QUANTIFYING AND MAPPING CHANGES IN HYDROLOGIC ECOSYSTEM SERVICES FROM A LARGE MAGNITUDE WILDFIRE IN SHASTA AND TEHAMA COUNTIES, CALIFORNIA

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Daniel Larson

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TABLE OF CONTENTS

PAGE

LIST OF TABLES

LIST OF FIGURES

FIGURE PAGE

ABSTRACT

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Ecosystem services are considered goods and services that sustain and enhance human life. Humans have degraded ecosystems due to land use or land management practices favoring more profitable land use scenarios at the expense of other valuable ecosystem services. Tools and models have emerged to quantify and value ecosystem services. InVEST is an open-source, spatially explicit, ecosystem service appraisal tool which has been widely-used. A review determined that only 18% of 153 publications on ecosystem services validated their results against observed data, and only one third of the studies had any sound basis for their results. There are few studies which have conducted a sensitivity analyses on InVEST model outputs. This study assessed predicted changes in the InVEST Hydropower Water Yield due to an episodic

disturbance (e.g., fire), validated model results against observed data, and determined how variation in spatial resolution (30, 100, 120, 250, 500, 1000, 4000 m) of input data affected model results. Results show that the model did not detect the fire event on water yield. The location of the fire is a more important element for producing a significant change in water yield. After model results were calibrated against observed stream gauge data, the model under-predicted by less than 1%. Annual or 30 year normal precipitation inputs do not significantly affect model results. Prior to calibration, normal precipitation resulted in a larger percentage of error 64% compared to annual data 44%. Calibration is a crucial and understated step. Resolution of land cover and soil data does not always create significant variation in model results. InVEST may be driven more by parameter values (e.g., seasonality constant, root depth, and evapotranspiration coefficients) than the resolution of soil and land cover data. These findings should assist InVEST users to more accurately parameterize their models and think critically about the appropriate study area, data inputs, and accuracy in results allowing for a widespread assessment of ecosystem services so that they may sustainably managed.

CHAPTER I

INTRODUCTION

Ecosystem services are considered goods and services that sustain and enhance human life. The knowledge of these benefits has been known for some time, but were operationalized by Daily (1997) and Costanza et al. (1997) as documented by (Nemec and Raudsepp-Hearne 2013). In 2005, the United Nations sponsored the Millennium Ecosystem Assessment (MEA 2005). The report highlighted that there were considerable gains in the last 50 years in terms of human well-being and economic development in some regions. These gains have come at the cost of environmental degradation and increasing regionalized social inequities (Ausseil et al. 2013). The MEA outlined the consequences of ecosystem degradation on human welfare (Nemec and Raudsepp-Hearne 2012). This report furthered the notion of ecosystem services and grouped them into four categories: provisioning, regulating, cultural services, and supporting services. There are a multitude of ecosystem services within these categories. Provisioning services include food and fiber production; regulating services represent issues such as flood mitigation and global climate regulation. Cultural services provided by the environment include recreational opportunities or places of spiritual benefit. Supporting services are the benefits provided by functional environmental landscapes such as water quality or healthy soil formation (MEA 2005). The MEA found that humans have significantly altered the natural landscape more rapidly in the last 100 years

than any other period in human history (MEA 2005). The MEA did not, however, provide a means for operationalizing the ecosystems services concept for researchers and policy-makers (Seppelt et al. 2011). Consequently, there is considerable debate around the terminology of what constitutes an “ecosystem service” versus an “ecosystem process”. The notion of a ‘service cascade’ with biophysical structures and processes at the top then cascading into function, service, benefit, and value was introduced by Haines-Young and Potschin (2010) and later adopted by Salles (2011). This notion infers a linkage with biophysical process and human well-being. Despite the lack of consensus on ecosystem service terminology (Potschin and Haines-Young 2011) most studies refer to supporting services (e.g., water yield) and provisioning services (e.g., water available for human consumption) as ecosystem services (MEA 2005; Turner and Daily 2008; Vigerstol and Aukema 2011; Crossman et al. 2013).

Humans have degraded ecosystems due to land use or land management practices favoring more profitable land use scenarios at the expense of other valuable ecosystem services (Polasky et al. 2011). This degradation is a byproduct (i.e., environmental externality) of land management/land use trade-offs which favor one output such as agricultural production at the expense of decreased water quality (Ausseil et al. 2013; Polasky et al. 2011). Supporters of the ecosystem service paradigm argue that demonstrating the economic and ecologic value of ecosystem services will enable people to make more sustainable land use decisions (Nemec and Raudsepp-Hearne 2013).There are many critics, however, of the ecosystem services paradigm. Some argue that nature possesses intrinsic value and that conservation should not be pursued only because it is profitable (McCauley 2006). In addition, the market-based approach of ecosystem

services fails to fully capture how people connect with nature (Jax et al. 2013). Detractors of ecosystem services argue that it will be difficult to determine how climate change will affect ecosystem function and investment in ecosystem service policies, that economic reasons will prevail in conservation considerations, optimizing ecosystems valuable to humans will come at the expense of biodiversity protection, and some services can be valued (e.g., water, carbon sequestration) where valuation of other services (e.g., aesthetic, cultural) is more difficult (Redford and Adams 2009; Chee 2004). Despite these criticisms, the importance of incorporating ecosystem services into decision making has grown. There is a need for tools and models to quantify and value services, and to determine the effects management decisions have on the provisioning of various services (Vigerstol and Aukema 2011). Ecosystem service assessments also provide an opportunity to increase our understanding of ecosystem processes and their associated feedback loops (e.g., agricultural production over water quality, water supply at the expense of biodiversity). These tools must be rigorously tested and validated to prove the efficacy of their results. A review from Seppelt et al. (2011) determined that only 18% of 153 publications on ecosystem services reviewed validated their results against observed data, and only one third of the studies had any sound basis for their results. This coupled prompted this study to focus on transparency and accuracy.

A geographical perspective has a lot to offer in advancing the ecosystem services paradigm since geography explores the spatial relationship between people and the environment over time (Potschin and Haines-Young 2011). Ecosystems are heterogeneous, and differ greatly both spatially and temporally. The use of geographic information systems (GIS) enables users to visualize and analyze the spatial distribution

of ecosystem services (Baral et al. 2009; Nemec and Raudsepp-Hearne 2013) at a variety of scales and time steps. Researchers are able to model how changes in land-use/land-cover (LULC), management decisions, and population pressures affect ecosystem service provision and the value of these services (Kareiva et al. 2012). Landscape modeling is inherently uncertain due to model structure, inputs, and natural variability (Hou et al. 2012). The primary source of these uncertainties is spatial and temporal scaling. When information is transferred to different spatial and temporal scales a loss of accuracy occurs which affects the reliability of model results. This concept originates from the hierarchy theory which states that each ecological process operates with individual spatial and temporal characteristics (Hou et al. 2013; Müller 1992). Due to intricacy of quantifying social-ecological interactions, a large amount of uncertainty exists in ecosystem service assessment. Spatially explicit ecosystem service assessments which focus on historical conditions and future scenarios are good approaches to capturing the temporal aspects of landscape change (Hou et al. 2012). In order to ensure data quality and integrity, strategies must progress to reduce uncertainty. Hou et al. (2012) proposes that the most direct way for reducing uncertainty is improving data quality. Some approaches include improved sampling methods, augmenting quantification methods and tools, a using high quality data. Hou et al. (2012) does not indicate a specific spatial or temporal resolution for data to be considered, ‘high quality’. The appropriate scale of data must depend on the underlying ecosystem process for the services in question. Egoh et al. (2012) argues that site-specific services (e.g., pollination) will require higher spatial resolution data, whereas more generic services such as carbon sequestration for climate regulation may employ coarser resolution data, as documented by (Crossman et al. 2013).

Regardless, there is an association between high quality and high resolution data and users should attempt to use this data when possible. Another effective method for decreasing uncertainty in modeling methodologies is by verification, calibration, and validation of data (Hou et al. 2012). In a quantitative review of 153 ecosystem service regional case studies Seppelt et al. (2011) and Seppelt et al. (2012) found that between 45-80% of the studies did not provide necessary information on model results' uncertainty and validation. Reporting uncertainty is crucial for determining how tools unravel the biophysical processes of ecosystems and to convey accurate information to stakeholders and policy-makers so they are able to make informed decisions (Hou et al. 2012).

CHAPTER II

BACKGROUND

The inception of ecosystem service modeling began with the Millennium Ecosystem Assessment (MEA 2005). The MEA was comprised of a team of 1360 scientists from 95 countries (Tallis and Kareiva 2006). Researchers created numerous possible future global ecological scenarios. These scenarios were designed to predict changes in provisioning, regulating, and supporting ecosystem services based on land use policy decisions. These scenarios were conducted at the global scale (Tallis and Kareiva 2006). The complex biophysical and economic models (e.g., IMPACT, IMAGE, WaterGap, Ecopath with Ecosim) used in the MEA coupled with its coarse global scale stimulated the emergence of more accessible, reproducible, and scale-relevant modeling frameworks (Nelson and Daily 2010; Tallis and Kareiva 2006). Modeling software ranges from freely available to proprietary. Ecosystem service modeling relies on two fundamental approaches: benefit transfer and ecological production function. The benefit transfer method is where results from ecosystem service analysis in a particular area are applied to another setting where necessary data is unavailable (Wilson and Hoehn 2006; Nemec and Raudsepp-Hearne 2013). The second approach to assessing ecosystem services is to use an ecological production function as the output of ecosystem services based on an ecosystems composition and structure. The ecological production function method is believed to be more accurate because it can detect subtle changes in

ecosystems, and changes in service provision can be explained by changes in ecosystem processes within the landscape (Nelson and Daily 2010). This approach is ideally suited for informing decision-makers by developing different land management scenarios and assessing the trade-offs among services (Nemec and Raudsepp-Hearne 2013).

There are a few examples of proprietary GIS-based ecosystem service models which utilize the benefit transfer approach at the site to landscape scale that have recently emerged. EcoMetrix is a proprietary ecosystem service software package used in conjunction with field observations developed by Parametrix, Inc. and intended for use at the site-scale (i.e. 0.5 km^2) to determine environmental credits from restoration activities. The Natural Assets Information System (NAIS) is a decision support framework which generates ecosystem service values for a specific habitat type (Nemec and Raudsepp-Hearne 2013; Bagstad et al. 2013). NAIS is scalable based on project needs. The disadvantage of proprietary software is that it limits wide-scale experimentation, documentation, and user input (Nemec and Raudsepp-Hearne 2013).

In contrast, the Artificial Intelligence for Ecosystem Services (ARIES) (2009) is an example of an open-source ecosystem service modeling software which has been extensively published and enables user experimentation and input using the benefits transfer approach (Nelson and Daily 2010). ARIES is a web-based software developed by a consortium of universities and conservation NGOs (Nemec and Raudsepp-Hearne 2013). ARIES creates models of ecosystem service provision and use within a landscape. ARIES uses spatial Bayesian networks to map ecologic and economic factors which contribute to the provision and use of a particular service (Villa et al. 2008; Nemec and Raudsepp-Hearne 2013). Each point on a landscape is given an ecosystem service

provision and value based on its land use/land cover (LULC). These values are derived from study sites with similar landscape attributes (Nelson and Daily 2010). The final product is a list of ecosystem assets for a specified area, assessments of realized and potential value, and the relationship between economic and ecologic data inputs and results which can then be downloaded into GIS for further manipulation and analysis.

ARIES may be more advantageous to users because data can be acquired from existing databases and literature reviews. ARIES also requires less expertise to operate than other ecosystem service analysis models (Vigerstol and Aukema 2011; Nelson and Daily 2010). A drawback of ARIES is the difficulty of interpreting its model code (Vigerstol and Aukema 2011). This may make explanation of model relationships and results difficult, thereby creating reluctance for its use among decision-makers (Vigerstol and Aukema 2011). Another issue with ARIES and other benefit transfer models are the errors they can produce when transferring, because of the differences that exist from one site to another. The user must strive to employ the same methods of valuing one site to the next (Tallis and Polasky 2009). The benefit transfer method also assumes that every habitat type is of equal value, making it difficult for land managers to determine ecosystem service provision and value under changing conditions (Nelson et al. 2009; Nemec and Raudsepp-Hearne 2013). As ARIES use and validation grows, its strengths and weaknesses will become more apparent (Nemec and Raudsepp-Hearne 2013).

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InVEST is an open-source, spatially explicit, GIS application which relies on the ecological production function approach. It has been developed by the Natural Capital Project, a partnership between Stanford University, The Nature Conservancy, and the World Wildlife Fund (Bai et al. 2011). Current versions of InVEST can be run within ArcGIS (ESRI 2013) or as stand-alone models. InVEST is a suite of biophysical and economic tools which can model marine, terrestrial, or freshwater ecosystem services (Bagstad et al. 2013). InVEST uses ecological production functions within deterministic models driven primarily by LULC relationships. These simplified models, performed at the landscape scale, are used as a decision support tool for scenario development. The models run on a raster map at an average annual time step. Model outputs are predictive raster maps and numeric data tables of biophysical or monetary values (Bagstad et al. 2013; Terrado et al. 2014). InVEST has been used in many places globally such as Hawaii, Oregon, Columbia, Indonesia, and China (Vigerstol and Aukema 2011; Long et al. 2011).

InVEST has been criticized for its over-simplification of hydrologic processes and landscape relationships (Vigerstol and Aukema 2011; Nemec and Raudsepp-Hearne 2013). However, these limitations are noted in the InVEST procedures (Tallis et al. 2013). Locally relevant data sources (precipitation, LULC, hydrology, and geology) and efficient model parameterization have also been a critique of this model (Vigerstol and

Aukema 2011; Nemec and Raudsepp-Hearne 2013). Users interested in specific hydrological ecosystem services may benefit more from using sophisticated ecosystem process models such as SWAT (Soil and Water Assessment Tool) and VIC (Variable Infiltration Capacity). These tools may provide more accurate results. However, these tools require significantly more data and expertise to implement (Vigerstol and Aukema 2011). InVEST must be tested and validated in a variety of different ecosystems before a high degree of confidence in the tool can be achieved (Nemec and Raudsepp-Hearne 2013). A recent study assessing the viability of various ecosystem service tools (e.g., ESR, InVEST, ARIES, LUCI, MIMES, EcoServ, Co$ting Nature, SolVES, Envision, EPM, InFOREST, EcoAIM, ESValue, EcoMetrix, NAIS, Ecosystem Valuation Toolkit, Benefit Transfer and Use Estimating Model Toolkit) concluded that InVEST would be more feasible for widespread use if a system of online data sharing existed for users to obtain, spatial, ecological, and economic data to aid in model parameterization and use. Bagstad et al. (2013) provides a detailed table illustrating the strengths, weaknesses, intended use, appropriate scale, and input parameters of each aforementioned model. To date no single site or resource agency has all the data necessary to run the InVEST model. The US EPA, however, is working on an “ecological production function library” which could solve the problem of data availability and expedite modeling efforts for future users. A lesson from a recent review of ecosystem service integration into decision making (Bagstad et al. 2013) discovered that InVEST requires many assumptions, and outputs of such models can have a larger degree of uncertainty. Tools must evolve to better represent uncertainty in maps and other outputs while simultaneously preserving the merit of important findings. Despite these criticisms, InVEST still remains to be the

most peer-reviewed of the generalizable open-source ecosystem service assessment tools (Bagstad et al. 2013). InVEST is generally used to determine how human activities and climate change will affect the delivery of ecosystem services on a landscape. The model is tailored to the development and analysis of LULC management scenarios derived from stakeholder engagement (Tallis and Polasky 2009; Guerry et al. 2012).

Climate change is likely to increase the frequency and severity of episodic landscape disturbances (DiMento and Doughman 2007, 232). Incorporating climate change into planning for hazard mitigation is a new approach for public and private entities (Ruckelshaus et al. 2013). Hazard mitigation services are determined by the ability of different land cover types to retain water in soil and vegetation to prevent flooding, or attenuate wave energy from storms and tsunamis (Crossman et al. 2013). There are many cases in the literature of mitigating services. There is very little evidence, however, of ecosystem service models used to assess changes in ecosystem services due to abrupt episodic events such as floods, fires, tornadoes, earthquakes, etc. A recent study using InVEST in China (Wang et al. 2012), analyzed the impact of the 2011 Wenchuan earthquake on ecosystem services. The earthquake had a magnitude of 8.0 and devastated the southeast region of China. This study assessed the biophysical impact of the earthquake on water retention, soil conservation, and carbon storage. Researchers concluded that only two percent of the entire 76,140 km^2 study area was directly affected by the earthquake; however, ecosystem services in the study area suffered severely detrimental effects (Wang et al. 2012). Clearly, it is necessary to determine how not only human activities but natural large magnitude episodic events affect the function and provision of ecosystem services.

This study seeks to answer the following questions using the InVEST Hydropower Water Yield and Water Scarcity model to validate model results against observed field data, while striving to be transparent in reporting uncertainties and model limitations:

1. Is the InVEST hydrological process model sensitive enough to detect the effects of a large magnitude episodic disturbance (e.g., fire) in land cover on water yield in a catchment?
2. Can InVEST produce accurate and consistent results in relation to stream gauged data?
3. How does varying spatial resolution in input data effect model results?

There are discrepancies in the literature about water yield. Some studies refer to water yield as “water provisioning” (Terrado et al. 2014; Bangash et al. 2013) while others refer to water yield as “water supply” (Long et al. 2011). Water yield is the amount of rainfall that flows out of the landscape as runoff. This is determined by precipitation minus storage and evapotranspiration (Tallis et al. 2013). In addition, there can be water consumption within a catchment (i.e., industrial, residential, agricultural uses) which impacts the total annual water yield available at the mouth of a catchment for hydropower production or drinking water (Bangash et al. 2013). To simplify the nomenclature, water provisioning services will refer to water yield minus water scarcity resulting in the realized supply of available water.

CHAPTER III

DATA AND METHODS

Study Area

The study site was the Battle Creek Watershed (BCW) located in the north eastern Sacramento Valley in northern California (Figure 1). The Battle Creek Watershed drains an area of approximately 924.6 square km. The watershed includes the Latour Buttes, the western slope of Mt. Lassen, and the mountains south of Mineral, CA. Battle Creek drains from an elevation of 10,400 ft. through stream channels developed in basalt to its confluence with the Sacramento River at Cottonwood, CA (Ward and Moberg 2004).

BCW is unique for its volcanic terrain, geology, and abundant perennial cold stream flows (TCRCD 2013). The BCW has a Mediterranean climate characterized by wet winters and dry summers (CalFire 2011). Annual precipitation exceeds 317 cm/year on the top of Lassen Peak and less than 76 cm/yr at the confluence of Battle Creek and the Sacramento River. The average annual precipitation over the entire 925 square kilometer watershed is 122 cm/yr. Analyzed through USGS stream gauge #11376550 below Coleman Fish Hatchery, the mean annual discharge for Battle Creek averages 500 cfs, with the highest recorded mean annual discharge at 926 cfs and the lowest at 238 cfs (U.S. Geological Survey 2013, 1-3).. The annual water yield dispersed over the entire

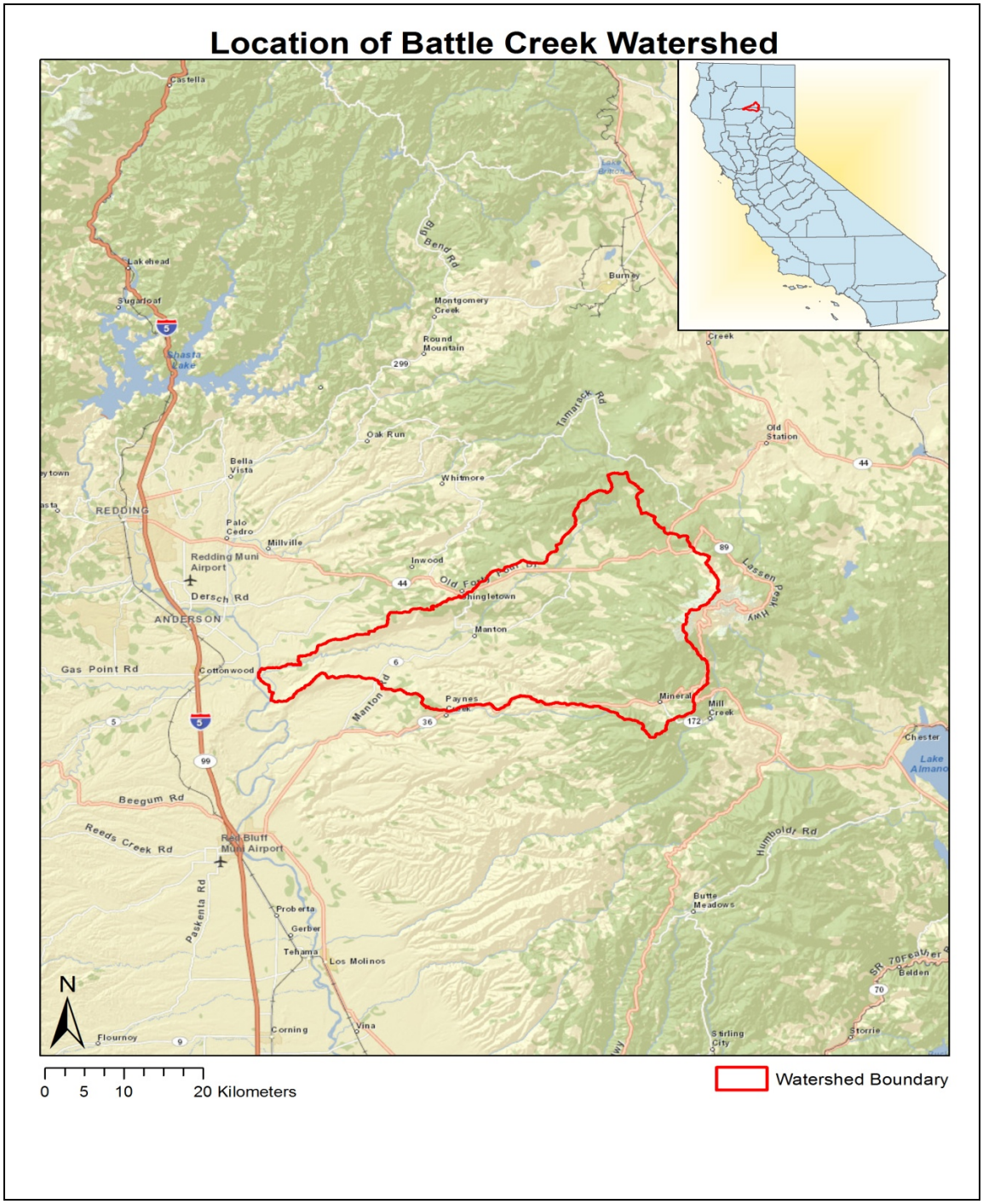

Figure 1. Location of study area.

watershed is 482.6mm/year. Peak flows are from January through March with low flows from August through October (Myers 2012).

Battle Creek and its tributaries have a history of supporting productive anadramous fish populations. The construction of dams and hydroelectric power infrastructure on the Sacramento River and Battle Creek threaten these populations. Anadramous fish restoration efforts within the Sacramento River watershed have identified Battle Creek as a high-priority watershed (Ward and Moberg 2004). Land use within the watershed ranges from residential development such as Manton, Shingletown, and Viola to protected wilderness areas within Lassen National Park. Large-scale timber harvesting is the primary land use activity within the watershed. Other land uses include livestock grazing, agricultural development such as orchards and vineyards, watershed restoration projects, wildfire suppression, roads, and recreation (TCRCD 2013; CalFire 2011). Land ownership is a patchwork of federal land management agencies (i.e., US Forest Service, Bureau of Land Management, Lassen National Park) private timber companies (Sierra Pacific Industries), conservation groups (The Nature Conservancy), and many smaller private landholders (TCRCD 2013). Logging intensity and scale has increased with the implementation of clear-cutting in 1998. The increase in logging activities has raised concerns that this practice will adversely affect water quality, increase water temperature, increase sediment transport, and decrease available anadromous fish habitat (Myers 2012).

In August-September 2012, the BCW was affected by the 247.11 square kilometer Ponderosa Fire which burned across the South Fork of Battle Creek then continued to burn north and east into Battle Creek's North Fork (Figure 2). The fire was caused by lightning and destroyed 52 residences and 81 outbuildings (CalFire 2014). A considerably large amount of forest and riparian cover was lost from this event (TCRCD

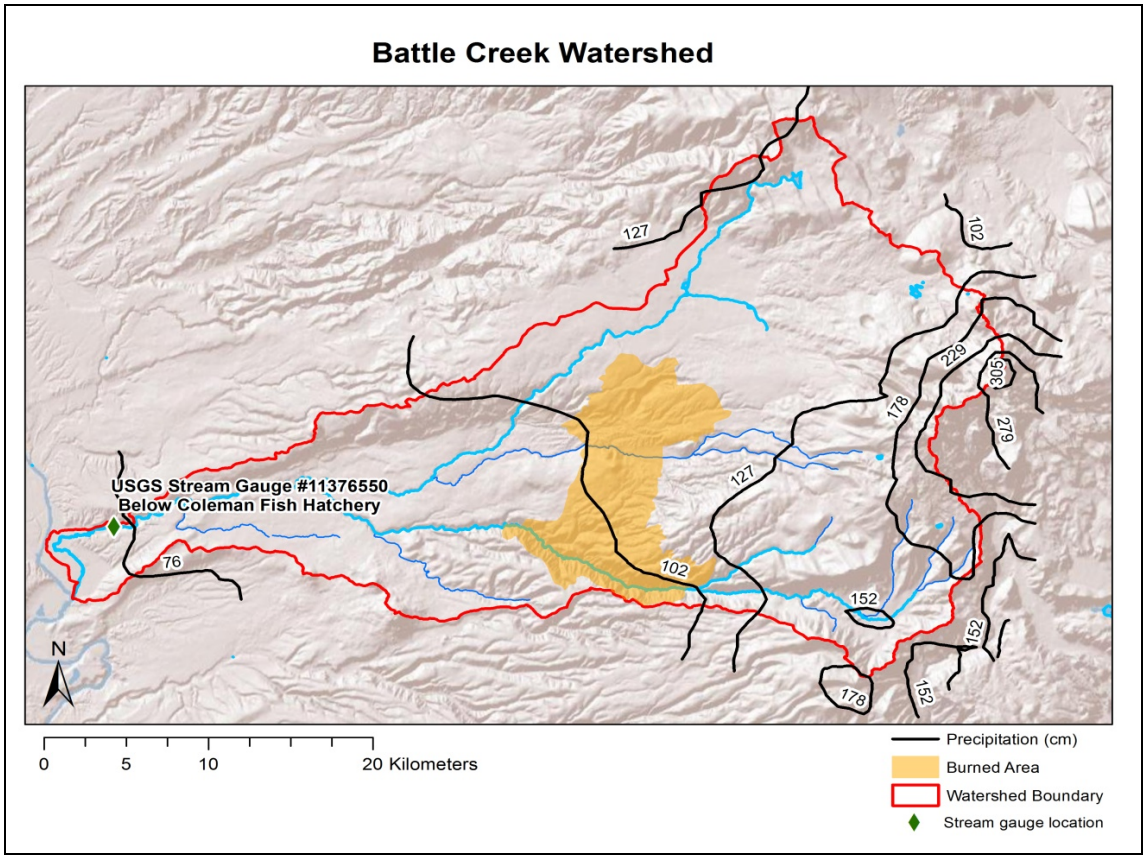

Figure 2. Battle Creek Watershed depicting stream gauge, 2012 Ponderosa Fire scar, and precipitation contours.

2013). The significant contrast change in land cover due to this event was the impetus for an analysis of how ecosystem services are affected by episodic changes in land cover.

Data and Analysis

Inputs for the water yield model vary from raster and vector GIS files and accompanying attribute values. See Table 1 for a description of ecosystem services, model steps, data inputs, processes, outputs, and data sources.

Catchment and sub-catchment delineations were derived from the USGS National Hydrography Dataset (NHD). The water yield model uses data on land use/land

Table 1. InVEST 2.5.6 Hydropower Production model data inputs and outputs

Ecosystem Service	Step	Data Input	Process	Output
Water Provisioning	Water Yield	-LULC -Average annual precipitation (mm) -Average annual reference evapotranspiration (mm) -Soil depth (mm) -Plant available water content (0-1) -Root depth (mm) -Evapotranspiration coefficient -Zhang seasonality constant (0-10)	Calculates yield per pixel as the difference between precipitation and water stored or transpired by plants	Annual average water yield mm/yr or m^3/year
	Water Scarcity	Water demand per LULC category (m^3/yr)	Subtracts consumptive uses from annual average water yield	Realized supply of water at a point of interest for water provisioning in m^3/year

cover (LULC), annual precipitation, annual reference evapotranspiration, plant available water content, effective soil depth, root depth, and a Zhang coefficient (seasonal rainfall constant). Raster-based LULC scenario maps of BCW were obtained for 1992, 2001, and 2006 from the National Land Cover Database (NLCD), at 30 meter resolution (Figure 3). The classification system used by the National Land Cover Dataset 1992 (NLCD 1992), the National Land Cover Database 2001, and the National Land Cover Database 2006 are modified from the Level II Anderson Land Cover Classification System (Multi-Resolution Land Characteristics Consortium 2014; Anderson et al. 1976). The National Landcover Dataset (NLCD) 1992, and the National Landcover Database (NLCD) 2001 are two different data products with differences in imagery and legends. Legend incompatibilities existed for developed, agricultural, and barren land use classes (Fry et

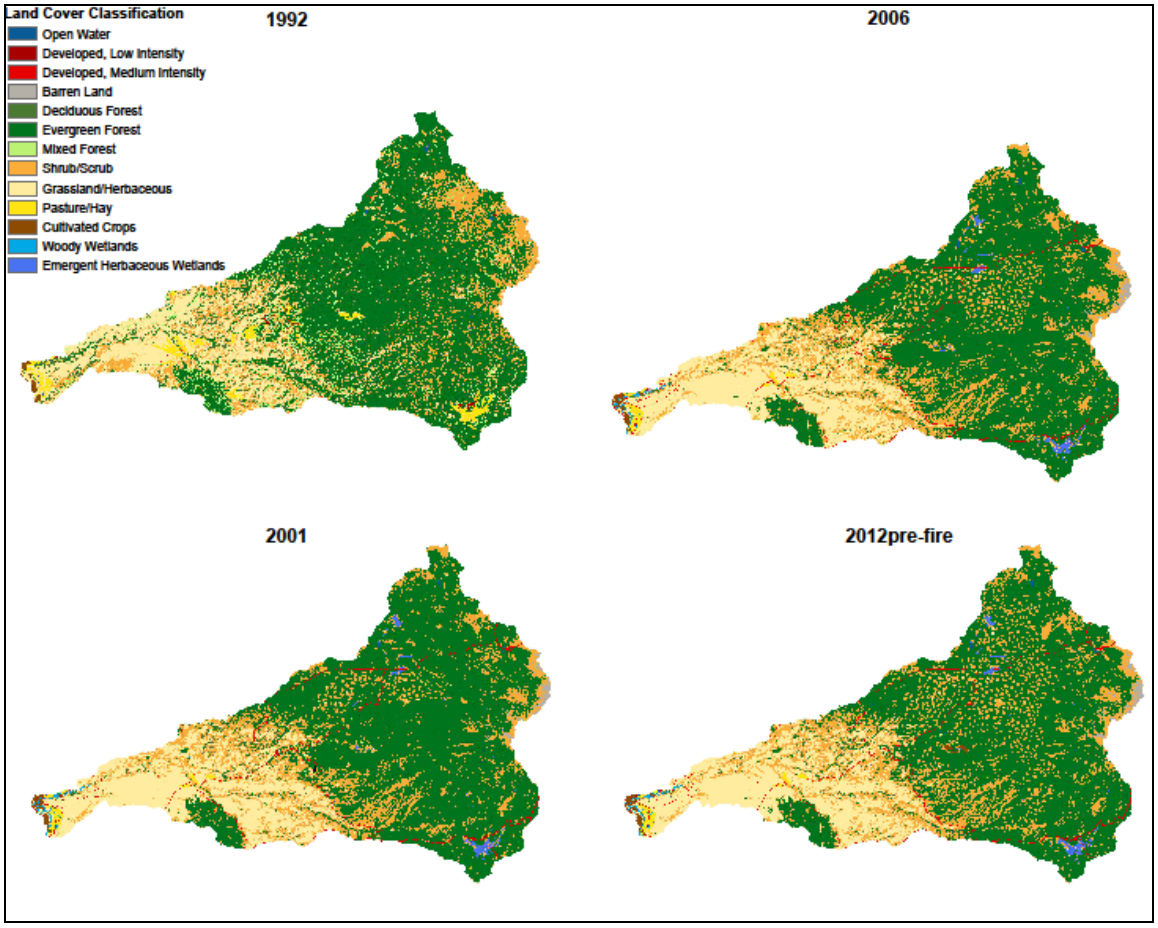

Figure 3. National land cover dataset land use/land cover (1992-2012pre-fire)**.**

al. 2009). This made a direct comparison difficult. A National Land Cover Database (NLCD) 1992-2001 Land cover change retrofit product was created. Land cover classifications were cross-walked to an Anderson Level I classification system, resulting is a Modified Anderson Level I land cover product (Fry et al. 2009). This Modified Anderson Level I classification still prevented a direct comparison of the retrofit product to the NLCD 2001 product which utilized an Anderson Level II classification system. To allow for a direct comparison from 1992 to 2001 land cover classes were manually reclassified in ArcMap (2013). The National Landcover Database (NLCD) 2001 and

2006 were created using the same methodology to allow for direct comparisons. No reclassification was required.

A 2012pre-fire and 2012post-fire LULC rasters were derived to model the sensitivity of water yield from the large episodic Ponderosa fire. The 2012pre-fire data set was created in GIS by heads-up digitizing from a July 2012 Landsat 7 image set to a mapping scale of 1:50,000 for forestry clear-cuts, agricultural operations, and expanded urban uses not present in the 2006 NLCD map. The same procedure was used using an October 2012 Landsat 7 image to digitize the area burned by the Ponderosa fire to create 2012post-fire data set. These digitized features were converted to raster at a 30 meter resolution and mosaicked into the original 2006 NLCD raster file, which essentially replaced the older cell values with the newer cell values, to create the two new LULC files. Given the intensity of the fire and complete loss of understory vegetation, the burned area was given a new designation and coded as fire scarred in the NLCD coding system (Appendix A).

InVEST does not account for water removed from the catchment by water infrastructure. A 2012 USGS Water Data Report (U.S. Geological Survey 2013, 1-3) for the catchment indicated that North Fork Battle Creek Reservoir retains 1090 acre-feet per year and McCumber Reservoir retains 430 acre-feet. In order to account for water removed by infrastructure, a single pixel was placed in each location and given a unique LULC code. In-catchment water demands (i.e., residential, agricultural) by land cover type are stored in a water demand table. Consumption by North Fork Battle Creek Reservoir and McCumber Reservoir were added to the water demand table. The water scarcity step of the model then removes water consumed in-catchment from the water

balance equation. The final LULC scenarios represent 1992, 2001, 2006, 2012pre-fire, and 2012post-fire.

In order to address inherent uncertainties in scale transfer and how InVEST performs under degrading spatial resolutions, the methodological approach of Wu et al. (2008) was emulated to determine the effects of cell size on watershed runoff. The 30 meter LULC and soil baseline datasets were resampled to 100, 120, 250, 500, 1000, and 4000 meters. These resolutions were chose based on some of the regional, national, and global data available from the Natural Capital Project (The Natural Capital Project 2014).

In order to account for the coarser cell aggregations leaving “no data” values in gaps along the catchment boundary, the watershed boundary was buffered by 4km and this buffer was used to clip the land cover and soils layer. When land cover and soil layers were resampled, all input data extended beyond the watershed boundary, however the model only calculates data within the watershed boundary.

Monthly precipitation data was obtained from PRISM Climate Group (2014) for 1992, 2001, 2006, 2012, and 2013. This monthly data was then processed in GIS to produce a rainfall raster grid which corresponded to the California water year, not calendar year values. A water year is defined as a 12-month period from October 1st to September 30th of the following year. The use of water year data allows for a direct comparison with USGS annual surface water supply reports. Water year precipitation data is available at a spatial resolution of 4km. A study by Berne et al. (2004) found that precipitation input data of 5.2 km resolution or less is appropriate in Mediterranean climates where a high degree of accuracy is required. In addition, 30yr. normal precipitation values from 1981 to 2010 at a resolution of 800m were also obtained from

PRISM. Normal precipitation scenario outputs were then compared to the average discharge for 1981 to 2010 for model validation. Normal precipitation data was used to control for the seasonal variation in precipitation. This enabled an assessment of how land cover alone influenced water yield from one year scenario to the next. Average annual evapotranspiration was derived from the Sanford and Selnick (2013) national data set at 800m resolution. Plant available water content and effective soil depth were acquired from the National Resource Conservation Service (NRCS 2013) Soil Survey Geographic (SSURGO) database (http://datagateway.nrcs.usda.gov/).

The InVEST User Manual suggested using Canadell et al. (1996), to derive rooting depth by vegetation type at the global scale. Previous InVEST case-studies (Bangash et al. 2013; Long et al. 2011; Sanchez-Canales et al. 2012) have also used this comprehensive review to determine root depth. Vegetation types listed in the review were matched to root depth studies taken from locations with landscape characteristics similar to the BCW. Each LULC class has a different corresponding root depth listed in millimeters (Appendix B). Developed areas, barren land, and burned areas will have a root depth value of *1* to signify the absence of vegetation. The Zhang seasonal rainfall constant (z) is a floating point value between 0 and 10. In regions with large winter rains the (z) value is closer to 10, in humid regions with rainfall evenly distributed throughout the year the (z) value will be closer to 1. The Zhang constant for Mediterranean climates varies be between 7 and 9 (Sanchez-Canales et al. 2012.). The chosen (z) value for the study area was 9 given the abundance of heavy winter storms in Northern California especially in the Southern Cascades.

The USGS operates a stream gauge (#11376550) near the mouth of the BCW catchment before it flows into the Sacramento River (Figure 2). Gauges collect a variety of hydrologic data from stream flow volume, stream height above a reference point, and water quality measures. The gauge below Coleman Fish Hatchery records gauge height and stream flow volume (measured in cubic feet per second) (U.S. Geological Survey 2013). Observed flow data from stream gauges can be converted from (cfs) to total annual runoff volume (m^3/year) and compared against InVEST results to validate the overall accuracy of the model.

The potential change in ecosystem services was evaluated by varying the three main landscape parameters (LULC, effective soil depth, and plant available water content) at seven spatial resolutions (30, 100, 120, 250, 500, 1000, and 4000m) among the twelve sub-watersheds within BCW to assess how water yield varied from 1992 to 2012post-fire, and how it varied from the episodic disturbance which occurred between 2012pre-fire and 2012post-fire. Holding precipitation constant by using 30 yr. normal data will control against annual variation in precipitation volumes to determine if the Ponderosa fire prompted a change in mean water yield. This longitudinal study of the change in water yield among the twelve sub-watersheds over the years required a repeated measures ANOVA. A repeated measures ANOVA is used when all observations in the same population are measured under different conditions (e.g. year). A standard One-Way ANOVA would neglect the correlation between repeated measures, violating the assumption of independence, that there is no relationship between water yield observations among the sub-watersheds over the years. Using SPSS (IBM 2014), a One-Way repeated measures analysis of variance (ANOVA) was conducted on annual and

30yr. normal precipitation scenarios to determine if there was a significant change in mean water yield from 1992 to 2012post-fire, then repeated for each spatial resolution. Results will be deemed significant at the 90% level ($p < 0.1$). A *post-hoc* Bonferroni test assessed the pairwise comparisons among the sub-watersheds to determine which ones drove any significant differences found in the overall analyses.

Reservoir Hydropower Production Model:
Water Provisioning Services

The water yield and water scarcity steps of the InVEST 2.5.6 Reservoir Hydropower model were used to determine water provisioning within the study area. The InVEST water yield step (Tallis et al. 2013) is based on the Budyko curve (Budyko 1974) and annual average precipitation. Annual water yield (Y_{xj}) is calculated for each pixel on the landscape (indexed by x = 1,2,..., X) as indicated below:

$$Y_{xj} = \left(1 - \frac{AET_{xj}}{P_x}\right) \cdot P_X \tag{1}$$

Where AET_{xj} is the annual actual evapotranspiration on pixel x for LULC j (LULC code such as 22 for residential, 11 for open water, 43 for mixed forest), P_x is the annual precipitation on pixel x.

The approximation of the Budyko curve (Zhang et al. 2001) is used to calculate the evapotranspiration partition of the water balance, $\frac{AET_{xj}}{P_x}$, given as:

$$\frac{AET_{xj}}{P_x} - \frac{1 + w_x\ R_{xj}}{1 + w_x\ R_{xj} + \frac{1}{R_{xj}}} \tag{2}$$

Where R_{xj} is the Budyko dryness index on pixel *x* for LULC *j*, defined as the ratio of potential evapotranspiration to precipitation (Budyko 1974) and w_x is a modified ration of plant available water storage to expected precipitation during the year (Zhang et al. 2001). Zhang et al. (2001) define a non-physical parameter to characterize soil-climate interactions.

$$W_x = Z \frac{AWC_X}{P_x} \tag{3}$$

Where W_x is the plant available water coefficient on pixel x. *Z* is the seasonal rainfall factor developed by Zhang et al. (2001). AWC_x is the volumetric (mm) plant available water content on pixel *x*. Soil characteristics determine the amount of water that can be stored and transpired by a plant. The Budyko dryness index where R_{xj} values are greater than 1 signify that pixels are potentially arid (Budyko 1974).

$$R_{xj} = \frac{k_{xj} \cdot ETo_x}{P_x} \tag{4}$$

ETo_x is the reference evapotranspiration from pixel *x*, and k_{xj} is the evapotranspiration coefficient for a particular LULC *j* on pixel *x*. ETo_x is an index of climatic demand while k_{xj} is determined pixel's vegetation characteristics. The water yield model generates total and average water yield at the sub-watershed level (Tallis et al. 2013).

Water consumption within the watershed is factored into the water scarcity model before a realized supply of water provisioning services can be determined. Water provisioning is defined as the difference between total water yield and total consumptive use within the watershed.

$$V_{rs} = Y - u_p \tag{5}$$

Where V_{rs} is the total realized supply. Y is the total water yield from the watershed upstream of point of interest p and u_p is the total volume of water consumed in the watershed upstream of point of interest p (Tallis et al. 2013).

Dividing observed values from the USGS stream gauge by realized supply produces a calibration constant. This calibration constant was then used in the Hydropower Calibration Table in the Water Scarcity model. This enabled the model to produce actual results resembling direct comparison to USGS observations (Tallis et al. 2013).

Agriculture is the primary water demand within BCW (California Department of Water Resources 2013). The primary crop types are irrigated pasture, vineyards, and orchards. Surface water is the main water source for irrigation purposes. Residential water needs are met through groundwater extraction. The Reservoir Hydropower model ignores surface-groundwater interactions. Given the sole reliance on groundwater for residential use, small population of approximately 13,360 people, and absence of industrial water consumption within the BCW (Sacramento River Watershed Program 2013; California Department of Water Resources 2013), residential and industrial water consumption were neglected in calculating water demand within the study area.

CHAPTER IV

RESULTS

Episodic Disturbances: The Influence of Fire on Water Provisioning Services While Controlling for Precipitation Inputs

Repeated measures ANOVA results for 30 meter annual actualized precipitation show that there is a significant difference ($p < 0.000$) between the changes in mean annual water yield among the sub-watersheds from 1992, 2001, 2006, 2012pre-fire, and 2012post-fire respectively. The effects of the fire (2012pre-fire vs. 2012post-fire) on mean annual water yield were significant (p =0.002) in a *post-hoc* test of significance. The post-hoc test indicated that 1992 was not significantly different from 2001, but was significantly different from 2006, 2012pre-fire, and 2012post-fire. A *post-hoc* test also showed that 2006, 2012pre-fire, and 2012post-fire were significantly different from all other years

30-year normal precipitation data scenario (30 meter resolution) was used to control for the seasonal variation in precipitation. Controlling precipitation enabled an assessment of how changes in LULC alone would influence the mean water yield among the sub-watersheds. Repeated measures ANOVA results showed a significant difference (p = 0.046) in mean water yield among the sub-watersheds from 1992, 2001, 2006, 2012pre-fire, to 2012post-fire. However, a *post-hoc* test of significance revealed that no significant change (p = 0.71) in water yield was noted for 2012pre-fire vs. 2012post-fire.

The InVEST hydrological process model was unable to detect the effects of the fire on water yield within the catchment.

InVEST Results and Observed Stream Gauge Data

InVEST results of realized supply (available water) were compared against water year discharge data from the USGS stream gauge. Results from the water year precipitation comparison show that InVEST reported water year runoff volumes similar to those observed by the stream gauge (Table 2). For scenarios using 30yr. normal precipitation at 30m, InVEST under-predicted values (0.48 to 1.45%; 0.71% mean) when compared to USGS average water year runoff volume corresponding with the precipitation data (e.g., 1980-2010). Scenarios using actualized precipitation at 30m exhibited an under-prediction of 0.19% to 2.43% with a mean of 0.82%.

Table 2. Predicted vs. observed available water for normal and actualized precipitation (30m resolution)

Year	InVEST Normal Precipitation (m^3/year)	InVEST Actualized Precipitation (m^3/year)	USGS Observations Normal	USGS Observations Actualized	% Error Normal	%Error Actualized
1992	437822773.46	223610113.48	444282473.68	229182256.20	1.45	2.43
2001	441758184.83	285000052.35	444282473.68	285552703.50	0.57	0.19
2006	442140180.41	724375590.22	444282473.68	726401672.10	0.48	0.28
2012	441994552.52	320020953.30	444282473.68	321570582.30	0.51	0.48
2013	441833628.18	349205562.87	444282473.68	351667713.90	0.55	0.70
Mean	441109864	380442454	444282473.68	382874986	0.71	0.82
Std.	1843457.6	197877575	0	197380210	0.41	0.92

Spatial Resolution and Model Results

Varying spatial resolution in the model inputs revealed that as the cell size of landscape parameters (e.g., LULC and soil data) increased, total water year runoff volume fluctuated slightly, but exhibited an overall decrease throughout the watershed for annual and normal precipitation scenarios (Figures 4 and 5). Annual precipitation

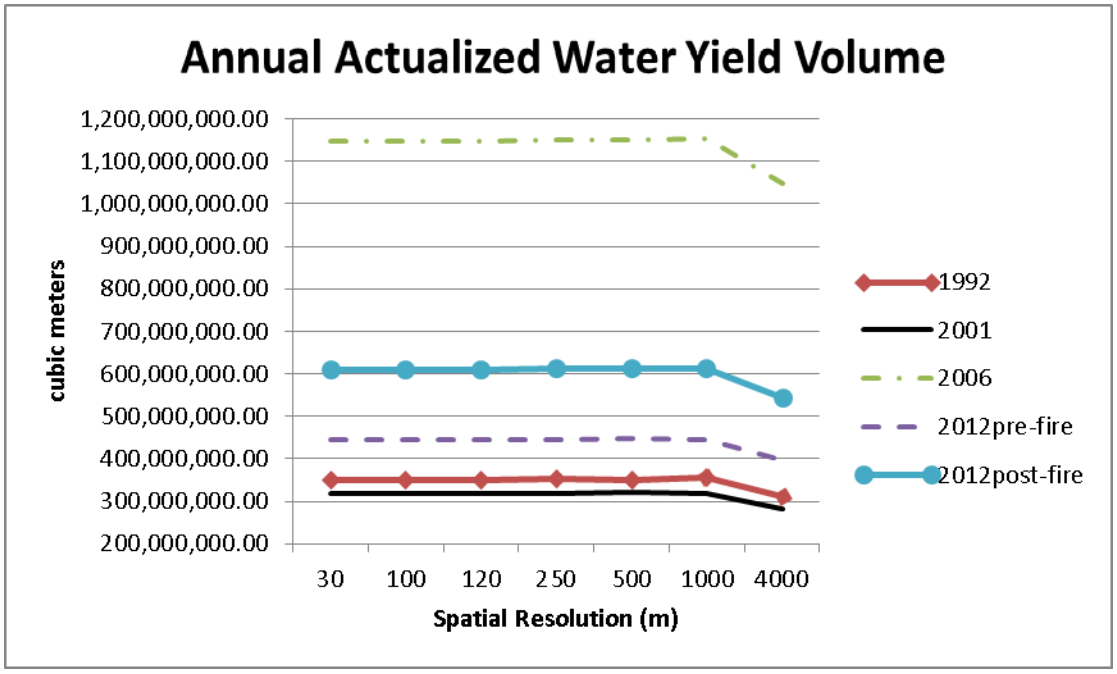

Figure 4. Spatial resolution and water yield, annual actualized precipitation 1992-2012post-fire.

scenarios show an average decrease of 1.5% over the years as spatial resolution decreased. There is an increase of 0.2% at 250m; a 0.32% increase from 250 to 500m; an increase of 0.14% from 500 to 1000m followed by a sharp decrease of 11.3% from 1000 to 4000. Normal precipitation scenarios show an average decrease in water yield of 0.92% over the years as cell size increased. Water yield increases at 500m, decreases

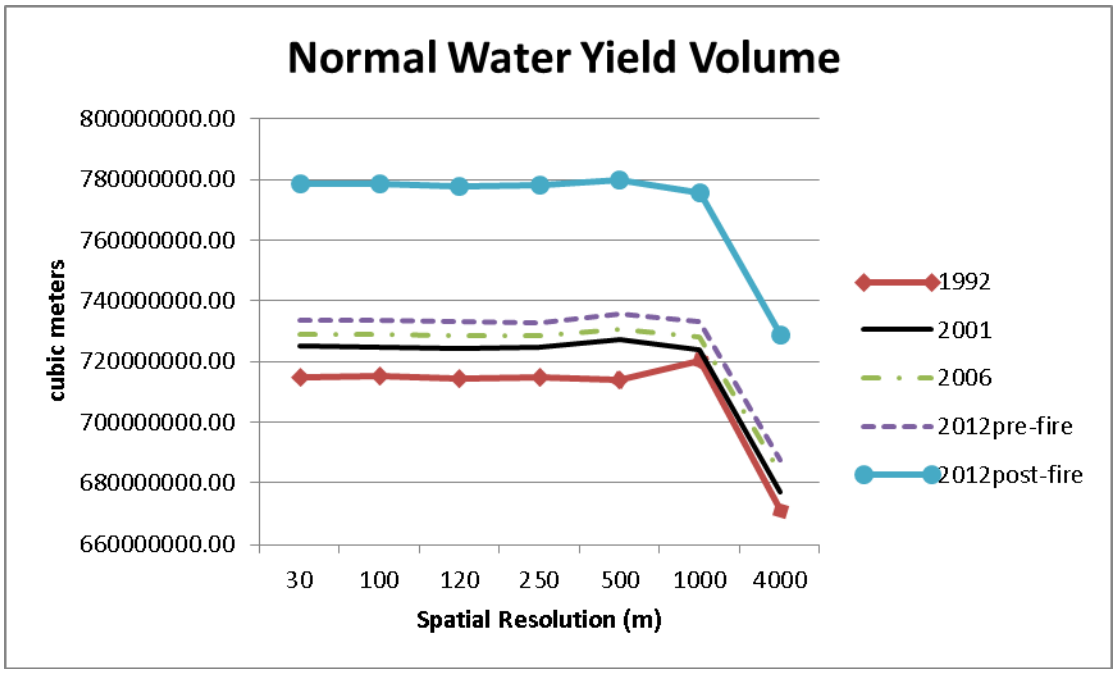

Figure 5. Spatial resolution and water yield, normal Precipitation 1992-2012post-fire.

from 500 to 1000m and drops precipitously from 1000 to 4000m. This pattern is true for all years except 1992. The 1992 scenario shows slight fluctuation with a decrease at 500m, a sharp increase from 500 to 1000m followed by a sharp drop from 1000 to 4000m.There is a fluctuation of 0.2% between 30m and 250m and a marked decline at resolution greater than 250m. This pattern of minimal fluctuation between 30m and 250m, coupled with the decrease in water volume was true for all years under both actualized and 30yr. normal precipitation scenarios.

Sub-watershed water yield results were grouped by resolution (30, 100, 120, 250, 500, 1000, 4000m) for each scenario year and a repeated measures ANOVA was performed to assess if input resolution causes a significant variation in water yield results. Analyses were performed for annual and normal precipitation scenarios. Results from the repeated measures ANOVA (Table 3) show that

Table 3. ANOVA results for annual and normal precipitation scenarios

Year	Annual ANOVA Results	Normal ANOVA Results
1992	$p = 0.043$	$p = 0.259$
2001	$p = 0.200$	$p = 0.360$
2006	$p = 0.199$	$p = 0.410$
2012pre-fire	$p = 0.203$	$p = 0.409$
2012post-fire	$p = 0.138$	$p = 0.345$

The variation among the means was not significant for annual and normal precipitation for each scenario year, except 1992. A plot of the means for annual precipitation demonstrates that mean water yield volume fluctuates slightly but remains constant until 1000m when mean water yield decreases suddenly. A plot of the means for normal precipitation shows more fluctuation with increases after 250m with a decrease after 500m followed by a sharp decline after 1000m. This pattern is true for all normal precipitation scenario years except 1992. A *post-hoc* test of significance revealed that there is no significant variation among mean water yield for both precipitation scenarios at all resolutions when compared to the 30m baseline data. The variation among the means for 1992 was significant, however a *post-hoc* test did not reveal at which resolution this variation became significant.

CHAPTER V

DISCUSSION

The Effects of Episodic Disturbances in Land Second pCover

Water is extremely sensitive to the disturbance of vegetation and soils (Baker 1990). Wildfire is a disturbance which has the greatest likelihood of altering watershed conditions. Wildfire reduces infiltration, facilitates water repellency, reduces the interception of water by forest floor cover, and decreases evapotranspiration (Ice et al. 2004). The adverse effects of wildfires on watershed conditions can last very short periods or many years (Baker 1990). Given the heterogeneity of the study area, the sub-watersheds within the BCW vary in water yield based on size, slope, elevation, vegetation cover, soil characteristics and precipitation/evapotranspiration rates. There was an increase at the entire watershed level in water yield and available water from 2012pre-fire conditions to 2012post-fire conditions. However, determining increases or decreases in water yield and available water at the sub-watershed level provides a more accurate picture of changes to hydrologic processes as a result of the fire.

The Battle Creek Watershed is divided into twelve sub-watersheds (Figure 6). The sub-watersheds affected by the fire (labeled 4, 7, 11, and 12) show an increase from 2012pre-fire to 2012post-fire conditions for both precipitation scenarios (Figure 7). The burned sub-watersheds experienced an increase in water yield of up to 70% under

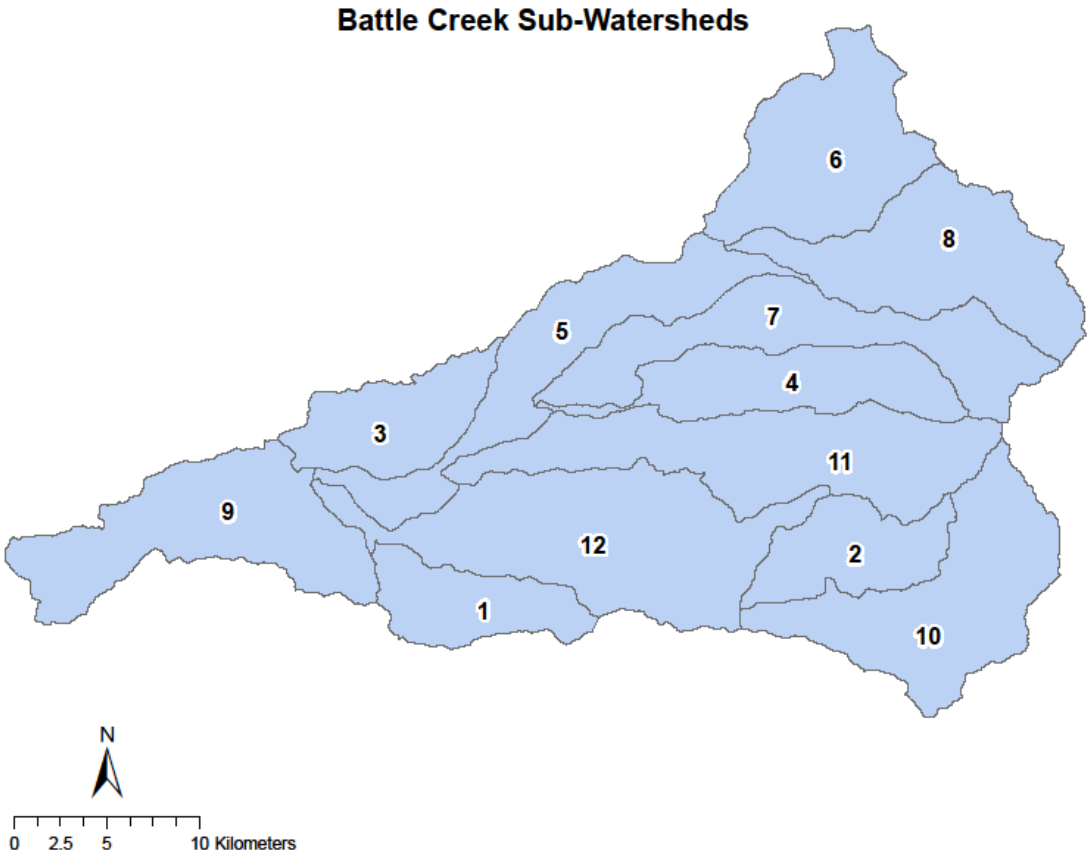

Figure 6. Battle Creek watershed, sub-watersheds.

actualized precipitation scenarios with an average increase of 54%. Under 30yr. normal precipitation scenarios increases of up to 27% in one sub-watershed and an average increase of 17% were shown.

The use of 30yr. normal precipitation data controlled for annual water year variations; allowing for an assessment of how land cover alone influences water yield. Outputs from 30yr. normal precipitation scenarios show that the change in water yield is higher from 2012pre-fire to 2012post-fire in sub-watersheds affected by the fire Sub-watersheds 4, 7, 11, and 12 saw an increase of 26.58%, 6.56%, 9.05%, and 24.55% respectively. The Ponderosa fire caused a 12% reduction in forest and grassland cover,

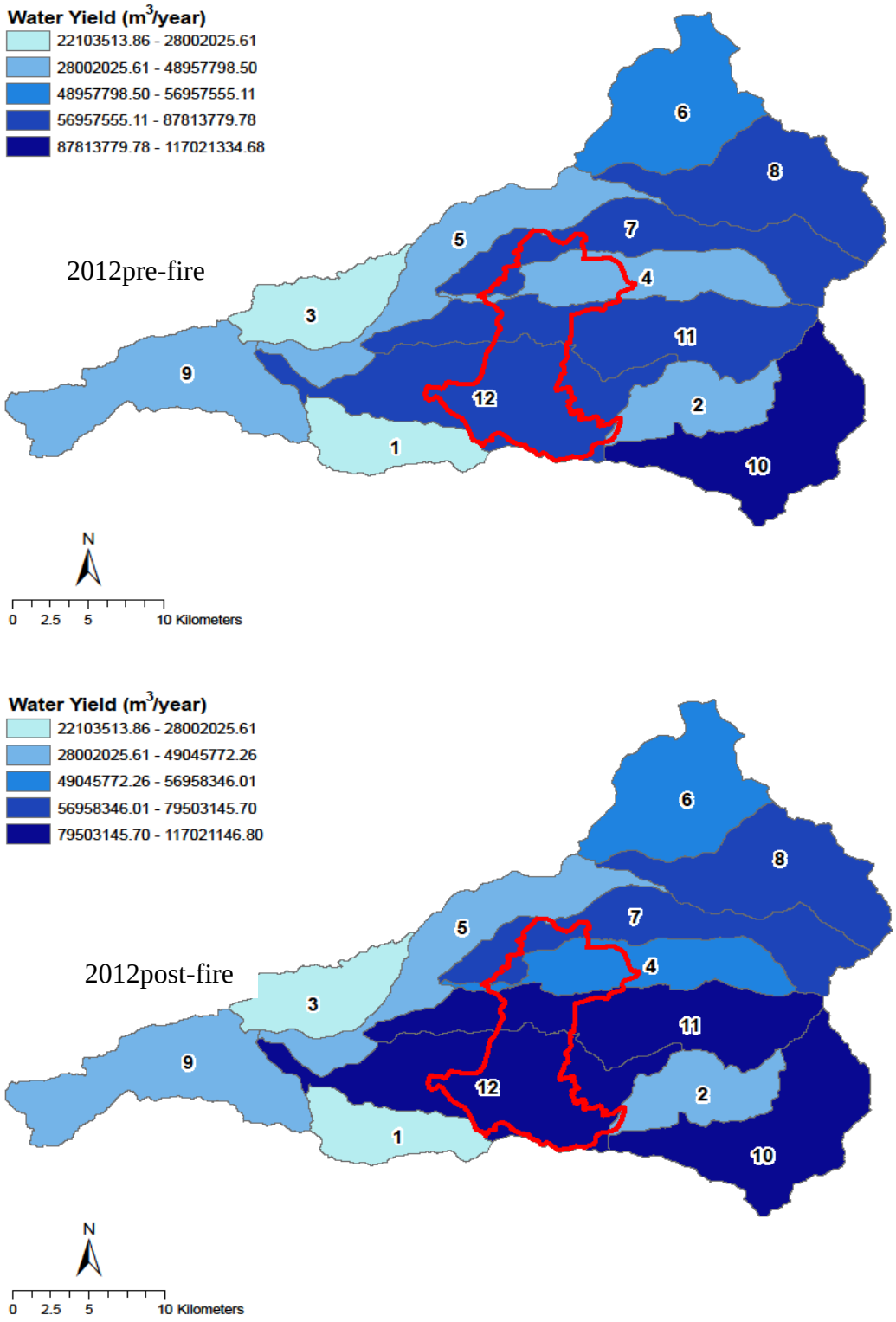

Figure 7. Change in water yield due to fire

converting it to barren ground. This sudden reduction in vegetation and alteration of soil characteristics should have increased water yield due to excess runoff and lack of soil infiltration. Perhaps the fire was not large enough, or occurred in a location within the catchment less responsible for the contribution to water yield. The topography of the area may answer questions why certain sub-watersheds consistently contribute more to water yield than others. The watershed begins in the floor of the Sacramento Valley and increases in elevation until it reaches Mt. Lassen. Precipitation is not evenly distributed throughout the watershed. The map (see Figure 2) depicting Isohyetal contours demonstrates the relationship between increasing elevation and precipitation, which is strongly orographic and typical of California's precipitation regime. A simple overlay analysis of the fire scar polygon within the sub-watersheds primarily responsible for water yield (labeled 2, 7, 8, 10) revealed that only 9.73% of the Ponderosa fire occurred within the major water contributing watersheds. The fire burned in sub-watersheds which contribute less to water yield than sub-watersheds located in the upper reaches of the catchment. The location of the fire is just as important as the drastic alteration of land cover when assessing freshwater ecosystem services. To determine the importance of the fire's location on service provision, future research should create a hypothetical fire scenario of equal size located in major contributing sub-watersheds to determine if location is a primary factor in determining water yield within the catchment, as opposed to only the extent of a fire.

Accuracy assessments must be considered when inferring land use behavior based upon National Land Cover Database data. Thematic accuracy assessments were conducted for 1992, 2001, and 2006 National Land Cover Data. Assessments at the

classification level used in this study show that the overall accuracy for 1992, 2001, and 2006 is 70%, 78.7%, and 78% respectively (Wickham et al. 2006; Wickham et al. 2010; Wickham et al. 2013).

The catchment experienced changes in land use patterns from 1992-2012 (Table 4). Those experienced from 1992 to 2001 can be attributed to different versions of land cover data products. A 1992 retrofit was created which used a Modified Anderson

Table 4. Land Use /Land Cover Percentage 1992-2012 post-fire

LULC Code	Land Use Type	1992	2001	2006	2012pre-fire	2012 post-fire
11	Open Water	0.14	0.10	0.10	0.10	0.10
22	Developed Low Intensity	0.09	0.07	0.07	0.07	0.06
23	Developed Medium Intensity	0.03	1.13	1.13	1.13	0.99
31	Barren Land	0.12	0.47	0.47	0.47	0.47
41	Deciduous Forest	3.43	1.21	1.18	1.18	1.06
42	Evergreen Forest	57.74	59.15	57.36	55.35	49.18
43	Mixed Forest	4.19	0.10	0.09	0.09	0.07
52	Shrub Scrub	16.22	19.50	21.07	23.08	19.42
71	Grassland Herbaceous	15.85	16.77	17.03	16.97	15.97
81	Pasture Hay	1.89	0.45	0.45	0.45	0.45
82	Cultivated Crops	0.31	0.25	0.25	0.34	0.34
90	Woody Wetlands	0.00	0.20	0.20	0.20	0.20
95	Emergent Herbaceous Wetlands	0.00	0.58	0.59	0.56	0.56
96	Fire Scarred	0.00	2.00	0.00	0.00	11.13

Level I classification system. After the completion of the 1992 retrofit product researchers found that 97% of the land cover remained unchanged from 1992 to 2001. The 3% change occurred amongst forest, grass/shrub, and agriculture classes (Fry et al. 2009). The manual reclass in this study yielded similar results to those found by Fry et al.

(2009) with an approximate 3% change amongst forest, grass/shrub, and agricultural land cover classes. This may explain the increase in evergreen forest from 1992 to 2001, although the practice of heavy clear-cutting began in 1998 (Myers 2012). There was a 14.35 km^2 reduction in agricultural land cover from 1992 to 2001. Agricultural land was converted to grasslands or emergent wetlands. Medium-intensity development increased from 1992 to 2001 and remained consistent until 2012post-fire conditions. Shrub/scrub increased from 1992 to 2012pre-fire, only to decrease after the fire. Clear-cut patches are coded as shrub/scrub indicating that the fire occurred in areas where abundant clear-cut forestry had occurred (Figure 3).

The rapid reduction in forest and shrub cover may also contribute to an increase in water yield. Examining the Normal precipitation scenario, to control for seasonal variation, at 30m resolution shows that there is no significant change (p=1) among the mean water yield among the sub-watersheds from 1992 to 2001 as a result of increased clear-cutting.

Many ecosystem service studies in the literature investigate anthropogenic disturbances to an ecosystem. There are a few studies which have assessed the ways in which large disturbances alter ecosystem structure and function (Millward and Kraft 2004; Sinclair and Byron 2006). Studies using InVEST to document the spatial variation in ecosystem services as a result of an episodic disturbance are rare (Wang et al. 2012). In the face of climate change, the frequency and intensity of extreme weather and fire events is likely to increase (Westerling and Bryant 2008; DiMento and Doughman 2007, 232) in the Western US. Sudden episodic disturbances (i.e., fire, flood, hurricane) immediately impact ecosystem structure and service provision (Chiang et al. 2014). Studies like this

one will be important for informing future ecosystem service assessments commissioned in response to environmental hazards.

Year 2013 was the driest year in recorded history for many regions of California prompting Governor Edmund G. Brown to declare a drought state of emergency (California Department of Water Resources 2014). As impacts of the drought are felt throughout California, this study has promising implications for land managers seeking to maximize water yield. In South Africa's Western Cape province, prescribed fires are utilized extensively by land managers to rejuvenate vegetation, reduce wildfire risk, increase water yield, and control the spread of invasive species (Richardson et al. 1994). The Catchment Management System (CMS) in South Africa is a suite of GIS-based models which prioritizes areas for prescribed burns and monitors results after the burn treatment (Richardson et al. 1994). In California, the similar Mediterranean climate, high wildfire risk, and water scarcity issues could prompt land managers to employ innovative solutions, similar to those used in South Africa, to increase water supply.

If parameterized correctly, land managers could generate hypothetical prescribed burn scenarios to determine how much water would be yielded from different parts of a landscape. The CalFire fire history database contains data on wildfire and prescribed fires from 1878 to 2012. The following map (Figure 8) depicts fire frequency in Battle Creek Watershed for recorded fires in 1911 to 2012. Parts of the watershed that have never burned during this 101 year time period are coded as "0" and the most frequent fire occurrences are coded as "6" for six times. Looking at the distribution of past fires within the catchment most occur in or around the perimeter of the 2012 Ponderosa fire. This fire frequency data could be used to reduce wildfire risk and increase

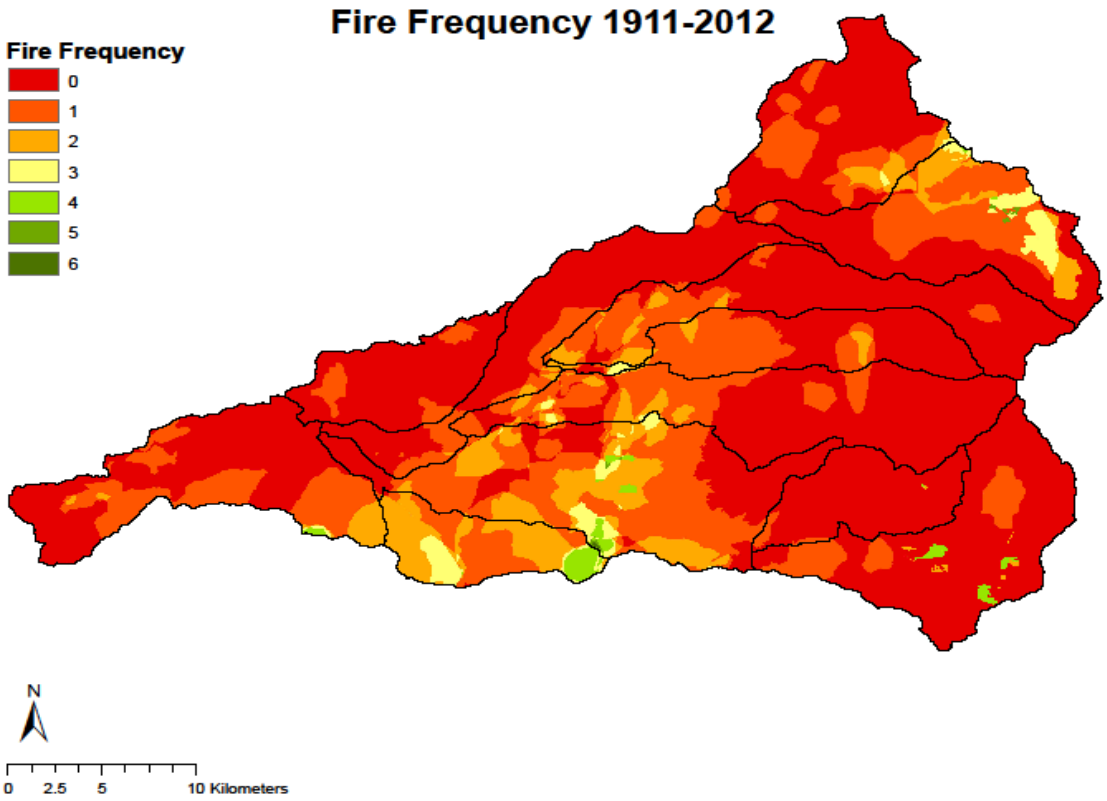

Figure 8. Fire frequency, 1911-2012.

water yield by identifying management “burning blocks” based on the dominant vegetation type and past fire return intervals by sub-watershed. The accessibility and lack of expertise required for the use of InVEST could result in the rapid assessment of potential burn blocks to respond to the effects of the drought. The volatilization of organic material from fires reduces the infiltration capacity of the soil (Letey 2001). Fire also decreases evapotranspiration by removing vegetation from the landscape (Ice et al. 2004). The desire to increase water yield through the use of prescribed burning programs must be moderated by the need to conserve biodiversity, reduce soil erosion, sediment redistribution, and mass wasting events which result from fires (Shakesby and Doerr

2006). Used appropriately, InVEST and this method for selecting and evaluating burn blocks could develop the possibility to increase water supply over the long term.

Spatial Resolution and Biophysical Models

There are many studies on ecosystem quantification and mapping, many of which are at a variety of scales (Crossman et al. 2013). Studies evaluating whether there is a threshold for when certain spatial resolutions begin to produce inaccurate data are absent in the literature. A review of 122 studies focusing on modeling and quantification methods found that 51% were at the regional scale, and 15% were at the national scale (Crossman et al. 2013). A typical spatial scale in the US would be modeling at 30 meter cell size, in other regions where data availability is poor, 1 to 10 km cell sizes may be used (Vigerstol and Aukema 2011). The use of coarse resolution data seems common for InVEST modeling based on the database of data sources from the Natural Capital Project (2014). If coarse global data is being used to quantify ecosystem services for fine local scale policy questions, will the InVEST model produce inaccurate results? This study analyzed if there is an appropriate or inappropriate scale of data for model inputs (e.g., land cover and soil data). When a model is used at a scale different from the one it was designed, the model loses competence (Hou et al. 2012).

This study showed that as cell size increased, water yield volume fluctuated but showed a general decrease. Results from this study are similar to those reported in Wu et al (2008), as cell size increases water yield volume decreases. It is important to note that while volume decreases as cell-size increases, the mean and variation among resolutions (e.g. 100, 120, 250, 500, 1000, 4000m) are not statistically significant when

compared to 30m baseline model results. The only significant variation that occurred was in the annual 1992 scenario (p=0.043). It was puzzling that the 1992 annual precipitation scenario was the only scenario to exhibit a significant variation among the means for each resolution. A *post-hoc* test of significance did not show at which resolution the variation became statistically significant. Further investigation revealed that cell aggregation as a result of resampling was responsible for the significance in variation. As land cover was resampled at higher cell-sizes the percentages of land cover classes changed. See Appendix C for land cover change by resolution for each year. In 1992 the predominant land cover classes were Evergreen Forest, Shrub/Scrub, and Grassland/Herbaceous. Evergreen Forest decreased from 53.7% to 43.2% when resampled from 30m to 4000m (Table 5).

Table 5. Percent Land Cover Change by Resolution, 1992

		Land Cover						
LULC Code	Land Use Type	30.000	100	120	250	500	1000	4000
11	Open Water	0.24	0.23	0.22	0.26	0.25	0.30	0.94
22	Developed Low Intensity	0.08	0.09	0.09	0.09	0.04	0.12	0.00
23	Developed Medium Intensity	0.04	0.04	0.04	0.02	0.01	0.00	0.00
31	Barren Land	0.43	0.41	0.43	0.42	0.31	0.71	0.00
41	Deciduous Forest	4.30	4.30	4.30	4.29	4.26	4.58	4.72
42	Evergreen Forest	53.77	53.66	53.81	54.07	53.77	52.68	47.17
43	Mixed Forest	4.93	4.98	4.95	4.96	4.97	4.46	2.83
52	Shrub Scrub	15.34	15.30	15.18	15.04	15.44	16.65	19.81
71	Grassland Herbaceous	18.32	18.44	18.42	18.30	18.34	17.95	18.87
81	Pasture Hay	1.97	1.99	1.98	1.94	1.95	2.20	3.77
82	Cultivated Crops	0.58	0.58	0.58	0.61	0.61	0.24	0.00
90	Woody Wetlands	0.00	0.00	0.00	0.00	0.00	0.00	0.00
95	Emergent Herbaceous Wetlands	0.00	0.00	0.00	0.00	0.00	0.00	0.00
96	Fire Scarred	0.00	0.00	0.00	0.00	0.00	0.00	0.00
98	McCumber Reservoir	0.00	0.00	0.00	0.00	0.01	0.06	0.94
99	North Fork Battle Creek Reservoir	0.00	0.00	0.00	0.00	0.01	0.06	0.94

While Evergreen forest decreased, Shrub/Scrub, Grassland/Herbaceous, and Pasture/Hay land cover classes increase. These land cover classes have much lower evapotranspiration coefficients in the InVEST biophysical table (e.g., Evergreen Forest = 988; Shrub/Scrub = 500). The evapotranspiration coefficients are based on plant physiological characteristics for each land cover class and are used to modify the evapotranspiration input dataset (Tallis et al. 2013). Since InVEST assumes that the amount of water running off each pixel as precipitation minus evapotranspiration, changes in model calculated evapotranspiration can significantly change water yield results. The following maps (Figures 9-13) depict 1992 dominant land cover layers (e.g. Deciduous Forest, Evergreen Forest, Mixed Forest, Shrub/Scrub, Grassland/Herbaceous, Pasture/Hay), water yield, and evapotranspiration at increasing cell sizes.

A visual observation shows that evapotranspiration is highest in areas of the watershed dominated by Evergreen Forest. Water yield is also the lowest in these areas. Water yield is highest in the upper reaches of the watershed where there is some Evergreen Forest however the strongly orographic precipitation regimes discussed in the previous section are responsible for this. Water Yield and Evapotranspiration for 1992 have decreased from 30m to 500m. In contrast, water yield and increases as evapotranspiration decrease from 500m to 1000m. Then from 1000m to 4000m, water yield decreases as evapotranspiration increases.

This fluctuation of water yield and evapotranspiration is not evident in other years. In 2012, water yield and evapotranspiration decrease as cell size increases. There is more generalization that occurs with larger cell sizes with less land cover classes represented at 4000m than in 1992 (Table 6).

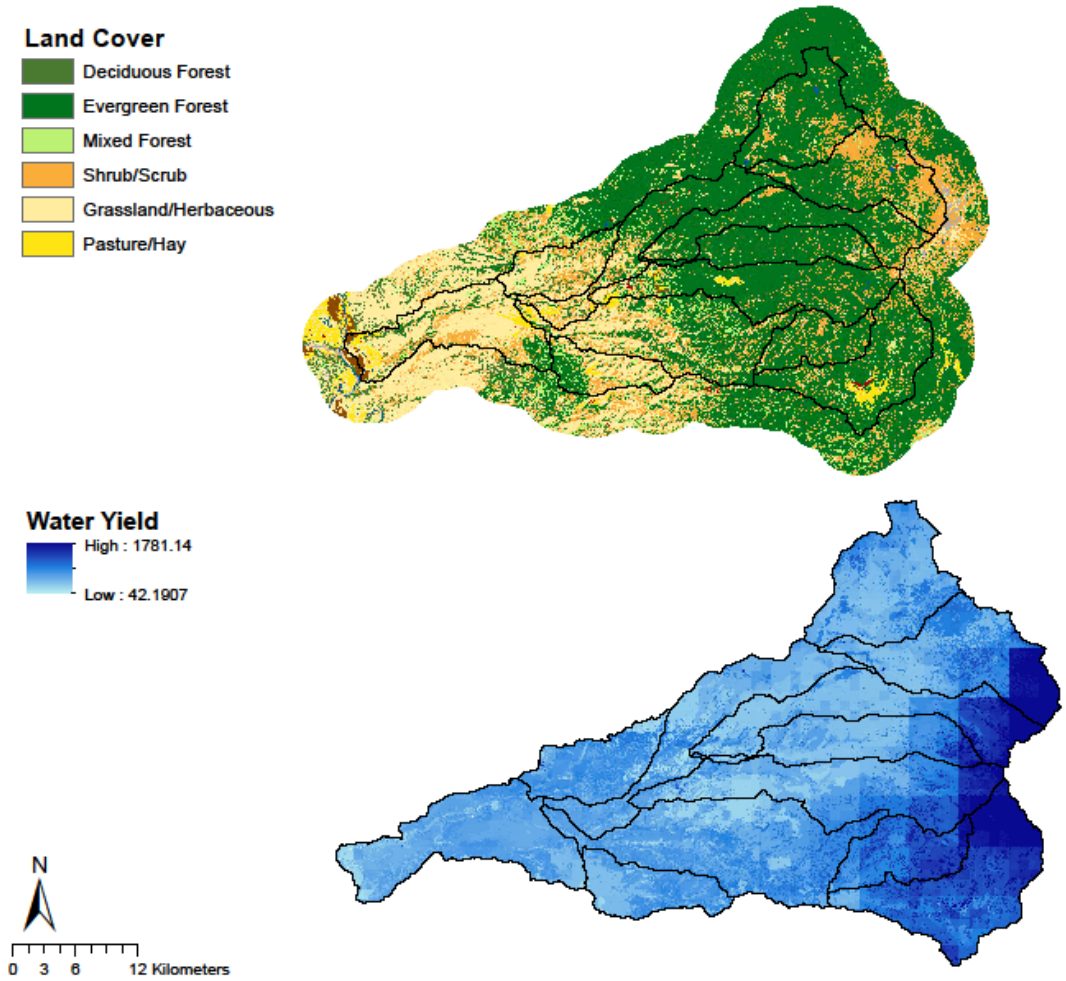

Figure 9. Land cover and water yield 1992, 30m.

The following maps (Figures 14-17) show that in 2012 as cell size increases water yield and evapotranspiration decrease, however this decrease does not create a statistically significant variation among water yield means at various resolutions.

The generalization due to cell aggregation results in the loss of land cover layers that increase or decrease water yield. InVEST sums and averages water running off each pixel to the sub-watershed level. As cell size increases, certain land cover layers begin to dominate many of the sub-watersheds. This calculation of water yield to the

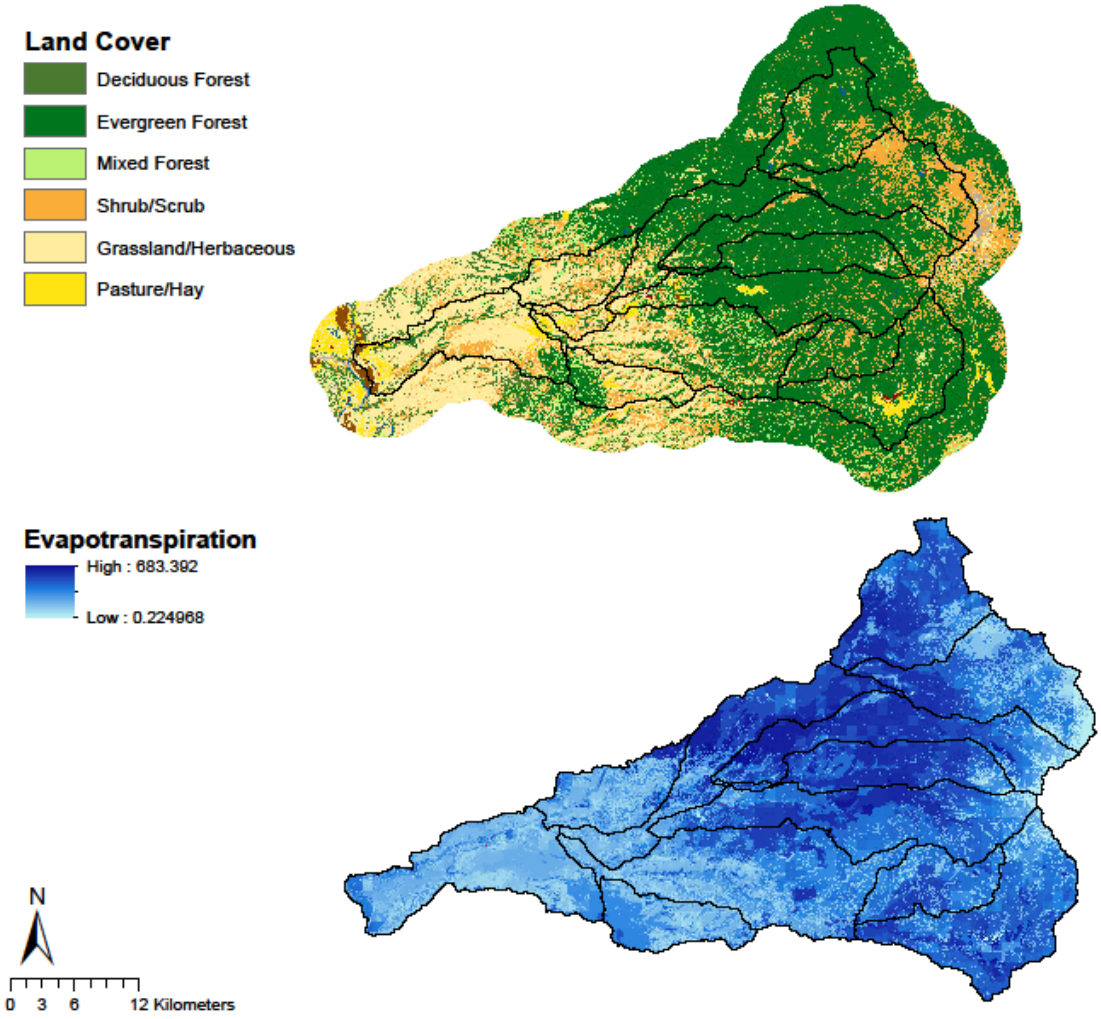

Figure 10. Land cover and evapotranspiration 1992, 30m.

sub-watershed boundary presents problems when a single cell may encompass the entire sub-watershed. The fluctuations in evapotranspiration and water yield in 1992 compared to the steady decrease in water yield and evapotranspiration in 2012 demonstrates that cell size does not always significantly influence water yield, rather the aggregation of cells and their distribution among the sub-watersheds can have a more significant influence on water yield. Ultimately this indicates that resolution of soil and land cover data is not what is driving the model, if water yield is calculated as precipitation minus

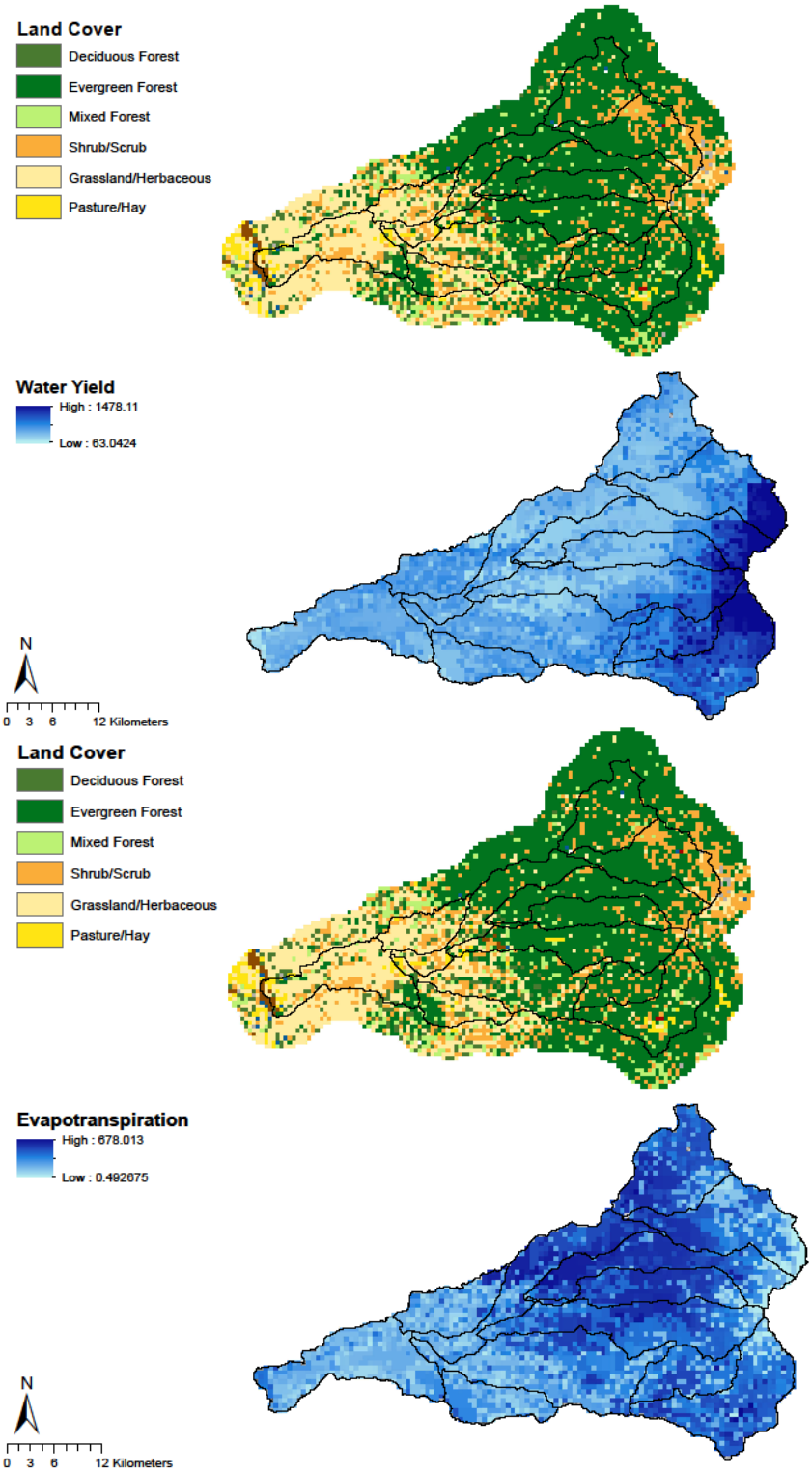

Figure 11. Land cover, water yield, and evapotranspiration 1992, 500m.

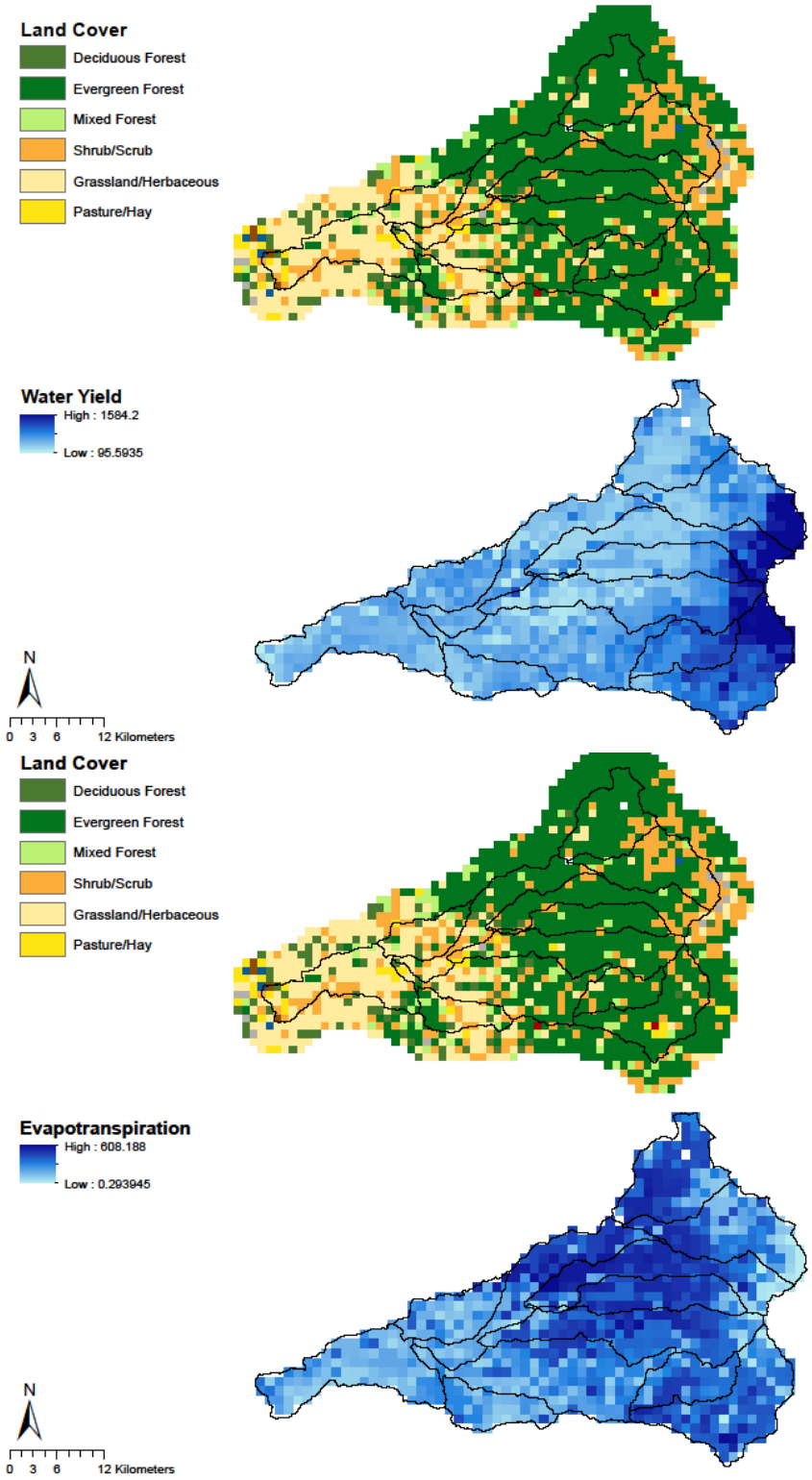

Figure 12. Land cover, water yield, and evapotranspiration 1992, 1000m.

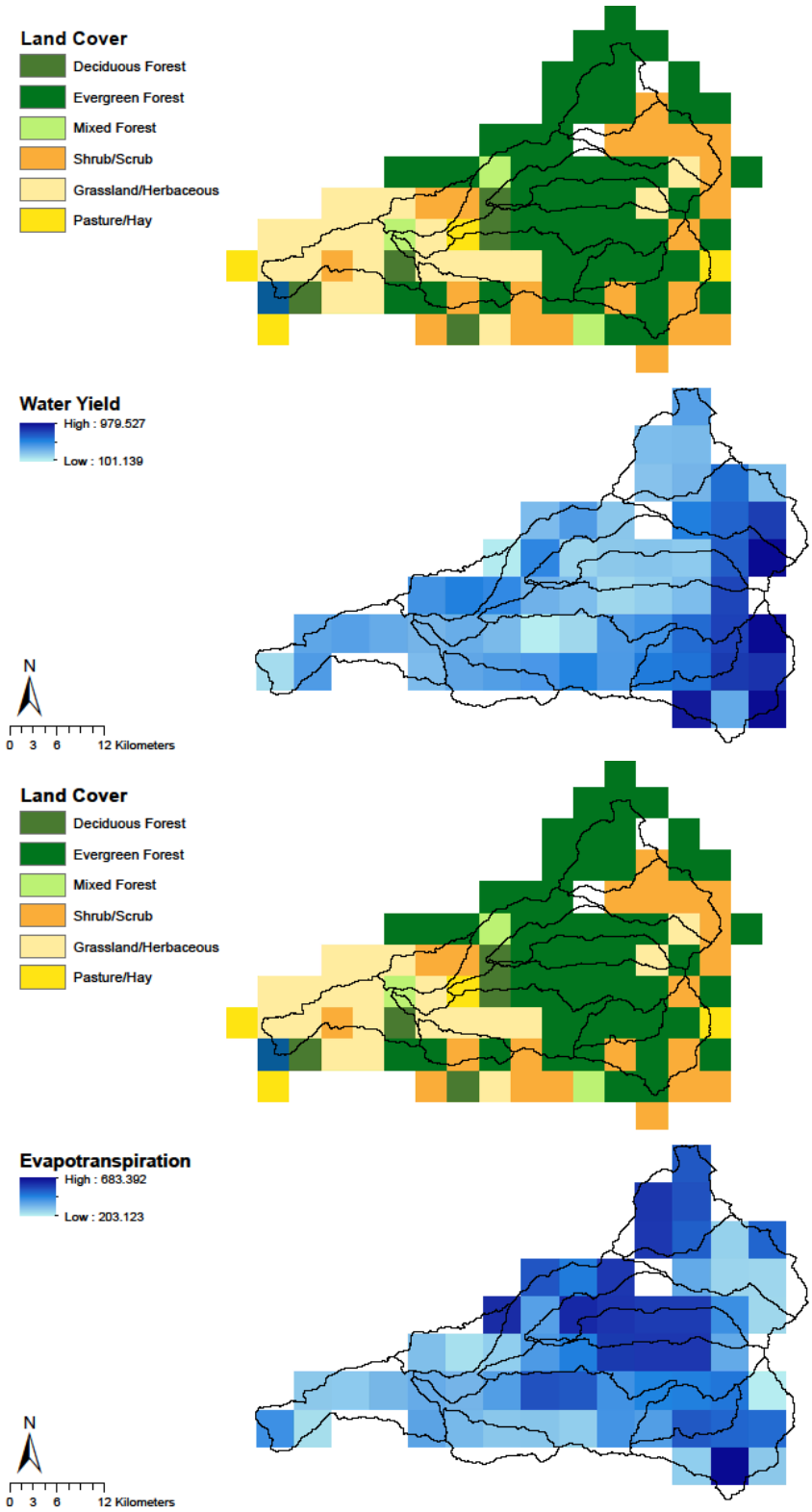

Figure 13. Land cover, water yield, and evapotranspiration 1992, 4000m.

Table 6. Percent Land Cover Change by Resolution, 2012

LULC Code	Land Use Type	Land Cover 30	100	120	250	500	1000	4000
11	Open Water	0.25	0.25	0.25	0.24	0.21	0.24	0.00
21	Developed Open Space	1.55	1.53	1.51	1.40	1.72	1.42	0.95
22	Developed Low Intensity	0.12	0.13	0.11	0.11	0.16	0.00	0.00
23	Developed Medium Intensity	0.04	0.04	0.04	0.03	0.04	0.00	0.00
24	Developed High Intensity	0.04	0.05	0.04	0.04	0.04	0.06	0.00
31	Barren Land	0.81	0.82	0.81	0.84	0.93	0.89	1.90
41	Deciduous Forest	1.13	1.13	1.12	1.12	1.09	1.19	0.95
42	Evergreen Forest	52.95	52.96	52.93	53.11	52.62	53.23	51.43
43	Mixed Forest	0.09	0.09	0.09	0.08	0.06	0.00	0.00
52	Shrub Scrub	21.79	21.83	21.85	21.74	22.25	22.11	20.00
71	Grassland Herbaceous	18.97	18.95	19.00	18.90	18.68	18.26	20.95
81	Pasture Hay	0.79	0.80	0.80	0.82	0.78	1.13	0.00
82	Cultivated Crops	0.66	0.66	0.65	0.69	0.68	0.77	0.00
90	Woody Wetlands	0.33	0.33	0.32	0.38	0.33	0.18	0.95
95	Emergent Herbaceous Wetlands	0.47	0.46	0.48	0.47	0.37	0.41	0.95
96	Fire Scarred	0.00	0.00	0.00	0.00	0.00	0.00	0.00
98	McCumber Reservoir	0.00	0.00	0.00	0.00	0.01	0.06	0.95
99	North Fork Battle Creek Reservoir	0.00	0.00	0.00	0.00	0.01	0.06	0.95

evapotranpsiration, evapotranspiration can be modified by changes in land cover layers due to resolution thus changing water yield. This reinforces the findings made by Sanchez-Canales et al. (2013) in their sensitivity analysis of InVEST in the Llobregat basin in Spain. Results from this study revealed that effects of the seasonality constant on model results are negligible. Precipitation and evapotranspiration are more important for obtaining precise model outputs (Sanchez-Canales et al. 2013). The InVEST Manual states that the Hydropower Model is driven more by parameter values (e.g. seasonality constant, root depth, evapotranspiration coefficient) than by individual hydrologic processes within the catchment (Tallis et al 2013). These findings have compelling

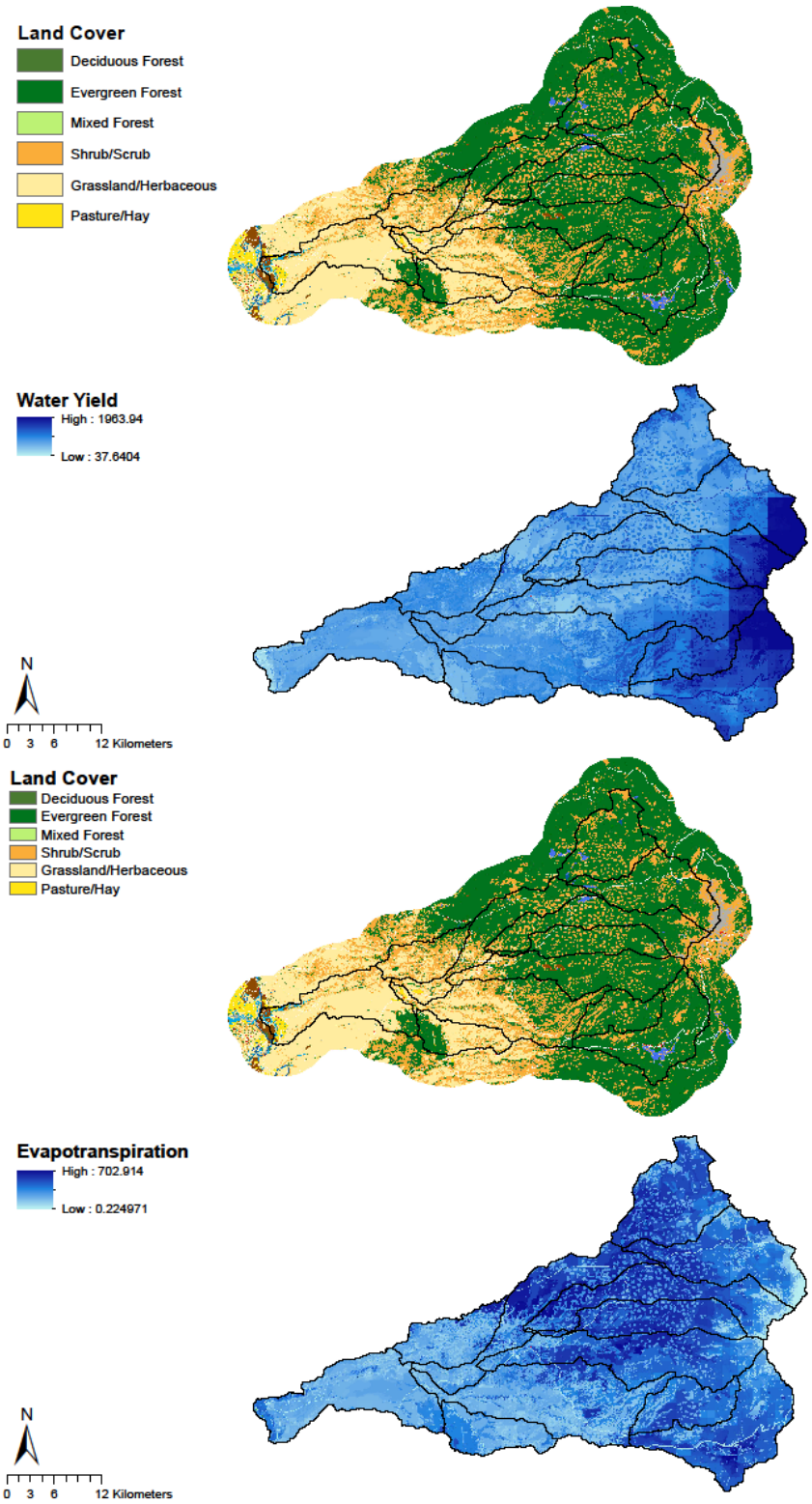

Figure 14. Land cover, water yield, and evapotranspiration 2012, 30m.

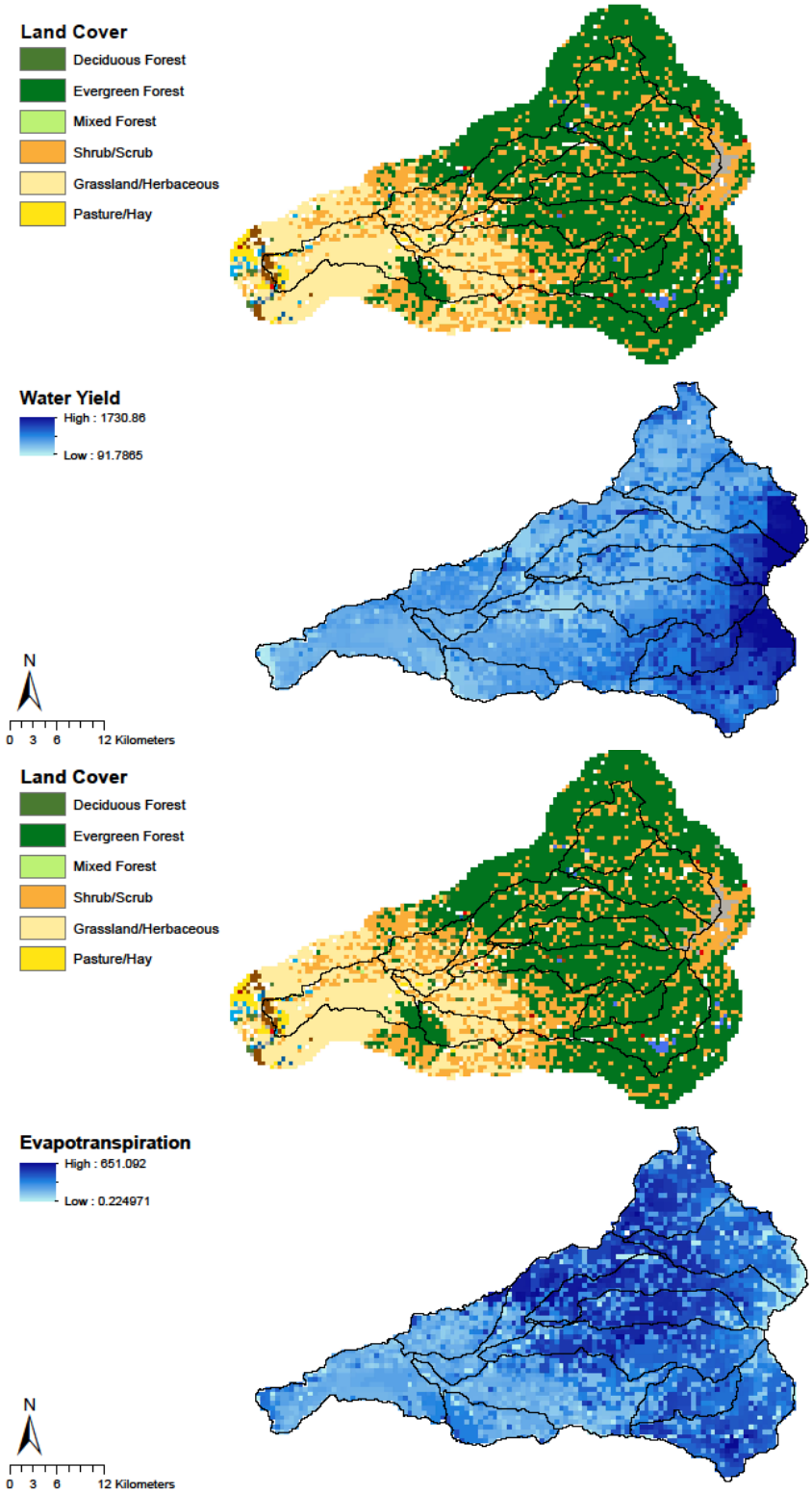

Figure 15. Land cover, water yield, and evapotranspiration 2012, 500m.

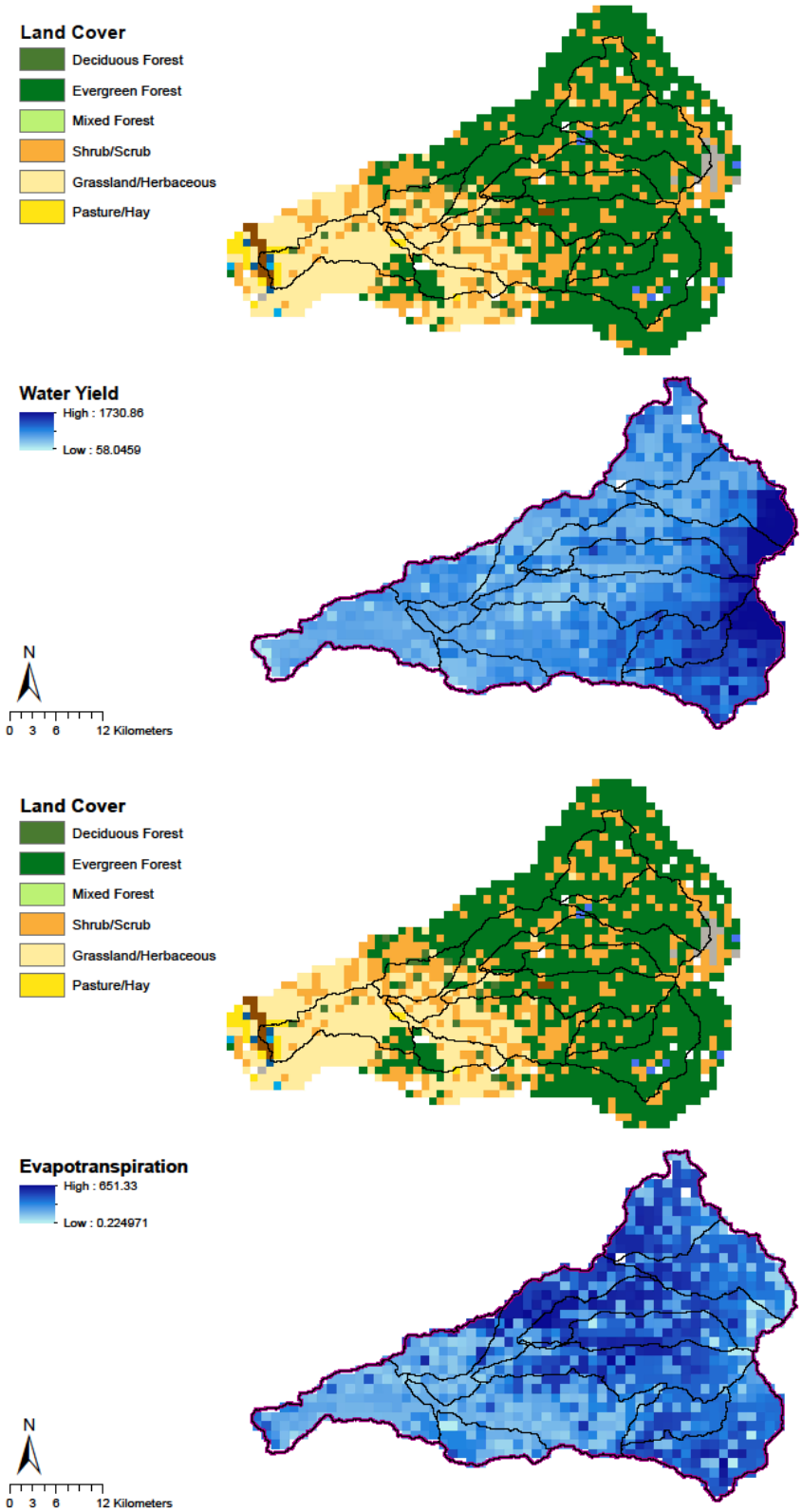

Figure 16. Land cover, water yield, and evapotranspiration 2012, 1000m.

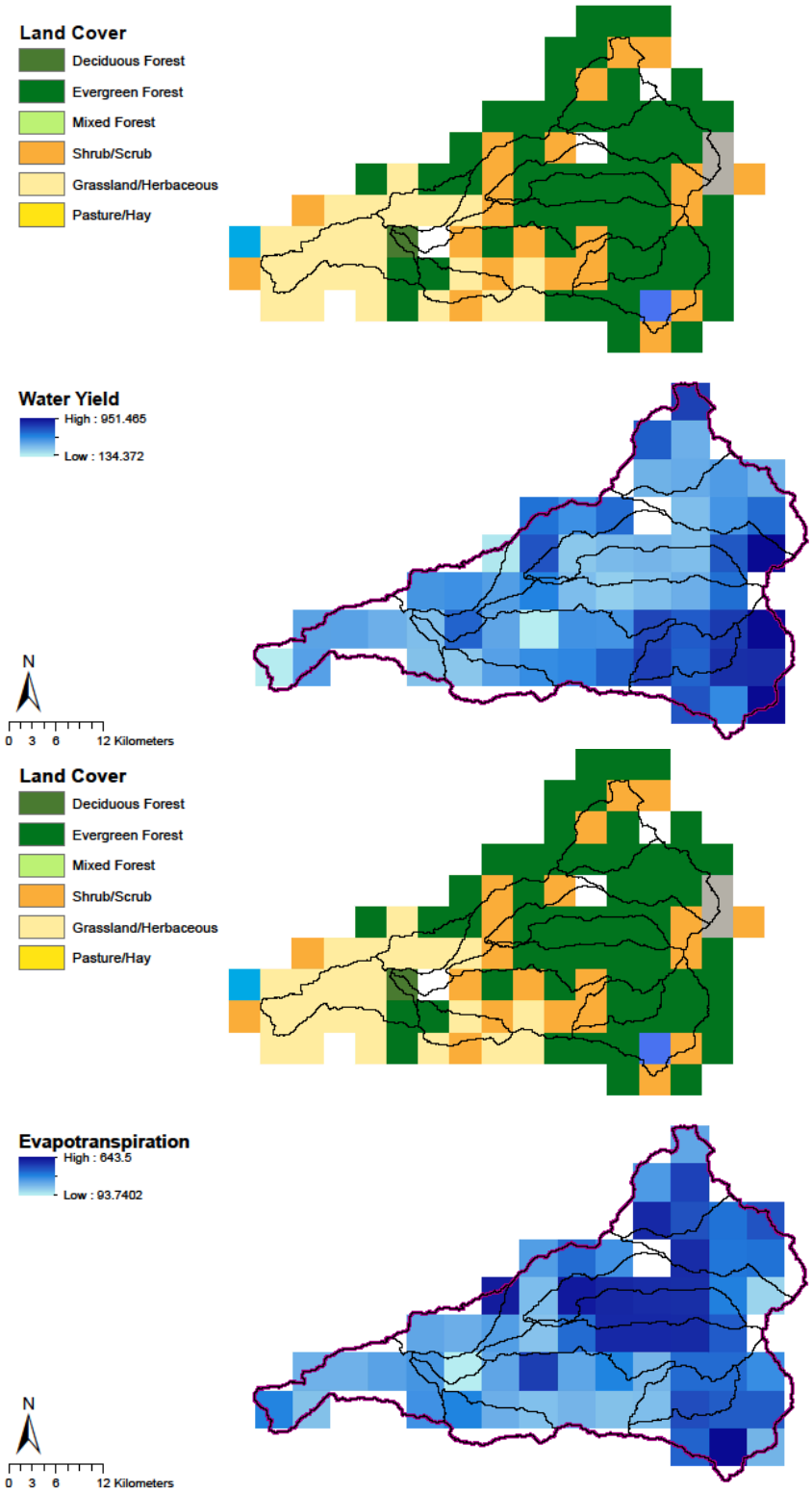

Figure 17. Land cover, water yield, and evapotranspiration 2012, 4000m.

implications for practitioners using the InVEST Hydropower Model. If coarse resolution data for land cover and soil data can be used without significantly changing model outputs, it makes widespread use more feasible in places where high-resolution data is non-existent. If the model is driven by precipitation, evapotranspiration, root depth, and evapotranspiration coefficients, users will require considerably more hydrological and applied climatological expertise in order to accurately parameterize the model.

This study utilized 4km precipitation input data. A study conducted by Berne et al. (2004) found that the appropriate resolution of precipitation data for urban hydrology applications requiring a high degree accuracy in Mediterranean climates was a minimum of 5.2 km. The need for accuracy in ecosystem service assessments coupled with similar climate regimes justified the use of 4km precipitation data in this study. This coarse precipitation data differs greatly from the higher resolution evapotranspiration dataset (800m). Monthly precipitation data was required in order to create “water year” annual precipitation scenarios. Monthly 800m resolution data is not freely available from PRISM Climate Group (2014). Perhaps if precipitation resolution had matched evapotranspiration data results would have been different.

InVEST Model Limitations

InVEST assumes that all precipitation not lost through evapotranspiration will arrive at the watershed outlet. The model does not take into account water diverted by water infrastructure. Deductions from the water balance equation are those allocated to human consumptive uses such as agricultural, industrial, and residential. The model ignores surface-groundwater interactions and does not account for seasonal or sub-

seasonal variability (Tallis et al. 2014). There is a significant amount of water infrastructure present in BCW, even for a rural watershed. Some of these facilities include Coleman National Fish Hatchery, Volta Powerhouse, Eagle Canyon Dam, South Dam, Inskip Dam, Coleman Dam, North Fork Battle Creek Reservoir, and McCumber Reservoir (Figure 18). Initial modeling attempts ignoring the presence of water

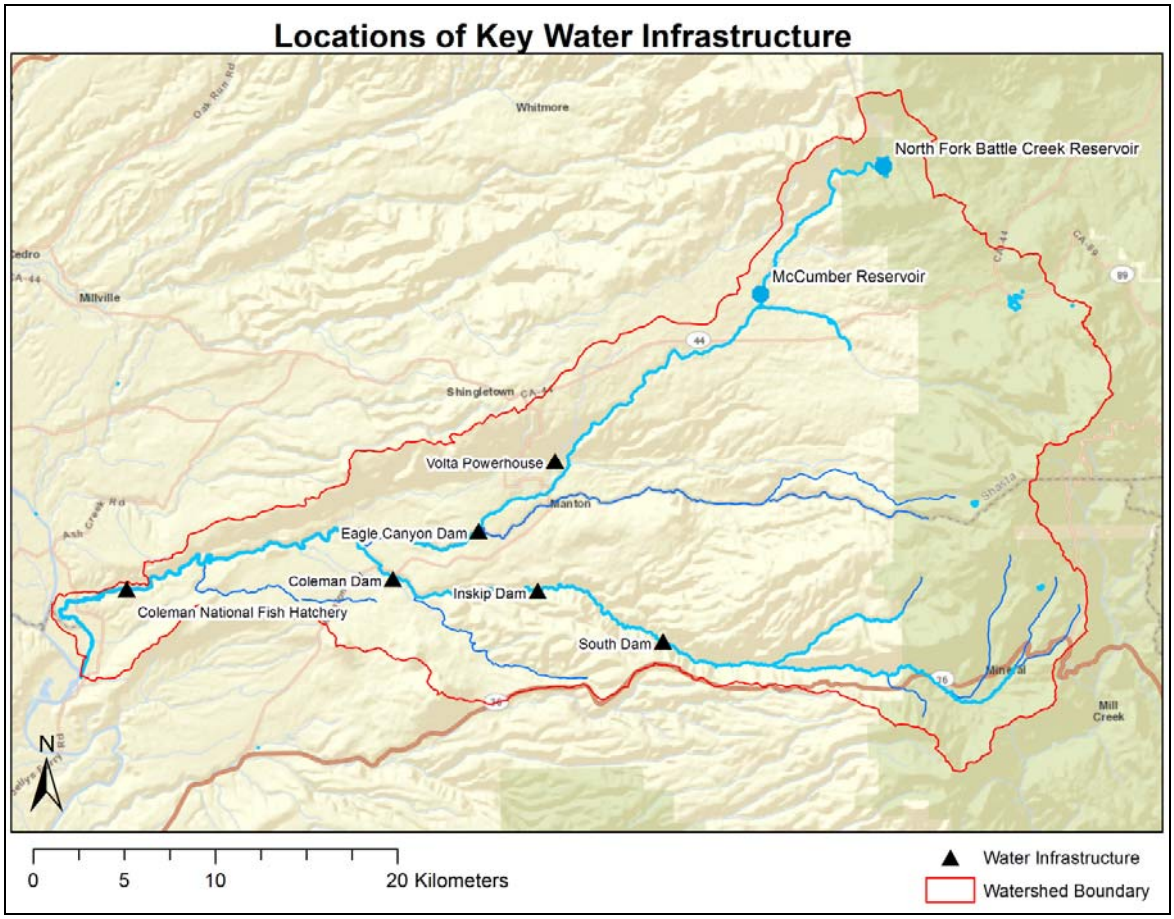

Figure 18. Water infrastructure locations.

infrastructure led to the InVEST Hydropower model over reporting by 118% when compared to USGS runoff observations. After North Fork Battle Creek Reservoir, and McCumber Reservoir were incorporated into the LULC raster and the InVEST water

balance equation, results as documented in this study were much closer to observed runoff. Volta Powerhouse, Eagle Canyon Dam, South Dam, Inskip Dam, and Coleman Dam were omitted from water balance considerations due to the lack of hydrologic data on these structures and their impact within the watershed

A report on the water balance in BCW (Freeman 2009) found that the catchment had a 40% runoff recovery rate. The report postulated that approximately 250 cfs consistently leaks out of the Battle Creek basin and enters the Sacramento River in the form of sub-surface flow. The InVEST model developers acknowledge limitations that ignoring surface-groundwater interactions may cause errors in areas of karst geology (Tallis et al. 2014). While BCW is not located within a karst landscape, the unique volcanic geology of the southern Cascade Mountains may generate similar errors since lava tubes and fractured basalt are known throughout the region. Never the less, the accurate results of this analysis are promising for future researchers planning to conduct InVEST analyses in southern Cascade landscapes. These findings are promising to InVEST developers and supporters of the ecological production function methodology. Perhaps InVEST is generalizable enough to be applied anywhere globally, yet able to be tailored to local conditions to produce accurate results (Nelson and Daily 2010).

InVEST has many temporal limitations. The model runs on an annual time step. The InVEST procedures (Tallis et al. 2014) encourage validation of hydrological model outputs against observed data (e.g., stream gauges). Stream gauges and water data reports function on a ‘water year’. The USGS defines a water year as a 12-month period from October 1st to September 30th of the following year. Users should consider their regions definition of a water year when validating model results. The use of annual

calendar data can be a limitation. This study circumvented this by manipulating the model inputs. Annual monthly precipitation was processed into water years allowing for direct comparison to published USGS water data reports on annual surface runoff. Researchers should consider their model approach to allow for a direct comparison during the validation stage of their modeling efforts. The calibration step is understated in the InVEST Manual but a crucial step towards producing accurate results. Before using a water year calibration the percent error of model results when compared to observed runoff was an average under-prediction of 64% for 30yr. normal precipitation and 44% for actualized precipitation across all scenarios. After calibration the average percent error was under-predicting approximately 0.71% and 0.82% respectively. It is assuring to see the model produce more conservative results as smaller under-prediction than a larger over-prediction. It is also interesting to note that calibration created a similar margin for error between 30yr. normal and actualized precipitation. This is more evidence of the flexibility of InVEST. Regardless of which precipitation inputs are used, after calibration the results are similar from one data input to the next and closely resemble observed field values. The original inception of InVEST called for Tier 2 models which could be run at daily time steps (Ruckelshaus et al. 2013). More robust Tier 2 and Tier 3 models are currently under development capable of capturing daily to monthly time steps (Tallis et al. 2014).

Some challenges during the course of the research required creating water year precipitation inputs and processing LULC rasters to account for water infrastructure classes. In addition, while the hydrologic component in InVEST is simplified compared to more complex models such as SWAT or VIC (Vigertsol and Aukema 2011), users

would benefit from consulting with a local hydrologist to aid in model parameterization, expediting analysis, deciphering model outputs, and validating the model against observed data.

The InVEST manual states that the model is driven by parameter values (i.e., seasonality constant, evapotranspiration coefficients, and root depth) rather than the individual hydrologic processes occurring in the catchment (Tallis et al. 2013). Plant root depth is still derived from the exhaustive review by Canadell et al. (1996) which provides maximum rooting depths of various vegetation types at a global scale. This review concluded that rooting depths among similar ecoregions and species are more consistent than previously believed. Users should be able to derive root depth values from this review with a high degree of confidence. Future investigation regarding accurate model parameterization should focus on the derivation of evapotranspiration coefficients. Evapotranspiration coefficients are difficult to derive, however, an evapotranspiration calculator has been made available on the Natural Capital Project (2014) website. Future users should consult with a hydrologist or applied climatologist to derive evapotranspiration coefficients appropriate to their study area.

CHAPTER VI

CONCLUSION

A review from Seppelt et al. (2011) determined that only 18% of 153 publications on ecosystem services reviewed validated their results against observed data, and only one third of the studies had any sound basis for their results. This coupled with suggestions for increasing clarity and honestly reporting uncertainty in Ruckelshaus et al. (2013) prompted this study to focus on transparency and accuracy.

InVEST has already been used to aid decision-making in a variety of geographic contexts. This study and accuracy of model results are a testament to the generalizability of the InVEST Hydropower model. This study strove for transparency and model validation against observed field results was a primary objective. Model users should attempt to conduct assessments of hydrologic services in catchments with accurate stream gauge data. Without model calibration based on stream gauge data, results will not be as accurate. However, there are many places in which an ecosystem service assessment is necessary, but stream gauges or other instrumentation may not be available for the validation of model results. If model users were conducting an InVEST assessment in a similar ecoregion to the southern Cascades and stream gauges for validation were unavailable, based on the results of this study, users could expect an average under-prediction of 64% with Normal precipitation inputs and 44% with annual actualized precipitation inputs. Studies should attempt to more honestly explain the

uncertainty of their findings to determine if more trends emerge. In addition to validation, the type of input data should be considered. Precipitation normals (30 year averages) result in a higher percentage of error than actualized annual precipitation. Spatial resolution of input data also influences model results. Generally, as spatial resolution decreases, so does the predicted water yield volume.

The biophysical assessment and mapping of ecosystem services is crucial to our understanding of their importance. InVEST is a widely used tool for mapping the distribution of these services. InVEST is primarily used to determine the trade-offs certain management strategies will have on ecosystem services. As the effects of climate change continue to enter public discussion, decision-makers must determine how large magnitude disturbances (e.g. fires, floods, earthquakes) influence ecosystem service provision. To determine how changes in land cover influence water yield, this study controlled the annual variation in precipitation by using normal precipitation data. Results show that the InVEST Hydropower model was unable to detect a significant change in water yield due to the abrupt conversion of land cover from the Ponderosa Fire when compared to water yield from 1992, 2001, 2006, 2012pre-fire, and 2012post-fire. InVEST and fire is a novel concept. InVEST could be used to create hypothetical burn blocks for the purpose of increasing water yield to combat drought.

When InVEST available water results were compared to observations from a USGS stream gauge at the mouth of the catchment, InVEST predictions closely resembled observed annual runoff values. These values were even more consistent after calibration. This study sought to determine if there is an optimum resolution for input data in order to produce accurate results. Results show that resolution may not necessarily

be the driving factor in predicting water yield. The aggregation of land cover types amongst the sub-watersheds is more crucial for water yield predictions. These findings indicate that other parameters (e.g., precipitation, evapotranspiration, evapotranspiration coefficients, and root depth) are more important for the determination of water yield than the resolution of soil and land cover data.

Above all else this study attempted to validate its results and strive for transparency. An element lacking in many ecosystem service studies (Seppelt et al. 2011). Findings will help future InVEST users more accurately parameterize their models and think critically about the appropriate study area, data inputs, and precision in results.

This study is evidence of the efficacy of land cover based proxy mapping of ecosystem services. Land cover based proxy studies are often criticized for being imprecise, stating that the “fit of proxy data to observed data may be poor” (Eigenbrod et al. 2010). The use of primary data based on field sampling will produce the most accurate assessments of ecosystem services. However, primary data is expensive and often unavailable. This study shows that predicted values closely resemble observed values. Observed vs. predicted values can become more consistent with rigorous model calibration and parameterization.

While conducted entirely from publicly available data this study provided accurate ecosystem service information on a watershed which, other than the Sacramento River, can maintain breeding populations of Steelhead and provides habitat for all seasonal runs of Chinook Salmon. The catchment is also a major tributary of the Sacramento River Basin which provides water to residents of Northern and Southern California. The transparency of this study and the detailed description of the modeling

approach will aid in the reproducibility and generalizability of future studies, allowing for a widespread assessment of ecosystem services so that they may sustainably managed.

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APPENDIX A

Table A1. NLCD land cover classification legend.

Code	Name	Description
11	Open Water	Areas of open water, generally with less than 25% cover of vegetation or soil.
21	Developed Open Space	Areas with a mixture of some constructed materials, but mostly vegetation in the form of lawn grasses. Impervious surfaces account for less than 20% of total cover. These areas most commonly include large-lot single-family housing units, parks, golf courses, and vegetation planted in developed settings for recreation, erosion control, or aesthetic purposes.
22	Developed Low Intensity	Areas with a mixture of constructed materials and vegetation. Impervious surfaces account for 20% to 49% percent of total cover. These areas most commonly include single-family housing units.
23	Developed Medium Intensity	Areas with a mixture of constructed materials and vegetation. Impervious surfaces account for 50% to 79% of the total cover. These areas most commonly include single-family housing units.
31	Barren Land	Areas of bedrock, desert pavement, scarps, talus, slides, volcanic material, glacial debris, sand dunes, strip mines, gravel pits and other accumulations of earthen material. Generally, vegetation accounts for less than 15% of total cover.
41	Deciduous Forest	Areas dominated by trees generally greater than 5 meters tall, and greater than 20% of total vegetation cover. More than 75% of the tree species shed foliage simultaneously in response to seasonal change.
42	Evergreen Forest	Areas dominated by trees generally greater than 5 meters tall, and greater than 20% of total vegetation cover. More than 75% of the tree species maintain their leaves all year. Canopy is never without green foliage.
43	Mixed Forest	Areas dominated by trees generally greater than 5 meters tall, and greater than 20% of total vegetation cover. Neither deciduous nor evergreen species are greater than 75% of total tree cover.
52	Shrub Scrub	Areas dominated by shrubs; less than 5 meters tall with shrub canopy typically greater than 20% of total vegetation. This class includes true shrubs, young trees in an early successional stage or trees stunted from environmental conditions.
71	Grassland Herbaceous	Areas dominated by gramanoid or herbaceous vegetation, generally greater than 80% of total vegetation. These areas are not subject to intensive management such as tilling, but can be utilized for grazing.

Table A1 (continued)

Code	Name	Description
81	Pasture Hay	Areas of grasses, legumes, or grass-legume mixtures planted for livestock grazing or the production of seed or hay crops, typically on a perennial cycle. Pasture/hay vegetation accounts for greater than 20% of total vegetation.
82	Cultivated Crops	Areas used for the production of annual crops, such as corn, soybeans, vegetables, tobacco, and cotton, and also perennial woody crops such as orchards and vineyards. Crop vegetation accounts for greater than 20% of total vegetation. This class also includes all land being actively tilled.
90	Woody Wetlands	Areas where forest or shrubland vegetation accounts for greater than 20% of vegetative cover and the soil or substrate is periodically saturated with or covered with water.
95	Emergent Herbaceous Wetlands	Areas where perennial herbaceous vegetation accounts for greater than 80% of vegetative cover and the soil or substrate is periodically saturated with or covered with water.
96	Fire Scarred	Areas completely burned by fire. Complete loss of understory and forest floor cover.

APPENDIX B

Table B1. Root depth

Code	Name	Root Depth (mm)
11	Open Water	1
21	Developed Open Space	1
22	Developed Low Intensity	1
23	Developed Medium Intensity	1
31	Barren Land	1
41	Deciduous Forest	4300
42	Evergreen Forest	3500
43	Mixed Forest	7300
52	Shrub Scrub	3700
71	Grassland Herbaceous	2100
81	Pasture Hay	2300
82	Cultivated Crops	2500
90	Woody Wetlands	3000
95	Emergent Herbaceous Wetlands	2000
96	Fire Scarred	1

APPENDIX C

Table C1. Percent land cover change by resolution, 1992

	1992	Percent Land Cover						
LULC Code	**Land Use Type**	**30.000**	**100**	**120**	**250**	**500**	**1000**	**4000**
11	Open Water	0.24%	0.23%	0.22%	0.26%	0.25%	0.30%	0.94%
22	Developed Low Intensity	0.08%	0.09%	0.09%	0.09%	0.04%	0.12%	0.00%
23	Developed Medium Intensity	0.04%	0.04%	0.04%	0.02%	0.01%	0.00%	0.00%
31	Barren Land	0.43%	0.41%	0.43%	0.42%	0.31%	0.71%	0.00%
41	Deciduous Forest	4.30%	4.30%	4.30%	4.29%	4.26%	4.58%	4.72%
42	Evergreen Forest	53.77%	53.66%	53.81%	54.07%	53.77%	52.68%	47.17%
43	Mixed Forest	4.93%	4.98%	4.95%	4.96%	4.97%	4.46%	2.83%
52	Shrub Scrub	15.34%	15.30%	15.18%	15.04%	15.44%	16.65%	19.81%
71	Grassland Herbaceous	18.32%	18.44%	18.42%	18.30%	18.34%	17.95%	18.87%
81	Pasture Hay	1.97%	1.99%	1.98%	1.94%	1.95%	2.20%	3.77%
82	Cultivated Crops	0.58%	0.58%	0.58%	0.61%	0.61%	0.24%	0.00%
90	Woody Wetlands	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%
95	Emergent Herbaceous Wetlands	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%
96	Fire Scarred	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%
98	McCumber Reservoir	0.00%	0.00%	0.00%	0.00%	0.01%	0.06%	0.94%
99	North Fork Battle Creek Reservoir	0.00%	0.00%	0.00%	0.00%	0.01%	0.06%	0.94%

APPENDIX D

Table D1. Percent land cover change by resolution, 2001

	2001	Percent Land Cover						
LULC Code	**Land Use Type**	**30**	**100**	**120**	**250**	**500**	**1000**	**4000**
11	Open Water	0.24%	0.35%	0.25%	0.23%	0.22%	0.24%	0.00%
21	Developed Open Space	1.55%	2.19%	1.50%	1.39%	1.67%	1.19%	0.95%
22	Developed Low Intensity	0.11%	0.16%	0.10%	0.11%	0.15%	0.00%	0.00%
23	Developed Medium Intensity	0.01%	0.02%	0.01%	0.01%	0.01%	0.00%	0.00%
24	Developed High Intensity	0.00%	0.00%	0.00%	0.00%	0.01%	0.00%	0.00%
31	Barren Land	0.81%	1.18%	0.81%	0.84%	0.95%	0.89%	1.90%
41	Deciduous Forest	1.15%	1.65%	1.14%	1.16%	1.12%	1.24%	0.95%
42	Evergreen Forest	55.41%	79.80%	55.36%	55.46%	54.95%	55.96%	54.29%
43	Mixed Forest	0.10%	0.13%	0.09%	0.08%	0.06%	0.00%	0.00%
52	Shrub Scrub	19.61%	28.29%	19.67%	19.65%	20.21%	19.74%	17.14%
71	Grassland Herbaceous	18.82%	27.07%	18.87%	18.74%	18.53%	18.26%	20.95%
81	Pasture Hay	0.79%	1.15%	0.80%	0.82%	0.78%	1.13%	0.00%
82	Cultivated Crops	0.61%	0.87%	0.61%	0.63%	0.65%	0.65%	0.00%
90	Woody Wetlands	0.33%	0.47%	0.31%	0.39%	0.30%	0.18%	0.95%
95	Emergent Herbaceous Wetlands	0.47%	0.67%	0.48%	0.49%	0.35%	0.41%	0.95%
96	Fire Scarred	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%
98	McCumber Reservoir	0.00%	0.00%	0.00%	0.00%	0.01%	0.06%	0.95%
99	North Fork Battle Creek Reservoir	0.00%	0.00%	0.00%	0.00%	0.01%	0.06%	0.95%

APPENDIX E

Table E1. Percent land cover change by resolution, 2006

	2006	Percent Land Cover						
LULC Code	**Land Use Type**	**30**	**100**	**120**	**250**	**500**	**1000**	**4000**
11	Open Water	0.25%	0.25%	0.25%	0.24%	0.21%	0.24%	0.00%
21	Developed Open Space	1.55%	1.53%	1.51%	1.40%	1.72%	1.42%	0.95%
22	Developed Low Intensity	0.12%	0.13%	0.11%	0.11%	0.16%	0.00%	0.00%
23	Developed Medium Intensity	0.04%	0.04%	0.04%	0.03%	0.04%	0.00%	0.00%
24	Developed High Intensity	0.04%	0.05%	0.04%	0.04%	0.04%	0.06%	0.00%
31	Barren Land	0.81%	0.82%	0.81%	0.84%	0.93%	0.89%	1.90%
41	Deciduous Forest	1.13%	1.13%	1.12%	1.13%	1.09%	1.19%	0.95%
42	Evergreen Forest	54.10%	54.10%	54.08%	54.21%	53.94%	54.48%	52.38%
43	Mixed Forest	0.09%	0.09%	0.09%	0.08%	0.06%	0.00%	0.00%
52	Shrub Scrub	20.65%	20.70%	20.70%	20.61%	20.98%	20.98%	19.05%
71	Grassland Herbaceous	19.00%	18.97%	19.03%	18.95%	18.68%	18.26%	20.95%
81	Pasture Hay	0.79%	0.80%	0.80%	0.82%	0.78%	1.13%	0.00%
82	Cultivated Crops	0.60%	0.60%	0.60%	0.63%	0.64%	0.65%	0.00%
90	Woody Wetlands	0.33%	0.33%	0.32%	0.38%	0.33%	0.18%	0.95%
95	Emergent Herbaceous Wetlands	0.48%	0.47%	0.49%	0.50%	0.37%	0.41%	0.95%
96	Fire Scarred	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%
98	McCumber Reservoir	0.00%	0.00%	0.00%	0.00%	0.01%	0.06%	0.95%
99	North Fork Battle Creek Reservoir	0.00%	0.00%	0.00%	0.00%	0.01%	0.06%	0.95%

APPENDIX F

Table F1. Percent land cover change by resolution, 2012pre-fire

	2012pre-fire	**Percent Land Cover**						
LULC Code	**Land Use Type**	**30**	**100**	**120**	**250**	**500**	**1000**	**4000**
11	Open Water	0.25%	0.25%	0.25%	0.24%	0.21%	0.24%	0.00%
21	Developed Open Space	1.55%	1.53%	1.51%	1.40%	1.72%	1.42%	0.95%
22	Developed Low Intensity	0.12%	0.13%	0.11%	0.11%	0.16%	0.00%	0.00%
23	Developed Medium Intensity	0.04%	0.04%	0.04%	0.03%	0.04%	0.00%	0.00%
24	Developed High Intensity	0.04%	0.05%	0.04%	0.04%	0.04%	0.06%	0.00%
31	Barren Land	0.81%	0.82%	0.81%	0.84%	0.93%	0.89%	1.90%
41	Deciduous Forest	1.13%	1.13%	1.12%	1.12%	1.09%	1.19%	0.95%
42	Evergreen Forest	52.95%	52.96%	52.93%	53.11%	52.62%	53.23%	51.43%
43	Mixed Forest	0.09%	0.09%	0.09%	0.08%	0.06%	0.00%	0.00%
52	Shrub Scrub	21.79%	21.83%	21.85%	21.74%	22.25%	22.11%	20.00%
71	Grassland Herbaceous	18.97%	18.95%	19.00%	18.90%	18.68%	18.26%	20.95%
81	Pasture Hay	0.79%	0.80%	0.80%	0.82%	0.78%	1.13%	0.00%
82	Cultivated Crops	0.66%	0.66%	0.65%	0.69%	0.68%	0.77%	0.00%
90	Woody Wetlands	0.33%	0.33%	0.32%	0.38%	0.33%	0.18%	0.95%
95	Emergent Herbaceous Wetlands	0.47%	0.46%	0.48%	0.47%	0.37%	0.41%	0.95%
96	Fire Scarred	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%
98	McCumber Reservoir	0.00%	0.00%	0.00%	0.00%	0.01%	0.06%	0.95%
99	North Fork Battle Creek Reservoir	0.00%	0.00%	0.00%	0.00%	0.01%	0.06%	0.95%

APPENDIX G

Table G1. Percent land cover change by resolution, 2012post-fire

	2012post-fire	**Percent Land Cover**						
LULC Code	**Land Use Type**	**30**	**100**	**120**	**250**	**500**	**1000**	**4000**
11	Open Water	0.25%	0.24%	0.25%	0.23%	0.21%	0.24%	0.00%
21	Developed Open Space	1.47%	1.46%	1.42%	1.32%	1.63%	1.36%	0.95%
22	Developed Low Intensity	0.11%	0.12%	0.11%	0.11%	0.13%	0.00%	0.00%
23	Developed Medium Intensity	0.04%	0.04%	0.04%	0.03%	0.04%	0.00%	0.00%
24	Developed High Intensity	0.04%	0.05%	0.04%	0.04%	0.04%	0.06%	0.00%
31	Barren Land	0.81%	0.82%	0.81%	0.84%	0.93%	0.89%	1.90%
41	Deciduous Forest	1.07%	1.07%	1.05%	1.06%	1.02%	1.19%	0.95%
42	Evergreen Forest	49.45%	49.46%	49.46%	49.58%	49.30%	50.09%	48.57%
43	Mixed Forest	0.08%	0.07%	0.07%	0.06%	0.06%	0.00%	0.00%
52	Shrub Scrub	19.72%	19.74%	19.76%	19.70%	20.01%	20.04%	18.10%
71	Grassland Herbaceous	18.41%	18.39%	18.43%	18.34%	18.10%	17.49%	20.00%
81	Pasture Hay	0.79%	0.80%	0.80%	0.82%	0.78%	1.13%	0.00%
82	Cultivated Crops	0.66%	0.66%	0.65%	0.69%	0.68%	0.77%	0.00%
90	Woody Wetlands	0.33%	0.33%	0.32%	0.38%	0.33%	0.18%	0.95%
95	Emergent Herbaceous Wetlands	0.46%	0.45%	0.47%	0.47%	0.37%	0.41%	0.95%
96	Fire Scarred	6.31%	6.30%	6.30%	6.32%	6.34%	6.05%	5.71%
98	McCumber Reservoir	0.00%	0.00%	0.00%	0.00%	0.01%	0.06%	0.95%
99	North Fork Battle Creek Reservoir	0.00%	0.00%	0.00%	0.00%	0.01%	0.06%	0.95%