Integrating Ecological Suitability and Socioeconomic Feasibility at Landscape Scale to Restore Biodiversity and Ecosystem Services in Southern Chile

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Introduction

Deforestation and forest degradation are major processes driving biodiversity loss and global environmental change (Lewis et al. 2019). From 2010 to 2015, natural forest area decreased by a net total of 4.7 million ha per year (FAO 2020). In rural landscapes, even larger areas are subject to degradation from low-intensity chronic disturbances such as grazing, selective logging, invasion of exotic species, and fire. These disturbances lead to changes in species diversity and composition, with major consequences for ecosystem functioning (Cadotte et al. 2011; Baraloto et al. 2012). The drivers and intensity of degradation vary by region, but the impact of forest loss and degradation can be felt at all scales, from global climate change to the declining economic value of forest resources and biodiversity and threatened local livelihoods (Mitchell et al. 2017).

Deforestation is relatively simple to detect and predict (Cayuela et al. 2006a; Zamorano-Elgueta et al. 2015; Miranda et al. 2015), but the identification of degraded forests at a landscape scale is not straightforward. Forest degradation has been defined as the reduced capacity of a forest to provide goods and services (Chazdon 2008). Remote-sensing methods for the identification of degraded forests are expensive, relying on high-resolution satellite imagery or LIDAR datasets to detect changes in canopy cover or above-ground biomass (Riva et al. 2017; Mitchell et al. 2017). In addition, imagery analysis methods are often ineffective at detecting below-canopy degradation, such as the removal of understorey vegetation or lack of overstorey recruitment (Mitchell et al. 2017). Alternatively, as shown by previous research in Latin America, socio-economic factors can be used to predict forest degradation (Reyes 2004; Reyes et al. 2015; Zamorano-Elgueta et al. 2012; 2014; 2015). Degradation can be the result of anthropogenic disturbances, including alien invasive species, firewood extraction, livestock grazing (causing soil compaction as well as trampling and removal of understorey) and other disturbances (Vásquez-Grandón et al. 2018). Prioritizing degraded forests for restoration is advantageous as restoration efforts tend to be more successful and cost-effective in these forests compared to deforested areas (Lewis et al. 2019; Austin et al. 2020). Considering both deforested land and degraded forests as target areas for forest restoration could thereby provide valuable policy guidance for governments and planners, especially in the context of global commitments to mitigate greenhouse gas emissions. Determining changes in forest cover using remote sensing can initially involve comparing images before and after an event, but this approach may oversimplify forest degradation processes. To robustly assess degradation requires integrating field data and remote sensing technologies (Aryal et al. 2021). Moreover, degradation events frequently occur at a fine spatial scale, sometimes smaller than the resolution of standard Earth observation tools, necessitating subpixel analysis for detection (Bullock et al. 2020).

Nature's contributions to human wellbeing include services such as firewood/energy production, food provision, climate change mitigation, water purification, and erosion regulation (IPBES 2021). Nature-based climate change mitigation strategies, leveraging natural ecosystem processes for carbon sequestration and storage, have long been acknowledged for their potential to significantly reduce global greenhouse gas (GHG) emissions (Adams et al. 1999). The Intergovernmental Panel on Climate Change (IPCC) underscores that such strategies, including afforestation, reforestation, improved forest management, and avoiding forest conversion, are crucial components of the global abatement portfolio (Smith et al. 2014). National and international commitments to mitigate GHG emissions prominently feature activities within the forest sector (Grassi et al. 2017). Previous research indicates that these mitigation efforts could contribute between one-quarter and one-third of the reductions necessary to achieve climate stabilization targets by 2030 (Busch et al. 2019). In this scenario, ecosystem restoration is recognised as a global priority (Aronson and Alexander 2013). For example, Chile has set the goal of achieving carbon neutrality by the second half of this century, with the land use, land-use change, and forestry sector playing a crucial role (Hoyos-Santillan et al. 2021). Key climate actions include afforesting 200,000 hectares, reducing forest degradation and deforestation, and implementing a landscape restoration plan covering 1 million hectares (Hoyos-Santillan et al., 2021). Limited funding, however, requires that any forest restoration initiative should be based on a careful identification of priority areas to maximise the ecological and socioeconomic returns of available resources (Perring et al. 2015). These priorities can be based on the area's biodiversity values, vulnerability to threats, contribution to landscape-scale processes or some combination of all these factors (Myers et al. 2000; Strassburg et al. 2020, Silva et al. 2023).

The identification of restoration priorities at the landscape scale is a multi-objective planning problem, in which nature conservation is balanced against social and economic outcomes (Kangas and Leskinen 2005; Silva et al. 2023). Multicriteria Analysis (MCA) provides a useful framework to address similar problems, enabling exploration of the implications of different planning options (Geneletti and Ferretti 2015). MCA allows the assessment and comparison of alternatives against a set of explicitly defined criteria, which account for the most important aspects in a specific decision problem (Adem Esmail and Geneletti 2018; Veronesi et al. 2017). A key feature of MCA is that it can handle the use of criteria that cannot be directly combined

using the same unit of measurement, in contrast to other approaches such as Cost Benefit Analysis (Geneletti 2010). In addition, MCA allows the explicit incorporation of the values and perspectives of different stakeholders, and simulation of the effects of these values on the final outcomes of the decision processes (French et al. 2009). This makes the approach preferable to other methods (such as linear programming) when the objectives of the analysis are to present a range of possible options, acknowledging that the most preferable option(s) will depend upon the values and priorities of the actors involved in the process.

The interpretation of the outcome of MCA may be less straightforward when compared to other methods, such as linear programming or cost-benefit analysis (Mouter et al. 2020). However, MCA provides a more comprehensive, adaptable, and inclusive approach for identifying priority areas for forest restoration, particularly when addressing the integration of multiple ecological and socioeconomic variables (Geneletti 2019; Mouter et al. 2020). By coupling MCA with Geographical Information Systems (GIS), spatial MCA tools allow outputs to be translated into readily understandable maps of restoration priorities at the landscape scale (Orsi and Geneletti 2010; Borda-Niño et al. 2017).

MCA approaches tools have been widely used, for example to: define restoration priorities (Marjokorpi and Otsamo 2006; Uribe et al. 2014), including wetlands (Qu et al. 2019); design natural protected areas and conservation networks (Geneletti 2004); inform the restoration of ecosystem services at the watershed scale (Trabucchi et al. 2014); guide landholder decisions at the property level based on conservation priorities in the surrounding landscape (Oakleaf et al. 2017); and to underpin climate adaptation strategies in multi-use landscapes (Maxwell et al. 2019). The incorporation of criteria related to socioeconomic feasibility is, however, less common. Spatial MCA methods which incorporate socioeconomic feasibility are likely to be particularly relevant in the context of ambitious global forest restoration targets such as those proposed by the Bonn Challenge, the UN Declaration on Forests, the 20 × 20 Initiative, and the IPCC. For example, Bastin et al. (2019) estimated that 0.9 billion ha of land have potential for forest restoration at the global scale. This assessment is based purely on potential tree cover under existing and future climate scenarios, however, and includes no assessment of how conflicting land uses are likely to influence restoration attempts. Moreover, identifying priority areas for restoration has typically targeted deforested areas due to the difficulty of identifying degraded areas at a landscape scale (Orsi and Geneletti 2010). The accurate identification of degraded areas is important because restoring such areas can be faster and cheaper than restoring deforested areas (Chazdon 2008).

The present study develops a context-specific spatial MCA model to identify priority areas for forest restoration on both deforested land and degraded forest areas in rural landscapes, with the purpose of enhancing biodiversity and relevant ecosystem services such as water provision and erosion regulation. The model proposed integrates ecological suitability with the socioeconomic feasibility of restoration interventions. Suitability is defined as the ability of a land parcel to maintain the provision of biodiversity and ecosystem services. Feasibility is a measure of how likely restoration interventions are to succeed given the socioeconomic context (e.g. probability of land cover conversion). The proposed model is illustrated with a case study in the Cordillera de la Costa (Coastal Range) of the Los Ríos region, southern Chile, and represents an innovative integrated model with clear potential beyond the region. This area has been classified as a biodiversity hotspot for global conservation due to high rates of biodiversity and species endemism (Myers et al. 2000; Tilman et al. 2017), reflecting the location of vegetation refugia during the last glacial period (Armesto et al. 1995). The area is also among the most threatened eco-regions in the world (Tilman et al. 2017), with forests under threat from conversion to exotic tree plantations (for pulp and paper products), clearing for pasture, human-set fires, and logging (Miranda et al. 2015; Zamorano-Elgueta et al. 2015). These patterns of degradation are particularly visible in the central and eastern parts of the study area, where most of the tree plantations are found and rural communities are concentrated. However, a significant proportion of well-conserved old-growth forest still remains.

Methods

Study Area

The study area is in the Valdivian rainforest ecoregion of Chile, and forms part of the Coastal Range. It covers approximately 2700 km^2 (Fig. 1) and ranges from 4 to 1000 m asl. The predominant climate is temperate with Mediterranean influence and annual precipitation between 2000 to 4000 mm. Soils derive from metamorphic and granitic rocks (IREN-CORFO 1964).

Broadleaved evergreen forests are the dominant vegetation type, comprising 79% of the total forest cover in the Los Ríos region (CONAF-CONAMA 2008). In the study area, the dominant land uses are commercial forestry (81,100 ha, 30% of the study area), private protected areas (52,000 ha, 19.3%), small farming properties (i.e. <200 ha as defined by the Chilean laws; 46,827 ha, 17%), medium-large farming properties (45,663 ha, 16.9%), and national protected areas (26,000 ha, 9.7%).
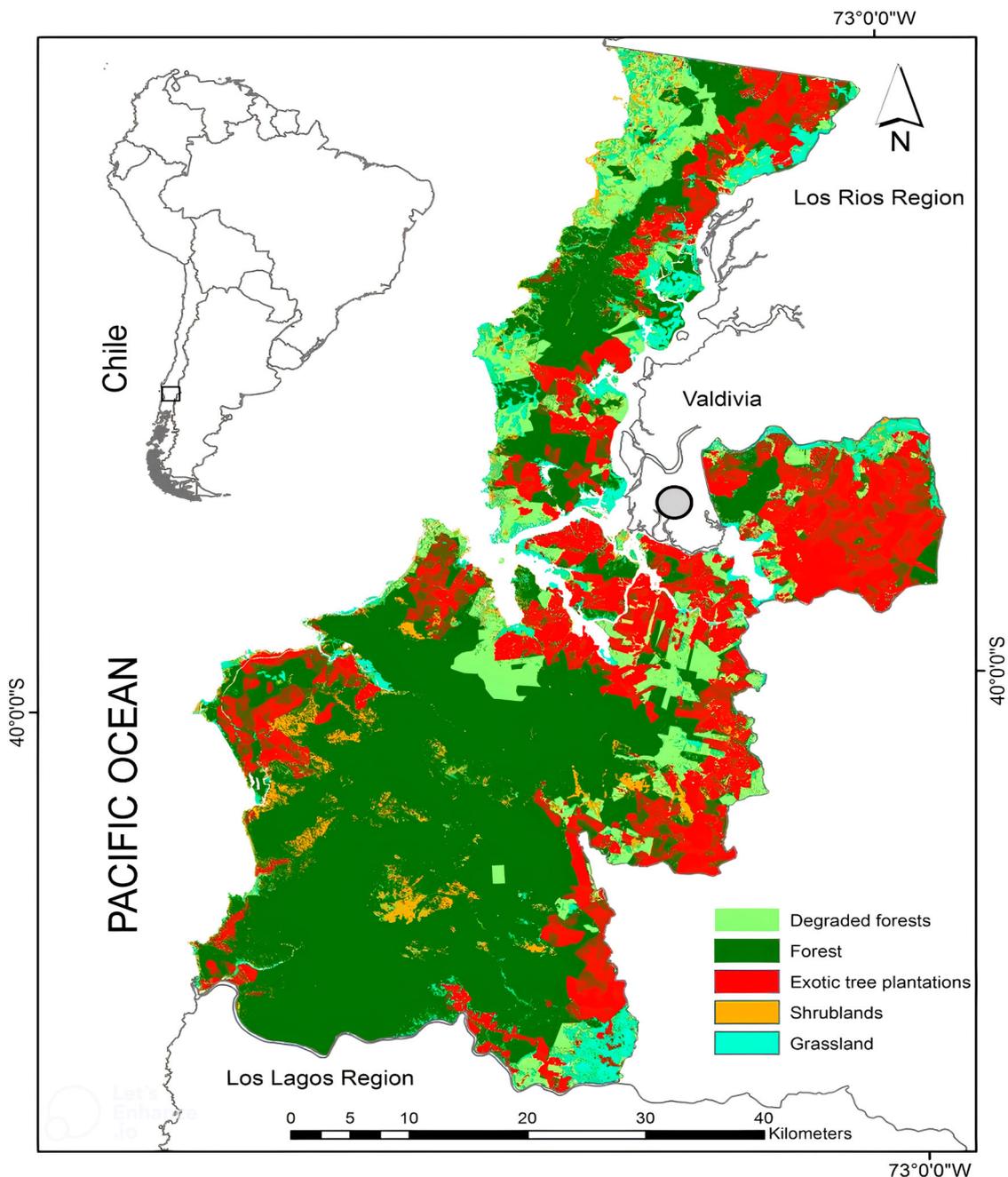
Fig. 1 Map of the study area showing major land cover types within the Coastal Range of the Los Ríos region, Chile. Adapted from Zamorano-Elgueta et al. (2015)

Multicriteria Analysis (MCA)

We followed four sequential steps to identify priority areas for forest restoration in our multicriteria analysis (Fig. 2): (i) defining target areas for restoration; (ii) assessing the suitability of land to restoration based on ecological criteria; (iii) assessing restoration feasibility based on socioeconomic criteria; and (iv) combining spatial suitability and feasibility to identify priority areas for forest restoration.

Defining target areas for restoration

Both deforested land and degraded forests were considered as target areas for forest restoration. All non-forested areas, including agricultural land, grassland, exotic tree plantations and bare ground were defined as deforested areas, since historically they were once part of a continuous forest landscape (Lara et al. 2012). A land use and land cover map was used to identify deforested areas. This map was obtained through the classification of a 30×30 m pixel
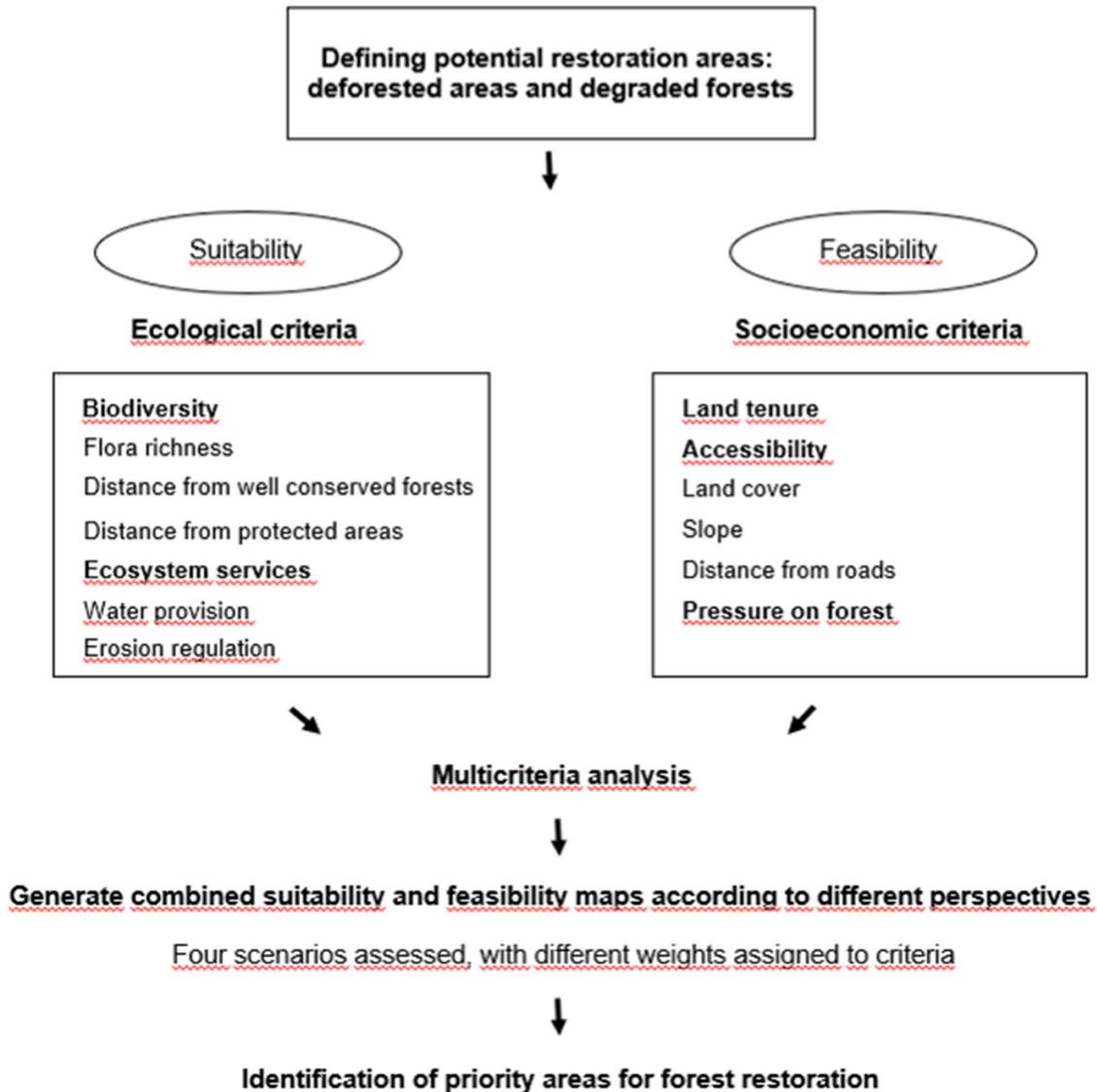
Fig. 2 Flow chart of the proposed approach. Bold headings indicate the general framework which may be applied across regions, while regular font indicates the specific approach taken for the Coastal Range region

resolution Landsat™ image (path 233, row 88) for the year 2014. The image was pre-processed using geometric, atmospheric, and topographic corrections, and then classified using a supervised maximum likelihood classification into the following land cover types: native forest, exotic tree plantations, shrubland, agricultural/grassland, and bare ground (Fig. 1; Zamorano-Elgueta et al. 2015). The 2014 image provides an appropriate spatial and temporal resolution for the study's objectives, as the main land cover changes in the study area occurred mainly during the 1985–1999 period (Zamorano-Elgueta et al. 2015; Miranda et al. 2015). Although some changes may have occurred in recent years, the general patterns relevant for the multi-criteria prioritization of restoration areas remain stable, allowing for the development of a robust methodology that can be easily updated with new images in the future.

In the Coastal Range region, socioeconomic factors strongly predict both the likelihood and intensity of cattle ranching and selective logging, which are major regional drivers of forest degradation in rural areas (Zamorano-Elgueta et al. 2014). High frequency and intensity of these disturbances in the study region are typically associated with small properties under 55 ha (Reyes 2004). Thus, forested areas on land units <55 ha were included in the MCA as potentially degraded areas.

Assessing the suitability of land to restoration based on ecological criteria

We assumed that the restoration suitability of a land parcel is directly related to its capacity to contribute, once restored, to biodiversity conservation, water provision and erosion

regulation. Such capacity was assessed at the pixel level according to different metrics that were mapped in a raster format, as described below.

Biodiversity Three sub-criteria, namely potential floristic richness, distance to well-conserved forests, and distance to protected areas, were used to assess the potential of each 30×30 m raster cell to support high levels of biodiversity. They were selected based on a review of the literature (Ianni and Geneletti 2010; Orsi and Geneletti 2010; Trabucchi et al. 2014) and data availability for the study area (see below). Potential floristic richness was used as a proxy of overall biodiversity levels. Distance to well-conserved forests represents distance to source populations for colonisation and ongoing landscape connectivity for maintenance of ecological processes such as seed dispersal, mating, and pollination (WCMC 2000). Distance from protected areas represents areas that play a major role in biodiversity conservation, given that protected areas are usually a representative sample of the regional biodiversity and buffer external threats such as exotic plant invasions (Margules and Pressey 2000). Reforesting in and around a protected site means both enhancing the forested ecosystem and creating a buffer zone that prevents the site from being disturbed (Orsi and Geneletti 2010).

To estimate the potential floristic richness, we assembled predicted species ranges, following a "predict first, assemble later" approach (Ferrier and Guisan 2006). To generate single species distribution models, we used an existing dataset comprising flora survey of 129 25×20 m field plots randomly allocated in the evergreen forests (the dominant native forests) of the study area (see Zamorano-Elgueta et al. (2014) for a further description of the sampling design). The average distance between field plots was 1600 m. To generate reliable statistical models, only the most abundant vascular flora species (i.e. those present in at least 10 plots) were analysed. Species distribution models were generated using the maximum entropy (Maxent) algorithm, a machine learning approach that has been widely applied in recent years to model species niches and distributions based on presence-only data (Phillips et al. 2017). Species distributions were modelled as a function of the following environmental variables: i) climate, including temperature (average temperature of the coldest month) and rainfall (average rainfall of the driest month); ii) topography, including elevation, slope, and aspect; iii) soil properties, including soil pH, soil depth, and drainage; and iv) land cover. Most of these data were obtained from the *Infraestructura de Datos Espaciales* (www.ide.cl), *Sistema Nacional de Información Ambiental* (www.sinia.cl), and Global Climate Data (http://www.worldclim.org/current) sources. Each model was then

converted from a continuous suitability map to a presence/absence binary map using threshold criteria (Jiménez-Valverde and Lobo 2007; Freeman and Moisen 2008). These maps were summed to obtain a richness model (Benito et al. 2013).

The distance to well-conserved forest areas was calculated using the Euclidean distance from each raster cell. Both well-conserved forest patches and protected areas were defined using the land cover raster shown in Fig. 1. The potential biodiversity value of each raster cell was then calculated by overlaying the resultant raster maps.

Ecosystem services Two measures of ecosystem services, water provision and erosion regulation, were estimated for the study region. Water provision was estimated using the relationship between mean annual direct runoff coefficient and native forest cover in each watershed, as proposed by Lara et al. (2009). This relationship follows the equation:

$$Y_i = 0.2271 \times FC_i + 0.2725$$

where Y_i is the mean annual direct runoff coefficient (quickflow/precipitation; Longobardi et al. 2003), and FC_i is the native forest cover (%) in a watershed. Watersheds were defined from high-resolution digital elevation data (ASTER GDEM, available at http://gdem.ersdac.jspacesystems.or.jp). The annual runoff coefficient can be used as an indicator of water provision. For example, Lara et al. (2009) showed a positive correlation between native forest cover and the annual runoff coefficient and, conversely, a negative correlation between the annual runoff coefficient and cover of exotic tree plantations.

Erosion regulation was estimated through a potential erosion map developed by the *Centro de Información de Recursos Naturales of Chile* (CIREN), which stems from the IREPOT model that integrates soil physical characteristics (e.g. texture, organic matter content, permeability, soil depth), topography (e.g. altitude, slope), and climate variables (e.g. rainfall), and has a spatial resolution of 30 m. The resulting map is categorised into five erosion classes: low, moderate, severe, very severe, and other uses. According to this map, the study area mostly belongs to three erosion classes: very severe (46%), severe (34%), and moderate (12%). Further information about this procedure can be found in CIREN (2010).

Assessing restoration feasibility based on socioeconomic criteria

Restoration feasibility was assessed by means of three variables - land tenure, accessibility, and pressure on forest, assuming that restoration interventions are in general more feasible where: (a) land tenure is stable (i.e. ownership is not likely to change in the short and medium term), (b)

access is easier and thus restoration is less expensive, and (c) distance from urban areas is greater and therefore the risk of forest degradation is lower (Sunderlin et al. 2005).

Land tenure For this study, the stability of land tenure was assessed in terms of property size and ownership. Properties were classified by size into small properties (<200 ha) and large properties (≥200 ha). Small properties were further divided into either sustainable (i.e. natural resources can be exploited in a sustainable way) and unsustainable properties based on an area threshold of 55 ha, as estimated for the study area by Reyes (2004). This area is the minimum amount of land necessary for a standard family of five members to sustainably exploit natural resources. This threshold was estimated integrating economic (e.g. balance between intra and extra-property incomes, estimation of the potential sustainable productivity), social (e.g. local organisation, social cohesion), and environmental features (e.g. forest and grassland productivity in terms of livestock capacity, size of the property and other land uses; Reyes 2004).

Large privately-owned properties, particularly those owned by *Forest Stewardship Council* (FSC) certified companies, are likely to have the highest restoration feasibility due to alignment with the goals of the FSC certification and, in general, the availability of financial resources for restoration actions. In public lands (excluding protected areas), feasibility is reduced due to widespread exploitation of these areas by neighbouring landholders as a source of fodder and refuge for cattle and for firewood production, with no control over livestock density or logging intensity. Based on these considerations, tenure types in the study area were ranked in decreasing order of restoration feasibility (and assigned related feasibility values) as presented in Table 1.

Table 1 Values assigned to convert qualitative or categorical criteria related to land tenure into a semi-quantitative range

Land tenure classes	Feasibility value
Land owned by forest companies certified by the Forest Stewardship Council (FSC)	1
Protected areas (public and private)	2
Large properties (>200 ha)	3
Other land owned by forest companies	4
Sustainable small properties (>55 ha and <200 ha)	5
Public properties	6
Unsustainable small properties (<55 ha)	7
Land with uncertain property rights	8

Lowest values correspond to maximum feasibility and the highest ones to minimum feasibility.

Accessibility Accessibility was assessed as a combination of land cover, slope, and distance from roads. Land cover classes were ranked according to the difficulty involved in converting the existing vegetation community to forest, i.e. degraded forests (1), shrubland (2), agricultural/grassland (3), and exotic tree plantations (4); noting that they are listed in decreasing order of feasibility for restoration, and the number in parentheses correspond to the assigned value. Increasing slope and distance from roads are both considered to decrease accessibility and therefore feasibility for restoration. Hence, for example, degraded forest or shrubland on gently sloping land which is situated close to a road would have high restoration feasibility.

Pressure on forests Pressure on forests, namely the likelihood that forests can be degraded by human activities, was measured as the combination of distance from human settlements, population density, and slope. Areas close to human settlements are at a higher risk of disturbance from harvesting of forest resources and fires. This risk is increased in areas with high population density but decreases in steeper areas (Bar-Massada et al. 2014).

Combining suitability and feasibility maps to identify restoration priority areas

Suitability and feasibility maps required the combination of the criteria described above, for which value ranges and measurement units were different. To produce comparable maps, all metrics were therefore converted to a 0–1 range by means of value functions (Keeney and Raiffa 1993). Value functions are the mathematical representation of human judgement on criterion scores and transform the original score of a given criterion into dimensionless values between 0 and 1, where 0 corresponds to minimum desirability and 1 to maximum desirability (Geneletti 2005). The value functions selected for this study belong to four main categories, depending on whether the relationship between desirability and the original score was linear (erosion regulation, potential biodiversity, land cover, slope, water provision, distance from forests, distance from protected areas, and land tenure) or quadratic (distance from roads and pressure on forests), and whether maximum desirability was attained at high or low original scores. Criteria that were expressed through classes (e.g. land tenure, land cover) were converted into the 0–1 range using look-up tables (Table 1).

After scaling each criterion to a 0–1 range, overall suitability and feasibility maps were generated by means of weighted summation of the relevant criterion rasters. Given that the assignment of weights is a subjective procedure and that this study did not rely on consultations with stakeholders or decision-makers to derive weights, four scenarios, both for suitability and feasibility, were considered to simulate

Table 2 Assignment of weights to each criterion under four different scenarios of suitability and feasibility of forest restoration

Suitability criteria	Scenarios of forest restoration			
	Biodiversity	Water provision	Erosion regulation	Balanced
Biodiversity	**0.5**	0.25	0.25	0.33
Water provision	0.25	**0.5**	0.25	0.33
Erosion regulation	0.25	0.25	**0.5**	0.33
Feasibility criteria	**Land tenure**	**Accessibility**	**Pressure on forest**	**Balanced**
Land tenure	**0.5**	0.25	0.25	0.33
Accessibility	0.25	**0.5**	0.25	0.33
Pressure on forest	0.25	0.25	**0.5**	0.33

Each scenario gave more weight to one particular criterion (0.5 weight, bold values) than to the others (0.25 weight) except the balanced scenario for which all criteria were assigned the same weight (0.33)

different stakeholders' perspectives (Table 2). Three scenarios assigned a greater weight to one specific criterion, whereas the fourth one assigned the same weight to all criteria.

The basic assumption guiding the identification of priority areas for forest restoration was that restoration should only be attempted if the proposed site is both suitable for restoration (i.e. able to support a functional, sustainable and diverse ecosystem) and feasible to restore within the existing socioeconomic context. Binary rasters were created for each of the four suitability and feasibility scenarios by assigning a value of 1 to the 10% most suitable/feasible cells and a value of 0 to all other cells. Then, the four binary rasters of both suitability and feasibility were summed, generating two rasters, one for suitability and one for feasibility, with cell values ranging between 0 (i.e. cells never identified as priorities) and 4 (i.e. cells identified as priorities in all the four scenarios). Finally, priority areas for restoration were selected as those that were prioritised under all suitability and feasibility scenarios (i.e. cells with value equal to 4 in both maps).

Results

Suitability and Feasibility for Forest Restoration

Cells of the suitability maps presented a high variation of values yet most of them showed intermediate values for all scenarios. They ranged between 0 and 0.94 for the water provision and biodiversity scenarios, and between 0 and 0.93 for the erosion regulation and balanced scenarios (Fig. 3). Cells ranked in the upper 10% for suitability (i.e. suitability = 1) for all evaluated scenarios represented an area of 53,193 ha (19.7% of the study area), whereas the area with the highest feasibility values for all scenarios was 59,136 ha (21.9%, Fig. 4). A large part of the territory had a value of zero for all scenarios, mostly corresponding to well conserved, old-growth forest areas. These areas were concentrated in the southern region of the study area (Figs. 3–5).

Priority Areas for Forest Restoration

Cells classified as potential areas for forest restoration were concentrated in the north, central and eastern parts of the study area (Fig. 5). These areas comprised 115,500 ha (42.78% of the study area), of which 86,563 ha (32%) were deforested areas and 26,894 ha (10%) were degraded forests.

Priority areas for forest restoration, defined as those prioritised under all four scenarios, were concentrated in the northern, central, and eastern parts (Fig. 6). The total amount of priority restoration areas accounted for 28,985 ha (10.7% of the study area), of which 20,052 (7.4%) were deforested areas and 8933 ha (3.3%) were degraded forests. They mainly occurred on land owned by forest companies certified by the FSC (15,817 ha, 5.85%), sustainable small properties (8781 ha, 3.25%), and protected areas (2345 ha, 0.87%).
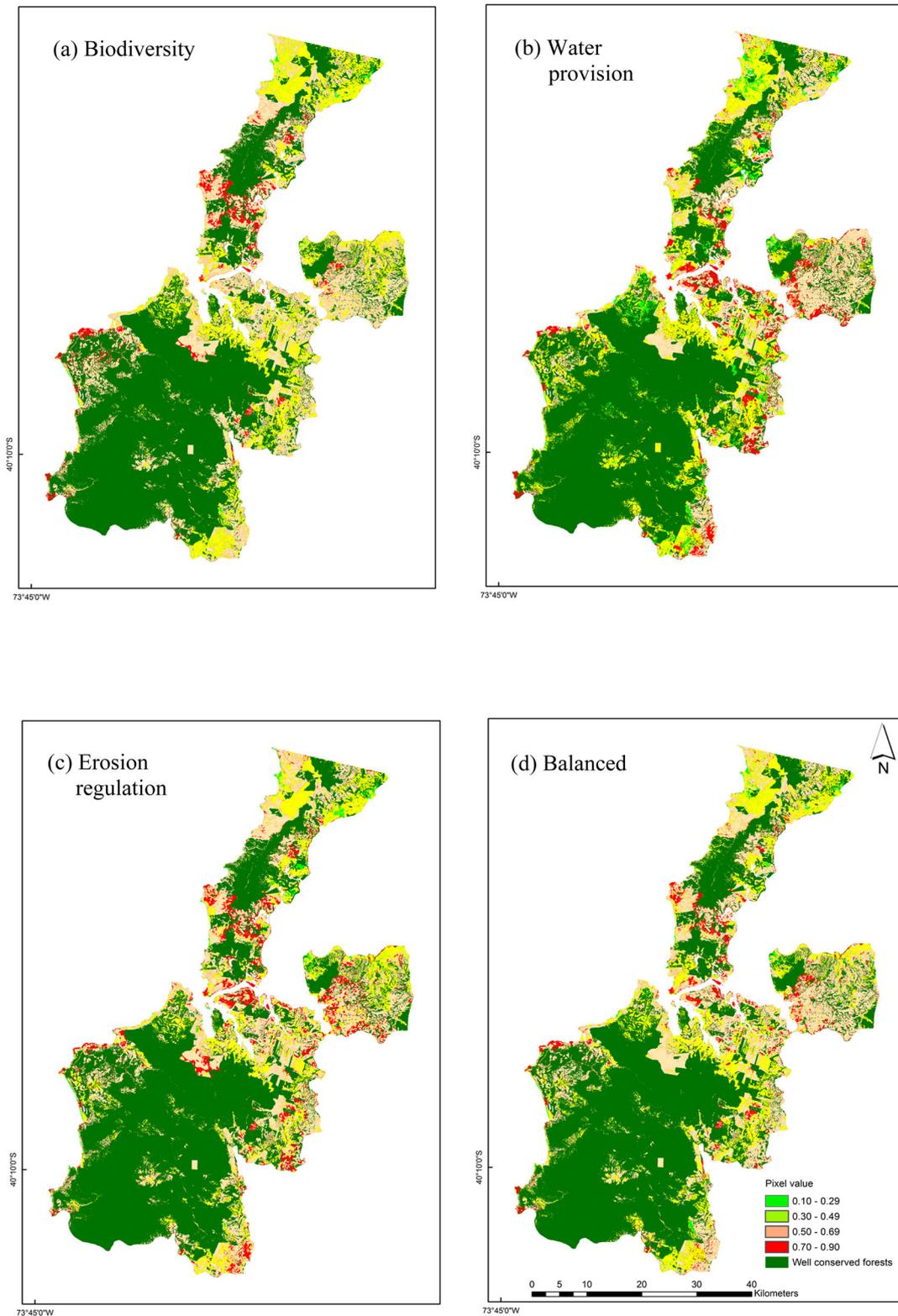
Fig. 3 Suitability maps and cell value ranges for the **a** biodiversity, **b** water provision, **c** erosion regulation, and **d** balanced scenarios, where 0 corresponds to minimum suitability and 1 to maximum suitability for forest restoration

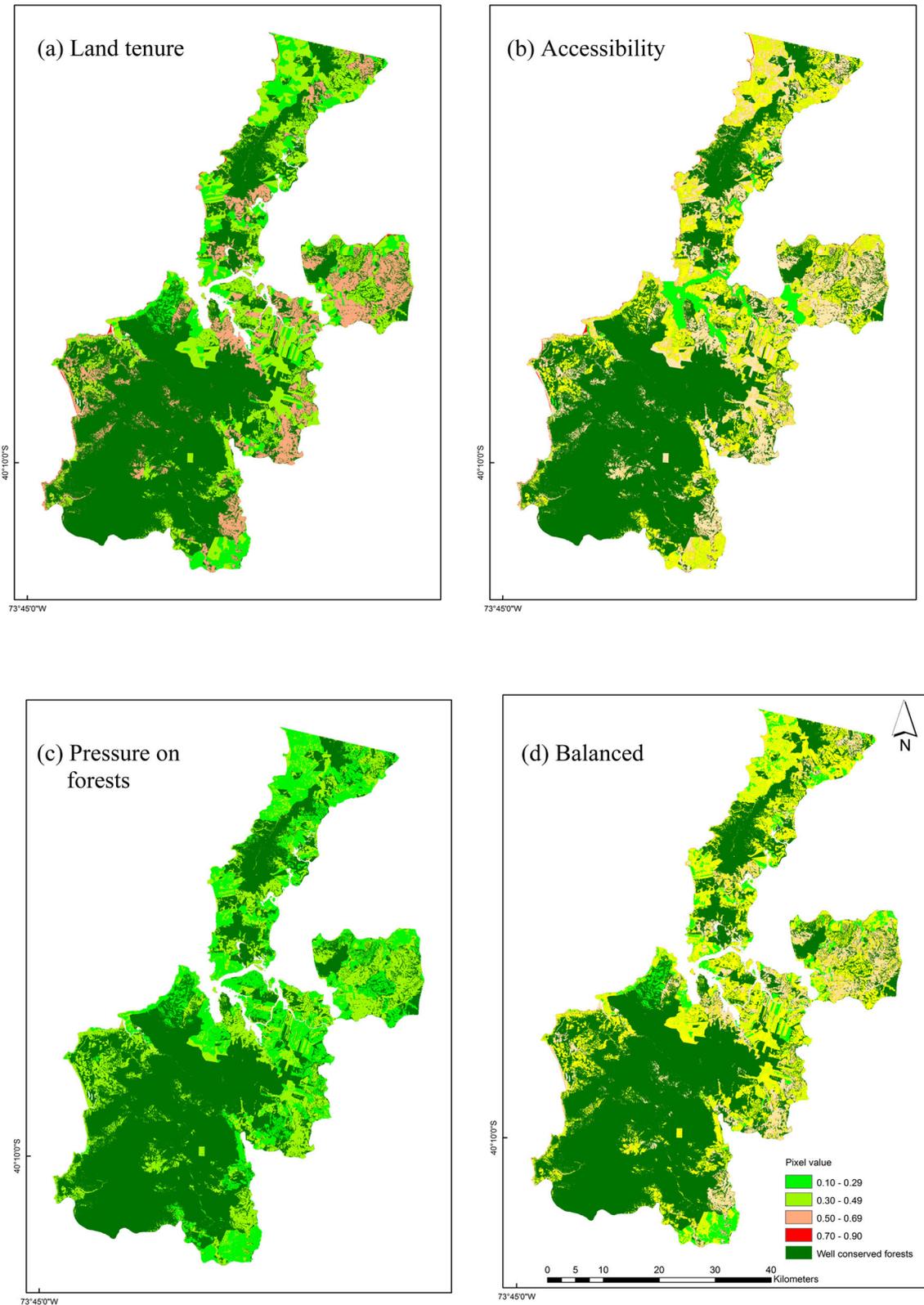**Fig. 4** Feasibility maps and cell value ranges for the **a** land tenure, **b** accessibility, **c** pressure on forest, and **d** balanced scenarios, where 0 corresponds to minimum feasibility and 1 to maximum feasibility for forest restoration
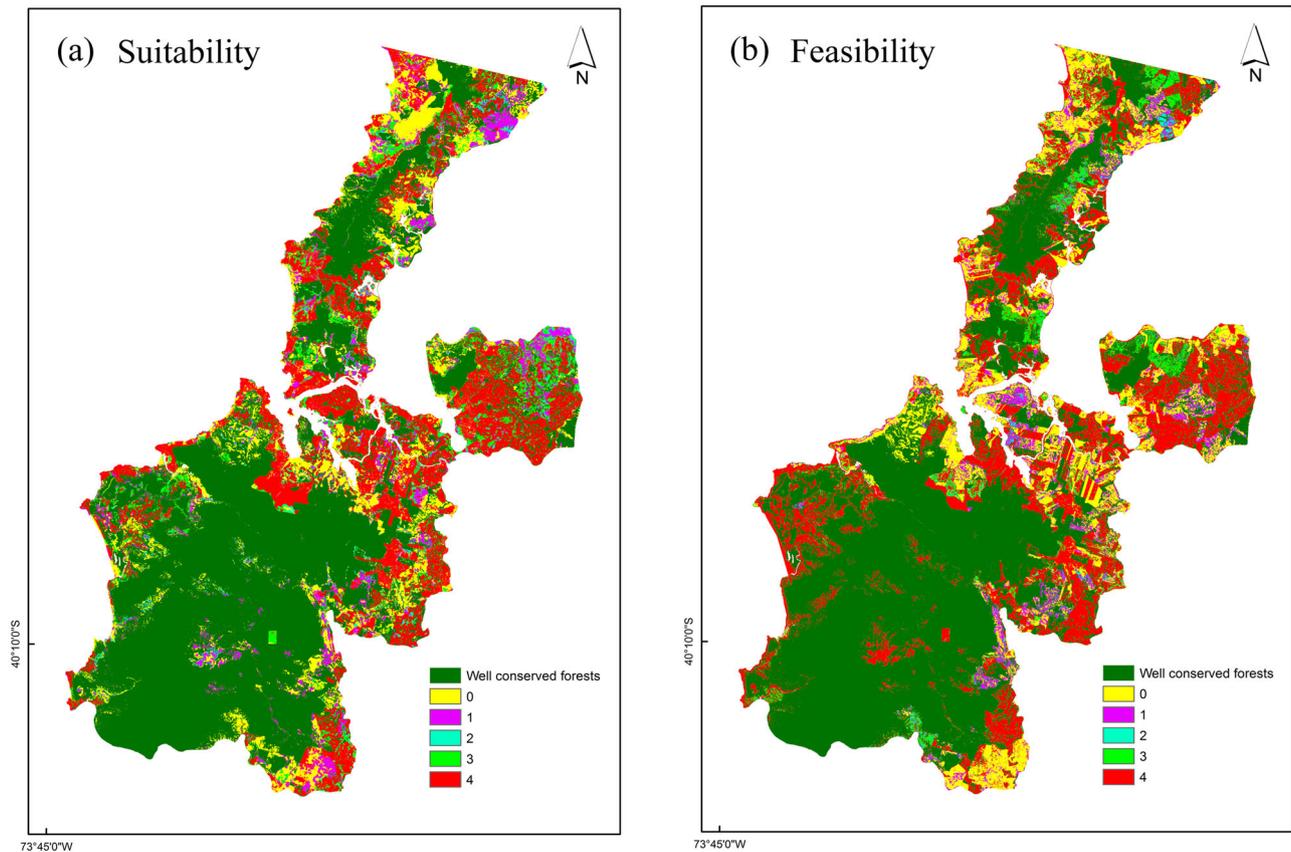
Fig. 5 Maps of **a** suitability and **b** feasibility scores for forest restoration after summing up the individual suitability (Fig. 3) and feasibility rasters (Fig. 4) generated for each of the four scenarios (0 indicates minimum suitability/feasibility and 4 maximum suitability/feasibility)

Discussion

In this study, we proposed a methodology to identify priority areas for forest restoration at the landscape scale, which incorporates both the potential value of restored forests in terms of biodiversity and the provision of eco-system services, and the socioeconomic feasibility of restoration interventions. Our methodology was able to identify priority restoration areas in potentially degraded forests and in partially or completely deforested areas. This is a significant asset given that restoration is typically more successful and cost-effective in degraded rather than com-pletely deforested areas (Lewis et al. 2019).

Restoration, Not Just Reforestation: Overcoming Difficulties in Identifying Forest Degradation at the Landscape Scale

Our study included both degraded forests, which were priority target areas for restoration, as well as partially or completely deforested areas. Given the complexity involved in using remote-sensing methods and need for extensive field work to detect forest degradation, which

are often prohibitively expensive or highly time con-suming (Cayuela et al. 2006b), it is unsurprising that previous studies have often limited their analyses to deforested areas only (e.g. Orsi and Geneletti 2010). Here, we use an alternative approach, integrating previously identified socioeconomic correlates of forest degradation occurrence and intensity to identify degraded areas. Although this approach was based on previous research within the studied landscape, similar correlates are likely to be available in many landscapes given that most forest degradation is the direct or indirect result of human dis-turbance (Vásquez-Grandón et al. 2018). As shown here, these correlates can be a useful proxy when sufficiently detailed spatial datasets are not accessible or suitable. Our identification of degraded forest was consistent with pre-vious work on forest degradation in the study area (Reyes 2004; Nuñez et al. 2012; Vergara-Díaz et al. 2017). According to these authors, forest exposed to fragmenta-tion, logging and cattle activity are concentrated in the north of the region, with impacts that are threatening several flora and fauna species (Nuñez et al. 2012); whereas southern parts of the region are characterised by well-conserved and continuous forest.

Fig. 6 Map of priority areas for forest restoration that results from combining the highest cell values of the suitability (Fig. 5a) and feasibility (Fig. 5b) maps. The identified areas correspond to those cells that were priorities for all suitability and feasibility scenarios (i.e. cell value equal to 4)
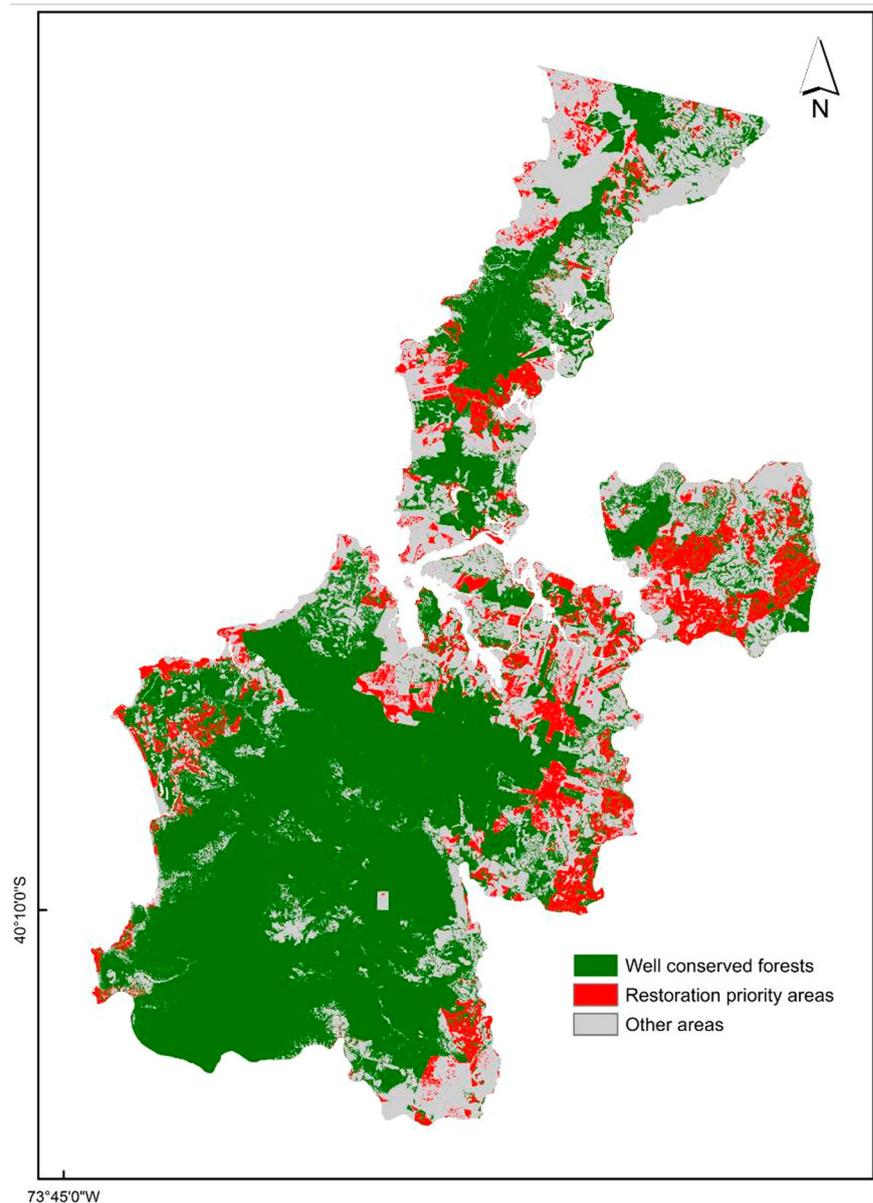
By including degraded areas in our analyses, we were able to shift the focus of our prioritisation from reforestation to regeneration. This differs from existing restoration prioritisation studies, such as the method proposed by Orsi and Geneletti (2010), which is to some extent similar to that presented here. However, their work differs from this study in that they (a) defined priority areas for reforestation exclusively focused on potential for biodiversity conservation, (b) considered aspect and elevation heterogeneity as proxy variables to estimate tree species richness, and (c) addressed the potential for reforestation rather than forest restoration. Here, the term "reforestation" refers to tree establishment from planted seedlings or seeding, including land that has a history of forest cover but has been deforested for some extended time period (Murray

2000; Nolan et al. 2018). On the other hand, forest restoration instead implies an intentional activity that initiates or accelerates the recovery of an ecosystem with respect to its ecological integrity and sustainability. Unlike reforestation, restoration represents an indefinitely long-term commitment of land and resources, and a proposal to restore an ecosystem requires thoughtful deliberation (SER 2004; Gann et al. 2019).

Incorporating Ecosystem Services and Socioeconomic Feasibility to Increase Restoration Success

Landscape-scale studies aiming to develop methods for prioritizing forest restoration that integrate ecological

suitability with socioeconomic feasibility to maintain and enhance biodiversity and ecosystem services are scarce. Such integration is critical as it enables the efficient allocation of restoration resources in the short term to areas with high potential value, while also identifying socioeconomic scenarios that may enhance long-term restoration success (Reyes-García et al. 2018). This approach can inform policy guidance and promote initiatives to reverse or mitigate forest degradation. However, further investigation is needed to empirically define degraded forests spatially and to comprehensively understand the various factors influencing forest function, in order to develop and promote effective measures for their restoration and conservation.

A significant number of studies have revealed the role of biodiversity in supplying ecosystem services (Rey Benayas et al. 2009; Meli et al. 2017; Moreno-Mateos et al. 2017). Biodiversity has positive effects on ecosystem services such as erosion regulation and nutrient cycling services (e.g. Balvanera et al. 2006). Our methodology involved using stacked species distribution models, derived from an extensive field dataset, to estimate potential flora richness as a direct proxy for potential biodiversity. In addition, we considered the distance to well-conserved forests and protected areas to capture the potential for natural regeneration of restoration areas, which will in turn influence their potential to regain ecosystem function and enhance ecosystem services. Both approaches are particularly relevant in the context of the Kunming-Montreal Global Biodiversity Framework (GBF), which emphasizes the need to protect 30% of the planet by 2030, with a specific focus on biodiversity (https://www.cbd.int/gbf).

Forest regeneration is the process that ensures successive generations of trees (Barnes et al. 1998), and is essential in maintaining the long-term ecological functions and values of forests (Donoso and Nyland 2005). Natural regeneration is the cheapest and technically easiest method for forest restoration (Lewis et al. 2019; Chazdon et al. 2020). It mitigates or reverses early degradation processes, and thus increases the efficiency of restoration activities. Natural forest regeneration ensures successive generations of trees (Barnes et al. 1998) and is essential in maintaining the long-term ecological function and values of forests (Donoso and Nyland 2005). While promoting natural regeneration can be a viable strategy to achieve ambitious restoration commitments, however, predicting where native forest cover is likely to expand is challenging (Borda-Niño et al. 2020). In the tropical and subtropical areas of the world, Borda-Niño et al. (2020) showed that forest cover increased more often on steeper slopes, close to forest remnants, inside protected areas, and far from human settlements. However, the effects of most drivers varied among scales of evaluation and may be further affected by the scale of forest regrowth. In Chilean temperate forests, Zamorano-Elgueta et al. (2015)

found a relatively high rate of natural conversion from shrubland to native forest, suggesting a high potential for natural regeneration. It is important to note that native forests established by natural regeneration differed in diversity, structure, and functionality from old-growth forests, and responded differently to human impacts (Lu et al. 2003; Zamorano-Elgueta et al. 2014; 2015).

In addition to the inherent regeneration and biodiversity potential of degraded areas, land tenure and socioeconomic context are likely to strongly influence the feasibility and long-term success of restoration efforts. In Chilean forests, high-intensity human disturbances are associated with the dominant subsistence farming system. Firewood from native forests represents the main household income in small rural properties in the study area (Reyes et al. 2015; Manuschevich 2020). Under certain conditions of isolation and poverty (few off-farm incomes opportunities, low opportunity cost of labour, and large families), firewood production leads to chronic forest degradation (Reyes et al. 2015). Cattle grazing also increases forest degradation by reducing the regeneration of some tree species (Zamorano-Elgueta et al. 2012; 2014), and these impacts tend to be exacerbated on small rural properties (Reyes 2004). In addition, the longevity of restored ecosystems has a temporal dimension that is variable and often related to ownership. Thus, landowners that received most of their income from farming have land with shorter periods for forest recovery (Reid et al. 2017).

As a result, priority restoration areas identified in this study occur on larger properties. In the study area, large properties are owned mainly by forest companies or are natural protected areas. Other owners of large properties live in urban areas outside the region, and their income does not come from rural property. Large properties owned by forest companies are mostly used for industrial exotic tree plantations, which were heavily promoted in the last 50 years by the Chilean government (Manuschevich 2020). In fact, Chile is a leading country in forest exports, in large part because plantations of *Pinus radiata* (Monterey pine) can provide one of the highest returns on investments worldwide (Cubbage et al. 2007). The exotic tree plantations severely affect water provision during the dry summer season (Lara et al. 2009; Álvarez-Garretón et al. 2019). In Chile, their management, mainly through extensive clear cutting, leaves the soil unprotected for several years until the new plantation regrows. It is essential that sustainable forest management ensures the retention of significant canopy cover to prevent soil erosion and foster nutrient retention and cycling (Pérez 1999; Banfield et al. 2018). This is particularly critical as restoration priorities are concentrated in land owned by forest companies. Moreover, outside protected areas, exotic tree plantations surround most native forest remnants.

Applications

Our approach relies on the combination of multiple criteria using value functions, and the definition of appropriate weighting schemes that reflect the relative importance of the different criteria. This approach may imply uncertainties, as modifying a value function may significantly influence the outcome. However, it is important to note that the MCA approach was explicitly developed to deal with conflicting objectives. As an example, the distance from urban areas and distance from roads can have both positive and negative influences on restoration priority: areas closer to roads are easier to access and therefore to restore, but at the same time they are more exposed to disturbances. Hence, the result that we obtain is a compromise between all criteria, and clearly shows their trade-offs. While value functions are inherently subjective, we designed a model framework that can be easily and transparently modified by adjusting value functions and weights to find solutions that are more in line with various stakeholders. Our approach is conservation-oriented as all the value judgements prioritise ecological over economic benefits (e.g. firewood). Yet, the value judgements can be modified by actors according to their priorities and obtain results that are more balanced (in terms of ecological vs economic effects) to provide more immediate benefits to local communities. Additionally, depending on their regional context and the availability of reliable data, stakeholders could integrate additional information into the proposed method to further refine the identification of priority areas for restoration. Examples of additional information include land use regulations or zoning, including the locations of approved large-scale infrastructure developments or exploration/extraction licenses for mining. They may also include data related to alternative or additional ecosystem services and biodiversity values to those used here, such as cultural services or the promotion of ecological corridors, even though the study area is characterized by continuous forests and low fragmentation. Thus, our proposal could be enhanced with criteria related to landscape metrics that address forest fragmentation, such as reducing the distance between fragments (e.g., nearest-neighbour distance) and prioritize areas that are closest to each other.

When this approach comes to practical application, value judgements may be informed by the opinion of stakeholders, experts and local communities to identify restoration plans that are ecologically suitable, socially acceptable, and economically feasible in the region of interest. In this respect, the method provides a robust framework. Future studies should emphasise participatory processes at all levels of the modelling steps and the integration of other ecological, social, or economic considerations. Moreover, decision-makers and practitioners have considerable leverage to increase the probability that restored ecosystems persist into the future. Locally, practitioners can engage communities to build stakeholder support and facilitate training to improve technical capacity (Reid et al. 2017). Our proposal can also complement other prioritization tools not intended for detailed, district-level planning, such as the ROAM (Restoration Opportunities Assessment Methodology, IUCN/WRI 2014), a framework used to identify and assess potential areas for ecological restoration. The ROAM provides a structured approach for decision-makers to evaluate restoration opportunities, ensuring that the most impactful and sustainable projects are selected. This methodology is commonly used in conservation and land management to guide restoration efforts and optimize the allocation of resources.

Future studies should also consider a multi-scale approach to defining socioeconomic and ecological variables. The response of several variables is not limited to a single spatial scale and are often associated with multiscale processes and phenomena (Bhakti et al. 2018). As an example, at the property scale lower household incomes in our study region are positively associated with forest degradation, due to the associated need to exploit forest resources for subsistence and livelihoods. At a regional scale, however, low-income regions are typified by less investment in major infrastructure and development, and therefore support more remnant vegetation and biodiversity than richer and more intensively developed regions. We therefore recommend consideration of how variables of interest vary at different scales, and/or careful definition of the study area and raster cell resolution at an appropriate scale for which the effects of each variable are well-understood.

Prioritizing Forest Restoration in a Context of Rapid Changes

Wisely-planned forest restoration is one of the most effective strategies for biodiversity conservation and climate change mitigation (Lewis et al. 2019). While the case study presented here is context specific, the approach and methodology used can be readily adapted to various scenarios and objectives. This would require careful consideration of the biophysical characteristics of the landscape, availability of suitable landscape-scale data, and the priorities of relevant stakeholder groups. This approach allows for clear and transparent prioritisation of available funding, with an explicit mechanism for incorporating stakeholder consultation and expert advice to balance the ecological value of potential restoration sites with socioeconomic feasibility. Ecological restoration serves as a strategy for mitigating and adapting to climate change, yet its outcomes are also vulnerable to the very impacts it seeks to address (Simonson

et al. 2021). As predicted changes in climate will affect forest distribution, it is crucial to consider these impacts into the prioritization of restoration areas in future proposals. Simonson et al. (2021) identified seven key aspects of restoration design and implementation where climate change must be considered: defining restoration goals, selecting sites and ensuring connectivity, choosing appropriate target species and ecosystems, managing essential ecosystem interactions and micro-climates, addressing site-specific climate change risks, aligning restoration efforts with long-term policies, and developing a monitoring framework to support adaptive management. However, forest restoration projects are at risk from climate change impacts, including rising temperatures, droughts, insect infestations, and increased wildfire frequency. These threats undermine the long-term success of restoration initiatives, which aim to sequester carbon in biomass and soils over project lifespans of 50–100 years or more (Koch and Kaplan 2022). In this context of rapid changes, it will be essential to develop prioritization systems to restore forests that integrate the future potential distribution of forests and their role for climate change mitigating and adaptation.

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Compliance with Ethical Standards

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