Journal of Hydrology 616 (2023) 128681

Contents lists available at ScienceDirect

Journal of Hydrology

journal homepage: www.elsevier.com/locate/jhydrol

Review papers

Forest hydrology in Chile: Past, present, and future

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ARTICLE INFO

This manuscript was handled by Marco borga, Editor-in-Chief, with the assistance of Daniele Penna, Associate Editor

Keywords:
Exotic plantations
Chilean native forests
Sediment yield
Water yield
Land use planning

ABSTRACT

This paper reviews the current knowledge of hydrological processes in Chilean temperate forests which extend along western South America from latitude 29° S to 56° S. This geographic region includes a diverse range of natural and planted forests and a broad sweep of vegetation, edaphic, topographic, geologic, and climatic settings which create a unique natural laboratory. Many local communities, endangered freshwater ecosystems, and downstream economic activities in Chile rely on water flows from forested catchments. This review aims to (i) provide a comprehensive overview of Chilean forest hydrology, to (ii) review prior research in forest hydrology in Chile, and to (iii) identify knowledge gaps and provide a vision for future research on forest hydrology in Chile. We reviewed the relation between native forests, commercial plantations, and other land uses on water yield and water quality from the plot to the catchment scale. Much of the global understanding of forests and their relationship with the water cycle is in line with the findings of the studies reviewed here. Streamflow from

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https://doi.org/10.1016/j.jhydrol.2022.128681
Received 21 February 2022; Received in revised form 14 September 2022; Accepted 20 October 2022
Available online 9 November 2022

forested catchments increases after timber harvesting, native forests appear to use less water than plantations, and streams draining native forest yield less sediment than streams draining plantations or grassland/shrublands. We identified 20 key knowledge gaps such as forest groundwater systems, soil–plant-atmosphere interactions, native forest hydrology, and the effect of forest management and restoration on hydrology. Also, we found a paucity of research in the northern geographic areas and forest types (35-36°S); most forest hydrology studies in Chile (56%) have been conducted in the southern area (Los Rios Region around 39-40° S). There is limited knowledge of the geology and soils in many forested areas and how surface and groundwater are affected by changes in land cover. There is an opportunity to advance our understanding using process-based investigations linking field studies and modeling. Through the establishment of a forest hydrology science "society" to coordinate efforts, regional and national-scale land use planning might be supported. Our review ends with a vision to advance a cross-scale collaborative effort to use new nation-wide catchment-scale networks Long-term Ecosystem Research (LTER) sites, to promote common and complementary techniques in these studies, and to conduct transdisciplinary research to advance sound and integrated planning of forest lands in Chile.

1. Introduction

In Chile, forests occupy 17.7×10^6 ha or 23 % of the country and presently comprise 3.1×10^6 ha of exotic plantations and 14.6×10^6 ha of native forests (Ministry of Agriculture, 2021). This represents about 0.4 % of the more than four billion hectares of global forests (Food and Agriculture Organization, 2020), approximately 278 million hectares of which are commercial plantations (Payn et al., 2015). Since the mid 20th century, plantation forests for wood production have profoundly changed the landscapes across all permanently inhabited continents. The current land cover of central Chile is the result of a long history of food and wood production in both the pre- and post-Columbian periods.

The expansion of timber plantations commenced in the 1970′s, mostly on the coastal mountains of central Chile (Heilmayr et al., 2016). The remaining native forests are a globally important repository for Mediterranean and temperate forest species and biodiversity (Myers et al., 2000). This is particularly true in the coastal mountains and the central valley where native forests have been fragmented by agriculture and commercial plantations of mostly *Pinus* spp. and *Eucalyptus* spp. (e.g., Miranda et al., 2016; Echeverría et al., 2006, 2008). For example, from the post-Columbian (ca. 1550) to 2007, 51 % of native forests were lost due to land cover conversion to grassland and agriculture (Lara et al., 2012). When plantations replace native forest cover, there is concern about reduction of ecosystem services, including recreation, biodiversity, and water supply (Han et al., 2017; Kim et al., 2017). For instance, a long-standing debate about where forests should be located, particularly on the coastal mountains, originated decades ago (Torrejón et al., 2011; Cunill, 1970; Elizalde, 1970). This debate continues, but there is now an emphasis on how to locate plantations and manage native forest to maximize carbon sequestration and wood production, while preserving water yield and other ecosystems services. Managing the trade-offs between carbon sequestration, wood production, and water security will require an understanding of the regional relationship between forests, productivity, and water use. In terms of water footprint and carbon sequestration capability, wood stands over other structural materials (see Falk (2009) for wood and steel comparison).

Water supply is the most significant concern regarding the effect of plantations and a great deal of research has been conducted to quantify the effect of commercial plantations on water yield and quality. The area occupied by plantation forests is also the most densely populated region of Chile (Armesto et al., 2007; Fuentealba et al., 2021), creating significant pressures on the land's availability for food production, timber growth, and water supply. The need to store carbon in this region to achieve the global needs for emissions neutrality by 2050 (Höhne et al., 2020) will intensify pressures on this area of Chile. In response, Chile has committed to the National Determined Contributions (NDC), which included the afforestation (or recovery) of 200,000 ha with a minimum of 70,000 ha of native forest and 100,000 ha of permanent forests by 2030 in conservation priority areas and/or in suitable sites for tree growth (Chilean Government, 2021). Therefore, a solid base of knowledge regarding forest hydrology and its drivers is critical to better management of land and water in Chile with a future net-zero emission status (Höhne et al., 2020).

Commercial plantations have also been associated with a range of negative effects on a broad range of environmental values. These include increased rates of evapotranspiration (Huber et al., 1985, 2008; Oliveira et al., 2005; Oyarzún and Huber, 1999), reduced streamflow and water yield (Lara et al., 2009; Little et al., 2009), water quality changes (Fierro et al., 2016, 2017), stream disturbances (Guevara-Cardona et al., 2006; Guevara et al., 2018), nutrient cycling (Oyarzún et al., 2005, 2007), reduced biodiversity (Nahuelhual et al., 2012; Zhang, 2012), and socio-cultural and environmental issues (Cannell, 1999; Durán and Barbosa, 2019).

Among the positive effects of forests, the erodibility of soils on the coastal range has been reduced by deep-rooted vegetation enhancing a sustainable option represented by less intensively managed planted forests, or native forests. Also, it has been discussed the potential benefits of plantations such as the reduction of the runoff and, therefore, the deep percolation of water into soil (Pizarro et al., 2022). An additional positive effect of plantations is the potential for efficient carbon sequestration. White et al. (2021) found that *E. globulus* can be five times more efficient in wood production per unit of water consumed than a native forest. Thus, sustainably managed plantations can mitigate the effects of climate change.

Despite the relevance of natural forests and tree plantations for the environment, regional economies, and quality of life (Glatzer, 2014), there has not yet been a complete synthesis of the biophysics of forest hydrology in Chile. The Chilean research on forest hydrology has focused mainly on the water balance of commercial plantations (e.g., Huber and López, 1993; Huber et al., 1998; Huber and Iroumé, 2001; Huber and Trecaman, 2004; Pizarro et al., 2006; Echeverría et al., 2007; Huber et al., 2010) and much less is known about the hydrological cycle in native forests (Frêne et al., 2020; Gutiérrez et al., 2014; Oyarzún et al., 2011a; Donoso et al., 2014). Also, the effects of forest management practices on streamflow and water quality (e.g., sediment and nutrient export) have not been synthesized. Although no general review has been published, some aspects have been covered in previous synthesis papers. These include Jofré et al. (2014) who analyzed the link between forest science and policy, García-Chevesich et al. (2017) evaluated the link between forest management and water yield in 13 countries, including Chile, and Pizarro et al. (2019) who addressed common questions regarding the effects of land use (comparing exotic plantations and native forest) on water supply. Soto-Schönherr and Iroumé (2016), reviewed all published studies about interception losses in native forests and commercial plantations in Chile. Finally, Jones et al. (2017) looked at the state of forests in South America and reviewed forest science and practice in the continent. While these papers recommended some research actions, they were either specific to one topic or too broad to the forest hydrology research variability within Chile. Therefore, a comprehensive review with a common research vision is necessary to identify knowledge gaps for future landscape planning and forest management.

1.1. Objectives, approach, and reference framework

The objectives of this review are (i) to provide a comprehensive overview of Chilean forest hydrology, to (ii) review prior research in forest hydrology in Chile, and (iii) to identify knowledge gaps, and provide a vision for future research on forest hydrology in Chile. We summarize 47 years (1975–2022) of research on forest hydrology in Chile, with a focus on understanding the effects of land cover change from native vegetation to forest plantations and grasslands/shrublands on the water cycle and sediment yield at different scales. While we focused on Chile, we expect that this review will contribute to a global understanding of forest hydrology.

The review is organized as follows: (i) Chilean forest hydrology (climate, hydrology, land cover history, human impacts in water demands, and climate change); (ii) prior research in forest hydrology in Chile which addresses questions and topics prevalent in the literature and among stakeholders, particularly the community. These questions include: Do forests consume more water than grassland and shrublands in Chile? Do plantations consume more water than native forest in Chile? Do Eucalyptus plantations use more water than Pinus plantations in Chile? How do plantations affect peak flow and low flow? How does the water balance of plantations and native forest vary across the plantation management cycle? How do plantations and their management affect water quality? The answer to these questions form the basis for (iii) filling knowledge gaps and a vision for the future of Chilean forest hydrology.

We used Web of Science and Google Scholar to find articles to include in this review. Using search strings including “Chile + land cover change”, “Chile + Water balance”, “Chile + forest hydrology”, “Chile + hydrology + native forest”, “Chile + evapotranspiration + plantations + native forest”, and looking for terms related to runoff, low flow, baseflow, transpiration, interception, throughfall, and stemflow, suspended sediment yield, forest/plantation water quality, forest management (sediment and water), and modelling and climate change (streamflow and water balance topics). The studies were grouped by the main areas of consensus and questions that emerge from the global research on forest hydrology and their application in Chile, namely:

1. Forests use more water and yield less streamflow than annual pastures or dryland (non-irrigated) and agriculture (Olivera-Guerra et al., 2014; van Dijk and Keenan, 2007; Farley et al., 2005; Zhang et al., 2001).
2. Exotic plantations appear to consume more water than native forest and reduce streamflow, but this result is less universal than the comparison between forest and grassland or shrublands (e.g., Galleguillos et al., 2021; Smethurst et al., 2015).
3. *Eucalyptus* plantations appear to use more water and the resulting streamflow reduction is more pronounced than that has been observed in *Pinus radiata* plantations (e.g., Scott and Lesch, 1997).
4. Removal of forest cover increases streamflow (Bosch and Hewlett, 1982; Hornbeck et al., 1993; Ice and Stednick, 2004; Jones and Post, 2004; Brown et al., 2005; Jones et al., 2009; Dai et al., 2010, Lara et al., 2021), resulting in larger peak flows (Beschta et al., 2000; Buttle et al., 2019) and increases the sediment export from catchments or logged stands (Safeeq et al., 2020; Gomi et al., 2005).
5. The effect of minor forest disturbances, such as thinning, pruning, or disease control, are often difficult to detect at the catchment scale. Therefore, stand scale studies are more appropriate for detecting change on hydrological processes (e.g., Nikooy et al., 2020). Similarly to a partial afforestation, the removal of < 20 % of a catchment cover appears not to be detectable in paired catchment studies (Brown et al., 2005).
6. The presence of forest cover plays an important role in the reduction of soil erosion and sediment loss (see Marden, 2012) but the difference between the cover of different forest types is still unclear (e.g., Pizarro et al., 2020) especially when different soil parental materials and the lithology of the headwater catchment are considered (e.g., Bywater-Reyes et al., 2017).
7. Even though the effects of land cover and forest management on low flows are usually associated with a reduction of stream flows (e.g., Coble et al., 2020; Perry and Jones, 2017; Iroumé and Palacios, 2013), the direction of change, magnitude, and the scale of the impact on catchment water storage appear to be unclear and highly site specific (e.g., Goeking and Tarboton, 2020).
8. Climate Change will exacerbate the hydrological responses in forested catchment with decreasing streamflows, due to higher evapotranspiration requirements of trees, with higher temperatures and when decreases in precipitations are projected (e.g., Galleguillos et al., 2021; Martínez-Retureta et al., 2021).

Our review includes, but distinguishes between measurements made at the plot scale, stand scale, and catchment scale. The implications of discrepancies between studies made at different scales are discussed. This collaborative document brings together the active and diverse national forest hydrology community to agree on priorities for supporting science for policy-makers and the sustainable management of water resources.

2. An overview of the Chilean forest hydrology

2.1. Climate and hydrology

Chile has 18 Köppen-Geiger climate types (Beck et al., 2018), the distribution of which is affected by latitude and topography (Fig. 1a). A common feature in all these climatic zones is that the rainfall is concentrated in winter. Annual rainfall increases from < 100 mm at semiarid latitudes (30° S) to > 3,000 mm at temperate latitudes (about 40° S) (Fig. 1b). Orographic effects create zones of high and low rainfall. One example is the coastal range, where the highest peak (1,300 m asl) of the Nahuelbuta Range (37° S) creates an area of very high rainfall (González and Garreaud, 2019) (Fig. 1b). South-Central Chile has four main topographic features, moving from west to east through the coastal plain; the coastal mountain range, the intermediate depression or central valley, and the Andes Mountain Range (with altitudes above 5,000 m). This topography influences precipitation and temperature and therefore the distribution of vegetation.

Along the latitudinal 17° S to 56° S range, Chile comprises 101 basins (Fig. 1c) according to Chile’s General Water Department (DGA). Of these 101 basins, 88 are covered by forest, including commercial plantations. The hydrological regimes of these basins include rain dominated, snow dominated, and a mix of both (McPhee, 2018). The intermediate depression, the coastal plains, and the coastal mountain range are mainly rain dominated, with some snow dominated areas in the higher peaks of the coastal mountain range, such as the Nahuelbuta range. The basins in the Andes are mainly snow dominated and where water is stored as snow in winter and released in spring to feeding springs and rivers that flow to the Pacific Ocean. Most summer flows are sourced from water storage.

Of the 101 defined basins, the average annual precipitation is 1,525 mm and the mean runoff is 1,220 mm (Dirección General de Aguas, 2016). Boisier et al. (2018) reported negative trends in rainfall in central and south-central Chile during the last four decades. In the last two decades, Chile has experienced one of the driest periods on record (Boisier et al. 2016). In the last decade, an unprecedented megadrought has affected an extensive latitudinal range (30° S to 38° S), with reduced rainfall of about 40 % below the long-term average (Garreaud et al., 2017, 2020).

2.2. Native forests in Chile

In Chile, native forests grow under semiarid and temperate climates (29° and 56° S) (see Fig. 2a). Donoso (1981) classified the Chilean native

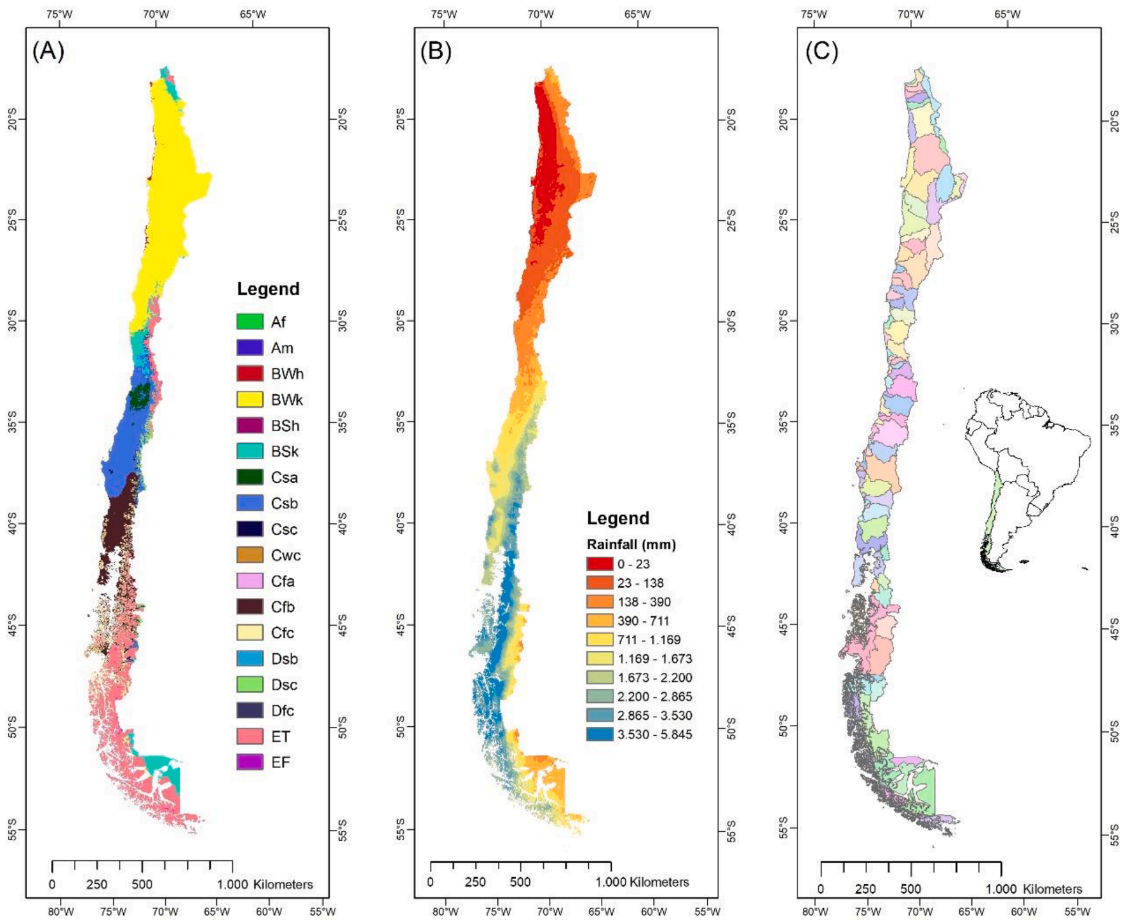

Fig. 1. (a) distribution of Köppen-Geiger climate types (0.0083° resolution) within Chile (Beck et al., 2018). (b) average rainfall (1979–2020) within central-southern Chile from CR2 products (http://www.cr2.cl/datosproductos-grillados). Most of exotic plantations occur between 34° S and 44° S (Salas et al., 2016). (c) Main Chilean basins (1 0 1) (DGA, 2016).

forests into 12 types. In the region with the greatest human population density, where commercial plantations are present, the main types of natural vegetation are the sclerophyllous forest, Roble – Raulí – Coigüe (respectively, *Nothofagus obliqua* (Mirb.) Oerst., *Nothofagus alpina* (Poepp. & Endl.) and *Nothofagus dombeyi* (Mirb.) Oerst.), evergreen (e. g., Ulmo, *Eucryphia cordifolia* Cav. and the Olivillo, *Aextoxicon punctatum* R. et Pav.), and Lenga (*Nothofagus pumilio* (Poepp. & Endl.) Krasser) (Luebert and Pliscoff, 2018; Salas et al., 2016). Evergreen (which develops in the Valdivian and Northern Patagonia regions, 38-46°S) and Lenga forest types represent ~ 50 % of the Chilean Native forest (Corporación Nacional Forestal, 2022a, available at https://www.conaf.cl/nuestros-bosques/bosques-en-chile/catastro-vegetacional) distributed ~ 25 % on each forest type.

Around 10.4x10^6 ha of native forest cover can be managed for commercial forest products (e.g., timber) (Salas et al., 2016), and this total includes around 5,000 ha of plantations of the native forest species, mainly *Quillaja saponaria* Molina and *Nothofagus* spp. (Corporación Nacional Forestal, 2020). Most of the native forest surface is owned by private stakeholders, and the harvested wood is used as timber (mainly for Oriented Strand Boards and sawn timber) or for firewood production, often with poor management practices (Lara et al. 2006, 2009, 2018). The Mediterranean and the Valdivian native forest in south-central Chile are one of the 35 biodiversity hotspots recognized globally, due to their high degree of endemism and the increasing number of threatened species (Myers et al., 2000; Mittermeier et al., 2004). More than 50 % of the broad-leaved, evergreen temperate forest in the southern hemisphere occurs in Chile (e.g., Coigüe, Ulmo, and Olivillo) (Donoso, 1993).

2.3. Commercial plantations in Chile

Commercial plantations are located mostly in central-southern Chile, between 34° and 40° S (Fig. 2b), a territory occupied by five million people (30 % of total Chilean population) (Instituto Nacional de Estadística [INE], 2018). Sixty-one percent of plantations are *Pinus radiata,* and 33 % are *Eucalyptus spp*. (mainly *E. globulus, E. nitens, and E. globulus* × *nitens* hybrids) (Ministry of Agriculture, 2021). Plantations were introduced in 1907 (Albert, 1907) to control soil erosion that arose from clearing and burning of native forest for agriculture stretching back to the pre-Hispanic period (Pizarro-Tapia et al., 2021a; Armesto et al., 2010). Since then, large areas of native forest have been logged to produce lumber for mining and cleared for cattle grazing, which led to > 200 years of intense deforestation, concentrated in south-central Chile (Armesto et al., 2010). Land clearing across many areas was largely unplanned, and by 1950 > 19 million hectares of Chile were assessed as severely eroded (García-Chevesich et al., 2017). Erosion was particularly severe on the highly erodible metamorphic sediments of the western coastal mountains (Elgueta and Jirkal, 1942) and the granite derived soils of the eastern coastal ranges.

Monterrey pine (*Pinus radiata* D.Don) were then chosen for reclamation of this severely eroded land in central Chile (Albert, 1907; Camus, 2003; Ramírez, 1992). Erosion continued to worsen into the 1970 s until the Chilean Government generated incentives for afforestation, providing (Decree Law 701 of 1974, Jelvez et al. (1990)) the impetus for the creation of the Chilean forestry sector (Cabaña, 2010). The rapid early growth and resultant short-rotation lengths of *P. radiata* made it more profitable than other alternatives and gave the Chilean

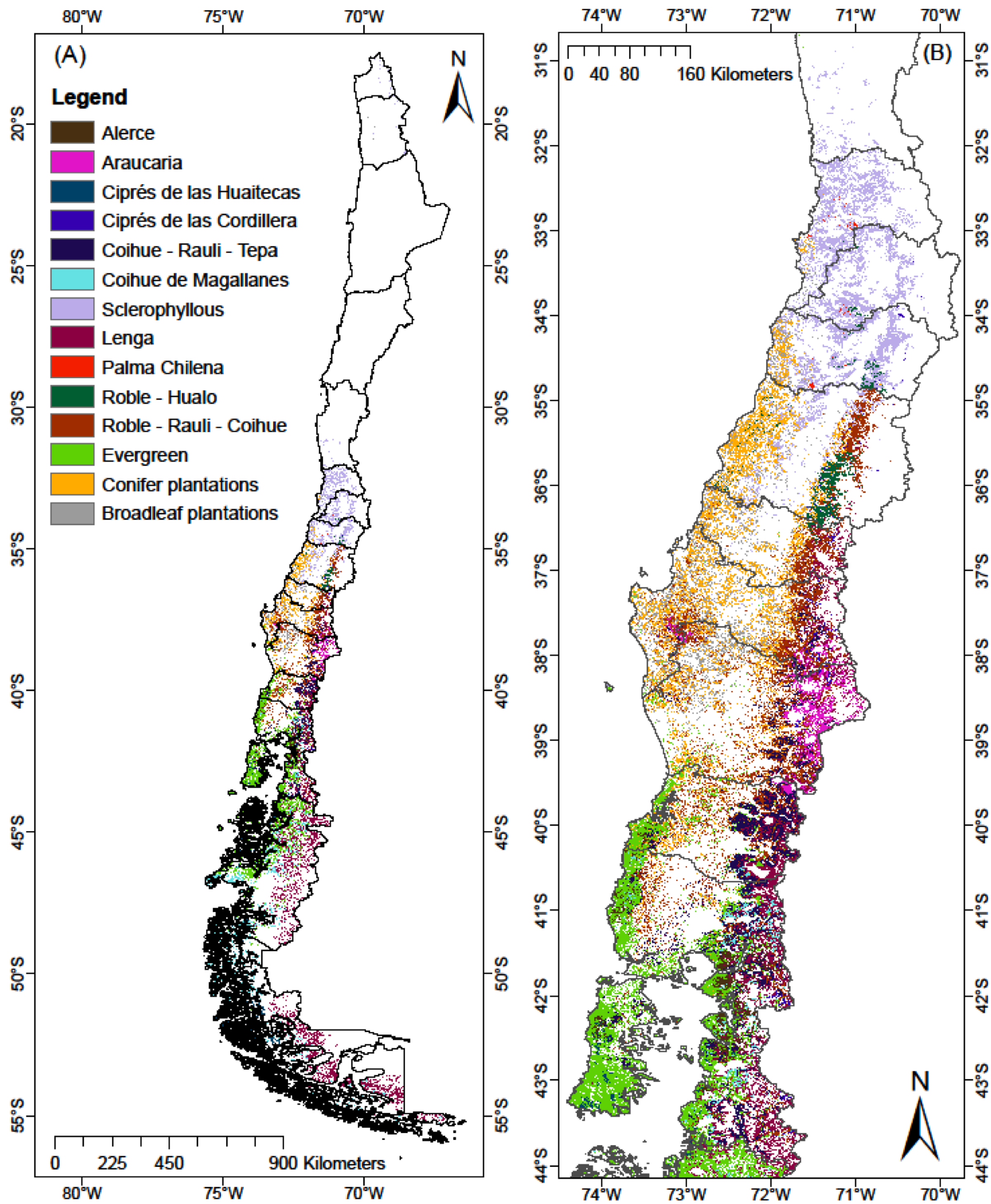

Fig. 2. (a) land cover classification of the original layer with Native Forest (adapted from Corporación Nacional Forestal, 2022a) and plantations (according to Zhao et al., 2016). Most of the forest starts around latitude 33° S, but there are some small relicts of forest in semiarid Chile, 30° S. (b) shows area where most of the exotic plantations occur (between 34° S and 44° S).

forestry sector a competitive advantage over other countries. Other exotic species were later introduced between 1960 and 1980, and large-scale afforestation with these tree species in addition to *P. radiata* ensued (Fuentealba et al., 2021). These exotic species used for reclamation included *Eucalyptus nitens* (Deane et Maiden) Maiden; *Eucalyptus camaldulensis* Dehn.; *Pinus ponderosa* Dougl. ex Laws.; *Pinus contorta* Dougl. ex Loud.; *Pseudotsuga menziesii* (Mirb.) Franco; and *Acacia saligna* (Labill.) HLWendl (Instituto Forestal, 2019).

The area of plantations has increased by several million hectares in the last 50 years (Fig. 3). A proportion of this area represented substitution of native forest. For instance, the Rio Maule-Cobquecura area (ca 600,000 ha, around 35 – 36° S) suffered a decline in native forest surface from 21 to 7 % (1975– 2000) which was attributed to exotic plantation expansion (Echeverría et al., 2007). Specifically, commercial plantations (exotic and native forest, including forestation and reforestation) increased on average around 50,000 ha per year between 2000 and 2010, while the 2015– 2019 period this area diminished to ca 2,000 ha per year (Corporación Nacional Forestal, 2022b, available at

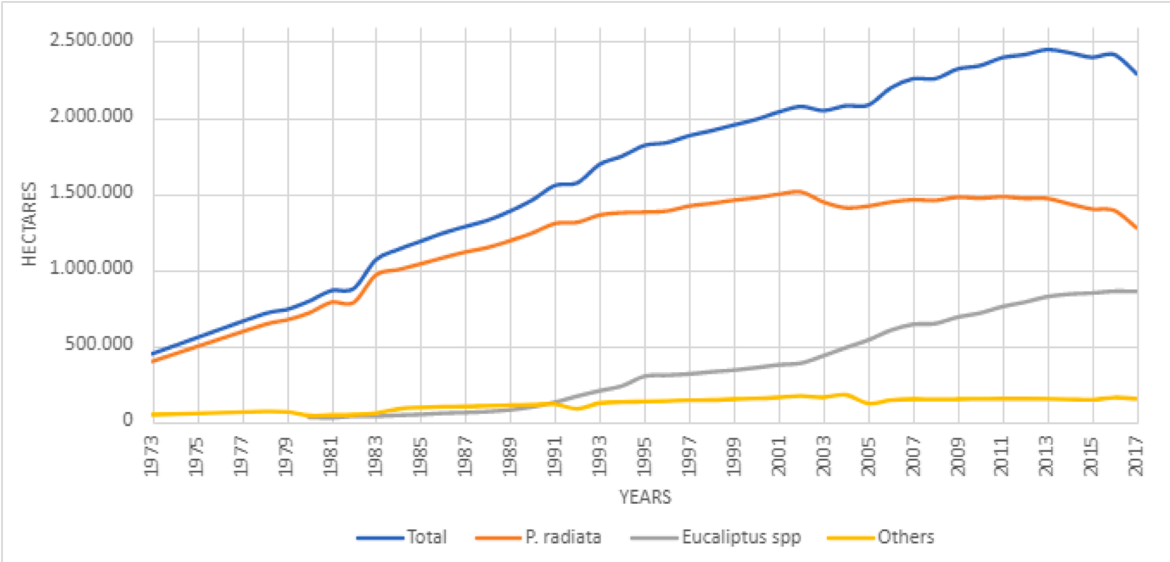

Fig. 3. Plantation evolution between 1973 and 2017.
adapted from Barros, 2018

https://www.conaf.cl/nuestros-bosques/bosques-en-chile/estadisticas-forestales/; Food and Agriculture Organization, 2020). Thus, there is a growing concern regarding the effect of this change on water resources across soil types, climate, topography, and geological setting (e.g., Jones et al., 2017; Martínez-Retureta et al., 2020).

2.4. *Population growth, water consumption per capita, and climate change*

In Chile, total water demand by agriculture, industry, and human consumption has increased up to two times faster than population growth during the last century (Rivas et al., 2020). More than 3 million people live in the coastal cities of Concepción, Valparaiso, and Valdivia (33– 40° S) and many smaller cities and towns that receive water from rivers that originate in the heavily forested headwater catchments of the coastal mountain range. Much of the commercial forestry is also located in these catchments (Armesto et al., 1998; Salas et al., 2016) (Fig. 2). The establishment of these plantations has resulted in alterations of water quality (Cuevas et al., 2006, 2018; Hervé-Fernández et al., 2016a; Oyarzún et al., 2007, 2015; Uyttendaele and Iroumé, 2002), quantity (Little et al., 2009, 2015), and timing of streamflow and spring flow (Rivas et al., 2020). Recently, the pressure on water resources created by this co-location of major population centers with commercial timber/pulp production has been intensified by a pronounced drying trend in the local climate. Runoff generation and water supply could be affected because water demand is projected to increase within this region during the coming decades (Fernández et al., 2018) and that current negative trends of rainfall are expected to increase under climate change scenarios (Barría et al., 2017; Bozkurt et al., 2018). Therefore, a detailed knowledge of the water balance of plantations and alternatives with high resolution —in time and space— are needed to ensure appropriate water resources planning into the future (Ellison et al., 2012). For instance, water restrictions were common in rural sectors (Molinos-Senante and Donoso, 2021). Water scarcity has been associated with both climate change and water use patterns as important drivers (Barría et al., 2021a, 2021b; Gosling and Arnell, 2016).

3. A review of Chilean forest hydrology research

This section summarizes the findings of 75 studies regarding the comparative effects of natural forests and exotic plantations on the water balance and water quality conducted primarily in 54 (out of 56) studied small catchments and ca 42 basins (Fig. 4). Knowledge gaps and priorities for research were identified. Additionally, Table A1 presents reported hydrologic information (i.e., precipitation, streamflow, low flow-baseflow, runoff, throughfall, stemflow, interception, and evapotranspiration).

The first forest hydrology work published using a Chilean dataset compared three small catchments (3.4–28 ha) in Central Chile (35° S) (Jones et al., 1975). These catchments were covered by small native trees and shrubs, and two of them were partially burned. This study found that peak flows and sediment yield were higher within the burned catchments in the period immediately after the fire.

Studies regarding water quality and quantity are mainly concentrated between 35° S and 41° S (Fig. 4). Of the 73 studies, 38 of them (56 % minus the extreme south studies below 41° S) were conducted in the Los Rios Region (i.e., at least 1 site was located in this region). Most of the studies were conducted between 2010 and 2022 after 18 years with very few publications (Fig. 5). Long-term experimental data is very scarce and since 1950 has been focused on streamflow measurements in large catchments (<100 km^2, in charge of the Chilean Directorate of Water [DGA]) (Fig. 6) and precipitation from 1979 at country level (Boisier et al., 2018).

3.1. *Interception and throughfall*

3.1.1. *Partitioning of rainfall among interception, throughfall and evapotranspiration – Effects of forest type*

Interception varies between vegetation and forest types (Table 1). It is also influenced by rainfall intensity and amount, and plantation density and leaf area index. In a comprehensive study of canopy interception by plantations and broadleaf species, Soto-Schönherr and Iroumé (2016) presented interception values for different plantation and native forest ages across a rainfall gradient. The average annual interception across all rainfall and forest type combinations was around 21 % of rainfall and interception losses were correlated with annual rainfall and basal area. Results show that broadleaf native forests intercepted more water than pine plantations. The reported value matches global observations (Yue et al., 2021). For example, Frêne et al. (2022) found, in average, a 33 % of interception in a native forest plot study in southern Chile (42° S, ca 2,000 – 2,500 rainfall). There was a higher throughfall in coniferous (Alerce, Monterrey Pine, and Douglas-Fir) than

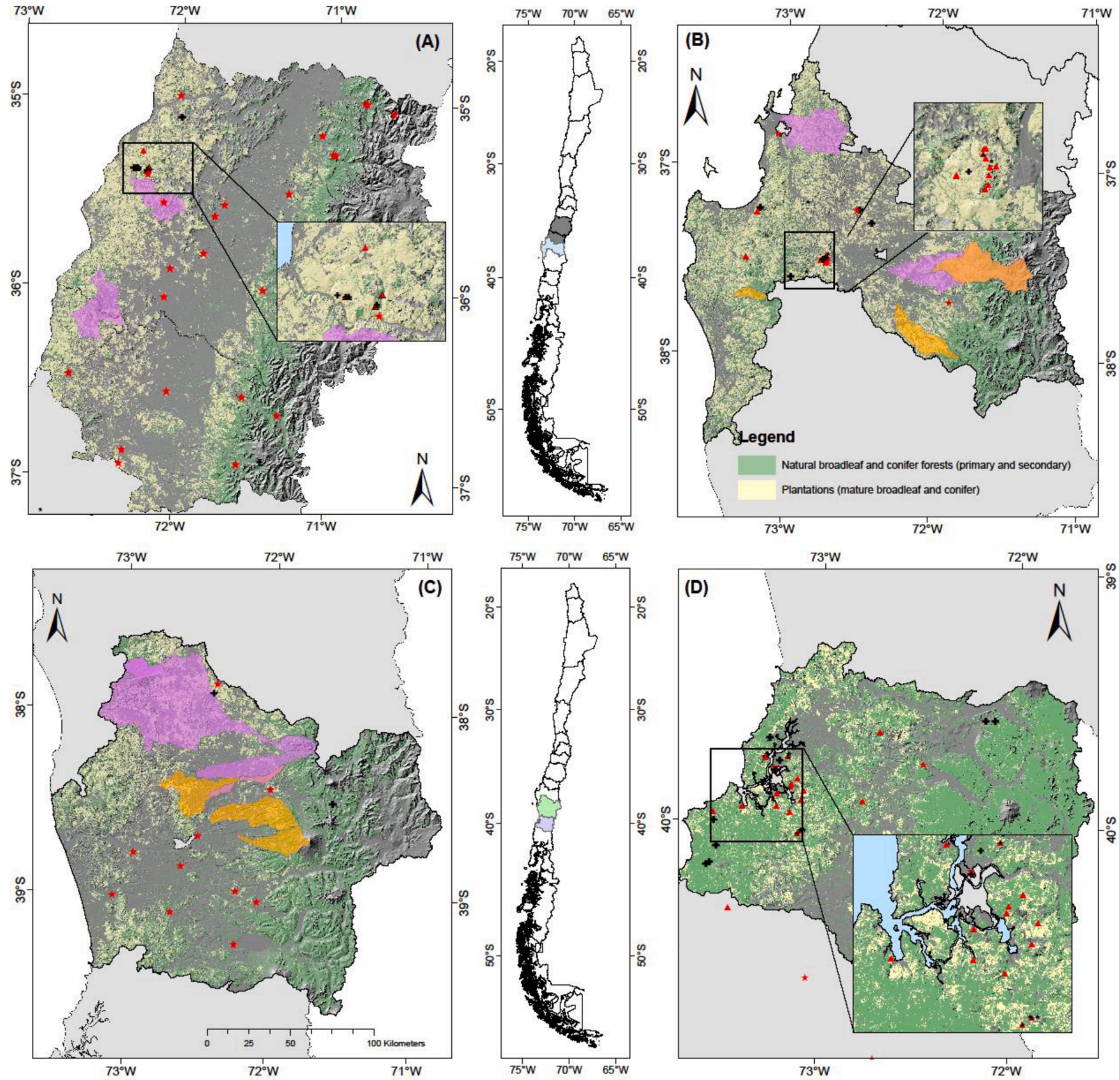

Fig. 4. Location of forest hydrology studies from south-central Chile (catchment, small-catchment, stand, and plot scales) compiled in this review. For illustration purposes land cover was grouped into forestry plantations (broadleaf and conifers) and natural, non-planted forests (Zhao et al., 2016). Central-southern Chile was split into four sections for clarity; Zone (a) extends from 35° to 37°S (from Maule to Ñuble Region), zone (b) from 37° to 38° S (BioBio Region), zone (c) from 38° to 39° S (La Araucania Region), and zone (d) from 39° to 40° S (Los Rios region). Red triangles are micro-catchment studies (outlet locations, <100 km^2), black crosses are plot studies, black stars are sediment studies, red pushpins are stand studies, orange areas indicate large study catchments (>100 km^2), red stars are Pizarro et al. (2022) metanalysis catchment outlet, and in purple catchment modelling studies. Main cities are shown in gray. There were two study sites below 41° S that were excluded from this map because exceeded the exotic plantations range. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

in broadleaved forests (native and *E. nitens*) but no significant differences in a 29-plot study over different land covers across central-southern Chile (Huber and Iroumé, 2001). Throughfall was similar in a broadleaf native forest and in planted Douglas fir in the southern Andes (79 % in broadleaved and 72 % for Douglas fir), but interception was higher in Douglas fir (22 % versus 14 %) (Iroumé and Huber, 2002). However, the difference was more attributable to the rainfall type and the meteorological conditions.

In mature pine plantations near Valdivia (39° S), average annual canopy interception was 10 %, but the proportion of canopy interception varied monthly and was negatively correlated with rainfall intensity (Huber and Oyarzún, 1983). In a subsequent study in southern Chile (39° S), interception losses were in the order of 15 % and throughfall of ca. 85 % with all these values also affected by rainfall intensity and amount and the temporal distribution of rainfall (Huber and Oyarzún, 1990). In general, higher intensity and longer duration rainfall is associated with lower interception as a proportion of rainfall (Huber and Oyarzún, 1984). In a more recent study conducted along a rainfall gradient in central Chile (35° S, 37° S, and 39° S), the rainfall interception by *P. radiata* on high rainfall sites (> 2,000 mm) was 15 % but when rainfall was < 1,200 mm, interception was much higher at 36 to 40 % of annual rainfall (Huber et al., 2008).

Results of Chilean studies on *Pines* and *Eucalyptus* are consistent with global comparison. That is, for a given rainfall regime the proportion of rainfall intercepted by *Eucalyptus* canopy is lower than the proportion intercepted by a *Pinus* canopy (Benyon and Doody, 2015). For instance, Oyarzún and Huber (1999) reported averaged values of 4.4 % for *E. globulus* and 3.8 % for the *P. radiata* of canopy interception. In another pine and *E. globulus* comparison, canopy interception varied from 10 to 17 % with *E. globulus* the lowest (Huber et al., 2010). In the period following the recent Chilean megadrought, the proportion of annual rainfall intercepted by 21-year-old *P. radiata* and *E. globulus* stands were 15 % and 13 % of rainfall (White et al., 2021) and were similar to the range of values observed for the same species in southern Australia (Benyon and Doody, 2015).

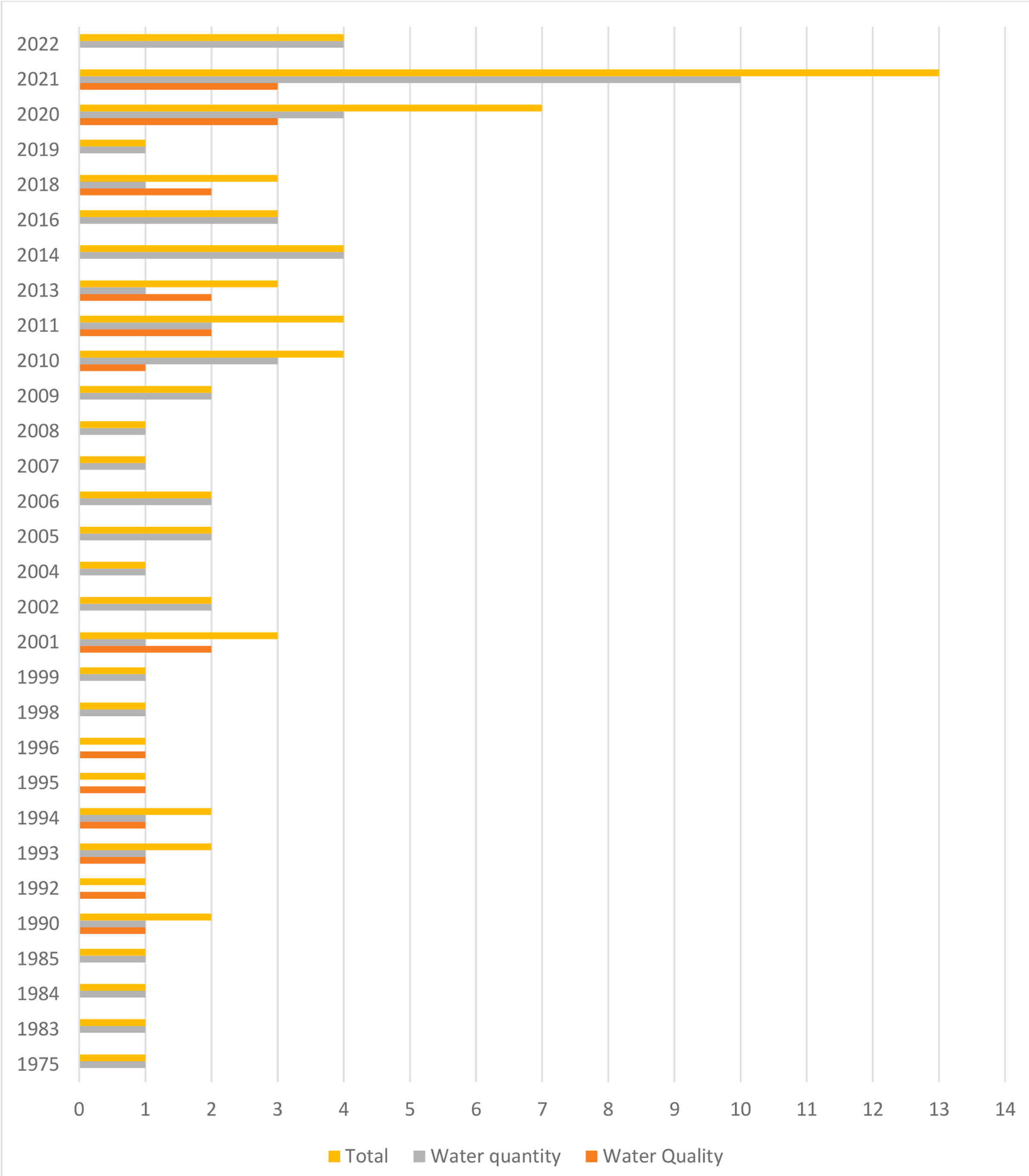

Fig. 5. Numbers of studies per year since 1975. Four studies have analyzed both water quantity and quality (Otero et al., 1994; Birkinshaw et al., 2011; Little et al., 2015; Frêne et al., 2021). Years without publications have been omitted for visualization purposes.

3.2. Forest evapotranspiration

One of the key questions in forest hydrology research is how much water is taken up by forest. Prior research has examined how evapotranspiration (ET) differs between grasslands, shrublands, native forest, and plantation forests. Unfortunately, there are few published studies that have measured evapotranspiration under forested systems (i.e., White et al., (2021) with sap flow sensors, and Meza et al. (2018) and Melo et al. (2021) with eddy covariance systems). All other studies have derived indirectly ET from water balance approaches, complemented with remote sensing information in a few cases.

3.2.1. Do forests consume more water than grassland or shrubland in Chile?

In Valdivia (39° S), south-central Chile, where annual rainfall often exceeds 2,300 mm, evapotranspiration was approximately 76 % of net precipitation in mature and young pine plantations and only 29 % of rainfall in grassland (Huber et al., 1985). When grassland is compared to a native forest in southern Chile (38° S, 25 km northeast from Valdivia), evapotranspiration was only 13 % of rainfall (3,204 mm between September 1992 and August 1993) in grasslands, while it was 33 % in native forest (Echeverría et al., 2007). All studies that have compared grassland with plantations in Chile have confirmed that runoff from forests is lower than runoff from grass covered catchments (Huber and

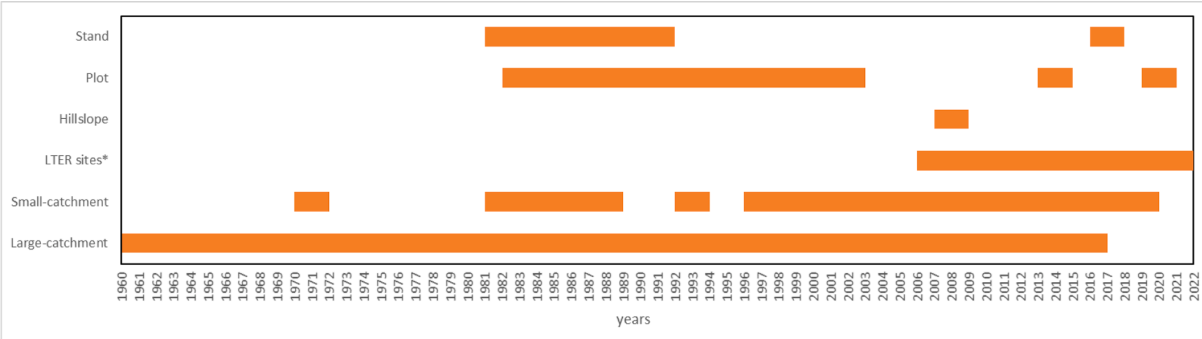

Fig. 6. Temporal distribution of studies in Chile (water quantity and quality). LTER sites refer to Lara et al. (2021) and Balocchi et al. (2021b) small catchments network dataset. Large catchment refers to the Directorate of water (DGA) catchment administration (<1,000 ha).

Table 1
Interception reported values within the Chilean literature. Plot scale area range from 400 to 2,500 m^2 and small-catchment areas were < 100 km^2.

Latitude	Study precipitation (mm)	Scale	Species	Interception	Reference
37° S	2768	Stand	15–25 yr old N. obliqua	11 %	Fuentes et al., 1994
			23-yr old P. radiata	26 %	
			13-yr old P. radiata	17 %	
37° S	1090	Plot	8-yr old E. nitens (high density)	30 %	Huber et al., 1998
			8-yr old E. nitens (medium density)	27 %	
			8-yr old E. nitens (low density)	*24 %*	
41° S	2100	Stand	60-yr old Young secondary	20 %	Gutiérrez et al., 2014
			>500-yr old Old-growth		
35-42° S	667–4500	Plot	Multispecie (Pine, Eucalyptus, Nothofagus, etc.)	21 %	Soto-Schönherr and Iroumé, 2016
37° S	1003	Stand	15 yr old P. radiata	32 %	Huber and Trecaman, 2000
39° S	2000	Stand	26 yr-old P. radiata	10–15 %	Huber and Oyarzún, 1983; Huber and Oyarzún, 1984; Huber and Oyarzún, 1990
35–37-39° S	< 1200	Plot	13–17 yr old P. radiata	36–40 %	Huber et al., 2008
	< 2000			15 %	
39° S	2300	Plot	4 yr-old P. radiata	3.8 %	Oyarzún and Huber, 1999
			2 yr-old E. globulus	4.4 %	
37° S	1150	small-catchment	23yrs old P. radiata	16–17 %	Huber et al., 2010
			9 yr-old E. globulus	10–11 %	
37–38-39-40°S	1000–3500	Plot	Conifer (pantation and native forest)	11–39 %	Huber and Iroumé, 2001
			Broadleaf (plantation and native forest)	10–37 %	
38° S	2341	Plot	Douglas-fir	22 %	Iroumé and Huber, 2002
			Native forest (southern andes)	14 %	
37° S	2149	Small-catchment (12–21 ha)	23-yr old P. radiata (12 ha)	16 %	Huber et al., 2010
			23-yr old P. radiata (14 ha)	15 %	
			9-yr old E. globulus (16 ha)	10 %	
			9-yr old E. globulus (21 ha)	10 %	
37° S	1900	Stand	21- yr old P.radiata	15 %	White et al., 2021
			21- yr old E. globulus	8 %	
			Native forest	13 %	
35° S	524		13-yr old P.radiata	33 %	
			Native forest	21 %	
41° S	2,500	Plot	Native forest	33 %	Frêne et al., 2022

López, 1993; Huber et al., 1985).

Other studies have demonstrated lower ET values for shrubland and grassland when compared to forest covers. This is the case of the study of Galleguillos et al. (2021) who computed lower ET values at the hydrological response unit (HRU) scale for shrublands in South-Central Chile (430 mm in average) instead of 709 to 623 mm from *Pinus Radiata* using the SWAT process-based model. Those differences are consistent with results obtained from ASTER satellite imagery where surface energy models showed similar differences among shrublands and forests for the same study region (Olivera-Guerra et al., 2014). This study also reported lower water consumption of grassland than forests. These higher ET values in forest are in agreement with direct measurements of transpiration and ET. White et al. (2020) reported values of 514 mm of evapotranspiration for native forests in central Chile (35° S) and Meza et al., (2018) reported values of ET obtained from a eddy covariance systems in the order of 150 mm in shrublands in a more arid context (33° S).

In Chile, the greatest absolute differences in evapotranspiration between different land uses occur in high rainfall areas (Table 2), which is consistent with the conclusion of Zhang et al. (2001) who analyzed studies from around the world.Table 3..

3.2.2. Do plantations consume more water than native forest in Chile?

There is now evidence, collected at both the catchment (Iroumé et al., 2021; Lara et al., 2009) and the stand scales (White et al., 2021), that commercial tree plantations consume more water and consequently release less water downstream than mature native forest. The greater water consumption from commercial plantations is particularly high

Table 2
Evapotranspiration (ET) studies in Chile.

Latitude	Study precipitation (mm)	Scale	Species	ET (% over rainfall)	Reference
37° S	2768	Stand	15–25 yr old N. obliqua	89 %	Fuentes et al., 1994
			23-yr old P. radiata	72 %	
			13-yr old P. radiata	66 %	
41° S	2100	Stand	60-yr old Young secondary	30 %	Gutiérrez et al., 2014
			>500-yr old Old-growth	34 %	
39° S	1023	Stand	32-yr old P. radiata	93 %	Huber and López, 1993
			Grassland	54 %	
39° S	2220	Stand	26-yr old P. radiata	62 %	Huber et al., 1985
			9-yr old P radiata (managed)	60 %	
			9-yr old P radiata (cattle)	58 %	
			Grassland	29 %	
38° S	3024	Plot	Meadow	13 %	Echeverría et al., 2007
			Native forest	34 %	
37° S	2149	Small-catchment (12–21 ha)	23-yr old P. radiata (12 ha)	65 %	Huber et al., 2010
			23-yr old P. radiata (14 ha)	64 %	
			9-yr old E. globulus (16 ha)	75 %	
			9-yr old E. globulus (21 ha)	70 %	
35-39° S	1000–2200	Plot	11–15-yr old P. radiata (multiple sites and densities)	40 %-84 % (66 % average)	Huber and Trecaman, 2004
37° S	1090	Plot	8-yr old E. nitens (high density)	79 %	Huber et al., 1998
			8-yr old E. nitens (medium density)	77 %	
			8-yr old E. nitens (low density)	71 %	
35-39° S	1000–2200	Plot	15-yr old P radiata	Net ET 32 %-55 %	Huber et al., 2008
37° S	1448	Stand	21-yr old P.radiata	45 %	White et al., 2021
			21-yr old E. globulus	45 %	
			Native forest	41 %	
35° S	524		13-yr old P.radiata	76 %	
			Native forest	55 %	
36° S	814	Catchment (HRU)	*Pinus. Radiata Native Shrubs*	65–77 %	Galleguillos et al., 2021
				57 %	
38° S	1090	Catchment (HRU)	Native Shrubs	36 %	Gimeno et al., 2022
			Eucalyptus. sp	70 %	
			Native forest	50 %	

HRU: hydrological response unit for SWAT model that represent the smallest modeĺs spatial unit (unique combination of land use, soil, and slope characteristics).

when compared to native forests of north-central Chile, mainly covered by *Nothofagus* species.

White et al. (2021) recently found that trees of *E. globulus* (21 years-old in central sites) and *P. radiata* (13 years-old in dry sites and 21 years-old in central sites) used a little over 100 mm more water (around 5 to 6 % of the precipitation) per year than native tree species (*Nothofagus obliqua* –*Nothofagus glauca*, 80 years-old on average) forest in a dry (1,000 mm precipitation) and central (1,340 mm precipitation) locations within the coastal range at stand scale.. The greater water consumption of plantations compared to native Roble-Hualo forest type was also observed at catchment scale studies (140 to 1,462 ha). For instance, Lara et al. (2009) found a greater annual direct runoff (quickflow/precipitation) as the native forest cover increased (old-growth and second growth native forest), when compared with commercial plantations (*P. radiata*, *E. nitens,* and *E. globulus*). Gimeno et al. (2022) analyzed ET dynamics at the hydrological response unit (HRU) scale in a catchment in central-southern Chile (38° S) and showed that the ratio of the actual evapotranspiration (ETa) and rainfall was 70 % for tree farms (plantations) and 50 % in native forest (Table 2). The agreement of catchment and stand scale studies supports the conclusion that native forests consume less water than commercial plantations in central Chile.

While it seems that intensively managed plantations consume on average more water than native forests, some empirical studies have not found a significant difference in the water balance between native forest and plantations. In fact, there is one study that reported greater native forest evapotranspiration (89 % of rainfall) values than a *Pinus* plantation (between 66 and 72 %) (Fuentes et al., 1994). In the semiarid north-central Chile (35° S), Pizarro et al. (2005) found that 50 % substitution of native forest by plantations in a 264 km^2 basin did not affect annual water yield over 40 years. However, in the same basin a decrease in summer runoff due to the increase in the percentage of plantations was found (Little et al., 2009). In contrast, at the wetter end of the rainfall range in central Chile (2,000 mm precipitation), Iroumé and Palacios (2013) observed that a 16 % substitution of native forest by plantations was sufficient to reduce streamflow in their six large study basins (94 to 1,545 km^2). Huber et al. (2008) study found that net evapotranspiration values varied from 32 % of the incoming precipitation in the south (40° S) to 55 % of the incoming precipitation in northernmost (and drier) sites (33° S). A remote sensing study estimating evapotranspiration on agricultural land, native forest, plantations, and shrubland, found that the rates of evapotranspiration in commercial plantations were higher than in native forests, and these latter were higher than the mixed forests (Olivera-Guerra et al., 2014).

While these results may appear contradictory, when they are considered in the context of the model of Zhang et al. (2001) some patterns emerge. The largest proportional difference between plantations and native forests seems to occur in Chile's medium volume rainfall area where the climate wetness index (CWI) (ratio of precipitation to potential evaporation) is between 0.8 and 1.2 (e.g., White et al., 2021). Areas with climate wetness index $<$ 0.7 are water-limited, whereas areas with climate wetness index $>$ 1.5 are energy-limited. These limitations lead to smaller differences in transpiration between species. The CWI definition does not mean that differences do not exist in water or energy energy-limited areas, only that they are more challenging to detect. In this sense the study of Alvarez-Garreton et al. (2019) reported the same patterns with higher streamflow differences when forest plantations replaced other land cover in drier than in humid catchments.

Tree water sources revealed by stable isotopes in the water molecule (i.e. δ^2H and δ^{18}O) have shown that native evergreen forests in southern Chile use water that has been interpreted as slow or static water and replenished during autumn precipitation events. In comparison, exotic *Eucalyptus nitens* appear to use more mobile water that is recharged during the year (Hervé-Fernández et al., 2016b).

Table 3
Modelling and climate change main findings.

Latidude	Rainfall (mm)	Scale	Model	Type of change simulated	change in annual streamflow/ water yield/summer flow (m^3/s or %)	Reference
37-38° S	1650	Catchment (4340 km2)	SWAT	39 % plantations – 24 % native forest (Baseline) 76 % plantations – 24 % native forest 76 % agriculture-23 native forest 62 % plantations-16 % native forest	Annual mean streamflow w/ respect to baseline – 5 % + 5 % – 2.4 %	Stehr et al., 2010
37-38° S	1650	sub-catchment (678,427,401 km2)	SWAT	Baseline 60 % plantations expansion (2015–2020) 68 % plantations expansion (2025–2030) 74 % plantations expansion (2035–2040)	Summer flow −34 % −21 % −55 % −39 % −24 % −58 % −42 % −31 % −61 %	Aguayo et al., 2016
36° S	814	Catchment (HRU, 620 km2)	SWAT	Current land cover (Baseline 2006–2018) Stand Forest management Projected Forest policy Extreme plantation expansion Forced land displacement Massive Restoration strategy	Annual streamflow w/ respect to baseline; and future (2035–2050) No variation; −32 % No variation; −31.7 % −2.5 %; −34 % −17 %; −46.6 % 2.3 %; −29.5 % 10.9 %; −22.3 %	Galleguillos et al., 2021
36° S	1250	Catchment (742 km2)	SWAT	Plantation expansion scenarios (1984–2013) LU_1986 LU_2001 LU_2011	LU_1986-LU_2001 −3.87 m3/s per year LU_2001-LU_2011 −21.19 m3/s per year LU_1986-LU_2011 −25 m3/s per year	Martínez-Retureta et al., 2020
37-38° S	1090	Catchment (HRU, 1026 km2)	SWAT	Baseline (1990–2015 land cover) NFPR (native forest recovery) PR (potential native forest)	Mean annual flow 0.33 % (NFRP/baseline) 2.52 % (PR/baseline)	Gimeno et al., 2022
38° S	3800	Catchment (540 km2)	SWAT	Native forest (scenario 2001–2018) Mix (Native and plantations) (scenario 2001–2018) Agriculture (scenario 2001–2018)	Water yield (WY) −0.05 % WY 1.31 % WY 0.52 %	Esse et al., 2021
37° S	1160	Small-catchments (7–414 ha)	TETIS	25 combinations of climate scenarios on Current cover (2015) Partial harvest (50 %) Native forest (50 % increase on the harvested area)	20 % decrease in rainfall and ET: −20–61 % runoff 20 % increase in rainfall and 20 % decrease in ET: 21–55 % runoff	Barrientos et al., 2020

3.2.3. Do Eucalyptus plantations use more water than Pinus plantations in Chile?

There is a common expectation that plantations of *Eucalyptus* species evapotranspire more water, and therefore have a larger effect on streamflow, than plantations of *P. radiata*. This view is prevalent in both the general public and amongst scientists and is prevalent worldwide. Given the ubiquity of this view, it is surprising that very few papers worldwide have directly compared estimates of evapotranspiration of *Pinus* and *Eucalyptus* plantations. In fact, we could only find two such publications (Scott and Lesch, 1997; Benyon and Doody, 2015). In Chile, studies that showed greater water use in *Eucalyptus* than *Pinus* were all conducted at the catchment scale (e.g., Huber et al., 2010). However, stand scale measurements did not detect a difference in annual evapotranspiration between *E. globulus* and *P. radiata* within northern (35° S) and central (37° S) sites (White et al., 2021). After radial sapflux density corrections, these latter authors found annual transpiration values of 492 mm for *P. radiata* and 472 mm for *E. globulus*, with no significant differences between tree species.

Turning first to the catchment studies, Huber et al. (2010) measured streamflow in two pine and two eucalyptus catchments (12.6 to 21.1 ha) in central Chile (37° S) and found that catchments with eucalyptus had lower streamflow than those with pine (469 vs 706 mm). Using the simplified catchment scale equation $P = Q + ET + \Delta S$ and assuming no change in storage, Huber et al. (2010) calculated an average evapotranspiration of 1,468 mm for the *E. globulus* (rainfall was 2,060 mm) and 1,384 mm for *Pinus* (rainfall was 1,930 mm). It is not possible to test for statistical significance of this difference, but these amounts of water may be important. The Priestley-Taylor potential evaporation for the area where these measurements were made is ~ 1,300 mm, so these estimates of evapotranspiration give crop factors of about 1.1 for both species. These results suggest that some water may have been passing below the measurement v-notch weirs in these catchments or that soil storage increased by around 100 mm in the measurement year. Thus, the absolute values of streamflow should be treated cautiously, but they may be important and should be corroborated by other studies.

In a recent stand scale comparison White et al. (2021) measured much lower rates of annual evapotranspiration than Huber et al. (2010) in stands of *E. globulus* and *P. radiata* species (745 mm in *E. globulus* and 731 mm in *P. radiata*) and found no difference between these species. The White *et al.* sites were about 60 km north of the Huber et al. (2010) sites in central Chile (37° S) but with about 500 mm less rain per year. White et al. (2021) observed that annual water use (ET) by both species (*E. globulus* and *P. radiata*) was around 740 mm, about 60 % of rainfall, but that water use by *E. globulus* was very rapid early in spring, while *P. radiata* deferred water use until later in the dry season.

Not all catchment studies have shown differences between the water balances of *Eucalyptus* and *Pinus* covered catchments. An analysis of water storage within 15 catchments (7 – 411 ha) in central southern Chile (37° S) with different land cover (Pine, *Eucalyptus*, and mixed) showed highly variable results (Barrientos and Iroumé, 2018). In these studies, plantation density, age, growth, and soil water storage were all correlated with water consumption. In the same study, rainfall and elevation were the main drivers of catchment storage capacity. In the same location and at 11 of the 15 catchments described above, another study reported a greater reduction in deep percolation under *Eucalyptus* than under *Pinus* (Iroumé et al., 2021). Also, after 10 years of streamflow measurements, Iroumé et al. (2021) did not find significant differences between *Eucalyptus* and *Pinus* covered catchments. However, the age of the pine plantations was greater than the usual commercial harvesting rotation (21–33 years old). For example, Huber and Trecaman (2004) at

Table A1
Hydrologic variables values reported in Forest Hydrology studies in Chile. These variables are precipitation, streamflow, low flow-baseflow, runoff, throughfall, stemflow, interception, and evapotranspiration). n/r: not reported. Pr, *Pinus radiata*. Mn = managed.

Land cover	Year	Precip (mm)	Streamflow (mm)	Runoff (mm)	Throughfall (mm)	Stemflow (mm)	Interception (mm)	ET (mm)	Reference
Native	1970–1972	92.5	15.7						Jones et al. (1975)
Pr	1981–1982	1769.2			1379.3	208.4	181.5		Huber and Oyarzún (1983)
Old Pr (26 yr)	1982–1983	2229.2			1882.7		346.5	1057	Huber et al. (1985)
Pr (9 yr)		2229.2			1796.6		432.6	1009	
Mn Pr (9 yr)		2229.2			2004.3		224.9	999	
Grassland		2229.2			2229.2		0	501	
Old pr	1982–1988	1969.7			1459.1	215.8	294.7		Huber and Oyarzún (1990)
Pr	1990	n/r						1190	Huber et al. (1990) *in german, no
Pasture		n/r						946	access to the publication
Native -Pr- grassland	1983–1984	2170	1150						Iroumé (1992)
Native -Pr- grassland	1988–1989	1485	655						
Pr before harvesting	1988–1989	1357						1031	Huber and López (1993)
Pr after harvesting	1989–1990	1175						489	
Pr after harvesting	1990–1991	1940						501	
Grassland	1988–1989	1357						601	
Grassland	1989–1990	1175						547	
Pr 1	1993–1994	3289	2739						Otero et al. (1994) *we average the
Pr 2		3289	1637						specific streamflow and then we
Native forest 1		3289	2532						converted to mm/year
Native forest 2		3289	2018						
Pr no residues	1991	1537		289					Oyarzún and Peña (1995)
Pr residues		1537		212					
Pr control		1537		54					
Pr no residues	1992	n/r		217					
Pr residues		n/r		123					
Pr control		n/r		70					
Pr 69	1991–1992	2000.3		59.72					Ibarra et al. (1996)
Pr 79		2000.3		73.38					
Native forest		2000.3		42.41					
E. nitens T1	1996–1997	1089.6			761.8		327.8	861.2	Huber et al. (1998)
E. nitens T2		1089.6			797.5		292.1	836.2	
E. nitens T3		1089.6			829.5		260.1	776.1	
Pr 90	1994–1995	1980.9			1850.2	69.9	60.8	639	Oyarzún and Huber (1999)
Pr 90	1995–1996	2127.5			1934.2	115.3	78	431	
Pr 90	1996–1997	1688.7			1512.2	100.1	76.4	936	
E. globulus 92	1994–1995	1980.9			1922	2.1	56.8	610	
E. globulus 92	1995–1996	2127.5			2037.7	7.8	82.4	475	
E. globulus 92	1996–1997	1688.7			1564.6	14	110.1	910	
Pr (15yr)	1997–1998	1003.8			777.4	29.6	326.5		Huber and Trecaman (2000)
Pr (Valdivia a)	1982–1988	1988.1			1459.3	215.8	313.8		Huber and Iroumé (2001)
Pr (Valdivia b)	1992–1994	2500			1771	244.5	484.5		*average values per period
Pr (Valdivia c)	1992–1998	2328.8			1759.4	185.2	384.2		
Pr (Valdivia d)	1992–1998	2328.8			1869.4	229.6	339		
Pr (Valdivia e)	1996–1997	2373					367		
Pr (Valdivia f)	1992–1993	2648					579		
Pr (Valdivia g)	1992–1994	2648					538		
Pr (Valdivia h)	1992–1995	2648					515		
Alerce	1982–1983	4603			3662	403	537		
Mixed broadleaf (a)	1982–1983	3563			2650	131	782		
Mixed broadleaf (b)	1986–1995	227.5			1496.7	37.8	692.8		
Pr (Nacimiento a)	1991–1992	1971			1359	88	523		
Pr (Nacimiento b)	1991–1992	1971			1557	68	338		
N. Obliqua (Nacimiento c)	1991–1993	1971			1697	49	225		
Pr (Collipulli a)	1996–1998	1448.5			967.5	38.5	442		
Pr (Collipulli b)	1996–1998	1448.5			1008.5	26	414		
Pr (Collipulli c)	1996–1999	1210.3			847.3	26.3	336.3		

(continued on next page)

Table A1 (*continued*)

Land cover	Year	Precip (mm)	Streamflow (mm)	Runoff (mm)	Throughfall (mm)	Stemflow (mm)	Interception (mm)	ET (mm)	Reference
Pr (Collipulli d)	1996–1997	1039			758	12	269		Huber and Iroumé (2001)
E. nitens (Collipulli e)	1996–1999	1210.3			779.6	45.3	385		*average values per period
E. nitens (Collipulli f)	1996–1999	1210.3			825.3	35	350		
E. nitens (Collipulli g)	1996–1997	1039			755	20	264		
Douglas–fir (a)	1998–1999	1346			812	81	452		
Mixed broadleaf (b)	1998–1999	1346			981	105	350		
Pr (Los Angeles I a)	1998–1999	1005			562	54	389		
Pr (Los Angeles I b)	1998–1999	1005			721	44	240		
Pr (Los Angeles I c)	1998–1999	1005			550	64	391		
Pr (Los Angeles I d)	1998–1999	1005			677	23	306		
Pr (Los Angeles II a)	1998–1999	1038			745	14	280		
Pr (Los Angeles II b)	1998–1999	1038			721	25	292		
Pr (12yr – w/o management)	1998–2000	1093.5						448	Huber and Trecaman (2002)
Pr (12yr – with management)		1093.5						431	
Native broadleaf (managed)	1998–2000	3805			2993	271	541		Iroumé and Huber (2002)
Douglas–fir		3805			2754	223	828		
Pr (Palhuen T1)		1271			838			978	Huber and Trecaman (2004)
Pr (Palhuen T2)		1271			786			1063	
Pr (San Ignacio T1)		1133			742			813	
Pr (San Ignacio T2)		1110			687			834	
Pr (San Ignacio T3)		1238			866			838	
Pr (San Ignacio T4)		1238			1015			672	
Pr (San Ignacio T5)		1199			948			528	
Pr (San Ignacio T6)		1199			1008			484	
Pr (San Ignacio T7)		1276			979			729	
Land cover	**Year**	**Precip (mm)**	**Streamflow (mm)**	**Runoff (mm)**	**Throughfall (mm)**	**Stemflow (mm)**	**Interception (mm)**	**ET (mm)**	**Reference**
Pr (San Ignacio T8)		1276			1030			658	Huber and Trecaman (2004)
Pr (Povernir T1)		1090			714			920	
Pr (Povernir T2)		1288			813			1056	
Pr (Povernir T3)		1256			938			902	
Pr (Povernir T4)		1256			1023			846	
Pr (Povernir T5)		1219			877			954	
Pr (Povernir T6)		1219			961			917	
Pr (Povernir T7)		1321			959			992	
Pr (Povernir T8)		1321			1022			900	
Pr (Huape T1)		1943			1628			933	
Pr (Huape T2)		2219			1903			1010	
Pr La reina pre–harvesting	1997–1998	183.1		61.4					Iroumé et al. (2005) *summer period presented only (December to March)
Pr La reina pre–harvesting	1998–1999	210.2		20.9					
Pr La Reina post–harvesting	1999–2000	388.2		67.9					
Pr La Reina post–harvesting	2000–2001	465.6		214					
Pr Los Ulmos 1	1999–2000	443.8		132.7					
Pr Los Ulmos 1	2000–2001	461.2		250.3					
Pr Los Ulmos 2	1999–2000	443.8		139.4					
Pr Los Ulmos 2	2000–2001	461.2		270.4					
Pr La reina pre	1997–1999	2286		917					Iroumé et al. (2006)
Pr La reina post	2000–2002	2907		2033					
E. nitens Los Ulmos 1	2000	2907		1800					
E. nitens Los Ulmos 1	2001	2645		1623					
E. nitens Los Ulmos 1	2002	3161		1690					
E. nitens Los Ulmos 2	2000	2907		1859					
E. nitens Los Ulmos 2	2001	2645		1673					
E. nitens Los Ulmos 2	2002	3161		1907					

Forest cover	Year	Reference

(*continued on next page*)

Table A1 (*continued*)

Land cover	Year	Precip (mm)	Streamflow (mm)	Runoff (mm)	Throughfall (mm)	Stemflow (mm)	Interception (mm)	ET (mm)	Reference
		Precip (mm)	**Streamflow (mm)**	**Runoff (mm)**	**Throughfall (mm)**	**Stemflow (mm)**	**Interception (mm)**	**ET (mm)**	
Native forest	1992–1993	3100					551	1033	Echeverría et al. (2007)
Meadow		3100						398	
Pr Palhuen	200–2003	1015			581	26	407	544	Huber et al. (2008)
Pr San Ignacio	1999–2001	1137			650	57	430	429	
Pr Porvenir	1999–2000	1189			752	61	426	563	
Pr Huape	1999–2000	2081			1720	45	316	656	
Pr 1	2000–2009	2149	705	188			357	1405	Huber et al. (2010)
Pr 2		2149	707	185			337	1388	
E. globulus 1		2149	439	130			215	1630	
E.globulus 2		2149	500	124			235	1511	
Pr La reina pre–harvesting	1997–1999	2286		917					Iroume et al. (2010)
Pr La Reina post–harvesting	2000–2009	n/r							

plot scale compared evapotranspiration rate (net ET / tree volumetric increase) of *P. radiata* within a latitudinal gradient (35° S, 37° S, and 39° S) and at same age (11–15 years old). The highest evapotranspiration rates were found in the northern (and driest) sites.

The observation of differences between catchments may not be inconsistent with the absence of differences at the catchment scale. Also, catchment studies integrate all the land uses in a catchment. It is possible, that areas not occupied by plantations such as the riparian edges behaved differently in *Eucalyptus* and *Pinus* covered catchments due to the different water use seasons (i.e., *E. globulus* used water much more rapidly in spring than *P. radiata*, while *P. radiata* used more water than E. *globulus* late in summer (White et al., 2021). Whatever the reasons for the apparent differences between catchment and stand studies in Chile and elsewhere, it is crucial that the results across scales are reconciled, most easily using a combination of long-term catchment datasets (particularly actual evapotranspiration measurements along streamflow and soil moisture), combined with remotely sensed data and modelling approaches.

3.3. Streamflow in forested catchments

Differences in transpiration between forest types can lead to variation in streamflow and in the extent of the effect of land use change and forest management on low flow, peak flow, streamflow, and the catchment water balance.

Alvarez-Garreton et al. (2019) fitted a regression model to runoff data from 25 forested catchments and the proportion of land cover. They showed that annual runoff decreased with increased plantation of 10,000 ha by between 2.2 % and 7.2 % across a gradient of rainfall from higher to lower. Pizarro et al. (2022) found contrasting results in a study based of 42 large catchment (200 to 24,000 km^2) mainly covered by native forest, plantation, or a mix of both and showed similar streamflow even in a period of low rainfall (see Esse et al. (2021) for similar results using SWAT hydrological model).

3.3.1. How does the water balance of plantations and native forest vary across the plantation management cycle?

3.3.1.1. Plantation management. Most of the results summarized in section 3.2. are concerned with annual catchment and stand water balances measured at a particular time within a rotation. The management or life cycle is very different for different types of native forest and particularly for *Pine* and *Eucalyptus* plantations in Chile. The forestry operations and natural disturbances (e.g., effect of wildfires over the water balance in Balocchi et al. (2020, 2022a) and White et al. (2020)) that occur during the life cycles of native forests and plantations may all change the water balance and, therefore, affect the water supply for other users. The comparison between forest types at different life stages is limited for Chile and elsewhere in the world, but there have been several studies in each forest type under consideration.

Perhaps the most dramatic operation in a plantation's life cycle is the harvest. After harvesting *Pinus* plantations are replanted with improved, site matched genotypes, while plantations of *Eucalyptus* can be either replanted or allowed to re-sprout from stumps. In general, streamflow increases for a period after the harvest of a *Pinus* plantation. After harvesting of 79 % of a *P. radiata* plantation that covered 34.4 ha catchment in southern Chile (39°-40° S), runoff increased by 110 % compared to pre-harvest mean (Iroumé et al., 2006), and in another study of the same 34.4 ha, runoff increased significantly for four years after harvesting and this runoff increase persisted for over eight years (Iroume et al., 2010). In a similar, high rainfall area, (39° S) *P. radiata* used about 13 % less water after harvest than before but continued to use more water than the neighboring grassland (Huber and López, 1993). However, a more recent study did not observe marked increase in runoff after harvesting of either 20–22 old year pine or 7–8 old year eucalyptus catchments in central Chile (Iroumé et al., 2021).

In Chile, we lack data quantifying the effect of initial stocking or thinning on either stand or catchment-scale water balance. In one plot-scale study, evapotranspiration estimated in low, medium, and high tree densities of an 8-year-old *E. nitens* plot was 79, 77, and 71 % of annual rainfall (Huber et al., 1998) while in a similar study of two densities of 12-year-old *P. radiata* in central Chile (37° S) net evapotranspiration was similar in both stands (Huber and Trecaman, 2002). The results for thinning plantations are consistent with results from Australia (White et al., 2014).

It is reasonably well established that tree water use increases up to the time of canopy closure (less than four years in *Eucalyptus* and seven to nine years in *Pinus*) (e.g., White et al., 2010). This pattern of water use by plantations was also found in a study in *E. globulus* in Chile, where evapotranspiration rates values were up to 30 % of the annual rainfall in the first year and to 58 % in the third year of measurements (Oyarzún and Huber, 1999).

3.3.1.2. Native forest. Several studies have observed increased runoff associated with wider riparian buffers of native forest (e.g., Alvarenga et al., 2017). Little et al. (2015) estimated that for each increase of 1 m in the width of the native forest riparian buffer, runoff increased by 1.4 % in southern Chile (39° S). In a northernmost site, Iroumé et al. (2021) found that in central Chile (37° S), catchments with a wider native forest buffer not only yielded more water but were also more resilient to drought.

There is little information from around the world on the water balance of recently planted native forest. Lara et al. (2021) in a decadal restoration study (from *Eucalyptus* cover to native forest) in southern

Chile (in a high rainfall area near Valdivia, 39° S) found that after 9 years of native forest growth (within 3 catchments of 3.74, 3.43, and 4.28 ha) streamflow and baseflow increased gradually when compared to the *Eucalyptus* plantation streamflow (5.26 ha). Frêne et al. (2020) analyzed the effects of different successional native forest stages near Valdivia (39° S) on streamflow. They found a greater streamflow in the old-growth catchment compared to a secondary 110-old year forest and shrub covered catchments. Within the same study site, Frêne et al. (2021) analyzed streamflow yield at different rainfall intensities. They found a greater streamflow in a secondary forest than in old-growth forest and dense scrub vegetation within a high rainfall intensity. In another study in an old-growth Patagonian rainforest (42° S) stand, evapotranspiration was greater than that of a younger stand (Gutiérrez et al., 2014). In an Andean range native forest, water balance study (around 5,000 mm rainfall) in southern Chile (39° S) Oyarzún et al. (2011a) compared seven stands of evergreen, deciduous and mixed native forest species (pristine, harvested, unmanaged, and managed). A relationship was found between the structure of the forests and the rainfall partitioning (throughfall, stemflow, and interception). For instance, interception varied from 11 to 36 % (median monthly) with one of the evergreen pristine stands with the highest interception values and the harvested evergreen stand with the lowest. These results suggest that revegetation with native forest will enhance streamflow regardless of which type of mature forest is replaced.

3.3.2. How do plantations affect peak flow and low flow?

While the effects of native forests and different plantation species on annual streamflow are important for large scale water planning, the effect on summer or low streamflow is more important for the rural communities, which extract water directly from streams and wells within and near the forests. Peak storm flows and the rate of recession are important hydrological parameters that can be used in the prediction of low or summer flow (e.g., Eng and Milly, 2007).

3.3.3. Low flows, baseflow, and summer flows.

The establishment of plantations has generally been associated with reduced summer (December to March) flow and baseflow (Oyarzún and Huber, 1999). Replacement of native forest with pine plantations can reduce streamflow in summer by as much as 28 %, although, in some cases, pine has also been associated with an increase in flow (359 L s^{-1} km^{-2} in pine and 274 L s^{-1} km^{-2} in native forest) in high rainfall period near Valdivia (Otero et al., 1994). Lara et al. (2009) developed a relationship considering six 140 to 1,462 ha catchments between proportional native forest cover and summer flow and found that a 10 % increase in native forest cover resulted in a 14 % increase in average summer flow relative to that coming from commercial plantations. The same difference was found in larger catchments in central Chile (e.g., Little et al., 2009), but other authors did not find significant differences in the other larger catchments in south central Chile (38° S) (e.g., Iroumé and Palacios, 2013).

While most studies show that plantation expansion and land cover change might reduce flows during the summer months (January, February, and March), there are some exceptions in the literature. One way of quantifying the effect of a change in land-cover on streamflow responses, and baseflow, is to fit an exponential decay function to the relationship between flow and time and estimate the decay parameter (α). In a latitudinal transect (35 to 40° S latitude) considering eight 13 to 112 ha catchments, in south-central Chile, Balocchi et al. (2021a) showed that α was not affected by forest type (*P. radiata*, *E. globulus*, *E. nitens* and native forest) in winter (May to September), but however more evident but not significantly different between the different forest types including native forest in summer (October to April). Average α values were not significantly different between native forest and pine plantations, nor between native forest and *Eucalyptus* spp. plantations. After harvesting, summer streamflow increased over the preharvest values (Iroumé et al., 2005), and this effect of water loss persists for up to seven years after harvesting and re-planting (Iroume et al., 2010).

3.3.4. Peak flows

The peak flow in response to a rainfall event is also an important characteristic of the hydrograph, which is relevant to rural communities due to potential damage to water supply facilities and water quality issues (e.g., sediment load). Peak flows are also affected by the establishment and management of plantations. When 79 % of a 34.4 ha *P. radiata* catchment in southern Chile (39° S-40° S) was harvested, a mean increase in peak flow of 32 % was observed (Iroumé et al., 2006), and the effect was similar for winter and summer storms (Iroumé et al., 2005). The post-harvesting effect of increased peak flow can persist for up to eight years after replanting (Iroumé et al., 2011). While most studies have observed an increase in peak flows after harvesting, at least one example shows no change in peak flow after 80 % removal of the pine plantations (Birkinshaw et al., 2011). Bathurst et al. (2020) found for a 34.4 ha catchment (40° S) that at lower rainfall events, forest cover may decrease peak discharge, but at higher rainfall events forest cover appears to have a lesser effect on the reduction of peak flows which likely depend on the antecedent soil moisture. Bathurst et al. (2022) analyzed the effect of afforestation on peak flow in four 434–1,545 km^2 large catchments in southern Chile. While they found differences when land cover change occurs (i.e., changes in annual evapotranspiration and runoff), the effect on peak flows was unclear.

Four studies have compared peak flow among catchments with different forest types, and none have observed any differences. The latter is true for a comparison of *E. nitens* and younger *P. radiata* (Iroumé et al., 2006), conversion from native forest to plantations (Pizarro et al., 2005, 2006) and a *meta*-analysis of data from 42 200–24,000 km^2 catchments with native forest, plantation and agriculture as land cover (Pizarro et al., 2022, 2021).

3.4. Water quality

3.4.1. How do plantations and their management affect water quality?

3.4.1.1. Sediment yield and assessment mechanisms.

We found 21 studies that reported results on forest cover and management effects on sediment transport, suspended sediment, and/or sediment modelling. In Chile, this work commenced in the early 1990 s (Iroumé, 1990, 1992).

As noted earlier, afforestation with *P. radiata* was an important strategy for controlling erosion and soil loss from land previously cleared for agriculture (Toro and Gessel, 1999). Forest cover, including *P. radiata* stands of different ages and management, and second growth *N. obliqua* stands were found to reduce sediment yield 97.5 % in native forest to 94 % in pine plantations relative to agricultural land cover (Ibarra et al., 1996). The native forest stands had the lowest erosion rates (Otero et al., 1994).

Results from comparisons between plantation types and comparison between plantations and native forest are variable and seem very dependent on the age and successional stage of the plantation. At hillslope scale, annual soil loss was, in general, not significantly different between native forest and plantations (five treatments, two 1-year old, two mature, and one harvested stand) in south-central Chile (37° S), but soil loss was highest from the youngest (1 year old) pine plantations (Aburto et al., 2021). At the catchment scale (<200 km^2), sediment exports decreased after plantation establishment (Pizarro et al., 2020). In one comparison between pine and *E. globulus* plantations, pine yielded more sediment than eucalyptus (Huber et al., 2010), but again this may be so because canopy closure in pine occurs at a later growth stage. When an *E. globulus*-planted catchment was compared to a native forest, suspended sediment was a bit higher in *E. globulus* (368 kg ha^{-1}) than native forest (305 kg ha^{-1}), and these differences were due to fact that the plantation was recently established, so it was in a transition from prairie to plantation (Oyarzún et al., 2011b). However, other results showed no significant differences between the pre- and post-harvest sediment yield in southern Chile in *E. globulus* catchments (Cuevas et al., 2018). Yet, this same study agreed with the minimum requirement for native forest riparian strips (22 m wide) for catchment protection for post harvesting sediment yield. Although, Little et al. (2015) concluded that an increasing width of riparian native forest can diminish nutrient loss. Within a study of native forest stages near Valdivia (old-growth, shrubs, and successional forest), Frêne et al. (2021) found a greater suspended sediment yield from catchments cover with shrub vegetation than in catchments cover with old-growth and secondary forest.

In summary, we found no measurable difference between the sediment yield from closed canopy plantations (<7 years) and native forest, but before canopy closure, more sediment can enter stream from plantations than from undisturbed native forest.

The distribution of harvesting operations in catchments and the management of harvesting residues will both have important consequences for reducing the potential for sediment transport after harvesting plantations. A comparison of harvested and undisturbed *P. radiata*, Oyarzún and Peña (1995) found that soil losses were much more significant from the harvested area than from the unharvested area, as the harvested area produced greater runoff. Oyarzún (1993) found greater soil loss when residues/burning was performed after clearcutting.

It was also found that microtopography acts as the first-order control of runoff generation and erosion processes in a small-scale rainfall simulation study in central Chile (Mohr et al., 2013). Mohr et al. (2014) within 3 small-catchments (12–9 ha) in south-central Chile annual suspended sediment increased between increased by 46–86 % after clearcutting of *P. radiata* compared to an uncut control. Iroume et al. (2010) studied the sediment yield within three small catchments (94, 12, and 17 ha) in southern Chile and they found a greater sediment load in the catchment with the lowest plantation cover (33 % of *P. radiata* and *Eucalyptus*) and the lowest sediment load in the bigger plantation area (81 % *Eucalyptus*). In another study, before and after harvesting within a small *P. radiata* catchment (34.4 ha, 40° S), measurements were not conclusive regarding the effects on sediment yield (Birkinshaw et al., 2011). In a paired catchment sediment study, Schüller et al. (2021) analyzed forest activities (harvesting and replanting). They found that the stream channel was the main source of sediment load pre harvest, and a catchment with forest activities yielded twice as much sediment compared to the control catchment. This higher sediment load was due to the activation of catchment slopes and forest roads. For example, from a sediment core analysis in the Lanalhue lake in central Chile (37° S) Alaniz et al. (2021) found a correlation of the increasing erosion and organic matter content within this lake and the afforestation with exotic plantations within the last 30 years.

Tracers have frequently been used to study hydrological processes, biogeochemistry and sediment sources at plot and catchment scale. Among the most used tracers in Chile, fallout radionuclides (FRN) have been used to trace and fingerprint sediment sources. Schüller et al. (2013) characterized sediment sources in a *P. radiata* covered catchments under a low (1,200 mm yr^{-1}) and high (2,500 mm yr^{-1}) rainfall regime using ^{137}Cs and $^{210}Pb_{ex}$. The relative contribution after harvest from slopes increased from 16 to 25 %, while the contribution from road increase from 37 to 45 %. Bravo-Linares et al. (2020) studied sediment sources in different land covers history. They found a greater sedimentation processes in forestry activities compared to wheat land cover, however, wheat farmland added an important sediment contribution in the 1950 s and 1970 s. Recently, studies using Compound specific stable isotopes (CSSI) done by Bravo-Linares et al. (2018) and Muñoz-Arcos et al. (2021) identified the sediment source within an exotic plantation catchment before and after wildfire. The main conclusion was that before the wildfire, sediment came mostly from unpaved roads, but the hillslope became the dominant source after wildfires. Similar changes in source contributions associated with clearcutting were documented for the high rainfall regime, where the relative contribution of slopes increased from 10.5 to 30 %, while the contribution from roads grew from 10 to 20 %. These results indicate that changes in sediment sources are closely related to clearcutting. However, they are also influenced by the amount of rainfall that occurs after clearcutting. The authors emphasize the need to implement better management practices during forest harvesting to reduce the increase in sediment mobilization from catchment slopes and roads, which can result in the loss of valuable soil and nutrients from the forest floor, causing degradation of the water quality of adjacent streams. Some of the applied solutions to this problem include using woody debris to create sediment barriers (Schüller et al., 2010) and using riparian strip vegetation to enhance sediment trapping and nutrient retention (Cuevas et al., 2018; Little et al., 2015; Oyarzún et al., 2007; Oyarzún et al., 2011a; Wang et al., 2005).

Some mathematical modelling has also been applied to suspended sediment in *P. radiata* covered catchments and after forest operations. Fuentes et al. (2001a, 2001b) found that the best predictive variables in a model to predict suspended sediments were monthly precipitation, water discharge, truck traffic, and forest road length.

3.5. Hydrological modelling and climate change

Although empirical studies of forest hydrology and ecohydrology are made under a unique set of climatic and hydrogeological conditions, their results have helped to feed hydrological models (i.e., for calibration purposes). This section looks at the process-based modelling that has been done in Chile to analyze and disentangle the effects of land use change and climate trends in the local forests and their hydrology (Table 3).

In general, hydrological modelling is scarce in Chile. Particularly it has been performed through two models (i.e., a physically based distributed precipitation-runoff model (SWAT) and TETIS). An exception is Flores et al. (2021) study who analyzed the performance of three 1D lumped hydrological models over four small catchments (22–40 ha) with different land cover in South-central Chile (i.e., native forest, Monterrey pine, and *Eucalyptus nitens*). They found the best potential evapotranspiration method for modelling streamflow within catchments as well the best hydrological model (i.e., GR6J).

Using the SWAT model at a mesoscale Andean watershed with changes in land use in central Chile it was predicted that the mean annual discharge increased 7 % under agriculture from that under native forest and decreased 10 % under pine plantation management (Stehr et al., 2010). In the same catchment the effect of future climate on streamflow, particularly during summer, was modelled and this indicated that from 2018 to 2028 and then to 2038, trends of reduced rainfall will cause a further reduction in streamflow from the present levels (Aguayo et al., 2016), and in some scenarios reductions in summer flow of up to 50 % were predicted. Another SWAT study, about the effect of land cover change in central Chile, indicated that a decrease in annual streamflow within the three scenarios studied (plantation coverage from 35 % to 63 % for 1986, 2001, and 2011) with an increased evapotranspiration and surface flow, and a reduced percolation and lateral flow (Martínez-Retureta et al., 2020). Gimeno et al. (2022) used SWAT to simulate the effect of an earlier application of forest conservation policies in a large catchment (1,026 km^2) in southern Chile. Within three scenarios (baseline, native forest recovery/protection, and pristine) they found little differences in inflow from baseline of 0.3 % in the native forest recovery scenarios and 3.5 % within the pristine scenario. However, larger differences are obtained reaching until 3.1 % and 21.2 % during the dry summer periods.

Climate Change analysis was introduced in the study of Martínez-Retureta et al. (2021) also with SWAT, where various components of the hydrological cycle in two 299–651 km^2 large catchments located in the foothills of the Andes (38° S). The model predicted increases in average annual evapotranspiration of 5–6 %, a decrease in percolation of 9–10 %, a reduction of surface flow of 2 %, and 7–8 % increase in depth to groundwater. These changes were associated with 5 % decrease in streamflow in the same catchments. The disentangling of land cover and climate changes effect on hydrological variables was investigated by Galleguillos et al. (2021) also using SWAT and results showed that change from shrubland to pine plantations decrease the annual mean streamflow between 2.5 and 17.3 % according to the amount of cover replaced in a coastal semi-arid catchment in central Chile. The study also forecast an increase in water yield of between 2 % and 11 % when pine was removed from the catchment headwater. Although, when future climate change scenarios are mixed with land use changes, annual streamflow decreases are much more severe and predicted to be between 23 % and 46 % according to contrasting climate change projections and the same land cover replacement scenarios (Galleguillos et al., 2021).

A simulation of trends for three forest management scenarios over 25 climate scenarios within eight forested catchments in the south-central Chile using TETIS (Vélez et al., 2005), indicated that smaller catchments (<20 ha) in the drier scenarios deliver more streamflow when reducing the forest cover (either plantations or native forest). Larger catchments (>50 ha) in all climate scenarios deliver more water when full forest cover is present. These results show how climate change (i.e., reduction of rainfall and/or increasing evapotranspiration) significantly influenced water yield (Barrientos et al., 2020).

4. A vision for Chilean forest hydrology; research gaps identification

Here we summarize the primary research gaps that we have identified from this review. In brackets is specified each research gap found.

1. Long-Term Ecosystem Research (LTER) sites. The only two publicly available and continuous dataset from small catchments (<100 km^2) are Balocchi et al. (2021b) which comprises data from ten 5 to 112 ha catchments in southern-central Chile (private catchment network) since 2008 (https://www.hydroshare.org/resource/4b517deaa07243aa8c46a58646dd4281/) and Lara et al. (2021) which comprises data from four 3.43 to 5.26 ha catchments in southern Chile since 2006 (https://www.cr2.cl/datos-cuencas-restauracion-reservavaldiviana). A detailed description of the catchments under research provides a comprehensive [1] forest structure characterization, soil types, root densities and other available characteristics which is uncommon in national forest studies. However, each dataset relies in streamflow and very little extra information (i.e., rainfall) is collected. More widely available data sets of these variables would help standardize studies and expand understanding of forest dynamics and water resources. At a regional and national scale, the CAMELS dataset (Alvarez-Garreton et al., 2018) holds hydrological variables such as daily streamflow, rainfall from different sources, reference evaporation (ETo), and the land uses within each large catchment (https://doi.org/10.1594/PANGAEA.894885). However, this dataset is the only one with free access in the country at landscape level monitoring despite the numerous small catchments (most of them paired) monitored by government and private institutions (the majority below to the privates). Most water balance data are for closed-canopy plantations and mature native forests. Therefore, there is an urgent need to quantify the water balance and its variables at plot, stand, and catchment scale of young stands of native species and plantations. The quantification of the water balance is needed to support vegetation recovery programs and to integrate space and time to scale the effects of plantations on water balance in addition to designing catchments with a range of plantation ages to mitigate risks to the local water supply. Therefore, these LTER sites should collect data and information for/from [2] water demands from forested catchments, [3] streamflow change among land cover, the [4] effects on timing and seasonality of peak flows between land covers, and the [5] availability of small catchment data within the country.
2. Sediment and water quality. Given the widely known impact of forest roads on stream water quality there is a surprising lack of studies on roads and their relationship to water inputs and sediment to streams. Forest activities require roads, gullies, etc., and their impact on water resources is still poorly known, even though these are the primary sources of sediment yield within forested catchments (e.g., Marden et al., 2014). While there are few research studies on sediment production in native forest and plantations (e.g., Pizarro et al., 2020), many more are needed; there is a gap in quantification of sediment production within forested catchments in the country. Another gap is the effect on water quality of forest activities such as pruning, thinning, logging (terrestrial and aerial) and harvesting, as well as the size necessary for buffer zones. Thus, [6] the effect of the forest operations impacts over water quality site specific (i.e. slopes, soil types, etc.) and [7] forest roads and its relationship with water and sediment yield (peak flows) is required.
3. Unknown forest groundwater systems. Despite the attempts by some papers to address groundwater topics no boreholes exist in any forest catchments in the country. Also, the National Hydrogeologic map is at a scale of 1:1,000,000 (Servicio Nacional de Geología y Minería, 2003) that makes it difficult to develop more complex studies at this resolution, especially within small catchments. Surface-groundwater interactions are also unknown as well as the relationship between soil water storage and the soil-vegetation complex. Detailed catchment studies require detailed digital elevation models and detailed geology, thus increasing the resolution of the National Hydrogeologic map would greatly assist attempts to understand the groundwater and the vadose zone. Additionally, it would significantly enhance progress if each new activity in monitored catchments includes activities such as geophysics, soil survey, etc., to verify and estimate below ground characteristics. Thus, we need better known [8] aquifer configuration and surface–groundwater interactions, [9] forest groundwater systems within coastal range (e.g., detailed geology, geophysics), and [10] the effects on timing and seasonality of low flows, and link to meteorological droughts.
4. Lack of knowledge of soil properties in the Andes and coastal range. In Chile there is only information on soils in the agricultural zones and therefore we have severe limitations in undertaking hydrological studies in forested catchments. There is a clear need for increased focus on soil properties studies and especially of native forest and plantations soils where, to our knowledge, there has been little or no previous work (see Soto et al. (2019) study for soil physical properties in different land covers). In fact, although the Natural Resources Information Centre of Chile (CIREN) soil database the only publicly available dataset in Chile covers around 24 % of the Chilean territory (Padarian et al., 2017), with an inadequate cover of native forest and plantation zones. Additionally, since private forest companies usually have valuable information, such as LIDAR data and several soil properties derived from soil pits, the science would be greatly enhanced if such data and products were more widely available. Then, the [11] soil physico-chemical and hydropedological properties of soils, the [12] erosion control soil activities (i.e., evaluation of tillage, trenches, etc.) is required. For example, a follow up study of Huber and Trecaman (2000) would have been valuable knowledge regarding soil water distribution under pine plantation.
5. Development/recovery of soils under forest cover. Most of the forest plantations were previously planted on land with wheat crops, native forest, or were heavily burned for agriculture and settlement expansion. There has been some research regarding soil characteristics (see Soto et al. (2019) and Cifuentes-Croquevielle et al. (2020)), however, there is not a single study on soil recovery from wheat-burned land on the coastal range, which later was afforested or under forest restoration. As, with hydrology, time matters and the effect of changes in soil properties can take several years or even centuries to become apparent (Andréassian, 2004). Therefore, [13]

soil properties evolution should be carefully studied to determine whether afforestation can recover water storage and carbon sequestration properties of these soils.

6. Extent of the effect of forest management and restoration. It has been a common approach in small catchment research to focus mainly on the effect of harvesting activities on streamflow. However, there is a lack of connections between large catchments and the effect of land cover change and other activities over the hydrological processes or the scaling effects from small catchments to large river basins. Water resources management in the public sector usually applies over large watersheds (>100 km^2). It has been discussed that extrapolating from small to large scale catchments can produce errors in the hydrological estimation (Kirchner, 2006) and therefore, when relying on observations at the small scale, in policy making strategies. Additionally, an important topic is the ecological restoration and its effect on the water cycle. We are aware of only one study regarding hydroecological restoration (i.e., Lara et al., 2021) in Chile. The [14] extent of the effect of forest management and restoration and [15] plantation densities and its effect in ET is require coping the effect of forest management.
7. Riparian/buffer effects. While riparian vegetation effects have been studied for impacts on water quality and biotic composition before and after forestry activities (e.g., Fierro et al., 2017; Guevara et al., 2018) or riparian restoration for forest fauna connectivity (e.g., Rojas et al., 2020), the effects of riparian zones on streamflow in forested catchments has been poorly explored in Chile (i.e., Little et al., 2015). Although riparian strips are part of stream management and land use activities, so the effect on streamflow of a new riparian zone, or an increase or reduction of such, versus a young or old plantation riparian zone is still not well understood (Segura et al., 2020). Also, the [16] native forest structure and composition in riparian zones and its effect over water balance and sediment yield is also required.
8. Native forest ecohydrological effects. Little is known regarding [17] native forest water balance within Chile. We can see that there are not many studies in native forest in the central zone (e.g., Huerta et al., 2019; Balocchi et al., 2020) nor in sclerophyllous forest or in sectors of pristine native forests in the central coastal mountain range (e.g., Galleguillos et al., 2021). As an alternative land use to plantations, the gap in native forest studies and their hydrological function should be addressed and particularly regarding the effects of climate change.
9. Climate change effects on water resources and land use change. We have strong evidence of the effects of climate change in Chile (Garreaud et al., 2017, 2020) on precipitation and temperature patterns. However, very little is known about its effect on water resources within forested and afforested catchments, particularly in central-southern Chile. While there are three studies that involve climate change and land use change (i.e., Barrientos et al., 2020; Martínez-Retureta et al., 2020; Galleguillos et al., 2021), they are recent denoting a lack in our knowledge in future climate scenarios. Additionally, we have not explored how land use change will impact regional climate (precipitation, temperature, and humidity) and how this impact will interact with climate change and the hydrologic cycle (Salazar et al., 2015). Special attention should be given to the trade-off between Carbon and Water cycle since tree planting is proposed as a strong mitigation option to climate change (Bastin et al., 2019). Thus, the [18] effect of climate change over a bigger number of catchments, the [19] understanding of the tradeoff water yield and carbon sequestration, and [20] a multimodel approach for a decision support system for public policies is required.

5. Towards a collective vision for the future of Chilean forest hydrology research

Overall, the science in the Chilean forest hydrology literature was found to be consistent with other studies worldwide. For instance, after a review of up to 75 scientific papers on hydrology in Chile (a) an increasing water yield was found in all studies after harvesting or forest cover removal, which is similar to other findings worldwide (e.g., Bosch and Hewlett, 1982). Similar findings in (b) peak flows increasing after forest canopy removal (harvesting) to other studies (e.g., Beschta et al., 2010) and consequently a (c) detriment in water quality as well (e.g., Safeeq et al., 2020). Also, most paired watershed studies have focused on the effect of forest harvest and regeneration on streamflow (McDonnell et al., 2018). However, these studies are small in spatial scale (<100 km^2) and, therefore, there is no quantifiable expression of the hydrogeological variability. Nearly all paired watershed studies show an initial increase in streamflow after forest harvest, followed by a gradual decline as forests regenerate, and in many cases seasonal or annual water yield falls below water yield in the reference catchment (see Goeking and Tarboton, 2020). Despite these similarities to other studies, there remain questions on the hydrological relationships between our Mediterranean climate ecosystems, the complex geology and topography in Chile.

Predicting the impacts of forestry on catchment processes and their associated effects on water flow is complicated and climate variability sometimes hides the cause-effect relationships that can exist between forest and streamflow, thus hindering or misinterpreting the effects of management forest on the water resources (Alila and Beckers, 2001). Hence, there is the need for attribution studies to disentangle impacts of different factors on the water balance at the catchment scale. Given this background, field measurements should be complemented with numerical modeling, which helps to understand better the cumulative effects of forest management on watersheds (Ziemer et al., 1991; Dunne, 2001). From a scientific perspective, models are important tools that help to interpret data and relationships between measurements at the plot and watershed scales (Hillel, 1986), and provide a medium to test our hypotheses and understanding of catchment processes. Modelling also provides an opportunity at minimal costs and time to investigate forest management effects on the water resources. Currently, there are numerous reviews of hydrological models with a marked emphasis on being able to interpret effects across forest management or climate change (Pike, 2003, 2007; Hutchinson, 2007; Werner and Bennett, 2009), which allows improved decision-making and reduced negative effects of forests on water resources.

We propose five key elements to advance a cross-scale collaborative effort to use new nation-wide catchment-scale networks, to promote common and complementary techniques in these studies, and to conduct transdisciplinary research to advance sound planning of forest lands in Chile. We strongly believe that the Chilean forest hydrology field can contribute not only locally but globally, especially in the context of climate change and the NDC framework.

In the following section, we discuss research gaps, and we propose a research agenda.

a. Long-term Ecosystem Research (LTER) sites: small-catchment (>100 ha) datasets have provided insightful knowledge regarding basic understanding on the land use change over the hydrologic cycle. Therefore, the implementation of a permanent monitoring program in small catchments with all the equipment needed to collect detailed field data on the water cycle variables and complementing these in larger catchments to capture hydrogeologic signals that in small catchments are not possible. The latter might be the case in soft rock systems and in areas where there has been much tectonic activity, such as Chile (Muñoz et al., 2016). These should represent different forest conditions across the country, from catchments with native forest (central zone), to catchments with only plantations (whenever possible). With this information, process-based models can be implemented to study the effects of climate change and land cover change with greater precision to identify resilient landscapes. Additionally, an intense program collecting rain, stream, soil, and other water sources samples for isotope analysis (oxygen 18, deuterium, tritium, etc.) which will add relevant information regarding water sources and ages (e.g., Cartwright et al., 2018). Dynamic distribution of tree water sources is an active field in hydrological sciences and should be further studied. Stable isotopes provide a myriad of simple yet significant information, which could help provide reliable data to stakeholders to be informed and decide regarding catchment water management decisions. Stable isotope signatures have been used to calculate the proportion of water that has been evaporated from soil (Gibson et al., 1993; Liu et al., 2006; Tsujimura and Tanaka, 1998). Although these estimations were yearly, the cost and processing time of these analyses has fallen significantly in recent years and, providing accurate and meaningful quantities, this would be a very worthwhile component of a monitoring program. In this regard, a network of several catchments monitors across the country can provide a unique dataset with the collaboration of several institutions (universities, research centers and the government) that can join in a multidisciplinary program. This will involve hydrologists, soil scientists, ecophysiologists, among other disciplines. For instance, the Forest Hydrology program from Arauco (Balocchi et al., 2021b) and CMPC timber companies (e.g., dataset in Iroumé et al., 2021), the Forest Institute (INFOR, Forest and Water program) and some others private forest companies such as MASISA (data not published), the drinking water companies (i.e., ESSBIO), and the FORECOS foundation (ONG for the Native Forest conservation). Therefore, a robust hydrology database developed with the private sector and the national universities should be created as an open database (e.g., Hydroshare website). The participation of the community and general public in the research is necessary. A community forest hydrology program could expand the current knowledge of forest systems in the coastal range (community data into a central location (data sharing) and a long-term monitoring such as Long Term Ecological Research Network – LTER, Critical Zone Observatories: Research and Application – OZCAR-RI in France, Critical Zone Observatories – CZO in the USA, etc.).

b. *Emerging technologies:* that can be applied in hydrology open an opportunity to fill in gaps. Remote sensing is a valuable and precise tool to study the water budget (Lillo-Saavedra et al., 2021). Since in Chile there are large areas of forest and plantations, this tool will help us to cover larger areas that are not possible to cover with plots or small catchments. It would be possible to understand larger areas and how can influence water cycle. Its utility includes the transport and storage of water, snow and water vapor in the soil–plant-atmosphere system and addresses the complex interaction between forest vegetation and the abiotic system, e.g., water uptake by roots, which is controlled by atmospheric conditions and photosynthetic activity of plants. However, remote sensing is only a tool measuring from a great distance and requires careful ground truthing to convert large scale images to quantitative measures of ground conditions and processes. To understand and predict energy-driven processes, such as evapotranspiration or snowmelt, it is necessary to know the radiation and energy balances of a forest watershed. For example, CubeSats can estimate river flow (Junqueira et al., 2021), Satellite-Borne GNSS-R for soil moisture measurements (Li et al., 2021), or other novel remote sensing data to determine reservoir characteristics (Guan et al., 2021). High accuracy LiDAR products can address important questions such as forest growth, canopy, and branch structure (Jaskierniak et al., 2021; Neuville et al., 2021) together with forest composition (Lopatin et al., 2015) within large areas. It also can be possible to characterize understory, riparian, and weed plant structure (e.g., Tatum and Wallin, 2021) and detailed geomorphology (e.g., Scheip, 2021). Internet of Things (IoT) will also add new insights in forest hydrology through the readily accessible data in real-time on any number of environmental attributes. New technologies could measure at a high accuracy, for example, how trees growth, transpire, and their spectral characteristics (Valentini et al., 2019). Another innovation for transpiration measurements is the Ribbonized sap flow (RSF) (Jones et al., 2020). RSF avoids the use of "needles" or thermocouples inserted in the tree trunk and replaces this with a "tape" with sensors that is inserted in the trunk, avoiding misalignment problems of other equipment (Ren et al., 2017). Also, advances in mobile mass spectrometry (MS) will help to reduce the laborious work to analyze environmental traces in hydrology (Popp et al., 2021a). This advance would allow us to involve different techniques to achieve new hydrologic analysis. Mobile MS in conjunction with Radon measurement will help find groundwater seepages to streams or young water travel times (Popp et al., 2021b).

c. A robust strategy for measuring water diversion from streams and rivers. The National Water Directorate (DGA) measures the main rivers in the country and the private water vigilance associations the main irrigations channels (agriculture use mainly). Nevertheless, there is a gap in knowledge for small communities that rely on water from headwaters which are most dependent on forest and forested catchments. Forest companies have been concerned about this issue and now some initiatives have been proposed to guarantee freshwater from forested land (e.g., Desafio Agua Arauco) but also from the sites protected by the research community/universities and public sector (e.g. SIMOL project for water community management and community-based water monitoring). Involving the people who rely on this resource in this monitoring is essential. The participation of the community can be through the well known positive impact of the "citizen science" (Kobori et al., 2016) that can contribute to expand, particularly, small catchments dataset (e.g., streamflow and rainfall measurements) to improve hydrological modelling for water management.

d. A Chilean forest hydrology Science Group. The national science funding institution (ANID, Chilean National Agency for Research and Development) has no guideline regarding the forest water supply and hydrology monitoring nor a specific expert committee, even though an important proportion of the water used by humans in Chile comes from forested catchment headwaters (e.g., Balocchi et al., 2022b). For communication, we propose to start with an annual Conference/Symposium to generate a national report/document as a result. This paper should be updated regularly to track improvements and promote collaboration. A Hydrological knowledge improvement which might be a 2–3 day workshop where the scientific community analyzes what we have done and what type of measurement instruments are held by each group, what data to can be shared, or further investigated. Also, because there are many Chilean experts in the field, there is the opportunity to run short courses or summer schools which can improve the local understanding and knowledge in the field. To complement all these endeavors, a Chilean Water Science Society Group could be formed as a unified voice for the scientific societies, stakeholders, and, particularly, to the local communities, taking advantage of ongoing science education program, such as Explora, that systematically approaches primary schools.

e. Involvement of stakeholders and the community onto the forest hydrology science. A decision support system (DSS) for water managers, stakeholder engagement is needed. Experimental catchments with community/stakeholders' involvement can ensure that they are understanding how the system behaves in their area. This DSS should be based on a broader understanding of the landscape in terms of when and where we can plant or restore. These activities must include serious efforts in data collection to assure the long term monitoring, in order to ensure reliable modeling output results. Therefore, there should be a program in forest hydrology modelling scenarios, including future climate and the definition of key indicators for national spatial planning legislation. This concerted cooperative effort outlined above should result in better coordination between catchment management and forest laws.

6. Conclusion

The rainfall variation across Chile and the history of land uses changes creates a unique natural laboratory in which to study climate change and forest hydrology. Much of the global understanding of forests and their effect on the water cycle is complemented by this review of Chilean studies. Streamflow from forested catchments increases after harvesting, native forests use less water than plantations, and native forests yield less sediment than plantation and grassland/shrublands. However, mechanisms that control groundwater flow and availability in forested systems and their differences and how climate change would affect native and plantations ecosystems remain unclear. In this regard, we found 20 research gaps which are related to lack of basic information (particularly in groundwater topics), climate, the need to improve the understanding of key hydrological processes and the relationship between forests, streams and the ecosystem.

One of our findings is that the southern Chile (39° S) is intensely studied, therefore we propose that the focus for conducting research and developing strategies for managing the trade-off between timber, carbon and water production should be placed in low rainfall areas including the Mediterranean-climate area and the boundary with the temperate zone (500–1,000 mm) where climate change scenarios project sharply rainfall reductions and thus affecting water resources.

Future research should focus on tackling down groundwater-surface water interaction in both native forest and plantations (McDonnell et al., 2018), and how plantations rotations and management affect hydrologic cycle and water quality. All this must be performed through the formatting (and expansion) of LTER sites across native forest and plantations, between different management scenarios and climates. Always bearing in mind that the conclusions that can be drawn at one spatial scale cannot necessarily be extrapolated to other spatial scales, and that more case studies are needed to advance the knowledge of this research topic. Also, to investigate the effects of climate change and land use change and the proportion of both variables affecting water quantity and quality.

We need also further collaboration between public and private institutions, sharing research data, sites, and results, and we need to incorporate in our research proposals and research teams other disciplines to understand our complex forest environment from top to bottom and bottom-up approaches (e.g. Pizarro et al., 2021). Fuentealba et al. (2021), for example, reviewed the impact of forest science in Chile but, despite the importance of water, water resources topic was absent from their analysis.

Traditionally, the effects of forests, and mainly plantations, on downstream water supply, have been analysed studying only part of the forest cycle. This partial analysis, since tree establishment to the final harvest and subsequent reforestation, does not allow exploration the relations between forests and water, from a larger perspective. The use of process-based modelling including scenario analysis could tackle this issue alongside measurements. However, in poorly gauged catchments, simplified modelling tools could also represent a valuable solution. Finally, potential downstream effects must be reduced or mitigated, to, within the framework of hydrosolidarity (Falkenmark and Folke, 2002; Lima et al., 2012), permit to sustain drinking and irrigation water demands.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

No data was used for the research described in the article.

Acknowledgements

We would like to express our gratitude to Anton Huber who is the pioneer in forest hydrology in Chile. Probably without his insightful studies in the field knowledge would be not as it is today. The authors wish to express their thanks for the support given by Julia Jones and Antonio Lara and their insightful and helpful comments for improving this document. Jeff McDonnell is acknowledged for participation in a scoping workshop and for comments on an earlier draft of this manuscript. Also, thanks to the Chilean National Agency for Research and Development (Agencia Nacional de Investigación y Desarrollo de Chile, ANID) through project FONDECYT 1210932, ANID/FONDAP/15130015, ANID/FONDAP/15110009, ANID BASAL FB210015 CENAMAD, FONDEF ID20I10338, and to the doctoral scholarship ANID-PFCHA/Doctorado Nacional/2021-21210861 for the support on lead author (FB). J. Armesto acknowledges the support from ANID project FB210006. Finally, thanks to Catalina Segura and the anonymous reviewer whose help us to significantly improve this document.

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