Arnold Schwarzenegger
*Governor*

# ASSESSING IMPACTS OF RANGELAND MANAGEMENT AND REFORESTATION OF RANGELANDS ON GREENHOUSE GAS EMISSIONS:

# A PILOT STUDY FOR SHASTA COUNTY

PIER PROJECT REPORT

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Public Interest Energy Research Program

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# Preface

The Public Interest Energy Research (PIER) Program supports public interest energy research and development that will help improve the quality of life in California by bringing environmentally safe, affordable, and reliable energy services and products to the marketplace.

The PIER Program, managed by the California Energy Commission (Energy Commission), conducts public interest research, development, and demonstration (RD&D) projects to benefit California's electricity and natural gas ratepayers. The PIER Program strives to conduct the most promising public interest energy research by partnering with RD&D entities, including individuals, businesses, utilities, and public or private research institutions.

PIER funding efforts are focused on the following RD&D program areas:

- Buildings End-Use Energy Efficiency
- Energy-Related Environmental Research
- Energy Systems Integration
- Environmentally Preferred Advanced Generation
- Industrial/Agricultural/Water End-Use Energy Efficiency
- Renewable Energy Technologies
- Transportation

In 2003, the California Energy Commission's Public Interest Energy research (PIER) Program established the **California Climate Change Center** to document climate change research relevant to the states. This Center is a virtual organization with core research activities at Scripps Institution of Oceanography and the University of California, Berkeley, complemented by efforts at other research institutions. Priority research areas defined in PIER's five-year Climate Change Research Plan are: monitoring, analysis, and modeling of climate; analysis of options to reduce greenhouse gas emissions, assessment of physical impacts and of adaptation strategies, and analysis of the economic consequences of both climate change impacts and the efforts designed to reduce emissions.

**The California Climate Change Center Report Series** details ongoing Center-sponsored research. As interim project results, the information contained in these reports may change; authors should be contacted for the most recent project results. By providing ready access to this timely research, the Center seeks to inform the public and expand dissemination of climate change information; thereby leveraging collaborative efforts and increasing the benefits of this research to California's citizens, environment, and economy.

*Assessing Impacts of Rangeland Management and Reforestation of Rangelands on Greenhouse Gas Emissions: A Pilot Study for Shasta County* is the final project report for the Preliminary Economic Analyses of Climate Change Impacts and Adaptation, and GHG

Mitigation project (contract number 500-02-004, MR-006), conducted by Applied Geosolutions, LLC, and the Complex Systems Research Center at the University of New Hampshire.

For more information on the PIER Program, please visit the Energy Commission's website www.energy.ca.gov/pier/ or contract the Energy Commission at (916) 654-5164.

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# Abstract

A pilot study was implemented to quantify impacts of rangeland management and reforestation of rangelands on net trace gas emissions and soil carbon sequestration as part of a study to assess potential carbon supply curves for afforestation of rangelands in Shasta County, California. This study utilizes spatially explicit GIS data on soils, climate, potential forest type, and current rangeland types and forest/rangeland management combined with two soil biogeochemical process models: Denitrification-Decomposition (DNDC) and Forest-DNDC. The study objectives were to (1) assess the impact of reforestation of rangelands in Shasta County on net trace gas emissions and soil carbon sequestration and (2) assess the impact of implementing more intensive grazing of range grasslands on soil carbon and trace gas emissions. Results from the analysis include: (1) predicted baseline carbon dynamics and greenhouse gas (GHG) emissions for existing rangelands in Shasta County, (2) the estimated impact of alternative rangeland management strategies (various grazing intensities) on soil carbon stocks and trace gas emissions, and (3) predicted impact of reforestation on soil carbon dynamics and GHG emissions across Shasta County. A complete GHG balance is compiled for Shasta County for the 50 years following afforestation and compared with baseline model results to map changes in net GHG balance. The study's results indicate that, in general, full accounting adjusted carbon sequestration potential in the county by less than 10 percent. The impact of soil carbon losses and trace gas emissions on the net GHG balance are modeled as percent offset of emissions, relative to carbon sequestered in woody biomass and forest floor, and vary spatially across forest types driven by difference in forest productivity and soil conditions. Impacts of fertilizer application and climate change on afforestation dynamics and effects of afforestation in local hydrology are also examined.

Keywords: Soil biogeochemical modeling, greenhouse gas emissions, rangeland, reforestation

# Executive Summary

## Introduction

Applied Geosolutions, LLC, in collaboration with University of New Hampshire, conducted a scoping study for the California Energy Commission (Energy Commission) to design a framework to assess the impacts of rangeland management and reforestation of rangelands on greenhouse gas (GHG) emissions within Shasta County, California. For this project, the research team employed two existing process-based biogeochemical models—Denitrification-Decomposition (DNDC, a cropland/grassland model) and Forest-DNDC—to simulate vegetation growth, soil carbon dynamics, and trace gas emissions under various land use change and management scenarios in Shasta County, California.

## Purpose

Land use change strategies—including the conversion of rangelands to forestry through afforestation and reforestation—are being considered in California for mitigating GHG emissions from rangelands. These changes in land use can affect both above- and below-ground carbon sequestration impacting carbon stocks and C and N dynamics. This scoping study evaluates the impact of land use change on net changes in carbon stocks and non-$CO_2$ GHG emissions, incorporating assessment of both pre-land use changes (existing rangeland emissions) and post-land use changes (reforestation and changes in above- and below-ground carbon stocks as well as trace gas emissions). This research represents a sub-component of ongoing research to assess carbon supply curves for changes in management of forest and rangelands in Shasta County, California.

## Project Objectives

The objectives of this scoping study are twofold:

- Design a framework to assess the impact of reforestation of rangelands on net trace gas emissions and soil carbon sequestration.
- Apply this framework as a demonstration to assess the impact of afforestation and implementing more intensive grazing of range grasslands on soil carbon and trace gas emissions for Shasta County, California.

This assessment includes an evaluation of short- and long-term benefits in terms of net 100-year global warming potential (GWP).

## Project Outcomes

Results from the Forest-DNDC analysis for Shasta County include the following:

- Predicted baseline carbon dynamics and GHG emissions for existing rangelands in Shasta County.
- Estimated impact of alternative rangeland management strategies (various grazing intensities) on soil carbon stocks and trace gas emissions.

- Predicted impact of reforestation on soil carbon dynamics and GHG emissions across Shasta County.

**Conclusions**

The DNDC model is employed to estimate soil carbon dynamics and trace gas emissions (nitrous oxide, $N_2O$, and methane, $CH_4$) for Blue Oak Woodlands, annual grasslands, and generic shrub classes. The regional patterns of total carbon sequestration indicate that the annual grassland areas in the Southwest region of the county are losing carbon, whereas the other rangeland regions are predominantly sequestering carbon. Annual carbon sequestration of rangeland area is 5,640 metric tons of carbon per year, with all rangelands sequestering on average 29 kilograms of carbon per hectare per year (kgC/ha/yr).

The impact of grazing intensity on soil carbon dynamics depends on site quality (initial soil organic carbon (SOC) content). For sites with higher initial SOC, increased grazing intensity will decrease the rate of carbon sequestration. In general, the DNDC model indicates that in the absence of grazing, rangelands of Shasta County would be sequestering ~50 kg C/ha/yr—an increase of 21 kg C/ha/yr over nominal grazing intensity.

Based on the model results, the research team compiled a complete GHG balance for 50 years following afforestation and compared it with baseline model results to map changes in net GHG balance. To do this, the team compared the magnitude of the carbon sink due to tree biomass and forest floor carbon accumulation with the net carbon source from the cumulative loss in soil carbon and trace gas emissions and baseline rangeland dynamics over the 50-year period. The results indicate that, in general, full accounting adjusted carbon sequestration potential by less than 10%. The impact of soil carbon losses and trace gas emissions (SCTG-E) on the net GHG balance are calculated as the percent offset of emissions relative to carbon sequestered in woody biomass and forest floor (C-Stock):

$$\text{SCTG-Eoffset} = \text{SCTG-E (CO2eq)} \ / \ \text{C-Stock (CO2eq)}$$

SCTG-Eoffset varies spatially across forest types driven by difference in forest productivity and soil conditions. Sites with higher SCTG-Eoffset were characterized by low forest productivity. Over 90% of the sites had SCTG-Eoffset values less than 12%. Areas with high SCTG-E had high SOC content prior to afforestation. Areas with higher SOC typically have larger losses of soil carbon during forest establishment, will oxidize more methane, and will have higher nitrous oxide emissions. The true net GHG benefits are also examined by accounting for not only SCTG-E effects, but also the net GHG balance of the rangelands prior to afforestation. In general, net GHG estimates are lower than C-Stock estimates, due to soil carbon loss during afforestation and removal of the net carbon sink of rangelands.

**Recommendations**

Full greenhouse gas accounting is possible with the development and maturation of process-based models and can be readily assessed using the framework outlined in this

report. Based on initial findings of this scoping study, several additional issues related to afforestation and reforestation of rangelands in Shasta County were identified as warranting further research, including the following:

- Potential benefits of using fertilizer to enhance forest productivity and net carbon sequestration.
- Impacts of climate change on afforestation dynamics.
- Impacts of rangeland afforestation on local hydrology.

**Benefits to California**

This scoping study contributes to ongoing scientific research supported by the California Energy Commission PIER five-year Climate Change Research Plan by providing an analysis of land use mitigation options for reducing GHG emissions from rangelands in California. The proposed adaptation strategies of aforestation and reforestation of rangelands can be actively adopted and employed by land managers in the state to reduce GHG emissions over a multi-year time frame. In addition, the scoping study identifies critical issues, such as climate change and changes in hydrology associated with land use changes, that can impact GHG emissions, and it addresses a need for further research in these areas.

## 1.0 Background

Several land use change strategies are being considered in California for mitigating greenhouse gas (GHG) emissions through carbon sequestration, including conversion of rangelands to forestry through afforestation and reforestation. These changes in land use affect not only aboveground carbon but also belowground carbon stocks and carbon (C) and nitrogen (N) dynamics. For example, reforestation can increase root biomass and root exudation, which is a major source of dissolved organic carbon (DOC) for fueling a wide range of soil microbial activities, including nitrous oxide ($N_2O$) or methane ($CH_4$) production/consumption. In general, in comparison with the static ecosystems, disturbances occurring during the land use transitions and changes in forest management can usually be favorable for trace gas emissions. Therefore, evaluation of the net impact of changes in forest management and land use change should consider the net changes in not only aboveground carbon stocks, but also belowground carbon storage and non-$CO_2$ greenhouse gas emissions. Quantification of impact of land use change in terms of net global warming potential (GWP) should incorporate assessment of both pre-land use changes (existing rangeland emissions) and post-land use changes (reforestation and changes in above- and belowground carbon stocks, as well as trace gas emissions).

Forest soils, for example, can be a significant source or sink of important greenhouse gases, such as nitric oxide (NO), $N_2O$, $CH_4$, and ammonia ($NH_3$), which may react in the atmosphere to produce airborne particulate matter (PM). However, the production of N-trace gases in soils is a function of both production and consumption microbial processes (e.g., nitrification and denitrification) that occur simultaneously and are highly variable in space and time. The fertility of the forest soil controls the availability of nitrogen and carbon in the soil, and hence affects NO and $N_2O$ emissions. Competition between plants and microbes can become a limiting factor for N-gas production, especially in poor soils (Jones and Richards 1977; Schmidt 1982; Binkley and Hart 1989). Competition of nitrifying organisms with plant uptake and microbial N immobilization has been observed to be responsible for the low rates of net nitrification in the Harvard Forest soils in the United States (Bowden et al. 1991). Some researchers have observed that the presence of plants lowered denitrification by depleting the N pool (Bouwman 1990; Bowden et al. 1991). Many researchers have noted rapid increases in net nitrification following cutting of forests (Likens et al. 1969; Vitousek et al. 1982; Matson and Vitousek 1987; Montagnini and Buschbacher 1988; Fisk and Fahey 1990). Forest age and vitality is also a factor to consider in determining the nutrient requirements of a forest (Prichett 1979). An aggrading forest will readily take up additional N inputs compared to a steady-state forest with slower growth, and therefore lower nutrient requirements (Vitousek and Reiners 1975; Bormann and Likens 1979).

## 2.0 Project Objectives

The overall objective of this project is to combine spatially explicit GIS data on soils, climate, potential forest type, and current rangeland types with our cropland and forest soils biogeochemical process models, Denitrification-Decomposition (DNDC) and Forest-DNDC, to simulate the spatial and temporal distribution of trace gas emissions and soil carbon dynamics for Shasta County in California. Denitrification-Decomposition is a process-oriented computer simulation model based on the biogeochemical concepts for predicting soil C and N biogeochemistry. It has been integrated with a forest physiology model, PnET, to serve forest biogeochemistry studies.

The specific objectives for this pilot project were to:

- Construct a geographic information systems (GIS) database for Shasta County, including data on:
  - Soils (U.S. Department of Agriculture's Natural Resources Conservation Service (NRCS), Soil Survey Geographic (SSURGO), and State Soil Geographic (STATSGO));
  - Climate (DAYMET and California Irrigation Management Information System (CIMIS) databases),
  - Topography, land cover, and land use based on Wildlife Habitat Relationship (WHR) classification system (California Department of Forestry and Fire Protection (CDF), Fire and Resource Assessment Program (FRAP), and California Land Cover Mapping and Monitoring Program (LCMMP), coordinate with Winrock databases), and
  - N-deposition rates.
- Adjust DNDC and Forest-DNDC for forest and rangeland ecosystems in California. To do this, the project team
  - collected data on forest growth, litter production, and litter quality;
  - collected data on rangeland forage production, litter production, and quality; and
  - adjusted physiological and phenological parameters embedded in DNDC and Forest-DNDC with the collected forest or rangeland data.
- Use DNDC to predict baseline carbon dynamics and GHG emissions for existing rangelands in Shasta County.
- Use DNDC to predict the impact of alternative rangeland management strategies (various grazing intensities) on soil carbon stocks and trace gas emissions.
- Use Forest-DNDC to predict the impact of reforestation on soil carbon dynamics and GHG emissions across Shasta County.

## 3.0 Development of GIS Databases

### 3.1. Basemap

The Denitrification-Decomposition process-oriented computer simulation model was developed based on biogeochemical concepts for predicting soil biogeochemistry given study area characteristics, such as landcover and cropping activities, soil characteristics, and climate drivers. The DNDC model operates on a set of spatial modeling units dividing the study area into unique land use, soil, and climate landscape characteristics driving the biogeochemical processing algorithms. The research team chose to define the spatial modeling units for DNDC in Shasta County based on the unique combination of rangeland, candidate forest types for afforestation, and soil types within the county. It was assumed that the derived spatial modeling units would be of sufficient size that climate would not vary significantly within these extents. The rangeland distribution coverage was derived from the CDF-FRAP multi-source land-cover map using the WHR landcover classification system. Candidate forest types for afforestation were provided by Winrock International. The soils database was derived from the USDA's Natural Resources Conservation Service STATSGO database. Derivation of the rangeland, candidate forest types and soils datasets are described in more details in the following sections.

A polygon overlay of the rangeland, candidate forest types, and soils coverages was performed to create the DNDC basemap coverage. The resultant geographic intersection of the three datasets defined the DNDC spatial mapping units for the study area, with each polygon in the soil-rangeland-forest overlay coverage assigned a unique DNDC identification number (ID). Polygons smaller than 4 hectares were not included in the analysis and were removed from the basemap coverage by "dissolving" the polygon into an adjacent rangeland polygon, or eliminating the polygon if no adjacent rangeland feature was present. Rangeland, candidate forest type, soils, and climate characteristics needed for input to DNDC were identified for each of the DNDC polygons in the DNDC basemap coverage. Statistics and input files were created for each of the DNDC input layers based on this basemap grid.

### 3.2. Climate Database

The Daymet database was used to provide a surface of precipitation, temperature, and radiation for each reference year, for input into the DNDC model. Twenty-four years of daily climate data (1980–2003) were downloaded from the Daymet database for the analysis, including three reference climate years to simulate an "average/nominal" year (1996) and two extremes years (wet/late spring (1983) and dry/early spring (1997)).

The Daymet database includes daily surfaces of temperature, precipitation, humidity, and radiation over large regions of complex terrain. Daymet was developed at the University of Montana, Numerical Terradynamic Simulation Group (NTSG), to model fine resolution, daily meteorological and climatological data needed to model plant growth dynamics. Daymet provides a 24-year daily dataset (1980–2003) of temperature, precipitation, humidity, and radiation as a continuous surface at a 1 kilometer (km) resolution. A wide range of summary and point daily data over the conterminous United States is also available from the Daymet website (www.daymet.org).

Climate data was downloaded from the Daymet website using an automated mining routine for each of the DNDC basemap polygon centroids. Where no data was available at the centroid location, a 0.1 degree spiral search was initiated to find available data closest to the polygon centroid center. Daily data from 1980–2003 for minimum and maximum temperature, average precipitation, and solar radiation was downloaded for each DNDC mapping unit for input into the DNDC model.

### 3.3. Soils Database

Soil data on carbon content, pH, bulk density, soil texture, and percent stone required for input to the DNDC model were complied using the STATSGO database. The 1:250,000 STATSGO dataset is generated from satellite image (Landsat) interpretation and generalization of the more-detailed SSURGO database, utilizing field surveys and aerial photograph interpretation. The database is designed to be used for broad planning and management uses covering state, regional, and multi-state areas. The STATSGO attribute database gives the proportionate extent of the component soils and their properties for each map unit and includes over 25 physical and chemical soil properties, interpretations, and productivity.

The STATSGO dataset was used to obtain the minimum and maximum ranges for the soil attributes required by DNDC (pH, clay content, bulk density, soil organic matter, and percent stone measuring between 2 millimeters (mm) to 3 inches) aggregated to the DNDC basemap units. Figure 1 shows minimum and maximum clay content for soils in Shasta County from the STATSGO database.

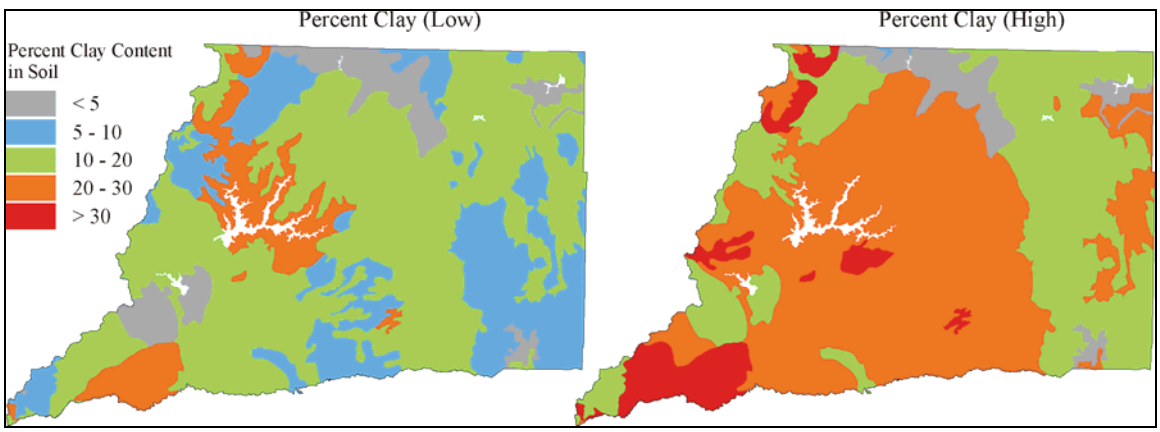


**Figure 1. Minimum and maximum clay content in Shasta County. White areas represent water bodies, which were not modeled in this analysis.**

The STATSGO database is arranged in a multi-layer format, where each polygon (referred to as "map unit" by STATSGO) can have multiple components and each component can have multiple layers. A soil component is a set of properties that are used to describe a certain soil type that exists. The percent areas that each soil component occupies within the STATSGO polygons are provided ("*comparea*" variable); however there is no information provided as to the actual spatial distribution of each component within the polygons.

It is evident that each STATSGO polygon has the potential for dozens of scenarios, based on multiple soil components and layers; however the DNDC model requires a single set of input ranges for the soil input variables. In order to take advantage of the detail that is available in the STATSGO database, an area-weighted approach was used to calculate DNDC soil input variables for each soil polygon. Because DNDC requires information for the surface layer only, researchers considered only STATSGO data that represented the top soil layer. Soil components representing 10% or more of the surface layer were area-weighted for each soil. The soil attribute information was then assigned by soil type to the basemap polygons for input into the DNDC program.

### 3.4. Rangeland and Afforestation Forest Types Database

Rangeland distribution in Shasta County was derived from the CDF-FRAP Multi-source land-cover map using the WHR landcover classification system. The WHR classes were further classified by Winrock International into rangeland, forest, and non-vegetated broad classes as per the "Module 3a: Carbon Supply Curves for California: Rangelands" report to the California Energy Commission and EPRI (Brown et al. 2003). The CDF-FRAP grid was initially clipped to the Shasta County extents. The WHR classes denoted as rangelands were then reselected from this grid to define the extent and distribution of rangelands in Shasta County.

A gridded database of likely candidate forest types for afforestation on rangeland was provided by Winrock International for analysis. The candidate forest type database also followed the WHR landcover classification system. This grid was clipped to the Shasta County extent and then masked to the rangeland extents described in the previous paragraph to obtain a mapping of candidate forest types over existing rangelands.

The defined rangeland and afforestation classes were recoded into 7 DNDC vegetative modeling classes for input into the model. Tables 1a and 1b show the recoding of the CDF-FRAP WHR classes into the final DNDC vegetation classes.

Figure 2 shows the distribution of rangelands and candidate forest types for rangelands in Shasta County. DNDC rangeland and candidate forest type designations were identified for each of the polygons in the DNDC basemap coverage to build the input files for the rangeland and forest DNDC analysis.

To characterize the variability in soil, climate, topography, and land cover types in the county the research team extracted a sub-sampling of the candidate forest types for this analysis. A sampling point coverage was created, containing 824 equidistance points positioned at intervals of 3.5 km across Shasta County. A total of 191 candidate forest-type polygons coinciding with the sampling point coverage were selected for the DNDC grazing management and forest analyses; the remaining sampling points did not fall on candidate forest type polygons and were therefore excluded from the analysis. Figure 3 shows the sampling point locations.

**Table 1a. CDF-FRAP WHR classification for rangelands and candidate forest type with DNDC recoded classification**

| CDF-FRAP WHR Class | DNDC Recode Class |
|---|---|
| Valley Foothill Riparian | Eliminate |
| Wet Meadow | Eliminate |
| Annual Grassland | Annual Grass |
| Blue Oak Woodland | 40% Oak, 60% Annual Grass |
| Blue Oak-Foothill Pine | 40% Oak, 60% Annual Grass |
| Valley Oak Woodland | 40% Oak, 60% Annual Grass |
| Eastside Pine | Pine |
| Juniper | Juniper |
| Klamath Mixed Conifer | Fir |
| White Fir | Fir |
| Montane Hardwood | Oak |
| Sierran Mixed Conifer | 10% Oak, 36% Fir, 36% Pine, and 18% Cedar |
| Bitterbrush | Shrub |
| Chamise-Redshank Chaparral | Shrub |
| Low Sage | Shrub |
| Mixed Chaparral | Shrub |
| Montane Chaparral | Shrub |
| Sagebrush | Shrub |
| Unknown Shrub Type | Shrub |

**Table 1b. DNDC rangeland and candidate forest type classes**

| DNDC ID | Rangeland and/or Forest Type Class |
|---|---|
| 1 | Annual Grass |
| 2 | Oak |
| 3 | Pine |
| 4 | Juniper |
| 5 | Fir |
| 9 | Cedar |
| 10 | Shrub |

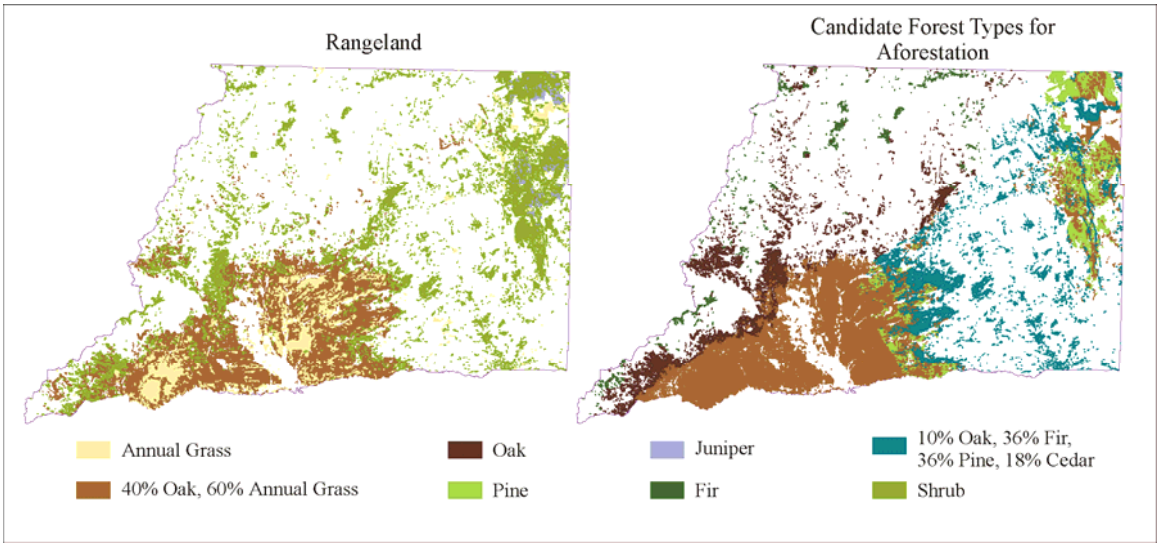


**Figure 2. Distribution of rangeland and candidate forest types for Shasta County**

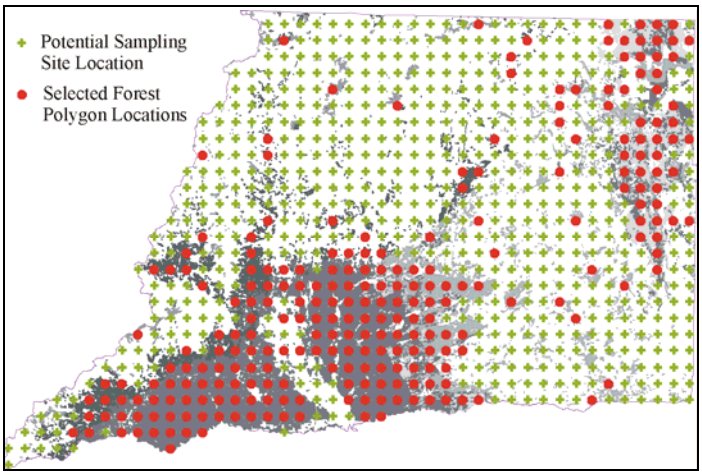


**Figure 3. Location of potential sampling sites and selected candidate forest polygon locations for forest DNDC analysis**

Figure 4A provides a histogram showing the distribution of soils variables; comparing the sampling subset to the full coverage of candidate forest polygons in Shasta County. This comparison shows that the soil sampling subset of the candidate forest types is representative of the soil conditions throughout Shasta County. In addition, given the equal spacing of the sampling grid, the research team feels that the climate variability across the county is captured by this subset.

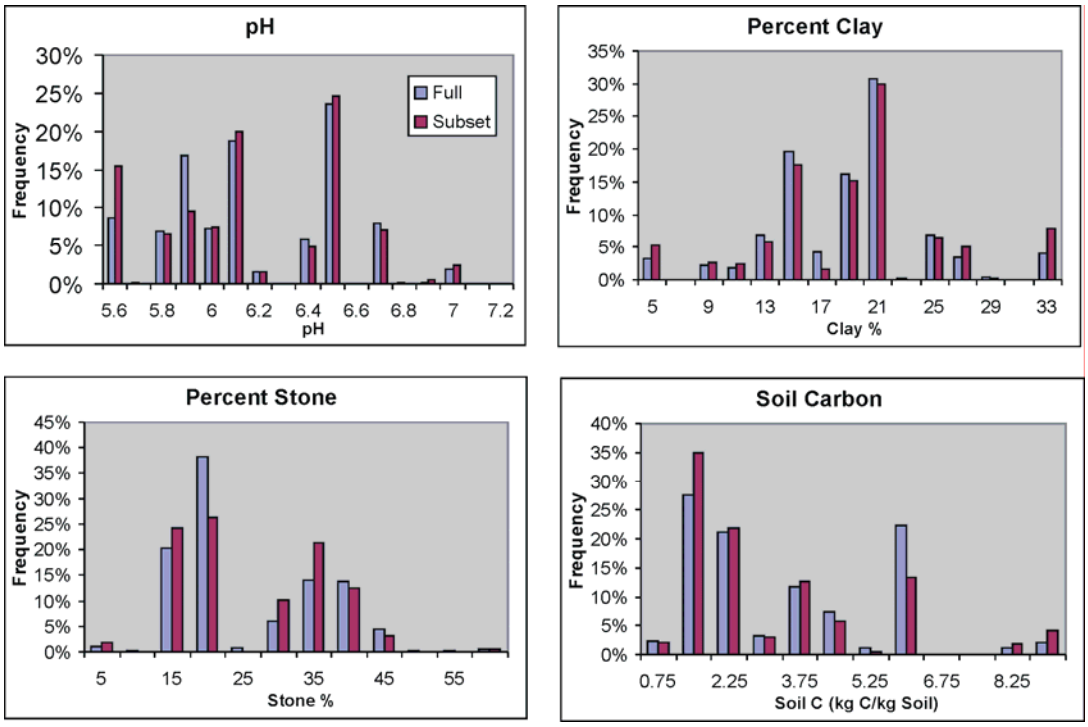


**Figure 4. Comparison of soil variables for all candidate forest polygons in Shasta County (full) and point sample subset of polygons for DNDC forest analysis (subset)**

## 4.0 Model Adjustments

This project used two existing process-based biogeochemical models to simulate vegetation growth: soil carbon dynamics, and trace gas emissions. The biogeochemical models have been used and tested across a wide range of land use, climate, and soil conditions. This chapter briefly describes each model and the assumptions/adjustments made to the model inputs to capture the vegetation and management conditions found in Shasta County.

### 4.1. DNDC and Forest-DNDC Models

To predict soil carbon dynamics, methane oxidation, and emissions of $N_2O$ and NO from both rangeland and forest soils, the research team used two existing models: Forest-DNDC and DNDC. Both models follow a similar framework (see Figure 5 for the Forest-DNDC example) and share similar sub-models. This section outlines the Forest-DNDC model.

The Forest-DNDC model was constructed by integrating PnET, a forest physiological model developed by Aber et al. (1996), with DNDC, a soil biogeochemical model (Li et al. 1992, 1994). The integration created a new modeling framework to meet some gaps existing in most forest models regarding the linkage between forest and soil processes. PnET simulates forest physiological processes but with weak components in soil biogeochemistry. DNDC possesses detailed soil processes but needs vegetation drivers. The integration of PnET with DNDC established a linkage to allow PnET and DNDC to exchange information at a daily time step. PnET predicts forest growth, litter production, water, and N demands; and DNDC receives litter incorporation, tracks its turnover, and quantifies water and N availability in the soils.

The integration substantially enhanced this study's modeling capacity for predicting C and N cycles in forest ecosystems. Based on the success, several new features specially related to anaerobic biogeochemistry were incorporated in the model for wetland simulations. The new features included: (1) division of soil profile by ground water table fluctuation, (2) soil Eh[1] evolution, (3) SOC turnover under anaerobic conditions, (4) methane production/consumption, and (5) wetland management practices (e.g., drainage, restoration). Three approaches were built in the forest version of DNDC to utilize water table data to drive biogeochemical simulations. The water table data can be obtained from field observations, empirical model simulations, or hydrological model simulations.

The Forest-DNDC model consists of two components. The first component—consisting of soil climate, forest growth, and decomposition sub-models—predicts forest carbon production, carbon allocation, litter incorporation, soil temperature/moisture/Eh profiles, and soil organic matter decomposition. The second component—consisting of nitrification, denitrification, and fermentation sub-models—simulates C and N transformation driven by the soil microbial activities, which are responsible for production and consumption of $CH_4$, nitrous $N_2O$, NO, and other trace gases in the soils.

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[1] *Eh* refers to soil redux potential.

The forest growth sub-model simulates photosynthesis, respiration, C allocation, and litter production driven by solar radiation, temperature, and canopy N content. The soil process sub-models (e.g., decomposition, nitrification, denitrification, fermentation) predicts C and N dynamics in forest soils including the forest floor and the mineral soil profile. Major management practices, such as deforestation, reforestation, thinning, burning, drainage, wetland restoration, fertilization, and others, have been parameterized and linked to the plant-soil processes. Equipped with the functions, Forest-DNDC is capable of simulating C and N cycles for both wetland and upland forest ecosystems. With the kinetic framework as well as its interacting functions, the Forest-DNDC model links ecological drivers to trace gas emissions. A description of the model can be found in Li et al. (2000). Tests for validating the model are published in a companion paper (Stange et al. 2000).

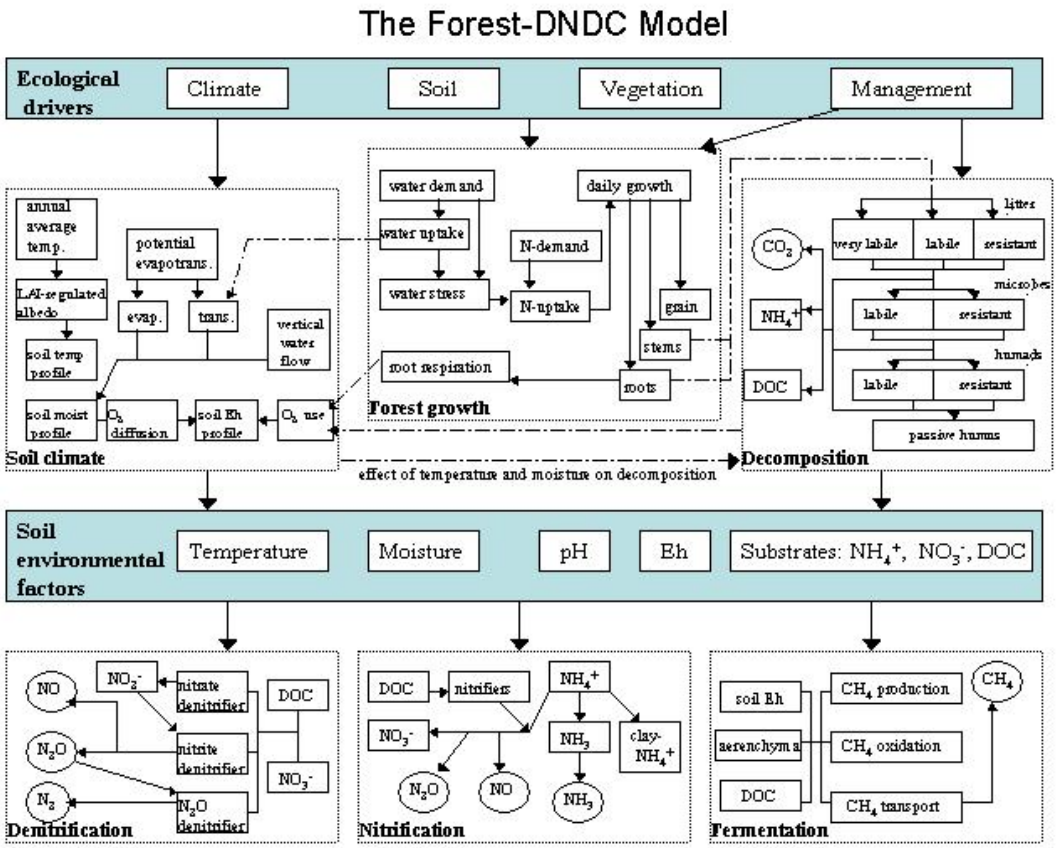


**Figure 5. Structure of the Forest-DNDC model**

The Forest-DNDC model simulates forest N uptake with the algorithms adopted from the PnET model (Aber et al. 1995, 1996) in conjunction with other functions. In the model, soil fertility is related to litter quantity and quality. The forest growth sub-model predicts litter production and litter C/N ratio. After litter fall, the decomposition sub-model allocates the fresh litter into the very labile, labile, and resistant litter pools. The decomposition rate of each organic matter pool is calculated separately based on its specific decomposition rate, temperature, and soil moisture (Li et al. 1992). If the fresh

litter has a low C/N ratio, larger fractions of the litter will be partitioned into the more labile pools. This will lead to a faster bulk decomposition rate. The laboratory studies by Prescott (1996) clearly revealed the effect of litter quality (C/N ratio) on organic matter decomposition rates. Since the quantity and quality of the litter vary from forest type to forest type (Table 3), the soil fertility in different types of forests can be different even they share similar climate conditions. The significant differences in soil organic matter (SOM) contents as well as in NO and $N_2O$ emissions between two adjacent forest stands (spruce and beech) have been observed by the researchers (Rothe 1998; Butterbach-Bahl et al. 1997; Papen and Butterbach-Bahl 1999; Gasche and Papen 1999) in the Höglwald Forest in Germany, although the two stands have similar climate, soil texture and forest age (see Stange et al. 2000).

Pre-forest land-use history also affects soil fertility. Magill et al. (1997) suggest that land-use history could be critical in determining current N status of soils and forests. Land-use history is recorded in SOM quantity and quality. Rather than distinguish moder, mor, or mull,[2] this study characterized SOM in forest soils based on amounts and C/N ratios of the organic matter (see details in Li et al. 1992, 1994). Initial pool sizes of organic matter were required as input parameters. Initial pools of SOM were set at the beginning of the afforestation process, based on 15-year simulations for each rangeland polygon.

Table 2 provides a list of the input parameters for Forest-DNDC. For this analysis, researchers simulated afforestation using our generic model for Pine, Fir and Oak (see Table 3 for our default forest parameters for these and other forest types). The research team obtained the Hessl et al. (2001) database of ecophysiological parameters for Pacific Northwest Trees. This database was created based on a detailed literature review. The database was developed to provide a centralized database of input parameters for ecosystem models (like Forest-DNDC).

### 4.2. Model Assumptions and Testing

For this project the research team created input parameters for California Blue Oak and a generic shrub class in order to use the plant growth sub-model, which is embedded in DNDC, to track dynamics of canopy and fine roots for woody crops (Blue Oak and Shrub classes are modeled as woody crops). California Blue Oak is an important tree species in California rangelands. The team parameterized the Blue Oak physiology/phenology library file based on leaf area index (LAI), lead mass density, annual foliar, twig and acorn production, and N concentration from three sources (Karlik and LcKay 2002; Jackson et al. 1990, and Knops et al. 1996). While there are a wide range of shrubs in California, the research team developed a generic shrub physiology/phenology library file based on biomass data (Martin et al. 1981) collected for big sagebrush (*Artemisia tridentata*), antelope bitterbrush (*Purshia tridentate*), snowbrush ceanothus (*Ceanothus velutinus*), and Greenleaf Manzanita (*Arctostaphylos patula*). Table 4 has the model parameters for the three rangeland classes modeled,

[2] Types of humus.

namely: annual grasses, Blue Oak Rangeland, and Shrubs. Annual grasses in California rangeland have root-to-shoot ratios from 0.6 to 1.8 (Center et al. 1988; Savelle 1977).

Rainfall and nitrogen are primary controlling factors on forage production for California's grasslands with its Mediterranean climate with hot dry summers and mild wet winters. Peak forage production of rangeland grasses varies considerably temporally and spatially. For example, forage production at the San Joaquin Experimental Range varied between 1000 and over 5000 kg/ha between 1935 and 2000 (George et al. 2001). Data for a site in Shasta County near Redding exhibited large inter-annual variations in production (See Figure 6). The research team has not been successful in obtaining the specific location and soils for this site to test its model estimates of inter-annual variability of forage production with these field observations.

**Table 2. Forest-DNDC Inputs**

| Parameter | Unit |
|---|---|
| Climate | |
| - daily maximum and minimum air temperature | °C |
| - daily precipitation | centimeters/day (cm day) |
| - photosynthesis actively radiation (PAR) * | µmole/$m^2$/sec |
| - atmospheric N deposition | parts per million (ppm) |
| Soil ** | |
| - organic C content at litter layer | kilograms of carbon per hectare (kg C/ha) |
| - organic C content at top of mineral soil | kg C/kg soil |
| - bulk density of mineral soil | grams per cubic centimeter (g)/$cm^3$ |
| - pH at litter layer | |
| - pH at mineral soil | |
| - texture of mineral soil | |
| - clay fraction in mineral soil | |
| - stone fraction of the soil | |
| Vegetation | |
| - forest type *** | |
| - forest age | Year |

* Default daily Photosynthetically Active Radiation (PAR) values are calculated based on latitude, day length, horizontal potential insolation, potential insolation, declination of the sun, radius vector of the sun, solar constant, and a default cloud index 0.5.

** Default annually litter accumulation rate is calculated by an empirical equation ($dL/dt = (5700 - 80 * Latitude) – L / 10^{[-0.8 + 0.04 * Latitude)}$ ), developed based on data in Olson (1963) and Bray and Gorham (1964)

*** Pine, spruce, fir, hemlock, mixed hardwoods, oak, birch, and beech.

**Table 3. Forest parameters**

| Forest type | Leaf-N (%) | Leaf C/N | Wood C/N | Litter C/N in the forest floor | Leaf retention (year) |
|---|---|---|---|---|---|
| Pine | 1.3 | 35 | 200 | 70 | 2.25 |
| Spruce | 1.2 | 37 | 200 | 74 | 4 |
| Fir | 1.1 | 43 | 300 | 86 | 4 |
| Hemlock | 1.1 | 42 | 350 | 84 | 4 |
| Hardwoods | 2.2 | 20 | 200 | 40 | 1 |
| Oak | 2.0 | 23 | 150 | 46 | 1 |
| Birch | 2.2 | 21 | 180 | 42 | 1 |
| Beech | 2.7 | 16 | 200 | 32 | 1 |

(Based on Aber and Federer 1992; Aber et al. 1995; Stump and Binkley 1992; Cortez et al. 1996)

**Table 4. DNDC model parameters for rangeland classes**

| Range Code | 11 | 70 | 75 |
|---|---|---|---|
| Range Class | Annual_grasses | Blue_oak | Shrubs |
| Total biomass C* (kg/ha) | 4520 | 7082 | 3200 |
| Grain fraction | 0.01 | 0.01 | 0.01 |
| Shoot fraction | 0.45 | 0.5 | 0.7 |
| Root fraction | 0.54 | 0.49 | 0.29 |
| Plant C/N ratio | 37.6 | 60.76 | 174.2 |
| Grain C/N ratio | 33 | 30 | 60 |
| Root C/N ratio | 50 | 80 | 200 |
| Shoot C/N ratio | 29 | 50 | 160 |
| Water requirement | 400 | 550 | 450 |
| Max LAI | 3 | 7 | 3 |
| TDD | 2500 | 3000 | 3000 |
| Initial efficiency of use of absorbed light (kg $CO_2$/ha/hr)/(J/m$^2$/sec) | 0.4 | 0.4 | 0.4 |
| Maximum rate of leaf photosynthesis, kg $CO_2$/ha/hr | 60 | 35 | 35 |

*For Blue Oak and Shrubs this represents the total annual litter (foliar, twigs and fine roots) production.

Note: J/m$^2$/sec = Joules per meter squared per second; kg $CO_2$/ha/h = kilograms of carbon dioxide per hectare per hour.

Due to the high variability in soil fertility and rainfall patterns it is difficult to test against the data in Figure 6 without specific location information. However, to test the sensitivity of our grassland model, the research team ran a 10-year simulation for a site Southeast of Redding, California. DNDC simulated large inter-annual variations in total forage production (Figure 7). While a direct comparison of these data (see Figures 6 and 7) is limited, it is clear that the range (1400–2400 kg/ha, which is equivalent to 1568–2688 pounds per acre, lbs/acre) of DNDC estimates of forage production has the same order of magnitude as the field observations. Since the research team did not have access to appropriate data for Shasta County, the team used data from the Hopland Research Station for model calibration and testing. Fourteen years of average forage production at Hopland site over 14 years and soils were obtained from Valerie Eviner and Charles Vaughn and compared with DNDC model estimates (Figure 8). While the field data exhibited greater inter-annual variability, the magnitude of the DNDC modeled forage production estimates were comparable to the field data.

### 4.3. Linking DNDC Rangeland Analysis with Forest-DNDC Model

Prior starting the 50-year reforestation simulation for each polygon, a 15-year rangeland simulation was conducted for each modeling polygon first to observe the SOC dynamics under present rangeland conditions. Because the soil properties, especially SOC content and soil texture, are highly variable across the modeled domain, the SOC dynamics could be very different across the polygons that share the same vegetation types. For example, with same shrub cover, a site with an initially high SOC content would constantly lose C, versus a site with an initially low SOC content, which could gain C. Thus, the research team designed the soil database to assign two values of SOC content for each polygon based on soil survey. This design allowed the team to estimate the average SOC trends for a polygon with heterogeneous soils by capturing the maximum and minimum SOC fluxes. A 15-year simulation provided an adequate time period to observe the SOC trends under the differentiated soil conditions for each polygon.

At the end of the 15-year rangeland simulation for each polygon, a series of soil C and N profiles were recorded, and they were used as the initial soil profiles for the reforestation simulations. In the forest modeling mode, a 50-year simulation was carried out for each polygon using the soil profiles inherited from the 10-year rangeland simulations with the forest physiological and phenological parameters for the specific forest type starting from a forest age of 1. For most forests, peak biomass will not plateau within 50 years. However, the period is long enough to observe the biomass and SOC dynamics and trends in trace gas emissions.

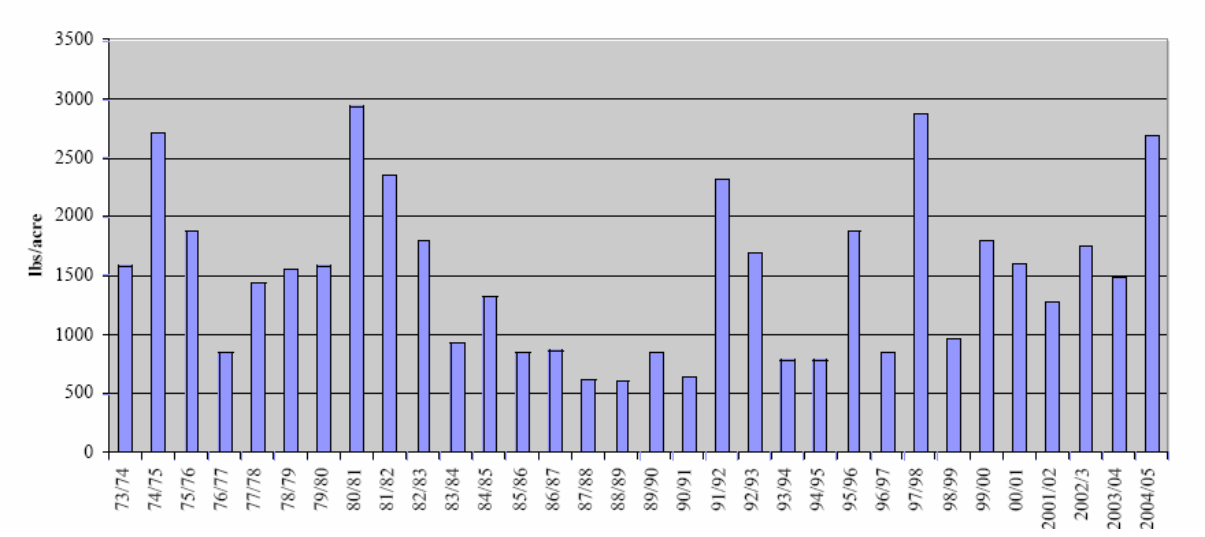


**Figure 6. Average forage production on annual range from 1973 through 2005, site outside of Redding, California**
**(Source: UCCE Sample Costs for Beef Cattle Yearling/Stocker Production, BV-SV-05)**

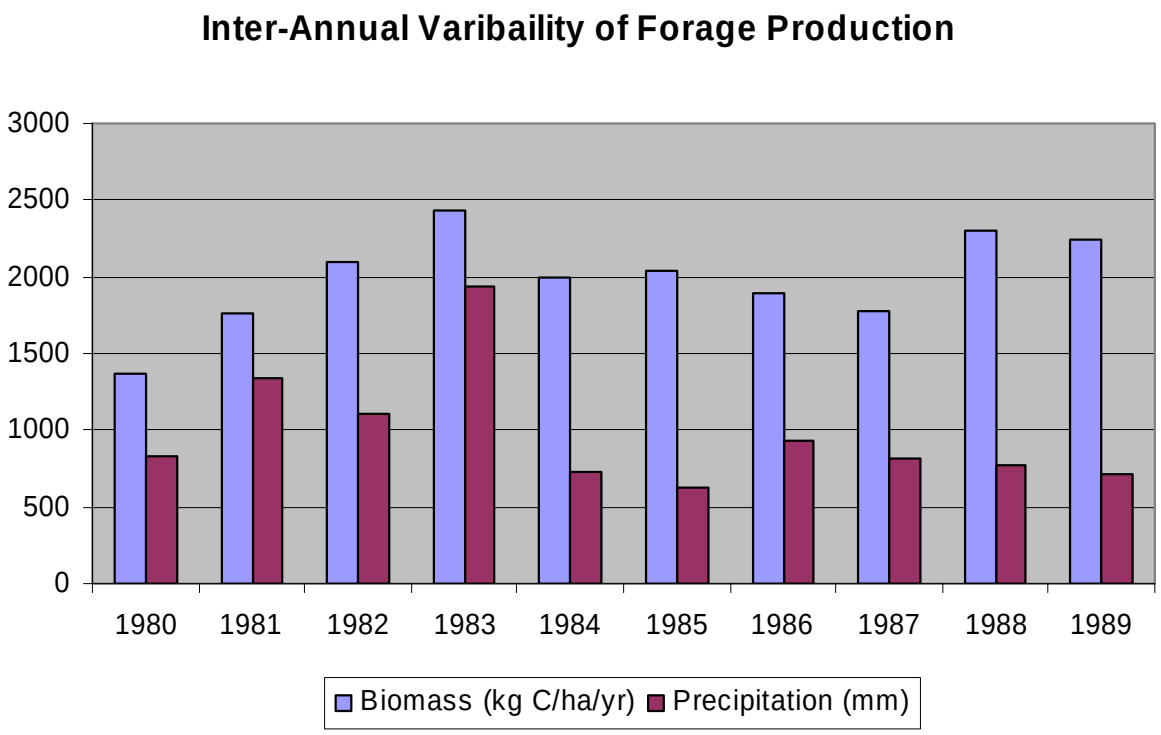


**Figure 7. DNDC modeled inter-annual variability of forage production with annual precipitation**

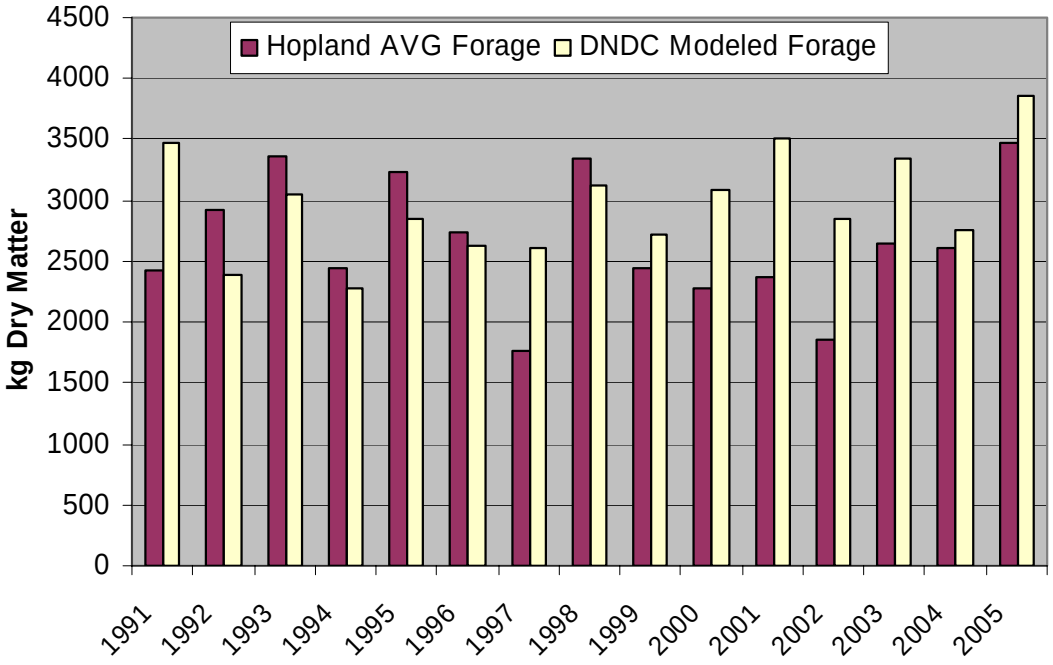


**Figure 8. Comparison of annual DNDC forage estimates with Hopland field data**

## 5.0 Management Scenarios

One goal of this scoping study was to evaluate the potential impacts of change grazing intensity on soil carbon stocks and trace gas emissions. The rationale was to assess the impacts of increasing the intensity of grazing as more rangelands are taken out of grazing use and reforested. Improvements in beef production per acre on annual rangeland due to intensive grazing management is more a result of improved utilization of forage, rather than an increase in forage production (pers. comm. Melvin George). However, some changes in ranch productivity have been documented under intensive grazing management including an improvement in the vigor and productivity of native and introduced perennial grasses (George et al. 1989). Stocking density (number of cattle per hectare) for California rangelands vary due to spatial and temporal variability of forage production and other management consideration. The research team set the nominal stocking density of one cow for every six hectares of grassland. Cattle grazing periods can range from winter grazing only, winter and spring grazing during peak forage production or year round grazing. This analysis assumed that winter and spring grazing periods ran from November 1 through June 1. The research team ran grassland simulations with three management scenarios:

1. No grazing: to examine soil carbon dynamics and trace gas emissions in the absence of grazing.
2. Nominal grazing: 1 cow for each 6 hectares.
3. Intensive grazing: 1 cow for each 3 hectares.

The DNDC model simulates the presence of grazing cattle by removing aboveground grass and depositing manure during the grazing periods. Cattle consume between 1.8% to 3.0% of their body weight daily (pers comm. Mel George). Thus, for a 1000 lb. cattle, this range translates into a daily consumption of between 3.3 to 5.5 kg C/cow/day. In Brown et al. (2004) total annual forage consumption was assumed to be 9495 lbs dry matter (DM)/cow/yr, which is equivalent to 4.7 kg C/cow/day. Based on these data, for this study's DNDC modeling researchers assumed that a cow consumes 4.7 kg forage-C/day, and produces 1.52 kg manure-C/day, which contains 0.1 and 0.13 kg N in the dung and urine, respectively. DNDC tracks animal demands for food and availability of forage (e.g., living grass and stubs) at daily time step. If there is deficit in forage availability, the manure production will be reduced accordingly.

## 6.0 Results

Field research and modeling analyses show that climate variability (primarily precipitation) can have a large impact on annual forage production and resulting carbon dynamics in the grasslands of California. Therefore, while this study has compiled 24 years worth of climate data (1980–2003), this report presents results from runs that have used a single climate year (1997) to illustrate how management (i.e., grazing intensity and reforestation) alone impacts soil carbon dynamics and trace gas emissions. Adding the impact of climate variability with these management alternatives is beyond this scoping effort.

Because, in general, SOC dynamics and $N_2O$ emissions are most sensitive to the initial SOC content (relative to soil pH, texture, and bulk density), the maximum and minimum SOC content (commonly obtained in a county soil survey data) are recorded in our GIS database for the DNDC and Forest-DNDC simulations (discussed in Section 3.3 above). DNDC and Forest-DNDC simulations were performed for each modeling area twice with the maximum and minimum SOC values, respectively. The two model runs produced two results of SOC change (or $N_2O$ flux). The two flux values formed a range, which will, with a high probability, be wide enough to include the "real" flux. The midpoint of these ranges are reported here, unless specified otherwise.

### 6.1. Baseline Shasta County Rangeland Carbon and GHG Emissions

Based on the model parameters listed in Table 4, the research team estimated soil carbon dynamics and trace gas emissions ($N_2O$ and $CH_4$) for Blue Oak Woodlands, annual grasslands, and our generic shrub classes. Rates of carbon sequestration varied significantly across the county. On average, shrubs were sequestering 90 kg C/ha/yr, with carbon sequestration rates varying from -881 to 975 kg C/ha/yr. Annual grasses, under a grazing intensity of 1 cow per 6 hectares, were essentially in carbon balance, losing on average less than 4 kg C/ha/yr. Grassland exhibited large variability with some regions sequestering up to 400 kg C/ha/yr and other regions losing soil carbon at a rate over 900 kg C/ha/yr. Grazed Blue Oak (modeled as 40% Blue Oak and 60% annual grasses) were a small source of carbon on average (~10 kg C/ha/yr), but also exhibited large spatial variability ranging from a moderate sink (~100 kg C/ha/yr) to a large source (~1000 kg C/ha/yr) of soil carbon. The regional patterns of total carbon sequestration (Figure 9 a,b,c) indicate that the annual grassland in the Southwest region of the county are losing carbon, where as most of the other regions of rangelands are sequestering carbon. Summing up the annual carbon sequestration of the total rangeland area modeled (all colored areas in Figure 9, other rangeland were not modeled as they were not suitable for afforestation) is 5,640 metric tons of carbon per year, with all rangelands sequestering, on average, 29 kg C/ha/yr.

In general, trace gas emissions from these rangelands are low. Table 5 provides average and range in methane oxidation (consumption), nitrous oxide emissions, and the net 100-year GWP. Some areas of annual grassland-exhibited relatively large $N_2O$ emissions, likely due to higher soil organic matter. From a net GWP perspective, the shrubs are a net sink in terms of $CO_2$-equivalents, due to high rates of carbon sequestration, while Blue Oak and annual grasslands are net carbon sources on average.

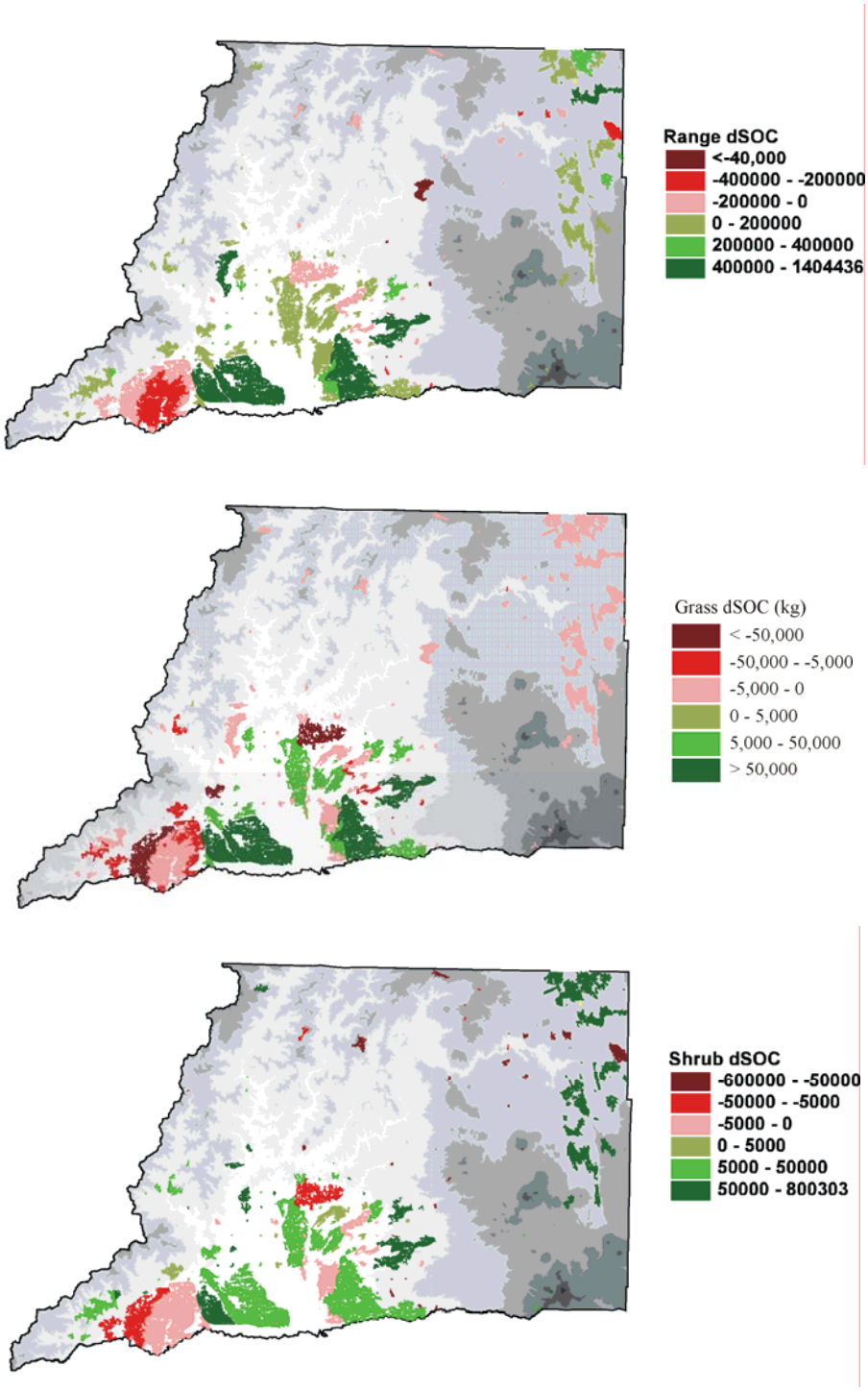


**Figure 9. Estimate of carbon sequestration in Shasta County rangelands. The top figure shows the net sequestration from all 3 rangeland classes (annual grasslands, blue oak woodlands, and shrubs). The middle figure shows the contribution from annual grasslands. The bottom figure shows the contribution of the shrub class. The shrub class dominates the carbon balance of the rangeland in the Northeast.**

**Table 5. Trace gas emissions and net GWP for rangelands**

| | $CH_4$ (kg C-$CH_4$/ha/yr) | | $N_2O$ (kg N-$N_2O$/ha/yr) | | Net GWP ($CO_2$eq/ha/yr) | |
|---|---|---|---|---|---|---|
| | Avg | Max | Avg | Max | Avg | **Range** |
| **Shrub** | -0.7 | -2.0 | 0.4 | 2.4 | -144 | **-831 to 975** |
| **Blue Oak** | -0.2 | -1.3 | 0.1 | 3.9 | 85 | **-353 to 5628** |
| **Grassland** | **-0.3** | **-1.9** | **1.1** | **7.2** | **535** | **-765 to 7043** |

### 6.2. Impact of Grazing Grasslands on Soil Carbon and GHG Emissions

The impact of grazing intensity on soil carbon dynamics depends on site quality (initial SOC content). For sites with higher initial SOC, increased grazing intensity will decrease the rate of carbon sequestration. In general, the DNDC model indicates that in the absence of grazing, rangelands of Shasta County would be sequestering ~50 kg C/ha/yr—an increase of 21 kg C/ha/yr over nominal grazing intensity. This is an increase of 4,390 metric tons of C per year. Figure 10 presents the patterns of net carbon sequestration by shifting to no grazing of annual grassland.

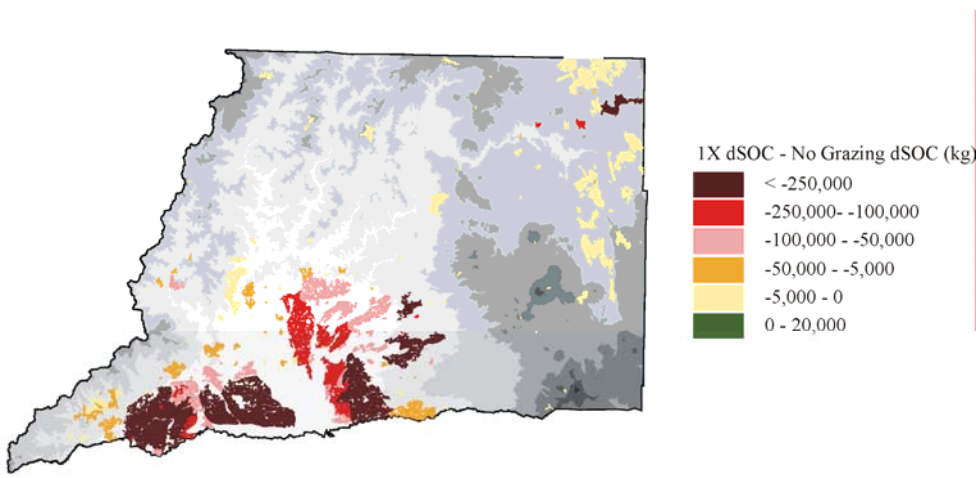


**Figure 10. Changes in annual carbon sequestration with no grazing of annual grasslands**

To estimate the impacts of a shift to more intensive grazing practices, the research team ran DNDC simulation with a doubling of the stocking density to one cow for every three hectares of annual grasslands. Based on a literature review, there appears to be a variable response of soil carbon stocks to grazing. For example, Milchunas and Lauenroth (1993) reviewed 34 studies, 40% of which showed an increase in soil carbon in response to grazing, with 60% showing a decrease in soil carbon. The present study's model results indicated a spatial variability of net impacts on soil carbon. While the majority of the regions had a net loss of soil carbon under intensive grazing, some

regions exhibited a increase in soil carbon relative to nominal grazing intensity. Based on these initial results, the impact of grazing intensity varies significantly depending on initial site fertility. There appears to be a physiologic response of the grazing that is dependant on site soil fertility. Sites with high soil carbon had very little difference in modeled below-ground carbon allocation in response to grazing intensity. However, grasses on areas with lower soil carbon had higher below-ground carbon allocation in response to higher grazing intensity. This impact can also been seen in the magnitude of soil heterotrophic respiration, as areas with lower initial SOC had an increase in heterotrophic respiration in response to the higher belowground carbon allocation. Hodgkinson and Baas Becking (1977) reported that after grazing, root mortality increases, possibly leading to large increases in SOC over short periods. However, without field data on the physiological response of grasses to grazing in Shasta County, the research team cannot say with confidence that intensive grazing in some regions of Shasta County will lead to increased soil carbon stocks.

### 6.3. Reforestation Carbon and GHG Emission Dynamics

Based on the distribution of candidate forest types for Shasta County (see Figure 2), the research team ran Forest-DNDC for three main afforestation types: pine, fir and oak. For each site and species, researchers ran a 50-year simulation and recorded annual estimates of net ecosystem exchange (NEE), net primary production (NPP), tree carbon, forest floor carbon, soil carbon, nitrous oxide, and methane emissions.

Each 50-year simulation was run twice, using the range in soil conditions following the study's rangeland simulation with the range of initial soil conditions. Simulations that start with the higher or lower end of the range in SOC content (derived from the NRCS STATSGO soil survey database) are labeled "max" or "min," respectively. The study assumes that at the establishment time of afforestation all native vegetation has been removed.

While the primary goal of modeling carbon and nitrogen dynamics during the reforestation process is to assess the potential magnitude of trace gas emissions and soil carbon dynamics, the research team also wanted to evaluate the study's forest growth projections with Winrock's forest biomass growth model, because significant differences would influence litter inputs and carbon allocation, which in turn influence soil carbon dynamics and trace gases emissions. Growth rates for our Fir, Pine and Oak classes varied slightly (Figure 11), with Pine exhibiting higher growth rates.

In addition to changing above-ground carbon stocks, afforestation of grasslands influences SOC stocks by changing soil structure, soil moisture, magnitude and dynamics of soil carbon and nitrogen inputs, and other factors. Forest-DNDC model simulation revealed an initial decrease in soil carbon following reforestation (see Figure 12), followed by an increase in soil carbon stocks as the forest stands become more established. However, these dynamics are again influenced by the initial soil carbon content and, to a lesser extent, forest type.

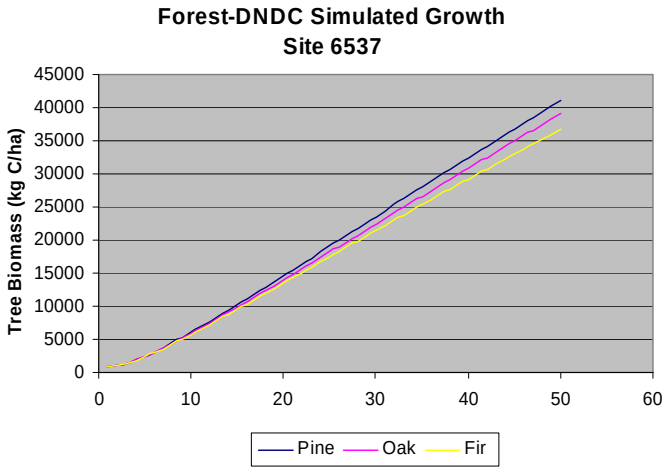


**Figure 11. Forest growth rates**

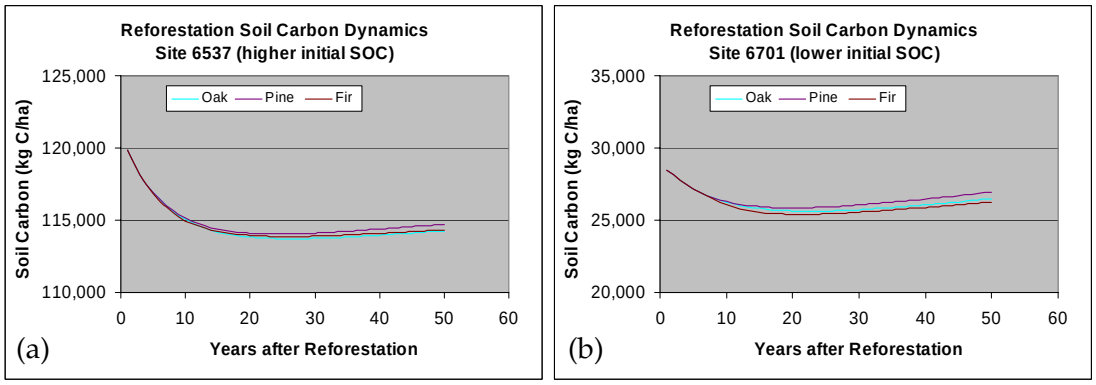


**Figure 12. Mineral soil carbon dynamics following afforestation. Rates of carbon loss from mineral soils during the first 50 years follow afforestation varies, based initial soil carbon content (12a site has higher initial soil organic carbon content than site in 12b), with small variation across forest types modeled after 50 years.**

Assessing the net impact of afforestation of rangelands on full GHG accounting requires an assessment of not only changes in tree and soil carbon stocks, but also changes in forest floor carbon stocks and trace gas emissions, as well as accounting for net GHG conditions of the rangelands prior to afforestation. After fifty years, carbon accumulation in the forest floor can reach upwards of 6500 to 7500 kg C/ha, or 15%–20% of the tree carbon stocks (Figure 13).

During afforestation, the amount and quality of fresh litter will impact the trace gas emissions. For example, if the fresh litter has a low C/N ratio, larger fractions of the litter will be partitioned into the more labile pools, resulting in faster decomposition rates. Because the quantity and quality of the litter vary from forest type to forest type, we can expect differences in soil organic matter contents as well as trace gas emission emissions for the same site with different afforestation species. For example, reforestation with oak, versus pine or fir, will potentially lead to higher nitrous oxide

emissions due to oak higher foliar nitrogen content. This can been seen in the model simulations which indicate that Oak stands would have approximately double the $N_2O$ emission rates after 40 years of stand development (Figure 14). During the initial stages of stand development, $N_2O$ emission rates would vary based on initial soil carbon content ("max" simulations had higher emissions than "min" simulations). However, these differences were substantially reduced as the system approached equilibrium with respect to forest floor dynamics and soil carbon and nitrogen inputs.

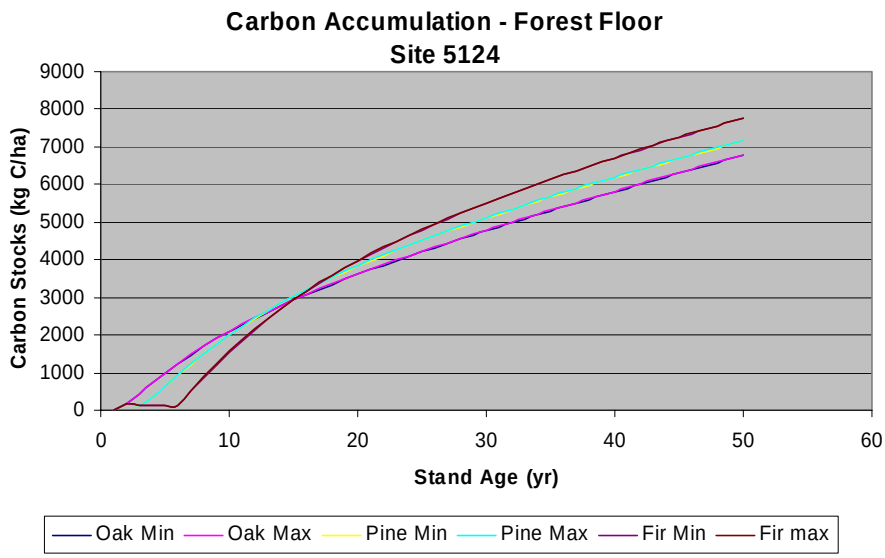


**Figure 13. Development of modeled forest floor carbon stocks following reforestation**

From this study's simulations with Forest-DNDC, the research team estimated carbon stocks of woody biomass after 50 years to be approximately 38, 40, and 42 tons C/ha for fir, oak, and pine forests, respectively. If the accumulation of carbon in the forest floor is included, then these stocks average 43, 42, and 47 tons C/ha for fir, oak, and pine forests, respectively. These stocks are significantly smaller than those provided in Brown et al. (2006), which were derived from look-up tables and summarizing forest inventory and analysis (FIA) data for Shasta County. The model simulations indicated that forest growth was severely limited by nitrogen stress. Annual deficiencies in nitrogen availability between 10% and 20% were common. Without nitrogen stress, this study's modeled forest yields would be significantly higher. In the example given in the last section of this report, a fertilization rate of 20 kg N/ha would result in an increase in forest biomass of 20%. Nevertheless, this higher production would still be lower than the estimates provided by look-up tables. While this scoping study was aimed at understanding the changes in soil carbon stocks and trace gas emissions, a change in biomass yields would have an impact on relative contribution of soil carbon dynamics and trace gas emissions on the full GHG balance.

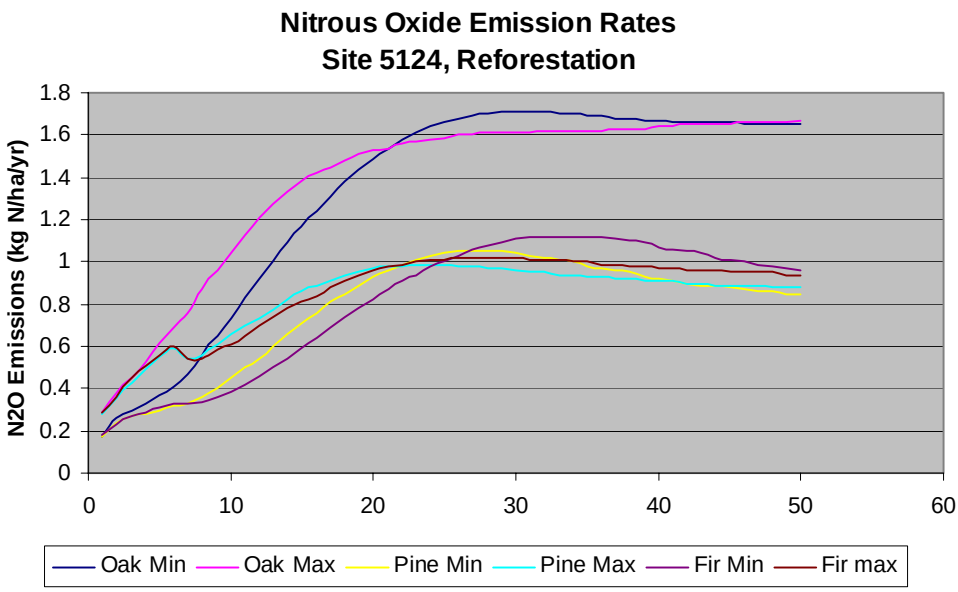


**Figure 14. Nitrous oxide emissions following reforestation with oak, pine or fir**

Based on the model results, the research team compiled a complete GHG balance for the 50 years from the onset of the afforestation. To do this, the team compared the magnitude of the carbon sink due to biomass and forest floor carbon accumulation with the net carbon source from the cumulative loss in soil carbon and trace gas emissions over the 50-year period. Often studies of afforestation or reforestation focus on carbon accumulation rates of woody biomass and forest floor, with little attention paid to soil carbon dynamics and trace gas emissions. This is primarily due to the difficulty in estimating soil carbon dynamics and trace gas emissions during afforestation. However, with the maturity of process-based biogeochemical and forest physiology models and the improved access of GIS databases, full accounting is now possible, albeit with high levels of uncertainty. To estimate the impact of soil carbon losses and trace gas emissions (SCTG-E) on the net GHG balance the research team calculated the percent offset of these emissions relative to carbon sequestered in woody biomass and forest floor (C-Stock) by the following simple ratio (expressed as a percentage):

$$\text{SCTG-Eoffset} = \text{SCTG-E } (CO_2\text{eq}) \ / \ \text{C-Stock } (CO_2\text{eq}).$$

The magnitude of SCTG-Eoffset varied spatially across forest types driven by difference in forest productivity (C-Stock) and soil conditions. SCTG-Eoffset ranged from a few percent to 35% (Figure 15). Sites with higher SCTG-Eoffset were characterized by low forest productivity. Over 90% of the sites had SCTG-Eoffset values less than 12%. The spatial patterns of SCTG-E for hardwoods are shown in Figure 16. Areas with high SCTG-E had high SOC content prior to afforestation. Areas with higher SOC typically have larger losses of soil carbon during forest establishment (see Figure 12), will oxidize more methane, and will have higher nitrous oxide emissions.

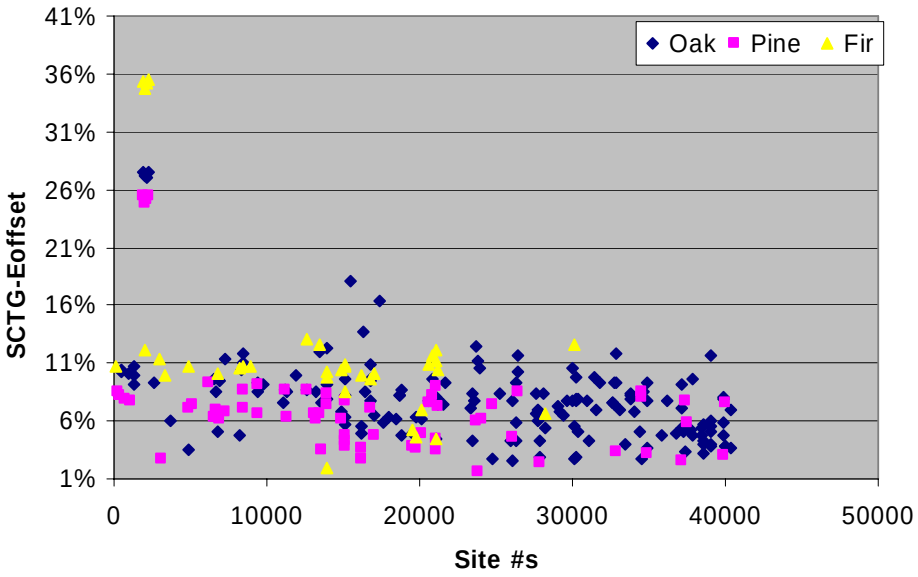


**Figure 15. Potential percent reductions in net carbon sequestration during afforestation, due to losses of soil organic carbon and elevated trace gas emissions**

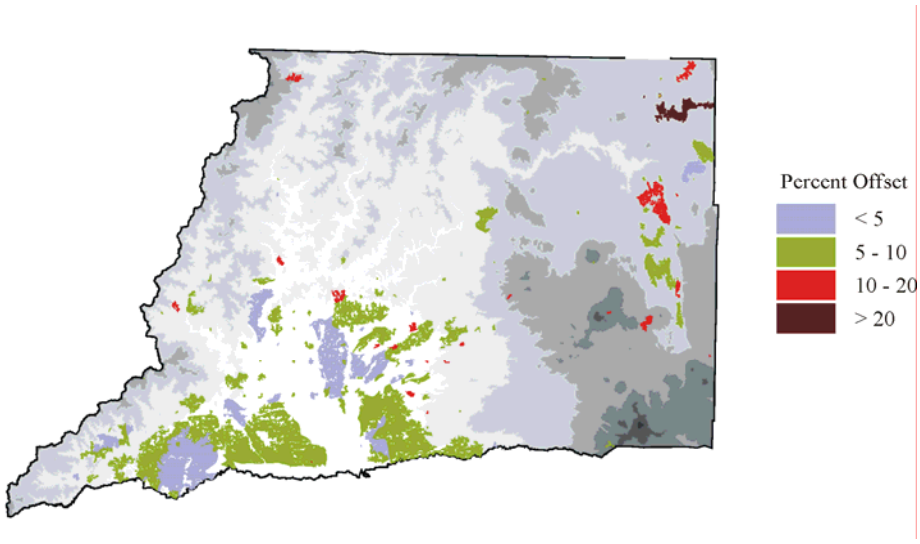


**Figure 16. Spatial patterns of SCTG-E offset for hardwoods**

This study also examined the true net GHG benefits by accounting for not only SCTG-E effects, but also the net GHG balance of the rangelands prior to afforestation. As shown in Table 5 and Figure 9, many of Shasta's rangelands are net carbon sinks. Conversion of these rangelands to forestlands can shift the soils from a net sink to a source. The research team estimated the net GHG benefits of afforestation as follows:

$$\text{Net GHG}_{\text{afforestation}} = \text{C-Stock} + \text{SCTG-E} + \text{NetGHG}_{\text{range}}$$

where each term is presented in $CO_2$equivalents, with a positive sign for sequestration and negative sign for sources. The difference between C-Stock and netGHG accounting for afforestation is illustrated with some results for subset of twelve sites that are indicative of Shasta County conditions (Figure 17). In general, NetGHG estimates are lower than C-Stock estimates, due to soil carbon loss during afforestation and removal of the net carbon sink of rangelands. In some rare cases, afforestation of rangelands may lead to a net increase in GHG emissions when rangelands, which are strong sinks of carbon, are replaced by very low-productivity forests with high trace gas emissions (for example, site 7 in Figure 17). However, in some cases, by accounting for netGHG balance, a greater rate of carbon sequestration than C-Stock is observed (site 5). Given that the study's model estimates of forest productivity are lower than estimates from look-up tables, the difference between C-Stock and netGHG estimates may be lower. In addition, the differences between C-Stock and netGHG accounting will change over time as soil carbon stock recover following initial disturbance of afforestation (Figure 12). Nevertheless, it is clear that full GHG accounting is helpful in understanding the spatial and temporal variability in benefits of afforestation for carbon sequestration.

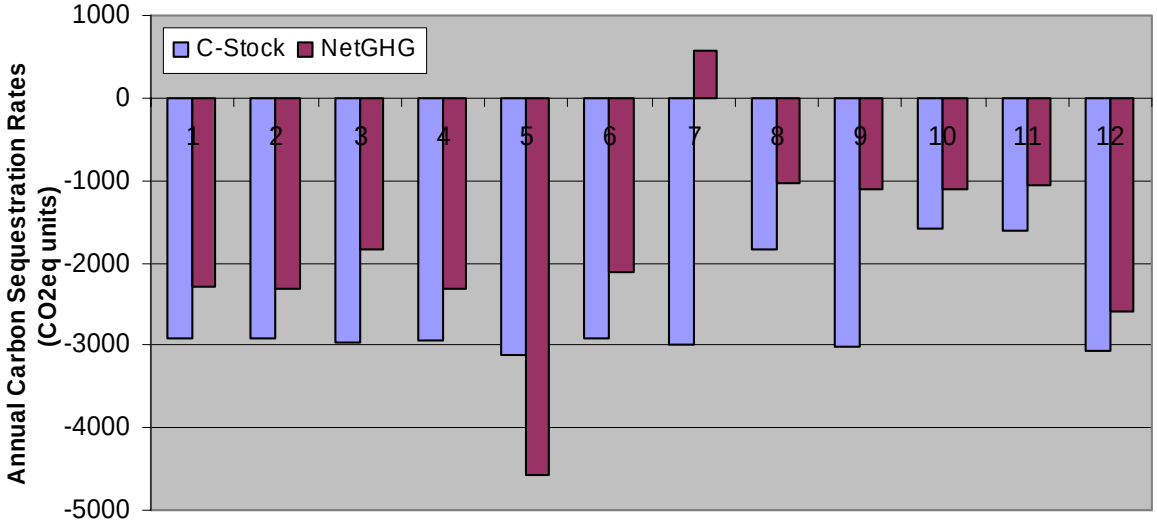


**Figure 17. Comparison C-Stock and NetGHG accounting for 12 sites in Shasta County**

## 7.0 Additional Research Needs

Based on the initial findings of this scoping study, the research team began to examine several additional issues related to afforestation of rangelands in Shasta County: (1) the potential benefits of using fertilizer to enhance forest productivity and net carbon sequestration, (2) how climate change might impact afforestation dynamics, and (3) how afforestation would impact the local hydrology in Shasta County.

### 7.1. Impacts of Fertilization and Climate Change

This study's baseline analysis assumed that the afforestation did not use fertilizer inputs to enhance productivity. In addition, the research team assumed steady climate conditions based on 1997 climate. The team ran a sensitivity analysis on the impacts of fertilization or temperature change on C and N dynamics during the afforestation process. Researchers selected a representative upland site in Shasta County modeled two forest types (fir and hardwoods) with three fertilizer treatments (no fertilizer, 10 kg N/ha per year, and 20 kg N/ha per year, applied on March 1) and three temperature conditions (1997, a 2°C temperature increase, and a 4°C temperature increase). The modeled carbon dynamics, trace gas emissions and net GWP for each scenario are shown for fir (Table 6) and hardwoods (Table 7).

Global warming potential was calculated based on

$$GWP = CO_2 + N_2O * 310 + CH_4 * 21$$

where

$$CO_2 = -(dBiomass\text{-}C + dFloor\text{-}C + dSoil\text{-}C) * (44/12);$$

$$N_2O = N_2O\text{-}N * (44/28);$$

$$CH_4 = CH_4\text{-}C * (16/12).$$

**Table 6. 50-year average annual C and N fluxes (units: kg C or N/ha per year) and other relevant items for the fir forest**

| | NPP | d(biomass C) | d(floor C) | d(soil C) | $N_2O$ | $CH_4$ | ET | GWP |
|---|---|---|---|---|---|---|---|---|
| | kg C/ha/yr | kg C/ha/yr | kg C/ha/yr | kg C/ha/yr | kg N/ha/yr | kg C/ha/yr | mm/yr | kg $CO_2$-equivalent/ha/yr |
| Baseline | 6373 | 709 | 79 | -39 | 0.23 | -4.73 | 431 | -2767 |
| 10 kg N | 6567 | 786 | 95 | -17 | 0.38 | -4.75 | 437 | -3116 |
| 20 kg N | 6651 | 809 | 103 | -7 | 0.51 | -4.76 | 440 | -3203 |
| 2°C | 6005 | 708 | 76 | -45 | 0.26 | -5.3 | 493 | -2731 |
| 4°C | 5460 | 709 | 73 | -50 | 0.28 | -5.8 | 552 | -2710 |

ET = evapotranspiration

**Table 7. 50-year average annual C and N fluxes (units: kg C or N/ha per year) and other relevant items for the hardwood forest**

| | NPP | d( biomass C) | d(floor C) | d(soil C) | $N_2O$ | $CH_4$ | ET | GWP |
|---|---|---|---|---|---|---|---|---|
| | kg C/ha/yr | kg C/ha/yr | kg C/ha/yr | kg C/ha/yr | kg N/ha/yr | kg C/ha/yr | mm/yr | kg $CO_2$-equivalent/ha/yr |
| Baseline | 693 | 591 | 46 | -70 | 0.24 | -4.7 | 267 | -2094 |
| 10 kg N | 946 | 675 | 56 | -55 | 0.36 | -4.7 | 278 | -2435 |
| 20 kg N | 1078 | 702 | 61 | -48 | 0.46 | -4.8 | 283 | -2532 |
| 2°C | 807 | 626 | 48 | -71 | 0.26 | -5.3 | 316 | -2233 |
| 4°C | 815 | 646 | 48 | -73 | 0.27 | -5.8 | 359 | -2277 |

ET = evapotranspiration

These model results indicate that:

1. Increase in the forest biomass and floor C pools dominates the GWP; Emissions of $N_2O$ or $CH_4$ account for only about 5% of GWP.

2. During the simulated 50 years, the forest biomass and floor C pools continuously increased, while C storage in the soil mineral profile decreased, due to its initial high SOC contents (0.12 kg C/kg and 0.036 kg C/kg for fir and hardwood, respectively) and reduction in litter input.

3. Fertilization increased the forest productivity (i.e., NPP) and hence C storage in the forest biomass pool (Figure 18) as well in the forest floor pool (Figure 19), which is the major pool receiving litter. Carbon in the soil mineral pool slightly decreased due to the enhanced decomposition with higher availability of free N (Figure 20). Fertilization increased $N_2O$ emissions slightly with little change in $CH_4$ oxidation. Based on this site result, it appears that fertilization may ameliorate differences in C-Stock and netGHG sequestration by enhancing productivity and reducing soil carbon losses.

4. Increase in temperature decreased fir but increased hardwood productivity (i.e., NPP). Total biomass increased for hardwood but remained unchanged for fir forest. Increase in temperature increased $N_2O$ emissions and $CH_4$ oxidation. Evapotranspiration rate increased from 431 to 552 mm/yr for fir; and from 267 to 359 mm/yr for hardwood, when temperature increased by 4°C.

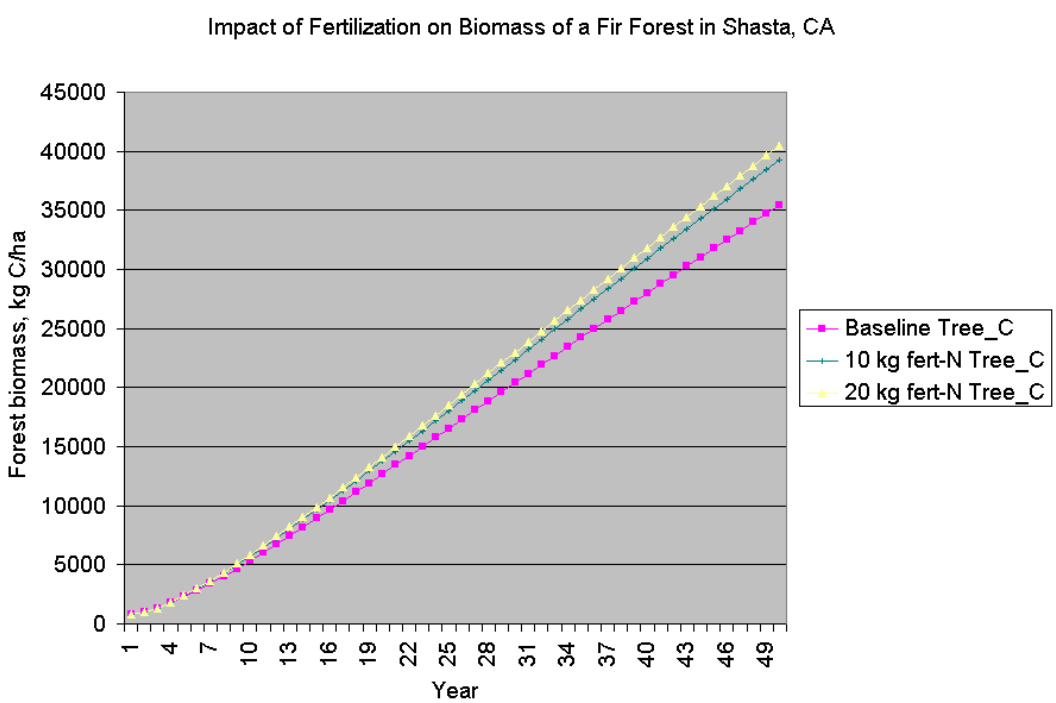


**Figure18. Impact of fertilization on forest biomass production. Annual application of 20 kg N/ha can increase forest productivity by over 10% over a 50-year period.**

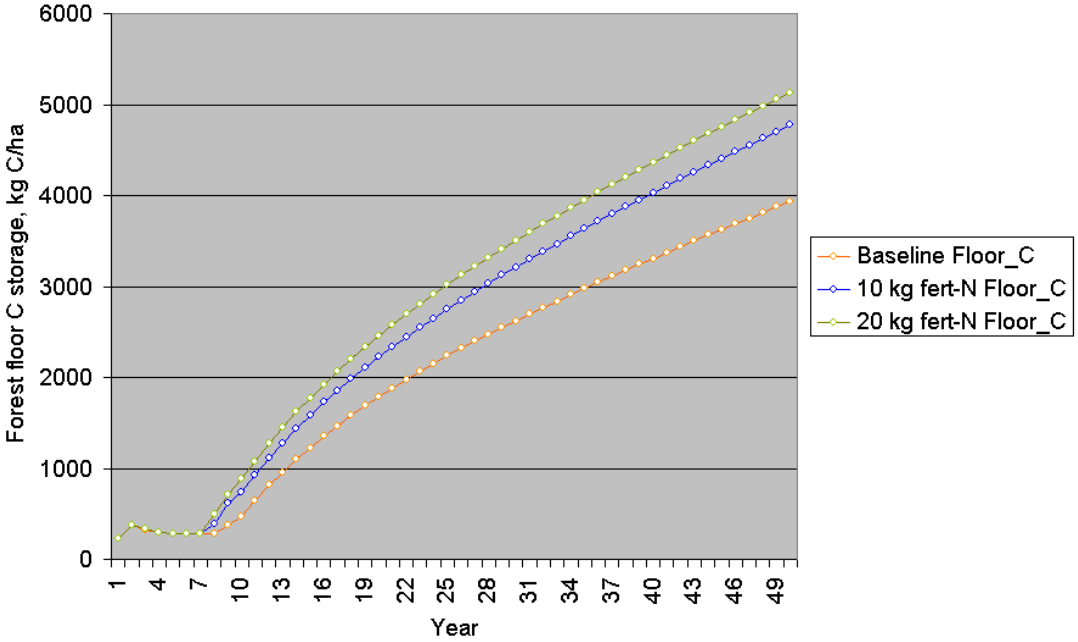


**Figure 19. Impact of fertilization on carbon accumulation in forest floor. Forest floor carbon accumulation is high under increasing fertilizer application rates.**

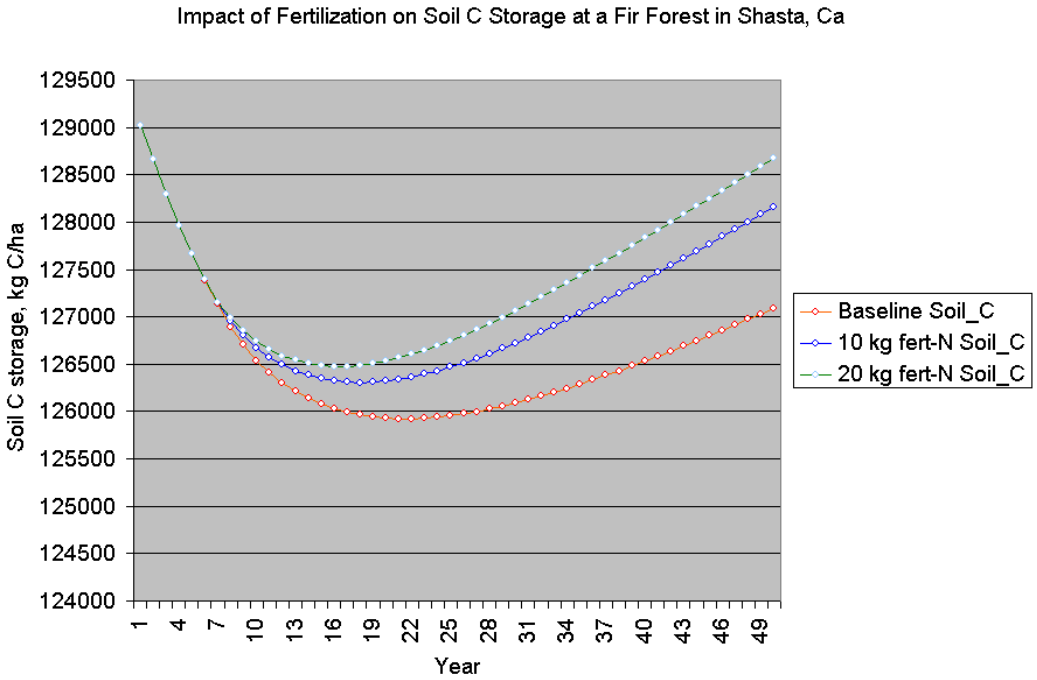


**Figure 20. Fertilizer impacts on soil carbon dynamics. Soil carbon stocks typically decline starting from the transition from rangelands to afforestation. Annual fertilizer application can reduce the soil carbon deficit during afforestation.**

## 7.2. Changes in Hydrology

Recent climate change modeling scenarios predict that soil moisture dynamics will likely change under conditions of elevated $CO_2$ and climate change in California, although the direction of that change is still being debated. Therefore, in addition to the soil carbon and trace gas emission analysis, the research team decided to use SWAT model to assess potential impact of reforestation on regional patterns of evapotranspiration. The Soil Water Assessment Tool (SWAT) is a continuous, distributed hydrologic model that estimates water flow as surface runoff, lateral subsurface flow, and vertical water table fluctuations based on physically based and conceptual models of flow processes (e.g., kinematic wave, SCS curve number methods). SWAT is capable of simulating surface water and groundwater fluxes as well as spatial and temporal water table dynamics at the landscape scale. SWAT consists of major water budget components such as weather, surface runoff, return flow, evapotranspiration, transmission losses, pond and reservoir storage, irrigation water transfer, groundwater flow, and channel routing. The model runs on a daily time step for short- or long-term predictions and operates in a distributed manner to account for spatial differences in soils, land use, crops, topography, channel morphology, and weather conditions. The SWAT model was developed as a nonpoint source pollution model with a strong emphasis on modeling the hydrologic cycle and water routing in watershed.

The long-term impact of afforestation of rangelands on system hydrology is also an important question to address in regards to rangeland management and impacts of afforestation on ecosystem services and climate. The SWAT model was employed to run initial water budget analyses under current rangeland conditions and future full reforestation scenario for selected sub-basins in Shasta County. Figure 21 shows results from a SWAT run for an area in the northeastern section of Shasta County dominated by montane vegetation. This figure compares evapotranspiration (ET) rates for current rangeland and a full afforestation scenario represented as the ratio of reforested ET to current rangeland ET.

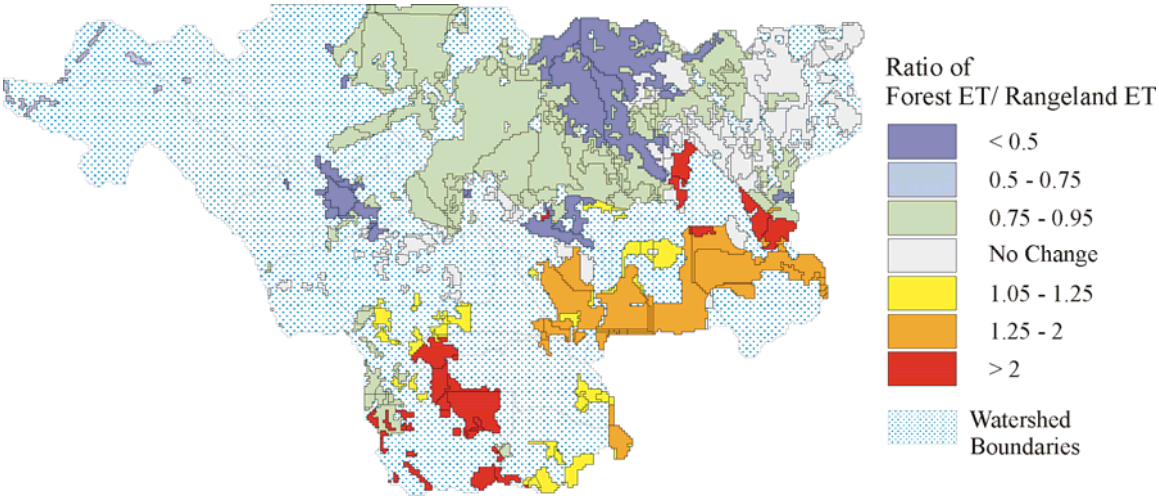


**Figure 21. Change in evapotranspiration (ET) rates for current rangeland and a full reforestation scenario**

The results of this preliminary SWAT analysis show a mix of both increases and decreases in ET occurring from afforestation. Areas exhibiting a marked decrease in ET are associated with shrubland rangeland system reforested to pine forest in the north-central part of the study area. A marked increase in ET was observed in the south-central part of the study area with annual grassland areas afforestated to mixed hardwood systems. Further analysis of ET predictions at more sites in Shasta County would be beneficial to understand the dynamics of hydrology in relation to afforestation and changing land use scenarios. An analysis of the net impact on surface runoff, infiltration, total ET, and water delivery to stream network could be performed by integrating SWAT with vegetation physiology in Forest-DNDC.

Figures 22 and 23 illustrate the impacts of reforestation on seasonal runoff for two sites with different soil characteristics. These figures show a decrease in surface runoff for reforested areas but only for those areas with higher clay content. From Figure 21 we may presume that changes in ET patterns under certain afforested conditions could result in more water being transferred to the atmosphere via evapotranspiration and resulting in less water leaving the system via surface runoff. It is assumed that infiltration dominates the sandier, lower clay content soils, and therefore any potential changes in ET water demand will have a comparatively smaller effect on the water balance.

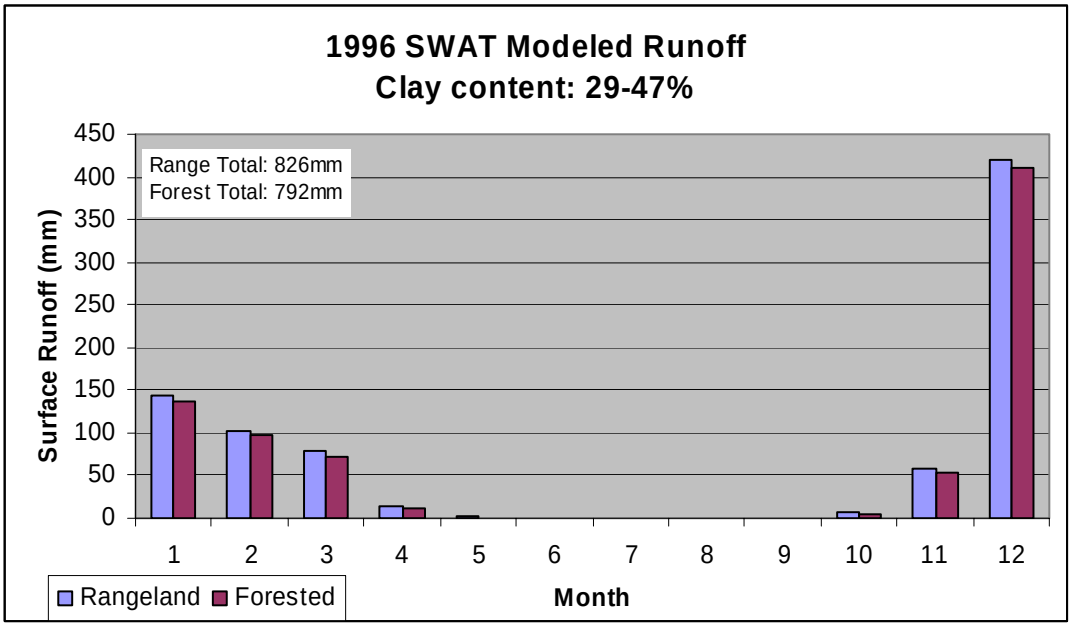


**Figure 22. Impacts of afforestation on seasonal runoff for a site with moderately high clay content**

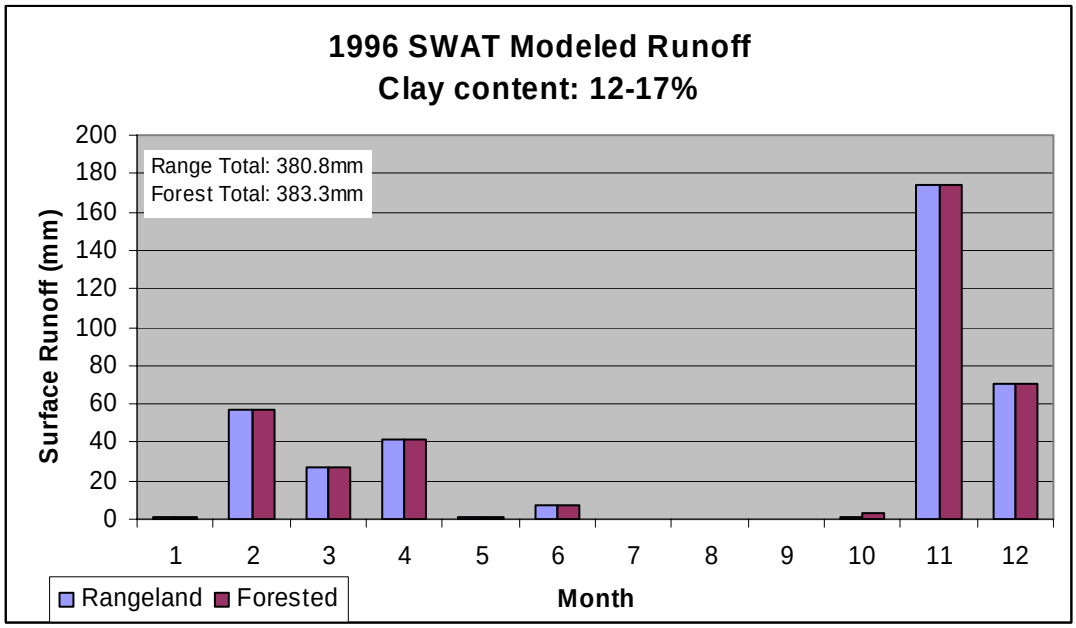


**Figure 23. Impact of afforestation on seasonal runoff for a sandy site**

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## 9.0 Glossary

| | |
|---|---|
| CDF | California Department of Forestry and Fire Protection |
| CIMIS | California Irrigation Management Information System database |
| DAYMET | A model that generates daily surfaces of temperature, precipitation, humidity, and radiation |
| DNDC | Denitrification-Decomposition |
| ET | evapotranspiration |
| FRAP | Fire and Resource Assessment Program |
| GHG | greenhouse gas |
| GIS | geographic information systems |
| GWP | global warming potential |
| LAI | leaf area index |
| LCMMP | California Land Cover Mapping and Monitoring Program |
| NEE | Net Ecosystem Exchange |
| NPP | Net Primary Production |
| NRCS | U.S. Department of Agriculture's Natural Resources Conservation Service |
| NTSG | Numerical Terradynamic Simulation Group |
| PAR | Photosynthetically Active Radiation |
| PM | particulate matter |
| PnET | forest physiology model |
| SCTG-E | soil carbon losses and trace gas emissions |
| SOC | soil organic carbon |
| SOM | soil organic matter |
| SSURGO | Soil Survey Geographic database |
| STATSGO | State Soil Geographic database |
| SWAT | Soil Water Assessment Tool |
| WHR | Wildlife Habitat Relationship |