AN ABSTRACT OF THE DISSERTATION OF

Julian A. Licata for the degree of Doctor of Philosophy in Forest Science presented on December 20, 2007.
Title: Impact of Exotic Ponderosa Pine (*Pinus ponderosa* Doug. ex Lawson) Plantations on Water Resources in Northwestern Patagonia, Argentina.

Abstract approved: ____
Barbara J. Bond

Exotic ponderosa pine (*Pinus ponderosa* Doug. ex Lawson) plantations are being planted within the natural distribution area of cordilleran cypress (*Austrocedrus chilensis* (D.Don) Pic. Ser. et Bizzarri) in Patagonia, Argentina. The productivity of these exotic plantations is much greater than that of native forests, suggesting greater water use. Before these plantations become widespread, it is crucial to determine what effects ponderosa pine plantations may have on scarce water resources in the region.

In the first study, I examined the seasonal variation in transpiration and soil water depletion for four forest stands: high and low density ponderosa pine plantations, and high and low density cordilleran cypress stands. Measurements were taken during two growing seasons with contrasting amounts of precipitation. Transpiration was greater in ponderosa pine than in cordilleran cypress stands of similar density. The high density ponderosa pine plot used a greater amount of water from deeper soil layers compared to all the other plots.

In the second study, I conducted measurements of above- and below-canopy rainfall, and water storage capacity of branches to estimate the proportion of water that it is intercepted by the canopy and evaporates back into the atmosphere without

reaching the soil. A simple analytical model was used to estimate annual interception losses by the four forest stands. Cordilleran cypress stands showed much larger interception losses than ponderosa pine stands of similar density. Thus, the negative effects of increased transpiration in ponderosa pine may be offset by their smaller interception losses when compared to cordilleran cypress.

In the third study, I used a detailed process-based model to close the annual water budget of the stands, and evaluate the balance between ET and precipitation under several climate change scenarios, based on the projections made in the IPCC's fourth assessment report. Model predictions suggested that high density ponderosa pine plantations are not hydrologically sustainable (ET > precipitation) under current conditions, and that none of the high density stands would be hydrologically sustainable under future scenarios. Management of stand density in both native cordilleran cypress and ponderosa pine plantations would be necessary to reduce the impacts of future climate change on water resources.

Impact of Exotic Ponderosa Pine (*Pinus ponderosa* Doug. ex Laws.) Plantations on Water Resources in Northwestern Patagonia, Argentina

by

Julian A. Licata

A DISSERTATION

submitted to

Oregon State University

in partial fulfillment of
the requirements for the
degree of

Doctor of Philosophy

Presented December 20, 2007
Commencement June 2008

Doctor of Philosophy dissertation of Julian A. Licata presented on December 20, 2007.

APPROVED:

Major Professor, representing Forest Science

Head of the Department of Forest Science

Dean of the Graduate School

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

Julian A. Licata, Author

ACKNOWLEDGMENTS

This thesis would not have been possible without the help of many people. First of all, I would like to express my immense gratitude to my advisor, Barbara Bond. Without her enthusiasm, support and scientific guidance, my dissertation would not have been possible. Since the first day I met her in Argentina, I was impressed by her creativity and enthusiasm on many scientific issues that I was also interested in. I knew at that very moment that, if I was going to pursue a graduate degree, I wanted her to be my advisor. Since then, she has been a lot more than a scientific advisor to me. She has treated me like a member of her family. I'm not sure if there are many graduate students that can say that they usually feel better *after* meeting with their advisors than *before*. Well, I'm one of those.

I would like to thank my committee members Dominique Bachelet, John Selker and Mike Unsworth for their support, encouragement and advice. They were always available to help and give me advice when I needed it. Thanks also to John Bliss and Kevin Boston, who agreed to serve as my graduate representatives.

I am very grateful to the PI of the project in Bariloche, Argentina, where I did all the field work for my dissertation; Tomás Schlichter, who became a role model for me and my career. I would also like to thank my colleagues in Bariloche, Gonzalo Caballé, Inés Bertoldi, Mariana Weigandt, Mauro Sarasola, Mercedes Urquiza, Fabian Jaque, Veronica Rusch, Santiago Varela, and Guillermina Dalla Salda. They made me feel at home in their lab and unselfishly shared everything with me. Special thanks to Maria Elena Fernández and Javier Gyenge, who not only helped me professionally, but also their encouragement and help made my relationship with my girlfriend (now wife) possible. My field work would not have been the same if not for Nora and Jose, who run the campground where I spent many nights during my field seasons. I'll never forget the dinners in their kitchen, talking in front of the warm stove, and their care and affection for me.

Many thanks also to my 'surrogate family,' the Reisings, in Bariloche that cared for me and made my many moves from one hemisphere to the other less painful. I was literally treated like another son by 'mom' Estela, and am very grateful for her kindness. I have no words to express my appreciation for all the help and support I got from my friend Carlitos and his lovely wife and daughter Luciana and Josefina.

Of all the wonderful things I got from coming to Corvallis, my friendship with Guido Corno is one of the most important to me. Thank-you for your advice, support, great political and scientific discussions, and for putting up with that crazy travel schedule to be able to come to my wedding and be my best man.

Of course, my 'real' family has been a great part of my success in life, and they are anxiously waiting for me to finish my degree. They are still hopeful that I'll be a little closer to home (Buenos Aires) in the next step of my life.

Above all, none of this would have been possible (and I'm speaking literally here) if not for the love, encouragement and help of my wife, Erika Adams.

I am grateful to the National Science foundation (Grant Number DEB 0316392), and INTA (Instituto Nacional de Tecnología Agropecuaria) for providing funding for my research. I gratefully acknowledge Rupert von Haniel for allowing me to carry out my research on his property, and the personnel of EEA INTA Bariloche for their administrative support. I am also thankful to the Forest Science Department here at Oregon State, which I’m very sad to be leaving.

CONTRIBUTION OF AUTHORS

Javier E Gyenge, Maria Elena Fernández, Tomás M Schlichter, Barbara J Bond were directly involved in the design and analysis of Chapter 2, and are co-authors on the publication resulting from this chapter. Thomas G. Pypker, Mariana Weigandt, Michael H. Unsworth, Javier E. Gyenge, Maria Elena Fernández, Tomás M. Schlichter and Barbara J. Bond assisted in the design, field work and analysis of Chapter 3, and are co-authors on the manuscript to be submitted for publication. Dominique Bachelet, Gonzalo Caballé, Javier E. Gyenge, Maria Elena Fernández, Tomás M. Schlichter and Barbara J. Bond were involved in collection of the field data and analysis of the model results for Chapter 4 and, as a result, will be co-authors on the paper to be submitted for publication.

TABLE OF CONTENTS

TABLE OF CONTENTS (Continued)

Page

LIST OF FIGURES

LIST OF FIGURES (Continued)

LIST OF FIGURES (Continued)

LIST OF TABLES

CHAPTER ONE: GENERAL INTRODUCTION

The fate of global climate and its interactions with carbon and water cycles are currently of great concern to scientists and policymakers. Evidence of the effects of human activities on the global climate is mounting (Le Trout et al., 2007). At the same time, a large effort is being made to elucidate the roles of terrestrial ecosystems in global climate, how each one feeds back on the other, as well as determine what the potential consequences of future climate change may be (Cramer et al., 2001; Nemani et al., 2003; Friedlingstein et al., 2006; Torn and Harte, 2006). Changes in forest cover and structure have been reported to induce changes in the local and regional climate (Lawton et al., 2001; Feddema et al., 2005), and hydrologic cycle (Kuczera 1985; Sahin and Hall, 1996; Bradshaw et al., 2007) through changes in the partitioning of energy and evapotranspiration.

Argentina has suffered dramatic reductions in forest cover area in the last century, primarily a result of human intervention (Merenson, 1992; SAyDS, 2006). Estimates of the total reduction in forest area range from 40 to 75% (Montenegro et al., 2004). During the last three decades, the Argentinean government has implemented several mechanisms to promote investment in fast-growing forestry plantations as a way to reduce the net reduction in forest land (SAGPyA, 1999; Laclau, 2003). The use of afforestation with fast-growing species has been suggested by many authors to be a sustainable option to reduce the logging of natural forests (Sedjo and Botkin, 1997; Binkley, 1999; Hartley, 2002; Evans and Turnbull, 2004; Friedman, 2006). The increase in productivity in certain areas would allow other areas to remain untouched for conservation. However, widespread afforestation in a particular region using fast-growing species could modify the local environment, resulting in deleterious consequences (Le Maitre et al., 2002; Kanowski et al., 2005; FAO, 2007). For water resources in particular, the increase in productivity by forest plantations is usually associated with increased water use (Farley et al., 2005; Nosetto et al., 2005). Afforestation is currently the most rapidly expanding land use activity in northern Patagonia, Argentina.

Patagonia, in southern South America, where my research site was located, has several similarities with the Pacific Northwest (PNW) of North America. The two regions are similar topographically. The Andes mountain range demarcates the western boundary of Patagonia in Argentina and creates a sharp longitudinal precipitation gradient, just as the Cascade Range does in the PNW. Dominant winds come from the west carrying humidity from the Pacific Ocean, leading to annual precipitations of 4000 mm in some places along the border between Argentina and Chile on the western slope of the Andes, and sharply decline to 300 mm per year on the extra Andean mesa only 100 km east of the highest peaks (Paruelo et al., 1998). The precipitation regime in both regions is Mediterranean, with more than 70% of the annual precipitation falling during their respective falls and winters. The severe precipitation gradient causes drastic differences in the vegetation types and productivity between the western and eastern slopes of the mountains. However, despite these many similarities, the species compositions and types of forests in both hemispheres are very different. In Patagonia, the western mountainous region is dominated by temperate evergreen broadleaf forests, the central region has mixed patches of conifer and broadleaf forests, and the dryer eastern region is contains primarily conifer forests that range from dense stands to savanna-type forests dominated by cordilleran cypress (*Austrocedrus chilensis* (D. Don) Pic. Serm. et Bizzarri). The boundaries between these forest types are irregular, following the orographic rainfall pattern, soil characteristics derived from glacial or volcanic activities, and fire regimes.

The distribution and structure of the vegetation in Patagonia has been largely influenced by the European colonization of the area. The use of fire to clear the woods and transform them into pasture and agricultural lands was the largest source of disturbances. Among the dominant tree species of the region, cordilleran cypress was affected the most by human-induced fires in the middle of the 19th century (Veblen et al., 1999). However, cordilleran cypress forests patches persisted in the rocky highlands and provided a seed source for recolonization (Kitzberger and Veblen, 1999). Episodic ENSO-related drought mortality events have also been reported to have influenced the distribution of cordilleran cypress in the area (Villalba and Veblen, 1998). Fire frequency declined in the early 20th century, but human exploitation of this species for its valuable wood, which was commonly used for construction in the area, continued. Despite the economic importance of cordilleran cypress and the changes threatening its survival within its own natural distribution area, most of the national parks and natural reserves were established in the western mountainous areas of Patagonia to protect the more emblematic humid *Nothofagus sp.* forests. Sheep grazing, now the most widespread agricultural use of areas within the natural distribution of cordilleran cypress, has largely prevented the reestablishment of cypress forests.

A new shift in land use is currently taking place in the forest-steppe ecotone; afforestation with exotic conifers from the PNW of the United States. In northern Patagonia, Argentina, exotic ponderosa pine (*Pinus ponderosa* Doug. ex. Laws) is being used in over three quarters of afforestation projects (SAGPyA, 1999; Gallo et al., 2005). Most of these forest plantations occupy areas where cordilleran cypress forests were previously displaced by fires and sheep ranches. In areas of the forest-steppe ecotone where grazing pressure was reduced, natural recolonization by cordilleran cypress can be observed, but is uncommon in areas where high density ponderosa pine plantations have been established. The wood productivity of ponderosa pine has been reported to be much higher than that of cordilleran cypress. While the wood productivity of cordilleran cypress stands under intensive management ranges from 4.5 to 11 m^3 ha^{-1} y^{-1} (Loguercio et al., 1999; Loguercio, 2005), that of managed ponderosa pine forests averages 20 m^3 ha^{-1} y^{-1} in the region (Gallo et al., 2005). Since both carbon fixation and transpiration are regulated by stomatal conductance, ponderosa pine stands are expected to transpire more water than cordilleran cypress stands. To date, only approximately five percent of the 2 million ha of land suitable for afforestation has been planted (SAGPyA, 1999). Understanding the interactions between ponderosa pine plantations and the natural environment is critical, before the exotic plantations become widespread, in order to prevent potential damages to the ecosystem.

In my dissertation, I focused primarily on the impacts of ponderosa pine plantations on water resources, which is a limiting resource in the region. Previous studies in the area (Gyenge et al., 2002; Nosetto et al., 2006) found greater water use by young ponderosa pine plantations compared to the native shrub/grasslands. In physiological comparisons among individual trees, Gyenge (2005) found that ponderosa pine trees had the capacity to transpire at higher rates than cordilleran cypress trees. However, no previous study had compared the water consumption of entire stands of exotic ponderosa pine with that of native forests during complete growing seasons. The research presented in this dissertation was designed to address several issues concerning the differences in water resource use between ponderosa pine plantations and native cordilleran cypress stands. Detailed measurements of stand structure, tree physiology and water fluxes throughout two growing seasons were taken to help us understand the interactions occurring between the stands and the environment and determine whether differences existed between the two stands that could lead to different total annual water use by each stand.

In the second chapter of this dissertation (the first data chapter) we took detailed measurements of soil water depletion at various depths and transpiration, to compare the water use of a high density ponderosa pine plantation stand (HDPP) and a pine-grass silvopastoral stand (Silvo; later called LDPP) against high and low density native cordilleran cypress forest stands (HDCip and LDCip, respectively) throughout the growing season.

The main hypothesis tested in this second chapter was that ponderosa pine plantations have higher total annual transpiration per hectare than native cordilleran cypress stands of similar density. As a secondary hypothesis, we investigated the potential mechanisms that would allow ponderosa pine stands to have higher transpiration. Specifically, we tested whether ponderosa pine stands: 1) have higher maximum transpiration rates, 2) utilize water over a longer season, 3) deplete soil water to a lower moisture content level, and 4) extract water from deeper soil layers than native cypress stands.

In the third chapter of the dissertation, we performed measurements of above and below canopy incoming precipitation in order to determine the contribution of interception losses to total evapotranspiration in the stands. The primary objective of this chapter was to test the hypothesis that rainfall interception was greater in native cordilleran cypress stands than in planted ponderosa pine forests with similar leaf areas. We also investigated the effect of the combination of transpiration and interception loss on water balance in these two forests.

In the fourth chapter, we used structural and physiological measurements from the same forest stands as in the previous chapters to parameterize a detailed process-based model. The model parameterization was calibrated and validated using each of the two separate growing seasons of transpiration measurements. The fully parameterized model was then used to assess the hydrological sustainability of the four forest stands under current climatic conditions. Hydrological sustainability, as defined in this study, occurred when total annual evapotranspiration was lower than total annual precipitation. The second objective of this chapter was to use the parameterized model to predict the potential impact of future climate scenarios on seasonal and annual water fluxes in all four forest stands.

CHAPTER TWO: INCREASED WATER USE BY PONDEROSA PINE PLANTATIONS IN NORTHWESTERN PATAGONIA, ARGENTINA COMPARED WITH NATIVE FOREST VEGETATION

Julian A. Licata[1*], Javier E. Gyenge[2], Maria Elena Fernandez[2], Tomás M. Schlichter[2] and Barbara J. Bond[1]

([1]) Department of Forest Science, Oregon State University, 321 Richardson Hall, Corvallis, OR 97331, USA.

([2]) INTA EEA Bariloche, Grupo de Ecologia Forestal, CC 277 (8400) Bariloche, Argentina.

Forest Ecology and Management

www.elsevier.com/locate/foreco

Article in press (2007)

Abstract

Ponderosa pine (*Pinus ponderosa* Doug. ex. Laws) plantations represent more than three quarters of afforestation projects in the Northwestern part of Argentinean Patagonia. Most plantations are located in the forest-steppe ecotone within the natural distribution area of cordilleran cypress (*Austrocedrus chilensis (D. Don)* Pic. Serm. et Bizzarri). The productivity of ponderosa pine plantations is much greater than that of the native cordilleran cypress forests, which would suggest a greater utilization of water resources. Our primary hypothesis was that ponderosa pine plantations have higher total annual transpiration per hectare than native cordilleran cypress stands of similar density. As secondary hypotheses we tested whether ponderosa pine stands: 1) have higher maximum transpiration rates, 2) utilize water over a longer season, 3) deplete soil water to a lower moisture content level, and 4) extract water from deeper soil layers than native cypress stands. Through continuous measurement of sap flow and of soil water content at various depths (from surface to 1.8 m), we examined the total amount of water used by four forest stands: a high density ponderosa pine timber plantation (HDPP), a low density silvopastoral ponderosa pine plantation (Silvo), a low density cordilleran cypress forest stand (LDCip) and a high density cordilleran cypress forest stand (HDCip). Measurements were carried out during two years that had contrasting amounts of summer precipitation. HDPP had the highest water use at the stand level, averaging 64 % and 33% greater than HDCip in the wet and dry years, respectively. Higher transpiration rate but not length of the transpiration season explained the difference in total annual transpiration between species. Water depletion

occurred simultaneously at all soil depths for all the plots even in the wet year. However, the HDPP plot used a greater amount of water from deeper soil layers compared to all the other plots. Our results emphasize the need for caution when planning afforestation projects at large scales, as the increase in transpiration due to conversion of native forests to high density ponderosa pine plantations could have a large impact on water resources.

Introduction

The increase in global demand for paper and wood products is increasing pressure on natural forests. Afforestation with fast-growing species has been proposed as a sustainable economic alternative that would reduce the harvest of native forests (Sedjo and Botkin, 1997; Binkley, 1999; Hartley, 2002; Friedman, 2006). Worldwide, increased wood productivity is usually achieved by planting exotic species (Sedjo, 1999; Hartley, 2002; Siry et al., 2005; Fritzsche et al., 2006). However, the introduction of exotic species could lead to a complex array of negative consequences (Le Maitre et al., 2002; Kanowski et al., 2005; Nosetto et al., 2005; FAO, 2007). In semi-arid regions, for example, stands of fast-growing exotic trees may deplete already-scarce water resources. The eastern slope of the Andes in Patagonia serves as the water source for vast regions of Argentina, and large-scale afforestation efforts in this region have the potential to drastically alter regional water resource availability. There are currently ~70,000 ha of planted forests in northwestern Argentinean Patagonia, with an additional ~2,200,000 ha that could be converted to plantation forestry (Laclau and Andenmatten, 2005). Understanding the potential effects of exotic pine plantations on water resources is critical in Patagonia to prevent potential negative consequences from the expansion of planted forests.

The Andean region of Patagonia in South America is geographically and climatically very similar to the Pacific Northwest (PNW) of North America, and most of the exotic plantations in this region utilize species from the PNW. The Andes range creates an exponential decrease in annual precipitation from west to east, which is the predominant direction of the strong winds that are typical of Patagonia (Paruelo et al., 1998). The western humid forests of the region are mostly protected in large national parks established by the Administration of National Parks of Argentina. Therefore, most of the plantations are located outside these parks in the drier forest-steppe ecotone. As a result, around 80% of the afforestation projects have used ponderosa pine, which is well adapted to dry environments like those found in the forest-steppe ecotone of Patagonia (Gallo et al., 2005).

The distribution and structure of the forest-steppe ecotone has been heavily influenced by fire disturbances (Mermoz et al., 2005). In particular, the use of fire by Native American hunters and, later, European settlers, to transform woodlands into pasture and agricultural lands has shaped the current distribution of cordilleran cypress (Veblen et al., 1999; Kitzberger and Veblen, 1999), the dominant native tree species in the area. Most ponderosa pine plantations occupy areas where cordilleran cypress forests were previously displaced by fires and sheep ranches. Natural recolonization by the cordilleran cypress is occurring in large areas of the forest-steppe ecotone, but it is uncommon in areas where high density ponderosa pine plantations have been established.

The productivity of ponderosa pine plantations is much greater than the productivity of the native vegetation they replace (Schlichter and Laclau, 1998). Wood productivity of native cordilleran cypress on intensively managed stands ranges from 4.5 to 11 m^3 ha^{-1} y^{-1} (Loguercio et al. 1999; Loguercio et al. 2005). In contrast, annual wood productivity of ponderosa pine plantations in the same region averages 20 m^3 ha^{-1} y^{-1} (Gallo et al., 2005), and has been reported to be as high as 40 m^3 ha^{-1} y^{-1} (Girardin and Broquen, 1995). The higher growth rates of ponderosa pine plantations compared with native cordilleran cypress in Patagonia suggests greater water use by the pines. In forest ecosystems, water transfer from soil to the atmosphere is mainly mediated by vegetation (Unsworth et al., 2004). Therefore, an increase in transpiration due to a change in land use could lead to a decrease in available water for other uses downstream.

Higher consumption of water by tree plantations compared to natural vegetation has been reported in other parts of the world (Vertessy and Bessard, 1999; Vertessy et al., 2002; Farley et al., 2005; Nosetto et al., 2005). Changes in groundwater level have also been observed following modification of forest cover (Xu et al., 2002; Pothier et al. 2003; Jackson et al. 2005; Jutras et al. 2006). Based on soil water measurements, Gyenge et al. (2002) estimated that young ponderosa pine plantations in Patagonia used about 90 mm more water through the growing season than the natural vegetation on the non-forested adjacent steppe. Physiological comparisons concerning water use among individual trees of ponderosa pine and cordilleran cypress have been done in the same study area (Gyenge et al., 2003; Gyenge, 2005); however, no previous study has compared the consumption of water by forest stands of exotic ponderosa pine with native forests.

Although ponderosa pine plantations are mainly planted in dense stands solely for the purposes of wood production, we also evaluated a low density ponderosa pine plantation, which had been studied previously to assess the feasibility of silvo-pastoral systems in the region (Fernandez et al., 2002; Gyenge et al, 2002). Silvo-pastoral systems are not well developed in the region, but have a large potential for acceptance by landowners who traditionally work with grazing systems.

The main objective of this study was to compare the water use of a high density ponderosa pine plantation stand (HDPP) and a pine-grass silvopastoral stand (Silvo) against high and low density native cordilleran cypress forest stands (HDCip and LDCip, respectively) in Andean Patagonia through the growing season. The main hypothesis we tested was that ponderosa pine plantations have higher total annual transpiration per hectare than native cordilleran cypress stands of similar density. As secondary hypotheses, we tested potential mechanisms that would allow ponderosa pine stands to have higher transpiration. Specifically, we tested whether ponderosa pine stands: 1) have higher maximum transpiration rates, 2) utilize water over a longer season, 3) deplete soil water to a lower moisture content level, and 4) extract water from deeper soil layers than native cypress stands.

Methods

Study site

The study was carried out over two growing seasons (from October 2004 to April 2006) in Estancia Lemú Cuyén, Valle de Meliquina (40.29º S, 71.13º W), in Lanín National Park, Patagonia, Argentina. The precipitation regime is of Mediterranean type with approximately 80% of the annual precipitation occurring during fall and winter, mainly as rainfall.

Four plots (approximately 0.25 ha each) were installed in an alluvial plane of the Caleufu River, at an altitude of 810 m. Soils are deep with a sandy loam texture, with pebbles and small rocks (mixed with the sediments) increasing in abundance with depth. Three plots were installed the first year (2004-2005), and were comprised of a dense stand of 25-year-old ponderosa pine plantation (HDPP), a 25-year-old silvo-pastoral managed stand of planted ponderosa pine with grasses (Silvo), and a natural pure cordilleran cypress stand (HDCip). The second year (2005-2006), a lower density cordilleran cypress plot (LDCip) was established on a gentle slope at the base of a hill, which aimed to be more representative of the native forest stands that occupy the hillslopes of the forest-steppe ecotone of the region. In the first year, measurements started on December 1st of 2004, and ended on March 15th of 2005, while in the second year measurements were started on October 15th of 2005, and ended on April 15th of 2006. Tree densities were 1135, 2662, 350 and 668 trees per hectare for HDPP, HDCip, Silvo and LDCip, respectively. Bifurcated stems were considered individual trees. Average dominant tree height in all the plots was approximately 15 m. The main criterion used to select the native forest plots (HDCip and LDCip) was the similarity of environmental conditions to the plots with planted pines. Thus, before deploying the measuring devices, we visually assessed the similarity of the leaf area, height, and soil type of the native forest plots to the ponderosa pine plots. Moreover, plots had to be located within a radius of 500 m from the ponderosa pine plots.

Due to the intensive measurements and the nature of the equipment required for this study (e.g., measurement systems for sapflow and soil water content requiring a central energy source and dataloggers), it was not possible to replicate the treatments. This was a case study, designed to provide detailed, stand-level measurements of water use in native and exotic pine stands at high and low densities.

Meteorological data

Meteorological data were acquired from a permanent meteorological station installed 7 km from the study plots. A portable meteorological station was located next to the permanent station to calibrate sensors and then in an open grassland near the study plots for 15 days during two different periods to check for possible discrepancies in the meteorological conditions between the two locations. The permanent station was a personalized Campbell Scientific (Logan, UT, USA) weather station with sensors for air temperature and relative humidity (Vaisala, CS500) at 1.5 m high, wind speed and direction (RM Young Wind Sentry, Model 03001-5) at 2 m, precipitation (TE525MM Tipping bucket rain gauge) at 0.4 m, solar radiation (LI200X pyranometer) at 2 m high, and soil temperature (107L temperature probe) at 0.05 m depth. The portable station had sensors for air temperature and relative humidity

(Vaisala, HMP45A) at 1.5 m high, wind speed (Met One Instruments, Grants Pass, OR, USA, model 014A-L) at 2 m high, photosynthetically active radiation (Li-190SA) at 2 m high, and precipitation (TE525MM Tipping bucket rain gauge) at 0.4 m. Sensor output was recorded every 15 sec and averaged every 30 min with a Campbell CR10X datalogger. Long term data (23 years of monthly precipitation averages) were obtained from a meteorological station (70 km south of the study site) at the Bariloche airport, managed by the Argentinean National Meteorological Service.

Leaf Area Index

The leaf area index (LAI) of all the plots was calculated using the total leaf biomass and the specific leaf area (SLA) of total leaf area for each of the plots. The SLA of cordilleran cypress plots was measured from 20 samples of foliage collected from full sun and shaded areas of the canopy, from 10 different trees at both cordilleran cypress plots. Leaf areas were determined using ImageJ software (Rasband, 2006) on digital images of the green foliage. SLA of ponderosa pine was obtained from Gyenge (2005), who measured that parameter in the same plots for a previous study. The mass of the dried samples was determined using a 0.001g precision balance. Total leaf biomass was calculated applying allometric equations described by Laclau (2003) for every tree in the plots that had DBH larger than 5 cm. Those equations were developed using trees of both species from different stands around the region, which included plots at the same location of this study.

Transpiration

We measured sap flow continuously using 20 mm constant power thermal dissipation probes (Granier, 1987). Sap flow sensors were installed in 15, 18, 18 and 24 trees, randomly selected, in the Silvo, HDPP, LDCip and HDCip plots, respectively. The sampled trees represented approximately 50 percent of the sapwood basal area in the plots. In eight trees of the HDCip plot that had stems of irregular circumference, multiple sensors per tree were installed to check for differences in sap flow density between the different parts of the stem rugosity (flat, convex and concave positions). There were no consistent differences in sap flow density among the flat, convex and concave positions of the stem surface. All sensors were installed in the outermost part of the xylem at 1.4 m height, on the southern side of the stems. All sensors were replaced between the two measurement years and new sensors were installed in different trees or at different positions in the same tree if no other trees were available in the plot. At least 10 cm of vertical and horizontal distance was left between old and new sensor installation placements. A ring of thermal and radiation insulation (Mexpol®, 9mm) was placed around the stems, covering the sensors and at least 20 cm above and below them, to avoid a thermal gradient effect on the stems. Sensor output was measured every 15 sec and averaged every 30 min with a Campbell CR10X data logger and AM416 multiplexer. These data were transformed to sap flow density on a sapwood area basis following Granier (1987).

Previous studies involving several tree species found higher sap flow density in the outer part of the sapwood area than in the zone closer to the heartwood (e.g. Cermák and Nadezhdina, 1998; Jiménez et al., 2000; Irvine et al., 2002; Nadezhdina et al., 2002; Delzon et al., 2004; Ford et al., 2004). Accounting for this decline in sap flow density across the stem is particularly important for ponderosa pine because it has a large sapwood depth, usually more than five times the depth of the sap flow sensors in the trees of this study. Therefore, at the end of both seasons (beginning in January 16th in 2005, and in March 17th in 2006, for a minimum of 15 days) sapflow sensors were installed at several depths in four trees at both the HDPP and the Silvo plots. Sensors were installed in depth by drilling a 16 mm diameter hole until the desired starting depth to make the measurement. Initially, sensors were installed at increasing depth in a spiral up the stem, incrementing the depth between each sensor by 1cm (sensors depths were: 0-2; 3-5; 6-8; 9-11; 12-15 cm). This pattern of sensor installation proved inappropriate due to heat build up during the night. Heat interference between sensors was verified by the increase in temperature during the night in unheated sensors. To remove heat interference among sensors, only three sensors per tree were left heated. In the second year only two or three sensors were installed per tree, depending on the sapwood depth, in four other ponderosa pine trees.

Similar to other authors' findings (Ford et al., 2004; Fiora and Cescatti, 2006) the ratio between the instantaneous sap flow densities of the outmost and inner sensors was variable among hours of the day and trees. To reduce this variation, ratios between the outermost sensor and the inner sensors were calculated using the daily values from each sensor. The position of the sap flow sensor was normalized by the total sapwood depth to make comparable the ratios between sensors in trees of different sizes. Although the best fit was given by a Gaussian function, it improved the Pearson's r^2 by only 3% compared to a simple linear equation. Therefore, for simplicity a linear decrease function was used to simulate the radial variation of sap flux density across the stem (Figure 2.1). Sapwood depth in the cordilleran cypress trees is much shallower than in ponderosa pine, rarely exceeding four centimeters. Consequently, the amount of error that could be produced by not accounting for the radial decline in sap flow density is much lower in cypress than pine. The same relative radial decline function was used to simulate a decrease in sap flow density in cordilleran cypress stems as in ponderosa pine stems.

Sapwood depth was measured by taking increment cores from 30 trees of each species growing adjacent to the study plots. Species-specific allometric equations derived from those measurements were used to calculate the sapwood area of trees inside the plots. Sapwood area per ground area was calculated through measurement of the diameter at breast height (DBH) of all the trees in the plots.

By integrating the sap flux density across the stem in a 1 cm increase in sapwood depth, the following function was derived to calculate a radial correction multiplier, depending solely on the stem radius.

$$\beta= \frac{\Sigma\, [(r-r_i)^2 - (r-r_{i+1})^2] * (r_i - r_h)/(r_0-r_h)}{(r^2-r_h)^2}$$

where r is the radius of the cambium from the pith; r_h is the radius of the heartwood from the pith; r_o is the radius of the outmost sap flow sensor; r_i is the radius from the pith of the i iteration.

The following equation was used for scaling transpiration to the stand level:

$$E = \bar{\beta}\rho_w * \bar{J} * SAI$$

where E is transpiration at the stand level; $\bar{\beta}$ is the radial correction multiplier for the average DBH of the stand; ρ_w is the density of water; $\bar{J}$ is the average sapflow density of the stand; SAI is the sapwood area per ground area.

The lengths of the measurement periods were different in the two years; the second year measurements were started a month and a half earlier than the first year and ended a month later. Therefore, in order to compare total seasonal transpiration between the two years, estimates of total transpiration were standardized to a 180-day period from the middle of the spring to the middle of the fall (the "transpiration season"). All the daily values throughout this period of time were added to get the total transpiration for each season. Missing daily values within the measurement period (<2% of the data) were assumed to be equal to the 30-day average around the day of the missing data; missing daily values for the beginning and the end of the first year, when there were no measurements available, were assumed to be equal to the average of the first and last 15 days of measured data from that same year, respectively. Data from the second year of measurements suggested that this was a reasonable assumption.

Soil water depletion

Multiple sensor, frequency domain capacitance probes (Sentek EnviroSMART, Sentek Pty Ltd., Adelaide, Australia) were used to continuously monitor volumetric soil water content; data were recorded every 15 seconds and 30-minute averages were stored in a CR10X Campbell Scientific datalogger. The Sentek measurement system employs long probes, which are inserted into 5.5 cm diameter PVC casings; the casings, in turn, are installed in holes augured into the soil. Our probes were 2 m long, with six sensors per probe at 10, 20, 40, 80, 140, and 180 cm depths. In the first year, we used four probes randomly distributed in each of the three plots. In the second year, we used three probes in each of the four plots. In the LDCip plot we could not use the 180 cm sensor due to shallower soils.

Daily average soil water content was determined for each sensor, and soil water depletion was calculated as the difference between successive days. We calculated the total volumetric water content through the 2 m depth of each probe assuming a linear gradient of soil water content between sensors.

Gravimetric measurements of soil water content were carried out during the second season to perform a site-specific calibration of the Sentek EnviroSMART sensors. A total of 642 soil samples were collected with an auger at approximately 60 cm from the probes, at intervals of 20 cm up to a depth of 160 cm, every two to three weeks throughout the second year of measurements. Samples were dried at 105 ^{o}C. The calibration equation used was:

$$WC_{ss} = 1.0752 * WC_{stk} + 4.766,\ r^2=0.7297,\ P>0.001.$$

where WC_{ss} is the percent of volumetric water content derived from soil samples; WC_{stk} is the water content reported by the Sentek sensors using the manufacturer’s calibration.

Reference Evapotranspiration

The FAO Penman-Monteith method was used to calculate values of “reference evapotranspiration” (ET_{ref}) following the procedure detailed in FAO Irrigation and Drainage Paper 56 (Allen et al., 1998) for daily climatic data. ET_{ref} represents the maximum potential evapotranspiration of a hypothetical reference crop, with no water deficiency, growing under the same meteorological conditions as measured in the field site.

Results

Stand structure was markedly different between the high and low density plots and relatively similar between plots of similar density (Figure 2.2). LAI was approximately doubled in the high density plots (9.1 and 9.0 for HDCip and HDPP, respectively) compared to the low density plots (4.6 and 3.1 for LDCip and Silvo, respectively; Figure 2.2). Basal area was 97, 65.5, 50.7 and 27.5 m^2 ha^{-1} for HDCip, HDPP, LDCip and Silvo, respectively, while sapwood area was 43.7, 59.7, 21.7 and 25.9 for HDCip, HDPP, LDCip and Silvo, respectively, resulting in a lower leaf area per sapwood area in ponderosa pine (0.15 and 0.12 m^2 cm^{-2} for HDPP and Silvo, respectively) than in cordilleran cypress plots (0.21 m^2 cm^{-2} for both HDCip and LDCip). For any particular measure of stand density, high density plots were about twice the value of the low density plot for the same species. Comparability of densities between species was based on the similarity of LAI. However, high and low density is used throughout the manuscript as a qualitative descriptor of the plots and not as a variable causal of response.

Precipitation was different between the two growing seasons. While the total cumulative precipitation from June to July was only 101.3 mm for the two years of measurements, summer precipitation between November and March was 154 mm greater in the second year (2005-2006) (Figure 2.3). Moreover, the first (dry) and second (wet) seasons of measurements correspond with the lower and upper limits of 1 standard deviation of the 23-year mean summer precipitation.

The HDPP plot had the highest total cumulative transpiration of all the plots in both years. Total estimated transpiration for the dry year (2004-2005) was 479.4, 361.6 and 332.9 mm for HDPP, HDCip and Silvo, respectively, compared with 901.6 mm for ET_{ref}. Total estimated transpiration for the wet year (2005-2006) was 657.8, 402.3, 394.8, 252.7 mm for HDPP, HDCip, Silvo and LDCip, respectively, compared with 851.7 mm for ET_{ref}.

As a consequence of more cloudy, rainy and cooler weather the second year, ET_{ref} decreased by 6% between the first and second year. However, all of the plots showed a notable increase in total transpiration in the second year. Total transpiration increased by 37%, 11% and 19% for the HDPP, HDCip and Silvo plots, respectively, between years one and two, thus enhancing the difference in transpiration between the HDPP and the rest of the plots.

During both years, ET_{ref} was greater than the actual transpiration rate for all the sites throughout the whole season (Figure 2.4). During the dry year, the daily transpiration rate of HDPP was greater than HDCip plots for the first two thirds of the measurement period. The Silvo plot had lower daily transpiration rates than HDPP and HDCip until the middle of the summer. Over this time, transpiration rates in the HDPP and HDCip plots steadily declined, while the transpiration rates of the Silvo plot remained relatively unchanged. In early February, transpiration rates of all plots were fairly similar (Figure 2.4); thereafter, the Silvo plot had higher transpiration rates than both HDPP and HDCip plots until the beginning of the fall rains in March.

In the wet year, HDPP had the highest daily transpiration rates throughout the whole growing season, and the LDCip site had the lowest. The Silvo and HDCip plots showed a pattern of seasonal transpiration similar to the previous year, but the Silvo plot transpiration started to be higher than the HDCip 10 days later than the previous year (Figure 2.4).

To compare the length of the “transpiration season” among plots, we expressed transpiration rates as a percent of the daily maximum of each season for each site, and arbitrarily defined as the end of the season whenever the daily values dropped and remained below 30% of the season’s maximum. There were no consistent differences in length of the transpiration season among sites of similar densities using this criterion. While in the dry year the transpiration season was 15 days shorter at HDPP than at HDCip, in the wet year the season was 10 days longer at HDPP than at HDCip (Figure 2.5). In both years the Silvo plot had a relatively high and steady rate of transpiration despite the summer drought period, while HDPP and HDCip rates were consistently decreasing. No differences were observed in the length of the season between the Silvo and the LDCip plots.

Despite the high frequency of rains in the wintertime, at the beginning of measurement periods of both years there was less total soil water to 1.8 m in HDPP than in all the other plots (Figure 2.6). The lower soil water content in HDPP persisted for the whole season in both years. This could be caused by incomplete refilling of the soil during the winter of the water used by the HDPP in the previous year, by greater water use in springtime before the measurement period, or by a lower water holding capacity of the soil. The latter is less likely to have occurred given the proximity of the plots and similarity in parent material of the soils. For all plots, the amount of soil water available at the beginning of the measurements period in the first (dry) year averaged 100 mm lower than in the second (wet) year.

Water depletion was observed simultaneously at all measured depths for all sites in both years (Figure 2.7); the amount of water withdrawn from deeper layers increased as the water content in upper layers decreased. However, water was depleted from deep layers of the soil even when there was water available near the surface. From the initiation of measurements until the first rain event in the dry year, more than 60% of the depleted water was supplied by the soil layers below 80 cm. In contrast, in the wet year more than 60% of the total amount of used water came from the upper 80 cm.

The soil water content measurements showed that rainfall was the only significant input of water into these systems, which allowed us to perform a mass balance analysis to compare water depletion from the soils with water use by trees, estimated from sapflow measurements. In general, the measured amount of water lost from soils to 180 cm depth exceeded the amount of water transpired by the trees (Figure 2.8). This is because our measurements did not account for all of the pathways of water leaving the soil, such as deep seepage, soil evaporation or transpiration by herbaceous plants. On the other hand, in HDPP there was an apparent net negative balance between soil water depletion and transpiration in both growing seasons (Figure 2.8). The unbalanced water budget in HDPP could be explained by a higher extraction of water below 1.8 m deep by deeper roots in the HDPP plot. The average daily differences between soil water depletion and sap flow were -0.98, 0.33 and 0.16 mm d^{-1} for HDPP, HDCip and Silvo, respectively, in the first year, and -0.44, 0.48, 0.65, -0.25 for HDPP, HDCip, Silvo and LDCip, respectively in the second year. In the early season of both years in Silvo, soil water depletion exceeded tree water use (Figure 2.8), most likely due to transpiration of the grasses and evaporation from the soil.

Discussion

In an effort to reduce the net loss of forest area, Argentina has been actively promoting forestry plantations with exotic fast growing species through subsidies and tax exemption policies (Laclau, 2003). The increase in productivity by the introduction of fast-growing species occurs at the cost of water resources in many situations (Van Wilgen et al., 1996; Vertessy and Bessard, 1999; Zhou et al., 2002; Jackson et al., 2005; Nosetto et al., 2005). This cost is particularly important for places where water is a limiting resource, such as in Mediterranean ecosystems like the forest-steppe ecotone of Argentinean Patagonia.

In Northern Argentinean Patagonia, ponderosa pine, which is native to the western states of the United States of America, is used in over three quarters of afforestation projects (Gallo et al., 2005). Ponderosa pine plantations have approximately 4 times the productivity of native cordilleran cypress growing in optimal conditions (Girardin and Broquen, 1995; Loguercio et al., 2005). As carbon uptake and water loss by plants occur mainly through the same pores, the stomata, we hypothesized that transpiration of ponderosa pine would be substantially greater than that of cordilleran cypress growing under similar conditions.

As expected, our measurements showed greater use of water resources throughout the year by ponderosa pine compared to the native cordilleran cypress plots growing under similar conditions and comparable levels of stand density. Differences in transpiration among stands were enhanced during the year with higher precipitation during the summer, when all the plots followed a seasonal transpiration pattern similar to a crop with no water limitations. While in the dry year HDPP had 33% greater total transpiration than the HDCip, that difference was almost doubled (64%) during the year with relatively moist summer conditions. As potential mechanisms for an increase in total annual water use, we tested the hypotheses that ponderosa pine had higher transpiration rates than cordilleran cypress, and/or that it utilizes water over a longer season. Either of these hypotheses implies that ponderosa pine is able to utilize soil water resources that are not used by native forests. Alternative possibilities are that ponderosa pine could access water from deeper soil layers, and/or that it could deplete soil water content to lower levels of soil moisture.

Transpiration rates

The greater total water use by ponderosa pine plantations compared to native cypress stands was associated with higher maximum transpiration rates throughout the transpiration seasons. Moreover, at comparable levels of stand density, transpiration rates of ponderosa pine were always higher than that of cordilleran cypress, suggesting a higher maximum stomatal conductance of ponderosa pine trees given the similar LAI for pine and cypress forests at comparable levels of stand density.

Stand-level maximum transpiration rates in the HDPP plot of this study were from 1.6 to five times greater than the values reported for ponderosa pine forests in North America (Anthoni et al., 1999; Ryan et al., 2000; Law et al., 2001; Irvine et al., 2002; Kurpius et al., 2003; Irvine et al., 2004; Simonin et al., 2007).

The increase in transpiration rates between the dry and the wet year were noticeably greater in the high density pine compared to the high density cypress (37%

and 11% increase, respectively), suggesting that it was capable of utilizing water resources at a faster rate when they were available, and that it was more sensitive to drought than the native forest. This could imply larger interannual fluctuations in productivity of ponderosa pine plantations compared to native cordilleran cypress forests. The differences in measured transpiration between the two years may be explained by a much larger depletion of water from the surface soil layers during the early to middle spring of the dry year (2004), even before our measurements were initiated. The lack of rainfall throughout the growing season kept the upper soil dry. This result is consistent with results reported by Warren et al. (2005), who found that the amount of water used by ponderosa pine was strongly dependent on water potential at 20 cm of soil depth, even when there was abundant water available to deeper roots. A potential explanation for this pattern is that the early depletion of soil water content during the dry year could have led to a change in soil-root resistance through the remainder of the season. It is well known that soil water conductivity decreases exponentially with water content depletion (Saxton et al., 1986; Brandyk et al. 1989). Furthermore, as the upper soil layers become depleted of water, the active root surface area to leaf area ratio also decreases, thus increasing the total resistance of the water pathway.

Length of the transpiration season

In a Mediterranean climate, where summer precipitation is low, the length of the transpiration season is often determined by the total amount of soil water that is available in the soil. It is possible for two species with similar rates of stand-level transpiration to have different annual water usage if one of them can sustain transpiration longer into the summer drought period. Similarly, it is possible for two species with different maximum transpiration rates to have similar annual water usage. A possible consequence of high maximum transpiration rates, such as in the dense ponderosa pine plantation of our study, might be a shortened period of active transpiration, which would ultimately result in similar total water use by the different stands, only distributed differently over time. However, for the pines to acquire more water over a similar length of season as the native trees would require an ability to extract water either at lower tensions or from greater depths.

Our results showed that the length of the transpiration season was similar for the high density ponderosa pine and native cordilleran cypress plot. The greater transpiration by pines was not simply a result of a longer period of transpiration. Also, there was no evidence of a tradeoff between high transpiration rates and a shorter season of transpiration. The ponderosa pine trees must have been able to extract more water from the soil throughout the season.

We did find different lengths of transpiration seasons between the high density pine and the silvopastoral plots during the dry year, when the HDPP showed signs of water stress and the Silvo did not. Moreover, although the silvopastoral and high density cypress plots used approximately the same total amount of water throughout the season, the silvopastoral plot followed temporal trend similar to that of a crop with no water limitations (ET_{ref}) throughout the summer, while the high density cypress stand sharply reduced its rate of transpiration in the second half of the summer. This pattern of sustained transpiration in the silvopastoral plot throughout the drought season is consistent with a previous study, which showed that ponderosa pine trees growing in low density plantations in Patagonia are able to sustain consistently high transpiration rates from spring to autumn (Gyenge et al., 2003). During the second year of measurements when water was more plentiful, all stands followed the ET_{ref} temporal trend more closely.

Access of water from soil layers

As the input of water to these ecosystems is mainly through precipitation during the winter and fall, a higher total annual transpiration can imply greater extraction of water from soil reserves. We examined two hypotheses (not mutually exclusive) as possible strategies for ponderosa pine to withdraw more water from the soil than cordilleran cypress: deplete soil water content to lower levels of soil moisture; or extract water from deeper soil layers.

Concerning the first strategy, the soil water content data gave us contradictory results. Although the high density pine plantation ended up with a lower level of soil water content in both years than the high density cypress, the low density silvopastoral plot had a higher level of soil water content than the low density cypress at the end of the season. Our results suggest that ponderosa pine are indeed able to deplete water to a lower soil water content level than the native cordilleran cypresses, but that the available soil water was sufficient to last throughout the season for the pines in the silvopastoral system.

To investigate the second hypothesized strategy, we measured the soil water content at six depths continuously over the two growth years, to a maximum depth of

1.8 m. The relative contribution of each soil stratum to daily total water depletion did not show any conclusive pattern of differential water use by depth per species, suggesting that both species could access water up to a 1.8 m of depth.

However, a mass balance examination of the water cycle in our study plots suggests another interpretation. In the two cypress plots there was a close match between estimated depletion of water from the soil and total use of water by trees. However, in the high density pine plot, transpiration estimated from sapflow greatly exceeded the soil water content depletion estimated from measurements in the 1.8 m soil profile. This difference between the soil water depletion and sap flow estimates suggests that approximately 33% in the dry year and 24% in the rainy year of the total annual water consumed by trees in the high density pine plot was extracted by deep roots below 1.8 m (the maximum measured in this study). In the case of the silvopastoral plot, the reverse situation occurred at the beginning of both seasons; soil water content depletion exceeded estimated water use by trees. This is likely due to evapotranspiration from soil and grass. If true, the partitioning of water resources in the Silvo plot over the season would amount to 84% of transpiration by trees and 16% of evapotranspiration from soil and grasses. It is also possible that deep drainage could have accounted for some additional soil water losses in the silvopastoral plot.

The mass balance approach supports the hypothesis that ponderosa pine can access soil water from deeper soil layers than can cordilleran cypress. Schulze et al. (1996) in a study along a precipitation gradient in Patagonia also found that although native woody species had deep roots, they did not find use of water from deep soil depths. Despite the fact that their conclusions were derived from a period of time when water was available at the upper layers of the soil, it suggests the possibility that native woody vegetation in Patagonia makes limited use of deep soil water resources.

Management implications

Despite the limited scale and scope of this case study, our results emphasize a need for caution when planning afforestation projects at large scales. The management practices used in the high density ponderosa pine plantation plot in our study are representative of those used in ponderosa pine plantations throughout Northwestern Patagonia, Argentina. If the differences in transpiration between treatments found in our study were extrapolated across a large region, the increase in evapotranspiration due to conversion of native forests to high density ponderosa pine plantations could have a large impact on water resources.

Exports of groundwater from this region are particularly important for large areas of Argentina that get water from the rivers originating in this mountainous area. The dominant winds in this region come from the west, carrying humidity from the Pacific Ocean. The rain shadow effect of the Andes causes a dramatic precipitation gradient from west to east, with annual precipitation of 4000 mm at some spots at the border between Argentina and Chile declining to 300 mm at the extra Andean mesa only 100 km away from the highest peaks. Precipitation remains consistently low across the mesa to the Atlantic coast, where most of the excess water from the Argentinean Patagonian Andes is discharged. Along this long path, water from the Andes provides critical services for the rest of the country, such as hydropower, which provides approximately 32% of the power consumption of the country, and irrigation of around 130,000 ha of agricultural land in the provinces of Neuquén and Rio Negro (Fiorentino, 2005).

At the watershed scale, runoff water from forested areas is particularly important for the traditional long-established activity of sheep grazing (Paruelo et al., 2000; Nosetto et al., 2006). The most valuable and productive areas in the region for grass production are wet meadows called "mallines" (Golluscio et al., 1998; Paruelo et al., 2000) usually located in the bottom of the valleys where there is an input of groundwater from the uplands (Gandullo and Schmid, 2001; Nosetto et al., 2006).

This study highlights the potential to reduce the consumption of water by ponderosa pine plantations through management of stand density at least up to levels that native cordilleran cypress may achieve in the region. Further studies are needed to assess potential management practices that would lead to a sustained reduction in leaf area along the rotation period. Moreover, it would be critical to explore the potential change in water use efficiency with stand density in order to be able to optimize the land use for both wood production and water conservation. Furthermore, although this study focused on estimates of evapotranspiration through sap flow and soil water depletion, there are other aspects of the hydrologic cycle that could be different between the native cordilleran cypress forests and the fast growing exotic conifers. We are conducting parallel studies to elucidate possible differences in rain interception and stemflow dynamics by the different types of stands, what will help to close the water budget for the different systems.

Figures

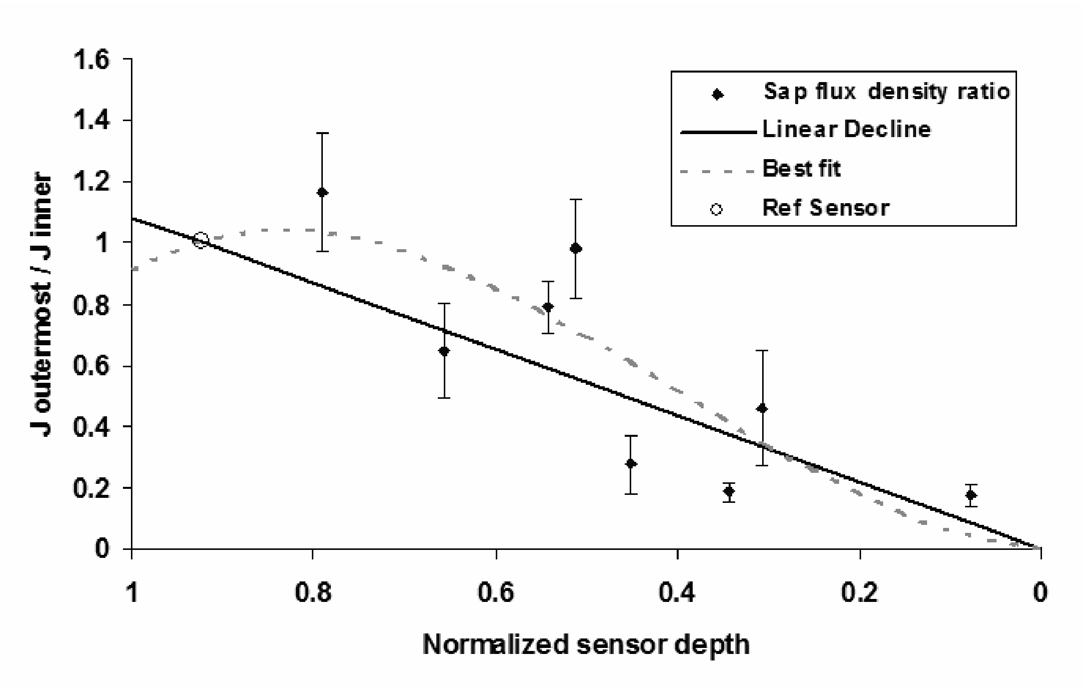


Figure 2.1 - Radial decline of sap flow density across the stem. X axis represents the relative distance from the heartwood (x = 0) to the cambium (x = 1). Y axis represents the daily sap flow density ratio of inner sensors relative to outermost sensor in the same tree. Each data point is a fifteen days average. Error bars represent standard errors. Dotted line represents the best fit function tested, and solid line represents the linear decline used in this study. Open circle represents the average position of the outermost sensors used as references.

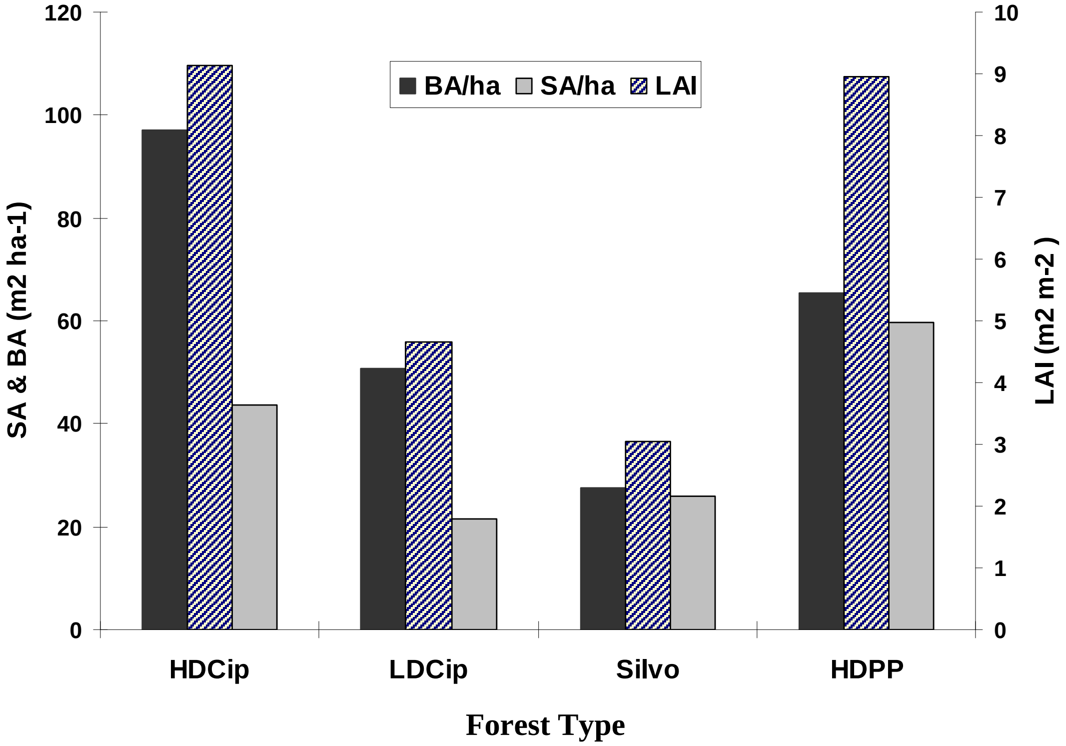


Figure 2.2 - Stand Density. Basal area (BA m^2 ha^{-1}), sapwood area (SA m^2 ha^{-1}) and Leaf Area Index (LAI m^2 total leaf area m^{-2} ground area) for the measured plots.

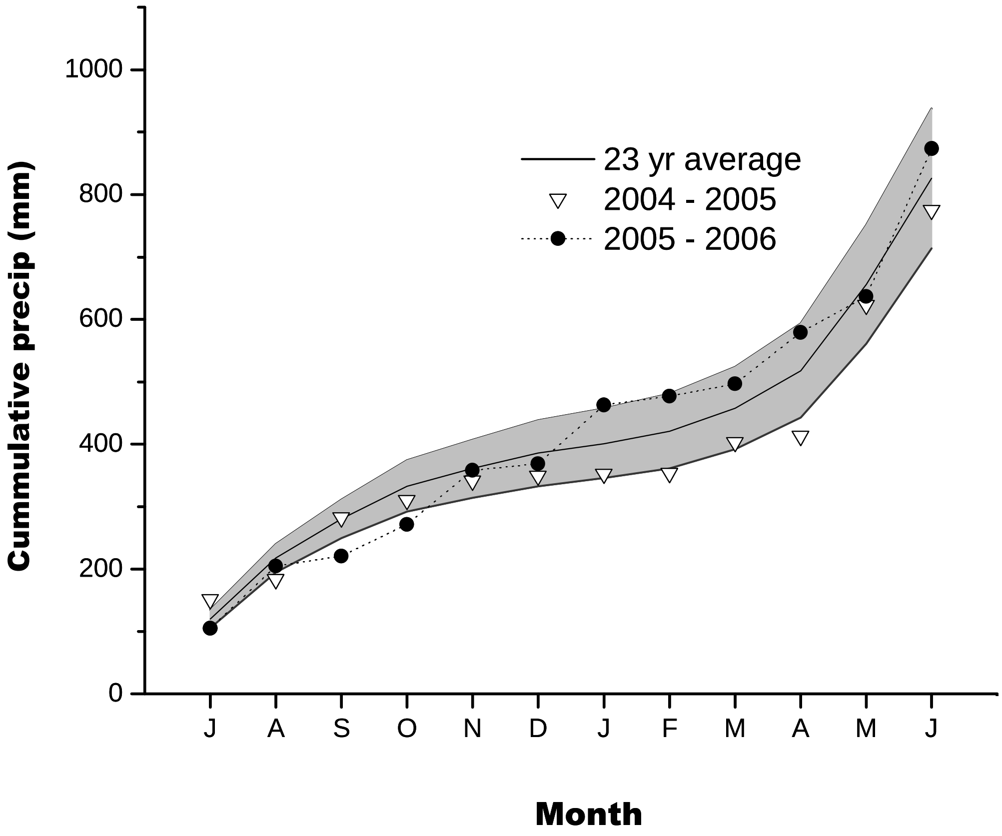


Figure 2.3 - Monthly cumulative precipitation of the measurement seasons compared to historic records (23 years monthly average) of precipitation data for Bariloche, Argentina (approx. 70km south from study site). Grey area represents ±1 standard error. Normal years show a Mediterranean regime of precipitation with a notorious summer drought (Dec-Mar).

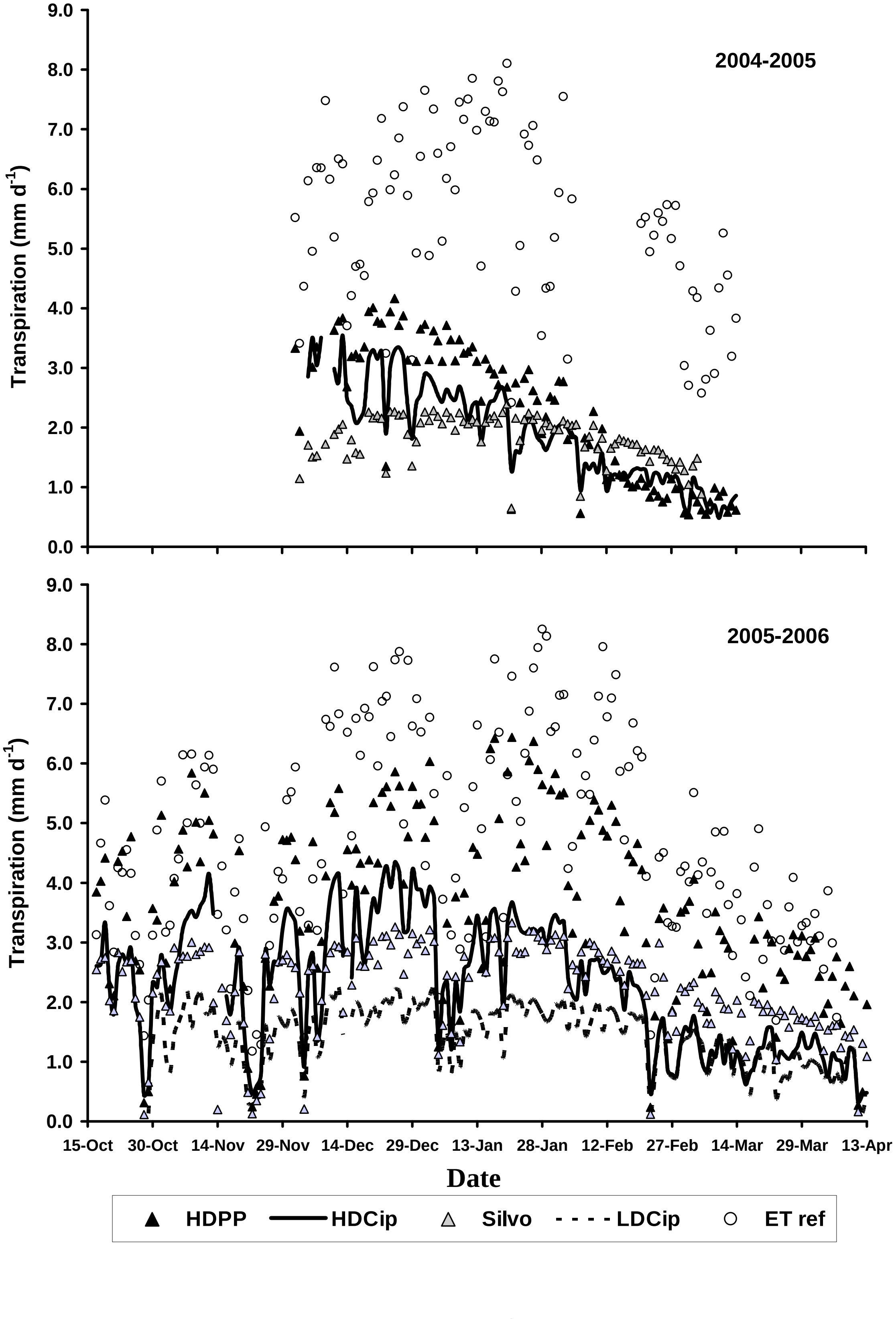


Figure 2.4 - Daily transpiration per plot (mm d^{-1}). Upper chart is 2004-2005

Figure 2.4 - (continued) (normal dry summer season). Lower chart is 2005-2006 (wet summer season). Open circles represent the potential evapotranspiration, solid triangles HDPP, grey triangles Silvo, solid line HDCip, and dotted line LDCip. The response to greater soil water content availability in the second year was more pronounced in the ponderosa pine sites (both HDPP and Silvo). Note that in the second season measurements were started earlier in the spring.

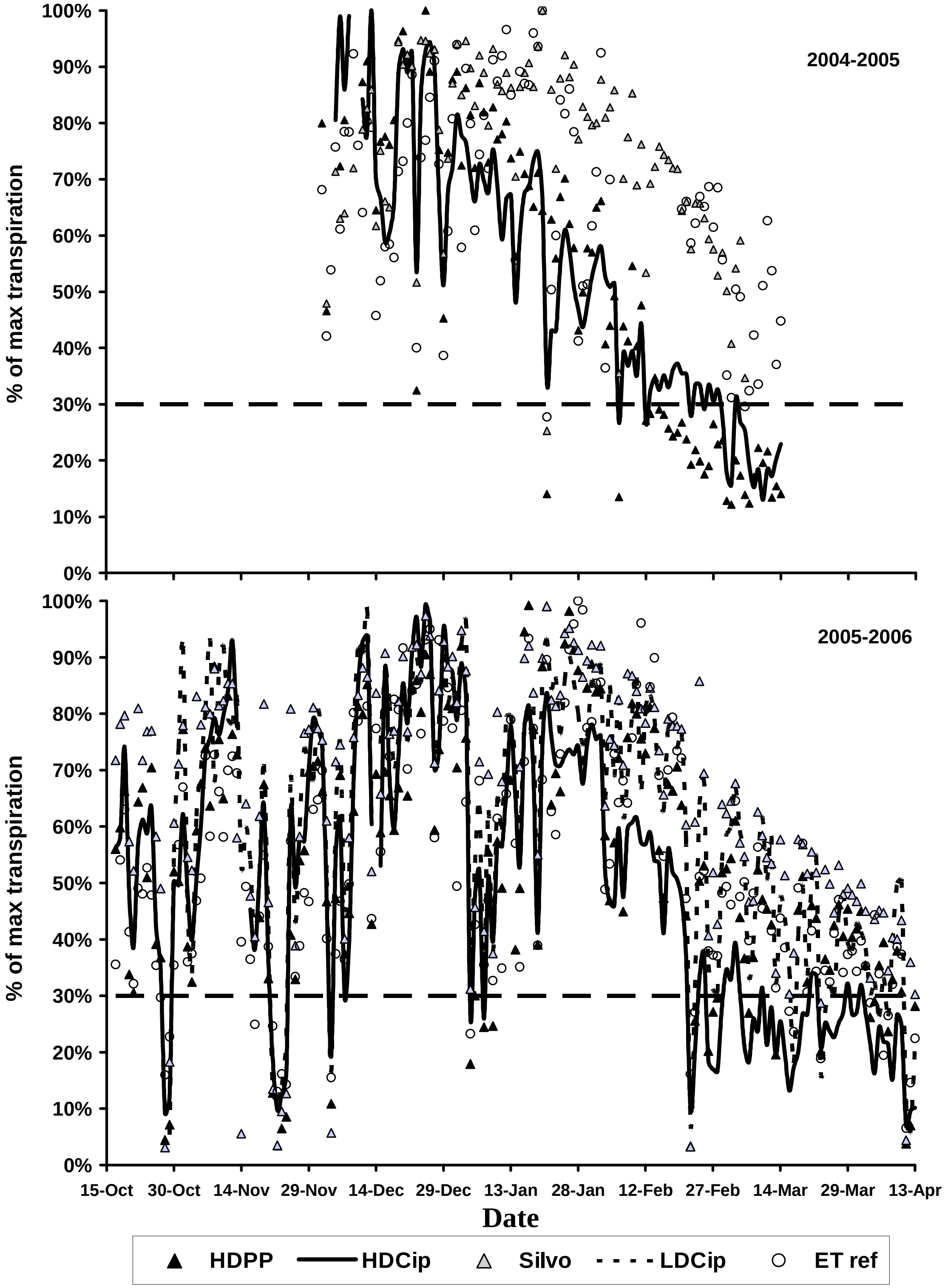


Figure 2.5 - Percent transpiration relative to maximum transpiration of the

Figure 2.5 (continued) season per site per day. Etref was standardized relative to the maximum calculated ETref rate of each season. Upper chart is season of year 2004-2005 (dry summer year). Lower chart is the 2005-2006 (wet summer year). Open circles represent the potential evapotranspiration, solid triangles HDPP, grey triangles Silvo, solid line HDCip, and dotted line LDCip.

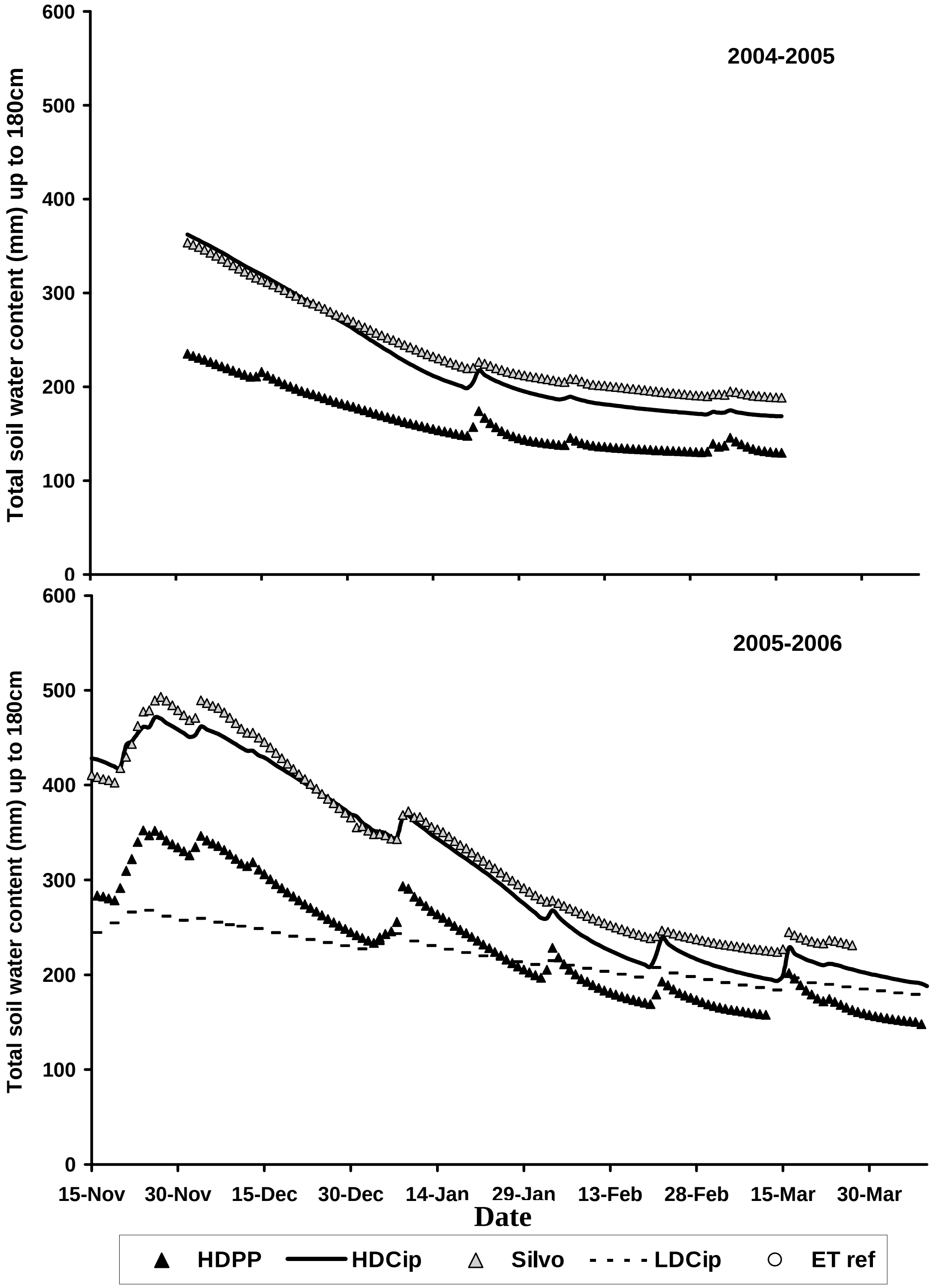


Figure 2.6 - Total soil water content (mm) of water stored in the soil up to 180 cm deep per site for both seasons. Calculated from daily averages of volumetric water

Figure 2.6 (continued) content measurements from 3-4 probes per site with 6 sensors per probe at different depths (10, 20, 40, 80, 140 and 180 cm from soil surface). Solid triangles represent HDPP, grey triangles Silvo, solid line HDCip, and dotted line LDCip.

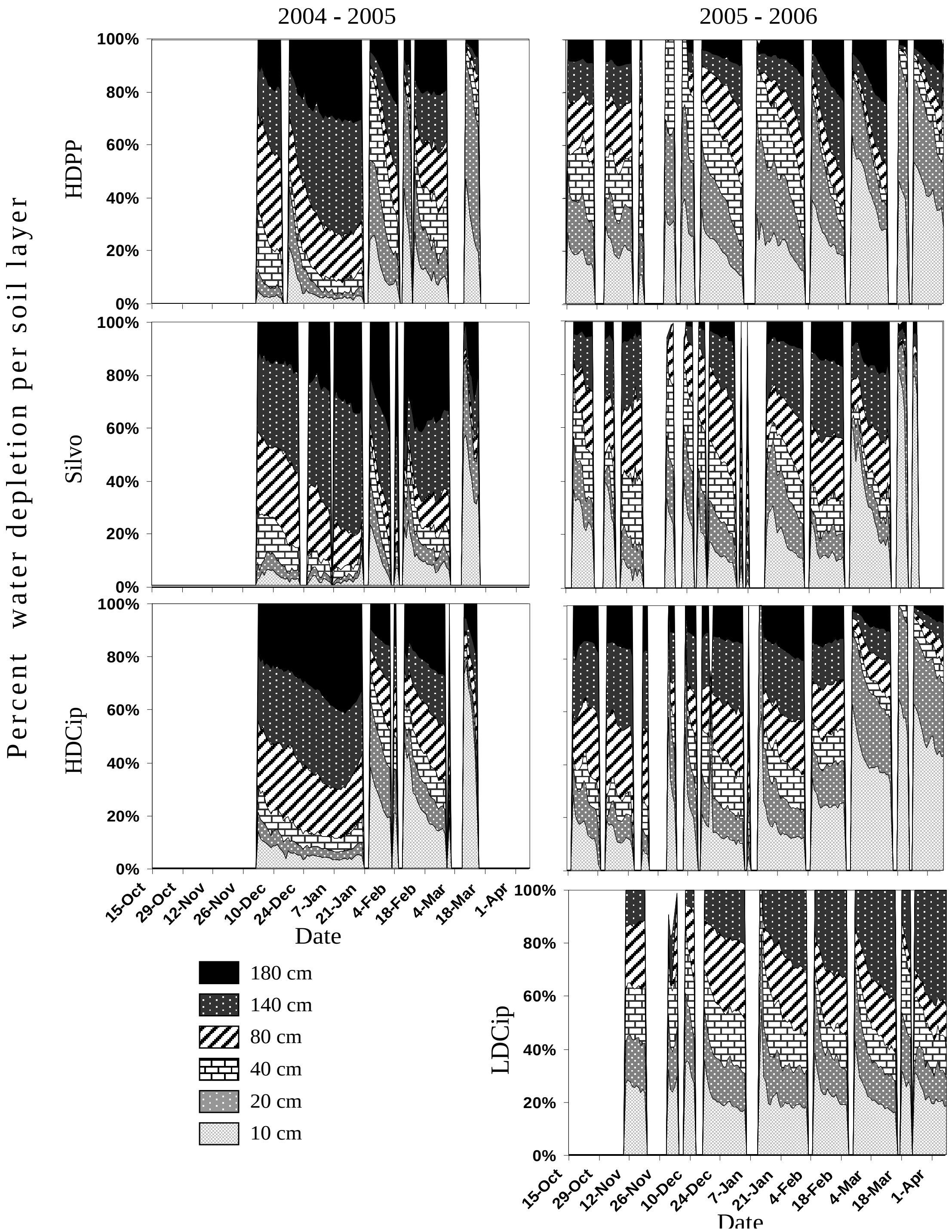


Figure 2.7 - Relative daily water depletion from individual layers in the upper 180 cm of the soil, calculated from daily averages of soil volumetric water content

Figure 2.7 (continued) measurements. Blank areas represent periods without measurements, or rain events plus two days, where water depletion was not calculated.

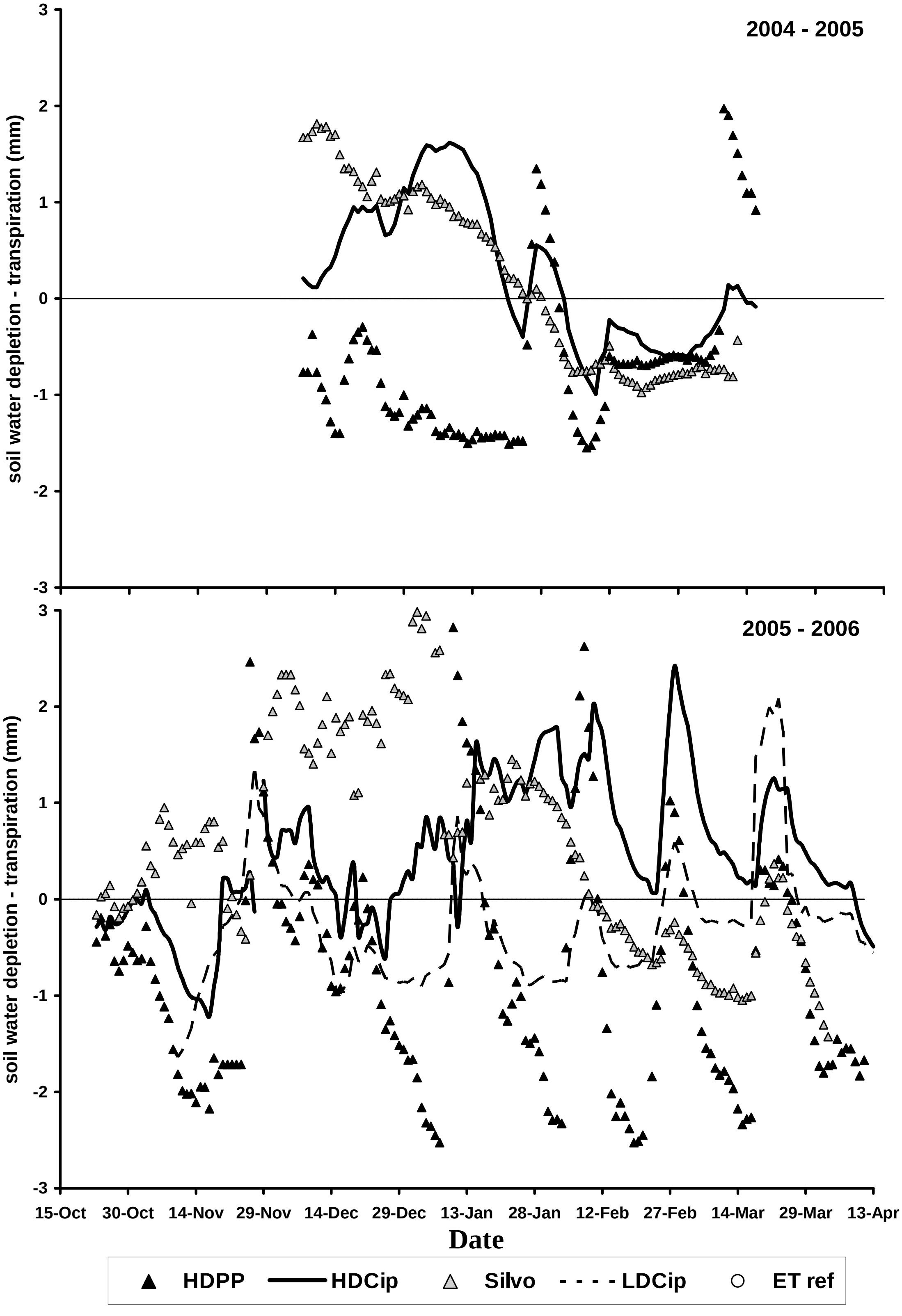


Figure 2.8 - Mass balance analysis. Difference between soil water depletion and

Figure 2.8 (continued) sapflow estimates of water use (mm d^{-1}). Positive values represent a higher estimate of water depletion from soil measurements than transpiration estimates from sapflow measurements. Solid triangles correspond to HDPP, grey triangles to Silvo, solid line to HDCip, and dotted line to LDCip.

CHAPTER THREE: AFFORESTATION AND RAINFALL INTERCEPTION: A CRITICAL FACTOR TO MINIMIZE IMPACTS ON WATER RESOURCES

Julian A. Licata[1*], Thomas G. Pypker[2], Mariana Weigandt[3], Michael H. Unsworth[1], Javier E. Gyenge[3], Maria Elena Fernández[3], Tomás M. Schlichter[3] and Barbara J. Bond[1]

([1]) Department of Forest Science, Oregon State University, 321 Richardson Hall,
Corvallis, OR 97331, USA.

([2]) School of Forest Resources & Environmental Science , 110 Noblet Building,
Houghton, MI 49931 USA.

([3]) INTA EEA Bariloche, Grupo de Ecologia Forestal, CC 277 (8400) Bariloche,
Argentina.

Abstract

In the forest-steppe ecotone at the base of the Andes Mountains in northwest Patagonia, Argentina, lands that were originally cleared for livestock grazing are now being converted to plantations. Exotic ponderosa pine (*Pinus ponderosa* Doug. ex. Laws) is being used in over three quarters of afforestation projects in the natural distribution area of cordilleran cypress (*Austrocedrus chilensis* (D.Don) Pic. Ser. et Bizzarri). Previous studies found that seasonal transpiration in ponderosa pine plantations was between 33 to 64 % greater than that of cordilleran cypress under similar growing conditions. Despite transpiration typically being the largest component of evapotranspiration (ET) in closed canopy forests, evaporative losses of rainfall water intercepted by the canopy (I_L) could account for a large proportion of ET in forest ecosystems. The structure of the canopy and the morphology of leaves are very different between ponderosa pine plantations and cordilleran cypress stands, suggesting that there may be potential differences in I_L. The primary objective of this study was to test the hypothesis that interception of rainfall is greater in native cordilleran cypress stands than in planted ponderosa pine forests at similar leaf areas. We also sought to compare the combined effect of transpiration and interception loss on water balance in these two forests located in Northern Argentinean Patagonia. Additionally, water storage capacity of cut branches was measured in the lab. We found that the I_L of cordilleran cypress stands was much larger than that of ponderosa pine stands. Total annual canopy ET did not differ between species, but was higher in high density plots as compared to low density plots.

Introduction

Since the beginning of the last century, the natural forests in Argentina have been dramatically reduced (Merenson, 1992; SAyDS, 2006). In just the last decade, the loss of natural forests was estimated to be as high as 200,000 ha yr^{-1} (Montenegro et al., 2004; UMSEF, 2007). To reduce pressure on natural forests, plantation forestry using fast-growing species has been proposed as an alternative to logging native forests (Sedjo and Botkin, 1997; Hartley, 2002; Evans and Turnbull, 2004). In this way, the productivity can be maintained by increasing productivity in certain patches of the landscape, while leaving the others untouched for conservation. Following this rationale, the Argentinean government has been actively promoting the expansion of forest plantations for the last three decades (SAGPyA, 1999; Laclau, 2003).

Although the use of fast-growing species could reduce logging pressure on native forests and at the same time support rural economies, this change in land vegetation cover could bring its own environmental costs. For water resources in particular, this vegetation change could reduce catchment water yields through an increase in evapotranspiration (ET; Vertessy, 2000; Jackson et al., 2000; Nosetto et al., 2005). In order to minimize the potential negative impact on the regional water cycle it is necessary to understand how changing the vegetation type will alter ET.

In the forest-steppe ecotone at the base of the Andes Mountains in northwest Patagonia, Argentina, lands that were originally cleared for livestock grazing are now being converted to plantations. In this area water is a crucial limiting resource, and exotic ponderosa pine (*Pinus ponderosa* Doug. ex. Laws) is being used in over three quarters of afforestation projects (Gallo et al., 2005; Sarasola et al., 2006). Little or no regeneration of the native cordilleran cypress (*Austrocedrus chilensis* (D. Don) Pic. Serm. et Bizzarri) occurs in typical ponderosa pine timber plantations. To date, only around five percent of the total amount of land suitable for afforestation has been planted (SAGPyA, 1999), but there is large potential for expansion of plantations in the region (Schlichter and Laclau, 1998; Laclau and Andenmatten, 2005). Knowledge of the differences in ET between cordilleran cypress and ponderosa pine will help land managers evaluate the potential tradeoffs between wood production and impacts on water resources before plantations become widespread.

In water-limited ecosystems, a large proportion of annual precipitation is returned to the atmosphere as ET, regardless of the vegetation type, leaving a relatively small proportion available for runoff as streamflow (Zhang et al., 2001; Wilcox et al., 2003; Le Maitre, 2004; Huxman et al., 2005). ET has three components, evaporation (E) of water from the soil surface, interception loss (I_L) of precipitation and subsequent evaporation from the wet canopy, and transpiration (T) which involves uptake of water by roots, transport through stems to foliage, and evaporation from leaves through stomata. Changes in the partitioning of ET among these components can have a critical impact on productivity, since transpiration is the only flux of the three that is directly related to productivity. If more water is available for transpiration, stomata can remain open for longer periods of time and leaves can assimilate more carbon through photosynthesis.

Transpiration is the dominant pathway of water from soil to the atmosphere in closed canopy forest ecosystems (Unsworth et al., 2004). In a previous study, we found that transpiration in ponderosa pine plantations was between 33 to 64 % greater than that of cordilleran cypress under similar growing conditions and stand density (Licata et al., in press). Despite transpiration typically being the largest term in ET for forests, I_L is often the primary source of differences in ET among forest ecosystems (Crockford and Richardson, 2000). Among the three components of ET, I_L is the least studied (Savenije, 2004), and the potential impact of ponderosa pine plantations on I_L, and consequently on ET partitioning, is unknown.

I_L is likely to change when the structure of the canopy or the morphology and distribution of leaves changes (Pypker et al, 2005; Roberts, 2007). These characteristics can vary greatly, as is the case between planted ponderosa pine and native cypress forests. Ponderosa pine plantations usually have a homogeneous distribution of tree crowns that would likely result in a different aerodynamic resistance than the more heterogeneous native cypress stands. Moreover, pine needles are longer (~15, 0.5 cm for ponderosa pine and cordilleran cypress, respectively) and have smoother surfaces than the scale-shaped leaves of cordilleran cypress, which may affect canopy water storage capacity (S). These structural and morphological differences between the two forest types are likely to translate into differences in I_L.

The primary objective of this study was to test the hypothesis that interception of rainfall is greater in native cordilleran cypress stands than in planted ponderosa pine forests at similar leaf areas. We also sought to compare the combined effect of trees transpiration and interception loss on water balance in these two forests located in northern Argentinean Patagonia.

Methods

Field Site

The study site is in northwestern Patagonia, Argentina (40.29º S, 71.13º W). Average annual rainfall (1980-98) at the site is approximately 800 mm (AIC, 2005) with a Mediterranean weather regime, where approximately 80% of the annual precipitation occurs during fall and winter, mostly as rainfall. Maximum and minimum annual average temperatures are 17.1°C±0.5(SE) and 4°C±2.1(SE), respectively. Soils are deep (> 1.8 m) with a sandy loam texture and small pebbles.

Four forest plots were installed in the Caleufu River valley, at an altitude of 810 m. Treatments consisted of a high density ponderosa pine (HDPP), a low density ponderosa pine with grass understory (LDPP), a high density cordilleran cypress stand (HDCip), and a low density cordilleran cypress stand (LDCip). Cordilleran cypress stands were the result of natural regeneration, whereas the ponderosa pine stands were planted for both timber production (HDPP) and timber and pasture production (LDPP). The size of each of the plots was approximately 400 m^2 and 200 m^2 for low density and high density plots, respectively. Plots were assessed visually and selected within larger stands based on leaf area, height, and soil type. All plots were located within a radius of 500 m of one another. Despite differences in the number of stems per hectare and basal area among the plots, leaf area index (LAI) was similar between plots with the same stand density (e.g., low density). High density plots had approximately double the LAI of low density plots of the same species (Table 1). Although the plots were carefully selected to be similar to those commonly planted in the region, the current study was installed as part of an existing trial that does not replicate treatments. Thus, the results should be considered a case study and not as representative of all ponderosa pine plantations in NW Patagonia. This study was designed to provide detailed, stand-level measurements of water fluxes in native and exotic pine stands growing under the same environmental conditions.

Meteorological data

Half hourly values for relative humidity, air temperature, wind speed, solar radiation and gross precipitation were measured by a permanent low-maintenance meteorological station within 7 km of the study plots, and daily values were obtained from two highly maintained meteorological stations located in the closest city to the study site (Bariloche, 70 km south of the study plots). We also used a portable meteorological station which was placed near the low-maintenance station and in an open grassland near the study plots to check for possible non-random discrepancies in the meteorological conditions between the low-maintenance station and the study plots.

Potential Evapotranspiration (PET)

Daily potential evapotranspiration (PET) was calculated following the FAO Penman-Monteith method as detailed in FAO Irrigation and Drainage Paper 56 (Allen et al., 1998) for daily climatic data. PET represents the maximum potential evapotranspiration of a hypothetical reference crop, with no water deficiency, growing under the same meteorological conditions as measured in the field site.

Net and Gross precipitation

Two independent measurement methods were employed to measure precipitation above (gross precipitation, P_G) and below the canopy (net precipitation, P_n) in the study plots: manual rain gauges (MRG) and tipping buckets with troughs (TBT). Manual rain gauges were used for measuring total precipitation data from individual rain events, whereas tipping buckets recorded cumulative precipitation every 10 minutes.

The manual rain gauges were employed in arrays of 25 PVC pipe rain gauges per plot. An additional set of 30 roving rain gauges were moved from one plot to another to check the accuracy of the permanent gauges. Water collected from the gauges was measured manually in a 100 ml graduated cylinder to a resolution of 1 ml. The collection area for each gauge was 86.7 cm^2. Gross precipitation was measured with five rain gauges placed in a 50-m-wide fire cut adjacent to the plots having a North-South orientation. To reduce the potential impact of evaporation on the gauges in the open fire cut, each was surrounded with reflective insulation material (Mexpol®, 9 mm). Measurements were taken on all plots after every rain event, from December 4th 2005 to May 26th 2006.

The second method consisted of two sets of five tipping buckets (model TE525MM, CSI, Logan, UT) with custom-made troughs and collectors, plus an additional tipping bucket located in the open. Troughs were set at 45 degree angles (relative to the ground), and consisted of two 200 cm long PVC tubes, each with a ~1.75 cm wide and 175 cm long opening cut into the top of the tube. Once cut, the width of the slot was measured every 10 cm to accurately calculate the collection area of each trough. Since only two plots could be measured at the same time, we measured the HDPP and LDCip plots first and the LDPP and HDCip later.

Evaporation of collected water and flow distortion of precipitation by wind are the most important factors creating systematic errors in rain gauge measurements (Michelson, 2004). Both factors would be expected have a larger effect on the rain gauges located in the open area, resulting in a potential underestimate of P_G. Underestimation of P_G would in turn generate underestimates of interception loss (I_L) at all the sites, as the same P_G value was used for all the sites. A rough estimate of the potential error that wind speed distortion could have generated (based on Michelson, 2004) was as high as 10.1 % under average wind speed (2.06 m s^{-1}) and low intensity rainfall (0.5 mm hr^{-1}) conditions typical at our site. However, our I_L results were higher than those published in the majority of rainfall interception studies (Carlyle-Moses, 2004; Llorens and Domingo, 2007), indicating that it is unlikely that our results underestimate P_G. Furthermore, even if we underestimated $P_{G,,}$ our conclusions would not differ.

Stemflow

We performed preliminary measurements to evaluate the importance of stemflow in our study plots. If stemflow is not accounted for explicitly, our measurement approach will overestimate I_L. Stemflow measurements were made on five trees in each of the HDPP and LDCip plots using a 2 cm diameter garden hose cut in half longitudinally and attached in a spiral fashion around the stem of the trees. The bottom of the hose fed into a plastic container where stemflow water was collected. We found that stemflow represented 3% and 1.5% of gross precipitation for ponderosa pine and cordilleran cypress, respectively. These values are relatively small and only slightly increase the differences in interception losses found between species. Consequently, we did not pursue these measurements further.

Canopy hydrologic parameters

A simplified version of the Gash model (Gash, 1979; Link et al., 2004; Pypker, et al. 2005) was used to calculate interception losses. This model requires knowledge of canopy hydrologic parameters, such as canopy storage capacity (S), direct throughfall (p) and average evaporation rate to rainfall rate ratio ($\bar{E}/\bar{R}$). A system of two linear regressions was fitted to P_n vs. P_g (Figure 3.1) and was used to calculate the canopy hydrologic parameters that regulate rainfall interception losses, as explained in the mean method in Klaassen et al. (1998) for rain gauges, and similarly applied in Link et al. (2004) and Pypker et al. (2005) for tipping bucket data. This approach was used to fit the results from the manual rain gauges (MRG; each point being an individual storm event), and the tipping buckets with troughs (TBT; each point being the ten minute average of the cumulative precipitation). The first linear regression corresponds to the pre-saturation period for the TBT data (i.e., the small storm events insufficient to saturate the canopy and be collected in the MRG) and has only one parameter (the slope, a_1). The second linear regression corresponds to the post-saturation period of the rain event for the TBT data (or the storm events sufficient to saturate the canopy for the MRG data), and has two parameters (slope and intercept, a_2 and b_2, respectively).

Therefore, canopy hydrologic parameters were calculated as follows:

Direct throughfall (p):

(1) $p = a_1$

Average evaporation rate to rainfall rate ratio ($\bar{E}/\bar{R}$):

(2) $\bar{E}/\bar{R} = 1 - a_2$

Canopy storage capacity (S), assuming evaporation during wet-up:

(3) $S = - b_2$

Amount of gross rainfall necessary to saturate the canopy (*Ps*):

(4) $P_s = \dfrac{a_2}{a_1 - b_2}$

Maximum Storage Capacity of Branches

A second method was used to corroborate the results found using the two linear regression method described above. A set of 9 branches of cordilleran cypress and 6 branches of ponderosa pine of different sizes were collected in the field and returned to the laboratory. Shortly thereafter, the fresh weight, length and diameter of the branches were measured. Branches were then suspended from a beam and wetted continuously with a sprinkler until 2 minutes after the branches started dripping. Branches were left to drain and were weighed immediately after dripping stopped. The water storage capacity was measured as the weight difference before and after wetting. All foliage from the branches was removed and dried at 70 °C to obtain the dry weight of the leaves per branch.

Calculation of Annual Interception Losses

The Gash model is commonly used to predict I_L and has been used in a broad variety of forest types under various climatic conditions (Loustau et al., 1992; Valente et al., 1997; Deguchi et al., 2006; Cuartas et al., 2007). It requires a small set of parameters and is constrained by the following assumptions (Gash, 1979): (1) rainfall is composed by discrete storm events separated in time long enough to allow the canopy to dry; (2) the meteorological conditions remain constant throughout each storm event; and (3) there is no drip from the canopy during wet-up.

The average evaporation to rainfall rate ratio ($\bar{E}/\bar{R}$) was calculated in the canopy hydrologic parameters, and generally is assumed to be representative of all the rain events in a particular forest stand. However, the time of the year when the manual rain gauge measurements were taken was warmer than when the majority of the rain events occurred (in the fall and winter, when the site was inaccessible by car due to snow-covered mountain passes), and we consider it inappropriate to use the $\bar{E}/\bar{R}$ value calculated from those measurements to predict I_L at other times of the year. Therefore, we corrected the $\bar{E}/\bar{R}$ value using the actual potential evapotranspiration (PET_d) for each day. To do so, we calculated the average PET for the days of rainfall measurements (PET_m), and the PET_d for the days with rainfall events for the two years of transpiration measurements. Then, for each storm event we included a correction coefficient equal to the ratio PET_d / PET_m. In addition, to make the model easier to use, we did not partition water flow and evaporation between stems and canopy. For each individual storm event the following simplified Gash model was used:

If $Pg < Ps$ (presaturation condition), then:

$$(5)\ I_L = (1 - p)\,Pg$$

If $Pg > Ps$ (post saturation condition), then:

$$(6)\ I_L = \bar{E}/\bar{R}\,\frac{PET_d}{PET_m}\ Pg + Ps\,(1\text{-}p - \bar{E}/\bar{R}\,\frac{PET_d}{PET_m})$$

Daily meteorological data were used to predict annual I_L in the study plots for each set of parameters. However, the use of daily data (vs. hourly data) can prove problematic when it rains for several days at a time. It is not possible to determine whether the canopy dried out between consecutive rainy days or whether the canopy remained saturated throughout continuous days of rain. Most likely, some combination of these two conditions occurs. To deal with this uncertainty, we performed two analyses for each set of parameters of the Gash model with different storm event size configurations, one with each day’s rainfall precipitation equaling a single storm event, and the other with each set of continuous days of rain grouped into one storm event. Averages of these two storm event size configurations were used to estimate I_L.

When analyzing the troughs and tipping buckets data we encountered two type of problems worth mentioning to improve the design of future studies. At certain times the tipping buckets below the canopy recorded precipitation that exceeded the one of the open area. During high intensity rains the volume of water captured by the troughs was not able to drain out of the device and started impeding the tipping mechanism. If there was not enough time to reach canopy saturation before the tipping mechanism was affected, then the data for that storm event was not used.

A second type of problem relates to the storm events of variable intensity or intermittent rain periods where partial drying and rewetting of the canopy occurs. Using data from this type of storm to parameterize the Gash model would induce a large overestimation of the evaporation to rainfall rate ratio. For that reason, in the parameterization of the canopy hydrologic parameters with the manual rain gauges we did not to use any rain event larger than 30 mm or any storm event longer than one day.

Results

Gross (Pg) and Net Precipitation (Pn)

Total annual gross precipitation was 122 mm higher the second year of measurements (May 2005 - April 2006), compared to the first year of measurements (May 2004 - April 2005). The distribution of precipitation throughout the year (Figure 3.2) also varied between the two years. The first year of measurements (hereafter, the "dry year") showed a typical Mediterranean distribution of precipitation with only 14 % of the total annual precipitation falling during spring and summer. In contrast, spring and summer precipitation represented 30% of the total annual precipitation in the second year of measurements (hereafter, the "wet year"). In absolute terms, spring and summer precipitation was 148.8 mm greater in the wet year than in the dry year.

Net precipitation was consistently lower below the canopies of the cordilleran cypress stands than below the ponderosa pine stands throughout the period of measurements with rain gauge measurements (Figure 3.3). For all the storms measured from December 4th, 2005 to May 26th, 2006 (n=13), relative net precipitation averaged 61% (S.E. 5.9%), 65.6% (S.E. 3.6%), 38.0% (S.E. 4.5%) and 41.5% (S.E. 4.8%) of gross precipitation for HDPP, LDPP, HDCip and LDCip, respectively. Average daily potential evapotranspiration for all of the manual rain gauge measurements was 2.98 mm (S.E. 0.34 mm, n=13). The percentage of water per storm event passing through the canopies increased with storm size (total rainfall per event) up to 30 mm of rain, and then remained constant in the ponderosa pine stands, while it decreased to a lower rate in the cypress stands (Figure 3.3). Storm events larger than 30 mm were events that comprised more than two consecutive days of rain, which could have allowed for partial drying and rewetting of the canopy. In all storms greater than 5 mm, more water reached the forest floor in ponderosa pine stands compared to cordilleran cypress stands (Figure 3.4). There were no consistent differences in the total or relative amount of P_n between stands of the same species with different densities.

Canopy Hydrologic parameters

For all the parameterization methods, HDCip had the highest canopy storage capacity (*S*). Parameterization using the manual rain gauge (MRG) data yielded values of *S* of 2.28, 0.70, 3.78 and 1.65 mm for HDPP, LDPP, HDCip and LDCip, respectively (Table 2). Parameterization using data from the tipping buckets with troughs (TBT) yielded values of *S* of 3.93, 3.69, 8.88 and 2.66 mm for HDPP, LDPP, HDCip and LDCip, respectively. The amount of gross precipitation necessary to saturate the canopy (*Ps*) followed a pattern similar to *S*, except for the LDPP plot where the TBT *Ps* estimate was double the MRG *Ps* estimate (Table 2). MRG estimates of *Ps* were 4.03, 2.57, 7.28 and 4.09 mm for HDPP, LDPP, HDCip and LDCip, respectively. TBT estimates of *Ps* were 4.56, 5.13, 9.47 and 5.63 mm of Pg for HDPP, LDPP, HDCip and LDCip, respectively (Table 2). Direct throughfall (*p*) was higher in the low density plots for both methods, except in the estimate obtained with the MRG data where both cypress stands had similar values of *p* (Table 2). Average evaporation to rainfall rate ratio ($\bar{E}/\bar{R}$) was also higher in the low density plots compared to the high density plots of the same species, except in the TBT estimate where both ponderosa pine stands had similar values of $\bar{E}/\bar{R}$ (Table 2).

The overall regression of modeled versus observed *Pn* for the MRG data produced adjusted r squared values of 0.99, 0.95, 0.93 and 0.94 for HDPP, LDPP, HDCip and LDCip, respectively, and none of the intercepts and slopes were different from a 1:1 line with a 95% CI. The sum of squared errors was 2.66, 8.91, 7.76 and 6.39 mm^2 for HDPP, LDPP, HDCip and LDCip, respectively. The overall regression of modeled versus observed *Pn* for the TBT data gave values of adjusted r squared above 0.98 for all sites, and none of the intercepts and slopes were different from a 1:1 line with a 95% CI. The sum of squared errors was 3.77, 10.04, 2.03 and 0.62 mm^2 for HDPP, LDPP, HDCip and LDCip, respectively.

Maximum Storage Capacity of Branches

Differences in branch water storage capacity confirmed the observed differences between *S* (canopy water storage capacity) measured in the different forest stands (Table 2). Water storage capacity normalized by the dry weight of the branches was significantly higher in cordilleran cypress compared to ponderosa pine branches (Figure 3.5_a). Average water storage was 1.5 (SE 0.38) and 0.33 (SE 0.04) g water g^{-1} leaf for cordilleran cypress and ponderosa pine respectively. Water storage per branch was highly correlated with basal area of the branch (Figure 3.5_b); correlation coefficients were 0.97 ($p<0.001$) and 0.98 ($p<0.001$) for cordilleran cypress and ponderosa pine, respectively. Moreover, basal area has been observed to be highly correlated with leaf area in both species in a previous study (Laclau, 2003), which used trees of both species from various stands in the region, including several plots in the same location as this study. Applying the equations developed by Laclau (2003) to transform storage capacity estimates to a ground area basis, estimated values of *S* were 3.1, 1.3, 7.1 and 3.6 mm for HDPP, LDPP, HDCip and LDCip, respectively.

Total annual interception loss (I_L) estimates

Total annual I_L was highest in HDCip during both years, and both cordilleran cypress stands clearly had higher I_L than the ponderosa pine stands in both years (Figure 3.6). In the dry year (2004-2005) total annual I_L was 224.5 (SE 75.8), 203.9 (SE 72.2), 337.0 (SE 54.2) and 315.0(SE 62.3) mm for HDPP, LDPP, HDCip and LDCip, respectively. In the wet year (2005-2006) total annual I_L was 281.4 (SE 53.2), 252.5 (SE 32.5), 409.0 (SE 72.1) and 353.2 (SE 51.9) mm for HDPP, LDPP, HDCip and LDCip, respectively. Relative to P_g, interception losses were as high as 26, 24, 39 and 37% in the dry year, and 30, 27, 43 and 38% in the wet year for HDPP, LDPP, HDCip and LDCip, respectively. Total annual Pg increased 122 mm from the dry to the wet year, and net precipitation for each plot increased 25.9, 34.2, 10.8 and 44.6 mm for HDPP, LDPP, HDCip and LDCip, respectively between the two years. However, as the distribution of precipitation was also different from the dry to the wet year, the net precipitation per plot for spring and summer increased 120.6, 121.6, 101.8, 122.3 mm for HDPP, LDPP, HDCip and LDCip, respectively. When total I_L was added to the transpiration estimates from the same plots (Licata et al. in press) in both years, as an estimate of total annual canopy ET, there were no significant differences between stands of similar levels of density (Figure 3.7). Total canopy ET during the dry year was 703 (SE 159), 536 (SE 141) and 698 (SE 110) mm for HDPP, LDPP and HDCip, respectively (there were no transpiration estimates for the LDCip plot that year). Total ET during the wet year was 939 (SE 168), 647 (SE 114), 811 (SE 135) and 605 (SE 88) mm for HDPP, LDPP, HDCip and LDCip, respectively.

Discussion

To reduce logging pressure on natural forests, the Argentine government is promoting industrial forestry plantations using exotic fast-growing species (Laclau, 2003). In the forest - steppe ecotone of southwestern Argentina, approximately 80% of afforestation projects have been planted with fast-growing ponderosa pine, which has greater productivity than the native tree species (Schlichter and Laclau, 1998). However, this increase in productivity is linked to an increase in transpiration (Licata et al., in press), which could have an impact on water resources, thereby altering water balance downstream. It is generally believed that increased canopy evapotranspiration (ET) will decrease runoff (R) in humid landscapes (Fahey and Jackson, 1997; Jackson et al., 2000), but it is debatable if the same decrease in R will happen in water-limited environments, like those in the forest-steppe ecotone in Patagonia (Zhang et al., 2001; Wilcox, 2002). In humid environments, gross precipitation usually exceeds ET and the remainder leaves the landscape as runoff. In contrast, there is minimal water remaining for runoff in water-limited environments (Wilcox et al., 2003), leaving little room for variation in total ET for plants to increase transpiration. However, changes in the partitioning of ET between interception loss and transpiration may allow for an increase in productivity without reducing runoff. In the forest-steppe ecotone in Patagonia, where water is a limiting resource of productivity, exotic ponderosa pine plantations have higher transpiration rates than native cordilleran cypress (Licata et al., in press). However, our results show that differences in interception loss between cordilleran cypress and ponderosa pine plantation stands of similar leaf area were large enough to compensate for the increased transpiration of the exotic pine plantation, resulting in a net difference in ET of zero between the two species.

The estimates of interception loss found in this study for cordilleran cypress are larger than those typically found in other conifer forests (Valente et al., 1997; Lankreijer et al., 1993; Pypker et al., 2005; Iida et al., 2006). However, when compared to other scale-leaved species growing in dry environments our interception loss values (37% to 43%) are within the range reported by other studies. Owens et al. (2006) reported an average interception loss of 35% in Juniper rangelands, whereas Thurow and Hester (1997) reported values as high as 70% when litter layer interception was also included in the calculation of I_L. Our estimates of ponderosa pine interception loss (24% to 30%) are on the higher end of the range of estimates reported by other studies for pinus sp. forests (Loustau et al., 1992; Gash et al., 1995; Valente et al., 1997; Bryant et al., 2005).

Interestingly, species differences were much more important than LAI in interception losses in our study. Despite the large difference in LAI between stands of different density, total annual interception losses were only slightly different between stands of the same species. This may be explained by a simultaneous decrease in canopy storage capacity and an increase in evaporation rate from wet surfaces when the number of trees per hectare decreased. The difference in tree density between high and low density plots of the same species was larger than the difference in leaf area. Therefore, while canopy storage capacity decreased with LAI, the reduction of trees per hectare may have increased the turbulence within the canopy and consequently increased the aerodynamic conductance. A higher aerodynamic conductance will, in turn, lead to an increase in evaporation rate from wet surfaces (Teklehaimanot et al., 1991). In addition, low density stands did not showed an increase in direct throughfall proportional to the decrease in canopy cover. This is likely due to the high winds predominant in the region, and the lack of gaps large enough to let the angled incoming precipitation reach the ground without touching the canopy. Transpiration, on the other hand, was dramatically different between sites of different densities (Licata et al. in press). Consequently, our results indicate that management of stand density could be more effective than species composition in reducing total annual ET, as well as decreasing impacts on water resources. It is important to note, however, that in the low density stands, other potential sources of ET (soil and understory) were not measured. While ET from soil and understory is expected to be similar between same-density stands of different species, these could account for the differences in canopy ET between stands of different densities.

Among the structural and morphological differences between the stands of ponderosa pine and cordilleran cypress, we believe that leaf morphology was the most important factor contributing to the larger interception loss of cordilleran cypress native forest. Canopy water storage capacity has been highlighted by various authors as the most important factor influencing I_L (Valente et al., 1997; Klaassen et al., 1998; Crockford and Richardson, 2000; Llorens and Gallart, 2000; Iida et al., 2005). Our experiment with artificially wetted branches showed very large differences in maximum canopy storage capacity per unit leaf weight and basal area between cordilleran cypress and ponderosa pine. Moreover, these two species showed contrasting values of stand canopy water storage capacity, irrespective of the method used to estimate it. These results suggest that scale-leaved species should be treated as a separate group from the rest of conifers for rainfall interception analyses, as their rainfall interception losses will most likely be higher than other vegetation types of similar leaf area.

Our findings showed that canopy ET was similar across species. An average of about 95% and 80% of total precipitation (in high and low density plots, respectively) was either lost by evaporation or transpiration. However, the partitioning of ET into interception and transpiration differed drastically between species, with cordilleran cypress having higher interception but lower transpiration than ponderosa pine. This raises interesting questions about the evolutionary processes that would lead to high interception rates, and has important implications for management.

From an evolutionary perspective, it seems probable that plants growing in a water limited environment would evolve towards a low canopy interception of precipitation in order to have more water available for transpiration, and consequently, more carbon assimilation. However, although at the stand level a low interception of precipitation could allow for an increase in transpiration, this mechanism will not guarantee that the benefits from letting more water enter the soil are going to be harvested by the plant that is actually making this happen. On the other hand, it is not clear how the increase of interception could benefit a particular plant unless they were able to take in some of the intercepted water at the leaf level. However, in Mediterranean climates where most of the precipitation occurs while the plants are not actively transpiring, having a large interception of precipitation might not be a competitive disadvantage either, as long as it lets enough water pass through the canopy to reach the maximum water holding capacity of the soil by the end of the rainy season. Indirect benefits could arise when the high interception plant has access to deeper water, or tolerates lower soil water contents than its neighbors. Then, by preventing neighboring plants from getting water, it would eventually eliminate the competition.

Concerning water management, there is increasing evidence that in water-limited environments total ET is more dependent on input precipitation than on type of vegetation cover (Zhang, et al. 2001; Wilcox 2002; Kuhn et al., 2007), as long as rooting depth and storage capacity of the soil are not changed (Seyfried and Wilcox, 2006). Therefore, replacement of vegetation cover using plants with lower interception losses could greatly affect the productivity of the landscape, as interception losses affect the amount of water that reaches the ground to refill the soil and will be available for transpiration.

There are two not mutually exclusive strategies that would allow plants to increase productivity in water-limited environments: enhancement of the intrinsic water use efficiency and increase of the ability to access water for transpiration. A great deal of work has been done to improve plant productivity in dry environments through genetic selection of material with higher water-use efficiency. However, low interception losses are not commonly taken into account in genetic improvement programs or in land management programs. Decreasing the interception of incoming precipitation would allow plants to access more water without the cost of other mechanisms like increasing rooting depth or decreasing minimum leaf water potential. Moreover, by reducing the interception losses without necessarily changing the intrinsic water use efficiency of the plants, the total water use efficiency - measured in terms of total annual ET/total annual productivity - could be largely improved at the landscape level.

Figures

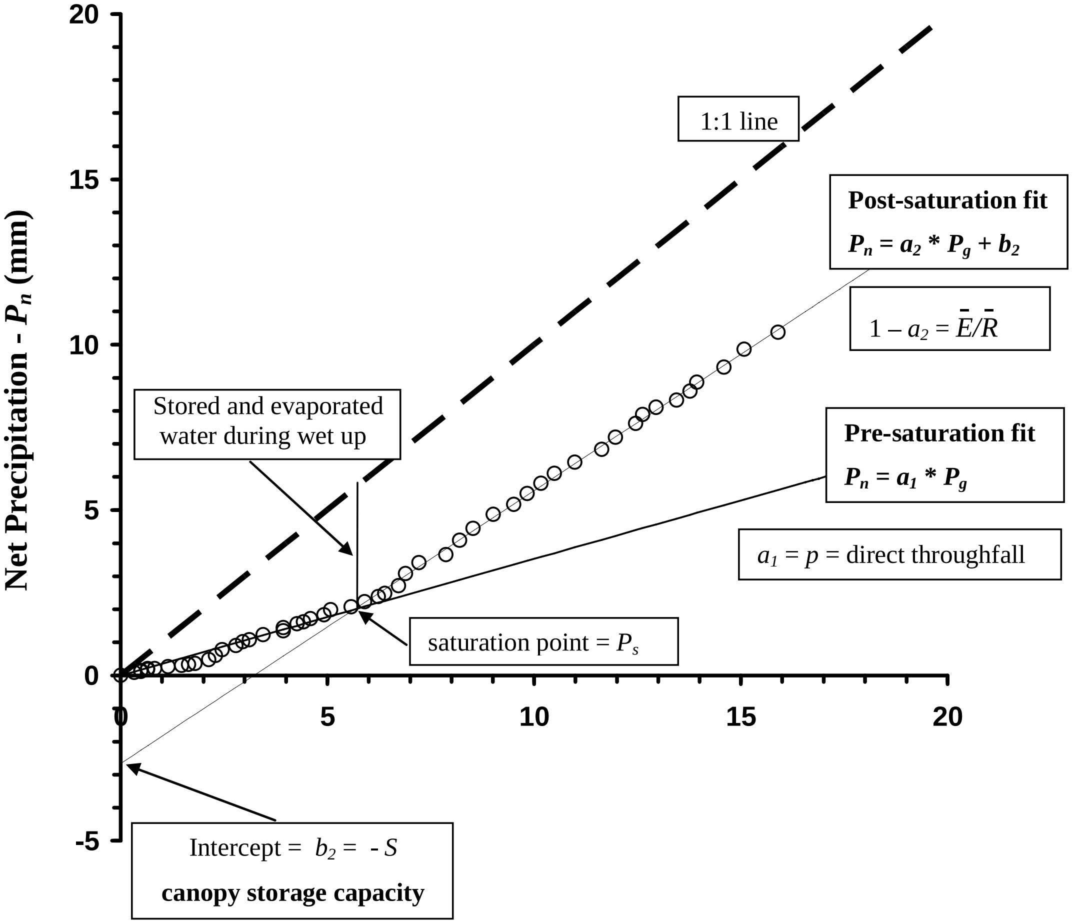


Figure 3.1 - Example of rainfall interception model parameterization using tipping buckets with troughs data. Rain event of day 108. Empty circles are 10 minute cumulative net precipitation (P_n) average in the low density cordilleran cypress stand. Dotted line is the 1:1 line and straight lines are linear regressions fitted to measured data.

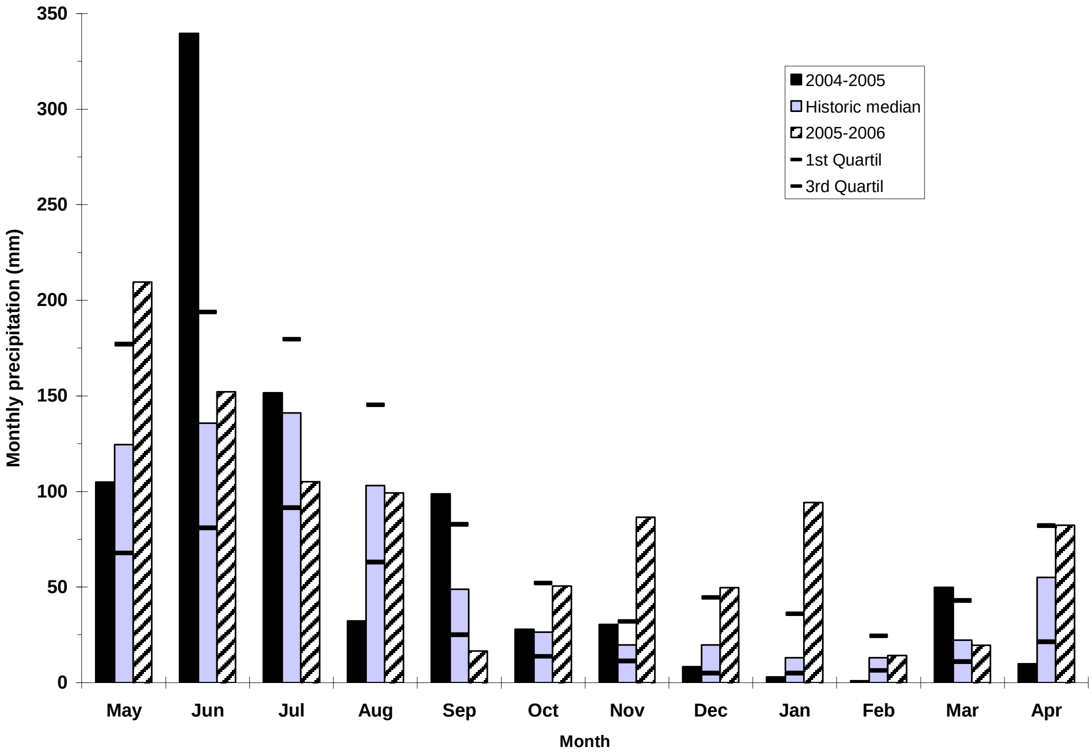


Figure 3.2 - Total monthly precipitation distribution of the measurement seasons compared to historic records (from 1981 to 2003) of precipitation data for Bariloche, Argentina (approx. 70 km south from study site).

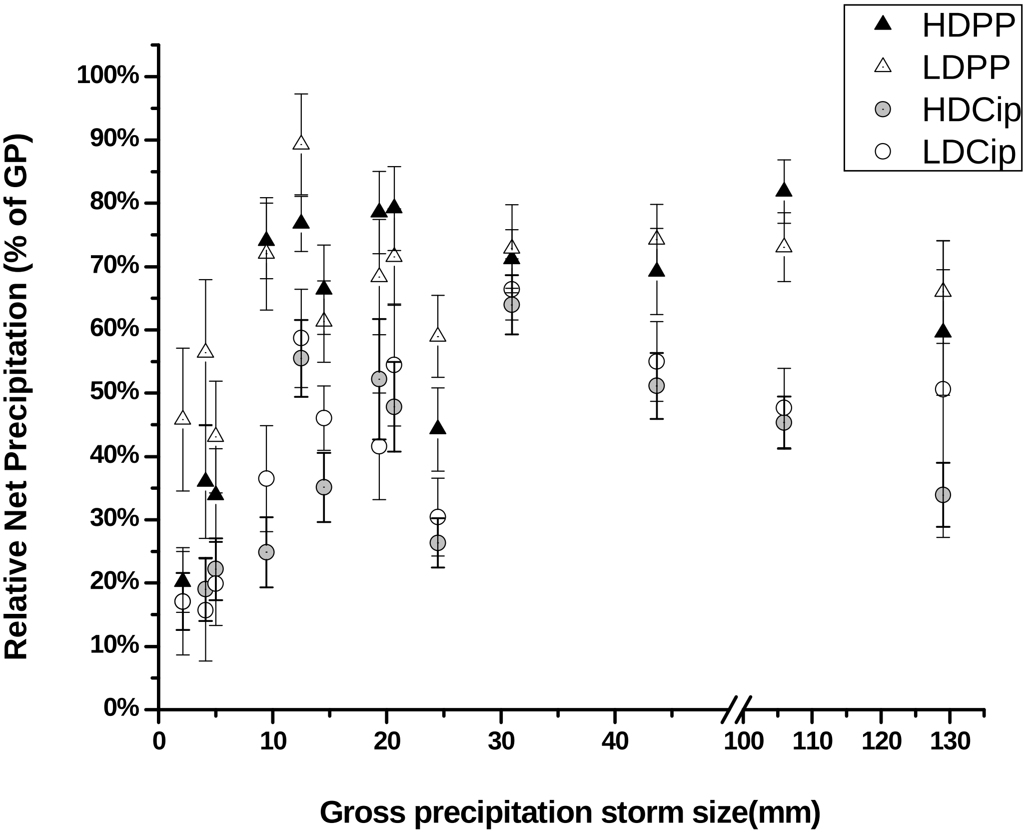


Figure 3.3 - Relative net precipitation per storm event as percent of gross precipitation per forest type, measured with PVC pluviometers from December 4th, 2005 to May 25th, 2006.

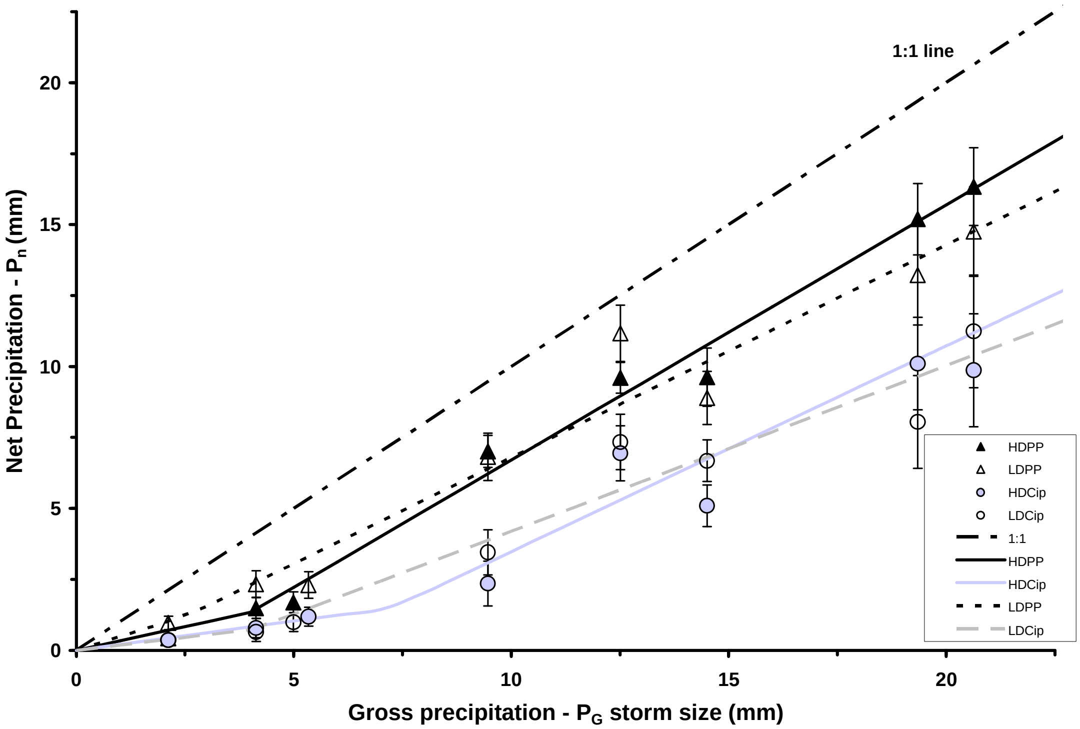


Figure 3.4 - Rain interception model parameterization. Two linear regression models were fitted to each forest type set of data of net precipitation (P_n) as a function of gross precipitation (P_G).

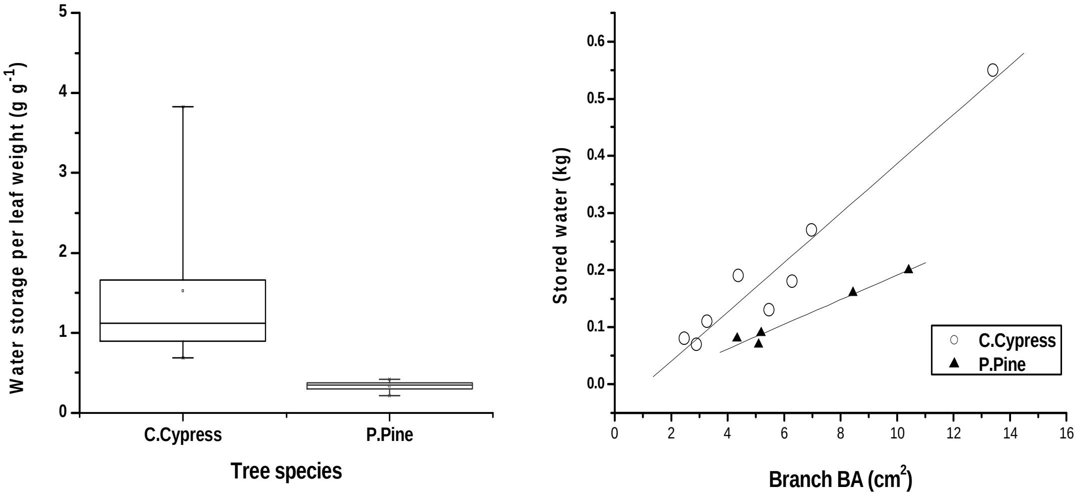


Figure 3.5 - Storage of water on branches under simulated rainfall. (a) Average water holding capacity per leaf biomass. (b) Stored water per branch as a function of basal area of the branch.

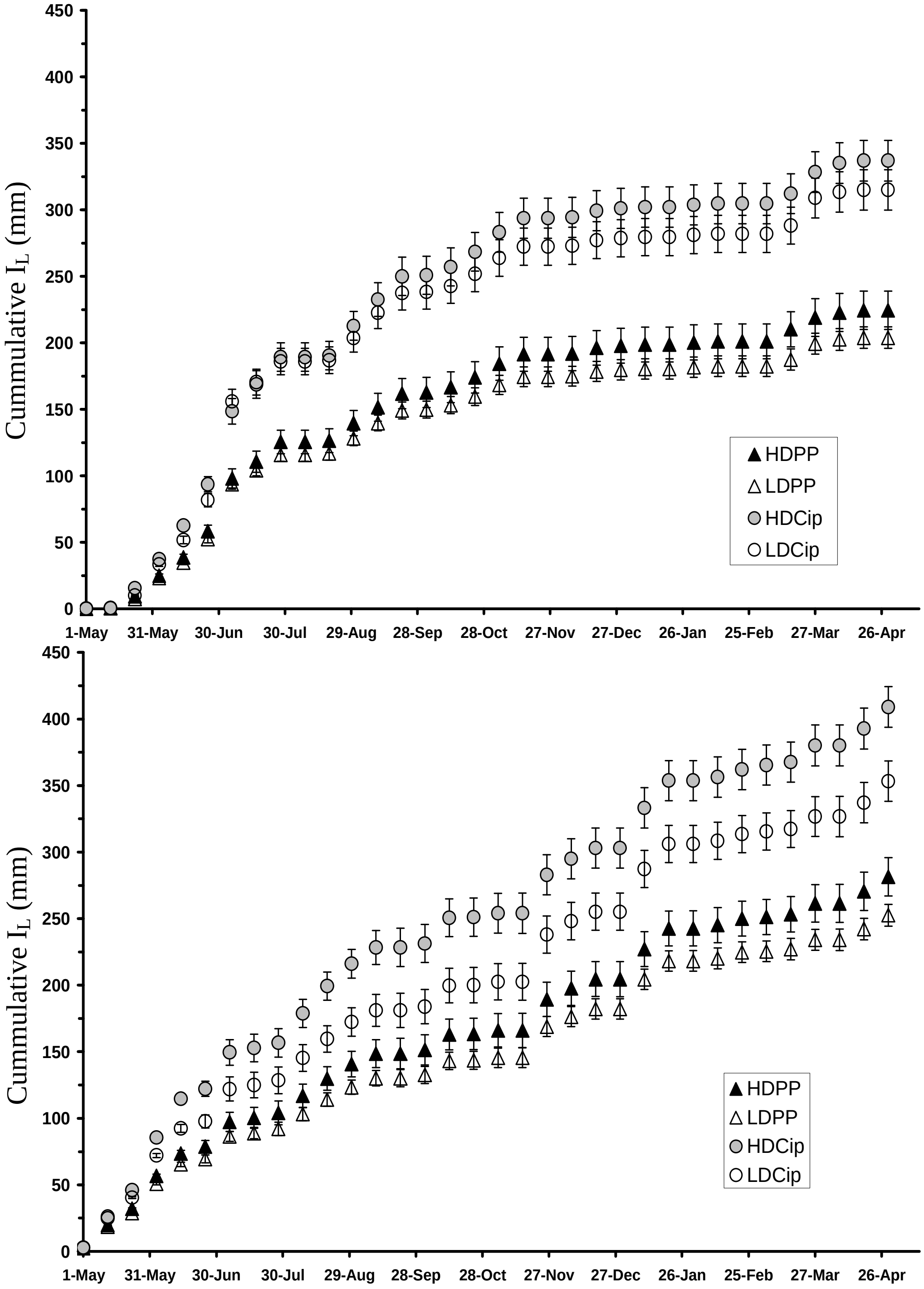


Figure 3.6 - Modeled Rainfall Interception Loss per site per year. Cumulative interception loss according to the simplified Gash model with corrected $\bar{E}/\bar{R}$ by the daily potential evapotranspiration. Error bars represent standard errors between the four model parameterizations per site.

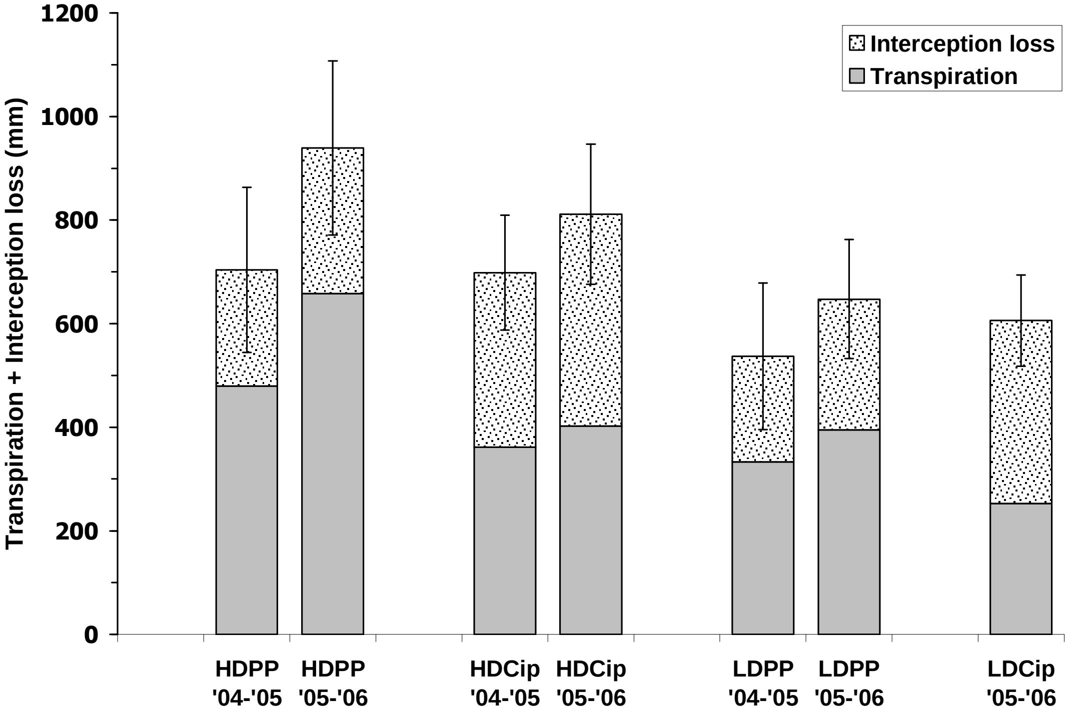


Figure 3.7 - Total Transpiration plus Interception Loss per site per year.

Cumulative interception loss according to the simplified Gash model. $\bar{E}/\bar{R}$ was corrected using the daily potential evapotranspiration. Error bars represent standard errors between the four model parameterizations per site plus standard errors of transpiration estimates.

Tables

Table 3.1 - Stand structure. Tree density (trees ha^{-1}), LAI (m^{2} leaf area m^{-2} ground area), Basal area (BA m^{2} ha^{-1}), sapwood area (SA m^{2} ha^{-1}), leaf morphology, presence of understory and spatial distribution of the measured plots.

	HDPP	**LDPP**	**HDCip**	**LDCip**
Trees ha -1	**1135**	**350**	**2662**	**668**
LAI	**9.0**	**3.1**	**9.1**	**4.6**
BA	**65.5**	**27.5**	**97.0**	**50.7**
Height	**14.5**	**13.7**	**14.1**	**15.1**
Leaf type	**15 cm needle**	**15 cm needle**	**1-3 mm scale**	**1-3 mm scale**
Understory	**None**	**Herbaceous**	**None**	**Herbaceous**
Distribution	**Homogeneous**	**Homogeneous**	**Clumped**	**Clumped**

Table 3.2 - Canopy hydraulic parameters per site and method of measurement. MRG are the parameters derived from the manual rain gauges measurements; TBT are the parameters derived from the tipping buckets with troughs measurements. *Direct throughfall (p),* gross precipitation necessary to saturate the canopy (*Ps*), canopy storage capacity (*S*) and average evaporation to rainfall rate ratio ($\bar{E}/\bar{R}$).

MRG	HDPP	LDPP	HDCip	LDCip
p	0.33	0.48	0.21	0.18
P_s (mm)	4.03	2.57	7.28	4.09
S (mm)	2.28	0.70	3.78	1.65
$\bar{E}/\bar{R}$	0.10	0.25	0.27	0.42
TBT	**HDPP**	**LDPP**	**HDCip**	**LDCip**
p	0.06	0.24	0.03	0.35
P_s (mm)	4.56	5.13	9.47	5.63
S (mm)	3.93	3.69	8.88	2.66
$\bar{E}/\bar{R}$	0.07	0.05	0.03	0.18

CHAPTER FOUR: USE OF A PROCESS-BASED MODEL TO ASSESS HYDROLOGICAL SUSTAINABILITY AND THE EFFECTS OF CLIMATE CHANGE ON PONDEROSA PINE (*PINUS PONDEROSA* DOUG. EX LAW.) PLANTATIONS IN NORTHWESTERN PATAGONIA, ARGENTINA

Julian A. Licata[1*], Dominique Bachelet[2], Gonzalo Caballé[3], Javier E. Gyenge[3], Maria Elena Fernández[3], Tomás M. Schlichter[3] and Barbara J. Bond[1]

([1]) Department of Forest Science, Oregon State University, 321 Richardson Hall, Corvallis, OR 97331, USA.

([2]) Department of Biological and Ecological Engineering, Oregon State University, 116 Gilmore Hall, Corvallis, OR 97331, USA.

([3]) INTA EEA Bariloche, Grupo de Ecologia Forestal, CC 277 (8400) Bariloche, Argentina.

Abstract

Understanding the interactions between the environment and vegetation is fundamental for determining the potential impacts of land use change on carbon and water cycles, as well as determine potential feedbacks to the atmosphere. In northern Patagonia, Argentina, the species most commonly used for afforestation projects is ponderosa pine (*Pinus ponderosa* Doug. ex. Laws), which has been used in approximately 80% of new plantations in the natural distribution area of native cordilleran cypress (*Austrocedrus chilensis (D. Don)* Pic. Serm. et Bizzarri. In previous studies we found that ponderosa pine plantations transpired more than cordilleran cypress stands of similar density during spring and summer, while cordilleran cypress stands had larger annual rainfall interception losses than ponderosa pine stands. However, there are no estimates available for the whole water budget for a complete year. Moreover, afforestation projects in temperate regions usually imply a long-term change in land use, and changes in global climate may alter forest ecosystems carbon and water fluxes. We parameterized a detailed process-based model to four forest stands of cordilleran cypress and ponderosa pine of different densities to predict annual water fluxes, and estimate potential outcomes under different climate change scenarios. Hydrological sustainability (defined as ET< precipitation) was evaluated under each scenario. High density ponderosa pine plantations were not hydrologically sustainable under any scenario, while high density cordilleran cypress stands were sustainable only under current climate conditions, but not in any future climate scenarios. Warmer and drier conditions, as projected by the

IPCC A1B scenario, indicate that stressful summer drought conditions will be worse than under current climate. Reductions of stand density in both native cordilleran cypress and ponderosa pine plantations would be necessary to reduce the impact of future climate change on water resources.

Introduction

The impact of human activities on carbon and water cycles, and their feedback on global climate are areas of major concern in the scientific community and the global political arena. Forest ecosystems are vital components of terrestrial environments that interact dynamically with carbon and water fluxes. Human activities regarding forest and land management have impacts on both water and carbon cycles. In the Kyoto protocol, afforestation with highly productive species has been proposed as a mechanism to increase the function of carbon sinks in land ecosystems (Schulze et al., 2002). However, increases in productivity are generally linked to an increase in water use (Farley et al., 2005; Nosetto et al., 2006). While carbon is being fixed through photosynthesis, water is being lost through transpiration. Both fluxes occur at the stomata level, which is regulated differently among plant species. Understanding the interactions between the environment and vegetation is fundamental for determining the potential impacts of changes in land use on carbon and water cycles, as well as potential feedbacks to the atmospheric environment.

In northern Patagonia, Argentina, the species most commonly used for afforestation projects is ponderosa pine (*Pinus ponderosa* Doug. ex. Laws), which has been used in approximately 80% of new plantations (SAGPyA, 1999). Recent studies have pointed out the lower invasiveness of ponderosa pine compared to the other species used in afforestation in the region (Sarasola et al., 2006), which makes it ‘safer’ to use than the other species. Nevertheless, the deliberate change in land vegetation cover by forest plantations from native to exotic species could have deleterious impacts on ecosystems processes. We will focus here on potential alterations of the water cycle by the introduction of ponderosa pine plantations compared to native cordilleran cypress forest stands (*Austrocedrus chilensis (D. Don)* Pic. Serm. et Bizzarri) in the forest-steppe ecotone at the base of the Andes Mountains in northwestern Patagonia, Argentina, where most of the plantations are being planted.

In a previous study, we found that planted ponderosa pine forests transpired more than cordilleran cypress stands of similar density during spring and summer (Licata et al., in press). Later, we measured differences in interception loss between the sites and found that cordilleran cypress stands had larger rainfall interception losses than ponderosa pine stands (Licata et al., in prep.). The magnitudes of the differences estimated for annual interception loss between the two species were comparable to the differences in spring and summer transpiration found between the stands. Unfortunately, however, while we feel confident in our measurements, the scope of our conclusions are somewhat limited by the fact that our data were not collected throughout the entire year. Also, there are no estimates available for all the components of the water cycle to close the annual water budget. Moreover, afforestation projects in temperate regions usually imply a long-term change in land use and climate change may alter forest ecosystems carbon and water fluxes (Nemani et al., 2003; Boisvenue and Running, 2006). Consequently, it is critical that reliable tools be used to predict the potential outcomes of natural and planted forests under future climate scenarios (Magnani et al., 2004).

Most climate models converge on an increase in temperature during the next century in southern South America (Boulanger et al., 2006; Vera et al., 2006).

Conversely, in a comparison of seven of the models used in the Intergovernmental Panel on Climate Change (IPCC) third assessment report, Boulanger et al. (2007) found that the predictions for precipitation strongly diverged from one model to another for South America. However, in the IPCC's most recent assessment report (fourth; AR4), the 21 climate models used unanimously predicted either no change or a decrease in precipitation for the southern Andean region of South America (Christensen et al., 2007). Under the A1B emission scenario, an intermediate scenario, the models used by the panel predicted a decrease in total annual precipitation of 25% by the end of the 21st century for the region that included our study site. In addition, the largest relative changes in precipitation were expected to occur during the summer months. In a drought-prone area, like in the forest-steppe ecotone where we conducted our study, where the driest part of the year naturally occurs during the summer, it becomes even more important to prevent changes in the water cycle since these could increase the risk of summer fires (Westerling et al., 2006), and drought-induced tree mortality (Allen, 2007). Understanding how changes in the physical environment could affect the vegetation and vice versa would help policymakers make sensible decisions about land use management. One approach for predicting changes in water fluxes through space and time is with the use of process-based models. Accurately describing the ecophysiological mechanisms involved in the control of water fluxes from the soil to the atmosphere allows us to predict what these fluxes would be under different environmental conditions.

The process-based model we used in this study is the soil-plant-atmosphere (SPA) continuum model (Williams et al., 1996). The SPA model is unique in that it explicitly inter-connects water and carbon dioxide fluxes by stomatal conductance behavior under different stimuli. It uses a detailed physiological approach, with a time step of thirty minutes and multiple canopy and soil root layers. Water supply to the leaf is regulated by the water potential gradient and the total soil-root-stem hydraulic resistance. Leaf water potential is calculated at each timestep and it is prevented to fall below a threshold value. Meteorological data like windspeed, photosynthetically active radiation (PAR), relative humidity, air temperature and precipitation are required to calculate micrometeorological conditions for each canopy layer. The SPA model has been applied successfully to several ecosystems, including temperate hardwood forests (Williams et al. 1996), Artic tundra (Williams et al., 2000), open ponderosa pine forests (Law et al., 2000; Williams et al., 2001a-b; Schwarz et al., 2004), and the Amazon rainforest (Williams et al., 1998; Fisher et al., 2007), demonstrating that it is an appropriate model to use for predicting seasonal changes in water and carbon fluxes for C_3 plant ecosystems.

The first objective of this study was to parameterize and validate the SPA model for forest stands of exotic ponderosa pine and native cordilleran cypress grown at different densities using data collected at our field sites, in order to estimate total annual water fluxes and assess their hydrological sustainability. Hydrological sustainability, for the purposes of this study, occurs when precipitation exceeds evapotranspiration.

The second objective was to use the parameterized model to predict the potential impact of future climate scenarios on seasonal and annual water fluxes in forest stands of exotic ponderosa pine and native cordilleran cypress.

Methods

Study site

Model parameterizations and validations were based on measurements made on four plots located in northwestern Patagonia, Argentina (40.29º S, 71.13º W). Plots consisted of a high density ponderosa pine (HDPP), a low density ponderosa pine with grass understory (LDPP), a high density cordilleran cypress stand (HDCip), and a low density cordilleran cypress stand (LDCip). Cordilleran cypress stands were the result of natural regeneration, whereas the ponderosa pine stands were planted for both timber production (HDPP) and timber and pasture production (LDPP). All plots were located within 500 m of each other in the Caleufu River valley. Edaphic conditions up to a tree height of 150 cm were assessed visually for similarity before installing the plots. The LAI of high density plots was approximately twice that of low density plots of the same species (Table 1). Ponderosa pine plots were pruned up to a height of four meters in 50 and 100 % of the trees in the high and low density plot, respectively.

Average annual rainfall (1980-98) at the site is approximately 800 mm (AIC, 2005) with a Mediterranean distribution, where approximately 80% of the annual precipitation occurs during fall and winter, mostly as rainfall. Maximum and minimum annual average temperatures are 17.1°C±0.5(SE) and 4°C±2.1(SE), respectively. Soils are deep (> 1.8 m) with a sandy loam texture and small pebbles.

The nature of this study required labor intensive measurements of stand level water fluxes and physiological characteristics of trees in each plot. Logistical and economical constraints limited the study to only one replicate per density and species.

However, plots were selected to be similar to those commonly planted in the region. Furthermore, the SPA model is highly mechanistic and accounts for most of environmental and biological variables known to influence transpiration and productivity in C3 plants. Thus, the experimental results presented here should be considered a case study, and modeled scenarios as indicative of potential outcomes of ponderosa pine plantations and cordilleran cypress having genetic backgrounds and developmental histories similar to the trees in this study.

Model description

The SPA model is a multiple layer process-based model, where the structure of the canopy is divided into 10 layers and the soil into 20 layers to represent the vertical variation in the physical and biological environment. The SPA model uses a pipe hydraulic scheme approach, and assumes that each canopy layer is served by an independent water supply system of roots. It also employs an unbranched hydraulic pathway scheme, using an electric circuit analogy approach. Every 30 minutes the absorption of photosynthetically active radiation (PAR) and other wavelengths, leaf boundary layer conductance, and soil water availability are determined for each canopy layer. Leaf water potential varies with transpiration, which is calculated using the Penman-Monteith equation. The variation of leaf photosynthetic capacity parameters changes with foliar nitrogen distribution among canopy layers, following Harley et al. (1992).

The key assumption governing the biological components of the model is that stomatal conductance (gs), is controlled to maximize carbon gain per unit nitrogen (N) within the limits set by the rate of water uptake and canopy water storage. The soil-roots sub-model has 20 layers, allowing for a detailed representation of root distribution through depth. The soil water content, conductivity and water potential of each layer are calculated every timestep, and regulate the soil-to-leaf water flux. Detailed explanations of the fundamental equations and structure of the model can be found elsewhere (e.g., Williams et al., 1996, Law et al., 2001; Williams et al., 2001). To better represent the vegetation of the low density stands, the original code in the SPA model was modified to allow for the presence of a low canopy layer that has access to water from the upper soil layers.

Field Measurements

Meteorological data

Half-hourly measurements of relative humidity, air temperature, wind speed, total solar radiation and gross precipitation were recorded by an automated low-maintenance meteorological station within 7 km of the study plots. A portable meteorological station was used to check for potential non-random differences between the study plots and the permanent meteorological station. Additional data from a highly maintained meteorological station located in the closest city (Bariloche, 70 km south of the study plots) was used to fill in gaps in the meteorological data. We determined the nonlinear relationships between the variables at both locations in order to fill in gaps in the field station data. The annual meteorological driver for the current-condition simulation model runs was calculated by averaging the meteorological conditions from the two growing seasons (2004-2005, 2005-2006) for which we had data. Precipitation data were not averaged between seasons, but instead half of the precipitation events measured in each season were included in the current conditions meteorological driver. Air temperature and precipitation were then corrected to match the historic (23 year) monthly averages.

Soil properties

Soil texture variation with depth was determined for each plot by taking samples every 20 cm, up to a depth of 160 cm. Each sample analyzed was a composite of 18 samples from different locations within each plot. Textural analyses were performed by the soil laboratory at INTA EEA Bariloche (Figure 4.1).

Soil water holding at field capacity was measured on undisturbed soil samples from four samples of the topmost 30 cm of the mineral soil profile. Plastic tubes 10.1-cm in diameter with a sharpened edge were pounded vertically into the soil. Care was taken to minimize compaction and to excavate samples while maintaining the soil structure intact. The lower end of the core was sealed with plastic which was then perforated in the lab to add water to the tubes until saturation was reached. Samples where capped to reduce evaporation and were allowed to drain until the weight was constant for 24 hours.

Bulk density was measured from 4 samples taken from each plot, at each of three depths, 10, 20 and 40 cm, and 2 samples from 80 and 140 cm at each plot (Figure 4.2).

Hydraulic conductivity and soil water potential of every 20 layers of soil are calculated by the SPA model at each timestep as functions of soil moisture content according to the generalized equations described by Saxton et al. (1986). The input parameters of these equations are percent of sand and clay, and proportion of saturated pores at field capacity. An improved and more commonly used tool (Rosetta; Schaap et al., 2001) than the equations described by Saxton et al. (1986) have been developed to predict soil hydraulic properties from several arrays of measured parameters from the soil. Therefore, instead of using the measured texture data as input parameters for the model, we generated the soil hydraulic parameters using measured values of sand, silt and clay percent, bulk density and water holding capacity as inputs of the Rosetta software to create the soil characteristic curves according to van Genuchten (1980) equations. Then, sand and clay percent parameters were varied as input on the Saxton equations extracted from the SPA model to match the soil characteristic curve of each plot generated by the Rosetta-van Genuchten method (Figure 4.3).

Leaf photosynthetic capacity

The algorithms used to model photosynthesis in the SPA model are based on kinetics described in Farquhar and Von Caemmerer (1982), where photosynthesis is limited by Vcmax (rubisco activation rate), Jmax (light saturated rate of carboxylation limited by electron transport), and R_d (rate of respiration in the presence of light). The maximum rates of these parameters were calculated by performing net CO_2 assimilation versus calculated internal carbon dioxide measurements (A–Ci curves). A-Ci curves were calculated in the field using a LiCor-6400 portable photosynthesis system (LICOR, Lincoln, Nebraska) on one-year-old leaves growing under full sun conditions, on five trees per species. The projected foliage area in the cuvette was quantified digitally using image J software (Rasband, 2006) on digital images of the green foliage. Average Vcmax, Jmax and Rd were used as input parameters for the top of the canopy and adjusted proportionally to the foliar nitrogen content of each canopy layer. Our photosynthetic parameters might underestimate the maximum potential for photosynthesis of both species, as measurements were taken towards the end of the growing season. Jmax and Vcmax usually reach their maximum values at the middle of the growing season (Panek, 2004).

Vertical distribution of leaf nitrogen

Total nitrogen concentration (% weight) of leaf samples was measured using samples taken from three vertical positions in the canopy (top, middle and bottom) and three positions in the horizontal distribution of foliage within the tree (external, middle and internal). Nine samples per plot were analyzed, and each analyzed sample was a pool of leaves from 5 different trees. Nitrogen concentration was transformed to a leaf area basis by multiplying it by the specific leaf area (SLA; cm^2 g^{-1}) of each canopy layer. SLA was measured at five heights on five trees per plot, except for the LDPP plot where the lower canopy layer was nonexistent due to pruning. The leaf area of these 95 leaf samples was determined by digitally scanning the fresh samples, together with pieces of black cardboard having known areas, and the images later analyzed using ImageJ software (Rasband, 2006). The pieces of cardboard used as references were cut into shapes similar to the leaves in order to avoid border effects on the calculation of the area. Leaf samples were dried at 70°C for 72 hours and weighed with a 1 mg precision balance. Leaf area vertical distribution was estimated from side pictures of the trees, which we considered more appropriate than assuming an equal distribution of leaf area from the top to the bottom of the canopy. Phenological qualitative observations were transformed into a quantitative change of leaf area, assuming a 25% leaf turnover per year and a linear increase in leaf area from bud brake until full stem expansion, and from beginning to end of leaves dropping.

Root Biomass distribution

Root biomass was measured at five locations per plot at 20, 40, 80, and 140 cm depths, and in three locations per plot at 180 cm deep. Samples were extracted with 10 cm diameter cores pounded horizontally in previously dug pits. Roots were not separated into live and dead roots. These data were used as a starting point to estimate fine root density per layer and rooting depth, through an iteration process where root density was varied as a proportion of measured data, up to 140 cm deep. An exponential decline was assumed for roots deeper than 140 cm.

Root resistivity in the model is constant along the rooting profile, which could lead to a misrepresentation of the rooting density profile determined by an iteration procedure as shallow roots may have higher resistivity than deep roots (Jackson et al., 2000). In addition, fine root biomass has been has been found to vary seasonally (Tierney et al., 2003), while it is maintained constant throughout the season in the model.

Model parameterization

The SPA model requires values for a large number of stand level parameters in order to accurately represent the processes involved in carbon and water fluxes.

Therefore, data available from previous studies at the same site or species were used when possible. Estimates of LAI (Licata et al., in press), minimum leaf water potential (Schwarz et al., 2004; Gyenge et al., 2005), and rainfall interception parameters (Licata et al., in prep) were available.

Two critical parameters describing stand hydraulics are practically impossible to measure accurately: above ground plant conductivity (G_p), and root resistivity (R_R). Consequently, several nested sensitivity analyses were performed and compared to the daily transpiration estimates from Licata et al. (in press) to determine the value of these parameters. The parameterization process can be divided in two phases. In the first phase, only G_p and R_R were varied to visually fit model output to transpiration and soil water content data. In the second phase, all measured and estimated variables were varied ± 20% to select for the best set of parameters through a more objective methodology. The goodness-of-fit indicators used to select the best set of parameters were the root mean square error (RMSE), and an integrated indicator (I_{index}) of the goodness-of-fit of the regression parameters between model vs. measured data. An I_{index} of 1.0 would indicate that all parameters in the regression between model vs. daily measured transpiration are equal to the ideal value (slope = 1 , intercept = 0, and $r^2 = 1$). Approximately 3,000 runs per plot were performed for the iterative parameterization process. Among all the runs, the set of parameters selected corresponded to the one with the lowest RMSE of the 10% with the highest I_{index}.

Model validation

Two growing seasons of transpiration estimates (2004-2005, and 2005-2006) were available for most of the stands, with the exception of the LDCip, where there were only estimates for the 2005-2006 season. Precipitation was markedly different between the two seasons. The 2004-2005 season had a drier than normal summer, while the 2005-2006 season was an exceptionally rainy summer. For each of the stands, one season of estimates was used to calibrate the stands and the other season was used to validate the parameterizations. Ponderosa pine stands were calibrated using the 2004-2005 transpiration data, while the cordilleran cypress stands were calibrated with the 2005-2006 measurements.

Potential future climate scenarios

As a reference for the potential future scenarios of global climate change, we used the predictions made in the IPCC report, scenario A1B (Christensen et al., 2007). There is general agreement among the models used for the report that air temperature in the region will increase about 2.0 ˚C (mean annual temperature), corresponding to a 2.5 ˚C increase during December, January and February (summer) and a 1.5 ˚C increase for the period of June, July and August (winter). Total annual precipitation, on the other hand, is expected to be reduced by approximately 25% by the end of the century. Due to the natural Mediterranean distribution of precipitation, the 25% decrease in mean precipitation corresponds to a 50% decrease during summer months, and a 20% decrease during the winter. None of the 21 models used in the IPCC predictions showed an increase in annual, winter or summer precipitation.

We generated nine meteorological data drivers (input data files) in which the IPCC predictions represented the center of variation among scenarios. The potential climate scenarios corresponded to a zero, two and four degree (˚C) increase in air temperature, and a zero, 25% and 50% decrease in total annual precipitation over current climate conditions, as well as all possible combinations of these changes.

The impact on water resources was evaluated by comparing the total annual and seasonal changes in evapotranspiration of the forest types under the potential future climate change scenarios against the current conditions predicted by the SPA model. All vegetation types were assumed to be growing under the same edaphic conditions for the future climate scenarios. A 70 % sand and 5% clay soil was used for the model runs. Soil profile was assumed to reach field water capacity at the beginning of each modeled year.

Results

Analysis of soil and vegetation properties used to parameterize the model

Both vegetation types had the same root conductivity of 0.2 mmol g^{-1} s^{-1} Mpa^{-1} (Table 4.2) and most of the plant resistance to water flow was located above ground (approximately 8 times larger than below ground). Above ground plant conductivity was 5.8 and 6 mmol m^{-1} s^{-1} Mpa^{-1} for high and low density ponderosa pine stands, respectively and 4 mmol m^{-1} s^{-1} Mpa^{-1} for both cordilleran cypress stands. Maximum photosynthetic parameters were similar between species, and were only different between stands because of the different nitrogen content per leaf area basis among the stands (Figure 4.4).

Analysis of root biomass distribution

Measured root biomass density was larger in the HDCip site than in all the other sites by 2.4, 1.9 and 1.5 times for HDPP, LDPP and LDCip respectively. Root biomass of both cordilleran cypress stands was similar in the first 10 cm, while both ponderosa pine stands had similar values of root density along the entire rooting profile (Figure 4.5). Fine root biomass results from the fitted parameterization of the SPA model yielded an average proportion of fine/total root biomass of 0.14, 0.06, 0.14 and 0.3 for the HDPP, HDCip, LDPP and LDCip, respectively. Larger fine over total root biomass of ponderosa pine stands compared to cordilleran cypress stands could be due to the older age of the cypress stands (approx. 30 years older). As fine roots have a faster turnover than coarse roots (Gill and Jackson, 2000), the proportion of fine/total root biomass will decrease over time as a buildup of coarse roots occurs. The fitted parameterization of SPA yielded deeper rooting depths ponderosa pine stands than cordilleran cypress stands. Fitted rooting depths were 3.5, 2.75, 3.3, and 2.5m for HDPP, HDCip, LDPP and LDCip, respectively. Moreover, while cypress stands had 60% of their fine roots in the upper 30 cm, ponderosa pine stands had only 36% of their fine roots in the upper 30 cm.

Model parameterization

The set of parameters selected through the iteration process for each of the forest stands accurately predicted the measured daily transpiration of the four forest stands throughout the season of measurements (Figure 4.6). Spa model predictions explained most of the variation registered throughout the calibration of the growing season, yielding r^2 values of 0.86, 0.90, 0.84 and 0.85 for HDPP, HDCip, LDPP and LDCip, respectively. Regression analyses of predicted against observed data (Figure 4.7a) revealed than none of the fitted curves differed from a 1:1 line. Slope values were 1.07, 0.99, 1.01 and 1.00, while intercept values were 0.00, 0.02, 0.02 and 0.11 for HDPP, HDCip, LDPP and LDCip, respectively. Root mean square errors were 0.54, 0.37, 0.18 and 0.24 mm d^{-1}, representing a relative error of 23, 13, 10 and 15 % for HDPP, HDCip, LDPP and LDCip, respectively.

Model validation

The parameterized SPA model accurately predicted daily transpiration measurements of the growing season data (Figure 4.8). Not surprisingly, the goodness-of-fit indicators for the validation year were slightly poorer than for the calibration year, but still satisfactory. In the ponderosa pine stands, where the calibration was performed during the dry year growing season, the model slightly underestimated transpiration during the rainy year growing season by an average of 0.13 and 0.4 mm d^{-1} for HDPP and LDPP respectively. On the other hand, in the HDCip stand, where the calibration was performed during the rainy year growing season, model predictions for the dry year growing season had the same bias of 0.01 mm d^{-1} as the rainy year (calibration) predictions. Model predictions for all the validation growing seasons yielded r^2 values of 0.82, 0.87 and 0.78 for HDPP, HDCip and LDPP, respectively (there were no data available to validate the LDCip parameterization). Regression analysis of predicted against observed data for the validation growing seasons (Figure 4.7b) yielded slope values of 0.97, 0.81 and 1.01 while intercept values were -0.01, 0.26 and -0.32 for HDPP, HDCip and LDPP, respectively. Root mean square errors were 0.68, 0.36 and 0.54 mm d^{-1}, representing a relative error of 17, 14 and 22 % for HDPP, HDCip and LDPP, respectively.

Model predictions for the current climate

The annual precipitation used for the current climate meteorological driver was 799 mm y^{-1}, distributed mainly in fall-winter (80% of the annual total). Mean annual air temperature was 7.4 ° C, with February being the warmest month with an average temperature of 15° C, and July being the coldest month with an average temperature of 1.6 ° C.

Total annual evapotranspiration estimates for current climatic conditions using the parameterized SPA model were largest in the HDPP stand (Figure 4.9). Absolute total annual ET was 960, 799, 628 and 557 mm y^{-1} for HDPP, HDCip, LDPP and LDCip, respectively. Transpiration was the main component of ET in all stands, representing 78.2, 65.6, 70.7 and 57.6 % of total annual ET for HDPP, HDCip, LDPP and LDCip, respectively. Both high density stands showed a decline in ET beginning in early summer (January) due to the reduction in soil water, while low density stands maintained their ET rate until the beginning of the fall (March; Figure 4.9). Modeled runoff was larger in the low density stands than in high density stands, and was larger in ponderosa pine stands than in cordilleran cypress stands, although this may be an artifact of the assumption that all forest stands started the water year with soils at field water capacity. Only in the high density ponderosa pine stand was the annual water balance negative, suggesting an imbalance between the input precipitation and the output of water from the natural system. The parameterized SPA model predicts a decline in soil water storage of 130 mm in the HDPP, while in the rest of the sites there was an excess of water of 1, 171, and 267 mm for HDCip, LDPP and LDCip, respectively.

Future climate scenarios - air temperature increase

The increase in air temperature without any changes in total precipitation caused an increase in total annual ET on all forest stands. However, the effect of increasing temperature on total annual ET was larger in the low density stands than in the high density stands independent of species. Total annual ET was 39.0, 26.1, 54.0 and 29.8 mm y^{-1} greater than current conditions for a 2° C increase in air temperature on HDPP, HDCip, LDPP and LDCip, respectively. The 4°C increase scenario yielded increases in ET of 68.8, 49.2, 107.4 and 56.9 mm yr^{-1} for HDPP, HDCip, LDPP and LDCip, respectively.

The seasonal variation in ET in the high density stands showed an earlier onset of summer drought under the warmer scenarios. While the increase in air temperature created a higher ET rate at the beginning of the season, by the middle of the summer warmer scenarios showed a smaller ET rate than the current climate (Figure 4.10). On the other hand, in the low density stands the increase in ET rate in warmer scenarios was maintained throughout the season (Figure 4.11). The absolute difference in ET under current conditions for the winter months (JJA) with a 2° C (4° C) increase scenario was 12.9 (13.8), 10.7 (11.1), 7.4 (8.8) and 6.6 (7.6) mm for HDPP, HDCip, LDPP and LDCip, respectively. The absolute difference in ET under current conditions for the spring months (SON) with a 2° C (4° C) increase scenario, was 32.0 (30.6), 15.3 (13.0), 18.4 (18.2) and 10.2 (9.2) mm for HDPP, HDCip, LDPP and LDCip, respectively. The absolute difference in ET under current conditions for the summer months (DJF) with a 2° C (4° C) increase scenario was 1.3 (-6.7), -3.2 (-3.9), 17.4 (16.3) and 6.0 (5.2) mm for HDPP, HDCip, LDPP and LDCip, respectively. The absolute difference in ET under current conditions for the fall months (MAM) with a 2 °C (4° C) increase scenario was -7.2 (-7.9), 3.3 (2.9), 10.8 (10.2) and 7.0 (5.6) mm for HDPP, HDCip, LDPP and LDCip, respectively.

Future climate scenarios - reduction in precipitation

The decrease in total annual precipitation with no changes in air temperature caused a decrease in total annual ET for all forest stands. However, the effect of reductions in annual precipitation on total annual ET was larger in the high density stands than in the low density stands, independent of species. Total annual ET decreased 51.6 (159.2), 40.5(124.3), 16.5(36.7) and 22.4 (56.5) mm y^{-1} from current conditions for 25% (50%) reductions in annual precipitation in HDPP, HDCip, LDPP and LDCip, respectively.

The seasonal variation of ET in high density stands showed an earlier onset and steeper reductions in ET because of drought summer stress under precipitation reduction scenarios, which caused a reduction in transpiration rates (Figure 4.12). The reduction of annual precipitation had a negligible impact on ET rates at the beginning of the season, but the reductions in ET caused in the high density stands starting in the middle of the summer were steeper than those caused by the temperature increase scenarios (Figures 4.10 and 4.12). On the other hand, in the low density stands, the reduction in precipitation rates caused reductions in ET, mainly through reductions in interception loss. ET differences among scenarios were larger in the LDCip plot than in the LDPP plot and were distributed evenly throughout the season (Figure 4.13). The absolute difference in ET under current conditions for the winter months (JJA) with a

25% (50%) decrease scenario was 0.1 (-0.9), 0.4 (-2.3), -0.2 (-1.4) and -0.1 (-1.8) mm for HDPP, HDCip, LDPP and LDCip, respectively. The absolute difference in ET under current conditions for the spring months (SON) with a 25% (50%) decrease scenario was -4.6 (-7.9), -10.8 (-11.9), -4.0 (-3.6) and -4.6 (-9.3) mm for HDPP, HDCip, LDPP and LDCip, respectively. The absolute difference in ET under current conditions for the summer months (DJF) with a 25% (50%) decrease scenario was -13.7 (-47.1), -14.8 (-35.5), -7.2 (-6.4) and -9.3 (-11.0) mm for HDPP, HDCip, LDPP and LDCip, respectively. The absolute difference in ET under current conditions for the fall months (MAM) with a 25% (50%) decrease scenario was -33.4 (-51.6), -15.3 (-34.4), -5.2 (-8.9) and -8.5 (-12.0) mm for HDPP, HDCip, LDPP and LDCip, respectively.

Future climate scenarios - combined effect of temperature and precipitation

Variations in total annual ET were caused mainly by changes in total annual transpiration (Figure 4.14). In a linear regression analysis of ET and transpiration for all the plots, the general r^2 was 0.94. The r^2 values from the individual plot regressions were 0.99, 0.91, 0.90 and 0.61 for HDPP, HDCip, LDPP and LDCip, respectively. The reduction in precipitation reduced differences in annual ET between temperature scenarios in the high density stands, while it enhanced the differences in the low density stands (Figure 4.15). Changes in total annual precipitation were more important than temperature changes in the high density stands regardless of the species. On the other hand, in the low density stands temperature changes had a greater effect on the total annual ET than changes in precipitation, regardless of the species (Figure 4.15). Total annual ET increased an average of 9.3, 9.7, 27.2 and 14.1 mm per degree Celsius increase in air temperature for HDPP, HDCip, LDPP and LDCip, respectively. Total annual ET decreased, on average, at a rate of 3.4, 2.3, 0.7 and 1.1 mm per every 1% reduction in annual precipitation for HDPP, HDCip, LDPP and LDCip, respectively.

Discussion

There is general consensus among scientists that global climate has been affected by human changes in the carbon cycle (Le Treut et al., 2007). Among the achievements of the Intergovernmental Panel on Climate Change (IPCC), one of the most important was sending an unequivocal message to the policymakers that climate change is occurring, and that actions should be taken to mitigate ensuing impacts on global, regional and local economies. The scientific community, then, must provide tools and information to help policymakers evaluate the costs and benefits to society and help them make rational decisions (Lawton, 2007). Decisions regarding land use management have direct influences on climate change and at the same time generate feedback on water and carbon fluxes (Marland et al., 2003). In northern Patagonia, the Argentinean government is subsidizing the use of fast-growing species in forest plantations to help develop rural economies and to reduce the environmental impact of the destruction of native forests (SAGPyA, 1999). The main species used for afforestation in the region is ponderosa pine, which is commonly planted at high densities (>1200 trees ha-1). Most of the plantations are being located in the natural distribution area of cordilleran cypress. Understanding the interaction between current vegetation cover, water resources and climate could help prevent potential damages to the environment before widespread changes in land cover occur. The purpose of this study was to parameterize a detailed process-based model to forest stands of exotic ponderosa pine and native cordilleran cypress grown at different densities, in order to estimate total annual water fluxes and assess their hydrological sustainability. In addition, we used the parameterized model to evaluate the potential effects that changes in temperature and precipitation of future climate scenarios could have on the seasonal and annual water fluxes of ponderosa pine and cordilleran cypress stands.

Model parameterization and validation

The detailed structural and physiological measurements taken on the four forest stands in our study allowed us to parameterize the SPA model, and accurately explain the interactions between the plants and their physical environment. The iteration process to fit the hydrologic parameters yielded values similar to those found by Schwarz et al. (2004) for a mature ponderosa pine stand in Oregon that also had a similar height as the stands in our study (~14 m tall). The fit against the measured data for the calibration runs was successful as it explained more than 80 % of the observed variation in daily transpiration for all forest stands; the values of slope and intercepts did not differ from 1:1. Despite lower goodness-of-fit indicators for the model validation, the parameterized SPA model accurately predicted transpiration in the second set of data (the growing season not used for calibration) that had a strongly contrasting precipitation regime, suggesting a good representation of vegetation processes.

The largest discrepancies between model predictions and field measurements of transpiration occurred during the two days after rain events in the summer of the dry year. The SPA model predicted larger transpiration rates for those days than were actually measured. A possible explanation for this pattern could be that after a long drought season the roots located in the upper layers of the soil lose functionality by cavitation. Therefore, transpiration response after a rain event is delayed until water reaches the functional roots, or roots become functional again. In the model predictions, on the other hand, roots do not lose their functionality, nor is their resistivity changed at any time throughout the season. Consequently, the model predicted that trees started transpiring at higher rates as soon as soil water was available.

Model predictions for the current climate

Estimates of total annual ET by the parameterized SPA model suggest that the high density ponderosa pine plantations (HDPP) typically used in timber production, like the one in this study, are not hydrologically sustainable as total ET exceeds input precipitation in the forest-steppe ecotone of northwest Patagonia under current meteorological conditions. This could translate into a reduction of stored soil water over time and a reduction in productivity when the stored water becomes depleted. It is also possible that the high density ponderosa pine system is using excess water from other areas of the landscape that drains into the root zone of the ponderosa pine plantation. In that case, water yield in the catchment would be reduced and, potentially, the water table level lowered, as water that usually leaves the system is now used to refill the soil under the high density ponderosa pine plantation. In the high density cordilleran cypress stand there was a close match between input precipitation and total ET. Despite the classification of ‘hydrologically sustainable’ adopted in this study, this result also suggests that if the meteorological data used in the study represents the average climatic conditions, then on half of the years the high density cordilleran cypress stand will be using more water than is input. In addition, soil water discharge is not considered in the hydrological sustainability concept and with precipitation occurring mostly when transpiration is minimal, then it is very likely that there is going to be some water discharge in most of the years unbalancing the water budget. In the low density forest stands of both species the hydrological sustainability does not appear to be compromised under current climate.

Interestingly, total annual ET predictions by the SPA model present a different picture than our previous estimates for canopy ET in the same stands (Licata et al., in prep). This could be attributed to several differences between the methods used for each estimate. In the previous estimates (Licata et al. in prep), transpiration was only calculated for the spring and summer and interception loss was estimated for the whole year, while in the SPA model predictions of ET corresponded to transpiration, soil evaporation and interception losses for the whole year. The meteorological data driver used in the SPA model predictions was not identical to the data from either of the growing seasons measured in the previous study, but was instead a mix of both growing seasons, plus data adapted from a nearby location for the end of the fall and winter. Moreover, the meteorological data driver was modified to make it match historic monthly averages of temperature and precipitation. Furthermore, the interception algorithms between the two methods are conceptually different and operate at different timescales. Although the estimates of interception loss between the two methods were fairly similar during the calibration period, the estimates from the previous method were double those from the SPA model for the rest of the year. In both cases, the cordilleran cypress stands had almost twice as much interception loss as the ponderosa pine stands. It is not possible to determine which of the methods is more accurate unless direct measurements of interception loss are carried out during the whole year. We have reason to believe that both methods have their strengths and weaknesses, and most likely the real values are somewhere in between. A greater interception loss on all the stands, would reduce the differences between stands of different species, but would also result in less sustainable situations for all forest stands under all the climate change scenarios we modeled.

Model predictions for future climate scenarios

SPA model predictions suggest that both natural cordilleran cypress and exotic ponderosa pine plantations in the region would be affected by changes in future climate, as predicted by the fourth IPCC assessment report. Warmer and drier environmental conditions will reduce transpiration more drastically during the summer in high density ponderosa pine plantations than in any other forest stand. Predicted changes in both precipitation (decrease) and temperature (increase) indicate that stressful summer drought conditions will be worse than under current climate conditions. Moreover, the negative impact on total ET will be larger in the high density stands than in low density stands. At the same time, high density stands showed a tighter relationship between ET and transpiration than low density stands, suggesting that high density stands may have higher variabilities in productivity than the low density plots because of climate variability.

The increase in air temperature caused an increase in evaporative demand and, therefore, an increase in the ET rate for all forest types when water availability was not limiting. Therefore, the effect of increasing temperature on total annual ET was larger in the low density stands than in the high density stands, independent of species. High density stands showed a larger increase in ET than low density stands at the beginning of the season because of the increase in air temperature, but also showed an earlier onset and a more severe drought impact starting in early summer.

The different impacts of temperature and precipitation changes on transpiration in the forest stands suggest that the identification of a limiting climatic driver to plant growth, as proposed by Nemani et al. (2003) on a global scale, could be modified at the local level by vegetation structure. While in high density stands of this study the limiting factor was water availability, in the low density stands, the limiting factor was mean air temperature.

Management implications

The hydrological sustainability results under current climatic conditions suggest the need for land use management regulations at the catchment level to mitigate the impact the typical high density ponderosa pine plantations will have on water resources before they become widespread. The complete coverage of a particular catchment with high density ponderosa pine plantations could induce an imbalance of the natural water cycle that would likely reduce the water that gets to streams. Modification of the natural cycle of water courses could result in permanent deleterious consequences on the environment.

Current climate change projections for the region would cause an earlier and more severe manifestation of summer drought in high density forest stands, which may also impact natural fire regimes. Earlier spring conditions and drought manifestation have already been shown to increase fire frequency (Westerling et al., 2006) and an increase in burn area was predicted under IPCC climate scenarios (Lenihan et al., 2007) in the western United States, where the precipitation regime is similar to that experienced in northern Patagonia.

Reductions in stand density would improve forest stand resistance to drought stress and help reduce fire risks (Allen et al. 2002). Higher densities and increased drought stress would also predispose forest stands to more frequent and severe insect attacks (Waring and Pitman 1985; Allen, 2007). The dispersal of *Sirex noctilio,* a damaging plague of pine trees already present in the region, is know to be favored by the presence of stressed trees (Wylie, 2001). Managing the stand density of forest stands could be crucial to protect forest investments.

Caveats

It is important to highlight the fact that the forest stands evaluated in this study were located in the best position in the landscape for forest growth; at the bottom of the valley with very deep (>2.5 m) and well-drained soils. To assess the sustainability of ponderosa pine plantations at the catchment level, it would be necessary to further estimate the ET requirements of forest stands in other positions in the landscape and determine how much of each landscape position makes up the area as a whole. It is very likely that plantations on steep hillslopes, in upper elevations or on shallower soils would have a lower ET than the trees measured in this study.

SPA model predictions for future climate scenarios may not be totally realistic since the assumption of soils under field capacity at the beginning of the season may not be achieved under lower precipitation scenarios for the high density stands, as assumed here (to provide similar initial conditions for the model runs). Under persistent water deficit conditions, plant communities would eventually modify their structure to reduce evapotranspiration and be in equilibrium with the available water. Changes in total leaf area, or modifications of the leaf area per sapwood area ratio have been documented in ponderosa pine stands (McDowell et al., 2006). However, none of these structural changes would occur without detrimental effects on the environment and plant growth. Reductions in leaf area imply tree mortality and modifications in leaf area to sapwood area ratio imply persistent drought stress conditions in individual trees.

Figures

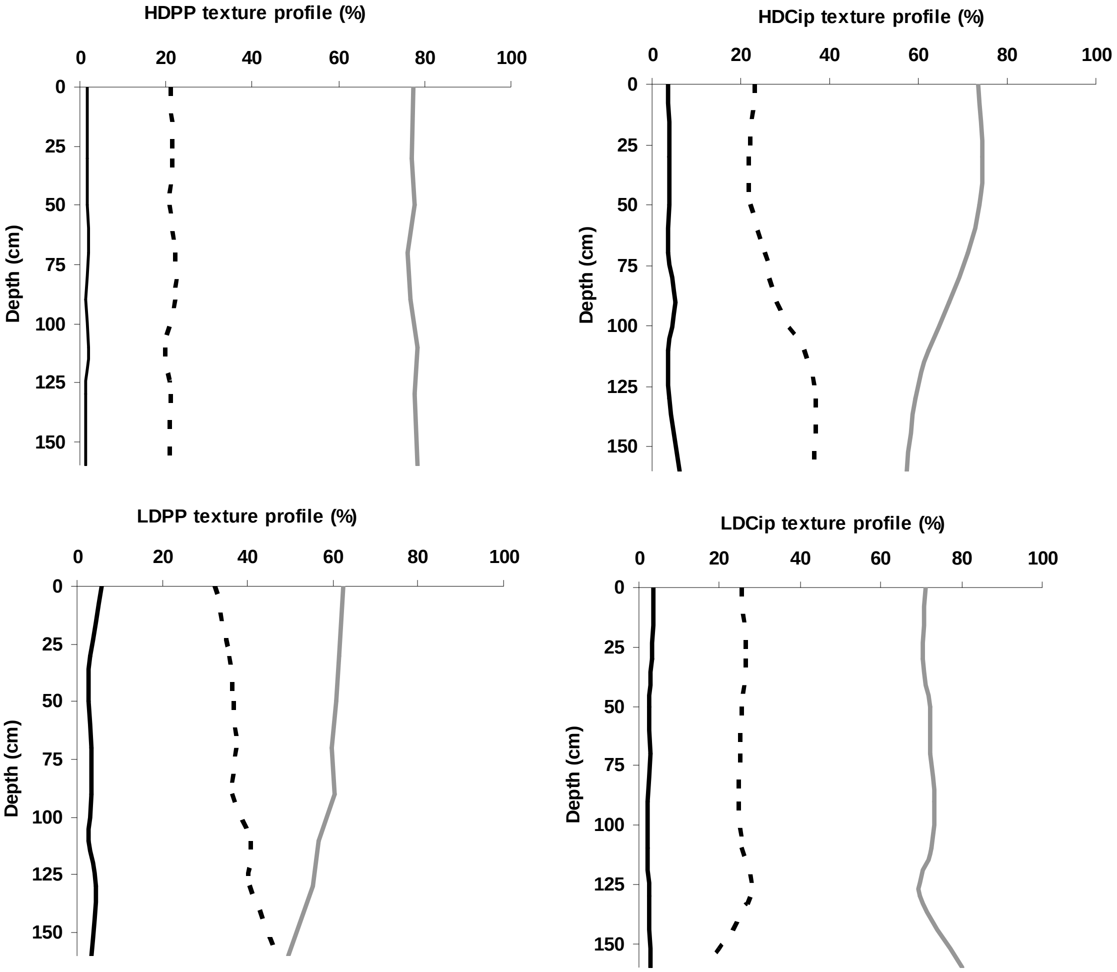


Figure 4.1 - Soil texture variation with depth per experimental plot. Measurements were taken every 20 cm up to a depth of 160 cm. Each analyzed sample was a pool of 18 smaller samples. The grey line represents the percent of sand, the dashed line the percent of silt and the solid black line the percent of clay.

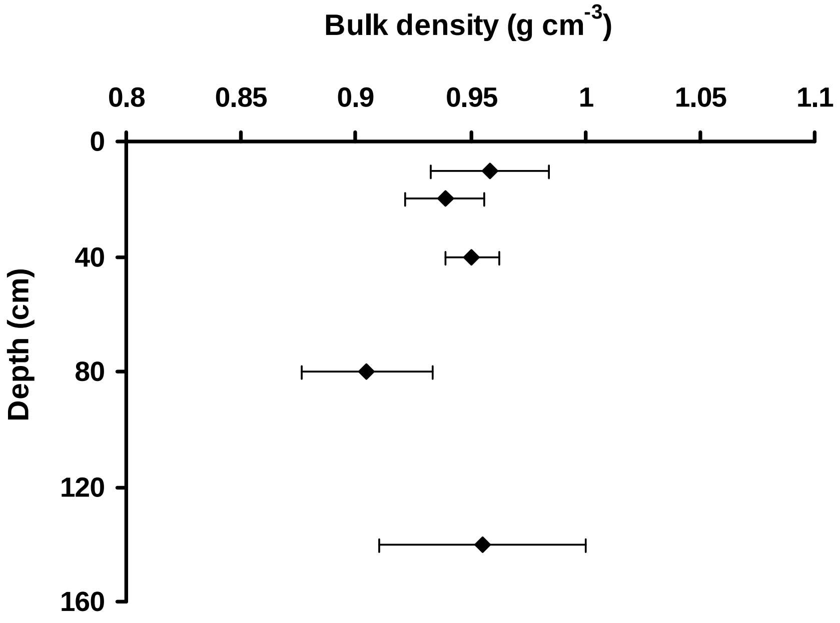


Figure 4.2 - Average bulk density at different depths. Error bars are standard deviations. Samples were taken from each of the four plots.

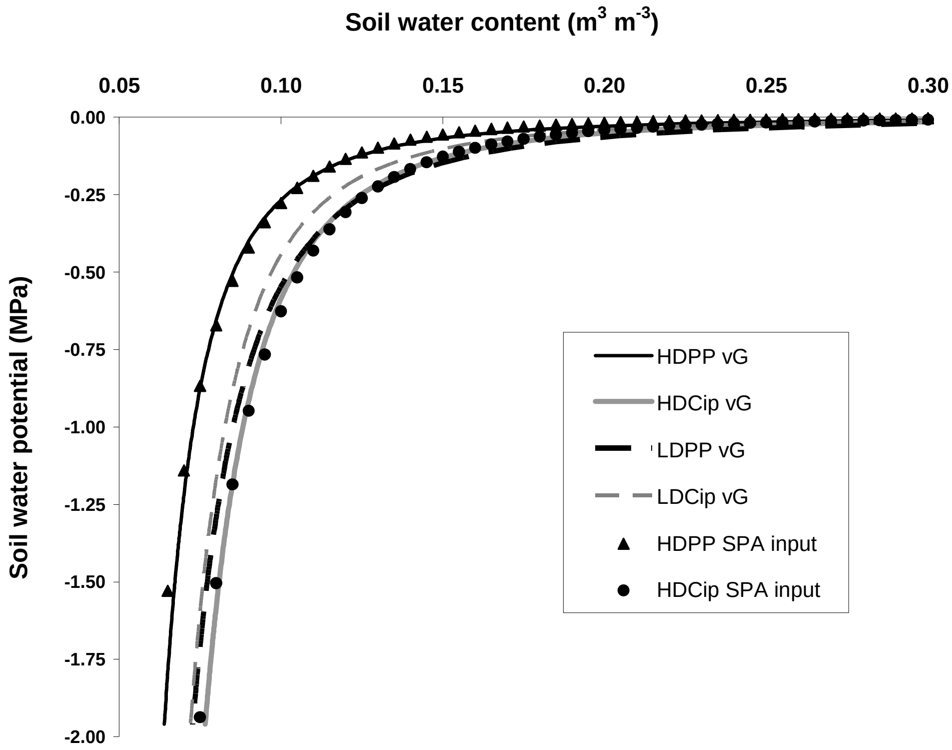


Figure 4.3 - Water release curves for all four experimental plots. Lines are the output of the van Genuchten equation, with hydrologic parameters obtained using the Rosetta software. Symbols represent the output of the water release curves used in the SPA model.

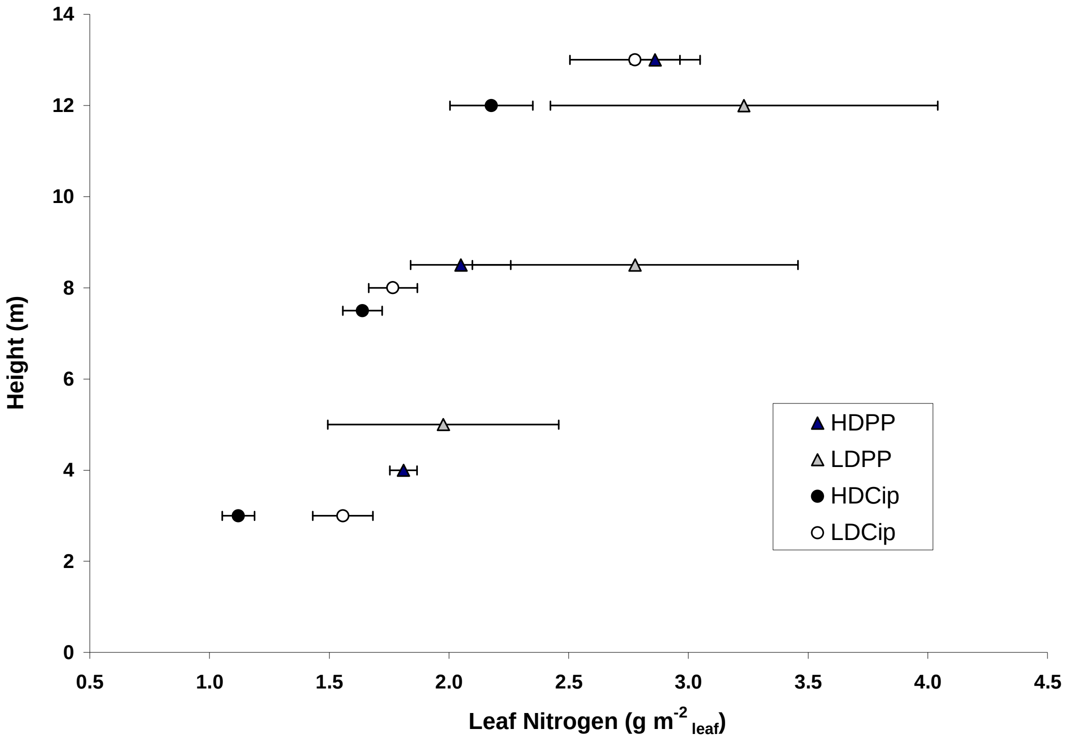


Figure 4.4 - Vertical profile of nitrogen content per unit of leaf area (g m^{-2}) by vegetation type. Error bars represent one standard deviation.

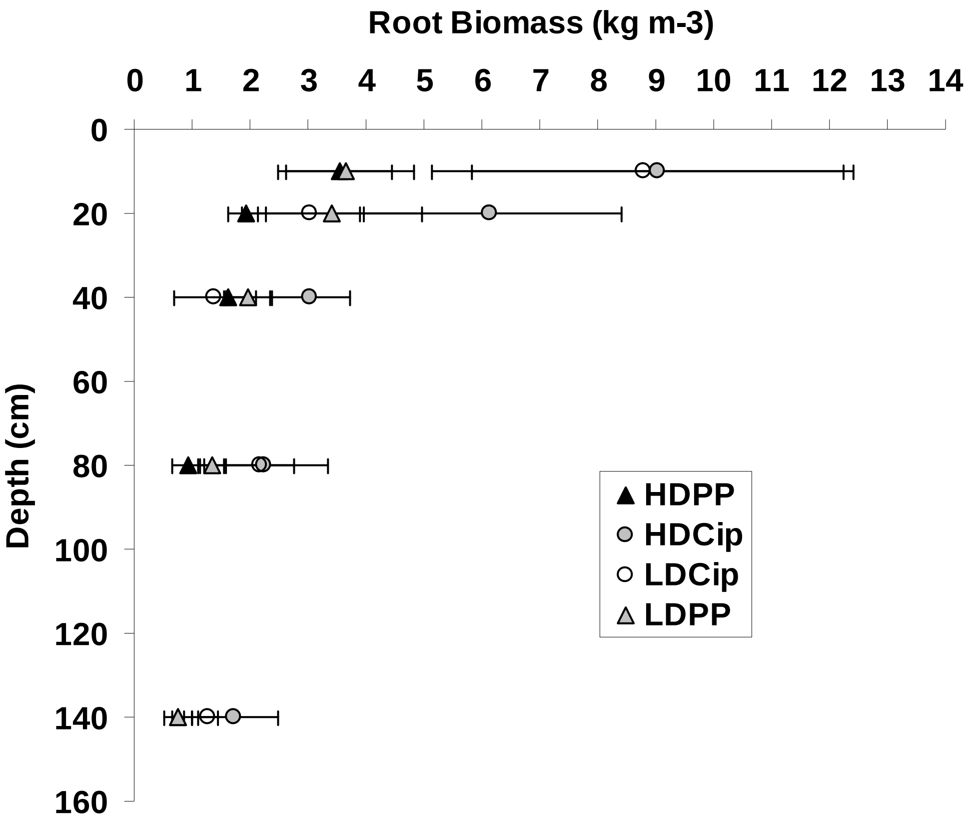


Figure 4.5 - Root Biomass distribution. Error bars represent one standard deviation.

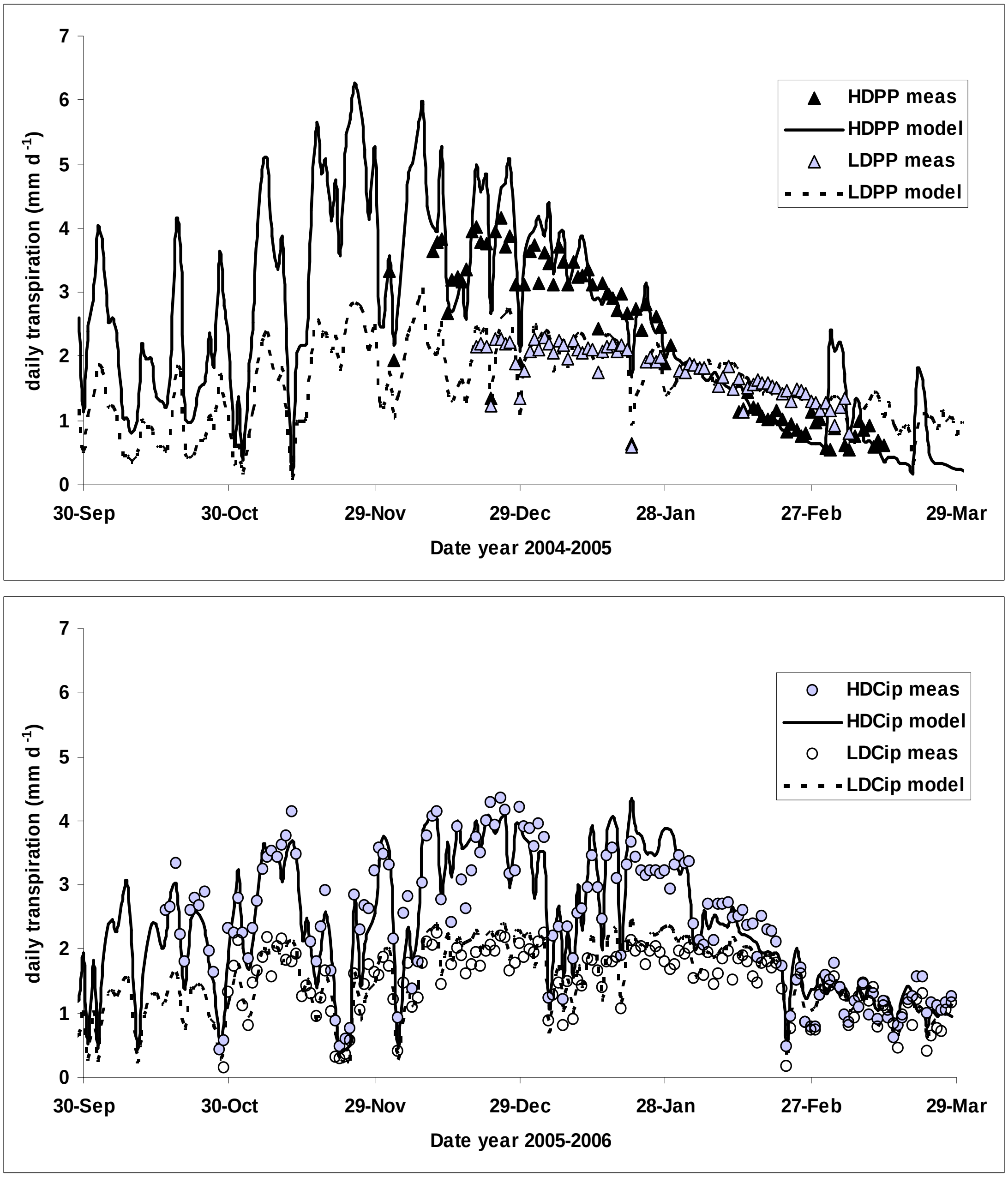


Figure 4.6 - SPA model parameterization. Measured and modeled daily transpiration (mm d^{-1}) for the parameterization runs in each forest stand during the two growing seasons for which we had data.

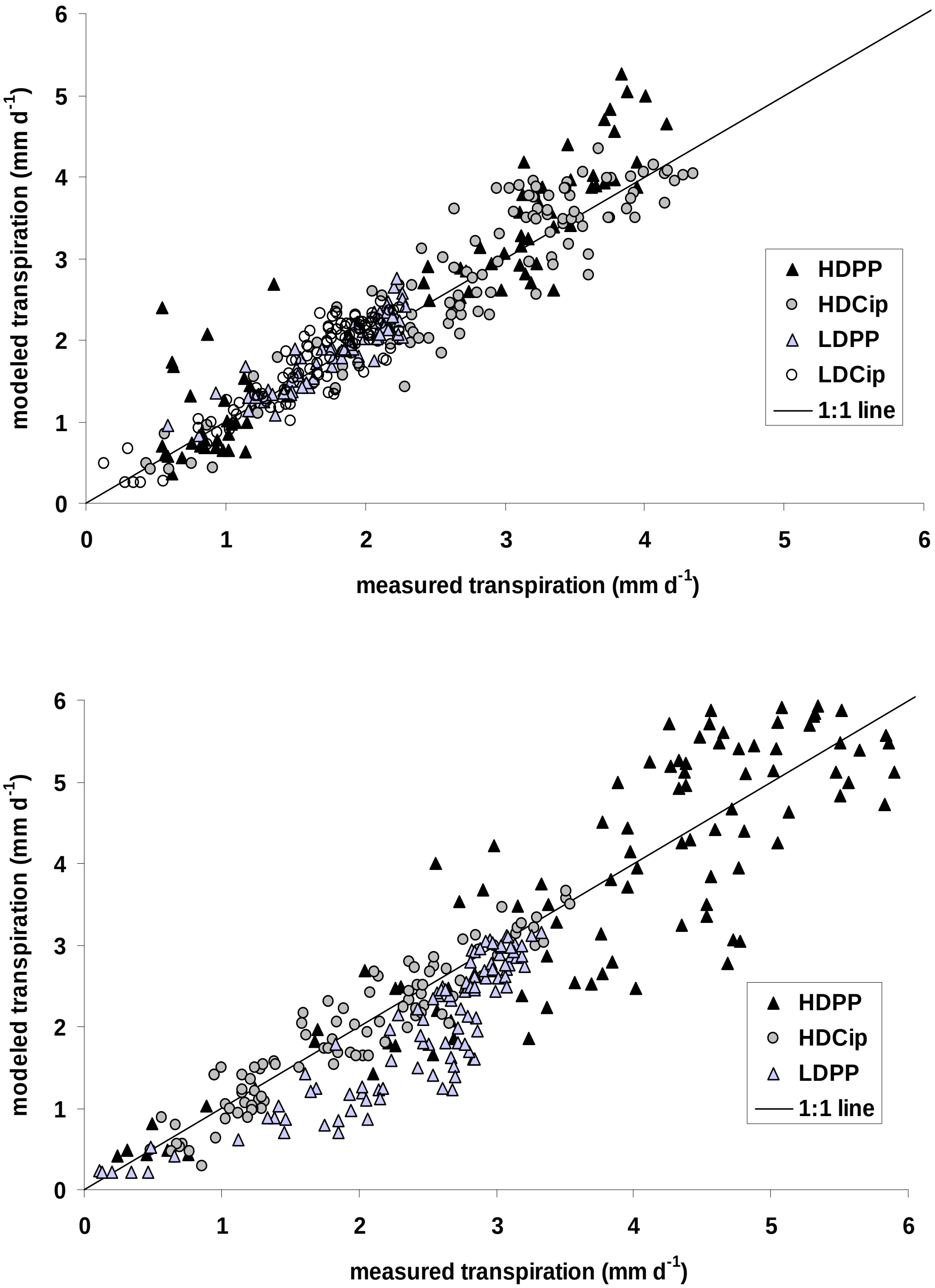


Figure 4.7 - Regression showing model fit. Modeled vs. measured daily transpiration of all stands for the calibration period (upper panel) and the validation period (lower panel).

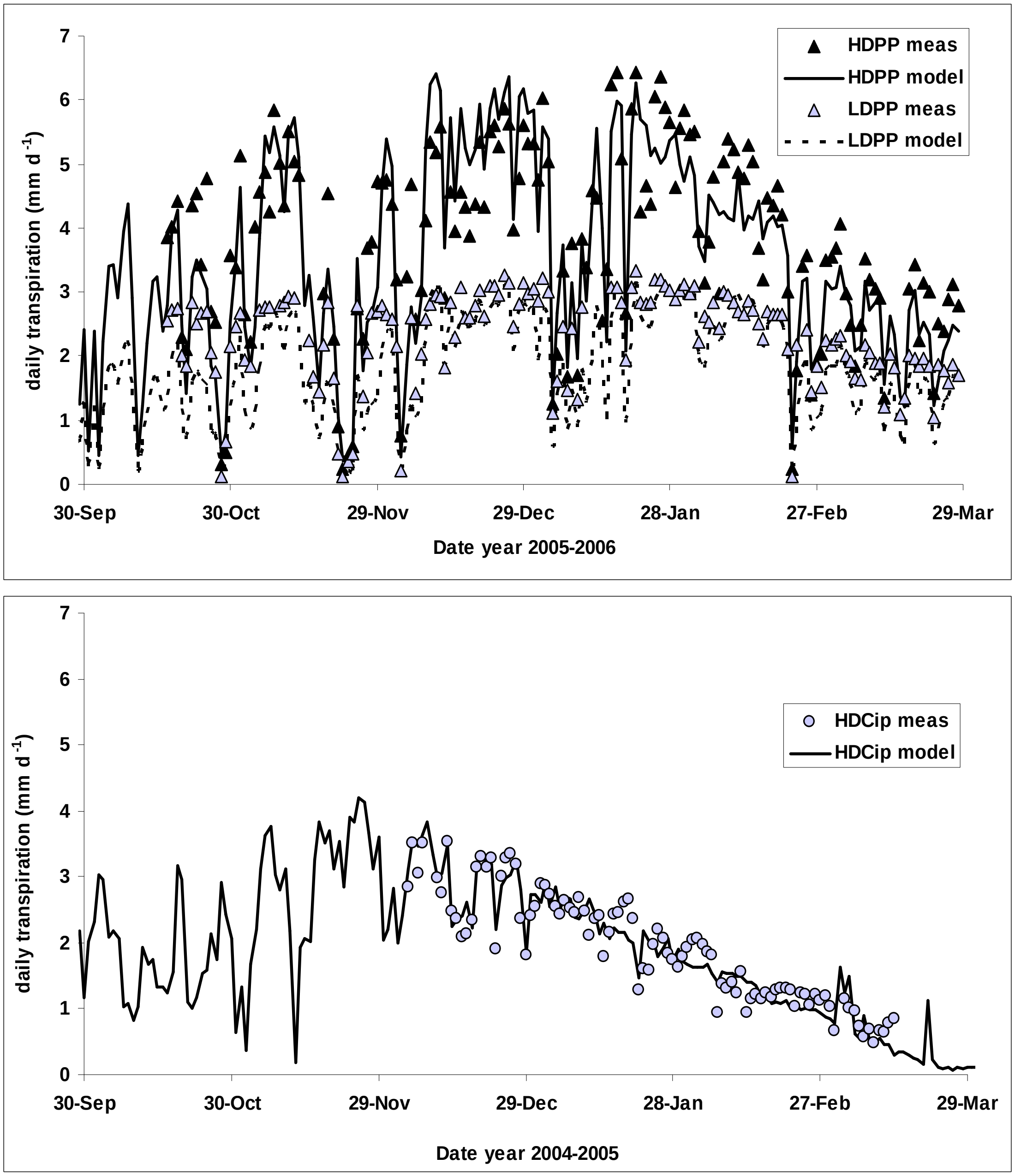


Figure 4.8 - SPA model validation. Measured and modeled daily transpiration (mm d^{-1}) for the validation growing seasons per forest stand. High density cordilleran cypress was validated using the 2004-2005 growing season data, and high and low density ponderosa pine were validated using the 2005-2006 growing season data.

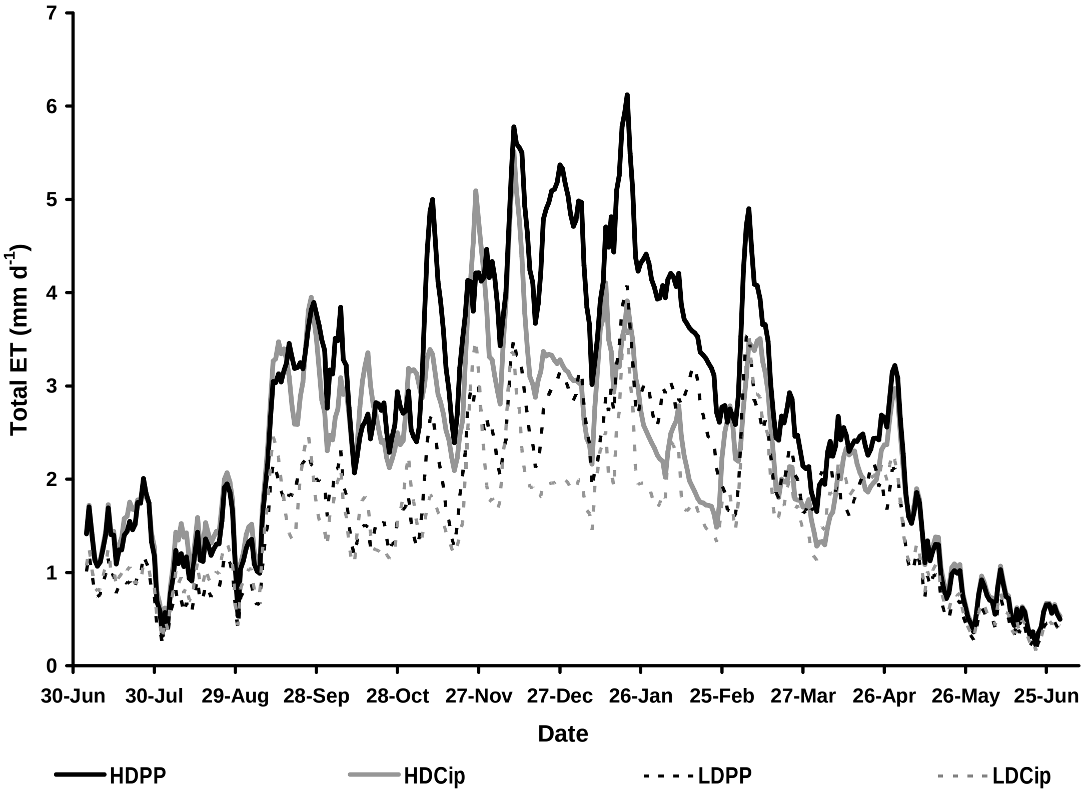
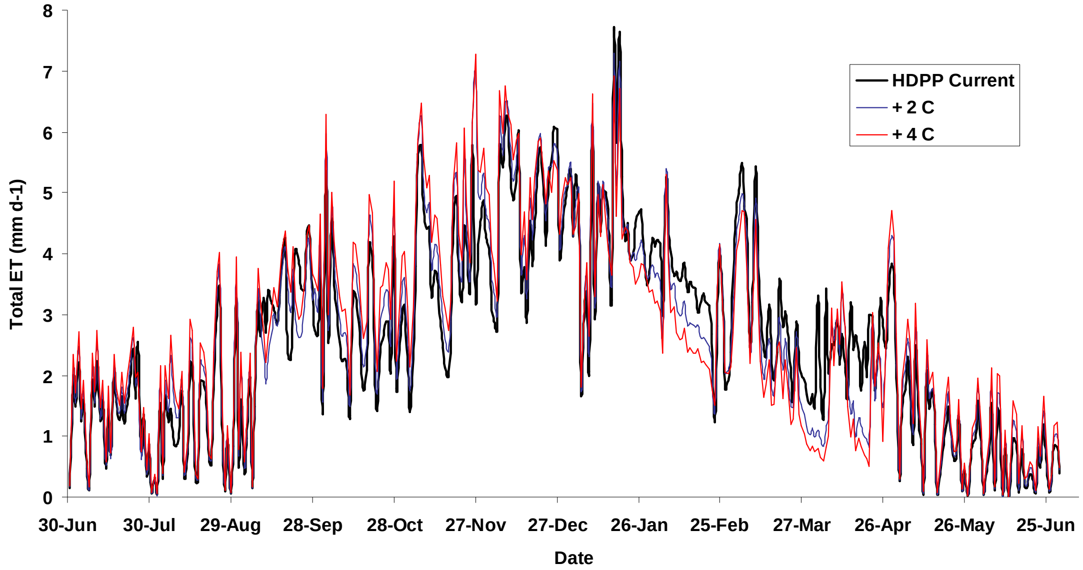


Figure 4.9 - Total daily ET in mm d^{-1} (Transpiration + Soil evaporation + Interception loss) per vegetation type under current climate conditions. Lines represent 5-day moving averages.



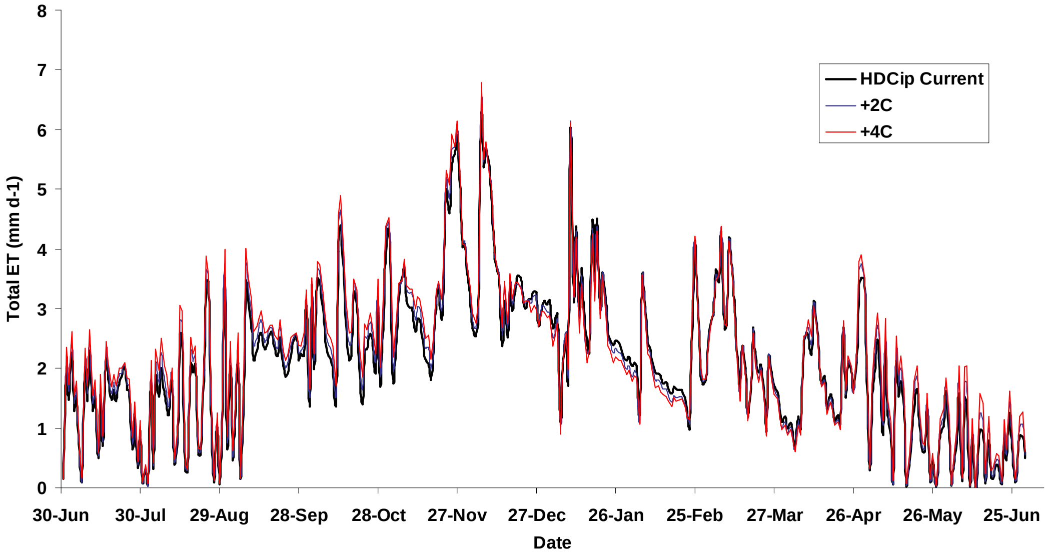


Figure 4.10 - Seasonal change of daily ET in high density ponderosa pine plantations (HDPP) and cordilleran cypress (HDCip) under different air temperature change scenarios. Solid black lines represent the modeled ET under current conditions, blue and red lines represent modeled ET under scenarios with an increase in air temperature of 2 and 4° C, respectively.

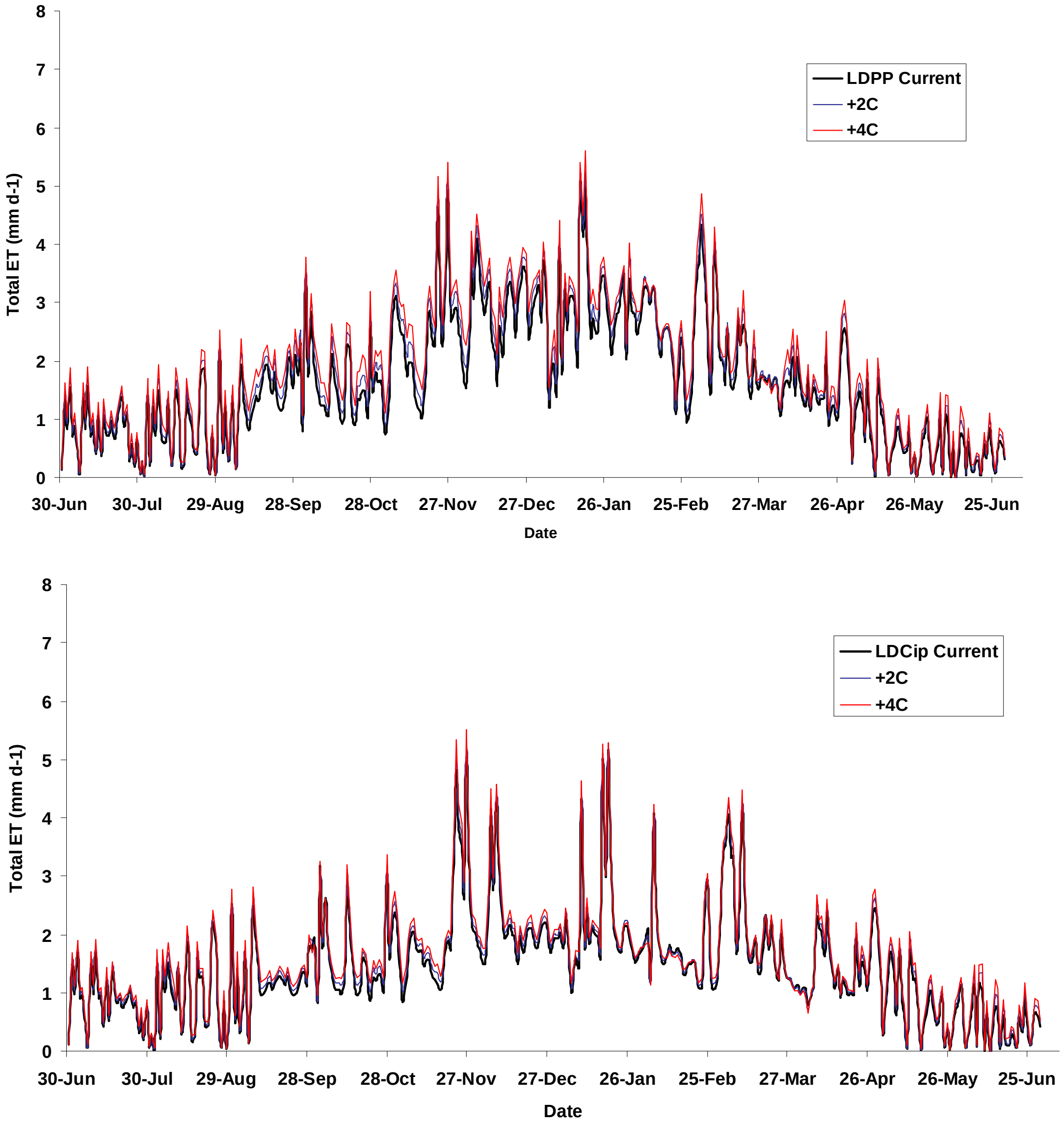


Figure 4.11 - Seasonal change of daily ET in low density ponderosa pine plantations (LDPP) and cordilleran cypress (LDCip) under different air temperature change scenarios. Solid black lines represent the modeled ET under current conditions, blue and red lines represent modeled ET under scenarios with an increase in air temperature of 2 and 4° C, respectively.

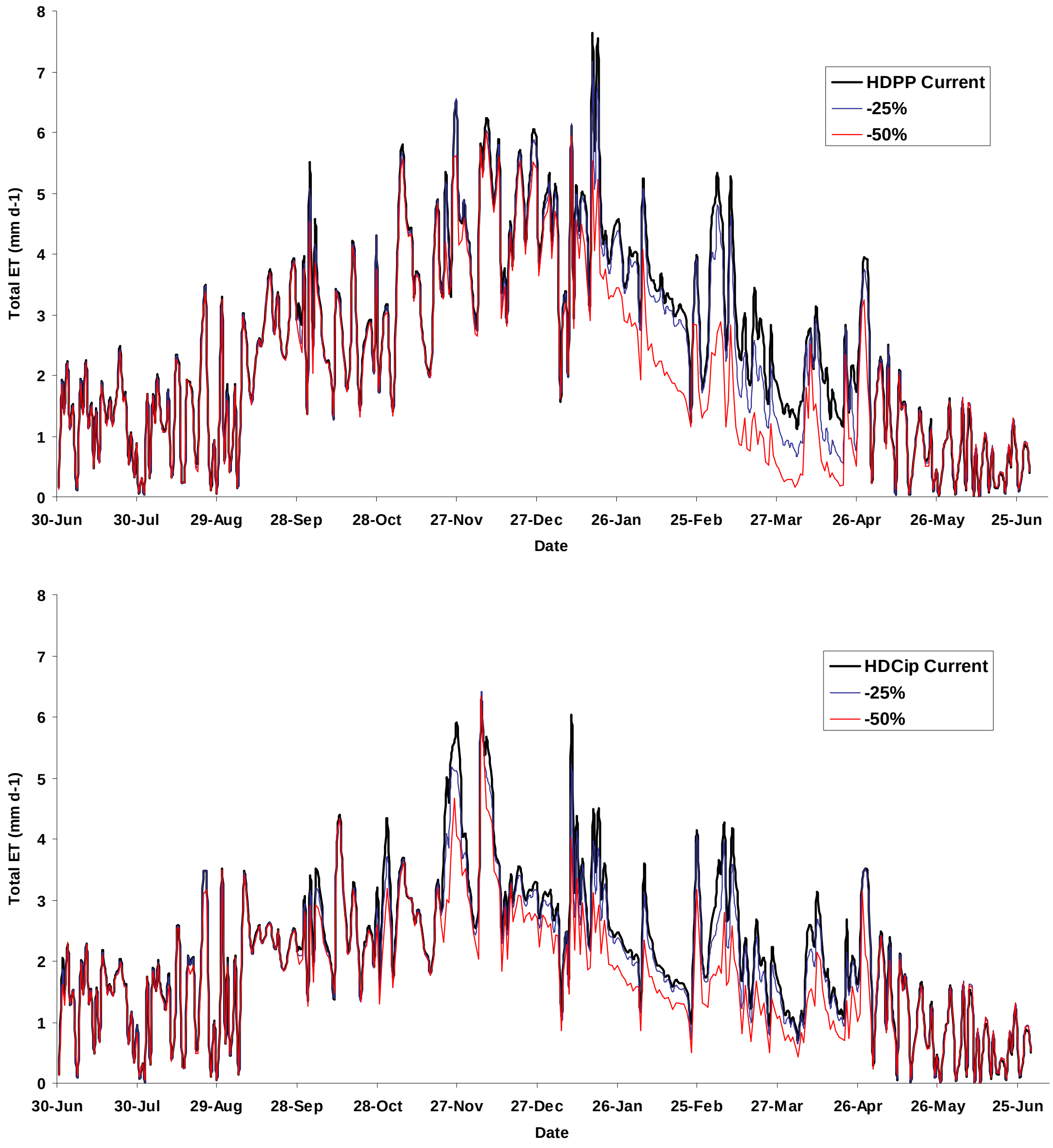


Figure 4.12 - Seasonal change of daily ET in high density ponderosa pine plantations (HDPP) and cordilleran cypress (HDCip) under different precipitation reduction scenarios. Solid black lines represent the modeled ET under current conditions, blue and red lines represent modeled ET under scenarios with a decrease in precipitation of 25 and 50 %, respectively.

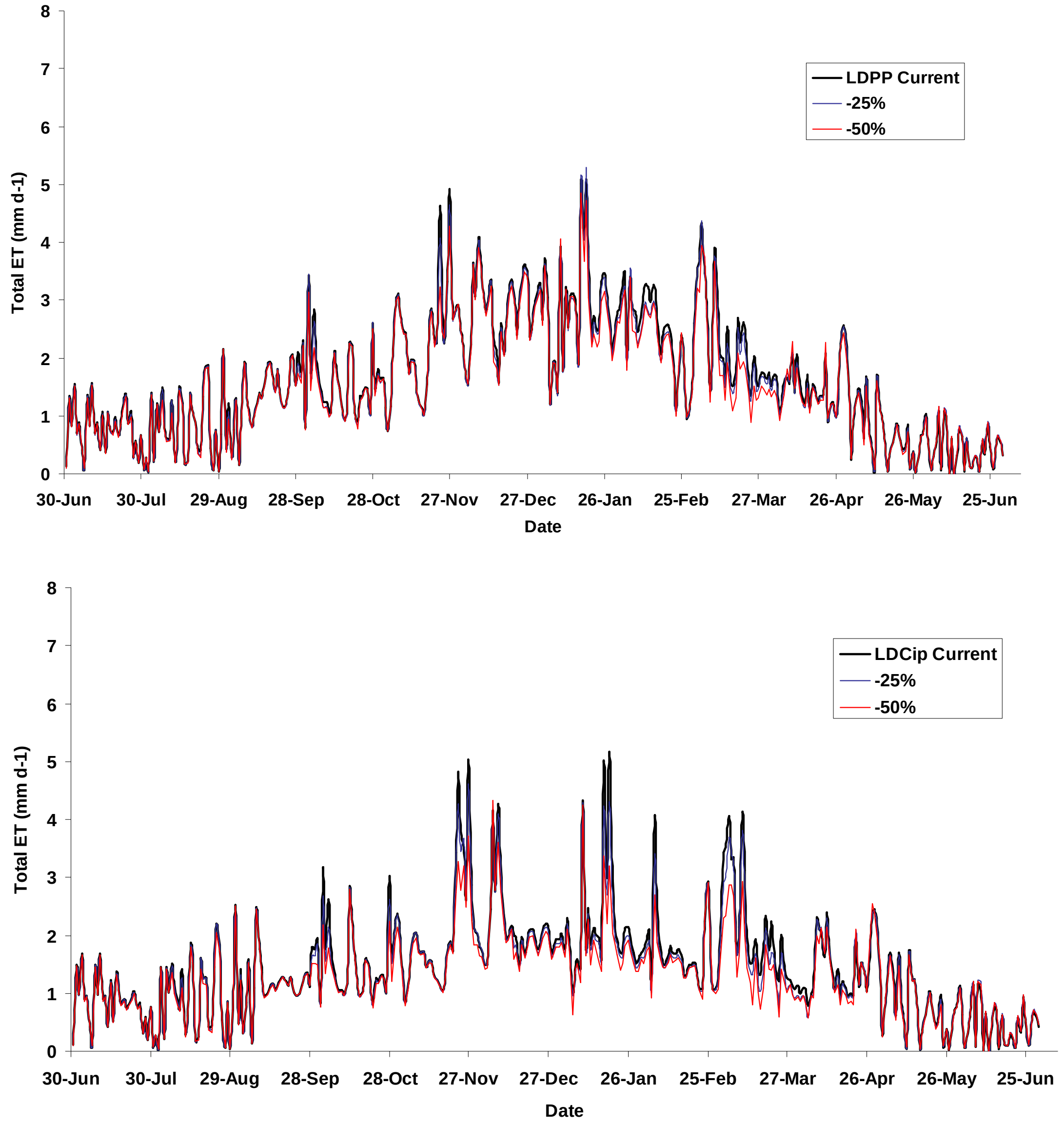


Figure 4.13 - Seasonal change of daily ET in low density ponderosa pine plantations (LDPP) and native cordilleran cypress (LDCip) under precipitation reduction scenarios. Solid black lines represent the modeled ET under current conditions, blue and red lines represent modeled ET under scenarios with a decrease in precipitation of 25 and 50 %, respectively.

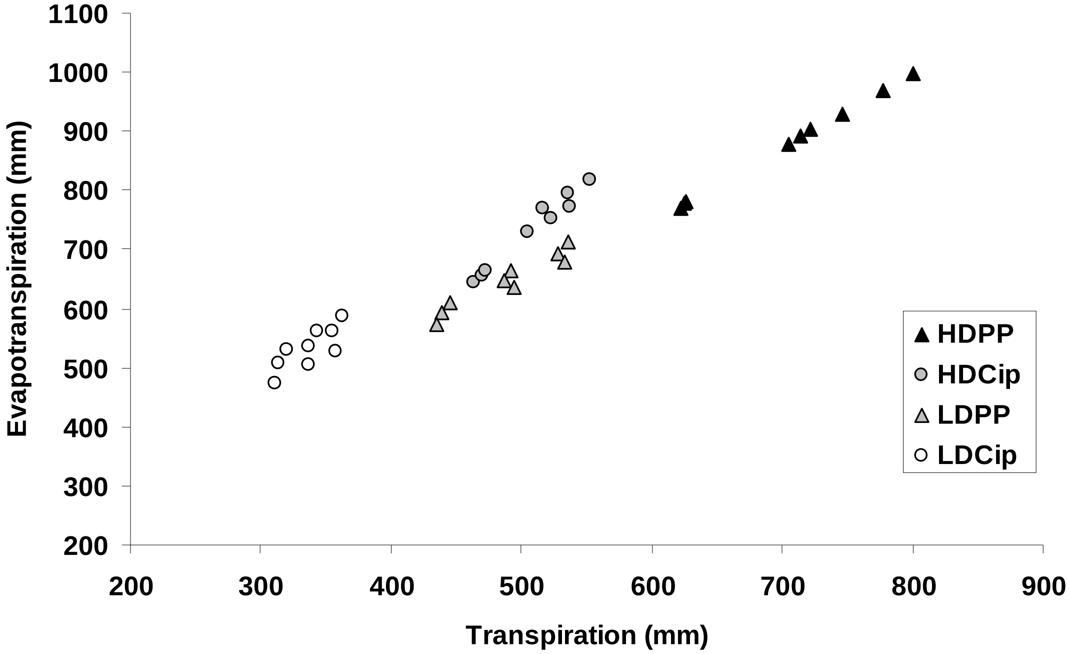


Figure 4.14 - Modeled annual ET vs. Transpiration (mm) for the four vegetation types for all climate scenarios.

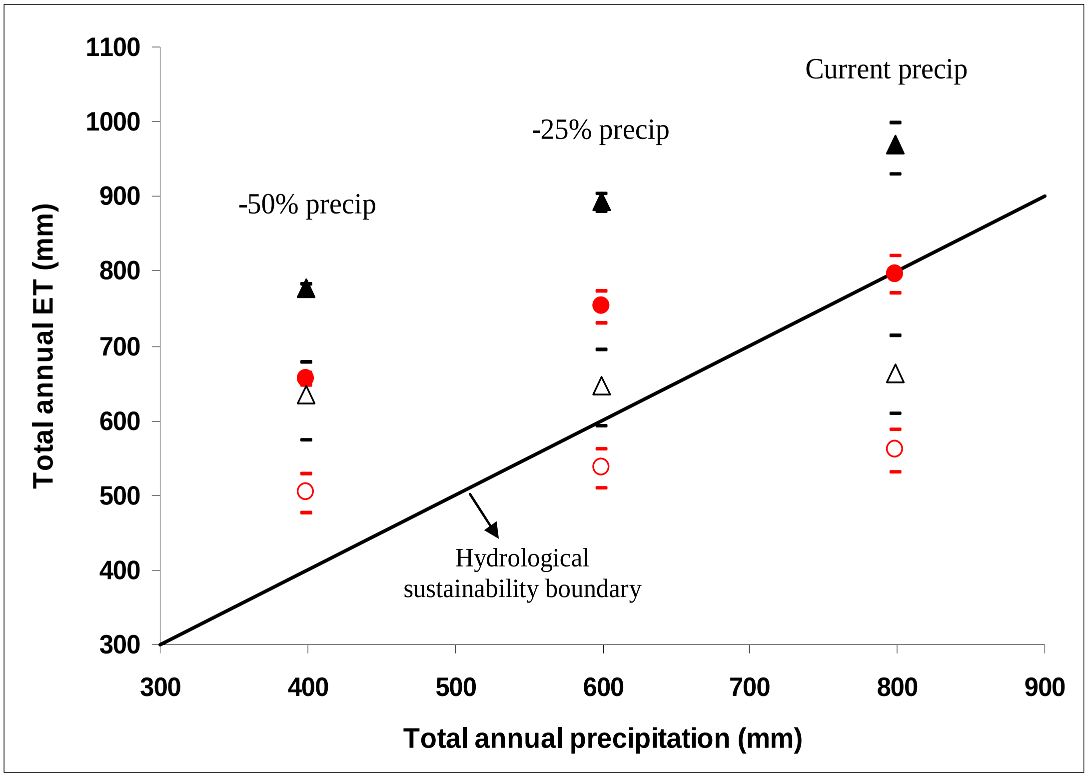


Figure 4.15 - Effects of changes in both precipitation and air temperature during potential future climate change scenarios on total annual ET (mm) for the four vegetation types. Open symbols are low density stands, filled symbols are high density stands. Triangles are ponderosa pine stands, circles are cordilleran cypress for scenarios with a two degree increase in mean temperature. Dashes represent a four degree increase in mean temperature (upper dash) and no increase in temperature (lower dash). Solid line represents the hydrologic sustainability boundary where total ET = total precipitation.

Tables

Table 4.1 - Model parameters obtained from previous studies.

	HDPP	HDCip	LDPP	LDCip	Units	Source
LAI	9	9	3.1	4.6	m2 m-2	Licata et al., in press
Min Leaf ΨL min	-2	-1.5	-2	-1.5	Mpa	Gyenge, 2005
Canopy Storage	3.1	7.5	2.2	2.7	mm	Licata et al., in prep
Direct throughfall	0.1	0.1	0.4	0.25		Licata et al., in prep
Height	14.5	13.7	14.1	15.1	m	Licata et al., in prep

Table 4.2 - Model soil and vegetation parameters measured and fitted in this study.

	HDPP	HDCip	LDPP	LDCip	Units
Top leaf Nitrogen	2.9	2.18	3.2	2.78	g m^{-2}
Above ground plant conductivity	5.8	4	6	4	mmol m^{-1} s^{-1} Mpa^{-1}
Root conductivity	0.2	0.2	0.2	0.2	mmol g^{-1} s^{-1} Mpa^{-1}
Full sun Vcmax	29	25	32.5	32	µmol m^{-2} s^{-1}
Full sun Vjmax	98	98	110	125	µmol m^{-2} s^{-1}
Average soil sand %	75	70	60	73	%
Average soil clay %	3.1	5	4	2.7	%
Rooting depth trees	3.5	2.75	3.3	2.5	m
Understory rooting depth	-	-	1.5	1.5	m
Understory LAI	-	-	0.5	0.5	m^2 m^{-2}

CHAPTER FIVE: GENERAL CONCLUSIONS

The results from the studies in this dissertation showed that high density ponderosa pine stands had the highest total annual transpiration of all the measured sites. High density stands had higher total annual transpiration than low density stands. When comparing among species, the higher observed total annual transpiration by ponderosa pine was due to consistently higher transpiration rates throughout the year by ponderosa pine compared to the native cordilleran cypress plots growing under similar conditions and comparable levels of stand density. Differences in transpiration among stands were greater during the year with higher precipitation during the summer, when all the plots followed a seasonal transpiration pattern similar to a crop with no water limitations. High density stands of both species showed a decline in transpiration rates due to summer drought during the year of measurements with lower summer precipitation, while the decline was almost unnoticeable in the low density stands.

In both species, soil water depletion occurred simultaneously (although at decreasing rates) at different depths up to 1.8 m deep. This suggested a very low root resistivity, which was then confirmed by the model parameterization results. The mass balance approach between the transpiration estimated by sapflow measurements and the soil water depletion indicated that high density ponderosa pine stands should be using water from deeper soil layers than the ones measured (>1.8m). Spa model parameterization suggested a rooting depth of up to 3.5 m in the high density ponderosa pine stand, and up to 2.7 m in high density cypress.

Rainfall interception losses are not trivial. Although the annual estimates produced by the different methods (chapter 3 vs. chapter 4) used to calculate interception losses were considerably different from one another, the relative differences between species persisted. In both cases, however, including interception losses in the results had a large impact on the final conclusions. The leaf morphology of cordilleran cypress likely creates a much larger water storage capacity than that of ponderosa pine. The amount of water that is lost as interception in cordilleran cypress is almost double that of ponderosa pine. This also means that more water is available for transpiration in the ponderosa pine stands than in cordilleran cypress, and would consequently translate in higher productivity. The strikingly different values observed between species, but not between stand densities, suggest that, generally speaking, any comparison in water use between species that have different leaf morphology should include some estimate of their relative effects on interception loss.

Finally, the use of a process-based model made the integration of the accumulated knowledge about the systems possible and helped close the water budget of the different forest stands. Estimates of total annual ET by the parameterized SPA model suggest that typical high density ponderosa pine plantation for timber production, such as the one in this study, are not hydrologically sustainable under current meteorological conditions in the forest-steppe ecotone of NW Patagonia.

SPA model predictions further suggest that warmer and drier environmental conditions, as projected by the IPCC AR4 for this region, will reduce transpiration more drastically during the summer in high density ponderosa pine plantations than in any other forest stand. Both, predicted changes in precipitation (decrease) and temperature (increase) indicate that stressful summer drought conditions will be worse than under current climate. None of the species high density stands would be hydrologically sustainable under these predicted future scenarios.

Higher temperature scenarios may increase productivity of the low density stands in both species, but would increase the summer drought severity in the high density stands, compromising any potential increase in productivity that may occur at the beginning of the season.

Overall, these dissertation results highlight the need for stand density management in order to reduce the ongoing impact of high density ponderosa pine plantations on water resources. Reductions of stand density in dense stands of both species would also improve forest stand resilience to drought stress and help reduce fire risks.

CHAPTER SIX: BIBLIOGRAPHY

Chapter 1

Binkley, C. 1999. Ecosystem management and plantation forestry: new directions in British Columbia. New Forests 18: 75-88.

Bradshaw C.J.A., Sodhi, N.S., Peh, K.S.-H., Brook, B.W. 2007. Global evidence that deforestation amplifies flood risk and severity in the developing world. Global Change Biology 13 (11), 2379–2395.

Cramer,W, Bondeau, A, Woodward, FI, Prentice, IC, Betts, RA, Brovkin, V, Cox, PM, and co-authors. 2001. Global response of terrestrial ecosystem structure and function to CO2 and climate change: results from six dynamic global vegetation models. Global Change Biol. 7: 357-373.

Evans, J., Turnbull, J.W. 2004. Plantation forestry in the tropics. Oxford University Press pp 13-22. 488pp

FAO 2007. State of the World's Forests. Food and Agriculture Organization of the United Nations, Rome.

Farley K.A., Jobbagy, E.G., Jackson, R.B. 2005. Effects of afforestation on water yield: A global synthesis with implications for policy. Global Change Biology 11: 1565–1576.

Feddema, J.J., Oleson, K.W., Bonan, G.B., Mearns, L.O., Buja, L.E., Meehl, G.A., Washington, W.M. 2005. The importance of land-cover change in simulating future climates. Science 310: 1674-1678.

Friedlingstein, P., Cox, P. M., Betts, R. A., Bopp, L., von Bloh, W. and co-authors. 2006. Climate–Carbon Cycle Feedback Analysis: Results from the C4MIP Model Intercomparison. Journal of Climate 19: 3337-3353.

Friedman, S.T. 2006. Environmental Aspects of the Intensive Plantation/Reserve Debate. Journal of Sustainable Forestry 21: 59-73.

Gallo, L., Martinez Meier, A., Azpilicueta, M.M., Marchelli, P., Mondino, V. 2005. III. 4. Subprograma coníferas y otras especies en la región patagónica, in: Norverto, C.A.(Ed.), Mejores arboles para mas forestadores. Secretaria de Agricultura, Pesca y Alimentacion, Buenos Aires, 95-116.

Gyenge, J.E. 2005. Uso de agua y resistencia a la sequía de las principales especies forestales del noroeste patagónico, pino ponderosa y ciprés de la cordillera. PhD. Thesis, Universidad Nacional del Comahue, Rio Negro, Argentina, 364 pp

Gyenge, J.E., Fernandez, M.E., Dalla Salda, G., Schlichter, T.M. 2002. Silvopastoral systems in northwest Patagonia II: water balance in a stand of Pinus ponderosa and native grassland. Agroforestry systems 55: 47-55.

Hartley, M.J. 2002. Rationale and methods for conserving biodiversity in plantation forests. Forest Ecol. Manage. 155: 81–95.

Jobággy, EG, Paruelo, JM, Leon, RJC. 1995. Estimacion del regimen de precipitacion a partir de la distancia a la cordillera en el noroeste de la Patagonia. Ecologia Austral 5: 47-53.

Kanowski, J., Catterall, C.P., Wardell-Johnson, G.W. 2005. Consequences of broadscale timber plantations for biodiversity in cleared rainforest landscapes of tropical and sutropical Australia. Forest Ecology and Management 208: 359-372.

Kitzberger, T., Veblen, T.T. 1999. Fire-induced changes in northern Patagonian landscapes. Landscape Ecology 14: 1-15.

Kuczera, G.A. 1985. Prediction of water yield reductions following a bushfire in ash-mixed species eucalypt forest. Catchment Hydrology Research Report No. MMBW-W-0014. Melbourne and Metropolitan Board of Works, p. 163.

Laclau, P. 2003. Biomass and carbon sequestration of ponderosa pine plantations and native cypress forests in northwest Patagonia. Forest Ecology and Management 180: 317-333.

Lawton, R.O., Nair, U.S., Pielke Sr., R.A., Welch, R.M. 2001. Climatic impact of tropical lowland deforestation on nearby montane cloud forest. Science 294: 584–587.

Le Maitre, D.C., Van Wilgen, B.W., Gelderblom, C.M., Bailey, C., Chapman, R.A., Nel, J.A. 2002. Invasive alien trees and water resources in South Africa: case studies of the costs and benefits of management. Forest Ecology and Management 160: 143-159.

Le Treut, H., R. Somerville, U. Cubasch, Y. Ding, C. Mauritzen, A. Mokssit, T. Peterson Prather, M. 2007. Historical Overview of Climate Change. In: Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change [Solomon, S., D. Qin, M. Manning, Z. Chen, M. Marquis, K.B. Averyt, M. Tignor and H.L. Miller (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Loguercio, G.A., Buduba, C., La Manna, L. 2005. Plantación de ciprés de la cordillera de 57 años de edad: una experiencia en el Parque Nacional Los Alerces. Patagonia Forestal 10: 7-8.

Loguercio, G.A., Burschel, P., Rey, M. 1999. El bosque de ciprés de la cordillera: su conservación y uso. Centro de Investigación y Extensión, Forestal Andino Patagónica, Folleto de Divulagación No. 14.

Merenson, C. 1992. Desarrollo Sustentable o Deforestación. Plan Forestal Argentino. Dirección de Recursos Forestales Nativos. Buenos Aires, 55 pp.

Montenegro, C., Gasparri, I., Manghi, E., Strada., M., Bono, J., Parmuchi, M.G. Unidad de Manejo del Sistema de Evaluacion Forestal. 2004. Informe sobre deforestacion Argentina. Secretaria de ambiente y desarrollo sustentable. Direccion de Bosques. Buenos Aires, 8 pp.

Nemani, RR, Keeling, CD, Hashimoto, H, Jolly, WM, Piper, SC, Tucker, CJ, Myneni, RB, Running, SW. 2003. Climate-driven increases in global terrestrial net primary production from 1982 to 1999. Science 300: 1560-1563.

Nosetto, M. D., Jobbágy, E.G., Paruelo, J. M. 2005. Land-use change and water losses: the case of grassland afforestation across a soil textural gradient in central Argentina. Global Change Biology 11: 1101-1117.

Nosetto, M. D., Jobbágy, E.G., Paruelo, J. M. 2006. Carbon sequestration in semiarid rangelands: Comparison of Pinus ponderosa plantations and grazing exclusion in NW Patagonia. Journal of Arid Environments 67: 142-156.

Paruelo, J.M., Beltrán, A., Jobbágy, E.G., Sala, O., Golluscio, R.A. 1998. The Climate of Patagonia: general patterns and controls on biotic processes. Ecología Austral 8: 85-101.

SAGPyA. Secretaria de Agricultura Pesca y Alimentacion. 1999. Argentina investment opportunities in plantation forests. Buenos Aires. 208 pp.

SAyDS. Secretaria de Ambiente y Desarrollo Sustentable. 2006. Tercer informe nacional para la implementación de la convención de las Naciones Unidas de lucha contra la desertificación. Buenos Aires, 77 pp.

Sahin, V., Hall, M.J. 1996. The effects of afforestation and deforestation on water yields. Journal of Hydrology 178: 293-309.

Sedjo, R.A. and D. Botkin. 1997. Using forest plantations to spare natural forests. Environment 30: 15-20.

Torn, M. S., Harte, J. 2006. Missing feedbacks, asymmetric uncertainties, and the underestimation of future warming. Geophys. Res. Lett. 33: L10703, doi:10.1029/2005GL025540.

Veblen,T.T., Kitzberger, T., Villalba, R., Donnegan, J. 1999. Fire History in Northern Patagonia -The Roles of Humans and Climatic Variation. Ecological Monographs 69: 47-67.

Villalba R, Veblen, T.T. 1998. Influences of large-scale climatic variability on episodic tree mortality in Northern Patagonia. Ecology 79: 2624-2640.

Chapter 2

Allen, G.A., Pereira, L.S., Raes, D., Smith, M. 1998. Crop Evapotranspiration – guidelines for Computing Crop Water Requirements. FAO: Irrigation and Drainage Paper 56. United Nations, Rome, Italy.

Anthoni, P.M., Law, B.E., Unsworth, M.H. 1999. Carbon and water vapor exchange of an open-canopied ponderosa pine ecosystem. Agricultural and Forest Meteorology 95:151–168.

Binkley, C. 1999. Ecosystem management and plantation forestry: new directions in British Columbia. New Forests 18: 75-88.

Brandyk, T., Romanowicz, R., Skapski, K. (1989) Determination of the exponential unsaturated hydraulic conductivity function by parameter identification. European Journal of Soil Science 40: 261–267.

Cermák, J., Nadezhdina, N. 1998. Sapwood as the scaling parameter – defining according to xylem water content or radial pattern of sap flow? Ann. Sci. For. 55: 509-521.

Delzon, S., Sartore, M., Granier, A., Loustau, D. 2004. Radial profiles of sap flow with increasing tree size in maritime pine. Tree Physiology 24: 1285-1293.

FAO 2007. State of the World's Forests. Food and Agriculture Organization of the United Nations, Rome.

Farley K.A., Jobbagy, E.G., Jackson, R.B. 2005. Effects of afforestation on water yield: A global synthesis with implications for policy. Global Change Biology 11: 1565–1576.

Fiora, A., Cescatti, A. 2006. Diurnal and seasonal variability in radial distribution of sap flux density: implications for estimating stand transpiration. Tree Physiology 26: 1217-1225.

Fiorentino, R. 2005. El riego en la provincia de Neuquen. Banco Mundial, Buenos Aires, Marzo 2005.

Fernandez, M.E., Gyenge,J.E., Dalla Salda, G., Schlichter, T.M. 2002. Silvopastoral systems in northwest Patagonia I: photosynthesis of Stipa speciosa under different levels of Pinus ponderosa cover. Agroforestry systems 55: 27-35

Ford, C.R., McGuire, M.A., Mitchell, R.J., Teskey, R.O. 2004. Assesing variation in the radial profile of sap flux density in Pinus species and its effect on daily water use. Tree Physiology 24: 241-249.

Friedman, S.T. 2006. Environmental Aspects of the Intensive Plantation/Reserve Debate. Journal of Sustainable Forestry 21: 59-73.

Fritzsche, F., Abate, A., Fetene, M., Beck, E., Weise, S., Guggenberger, G. 2006. Soil-plant hydrology of indigenous and exotic trees in an Ethiopian montane forest. Tree Physiology 26: 1043-1054.

Gallo, L., Martinez Meier, A., Azpilicueta, M.M., Marchelli, P., Mondino, V. 2005. III. 4. Subprograma coníferas y otras especies en la región patagónica, in: Norverto, C.A.(Ed.), Mejores arboles para mas forestadores. Secretaria de Agricultura, Pesca y Alimentacion, Buenos Aires, 95-116.

Girardin, J.L. and P. Broquen. 1995. El crecimiento de Pinus ponderosa Dougl. y Pseudotsuga menziesii (Mirb.) Dougl. En diferentes condiciones de sitio. Prov.NQN-R.A. Bosque 16: 57-67.

Gandullo, R. and Schmid, P. 2001. Analisis ecologico de mallines del parque provincial copahue, neuquen, argentina. Agro sur 29: 83-99.

Golluscio, R.A., Deregibus,V.A., Paruelo, J.M. 1998. Sustainability and range management in the Patagonian steppes. Ecología Austral 8: 265-284.

Granier, A. 1987. Evaluation of transpiration in a Douglas-fir stand by means of sapflow measurements. Tree Physiology 3: 309-320.

Gyenge,J.E., Fernandez, M.E., Dalla Salda, G., Schlichter, T.M. 2002. Silvopastoral systems in northwest Patagonia II: water balance in a stand of Pinus ponderosa and native grassland. Agroforestry systems 55: 47-55.

Gyenge J.E., Fernández M.E., Schlichter T.M., 2003. Water relations of ponderosa pines in Patagonia Argentina: implications for local water resources and individual growth, Trees 17: 147–423

Gyenge, J.E. 2005. Uso de agua y resistencia a la sequía de las principales especies forestales del noroeste patagónico, pino ponderosa y ciprés de la cordillera. PhD. Thesis, Universidad Nacional del Comahue, Rio Negro, Argentina, 364 pp.

Hartley, M.J. 2002. Rationale and methods for conserving biodiversity in plantation forests. Forest Ecology and Management 155: 81-95.

Irvine, J., Law, B.E., Anthoni, P.M., Meinzer, F.C. 2002. Water limitations to carbon exchange in old-growth and young ponderosa pine stands. Tree Physiology 22: 189-196.

Irvine, J., Law, B.E., Kurpius, M., Anthoni, P.M., Moore, D., Schwarz, P. 2004. Age related changes in ecosystem structure and function and the effects on carbon and water exchange in ponderosa pine. Tree Physiology 24:753-763.

Jackson, R.B., Jobbágy, E.G., Avissar, R., Roy, S.B., Barrett, D., Cook, C.W., Farley, K.A., le Maitre, D.C., McCarl, B.A., Murray, B.C. 2005. Trading water for carbon with biological carbon sequestration. Science 310: 1944-1947.

Jimenez, M.S., Nadezhdina, N., Cermák, J., Morales, D. 2000. Radial variation in sap flow in five laurel forest tree species in Tenerife, Canary Islands. Tree Physiology 20: 1149-1156.

Jutras, S., Plamondon, A.P., Hoekkae, H., Begin, J. 2006. Water table changes following precommercial thinning on post-harvest drained wetlands. Forest Ecology and Management 235: 252-259.

Kanowski, J., Catterall, C.P., and Wardell-Johnson, G.W. 2005. Consequences of broadscale timber plantations for biodiversity in cleared rainforest landscapes of tropical and sutropical Australia. Forest Ecology and Management 208: 359-372.

Kitzberger, T., Veblen T.T. 1999. Fire-induced changes in northern Patagonian landscapes. Landscape Ecology 14: 1-15.

Kurpius, M.R., Panek, J.A., Nikolov, N.T., McKay, M., Goldstein, A.H. 2003. Partitioning of water flux in a Sierra Nevada ponderosa pine plantation, Agricultural and Forest Meteorology, 117, 173-192.

Laclau, P. 2003. Biomass and carbon sequestration of ponderosa pine plantations and native cypress forests in northwest Patagonia. Forest Ecology and Management 180: 317-333.

Laclau, P., Andenmatten, E. 2005. La calidad de sitio y el negocio forestal. IDIA XXI Forestales, 230-232.

Law, B. E., Goldstein, A. H., Anthoni, P. M., Unsworth, M. H., Panek, J. A., Bauer, M. R., Fracheboud, J. M. and Hultman, N. 2001. Carbon dioxide and water vapor exchange by young and old ponderosa pine ecosystems during a dry summer. Tree Physiology 21:299-308.

Le Maitre, D.C., Van Wilgen, B.W., Gelderblom, C.M., Bailey, C., Chapman, R.A., Nel, J.A. 2002. Invasive alien trees and water resources in South Africa: case studies of the costs and benefits of management. Forest Ecology and Management, 160: 143-159.

Loguercio, G.A., Burschel, P., Rey, M. 1999. El bosque de ciprés de la cordillera: su conservación y uso. Centro de Investigación y Extensión, Forestal Andino Patagónica, Folleto de Divulagación No. 14.

Loguercio, G.A., Buduba, C., La Manna, L. 2005. Plantación de ciprés de la cordillera de 57 años de edad: una experiencia en el Parque Nacional Los Alerces. Patagonia Forestal 10: 7-8.

Mermoz, M., Kitzberger, T., Veblen, T.T. 2005. Landscape influences on occurrence and spread of wildfires in patagonian forests and shrublands. Ecology 86: 2705-2715.

Nadezhdina, N., Cermák, J., Ceulemans, R. 2002. Radial patterns of sap flow in woody stems of dominant and understory species: scaling errors associated with positioning of sensors. Tree Physiology 22: 907-918

Nosetto, M. D., Jobbágy, E.G., Paruelo, J. M. 2005. Land-use change and water losses: the case of grassland afforestation across a soil textural gradient in central Argentina. Global Change Biology 11: 1101-1117.

Nosetto, M. D., Jobbágy, E.G., Paruelo, J. M. 2006. Carbon sequestration in semiarid rangelands: Comparison of Pinus ponderosa plantations and grazing exclusion in NW Patagonia. Journal of Arid Environments 67: 142-156.

Paruelo, J.M., Beltrán, A., Jobbágy, E.G., Sala, O., Golluscio, R.A. 1998. The Climate of Patagonia: general patterns and controls on biotic processes. Ecología Austral 8: 85-101.

Paruelo, J.M., Sala, O.E., Beltran, A.B., 2000. Long-term dynamics of water and carbon in semi-arid ecosystems: a gradient analysis in the Patagonian steppe. Plant Ecology 150: 133–143.

Pothier, D., Prevost, M., Auger, I., 2003. Using the shelterwood method to mitigate water table rise after forest harvesting. Forest Ecology and Management 179: 573–583.

Rasband, W.S. 2006. ImageJ, U. S. National Institutes of Health, Bethesda, Maryland, USA, http://rsb.info.nih.gov/ij/

Ryan, M.G., Bond, B.J., Law, B.E., Hubbard, R.M., Woodruff, D., Cienciala, E., Kucera, E. 2000. Transpiration and whole-tree conductance in ponderosa pine trees of different heights. Oecologia 124: 553-560.

Saxton K.E., Rawls W.J., Romberger J.S. & Papendick R.I. (1986) Estimating generalized soil-water characteristics from texture. Soil Science Society of America Journal 90, 1031–1036.

Sedjo, R.A. and D. Botkin. 1997. Using forest plantations to spare natural forests. Environment 30: 15-20.

Sedjo, R.A. 1999. The potential of high-yield plantation forestry for meeting timber needs. New Forests 17: 339-359.

Schlichter, T., Laclau, P. 1998. Ecotono estepa-bosque y plantaciones forestales en Patagonia norte. Ecologia Austral 8: 285-296.

Schulze, E.D., Mooney, H.A., Sala, O.E., Jobbágy, E., Buchmann, N., Bauer, G., Canadell, J., RB Jackson, R.B., Loreti, J., M Oesterheld, M., Ehleringer, J.R. 1996. Rooting depth, water availability, and vegetation cover along an aridity gradient in Patagonia. Oecologia 108: 503-511.

Simonin, K., Kolb, T.E., Montes-Helu, M., Koch, G.W. 2007. The influence of thinning on components of stand water balance in a ponderosa pine forest stand during and after extreme drought. Agricultural and Forest Meteorology, 143: 266-276.

Siry, J. P., Cubbage, F. W., Ahmed, M. R. 2005. Sustainable forest management: global trends and opportunities. Forest Policy and Economics 7: 551-561

Unsworth, M.H., Phillips, N., Link, T., Bond, B.J., Falk, M., Harmon, M., Hinckley, T., Marks, D., Paw U, K.-T. 2004. Components and controls of water flux in an old-growth Douglas-fir/western hemlock ecosystem. Ecosystems 7: 1–14.

van Wilgen, B.W., R.M. Cowling and C.J. Burgers. 1996. Valuation of ecosystem services. BioScience 46: 184-189.

Veblen,T.T., Kitzberger, T., Villalba, R., Donnegan, J. 1999. Fire History in Northern Patagonia -The Roles of Humans and Climatic Variation. Ecological Monographs 69: 47-67.

Vertessy, R.A., Bessard, Y. 1999. Conversion of grasslands to plantations: anticipating the negative hydrologic effects. VIth International Rangeland Congress Proceedings (2); Townsville, QLD., Australia 679-683.

Vertessy, R., Zhang, L., Dawes,W. 2002. Plantations, river flows and river salinity. In: Gerrand (ed.) Proceedings of the Prospects for Australian forest plantations 2002 conference, 20-21 August 2002 Canberra, Australia. Bureau of Rural Sciences, Australia. 97-109.

Warren, J.M., F.C. Meinzer, J.R. Brooks, and J.C. Domec. 2005. Vertical stratification of soil water storage and release dynamics in Pacific Northwest coniferous forests. Agricultural and Forest Meteorology 130: 39-58

Xu, Y.-J., Burger, J.A., Aust, W.M., Patterson, S.C., Miwa, M., Preston, D.P. 2002. Changes in surface water table depth and soil physical properties after harvest and establishment of loblolly pine (Pinus taeda L.) in Atlantic coastal plain wetlands of South Carolina. Soil Tillage Res. 63: 109–121.

Zhou et al. 2002. Hydrological impacts of reafforestation with eucalypts and indigenous species; a case study in southern China. Forest Ecology and Management 167: 209-222.

Chapter 3

AIC. Administracion Interjurisdiccional de las Cuencas de los rios Limay, Neuquen y Negro. 2005. Informes hidrometeorologicos mensuales. Enero – Diciembre 2005. Año XII, vols: 140-151.

Allen, G.A., Pereira, L.S., Raes, D., Smith, M. 1998. Crop Evapotranspiration – guidelines for Computing Crop Water Requirements. FAO: Irrigation and Drainage Paper 56. United Nations, Rome, Italy.

Bryant, M.L., Bhata, S., Jacobs, J.M. 2005. Measurements and modeling of throughfall variability for five forest communities in the southeastern US. J. Hydrol. 312, 95-108.

Carlyle-Moses, D.E., 2004. Throughfall, stemflow and canopy interception loss fluxes in a semi-arid Sierra Madre Oriental matorral community. Journal of Arid Environments 58, 181–202.

Cuartas, L.A., Tomasela, J. Nobre, A.D., Hodnet, M.G. Waterloo, M.J., Munera, J.C. 2007. Interception water-partitioning dynamics for a pristine rainforest in Central Amazonia: Marked differences between normal and dry years. Ag. For. Met. 145, 69–83.

Crockford, R.H., Richardson, D.P. 2000. Partitioning of rainfall into throughfall, stemflow, and interception: effect of forest type, ground cover and climate. Hydrol. Process. 14:2903–2920.

Deguchi et al. 2006. The influence of seasonal changes in canopy structure on interception loss; Application of the revised Gash model. J. Hydrol. 318, 80-102.

Evans, J, Turnbull, J.W. 2004. Plantation forestry in the tropics. Oxford University Press pp 13-22. 488pp

Fahey, B., Jackson, R. 1997. Hydrological impacts of converting native forests and grasslands to pine plantations, South Island, New Zealand. Ag. For. Met. 84:69-82.

Gash, J.H.C. 1979. An analytical model of rainfall interception by forest. Q. J. R. Meteorol. Soc. 105, 43–55.

Gash, J.H.C., Lloyd, C.R., Lachaud, G. 1995. Estimating sparse forest rainfall interception with an analytical model. J. Hydrol. 170, 79-86.

Gallo, L., Martinez Meier, A., Azpilicueta, M.M., Marchelli P., Mondino, V. 2005. III. 4. Subprograma coníferas y otras especies en la región patagónica, in: Norverto, C.A.(Ed.), Mejores arboles para mas forestadores. Secretaria de Agricultura, Pesca y Alimentacion, Buenos Aires, 95-116.

Hartley, M.J., 2002. Rationale and methods for conserving biodiversity in plantation forests. Forest Ecol. Manage. 155, 81–95.

Huxman, T., Wilcox, B.P., Breshears, D.D., Scott, R.L., Snyder, K.A., Small, E.E., Hultine, K., Pockman, W.T., Jackson, R.B. 2005. Ecohydrological implications of woody plant encroachment. Ecology, 86:308-319.

Iida, S., Tanaka, T., Sugita, M. 2005. Change of interception process due to the succession from Japanese red pine to evergreen oak. J. Hydrol., 315:154-166.

Iida, S., Tanaka, T., Sugita, M. 2006. Change of evapotranspiration components due to the succession from Japanese red pine to evergreen oak. J. Hydrol., 326:166-180.

Jackson, R.B., H.J. Schenk, E.G. Jobbagy, J.Canadell, G.D. Colello, R.E. Dickinson, C.B. Field, P. Friedlingstein, M. Heimann, K. Hibbard, D.W. Kicklighter, A. Kleidon, R.P. Neilson, W.J. Parton, O.E. Sala, M.T. Sykes. 2000. Belowground consequences of vegetation change and their treatment in models. Ecol. Appl.10:470-483.

Klaassen, W., Bosveld, F., de Water, E. 1998. Water storage and evaporation as constituents of rainfall interception. J. Hydrol. 212-213: 36-50.

Kuhn, T.J., Tate, K.W., Cao, D., George, M.R. 2007. Juniper removal may not increase overall Klamath River Basin water yields. California Agriculture: 61:166-171.

Laclau, P. 2003. Biomass and carbon sequestration of ponderosa pine plantations and native cypress forests in northwest Patagonia. Forest Ecology and Management 180: 317-333.

Laclau, P., Andenmatten, E. 2005. La calidad de sitio y el negocio forestal. IDIA XXI Forestales, 230-232.

Lankreijer, H.J.M., Hendriks, M.J., Klaassen, W., 1993. A comparison of models simulating rainfall interception of forests. Ag. For. Met., 64, 187–199.

Le Maitre, D.C. 2004. Predicting invasive species impacts on hydrological processes: The Consequences of Plant Physiology for Landscape Processes. Weed Technology 18:1408–1410.

Licata, J.A., Gyenge, J.E., Fernandez, M.E., Schlichter, T.M., Bond, B.J. 2007 *in press*. Increased water use by ponderosa pine plantations in northwestern Patagonia, Argentina compared with native forest vegetation, Forest Ecol. Manage., doi:10.1016/j.foreco.2007.09.061.

Link, T.E., Unsworth, M.H., Marks, D., 2004. The dynamics of rainfall interception by a seasonal temperate rainforest. Ag. For. Met., 124, 171–191.

Llorens, P., Gallart, F. 2000. A simplified method for forest water storage capacity measurement. J. Hydrol 240:131-144.

Llorens, P., Domingo, F. 2007. Rainfall partitioning by vegetation under Mediterranean conditions: A review of studies in Europe. J. Hydrol. 335:37–54

Loustau, D., Berbigier, P., Granier, A., 1992. Interception loss, throughfall and stemflow in a maritime pine stand II. An application of Gash's analytical model of interception. J. Hydrol. 138, 469–485.

Merenson C. 1992. Desarrollo Sustentable o Deforestación. Plan Forestal Argentino. Dirección de Recursos Forestales Nativos. Buenos Aires, 55 pp.

Michelson, D.B., 2004. Systematic correction of precipitation gauge observations using analyzed meteorological variables. J. Hydrol. 290, 161–177.

Montenegro, C., Gasparri, I., Manghi, E., Strada., M., Bono, J., Parmuchi, M.G. Unidad de Manejo del Sistema de Evaluacion Forestal. 2004. Informe sobre deforestacion Argentina. Secretaria de ambiente y desarrollo sustentable. Direccion de Bosques. Buenos Aires, 8 pp.

Nosetto, M. D., Jobbágy, E.G., Paruelo, J. M. 2005. Land-use change and water losses: the case of grassland afforestation across a soil textural gradient in central Argentina. Global Change Biology 11: 1101-1117.

Owens, M.K., Lyons, R.K., Alejandro, C.L. 2006. Rainfall partitioning within semiarid juniper communities: effects of event size and canopy cover. Hydrol. Process. 20: 3179-3189.

Price, A.G., Carlyle-Moses, D.E., 2003. Measurement and modelling of growing-season canopy water fluxes in a mature mixed deciduous forest stand, southern Ontario, Canada. Ag. For. Met., 119, 69–85.

Pypker, T.G., Bond, B.J., Link, T.E., Marks, D., Unsworth, M.H. 2005. The importance of canopy structure in controlling the interception loss of rainfall:

Examples from a young and an oldgrowth Douglas-fir forest. Ag. For. Met. 130 (1–2): 113–129.

Roberts, J. 2007. The role of plant physiology in hydrology: looking backwards and forwards. Hydrology and Earth System Sciences 11: 256-269.

Savenije, H.H.G. 2004. The importance of interception and why we should delete the term evapotranspiration from our vocabulary. Hydrol. Process. 18: 1507–1511.

SAGPyA. Secretaria de Agricultura Pesca y Alimentacion. 1999. Argentina investment opportunities in plantation forests. Buenos Aires. 208 pp.

Sarasola, M., Rusch, V.E., Schlichter, T.M., Ghersa., C.M. 2006. Invasión de coníferas forestales en áreas de estepa y bosques de ciprés de la cordillera en la Región Andino Patagónica. Ecologia Austral 16: 143-156.

SAyDS. Secretaria de Ambiente y Desarrollo Sustentable. 2006. Tercer informe nacional para la implementación de la convención de las Naciones Unidas de lucha contra la desertificación. Buenos Aires, 77 pp.

Schlichter, T., Laclau, P. 1998. Ecotono estepa-bosque y plantaciones forestales en Patagonia norte. Ecologia Austral 8: 285-296.

Sedjo, R.A., Botkin,D. 1997. Using forest plantations to spare natural forests. Environment 30: 15-20.

Seyfried, M.S., Wilcox, B.P. 2006. Soil water storage and rooting depth: key factors controlling recharge on rangelands. Hydrol. Pocess. 20: 3261-3275.

Teklehaimanot, Z., Jarvis, P.G., Ledger, D.C. 1991. Rainfall interception and boundary layer conductance in relation to tree spacing. J. Hydrol. 123: 261-278.

Thurow, T.L., Hester, J.W. 1997. How an increase or a reduction in juniper cover alters rangeland hydrology. Juniper Symp. Proc. p. 9-22. Texas A&M Univ., San Angelo, Tex.

UMSEF. Unidad de Manejo del Sistema de Evaluacion Forestal. 2007. Monitoreo de Bosque Nativo. Secretaria de ambiente y desarrollo sustentable. Direccion de Bosques. Buenos Aires, 11 pp.

Unsworth, M.H., Phillips, N., Link, T., Bond, B.J., Falk, M., Harmon, M., Hinckley, T., Marks, D., Paw U, K.-T. 2004. Components and controls of water flux in an old-growth Douglas-fir/western hemlock ecosystem. Ecosystems 7: 1–14.

Valente, F., David, J.S., Gash, J.H.C. 1997. Modelling interception loss for two sparse eucalypt and pine forests in central Portugal using reformulated Rutter and Gash analytical models. J. Hydrol. 190: 141-162.

Vertessy, R. 2000. Impacts of Plantation Forestry on Catchment Runoff. In: Plantations, Farm Forestry and water: Workshop Proceedings Publication No. 01/20. Edited by E.K Sadanandan Nambiar and Alan G. Brown

Wilcox, B.P. 2002. Shrub control and streamflow on rangelands; a process-based viewpoint. Journal of Range Management 55: 318–326.

Wilcox, B. P., Breshears, D. D., Allen, C. D. 2003. Ecohydrology of a resource-conserving semiarid woodland: temporal and spatial scaling and disturbance. Ecological Monographs 73: 223–239

Zhang, L., Dawes, W.R., Walker, G.R. 2001. Response of mean annual evapotranspiration to vegetation changes at the catchment scale. Water Resources Research 37: 701–708.

Chapter 4

AIC. Administracion Interjurisdiccional de las Cuencas de los rios Limay, Neuquen y Negro. 2005. Informes hidrometeorologicos mensuales. Enero – Diciembre 2005. Año XII, vols: 140-151.

Allen, CD. 2007. Interactions Across Spatial Scales among Forest Dieback, Fire, and Erosion in Northern New Mexico Landscapes. Ecosystems 10:797-808.

Allen, C. D., M. Savage, D. A. Falk, K. F. Suckling, T. W. Swetnam, T. Schulke, P. B. Stacey, P. Morgan, M. Hoffman, and J. T. Klingel. 2002. Ecological restoration of southwestern ponderosa pine ecosystems: a broad perspective. Ecological Applications 12:1418–1433.

Boisvenue, C and Running, SW. 2006. Impacts of climate change on natural forest productivity – evidence since the middle of the 20th century. Global Change Biol. 12:862-882.

Boulanger J-P, Martinez F, Segura EC, 2006. Projection of future climate change conditions using IPCC simulations, neural networks and Bayesian statistics. Part 1: Temperature mean state and seasonal cycle in South America. *Clim Dyn* 27:233–259

Boulanger J-P, Martinez F, Segura EC, 2007. Projection of future climate change conditions using IPCC simulations, neural networks and Bayesian statistics. Part 2: Precipitation mean state and seasonal cycle in South America. *Clim Dyn* 28:255-271.

Christensen, J.H., B. Hewitson, A. Busuioc, A. Chen, X. Gao, I. Held, R. Jones, R.K. Kolli, W.-T. Kwon, R. Laprise, V. Magaña Rueda, L. Mearns, C.G. Menéndez, J. Räisänen, A. Rinke, A. Sarr and P. Whetton, 2007: Regional Climate Projections. In: Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change [Solomon, S., D. Qin, M. Manning, Z. Chen, M. Marquis, K.B. Averyt, M. Tignor and H.L. Miller (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Farley, K, Jobbágy, E.G. and Jackson, R.B. 2005. Effects of afforestation on water yield: a global synthesis with implications for policy. Global Change Biology 11:1565-1576.

Farquhar, G.D. ,von Caemmerer, S. 1982. Modelling of photosynthetic response to the environment, in Physiological Plant Ecology II: Encyclopedia of Plant Physiology, edited by O. L. Lange et al., pp. 549–587, Springer-Verlag, New York.

Fisher, RA, Williams, M, Lola da Costa, A., Malhi, Y, Costa, RF, Almeida, S, Meir, P. 2007. The response of an Eastern Amazonian rain forest to drought stress: results and modelling analyses from a throughfall exclusion experiment. Global Change Biology, 13:2361–2378.

Gill, R.A., Jackson, R.B. 2000. Global patterns of root turnover for terrestrial ecosystems. New Phytologist 147: 13–31.

Gyenge, J.E., Fernández M.E., Dalla Salda, G., Schlichter, T. 2005. Leaf and whole-plant water relations of the Patagonian conifer Austrocedrus chilensis (D. Don) Pic. Ser. et Bizzarri: implications on its drought resistance capacity. Ann. For. Sci. 62: 297–302.

Harley, P.C., Thomas, R.B., Reynolds, J.F., Strain, B.R. 1992. Modelling photosynthesis of cotton grown in elevated CO2. Plant, Cell and Environment 15: 271–282.

Jackson R.E., Sperry, J.S., Dawson, T.E. 2000. Root water uptake and transport: using physiological processes in global predictions. Trends in Plant science, 5:482-488

Law, B.E., Williams, M., Anthoni, P.M., Baldocchi, D.D., Unsworth, M.H. 2000. Measuring and modelling seasonal variation of carbon dioxide and water vapour exchange of a Pinus ponderosa forest subject to soil water deficit, Global Change Biol., 6, 613–630.

Law, B.E., Cescatti, A., Baldocchi D.D. 2001, Leaf area distribution and radiative transfer in open-canopy forests: Implications for mass and energy exchange, Tree Physiol., 21, 777–787.

Lawton, J. 2007. Ecology, politics and policy. Journal of Applied Ecology,44:465-474.

Lenihan, J.M., Bachelet, D., Neilson, R.P., Drapek, R. 2007. Response of vegetation distribution, ecosystem productivity, and fire to climate change scenarios for California. Climatic Change: DOI 10.1007/s10584-007-9362-0

Le Treut, H., R. Somerville, U. Cubasch, Y. Ding, C. Mauritzen, A. Mokssit, T. Peterson and M. Prather, 2007: Historical Overview of Climate Change. In: Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change [Solomon, S., D. Qin, M. Manning, Z. Chen, M. Marquis, K.B. Averyt, M. Tignor and H.L. Miller (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Licata, J.A., Gyenge, J.E., Fernandez, M.E., Schlichter, T.M., Bond, B.J. 2007 *in press*. Increased water use by ponderosa pine plantations in northwestern Patagonia, Argentina compared with native forest vegetation, Forest Ecol. Manage., doi:10.1016/j.foreco.2007.09.061.

Magnani, F., Consiglio, L., Erhard, M., Nole, A., Ripullone, F., Borghetti, M. 2004. Growth patterns and carbon balance of Pinus radiata and Pseudotsuga menziesii plantations under climate change scenarios in Italy. For Ecol Man, 202: 93-105.

Marland, G., Pielke, R.A., Apps, M., Avissar, R., Betts, Davis, K.J., Frumhoff, P.C.,and co-authors. 2003. The climatic impacts of land surface change and carbon management, and the implications for climate-change mitigation policy. Climate Policy 3:149-157.

McDowell, N.G., Adams, H., Bailey, J.D., Hess, M., Kolb, T.E. 2006. Homeostatic maintenance of ponderosa pine gas exchange in response to stand density changes. Ecological Applications 16: 1164–1182

Nemani, R.R., Keeling, C.D., Hashimoto, H., Jolly, W.M., Piper, S.C., Tucker, C.J., Myneni, R.B., Running, S.W. 2003. Climate-driven increases in global terrestrial net primary production from 1982 to 1999. Science 300:1560-1563.

Nosetto, M.D., Jobbágy, E.G., Paruelo, J.M. 2006. Carbon sequestration in semiarid rangelands: Comparison of Pinus ponderosa plantations and grazing exclusion in NW Patagonia. Journal of Arid Environments 67: 142-156.

Panek, J.A. 2004. Ozone uptake, water loss and carbon exchange dynamics in annually drought-stressed Pinus ponderosa forests: measured trends and parameters for uptake modeling. Tree Physiology 24: 277–290

Rasband, W.S. 2006. ImageJ, U. S. National Institutes of Health, Bethesda, Maryland, USA, http://rsb.info.nih.gov/ij/

SAGPyA. Secretaria de Agricultura Pesca y Alimentacion. 1999. Argentina investment opportunities in plantation forests. Buenos Aires, 208pp.

Sarasola, M., Rusch, V.E., Schlichter, T.M., Ghersa., C.M. 2006. Invasión de coníferas forestales en áreas de estepa y bosques de ciprés de la cordillera en la Región Andino Patagónica. Ecologia Austral 16: 143-156.

Saxton K.E., Rawls W.J., Romberger J.S., Papendick R.I. 1986. Estimating generalized soil-water characteristics from texture. Soil Science Society of America Journal 90: 1031–1036.

Schaap, M.G., Leij, F.J., van Genuchten, M.Th. 2001. Rosetta: a computer program for estimating soil hydraulic parameters with hierarchical pedotransfer functions. Journal of Hydrology, 251: 163-176.

Schulze, E.D., Valentini, R., Sanz, M.J., 2002. The long way from Kyoto to Marrakesh: implications of the Kyoto Protocol negotiations for Global Ecology. Global Change Biol. 8: 1–14.

Schwarz P.A., Law B.E., Williams M., Irvine, J., Kurpius, M., Moore, D. 2004. Climatic vs. biotic constraints on carbon and water fluxes in seasonally droughtaffected ponderosa pine ecosystems. Global Biogeochem. Cycles, 18: GB4006.

Tierney, G.L., Fahey, T.J., Groffman, P.M., Hardy, J.P., Fitzhugh, R.D., Driscoll, C.T. 2003. Environmental control of fine root dynamics in a northern hardwood forest. Global Change Biol, 9: 670-679.

van Genuchten, M.Th. 1980. A closed-form equation for predicting the hydraulic conductivity of unsaturated soils. Soil Sci. Am. J. 44: 892-898

Vera, C., Silvestri, G., Liebmann, B., González P. 2006. Climate change scenarios for seasonal precipitation in South America from IPCC-AR4 models, Geophys. Res. Lett. 33: L13707.

Waring, R. H., Pitman, G.B. 1985. Modifying lodgepole pine stands to change susceptibility to mountain pine beetle attack. Ecology 66: 889–897.

Westerling, A.L., Hidalgo, H.G., Cayan, D.R., Swetnam, T.W. 2006. Warming and earlier spring increase western U.S. forest wildfire activity. Science 313: 940-943. 10.1126/science.1128834

Williams, M., Rastetter, E.B., Fernandes, D.N., Goulden, M.L., Wofsy, S.C., Shaver, G.R., Melillo, J.M., Munger, J.W., Fan, S.M., Nadelhoffer, K.J. 1996. Modelling the soil-plant-atmosphere continuum in a Quercus-Acer stand at Harvard forest: The regulation of stomatal conductance by light, nitrogen and soil/plant hydraulic properties, Plant Cell Environ., 19: 911 –927.

Williams, M., Malhi, Y., Nobre, A.D., Rastetter, E.B., Grace, J., Pereira, M.G.P. 1998. Seasonal variation in net carbon exchange and evapotranspiration in a Brazilian rain forest: A modelling analysis, Plant Cell Environ., 21: 953– 968.

Williams, M., Eugster, W., Rastetter, E.B., McFadden, J.P., Chapin, F.S. 2000. The controls on net ecosystem productivity along an Arctic transect: A model comparison with flux measurements, Global Change Biol., 6: 116–126.

Williams, M., Bond, B.J., Ryan, M.G. 2001a. Evaluating different soil and plant hydraulic constraints on tree function using a model and sap flow data from ponderosa pine, Plant Cell Environ., 24: 679– 690.

Williams, M., Law, B.E., Anthoni, P.M., Unsworth, M.H. 2001b. Use of a simulation model and ecosystem flux data to examine carbon-water interactions in ponderosa pine, Tree Physiol., 21, 287– 298.