ECOSYSTEM SERVICE VALUE OF WATER SUPPLY BENEFITS PROVIDED BY FOREST STANDS IN THE MATTOLE RIVER WATERSHED, CALIFORNIA: A BIOECONOMIC AND BENEFIT TRANSFER - SPATIAL ANALYSIS APPLICATION.

By

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ABSTRACT

Ecosystem service value of water supply benefits provided by forest stands in the Mattole River Watershed, California: A bioeconomic and benefit transfer – spatial analysis application

Max Kaufman

This thesis outlines an approach based on valuing ecosystem services for assessing the trade-offs in yields from forestland management. The ecosystem services valuation approach integrates ecology and economics to help explain the effects of land management on forest and watershed ecosystem services. These services, including the cycling of clean air and clean water supplies and provisions of habitat for flora and fauna, are essentially free and considered positive externalities. As such, these services are undervalued and consequently underprovided because the beneficiaries do not pay for them.

This thesis presents a simple bioeconomic model for estimating ecosystem service values. By using the benefit transfer method, Geographic Information Systems applications, and developing a forest hydrology streamflow model, I estimated ecosystem service values, for water supply benefits to fisheries provided by forest stands for the case of the Mattole River Watershed in Northern California. Results indicate that the remaining old growth stands in the Mattole Watershed provide more than $1,910,800 a year in water supply benefits to the region. This information may be useful in future analysis of the total economic impacts of how forest management affects water related benefits and ecological services.

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INTRODUCTION

Forests produce benefits to society such as timber, recreational opportunities, open space, wildlife habitat, and air and water purification. Non-use and non-market services provided by forests including the cycling of clean air and water supplies and provisions of habitat for flora, fish, and fauna do not come with a price, are essentially free and are seldom accounted. The fundamental environmental-economic problem with forestland conversion is that the loss in ecosystem services are not paid for by those who benefit from the conversion because those benefits are external to the market factors of supply and demand.

Consider forestland growing near a rural residential area that can be used for sustainable forest management or converted into roads and new housing development. Forestland is bought and sold in markets; however, the market demand for forestland is based on the private benefits that flow to the sellers, such as revenues from timber harvest or from selling land to a developer. The market ignores the benefits received by others who do not pay for them. Further, it ignores the costs to society of benefits lost as a result of such transactions. Since the demand for forestland does not reflect the benefits of intact forests that flow to society, the market will allocate less forest than the socially optimal amount.

Learning to value ecosystem services and developing incentives for private landowners who practice sustainable management is a first step towards renewing natural resources and maintaining healthy forests. Original valuation research is the preferred method when attempting to value ecosystem services. However, cost and time constraints severely limit the application of primary research methods. In such cases, a benefit transfer is considered acceptable. A benefit transfer is an economic application, which entails taking previously researched monetary values obtained from one location, termed the study site, and using those values at a different site, called the` policy site.

This thesis applies the benefit transfer method and outlines an ecosystem services approach that integrates ecology and economics to help explain the effects of land management on a forested watershed ecosystem. I developed a simple bioeconomic model to estimate ecosystem service values, for water supply benefits to fisheries provided by forest stands for the case of the Mattole River Watershed in Northern California. Integrating the results of a forest cover and water supply model with monetary values allocated from the benefit-transfer and utilizing Geographic Information Systems technology, has allowed me to predict the response of streamflow benefits to alternative forestland management practices in the Mattole.

The goal of this thesis was to estimate ecosystem service benefits of water supply provided by forestland in the Mattole Watershed and how land use change affects ecosystem service values. Specifically, what is the difference in streamflow benefits between a mature forest stand compared to a young second growth stand. This thesis answers these questions 1) what are forest ecosystems services regarding water flow in the Mattole 2) How much are they worth and 3) how might changes in land-use practices affect these ecosystem service values and the benefits they provide?

This thesis developed a useful model and tool for evaluating the relative differences in ecosystem service outcomes among various management options. This model can be used in future analysis to enable decision makers to better interpret on the ground data and visualize how forest management affects water related benefits.

LITERATURE REVIEW

Ecosystem services are the natural functions and processes of the environment, from which people receive some benefit. A wide range of ecosystem services has been described and a strong delineation of ecosystem services is now in use worldwide (Costanza et al. 1997a, Daily 1997, Millennium Ecosystem Assessment 2005, Wilson and Troy 2005). Most lists include air and water purification, nutrient cycling, soil formation and erosion control, climate control, and maintenance of biodiversity, resource, aesthetic, recreational, and cultural benefits. For example, forests sequester carbon, provide spotted owl habitat, and provide timber while deserts provide bountiful aesthetic and recreational opportunities, and spiritual and cultural significance.

Global forest ecosystem services have been valued at more than $4.7 trillion (Larson 2002). Forests are a key indicator of the health of the planet and they are part of a complex system that supplies and cleans the air we breathe and the water we drink. Woodlands can increase soil nutrient concentrations and contribute to gas regulation through their roles as carbon sinks (Boutton et al. 1999, U.S. Environmental Protection Agency 2012). Because trees remove carbon dioxide out of the atmosphere and store it within their boles and limbs, forests play a significant role in moderating global climate change.

Forests are also vital in regulating water flows and filtering pollutants, enhancing water quality and providing clean drinking water to people and habitat for aquatic organisms (Myers 1997, Keppeler et al. 2003). Dense forest cover may also help to produce the very steep slopes commonly found among coastal watersheds in the Pacific Northwest region. Sidle and others (1985) concluded that very steep slopes on geologic units similar to those found in the region would not have been able to form unless the slopes were stabilized by the protection of dense forest. The forest stabilizes the slope in four major ways: (a) a dense tangle of roots within the soil and penetrating down into fractured, weathered bedrock tends to hold the soil in place and stabilize the slope; (b) the large amount of water used by the vegetation keeps the water table lower than it would be without this water use; (c) the trees break the impact of falling rain so that surface erosion is minimized, and (d) duff on the ground absorbs water and prevents surface flow and attendant erosion. Thus, the forest may have a significant effect in shaping the landscape itself. The ground behaves like a sponge absorbing water, while slowing runoff and filtering out sediment and other pollutants, thus sustaining downstream aquatic habitats and water supplies.

Consequently, when forests undergo change, ecosystem services may be disrupted or redistributed, and their benefits eliminated (Kreuter et al. 2001, Wilson and Troy 2005). Logging in the Pacific Northwest has had a significant impact on hydrological functions and has degraded aquatic habitat (Keppeler and Ziemer 1990, Cafferata and Spittler 1998, Lewis 1998). Fragmentation and conversion of forests are also significant factors that influence the flow of benefits from ecosystem services. For example Kagi (2000) suggested that reducing forest conversion and deforestation might significantly reduce carbon dioxide (CO_2) emissions.

Meanwhile, national trends of forest fragmentation and conversion to other land uses, mainly development are increasing (Shih 2002). Forestland owners have limited options for gaining financial returns and consequently face increasing pressure to convert their lands. On the California coast, for example, a continued decline in the amount of high value, large redwood logs and relatively high operating costs may reduce landowners’ interest to maintain large areas of forestland in sustainable forest products operations (Stewart 2007). In fact, Stewart (2007) observed that one of the most significant trends in the California redwood region is the increase and extent of rural residential land use.

Development in forested watersheds can degrade water quality, which in turn will require downstream communities to employ high cost filtration systems for their drinking water (Coombe 1994). The developed land area of the United States increased more than 14 million hectares between 1982 and 2003 (White et al. 2009). A U.S. Forest Service report predicted that by the year 2035, housing and development will have spread across private forests totaling the size of the State of Washington (Little 2006). Over the next decade, between 200,000 and 550,000 acres of undeveloped or underdeveloped land will be required to accommodate the needs of new urban residents in California (California Department of Forestry and Fire Protection 2010).

Learning to value ecosystem services and developing incentives for private landowners to practice sustainable forest management is a first step towards renewing natural resources and maintaining healthy forests that provide life sustaining and socio-economic benefits. A benefit-cost analysis is a commonly applied valuation method for determining the feasibility of management objectives by comparing the benefits and costs of a particular action. For example, a benefit-cost analysis can inform a forest owner whether the costs of harvesting are less than the timber is worth in the market. If so, then an owner would be more likely to harvest. With the passage of the National Environment Protection Act, in the early 1970s, and the passage of President Ronald Reagan’s Executive Order 12291 in 1981, subjecting new federal regulations to cost-benefit analysis, benefit-cost analysis has become more common (Hackett 2006). However some forest ecosystem services, such as the provision of regulated surface water flows and fish habitat, are not directly used by people and not easily assessed using cost benefit analysis.

Garrett Hardin's (1968) Tragedy of the Commons illustrated the tension in the way public and private goods are valued and who benefits from the use or non-use of natural resources. Applied to forest management for example, a single timber harvest, implemented by a private landowner, may have relatively minor environmental impacts on a given watershed. However, as other forest landowners file and implement harvest plans, the cumulative effects of individual landowners' self-interest can be significant. The costs borne by society, the public, from hydrologic disturbances, property damage from increased winter floods, agricultural and domestic water supply losses from low summer flows, as well as effects on commercial and recreational fishing are not accounted for.

In addition to direct changes in benefits to humans, existence values, also known as non-use values, for fish and other aquatic species are not captured in monetary terms and are mistakenly considered more valuable only after their populations reach near extinction levels. Non-use values include biodiversity and intrinsic values that people receive from the mere contemplation of the survival of endangered populations (Sierra Club vs. Morton 1972). Non-use values are seldom included in analyses of forest management, resulting in unsustainable and consequently uneconomical long-term decision-making. To only consider the economic benefits of timber harvest and disregard ecological costs can lead forest owners to harvest sooner or more than they would with full accounting, and it is likely that the costs, for example, in reduced salmon populations and water flows, will not be considered.

Classifying benefits into use and non-use categories will allow for the estimation of the total economic value of a watershed's resources (Barbier 2000). If the non-use benefits of forest ecosystems were valued and quantified, decision makers would be supplied with additional information. This would enable them to consider more fully the social and environmental costs and benefits of land-use management (Troy and Wilson 2006). Bingham et al. (1995) argued that the ability to estimate the total value of ecosystem services is instrumental to integrated environmental decision-making, sustainable business practice, and land-use planning at multiple geographic scales and socio-political levels.

Economists have developed valuation methods to estimate these ecological goods and services. Hotelling (1949) estimated the travel demand for National Parks, a common method used to measure recreational use benefits (Englin and Shonkwiler 1995, Sohngen et al. 2000, Parsons 2003). The travel cost method measures the time and the expenses incurred while traveling to visit a site, and therefore peoples' willingness to pay to visit the site can be estimated based on the number of trips that they make at different travel costs.

The contingent valuation method is another popular tool economists use to measure non-use environmental benefits (Loomis 1987, Bell et al. 2003, Carson et al. 2003). This involves directly asking people, in a survey, how much they would be willing to pay for specific environmental services.

Hedonic pricing is a method used to estimate ecological values that directly affect prices in the housing market that reflects the use value of local environmental attributes (Schultz and King 2001, Sieg et al. 2000). For example, properties adjacent to open space or parks typically have a higher per-acre value than similar properties located near major roads.

The avoided cost method estimates use values based on either the cost of avoiding damages due to loss or the cost of replacing ecosystem services. For example, New York City officials invested $1.5 billion on restoring the Catskill Watershed for its provision of water purification services instead of constructing a water filtration plant with an estimated cost of $8 billion (Coombe 1994). The city of Denver also has plans to match the U.S. Forest Service's

$16.5 million investment toward forest treatment and watershed protection projects through the From Forests to Faucets Partnership. Following the Buffalo Creek and Hayman fires, Denver Water has spent more than $10 million on water quality treatment, sediment and debris removal, reclamation techniques, and infrastructure projects (Denver Water 2012).

A more integrated approach to quantifying natural resources and the economies that depend on them is the use of bioeconomic models. Economic analysis of a fishery, for example is invariably based on a bioeconomic model, which combines an economic model with a biological model of population dynamics and biological parameters (Massey et al. 2006). For example, Loomis (1988) estimated losses of $1.7 million over 30 years in recreational and commercial fishery benefits as a result of future timber harvest in the Siuslaw National Forest, OR. He accomplished this by combining a series of fish habitat and population models that related sediment, temperature, debris and watershed conditions to alternative timber harvest levels and carrying capacity for fisheries. Once fish populations were projected, using the cost of the trip and number of trips taken, recreational and commercial catch-to-escapement ratios were determined and economic linkages were established.

Subsequently, Loomis et al. (1995a) quantified the economic benefits to deer hunters of maintaining more stands of oak in forested areas of northern California. Specifically, the authors valued oak woodlands lost through forest in-growth in the absence of fire to Douglas-fir (*Pseudotsuga menziesii*) or a mix of coniferous and hardwood trees. The authors suggested that the United States Forest Service standards for commercial timberlands might not adequately consider the potential value of hardwood components to maintain biodiversity. They found that the abundance of hardwoods in habitats used during the late summer and fall can directly influence the number of buck deer in forested habitats and, indirectly, the subsequent number of deer tags issued and the quality of hunting conditions. Using a bioeconomic model, the authors correlated deer use with basal area of oak dominated habitats to human use in order to calculate an economic use-value for deer.

Original research is the preferred method when attempting to value ecosystem services. However, cost and time constraints severely limit the application of primary research methods, such as those mentioned above. In cases where less rigorous approaches are acceptable, the benefit transfer is an innovative methodology that provides decision makers with the information necessary to perform benefit-cost estimates. A benefit transfer is an economic application, which entails taking previously researched monetary values obtained from one location, termed the study site, and using those values at a different site, called the policy site. Wilson and Hoehn (2006) tracked the first benefit transfers to the mid-1980s. Benefit transfers have become a common practice in environmental cost-benefit analysis. The United States Environmental Protection Agency (EPA) developed guidelines for its own benefit-cost analysis and suggested that such "off-the-shelf methodologies" and studies serve as the basis for benefit-cost analysis (Desvousges et al. 1992, Smith et al. 2002).

Although benefit transfers are quicker and less costly than doing primary valuation, the validity of benefit transfer estimates is often debated (Johnston et al. 2005, Desvousges et al. 1992). There are a number of sources of error in benefit transfer estimates. When estimates of non-market goods and services are obtained from an original study, any error inherent in the original study is carried forward. Common errors include incorrectly estimating the demand for a good, an incorrect assumption about the population and site characteristics, poor quality of research, and difficulties in measuring existence values (Desvousges et al. 1992, McConnell

1992). Consequently, benefit transfers can only be as accurate as the initial value estimate. Further, there will always be some error in the transfer of estimates to the policy site.

Tests of benefit transfers provide some empirical evidence on the relative validity of the transfer. Several tests have estimated a range of error for transfers of 4% –39% (Loomis 1992, Parsons and Kealy 1994). An important factor affecting the validity of benefit transfers is the degree of similarity between a study site and a policy site. Loomis et al. (1995b) and VandenBerg et al. (2001) discovered low transfer errors when intra-region transfers were compared to inter-region transfers and when affected populations shared common experiences and attitudes.

There are two general types of benefit transfer, value transfers and function transfers. Value transfer is the simplest method and is either, the transfer of a range of estimates from low to high, an average, or the reporting of a value as a single point estimate. For instance, if three separate valuation studies are found to estimate improved water quality benefits to be worth $1 million, $2 million, and $3 million, then a value transfer for a new policy site could potentially be worth between one and three million dollars, an average of $2 million, or any of the three point estimates. Function transfers involve applying functions or statistical models, which include explanatory variables such as demographics, education and income. Transferring a regression equation and coefficients from a study site to a policy site and then adjusting the variables to represent policy site characteristics, such as population, will generate a new benefit estimate.

Choosing between a value transfer and a function transfer method is mostly determined by the quantity and quality of original valuation data. Rosenberger and Loomis (2003) suggest when differences between study site and policy site are substantial and demand coefficients are available, function transfers may be appropriate. Value transfers are best suited when study sites and policy sites are similar in many respects, including type of resource or commodity, minimal differences in population size and attitudes, and similarity of site locations (Rosenberger and Loomis 2003).

John Loomis (1996) conducted a value transfer for the U.S. Army Corps of Engineers to measure the benefits of removing four dams on the Lower Snake River and restoring the ecosystem and the anadromous fishery. The federal agency was interested in non-use values of increased salmon populations. Loomis defined the policy site context as the Pacific Northwest and used a contingent valuation approach to estimate passive use values as per household annual willingness-to-pay for an increase of 47,471 native salmon. After conducting a literature review and screening the studies for relevance and quality, five original salmon passive use valuation studies were chosen. The original studies estimated households' willingness to pay between $32 and $227 per year to help increase salmon populations.

To determine the potential range in passive use values for the Lower Snake River:

$$[[\mathit{annual\ houshold\ willingness\ to\ pay} \times \mathit{number\ of\ households}] \div [\mathit{change\ in\ fish\ population\ at\ the\ policy\ site}]] \times [\mathit{change\ in\ fish\ population\ at\ the\ policy\ site}]$$

Total salmon passive use values were calculated resulting in estimates ranging from $151 million to $542 million for the Lower Snake River site. This range was obviously quite large. Yet, when the lower value, was combined with other use benefits of dam removal such as recreational benefits and improved quality of drinking water, and weighed against the projected costs of removing the dams, this benefit transfer contributed to answering the policy question in favor of dam removal.

In addition to new methods of economic valuation, spatial analysis tools can be applied to the assessment of ecosystem services. Geographic Information Systems (GIS) technologies have enhanced abilities to demonstrate the effects of land-use change by mapping land-use spatially and over time. Remote sensing provides high resolution hydrologic and vegetation data layers (USDA 2011). The U.S. Forest Service utilized GIS and remote sensing in impact assessment for watershed restoration (Fisk et al. 1996) and for mining activities in and around national forests (Maus et al. 2003). Li (2005) used GIS to assess environmental impacts from urbanization using indicators of quantity, quality, location, and morphology to characterize land development patterns.

While GIS based analyses are multiplying rapidly, only a few examples illustrate efforts to compile spatially explicit estimates of ecosystem service values. Kreuter et al. (2001) used economic valuation coefficients and LANDSAT imagery to quantify changes in urban sprawl and ecosystem service values. Although authors reported an estimated 29% increase in urbanized land use in San Antonio, Texas, there appeared to be only a 4% decrease in ecosystem service values due to a significant increase in the area of woodlands, a relatively higher valued ecosystem coefficient. Troy and Wilson (2006) designed a spatially explicit decision framework to estimate ecosystem service flow values by land cover class in three states and by alternative development scenarios.

Ecosystem management is inherently a multifaceted task considering the complexity and scales at which ecosystems function. Clearcutting, conversion, and other land-use changes disturb forest ecosystems and can result in a disruption and redistribution of the flow of benefits. Mapping changes in ecosystem service values can enable decision makers to better interpret on-the-ground data and visualize how land-use change in a specific location affects the provision of market benefits like timber as well as non-market benefits like water regulation and the supply of aquatic habitat.

In this thesis, I developed a simple bioeconomic model of the response of water supply benefits to alternative forestland management practices in the Mattole Watershed, northern California and how those could be valued in terms of benefits to fisheries. While a number of ecosystem service benefits related to water flows could have been chosen for valuation, anadromous fish runs are highly significant in the Mattole as evidenced by over 30 years of efforts in fisheries restoration by community groups including the Mattole Salmon Group and the Mattole Restoration Council (Mattole Salmon Group 2010).

In order to pursue these questions first, I established a relationship between differences in forest cover and changes in water supply. This involved the use of two models. These included a streamflow model that related increases in water supply to selective harvesting (Ziemer 2000, Lewis et al. 2001). The next model translated increases in forest age to decreases in evapotranspiration rates (Moore et al. 2004). Once the change in forest cover had been translated into changes in water supply, the economic linkages could be established.

Integrating the results of the forest cover and water supply model with the monetary values allocated from the benefit-transfer and then mapping those values according to forest cover, and utilizing Geographic Information Systems applications, allowed me to develop a spatially explicit model predicting how changes in forest cover would affect ecosystem service values.

STUDY SITES

The first step in the value transfer was defining the policy site; in this case the policy site was the Mattole Watershed in northern California. Following the description of the Mattole, site characteristics of the two study sites, Caspar Creek Experimental Watershed in the western Coast Range of California and the H. J. Andrews Forest in the western Cascades of Oregon are described. The study results from these two well-studied experimental watersheds were used to predict how changes in forest cover affects water supply in the Mattole Watershed. All three study sites are considered Pacific Northwest watersheds and naturally exhibit similar characteristics such as climate, hydrology, geology, and forest cover.

The Mattole River Watershed

The Mattole Watershed lies on the eastern side of the King Ranges, part of the Coast Range, about 42 km south of Arcata and 467 km north of the Golden Gate on San Francisco Bay (Figure 1). The river is 100 km long, and 877 km of perennial streams drain about 78,700 ha of watershed (Downie et al. 2003). The climate is characterized as Mediterranean with cool wet winters and dry warm summers. Average seasonal temperatures range from 1 to 38 ˚C and average annual rainfall between 1500 mm – 2500 mm, with extreme annual events measuring more than 6000 mm (Downie et al. 2003). More than 50% of the watershed's vegetation is comprised of mixed conifer and hardwood forests, with over half of the watershed covered by young stands that have an average size of 30-60 cm at breast height (Downie et al. 2003). The mature forests of the Mattole also provide nesting habitat for the Northern Spotted Owl (S*trix occidentalis*), and the Marbled Murrelet (*Brachyramphus marmoratus*).

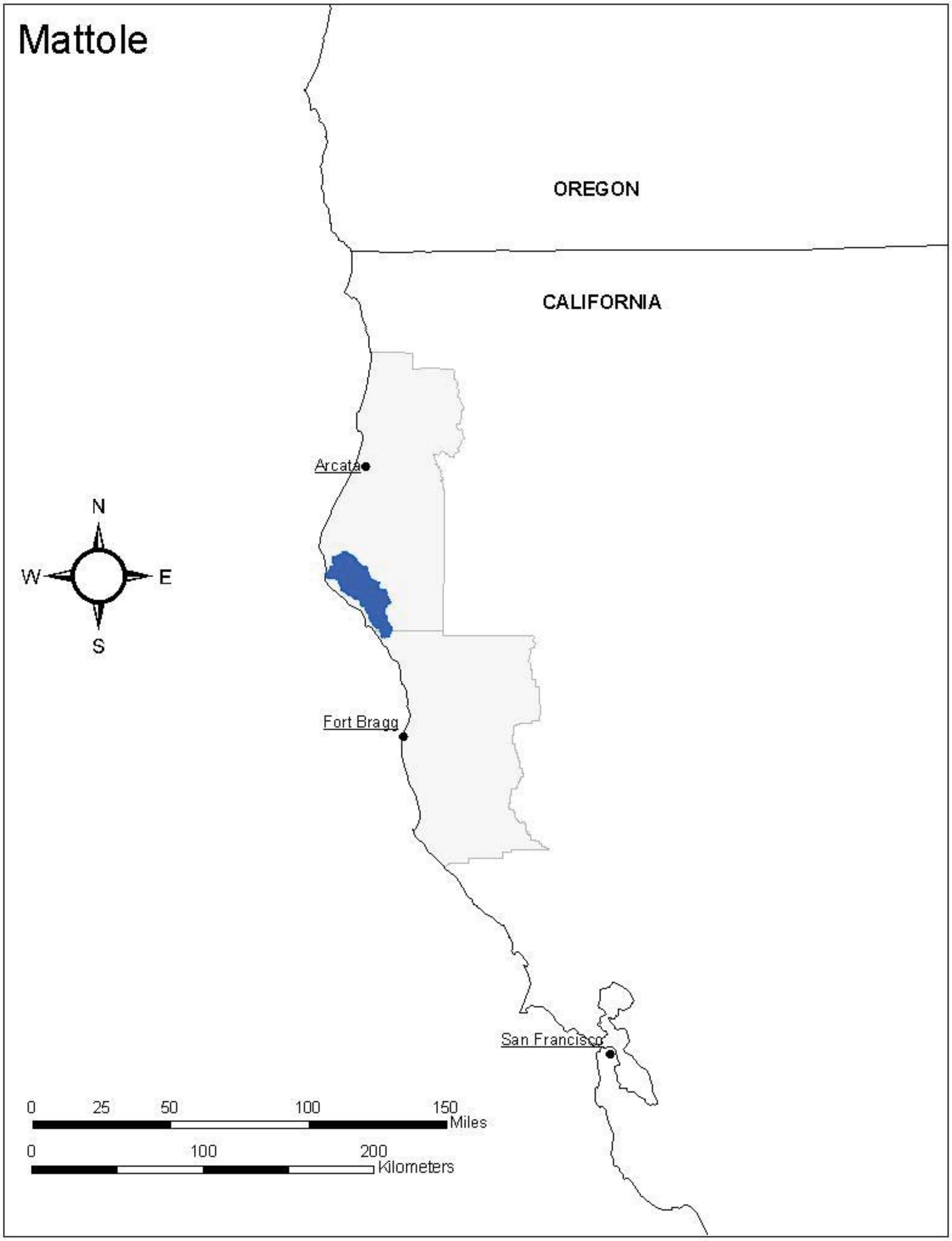

Figure 1. Study site location.

Native American settlement of the area dates back to 900 A.D. The Yurok displaced the Wiyot tribes and the Sinkyone and Mattole groups arrived in the region approximately 600 – 700 years ago (Whistler 1979). John Hill was the first recorded white explorer to enter the Mattole in 1854 (Downie et al. 2003). Many early ranchers followed, raising sheep and cattle to supply the gold rush market (U.S. Bureau of Land Management 2005). Around the turn of the 20th century the production of tannins from tan oak (*Lithocarpus densiflorus*) for processing leather emerged and the tanbark industry remained until 1940 (U.S. Bureau of Land Management 2005).

The first large scale timber harvesting began in the early 1950s. By 1974, close to half of the forests had been tractor logged and skidded downhill to landings and roads low on the slopes and often adjacent to streams (Figure 2). Only 8% of the mature redwood (*Sequoia sempervirens*) and Douglas-fir (*Pseudotsuga menziesii*) forest remain and are in old growth groves in private ownership or on public lands managed by the Federal government or the State of California. Beginning in the 1960s, a “back to the land” movement brought new settlers with the desire to live on the land with minimal environmental impact (U.S. Bureau of Land Management 2005).

Currently, the population is estimated at about 2,200 and is centered near the communities of Petrolia, Honeydew, Ettersberg, Thorn Junction, and Whitethorn (Figure 3). Descendants of early ranchers still manage more than one-third of the watershed in private forest and grazing lands. Another one-third is in parcels zoned rural-residential, fifteen percent is managed by the federal Bureau of Land Management (BLM), thirteen percent by industrial timber companies, and the remaining ten percent is split between the Sinkyone Wilderness State Park and other private lands (Figure 4).

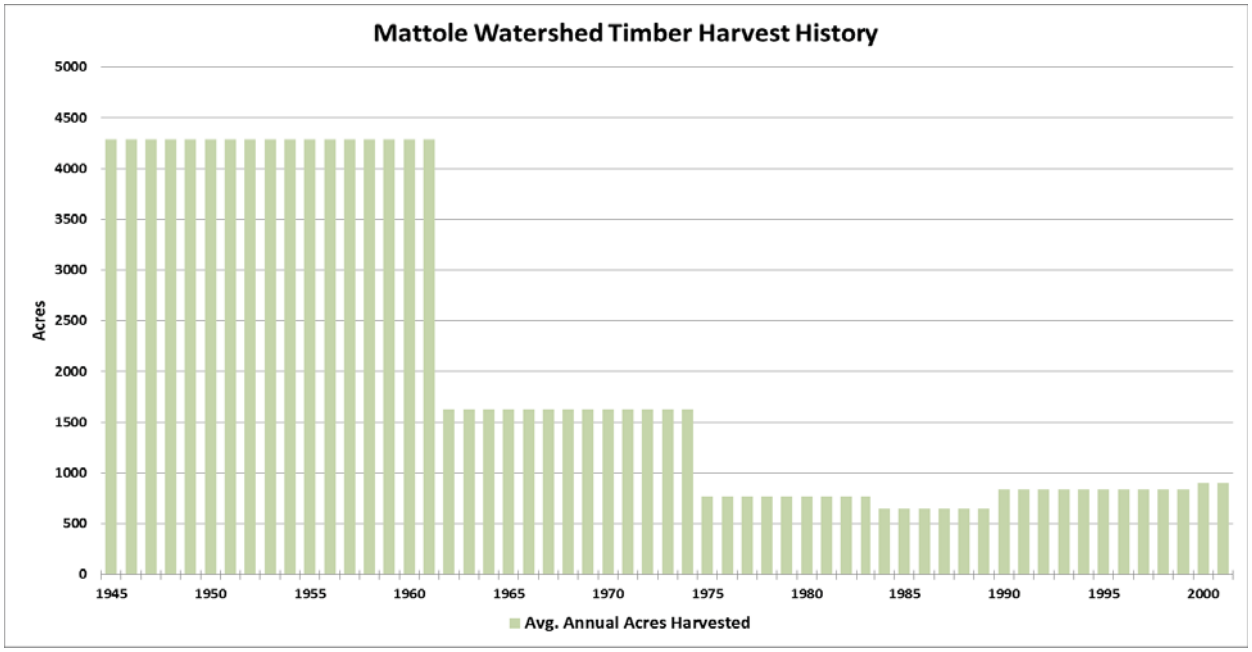

Figure 2. Timber harvest history, in the Mattole Watershed (Downie et al. 2003).

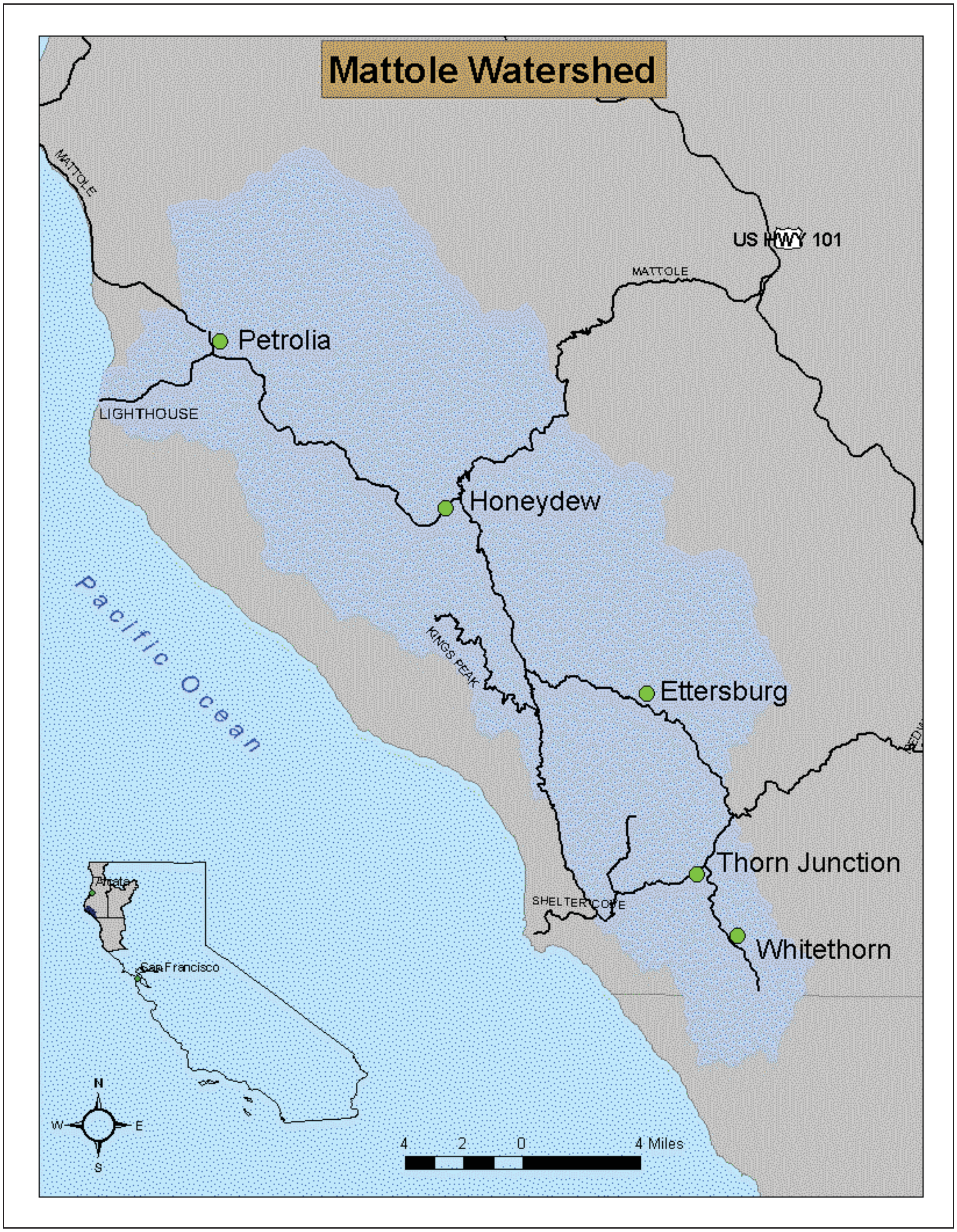

Figure 3. Population centers in the Mattole Watershed (Source: Humboldt County Community Development Services 2009).

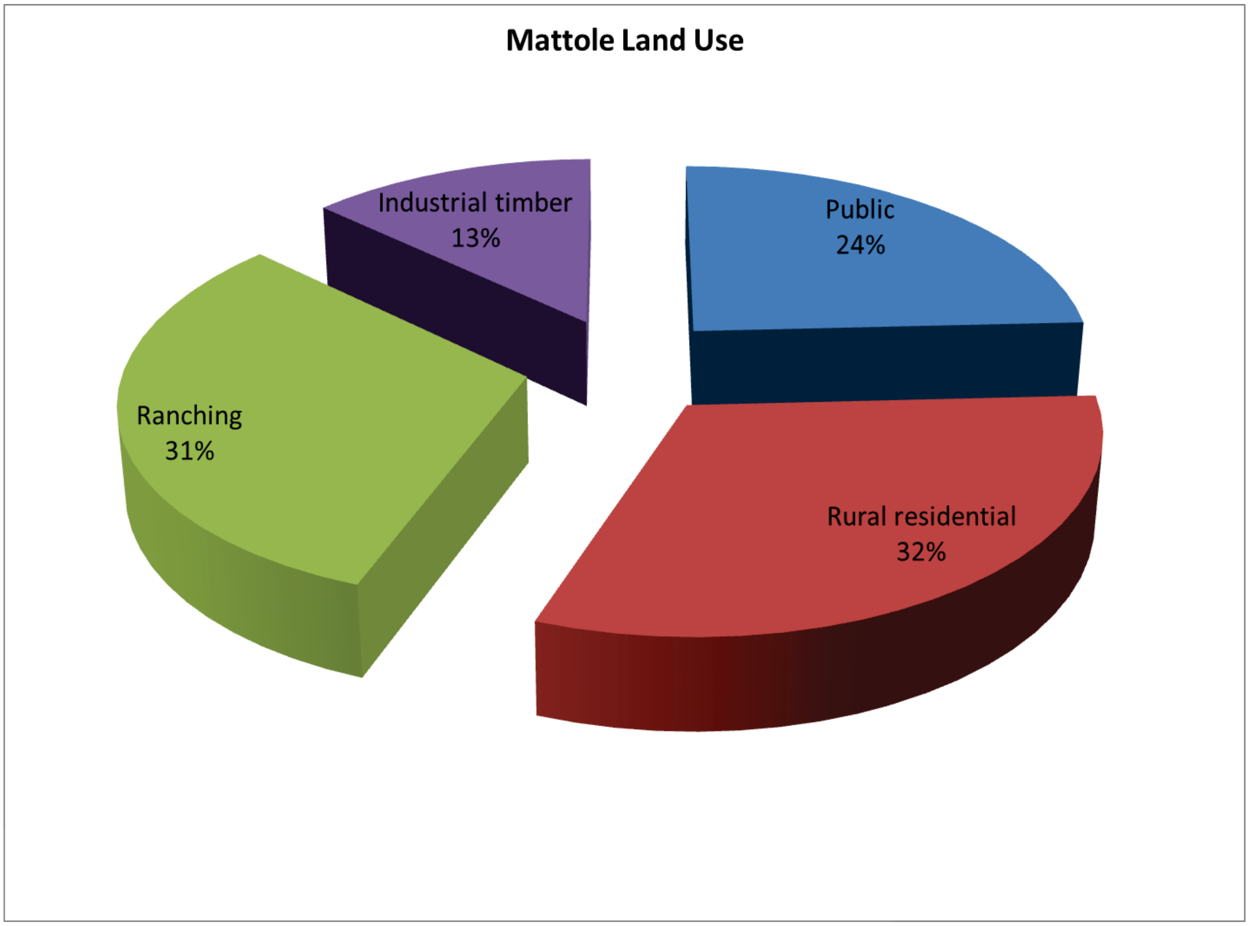

Figure 4. Land use in the Mattole Watershed (Downie et al. 2003).

Local unemployment was estimated at fifty percent in 1999. Much of the available work is seasonal and much of the area is alleged to harbor a large underground marijuana cultivation economy (Downie et al. 2003).

Although long-standing ranchers and back-to-the-landers often have conflicting views on land use, the people in the area are willing to work together to find common ground. The Mattole Restoration Council, The Mattole Salmon Group, and The Institute for Sustainable Forestry, local non-profit organizations, have been successful for several decades cooperating with landowners and government agencies on restoration efforts aimed at endangered species and sustainable timber management (U.S. Bureau of Land Management 2005).

In 2002, the Mattole River was declared a 303(d) impaired waterbody under the Clean Water Act (U.S. EPA 2003). The key concern was the decline of threatened coho (*Oncorhynchus kisutch*) and Chinook (*Oncorhynchus tshawytscha*) salmon and Northern California steelhead (*Oncorhynchus mykiss*) fisheries resulting from excess sediment load, elevated water temperatures, and summertime low flows associated with historic timber practices, grazing and agriculture (California Coastal Commission 2006).

The decline in fisheries coincided with the intensive logging carried out beginning in the 1950s. A 1960 USFWS study reported actual fish population of around 2,000 Chinook salmon, 5,000 coho salmon, and 12,000 steelhead trout. However, potential fish populations based on habitat characteristics, of over 15,000 Chinook salmon, 20,000 coho salmon, and 20,000 steelhead were estimated. The decline of the salmon has continued to be recorded (Figure 5). In 2010, The Mattole Salmon Group reported total basin populations of a couple hundred or fewer salmon (Mattole Salmon Chronicle 2010).

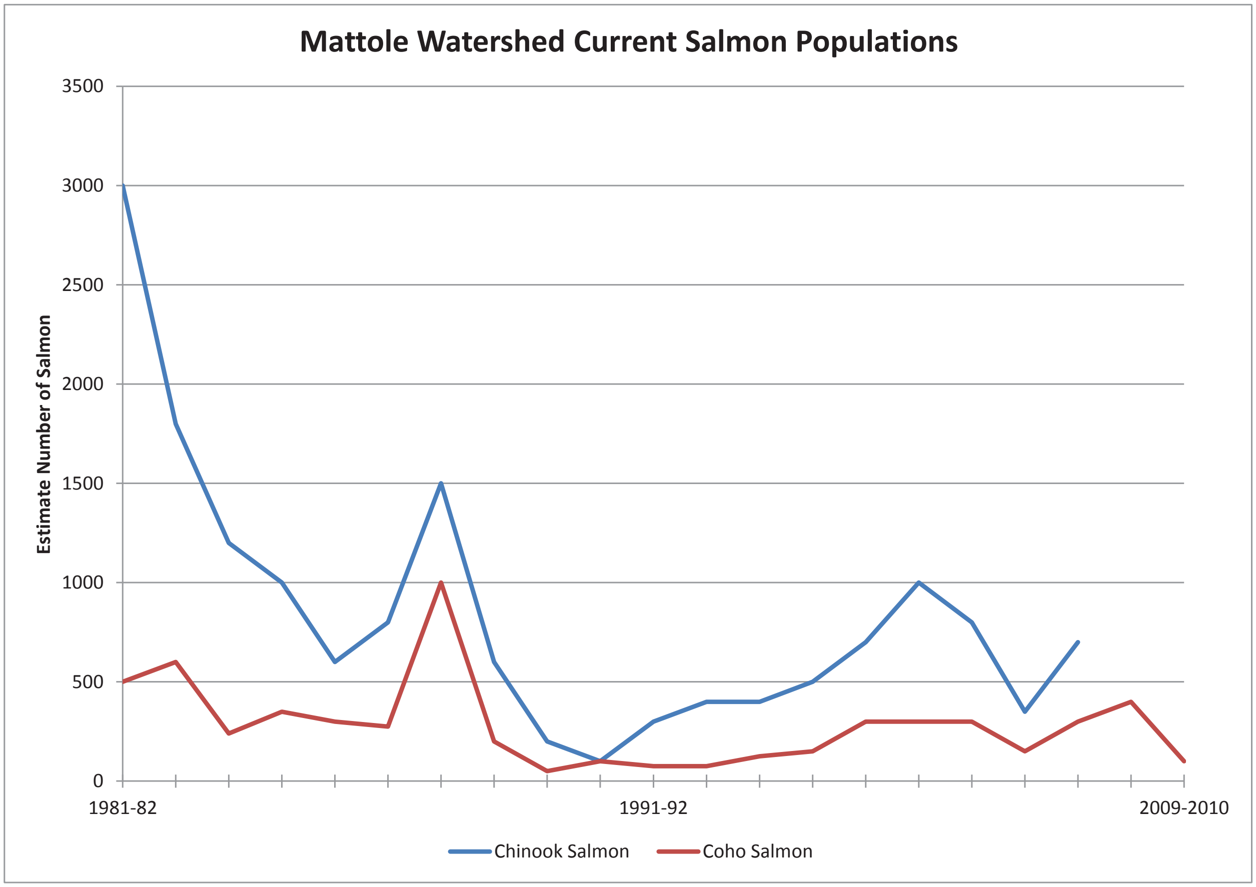

Figure 5. Mattole Watershed current salmon populations (Mattole Salmon Chronicle 2010).

Caspar Creek Watershed

Data used to develop the stream flow model for the Mattole were drawn from Caspar Creek in conjunction with data from the H. J. Andrews Experimental Forest. The 473-ha North Fork and the 424-ha South Fork of Caspar Creek Experimental Watershed are about 7 km from the Pacific Ocean, on the Jackson Demonstration State Forest, 10 km south of Fort Bragg, California (Figure 7). Prior to treatment, the watersheds supported a 90-year old second-growth forest dominated by coast redwood (*Sequoia sempervirens*) and Douglas-fir (*Pseudo Tsuga menziesii*),with minor associated western hemlock (*Tsuga heterophylla*), and grand fir (*Abies grandis*). The climate is characterized as Mediterranean, having dry summers with coastal fog and mild and wet winters with temperatures from 10 to 25 ˚C and average annual precipitation about 1200 mm (Ziemer et. al 1998).

Stream flow, suspended sediment, and bedload have been monitored since 1962. From 1963 to 1967, both forested watersheds were measured in an "untreated" condition. In 1967, logging roads were built in the South Fork. From 1971 through 1973, about 65% of the stand volume was selectively cut from the South Fork watershed, while the North Fork remained as an untreated control. Logging began in the North Fork in 1985 and ended in 1991. The timber volume removed from the North Fork watershed approximated that cut from the South Fork in the early 1970s, but clearcutting rather than selective harvest was used. The size of clearcut blocks in the North Fork ranged from 9 to 60-ha and occupied 35% to 100% of individual tributaries.

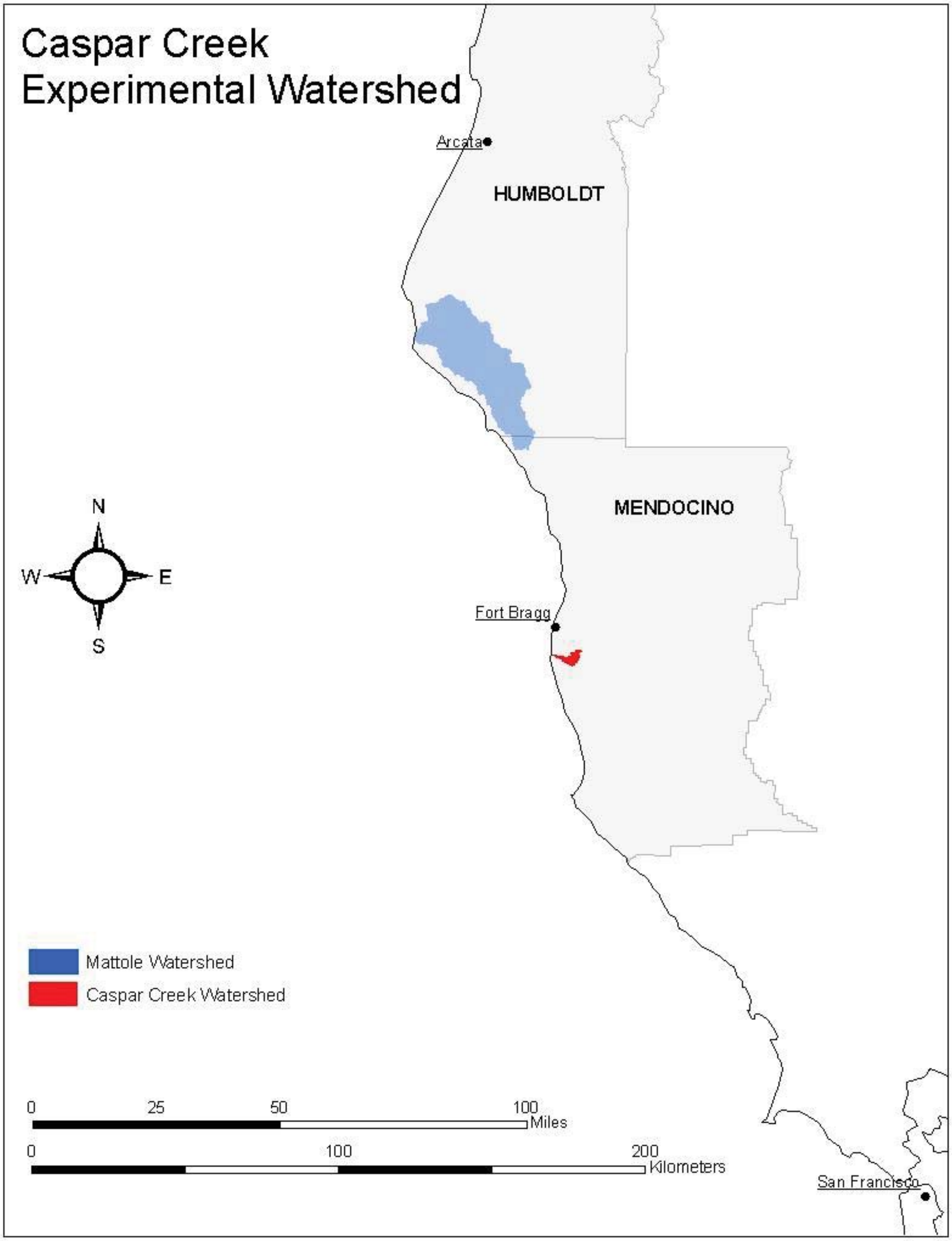

Figure 6. Caspar Creek Experimental Watershed near Fort Bragg, California.

Logging-induced changes in the South Fork's peak streamflow were greatest for the first storms following lengthy dry periods. There was no significant change in the largest peakflows (> ten-year return interval) after selectively logging the South Fork. Peak streamflows following clearcut logging in the North Fork behaved similarly.

Annual runoff in the South Fork increased between 9% and 30% for the first five years after selective logging (Ziemer 2000). This is equivalent to an average annual increase in water yield of 900 m^3 ha^{-1}. The increased annual water yield slowly declined and flows returned to pre-logging levels over the following 15 years. After clearcut logging in the North Fork, the increase in annual runoff varied from 9% to 58% in the first seven years, an average increase of 1119 m^3 ha^{-1} (Lewis et al. 2001). However, Keppeler et al. (2009) indicated there was still no trend towards a return to pre-logging levels.

Selective logging of the South Fork increased summer low flow for each of the first three years by about 80% from that predicted by the pre-logging regression. This increased summer flow declined with regrowth of the vegetation so that within seven to eight years after logging, summer low flow had returned to pre-logging levels. Clearcut logging about 50% of the North Fork produced minimum summer flows averaging 146% larger than predicted. The increased flow was greatest during the first two years after clearcutting, but seven years after cutting, summer flow was still 112% above prelogging levels.

Enhancement of stream flow at Caspar Creek can be explained by the magnitude of alteration of forest vegetation (Keppeler and Ziemer 1990, Rice et al. 2004). The removal of this forest cover greatly reduces water use and evapotranspiration processes and therefore results in increases of peak flows (Reid and Lewis 2007). Because less precipitation is intercepted by forest, an increase of run off and peak flows in addition to an increase in sediment loads, and a decrease of summer flow are possible (Keppeler and Ziemer 1990, Rice et al. 2004). Hence;

$$X_{2A} - X_{1A} = ET_A$$

$$ET_A = X_{2A} - X_{1A}$$

Where X_{1A} is the average annual water yield m^3 ha^{-1} of 90-year old forest stand type A, pre-harvest, X_{2A} is the average annual water yield m^3 ha^{-1} of 90-year old forest stand type A, post-harvest, and ET_A is the average annual water use m^3 ha^{-1} of 90-year old forest stand type A.

From Ziemer (2000) then, after selective harvesting a ha of 90-year old coast redwood (*Sequoia sempervirens*) and Douglas-fir (*Pseudo Tsuga menziesii*) dominated stand, there will be an average annual increase in water yield of 900 m^3 ha^{-1}. Then, average annual water usage of this forest stand type is represented by;

$$ET_A = 900 \text{ m}^3 \text{ ha}^{-1}$$

The H. J. Andrews Experimental Forest

Extensive data collection has been completed for the H. J. Andrews Experimental Forest in the western Cascades of central Oregon, and in particular, research has focused on how increased forest age affects water usage and evapotranspiration. The H. J. Andrews Experimental Forest is situated near Blue River, Oregon (Figure 7). The climate is characterized as Maritime, having dry cool summers with mild wet winters with mean monthly temperature ranges from 1 °C to 18 °C and average annual precipitation about 2300 mm (Moore et al. 2004). Research here has evaluated the effects of stand age, species composition and sapwood area on transpiration of two forests.

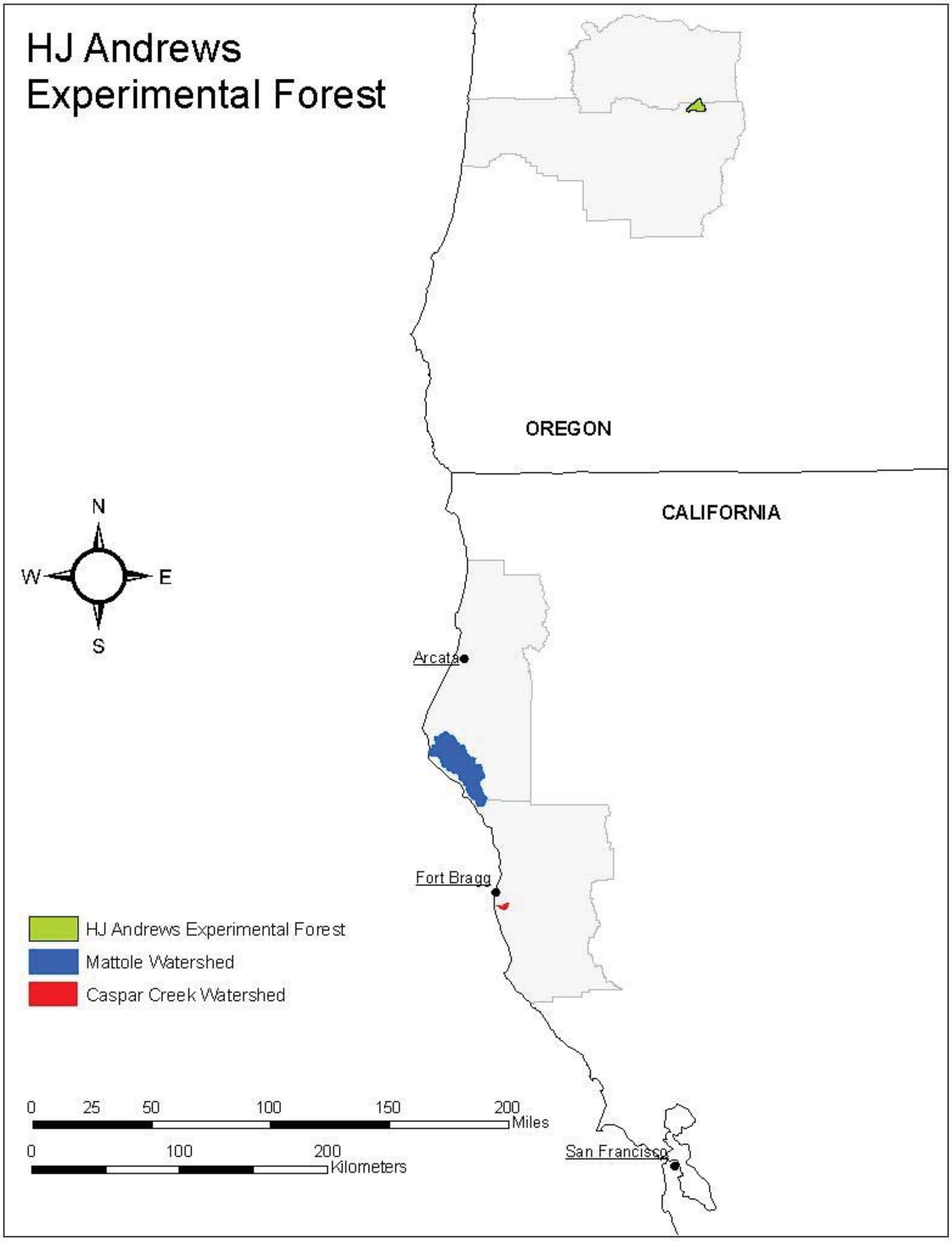

Figure 7. H. J. Andrews Experimental Forest, near Bend Oregon.

Two stands in an experimental watershed were compared. A 450-year old stand had a mix mainly of Douglas-fir (*Pseudo Tsuga menziesii*) and western hemlock (*Tsuga heterophylla*). The vigorously growing 40-year old stand consisted of mostly Douglas-fir (*Pseudo Tsuga menziesii*) and also a substantial angiosperm component (Moore et al. 2004).

During the growing season, sap flow was measured in both stands with constant-heat sap flow sensors, in Douglas-fir (*Pseudo Tsuga menziesii*) red alder (*Alnus rubra*), and western hemlock (*Tsuga heterophylla*) trees. Sap flow measurements were used to calculate the degree to which variances in age and species composition affected water use. Stand sapwood basal area was estimated based on a forest survey.

Estimated differences in water use as a result of differences in age, species composition and stand sapwood area were used to estimate transpiration from late June through October 2000 (Moore et al. 2004).

Transpiration was higher in the young stand because of greater sap flux density (sap flow per unit sapwood areas) by age class and size related differences, and species and greater total stand sapwood area. Overall, sapwood basal area was 21% higher in the young stand than in the old stand. In the old-growth forest, western hemlock is an important co-dominant, accounting for 58% of total sapwood basal area, whereas Douglas-fir is the only dominant conifer in the young stand. Angiosperms accounted for 36% of total sapwood basal area in the young stand, but only 7% in the old stand. For all factors combined, Moore et al. (2004) estimated 3.27 times more water use by vegetation of the young stand. The authors then extrapolated and suggested that water usage of 450 year-old stands is 3.27 times less than that of 40 year-old stands of this type (Moore et al. 2004). Tree age had the greatest effect on stand differences in water use, followed by differences in sapwood basal area, and finally species composition.

Hence;

$$ET_A = 3.27 \times ET_B$$

$$ET_B = ET_A \div 3.27$$

Where ET_A is the average annual water use $m^3\ ha^{-1}$ of 40-year old forest stand type, and ET_B is the average annual water use $m^3\ ha^{-1}$ of 450- year old forest stand type

MATERIALS AND METHODS

Determining the impact of logging practices on stream flows and assessing the economic value of flows may be a basis for identifying influences of forest management choices on ecosystem service values. To test this approach, I developed a bioeconomic model capable of calculating the economic benefits or losses from gains or depletions of stream flows after selective harvest.

First, data from the Caspar Creek Experimental Watershed in the western Coast Range of California (Keppeler and Ziemer 1990, Ziemer 2000, Rice et al. 2004) and the H. J. Andrews Experimental Forest in the western Cascades of Oregon (Moore et al. 2004) were utilized to model the effects of sustainable forestry practices and selective timber harvesting on stream flow volumes and transpiration, in the Mattole Watershed.

From Keppeler and Ziemer (1990) and Rice et al. (2004), if

$$X_{1A} = X_{2A} - ET_A$$

Where, X_{1A} is the average annual water yield $m^3\ ha^{-1}$ of 90-yr old forest stand type A, pre-harvest, X_{2A} is the average annual water yield $m^3\ ha^{-1}$ of 90-yr old forest stand type A, post-harvest, and ET_A is water use $m^3\ ha^{-1}$ of 90-yr old forest stand type A.

Keppeler and Ziemer (1990) and Rice et al. (2004) concluded that the increase in water yield post-harvest was due to the loss of evapotranspiration processes, which then result in additional water inputs to the river. Moreover, increased annual water yields, as a result of evapotranspiration losses, were approximately 900 $m^3\ ha^{-1}$ (Ziemer 2000). Hence,

$$ET_A = 900\ m^3\ ha^{-1}$$

In this thesis, I assume forests in the Mattole are comprised of this type A stand dominated by second-growth Douglas-fir (*Pseudotsuga menziesi*) and coast redwood (*Sequoia sempervirens*). Therefore;

$$ET_{Mattole} = 900 \text{ m}^3 \text{ ha}^{-1}$$

Where $ET_{Mattole}$ is the average annual water use m^3 ha^{-1} of 90-year old forest stand type A Similarly then, from Moore et al. (2004)

$$X_{1B} = X_{2B} - ET_B$$

Where X_{1B} is the H. J. Andrews' average annual water yield m^3 ha^{-1} of 450-yr old forest stand type B, pre-harvest, X_{2B} is average annual water yield m^3 ha^{-1} of 450-yr old forest stand type B, post-harvest, and ET_B is water use m^3 ha^{-1}of 450-yr old forest stand type B.

I used the Caspar Creek study and the H. J. Andrews' research as a basis for estimating the differences between water yields for two forest types, in the Mattole Watershed. Therefore,

$$\Delta X_{AB} = X_{1B} – X_{1A}$$

Where ΔX_{AB} is the difference between average annual water yield m^3 ha^{-1}of of 90 and 450-yr old forests, X_{1A} is the average annual water yield m^3 ha^{-1}of of 90-yr old forest stand type A, pre-harvest, X_{1B} is average annual water yield m^3 ha^{-1}of of 450-yr old forest stand type B, pre-harvest, Therefore,

$$\Delta X_{AB} = (X_{2B} - ET_B) – (X_{2A} - ET_A)$$

Where X_{2B} is the average annual water yield m^3 ha^{-1} of 450-yr old forest stand type B, post-harvest, X_{2A} is the average annual water yield m^3 ha^{-1} of 90-yr old forest stand type A, post-harvest, ET_A is the water use m^3 ha^{-1}of of 90-yr old forest stand type A, and ET_B is the evapotranspiration m^3 ha^{-1} of 450-yr old forest stand type B.

For the purposes of this thesis, I assumed a scenario in which both the old-growth and the 90-year old forest stands underwent identical harvesting practices which resulted in similar residual forest cover, therefore average annual water yields of both stand types, post-harvest, would be identical. Hence,

$$X_{2B} = X_{2A}$$

Consequently, the difference between average annual water yield in m^3 ha^{-1} of 90-year old and 450-year old forests, is the difference between each stand types' respective water use rate.

$$\Delta X_{AB} = (-ET_B) - (-ET_A)$$

Because average annual water use of forest type A was defined as 900 m^3 ha^{-1}, then the difference in average annual water yield between both stands is

$$\Delta X_{AB} = (-ET_B) - (-900 \text{ m}^3 \text{ ha}^{-1})$$

Next, I used the H.J. Andrews Forest study to evaluate the role of transpiration by old-growth forests and young stands on water yield. From Moore et al. (2004), transpiration from a 450-year old stand is 3.27 times less than the transpiration from a 40-year old stand and therefore,

$$ET_B = ET_A \div 3.27$$

Where ET_A is the water use in m^3 ha^{-1} of 40-year old stand and ET_B is the water use in m^3 ha^{-1} of 450-year old stand.

I assumed transpiration rates of 40-year old stands measured in the Cascades are comparable to transpiration rates of forest stands in the Mattole Watershed. Results from Ziemer (2000) indicated that within 15 years of selective harvesting the South Fork of Caspar Creek, average annual water yields returned to pre-harvest levels. In fact, a somewhat surprising result from the Caspar Creek study indicated that flow peaks and volumes 10 years after logging were similar to those in 100-year-old redwood forest (Lewis and Keppeler 2007). This suggests 15-year to 90-year old forest stands of this type exhibit relatively the same water yields $m^3\ ha^{-1}$ and therefore should exhibit comparable water use rates.

I substituted water yield increases for forest stands estimated for Caspar Creek stands into the above equation to estimate water use of 450-year old stands. Therefore,

$$ET_B = 900\ m^3\ ha^{-1} \div 3.27$$

If,

$$ET_B = 275\ m^3\ ha^{-1}$$

Then,

$$\Delta X_{AB} = -\ (275\ m^3\ ha^{-1}) + (900\ m^3\ ha^{-1})$$

Finally,

$$\Delta X_{AB} = 625\ m^3\ ha^{-1}$$

Where, ΔX_{AB} is the difference between average annual water yield in $m^3\ ha^{-1}$ of 90-yr old and 450-yr old forests. This increase in water supply is considered a measure of ecosystem service benefit provided from a hectare of old growth forest.

The next step in building a bioeconomic model was to assign monetary values to this particular ecosystem service. In this case the benefit transfer method was used to assign non-use values to water supply benefits. I incorporated Troy and Wilson's (2006) framework for spatially explicit value transfers and followed a seven-step process (Table 1).

The initial step in a value transfer is to define the policy context through a review of the current literature. In this thesis the policy site is defined as the Mattole Watershed, including its ecological and demographic characteristics and its history of land and resource use described in the "Study Sites" section.

Table 1. A decision framework for mapping ecosystem service values (Troy and Wilson 2006).

Step	Description
1	Define Policy Site
2	Forest Cover Typology Development
3	Economic Literature Search and Analysis
4	Mapping
5	Ecosystem Service Value Calculation
6	Geographic Summary
7	Scenario Analysis

Further described in the Study Sites section are the ecological and land uses in Caspar Creek experimental watershed and the H. J. Andrews State Forest. The study results from these two experimental watersheds are used to predict how changes in forest cover affects water supply in the Mattole Watershed. All three study sites are considered Pacific Northwest watersheds and naturally exhibit similar characteristics such as climate, hydrology, geology, and forest cover. Extensive data collections have been completed for both Caspar Creek and the H. J. Andrews Forest. The characteristics of these watersheds provide supporting data which are used to link effects of timber harvest on average annual water yields in the Mattole.

Step two was the development of land use and land cover typologies to further define the spatial landscape and biological conditions of the Mattole watershed (Troy and Wilson 2006). This began with a preliminary survey of available geographic information systems (GIS) data to determine the land cover types present, the watershed size, geography, water bodies, land-use, and ownership. Spatial data from the U.S. Forest Service, commonly referred to as CALVEG, were used to create a land cover typology (USDA Forest Service 2011). Updated in 2007, CALVEG maps, at a scale of 1:24,000, are comprehensive spatial databases that meet regional and national vegetation mapping standards. The extent of the watershed boundary was available from the California Spatial Information Library (CaSIL 2009). Parcels, land use, and ownership, were drawn from Humboldt County GIS parcel data (Humboldt County Community Development Services 2009).

Step three consisted of a search and analysis of the valuation literature that involved identifying original studies that estimated forest and watershed benefits. I explored journal articles, research reports, dissertations, published texts, and databases. I screened the original research and context corresponding to the Mattole in terms of ecological and socio-economic conditions. Initial reviews for forest and water ecosystem service values yielded more than 35 potential studies Cost and time constraints and the lack of demand coefficients available for the policy site prohibited the use of a function transfer in this thesis. The demographic and ecological similarity between the chosen study site and the policy site did validate the use of the value transfer method.

One study was selected as the best fit for transfer to the Mattole. This determination was based on the study site's similarity of ecological and geographical characteristics and for its utility for estimating watershed benefits including the provision of water supply. Values were adjusted to 2010 U.S. dollar equivalents using the consumer price index (Officer and Williamson 2011). The consumer price index is the relative cost of a bundle of goods and services in one year compared to the cost of that bundle in a base period.

The study site selected in the benefit transfer to the Mattole was drawn from an economic value study of Trinity River water (Douglas and Taylor 1999). Since damming the Trinity River in 1964, the loss of fish habit and a substantial portion of the Trinity River's flow resulted in a 90% decline in anadromous fish stocks (National Marine Fisheries Service 1994). These declines in streamflow and the viability of anadromous fish populations are related to existence benefits, which are tied to many coastal rivers with anadromous fish populations (Loomis et al. 1990, Lichatowich et al. 1995).

Researchers separated out existence benefits from recreation benefits from Trinity River resource users. The authors collected data utilizing the contingent valuation method to estimate the non-market benefits of augmenting Trinity River instream flows by letting more water flow down the Trinity River and transferring less water to the Sacramento River. A telephone and mail-out survey included Trinity River recreationists and a random sample of California, Oregon, Nevada, and Washington residents. The description of the water resource issue in the survey stressed the costly tradeoff between non-market benefits and development uses of Trinity River water. Five distinct flow and fish run related scenarios were included in the block of contingent valuation items.

Annual willingness-to-pay values, realized in monthly utility bills as payment vehicles, referred to five distinct flow levels in terms of the percentage diverted to the Sacramento River, the number of adult spawning anadromous fish, and the quality of recreational boating on the Trinity River. Authors compared the non-market benefits with the costs of foregone market uses of the diverted water, mainly hydropower and irrigation benefits. Benefit estimates were aggregated across California, Nevada, Oregon, and Washington populations. Annual benefits were estimated to be worth between \$161 million and \$1 billion, a value that greatly exceeded the cost estimate. Annual benefit values were divided by annual flow volumes (m^3) to establish annual benefit values (\$ m^{-3}).

The recommended flow regimes link two essential purposes deemed necessary to restore and maintain the Trinity River's fishery resources: 1) flows to provide physical fish habitat (i.e. appropriate depths and velocities, and suitable temperature regimes for anadromous salmonids) and 2) flows to restore the riverine processes that create and maintain the structural integrity and spatial complexity of the fish habitats (U. S. Department of the Interior 2000). Table 2 outlines the annual preservation benefits for Trinity River instream flows and increased fish runs. I selected the annual benefit estimate of \$0.64 m^3 calculated for an increase in 35,000 fish, Alternative 2, to be transferred to the Mattole River Watershed.

Table 2. Annual household benefits for Trinity River flows (Douglas and Taylor 1999).

Scenarios:	Flows (m^3 yr^{-1})	Mean values
	Size of annual fish run	$\frac{\text{Mean value}}{\text{Flow value}}$
Alternative 1	148,017,822	$157,000,000
	9,000 fish	$1.06 m^{-3}
Alternative 2	296,035,645	$189,000,000
	35,000 fish	$0.64 m^{-3}
Alternative 3	444,053,468	$367,000,000
	75,000 fish	$0.83 m^{-3}
Alternative 4	740,089,113	$757,000,000
	85,000 fish	$1.02 m^{-3}
Alternative 5	1,036,124,758	$1,180,000,000
	105,000 fish	$1.14 m^{-3}

Step four of the benefit transfer was to derive a final land cover map using ArcView software overlay and geo-processing tools. I used clip commands to extract vegetation layers from CALVEG, and land use and ownership parcels from Humboldt County GIS to the extent of the Mattole watershed boundary provided by CaSIL data.

I utilized the intersect command to merge layers and joined this dataset back to the original watershed layer. The final result was a layer with associated attribute tables consisting of area and cover types, land use and ownerships in the Mattole watershed.

Step five in the spatial analysis-benefit transfer application was to calculate the water supply values for the watershed. After assigning each mapping unit a forest cover type, ownership, land use, and corresponding benefit estimate, the values were summed and cross-tabulated by ecosystem benefit and land cover type. Adding up the individual values associated with that forest type and multiplying by the representative area in the Mattole produced the total ecosystem service value.

$$V(ES_i) = \sum A(LU_i) \times V(ES_{ki})$$

Where $A(LU_i)$ is the area of land use/cover type (i), $V(ES_{ki})$ is the annual value per ha for ecosystem service type (k) generated by land use/cover type (i).

In the case of the Mattole, ecosystem service type (k) (i) is the increase in average annual water yield in $m^3\,ha^{-1}$ supplied from the retention of old-growth mix conifer hardwood stands compared to average annual water yield in $m^3\,ha^{-1}$ after selective harvest practices, defined earlier as ΔX_{AB}. Therefore,

$$V(ES_i) = \sum A(LU_i) \times V(\Delta X_{AB})$$

Where $A(LU_i)$ is the area of land use/cover type (i), $V(\Delta X_{AB})$ is the annual economic value in ha^{-1} for ecosystem service type (k) generated by land use/cover type (i).

Once the economic value of water supply benefits per m^3 was determined from Table 2, then the annual value per ha of average annual water yield supplied by the retention of old growth mixed conifer hardwood stands can be calculated.

$$V(\Delta X_{AB}) = V(ESV) \times (\Delta X_{AB})$$

Where $V(\Delta X_{AB})$ is the annual economic value per ha of an increase in average annual water yield supplied by the retention of old growth mix conifer hardwood stands, $V(ESV)$ is the annual economic value m^{-3} of water supply benefits.

Finally, the total benefits of an increase in average annual water yield supplied by the remaining 4,777 ha of old growth forest in the Mattole can be calculated.

$$V(ES_i) = \sum A(LU_i) \times ((\$0.64\ m^{-3}) \times (625\ m^3\ ha^{-1}))$$

Where $V(ES_i)$ is the total value of water yield benefits supplied by old growth forest in the Mattole, and where $A(LU_i)$ is the area of old growth forest in the Mattole.

Finally, a scenario analysis was conducted by changing the inputs in steps 4 and 5 to predict how a proposed land-use change would affect the Mattole ecosystems and their flow of benefits.

The scenario analysis involved manipulating cover type area in the GIS to reflect a proposal for the conservation of mixed conifer forests on vacant rural residential parcels less than 40 acres (16.19 ha). A recalculation of ecosystem service values resulted in a new benefit estimate for the watershed and a map illustrating the probable economic and environmental impacts of the proposed land-use change.

RESULTS

The Mattole Watershed in northern California was defined as the policy context. The key concern in the Mattole was the decline of threatened anadromous fisheries. As of 2011, total basin populations of 200 or fewer salmon have been reported. However, potential coho and Chinook salmon populations were estimated to be more than 35,000 individuals (USFWS 1960). The decline in fisheries has been linked to excess sediment load, elevated water temperatures, and summertime low flows associated with historic timber practices, grazing and agriculture (US EPA 2003).

Figure 8 illustrates land cover typologies in the Mattole Watershed. GIS data indicated that 44,035 ha of the watershed are covered by mixed conifer and hardwood forests. Pure hardwood stands occupied 12,724 ha, coniferous forests without hardwoods occupy another 6,467 ha, and annual grasslands occupied 11,415 ha of the watershed. Old growth forest occupied 4,777 ha.

Figure 9 illustrates land use by parcel in the Mattole. Over a third of the Mattole Watershed is private land managed by ranchers for both timber and grazing. Another third is zoned rural residential. The federal Bureau of Land Management (BLM) manages fifteen percent. Thirteen percent by industrial timber companies and the remaining ten percent is split between the Sinkyone Wilderness State Park and other private lands.

The initial valuation search for forest and water ecosystem service values produced more than 35 credible studies. Table 3 lists some of the characteristics of these studies. The Trinity River study was selected as the study site to be used in the benefit transfer to the Mattole (Douglas and Taylor 1999). Figure 10 shows the Trinity River site in relation to the Mattole.

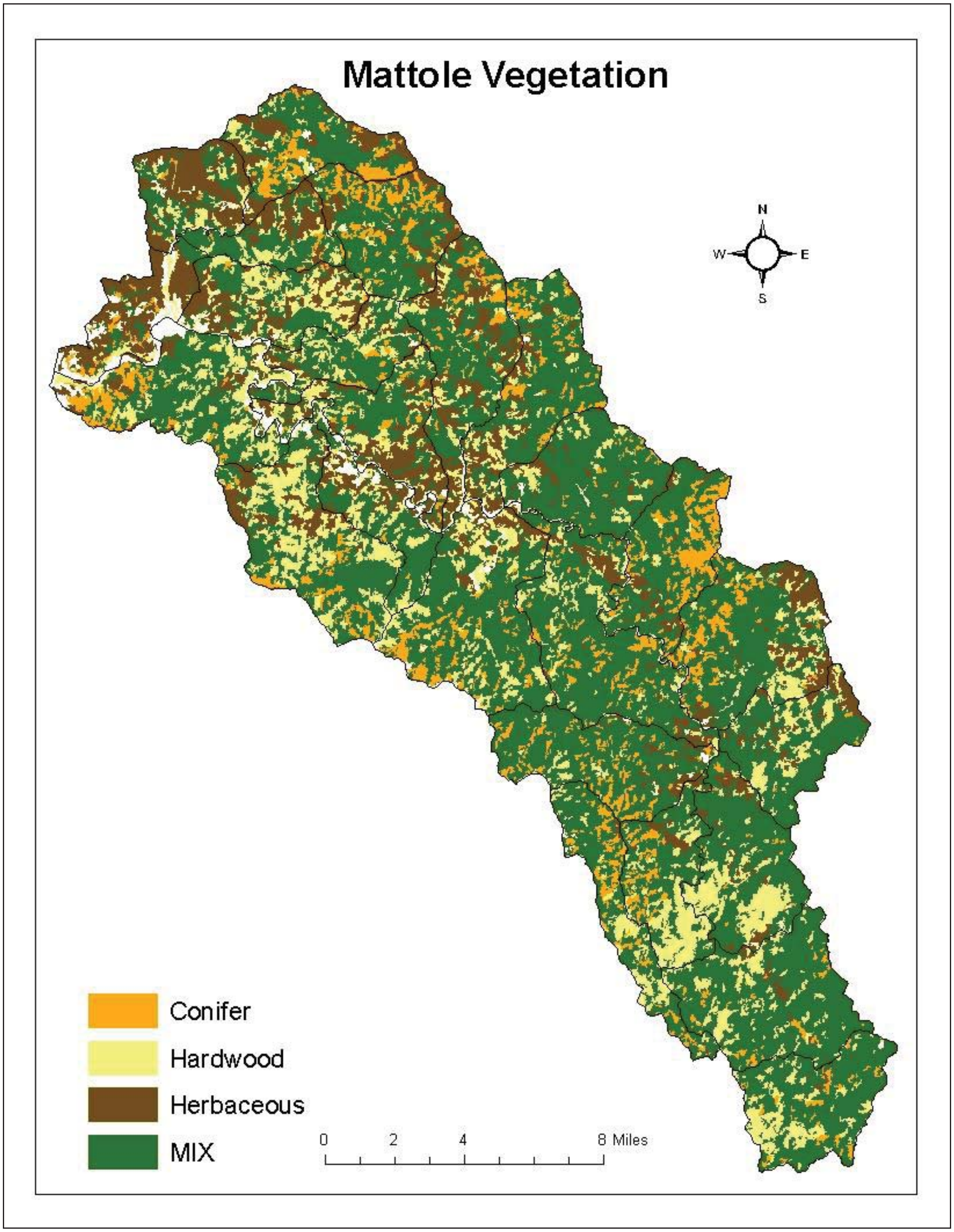

Figure 8. Forest cover typologies, in the Mattole Watershed (Source: Humboldt Community Development Services 2012).

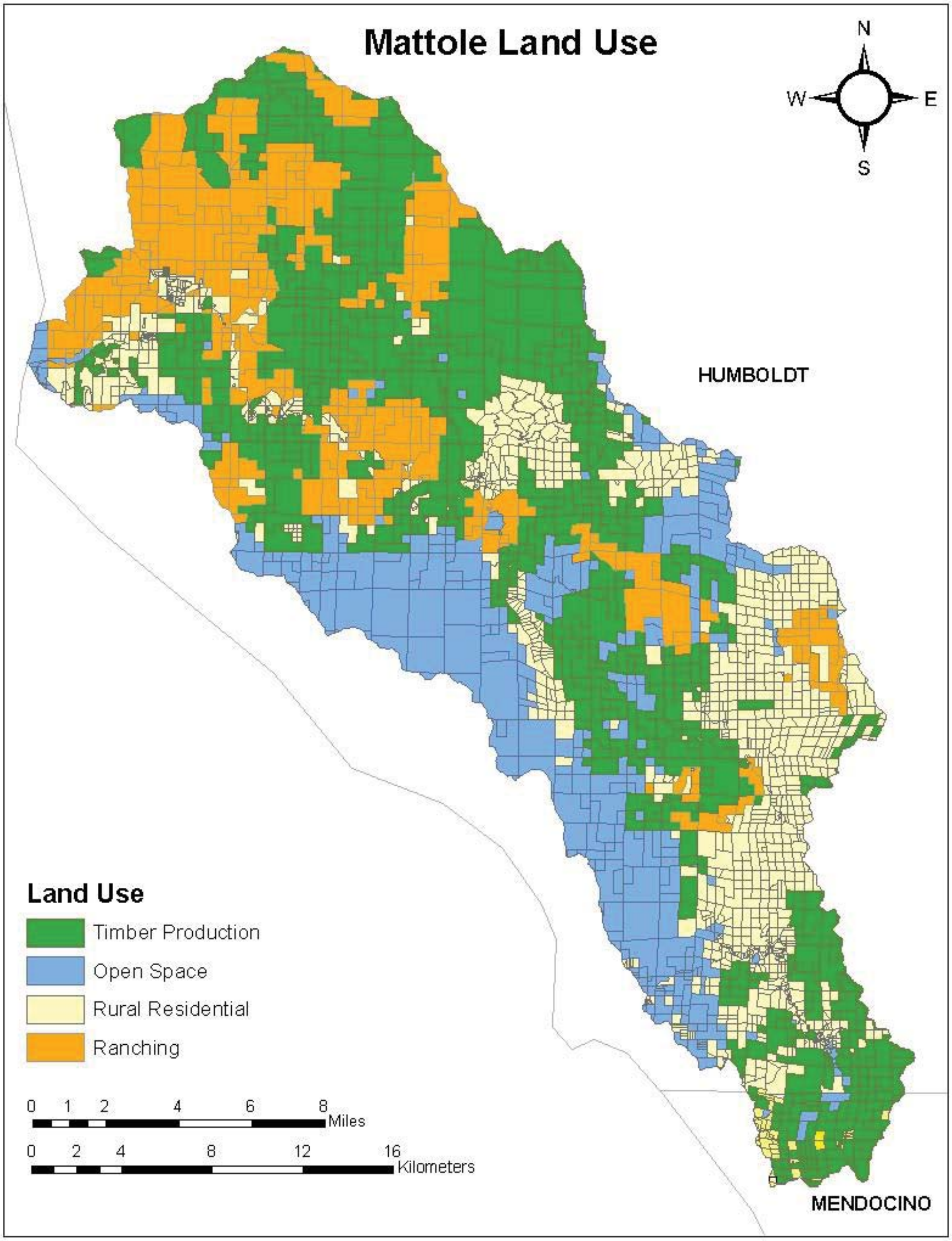

Figure 9. Land Use in the Mattole Watershed (Source: Humboldt County GIS 2012).

Table 3. Valuation literature search and analysis.

Study Site	Ecosystem Service	Average Estimate 2008 US Dollar	Scope
San Joaquin Valley, CA	Recreation	248 trip 0.46 m^{-3} yr^{-1}	Benefits from water increases to refuges and rivers (CA)[1]
The Trinity River, CA	Water supply	19 household WTP	Existence benefits from improved flows (CA, NV, OR, WA)[2]
Pacific Northwest	Erosion control	340 ha^{-1} yr^{-1}	Private timberland owners' willingness to forego harvest of riparian buffer (OR, WA)[3]
Siuslaw National Forest, OR	Fish habitat	32 fish	Cost to fisheries from timber harvest (Alsea River, OR)[4]
Columbia River Basin	Fish biodiversity	42 trip 35 fish 47 household	Existence value of doubling the run (ID, MT, OR, WA)[5]
Pacific Coast Region	Recreation	51 trip	Pacific Northwest[6]

[1] *Creel and Loomis (1992).*
[2] *Douglas and Taylor (1999).*
[3] *Kline et al. (2000).*
[4] *Loomis (1988).*
[5] *Olsen et al. (1991).*
[6] *Rosenberger and Loomis (2000).*

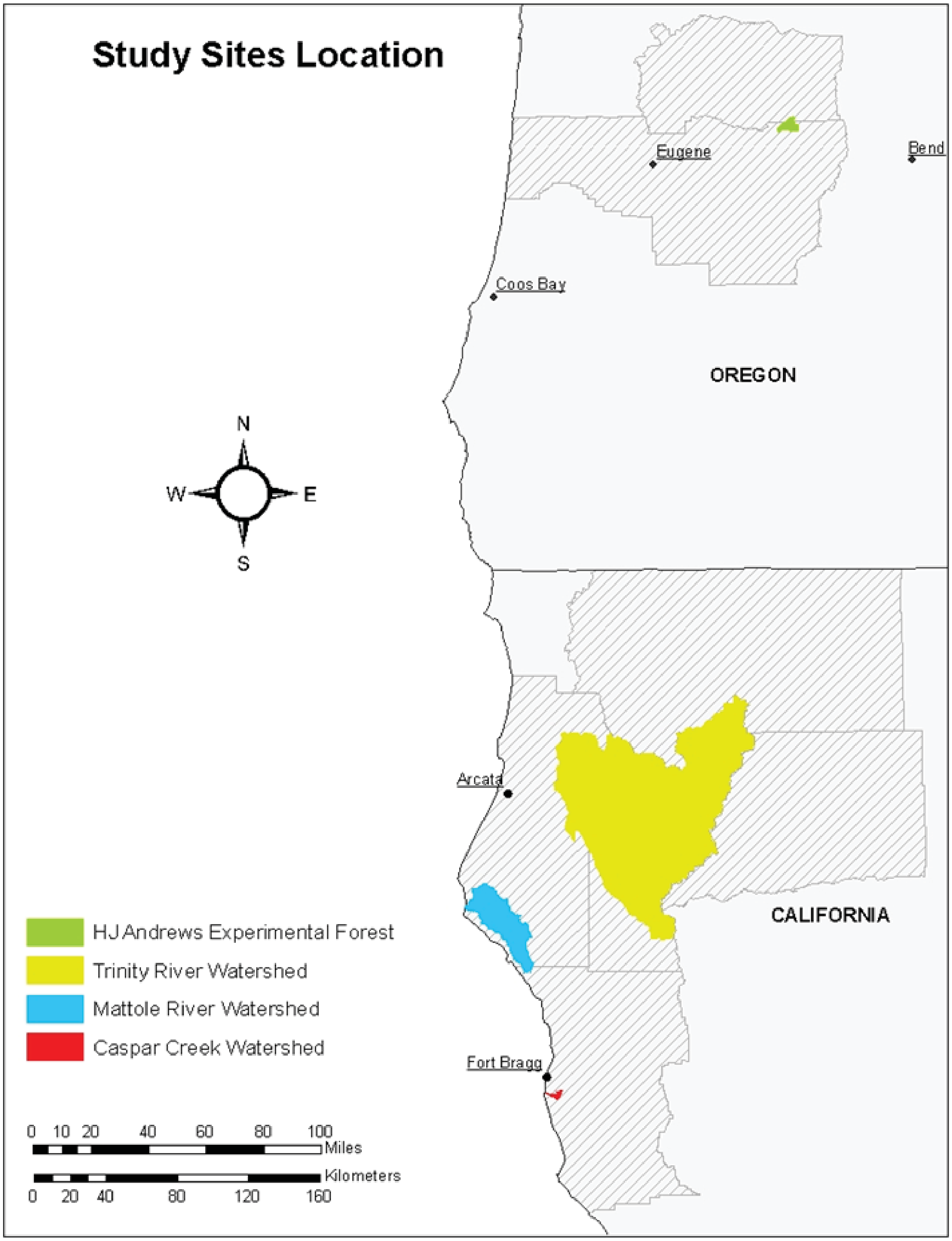

Figure 10. The Trinity River study site.

Annual benefits were estimated to be worth between \$161 million for the lowest instream flows and \$1 billion for the largest increase in flows. Again, the increase of instream flows increase salmon populations by restoring fish habitat and riverine processes that create and maintain the structural integrity and spatial complexity of the fish habitats (U. S. Department of the Interior 2000). Annual benefit values were divided by annual flow volumes (m^3) to establish annual benefit values (\$ per m^3). The annual household benefit for Trinity River stream flows that resulted in an increase of 35,000 fish was selected as the point estimate to be transferred to the Mattole. Therefore, the annual economic value of Mattole River instream flows was estimated to be worth \$0.64 m^{-3}.

Since the retention of mixed conifer hardwood forests resulted in an increase in average annual water yield of 625 m^3 ha^{-1}, the annual value of mixed conifer hardwood forests was calculated.

$$V(\Delta X_{AB}) = (\$0.64\ m^3) \times (625\ m^3\ ha^{-1})$$

Where $V(\Delta X_{AB})$ is the economic value ha^{-1} yr^{-1} of an increase in average annual water yield supplied by the retention of old growth mix conifer hardwood stands. Therefore, the annual ecosystem service value of old growth forests' capability of increasing average annual water yield equaled \$400 ha^{-1}.

The total benefits transfer of an increase in average annual water yield supplied by the remaining 4,777 ha of old growth forest in the Mattole was calculated.

$$V(ES_i) = \sum (4{,}777\ \text{ha}) * (\$400)$$

Where $V(ES_i)$ is the total ecosystem service value of annual average water yield from old growth forest in the Mattole Watershed. Therefore, the annual ecosystem service value of water yield benefits supplied by old growth forests in the Mattole Watershed was equal to \$1,910,800.

Figure 11 illustrates the scenario analysis, which focused on estimating potential benefits from the conservation and sustainable management of mixed conifer hardwood forests on rural residential parcels less than 40 acres (16.19 ha). A total of 6,540 ha were selected from the Mattole to fit the criteria listed above. I presumed these scenario parcels were to be managed for old-growth characteristics, such as uneven age structure. Conserving these parcels protects the integrity of ecosystem services and benefits contrary to clearcutting, developing, or converting these parcels to non-forest uses. The scenario analysis predicted that these parcels would provide an annual ecosystem service value of $2.6 million.

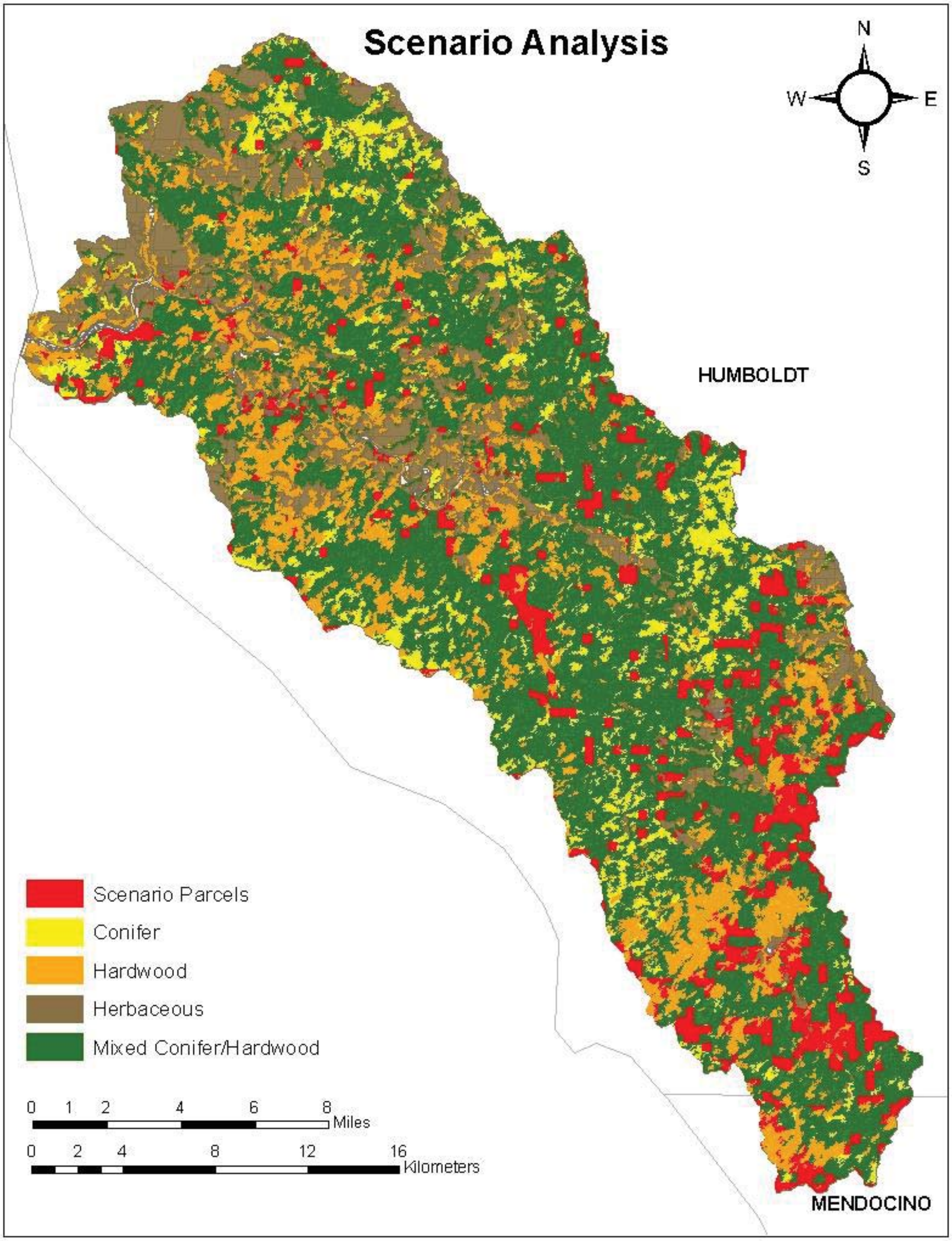

Figure 11. Scenario analysis of rural vacant parcels < 40 acres of mix forest.
(SOURCES: CalVeg and Humboldt County GIS dataset).

DISCUSSION

The goal of this thesis was to estimate ecosystem service benefits of water supply provided by forestland in the Mattole Watershed and how land use change affects ecosystem service values. Applying the seven-step framework for a point estimate benefit transfer-spatial analysis application, has illustrated how ecosystem services in the Mattole can be valued based on limited economic valuation studies and available spatial data. Although function transfers might have been more accurate then point estimate transfers, cost and time constraints and the lack of demand coefficients available for the policy site prohibited the use of a function transfer.

The Trinity River study did estimate existence values that were generated from potential increases in salmon stocks, as a result from increasing in-stream flows. The similarity between the Trinity River and the Mattole made the use of a value transfer appropriate and suitable for benefit transfer. These comparisons include physical and biological characteristics, type of resource or commodity, minimal differences in population size and attitudes (Rosenberger and Loomis 2003).

First, the Trinity River and the Mattole Watershed are considered Pacific Northwest coastal watersheds, at times both flowing through Humboldt County. The geographic distance between the Trinity River and the Mattole is small. As coastal watersheds, the Trinity River and Mattole ecosystems are subject to similar wet-dry climatic seasons. Most of the annual rainfall comes to both of these sites in the winter season, from October through March. Summers are dry. Topography and geology of the region is described as having steep and unstable slopes. Vegetation is dominated by mixed conifer and hardwood forest types. All of these factors affect the hydrogeological conditions, including winter high flows and summer low flows. Similar changes in hydrogeological conditions directly affect fish habitat conditions in both watersheds, such as spawning and rearing grounds.

Second, the size and type of commodity being valued is the essentially the same. The increase of instream flows for purposes of restoring fish habitat and hydrological conditions which promote threatened and endangered anadromous fish populations is the commodity being considered. The Trinity and the Mattole rivers are both federally listed as impaired and are subject to Total Maximum Daily Loading (TMDL) monitoring to assure that salmon habitat in streams is protected from excess sediment and temperature increases (USEPA 2001, USEPA 2003). The coho and Chinook salmon population potential in the Mattole has been estimated to be near 35,000 fish while potential fish populations in the Trinity River were estimated to increase by 9,000 – 105,000 fish.

Third, given the rural character of these mountain watersheds, similar low population densities and mixes of land use, I assumed that the residents, including the Hoopa tribe on the Trinity and the general tourist population, including recreational fishers visiting either the Trinity or Mattole area, most likely share similar views about the environment and conservation of endangered fish species.

Annual Trinity River instream flows were estimated to be worth between $157 million and $1.2 billion, or between \$0.64 m^{-3} and \$1.14 m^{-3}, depending on potential salmon population increases. The variation in fish run sizes with flow increases in the Trinity River, was one of the most critical types of data utilized and was based on best available scientific evidence. While the Trinity River is very similar to the Mattole biophysically and socio-economically, the context of the Trinity is centered on dams and the diversion of instream flows. This is a core debate as far as current environmental issues go in the region and may result in upward bias in willingness-to-pay estimates for increases in stream flow, in the Trinity. However, the goal at both the Trinity and Mattole sites is the survival of endangered salmon and an upward bias in benefit estimates of the Trinity over the Mattole is likely minimal. There is incredible local community effort in the Mattole to engage with the issue and restore salmon in the river. In fact, Trinity River benefit estimates for increases of 35,000 fish may likely underestimate benefits in the Mattole. An increase of 35,000 fish in the Trinity is a relatively small increase in Trinity River fish populations. However, a potential increase of only 35,000 fish in the Mattole is essentially a 100% increase in Mattole River fish populations. In other words, how much more are people willing to pay for a 100% increase in salmon populations in the Mattole compared to how much people are willing to pay for only a 50% increase in salmon populations in the Trinity?

Benefit estimates from the Trinity River study site are derived from a contingent valuation survey. This type of valuation method is inherently susceptible to biases based on an individual's interpretation of the questions in the survey. It is possible the survey takers' perception of instream flow benefits are the combination of ecosystem services that favor salmonids, such as pools, cool temperatures, and channel morphology. If so, then a benefit transfer to the Mattole represents a more holistic value of the total watershed ecosystem service values, and consequently may be reflected in a higher valuation estimate. I calculated annual Trinity River instream flows to be worth between \$0.64 m^{-3} and \$1.14 m^{-3}, depending on potential salmon population increases. I selected the most conservative value of \$0.64 m^{-3} to be transferred to the Mattole, based on the potential for an increase in 35,000 fish in both rivers.

The next step towards translating forest management impacts into economic gains or losses with regard to water supply benefits required building a stream flow model to predict how forest management practices affect instream flows. The stream flow model was developed using the Casper Creek paired watershed study data on water yield (Keppeler and Ziemer 1990, Ziemer 2000, Rice et al. 2004) in combination with the H. J. Andrews study data on transpiration (Moore et al. 2004). Casper Creek and H. J. Andrews forest cover is similar to forest cover in the Mattole and it is reasonable to assume that under the right conditions, a hectare of Mattole forest regulates water in much the same way.

From these studies, I assumed an indirect relationship between water use and stream flows. Because trees that were harvested no longer transpire and therefore do not use water, the surplus water enters the system and results in increase of average annual water yield. However, the increase in stream flow realized after a harvest was only for a relatively short period of time of about 15 years. In fact, young regenerating stands transpire and use about 67% more water than old-growth stands, thereby reducing water yields by this amount. I assumed then average annual water yield from a watershed comprised of young stands will be approximately 67% less than average annual water yield from a watershed comprised of old growth forest.

This may be a rough approach to estimating water supply increases and benefits provided by mixed conifer and hardwood stands in the Mattole. Although all three watersheds are influenced by coastal climate and have similar vegetation cover, aspects, topography, and soil type, at the onset of the Caspar Creek and the H. J. Andrews study the watersheds supported 90-year-old second growth forest and 450-year-old old growth forest respectively. Research shows 40-year-old conifer stands transpire 3.27 times more than old-growth stands, mostly due from noticeable differences in age and overall sapwood area of higher than 21% in the young stand (Moore et al. 2004). Because water yields in Caspar Creek returned to pre-harvest levels after 15 years, I assume that 15 year-old stands transpire the same as 90-year-old stands. Therefore, the difference between how much 90-year-old stands and 450-year-old stand transpire will either over- or under-estimate water yield and supply benefits. In fact, a somewhat surprising result from the Caspar Creek study indicated that flow peaks and volumes 10 years after logging were similar to those in 100-year-old redwood forest (Lewis and Keppeler 2007).

Caspar Creek and the Mattole differ in overall size. The study at Caspar Creek is comprised of relatively small isolated units. Larger basins like the Mattole are more ecologically complex and therefore more variability and a greater degree of uncertainty about streamflow predictions most likely exist. However, Lewis et al. (2001) reported that annual water yields were independent of watershed area. Therefore, these units can be seen as somewhat representative of larger scale watersheds.

Long-term paired watershed studies on forested watersheds in California and Oregon are specifically designed to monitor and study the effects of various land management practices on the quantity, timing, and quality of stream flow from mountain watersheds (Keppeler and Ziemer 1990, Jones and Grant 1996, Rice et al. 2004). The Casper Creek data represent the only long-term hydrological information from managed second-growth conifer forests in the western United States (Lisle 2003). The Caspar Creek paired watershed study is considered a case example for logging's effects on any northern California watershed having similar climate, soil, logging history, and vegetation (Rice 1979).

A recent study by Stubblefield et al. (2011) predicted an overall drop in average water use of Mattole forests in coming decades. A decline between 104 m^3 ha^{-1} and 682 m^3 ha^{-1} was estimated. This decrease in water use resulted directly from the greatly reduced numbers of young trees projected to comprise Mattole forests in the year 2055. Although mid and larger sized trees increased their water use over the time period, they were a much smaller proportion of total water use, so their impact was minimal.

Once the value for instream flows was calculated to be worth \$0.64 m^{-3} yr^{-1} and the supply of instream flows provided by forest stands to be 625 m^3 ha^{-1}, then finally the annual value per hectare for water supply benefits generated by forest stands can be assigned. Therefore, the annual ecosystem service value of water supply benefits provided by mature forest stands in the Mattole were estimated to be worth \$400 ha^{-1}.

This estimate is nearly the same as Costanza et al. (1997b) estimate of \$444 ha^{-1} for average annual global forest water quantity benefits. The Mattole benefit is more than twice as much as estimates for Chilean forests contribution to freshwater supply. Nunez et al. (2006) estimated water supply benefits from native temperate forests to be worth \$181 ha^{-1} for summer time flow and \$81 ha^{-1} for water supply the rest of the year. However, Myers (1997) estimated India's forest ecosystem service values for regulating and containing water to be worth \$1440 ha^{-1} per year. This is three and half times the water supply benefits transferred to the Mattole.

These comparisons are obviously from very different study sites and would not be appropriate for a benefit transfer to the Mattole. Other streamflow benefit estimates for the Pacific Northwest region range from \$40 ha^{-1} to \$60 ha^{-1} and include additional benefits such as hydroelectric power (Smith and Vaughn 1995). The differences between global and regional estimates are large, but the Mattole estimate does fall within this range. This is a clear example of the level of understanding of the complexity of ecosystem functions and methodologies used to estimate their worth and judgment required of the researcher applying benefit transfer.

One of the most current and relevant benefit transfer applications utilized Troy and Wilson's (2006) framework. Ganz et al. (2007) took into consideration both market and non-market values provided by California's natural landscapes that might be threatened by catastrophic fire. The authors' goal was not to "create" values for ecosystems, but rather to generate a conservative baseline estimate of the values people already hold with respect to these ecosystems through an assessment of the best available literature. In this study, authors' assessed the full suite of ecosystem valuation techniques.

The research team obtained more than 200 point estimates from a set of 84 studies. These results were then standardized to units of per acre per year to provide a basis for comparison. Using GIS analysis, a total of 16 different land cover typologies were assigned to Humboldt County. Most noteworthy were non-market values assigned to mixed conifer and hardwood forests worth \$953 ha^{-1} per year and to old growth redwood stands worth \$1096 ha^{-1} yr^{-1}. These estimates are more than twice the value of \$400 ha^{-1} yr^{-1} per year I assigned to the Mattole, using the bioeconomic model. The estimates from Ganz et al. (2007) were an aggregate of several ecosystem service values including carbon sequestration, habitat refuges, and aesthetic values. The higher value estimated from Ganz et al. (2007) compared to the Mattole estimate could most likely be the result from an aggregation of many ecosystem service benefits compared to only valuing one service in the Mattole, that being water supply with respect to fisheries habitat. The point is that older forests' water provision services should be considered a significant benefit in estimating ecosystem service values. Future studies of this nature valuing other ecosystem service benefits would likely show results from this study and Ganz et al. (2007) to have underestimated ecosystem service values.

The final step in the benefit transfer was a scenario analysis aimed at the conservation of mixed conifer and hardwood forest considered at risk to land-use change. Using this model to estimate ecosystem service values was a practical application useful in natural resource planning and cost benefit analysis regarding Mattole watershed restoration funds. Determining the impact of land management practices on streamflows and assessing the economic value of streamflows may be a basis for identifying influences of forest management choices on ecosystem service benefits. For example, the Humboldt County Planning Commission is projecting to have its review of the county's General Plan Update completed by the end of 2012. The General Plan designates land use on county lands by specific zoning and ordinances. Policies on the development of rural lands for residential use remain to be determined. The controversial Humboldt County General Plan Draft restricts subdivisions, on rural residential zoned land, to a 40-acre (16.19 ha) minimum parcel size to preserve the area's natural values (Humboldt County General Plan 2008). This model would allow decision makers to assess the costs and benefits of maintaining these parcels as primarily forested parcels and protecting the integrity of ecosystem services and benefits compared to harvesting or converting these parcels to non-forest uses.

Mixed conifer hardwood forests on vacant rural residential parcels less than 40 acres, or 16 ha, are defined as most at risk to land-use change, and comprise 6,540 ha within the watershed. Conservation scenario parcels are assumed to increase average annual water yield by 625 m^3 ha^{-1}. Selective harvesting or converting such parcels to other non-forest uses will alter the parcels ability to filter, store, and release water and maintain seasonal flows and channel dynamics. Changing the landscape hydrological conditions affects water supply related benefits. A scenario analysis for the conservation of 6,540 ha of forest resulted in more than $2.6 million in yearly water supply benefits.

In this thesis water supply benefits are assigned to non-use values applied to a narrow definition of mixed conifer hardwood forest. The relationship between the estimates reported in this thesis and Ganz et al. (2007) indicates that the model developed here for point estimate transfers of water supply benefits are reasonable and potentially make up almost half of the total economic value of these types of forest stands. The bioeconomic model presented here provides a good tool that can account for the nonmarket goods and services of water supply benefits provided by the forests in the region.

In the Mattole, the annual water provision benefits provided by mature mixed conifer hardwood stands were estimated to be worth \$400 ha^{-1}. Essentially, the ecosystem service value is limited solely to that specific typology and therefore applicable to only 57% of the total watershed. Further research that can value the other 43% of the land cover would allow for a more complete accounting of ecosystem service values, most likely resulting in a significant transfer of benefits.

CONCLUSIONS

This thesis developed a useful model for evaluating the relative differences in ecosystem service outcomes among various management options, and can be used in future analyses to enable decision makers to better interpret on the ground data and visualize how forest conservation affects the provision of water related benefits. This research helps answer several questions. What are forest ecosystems services regarding water flow in the Mattole? How much are they worth? How might changes in land-use practices affect these ecosystem service values and the benefits they provide?

While this thesis modeled a benefit transfer-spatial analysis of watershed ecosystem service values, it clearly illustrated how this application is limited by the available spatial data and paucity of economic valuation studies. The availability of economic valuation studies is one of the most significant constraints to benefit transfer. Inherent differences between methodologies make it difficult to compare and interpret benefits across a broad spectrum. There are no economic valuation studies for many important ecosystem services. The applicability of economic valuation studies that do estimate ecosystem services is further constrained by the fact that only those studies with a similar context to the Mattole could be used for the benefit transfer.

Determining whether a study site was similar to the Mattole was a challenge in itself. Due to the limited number of existing studies, tradeoffs had to be made among the biogeography, demographic, and level of scarcity of fish between sites. Considering these limitations, five study sites were initially found to be similar, each one estimating separate ecosystem service values, which were to be transferred.

Tying Caspar Creek water yield to benefits in the Mattole was an innovative attempt to link forest ecosystem services to measurable water quantity values. A significant portion of ecosystem benefit estimates for the Mattole was derived from the relationship between forests and water yield. Further watershed research that quantifies water use between even age and uneven age stands may provide an even more accurate assessment of water yield data and therefore a more accurate accounting of water quantity benefits.

My assumptions that stands in the Mattole transpire and regulate water the same amount as at H. J. Andrews Experimental Forest at Caspar Creek Experimental Watershed are open to debate. Further research focused on finding the correlation between age and sapwood area and water use would provide a more accurate estimate to water benefits. From this thesis I can only assume that water supply benefits are realized when stands are 450 year old. At what age does water use reflect old-growth criteria? A more accurate model would be one that can predict how much the current forest in the Mattole uses water compared to 450-year-old forests.

Double counting benefits are a recurring topic when accounting for ecosystem services. It is important to distinguish between use, direct use, and non-use values. For example, fishing is a direct use benefit, seasonal flows are an indirect use benefit, and existence values for salmon are non-use benefits and can be aggregated. However, it is difficult to define or separate exactly what ecosystem services are. Contingent valuation surveys generally result in higher prices for ecosystem services, due to the design of the survey and respondents' interpretation of the questions. Willingness-to-pay estimates for water quantity are based on perceptions that promoting salmonids survival includes improving flows, but as a result, cools temperatures, creates deep pools, and promotes large woody debris. All these benefits are reflected in a higher willingness-to-pay estimate and consequently double counting becomes more likely.

Finally, putting a price tag on nature does not go without controversy and is worthy of debate. One argues that nature is priceless and putting a price tag on her is a show of ignorance and arrogance of humankind and sets forth the motions of her ultimate use and ruin. In some ways I agree with this philosophy. However, I believe valuing nature’s services is a necessary and practical approach to reaching sustainability in today’s global economy. Not accounting for these ‘free’ benefits from nature is the exact reason for their overuse and degradation.

The timber resources of the Mattole were plenty, and in another decade or two it will be economically viable to harvest them again. However, historical harvesting practices without regard to soil, water, and impacts to fish have resulted in an enormous ecological cost. To only consider the monetary value of the timber resource is not an efficient or sustainable reality. When the timber harvest is relatively feasible, short-term profits from clearcutting may be significant. However, the resulting loss of ecosystem services over the longer term becomes visible only when the services are lost or destroyed and may prove too costly to restore. Finding the balance between resource extraction and maintaining water quality and aquatic habitat will be necessary in order to continue the rural lifestyles many of the Mattole residents seek to sustain. Finding tools to account for benefits of recreation, water quantity, water quality, fish habitat and biodiversity will result in a more balanced approach to ecosystem management and likely produce an overall gain to society. When original valuation methods are not feasible, the benefit transfer is considered a reasonable approach to measuring ecosystem benefits.

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