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El efecto de alteraciones antrópicas en los procesos hidrológicos de cuencas experimentales en un gradiente latitudinal en la Cordillera de la Costa entre las Regiones del Maule y Los Ríos, Chile.

Tesis para optar el grado de Doctor en Recursos Hídricos y Energía para la Agricultura

FRANCISCO JAVIER BALOCCHI CONTRERAS

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Profesor Guía: José Luis Arumí Ribera
Dpto. de Recursos Hídricos
Facultad de Ingeniería Agrícola
Universidad de Concepción
Profesor Co-Guía: Diego Rivera Salazar
Facultad de Ingeniería
Universidad del Desarrollo

Esta tesis ha sido realizada en el Departamento de Recursos Hídricos de la Facultad de Ingeniería Agrícola, Universidad de Concepción.

Profesor Guía

Dr. José Luis Arumí Ribera

Facultad de Ingeniería Agrícola

Universidad de Concepción

Profesor Co-Guía

Dr. Diego Rivera Salazar

Facultad de Ingeniería

Universidad del Desarrollo

Comisión Evaluadora:

Dra. Alejandra Stehr Gesche

Facultad de Ingeniería

Universidad de Concepción

Dr. Luis Octavio Lagos Roa

Facultad de Ingeniería Agrícola

Universidad de Concepción

Dr. Stephen Good

Biological & Ecological Engineering

Oregon State University

Director de Programa

Dr. José Luis Arumí Ribera

Facultad de Ingeniería Agrícola

Universidad de Concepción

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INDICE

RESUMEN

Chile se caracteriza por una alta diversidad de bosques naturales, plantaciones forestales, tipos de suelo, topografía, geología y patrones de precipitación en un amplio gradiente latitudinal, longitudinal y altitudinal. Además, el país presenta una larga historia de cambios de uso de suelo por efectos antrópicos desde la era precolombina (*i.e.*, tala de bosque nativo y quemas para habilitación de terreno). Desde las comunidades locales, la academia, la industria y el Estado existe la preocupación sobre el efecto del cambio de uso de la tierra (*e.g.*, reemplazo de praderas por plantaciones y los incendios) sobre los recursos hídricos como una alteración antrópica, tanto en cantidad, calidad, oportunidad y accesibilidad. Sin embargo, a pesar de más de 47 años de investigación de la hidrología forestal en Chile, aún no es claro el efecto de cambio de uso en los recursos hídricos y la interacción con el gradiente latitudinal de precipitaciones.

El objetivo de esta tesis fue evaluar la evidencia sobre el impacto del cambio de uso y los incendios, sobre la cantidad de agua en Chile centro-sur. En particular, se pone foco en el efecto de estas alteraciones en la generación de caudales base y estivales en cuencas forestales con diferentes tipos de cobertura vegetal en la macrozona forestal (entre las Regiones del Maule y de Los Ríos). En una primera aproximación (Capítulo II), se estudió la generación de caudales recesivos en ocho pequeñas cuencas forestales en Chile centro-sur, comparando el efecto de diferentes coberturas sobre estos caudales en un gradiente latitudinal, incluyendo variables morfométricas. En una segunda aproximación (Capítulo III), se estudió el efecto del mega incendio 2017 en tres microcuencas en Chile central a través de métricas hidrológicas tales como el coeficiente de escorrentía y el índice de caudal base, y mediciones del isótopo Tritio en el caudal. En el cuarto capítulo se identificó la necesidad de incentivar la investigación de la hidrología forestal tanto desde el Estado como del sector privado, así como la reforma de las políticas públicas, articulando las investigaciones en hidrología forestal con un imperativo sentido de integralidad, por lo que se presenta una propuesta de desarrollo integral y descentralizado de la investigación hidrológica forestal.

En el Capítulo II se determinó que, no obstante las diferencias morfométricas entre las cuencas estudiadas, de sus usos y clima, en época invernal no existen diferencias en la generación de caudales recesivos. Sin embargo, en verano es posible encontrar diferencias las cuales pueden deberse a diferencias en la evapotranspiración de las coberturas, y de la profundidad del suelo.

Para el caso del Capítulo II y el efecto del mega incendio sobre la hidrología de pequeñas cuencas en Chile central y contrariamente a la literatura, en la cuenca cubierta por bosque nativo no se observó un

aumento de los caudales punta, y con respecto a la escorrentía total no se observan alteraciones importantes dentro de los dos años siguientes al incendio. Para el caso de las dos cuencas con plantaciones, se observó un aumento de los caudales punta, posiblemente por efecto de la hidrofobicidad de los suelos, tal como se ha reportado en investigaciones en otros lugares.

Dadas ambas aproximaciones con respecto a dos alteraciones antrópicas diferentes (*i.e.*, plantaciones exóticas e incendios), es posible concluir que, a nivel de pequeñas cuencas experimentales, el suelo, la geología y particularmente el sistema de rocas fracturadas presente en ellas, jugarían un rol crucial en la interacción aguas superficiales-subterráneas de esos sistemas forestales. Por lo tanto, futuras investigaciones deberán estar enfocadas en los procesos subsuperficiales y en la interacción de aguas superficiales y subterráneas. Por ejemplo, el coeficiente de agotamiento de la curva recesiva como métrica hidrológica puede ayudar a desarrollar estrategias de planificación territorial con el fin de incrementar la provisión de agua en cuencas abastecedoras de agua potable.

ABSTRACT

Chile is characterized by a high diversity of natural forests, forest plantations, soil types, topography, geology, and precipitation patterns along a wide latitudinal, longitudinal, and altitudinal gradient. In addition, the country has a long history of land use changes due to human activities, such as deforestation and burning for land clearing, dating back to pre-Columbian times. Local communities, academia, industry, and the government are concerned about the impact of land use changes, such as replacing grasslands with plantations and forest fires, on water resources in terms of quantity, quality, timing, and accessibility. However, despite more than 47 years of research on forest hydrology in Chile, the effect of land use change on water resources and its interaction with the latitudinal precipitation gradient is still unclear.

The objective of this thesis was to evaluate the evidence of the impact of land use change and forest fires on water quantity in central-southern Chile. In particular, the focus was on the effect of these disturbances on baseflow and summer flow generation in forested catchments with different land cover in the forest macrozone (between the Maule and Los Rios regions). In the first approach (Chapter II), baseflow generation was studied in eight small forested catchments in central-southern Chile, comparing the effect of different land covers on baseflow along a latitudinal gradient, including morphometric variables. In the second approach (Chapter III), the effect of the 2017 mega fire on three small catchments in central Chile was studied using hydrological metrics such as the runoff coefficient and baseflow index, and measurements of the Tritium isotope in the flow. The fourth chapter identified the need to encourage forest hydrology research both by the government and the private sector, as well as reforming public policies to integrate forest hydrology research with an imperative sense of integrality. Also, Chapter IV presents a proposal for an integral and decentralized forest hydrological research development.

It was determined in Chapter II that, despite the morphometric differences between the studied catchments, their land use, and climate, there were no differences in baseflow generation during the winter. However, differences in summer flow generation may be due to differences in vegetation cover evapotranspiration and soil depth.

In Chapter III and regarding the effect of the mega fire on the hydrology of small catchments in central Chile, contrary to the literature, no increase in peak flows was observed in the native forest cover catchment, and no significant alterations in total runoff were observed within the two years following the fire. In the case of the two catchments with plantations, an increase in peak flows was observed, possibly due to the hydrophobicity of the soils, as reported in research elsewhere.

Finally, based on both approaches to two different anthropogenic disturbances (i.e., exotic plantations and forest fires), it is possible to conclude that, at the level of small experimental catchments, the soil, geology, and particularly the fractured rock system presents in them, play a crucial role in the interaction between surface and groundwater in those forest systems. Therefore, future research should focus on subsurface processes and the interaction between surface and groundwater. For example, the recession curve depletion coefficient as a hydrological metric can help develop territorial planning strategies to increase water supply in drinking water supply catchments.

CAPÍTULO I: INTRODUCCIÓN

1. Introducción

En Chile 17,7 millones de hectáreas (ha) del territorio (23%) presentan cobertura boscosa y 3,1 millones de ha corresponden a plantaciones exóticas (Ministerio de Agricultura, 2021). Desde ca. 1550 hasta el 2007, se estima que la pérdida del bosque nativo fue del orden del 51% debido, principalmente, a la habilitación de tierras para la agricultura y pastizales (Lara *et al.*, 2012). Así, el paisaje actual de Chile centro-sur resulta de una larga historia de agricultura intensiva y de producción de madera para diferentes usos en los periodos pre y post colombino (Balocchi *et al.*, 2022a).

El efecto de las plantaciones sobre la cantidad y calidad de agua ha sido uno de los principales focos de investigación desde la academia y desde la comunidad en general. Por ejemplo, entre el año 1975 al 2022 existen 75 artículos que han investigado el efecto anteriormente descrito, concentrados principalmente entre los años 2010 y 2022 (Balocchi *et al.*, 2022a). Aunque es clara la importancia del bosque desde lo ambiental y desde los servicios ecosistémicos, desde la economía regional y del bienestar de la comunidad, aún existe la inquietud en cuanto al real uso del agua de las diferentes coberturas y cómo éstas varían en el territorio, y del efecto sobre los procesos hidrológicos a nivel local y global (Balocchi *et al.*, 2021a). Por lo tanto, el estudio del efecto de las alteraciones antrópicas, como la plantación de especies exóticas, cosecha o incendios, sobre los procesos hidrológicos en Chile permite esclarecer si existe evidencia de que el establecimiento de especies exóticas ha intensificado la presión sobre los recursos hídricos (Lara *et al.*, 2010).

1.1. Bosque, agua, cambio climático y demanda de agua

Son tres las variables que complejizan el entendimiento del efecto de la cobertura vegetal sobre los procesos hidrológicos y, en particular, en la generación de caudales (*i.e.*, cantidad de agua). Por un lado, se tiene la evidente disminución de las precipitaciones en la zona centro sur de Chile y el aumento de la temperatura (Bates *et al.*, 2008), que tiene como consecuencia el aumento de la evapotranspiración y la disminución de agua para la generación de caudales. Boisier *et al.* (2016), por ejemplo, han estimado un déficit de precipitaciones del orden del 30% en el norte y centro de Chile. En segundo lugar, el cambio de uso de suelo - establecimiento de plantaciones y los incendios forestales -puede tener diferentes efectos sobre los procesos hidrológicos. Por un lado, el establecimiento de una plantación puede producir una disminución del caudal anual y de los caudales de verano (*e.g.*, Lara *et al.*, 2009; Little *et al.*, 2009; Iroumé y Palacios, 2013), pero en otros estudios no se han encontrado cambios (cf. Pizarro *et al.*, 2022;

Pizarro *et al.*, 2006; Brown, 1971). Por ello, la evidencia sobre efectos negativos no es confluyente, aún. De los efectos positivos de las plantaciones, por ejemplo, se encuentran la protección de suelos degradados y la capacidad de fijar grandes cantidades de carbono (*e.g.*, White *et al.*, 2021). La tercera variable se relaciona con la demanda de agua. Esta demanda se ha incrementado cerca de dos veces en relación con el crecimiento de la población en el último siglo (Rivas *et al.*, 2020). Además, se espera que la demanda por agua se incremente en las siguientes décadas (Fernández *et al.*, 2018) sumándose a esto las tendencias negativas en la precipitación en Chile (Barria *et al.*, 2017; Bozkurt *et al.*, 2018) lo que produciría un desbalance oferta demanda importante.

1.2. Cuantificación del uso de agua en diferentes coberturas vegetales

En Chile las experiencias en investigación en la hidrología forestal se iniciaron con trabajos enfocados en intercepción de plantaciones y bosque nativo (*e.g.*, Huber y Oyarzun, 1983, 1984, 1985). Sin embargo, los estudios se han concentrado, principalmente, en evaluar el balance hídrico en plantaciones y bosque nativo en cuencas pareadas de tamaños pequeños (<1000 ha) (*e.g.*, Huber y López, 1993; Huber *et al.,* 1998; Huber e Iroumé, 2001; Huber y Trecaman, 2004; Echeverría *et al.*, 2007; Huber *et al.*, 2010), pero el efecto de las actividades forestales (*e.g.*, silvicultura) sobre la cantidad y calidad del agua no ha sido desarrollado mayormente (Iroumé, 2003; Little y Lara, 2010; Oyarzún et al. 2011). McDonnell *et al.* (2018) debaten sobre la falta de información que se tiene sobre los procesos subsuperficiales y como estos no se están incluyendo en los análisis del efecto de las plantaciones sobre las variables hidrológicas. El reemplazo de pradera y/o bosque nativo por *Pinus radiata* o *Eucalyptus* spp. provoca una disminución de los caudales anuales y los de verano en la zona sur (la concentración de estudios está en las cercanías de la ciudad de Valdivia). Este efecto se acentúa en la zona centro por la disminución de las precipitaciones y la mayor temperatura. Sin embargo, existe una alta variabilidad en los estudios y se advierten algunas brechas y oportunidades de investigación en Chile (consultar Balocchi *et al.*, 2022a). Por ejemplo, para cuencas de gran tamaño, existe poca evidencia que el cambio de bosque nativo por plantaciones genere un decrecimiento marcado en los caudales (Pizarro *et al.*, 2022; Esse *et al.*, 2021; Pizarro *et al.*, 2006). El efecto sobre el caudal anual es notorio en cuencas pequeñas, pero este efecto se ve reducido cuando la cuenca es reforestada o se recupera naturalmente (Tamai *et al.*, 2020).

El efecto de la cobertura vegetal sobre el caudal estival ha sido investigado por variados autores, tanto a nivel global (*e.g.*, Yao *et al.*, 2012; Jones *et al.*, 2020) como en Chile (*e.g.*, Iroumé *et al.*, 2005; Little *et al.*, 2009; Aguayo *et al.*, 2016). Sin embargo, el análisis de los caudales recesivos en cuencas forestales (y experimentales) no ha sido cubierta. Con respecto al efecto de las actividades forestales en lo referente a las aguas subterráneas son aún incipientes en materia de investigación (McDonnell *et al.*, 2018).

1.3. Evaluación de alteraciones antrópicas en los procesos hidrológicos en cuencas forestales; plantación de exóticas e incendios forestales

Los procesos hidrológicos en cuencas forestales han sido estudiados desde una mirada más bien clásica (*i.e.*, análisis aguas superficiales y balances hídricos simples) y no integrada. La dinámica de los procesos hidrológicos en cuencas de alta pendiente, como es el caso de la Cordillera de la Costa en Chile que es en donde se concentra la mayor parte de bosques y plantaciones, es afectada por la topografía (Hotta *et al.*, 2010) y complejiza el análisis. Por ejemplo, pendientes fuertes se asocian con niveles freáticos de mayor profundidad, pero la topografía controla la entrega de agua principalmente (superficial y sub-superficial), además de la geología y el clima (Wolock *et al.*, 2004; Devito *et al.*, 2005). Para pendientes moderadas y empinadas, la gravedad es el principal impulsor del flujo de agua subterránea (Troch *et al.*, 2002). Para pendientes poco pronunciadas, el drenaje difuso tiene un impacto importante en el flujo de agua subterránea (Troch *et al.*, 2003). En este sentido, un análisis específico de las curvas de recesión en diferentes coberturas ha sido poco investigado (*e.g.*, Yang *et al.*, 2018) a pesar de que este análisis puede llevar a interesantes resultados como lo es el efecto o incerteza del espesor del acuífero (Dewandel *et al.*, 2003). Así, cuando se desconoce la configuración del subsuelo es complejo atribuir la disminución o aumento de los caudales recesivos o de los caudales base solamente a la cobertura o aun cambio de cobertura. Debido a que las características geomorfológicas y geológicas de una cuenca no varían en escalas temporales anuales, la variabilidad estacional de los caudales recesivos está gobernada por la climatología de la cuenca, así como también por las variaciones estacionales de la cobertura (Wittenberg, 2003). También, es plausible una relación no lineal entre la capacidad de almacenar agua del suelo y la dinámica de entrega de agua de este debido principalmente a la alta variabilidad de las precipitaciones.

En relación con los incendios forestales, en Chile, el 99% son producidos por efectos antrópicos (intencionales, descuidos, entre otros) (Corporación Nacional Forestal [CONAF], 2023). En cuanto a la superficie afectada, en el periodo 2021-2022 existió un aumento del 41% de incendios a nivel nacional (CONAF, 2023). Sin embargo, a pesar de los efectos negativos de los incendios sobre vidas humanas, la biodiversidad y los servicios ecosistémicos, desde la hidrología existe un número limitado de estudios ligados a los efectos de los incendios sobre los procesos hidrológicos. Estos estudios se reducen a cinco, específicamente vinculados a la hidrología forestal (1975, 2010 - 2021). Jones *et al*. (1975) utilizaron un conjunto de datos comparando tres cuencas (35°S) que estaban cubiertas por un sistema forestal degradado en donde dos cuencas estaban parcialmente quemadas. Este estudio encontró que los caudales punta y la producción de sedimentos fueron mayores dentro de las cuencas quemadas en el período inmediatamente posterior al incendio. García-Chevesich *et al.* (2010), estudiaron la hidrofobicidad de suelos post-incendio en plantaciones de *P. radiata*. García-Chevesich *et al.* (2019) realizaron un estudio en rodales dominados por *Nothofagus glauca* y por *P. radiata*, encontrando suelos hidrofóbicos post-incendio. Balocchi *et al.* (2020) analizaron el caudal antes y después del incendio en una cuenca de bosque nativo dominada por *N. glauca*. Los resultados de estos últimos autores muestran que los incendios forestales podrían aumentar la infiltración del suelo en cubiertas con bosque nativo pues el caudal no aumentó a pesar de la severa reducción del área foliar del rodal. White *et al*. (2020), dentro de la misma cuenca de Balocchi *et al.* (2020), compararon el balance hídrico antes y después del incendio a escala de parcela. Encontraron una reducción del 63% en la evapotranspiración después del incendio en comparación con los valores previos al incendio. Sin embargo, el balance hídrico se recuperó sustancialmente después de dos años. Por lo anterior, existe una falta de estudios sobre los efectos de los incendios forestales sobre los procesos hidrológicos en Chile.

1.4. Estructura del documento

La estructura del presente documento consiste en cinco capítulos. El CAPÍTULO I corresponde a la introducción donde se describe el planteamiento del problema, además de las hipótesis y de los objetivos. El CAPÍTULO II analiza en pequeñas cuencas con diferentes coberturas forestales, el comportamiento de los caudales recesivos en invierno y verano. El CAPÍTULO III analiza a través de uno de los isótopos del agua (Tritio) y métricas hidrológicas, el comportamiento hidrológico pre y post-incendio en microcuencas de Chile central. El CAPÍTULO IV corresponde al análisis de la hidrología forestal y una propuesta de actuación a futuro para incrementar la investigación en el área. Finalmente, el CAPÍTULO V versa sobre las principales conclusiones y futuras líneas de investigación.

1.5. Hipótesis de trabajo

a) Las diferencias en la generación de caudales base y estivales entre cubiertas (bosque nativo, *eucalipto spp*. y *Pino radiata*), en igual condición de forzantes, se explican por aspectos topográficos y geomorfológicos.

b) El efecto antrópico de un incendio afecta en diferente magnitud y dirección los procesos hidrológicos y es dependiente del tipo de cobertura vegetal (bosque nativo, *eucalipto spp*. y *Pino radiata*).

1.6. Objetivos

Objetivo general

Evaluar el efecto de las alteraciones antrópicas en la generación de caudales base y estivales entre cuencas forestales con diferentes coberturas vegetales en la macrozona forestal (entre las Regiones del Maule y de Los Ríos).

Objetivos específicos

OE1: Analizar el efecto de las alteraciones antrópicas en la generación de caudales base y estivales en diferentes tipos de coberturas vegetales en Chile centro-sur a través del estudio de las curvas recesivas.

OE2: Evaluar el efecto de los incendios en la generación de caudales base y estivales en cuencas forestales en Chile centro-sur a través de métricas hidrológicas.

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CAPÍTULO II: Comparison of streamflow recession between plantations and native forests in small catchments in Central-Southern Chile[1]

Francisco Balocchi, Neftali Flores, José Luis Arumí, Andrés Iroumé, Don A. White. Richard P. Silberstein y Pablo Ramírez de Arellano

2.1. Resultados clave

-No existen diferencias entre coberturas en la generación de caudales recesivos en invierno

-No existen diferencias significativas entre los coeficientes de agotamiento de bosque nativo y plantaciones

-Existen diferencias en los caudales recesivos en verano, lo que se atribuye a la morfología de las cuencas; en los sitios del norte se aprecian diferencias que parecen ser debido a la cobertura

2.2. Resumen en extenso

Los caudales recesivos reflejan la liberación de agua desde el acuífero al cauce luego de una tormenta, los que no han sido ampliamente estudiados en Chile y tampoco se ha analizado su comportamiento bajo diferentes usos de suelo. En esta investigación, se estudiaron los caudales recesivos bajo diferentes usos de suelo en un gradiente latitudinal en Chile centro-sur.

La constante de recesión es un método bien establecido para comprender el efecto del clima y las perturbaciones en los caudales mínimos y caudales base, y puede mejorar los análisis locales y regionales de los procesos hidrológicos. El análisis de la recesión del caudal después de las lluvias en pequeñas cuencas, especialmente durante el verano, es vital para la gestión de los recursos hídricos en áreas donde el establecimiento de plantaciones se ha producido en un clima seco. Por lo tanto, la tasa de disminución del caudal después de un evento de lluvia es una buena medida del riesgo de que el caudal disminuya por debajo de un umbral crítico. Este riesgo de flujo bajo se puede cuantificar usando un coeficiente de recesión u agotamiento (α) que es la pendiente de una función de decaimiento exponencial que relaciona

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el flujo con el tiempo transcurrido desde la lluvia. Se utilizó un análisis de conglomerados jerárquico (*i.e.*, Scott-Knott) para identificar diferencias entre el valor medio del coeficiente de agotamiento (α) para diferentes regiones y tipos de bosques en invierno y verano. Para estimar el coeficiente α se utilizaron diferentes modelos matemáticos (*e.g.*, Boussinesq) en 142 tormentas (64 en verano, 78 en invierno) en ocho cuencas monitoreadas entre 2008 y 2017. Todas estas cuencas tienen una geología similar, se extienden entre los 35º a 39º S, su altitud varía entre 207 y 530 msnm y se encuentran ubicadas en la Cordillera de la Costa del centro-sur de Chile. Las coberturas variaron entre cuencas con bosque nativo y plantaciones de *Pino radiata* y *Eucalyptus spp*.

El análisis de las curvas de agotamiento arrojó que no hubo diferencias significativas en invierno ($p<0.05$) (representados en el coeficiente de agotamiento, α) entre las diferentes cuencas y usos de suelo. En este sentido y dado estos resultados, en invierno y con los suelos saturados, no se evidenciaron diferencias entre caudales recesivos. Por lo tanto, en invierno, la cobertura no juega un rol preponderante en la generación de caudales, no cumpliéndose completamente la hipótesis de trabajo propuesta, la que indica que las diferencias en la generación de caudales base y estivales entre cubiertas (bosque nativo, *Eucalipto spp.* y *Pino radiata*) en igual condición de forzantes, se explican por aspectos topográficos y geomorfológicos.

En verano, se encontraron diferencias entre coberturas, pero dependen de la morfología de las cuencas. En la zona norte es dable indicar que el uso de suelo es importante en la producción de agua en cuencas pequeñas. Además, α para cuencas con bosque nativo fue similar a aquellas con plantaciones de pino, aunque no hubo diferencia ($p<0.05$) entre estas y las plantaciones de eucalipto. En el caso del verano, la topografía solo explicó una parte de las diferencias entre cuencas.

En Chile centro-sur, muchas comunidades dependen del agua que obtienen de pequeñas cuencas en la cordillera de la costa. La seguridad del agua para estas comunidades es más vulnerable durante la estación seca y sumado a la disminución de las precipitaciones entre un 20 y 40% entre el periodo 2010-2017, se hace relevante estudiar el posible efecto del uso del suelo sobre la generación de caudales.

El efecto de la cobertura sobre la generación de caudales parece estar relacionado en mayor medida con aspectos topográficos y geomorfológicos más que la cobertura como tal, especialmente en invierno. Aun cuando se compararon coberturas que, en general, se asocian a mayor uso de agua (*e.g.*, Balocchi *et al.*, 2022a), no se encontraron diferencias significativas. Esto puede suceder por variados motivos, entre ellos, la baja evapotranspiración de las coberturas en esta época y el mayor aporte de lluvia (clima mediterráneo). En verano, sin embargo, esto cambia especialmente en los sitios más secos (ca. 800mm de precipitación). No obstante, no es claro si es directamente efecto de la cobertura o es efecto de la comparación entre suelos altamente degradados (*i.e.*, en donde hoy hay plantaciones) y suelos ricos en materia orgánica y estructuralmente superiores en sus características fisicoquímicas (*e.g.*, Cifuentes-Croquevielle *et al.*, 2020). En este mismo sentido, no fue posible diferenciar si la diferencia en la generación de caudales recesivos fue por un efecto de sólo la cobertura o existe un efecto del cambio climático. Por ejemplo, en Pizarro *et al.* (2022) la tesis principal del estudio es que la combinación de bosque nativo con plantaciones tiene un efecto positivo en la escorrentía de las cuencas y este efecto es mayor que el del bosque nativo solo. Así, estos autores efectivamente encontraron que, a mayor proporción de bosque nativo en cuencas con plantaciones, los caudales tendían a mantenerse a pesar de la disminución de las precipitaciones.

Finalmente, este capítulo abre la posibilidad de investigar, por ejemplo, el efecto de la cobertura en cuencas de mayor tamaño y también el posible efecto del cambio climático. Asimismo, existe la necesidad de incluir más métricas para analizar con mayor exactitud si existe o en que magnitud diferencias en la generación de caudales recesivos en Chile centro-sur entre diferentes usos de suelo y gradiente de precipitaciones.

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RESEARCH AND OBSERVATORY CATCHMENTS: THE LEGACY AND THE FUTURE

WILEY

Comparison of streamflow recession between plantations and native forests in small catchments in Central-Southern Chile

Francisco Balocchi[1,2] | Neftali Flores[3] | José Luis Arumí[2,4,5] | Andrés Iroumé[6] | Don A. White[7,8] | Richard P. Silberstein[8,9,10] | Pablo Ramírez de Arellano[1]

[1]Bioforest SA, Camino a Coronel s/n, km 15, Coronel, Chile
[2]Water Resources and Energy for Agriculture PhD Program, Water Resources Department, Universidad de Concepción, Chillán, Chile
[3]Facultad de Ciencias Forestales, Universidad de Concepción, Concepción, Chile
[4]Departamento de Recursos Hídricos, Facultad de Ingeniería Agrícola, Universidad de Concepción Chillán, Chillán, Chile
[5]CHRIAM Water Center ANID/FONDAP/15130015, Universidad de Concepción Chillán, Chillán, Chile
[6]Facultad de Ciencias Forestales, Universidad Austral de Chile, Valdivia, Chile
[7]Whitegum Forest and Natural Resources Pty. Ltd., Midland, Western Australia, Australia
[8]School of Science, Edith Cowan University, Joondalup, Western Australia, Australia
[9]Hydrological and Environmental Scientific Solutions Pty Ltd., Subiaco, Western Australia, Australia
[10]School of Agriculture and Environment, University of Western Australia, Nedlands, Western Australia, Australia

Correspondence
Francisco Balocchi, Bioforest SA, Camino a Coronel s/n, km 15, Coronel, Chile.
Email: francisco.balocchi@arauco.com

Abstract

In central Chile, many communities rely on water obtained from small catchments in the coastal mountains. Water security for these communities is most vulnerable during the summer dry season and, from 2010 to 2017, rainfall during the dry season was between 20% and 40% below the long-term average. The rate of decrease in stream flow after a rainfall event is a good measure of the risk of flow decreasing below a critical threshold. This risk of low flow can be quantified using a recession coefficient (α) that is the slope of an exponential decay function relating flow to time since rainfall. A mathematical model was used to estimate the recession coefficient (α) for 142 rainstorm events (64 in summer; 78 in winter) in eight monitored catchments between 2008 and 2017. These catchments all have a similar geology and extend from 35 to 39 degrees of latitude south in the coastal range of south-central Chile. A hierarchical cluster analysis was used to test for differences between the mean value of α for different regions and forest types in winter and summer. The value of α did not differ ($p < 0.05$) between catchments in winter. Some differences were observed during summer and these were attributed to morphological differences between catchments and, in the northernmost catchments, the effect of land cover (native forest and plantation). Moreover, α for catchments with native forest was similar to those with pine plantations, although there was no difference ($p < 0.05$) between these and *Eucalyptus* plantations. The recession constant is a well-established method for understanding the effect of climate and disturbance on low flows and baseflows and can enhance local and regional analyses of hydrological processes. Understanding the recession of flow after rainfall in small headwater catchments, especially during summer, is vital for water resources management in areas where the establishment of plantations has occurred in a drying climate.

KEYWORDS
Chilean native forest, forest catchment, mathematical modelling, Mediterranean climate, plantation

1 | INTRODUCTION

From 2010 to 2017 central Chile experienced an unprecedented, sustained, period of low rainfall (Garreaud et al., 2017) that resulted in persistent low flows in the streams of the coastal ranges (Carroll et al., 2019; Garreaud et al., 2020). During the drought, low rainfall coincided with heat wave conditions. This coincidence of dry and hot conditions is forecast to occur more often in the future (Comisión Nacional del Medio Ambiente - Departamento de Geofísica, 2006; Intergovernmental Panel on Climate Change, 2007). The coastal mountains of central Chile are dotted with small rural communities that co-exist with commercial plantations of *Pinus* and *Eucalyptus*. Many of these communities rely on small streams and wells for their water supply. Understanding and managing the effect of land use on water resources in the context of climatic trends is therefore crucial for developing strategies to ensure water security in the region. In this region, the climate is a Mediterranean type (Armesto et al., 2015) so that water supply is most vulnerable during summer. Summer is the time of year when stream flow is lowest and when the effect of changes in rainfall regime will be most likely to reduce flow to levels low enough to stress local supply. Developing approaches for ensuring water security should be based on an understanding of the impact that climate change and land management will have on summer flow regimes in this region.

Increased forest cover often reduces summer flow. In China, Yao et al. (2012) found that summer flow decreased after afforestation. Similarly, in a study of the effect of land cover on summer flow in Chile, Iroumé et al. (2005) observed an increase in summer flow in catchments with a lower proportion of perennial vegetation cover. Some studies have investigated the relative contribution to flow reduction of changes in land cover and climate change. In the south-west of Australia where the majority of formerly perennial streams have become ephemeral, reduced summer flow has been attributed to a combination of increased sapwood area index and climate change (Hughes et al., 2012; Petrone et al., 2010). A decrease in streamflow will affect communities that rely on surface water sources (Wittenberg, 1999). Studies in various parts of the world indicate that land use and climate change will act in concert to reduce streamflow, but attribution and the magnitude of these impacts have not yet been generalised.

Flow recession is the reduction in flow from the peak of a runoff event. The rate of this recession is indicative of the amount of water stored in the catchment and the capacity of this catchment to sustain flow. A rapid recession of flow or a high slope of the recession coefficient after rainfall indicates a smaller contribution from baseflow and a more vulnerable system in which water leaves the catchment rapidly (Kirchner, 2009; Pizarro, 1993). As well as providing information that can and has been used to interpret storage and aquifer characteristic (Clark et al., 2009; Sayama et al., 2011) and hydrogeology (Kumar & Sen, 2018), the shape of the recession curve can be used to develop models to estimate summer flows and to improve understanding of the aquifer configuration which governs the outflow derived from groundwater and hence the baseflow (Frohlich et al., 1994; Jakada et al., 2019). It is therefore important to understand how recession flows are being modified by climate change and land use change to generate water management strategies.

Hydrological and mathematical modelling can be important tools for water resources management, as they can help us to understand how the water cycle responses to the main drivers of climate, geology and land-use (Refsgaard & Abbott, 1996). Moreover, mathematical modelling can be used to study complex systems where some processes are hard to evaluate (Sujono et al., 2004). Estrela (1992) noted that mathematical models can be applied to solve theoretical hydrologic problems and to test our understanding of systems. One approach for quantifying the change in summer flow is to model the effect of change on the slope of the recession curve or the rate of change in flow after a rainfall event. The slope of these relationships or recession coefficient (α) is influenced by geology, geomorphology, soil and climatic factors (Van Dijk, 2010), as well as land cover, human activity (Price, 2011) and drainage network morphology (Biswal & Nagesh Kumar, 2014). Brandes et al. (2005) observed that in small catchments (<130 km^2) the geomorphology, geology and soil characteristics determined the rate of recession in flow rates, with the main factors being drainage density, slope, bedrock and infiltration rate. Peña-Arancibia et al. (2010) found that the mean catchment slope and elongation are geomorphological factors that are correlated with the recession coefficients in small catchments, while the main climatological factor is mean annual precipitation. The recession coefficient (α) is known to be dynamic and depends on the wetness state of the catchment (Biswal & Nagesh Kumar, 2014; Shaw et al., 2013). Therefore, the prediction of the flow recession relies on the individual analysis of α, storm by storm (Reddyvaraprasad et al., 2019).

While quantitative approaches, including the analysis of temporal variation in α, show promise for quantifying the relative effect of different drivers on summer flow (Balocchi et al., 2014; Cirugeda, 1985; Parra et al., 2019; Pizarro et al., 2013; Stoelzle et al., 2015), studies of baseflow recession coefficient in small catchments (Yang et al., 2018), where baseflow is a large proportion of the total streamflow (Hewlett & Hibbert, 1967) are rare. As far as we are aware, no studies have yet compared the effect of forest cover and type on the streamflow recession in the Mediterranean region of south-central Chile.

During the longest drought experienced to date in central Chile, we analysed the recession coefficient for winter and summer storms in eight forested catchments in south-central Chile in order to identify differences between them. These catchments have different land covers and climate but the same metamorphic geology. Their land covers are the main ones found in this part of Chile, namely, native forest and plantation of Monterrey Pine (*Pinus radiata*) and *Eucalyptus* species (*E. globulus*, *E. nitens* and a hybrid). These latter plantation types are also important production forest types in many other parts of the world. The catchments are on a N-S transect and precipitation gradient within the Chilean coastal range. The most Northern catchments are in the semi-arid coastal zone of central Chile (Mean Annual Rainfall 900 mm), the north-central catchments are in the humid coastal zone (1200–1800 mm), the south-central catchments in the cool high rainfall zone near Valdivia in southern Chile (2000 mm).

Based on the presumption that the predominant influence on stream flow recession will be due to the geology and geomorphology, we would expect little or no difference in recession coefficients between the catchments in our dataset. Thus, in this article we test two hypotheses:

i. That flow recession will not be affected by annual rainfall or by differences in rainfall between summer and winter, and
ii. That flow recession in streams will not have a dependence on the type of forest cover in the catchments.

To test for the effect of vegetation we used hierarchical cluster analysis on streamflow recession from catchments with native forest, and plantations of *P. radiata* and *Eucalyptus*, and to test the response to rainfall gradient we analysed stream flow recessions in catchments across a wide range of rainfall domains and in winter and summer. In undertaking this analysis, we also tested the efficacy of three simplified analytical models for catchment flow recession to determine if there were significant differences, and whether a single model was to be preferred for the analysis of these catchments.

2 | METHODS

2.1 | Study sites

2.1.1 | Location, geology, soils and land cover

We studied eight coastal catchments in south-central Chile extending from just North of Constitución (35°S) to Valdivia (40°S) (see Figure 1 for a locality map). These eight catchments are in three regions and five locations and all are on land owned by Forestal Arauco S.A., a global manufacturer of forest products. Two are in the Maule Region, north-east of the city of Constitución (35.20°S) and are called Quivolgo 2 (Q2) and Quivolgo 3 (Q3). Four of our catchments are located in the Biobío region. Two of these are located to the south-east of Laraquete (37.24°S) and are known as Maria Las Cruces 1 (MLC1) and Maria Las Cruces 2 (MLC2). In the same Biobío region, two more are located about 50 km further south and about 10 km east of the town of Curanilahue (37.49°S). These catchments are called Bajo las Quemas 1 (BLQ1) and Bajo las Quemas 2 (BLQ2). The remaining two catchments are in the Los Ríos Region, one (San Gabriel, SG) is located around 15 km southeast of Valdivia city (39.91°S) and the second (Entrerios, ER) is further east near Máfil (39.63°S). All these catchments are rain-fed.

These parts of the coastal-range are underlain by a metamorphic rock complex (Mordojovich, 1974) ranging in age from the Silurian–Carboniferous in the northern and central area to Palaeozoic–Triassic in the southern part of the study site (Servicio Nacional de Geología y Minería, 2003). The soils that are derived from this bedrock have clay loam (Q2, Q3, MLC2, BLQ1 and BLQ2), clay (MLC1) or loam (SG and ER) textures (Table 1). The soils in Q2, Q3, MLC1 and MLC2 are all less than 1 m deep with a high percentage of stone while soils at BLQ1, BLQ2, SG and ER are generally deeper than 1 m.

The forest covers of the eight catchments are either native forest, *Pinus* or *Eucalyptus* plantations. Quivolgo 2 has a native forest cover dominated by *Nothofagus glauca* (Phil) Krasser (Balocchi et al., 2020). Quivolgo 3 is 61% *P. radiata* (D.Don), predominantly in the upper parts of the catchment and 25% covered by a similar native forest to Q2, and the remainder a predominantly evergreen forest mix in the lower gullies. Maria Las Cruces 1 has native forest dominated by *Nothofagus obliqua* (Mirb.) Oerst. Maria Las Cruces 2 is 74% covered by *P. radiata* planted in 1994 with the balance a mix of other land uses. Bajo Las Quemas 1 and 2 were planted with *E. nitens* (Deane and Maiden) in 2011. These plantations cover 88% of BLQ 1 and 80% of BLQ2. San Gabriel was planted to *P. radiata* in 1990 and ER is occupied by a Valdivian temperate rainforest. The areas of the eight catchments are between 13 and 112 ha with mean slopes between 21% and 51%. Other catchment attributes such as mean altitude, outlet location and so forth, can be found in Table 1.

2.1.2 | Climate

All catchments have a summer dry season between October and April and a rainy season between late Autumn and early Spring (May to September, Figure 2). From 2000 to 2016 the mean annual precipitation at Quivolgo was 1000 mm, 90% of which occurred between May and September (winter season). During the same period the mean annual precipitation was 1200 mm at MLC (80% in the winter) and 1800 mm at BLQ (80% in winter). While rainfall was also concentrated in winter at the southern catchments in San Gabriel and Entrerios, an average of more than 50 mm fell in each month of the year. The average annual rainfall between 2000 and 2016 was 1700 mm at Entre Rios and 1950 mm at San Gabriel, 70%–75% of which fell in the winter months. Thus, there is a general trend of increasing rainfall and decreasing winter dominance of rainfall from north to south in this region. From north to south the average maximum temperatures in the hottest month (January), decreased from 26°C at Quivolgo in the north to 24°C at ER and SG in the south while the average minimum in the coldest month (July) temperatures increased from 1.3°C in Quivolgo, to over 3°C in ER and SG in the south (Figure 2).

2.2 | Streamflow and rainfall measurements

A 90° V-notch weir was installed at the outlet of each catchment. The height of the water in the weir was measured with a pressure transducer flow gauge (KPSI 550, Pressure Systems Inc., Virginia, USA, and OTT Orpheus Mini, Kempten, Germany, more details in Balocchi et al., 2021) and the streamflow was estimated every 5 min from the theoretical rating curve. Q2 has streamflow records since 2009, Q3 since 2013, BLQ1 and 2 since 2013, MLC1 and MLC2 since 2008, SG since 2008 and ER since 2014 (Figure 1). The Quivolgo area was

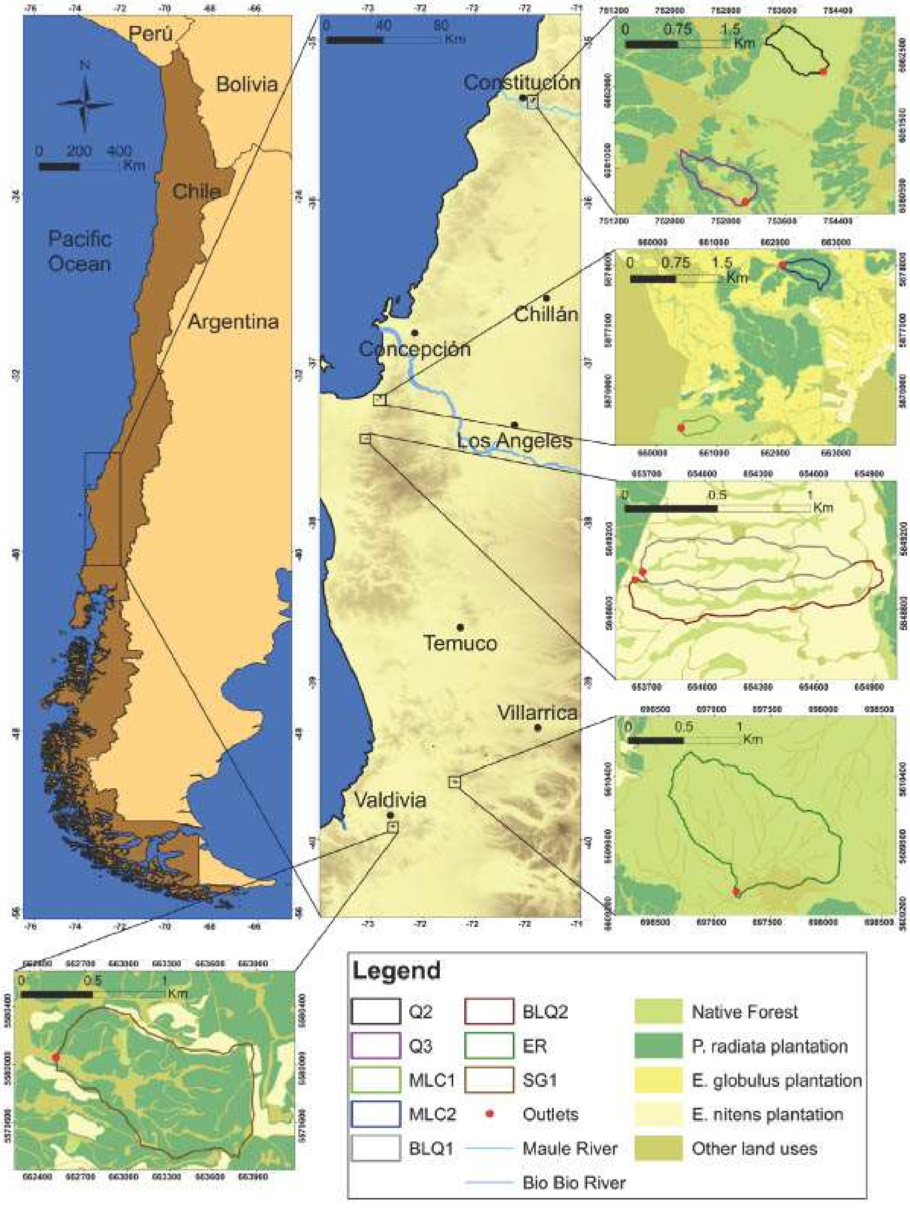

FIGURE 1 Geographic location of the eight catchments in Chile, South America and in relation to cities in central Chile. Also included are the outline and land use of the individual catchments

severely affected by the megafires of summer 2017 (Balocchi et al., 2020; White et al., 2020) and, therefore, in Q2 and Q3 only streamflow recorded before 2016 was considered.

In the Quivolgo area, rainfall was measured at the top of the Q2 catchment as described in Balocchi et al. (2020) study. For the other catchments we used rainfall data from the nearest DGA (Directorate

TABLE 1 Descriptive characteristics of the eight catchments including annual rainfall, location, catchment morphology and soil texture classes

	Quivolgo 2 (Q2)	Quivolgo 3 (Q3)	María Las Cruces 1 (MLC1)	María Las Cruces 2 (MLC2)	Bajo las Quemas 1 (BLQ1)	Bajo las Quemas 2 (BLQ2)	San Gabriel (SG)	Entre Ríos (ER)
Land cover	Native forest	*Pinus radiata*/ N. forest	Native Forest	*P. radiata*	*Eucalyptus nitens*	*E. nitens*	*P. radiata*	Native forest
Annual precipitation (mm)	966	966	1200	1200	1800	1800	1950	1900
Period of max rainfall		May–September						
Area (km^2)	0.33	0.4	0.13	0.18	0.23	0.22	0.89	1.12
Mean Altitude (m a.s.l)	394.9	442.3	250.26	317.1	527.6	530	207	400.7
Mean slope (%)	51.6	44.8	30.04	43	40	40.4	21.65	41.8
Catchment perimeter (km)	2.54	3.1	1.61	2.59	3.04	3.13	4.21	4.79
Mean slope of stream (%)	35.1	34.7	25.7	35.2	27.5	22.9	31.9	25.1
Length of main water course (km)	1.1	1.33	0.8	1.02	1.25	1.3	1.59	1.48
Time of concentration (h)[a]	0.12	0.15	0.06	0.12	0.14	0.15	0.19	0.17
Elongation ratio	0.74	0.6	0.64	0.63	0.47	0.39	0.73	0.9
Outlet latitude (°S)	35.35692	35.37396	37.23973	37.21539	37.47853	37.47854	39.89811	39.62698
Outlet longitude (° O)	72.20312	72.21479	73.19153	73.17342	73.26213	73.2626	73.0991	72.70242
Type of climate	Temperate semi-oceanic						Rainy temperate	
Average clay content (%)	35	33.3	46.3	37	36.2		19.1	16.2
Average silt content (%)	37	37.4	24.9	34.8	34.6		39.5	39.3
Average sand content (%)	28	29.3	28.8	28.2	29.2		41.3	44.5
Main texture class (USDA)	Clay Loam		Clay	Clay Loam			Loam	

[a]Time of concentration was calculated using the California Culvert Practice method (Li & Chibber, 2008).

General de Agua) meteorological stations. The data for MLC was from Rio Carampangue (~8 km southwest of the catchments). Rainfall for BLQ was measured at Rio Curanilahue about 6 km to the southwest of the catchments. For the SG catchment, the rainfall was measured at Llancahue which is around 9 km to the west of the catchment. For ER there is no nearby DGA station, so the CR2MET (Center for Climate and Resilience Research Meteorological dataset) rainfall product (5-km grid for continental Chile) was used (more details in Balocchi et al., 2021).

2.3 | Recession flow models

Exponential decay models are commonly used to describe the decrease in flow after the peak associated with a rainfall event (Balocchi et al., 2014; Fiorillo et al., 2012; Jakada et al., 2019). We used three mathematical models to describe the shape of the relationship between Q and t (Equations 1–3). The first exponential recession equation (Equation 1, model M1) was originally formulated by Boussinesq (1877) and used by Maillet (1905) as:

$$M1: Q(t) = Q_0 * e^{(-\alpha t)}, \quad (1)$$

where Q_0 is the initial flow, $Q(t)$ is flow at time t after the peak, and α is the exponential decay constant which is also called the recession coefficient or recession constant. In our study we have limited, or no data on the aquifer configuration and have therefore assumed a linear reservoir configuration (Peters et al., 2005) as is represented by Equation (1). Equation (2) is a potential function developed by Boussinesq (1903, 1904) (M2), and Equation (3) is a variation of the exponential model developed by Pizarro et al. (2013) (M3).

$$M2: Q(t) = Q_0 * (1 + \alpha t)^{-2}, \quad (2)$$

$$M3: Q(t) = Q_0 * e^{(-3\alpha^2 t)}. \quad (3)$$

Models M1 and M2 are solutions of the diffusion equation, however, they differ mainly in their considerations and simplifications. Model M1 is an approximate analytic solution of the diffusion equation and consists in the linearized flow version assuming Dupuit–Boussinesq aquifer model but with no dependence on vertical flow and the capillarity effect above the water table is negligible. M2 model assumes a Dupuit–Boussinesq aquifer model, that is groundwater flow moves shallower in the subsoil and it follows hydrostatic pressure (Wu et al., 2018), 'and the simplifying assumptions of a porous, free, homogeneous, isotropic, aquifer with no capillary effect, that is limited by an impermeable horizontal layer at the level of the outlet that is considered to be localized' (Boussinesq, 1877) to give an exact solution of the diffusion equation (Dewandel et al., 2003). Finally, model

M3 is a modification of M1 presented by Pizarro et al. (2013) which implies a greater participation of the recession coefficient at the time of estimating Q(t), thus yielding α values significantly higher than those obtained by the remaining models.

2.4 | Recession flow estimation

Using the rainfall data described above, rain events were identified in both dry and wet seasons for each of the eight catchments. An analysis of the streamflow data after rainfall indicated that flow receded over more than 5 days in summer and about 3 days in winter. Suitable events used in the recession analysis were selected according the following criteria: peak flows are clear and easy to identify in the hydrograph; the recession curves have durations of at least 5 days in summer and 3 days in winter; and, during the recession period, there are no rain events that affect the natural flow decay.

There are several methods to define the recession onset such as those by Chapman (1999) and Vogel and Kroll (1996) baseflow estimators. The most widely used method for the identification of recession onset was developed by Linsley et al. (1949) in which the recession curve of the hydrograph is plotted as the log of Q against time and the second break point (i.e., the second change in the logarithmic flow recession slope) after peak flow is taken to indicate the point after which groundwater makes up all of the flow (an example of second break points in the recession curve is presented as point D in Figure 3(a)) (Bedient et al., 2008). Break points of the recession indicate a change in slope or rate of the recession flow. Some authors also refer to this break point as indicating different micro-regimes of the aquifer discharge into the stream (Kresic, 2007). In this study, for each rain event between 2008 and 2017, the beginning of the recession flow curve (Q_0) was identified, using a modification of the method of Bedient et al. (2008) proposed by Pizarro et al. (2013), which allows more accurate estimations of the baseflow recession behaviour than the original method and has been previously tested in a watershed in south-central Chile (Balocchi et al., 2014). The modified method uses the third break point of the recession curve (point E in Figure 3(b)) to redefine the Q_0 flow and $t = 0$ at this point.

Typically, flow records of 4–10 days are required for the calculation of the recession coefficient when analysing the recession curve at a daily event-scale (Biswal & Marani, 2010; Howe, 1966; Shaw & Riha, 2012). Therefore, we have fitted each model (Equation 1–3) to the endpoints of a succession of time intervals (24 h increment) such as 0–24 h ($t = 24$), 0–48 h ($t = 48$), 0–72 h ($t = 72$) and so forth until 0–240 h ($t = 240$). The results from all flow events were averaged for each time interval to give an overall estimate of the α value at the difference stages of the flow recession. Summer and winter events were analysed separately using the same approach to provide a basis for comparison and test for any change in trends of the α coefficient through time per seasons and catchments.

2.5 | Model performance and data analysis

2.5.1 | Model performance

The mathematical modelling efficiency was evaluated using the Nash–Sutcliffe efficiency (NSE; Equation (4) (Nash & Sutcliffe, 1970) and the standard error of the estimate (SEE) (Romero & Casimiro, 2015). This evaluation was repeated for the different models for each combination of catchment and season.

The NSE criterion (Equation (4) determines the relative magnitude of residual variance with respect to the variance of the observed data. It takes values between $-\infty$ and 1, with values greater than 0 indicating that the model gives a better estimate than a simple mean. In general, values over 0 are accepted, with values over 0.5 indicating a satisfactory model efficiency, values over 0.65 indicating a good efficiency and values over 0.75 indicating a very good efficiency (Moriasi et al., 2007).

$$\mathrm{NSE} = 1 - \left[\frac{\sum_{i=1}^{n}\left(\hat{Q}_i - Q_i\right)^2}{\sum_{i=1}^{n}\left(\bar{Q} - Q_i\right)^2}\right], \tag{4}$$

where:

$\hat{Q}_i$ = ith simulated streamflow (mm)

Q_i = ith observed streamflow (mm)

$\bar{Q}$ = Average of observed streamflows (mm)

n = Sample size.

The SEE (Equation 5) determines the root mean square difference between observed and estimated streamflows, with the optimum result being a value near 0.

$$\mathrm{SEE} = \sqrt{\frac{\sum_{i=1}^{n}\left(\hat{Q}_i - Q_i\right)^2}{n-2}}. \tag{5}$$

2.5.2 | Recession coefficient analysis

The coastal mountains are currently affected by a period of low rainfall known locally as the megadrought (Carroll et al., 2019; Garreaud et al., 2017) so that our analysis was for a dry period rather than a normal or wet year. Recession analysis was done separately in winter, which includes rainstorms between May and September, and summer, which is the period between October and April. To test for significant differences (p-value < 0.05) among catchments and seasons, the Scott and Knott hierarchical test (Scott & Knott, 1974) was used with support of the Scott and Knott R package (Jelihovschi et al., 2014). In addition, we have explored some morphometric parameters such as drainage density, catchment mean slope and soil depth of each catchment, and related them to the recession coefficient behaviour in both winter and summer seasons using information and similar relationships from the literature. As noted in the introduction, the catchments are in four different rainfall zones. The North region included Q2 and Q3, the north-central region MLC1, MLC2, BLQ1 and BLQ2, the south-central included ER while SG was in the South region. Given the similarity in

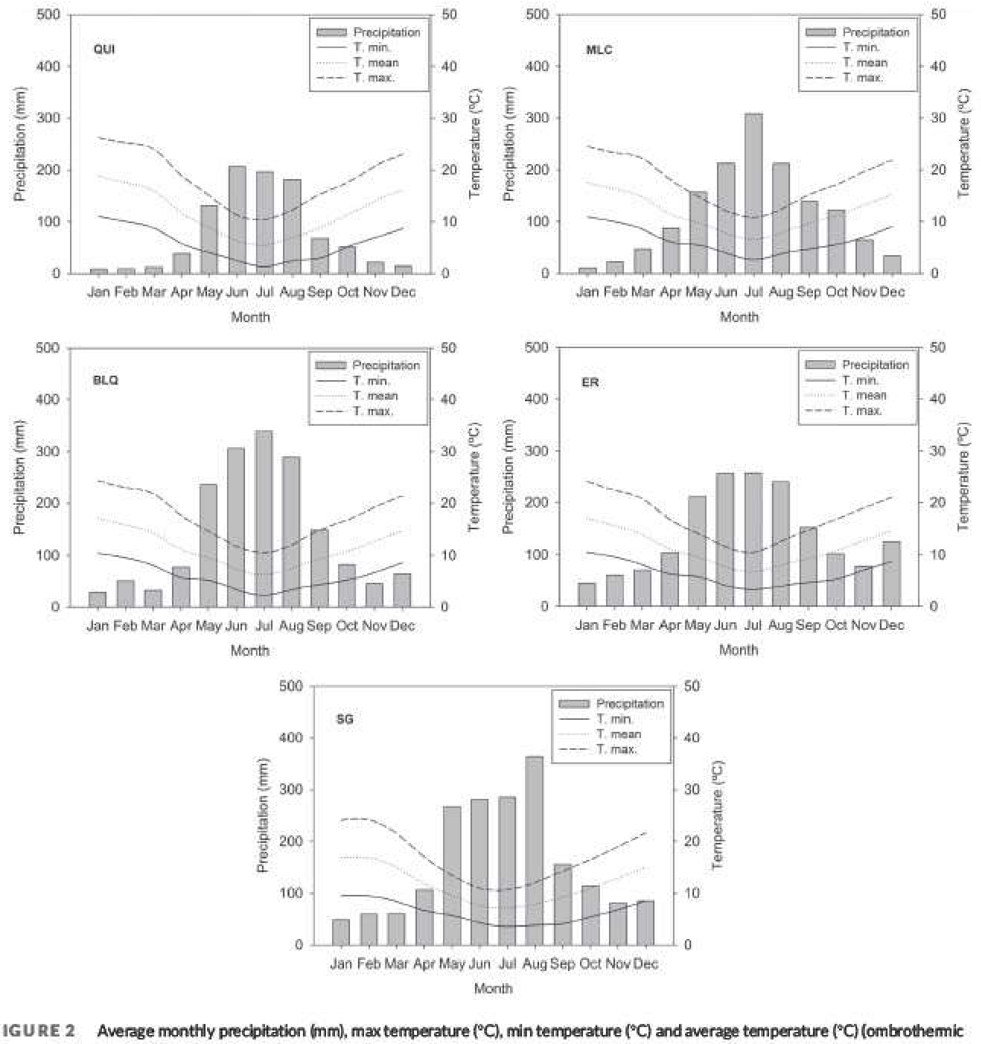

FIGURE 2 Average monthly precipitation (mm), max temperature (°C), min temperature (°C) and average temperature (°C) (ombrothermic diagrams) between 2010 and 2016 at five locations within the study sites

the geology of these regions, any test for the effect of region on the recession coefficient is therefore a test of the effect of annual rainfall.

3 | RESULTS

3.1 | Comparative performance of the mathematical models

The data covers the period from 2008 and 2018 in some catchments; MLC2 and SG have the longest streamflow record (10 years) and Q3 the shortest (3 years) (Balocchi et al., 2021). All the models were more efficient predictors of flow in winter than summer. In the wet months, M2 was more efficient predictor of flows (Figures 4(d) and 5(d)) in the first 5 days (120 h) after Q_0, while M1 and M3 were more efficient predictors of flows after the fifth day (Figures 4(a,f) and 6(a,f)). Similarly, in the dry season M2 tended to be more efficient in the first days after Q_0 (Figures 4(c) and 5(c)), and M1 and M3 were more efficient in the latter days (Figures 4(a,e) and 5(a,e)). The exponential decay model (M1) and the combination model (M3) were more efficient predictors of flow than the potential function (M2) and had the same efficiency even though α was scaled three times in M3. While

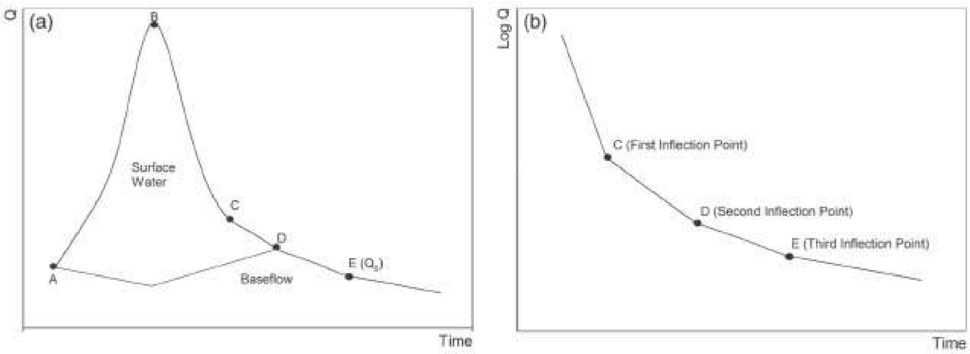

FIGURE 3 A typical storm hydrograph. Point A indicates the point of runoff initiation, point B indicates peak flow and point C is the time at which direct runoff ends. Point E corresponds to the point at which the model is initiated (Q_0) or $t = 0$ (*Source:* Modified from Balocchi et al. (2014))

for M2 and M3 model efficiency was equal, α values were not. For the analysis of the effects of season, catchment region and land use, M1 was used to estimate α as it had good results after 5 days which represented later stages of the recession. M1 also reduced bias and variability if recession length increased (Vogel & Kroll, 1996).

In the summer months, when using periods of $t \leq 120$ h, model M2 reached the highest efficiency for catchments Q2, Q3, BLQ1 and ER, for $t \leq 144$ h M2 had the highest efficiency for catchments MLC2 and SG, and for $t \leq 72$ h M2 had the highest efficiency for MLC1. In the later periods in all catchments, M1 and M3 had the highest efficiency. In addition, M2 has greater NSE values than M1 and M3 for MLC2 at $t \leq 48$ h, for MLC1, BLQ1 and BLQ2 at $t \leq 72$ h and for Q2, Q3, ER and SG at $t \leq 96$ h, while models M1 and M3 do so for all catchments in the latter periods. Meanwhile, SEE is significantly greater in ER and SG with all three models ($p < 0.05$). When using $t \geq 144$ h, the catchments that allow significantly greater NSE values to be obtained are MLC1 (NSE > 0.8) and Q2, MLC2 and ER (NSE > 0.5) with models M1 and M3 in the dry season.

When using $t \geq 144$ h in the wet season, catchments MLC1, MLC2, BLQ1, BLQ2, ER and SG reach significantly higher NSE values than those obtained in Q2 and Q3 by models M2, M1 and M3 ($p < 0.05$). Meanwhile EEE is significantly greater in catchments MLC1 and SG with all models ($p < 0.05$).

3.2 | Recession coefficient

Sixty-four suitable events were identified in summer (10 in Q2, 3 in Q3, 10 in BLQ1, 10 in BLQ2, 3 in MLC1, 8 in MLC2, 10 in SG and 10 in ER) and 78 in winter (10 in Q2, 10 in Q3, 9 in MLC1, 9 in MLC2, 10 in BLQ1, 10 in BLQ2, 10 in SG and 10 in ER). These events had minimum Q_0 flows above 0.0044 m^3/s in winter and above 0.0019 m^3/s in summer in all catchments. Quivolgo 3 (Q3) and MLC1 had the lowest streamflow values of all the studied catchments which limited the number of significant rainstorms observed in the catchment during summer months. Table 2 (summer) and Table 3 (winter) summarize α values per catchment and time intervals (mean values and SD). In the summer months α did not change with time after an event for any of Q2, Q3 or BLQ1 (p-value < 0.05) while in BLQ2, ER and SG the α values obtained for $t = 24$ and 48 h were significantly greater than those obtained for the later periods (p-value < 0.05). The values of α in MLC2 were significantly higher at $t \leq 120$ h than for later intervals, and in MLC1 α did not differ significantly between periods.

In the winter months no significant differences were observed between α values for the different periods in catchments Q3, MLC1 and ER (p-value < 0.05). In Q2, MLC2 and BLQ2 α values were significantly higher for $t \leq 72$ h than in any other period. At BLQ1 the values for $t \leq 48$ h were higher than later values and at SG the threshold for significant change in α was at $t \leq 96$ h (p-value < 0.05).

In the summer months, catchments BLQ2, SG and ER had values of α that were significantly greater than observed in the other catchments ($p < 0.05$) at $t \leq 48$ h, while in the later periods, BLQ2, ER, SG and MLC1 had values of α that were significantly greater than observed in the other catchments ($p < 0.05$). The highest values of α are observed in BLQ2 at $t \leq 168$ h and later in MLC1. In the winter months no significant differences between catchments were observed in most of the intervals analysed; at $t = 240$ h α was significantly higher in MLC1, BLQ1 and MLC2 (p-value < 0.05) than elsewhere. The highest coefficients were observed in Q3 at $t = 24$ h, MLC2 at $t = 48$ h and MLC1 at $t \geq 72$ h (Figure 6).

The α values tended to be higher in the winter months than in summer for all the periods in catchments Q2, MLC2 and BLQ1 ($p < 0.05$), with higher values in the first days after Q_0 and subsequently decreasing as the period increases. In BLQ2, α was significantly higher at $t \geq 96$ h and in MLC1, at $t \geq 192$ h. In Q3 it was also observed that the coefficients for winter events tended to be higher than those of summer for all the periods. However, the smaller sample size of the summer events (n = 3) limits the significance of this result.

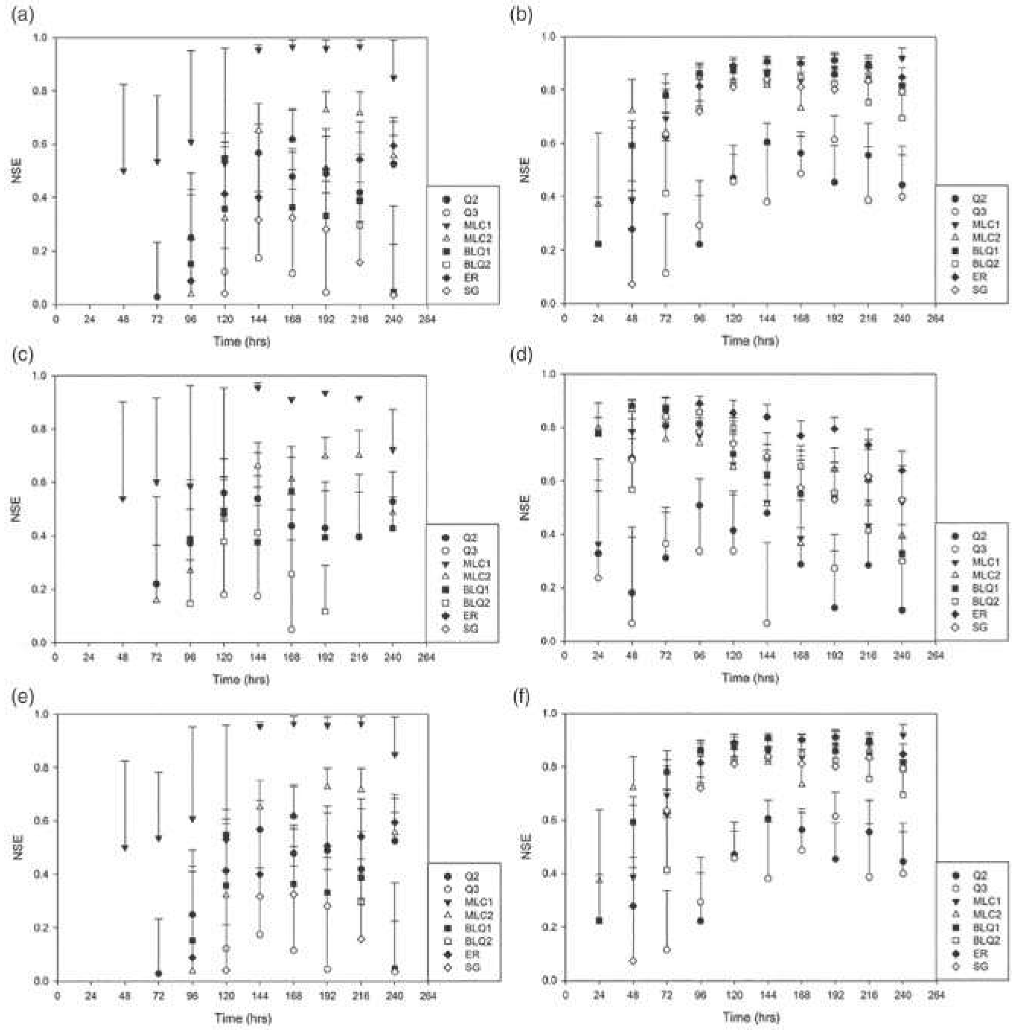

FIGURE 4 The mean and standard error of the Nash–Sutcliffe efficiency (NSE) for (a) M1 summer, (b) M1 winter, (c) M2 summer, (d) M2 winter, (e) M3 summer and (f) M3 winter. For clarity, NSE varies between 0 and 1. Catchments in the figure legend are arranged in order from the northerly catchments (Q2–Q3) to those on the south (ER-SG). In most of the catchments NSE increases as the time interval also increases. In the summer months (October to April) MLC1 and MLC2 (inverted black and white triangle, respectively) reached the highest NSE values. In winter months (may to September) highest NSE values are reached by catchments of the north-central, central and southern zones

In ER, α was significantly higher in winter than in summer only in periods longer than 192 h ($p < 0.05$). Within SG α was significantly greater in the winter months than in summer for periods longer than 76 h (p-value < 0.05).

In summary, winter values of α were about three times the values in summer (mean values 0.039 and 0.012, respectively).

In Figure 7, α values for each time step from all catchments are averaged to illustrate the predominant seasonal behaviour, which clearly shows the differences which were significant ($p < 0.05$). In Figure 7(b,c), the catchments in each regional zone are averaged to determine if there are regional differences in addition to the seasonal differences. While the figure shows that in the summer dry

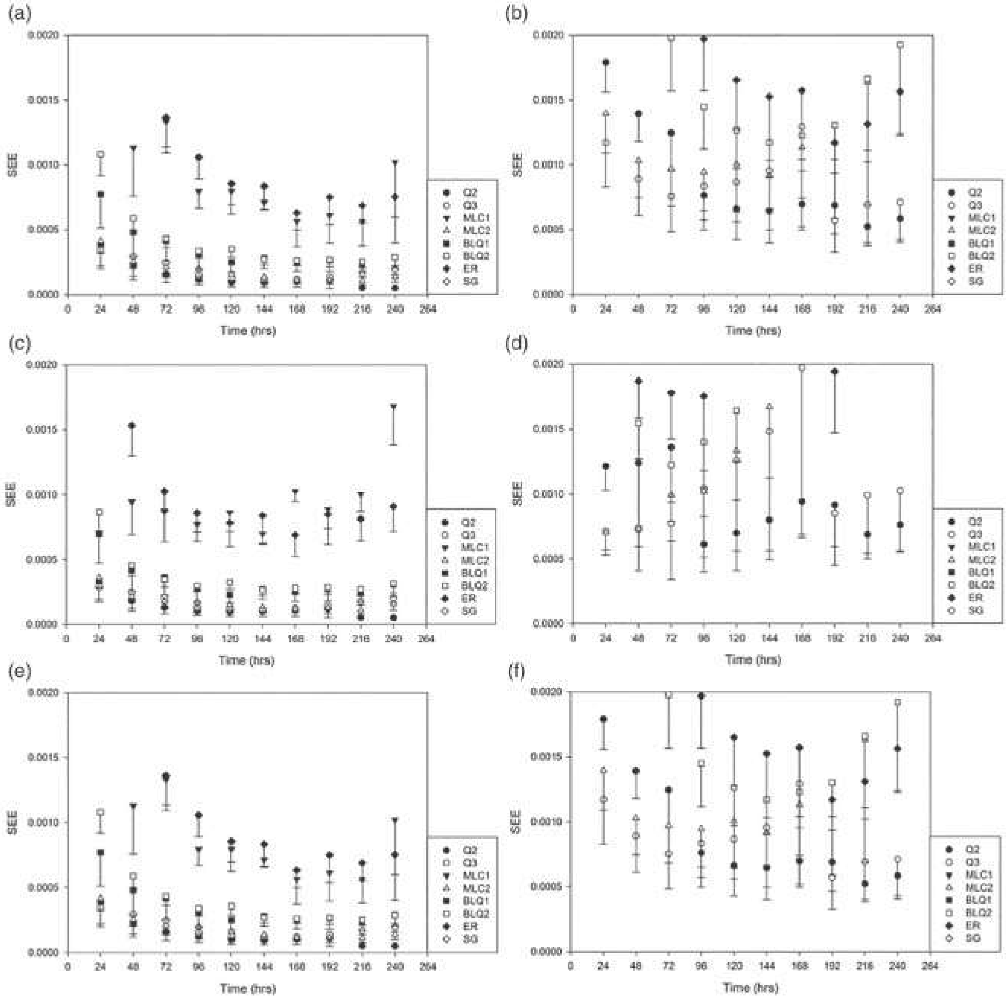

FIGURE 5 The mean and standard error of the standard estimation error (SEE) for (a) M1 summer, (b) M1 winter, (c) M2 summer, (d) M2 winter, (e) M3 summer, and (f) M3 winter. For clarity, SEE varies between 0.002 and 0. Catchments in the figure legend are arranged in order from the northerly catchments (Q2–Q3) to those on the south (ER-SG). It should be noted that most of the catchments reached lower SEE values in summer months than in winter months, being the northern catchments (i.e., Q2–Q3) those that reached the lowest values in SEE (see black and white circles, respectively)

season (Figure 7(b)) the more southern (higher rainfall) catchments have the higher α and in winter (Figure 7(c)) the northern (drier) catchments have the higher α, these differences were not significant (p-value < 0.05).

In Figure 8, we have averaged α values within each region per season separated by region (Figure 8(a)), and for winter and summer in the northern region (Figure 8(b) which includes North and north-central catchments), and the southern region (Figure 8(c) which includes south and south-central catchments). In Figure 8(a), we did not find differences between catchments in all seasons (p-value < 0.05). In Figure 8(b), even though we have found a graphical difference these were not significant (p-value < 0.05).

Finally, in Figure 9, we have combined all data for each vegetation type. That is all Pinus, native forest and *Eucalyptus* land cover

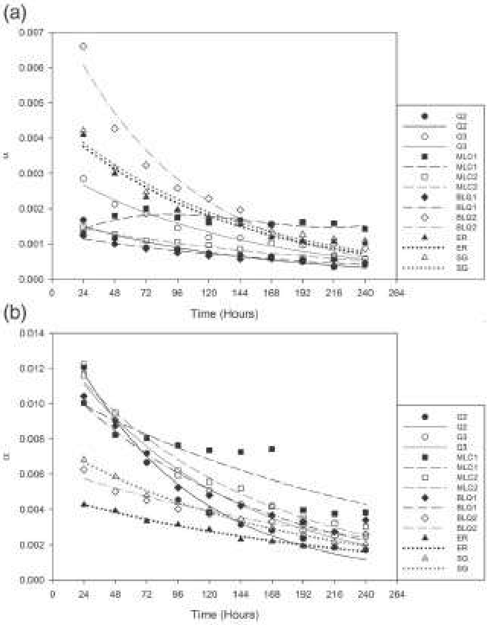

FIGURE 6 The relationship between the recession coefficient (α) and time after an event for the eight catchments in (a) summer and (b) winter. The symbols represent the average of α for combinations of catchment, season and time interval. The lines show the trend in α for different catchments and seasons

together. Our results showed that catchments with *Eucalyptus* had lower α than the other two forest types although this difference was not significant (p-value < 0.05).

4 | DISCUSSION

We tested for the effect of time since rainfall and land use on the rate of decline in flow after the rainfall event. We found that rate of decline was not different between the catchments in the winter, but some significant differences were evident in the summer when the drier catchments (i.e., Q2 and Q3) exhibited a more rapid decline (greater α) than the higher rainfall areas in the south. Therefore, our hypothesis (i) That flow recession will not depend on rainfall gradient among catchments and within winter dry season and winter wet season was supported in winter but was rejected in summer when rainfall did influence the recession coefficient. Regarding our hypothesis (ii) That flow recession in streams will not depend on the type of forest cover in the catchments, we did not find differences in winter but there were differences in summer and especially within regions. Our catchments have the same underlying geology, but they differ in land cover and rainfall and the latter seems to be the main determinant of the rate of recession. The conclusion that rainfall is a more important determinant of flow recession than land cover is potentially important given the political sensitivity of land use change in the region. Given this sensitivity it will be important to further corroborate these results in these and other catchments.

TABLE 2 Mean and standard deviation (SD) values for α values at each time interval from events in the summer season, as determined by model M1

		Time interval (h)									
Watershed	α	24	48	72	96	120	144	168	192	216	240
Q2	Mean	0.0017	0.0012	0.0009	0.0008	0.0007	0.0007	0.0006	0.0005	0.0004	0.0004
	SD	0.0016	0.0013	0.0009	0.0007	0.0006	0.0005	0.0005	0.0004	0.0002	0.0002
Q3	Mean	0.0028	0.0021	0.0019	0.0014	0.0012	0.0012	0.0010	0.0010	0.0007	0.0006
	SD	0.0019	0.0014	0.0009	0.0005	0.0003	0.0005	0.0003	0.0004	0.0001	0.0000
MLC1	Mean	0.0013	0.0018	0.0020	0.0017	0.0016	0.0017	0.0015	0.0016	0.0016	0.0014
	SD	0.0007	0.0004	0.0006	0.0006	0.0005	0.0008	0.0007	0.0007	0.0007	0.0009
MLC2	Mean	0.0015	0.0013	0.0011	0.0010	0.0010	0.0008	0.0006	0.0006	0.0006	0.0006
	SD	0.0009	0.0009	0.0006	0.0006	0.0006	0.0005	0.0002	0.0001	0.0002	0.0002
BLQ1	Mean	0.0012	0.0010	0.0009	0.0007	0.0007	0.0006	0.0006	0.0005	0.0005	0.0005
	SD	0.0017	0.0010	0.0008	0.0006	0.0005	0.0005	0.0004	0.0003	0.0003	0.0002
BLQ2	Mean	0.0066	0.0043	0.0032	0.0026	0.0023	0.0020	0.0016	0.0011	0.0010	0.0009
	SD	0.0049	0.0033	0.0022	0.0017	0.0013	0.0012	0.0013	0.0004	0.0002	0.0001
ER	Mean	0.0041	0.0030	0.0023	0.0020	0.0017	0.0015	0.0012	0.0011	0.0011	0.0010
	SD	0.0032	0.0019	0.0014	0.0011	0.0009	0.0007	0.0004	0.0004	0.0003	0.0004
SG	Mean	0.0042	0.0032	0.0025	0.0020	0.0018	0.0016	0.0014	0.0013	0.0012	0.0011
	SD	0.0020	0.0017	0.0013	0.0009	0.0008	0.0007	0.0006	0.0005	0.0004	0.0004

TABLE 3 Mean and standard deviation (SD) values for α values at each time interval from events in the winter season, as determined by model M1

		Time interval (h)									
Watershed	α	24	48	72	96	120	144	168	192	216	240
Q2	Mean	0.0120	0.0091	0.0072	0.0046	0.0038	0.0032	0.0028	0.0024	0.0018	0.0017
	SD	0.0118	0.0082	0.0061	0.0019	0.0016	0.0013	0.0011	0.0009	0.0003	0.0003
Q3	Mean	0.0122	0.0083	0.0067	0.0062	0.0051	0.0043	0.0042	0.0028	0.0025	0.0025
	SD	0.0136	0.0073	0.0060	0.0058	0.0054	0.0043	0.0035	0.0016	0.0015	0.0014
MLC1	Mean	0.0100	0.0082	0.0081	0.0077	0.0074	0.0073	0.0074	0.0040	0.0038	0.0038
	SD	0.0085	0.0077	0.0087	0.0097	0.0099	0.0104	0.0106	0.0011	0.0010	0.0011
MLC2	Mean	0.0116	0.0095	0.0081	0.0059	0.0056	0.0052	0.0042	0.0035	0.0032	0.0030
	SD	0.0070	0.0048	0.0034	0.0021	0.0022	0.0023	0.0017	0.0020	0.0018	0.0016
BLQ1	Mean	0.0104	0.0087	0.0067	0.0053	0.0048	0.0042	0.0037	0.0033	0.0027	0.0034
	SD	0.0058	0.0045	0.0031	0.0020	0.0017	0.0015	0.0014	0.0014	0.0016	0.0008
BLQ2	Mean	0.0063	0.0050	0.0045	0.0040	0.0037	0.0034	0.0033	0.0032	0.0027	0.0026
	SD	0.0017	0.0010	0.0009	0.0009	0.0009	0.0009	0.0009	0.0009	0.0006	0.0006
ER	Mean	0.0043	0.0039	0.0034	0.0032	0.0029	0.0023	0.0022	0.0019	0.0018	0.0017
	SD	0.0031	0.0027	0.0024	0.0021	0.0019	0.0013	0.0013	0.0005	0.0005	0.0004
SG	Mean	0.0068	0.0059	0.0048	0.0044	0.0040	0.0032	0.0030	0.0026	0.0021	0.0021
	SD	0.0046	0.0037	0.0029	0.0025	0.0020	0.0013	0.0012	0.0010	0.0008	0.0008

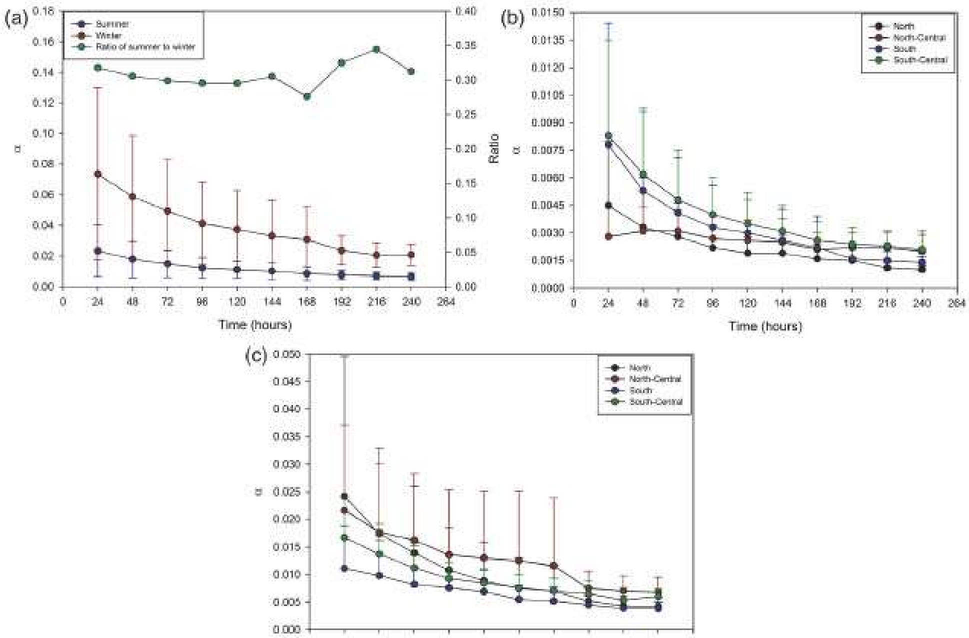

FIGURE 7 Recession coefficients, α, of all catchments (a) in winter and summer, and separated by region, (b) in summer and (c) in winter. The North region includes Q2 and Q3, north-central includes MLC1-MLC2-BLQ2-BLQ2, south-central includes EG and south includes SG

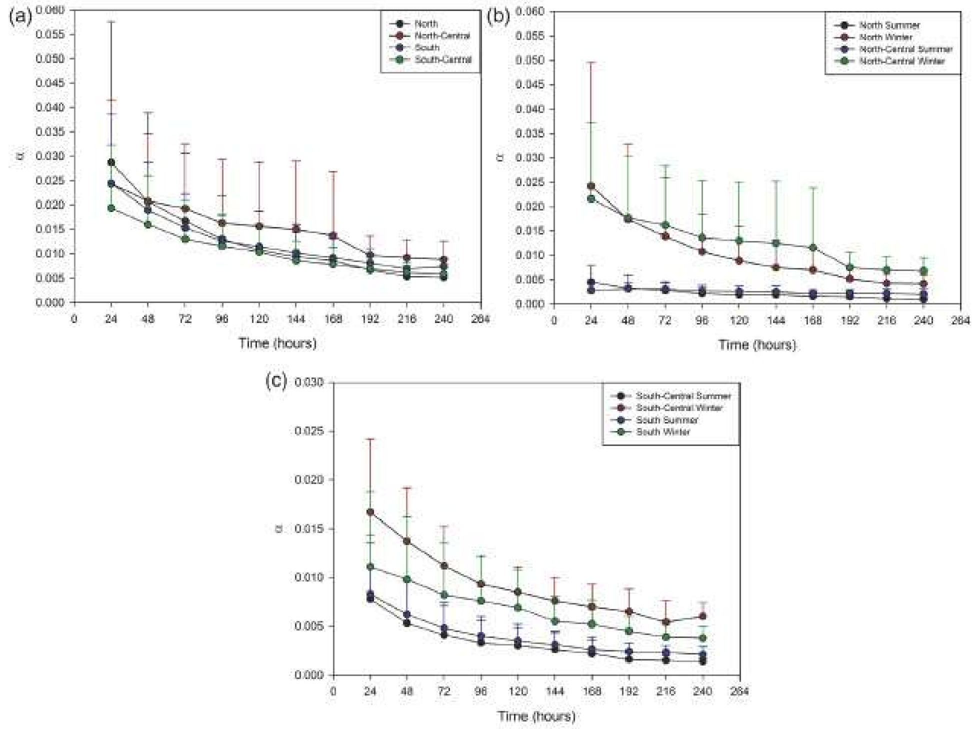

FIGURE 8 Recession coefficient separated by (a) region, and for winter and summer in (b) the northern regions and (c) the southern regions. The north region includes Q2 and Q3, north-central includes MLC1-MLC2-BLQ2-BLQ2, south-central includes EG and south includes SG

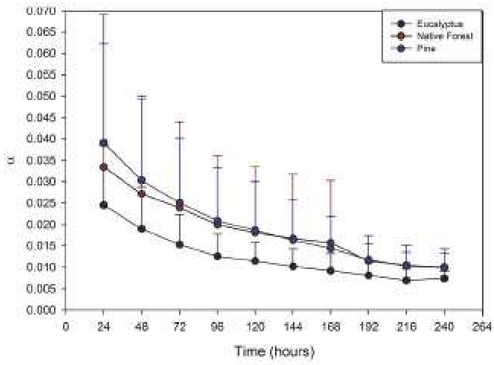

FIGURE 9 Recession coefficients for catchments with forest types combined

4.1 | Model performance

Recession flow characterization is complex as it is affected by the heterogeneity of the subsoil and aquifer (Beven, 2001), and the hydraulic gradient within the aquifer contributing stream flow. Although in this case there is a lack of information on the aquifer characteristics, our catchment hydrographs show a typical decay through time (Fiorotto & Caroni, 2013). We found that M2 was more efficient at early stages of recession suggesting a shallower groundwater input within the first 5 days. The better results of the M2 model may be due to a more realistic representation of shallower groundwater flow in the subsoil which dominates the recession in catchments with steeper slopes. However, as time passes, the influence of other components such as evapotranspiration in the spring–summer months results in better fits of exponential model (M1).

Overall, the exponential decay model (M1) and the combination model (M3) were more useful as predictors of flow than the power potential function (M2). Model M1, which was a good general model in this study, has been widely applied in studies of hydrograph recession after rainfall (Luo et al., 2016; Nathan & McMahon, 1990). M1, which is more efficient in the winter season in all catchments, also had satisfactory efficiency in the dry season (Amit et al., 2002). The M3 model, which showed promise here in higher rainfall catchments, was created and tested in the lower Andes, at a flow gauge located

35°S–71°W at 450 m.a.s.l., Maule Region, Curico Province, Chile, where the annual rainfall is ~1800 mm/y (Pizarro et al., 2013).

4.2 | Recession coefficient (α)

In the eight catchments studied, the underlying geology was similar and thus in this study, the focus was on the effect of vegetation cover and rainfall on the patterns of flow. The decline in flow was most rapid in wet catchments and in winter. The recession coefficient, α, varied between seasons (wet and dry) and was highest in winter wet season (Figure 7) in the northern lower rainfall catchments which, again, results in a rejection of our first null hypothesis. There were some visible differences between regions within seasons (Figure 7(b,c)), but these were not significant (p-value < 0.05). Thus, we reject the hypothesis (i) on a seasonal basis (wet and dry seasons) but not on the basis of the rainfall gradient. In the following paragraphs this general observation is discussed with respect to the effect of annual rainfall, seasonal patterns of rainfall and vegetation cover. Some consideration is also given to variation in soil type and depth between the catchments.

The rate of decline in the flow relative to the peak tended to decrease with time since rainfall. This effect was significant for all catchments in the winter and for the higher rainfall catchments in the summer. In all catchments and in both seasons the coefficient α reached a constant minimum value more than 240 h after the rain. This constant value at later times is consistent with an earlier study by Patnaik et al. (2015) in the United States. This pattern of flow is due to a progressive decrease in the contribution of soil stored water with time after a storm. The input of stored water to baseflow decreased resulting in a decrease in the recession coefficient (Biswal & Marani, 2014; Biswal & Nagesh Kumar, 2014).

These catchments covered a large range of annual and winter rainfall from the wet south to the dryer north. The winter dominance of rainfall in this part of Chile means that the relatively shallow soils were probably saturated, even in the northern catchments which had lower rainfall. In addition, in winter the rate of potential evaporation is similar in the north and south of this region and during this season evapotranspiration was similar to potential evaporation, even in the northern catchments (White et al., 2020). In this season, rainfall exceeds evapotranspiration (Ficklin et al., 2016). However, when examined by season (Figure 8(b,c)) we can see there was a separation between regions, particularly in summer, although, once again, this was not significant at $p < 0.05$.

A significant difference in recession coefficient, α, was observed between winter and summer for all catchments, and this was much stronger in the northern catchments than the southern ones. This difference was likely because in the northern catchments potential evaporation exceeded rainfall by more than soil water storage during the dry season. This effect was also observed by Wittenberg (2003) who attributed the recession coefficient differences between seasons to evapotranspiration reducing storage in the vadose and saturated zones in the regolith. In addition, summer drying of the lower rainfall catchments causes a reduction in the slope of the recession curve (Shaw et al., 2013; Tallaksen, 1995; Weisman, 1977).

It is noteworthy that during summer the recession coefficient did differ between the two northern catchments (Q2 and Q3, $p < 0.05$). While these catchments have similar rainfall and geology, one is planted to *P. radiata* (Q3) and the other (Q2) is occupied by a high conservation value native forest (White et al., 2020). It should be noted there were not any *Eucalyptus* species in these two catchments. They share similar rainfall amounts. Therefore, the variation in α between them was determined by geomorphology and forest type. The main geomorphological difference between them is drainage density (11.44 km/km^2 in Q2 and 8.67 km/km^2 in Q3), along with elongation (0.74 in Q2 and 0.60 in Q3). The difference between *P. radiata* and native forest species was not enough, given the small sample size, to produce a difference in the summer recession characteristics between these two land covers.

The difference between northern catchments (Q2 and Q3) was expected due to differences in the spring and summer rates of evapotranspiration between land uses. These differences could cause changes in the recession characteristics of catchments and therefore in the reliability of summer flow (Shaw et al., 2013). While it is quite well established that annual evapotranspiration of *Pinus* plantations can exceed that of native forests (Brümmer et al., 2012; Rao et al., 2011) the differences are small relative to annual rainfall and annual potential evaporation and thus studies often find that changes in recession coefficient are not sufficiently large or consistent to be attributable to variation in evapotranspiration for different land-uses. In our study, potential evaporation was 1300 mm in the northern catchments and around 1000 mm in the southern catchments. While differences between catchments in summer are consistent with the rainfall gradient, the subtler differences between vegetation types within regions were not large enough to be statistically different against this background. It is not clear whether this is driven by the forest differences or a coincident pattern in rainfall or difference in geomorphology. While there are clear differences on average between the regions (Figure 8), more catchments with these vegetation covers may be needed to detect differences.

While it is true that the underlying geology is similar among our eight catchments which are characterized by a uniform metamorphic rock typical of the region (Servicio Nacional de Geología y Minería, 2003), there are differences in soil depth between our catchments. The thinner soils and steep hillslopes at Quivolgo and MLC might to cause an increase in the recession coefficient due to greater runoff response to incoming precipitation (Peters et al., 1995). As catchment hillslopes are steep (mean slope over 40%, Table 1) we would expect that these hydrological systems worked as a 'storage excess sub-surface flow' (De Moraes et al., 2006) or shallow subsurface flow. Thus, the early stages of the recession will be fast due to these steep slopes and the characteristic shallow soils of steep slopes (Savenije, 2010). This behaviour can be seen in MLC and QUI catchments. In fact, drilling for access tubes and piezometers indicated that the soil depth was between 50 and 80 cm in both MLC1 and MLC2. In addition, and because M2 represents the shallower subsoil flow (or saturated soils and steep slopes) this model fits better in the early stages of the recession than M1. By contrast, M1 performed better later in the flow recession cycle (i.e., 5 days after Q_0 in our case). The M1 model does not depend on vertical flow (Hall, 1968). This lack of dependence on vertical flow is one possible reason for the better performance of this model in our catchments.

As infiltration is inversely related to slope, an increase in α might be attributable to shallows soils and/or steeper catchments. The recession coefficient denotes the storage factor delay. The α value decreases in areas with lower slopes so there will be a greater opportunity for infiltration of rainfall to groundwater recharge. Groundwater flow may therefore be slower, due to lower gradients, but persist for longer (Jakada et al., 2019). In areas with steeper slopes the infiltration may be less and, if hydraulic gradients are similarly steep, groundwater will seep faster to the stream (Troch et al., 2002). Thus, the value of the recession coefficient depends on the degree of saturation of the catchment prior to the recession stage. Therefore, a catchment with greater saturation results in a more stable groundwater contribution to the stream and, vice versa, streams with a more persistent baseflow indicate a catchment with greater saturated storage (Biswal & Marani, 2014; Penna et al., 2011; Weisman, 1977).

No differences between the coefficients in BLQ2, ER and SG were observed (p-value < 0.05) in either season, probably attributable to similar precipitation.

Topography and bedrock have been demonstrated as the main drivers of flow processes at hillslope scale (Duffy et al., 1994; Savenije, 2010; Troch et al., 2003). While our study did not include a detailed topographic analysis, it seems likely that similar catchment hillslope shapes between them is a key reason for the similar recession flow response observed across our catchments (Matonse & Kroll, 2009).

5 | CONCLUSIONS

The impact of plantations on stream flow is a global issue that will increase in importance as the role of forests in carbon neutrality is increased. Our comparison of plantations with native forest is therefore timely and the hydrologic differences between plantations and native forest were not large enough to be detected in flow recession analyses. In central Chile, where the majority of rain falls in winter, there was no conclusive evidence of an effect of land use on the rate of streamflow recession in either summer or winter. In the summer dry season, the differences between plantation and native forests were more evident, especially in the drier, northern catchments. It is important that further studies are made in the drier Mediterranean climate zones of Chile to further quantify the effect of land use change on summer recession and therefore on the vulnerability of dry-season flow to land use changes, particularly as these areas are the most populous in Chile.

In our study, streams receded more rapidly in the winter rainy season than in the summer dry season and this effect was stronger in northern, drier, catchments and in those with shallow, skeletal, soils than those with deeper loams. The differences observed between catchments in summer are attributable mainly to geomorphological factors such as drainage density and catchment elongation, but also soil moisture storage at the beginning of summer which depends both on characteristics of summer rainfall and on forest type. After rainfall, streams did not recede differently in catchments covered by native forest from those covered by pine plantations, but those with *Eucalyptus* had lower recession coefficients.

The recession coefficient could be a reasonable indicator of the vulnerability of summer flow by using the seasonal differences we have identified as surrogates for continuing climate change. Climate projections are for a continuing of the current drying trend and, as such, it is reasonable to extrapolate from wet seasons to current dry seasons into a future with less rainfall altogether. More and longer sequences of data would be needed to test such conjecture. Additional factors, such as the depth and texture of the vadose zone also have an important influence on the rate of recession and further work, particularly greater knowledge of soil profiles and water holding characteristics are required to further understanding of these features.

Finally, the analyses presented here are limited to this set of data and watersheds. As the Chilean topography and rainfall amount is highly variable, the exploration of the recession coefficient in a wide range of watersheds and climates is still necessary for a better understanding of recession flow. Our next steps are to investigate in more detail the complexity of the hillslope and its relationship with the recession flow in a broader scale from small to big catchments as topography may play a significant role in the water dynamics within a watershed (Troch et al., 2002). Additional factors, such as the depth and texture of the vadose zone also have an important influence on the rate of recession and further work, particularly greater knowledge of soil profiles and water holding characteristics are required to further understanding of these features.

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DATA AVAILABILITY STATEMENT

The data of this study are available from the corresponding author upon reasonable request.

ORCID

Francisco Balocchi https://orcid.org/0000-0002-9171-2382
Andrés Iroumé https://orcid.org/0000-0001-8148-1309

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CAPÍTULO III: An Analysis of the Effects of Large Wildfires on the Hydrology of Three Small Catchments in Central Chile Using Tritium-Based Measurements and Hydrological Metrics[2]

Francisco Balocchi, Diego Rivera, José Luis Arumi, Uwe Morgenstern, Donald A. White, Richard P. Silberstein y Pablo Ramírez de Arellano

3.1. Resultados claves

- El tiempo medio de residencia del agua varía entre 5 a 30 años
- Caudales máximos y base aumentaron post-incendios en cuencas con presencia de plantaciones
- Caudales máximos y base disminuyeron post-incendio en la cuenca con bosque nativo
- Primer estudio con tritio en el centro-sur de Chile y en cuencas con cobertura forestal

3.2. Resumen en extenso

Debido a los efectos del cambio climático se espera el aumento de incendios a nivel global (Goss *et al.*, 2020). A pesar de que los incendios forestales son una perturbación antrópica importante que afecta el suelo de las cuencas y los procesos hidrológicos, el efecto ha sido poco estudiado en Chile desde un punto de vista hidrológico, encontrándose solo cinco estudios relacionados (Jones *et al.*, 1975; García-Chevesich *et al.*, 2010; García-Chevesich *et al.*, 2019; Balocchi *et al.*, 2020; White *et al.*, 2020). En este marco y dada la oportunidad de estudiar los efectos de los incendios en cuencas forestales pre y post-incendio del año 2017, se encontró un efecto hidrológico diferente dependiendo de la cobertura analizada. Así, los caudales máximos y base aumentaron en dos cuencas (plantaciones y mixta), pero disminuyeron en la tercera (bosque nativo). Para el caso de la disminución de caudales máximo y disminución de los bases, Balocchi *et al.* (2020) concluye que se debe principalmente a dos condiciones; (i) al aumento de la infiltración en la cuenca y recarga de agua del suelo, y/o (ii) post-incendio el caudal está pasando por debajo del punto de medición y, por lo tanto, no se está midiendo. Sin embargo, en la cuenca con bosque nativo, los valores de transpiración medidos no coinciden (*i.e.*, no son lo suficientemente altos) para explicar la baja de caudal. Por otro lado, en las cuencas en donde aumentaron

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los caudales máximos, se explica principalmente por la hidrofobicidad del suelo post-incendio, lo que aumentó el caudal más de 30 veces en una de las cuencas estudiadas. Por lo tanto, la hipótesis propuesta con respecto a un efecto hidrológico diferente de acuerdo con el tipo de cobertura se acepta.

Con respecto a los tiempos medios de tránsito (MTT) del agua en las tres cuencas costeras de tipo climático mediterráneo analizadas, se utilizaron modelos de parámetros agrupados (LPM) para obtener sus MTT. Las concentraciones de Tritio indicaron edades del agua de entre 5 a 30 años. Estos valores de MTT antes y después del incendio no fueron significativamente diferentes. Por lo tanto, no hay evidencia concluyente de cambios hidrológicos a nivel de las aguas subterráneas debido a incendios forestales en esta etapa temprana. Evidentemente, dado los tiempos de residencia del agua, el efecto de los incendios a largo plazo solo podrá ser evaluado en el futuro. Por lo tanto, las mediciones de Tritio para este caso en particular, no parecen ser la mejor opción para este tipo de estudios.

Las métricas utilizadas en este estudio permitieron identificar el efecto de los incendios en diferentes coberturas. Sin embargo, este análisis abre la discusión con respecto a que otros factores influyen post-alteraciones antrópicas como un incendio. Por ejemplo, el suelo es uno de los factores que afectan el comportamiento hidrológico post-incendios debido a (i) la fuerte degradación de estos en donde las plantaciones están establecidas en comparación a los suelos de sistemas nativos con menor intervención y (ii) la disimilitud de efectos en áreas similares y la formación de hidrofobicidad (*e.g.*, Garcia-Chevesich *et al.*, 2019).

En el caso particular de este estudio, el cambio de los procesos hidrológicos está relacionado, principalmente, a la interacción entre la cobertura y el suelo, y estos con la generación de escorrentía superficial e infiltración de agua. Finalmente, dentro de los años siguientes a este estudio, el Centro de Investigaciones Forestales Bioforest mantendrá un programa de muestreo para continuar investigando tanto la sequía a largo plazo como el efecto de los incendios forestales en estas cuencas.

hydrology

Article

An Analysis of the Effects of Large Wildfires on the Hydrology of Three Small Catchments in Central Chile Using Tritium-Based Measurements and Hydrological Metrics

Francisco Balocchi [1,2,*], Diego Rivera [3,4], José Luis Arumi [4,5], Uwe Morgenstern [6], Donald A. White [7,8], Richard P. Silberstein [8,9,10] and Pablo Ramírez de Arellano [1]

1 Bioforest SA, Camino a Coronel s/n km 15, Coronel 4190000, Chile; pablo.ramirez@arauco.com
2 Water Resources and Energy for Agriculture PhD Program, Water Resources Department, Universidad de Concepción, Chillán 3812120, Chile
3 Centro de Sustentabilidad y Gestión Estratégica de Recursoe (CiSGER), Facultad de Ingeniería, Universidad del Desarrollo, Santiago 7760609, Chile; diegorivera@udd.cl
4 CHRIAM Water Center ANID/FONDAP/15130015, Universidad de Concepción, Chillán 3812120, Chile; jarumi@udec.cl
5 Department of Water Resources, Faculty of Agriculture Engineering, Universidad de Concepción, Chillán 3812120, Chile
6 Isotope Hydrology & Water Dating Lab, GNS Science, Lower Hutt 5040, New Zealand; u.morgenstern@gns.cri.nz
7 Whitegum Forest and Natural Resources Pty. Ltd., Midland, WA 6056, Australia; whitegumfnrm@gmail.com
8 School of Science, Edith Cowan University, Joondalup, WA 6027, Australia; richard.silberstein@hydroenviro.com.au
9 Hydrological and Environmental Scientific Solutions Pty Ltd., Perth, WA 6872, Australia
10 School of Agriculture and Environment, University of Western Australia, Crawley, WA 6009, Australia
* Correspondence: francisco.balocchi@arauco.com; Tel.: +56-932-498-759

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Abstract: Wildfires are an important disturbance affecting catchments' soil and hydrological processes within. Wildfires are predicted to increase in both frequency and severity under climate change. Here, we present measurements of tritium (^3H) in surface water of three streams before and after the 'las Máquinas' megafire of January 2017 in central Chile and streamflow metrics. Mean transit times (MTTs) of water were calculated in three coastal catchments with the Mediterranean climate type, covered by native forest, a mixture of native forest and *Pinus radiata* D. Don, and *P. radiata*. Lumped parameter models (LPMs) were used to obtain MTTs. Tritium activities from 2012 to 2018 ranged from 0.597 to 0.927 Tritium Units (TU), with the lowest TU activity in 2018. These ^3H concentrations indicated water ages from 5 to 30 years. Following the fire, peak flows and baseflow have increased in two catchments but decreased in the third. Even though we have seen changes in the hydrological responses within the three catchments, pre- and post-fire MTT values were not significantly different. Therefore, there is no conclusive evidence of hydrological changes at the groundwater level due to wildfire at this early stage. However, since the MTT ranges from 5 to 30 years, it is likely that more time is required for the changes in the hydrograph to be clearly reflected in the tritium signal even though there are noticeable changes in streamflow metrics such as runoff and baseflow. Within the following years from this study, a sampling schedule to continue to investigate both the long-term drought and the effect of wildfire on these catchments will be maintained.

Keywords: tritium; land cover; native forest; monterey pine; wildfires; *Nothofagus glauca*

1. Introduction

Management of the effect of fires on the supply of water to regional communities requires knowledge of their effect on forest structure, soils, and hydrogeology and on the local and regional water balance. In January of 2017, wildfires burned a few more than 550,000 hectares of mostly forested land in Central Chile [1]. The most intense and damaging fires occurred in the Maule Valley, east of the coastal city of Constitución, and have become known as the Las Mâquinas fires. While they were historically severe, it is expected that climate change will further increase the frequency and intensity of wildfires in Central Chile [2,3]. These fires occurred principally in the coastal range of Central Chile. The Coastal mountains, while not particularly high, are a steep and rugged landscape that is forested and dotted with small, rural communities. The forests are a combination of commercial plantations of *Pinus radiata* and *Eucalyptus* spp. and the fragmented and vulnerable Roble-Hualo native forest which is named after the deciduous *Nothofagus* species that dominate the overstorey. The plantations provide employment for rural Chileans while the native forest has high conservation value. The water that flows to the Maule River is a regionally important supply of drinking water and irrigation and is important to the many rural communities in the ranges.

The effect of fire on water balance depends upon the combination of many factors, including the rainfall regime before and after the fire and the effects of the fire on vegetation and the soil micro- and macro-structure. Central Chile, where the fires occurred, has a Mediterranean climate with an annual dry season extending from November to March or April. The 8 months that preceded the fires were unusually dry, in particular between July and October. The lack of the winter rains and a very hot start in January created the risk of severe fire conditions [4]. The climatic conditions that increased the fire risk also created a dry catchment with historically low moisture storage [5]. The fires further changed the local controls on the water balance by causing a rapid change in leaf area and, therefore, transpiration and interception [6–8]. The combined effect of this period of drying, followed by a rapid reduction of cover may have caused complex and hard to predict changes in catchment storage and hydrology. Therefore, understanding these hydrological changes after fires is a key factor in water resources management [9].

One measure of the hydrological behavior within a catchment is the Mean Transit Time (MTT) of water. MTT is the average time of a water molecule traveling through a catchment, from when it enters the system to when it exits at any point in the stream as discharge [10]. Therefore, MTT integrates all the hydrologic processes in a single measure of catchment behavior. Tritium activity (TU, where 1 TU represents a $^{3}H/^{1}H$ ratio of 10^{-18}) in water has been used to estimate the MTT of water in forested catchments worldwide (e.g., [11,12]). This has improved the understanding of how, and from which sources, the hydrological system is recharged (e.g., [13]). To date, tritium has not been used to investigate changes in MTT after a wildfire or in forested catchments in Mediterranean climate areas, such as drought-prone Central Chile. As noted above, fire destroys or damages the canopy, can change soil water repellency, and can open or close soil macropores. Combined, these changes may result in profound hydrologic change and affect the rate of groundwater recharge, streamflow, baseflow, and soil moisture storage [14]. Moreover, extremely dry conditions lead to decreasing stored groundwater and soil water [5] and interact with the fire in the affected catchment. How these hydrological processes interact to reach a new state depends on each catchment, but site monitoring, water age estimation, residence time, and hydrological models are all helpful to support observations and planning.

As well as affecting the vegetation, and therefore transpiration and throughfall, wildfire can also alter important soil properties that can have implications for the water cycle [15]. Commonly reported changes after a fire include an increase in peak flows and storm flows and a severe reduction in baseflow, baseflow recession, and low flow [16,17]. These changes in streamflow generation processes have been attributed to the formation of a hydrophobic layer [18] or soil sealing [19]. There have been a large number of investigations of the effect of fire on streamflow dynamics (e.g., [20,21]) that have applied a range of methods including cluster and regression analysis [22], runoff generation at hillslope scale [23], chemical analysis for groundwater interaction [24], geochemical and end-member mixing analysis [25], isotope mass balance methods [26], catchment runoff modelling (e.g., [27]), and paired catchment experiments [28,29].

While there have been many studies of the effects of fire on forest hydrology, to date, none of these have related the concentration of tritium to pre- and post-fire streamflow. McDonnell et al. [30] discussed the limitations of standard hydrography and hydrometric for understanding sub-soil hydrological processes. McDonnell and Beven [31] argued for the use of tracers in hydrology to complement hydrometrics and allow the more complete investigation of changes to hydrological processes caused by disturbances such as wildfire. Thus, this paper aims to (i) determine if we can detect changes of catchment groundwater water transit that may be ascribed to the impact of the fires, and (ii) unveil changes in hydrologic variables (i.e., runoff and baseflow). Within the area that was burnt in January 2017 in Central Chile [1], there are three experimental catchments where high and low baseflow tritium concentration of stream water has been monitored since 2009. All three catchments were completely burned by this high severity fire. This study reports tritium isotopic composition of water in the stream (high and low baseflow) and rain in these burnt catchments before and two years after the fire and uses these data to test the hypothesis that the MTT of water in these catchments is not affected by wildfire.

2. Materials and Methods

2.1. Study Sites Characteristics

The study site is in the locality of Quivolgo near the city of Constitución, in the Maule Region. All measurements were made in three small catchments located around 35°23′ S, 72°13′ W that flow to the Maule River (Figure 1). All catchments have similar geology and aspects so the main difference amongst them is their vegetation cover, size, and slopes. Catchment 1 (Q1) is covered by a *Pinus radiata* D. Don plantation that was established in 2003 and has an area of 0.1895 km^2 and a mean slope of 22%. Catchment 2 (Q2) is covered by native forest with an overstorey of *Nothogafus glauca* (Phil.) Krasser and has an area of 0.3302 km^2 with a mean slope of 51%. A complete description of the forest structure in Q2 can be found in [8]. Catchment 3 (Q3) is a mixed catchment covered by Pinus radiata (62%), planted in 2001, and native forest (34%) with *N. glauca* as the main species, and has an area of 0.4014 km^2 and a mean slope of 44%. The P. radiata plantations on Q1 and Q3 were established at 700 trees ha^{-1} and were thinned at age six (2007 in Q1 and 2009 in Q3) to 450 trees ha^{-1} [32]. At the same time, the retained trees were pruned to a height of 2.1 m. The size, slope, channel length, and other characteristics of the three catchments are summarized in Table 1.

The geology of the catchments is mainly metamorphic bedrock [33], described as "Dollimo Complex" [34]. Soil texture, bulk density, and organic matter have been estimated from samples taken in ten soil pits dug in and around the catchments, with estimated average bulk density and organic matter presented in Table 1.

The climate of this part of Central Chile is Mediterranean-type characterized by a pronounced summer dry season and strongly winter-dominant rainfall [35]. The average annual rainfall from 2009 to the present was 951 mm. Rainfall in the year before the fire was the lowest recorded since 2009 (697 mm) and the year after the fire was the wettest since 2009 (1460 mm) [20]. The average maximum temperature in the hottest month was around 26 °C (January), the average minimum in the coldest month was 1.3 °C (July) and the average temperature was 12 °C [20].

The Las Máquinas fire in January 2017 burned all the forest cover in Q1, Q2, and Q3. All the pine trees in Q1 and Q3 were killed by the fire and were replaced by a carpet of seedlings before the end of 2017. The native forest in Q2 regenerated rapidly. All the native forest species have resprouted from the base and the *N. glauca* also regenerated from the crown [8]. Resprouts were evident less than two months after the fire.

2.2. Streamflow and Rainfall

Streamflow has been measured since 2009 in Q1 and Q2, and from May 2013 in Q3. A 90° v-notch weir was built in each of the catchment outlets. Water depth in the weir was measured every 5 min using a pressure transducer and streamflow was calculated using a rating curve calibrated for each weir. This calibration was checked using monthly manual flow and depth measurements. Due to weir and sensor damage caused by the wildfires, no flow data was recorded between January 2017 and February 2017. There are some flow gaps within the Q3 dataset due to weir repairs from May 2018 to July 2018, and a small gap in August 2017.

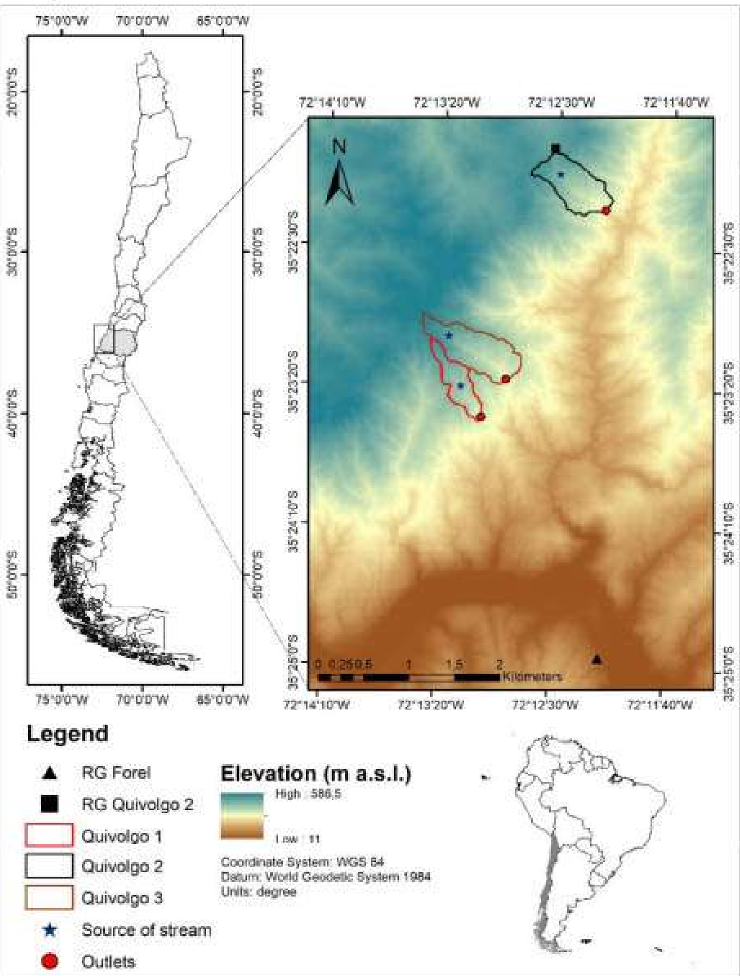

Figure 1. Catchment location and sampling points within the study site. RG: rain gauge.

Rainfall was recorded using a rain gauge at the top of Q2, and a second gauge at the site of Forel, around 4 km southeast of the catchments (Figure 1). The gauge at the top of Q2 was used as the primary source. The relationship between the gauge at Q2 and the weather station at Forel was used to estimate rainfall at the catchments if data from either location was available. For the periods where neither the gauge at Q2 nor at Forel were operating, data obtained from nearby stations maintained by the General Directorate of Water (DGA) at Nirivilo was used to estimate rainfall at the catchments, using linear regression as in [20].

Table 1. Characteristics of three investigated catchments in Quivolgo, Constitución, Chile.

	Q1	Q2	Q3
Land cover	*P. radiata*	Native forest	*P. radiata*/Native forest
Annual precipitation (mm) *		966	
Period of max rainfall *		May–September	
Maximum monthly rainfall (mm month^{-1}) *		283	
Surface (km^2)	0.1895	0.3302	0.4014
Mean Altitude (m a.s.l)	422	395	442
Mean slope (%)	22.1	51.6	44.8
Catchment perimeter (km)	2.50	2.54	3.10
Outlet altitude (m a.s.l.)	305	265	306
Mean slope of stream (%)	28.6	35.1	34.7
Length of main stream (km)	1.11	1.10	1.33
Time of concentration (hrs.) **	0.13	0.12	0.15
Type of climate *		Temperate semi-oceanic	
Clay content (%) ***	39	40	39
Silt content (%) ***	36	36	36
Sand content (%) ***	25	24	25
Organic matter (g g^{-1}) ***	1.265	1.135	1.265
Bulk density *** (g cm^{-3})	1.375	1.365	1.375
Main textural class (USDA)	Clay Loam	Clay	Clay Loam

* [36]. ** California Culvert Practice [37]. *** estimated from nearby soil pits as the average value.

2.3. Tritium Sampling

Samples of water were collected from the streams in Q1, Q2, and Q3 on eight occasions between 2012 and 2018. After 2014, each stream was sampled, as proposed by [38], at a time of low baseflow (end of summer) and at a time of high baseflow (end of spring). Originally, the objective was to sample tritium concentration in baseflow at different land uses and follow up the MTT in small catchments. Therefore, prior to 2018, all samples were collected from the weir at the catchment outlet. This approach and subsequent data analysis assumed that streamflow originated from a single source [39]. In 2018, samples were also collected from the point where the streamflow started in the catchment headwaters (sampling points in Figure 1). This additional sample was taken with the aim of estimating the age of water sources along the stream. Sampling followed the Geological and Nuclear Sciences (GNS) procedure and used a Nalgene Narrow-Mouth Square HDPE 1L bottle. Tritium (^{3}H) concentration of the samples was measured at the tritium and Water Dating Laboratory, Geological and Nuclear Sciences, New Zealand, according to [40].

2.4. Lumped Parameter Models for Estimating the Mean Transit Time (MTT) of Water

Several methodologies have been proposed to calculate the mean transit time (MTT) of water in catchments. Those that apply mixing and decay methods (e.g., [41]), such as lumped parameter models (LPMs) [42–44] have been widely used in hydrology studies [45,46]. Jurgens et al. [47] introduced some improvements to one of these LPMs (TRACERMODEL [48]) and created the TracerLPM Excel workbook. This has been used extensively [49–51] and was applied in this study. As there is no information regarding aquifer characteristics within the site, the exponential flow model (or exponential mixing model, EMM), the exponential-piston flow model (EPM), and the dispersion model (DM) were all used in order to compare MTTs obtained with different models [38]. These three models are the LPMs that are mostly applied to estimate mean transit times (e.g., [52]). A detailed explanation of each model can be found in [47]. As the EPM describes a mixture of exponential and piston flow portions within an aquifer, the TracerLMP software can define the proportion or contribution of the piston and exponential flow [47]. The EPM ratio is 1/f − 1, where f is the proportion of aquifer volume exhibiting exponential flow [53]. As the aquifer configuration is unknown, we performed the EPM modelling using f values of 0.7 and 0.8, which give EPM ratios of 0.43 and 0.25, respectively, following the work in [54].

Estimation of MTTs via LPMs requires the comparison of the tritium concentration in the stream (output) to rainfall (input). No continuous tritium rain record is available for Chile. However, tritium samples have been collected from Chilean rain stations sporadically and measured by the IAEA (IAEA and WMO, 2020). Chile and New Zealand receive rain from a similar Southern Ocean maritime climate. The tritium concentrations of Chilean and New Zealand rain are therefore expected to be similar. This is confirmed by the nearly identical records between Puerto Montt and Kaitoke, which both lie near the west coasts at similar latitudes. Figure 2 shows tritium concentrations in rain from the IAEA station Puerto Montt, Chile, in comparison to those from Kaitoke, New Zealand [40]. Moreover, results of rain samples collected between 2012 and 2014 near Constitución and Valdivia to refine the tritium input for local catchment studies are shown. The tritium concentrations between 1965 and 1975 at Puerto Montt and Kaitoke sites, during the period following the atmospheric nuclear weapons tests, match very well (R^2 0.83). The records also match well in the later period, between 2003 and 2009, however with a slight bias due to natural effects such as solar activity (neutron flux), formation of clouds, and the evaporation contribution to precipitation (more details in [55]), clearly indicated by a few higher concentration data, but still with a good agreement (R^2 0.77). Moreover, tritium concentrations of two rain samples collected in 2014 at Valdivia, at a similar latitude, match those of Kaitoke (insert Figure 2).

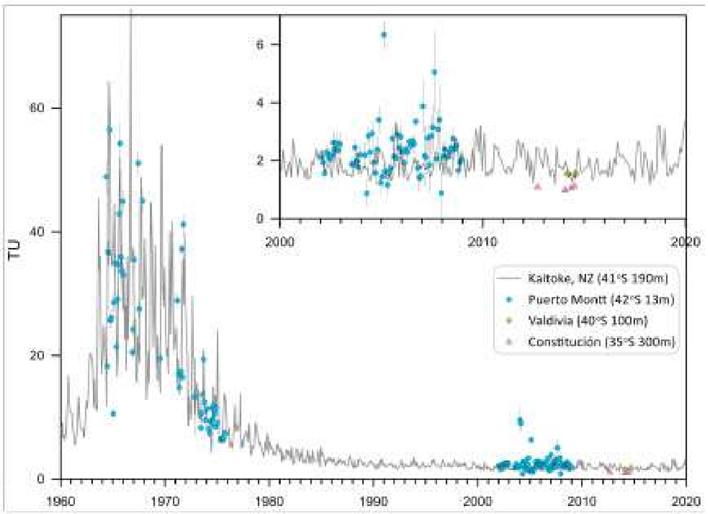

Figure 2. Tritium concentration of Chilean rain compared to rain from Kaitoke, New Zealand. Insert shows data with higher resolution for the last 20 years up to ≈ 6 TU. Also listed in the legend are the altitudes of the rain collection stations.

Due to their lower latitude, tritium concentrations of rain in catchments near Constitución are expected to be lower than at Kaitoke because tritium-rich atmospheric moisture originating from higher latitudes becomes increasingly diluted by low-tritium oceanic moisture on its way to lower latitudes. To determine a baseline tritium level in rain at Constitución, a scaling factor was determined for tritium levels between Constitución and Kaitoke. Four rain samples were collected at the study site (1.091 ± 0.028 TU in September 2012, 0.991 ± 0.027 in January 2013, 1.058 ± 0.028 in May 2014, and 1.124 ± 0.028 in June 2014), leading to a scaling factor of 0.62 following standard correlation procedures (e.g., [56]). The scaling factor, after applying to the Kaitoke rain TU activities, results in an estimate of the theoretical TU in the rain at the study site.

2.5. Data Analysis

The annual streamflow in the three catchments was determined for a period before and a period after the fires of January 2017. These intervals were 2010–2013 and 2017–2018 for Q1, 2010–2015 and 2017–2018 for Q2, and 2014–2016 and 2017 for Q3. For each catchment, annual runoff coefficients (RC, the ratio of streamflow to rainfall) and summer flows (between January and March) were calculated (e.g., [22]). We also estimated the annual average flow (mm, average from 2010 to 2016), the highest annual flow (mm), the lowest annual flow (mm), and the highest and lowest summer flow for each catchment.

The baseflow yield was calculated using the Recursive Digital Filter [57] passing forward, backward, and forward over the data with a filter of 0.925 for each full year of data. The results of each LPM were then compared with the hydrometrics results and rainfall intensity in 1 h versus instantaneous flow was also analyzed. Baseflow, Baseflow Index (BFI), and RC along with TU and MTTs were quantified and analyzed.

In order to compare water ages on each catchment and check if they have significant differences before and after a fire, the Scott and Knott hierarchical cluster analysis [58] was performed (p-value < 0.05) (hereafter referred to as SK test), using the ScottKnott R package.

3. Results

3.1. Rainfall and Streamflow

Average rainfall during the winter (June to August) for the 2010–2018 period was 579 mm and the driest winter was in 2016 (250 mm) while the wettest was in 2017 (832 mm). The average summer (Jan-Feb-Mar) rainfall was 25 mm, with the driest summer in 2017 and 2018 (5 and 4.8 mm, respectively) and the wettest in 2011 (53 mm). The six-month period immediately before the fire was unusually dry while the winter after the fire was much wetter than average (Figure 3).

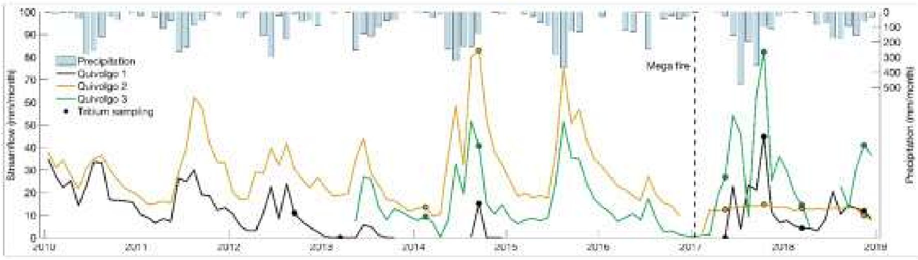

Figure 3. A time course of monthly streamflow (in mm) for the three study catchments between 2010 and 2018. The dates of water sampling for tritium analysis are indicated by colored dots. Note that the stream in Q1 (P. radiata plantation) ceased to flow in November 2013 and recommenced in August of 2014. Flow ceased again in October 2014 and did not recommence until the first winter (May–June 2017) after the fires.

In Q1 (P. radiata), annual, maximum, and minimum streamflow were similar before and after the fire while these measures of flow were greatly reduced in Q2 (native forest) and nearly doubled in Q3 (P. radiata) (Table 2). Of the three catchments, Q2 had the highest average annual flow (367 mm) and summer flow (64 mm) before the fires. After the fires, Q3 had the highest annual average flow (364 mm) and the highest summer flow (33 mm) (Table 2).

In the Q2 catchment, the lowest monthly flow before the fire was 9.7 mm in July 2016, and after the fire was 7.9 mm in December 2018. In contrast, for Q3 (P. radiata) the maximum flow was in the first year after fire (82 mm) and the highest summer flow was the second year after fires (62 mm). Moreover, for Q3, the maximum monthly flow before fires was in August 2014 (5 7 mm) and the minimum in April 2014 (0.4 mm). The minimum monthly flow was, interestingly, right after the fires in January 2017 (0.3 mm).

Table 2. Streamflow characterization within the experimental catchments at Quivolgo (average values pre- and post-fire).

	Q1		Q2		Q3	
Catchments	Before	After	Before	After	Before	After
Annual average flow (mm)	147.8	122.6	367.3	141.4	181.8	363.7
March low flow (mm)	0	4.37	19.7	12.7	7.3	7.8
* Average summer flow (mm)	32.4	17.1	63.8	26.5	26.4	32.7
** Runoff coefficient	0.21	0.10	0.37	0.12	0.17	0.28

* January to March flow. ** Sum of streamflow when is available to the sum of the rainfall when streamflow is available. Q1 streamflow available from 2010–2013 and 2017–2018, for Q2 from 2010–2015 and 2017–2018, and for Q3 2014–2016 and 2017 for Q3. Specifically, Q2 2016 flow is complete until December and the 2017 flow started in February due to the wildfire: Q3 2014 flow does not include October-November, 2017 August, and 2018 from April-July due to weir reparation. Q2 is a complete set of daily streamflow data from 2010 to 2018.

3.2. Runoff Coefficient (RC, the Ratio of Streamflow to Rainfall)

The average runoff coefficient (RC) after the fire in Q1 and Q2 is half of that before the fire (Table 3), while in Q3 after the fire, RC values are 60% greater than those observed before the fire. In Q1 (P. radiata), the RC decreased immediately after the fire but has since started to increase. After the fire, the RC in Q2 decreased and was similar to that of Q1.

Table 3. Annual runoff coefficients pre- and post-fire (January 2017).

	Pre-Fire							Post-Fire	
Catchment	2010	2011	2012	2013	2014	2015	2016	2017	2018
Q1 PR	0.32	0.21	0.14	inc	inc	n/d	n/d	0.09	0.13
Q2 NF	0.41	0.42	0.36	0.34	0.37	0.37	0.34	0.09	0.17
Q3 PRM	n/d	n/d	n/d	n/d	0.17	0.21	0.14	0.25	0.39 *

NF: native forest, PR: radiata pine, PRM; mix radiata pine and NF. Inc: incomplete dataset, n/d: no data. * January to April period.

To complement runoff coefficient results, we have selected some high rain intensities from before and after the fire (when available) and analyzed the instantaneous streamflow (L s^{-1}) one hour before and after the selected rainfall event (Table 4). There was no flow over the gauging weir in Q1 in the year preceding the fires. In Q3 rainfall of similar intensity yielded more flow after than before the fires. In Q3, rainfall of similar intensity yielded 20 times more water than Q2 and twice that of Q1 (Table 5, Q + 1). Conversely, Q2 required more rainfall after the fire to yield a similar flow to that generated before the fire.

3.3. Baseflow

Baseflow index (BFI), the ratio of baseflow to annual flow, varied from 31 to 94% across the three catchments (Table 5). In Q1, the BFI was higher after the fire than before. In Q2, the ratio was similar before (86%) and after (93%) the fire. In Q3 the baseflow proportion before the fire ranged from 54 to 71% (mean 79%) but decreased after the fires.

3.4. Tritium Concentrations and the Age of Water

Streamflow was measured on the day when water samples were collected (Table 6), and tritium concentration is summarized in Tables 6 and 7. In 2013 and May 2017, there was very little or no water flowing through the weir in Q1 (*P. radiata*). Of the three catchments, Q3 (*P. radiata* and Native Forest) had the highest monthly flow rate at the time of sampling after the fires. Flow rate at sampling was lower in all three catchments in the second year after fire than before the fire. No significant relationship was observed between monthly flow and either tritium concentration or Mean Transit Time (MTT).

Table 4. Rainfall intensity in 1 h (I) and instantaneous streamflow yield (L s^{-1}) pre- and post-fire (when data were available).

	Date	I (mm h^{-1})	Q1		Q2		Q3	
			Q_{t-1}	Q_{t+1}	Q_{t-1}	Q_{t+1}	Q_{t-1}	Q_{t+1}
Pre-fire	16 August 2016	8.8	-	-	2.50	3.21	1.38	1.98
	16 October 2016	4.6	-	-	1.98	2.31	0.90	1.46
	31 October 2016	4.2	-	-	1.54	1.83	0.29	1.14
Post-fire	15 June 2017	18.8	-	-	2.16	2.28	45.22	75.19
	10 April 2017	14.4	3.78	22.76	2.14	2.08	13.76	39.04
	22 June 2017	14	23.11	40.24	2.29	2.56	69.77	79.62

Q_{t-1}: streamflow 1 h before; Q_{t+1}: streamflow 1 h after.

Table 5. Proportion of baseflow over total flow within each catchment.

Year	Q1 BFI	Q1 Baseflow (mm)	Q2 ** BFI	Q2 ** Baseflow (mm)	Q3 *** BFI	Q3 *** Baseflow (mm)
2010	0.55	154.8	0.88	269.6	n/d	n/d
2011	0.56	103.3	0.77	277.4	n/d	n/d
2012	0.44	49.8	0.81	251.7	n/d	n/d
2013	n/d	n/d	0.83	202.9	n/d	n/d
2014	n/d	n/d	0.69	305.6	0.54	117.3
2015	n/d	n/d	0.83	321.7	0.69	162.2
2016	n/d	n/d	0.85	190.8	0.69	64.0
2017	0.31 *	41.2	0.93	127.5	0.51	185.9
2018	0.66	73.4	0.94	136.6	0.71	149.7

n/d: no data: * Q1 started to flow again in June 2017 after fires: ** Q2 2016 flow is complete until December and 2017 flow started in February due to the wildfire: *** Q3 2014 flow does not include October-November, 2017 August, and 2018 from April-July due to weir reparation.

Table 6. Monthly flow (mm) at the time of sample collection for tritium analysis in the three Quivolgo catchments.

Date of Sampling	Q1	Q2	Q3
7 September 2012	11.0	-	-
4 March 2013	0.0	-	-
17 February 2014	-	13.6	9.3
15 September 2014	15.2	82.9	40.6
18 May 2017	0.0	12.5	26.9
17 October 2017	44.9	14.8	82.3
6 March 2018	4.4	13.1	14.5
21 November 2018	11.9	9.9	41.0

In Q1 (P. radiata land cover) mean transit times varied from 5 to 15 years (Table 8) between 2012 and 2018. A slight increase in MTT from before to after fires was observed in Q1. Moreover, in Q1 the estimate for the MTT did not differ between samples collected at the stream source and the weir in 2018. In Q2 (native forest land cover), estimates of MTT ranged between 9.5 and 30 years (Table 8) and no change in MTT was evident from before to after the fire. In Q2, MTT from the stream source is about half of the age at the outlet in both sampling campaigns (summer/spring 2018) indicating groundwater contributions from deeper (longer) flow paths further down in the catchment. In Q3 (P. radiata and native forest land cover), MTT estimates ranged from 8 to 29 years (Table 8). As noted by using hydrological metrics, there was no noticeable pattern of change in MTT due to the fires in this catchment, and there was no significant difference in MTT between the water source and outlet.

Table 7. Tritium concentrations (TU) at sampling dates in three experimental catchments at Quivolgo and lab analysis dates.

Stream Sampling Date	Q1 Stream PR	Q2 Stream NF	Q3 Stream PRM	Lab Analysis Date
		TU		
7 September 2012	0.831 ± 0.026	-	-	1 June 2013 *
4 March 2013	0.801 ± 0.025	-	-	1 June 2013 *
17 February 2014	-	0.597 ± 0.020	0.613 ± 0.018	23 March 2015
15 September 2014	0.927 ± 0.027	0.779 ± 0.022	0.819 ± 0.022	23 March 2015
18 May2017	0.784 ± 0.029	0.615 ± 0.027	0.754 ± 0.029	22 July 2019
17 October 2017	0.818 ± 0.029	0.704 ± 0.027	0.726 ± 0.027	22 July 2019
6 March2018	0.689 ± 0.029	0.614 ± 0.021	0.652 ± 0.021	22 July2019
21 November 2018	0.727 ± 0.023	0.540 ± 0.025	0.621 ± 0.026	8 June 2020
Source sampling date	Q1 source PR	Q2 source NF	Q3 source PRM	Lab analysis date
		TU		
6 March 2018	0.708 ± 0.027	0.766 ± 0.023	0.624 ± 0.021	22 July 2019
21 November2018	0.792 ± 0.024	0.689 ± 0.023	0.615 ± 0.026	8 June 2020

NF: native forest, PR: radiata pine, PRM; mix radiata pine and NF. * An approximate date due to precluded lab database.

Table 8. Mean transit time (years) for all catchments by each of the models.

Stream Sampling Date		**Q1 Stream PR**		
MTT (Years)	**EMM**	**EPM (0.25)**	**EPM (0.43)**	**DM (0.8)**
7 September 2012	9 a	8.5 a	8 a	11 a
4 March 2013	11 a	9 a	8.5 a	12 a
17 February 2014	-	-	-	-
15 September 2014	5 a	5.5 a	5.5 a	6 a
18 May 2017	9 a	8 a	8 a	10 a
17 October 2017	8 a	6.5 a	6.5 a	8 a
6 March 2018	15 a	12 a	11.5 a	15 a
21 November 2018	13 a	10 a	9.5 a	13 a
6 March 2018 *	13.5	11	10.5	14
21 November 2018 *	9.5	8	7.5	9.5
Stream sampling date		Q2 stream NF		
MTT (years)	EMM	EPM (0.25)	EPM (0.43)	DM (0.8)
17 February 2014	30 a	22 a	19 a	29 a
15 September 2014	11 a	10 a	9.5 a	13 a
18 May 2017	22 a	17 a	17 a	23 a
17 October 2017	14 a	11 a	11 a	14 a
6 March 2018	22 a	17 a	15 a	21 a
21 November 2018	30 a	22 a	20 a	30 a
6 March 2018 *	10	9	8.5	11
21 November 2018 *	15	11	11	15
Stream sampling date		Q3 stream PRM		
MTT (years)	EMM	EPM (0.25)	EPM (0.43)	DM (0.8)
17 February 2014	29 a	20 a	18 a	25 a
15 September 2014	9 a	8.5 a	8 a	11 a
18 May 2017	11 a	10 a	10 a	12 a
17 October 2017	13 a	10.5 a	10 a	13 a
6 March 2018	18 a	14 a	13 a	18 a
21 November 2018	21 a	16 a	15 a	20.5 a
6 March 2018 *	21	16	15	21
21 November 2018 *	21.5	15	16	21

NF: native forest, PR: radiata pine, PRM; mix radiata pine and NF. *: Sample at the source of headwater. a/b: The same letter means no significant difference within MTT per model and per catchment.

For Q2, tritium analysis in 2018 indicated that the water at the stream source was about half the age of the water at the stream outlet. This indicates that water from longer flow paths enters the stream along its length [59]. Q1 had a similar MTT of 8.5 years at extremely low (March 2013 and May 2017) and at high flow. In 2018 the water was slightly older. Q2 had older water of about 18 years, except in October 2017 when it was younger than the 2018 samples.

Additionally, looking at mean MTT against the sampling month shows a decline in MTT from during the wet months and an increase in the dry season (Figure 4). Moreover, we see that MTT in Q2 has always been greater (around two times) than in Q1 and about 1.5 times that in Q3.

The Skott-Knott (SK) test showed no significant differences (p-value < 0.05) between water ages pre- and post-fire in all three catchments with all four LMP models (Table 8).

Within a catchment the was no clear relationship between the mean transit time and either BFI or RC (Figure 5). For example, in Q1 and Q3 when RC increased after the fire the mean MTT increased while in Q2 mean MTT did not change in the first year after fires. In the BFI in Q1, the more baseflow meant older water, but there is not a clear trend in BFI in Q2/Q3 with similar ages pre- and post-fire. However, there does seem to be a relationship between MTT and BFI across catchments, with catchments with more baseflow having a higher MTT, such as Q2.

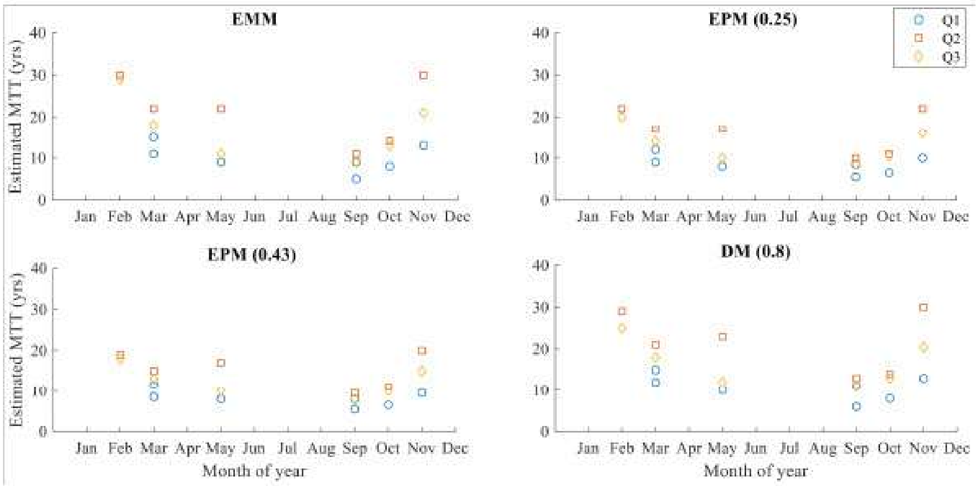

Figure 4. Mean transit times by month for each of the four models.

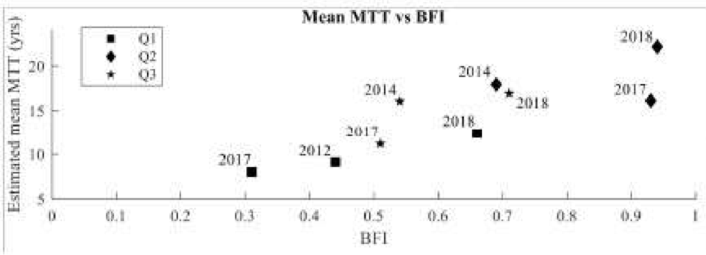

Figure 5. *Cont.*

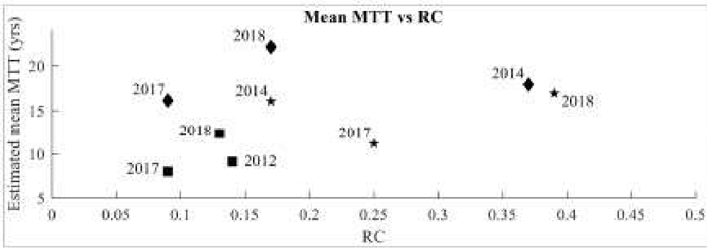

Figure 5. Baseflow Index (BFI) and Runoff Coefficient (RC) versus Mean MTT and TU. Mean MTT was calculated averaging the two samples of 2017 and 2018 as a baseflow annual MTT. For Q1, we took as pre-fire the 2012 samples; for Q2/Q3 the 2014 pre-fire samples were averaging as pre-fire MTT.

4. Discussion and Conclusions

In this paper, we used a combination of streamflow, rainfall, and tritium data from three catchments to test whether the Mean Transit Time of water through these catchments was unaffected by the 'Las Máquinas' wildfire of late January 2017. This was proposed despite previously reported changes in the peak and low flows in at least one of the catchments [32]. This expectation was based on two pieces of evidence. Firstly, the pore flow pathways were not affected by the fire (e.g., [60]) and, secondly, the peak and low flows reflect the celerity of water travel while the MTT is determined more by the bulk velocity of water movement [31]. The data support this hypothesis and indicate that the effect of the fire on streamflow magnitude varied between the three catchments, but that MTT was fairly constant or even increased slightly. The poor relationship between measures of flow and the MTT of water for the Quivolgo catchments and the observation that MTT was not greatly changed after the fire suggests that, for these catchments, fire has affected storage and celerity but has not affected the velocity of water in these catchments. This change in the balance between celerity and velocity likely results from changes in infiltration, storage, and drainage in these catchments.

The megafire burned a significant amount of litter and humic soil within the *N. glauca* forest in the Quivolgo area [61]. These findings together with the fact that rainfall is not generating high peak flows in Q2 suggest that deep soil infiltration may have been enhanced within the first year after the fire. This enhancement may be due to soil cracking during the drought [62], particularly after the removal of the leaf cover, and/or to the rapid development of vacant old root channels as preferred infiltration pathways after the fire. (e.g., [63,64]). Additionally, according to [20], evapotranspiration from Q2 accounts for a small proportion of rainfall and, because streamflow has not increased, there is a high probability of this unaccounted water being stored deeper in the soil where it is mixing with stored water but not yet affecting the average water age.

The peak flow in Q2 after the fire was much lower than Q1 and Q3, and much lower than before the fires. In [20] it was shown that after evapotranspiration of native forest in Q2 was accounted for, the reduced streamflow still left a net unaccounted water balance of 915 mm in 2017 and 421 mm in 2018. This suggests that the system has changed to one in which flow has decreased, even though the rainfall increased. It is an unusual behavior but is not without precedent (e.g., [64,65]). In [20], it was hypothesized that before the fire, infiltration had been blocked by a combination of filled root channels and very dry soil, whereas after the fire these potential preferred pathways opened, increasing the infiltration to the deep fractured rock system.

Runoff and water yield seem not to have an evident association with MTTs. In Q1, where runoff and water yield were not as high as Q2 before the fire, indeed virtually zero in 2015 and 2016, MTTs were shorter than the native forest. MTT in Q3 was similar to Q2 both before and after the fires (Table 6) and more than 50% greater than Q1. Stormflow response was greatest in Q3, before the fires, and was 1.5 times that of Q2. After the fires, when all catchments were flowing, the Q1 response to the storm was 300 times that of Q2, and Q3 was 155 times that of Q2. It may be that catchment volume and size is playing an important role in water age and streamflow. Q2 is the biggest catchment, followed by Q3, which is one explanation for why transit times are shortest in Q1 (the smallest catchment) and longest in Q2. However, it is out of our study scope, and in the future, we plan to explore catchment geomorphology effects on MTT.

After the fires, MTT in Q1 increased. However, the native forest catchment, Q2, MTT seems unaffected, but runoff and water yield were reduced. In Q2, the lowest monthly streamflow in the 2009–2016 period was 9.6 mm in 2014. However, the lower monthly streamflow post-fires were 12.0 mm in February 2017, indeed stream flow seems less variable through the seasons. Precipitation occurs mainly in winter, when the main species (*N. glauca*) drops all its leaves, therefore, more water gets into the soil, there is lower evapotranspiration and, while we would expect more water to be stored in the soil [66], often this is also the period of highest streamflow.

After fires, in Q1 and Q3, water ages increased which may be due to (1) more new water infiltrating deeper in the regolith and not passing through the soil matrix to the stream [67], or (2) new water entering the system pushing old water within the macropores to streamflow [68,69]. Thus, two different mechanisms could produce a similar result in the stream. Peak flows increase following a fire are often observed and usually ascribed to increased hydrophobicity and lower ET; in our case, this occurred in Q1 and Q3 but not Q2. For instance, Q3 BFI in 2015 and 2016 was 0.69 whereas in 2017 it was 0.51 which means half of the flow was surface runoff. Particularly in Q1, in the first-year post-fire surface runoff was almost 70% of the total flow, while in the second-year post-fire that surface flow dropped to only 34% of streamflow. However, in the first year after the fire, the Q1 catchment started to flow in May–June 2017 in a year with above the average rainfall (1400 mm), and some soil disturbing phenomena such as hydrophobicity, soil sealing, hyper-dry conditions, could also explain the high runoff. Nonetheless, the second-year post-fire BFI increased suggesting new water displacing older water to the stream.

An earlier study at these sites in 2015 [70] found transit times between 9 and 15 years (Q1 and Q2). In Q1 we have not found a significant difference in MTTs over time, however, this catchment went dry for a period, and we might expect older water in the stream. However, there is still little difference between MTTs before and after fires.

With almost 20 ha of native forest, Q3 seems to behave similarly to Q2, as we can see in the similar MTTs in summer 2014 and no statistically significant differences between these two catchments. This might be due to the similarities in the native forest composition and cover in Q3 and Q2, particularly in the wet zones near the streams. Moreover, in the spring of 2014 within all catchments, MTTs were at their lowest in the study period. In this pre-fire sampling, 1250 mm of rain had fallen in that year, the second wettest after the 2017 rainfall and it is possible that such an amount of water has mixed with older water decreasing mean water age.

When comparing source and outlet samples, the mature native forest (Q2) catchment had the greatest differences in MTTs, while Q1 and Q3 had no significant differences. In Q2, water sampled at the outlet had almost double the age of samples from the stream source in all LPMs results, despite the samples taken no more than 600 m apart. In 2018, the source sample MTT is about 8.5–11 years old, but the sample at the outlet is about 22–15 years old (Table 6). In Q1, we could not identify a difference between the age of source and outlet samples. A mixing process might be occurring in the soil, where new water is being mixed with the old due to an increase in infiltration rates. However, there could also be a bias of the methods with a probability of underestimating the water age due to limitations of the LPMs [71].

Cartwright and Morgenstern [39] found that the MTT differences between catchments can be related to the pattern of evapotranspiration. However, in this study, it was not possible to measure evapotranspiration after the fire within the pine catchments and was possible only in the native forest [20]. After fires, both Q1 and Q3 have increasing runoff

(Table 3), and water is getting older probably because after the initial increase in surface runoff component (BFI reducing) new water is entering the soil matrix displacing older water as seepage to the stream. In Q2, water flowing out at the source is around half the age of the stream water at the outlet, which could reflect longer flow paths in this catchment, a more complex groundwater flow entering along the stream at different points, or simply more subsurface flow entering the stream. Different water ages in samples from source and outlet introduce uncertainties in the calculations of MTTs [72].

Wildfires are a well-known disturbance that affects the hydrologic cycle. With an expectation of more fires in the future [73], how these fires affect the hydrology of a site will be of increasing concern, especially in drier zones such as the Mediterranean coastal zone of Chile. Age tracers, such as tritium, can help us understand how water moves within the different flow pathways within a catchment [74]. After 10 years of drought and with a mega-fire in 2017, MTTs have not materially changed. Sampling both the source and at the outlet, showed essentially no difference in MTT in Q1, a little difference in Q3, and a significant difference in Q2. Within the following years from this study, a sampling schedule to continue to investigate both the long-term drought and the effect of wildfire in these catchments will be maintained.

Finally, to improve our knowledge of our sites and in conjunction with hydrometrics and tritium sampling, a geophysical sampling schedule (e.g., electrical resistivity or seismic waves) and boreholes should be performed in order to check water table fluctuations and their relationship with water age and other hydrologic variables. Moreover, a sampling schedule for other isotope sampling (i.e., deuterium and oxygen-18) and a continuum tritium sampling in water and rain schedule (days or monthly) to increase MTT parameterization within sites should be performed in the future.

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CAPÍTULO IV: Cooperación interinstitucional para desarrollar un nuevo enfoque hidrológico forestal para apoyar la silvicultura sostenible en Chile[3]

Francisco Balocchi, José Luis Arumi y Andrés Iroumé

4.1. Resultados claves

- Existe la necesidad de un ente que agrupe a las diferentes instituciones y/o grupos de investigación ligados a la hidrología forestal

- No existe un financiamiento base o concursos dedicados a la investigación hidrológica

- Existe una alta calidad de investigadores en Chile, pero las investigaciones son dispersas y muchas similares

4.2. Resumen en extenso

El rol de los bosques sobre el ciclo hidrológico es conocido, sin embargo, no ampliamente estudiado en Chile. Si bien el primer artículo científico que estudia la escorrentía en cuencas es del año 1975 (Jones *et al.*, 1975), solo 75 estudios existen a la fecha relacionados con cantidad y calidad de agua que involucren cambio de uso de suelo (Balocchi *et al.*, 2022a). En Chile 17 millones de hectáreas de bosques y más de 3 millones de plantaciones juegan un papel importante en el ciclo hidrológico y su balance, especialmente desde la zona centro norte al sur. Bajo este escenario, se requiere contar con un nuevo impulso en las investigaciones ligadas a la hidrología forestal, tanto desde el Estado como del sector privado, así como la reforma a las políticas públicas, articulando las investigaciones en hidrología forestal con un imperativo sentido de integralidad.

Este documento presenta una propuesta de desarrollo de la investigación en hidrología forestal con un enfoque de integralidad y descentralización, cuyo sustento es la alta capacidad del capital humano científico en Chile. La propuesta se centra en la creación de un ente técnico público/privado que sea

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capaz de articular la discusión entre los actores del sector desde un punto de vista científico/técnico y desde ahí a políticas públicas. Sin embargo, no existe financiamiento de este tipo de iniciativas por lo que se propone crear un fondo público-privado que sea capaz de financiar proyectos de investigación para el apoyo o creación de políticas públicas en el país. Particularmente, el financiamiento de sitios de monitoreo a largo plazo no existe a nivel público.

Si bien, este artículo de opinión no apunta directamente a aceptar (o no) las hipótesis de trabajo, discute la problemática del sector forestal desde la cantidad y calidad agua; desde donde una proporción importante de población usa agua desde cuencas forestales (bosque nativo y plantaciones) (*e.g.*, Constitución y Coronel) y que esto no está considerado en llamados a fondos concursables de investigación, ni tampoco ha sido volcado a políticas públicas concretas con el fin de aumentar la seguridad hídrica de la población.

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Cooperación interinstitucional para desarrollar un nuevo enfoque hidrológico forestal para apoyar la silvicultura sostenible en Chile

Inter-institutional cooperation to develop a new forest hydrological approach to support sustainable forestry in Chile

Francisco Balocchi [a,b,*], José Luis Arumí [b,c], Andrés Iroumé [d]

*Autor de correspondencia: [a] Bioforest SA, Ecosistema, Productividad y Cambio Climático, camino a Coronel s/n km 15, Coronel, Chile, francisco.balocchi@arauco.com/frabalocchi@udec.cl

[b] Universidad de Concepción, Programa de Doctorado en Recursos Hídricos y Energía para la Agricultura, Chillán, Chile.

[c] Universidad de Concepción, Facultad de Ingeniería Agrícola, Departamento de Recursos Hídricos, Centro FONDAP CRHIAM, avenida Vicente Méndez 595, Chillán, Chile.

[d] Universidad Austral de Chile, Facultad de Ciencias Forestales y Recursos Naturales, Valdivia, Chile.

SUMMARY

There are more than 4 billion hectares of forests in the world, which corresponds to 31 % of the surface of our planet. In Chile, 17 million ha of forests cover the country where more than 3 million correspond to plantations. The role of vegetation cover in the dynamics of precipitation, in the generation of runoff and in the protection and conservation of degraded land is known. This cover plays an important role in the hydrological cycle and its balance, especially in the north central zone to the south. A new impulse is required in research, both from the Government and the private sector, as well as reformation to public policies, thus articulating research in forest hydrology with an imperative sense of integrality. Therefore, this document presents a proposal for the development of research in forest hydrology with an integral and decentralization approach. Highly capable scientific human capital in Chile, organized in geographically defined nuclei, would be capable of answering relevant research questions in forest hydrology that would support public policies based on scientific evidence. All this, under the auspices of a technical integrating entity to allow a fluid conversation and discussion of the actors of the forestry sector, both in research matters and in the generation of public policies from an integral and multidisciplinary perspective at the catchment level.

Keywords: water resources, plantations, native forest, stakeholders, forest hydrology.

RESUMEN

Las más de 4 mil millones de hectáreas de bosques en el mundo corresponden al 31 % de la superficie planetaria. Chile posee 17 millones de hectáreas de bosques y más de 3 millones corresponden a plantaciones. El rol de la cobertura vegetal en la dinámica de la precipitación y en la generación de escorrentía, y en la protección y conservación de suelos degradados, es conocido. Esta cobertura juega un papel importante en el ciclo hidrológico y su balance, especialmente desde la zona centro norte al sur. Se requiere de un nuevo impulso en las investigaciones, tanto desde el Estado como del sector privado, así como la reforma de las políticas públicas, articulando las investigaciones en hidrología forestal con un imperativo sentido de integralidad. Este documento presenta una propuesta de desarrollo de la investigación en hidrología forestal con un enfoque de integralidad y descentralización, cuyo sustento es la alta capacidad del capital humano científico en Chile, que organizándose en núcleos definidos geográficamente sería capaz de responder las principales preguntas de investigación en hidrología forestal para sustentar políticas públicas basadas en evidencia científica. Todo esto, bajo el alero de un ente integrador técnico que permita una conversación y discusión fluida de los actores del sector forestal, tanto en materias de investigación como en la generación de políticas públicas, desde una mirada integral y multidisciplinaria a nivel cuenca.

Palabras clave: recursos hídricos, plantaciones, bosque nativo, tomadores de decisión, hidrología forestal.

INTRODUCCIÓN

Las primeras cuencas experimentales establecidas para estudiar el efecto del bosque en la hidrología se implementaron a comienzos del 1850 (Andreassian 2004), debido a la observación de que el bosque podía modificar el clima, y con ello el caudal. Luego de décadas de investigación, esta preocupación sigue vigente, ya que no solo es necesario saber cómo el bosque influye en el caudal, sino que también es necesario saber cuánto y cómo condiciones entre cuencas, clima, y diferentes tipos de bosque y su manejo afectan los cambios (*e.g.* Pinchot 1905). A lo anterior se suma la dinámica del cambio global (cambios en el uso del suelo, agua y clima), que se ha acelerado en las últimas décadas y que crecientemente estresa los procesos hidrológicos, que obliga a una visión a largo plazo en cuanto a mediciones en campo y con ello la ciencia detrás para su análisis. Los efectos del cambio climático en la hidrología pueden alterar el retorno financiero de plantaciones, pero principalmente la escasez de agua que impactaría a las comunidades locales.

En Chile, la primera mención del efecto de los bosques y las lluvias se encuentra en una Carta que Lord Cochrane envió a Bernardo O´Higgins en 1820 luego de liberar Valdivia. Pero, los estudios relacionados con la hidrología forestal comienzan en el siglo XX (Cabaña-Chávez *et al.* 2013) y desde entonces se han intensificado (Jones *et al.* 2016). Posteriormente, Jones *et al.* (1975) publican el primer artículo científico que presenta y discute el caudal de tres cuencas pequeñas en Chile Central. La época de mayor productividad científica fue el año 2021 con 13 estudios relacionados con la producción de agua y plantaciones, y producción de sedimentos desde cuencas forestales. Se suman a lo anterior los esfuerzos de la Unidad de Cuencas de la Corporación Nacional Forestal (CONAF) con el plan de ordenación de cuencas en Magallanes en 1974, los proyectos CONAF-JICA, CONAF-DF-ID (modelo SHETRAN), y algunas cuencas experimentales de mediados de 1995 (Cabaña-Chávez *et al.* 2013). Estos estudios fueron financiados principalmente por programas de las Naciones Unidad como la FAO, PNUD y CEPAL, y el Banco Interamericano de Desarrollo (BID), además del apoyo de organizaciones nacionales como la Dirección General de Aguas (DGA) y la Comisión Nacional del Medio Ambiente (CONAMA), y organizaciones internacionales como la German Technical Cooperation, la Overseas Development Agency de Gran Bretaña.

A pesar de que el sector forestal es la segunda actividad económica más importante en Chile, que aporta 4,9 mil millones de pesos al PIB y con 3,1 millones de hectáreas plantadas entre las regiones de O´Higgins y Los Lagos (INFOR 2021), no existe financiamiento público o líneas de investigación ligadas directamente con la hidrología forestal. El Instituto Forestal (INFOR), institución de investigación dependiente del Ministerio de Agricultura y su programa Agua y Bosque, es la única institución que aborda la hidrología forestal desde lo público, no universitario.

Uno de los pocos ejemplos de concursos orientados al sector forestal es el Fondo de Investigación por el Bosque Nativo (FIBN) de la CONAF. Con montos bajos y plazos acotados, que no permiten mantener proyectos de largo plazo como son los estudios hidrológicos. Otros programas nacionales como el Fondo de Financiamiento de Centros de Investigación en Áreas Prioritarias (FONDAP) y Centros I+D de la Corporación de Fomento de la Producción (CORFO) apuntan a otras actividades productivas como la agricultura y la minería, pero no han incluido el sector forestal. Es posible que esto se deba a que la cobertura forestal no necesita de derechos de aguas, por lo tanto, desde la mirada nacional, deja de ser atractivo, a pesar de la influencia de los bosques en el ciclo hidrológico. Además, sería importante incluir al sector forestal, porque influye en otros servicios ecosistémicos y, por lo tanto, puede incidir en la sostenibilidad de los territorios y sus conflictos sociales.

Al no existir un fondo público específico, desde la necesidad de conocer el efecto de las plantaciones en el ciclo hidrológico y también desde los requerimientos de las certificaciones forestales, el sector privado ha financiado programas de investigación. Por ejemplo, Forestal Arauco a través de Bioforest, posee una red de 16 cuencas experimentales entre las regiones del Maule y de Los Ríos, en donde algunas de ellas se monitorean desde el año 2008 (Balocchi *et al.* 2021). Forestal Mininco posee 18 cuencas experimentales localizadas en las comunas de Nacimiento, Los Ángeles, y Coronel. Algunas de estas cuencas se monitorean desde el año 2008 y se encuentran principalmente cubiertas con plantaciones forestales (*e.g.* Iroumé *et al.* 2021). Finalmente, MASISA inició el año 2008 el monitoreo de 6 cuencas forestales, programa que actualmente se ha discontinuado. Desde el sector privado, la información recogida se debería compartir, analizar y evaluar con el fin de generar una discusión compartida aunando esfuerzos con la comunidad, la academia, el sector privado, y el Estado.

Ha existido una fragmentación de estudios en el país, con diferentes objetivos y no se ha buscado complementariedad y/o sinergias entre las instituciones. Además, la inversión estatal en investigación en hidrología forestal no posee un correlato con la importancia del sector. Si bien en el año 2017-2018 la entonces Comisión Nacional de Investigación Científica y Tecnológica (CONICYT, hoy la Agencia Nacional de Investigación y Desarrollo de Chile, ANID) definió entre sus ejes estratégicos al área de los recursos hídricos como prioritarios, una búsqueda en su base de datos (ANID, 2021) utilizando palabras claves como hidrología, erosión, uso de suelo, recursos hídricos, sedimentos, cambio de uso, plantaciones, pino insigne y eucalipto, arrojó que entre el 2001 y 2019 se adjudicaron 32 proyectos asociados a estas palabras clave de un total de 23.383 proyectos aprobados (se incluyeron todos los instrumentos CONICYT/ANID). Sin embargo, el plazo de estos proyectos es acotado, generando la necesidad de aumentar el horizonte con el fin de recolectar la mayor cantidad de información y variabilidad del sitio en estudio y evaluar el efecto del cambio climático a largo plazo (Garreaud *et al.* 2020).

La relevancia del sector forestal en nuestro país con respecto a la producción y conservación de agua para consumo humano es poco conocida o desestimada. Un análisis realizado con la ubicación de las captaciones de Agua Potable Rural (APR) en las regiones de la Araucanía y Los Lagos (información solicitada por la Ley 20285 de Acceso a la Información Pública) sobrepuestas sobre el mapa de usos de Zhao *et al.* (2016) de 287 captaciones (superficiales y subterráneas), determinó que el 41 % de las ubicaciones de APR se encuentra en cuencas con algún uso con cobertura vegetal no agrícola (matorral, bosque nativo, plantaciones y praderas). Otro ejemplo de lo anterior es el de Forestal Arauco, que mantiene en sus predios alrededor de 1.200 captaciones y estructuras anexas (879 de estas captaciones son superficiales) distribuidas en su patrimonio, abasteciendo alrededor de 630.000 personas y donde es la propia empresa que facilita su gobernanza. Forestal Mininco, por otro lado, administran 352 bocatomas a lo largo de su patrimonio. Aun así, el sector forestal no está considerado en las discusiones de políticas públicas, y participa de forma esporádica y no activamente de las acciones gubernamentales, a pesar de su disponibilidad para integrase a estas discusiones. Por ejemplo, Nuñez (2004) estimó un valor entre \$11 - \$25 el m^3 de agua desde la cuenca de Llancahue, en Valdivia, cubierta principalmente de bosque nativo, lo que se traduce en \$74.971 - \$170.389 por ha de bosque nativo. Por otro lado, en muy pocas de las diversas mesas del agua existentes participan representantes del sector forestal. Un caso de lo anterior es la Mesa Nacional del Agua (MOP 2019), en donde no participa el sector forestal a pesar de la importancia, rol e influencia de la cobertura vegetal en el ciclo hidrológico. Tampoco participan miembros de la sociedad civil relacionados con el recurso hídrico, ni universidades o centros de excelencia, encontrándose una alta participación de gremios productivos principalmente ligados al sector agrícola (*e.g.* Juntas de Vigilancia). Entonces, existe una desconexión de la ciencia (revisada por pares) desde el sector forestal con el Estado e incluso desde la misma ciencia con el sector.

Se debe destacar el capital humano presente en el país con investigadores altamente capacitados. Esta capacidad abre la posibilidad de lograr mayor eficacia y eficiencia en el impacto de la ciencia. Hoy existe una serie de temáticas similares abordadas por diferentes universidades o centros de investigación, pero es poco común que se aúnen esfuerzos relacionados con la hidrología forestal.

El sector forestal público y privado tiene un rol relevante en la adaptación al cambio climático, la provisión de agua y otros múltiples servicios ecosistémicos en Chile. Si bien por ahora, no hay conflictos entre usuarios con derechos de aguas, desde la protección del suelo y de los recursos hídricos, los servicios ecosistémicos y lo social, hacen sumamente relevante que el sector sea parte de las discusiones y de las políticas públicas. Así, es necesario que este sector participe activamente y con investigaciones diseñadas para construir una agenda efectiva de políticas públicas.

La preocupación por parte de las comunidades sobre la provisión de agua y el cambio climático, la apertura de actores clave (*i.e.* privados), un cambio de gobierno, y el desarrollo de importantes investigaciones parecen hacer de este un momento crítico para abordar los problemas del agua y su relación con la cobertura vegetal en Chile. Por ejemplo, lo reportado por Pizarro *et al.* (2022) en grandes cuencas donde el uso mixto de cobertura vegetal (bosque nativo y plantaciones) parece ser una alternativa viable frente al cambio climático, o que, en un ejercicio de modelación, las diferencias entre caudales de escenarios de bosque nativo prístino y plantaciones son mínimas (Gimeno *et al.* 2022).

Todos estos factores mencionados abren la necesidad y oportunidad para un nuevo enfoque hidrológico forestal. La situación es muy adecuada para adoptar un enfoque de gestión adaptativa con la participación de todos los sectores: ciencia, industria, gobierno, comunidades y organizaciones no gubernamentales.

PROPUESTA DE UNA INVESTIGACIÓN DESCENTRALIZADA Y COOPERATIVA

Dada la experiencia de la hidrología forestal nacional, se hace necesario hacer ciencia coordinada, aprovechando el capital humano avanzado presente en universidades, centros de investigación y el sector privado. Existen sitios de estudio en Chile en donde se puede hacer investigación multidisciplinaria y con ello aportar a la discusión del efecto de la cobertura vegetal sobre los recursos hídricos. Sin embargo, esto conlleva a un compromiso tanto de los investigadores como del Estado, por cuanto estos estudios son de largo plazo y necesitan financiamiento. Un ejemplo interesante es la red de bosques y cuencas experimentales establecidos por Estados Unidos (EE. UU.) en los primeros años del siglo XX, donde desde los años 1930s se han realizado múltiples experimentos a largo plazo y a escala de cuencas para analizar y evaluar los efectos de diversas prácticas forestales en la hidrología (Lugo *et al.* 2006, Adams *et al.* 2008). Desde 1980, los bosques experimentales del Servicio Forestal han recibido financiamiento del programa LTER de la National Science Foundation de los EE. UU. (Lugo *et al.* 2006), lo que permitió agregar estudios detallados de clima, suelo y cambios de vegetación y sus efectos (Peters *et al.* 2011), disponibles a la sociedad (USDA-FS 2021).

En Chile, dado que existen variadas cuencas en estudio, una red de cooperación interinstitucional formal parece ser una buena estrategia (ejemplos de éxito en EE. UU. se describen en Driscoll *et al.* 2012 y Swanson *et al.* 2021). Esta red concentraría a los investigadores que se encuentran cercanos a los sitios de estudio, descentralizando las investigaciones en determinadas universidades, pero sin perder la cooperación. Además, esto permitiría centralizar el rescate de datos hidrológicos en una única plataforma con el fin de globalizar datos y estudios, como lo es, por ejemplo, la red experimental de cuencas del United States Department of Agriculture - Agricultural Research Service (Goodrich *et al.* 2021). Luego, sería posible extender el conocimiento generado para aportar a la creación de políticas públicas y nuevos programas de investigación. Esta red, sumado a la ya reconocida experiencia de la ciencia chilena, creemos que pueden promover nuevos bríos en la investigación hidrológica nacional. Para ello, debiese existir un foco inicial en (i) cantidad de agua y su relación con las diferentes coberturas y (ii) calidad de agua como punto de partida, desde una mirada multidisciplinaria. Esto permitiría obtener resultados no solo para políticas públicas, sino que para la comunidad y sector privado, con el fin de garantizar un balance con respecto a la captura de carbono, provisión de agua y de productos forestales (*i.e.* pulpa).

Sin embargo, es necesario crear un ente integrador técnico que permita una conversación y discusión fluida de los actores del sector forestal, tanto en materias de investigación como en la generación de políticas públicas desde una mirada integral y multidisciplinaria a nivel de cuenca. Su mayor foco sería la investigación en el sector forestal, por lo tanto, debiera depender del Ministerio de Ciencia y Tecnología. Este ente debería ser capaz de traducir los resultados de la investigación a recomendaciones o incentivar la discusión informada de futuras políticas públicas. Un caso similar, que puede ser un ejemplo, es el Cooperative Research Centre for Catchment Hydrology establecido en Australia en los años noventa y cuyo fin fue crear un sistema de apoyo para decisiones en la hidrología con foco en la cuenca. Además de la creación de este ente, se debería crear un consejo consultivo integrado por los usuarios del agua, es decir, sociedad civil, empresas sanitarias, empresas forestales, universidades y el Estado, a través de la ANID o de la CORFO, y de los diferentes estamentos públicos que tienen relación con los recursos hídricos.

El financiamiento debería venir de variadas fuentes, pero principalmente desde el sector público. Actualmente, el sector privado posee líneas propias de investigación en hidrología forestal autofinanciadas. Por lo tanto, debería existir una interacción entre financiamiento público (desde la ANID o desde CORFO) con el fin de evitar la duplicidad de esfuerzos.

CONCLUSIONES

Existe en Chile una capacidad instalada de investigadores de alto nivel. Sin embargo, la investigación no tiene una guía clara del Estado, por lo que es difícil aportar a la discusión de políticas públicas sin un eje nacional definido. Tampoco se visualiza una estrategia desde los investigadores que ayude al Estado a definir materias específicas para un territorio sustentable. Así, se presenta una recomendación de acción para poder incentivar el apoyo del Estado a la investigación ligada a la hidrología forestal, además de una estrategia con respecto a cómo se pueden desenvolver espacialmente las diferentes instituciones, sin perder la cooperación interinstitucional. Creemos que esto es posible hacerlo en Chile con (i) la formalización de experimentos a largo plazo en las empresas forestales y grupos públicos y privados con patrimonio para estudiar la interacción de las coberturas vegetales en el ciclo hidrológico, con (ii) la capacidad para diseñar e implementar proyectos de investigación con el apoyo de la academia en conjunto con el sector privado, la comunidad (entregando herramientas y educando con respecto al rol de la cobertura vegetal), el Estado (ANID) y las ONGs, y (iii) el Estado, el cual tiene la capacidad de promulgar leyes basadas en los resultados de cooperación aquí propuestos. Esto conlleva a la creación de un ente integrador de las investigaciones que sea capaz de comunicarse con la academia y los tomadores de decisión para una gobernanza del agua integrada e informada.

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CAPÍTULO V: CONCLUSIONES GENERALES[4]

5.1. Caudales recesivos y usos de suelo en Chile centro-sur

El análisis de los caudales recesivos permitió establecer que no existen diferencias significativas en la magnitud de los caudales recesivos a nivel de microcuenca y en un gradiente latitudinal, y en coberturas de bosque nativo, *P. radiata* y *Eucalyptus spp.* en invierno. Asimismo, el promedio histórico de estos caudales no presenta diferencias entre coberturas. Sin embargo, en época estival, es donde se evidencian las mayores diferencias entre coberturas y gradiente latitudinal de precipitación (Balocchi *et al.*, 2021b). Por lo tanto, en el futuro sería necesario medir otras variables, especialmente aquellas relacionadas con la evapotranspiración de los diferentes usos y su relación con la humedad de suelo. Asimismo, es necesario investigar con mayor detalle el acuífero de cada cuenca. Con esto, la elección del modelo recesivo y el análisis posterior de los caudales recesivos garantizarían un manejo con mayor exactitud de los recursos hídricos de estas cuencas.

A pesar de que no fue posible determinar con exactitud la diferencia entre usos en la época estival, el coeficiente de agotamiento como variable principal de la modelación de caudales recesivos, permite abordar la modelación hidrológica con mayor detalle ya que representa características del acuífero. Por lo tanto, es importante para el manejo de recursos hídricos en un contexto de cambio climático por cuanto esta metodología está ligada al proceso de liberación del almacenamiento de agua desde el acuífero.

5.2. Efectos del mega incendio 2017 en la hidrología de cuencas forestales

Las métricas seleccionadas (*e.g.*, índice de flujo base y el coeficiente de escorrentía) permitieron identificar cambios importantes en la escorrentía superficial y en los patrones de infiltración del agua en la cuenca con bosque nativo que no suceden usualmente luego de un incendio. Estos cambios inusuales identificados en la cuenca con bosque nativo se deben a que no aumentaron los caudales punta y que la escorrentía superficial disminuyó, lo que permite concluir que existe un aumento de la infiltración en el suelo y confirmar la conclusión de Balocchi *et al.* (2020).

Por otro lado, el uso del radioisótopo Tritio como indicador de cambios en una cuenca y dado los tiempos de residencia encontrados en este estudio, no permitieron agregar información al impacto del mega incendio en la hidrología de las cuencas en estudio en un corto-mediano plazo del efecto de esta alteración antrópica. Por lo tanto, en futuras investigaciones sería necesario agregar el análisis de otros

[4] Las referencias utilizadas en este capítulo se encuentran en el numeral 1.7 de este mismo documento.

trazadores naturales como isotopos estables (*e.g.*, deuterio), isotopos radiogénicos (*e.g.*, radioisótopos de gases nobles), etc. (Cartwright y Morgernstern, 2016).

5.3. La hidrología forestal en Chile y futuras líneas de investigación

Como se expuso en el capítulo IV, existe la necesidad de contar con un ente unificador de no sólo ideas si no que se produzcan sinergias entre instituciones estatales, privadas y la academia (Balocchi *et al.*, 2022c). Esto, por cuanto la hidrología forestal en el país es dispersa, concentrada en algunos sectores, pero principalmente no está consensuada desde los pocos grupos que trabajan en este sector productivo. A pesar de la dispersión de las investigaciones y la concentración de estudios en el sur de Chile, la fortaleza principal del sector es el alto nivel académico de los investigadores, situación que permitiría re-enfocar esfuerzos en pro de políticas públicas apoyadas desde la ciencia.

Para ello, no solo es necesario el apoyo económico de instituciones como la Agencia de Investigación y Desarrollo (ANID), sino que es necesario cubrir algunas temáticas para poder realizar recomendaciones hacia un territorio productivo sustentable. Algunas de estas temáticas corresponden a la falta de detalle sobre los suelos forestales en Chile centro-sur, la relación aguas superficiales-subterráneas y la relación del complejo atmósfera-suelo-vegetación (Balocchi *et al.*, 2022a).

Con respecto a futuras líneas de investigación, a pesar de la conocida robustez de las curvas recesivas para el entendimiento de cómo funciona el acuífero y la liberación de agua al cauce, es necesario complementar con modelos hidrológicos con el fin de (i) identificar el impacto del cambio de uso de la tierra y (ii) el efecto que el cambio climático puede tener en la generación de caudales (principalmente estivales). Avances en estas líneas son la modelación híbrida con modelos físicos y redes neuronales (Balocchi y Rivera, 2022d) y ajuste de modelos con nuevas metodologías de parametrización (Balocchi y Moltedo, 2022e).

Con respecto al efecto de los incendios es necesario (i) identificar las brechas en la investigación hidrológica en este ámbito, por cuanto existen reducidos trabajos relacionados con esta temática (*e.g.*, Balocchi, Corti y Castillo, 2023) y (ii) continuar con el monitoreo continuo de cuencas y aumentar estas cuencas con diferentes coberturas ya que existe una falta de cuencas cubiertas con bosque nativo.