throughout the basin, peak flows occur in the spring (April-June) and declining flow through summer and early fall (Huntington, 1993). Although reduced stream flows are one of the primary concerns in the Grande Ronde River basin, there is no water withdrawal for irrigation in the study basin.

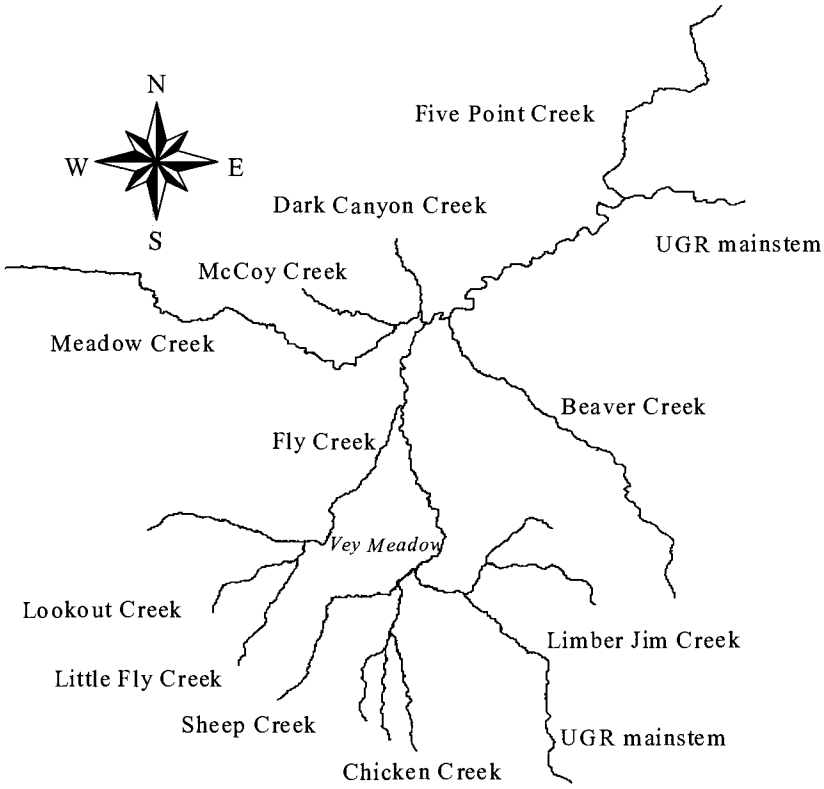

Figure 1.2: Study area in the Upper Grande Ronde River basin

The land ownership consists of both public and private. Most public lands (primarily owned by the U.S. Forest Service) are located in the source areas of each stream, as well as the lower part of Fly creek and the middle stretch of the UGR mainstem. Forested area accounts for 63 percent of the riparian area (30 meter width in each side from the stream) in the study basin, followed by herbaceous upland (8 percent), scrub-shrub (7 percent) and agriculture (6 percent). Agricultural lands in the study basin are primarily used for cattle grazing (ODEQ, 2000).

Water quality deterioration in the form of elevated temperatures is one of the primary natural resource concerns in the UGR basin. Figure 1.3 shows the maximum daily temperature in the mainstem at Vey Meadow in the summer of 1999. Temperatures were particularly high in

late July (date 210) and late August (date 240) when the maximum daily temperatures reached 25 °C.

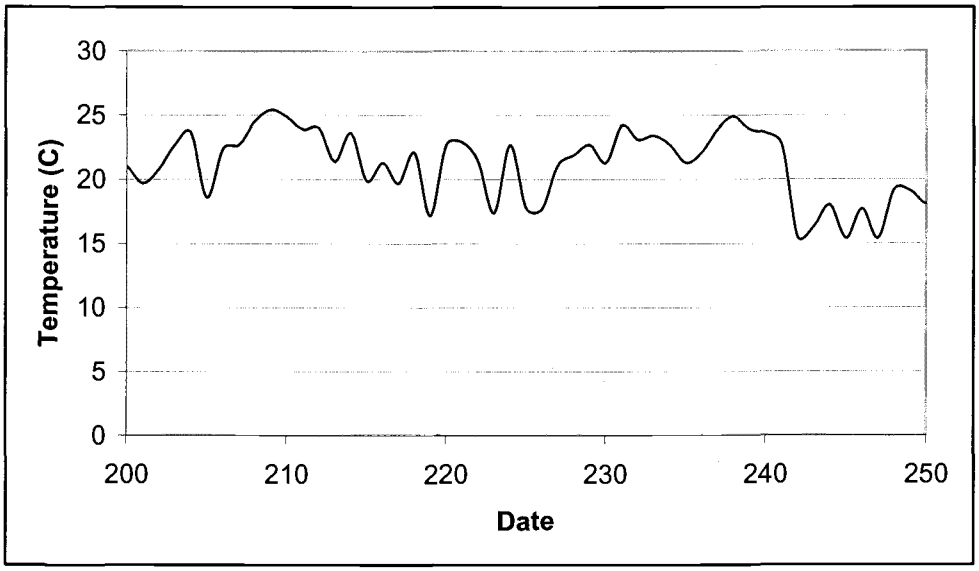

Figure 1.3: Maximum daily temperatures at Vey Meadow in the UGR mainstem in 1999

Note: Date is counted from January 1st.

Water quality in the Grande Ronde River basin has deteriorated to the point that the mainstem and tributaries of the Upper Grande Ronde River (UGR) violate water quality standards in terms of temperature, aquatic weeds or algae, bacteria, dissolved oxygen (DO), flow modification, habitat modification, nutrients, pH and sedimentation. Table 1.1 summarizes the streams in the study basin that violate temperature standards. It shows that in many segments of the streams, the water temperature criterion of salmonid rearing 64 °F (17.8 °C) has been violated.

Table 1.1: 1998 § 303(d) listed segments and applicable criterion

Stream	Segment	Criterion
Beaver Cr.	Mouth to La Grande Reservoir	Rearing 64F (17.8C)
Chicken Cr.	Mouth to West Chicken Cr.	Rearing 64F (17.8C)
Chicken Cr., West	Mouth to end of meadow in Section 15	Rearing 64F (17.8C)
Dark Canyon Cr.	Mouth to Headwaters	Rearing 64F (17.8C)
Fly Cr.	Mouth fo Umapine Cr.	Rearing 64F (17.8C)
Fly Cr., Little	Mouth to Headwaters	Rearing 64F (17.8C)
Grande Ronde R.	Limber Jim Cr. To Clear Cr.	Oregon Bull Trout 50F (10C)
Grande Ronde R.	La Grande to Limber Jim Cr.	Rearing 64F (17.8C)
Limber Jim Cr.	Mouth to Marion Cr.	Rearing 64F (17.8C)
Limber Jim Cr.	Marion Cr. To Headwaters	Oregon Bull Trout 50F (10C)
McCoy Cr.	Mouth to Headwaters	Rearing 64F (17.8C)
Meadow Cr.	Mouth to Headwaters	Rearing 64F (17.8C)
Sheep Cr.	Mouth to Warm Mineral Springs	Rearing 64F (17.8C)
Sheep Cr. E.F.	Mouth to Headwaters	Rearing 64F (17.8C)

Source: ODEQ (2000)

Note: Rearing 64 °F is applied to a basin for which salmonid fish rearing is a designated beneficial use. Bull trout 50 °F is applied to waters to support or to be necessary to maintain the viability of native Oregon bull trout.

In the Upper Grande Ronde River basin, the TMDL regulations have been in place since 2000. The TMDL targets have been set in terms of effective shade, channel width, sinuosity, width to depth ratio and discharge (ODEQ, 2000).[2] The ODEQ then performed a simulation analysis to calculate the temperatures that result from the riparian conditions and stream conditions which meet the TMDL targets. According to the TMDL document, most parts of the mainstem in the study basin attain the 64 °F (17.8 °C) temperature criterion for rearing

[2] In fact, these are surrogate measures. The formal TMDL target has been set in terms of solar radiation load (ODEQ, 2000).

salmonid juveniles, once the targets are met (ODEQ, 2000).[3]

High summertime water temperatures in the mainstem UGR and the lower portions of its tributaries create thermal barriers, restricting movements of adult salmonids and limiting the quantity and quality of available rearing habitat for juveniles (Ebersole, 2001). As a result, populations of several species of anadromous fish native to the Grande Ronde Basin are extinct or are threatened with extinction. Sockeye, coho and early-fall chinook salmon populations are now extinct (ODFW *et al.*, 1990, cited by Huntington, 1993). Snake River fall and spring chinook salmon are listed as threatened under the ESA and Snake River steelhead are listed as threatened in the UGR basin (ODEQ, 2000). Bull trout have also been petitioned for listing under the ESA (Huntington, 1993). Thus, summer steelhead , bull trout, and spring/fall chinook salmon are the primary fish species of concern.

Conservation activities have been actively implemented in the basin. For example, the Grande Ronde Model Watershed Program was established in 1992 to serve as an example for the establishment of watershed management partnerships among local residents, state and federal agency staffs, and public interest groups. Since its establishment, the GRMWP has assisted in the development of many projects for habitat restoration and water quality improvement in the Grande Ronde River basin in close coordination with agencies including the Oregon Department of Agriculture, Oregon Department of Fish and Wildlife, Soil and Water Conservation District, and USDA Forest Services. More than 30 million dollars have been spent since 1985 in restoration activities in the Grande Ronde Basin (GRMWP unpublished data, 2002).

[3] For those stretches where 64 °F temperature criterion for rearing salmonid juveniles are not attainable, the water temperature standard is that there is "no measurable surface water temperature increase resulting from anthropogenic activities."

1.4 Organization of the dissertation

The rest of this dissertation is structured as follows. Chapter 2 reviews the literature on the efficient allocation of water resources associated with water quality. Chapter 3 presents a theoretical model on the optimal allocation of conservation efforts in a watershed. Chapter 4 explains the methodology of the simulation study, followed by the results and discussion in Chapter 5. This dissertation is concluded by summarizing major findings in Chapter 6.

Chapter 2

Literature Review

The overall objective of this study is to examine an efficient allocation of conservation practices in the Grande Ronde River basin to decrease water temperature and to improve habitat for salmonids. To date, there do not appear to be any studies that specifically focused on the spatial configuration of these conservation practices associated with water temperatures in a watershed context. But there do exist studies that have examined an efficient allocation of water resources associated with water quantity and water quality. Thus, these studies are the first focus of this literature review.

Once an efficient spatial configuration of conservation practices in a watershed is determined, the next step is how to attain this allocation. In the western part of the United States, the primary rule that governs water allocation is the "prior appropriations doctrine." Under this doctrine, older water uses get priority over newer water uses; new users would have to wait their turn and divert water only after older uses had been satisfied (Bastasch, 1998). To improve water allocative efficiency under this doctrine, primary policy tools include water markets and conservation programs such as the Conservation Reserve Program. The latter is the focus of this study. Thus, the second section of this chapter focuses on conservation programs and related studies.

The third step of this research is to link the conservation practices with water temperatures and fish abundance, which are the policy targets. Thus, in the third section, literature on the relationships between riparian conditions, water temperatures, and fish abundance are reviewed.

2.1 Efficient water allocation and water quality improvement

In the United States, competition for water resources has increased as the demand for water has risen and as the recognition of the importance of instream flows has increased. Academics and policy makers have called for more efficient use of water resources, and as a result, there exist a number of theoretical as well as empirical studies that examined alternative allocations of water resources. Some of these studies also consider water quality issues. Kanazawa (1991) derived a condition of efficient water allocation, considering a saline water quality problem. Griffin and Hsu (1993) developed a model of water uses along a stream, considering instream flow values as well as return flows. They showed that for the water allocation to be efficient, water consumption in upstream areas should be less than that in downstream areas. Weber (2001) developed a more general theoretical model of water consumption and pollutant discharge along a stream without a tributary, with a condition that a minimum flow and a water quality requirement are met. She showed that the social cost of discharging pollutants into a stream decreases as one moves downstream. These studies examined conditions of efficient water allocation along a stream, but the effects of tributaries were not considered.

There have also been a large number of empirical studies on the allocation of water resources. For example, Meier and Beightler (1967) developed a dynamic programming technique to compute an efficient allocation of consumptive water uses along a river with a tributary. Scherer (1977) expanded this analysis to include water quality (salinity) issues, and derived a condition of the efficient allocation of irrigation water use, using a hypothetical numerical example. Booker and Young (1994) examined salinity problems in the Colorado River basin, and showed that efficient allocation would require large transfers from existing consumptive users in the upper basin. Paulsen and Wernstedt (1995) applied an optimization

framework to the Columbia River basin to examine the cost and biological tradeoffs to rebuild salmonid populations. Willis and Whittlesey (1998) examined a least cost method that attains a minimum stream flow objective in the Walla Walla River basin, Washington. Hurd *et al.* (1999) constructed spatial equilibrium models for the four major river basins in the United States. The models maximized the social benefit (sum of consumer and producer surplus) under different water allocation scenarios, subject to physical, economic and institutional constraints. Stevens *et al.* (2000) estimated the benefits of stream flow augmentation and compared the costs of achieving enhanced summer stream flows for the middle Deschutes River in Oregon. They found that the combination of leasing water rights from irrigators and repairing canals were the least-cost method. These theoretical and empirical studies are similar to this study in terms of general allocation issues, but none has explicitly considered the impact of riparian conditions on water temperatures and fish abundance, which is the focus of this study.

Another key issue in dealing with environmental quality management is the existence of heterogeneity. Sanchirico and Wilen (1999) focused on the role of heterogeneous resources in an evaluation of how a patchy environment affects biological as well as economic efforts in a marine environment. They showed that where there is a human influence, equilibrium depends on both economic and biological parameters. An implication of their research to the present study is that the pattern of restoration activities must consider the heterogeneous nature of habitat conditions in the basin. Fish responses to a change in temperature are likely to vary across stream segments due, for example, to different riparian conditions. An efficient allocation of restoration efforts in a riverine setting should therefore consider this heterogeneity.

2.2 Conservation programs and related studies

There are three major conservation programs currently being implemented in Oregon: Conservation Reserve Program (CRP), Conservation Reserve Program - Oregon Enhancement Program (CREP), and Environmental Quality Incentives Program (EQIP). The major features of each of these programs are presented here.

(1) Conservation Reserve Program (CRP)[4]

The Conservation Reserve Program (CRP) is a voluntary program and is administered by the Farm Service Agency (FSA) of the United States Department of Agriculture (USDA). The CRP encourages farmers to convert highly erodible cropland or other environmentally sensitive acreage to vegetative cover, wildlife plantings, trees, filter strips, or riparian buffers. In return, farmers receive an annual rental payment, and cost sharing is provided for initiating the vegetative cover practices. The goals of the CRP are to reduce soil erosion, protect the Nation's ability to produce food and fiber, reduce sedimentation in streams and lakes, improve water quality, establish wildlife habitat, and enhance forest and wetland resources. Eligible land must be either cropland that is susceptible to erosion or marginal pastureland that is suitable for use as a riparian buffer strip. The selection of lands to be enrolled in CRP contracts is determined based on the Environmental Benefit Index (EBI), which takes into account the following factors:

- Wildlife habitat benefits resulting from establishment of vegetative cover on contract acreage;
- Water quality benefits from reduced erosion, runoff, and leaching;

[4] http://www.fsa.usda.gov/pas/publications/facts/crp02.pdf and

- On-farm benefits of reduced erosion;
- Benefits that will likely endure beyond the contract period; and
- Cost.

CRP provides additional incentives for continuous signup. To be eligible for this continuous signup, the land must meet the basic CRP eligibility requirements. In addition, the acreage must also be determined by USDA's Natural Resources Conservation Service (NRCS) to be suitable for practices such as riparian buffers, filter strips, grassed waterways, shelter belts, field windbreaks, and living snow fences. Table 2.1 provides a summary of CRP practices and their acreages in Oregon for all program years (1987-2003).

Table 2.1: Summary of CRP practices acreages in Oregon for 1987-2003

Practices		Acreage	
Introduced grasses	CP1	105,825	23%
Native grasses	CP2	31,131	7%
Wildlife habitat	CP4	13,271	3%
Established grass	CP10	289,816	64%
Riparian buffers	CP22	8,759	2%
Others		6,359	1%
Total		455,161	100%

Source: http://www.fsa.usda.gov/crpstorpt/07approved/r1pracyr/or.htm (cited 5/2003)

(2) Conservation Reserve Enhancement Program (CREP)[5]

The Conservation Reserve Program - Oregon Enhancement Program (CREP) is a voluntary program and is administered by FSA and the State of Oregon. The CREP has been

http://www.fsa.usda.gov/pas/publications/facts/html/crpcont00.htm (cited 5/2003)

implemented since 1998 in Oregon. The objective is to improve the water quality of streams that provide habitat for salmon and trout species listed under the Federal Endangered Species Act by reducing water temperature to natural levels, reducing by 50 percent the sediment and nutrient pollution from agricultural lands adjacent to streams, and stabilizing stream banks along critical salmon and trout streams. The project area includes all streams in Oregon on agricultural land that provide habitat for endangered salmon and trout. Eligible land must contain acreage along salmon and trout streams and must be eligible for CRP. Eligible practices include filter strip, riparian buffer, and wetland restoration. Land rental cost and 75 percent of the cost of establishing conservation practices are provided for participating farmers. The CREP in Oregon is authorized to enroll up to 95,000 acres of riparian buffers and filter strips, plus 5,000 acres of wetlands. The total program cost is estimated at $250 million.

(3) Environmental Quality Incentives Program (EQIP)[6]

Environmental Quality Incentives Program (EQIP) is a voluntary conservation program and has been administered by NRCS since 1997. The objective is to promote agricultural production and environmental quality as compatible national goals. The eligible land includes cropland, rangeland, pasture, private non-industrial forestland, and other farm or ranch land. Through EQIP, farmers and ranchers may receive financial and technical help to install or implement structural and management conservation practices on eligible agricultural land. EQIP may pay up to 75 percent of the costs of certain conservation practices important to improving and maintaining the health of natural resources. In 2001, approximately $3.5 million dollars were obligated and the contracts covered nearly 116,000 acres in Oregon. Since 1997 when the

[5] http://www.fsa.usda.gov/pas/publications/facts/orcrep.pdf (cited 5/2003)
[6] http://www.nrcs.usda.gov/programs/farmbill/2002/pdf/EQIPFct.pdf (cited 9/2002)

program started, approximately $16 million have been obligated in Oregon (but none in the Grade Ronde River basin).[7]

(4) Studies on conservation programs

These conservation programs, although playing an important role in watershed enhancement, do not seem to have received as much attention as water markets. Some studies, however, have argued that the conservation programs themselves have not been implemented efficiently. For example, Ribaudo (1986) argued that the conservation programs have historically been designed to protect specific resources, managed by different agencies, and targeted on the basis of onsite physical criteria, such as soil erosion rates, rather than on the values (benefits) of environmental services provided. Reichelderfer and Boggess (1988) examined the performance of CRP in 1986 and found that the implementation was suboptimal in the sense that the net government cost of the program could have been reduced while simultaneously increasing the level of erosion reduction and supply control achieved. Recently, Wu and Boggess (1999) developed a theoretical model that showed that in the presence of threshold effects and the correlation between alternative environmental benefits, the allocation of conservation funds based on onsite physical criteria could result in little environmental quality improvement.[8] Then, Wu *et al.* (2000) and Wu and Skeleton-Groth (2002) empirically demonstrated the existence of threshold effects in the relationship between riparian conditions and fish abundance in the John Day River basin in eastern Oregon.

These studies have played an important role in improving the design of conservation programs. But none of them has evaluated the efficiency of the allocation of conservation practices in a watershed context. Although Wu *et al.* (2000) and Wu and Skeleton-Groth (2002)

[7] http://www.nrcs.usda.gov/programs/eqip/2001summaries/OREQIPdo.pdf (cited 5/2003)

examined riparian restoration efforts in two streams, the cumulative effects of streams (water quality in the upstream area affects water quality in the downstream area) were not considered. This aspect is also taken into account in this study.

2.3 Riparian conditions, water temperature and fish abundance

The conservation practices discussed above are expected to increase fish abundance through riparian condition improvements and water temperature decreases. In this section, studies on these relationships are reviewed. Since these effects are site-specific, studies made at the sites similar to the Grande Ronde River basin in northeastern Oregon are the primary focus.

2.3.1 Water temperature and riparian conditions

Water temperature is an expression of heat energy per unit of time, and there are six processes that allow heat energy exchange between a stream and its environment (Boyd and Sturdevant, 1997). These processes are solar energy, longwave radiation, evaporation, convection, streambed conduction, and groundwater inflow/outflow. Among these six processes, Brown (1970) found that the principal source of heat energy for streams is solar energy striking the stream surface directly. If riparian conditions are improved, there will be a greater level of riparian canopies (shading), and the surface area of a stream flow where solar energy strikes will be reduced. This is a direct impact of shading on water temperatures. Beschta (1997) also explained an indirect effect of shading on water temperatures. According to Beschta, where streamside vegetation is removed or reduced, a loss of root strength encourages stream bank erosion and channel widening. Additional stream width typically results in

[8] These effects are called "cumulative effects" in Wu and Boggess (1999).

relatively shallow stream depth during summertime flows, and the surface area of stream increases geomorphologically (Beschta, 1997).

The importance of shading on water temperatures, however, has been debated. Larson and Larson (1996) argued that shading is unable to control water temperature and is only one component of many watershed attributes such as air mass characteristics, elevation gradient, channel width and depth, water velocity, surrounding landscape, and interflow inputs. Bohle (1994), however, measured water temperatures in the Upper Grande Ronde River basin in the summer of 1991 and 1992 and examined the relationship between water temperatures and riparian vegetation as well as channel morphology. He found that stream cover (shading) plays an important role in moderating stream temperatures and reducing diel fluctuations during warm summer days. He also found that flow velocity and percent undercut banks have an effect of moderating stream temperatures. Beschta (1997) also argued that increased levels of shading for water quality limited streams would greatly improve (reduce) summertime stream temperatures in most situations in the Intermountain West. Moore and Miner (1997) also stated that "(s)hade is very important as a means of intercepting sunlight and reducing the energy that is transferred to the surface of a stream." Boyd and Sturdevant (1997) also argued the importance of shading on stream temperature. They identified two areas that need to be worked to control water temperatures. The first is that streams must experience long duration quality shade. The second is that the stream surface area exposed to solar radiation should be reduced by bank stability augmentation. These studies show that in an eastern Oregon setting, shade is an important factor that controls water temperature.

Models have been developed to predict water temperatures. Boyd (1996) developed the Heat Source model that predicts water temperatures at the reach scale. The Heat Source model has been used by the Oregon Department of Environmental Quality in TMDL studies. Cox

(2002) developed the WET-Temp model that predicts water temperatures. A desirable feature of the WET-temp model is its ability to incorporate spatial GIS data. It is also less information intensive than other temperature models such as the Heat Source model. In this study, the WET-Temp model is employed as a temperature model.

2.3.2 Riparian condition, water temperature and fish abundance

In eastern Oregon where many streams are characterized by low summer flows and elevated stream temperatures, riparian conditions and water temperatures play an important role in determining fish abundance (*e.g.*, Platts and Nelson, 1989; Li *et al.*, 1994; Baigun *et al.*, 2000; Ebersole, 2001; and Welsh *et al.*, 2001). Platts and Nelson (1989) examined the impacts of stream canopy and other variables on the salmonid biomass in the northern Rocky Mountains and Great Basin[9]. They found that canopy density was positively related to the salmonid biomass ($P<0.01$). Sun arc and thermal input also exhibited a negative relationship with the salmonid biomass ($P<0.05$).

Li *et al.* (1994) examined the relationship between the densities of rainbow trout and riparian conditions as well as maximum temperatures in the John Day River basin in eastern Oregon. They found that the maximum temperature is negatively correlated with rainbow trout biomass in Rock, Mountain and Fields Creeks ($P<0.01$), while it was not significantly correlated (although the sign was negative) in Alder and Service creeks. This is probably because the water temperatures were so high in Alder and Service creeks that the creeks were devoid of fish. They also found that many riparian characteristics such as vegetative use, bank stability and soil alteration were correlated with the density of rainbow trout in Alder and Service Creeks ($P<0.05$), while they were not significantly so in Rock, Mountain and Fields Creeks. This is

probably because water temperature is cold in the latter creeks and that riparian conditions were less influential on fish densities.

Baigun *et al.* (2000) examined the influence of water temperature in deep pools on summer steelhead in Steamboat Creek, a tributary of the North Umpqua River in southern Oregon. They found that mean bottom water temperatures were negatively correlated with the abundance of adult summer steelhead in August-September 1991 and 1992 ($r=-0.47$).

Welsh *et al.* (2001) examined a relationship between the existence of coho salmon and water temperatures in tributaries of the Mattole River in northern California. They found that all but two tributaries whose maximum weekly maximum temperature exceeded 18.0 °C (64 °F) degrees did not have coho salmon.

Ebersole (2001) also examined the relationship between fish abundance (chinook salmon and rainbow trout), maximum water temperature, and riparian conditions in Grande Ronde River in eastern Oregon. He found that chinook salmon density was negatively correlated to channel wetted width to depth ratio ($P<0.05$), and positively correlated with proportional pool area ($P<0.05$). Maximum temperature was also negatively correlated with chinook density ($P<0.1$). He found that maximum temperature was the most significant factor ($P<0.05$) affecting rainbow trout density along with mean substrate embeddedness ($P<0.05$). These studies show that both riparian conditions and maximum water temperature play an important role in determining fish density in streams characterized by low summer flows and elevated stream temperatures.

Recently, the importance of coldwater patches in elevated stream temperatures has been recognized. For example, Nielsen and Lisle (1994) examined the relationship between coldwater patches (called thermally stratified pools) and their use by steelhead trout in northern

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[9] The principal salmonid species included fry of chinook salmon, resident rainbow trout and steelhead,

California streams. They found that 65 percent of the juvenile steelhead found in the Racheria Creek study reaches moved into adjacent stratified pools during periods of high ambient stream temperatures (73-82°F (23-28°C)). Ebersole (2001) also examined the effect of coldwater patches on fish density in Grande Ronde River in eastern Oregon. He found that coldwater patch frequency was positively correlated with chinook salmon densities ($P<0.05$). As for rainbow trout densities, both coldwater patch frequency and coldwater patch areas were positively correlated with the densities ($P<0.05$). These studies show that in addition to riparian conditions and maximum water temperatures, the existence of coldwater patches play an important role in affecting fish densities.

There are studies that combine these biological and hydrological relationships to evaluate policy alternatives, and some of them include economic analyses. Theurer *et al.* (1985) integrated ecological and biological effects to examine the impact of different riparian vegetation and discharge scenarios on water temperatures and salmonid abundance in the Tucannon River, Washington. They considered four scenarios involving different riparian vegetation and stream morphology conditions. They found that estimated juvenile fish production would more than double when the climax vegetation is restored. They also conducted a cost-benefit analysis and argued that the benefit (an increase in the return of adult salmonid species) from the climax vegetation would far exceed the costs. Bartholow (1991) evaluated the effectiveness of alternatives to reduce summer maximum water temperatures for a 30 km stretch of the Cache la Poudre River, Colorado, using Stream Network Temperature Model (SNTEMP) developed by the US Fish and Wildlife Services. The alternatives considered included increasing discharge, doubling riparian shading, and halving stream width; an increase in discharge was determined to be the most effective in reducing water temperature. More

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cutthroat trout (Oncorhynchus clarki), bull trout, and brook trout (Salvelinus fontinalis).

recently, Hickey and Diaz (1999) developed an integrated model (AQUARIUS) of fish population, physical habitat, water temperature and water allocation, and analyzed alternative water allocation regimes to increase low winter flows in Colorado. They compared five alternative regimes, all intended to increase low winter flows, and found that while only the 25-cfs minimum flow regime would elevate rainbow trout populations, all alternatives would significantly enhance the brown trout fishery.

Like the research in this dissertation, the previous studies evaluate policy options associated with riparian restoration. However, these other studies only compared conservation scenarios and did not solve for optimal patterns of restoration. In addition, restoration alternatives are limited to the reaches in the mainstem, and thus ignore the tributaries. Therefore, it is difficult to gain insights on spatial configuration of restoration practices from these studies.

Chapter 3

Theoretical Framework

This chapter presents a general theoretical framework for the derivation of an efficient allocation of water resources within a watershed. Water resources in a watershed are assumed to be allocated efficiently if the sum of the social benefits derived from water resources in the watershed is maximized. Section 3.1 presents an overview on the development of a general theoretical framework on efficient water allocation. Section 3.2 and 3.3 develop such a theoretical framework within which to analyze conditions for efficient allocation of irrigation water use and riparian restoration activities along a stream with and without a tributary. Section 3.4 provides insights and conclusions gleaned from the theoretical framework.

3.1 Overview of an efficient allocation of water resources

In examining the efficient allocation of water resources, it is important to consider spatial effects of water uses. A socially efficient water allocation is attained if and only if the marginal benefits of water use at each diversion point are equal.[10] If the marginal benefit of water use at one diversion point, for example, is lower than that at other points, the amount of water diversion at the former will be reduced and that water will be allocated to other points of diversion. The total (social) benefits derived from water uses in the watershed can be increased by this reallocation.

This simple view, however, needs to be modified if water quality is also taken into account. Upstream water uses and the discharge of wastewater and/or irrigation return flows may adversely affect water quality in the downstream area. For example, it has been found that

[10] A diversions point is a point where water is withdrawn from a stream to be used, for example, for irrigation activities.

in the Upper Colorado River basin, irrigation return flows discharge dissolved salt into the stream. As a result of this high saline concentration, the value of stream flow to agricultural productivity in the downstream areas is diminished (*e.g.*, see Kanazawa, 1993). This is a typical negative externality problem. In such cases, there is a divergence between private and social marginal benefit of water uses. To be efficient, the social marginal benefit at each water use in a watershed should be equal.

It has also been recognized over the past two decades that the beneficial uses of water resources are not limited to consumptive uses; water in stream is also beneficial. Instream flows are known to have multiple functions including navigation, recreational activities such as fishing and rafting, and providing habitat for aquatic species such as fish. These functions yield benefits, and therefore, they also need to be taken into account in examining the efficient allocation of water resources.

There are two ways to incorporate the benefits of instream flow. The first is to maximize the benefits of consumptive water uses with a constraint that certain levels of discharge and water quality are not violated. Weber (2001), for example, examined the conditions of efficient allocation of water consumption and pollutant discharge with the constraint that these activities do not violate specified minimum levels of water quality and discharge. This approach, however, is not likely to result in a socially optimal level of water consumption at each diversion point since the minimum levels of water quality and discharge are determined exogenously and there is no reason to believe that these levels are set at socially optimal levels.

The second method is to maximize the benefits of both consumptive uses and instream flows. In this approach, the level of water quality and discharge are endogenously determined, and therefore, the allocation derived is socially optimal. Griffin and Hsu (1993), for example,

considered efficient allocation of water resources along a stream by maximizing the sum of benefits of consumptive uses as well as instream flows. One of the difficulties of this approach, however, is estimating the value of instream flows. To compare the value of instream flow with that of consumptive uses, accurate marginal values of instream flows must be available. There are a number of studies that investigated the marginal values of instream flow (see *e.g.* Brookshire *et al.*, 1980; Daubert and Young, 1981; Loomis, 1987; Ward, 1987; Johnson and Adams, 1988; Colby, 1989; and Duffield *et al.*, 1992). It has been found, however, that instream flow values depend on many site-specific factors, including the location in the stream, fish species in the river, and potential for downstream uses (Johnson and Adams, 1988). Thus, it is difficult to undertake an empirical analysis of an efficient allocation of water resources between consumptive uses and instream flows. This is the reason that regulatory frameworks such as the total maximum daily load (TMDL) regulations rely on the first approach; they set target levels of pollutant discharge without explicitly considering the economic benefits of instream flow which accrues from water quality and discharge improvements.

The theoretical framework developed in the following section employs the second approach since its objective is to gain general insights on the efficient allocation of water resources and riparian restoration investments. The theoretical framework developed here extends previous models in the following ways:

- The water quality problem focuses on water temperature, which is one of the most important water quality components in the western U.S., but has not been explicitly analyzed in a theoretical context.
- The model recognizes the possibility of corner solutions; the previous literature has always assumed interior solutions.

- The model analyzes the effect of tributary flows on an efficient allocation of water resources. Most existing literature has developed analyses assuming there is no tributary.

3.2 The case of no tributary

3.2.1 Introduction

This and the following sections present a theoretical framework to analyze the optimal allocation of conservation practices along a stream. Conservation practices considered are riparian restoration efforts such as fencing and vegetation planting, and instream flow augmentation through water conservation efforts and a lease or purchase of irrigation water. Riparian restoration efforts (passive and active restoration) are the most popular restoration activities in the Grande Ronde River basin (Grande Ronde Model Watershed Program, unpublished data, 2002). The importance of instream flow augmentation in the arid West has been discussed extensively in the economic literature over the past two decades as seen in the previous section. This analysis is primarily applied to 3rd and larger order[11] streams that are the key habitat for anadromous salmonid species in the Pacific Northwest. Many salmonid populations are in decline due to a range of factors, including high water temperatures in summer and fall, and reduced stream flows during critical periods in the salmonid lifecycle. Streams with these problems are common in the arid portions of the Pacific Northwest. Therefore, this analytical framework closely reflects the characteristics of streams and riparian conditions in the arid Pacific Northwest. Research questions examined in this and the following sections include the following:

[11] Order is a quantitative description of stream networks. Streams with no tributaries are designated 1st – order streams, the confluence of two 1st-order streams is the beginning of a 2nd-order streams, etc. (Dingman, 2002).

- Conditions required for the optimal allocation of conservation practices in a stream with and without a tributary, and
- Effects of budget constraints on an optimal allocation of conservation practices.

For the purposes of the conceptual analyses presented here, envision a stream that has I reaches which are ordered from reach 1 (located at the source of the stream) to reach I (located at the lowest portion of the stream). It is assumed that water is diverted currently for irrigation based on water rights and that there are no return flows to the stream from irrigation uses. Let the discharge in reach i be denoted as v_i,[12] and the water temperature T_i. It is also assumed that currently there is no riparian vegetation along the stream due to agricultural and livestock activities. Riparian restoration efforts implemented at i are denoted as r_i. Given an assumption that there is no riparian vegetation currently, riparian restoration efforts also represent the level of riparian conditions. Thus, in the following analysis, riparian condition and riparian restoration efforts are used interchangeably. The range of r_i is $0 \leq r_i \leq \bar{r}_i$, where $\bar{r}_i$ is the maximum level of riparian vegetation; *i.e.* 100 percent shading. Per unit cost of riparian restoration effort in reach i is denoted as w_i.[13]

Let the volume of flow augmentation through water conservation efforts and the lease or purchase of irrigation water at i be q_i. The maximum volume of q_i is the amount specified by the water right, which is denoted as $\bar{q}_i$. Thus, the range of q_i is $0 \leq q_i \leq \bar{q}_i$. The cost of water return at i is the loss in agricultural benefit resulting from reduced water diversion at that point,

[12] Discharge is defined here as the total volume of water moving past a point in a certain period of time (Allan, 1995).

[13] For example, w_i represents the cost of increasing riparian shading by 1 percent in reach i.

which is referred to as $h(q_i, i)$. It is assumed that $h_q > 0$ and $h_{qq} \geq 0$: the marginal cost of conserving water increases as its volume increases. The budget available for overall habitat restoration efforts in this stream is fixed, and is denoted as *B*. The goal of the conservation agencies is to maximize the benefit from conservation activities subject to the budget constraint.

The benefit of conservation efforts is derived from fish abundance, and the benefit in reach *i* is denoted as $G(T_i, v_i, r_i, i)$. While fish abundance is affected by many factors, it is assumed here that it is a function of three variables, maximum water temperature (T_i), discharge (v_i), and riparian conditions (r_i). As discussed in chapter 2, it has been shown empirically that maximum water temperatures affect the population of juvenile salmonid species in the Grande Ronde River and John Day River (Li *et al*, 1994; Drake, 1999; and Ebersole, 2001). Discharge is also a statistically significant variable in explaining the density of salmonid species in the John Day River (Li, *et al.* 1994). Riparian conditions such as riparian canopy density and channel wetted width to depth ratio also affect the density of salmonid species in the Grande Ronde River (Ebersole, 2001). Beschta (1997) has also shown that riparian vegetation affects aquatic habitat through litter inputs, insect drop, large and small woody debris recruitment, and nutrient transformations. These studies demonstrate that salmonid fish density is negatively affected by maximum water temperature ($G_T < 0$), and positively affected by riparian vegetation ($G_r > 0$) and discharge ($G_v > 0$). It is also assumed that these variables have a concave relationship with the benefit of instream flow ($G_{TT} < 0$, $G_{vv} < 0$, and $G_{rr} < 0$), but these assumptions are relaxed later. With respect to an interactive effect, we assume $G_{rv} < 0$, which implies that the marginal effects of riparian restoration on the stream flow benefits decreases as the discharge increases. Later this

assumption is also relaxed. For the interaction associated with water temperatures, it is assumed that the marginal effect of an increase in discharge or riparian shading is not affected by temperature levels ($G_{Tv} = 0$ and $G_{Tr} = 0$).

Fish abundance is also affected by factors other than maximum water temperature, discharge and riparian conditions. One of the important variables that affect fish abundance in the arid portions of the Pacific Northwest is the frequency of coldwater patches (Ebersole, 2001). Coldwater patches are generally produced by, for example, hyporheic and groundwater inflows and tributaries. Although the location of coldwater patches is hard to predict, coldwater patch frequency is to have a weak relationship with the width to depth ratio (r=0.22, Ebersole, 2001), which is also affected by riparian conditions. Therefore, although an impact of the frequency of coldwater patches on fish abundance is not explicitly considered in this analysis, the assumption that the benefit of instream flow is a function of riparian conditions takes into account a part of the important relationship between coldwater patches and fish abundance.

Water temperature is expected to rise as one moves downstream, given the initial condition (there is no riparian shading). However, riparian restoration effort (r_i) and discharge (v_i) affects the level of temperature increase. A change in water temperature in eastern Oregon streams of the type of interest in this analysis can be represented by the Brown equation (Brown, 1970):

$$\Delta T = \frac{A(r) * N}{v} * C \qquad (1)$$

where A = Surface area of a specific stream reach

N = Net heat exchange per unit of surface area

C = Coefficient

Thus, a change in water temperature can be denoted as $\Delta T = f(r_i, v_i, i)$. As the Brown equation shows, an increase in riparian restoration efforts and discharge reduces the level of temperature increase, and therefore, the expected signs are $f_r < 0$ and $f_v < 0$, respectively. In addition, the interaction between riparian restoration efforts and discharge is $f_{rv} > 0$ because the impact of riparian restoration on water temperature will decrease as discharge increases.

3.2.2 Optimal allocation of conservation practices

The first case involves the optimal pattern of conservation practices when there is no tributary to the stream or reach of interest. The control variables are water conservation efforts or the lease / purchase of irrigation water (q_i) and riparian restoration efforts (r_i). The social problem is to maximize the sum of the benefits of instream flow across I reaches in the study stream, subject to constraints on the state variables (discharge and water temperature), the range of control variables, and the budget for such activities. The problem can be specified as follows:[14]

$$\underset{q_i, r_i}{Max} \sum_{i=1}^{I} G(T_i, v_i, r_i, i) \tag{2}$$

s.t. $$v_{i+1} - v_i = -\overline{q}_i + q_i \tag{3}$$

$$T_{i+1} - T_i = f(r_i, v_i, i) \tag{4}$$

$$\sum_{i=1}^{I} w_i r_i + \sum_{i=1}^{I} h(q_i, i) \leq B \tag{5}$$

[14] Adapted from the model presented by Weber (2001). This set up assumes that the discharge decreases as one moves downstream due to water withdrawal.

$$0 \le q_i \le \overline{q}_i,\ 0 \le r_i \le \overline{r}_i \tag{6}$$

$$v_1 = \overline{v}_1, T_1 = \overline{T}_1, v_I \ge \overline{v}_I \tag{7}$$

This is a general constrained control problem with two control variables, q_i and r_i, and two state variables, v_i and T_i. Equations (7) are the initial and terminal conditions on the state variables. It is necessary that water discharge be greater than $\overline{v}_I$ at the terminal reach (reach *I*) to provide an adequate amount of water to water right holders located below the study area. We assume that the benefit of instream flow is continuously differentiable with respect to water temperature. It is assumed that the constraint qualification is satisfied (Kamien and Schwartz, 1991). Since it is not possible to generate the Hamiltonian directly from the above maximization problem because of the inequality integral constraint in equation (5), let Γ_i represent the following (Chiang, 1992):

$$\Gamma_i = -\left[\sum_{i=1}^{i} w_i r_i + \sum_{i=1}^{i} h(q_i, i)\right] \tag{8}$$

Given this integration, a change in Γ_i can be represented as follows:

$$\Gamma_{i+1} - \Gamma_i = -w_i r_i - h(q_i, i) \tag{9}$$

The terminal value for Γ_i is:

$$\Gamma_I = -\left[\sum_{i=1}^{I} w_i r_i + \sum_{i=1}^{I} h(q_i, i)\right] \geq -B \tag{10}$$

Using (9) and (10), we can restate the maximization problem as follows:

$$\underset{q_i, r_i}{Max} \sum_{i=1}^{I} G(T_i, v_i, r_i, i) \tag{11}$$

s.t. $$v_{i+1} - v_i = -\overline{q}_i + q_i \tag{12}$$

$$T_{i+1} - T_i = f(r_i, v_i, i) \tag{13}$$

$$\Gamma_{i+1} - \Gamma_i = -w_i r_i - h(q_i, i) \tag{14}$$

$$0 \leq q_i \leq \overline{q}_i,\ 0 \leq r_i \leq \overline{r}_i \tag{15}$$

$$v_1 = \overline{v}_1, T_1 = \overline{T}_1, v_I \geq \overline{v}_I \tag{16}$$

$$\Gamma_I \geq -B \tag{17}$$

Then, the Hamiltonian for this modified maximization problem is:

$$H = G(T_i, v_i, r_i, i) + \mu_i\left[-\overline{q}_i + q_i\right] + \eta_i f(r_i, v_i, i) - \lambda_i\left[w_i r_i + h(q_i, i)\right] + \alpha_i q_i + \beta_i\left[\overline{q}_i - q_i\right] + \gamma_i r_i + \delta_i\left[\overline{r}_i - r_i\right] \tag{18}$$

where μ_i, η_i and λ_i are the costate variables for discharge, water temperature, and budget, respectively. $\alpha_i, \beta_i, \gamma_i$ and δ_i are the Lagrangian multipliers associated with the constraints.

The necessary conditions are set out in the equations below:

$$\mu_i - \lambda_i h_{q_i} + \alpha_i - \beta_i = 0 \tag{19}$$

$$G_{r_i} + \eta_i f_{r_i} - \lambda_i w_i + \gamma_i - \delta_i = 0 \tag{20}$$

$$\mu_{i+1} - \mu_i = -\left[G_{v_i} + \eta_i f_{v_i}\right] \tag{21}$$

$$\eta_{i+1} - \eta_i = -G_{T_i} \tag{22}$$

$$\mu_I \geq 0, v_I \geq \bar{v}_I \text{ , } (v_I - \bar{v}_I)\mu_I = 0 \tag{23}$$

$$\eta_I = 0 \tag{24}$$

Equation (23) and (24) define the transversality conditions for discharge and water temperature, respectively. For this problem, the following Kuhn-Tucker conditions also need to be satisfied:

$$\alpha_i q_i = 0, \ \beta_i\left[\bar{q}_i - q_i\right] = 0 \tag{25}$$

$$\gamma_i r_i = 0, \ \delta_i\left[\bar{r}_i - r_i\right] = 0 \tag{26}$$

$$\lambda_I \geq 0, \ \lambda_I\left[\Gamma_I + B\right] = 0 \tag{27}$$

Equation (27) is associated with the budget constraint of the conservation efforts. Since budget is always scarce, it is reasonable to assume $\lambda_I > 0$, and therefore, we must have $\Gamma_I = -B$. This implies that all the budget is consumed between reaches *1* and *I*.

In equation (19), μ_i is the shadow price of discharge, which is positive and equal to the marginal benefit of an increase in discharge. λ_i is the shadow price of the budget constraint and is positive. h_{q_i} is the marginal loss in agricultural benefits when an additional unit of irrigation water is reduced from agricultural uses. The value of h_{q_i} depends on q, and it increases as q increases (by assumption, $h_q > 0$, $h_{qq} \geq 0$). In the case of an interior solution ($\mu_i = \lambda_i h_{q_i}$), the marginal benefit of instream flow increase (μ_i) must be equal to the marginal social cost of a reduction in agricultural water use ($\lambda_i h_{q_i}$).

If $\mu_i > \lambda_i h_{q_i}$, then equation (19) indicates $\mu_i - \lambda_i h_{q_i} + \alpha_i = \beta_i > 0$. The Kuhn-Tucker condition (25) implies $\overline{q}_i = q_i$, which means when the marginal benefit of an increase in discharge is greater than the marginal decrease in agricultural benefit, flow augmentation will be made to its maximum ($\overline{q}_i$). On the other hand, if $\mu_i < \lambda_i h_{q_i}$, then $\lambda_i h_{q_i} - \mu_i + \beta_i = \alpha_i > 0$, then the Kuhn-Tucker conditions imply $q_i = 0$. In this case, no flow augmentation efforts will be made. Only if $\mu_i = \lambda_i h_{q_i}$, then $\alpha_i = \beta_i = 0$, and we have an interior solution ($0 < q_i < \overline{q}_i$).

The optimal path of μ_i is given by the equation of motion (21). G_{v_i} is the marginal effect of an increase in discharge on the value of instream flow. f_{v_i} is the marginal decrease in water temperature resulting from an increase in discharge, and η_i is the shadow price of water temperature. So $\eta_i f_{v_i}$ represents the marginal benefit (of a temperature reduction) of an increase in discharge. Then, $G_{v_i} + \eta_i f_{v_i}$ represents the sum of direct and indirect (through temperature changes) marginal benefits of water conservation efforts. Since G_{v_i} is positive and

both η_i and f_{v_i} are negative, it is always the case that $\mu_{i+1} - \mu_i < 0$, implying that as one moves downstream the shadow price of discharge should decrease at the rate at which an increase in discharge is contributing to the overall benefit of instream flow.[15] The reason why the shadow price of discharge will be higher in the upstream area is that a decrease in discharge in the upstream area also decreases discharge in the downstream area, and therefore it has a higher opportunity cost. If $\mu_i > \lambda_i h_{q_i}$ holds in the upstream area, then we must have $q_i^* = \bar{q}_i$. Assuming that $\lambda_i h_{q_i}$ is constant along the stream, since μ_i will decrease as one moves downstream, at a certain reach (i_1^q) the value of μ_i becomes equal to $\lambda_i h_{q_i}\big|_{q=\bar{q}}$. At the reaches between i_1^q and i_2^q, the optimal water allocation requires that $0 < q_i^* < \bar{q}_i$. For the reaches below i_2^q, we have $\mu_i < \lambda_i h_{q_i}\big|_{q=0}$, and no water conservation efforts will be undertaken. This analysis is summarized graphically by Figure 3.1. The optimal pattern of flow augmentation along a stream is shown in the lower portion of Figure 3.1 by the solid bold line.

[15] It is assumed that discharge is large compared to the volume of water withdrawal (q_i), and therefore the values of G_{v_i} and f_{v_i} are not affected by water withdrawal and flow augmentation efforts ($\bar{q}_i$).

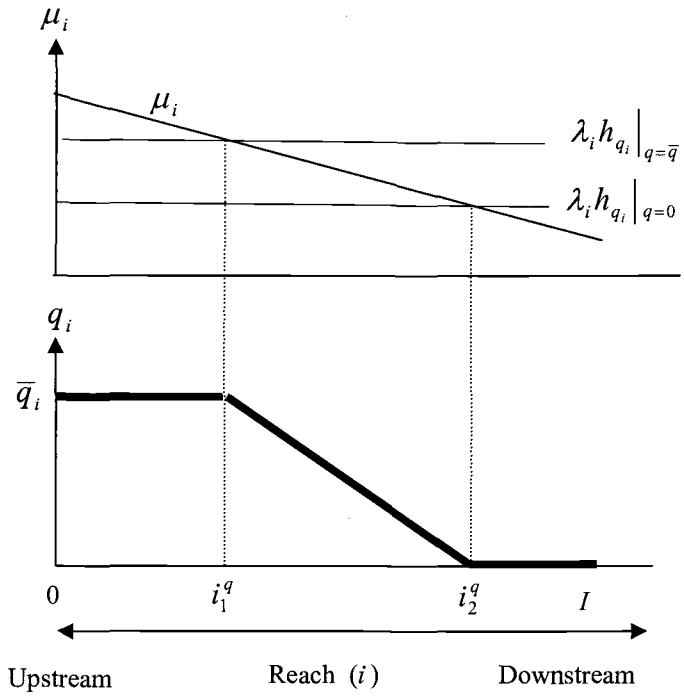

Figure 3.1: An optimal pattern of flow augmentation along a stream

Figure 3.1 implies that if the cost of flow augmentation efforts decreases by, for example, development of irrigation technologies, then the line $\lambda_i h_{q_i}\big|_{q=\bar{q}}$ shifts downward, and more conservation activities will be implemented along the stream. This is because development of irrigation technology decreases the marginal loss in agricultural benefits resulting from water conservation, while its marginal benefit in the value of instream flow remains the same. Similarly, where an increase in discharge has a significant effect on the marginal benefit of instream flow (G_v is large), then $\mu_{i+1} - \mu_i$ is large, and the line μ_i becomes steeper. In this case, an efficient policy is to increase flow augmentation efforts.

Similar analyses can be made with respect to the allocation of riparian restoration activities along the stream. In equation (20), G_{r_i} is the marginal benefit of riparian restoration

activities. f_{r_i} is the marginal effect of riparian restoration efforts on water temperature. η_i is the shadow price of water temperature. Thus, $\eta_i f_{r_i}$ represents the marginal benefit (of a temperature reduction) of riparian restoration efforts. Then, $G_{r_i} + \eta_i f_{r_i}$ represents the sum of direct and indirect (through temperature changes) marginal benefits of riparian restoration efforts. On the other hand, $\lambda_i w_i$ is the marginal social cost of riparian restoration efforts. In the case of an interior solution, the sum of direct and indirect marginal benefits of riparian restoration efforts ($G_{r_i} + \eta_i f_{r_i}$) must be equal to the marginal social cost of riparian restoration efforts ($\lambda_i w_i$).

The value of $G_{r_i} + \eta_i f_{r_i}$ varies depending on the level of r_i. If we assume that the value of instream flow is concave with respect to riparian restoration efforts ($G_{rr} < 0$), then $G_{r_i} + \eta_i f_{r_i}\big|_{r=0}$ is larger than $G_{r_i} + \eta_i f_{r_i}\big|_{r=\bar{r}}$ because $G_{r_i}\big|_{r=0} > G_{r_i}\big|_{r=\bar{r}}$. If $G_{r_i} + \eta_i f_{r_i} > \lambda_i w_i$, then $G_{r_i} + \eta_i f_{r_i} - \lambda_i w_i + \gamma_i = \delta_i > 0$. The Kuhn-Tucker condition (26) implies $r_i = \bar{r}_i$. This indicates that when the marginal benefit of riparian restoration is greater than the marginal social cost of that effort, then the restoration investment will be implemented to its maximum level. On the other hand, if $G_{r_i} + \eta_i f_{r_i} < \lambda_i w_i$, then $r_i = 0$, and no restoration activity will be undertaken. Only if $G_{r_i} + \eta_i f_{r_i} = \lambda_i w_i$, will there be an interior solution, $0 < r_i < \bar{r}_i$.

The optimal path of η_i is determined by equation (22). G_{T_i} is the marginal loss in stream flow benefits resulting from an increase in water temperature. Since G_{T_i} is negative, $\eta_{i+1} - \eta_i$ is always positive, implying that the absolute level of the shadow price of water

temperature (η_i) should decrease as one moves downstream. This is because a temperature increase in the upstream area also affects the temperature in the downstream area, and therefore, a temperature increase in the upstream areas has a higher opportunity cost than in the downstream area. The fact that the absolute level of η_i should decrease indicates that $G_{r_i} + \eta_i f_{r_i}$ becomes smaller as one moves downstream (note that both η_i and f_{r_i} are negative).[16] If $G_{r_i} + \eta_i f_{r_i} \big|_{r=\bar{r}} > \lambda_i w_i$ holds in the upstream area, then we must have $r_i^* = \bar{r}_i$. However, since $G_{r_i} + \eta_i f_{r_i} \big|_{r=\bar{r}}$ decreases as one moves downstream, at a certain reach (i_1^r), the value of $G_{r_i} + \eta_i f_{r_i} \big|_{r=\bar{r}}$ becomes equal to $\lambda_i w_i$. At the reaches between i_1^r and i_2^r, $G_{r_i} + \eta_i f_{r_i} = \lambda_i w_i$ and the optimal allocation of riparian restoration is $0 < r_i^* < \bar{r}_i$. For any reaches below i_2^r, we have $G_{r_i} + \eta_i f_{r_i} \big|_{r=0} < \lambda_i w_i$, and no riparian restoration efforts will be implemented ($r_i^* = 0$). These analyses are summarized in Figure 3.2. The optimal allocation of riparian restoration efforts is shown by the solid bold line in the lower portion of Figure 3.2.

[16] Again, it is assumed that discharge is large compared to the volume of water withdrawal (q_i), and therefore the values of G_{r_i} and f_{r_i} are not affected throughout the study stream by water withdrawal. Otherwise, it is possible that $G_{r_i} + \eta_i f_{r_i}$ becomes larger as one moves downstream. In this case, riparian restoration efforts may be implemented only in the lower stretches of the stream.

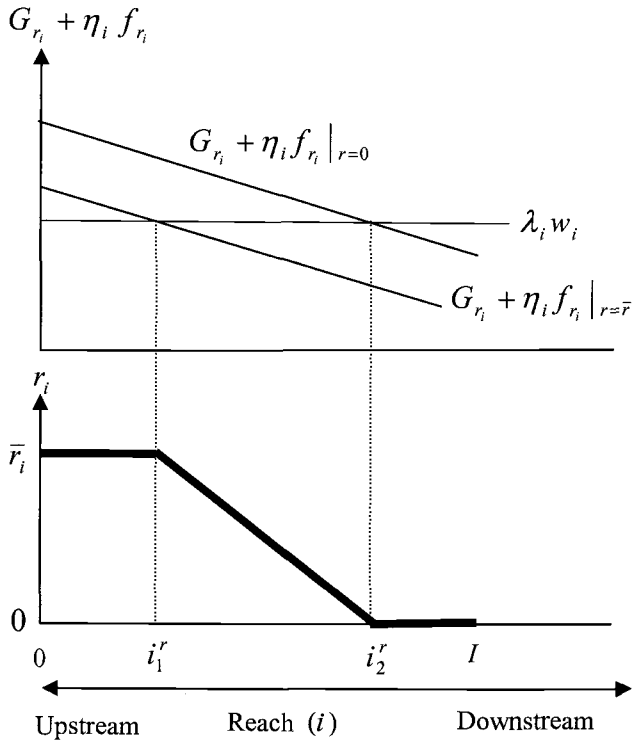

Figure 3.2: An optimal allocation of riparian restoration efforts

Figure 3.2 implies that if the cost of restoration efforts decreases, then the line $\lambda_i w_i$ shifts downward, and more restoration activities will be implemented. The intuition behind this result is that while the cost of restoration efforts has been decreased, its marginal benefit has been unaffected. Thus, there is a net benefit due to this change, and more restoration efforts will be implemented. These analyses on the optimal allocation of flow augmentation and riparian restoration activities lead to Proposition 1.

Proposition 1: If there is no tributary inflow, then more water will be left in stream (less water will be diverted to out of stream uses such as agriculture) and more riparian restoration activities will be implemented in upstream areas than in downstream areas.

In the above analysis, it was assumed that the cost of riparian restoration is the same at any point along the stream. However, this assumption may not hold since, in general, agricultural land is more productive in downstream areas (due to better soils and/or the proximity to urban areas), and therefore it would be more costly to implement restoration efforts. Then, $\lambda_i w_i$ has an upward slope as is shown in Figure 3.3. In this case, it can be noted that i_1^r and i_2^r shift upstream, and riparian restoration efforts will be concentrated in a shorter stretch of the stream. Likewise, an increase in the cost of riparian restoration efforts such as fencing also shift $\lambda_i w_i$ upward, which results in a lower level of restoration activities.

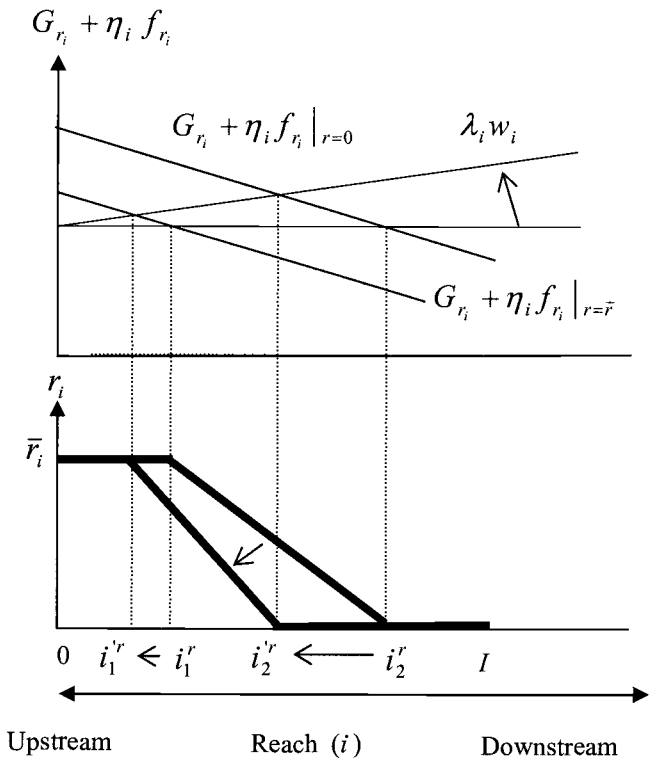

Figure 3.3: An optimal allocation of riparian restoration efforts when restoration costs increase as one moves downstream

Thus far, we have assumed that the benefit of instream flow is concave with respect to riparian restoration ($G_{rr} < 0$). However, it is possible that the relationship between riparian conditions and the benefit of instream flow is convex ($G_{rr} > 0$). For example, Wu *et al.* (2000) found that the stability of stream banks has a threshold effect on the benefit of instream flow. In this case, $G_{r_i} + \eta_i f_{r_i}|_{r=\bar{r}}$ may be larger than $G_{r_i} + \eta_i f_{r_i}\Big|_{r=0}$ because $G_{r_i}|_{r=\bar{r}} > G_{r_i}|_{r=0}$, and we have a corner solution: an optimal level of riparian restoration activity is either at the maximum level of restoration or zero. An optimal allocation of riparian restoration will then be modified as is shown in Figure 3.4:

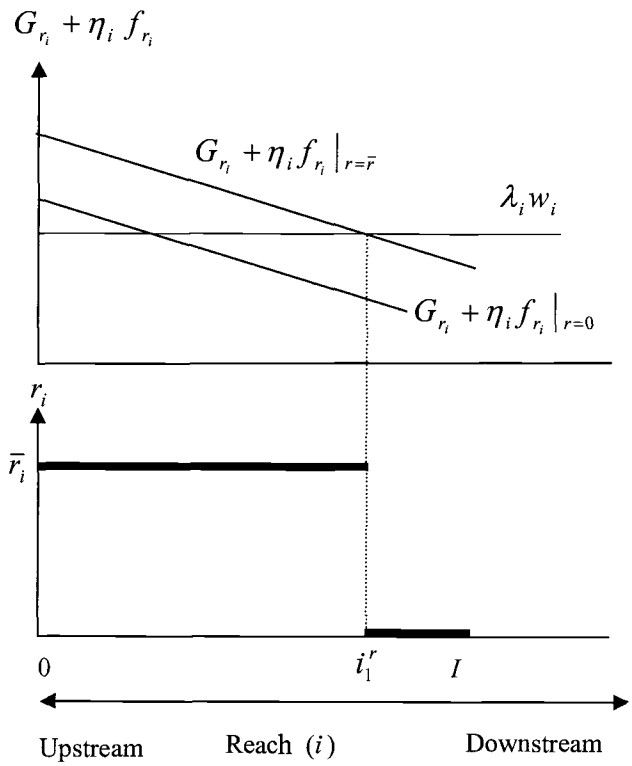

Figure 3.4: An optimal allocation of riparian restoration efforts in the presence of a threshold effect

As the Figure 3.4 shows, an optimal level of riparian restoration in the case of convex relationship is either at the maximum level of restoration or zero.

3.2.3 The effect of the budget constraint

Optimal levels of flow augmentation and riparian restoration efforts are, in practice, linked through the budget constraint. How these two conservation activities are allocated under a budget constraint is examined in this section. Equations (19) and (20) determine the optimal

allocation of the budget between these two conservation practices. Solving the equations for λ_i yields the following condition:

$$\frac{\mu_i + \alpha_i - \beta_i}{h_{q_i}} = \frac{G_{r_i} + \eta_i f_{r_i} + \gamma_i - \delta_i}{w_i} = \lambda_i \tag{28}$$

Equation (28) indicates that both water conservation and riparian restoration activities will be implemented until the marginal benefit of each activity becomes equal to the shadow price of the budget constraint. Figure 3.5 shows how each of the conservation activities is implemented along a stream.

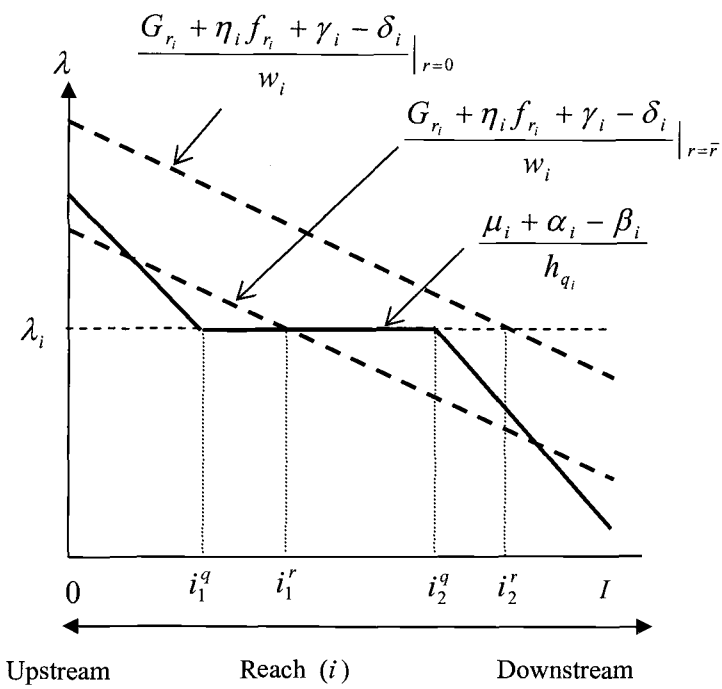

Figure 3.5: Optimal allocation of conservation practices under a budget constraint

Figure 3.5 has a similar interpretation to Figures 3.1 and 3.2. Specifically, with respect to flow augmentation, $q_i^* = \bar{q}_i$ between reaches *1* and i_1^q, $0 \le q_i^* \le \bar{q}_i$ between i_1^q and i_2^q, and $q_i^* = 0$ between i_2^q and *I*. With regard to riparian restoration, $r_i^* = \bar{r}_i$ between reaches *1* and i_1^r, $0 \le r_i^* \le \bar{r}_i$ between i_1^r and i_2^r, and $r_i^* = 0$ between i_2^r and *I*.

If the cost of riparian restoration efforts (w_i) increases, or the value of $G_{r_i} + \eta_i f_{r_i}$ decreases, then both $\left.\frac{G_{r_i} + \eta_i f_{r_i} + \gamma_i - \delta_i}{w_i}\right|_{r=\bar{r}}$ and $\left.\frac{G_{r_i} + \eta_i f_{r_i} + \gamma_i - \delta_i}{w_i}\right|_{r=0}$ shift downward, while $\frac{\mu_i + \alpha_i - \beta_i}{h_{q_i}}$ remain the same. Figure 3.6 shows an optimal allocation of the two conservation practices under this case, where the stretch in which riparian restoration efforts are implemented has now been reduced.

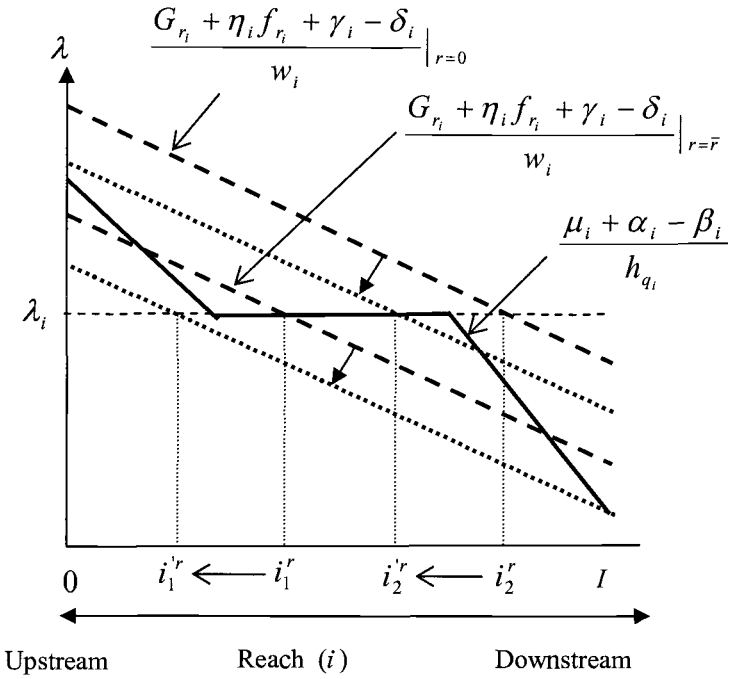

Figure 3.6: Optimal allocation of conservation practices under a budget constraint when the cost of riparian restoration increases

Figure 3.6 implies that at a stream segment where riparian restoration is costly (*i.e.* stream banks have been severely degraded or are used for non-agricultural land uses) or riparian restoration is not effective in enhancing the benefit of instream flow ($G_{r_i} + \eta_i f_{r_i}$ is low) because, for example, riparian conditions are not the limiting factor for the benefit of instream flow, riparian restoration efforts may not be an appropriate conservation practice. In this case, flow augmentation will instead be pursued.

On the other hand, if the cost of water conservation efforts increases (by, for example, an increase in the price of agricultural products), then $\frac{\mu_i + \alpha_i - \beta_i}{h_{q_i}}$ shifts downward, while

$\frac{G_{r_i} + \eta_i f_{r_i} + \gamma_i - \delta_i}{w_i}\Big|_{r=\bar{r}}$ and $\frac{G_{r_i} + \eta_i f_{r_i} + \gamma_i - \delta_i}{w_i}\Big|_{r=0}$ are not affected. This case is shown in Figure 3.7, where water conservation efforts will now be reduced and concentrated in a shorter stretch of the upstream area.

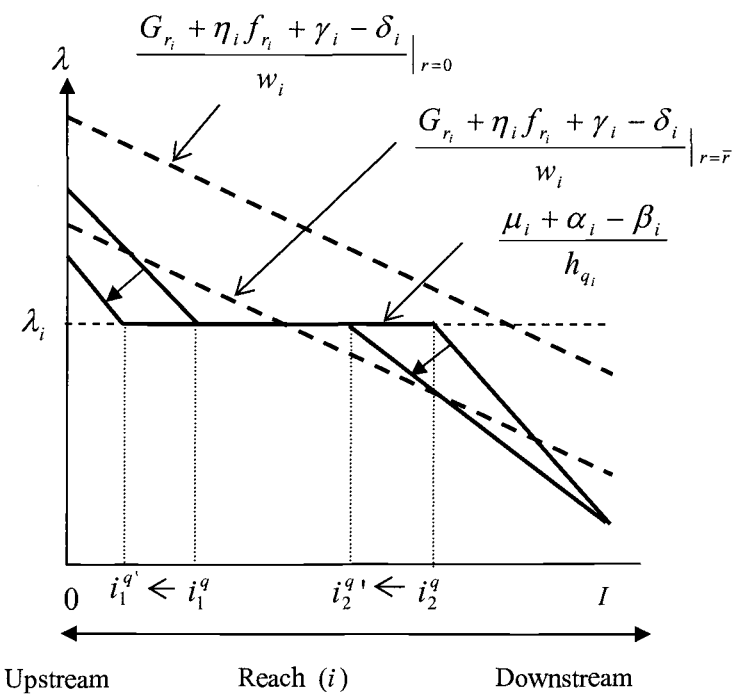

Figure 3.7: Optimal allocation of conservation practices under the budget constraint when the cost of water conservation increases

Figure 3.7 implies that at a stream segment where water conservation is costly (*i.e.* agriculture is producing high-valued crops or state-of-the-art irrigation technology is already employed) or discharge is not the limiting factor, then water conservation efforts might not be effective as a conservation practice.

If the budget is reduced, the shadow price (λ_i) will shift upward. Then, both water conservation efforts and riparian restoration activities will be reduced. Their distribution will be concentrated in a shorter stretch of the upstream area where the marginal social benefits of conservation practices are high. The intuition is that a budget decline and an increase in the shadow price of the budget require that any activity should yield a higher marginal social benefit. These analyses lead to Proposition 2.

Proposition 2: Under a budget constraint, only those conservation practices whose marginal benefits are greater than or equal to the marginal social cost of the constraint are implemented. Flow augmentation efforts are emphasized when the marginal benefit of agricultural water use is low and / or the marginal benefit of an increase in discharge is large. On the other hand, riparian restoration is the choice if the cost of riparian restoration is low and / or the marginal benefit of an improvement in riparian conditions is large. A budget decline results in a reduction of both conservation activities.

3.3 The Case of a tributary

3.3.1 The effect of a change in initial discharge or water temperature

In order to gain insights on the effects of a tributary on the efficient allocation of conservation practices, the effects of a change in initial discharge and water temperature on the allocation of conservation efforts are now examined. An increase in initial discharge occurs when water withdrawal above the study stream decreases or an improvement in riparian conditions above the study stream results in an increase in stream flow (*i.e.* riparian vegetation improvement may lead to an increase in hyporheic inflow). The latter case may also result in a decrease in initial water temperature. Conversely, clear cutting of forests in the source areas

above the study stream may result in an increase in initial water temperature. In this section, the effects of an increase in initial discharge and a decrease in water temperature are examined separately.

(1) Change in initial discharge

First, assume that the initial discharge has been increased from $\bar{v}_1^1$ to $\bar{v}_1^2$ ($\bar{v}_1^1 < \bar{v}_1^2$). From equation (21), a change in the shadow price of discharge should follow $\mu_{i+1} - \mu_i = -\left[G_{v_i} + \eta_i f_{v_i}\right]$. If the marginal value of instream flow decreases as the flow increases ($G_{vv} < 0$), then G_{v_i} becomes smaller with an increase in initial discharge ($G_{v_i}\big|_{v_1=\bar{v}_1^1} > G_{v_i}\big|_{v_1=\bar{v}_1^2}$). In addition, the absolute value of f_{v_i} also becomes smaller ($\left|f_{v_i}\big|_{v_1=\bar{v}_1^1}\right| > \left|f_{v_i}\big|_{v_1=\bar{v}_1^2}\right|$)[17]. Since both G_{v_i} and the absolute value of f_{v_i} become smaller, the absolute level of $\mu_{i+1} - \mu_i$ becomes smaller. Given the transversality condition specified by equation (23), the optimal path of μ_i then becomes flatter, as is shown in Figure 3.8.

[17] From the Brown equation (1), we have $f_{vv} > 0$

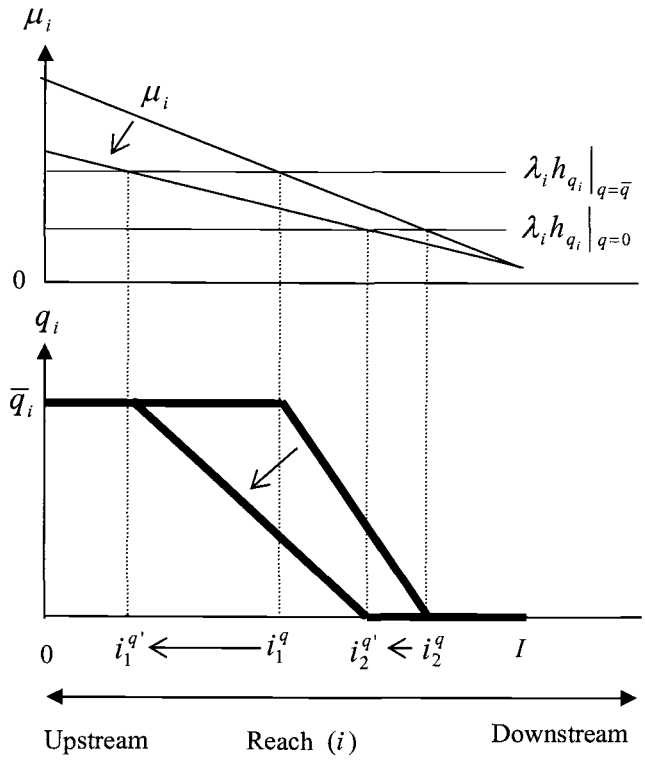

Figure 3.8: The optimal pattern of flow augmentation when initial discharge increases ($G_{vv} < 0$)

Figure 3.8 shows that with an increase in initial discharge, flow augmentation efforts will be concentrated in a shorter stretch of the upstream area. The intuition behind this result is that while the marginal benefit of an increase in stream flow decreases due to an increase in initial discharge, the marginal cost of flow augmentation is not changed. Therefore, water conservation efforts will be reduced. On the other hand, if the initial discharge has been decreased, then G_{v_i} and the absolute value of f_{v_i} increases. Thus, μ_i becomes steeper, and water conservation efforts will be implemented in a longer stretch of the stream.

It is possible that the marginal value of instream flow increases as discharge increases ($G_{vv} > 0$) when discharge is the primary limiting factor for the benefits of instream flow. Then G_{v_i} becomes larger with an increase in initial discharge. In this case, the direction of a change in $\mu_{i+1} - \mu_i$ is indeterminate because the absolute value of f_{v_i} still becomes smaller, and G_{v_i} and $\eta_i f_{v_i}$ move in opposite directions. This is also the case when the initial discharge is decreased.

The second analysis is of the impact of an increase in initial discharge on the allocation of riparian restoration activities. By equation (20), the optimal allocation of riparian restoration efforts is determined by the path of $G_{r_i} + \eta_i f_{r_i}$. Assume that the marginal effect of riparian restoration activities on the value of instream flow decreases as the volume of stream flow increases ($G_{rv} < 0$). Then, G_{r_i} decreases as the initial discharge increases ($G_{r_i}\big|_{v_1=\bar{v}_1^1} > G_{r_i}\big|_{v_1=\bar{v}_1^2}$). The absolute value of f_{r_i} also decreases ($\left|f_{r_i}\big|_{v_1=\bar{v}_1^1}\right| > \left|f_{r_i}\big|_{v_1=\bar{v}_1^2}\right|$) because, with a larger discharge, riparian conditions such as shading are less effective in decreasing water temperatures. In this case, the value of $G_{r_i} + \eta_i f_{r_i}$ becomes smaller. The new path of $G_{r_i} + \eta_i f_{r_i}$ is shown by a solid line in the upper portion of Figure 3.9.

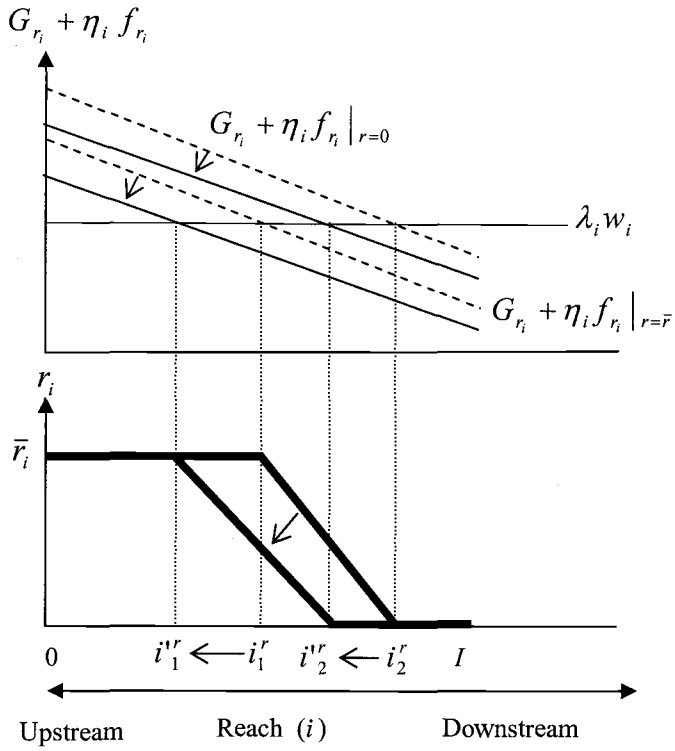

Figure 3.9: An optimal allocation of riparian restoration efforts when initial discharge increases ($G_{rv} < 0$)

Figure 3.9 shows that when initial discharge increases, riparian restoration activities will be decreased. This is because an increase in stream flow makes riparian restoration less effective while the costs of such activities remain the same. With a decrease in initial discharge, the absolute values of both G_{r_i} and f_{r_i} increase, which makes the value of $G_{r_i} + \eta_i f_{r_i}$ larger. In this case, riparian restoration activities will be implemented in a longer stretch of the stream.

On the contrary, if an increase in stream flow enhances the effectiveness of riparian restoration ($G_{rv} > 0$), then the result becomes indeterminate. This is because in the case of an initial water increase (decrease), G_{r_i} increases (decreases) while the absolute value of f_{r_i}

decreases (increases). The two terms move in opposite directions, and the net effect is determined by their relative magnitudes.

(2) Change in initial water temperature

Next, assume that the initial water temperature decreases from $\overline{T}_1^1$ to $\overline{T}_1^2$ ($\overline{T}_1^1 > \overline{T}_1^2$). Again the optimal allocation of flow augmentation efforts is determined by the relative magnitude of μ_i and $\lambda_i h_{q_i}$, and that μ_i should follow $\mu_{i+1} - \mu_i = -\left[G_{v_i} + \eta_i f_{v_i}\right]$. We assume that the value of instream flow is concave with respect to water temperature ($G_{TT} < 0$). Then, with a decrease in initial water temperature, the marginal benefit of an increase in discharge decreases ($G_{v_i}\big|_{T_1=\overline{T}_1^1} > G_{v_i}\big|_{T_1=\overline{T}_1^2}$). Since f_{v_i} is not affected by the levels of water temperature ($f_{v_i}\big|_{T_1=\overline{T}_1^2} = f_{v_i}\big|_{T_1=\overline{T}_1^2}$), the absolute level of $\mu_{i+1} - \mu_i$ becomes smaller, and the optimal path of μ_i becomes flatter. Then, flow augmentation efforts will be implemented in a shorter stretch of the stream. Figure 3.8 is applicable to this case. The intuition behind this result is that if water temperature decreases, then the marginal effect of an increase in discharge on the benefit of instream flow decreases, while the cost of water conservation is not changed. Therefore, water conservation is now more costly and will be reduced. On the other hand, if the initial water temperature increases, then the absolute level of $\mu_{i+1} - \mu_i$ becomes larger and the optimal path of μ_i becomes steeper. In this case, water conservation will be implemented in a longer stretch of the stream.

It is also plausible that as the water temperature decreases, the marginal benefit of instream flow increases ($G_{TT} > 0$). In this case, the absolute level of $\mu_{i+1} - \mu_i$ becomes larger

and the optimal path of μ_i becomes steeper. Then flow augmentation efforts will be implemented in a longer stretch of the stream in order to take an advantage of a decrease in initial water temperature. On the other hand, if the initial water temperature increases, then the absolute level of $\mu_{i+1} - \mu_i$ becomes smaller and the optimal path of μ_i becomes flatter, which will result in a lower level of flow augmentation efforts.

Finally, the impact of a decrease in initial water temperature on the allocation of restoration efforts is analyzed. As before, the optimal allocation of restoration efforts is determined by the path of $G_{r_i} + \eta_i f_{r_i}$, and a change in η_i should follow $\eta_{i+1} - \eta_i = -G_{T_i}$. We assume that the benefit of instream flow is concave with respect to water temperature ($G_{TT} < 0$). Then, given a decrease in water temperature, the absolute value of G_T decreases. In this case, the value of $\eta_{i+1} - \eta_i$ is reduced, and the optimal path of η_i becomes flatter, which decreases the value of $G_{r_i} + \eta_i f_{r_i}$. Figure 3.9 is applicable to this case. The intuition behind this result is that as water temperature decreases, the marginal effects of riparian restoration on stream flow benefits is decreased due to the assumption ($G_{TT} < 0$), while its cost has not changed. Thus, riparian restoration is more costly and will be reduced. On the other hand, if the initial water temperature increases, then $\eta_{i+1} - \eta_i$ becomes larger, and $G_{r_i} + \eta_i f_{r_i}$ increases. In this case, riparian restoration efforts will be implemented in a longer stretch of the stream.

Conversely, if the benefit of instream flow is convex with respect to water temperature ($G_{TT} > 0$), a decrease (an increase) in initial water temperature increases (decreases) $G_{r_i} + \eta_i f_{r_i}$ because the value of $\eta_{i+1} - \eta_i$ is increased (decreased). Then more (less) riparian restoration activities will be implemented.

3.3.2 Efficient allocation of conservation efforts when there is a tributary

An efficient allocation of conservation practices when there is a tributary inflow can be examined as an analogy to a change in initial discharge and water temperature. With a tributary, the mainstem below the confluence always has the largest discharge, followed by the mainstem above the confluence, and then by the tributary. But the temperature may either increase or decrease depending on the temperature of the tributary. The water temperature immediately below the confluence is the (discharge) weighted average of water temperatures in the mainstem and the tributary immediately above the confluence (Brown, 1970).

We examine the optimal allocation of conservation practices between the three reaches (above and below the confluence in the mainstem and the tributary). In the following discussion, we will refer to the reaches above and below the confluence in the mainstem as Reach 1 and Reach 2, and the tributary to Reach 3, respectively. The analysis in the previous section showed that when the discharge has a threshold effect with respect to instream flow benefits ($G_{vv} > 0$) or when the marginal effect of riparian restoration is enhanced as the discharge increases ($G_{rv} > 0$), then an optimal allocation of conservation activities are indeterminate. Thus, in this section, we always assume $G_{vv} < 0$ and $G_{rv} < 0$. The above analyses indicate that the priority of conservation activities between the three reaches is the same for both flow augmentation effort and riparian restoration activities. Therefore, the following analysis, which introduces the role of the tributary, focuses on the case of riparian restoration activities only.

(1) Tributary has the same temperature as the mainstem

All the three reaches have the same temperature, and an efficient allocation of restoration activities is solely determined by discharge. $G_{rv} < 0$ implies that riparian restoration activities will be allocated to those reaches that have the least discharge. Therefore, a priority will be given to Reach 3, followed by Reach 1, and then Reach 2. Thus, when the tributary has the same temperature as the mainstem, then riparian restoration activities will target the tributary first, followed by the reach of the mainstem upstream of the confluence.

Proposition 3: When the temperature of the tributary is close to that of the mainstem, then the tributary will be the target of conservation activities.

(2) Tributary is colder than the mainstem

First, we assume that the value of instream flow is concave with respect to water temperatures ($G_{TT} < 0$). It indicates that riparian restoration is the most effective where water temperature is the highest (Reach 1). On the other hand, $G_{rv} < 0$ implies that restoration effort is the most effective in a reach with the least discharge (Reach 3). Thus, in this case, an efficient allocation between the three reaches is indeterminate. However, if the discharge of the tributary (Reach 3) is as large as the mainstem above the confluence (Reach 1), the marginal effect of discharge on the stream flow benefit is almost the same between Reach 1 and Reach 3. Then, the priority is primarily determined by the temperature, and Reach 1 will be the first priority.

Conversely, if water temperature has a threshold effect with respect to instream flow benefits ($G_{TT} > 0$), priority for riparian restoration efforts will be given to those streams with the lowest water temperatures. Then, Reach 3 will be the priority, followed by Reach 2, and

then Reach 1. In this case, regardless of the levels of discharge in the tributary, riparian restoration efforts will target the tributary (Reach 3) first.

Proposition 4 (Tributary is colder than the mainstem): If water temperature has a threshold effect with respect to instream flow benefits, then the tributary is the target of conservation efforts. On the other hand, if the value of instream flow is concave with respect to water temperatures, then the optimal allocation of conservation activities is indeterminate unless the tributary is as large as the mainstem, in which case the mainstem above the confluence is the priority.

(3) The mainstem is colder than tributary

First we assume $G_{TT} < 0$. It implies that riparian restoration efforts will be the most effective where water temperature is the highest (Reach 3). $G_{rv} < 0$ also indicates that restoration efforts will be implemented in a reach with the least discharge (Reach 3). Both conditions imply that Reach 3 will be the first priority.

On the other hand, if temperature displays a threshold effect with respect to the value of instream flow ($G_{TT} > 0$), then the optimal solution is indeterminate because the discharge effect requires that Reach 3 be targeted first, while the temperature effect requires that Reach 1 be targeted first. However, it is likely that Reach 1 will be a priority if the discharge of the tributary (Reach 3) is as large as the mainstem above the confluence because the temperature effect exceeds the discharge effect.

Proposition 5 (Tributary is warmer than the mainstem): If the value of instream flow is concave with respect to water temperatures, then conservation efforts will target the tributary. Conversely, if water temperature has a threshold effect with respect to instream flow benefits, then the allocation of conservation activities is indeterminate. However, if the discharge of the tributary is as large as the mainstem, the mainstem above the confluence will be the priority.

3.4 Summary of the theoretical analyses

This section summarizes the results of the theoretical analyses performed in this chapter. Conservation practices considered are riparian restoration efforts such as fencing and planting as well as flow augmentation efforts such as water conservation and the lease or purchase of agricultural water. The analyses can be broadly divided into two cases. The first case deals with a stream without a tributary. Since streams almost always have tributary inflows, this first case may be applicable to a series of reaches of a given stream that have no tributary inflows. The second case is the analysis of a stream with tributaries.

A number of general insights are gained from the analyses. Where there are no tributary inflows, if stream characteristics are identical across reaches, it is always beneficial to have a greater level of riparian restoration efforts in the upstream area than in the downstream area. This is primarily because any water temperature decrease in the upstream area reduces water temperatures in the downstream area as well. Likewise, flow augmentation efforts will be targeted to the upstream area because an increase in stream flow in the upstream area augments stream flow in the downstream area as well.

If the reaches are not identical, these results need to be altered. In general, conservation practices will target reaches where the marginal benefits of such activities are high or the costs of doing such activities are low. In most real world watershed situations, for example, we expect

to have a more productive agricultural land in the downstream area, and as a result, riparian restoration efforts and stream flow augmentation are more costly. This situation reinforces the above conditions that more conservation activities be implemented in the upstream area.

When there are tributaries, the optimal allocation of conservation practices become more complex. If the benefits of instream flow are convex with respect to stream flow ($G_{vv} > 0$) or an increase in discharge increases the marginal effect of riparian restoration efforts on the stream flow benefits ($G_{rv} > 0$), the optimal allocation of conservation practices is indeterminate. The allocation would therefore be determined through an empirical or a simulation analysis with numerical values. Table 3.1 summarizes the efficient allocation of conservation practices when $G_{vv} < 0$ and $G_{rv} < 0$. The allocations shown in Table 3.1 apply to both flow augmentation efforts and riparian restoration activities.

Table 3.1: Efficient allocations of conservation practices when there is a tributary ($G_{vv} < 0$ and $G_{rv} < 0$)

Temperature in tributary relative to the mainstem	Relationship between stream flow benefit and water temperature	Priority reach	Condition
Same		Tributary	
Colder	Concave ($G_{TT}<0$)	Mainstem above the confluence	Tributary is as large as the mainstem
	Convex ($G_{TT}>0$)	Tributary	
Warmer	Concave ($G_{TT}<0$)	Tributary	
	Convex ($G_{TT}>0$)	Mainstem above the confluence	Tributary is as large as the mainstem

Table 3.1 shows that the efficient allocation is primarily determined by the temperature levels in the tributary relative to that of the mainstem as well as the curvature of the relationship between the stream flow benefit and water temperature. The table also shows that under the assumptions of $G_{vv} < 0$ and $G_{rv} < 0$, the priority of conservation practices will always be the reaches in upstream areas (either the mainstem or tributary).

This analytical model abstracts from many of the complicated aspects of riparian characteristics, stream morphology and instream flow benefits in a Pacific Northwest watershed. However, it does incorporate the fundamental relationships between riparian vegetation, water temperature, discharge, and fishery benefits. As a result, this analytical model provides an important and useful framework with which to address the allocation of riparian restoration activities and water conservation efforts in a watershed. For some of these relationships, the optimal solution to the allocation of conservation practices is indeterminate. In addition, the relationships between the benefits of instream flow (fish abundance) and variables such as water temperature, riparian conditions, and discharge, are not stable. They vary depending on sections of a stream, life-stage of salmonid species, and the absolute levels of stream flow and water temperature. They are also likely to be different across years. This observation indicates that to gain insights on an optimal allocation of conservation practices, site-specific empirical models are necessary. Thus, in the subsequent chapters, simulation analyses are implemented for streams in the Upper Grande Ronde River basin in northeastern Oregon.

Chapter 4

Procedures

This chapter explains the methodology employed in developing a simulation model that embeds the key theoretical relationships identified in the preceding chapter and that reflects the physical and economic conditions in the Grande Ronde River basin. Because of the interdisciplinary nature of the study, multiple steps are taken in developing and implementing the simulation model. They are presented below.

4.1 Divide the study basin into reaches

The study basin is divided into 41 reaches. Figure 4.1 shows a schematic of these 41 reaches. The numbers in Figure 4.1 are used to identify stream segments with multiple reaches. The division of the study basin into reaches is based on stream orders, geomorphologic characteristics and land ownership patterns. For example, each stream is divided into reaches at each confluence. The division of reaches in Sheep Creek and the UGR mainstem follow differences in stream morphology (Personal communication with Joe Ebersole). For example, reaches 7 and 8 in the UGR mainstem are distinguished from other reaches on the basis of alluvial characteristics. These divisions of study streams frequently match the borders of landownership patterns (public and private), given that the upland areas are typically Forest Service lands, while the valleys (which tend to be alluvial) are in private ownerships. The identification of these reaches is provided in Table 4.1.

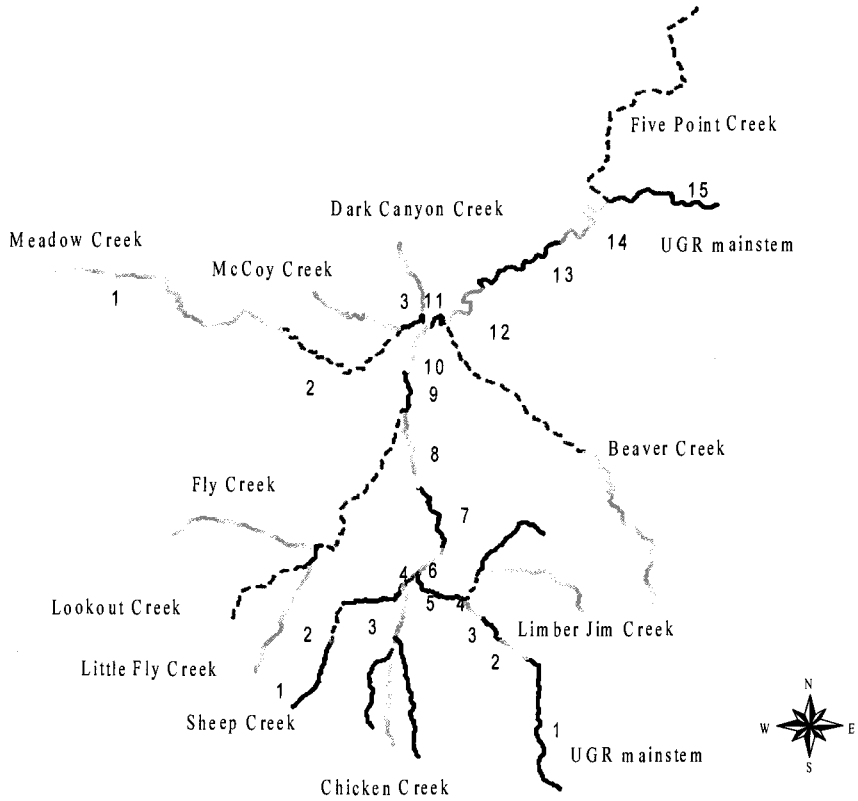

Figure 4.1: The Upper Grande Ronde River basin and reaches

The mainstem of the Upper Grande Ronde (UGR) River flows northward starting in reach "UGR mainstem 1" and then eastward to reach "UGR mainstem 15". In general, reaches in the lower (southern) part of the map occur in higher elevations. The riparian zone in each reach is further divided into 10 vegetation/land use types, which are shown in Table 4.1. The riparian vegetation data were collected by the USEPA, and the division of each reach into sub-reaches follows their classification.[18] We will refer to these vegetation / land use types as sub-reaches.

[18] This division follows the land cover classification system described in a memorandum "Land Cover Classification for the Upper Grande Ronde" by D.J. Norton (1993).

Table 4.1: Riparian vegetation / land use types in each reach

Reach	Length	AG	EM	SS	HU	Forest (Height, m)					Others
	(mile)					6-12	12-18	18-24	24-30	30-	
UGR mainstem 1	6.61	0%	3%	2%	0%	0%	0%	18%	74%	0%	3%
UGR mainstem 2	1.51	0%	0%	36%	8%	0%	0%	17%	9%	0%	30%
UGR mainstem 3	1.68	0%	0%	3%	17%	0%	0%	27%	49%	0%	4%
UGR mainstem 4	2.20	0%	7%	4%	29%	0%	0%	0%	56%	0%	3%
UGR mainstem 5	3.37	49%	26%	1%	8%	0%	0%	0%	15%	0%	0%
UGR mainstem 6	2.20	31%	50%	2%	4%	0%	1%	1%	1%	0%	10%
UGR mainstem 7	3.71	0%	2%	24%	17%	2%	0%	1%	40%	0%	14%
UGR mainstem 8	4.02	0%	0%	35%	4%	0%	0%	0%	53%	2%	6%
UGR mainstem 9	2.09	0%	1%	28%	9%	6%	0%	3%	19%	0%	35%
UGR mainstem 10	2.72	18%	2%	22%	5%	0%	2%	2%	8%	0%	42%
UGR mainstem 11	1.52	0%	0%	9%	18%	0%	0%	0%	25%	0%	48%
UGR mainstem 12	3.15	0%	0%	13%	17%	0%	0%	0%	14%	0%	56%
UGR mainstem 13	5.41	13%	9%	4%	10%	0%	0%	0%	17%	0%	47%
UGR mainstem 14	4.47	2%	1%	17%	10%	0%	1%	1%	12%	0%	56%
UGR mainstem 15	5.64	0%	0%	2%	6%	0%	6%	0%	13%	0%	73%
Limber Jim Cr. Source	5.93	0%	5%	3%	4%	0%	0%	23%	65%	0%	0%
Limber Jim Cr. Mouth	2.38	0%	1%	4%	79%	4%	0%	6%	6%	0%	0%
Limber Jim N.Fk. Cr.	4.09	0%	0%	2%	6%	0%	0%	11%	79%	0%	2%
Sheep Cr. 1	4.00	0%	3%	1%	36%	0%	0%	5%	55%	0%	0%
Sheep Cr. 2	2.56	77%	8%	2%	1%	0%	4%	7%	1%	0%	0%
Sheep Cr. 3	5.35	61%	35%	2%	1%	0%	0%	0%	1%	0%	1%
Sheep Cr. 4	1.53	2%	97%	0%	0%	0%	0%	0%	0%	0%	1%
Chicken Cr. Source	6.11	0%	1%	1%	18%	0%	1%	6%	72%	0%	1%
Chicken Cr. Mouth	3.51	42%	50%	1%	7%	0%	0%	0%	0%	0%	1%
West Chicken Cr. Source	4.62	0%	0%	0%	13%	0%	0%	11%	76%	0%	0%
West Chicken Cr. Mouth	0.94	0%	0%	0%	82%	0%	0%	14%	4%	0%	0%
W.West Chicken Cr.	4.11	0%	0%	0%	11%	0%	0%	9%	81%	0%	0%
Fly Cr. Source	8.34	24%	13%	2%	37%	0%	0%	15%	8%	0%	0%
Fly Cr. Mouth	9.22	2%	10%	21%	10%	2%	0%	3%	51%	0%	0%
Little Fly Cr. Source	6.06	18%	1%	2%	17%	1%	0%	27%	29%	0%	4%
Little Fly Cr. Mouth	1.09	67%	17%	0%	7%	0%	0%	9%	0%	0%	0%
Lookout Cr.	4.95	4%	2%	0%	9%	0%	0%	36%	49%	0%	0%
Meadow Cr. 1	14.05	0%	14%	1%	24%	0%	2%	27%	31%	0%	0%
Meadow Cr. 2	8.13	6%	1%	29%	33%	2%	0%	3%	23%	0%	3%
Meadow Cr. 3	1.45	26%	6%	25%	5%	0%	0%	3%	34%	0%	0%
Meadow Cr. 4	0.75	0%	0%	42%	1%	0%	0%	0%	47%	0%	10%
McCoy Cr.	4.98	28%	3%	17%	28%	4%	7%	2%	10%	0%	1%
Dark Canyon Cr.	3.90	0%	0%	39%	3%	0%	0%	1%	55%	2%	1%
Beaver Cr. Source	9.29	0%	1%	19%	16%	0%	0%	7%	45%	0%	10%
Beaver Cr. Mouth	9.71	1%	2%	10%	12%	0%	0%	1%	72%	0%	1%
Five Point Cr.	13.83	1%	0%	10%	22%	0%	3%	0%	63%	0%	0%

Source: ODEQ (2000)

Note: AG, Agriculture; EM, Emergent Vegetation[19]; SS, Scrub-Shrub; and HU, Herbaceous Uplands.

[19] Emergent vegetation primarily refers to wetland vegetation, and it consists of persistent, nonpersistent, narrow- and broad-leaved nonpersistent, and narrow-leaved persistent sub-classes (Norton, 1993).

The riparian vegetation data were collected by the USEPA in early 1990s. In the Grande Ronde River basin, riparian restoration activities have been implemented since the 1980s, but a rapid expansion in such activities took place in the mid-1990s.[20] In addition, there is a lag between the implementation of restoration activities and the effects on riparian vegetation. Therefore, it is assumed that riparian vegetation information in Table 4.1 reflects the conditions in the riparian zone before restoration activities are implemented.

Among these vegetation / land use classes, agricultural land (AG), emergent vegetation (EM), herbaceous upland (HU) and scrub/shrub (SS) are the sites for potential restoration activities. The decision of whether restoration activities are to be implemented is made at each sub-reach. Therefore, it is possible that one of the sub-reaches in a reach receives restoration activities while other sub-reaches in the same reach do not.

4.2 Identify conservation practices

Many conservation activities have the potential to lower water temperatures and a variety of conservation activities have been implemented in the basin. The most popular activities are passive and active restoration efforts as shown in Table 4.2. Passive restoration allows a riparian zone to recover naturally by eliminating activities causing degradation, such as cattle grazing. The primary means of passive restoration is building fences along the stream to prevent livestock grazing or other disturbances in riparian areas. Active restoration includes vegetation planting and silvicultural options to accelerate riparian forest development (Kauffman *et al.*, 1997). In this study, we assume that passive restoration activity is building fences along a stream and allowing the trees to grow naturally, while active restoration is

[20] Of the 600 projects implemented through the Grande Ronde Model Watershed Program, only 12 percent were implemented prior to 1992.

building fences and planting trees to accelerate tree growth. These restoration activities affect riparian conditions by changing the vegetative species and their rates of growth.

Table 4.2: Major restoration projects in the Grande Ronde River basin since 1985

Work Type	Share
Fencing	32%
Vegetation Planting	23%
Livestock Water Development	16%
Large Woody Material Placement	12%
Structure Placement - Rocks	8%
Construction	6%
Structure Placement - Logs	5%
Bank Stabilization	5%

Source: Grande Ronde Model Watershed Program, unpublished data, 2002

Note: The share is based on the number of sites in which each work type is implemented.

Other conservation practices, such as bank stability improvement and channel narrowing, are also available and have the potential to affect water temperatures. However, their impacts on water temperatures are harder to measure, and therefore are not considered in this study.

4.3 Estimate the costs of conservation alternatives

First, tree species that are well suited to the riparian zone in each reach of the study basin are identified. Based on literature reviews and personal communications with foresters and others who practice restoration activities in the basin, it is determined that the following tree species will grow in each vegetation / land use type (Table 4.3). It is assumed that in the case of passive restoration on agricultural lands (AG), each shrub, cottonwood and conifer grows in

one-third of the riparian zone in the sub-reach, respectively. Likewise, in scrub-shrub (SS) land, both shrubs and conifers are grown in the riparian zone (fifty percent for each species).

Table 4.3: Vegetation class and types of trees grown / planted

	AG	EM	SS	HU
Passive restoration	Shrub/ Cottonwood/ Conifer	Shrub	Shrub/Conifer	Conifer
Active restoration	Shrub/ Cottonwood/ Conifer	Shrub	NA	NA

Note: Shrub primarily represents willow and alder. No active restoration is implemented in HU and SS because it is difficult for planted trees to be established due to the lack of adequate moisture.

Given that potential tree height is the most important determinant of effects on water temperature, growth curves of each tree species are estimated (Figure 4.2). A growth curve for conifer was estimated based on the site index curves for Douglas-fir (King, 1966) and the information contained in the TMDL document on potential maximum height of ponderosa pine, Douglas fir, and Grand fir in the Upper Grande Ronde basin (ODEQ, 2000).[21] A growth curve for cottonwood was also obtained from a black cottonwood site classification table (USDA-FS, 1965) and the information contained in the TMDL document on potential maximum height. The TMDL document shows that the potential maximum height of black cottonwood in the riparian zone in the UGR study basin is 100 feet (ODEQ, 2000). The tree growth curve for shrubs (willow and alder) was estimated from studies undertaken in the Grande Ronde River basin

[21] They are 140 feet for Douglas fir, and 125 feet for Grand fir and ponderosa pine.

(Brookshire *et al.*, 2002; Case and Kauffman, 1997; Green and Kauffman, 1995; and Lytjen, 1998).

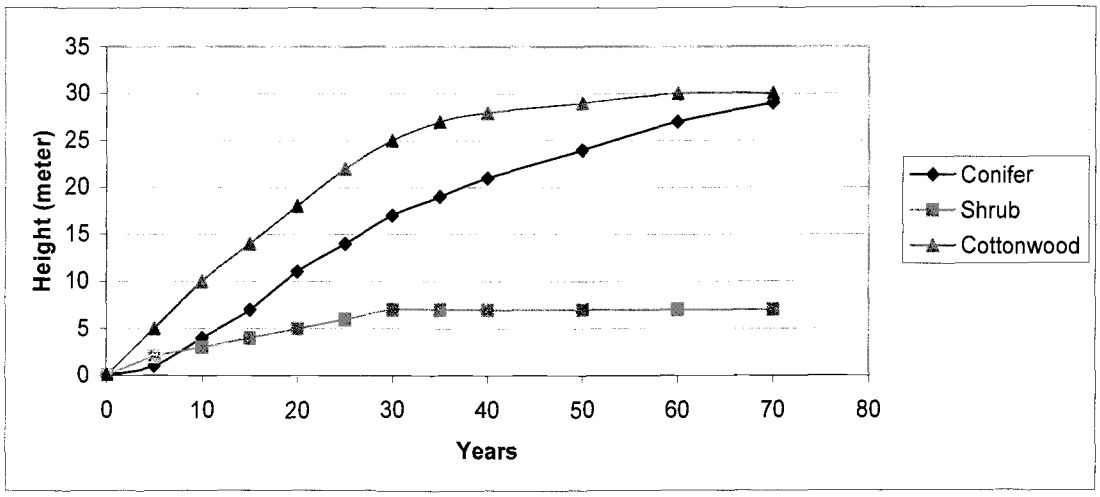

Figure 4.2: Tree growth curves

It is assumed that in the case of passive restoration, trees do not become established during the first three years after fences have been built. For active restoration, it is assumed that conifer, shrubs, and cottonwood are planted at the height of 0.5 meters (2 years of age), 1 meter (3 years of age) and 1 meter (1 year of age), respectively.

Using these tree growth curves, potential maximum tree heights in respective vegetation class resulting from passive or active restoration are then computed (Table 4.4).

Table 4.4: Potential maximum height in each vegetation / land use type

Landuse / vegetation type	Conservation Practice	In 10 years	In 20 years	In 40 years
		Vegetation height (m)		
AG	Passive restoration	4	9	18
	Active restoration / Shrub	4	6	7
	Active restoration / Conifer	5	12	22
	Active restoration / Cottonwood	11	19	28
EM	Passive restoration	3	5	7
	Active restoration / Shrub	4	6	7
SS	Passive restoration	2	9	10
HU	Passive restoration	4	8	14
	Existing height (m)	Mean height (m)		
Forest	6-12	17	22	26
	12-18	21	24	27
	18-24	24	26	28
	24-30	29	30	31
	30-	33	33	33

Note: It is assumed that restoration practices are not applied to forested areas. However, trees in forested areas are assumed to grow at rates established in the growth curves.

Table 4.4 shows, for example, it is possible to attain a 19 meter height in an AG landscape in 20 years if cottonwood trees are planted.

Using cost data from past projects implemented in the basin (Grande Ronde Model Watershed Program, unpublished data, 2002) as well as information from the Oregon Department of Forestry and other conservation agencies, the costs of each restoration activity are estimated. An average cost of fencing in Union County is estimated to be $1.60 per foot, which is equivalent to approximately $8,500 per mile. Livestock water development often accompanies the construction of riparian fencing, because when fences are built, cattle no longer have an access to water. Although the cost of livestock water development differs across sites, it is estimated that each water development costs about $2,000 per mile (personal communications with Lyle Kuchenbecker, Grande Ronde Model Watershed Program, and Tom

Smith, NRCD Wallowa). Therefore, in total, the cost of fencing per mile is estimated to be $10,500 per stream mile. Tree planting costs were primarily obtained from Oregon Department of Forestry for CREP projects (ODOF, unpublished data). They include the costs for site preparation, trees, labor for planting and maintenance. Differences in spacing as well as survival rates across tree species are also considered. Table 4.5 shows the costs of passive and active restoration per mile. These costs are assumed to be the same across reaches and across vegetation types.

Table 4.5: Cost of restoration activities

(Unit: dollar per mile)

Restoration types	First 5 meter width	Additional 5 meter width
Passive	10500	0
Active / Shrub	13340	2840
Active / Conifer	11330	830
Active / Cottonwood	12700	2200

In the case of passive restoration, expanding the width of restoration activities beyond 5 meter may incur additional costs to land owners. However, all the land designated as agriculture in the basin is actually for grazing, and the cost of setting aside an additional 5 meter width of riparian land is minimal for landowners. Therefore, it is assumed that the cost of increasing width associated with passive restoration is zero.

Using these data, the cost of passive and active restoration in each sub-reach in each time frame is estimated. The height used is the average vegetation height in each sub-reach. We assume that all the restoration activities are implemented in year 0. The type of restoration practices to be employed in each sub-reach is determined at this stage. Restoration types

presented in Table 4.6 represent the minimum cost activities in each vegetation type and in each time frame. Thus, once a certain sub-reach is identified as an optimal site to receive restoration, then the restoration activity specified in Table 4.6 is implemented in the sub-reach.

Table 4.6: Minimum cost restoration activities

		10 years	20 years	40 years
Restoration type employed	AG	Active restoration / cottonwood	Active restoration / cottonwood	Active restoration / cottonwood
	EM	Active restoration / shrub	Active restoration / shrub	Passive restoration
	SS	Passive restoration	Passive restoration	Passive restoration
	HU	Passive restoration	Passive restoration	Passive restoration
Cost per meter height per stream mile in dollars	AG	1155	668	454
	EM	3335	2223	1500
	SS	2625	1313	750
	HU	5250	1167	525
Cost for additional 5 meter of width per stream mile in dollars	AG	200	116	79
	EM	550	367	0
	SS	0	0	0
	HU	0	0	0

Table 4.6 shows that, for example, if the objective of conservation agencies is to have a 10 meter vegetation height in agricultural land in 40 years, then the minimum cost method is to apply active restoration with cottonwood trees. In this case, the cost per meter height is $454. According to the table, passive restoration is not a cost-effective method on agricultural land regardless of the time span considered in this study. It is interesting to see that in 10 year time frame, restoration in EM is less costly than HU, but in the 20 and 40 year time frames, restoration in HU is less costly than EM. This is because trees grow faster in EM in the short

run, but their maximum potential height is lower than for land class HU. Therefore, in the long run, it is less costly to apply passive restoration efforts in HU than in EM.

4.4 Estimate the relationship between riparian conditions and water temperature

A state-of-the-art temperature model, WET-temp, was used to estimate the temperature in the mainstem as well as in the tributaries in association with riparian vegetation height. The WET-temp was developed by Cox (2002) and was previously applied to a sub-watershed of the South Santiam River in western Oregon. A desirable feature of the WET-temp model is its ability to incorporate spatial GIS data. It is also less information intensive than other temperature models such as the Heat Source model used by the Oregon Department of Environmental Quality. The WET-temp model provides estimates of water temperature every 15 minutes at every 100 meters along the stream. Since we are interested in maximum water temperatures, the WET-temp model was calibrated to observed maximum daily temperatures at 3 points in the UGR mainstem in 1999 (Figure 4.3).[22] In general, WET-temp calibrates well as is shown in Table 4.7. In some of the tributaries, the WET-temp tends to overestimate temperatures. Thus, for those tributaries, the WET-temp estimates are adjusted using the ratio of actual to estimated data. This ensures that the estimates follow the actual temperature patterns.

[22] WET-temp estimates are maximum daily temperatures, but temperature standards such as TMDL are 7-day averaged maximum daily temperatures (maximum 7-day temperature). To convert maximum daily temperatures to maximum 7-day temperatures, maximum daily temperatures are multiplied by 0.95. This value was obtained by comparing the measured maximum daily temperatures and the measured maximum 7-day temperatures in multiple monitoring points in several years.

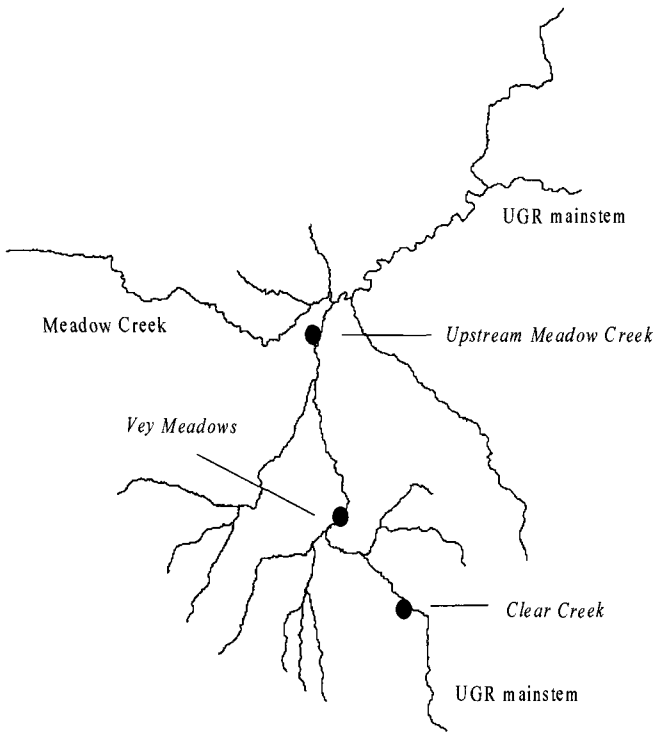

Figure 4.3: Calibration points of the WET-temp model

Table 4.7: Calibration results of the WET-temp model

	Site	Modeled temperature	Measured temperature
UGR mainstem	Clear Creek	15.11	14.32
	Vey Meadows	23.97	22.72
	Upstream Meadow Creek	24.44	24.50

Since the conservation practices (passive and active restoration) primarily affect the height and width of riparian vegetation, vegetation height and width are the control variables in WET-temp. In Oregon, riparian management area width is 30 meters (100 feet) for large

streams, 21 meters (70 feet) medium stream and 15 meters (50 feet) for small streams (ODOF, 2003).[23] However, the WET-temp simulations indicate that riparian vegetation wider than 10 meter has little effect on stream temperatures. Thus, in the simulation analyses, the width of restoration activities is set either at 5 meters or 10 meters.

Riparian vegetation provides various functions associated with water quality, including the following:

- Provides shade to moderate water temperature,
- Holds soil in place,
- Protects stream banks,
- Captures, stores and safely releases precipitation, and
- Filters nutrients from both ground water and surface runoff.

On the other hand, water temperatures are also influenced by many factors including the following:[24]

- Shading
- Volume of water flowing in the stream
- Width-to-depth ratio of the stream, and
- Groundwater recharge.

Passive and active restoration activities affect all these factors. However, among these interactions, we focus on the effects of riparian vegetation on water temperature through shading, because shade has the most direct impact to water temperatures (Beschta, 1997) and is

[23] Large, medium and small streams are categorized based on average annual flows. Large streams have an average annual flow of 10 cfs or greater, medium streams 2 to 10 cfs, and small streams less than 2 cfs.
[24] These lists are taken from the Upper Grande Ronde River Subbasin Local Agricultural Water Quality Advisory Committee (1999).

easier to capture than other links. This assumption suggests that the effects of passive and active restoration on water temperature reductions evaluated in this study are likely to be underestimated.

In estimating the relationship between vegetation height / width and water temperatures, the WET-temp model was run 2000 times for each time frame, each of which consists of different combinations of vegetation height and width in each sub-reach. Then, maximum water temperatures at representative points (or the average of maximum temperatures in each reach) estimated by the WET-temp model are regressed against vegetation height and width in each sub-reach located in their upstream areas. The following regression model was used to estimate the maximum water temperature at point $\tilde{j}$ in reach j:

$$temp(\tilde{j}) = \alpha_{\tilde{j}} + \sum_{i=1}^{j} \sum_{k=1}^{4} \beta_{ik} h_{ik} + \sum_{i=1}^{j} \sum_{k=1}^{4} \gamma_{ik} h_{ik}^2 + \sum_{i=1}^{j} \sum_{k=1}^{4} \delta_{ik} w_{ik} h_{ik} \tag{29}$$

where $i =$ reach ($i = 1,\ldots,j$ are located in the upstream area of reach j).

$k =$ AG, EM, HU, and SS

$h =$ vegetation height

$w =$ dummy variable ($w = 1$ if vegetation width is 10 meter, $w = 0$ if 5 meter)

The entire study basin consists of reaches numbered from 1 to I, and they are ordered such that a reach with a lower number is located upstream of a reach with a higher number. Squared height is included to capture a nonlinear relationship between vegetation height and water temperature. α_j is an intercept, and the sign of δ_{ik} is negative since an increase in vegetation width (from 5 meter to 10 meter) is expected to decrease water temperatures. In addition, either

β_{ik} or γ_{ik}, or both must be negative to ensure that an increase in vegetation height has a negative effect on water temperature. The R-squares of these regressions estimated here exceed 0.95, with the majority (except for 3 models) higher than 0.97.

4.5 Estimate the relationship between water temperature and fish density

Increases in salmonid populations are assumed to be the primary benefit of temperature reductions (ODEQ, 2000). To measure the increase in fish numbers, fish density models were first estimated for juvenile chinook salmon and rainbow trout, using the data collected by Ebersole (2001). Next, by applying the fish density models and using the information on habitat conditions in each reach, the total fish populations of chinook salmon and rainbow trout are estimated. The information on habitat conditions is obtained from the Aquatic Inventories Project (ODFW, 1999). Although the Aquatic Inventories Project provides the most comprehensive habitat dataset for Oregon, it includes only a limited number of parameters on habitat conditions. Therefore, the fish density models were estimated using data collected by Ebersole (2001) with habitat parameters available in the ODFW dataset only. In the regression analysis, the step-wise method is employed, including unsquared and squared terms of all the variables available in the ODFW dataset.

Table 4.8: Estimated coefficients for juvenile chinook salmon and rainbow trout density models

	Chinook salmon		Rainbow trout	
	Coefficient	t-statistics	Coefficient	t-statistics
Intercept	-1.1668 ***	-4.91	-0.6547 *	-1.56
7-day max water temperature	0.0930 ***	4.44	0.1244 ***	3.09
7-day max water temperature squared	-0.0023 ***	-4.37	-0.0034 ***	-3.41
Fines	-0.6268 **	-1.74	-0.7532 ***	-2.28
Fines squared	1.5115 ***	2.87	0.5188 *	1.53
Mean channel depth	3.3480 ***	2.97	-0.7442 **	-2.02
Mean channel depth squared	-7.0261 ***	-2.61		
R-square	0.68		0.43	
Number of observations	26		37	

Note: Three, two and one asterisks indicate statistical significance at the 5%, 10% and 15% levels, respectively.
Fines refers to the proportion (percent) of substrate transect points intersecting fine substrates (less than 16 mm in size)
Fish density and mean channel depth are log-transformed.
The incipient lethal limit for juvenile chinook salmon is set at 25.5°C based on Ebersole data (2001) as well as the TMDL document (ODEQ, 2000). If the temperature is above this level in any reach, it is assumed that chinook salmon is not present.

With the fish density models in Table 4.8, the total fish populations of chinook salmon and rainbow trout are estimated using the habitat data in each reach. However, the ODFW habitat data are available only in a limited number of streams, which are the UGR mainstem (except for Reach UGR mainstem 7 and 8), Sheep Creek and short stretches of some tributaries. In addition, the primary habitat area of chinook salmon in the upper Grande Ronde river basin is limited to the UGR mainstem and Sheep creek (personal communication with Joe Ebersole, 2003). Therefore, in estimating the total number of chinook salmon and rainbow trout, only reaches in the UGR mainstem and Sheep Creek are considered as a potential habitat.

It is important to understand the assumptions used in this fish analysis. The primary assumptions imposed are listed below.

(1) Spatial representation

Sites for Ebersole study (2001) were randomly selected from alluvial river segments within 3rd order and larger stream valleys in the Wenaha River/Lower Grande Ronde River, Lostine River, Catherine Creek, and Upper Grande Ronde basins (Ebersole, 2001). Access to 7 sites in the Upper Grande Ronde basin was denied due to private land ownership, and these sites were replaced with sites based on geographical proximity and similarity of geomorphic, vegetation and land-use characteristics. Therefore, in spite of a lack of access to 7 sites, it is believed that these study sites have a good spatial representation of the alluvial river segments within 3rd order and large stream valleys in the UGR (20 out of 37 study sites are in the UGR). One segment of the UGR mainstem (UGR mainstem 7 and 8) is not in an alluvial valley, and is excluded from the fish analyses.

(2) Temporal representation

Field data were collected in the summer of 1997. Since fish abundance, flow regimes and instream habitat can vary across years, the model may not be applicable to stream conditions, adult salmon and rainbow (spawner) abundances, or other environmental conditions that are not similar to those in 1997.

The water temperature model was calibrated to 1999 data, while fish data were collected in 1997. However, air temperatures in La Grande (the closest climate monitoring station to the study basin) in each month from June through September in 1997 and 1999 were within one standard deviation of the mean air temperature of the month, respectively. In addition, precipitation in both the 1997 and 1999 water years were within the one standard

deviation of the average precipitation. Therefore, integrating temperature and fishery analyses across the different two years is not expected to alter the results of this study.

(3) Constraints on fish populations: high water temperature

While many variables affect fish populations, our primary assumption is that elevated water temperature is the major constraint. Ebersole (2001) conducted a field study in late summer of 1997 when the potential impact of water temperature was expected to be the greatest. In an analysis of that data, he found that maximum temperature was a significant variable in a model of fish density. The relationship between fish abundance and water temperature, however, may not be straight forward; many other factors also play a role, such as the abundance and distribution of adult spawners producing the juvenile cohort(s) (Beard and Carline, 1991), availability of cover or thermal refuges (Ebersole, 2001), availability of productive foraging habitats and food (Nislow *et al.*, 1998), as well as interactions of temperature with other factors, such as food availability, disease, and competition (Reeves *et al.*, 1987; McCullough, 1999). For the purpose of this economic analysis, we assume that summer conditions, especially temperature, are a population "bottleneck" (limiting factor) and that improvements in summer habitat quality (i.e. decreased stream temperature maxima) due to riparian restoration, will result in increased abundances of salmonids. Specifically, we assume that the present population densities are limited by high summer temperatures in these reaches and that if more reaches of optimal temperature were available, population densities would increase. These estimates are only made for salmonids and the effects of changes in temperature on population densities of other native species are not included.

(4) Focus on juvenile salmonid density

The model focuses on the density of juvenile salmonids because juveniles are the life stage that is most immediately affected by changes in water quality and riparian conditions. Focusing on juveniles avoids additional assumptions such as ocean survival and harvesting rate, which are required in evaluating an impact on adult fish. However, abundances of spawners (parents) can strongly regulate juvenile abundance in Columbia and Snake River tributaries, thus juvenile densities may indirectly reflect ocean and mainstem conditions, in addition to stream habitat conditions. Adult chinook salmon escapement to the Upper Grande Ronde River, for example, has been extremely low over the past seven years (personal communication with Brian Jonasson, ODFW; Table 4.9). Fish-habitat relationships in this model are based upon observations of Ebersole (2001) during 1997. In the fall of 1996, only 22 chinook salmon redds (representing spawning activity of parents of the 1997 brood year) were observed in the Upper Grande Ronde, and this number has remained consistently low to the present (Table 4.9). Additionally, in 1996 spawners were only observed in the upper river above river kilometer (rkm) 307. Distribution and abundance of juvenile salmonids within the study area will likely differ during periods of higher adult abundance, or when spawners are distributed more widely across the study reaches. We have not simulated effects of increased spawner density or spatial distribution in this model; doing so would require more explicit information on spawner-recruit relationships than is presently available.

Table 4.9: Upper Grande Ronde River spring chinook spawning ground surveys

year	total redds	redds above weir	redds below weir	estimated spawners above weir	comments
1996	22	22	0	no estimate	no weir in 1996
1997	19	13	6	no estimate	no survey in Vey Meadows (6.5 miles)
1998	25	23	2	84	no survey in Vey Meadows (6.5 miles)
1999	0	0	0	4	no survey in Vey Meadows (6.5 miles)
2000	20	9	11	23	Vey meadows surveyed by helicopter
2001	15	8	7	34	Vey meadows surveyed by helicopter
2002	14	12	2	61	no survey in Vey Meadows (6.5 miles)

Note: Weir is located at rkm 307.

(5) Model specification

The ODFW habitat dataset (1999), although the most complete dataset on aquatic inventory for the stream, contain only a limited number of parameters. Thus, some important parameters, such as coldwater patch frequency, large wood frequency and embeddedness, had to be dropped from the model, which may have reduced the model's explanatory power and caused biased estimates. For example, it is known that coldwater patch frequency is an important variable affecting fish abundance (Ebersole, 2001). However, ODFW habitat data do not include the distribution of coldwater patches. Therefore, it was not included in the model. Instead, an attempt was made to replace the coldwater patch frequency by other variables which might be able to explain a part of the effects of the coldwater patch frequency.

(6) Effects of passive and active restoration

Riparian restoration can result in changes to instream habitat in addition to changes in stream temperature. However, the effects of passive and active restoration of stream banks on

geomorphology, spawning substrates, coldwater patches and instream community dynamics are not considered in these calculations of potential changes in fish density.

(7) Reach segmentation

Total fish population was estimated using the mean values of each habitat variable in each reach. Although reaches in the UGR mainstem and Sheep creek were divided primarily based on geomorphological aspects, this methodology assumed uniformity in habitat variables and did not consider potential variation within each reach.

(8) Influence of fish movement

Finally, it is assumed that fish densities within individual stream reaches in the model are a function of conditions in that reach, and that reach fish densities are not influenced by conditions elsewhere (*e.g.* movement between reaches is minimal).

4.6 Specify policy options

Three general optimization problems are specified here to evaluate a range of policy options. The first objective (policy option) is to invest in restoration activities to achieve a certain temperature reduction at a given point with minimum cost.

$$\underset{h_{ik},w_{ik}}{Min} \sum_{i=1}^{j}\sum_{k=1}^{4}\left[C_h(h_{ik}) + C_w(h_{ik}w_{ik})\right] \tag{30}$$

s.t. $$\Delta temp_j = \Delta temp_j(h_{11}, w_{11}, \ldots, h_{ik}, w_{ik}, \ldots, h_{j4}, w_{j4}) \leq \Delta T \tag{31}$$

where $temp_j$ =water temperature at a given point in reach j

i = reach (i = 1,…, j are located in the upstream area of reach j).

k= vegetation class (k = AG, EM, SS, and HU)

h_{ik} = riparian vegetation height in reach i, vegetation class k

w_{ik} = riparian vegetation width ($w_{ik} = 1$ if width is 10m, $w_{ik} = 0$ if 5 m)

$C_h(h_{ik})$ =restoration cost associated with vegetation height

$C_w(h_{ik} w_{ik})$ = restoration cost associated with vegetation width

ΔT = temperature change target

A change in temperature at a given point in reach j is the function of riparian vegetation height and width in each sub-reach located in the upstream area of reach j (reaches from 1 to j). Since a change in temperature is negative, the constraint equation means that the reduction in temperature needs to be larger than a targeted temperature change (ΔT).

The second policy option is to invest in restoration activities to maximize stream length where water temperature decreases by a certain degree with a given budget constraint.

$$\underset{h_{ik}, w_{ik}}{Max} \quad \sum_{i=1}^{I} [s_i * L_i] \tag{32}$$

s.t $\quad \Delta temp_{\hat{j}} = \Delta temp_{\hat{j}}(h_{11}, w_{11}, \ldots, h_{ik}, w_{ik}, \ldots, h_{j4}, w_{j4}) \quad j = 1, \ldots, I$ (33)

$$s_j = 1, \text{ if } \Delta temp_{\hat{j}} \leq \Delta T \qquad j = 1, \ldots, I \tag{34}$$

$$s_j = 0, \text{ if } \Delta temp_{\hat{j}} > \Delta T \qquad j = 1, \ldots, I \tag{35}$$

$$\sum_{i=1}^{I} \sum_{k=1}^{4} [C_h(h_{ik}) + C_w(h_{ik} w_{ik})] \leq B \tag{36}$$

where $temp_{\hat{j}}$ = the highest water temperature in reach j

s_i = dummy variable

L_i = length of reach i

B = budget

It is assumed that once a point that is the highest water temperature in each reach attains the temperature reduction target, the entire reach also attains the target. Then, the entire length of the reach is counted in the objective function. The temperature target is either a change in temperature or an absolute temperature level. In the latter case, deltas (Δ) in the above equations are omitted.

The third model specification is to simulate a policy to maximize fish numbers subject to a given budget constraint.

$$\underset{h_{ik},w_{ik}}{Max} \sum_{i=1}^{I}\left[Fish_i(temp_{\bar{i}}, a_i) * L_i\right] \tag{37}$$

s.t. $$temp_{\bar{j}} = temp_{\bar{j}}(h_{11}, w_{11}, \ldots, h_{ik}, w_{ik}, \ldots, h_{j4}, w_{j4}) \qquad j = 1,\ldots,\mathrm{I} \tag{38}$$

$$\sum_{i=1}^{I}\sum_{k=1}^{4}\left[C_h(h_{ik}) + C_w(h_{ik}w_{ik})\right] \leq B \tag{39}$$

where $temp_{\bar{i}}$ = average water temperature in reach i

$fish_i$ = fish density per mile in reach i and is defined by Table 4.8

a_i = other variables that affect fish density

These are mixed integer nonlinear programming (MINLP) models. The GAMS DICOPT solver is used to solve these problems.

Chapter 5

Results and Discussion

5.1 Longitudinal temperature profiles

The first simulation analysis performed here is to explore temperature changes in the mainstem without regard to the costs or locations of restoration activities. Using the WET-temp model, longitudinal temperature profiles in the UGR mainstem are estimated under the maximum restoration efforts in 10, 20 and 40 year time frames. As defined here, maximum restoration efforts mean that all the sub-reaches in the basin that have a potential for restoration efforts receive such restoration activities, for their entire length. These profiles are depicted in Figure 5.1 for the base case (current situation) and the three time frames of restoration.

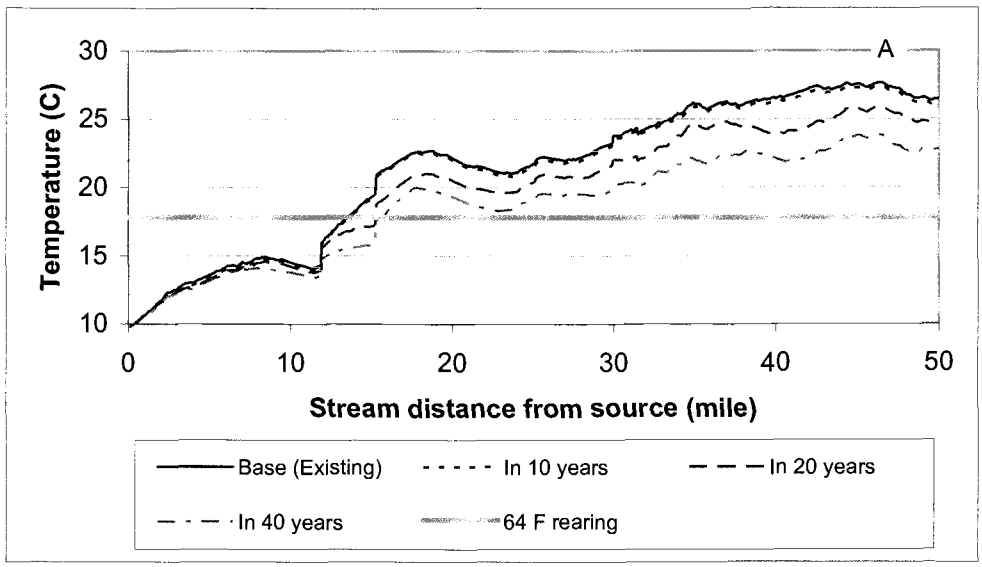

Figure 5.1: Longitudinal temperature profile in the UGR mainstem in 10, 20 and 40 year time frame when the maximum level of restoration efforts are implemented[25]

[25] Water temperatures reported here are the 7-day averaged maximum daily temperatures.

The figure shows that as the length of time increases, the magnitude of temperature reductions increases, because of an increase in shading brought about by tree growth. However, potential temperature decreases in the mainstem are limited to about 4 °C in 40 years, and less in the short run (10 years).

The water temperature criterion for rearing salmonid juveniles is 17.8 °C (64 °F). However, Figure 4 shows that it is not possible to attain the temperature criterion in most stretches of the UGR mainstem, even in 40 years, under an unlimited budget with the alternatives considered here. With the maximum level of riparian restoration, the increase in stream miles that attains the temperature criterion is only 1.7 mile in 20 years (from river mile 13.6 to 15.3) and 2.1 mile in 40 years (from river mile 13.6 to 15.7), respectively. However, it is important to note that this does not necessarily mean that the temperature criterion is not attainable in 40 years for most stretches of the mainstem. As mentioned before, this study considers the effect of riparian restoration on water temperature by means of shading only. Riparian restoration is also expected to reduce water temperature in different ways, such as improving stream banks and reducing the width to depth ratio. In addition, there are other restoration practices than passive and active restoration such as channel narrowing through engineering. If these activities are included, some additional cooling may result.

Contrary to the results for the mainstem, a greater magnitude of temperature reduction is likely in the tributaries. Figure 5.2 shows the longitudinal temperature profile in Fly Creek under the maximum restoration efforts in 10, 20 and 40 year time frames. It shows that a large portion of the Fly Creek attains the temperature criterion in 40 years if maximum restoration efforts are implemented. Although the magnitude of temperature reductions resulting from riparian restoration efforts varies across tributaries, it is a common feature of these analyses that

the magnitude of temperature reductions is larger in tributaries than in the mainstem. This implies that attaining the 17.8 °C (64 °F) temperature criterion has a greater potential and relevance in tributaries rather than in the mainstem.

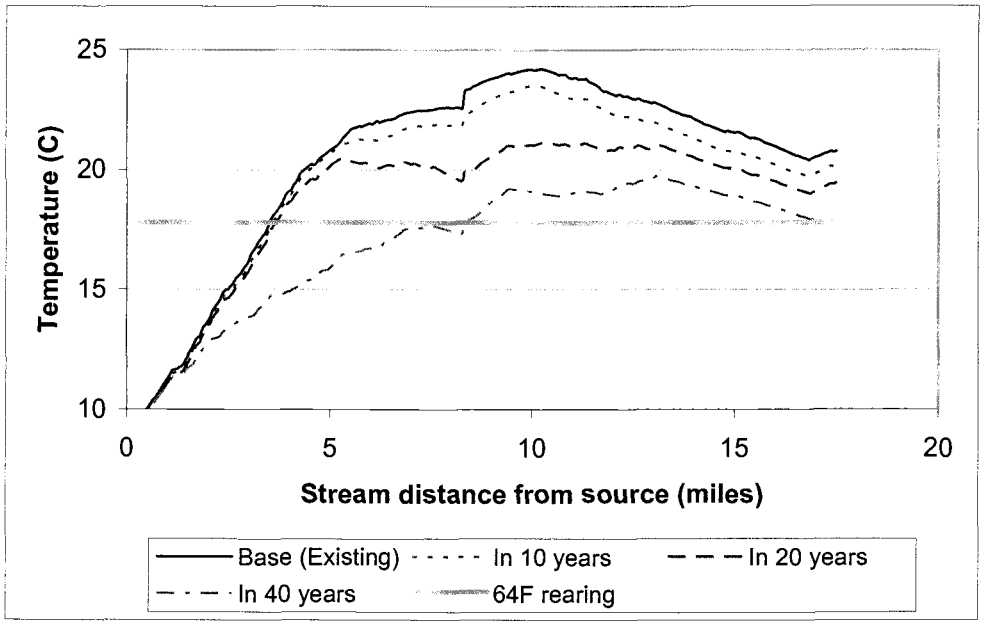

Figure 5.2: Longitudinal temperature profile in Fly Creek in 10, 20 and 40 year time frame when the maximum level of restoration efforts are implemented

While attaining the temperature criterion for rearing salmonid juveniles is not possible for most reaches of the mainstem, it is still important to decrease water temperatures for the purposes of expected benefits. As the fish density model shows, decreasing water temperature, even to levels above 17.8 °C, is still expected to increase salmonid populations. Therefore in the following analyses, spatial configurations of restoration efforts under different temperature targeting scenarios are examined.

5.2 Temperature change targeting

The first economic analysis focuses on minimum costs of temperature reductions. Specifically, Figure 5.3 shows the minimum costs associated with temperature reductions at a given point (point A) in Figure 5.1, in 20-year and 40-year time frames. Point A is chosen because it is the highest observed maximum temperature in the UGR mainstem. Figure 5.3 shows that it is not possible to decrease water temperature by more than 1.5 °C and 4.2 °C in 20 year and 40 years, respectively, even if an unlimited budget is invested in riparian restoration efforts. The figure also shows that the cost of temperature reductions is lower for small temperature reductions, but it increases rapidly once the magnitude of temperature reductions exceed 1 °C (1.8 °F) in the case of 20 year time frame and 3.5 °C (6.3 °F) in 40 year time frame. The curves also show that if temperature reductions are targeted over a 40 year time frame, then a much larger temperature reduction can be attained for a given cost.

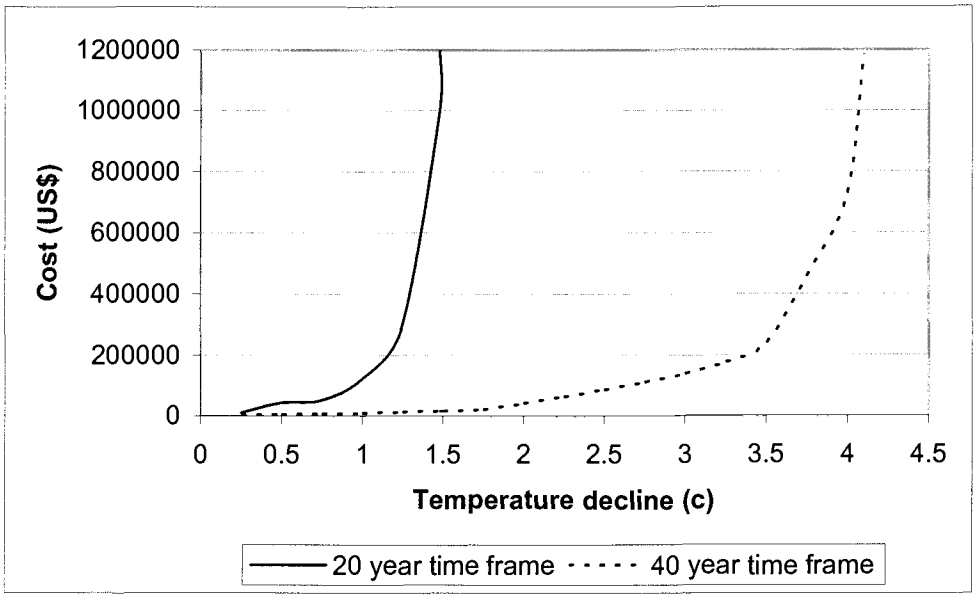

Figure 5.3: Costs of temperature reductions at a point in the lower UGR mainstem under 20 and 40 year time frames

The spatial configurations of restoration practices are examined for a range of water temperature decreases. Specifically, Figure 5.4 shows the minimum cost allocation of restoration activities when the water temperature at point A is decreased by 1, 2, 3, and 4 °C degrees in the 40 year time frame, respectively. The figure depicts those reaches where restoration activities are applied to at least one of the sub-reaches.

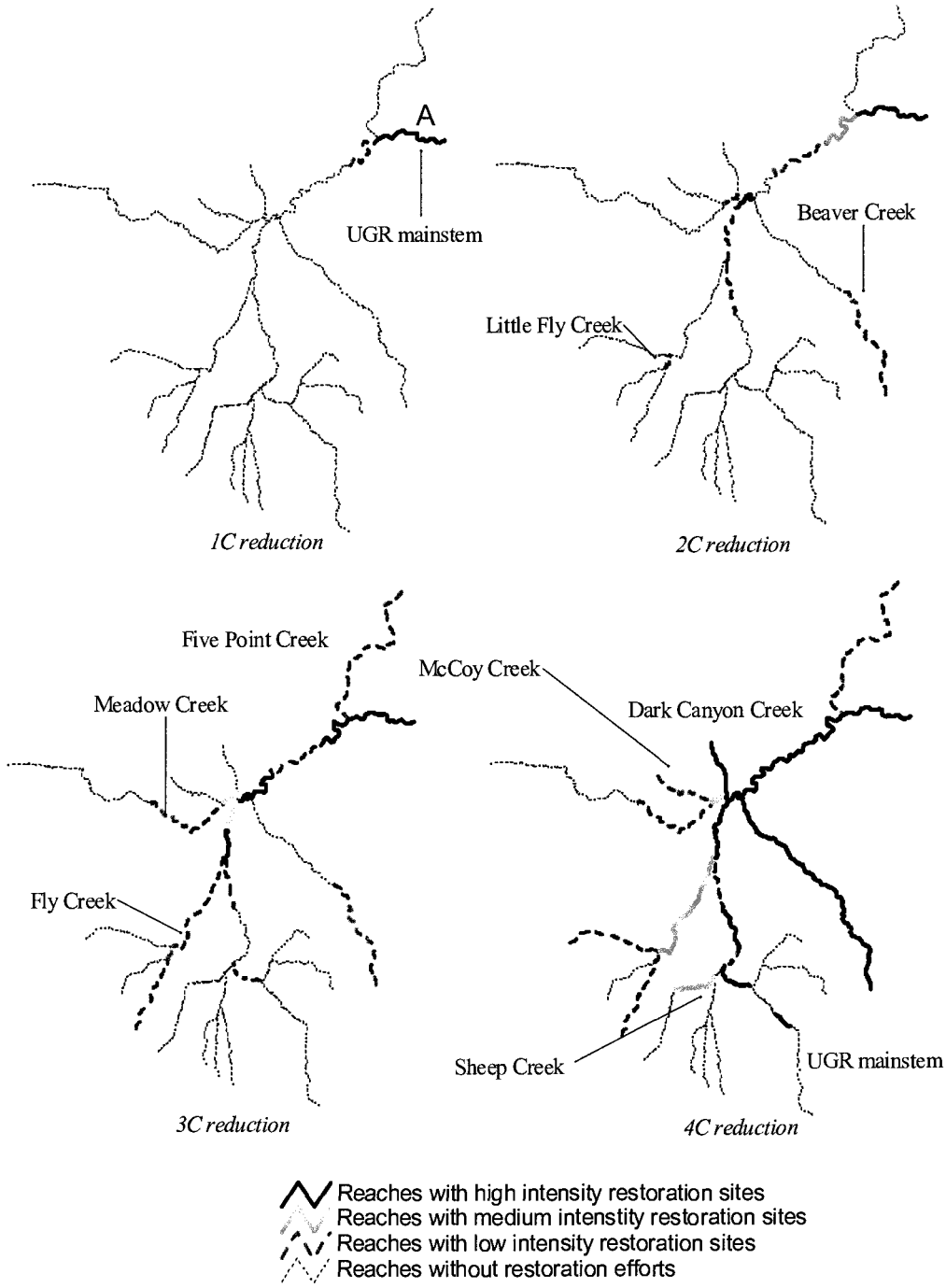

Figure 5.4: Minimum cost restoration efforts, by reach, when water temperature at point A is reduced by 1, 2, 3 and 4 °C for a 40 year time frame

Note: Reaches with high, medium, and low intensity restoration sites mean that restoration activities are implemented in more than 67 percent, between 33 percent and 67 percent, and less than 33 percent of the stream length of potential restoration sites in each reach (AG, EM, SS, and HU).

Figure 5.4 shows that when the magnitude of the desired temperature reductions is small, only the nearby reaches in the lower mainstem receive restoration efforts. As the magnitude of desired temperature reduction increases, restoration activities are intensified (the length in which passive or active restoration efforts are implemented in each reach) in those reaches that are already the sites of restoration activities. At the same time, restoration efforts are also expanded to the upper stream reaches in the mainstem, and then to the tributaries. This analysis shows that localized effects of restoration efforts are important in achieving small temperature reductions, while cumulative (longitudinal) effects are important when the magnitude of desired temperature reductions is large.

Figure 5.4 also shows that sometimes uptream reaches are given a higher priority than downstream reaches such as the case with Beaver Creek and Little Fly Creek in 2°C reduction scenario. This observation verifies the importance of spatial analysis that reflects the heterogeneous nature of temperature responses in the basin. This outcome results from the following two reasons. The first is that restoration efforts in these reaches are relatively effective compared to other reaches. In other words, temperature reduction per dollar of riparian restoration efforts is larger in these reaches than those in downstream reaches. The second is that the length of these sub-reaches is relatively small, and the total cost of riparian restoration efforts is low. Therefore, when other longer sub-reaches are not feasible for restoration efforts due to budget constraints, it is more cost-effective to apply restoration efforts to these small sub-reaches even though temperature reduction per dollar may be lower. Thus, these small reaches are chosen as a marginal source of temperature reductions. However, this second reason is primarily driven by the division of reaches, since reach length affects the cost of restoration efforts in each reach. Thus, it is necessary to be careful in identifying those reaches which should be given a priority over the downstream reaches.

Table 5.1 presents the cost allocation and contribution of each reach to temperature reductions at point A under the same scenario as in Figure 5.4. The reaches are divided into three groups: mainstem reaches located within 6 mile upstream of point A, the rest of the mainstem, and the tributaries. The table shows that, for example, in order to decrease temperature by 3 °C (5.4 °F) at point A, only 27 percent of the total cost is allocated to the nearby reaches in the mainstem, but these reaches accounts for 67 percent of the temperature reductions. As the magnitude of temperature reductions increases, a larger share of the restoration budget is allocated to other reaches in the mainstem (beyond 6 miles from point A) and to the tributaries. As a result, costs per unit of temperature reductions increase. Since the marginal effects of restoration efforts on temperature reductions in distant reaches are small, the marginal costs of temperature reductions increase rapidly. This is consistent with the results in Figure 5.3.

Table 5.1: Efficient cost allocations among reaches and their contribution to temperature reductions at point A

		Temperature reductions			
		1.0C	2.0C	3.0C	4.0C
		Cost allocation			
Reaches in the mainstem	Upstream within 6 mile	100%	54%	27%	5%
	Beyond 6 mile	0%	39%	55%	35%
Reaches in tributaries		0%	7%	17%	60%
Total Cost (dollars)		8888	40545	138172	728096
		Contribution to temperature reductions			
Reaches in the mainstem	Upstream within 6 mile	100%	87%	67%	50%
	Beyond 6 mile	0%	9%	26%	33%
Reaches in tributaries		0%	3%	7%	17%

Figure 5.5 looks at the same allocation of restoration activities as Figure 5.4 with a focus on the width of riparian restoration efforts. It shows that restoration activities should be implemented with the width of 10 meters in many reaches in the basin. This is consistent with the conservation practice standard on riparian forest buffers (NRCS, 2000). According to the forest practices standard, it is necessary to have a minimum of 35 feet (10.5 meter) of riparian vegetation along streams.

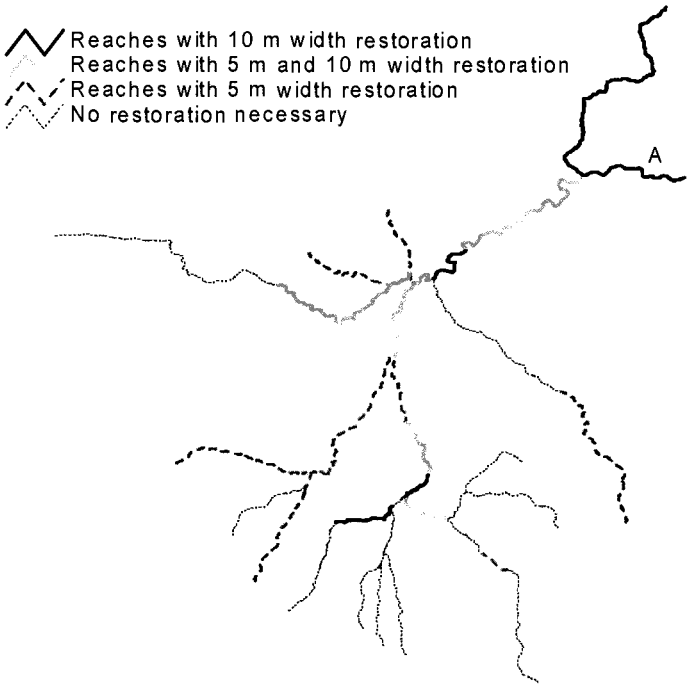

Figure 5.5: Restoration activities, by reach, with the width of 5 meters or 10 meters when water temperature at point A is reduced by 4 °C degrees (40 year time frame)

Note: For example, reaches with 10 meter width restoration means that all the sub-reaches that receive restoration efforts in that reach will have restoration activities with a width of 10 meters.

Whether passive or active restoration should be applied is also examined. Figure 5.6 shows the reaches where passive and/or active restoration activities are implemented under the same scenario as in Figure 5.4.

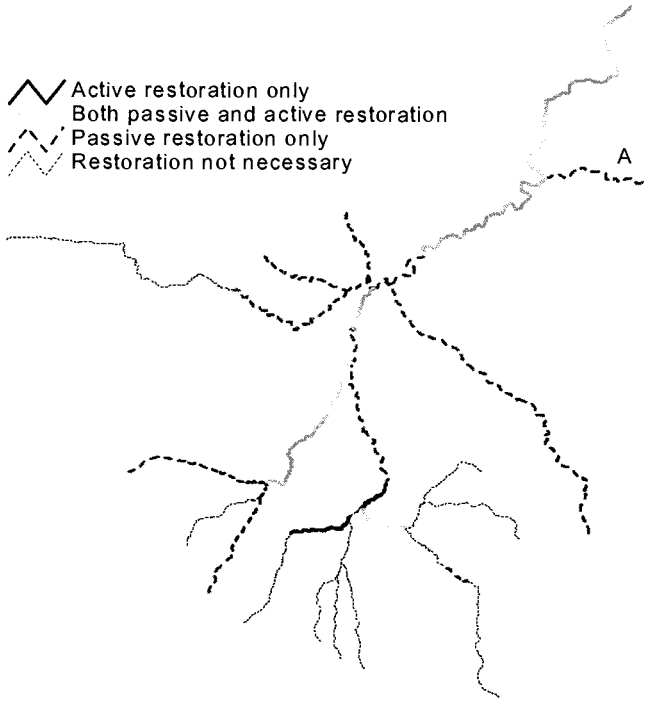

Figure 5.6: Passive and active restoration, by reach, when water temperature at point A is reduced by 4 °C degrees (40 year time frame)

Figure 5.6 does not show any specific features on spatial distribution of passive and active restoration efforts. Therefore, the impacts of time frame and the magnitude of desired temperature reductions on passive versus active restoration are further investigated by plotting

temperature changes and costs. Figure 5.7 shows the relationship between the magnitude of temperature reductions and the cost share of passive restoration for the 20 and 40 year time frames.

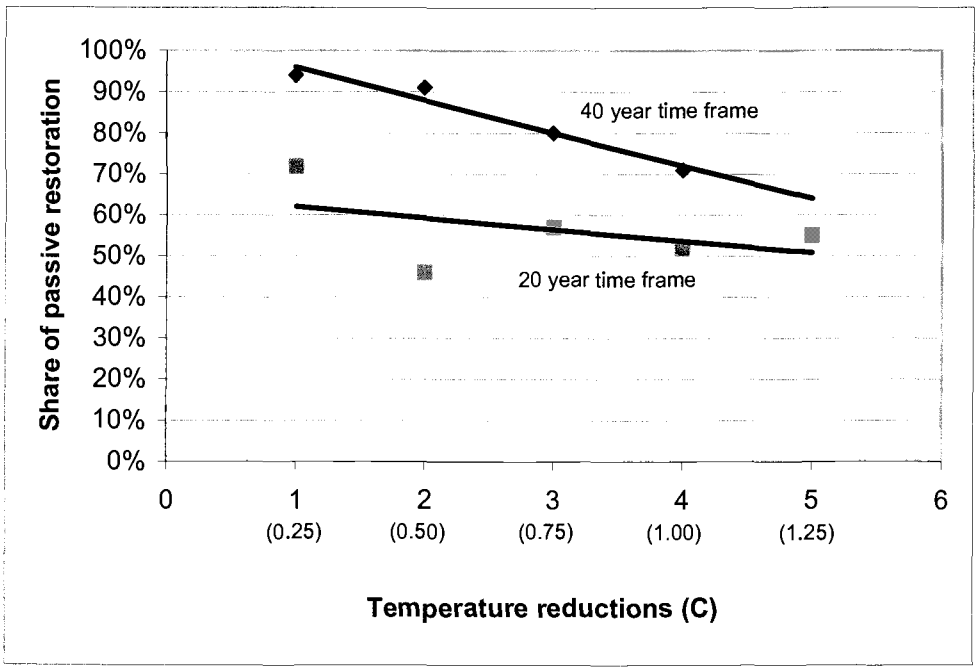

Figure 5.7: Cost share of passive restoration for a range of temperature reductions at point A in 40 and 20 year time frames

Note: The vertical axis represents the cost share of passive restoration when water temperature at point A is decreased by 1 °C – 4 °C degrees in a 40 year time frame and 0.25 °C to 1.25 °C degrees in a 20 year time frame. On the horizontal axis, the values in the upper row refer to temperature reductions in a 40 year time frame, and the values in the lower row (values in parenthesis) refer to temperature reductions in a 20 year time frame case.

Figure 5.7 shows that the cost share of passive restoration is always higher in the 40 year time frame than in the 20 year time frame. In addition, the share of passive restoration decreases as the magnitude of temperature reduction increases in both 20 and 40 year time

frame cases. Applying the same analysis to other points in the mainstem also shows that the cost share of passive restoration is higher in the 40 year time frame than in the 20 year time frame, and the share decreases as the magnitude of temperature reduction increases (in the case of 40 year time frame). These observations imply that in general passive restoration is preferred as the time frame increases and /or the magnitude of desired temperature reductions decreases. If the objective is to decrease water temperatures in a shorter time frame, and / or as the magnitude of desired temperature reductions increases, a larger share of budget will need to be allocated to active restoration activities.

The next analysis investigates the relationship between the cost of temperature reductions and stream flow (discharge). To examine this relationship, the point which is the highest observed temperature in each reach is selected, and the minimum cost of decreasing water temperature by 2 °C (3.6 °F) at each point is estimated. Figure 5.8 plots the discharge at these points and their corresponding costs. It also presents a straight trend line that minimizes the sum of the error squared.

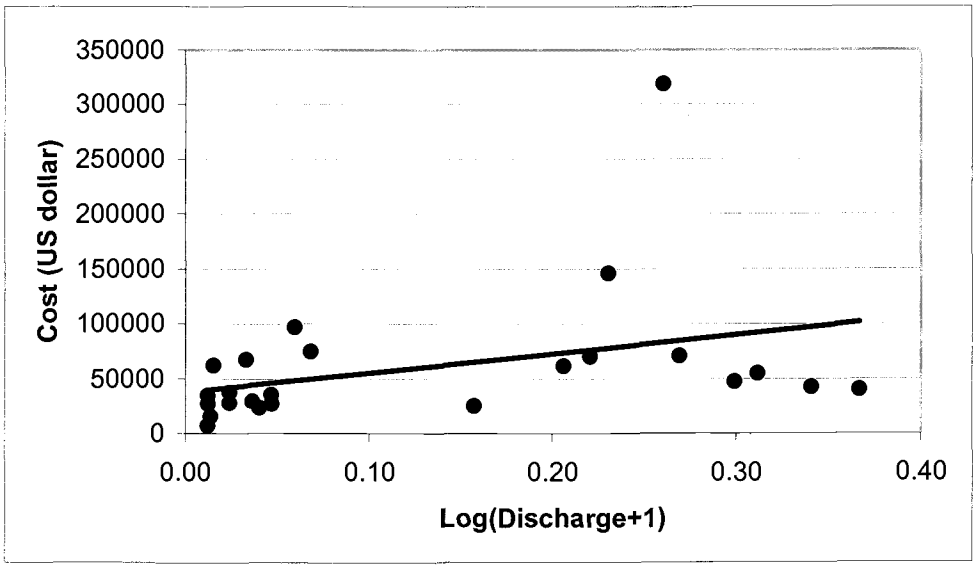

Figure 5.8: Relationship between the cost of temperature reductions by 2 °C (3.6 °F) and discharge[26]

Given equation (1) on temperature changes (Brown, 1970) in Chapter 3, it is expected that costs of temperature reductions will increase as the discharge increases. Such a relationship is only weakly observed here.[27] This suggests that, although discharge does affect the costs of temperature reductions, the heterogeneous nature of riparian characteristics such as stream morphology and the potential of restoration activities in the nearby upstream reaches, also influences the costs.

Thus far, our analyses have been associated with temperature changes at one monitoring point. However, conservation agencies are concerned with water quality in the entire basin. Therefore, in the following analysis, temperature changes in the entire basin are

[26] There are two outliers in the figure. They correspond to reach UGR mainstem 9 (the cost is $318,821) and 8 ($145,968), respectively. The costs of decreasing water temperatures in these two reaches are high because both reaches are located in and immediately below a valley where most of the riparian zone is already covered by forests, and therefore the potential of riparian restoration efforts in and around nearby upstream reaches are small.

[27] The discharge has a positive slope, and it is statistically significant at a 10 percent level (P=0.096).

considered. Specifically, the question is how the restoration efforts should be allocated within the basin if the objective is to maximize the stream length where water temperature is decreased by at least 1 °C with a given budget constraint. Maximizing the stream length that experiences temperature reductions has important implications for fish recruitment. Also, while temperature criteria are based on absolute temperature levels, conservation agencies may choose to target temperature changes, given that absolute temperature levels vary from year to year. Figure 5.9 presents those reaches where temperatures are decreased under different budget levels. It shows that tributaries such as Meadow Creek, McCoy Creek, Fly Creek, Little Fly Creek and Sheep Creek, as well as the portions of the mainstem, will be the first priority with a budget of $100,000. If the budget is expanded ($200,000), then the lower part of the mainstem as well as Five Point Creek and the lowest stretch of Chicken Creek will be targeted. Figure 5.9 shows that in general it is more efficient to decrease water temperatures in tributaries if the objective is to maximize stream length where temperature decreases by a certain degree.

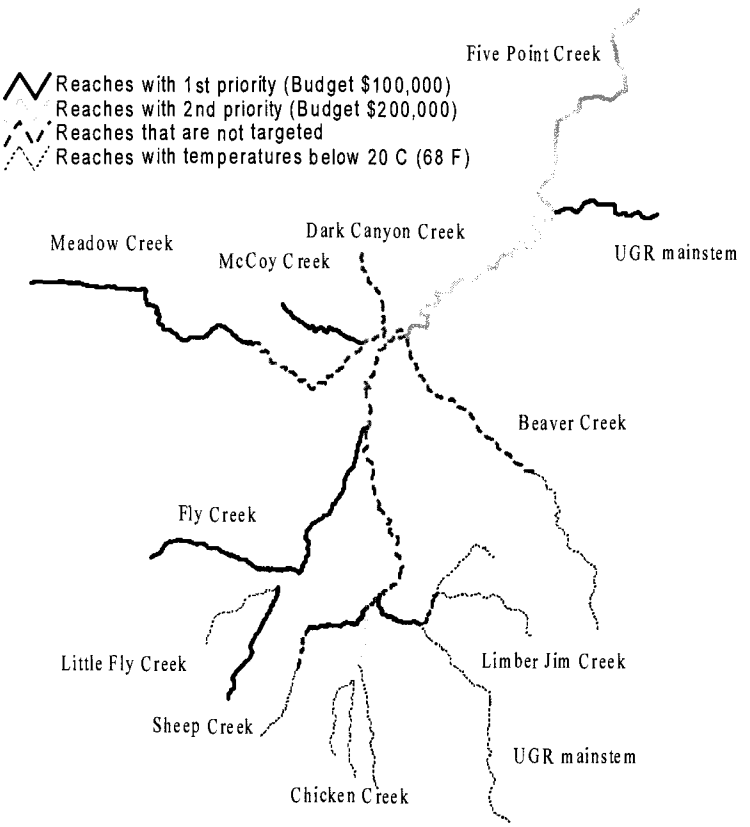

Figure 5.9: Targeted reaches when the objective is to maximize the stream length where temperature decreases by at least 1 °C

Note: It is assumed that if water temperature at a point with the highest water temperature in each reach decreases by one degree, then the water temperatures in the entire reach decreases by one degree.
Reaches with maximum temperatures lower than 20 °C (68 °F) are not subject to this temperature reduction because their temperature levels are already low.

5.3 Absolute temperature level targeting

Thus far, we have examined efficient allocations of restoration efforts in association with temperature changes. However, water quality standards are typically set based on absolute temperature levels, and many stream flow benefits such as the status of a fish population are

determined by absolute temperature levels. Thus, in the following analyses, we extend the analyses to absolute temperature levels.

First, we examine how the levels of temperature targets affect the spatial configuration of restoration efforts. Given the temperature needs of fish and varying budget constraints, conservation agencies may wish to pursue different temperature targets. For example, they may wish to minimize the stream length where water temperature is very high (*e.g.*, over 27 °C (80.6 °F) degrees) or they may want to target the stream reaches where water temperatures are already low (*e.g.*, below 20 °C (68 °F)) in order to improve habitat for coldwater fish species, ignoring reaches with high water temperatures. These different temperature targets will likely lead to different allocations of restoration activities and as a result have different impacts on the distribution of water temperatures. Figure 5.10 shows the efficient allocation of restoration efforts when the objective is to maximize stream length where temperature is below targeted levels (20 °C, 24 °C and 27 °C) with a given budget constraint (here, $100,000).

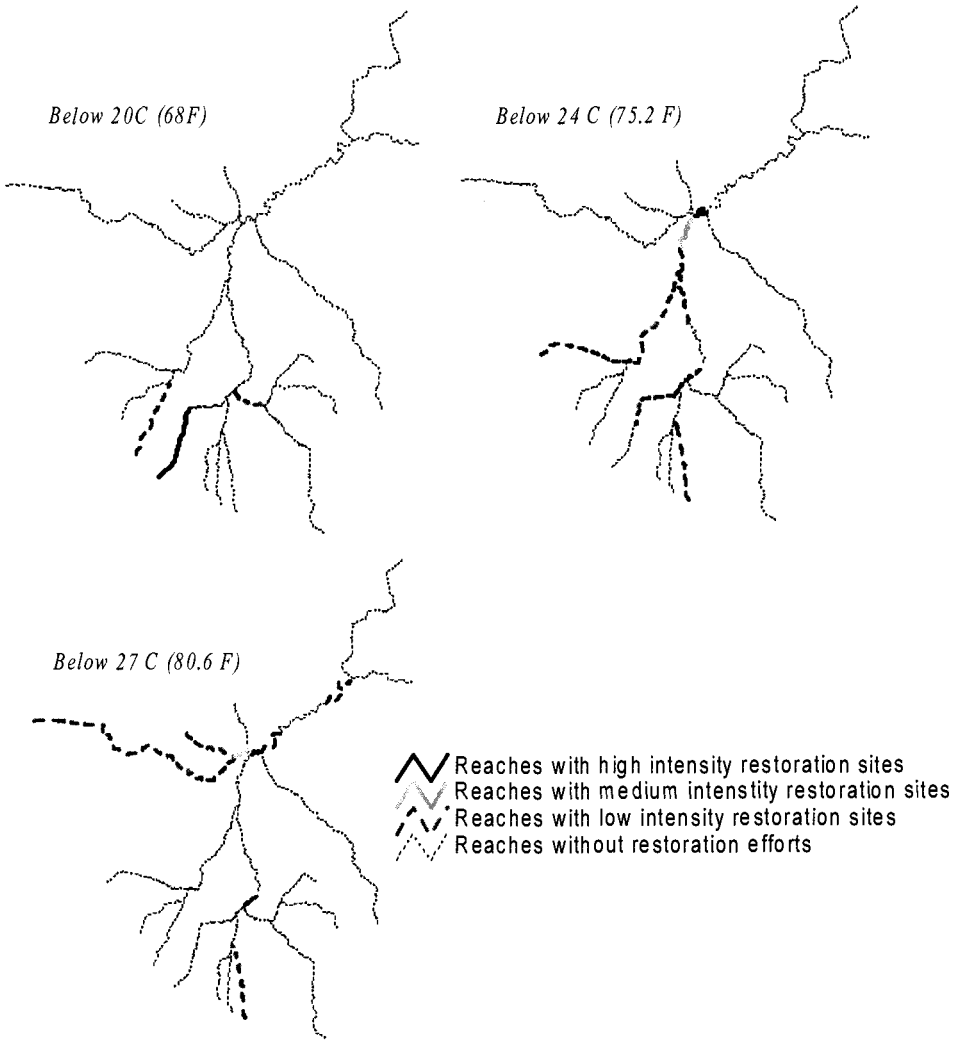

Figure 5.10: Efficient allocation of restoration efforts when the objective is to maximize stream length where temperature is below target levels with a given budget constraint

Note: Reaches with high, medium, and low intensity restoration sites mean that restoration activities are implemented in more than 67 percent, between 33 percent and 67 percent, and less than 33 percent of the stream length of potential restoration sites in each reach (AG, EM, SS, and HU).
It is assumed that if water temperatures at a point with the highest water temperature in each reach decrease below the targeted temperature level, then the water temperatures in the entire reach are below the target.

Figure 5.10 shows that depending on temperature targets, the spatial configurations of restoration efforts can vary greatly. It also shows that whatever the temperature target, reaches where temperatures are just above the target levels are given the first priority. Thus, as the temperature target rises, the sites for restoration shift northward, where elevation is lower and temperature is generally higher. These differences in the spatial configuration of restoration efforts have significant impacts on water temperatures.

Figure 5.11 shows the total stream length in each temperature range resulting from restoration efforts under the three different temperature targets. Specifically, Figure 5.11 shows the relative effects of targeting scenarios on stream reaches of different temperatures. For example, if conservation agencies wish to maximize the stream length where water temperature is below 24 °C (75.2 °F) with a budget constraint, then the minimum cost approach is to implement restoration activities in those reaches where water temperature is just above 24 °C. As a result, the length of reaches where water temperature is between 23 °C and 25 °C will be the longest among the three targeting scenarios (Compare the heights of the bar graph for this stream temperatures across the three target scenarios). Likewise, under the 20 °C targeting, the reaches where water temperature is just above 20 °C receive greater attention than in the 24 °C and 27 °C targeting cases, and the length of reaches where water temperature is below 20 °C will be the longest among the three targeting scenarios. This implies that the levels of temperature targets have significant impacts on the spatial allocation of restoration activities, and as a result, on the distribution of water temperatures in the basin.

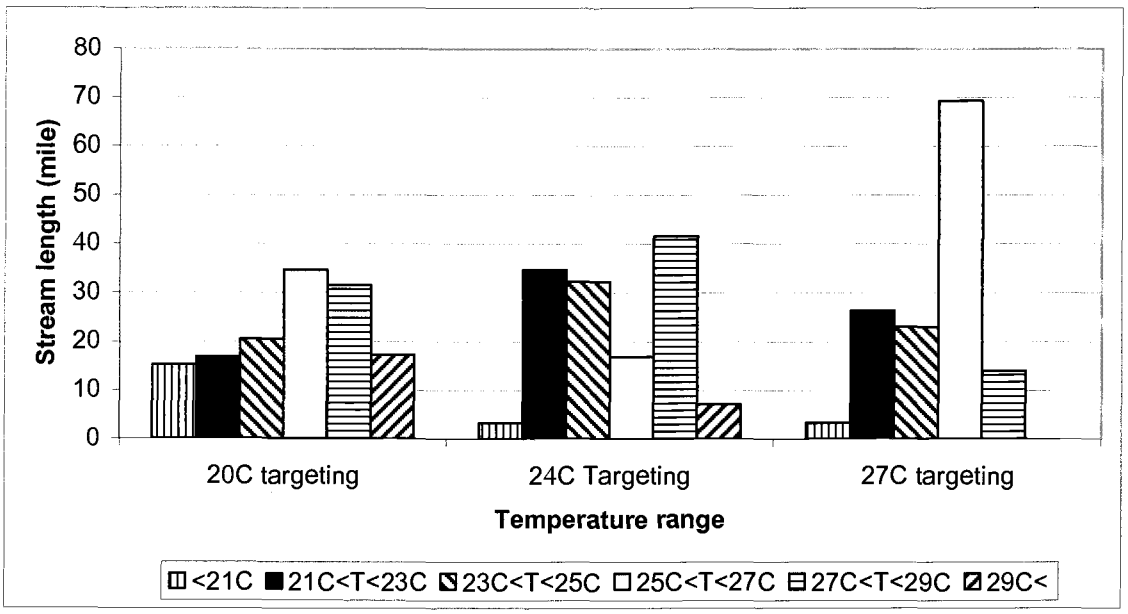

Figure 5.11: Stream length in each temperature range as a result of restoration efforts under different temperature targets

Note: Water temperature in each reach is determined by the highest water temperature in that reach. For example, if the highest water temperature in a reach is 24 °C, the entire length of the reach is categorized in the temperature range between 23 °C and 25 °C. Reaches where water temperatures are already below 20 °C are excluded from the figure.

5.4 Fishery benefits targeting

The final set of analyses draws on the efficient allocations of restoration efforts when the goal is to maximize fish populations. Figure 5.12 shows the effects of restoration efforts on juvenile chinook salmon populations. The estimated populations are computed under two different scenarios. The first is a no restoration scenario, and the second is a scenario in which a budget ($100,000) is allocated such that the chinook salmon populations in UGR mainstem and Sheep Creek are maximized.

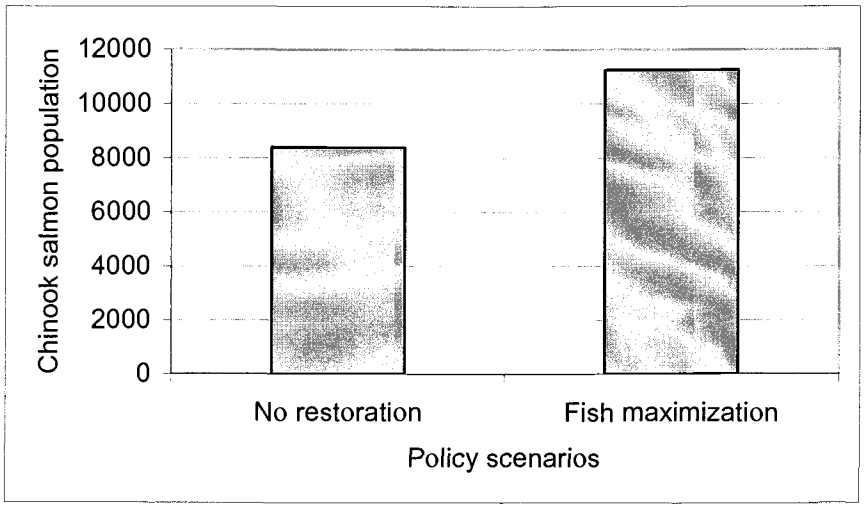

Figure 5.12: Estimated juvenile chinook salmon populations in 40 years

Note: Chinook salmon populations are estimated in the UGR mainstem and Sheep Creek only even after the restoration efforts have been implemented. Restoration activities can be applied to any sub-reach in the basin.

As shown in Figure 5.12, it is possible to increase juvenile chinook salmon by more than 30 percent with a budget of US $100,000 if the budget is allocated such that the chinook salmon population is maximized. It is important to note here that these impacts on fish populations are likely to be underestimated because of our assumptions specified in Chapter 4. Figure 5.13 presents the spatial configuration of restoration efforts within the basin under the juvenile chinook salmon population maximization scenario. The figure shows the reaches where restoration efforts are implemented.

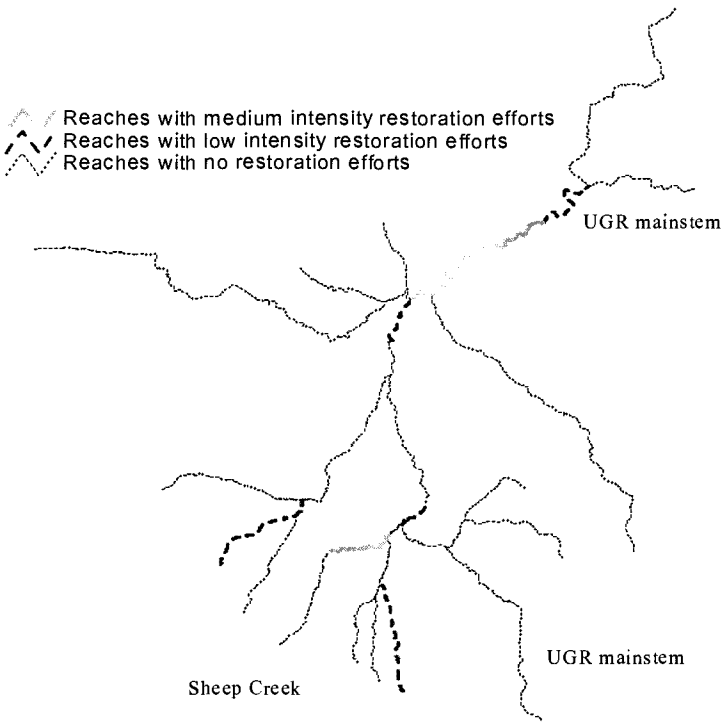

Figure 5.13: Restoration sites, by reach, to maximize juvenile chinook salmon populations with a budget constraint ($100,000)

Note: Reaches with medium and low intensity restoration sites mean that restoration activities are implemented in between 33 and 67 percent and less than 33 percent of the stream length of potential restoration sites in each reach (AG, EM, SS, and HU).

Figure 5.13 shows that restoration efforts are implemented primarily in the UGR mainstem as well as in Sheep Creek. The cost breakdown shows that more than 50 percent of the budget will be allocated to Sheep creek because of its productivity under reduced temperatures, although it consists of only 20 percent of the reaches suitable for chinook salmon habitat. This observation that the tributary (Sheep Creek) where temperature is higher than the mainstem will be given a greater share of the funding is consistent with the implications of the theoretical analysis reported in Chapter 3. Specifically, when a tributary is warmer than the

mainstem and the value of stream flow is concave with respect to water temperature, then the tributary will be the first priority.

The next analysis examines how the population of juvenile chinook salmon increases in the basin as the budget increases. The relationship between estimated juvenile chinook salmon populations and their minimum costs is shown in Figure 5.14.

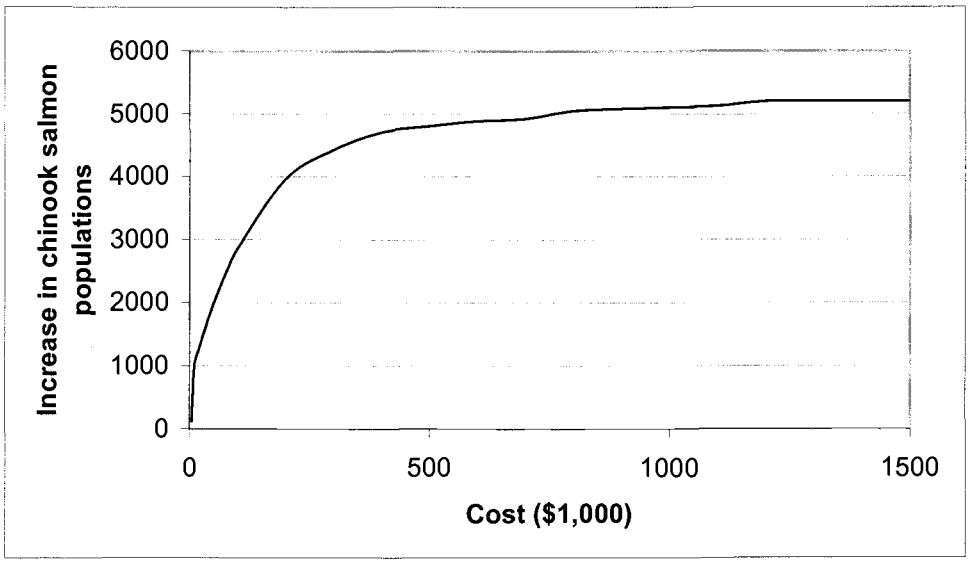

Figure 5.14: Costs of an increase in juvenile chinook salmon populations in the basin relative to the "no restoration" case

The vertical axis of Figure 5.14 shows the increase in juvenile chinook salmon populations relative to the no restoration case which is shown in Figure 5.12. Figure 5.14 indicates that chinook salmon populations increase rapidly when the budget is relatively small (less than $200,000). But as the budget level increases (and exceeds $500,000), marginal fishery benefits essentially fall to zero. This indicates that the marginal cost of an increase in juvenile

chinook salmon populations increases rapidly as the population is increased. This is primarily because as the water temperature decreases, further reductions in water temperature become more costly. As a result, the fishery benefits are minimal. This finding is consistent with Figure 5.3.

Next, the efficacy of temperature targeting scenarios relative to fish targeting scenarios is evaluated. The objective of the temperature targeting scenarios is to maximize stream length where temperature levels are below respective temperature targets with a budget constraint (again $100,000). Then juvenile chinook salmon populations under each scenario are computed.

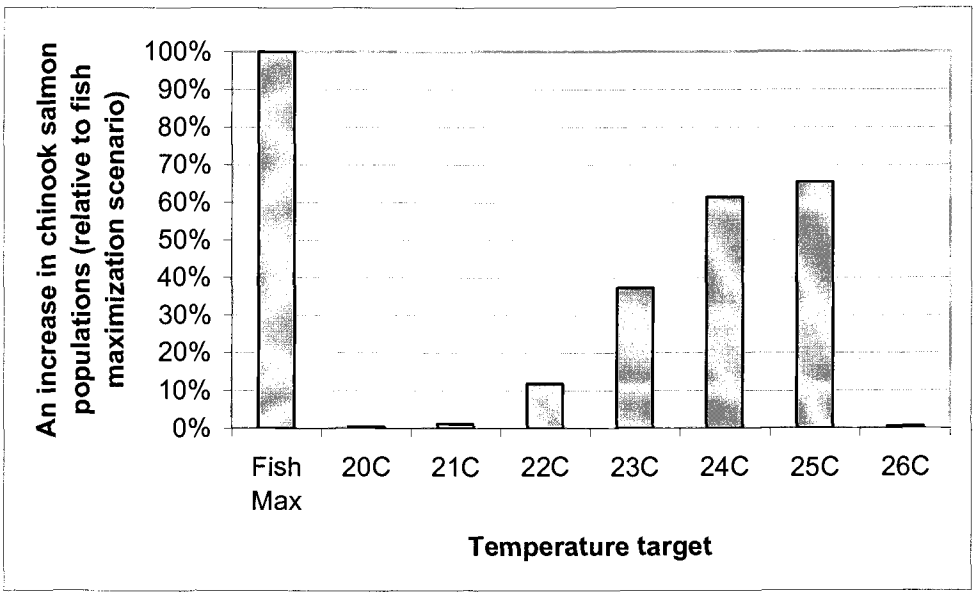

Figure 5.15: Total chinook salmon populations in a 40 year time frame under different targeting scenarios

Notes: The "no restoration scenario" and "fish maximization scenario" are the same as those in Figure 5.12.
It is assumed that chinook salmon are found only in the UGR mainstem and Sheep creek even after the restoration efforts have been implemented.[28]

[28] This assumption is reasonable due to the following two reasons. The first is that currently the UGR

Figure 5.15 shows that the temperature targeting scenarios are inefficient in the sense that they produce a smaller increase in chinook salmon populations than the fish maximization scenario with the same budget.[29] The difference is substantial (greater than 30% of expected fishery benefits). This indicates that for the allocation of restoration efforts to be efficient, fishery benefits should be considered. This result is consistent with past literature which found that targeting of physical criteria (such as water temperature) is inefficient compared to the targeting of environmental benefits (such as salmonid populations).

Figure 5.15 also presents the relative efficiency of varying temperature targeting scenarios in comparison to the fish maximization scenario. The figure shows that if the targeted temperature level is very high (*i.e.* 26 °C), then there is very little increase in juvenile chinook salmon. If the targeted temperature level is decreased, then chinook salmon populations increase compared to the "no restoration scenario," but as the target temperature level further decreases, the increase in population diminishes. Two factors explain this relationship between the targeted temperature levels and associated chinook salmon populations. The first is the curvature on the relationship between water temperature and chinook salmon populations. As seen in Chapter 4, the juvenile chinook salmon density is concave with respect to water temperature for water temperatures below 25.5 °C, and the absence of juvenile chinook salmon is assumed for water temperatures above 25.5 °C. Therefore, if water temperature target is 26 °C, reaches with water temperatures just above 26 °C are the focus of restoration efforts. In this case, there will be little effect on chinook salmon populations. However, once water temperature is below 25.5 °C, the

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mainstem and Sheep Creek are the primary habitat, and there is no model of recolonization of key habitat with the improvements of water temperatures in the basin. The second is that recolonization of chinook salmon is constrained by discharge levels in tributaries. If the discharge is small, adult chinook salmon will not be able to migrate into the tributary even if water temperatures have been improved. In this case, recolonization will be incomplete. Discharge in Sheep Creek is one of the largest among the study basin tributaries. (Personal communication with Joe Ebersole)

[29] Note the effects of a potential increase in thermal refugia in the UGR mainstem and Sheep Creek resulting from temperature decreases in tributaries are not considered in this analysis.

marginal benefit of the initial water temperature reduction on juvenile chinook salmon populations is very large. Subsequently, however, the marginal benefit decreases as the temperature levels decrease because chinook salmon population density is concave with respect to water temperature at this temperature range. This is one of the reasons why 25 °C targeting produces the largest juvenile chinook salmon populations among the temperature targeting scenarios. The second factor is associated with the spatial distribution of restoration activities. If targeted temperature levels are low (*i.e.* 20 °C), then conservation agencies will target streams with water temperatures just above 20 °C. Since reaches with cold water temperatures are in general located in upstream tributaries, they will be the primary target. For example, the shares of budget that are used in the UGR mainstem and Sheep creek are only 0 percent, 1 percent, and 18 percent for 20 °C, 21 °C, and 22 °C targeting scenarios, respectively. In this case, even if water temperatures in upstream reaches are decreased, it has little effect on the ambient water temperatures in the UGR mainstem and Sheep Creek. On the other hand, when targeted temperature levels are high, a large share of restoration activities is implemented in the lower part of the mainstem because high water temperatures are most frequently found in this stretch. As a result, 48% and 66% of the budget are used in the reaches in the UGR mainstem and Sheep Creek when targeted water temperature levels are 24 °C and 25 °C, respectively. This analysis indicates that the levels of temperature targets have significant impacts on the expected increase in chinook salmon populations.[30]

The efficient allocation of restoration efforts associated with rainbow trout are also examined here. Since the objective of this analysis is to compare spatial configurations of restoration activities for chinook salmon and rainbow trout, the same reaches considered as

[30] Again, in this analysis, the impact of cooling water temperatures in tributaries on coldwater patches in the mainstem and Sheep creek are not considered.

potential habitat for chinook salmon are also used as potential habitat for rainbow trout.[31] Figure 5.15 shows those reaches that receive restoration activities when the objective is to maximize rainbow trout populations in isolation (left) and when the objective is to maximize the sum of both chinook salmon and rainbow trout populations (right) with a budget constraint (again, $100,000).

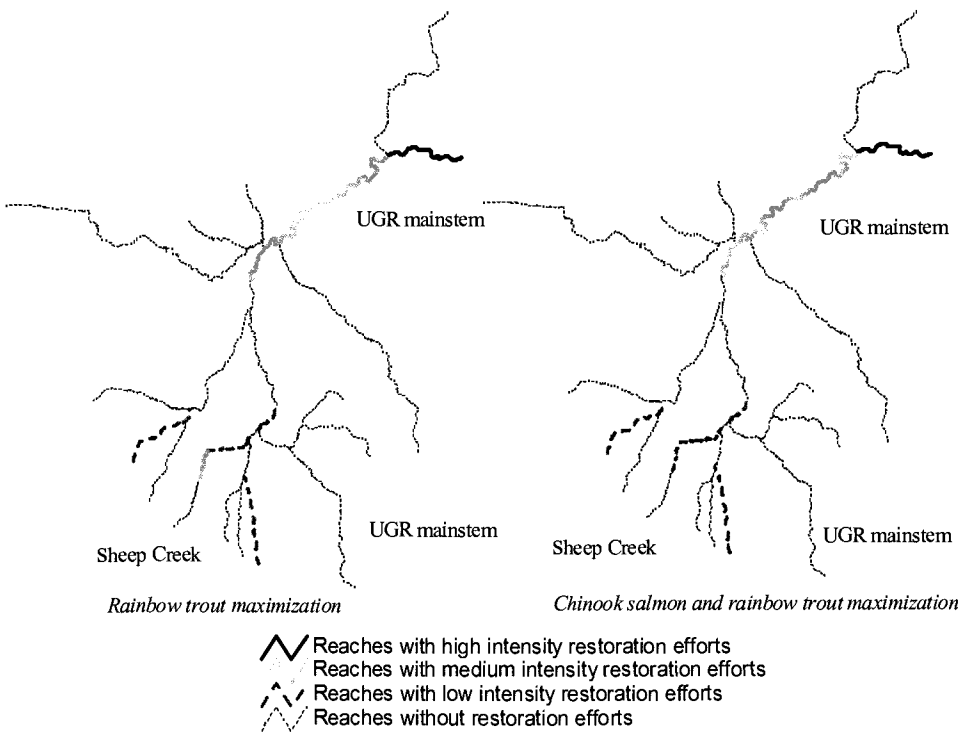

Figure 5.16: Restoration sites, by reach, when rainbow trout and the sum of chinook salmon and rainbow trout populations in selected reaches are maximized, with a budget constraint ($100,000)

[31] They are UGR mainstem (excluding reach UGR mainstem 7 and 8) and Sheep creek.

It is difficult to draw insights from Figure 5.16, because of the low R-squared of the rainbow trout density model and also because of the lack of information on habitat conditions in many reaches of the stream system which are potential habitat for rainbow trout. However, it is possible to observe that the efficient allocations of restoration efforts under the chinook salmon targeting (Figure 5.13) and rainbow trout targeting (Figure 5.16, left) closely resemble the allocation that maximizes the sum of both chinook salmon and rainbow trout populations (hereafter referred to as the combined scenario; Figure 5.16, right). In fact, the allocation under the combined scenario is almost identical to the rainbow trout targeting case. The only difference is the intensity in restoration efforts in one of the upstream reaches of Sheep creek. As a result, under the combined scenario, the expected rainbow trout population is nearly identical to the one obtained in the rainbow trout maximization scenario. Also, the combined scenario achieves about 90 percent of the chinook population increases realized under the chinook salmon maximization scenario. The reason the combined scenario performs well with respect to both chinook salmon and rainbow trout is the curvature of the fish density models. Table 4.8 shows both models have concave relationships between fish density and water temperatures for the typical water temperature ranges observed in the basin. Therefore, a response to a decrease in water temperatures is similar for both salmonid species.

There are two reasons why the combined scenario performs slightly better for rainbow trout than chinook salmon. The first is that the population of rainbow trout is larger than that of chinook salmon, and therefore in the combined scenario, rainbow trout receive a larger weight. The second is associated with the differences in temperature tolerance and habitat preferences between chinook salmon and rainbow trout. With respect to temperature tolerance, in the case of chinook salmon, it was assumed that reaches are not habitable if water temperature exceeds 25.5 °C. On the other hand, rainbow trout are more tolerant of high water temperatures than

chinook salmon and no upper limit in water temperatures are imposed.[32] Therefore, if water temperature is above 25.5 °C, it is assumed that there are no chinook salmon in that section, although rainbow trout may be present. In terms of the different preference of each species for habitat conditions, rainbow trout prefer riffles in summer, while chinook salmon prefer pools (personal communication, Joe Ebersole) and a larger number of riffles are found in the lower part of the UGR mainstem. These differences result in the combined scenario producing less of an increase in chinook salmon populations than the chinook salmon maximization scenario. Nevertheless, this analysis suggests that there is not a substantial trade-off in habitat restoration efforts between the two salmonid species.

Finally, implications of restoration efforts on coldwater patches and their effects on salmonid populations are discussed because coldwater patches play an important role in determining the populations and the distribution of salmonid species (Ebersole, 2001). A coldwater patch is defined as a discrete pocket of water more than 3 °C colder than the adjacent ambient streamflow (Ebersole, 2001). Coldwater patches are created either by surface tributary or groundwater inflows,[33] and riparian restoration efforts have implications for both types of coldwater patches. As for groundwater-fed coldwater patches, improvements in riparian conditions through restoration activities are expected to improve water retention capacity of stream banks, and as a result, the frequency of groundwater-fed coldwater patches is expected to increase. The effect of restoration activities on this type of coldwater patches is not considered in this study; this is another reason why the impacts of restoration activities on fishery benefits derived are likely to be underestimated.

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[32] Ebersole's data (2001) show that rainbow trout are present in a reach with water temperature of 27.5 °C.
[33] Following Ebersole (2001), the term groundwater inflow is used to describe any non-surface water inflows, such as inter-gravel and hyporheic inflows.

Tributary-fed coldwater patches are also expected to increase as a result of restoration efforts, but this effect is also not incorporated in this study. The preceding analysis on the relative effectiveness of water temperature reductions in the mainstem versus tributaries revealed that it is generally less costly to reduce water temperatures in tributaries than in the mainstem (Figure 5.9). Decreasing temperatures in tributaries will be beneficial for rainbow trout because their habitat is spread across the entire basin, including tributaries. On the other hand, the primary habitat for chinook salmon in the upper Grande Ronde basin is limited to the UGR mainstem and Sheep Creek. If tributaries are the targets, ambient water temperatures in the mainstem are not likely to decrease because temperature reductions in tributaries only have limited impacts on the mainstem. However, given current high water temperature levels in the mainstem, chinook salmon must rely on the existence of coldwater patches for their survival, and temperature reductions in the tributaries are expected to increase them in the mainstem. Therefore, even though ambient water temperature in the mainstem may not decrease significantly if the tributaries are the priority of restoration activities, chinook salmon in the mainstem are also expected to benefit from an increase in the frequency of coldwater patches.

A similar analysis can also be applied to Figure 5.14, in which chinook salmon populations under the fish maximization and temperature targeting scenarios are compared. The figure shows that as the targeted temperature levels decrease, an increase in chinook salmon populations diminishes. However, this analysis does not consider the effects of restoration efforts on coldwater patches. As the target temperature levels decrease (become more stringent), conservation efforts are increasingly applied to the reaches in higher elevations where water temperatures are in general lower. As a result, water temperatures in tributaries are decreased, which may result in an increase in coldwater patches in the mainstem and Sheep Creek. In this

case, the fishery benefits of, for example, 20 °C or 21 °C targeting scenarios are likely to be larger than those shown in Figure 5.14.

Incorporation of the effects of restoration efforts on coldwater patches and fishery benefits in the simulation analyses developed in this chapter was not feasible due to the lack of data on patch location and temperatures. However, as this qualitative analysis shows, it is important to consider coldwater patches when the interactions between riparian restoration efforts, water temperatures, and salmonid species are examined.

Chapter 6

Conclusions

The spatial and dynamic pattern of landscape changes has a profound effect on the supply of environmental services, including the provision of habitat for fish and wildlife. Spatial heterogeneity is a common feature of landscapes in the Pacific Northwest, most notably in areas important to the production of salmonid fish species. This heterogeneity complicates the design and implementation of policies to conserve the stocks of such species. To date, millions of dollars have been spent to improve habitat for salmonids, with mixed success.

The objective of this dissertation is to examine the spatial implications of riparian restoration activities in an Oregon watershed for the purpose of decreasing water temperature to benefit endangered salmonid fish species. To attain this goal, both theoretical and simulation analyses have been performed. The theoretical model defines an economically efficient allocation of conservation practices for a hypothetical stream with a range of hydrological and spatial characteristics. Conservation activities include flow augmentation, such as water conservation efforts as well as water leases, and riparian restoration efforts such as passive and active restoration. The theoretical framework is then used to develop an integrated hydrological, biological and economic modeling approach.

The empirical focus is on an important salmonid stream in Oregon. Specifically, the simulation analyses were applied to the upper Grande Ronde River basin of northeast Oregon, and a series of optimization (simulation) problems are investigated within the integrated models for different policy targets, including temperature reductions and enhanced fish populations. While there are numerous potential riparian restoration practices, the simulation analyses focused on passive and active restoration because they are the most popular restoration

activities in the basin. Such riparian restoration activities are expected to produce multiple benefits, including an increase in fish and wildlife populations and aesthetic improvements. Among these potential benefits, the focus in the simulation study is an increase in chinook salmon and rainbow trout populations only. Other cold water fish species, such as bull trout, will also benefit from riparian restoration, but they are not considered. In addition, passive and active restoration are expected to result in an increase in salmonid species in multiple ways, including a reduction in water temperatures through shading, recruitment of large and small woody debris, and providing nutrients through litter inputs and insects. Our focus here is on the effect of shading on water temperatures because high water temperature is the primary constraint of salmonid populations, and shading is the most important means to moderate water temperatures. As these assumptions suggest, it is important to recognize that the effect of restoration efforts on water temperatures and the populations of salmonid species considered in the simulation analyses are likely to be underestimated.

Through both the theoretical and simulation analyses, important insights on efficient allocations of restoration efforts have been gained. These key findings are summarized below. First, for this setting, the temperature criterion for rearing salmonid juveniles (17.8 °C (64 °F)) is not physically attainable in most stretches of the UGR mainstem, even in a 40 year time span, given the options considered here. On the other hand, it may be feasible to attain the temperature criterion in tributaries as the magnitude of temperature reductions obtained by shading is in general larger in tributaries. Second, localized effects of restoration efforts on temperature reduction are important for the achievement of small temperature reductions. However, as the desired magnitude of temperature reductions increases, the cumulative (longitudinal) effects become important. Third, it is possible that implementing restoration efforts in more distant reaches of the watershed is more efficient than efforts nearer the point of

monitoring. This kind of policy finding would not be possible without the spatial detail available for this study. Fourth, in terms of specific riparian restoration efforts, passive restoration is preferred to active restoration when the magnitude of desired temperature reductions is small and / or when the time frame considered is increased. Fifth, if the objective of conservation agencies is to maximize the stream length where water temperature decreases by a certain degree, then in general tributaries will be targeted first. Sixth, if agencies are concerned with absolute temperature levels, then the levels of those desired temperature targets have a significant impact on the spatial configuration of restoration efforts, and as a result, on the distribution of water temperatures in the basin. Seventh, temperature targeting scenarios are inefficient if the underlying goal is fish enhancement, given that they produce smaller numbers of fish populations than the fish maximization scenario, with the same budget. Thus, for the allocation of restoration efforts to be economically efficient, fishery benefits, not temperature reductions, should be considered. Finally, an efficient allocation of restoration efforts that maximizes the sum of the salmonid species (chinook salmon and rainbow trout combined) is likely to perform well compared to the individual species targeting scenarios, which suggests that there is not a substantial trade-off in habitat restoration efforts between these two key salmonid species.

While this type of analyses demonstrates the importance of representing spatial heterogeneity in riverine management, a number of extensions are needed. First, the role of changes in stream discharge needs to be explored. In many streams in arid portions of the Pacific Northwest, a reduction in discharge resulting from water withdrawal for irrigation is one of the primary reasons for elevated temperature levels; an increase in stream flows has been found to be a cost-effective method in decreasing water temperatures (Bartholow, 1991). Since there is no water withdrawal for irrigation in the study basin, we did not examine the effect of

stream flow augmentation. Further improvement in the WET-temp model, such as incorporating a discharge component, would allow analyses in settings where discharge changes and other options are possible.

A second need is to expand the analysis by including other benefits of riparian restoration efforts such as an increase in wildlife populations. In addition, incorporating other relationships between riparian vegetation and water temperature (other than shading) such as the effects through a change in the width to depth ratio, would also help improve the analyses.

The third extension is to develop this study into a benefit-cost analysis, which will provide insights on the optimal allocation of restoration activities. The optimal allocation would then be defined such that the restoration efforts are allocated so as to maximize the net present value of such efforts. For this goal, multiple steps are necessary. For example, more comprehensive benefits that accrue from riparian restoration efforts need to be captured. More complete habitat data in the basin is also needed to estimate the impact of riparian restoration efforts on fishery benefit. In addition, a relationship between juvenile salmonid populations and adult return will also need to be established. Finally, values would need to be attached to these outputs. If the goal of conservation activities is to attain an optimal allocation of conservation resources, then such extensions should be pursued.

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