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Protecting wilderness or rewilding? An ecoregion-based approach to identifying priority areas for the protection and restoration of natural processes for biodiversity conservation

Brenda Maria Zoderer [a,*], Thomas Marsoner [b], Erich Tasser [b]

[a] Institute of Landscape Development, Recreation and Conservation Planning, University of Natural Resources and Life Sciences, Vienna, Peter-Jordan Straße 65, 1180 Vienna, Austria

[b] Institute of Alpine Environment, Eurac Research, Drususallee 1, Bozen/Bolzano, Italy

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ABSTRACT

Amidst the global decline in biodiversity, there are growing calls for more ambitious conservation targets and practices, including a renewed focus on protecting and restoring natural processes. However, little is known about suitable areas for process-oriented conservation and its different strategies. In this paper, we identify priority areas for process-oriented conservation following an ecoregion-based approach. Using the Alpine Space programme area as a pilot study area, a Wilderness Quality Index is calculated and mapped based on spatial indicators reflecting variations in naturalness, human impact, remoteness, and ruggedness. To identify priority areas for process-oriented conservation, the 10% of areas with the highest wilderness quality are identified for each ecoregion ('ecoregional approach') and compared with the identification of the 10% wildest areas of the entire study area ('conventional approach'). The results show significant differences in priority areas between the two approaches, with those identified by the ecoregional approach being of lower wilderness quality, more dispersed across the study region and different elevation classes, and smaller in size. The ecoregional approach results in a greater coverage of ecosystem- and species-level diversity, yet it highlights a greater need for complementing the protection of wilderness in less modified regions with rewilding initiatives and the expansion of the protected area network in ecoregions with significant human activity. Based on these findings, we discuss the potential and challenges that an ecoregion-based identification of priority areas brings for biodiversity conservation, protection and restoration practice, and local communities. The ecoregion-based approach and the findings of this study can inform initiatives under the EU Biodiversity Strategy to 2030, in particular the target to 'strictly protect' 10% of the EU's land and sea.

1. Introduction

Global biodiversity loss represents one of the greatest challenges humanities is facing (Pörtner et al., 2021). Ongoing changes in land use, overexploitation, climate change, and other human activities are now threatening more species with global extinction than at any point in history (IPBES, 2019). There is mounting evidence that a quarter of species in each animal and plant group are currently at risk of extinction, with extinction rates up to 100 times higher than it has been averaged over the past 10 million years (ibid.). According to the Living Planet Index, nearly 70% of the world's wildlife populations declined between 1970 and 2018 (WWF, 2022). A similar pattern can be observed in Europe, where two-thirds of species protected under the EU Habitats Directive are in poor or bad conservation status (EEA, 2020a). The rapid loss of biodiversity and the realisation that past efforts to safeguard biodiversity are not sufficient have sparked debates on the need to (re) orient conservation practices towards the state of ecosystems and their processes, rather than solely focusing on the protection of individual species (Lorimer et al., 2015; Van Meerbeek et al., 2019). This renewed interest is also reflected in the EU Biodiversity Strategy 2030, which aims to protect at least 30% of the land and the sea in the EU while also ensuring that natural processes are protected and restored on at least 10% of the territory (i.e., the '10% target', EC, 2022).

Process-oriented approaches to biodiversity conservation focus on the maintenance and restoration of functionally intact and self-regulating ecosystems (Carver et al., 2021; Corlett, 2016; Fernández

* Corresponding author.
E-mail address: brenda.zoderer@boku.ac.at (B.M. Zoderer).

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et al., 2017; Perino et al., 2019). Such ecosystems can encompass both (near)-natural and restored (novel) ecosystems and are primarily governed by natural processes that determine ecosystem structure and function (Pettorelli et al., 2018; Van Meerbeek et al., 2019). Unlike traditional conservation approaches, which are focused on maintaining predefined species or ecosystem conditions, process-oriented strategies aim to maintain or restore natural processes by improving connectivity, introducing keystone species, and maintaining, or creating core wilderness areas, while reducing human control and pressure (Carver et al., 2021; Perino et al., 2019; Svenning et al., 2016). Most importantly, the low-intervention and open-ended nature of this strategy is seen as a more forward-looking approach to biodiversity conservation in times of rapid climate and socio-economic change, as it builds on the adaptive capacity and resilience of ecosystems rather than continuous human management interventions (Carroll & Noss, 2021; Svenning, 2020).

A strict process-oriented conservation strategy has traditionally been applied in protected areas with IUCN categories Ia (Nature Reserve), Ib (Wilderness Area), and the core zones of category II (National Parks). In contrast to other protected areas, such as IUCN category IV (habitat/species management areas), where the sustainable management of land is combined with the conservation of rare species and habitats, extractive activities like mining, agriculture, or forestry are excluded under the former protection categories. As a result, strictly protected areas are typically situated in remote, steep, and high-elevation locations with lower productivity potential and where threats from human activities are lowest (Cazzolla Gatti et al., 2023; Joppa & Pfaff, 2009; Venter et al., 2018). This location bias led to criticism about conservation efforts that tend to be more ambitious where they face less resistance and not where they are needed the most (Joppa & Pfaff, 2009; Pimm et al., 2018). Considering the important role of strictly protected areas in ensuring long-term biodiversity (Betts et al., 2017; Di Marco et al., 2019) and providing ecosystem services (Martin & Watson, 2016; Pimm et al., 2018; Watson et al., 2018), there have been calls to complement the strict protection of remote and less modified areas with the implementation of process-oriented approaches also in human-dominated regions (Bergin et al., 2024; Kennedy et al., 2019).

In intensively used regions, intact ecosystems are often confined to small, fragmented patches embedded in a landscape matrix that has been significantly altered by humans (Caro et al., 2012; Lindenmayer, 2019; Wintle et al., 2019). From a biodiversity conservation perspective, however, these areas are of particular importance as they often represent the remaining intact habitats for species that have lost their habitat elsewhere and are particularly vulnerable to threats from the surrounding landscape (Kennedy et al., 2019; Mokany et al., 2020). They can also exhibit remaining core areas from which species recolonisation and restoration of natural processes in surrounding areas can take place more easily (Baumann et al., 2020; Belote et al., 2021). In Europe, the protection of wilderness and remaining smaller, intact ecosystems may not be sufficient to overcome the biodiversity crisis (Dinerstein et al., 2017; Perino et al., 2021; Strassburg et al., 2020). While large areas of wilderness exist in other regions of the world such as Canada and Russia (Watson et al., 2016), few wilderness areas and smaller intact areas are left in Europe (Fisher et al., 2010; Sabatini et al., 2020; Strus & Carver, 2024). A recent study (Cazzolla Gatti et al., 2023) demonstrated that only 3.37% of the territory of the European Union currently hosts strictly protected areas of IUCN categories Ia/b or II and that many countries and biogeographical regions lack sufficient areas of low human impact to achieve the 10% strict protection target of the EU Biodiversity Strategy 2030 (EC, 2022). Thus, for the long-term conservation of biodiversity it will be necessary to additionally restore functionally intact, self-regulating ecosystems in currently modified landscapes by considering process-oriented restoration approaches such as rewilding.

Over the past decade, rewilding has emerged as a novel restoration strategy that aims to restore natural processes in landscapes where they have previously been lost or degraded (Carver et al., 2021; Perino et al., 2019). As rewilding gains traction, a range of different approaches have been developed and implemented in practice. Such approaches include the reintroduction of grazing regimes by large-bodied herbivores (i.e., trophic rewilding, Svenning et al., 2016), the restoration of top-down ecological regulation through the (re)introduction of important carnivores (Wolf & Ripple, 2018), and the promotion of natural ecological succession and recolonisation processes following the withdrawal or the reduction of agricultural management actions (i.e., passive rewilding, Carver, 2019; Navarro and Pereira, 2012). Despite the diversity of approaches and their implementation across various geographical contexts and spatial scales (Fernández et al., 2017; Schulte to Bühne, Pettorelli, et al., 2022), rewilding initiatives share the common goal of increasing the wildness and thus the ecological integrity and resilience of landscapes (Carver et al., 2021; Perino et al., 2019). Particularly in the European context, where most landscapes are cultural landscapes resulting from close interactions between humans and nature (Tieskens et al., 2017), it has been emphasised that rewilding should not only focus on the creation of wilderness areas but also promote the relative increase in wildness in landscapes inhabited and used by people to maintain and enhance biodiversity (Loth & Newton, 2018; Massenberg et al., 2023; Perino et al., 2019; Ward, 2019).

To contribute to a better understanding of the potential and challenges of these process-oriented strategies for biodiversity conservation, this article examines the spatial conditions and priority sites for such conservation strategies in the European Alps using an ecoregion-based approach. Ecoregions represent areas with relatively homogeneous ecological conditions in terms of climate, topography, and geobotanical features, and therefore host distinct assemblages of ecosystems, communities, and species (Bailey, 2014; Olson et al., 2001). Distributing conservation efforts across different ecoregions and protecting a sufficient fraction of their geographic extent has therefore been proposed as an effective strategy to achieve more holistic protection of different levels of biodiversity, including ecosystem and species diversity (Dinerstein et al., 2017; Smith et al., 2018). Using the Alpine Space Programme area as a pilot area, we spatially prioritise potential areas for process-oriented conservation strategies from an ecoregional perspective and compare this with a 'conventional approach' that prioritises areas that are most suitable within the administrative boundaries of the total study area. The ecoregional approach is chosen as a primary lens in this study to gain a better understanding of where efforts to implement process-oriented conservation should be prioritised to benefit biodiversity, and when wilderness protection or rewilding constitute the most appropriate strategies to realise process-oriented conservation. Following a detailed characterisation and comparison of priority areas from both approaches, we discuss the opportunities and challenges that an ecoregion-based identification of priority areas brings for biodiversity conservation, protection and restoration practice, and local communities. The insights gained in this study can inform current discussions and efforts to implement the EU Biodiversity Strategy up to 2030, in particular with regard to the '10% target'.

2. Methodology

A three-step approach was applied to identify priority areas for process-oriented conservation strategies (Fig. 1). First, the wilderness continuum concept, initially proposed by Nash (1993), was used to identify areas most suitable for process-oriented conservation strategies. The wilderness continuum describes the relative wildness of the landscape for a given geographical area and can thus provide an indication of the self-regulating capacity of the ecosystems it contains (Carver et al., 2021). Previous studies have used this concept to identify priority areas for wilderness protection (e.g., Cao et al., 2019; Kuiters et al., 2013; Plutzar et al., 2016; Radford et al., 2019) and areas suitable for rewilding (e.g., Ceaușu et al., 2015). Such priority areas were typically identified for pre-defined administrative units (i.e., conventional approach), ranging from federal regions (e.g., Zoderer et al., 2020), nations (e.g., Cao et al., 2019; Plutzar et al., 2016; Radford et al., 2019)

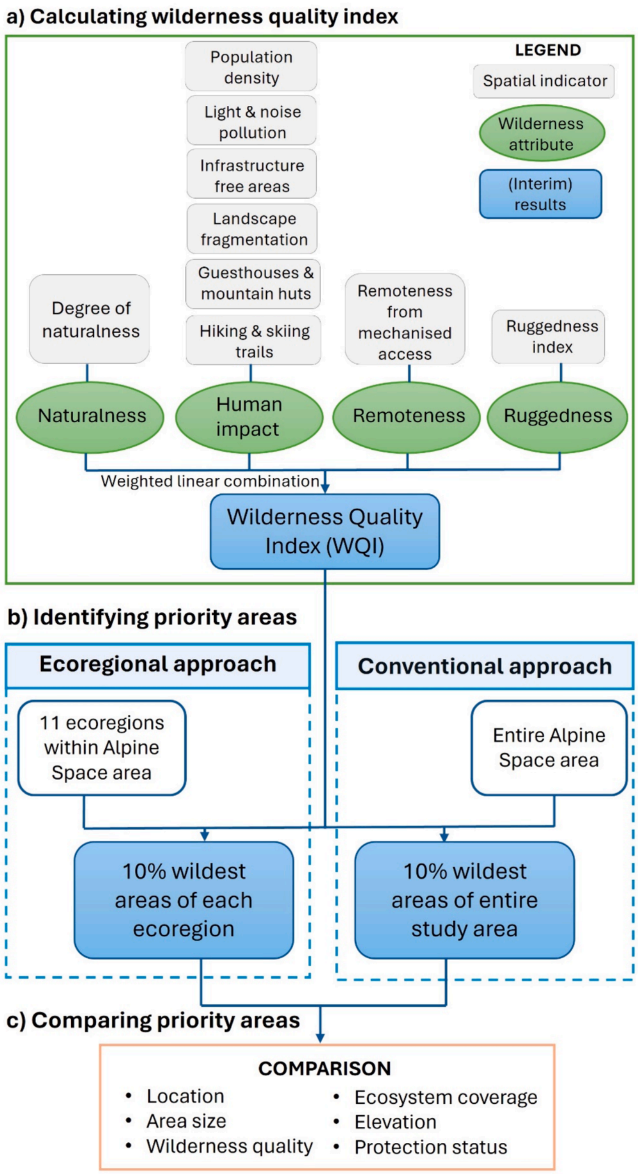

Fig. 1. Methodological overview of the three main steps undertaken in this study, including information on input data and (interim) results.

to supranational units like the European Union (e.g., Kuiters et al., 2013; Strus & Carver, 2024). While this approach can provide valuable insights into the least and most wild areas within the given administrative boundaries, it may fail to recognise the importance of considering different biogeographic realms when prioritising process-oriented strategies for biodiversity conservation. To overcome these limitations, we used information on the wildness of the landscape to identify areas with the highest self-regulating capacity within each ecoregion of the study area (i.e., ecoregional approach) and compared them with the selection of areas that have the highest self-regulating capacity within the administrative boundaries of the entire study region (i.e., conventional approach). Following the 'strict protection target' of the EU Biodiversity Strategy to 2030 (EC, 2022), we identified the 10% of areas considered most suitable for process-oriented conservation for each approach. Finally, the priority areas identified in each approach were compared in terms of their location, area size, wilderness quality, ecosystem coverage, distribution across elevation classes, and protection status.

2.1. Study site

We used the cooperation area of the Alpine Space Programme as a pilot area (Fig. 2). It encompasses Switzerland, Liechtenstein, Austria, Slovenia, as well as parts of France, Italy and Germany, and covers an area of approximately 390,000 km^2. With Mont Blanc as the highest elevation (4,807 m a.s.l.) and 0 m a.s.l. near the sea, the study area is topographically and climatically diverse. The high variability of the climate is due to the location of the Alps between the temperate Central European and the Mediterranean climate zones and the continental gradient running from west to east. The lowest annual precipitation (<500 mm) is found in the dry valleys of the Central Alps, while at higher elevations the precipitation can reach 3,500 mm. Almost half of the Alpine Space area is covered by forests (49%), 27% by agricultural land, 19% by high mountain landscapes with shrubs, natural grasslands, rocks and glaciers, 3.7% by artificial surfaces, and 1.1% by water bodies. Due to the high variability in topography, climate and land cover, 11 different ecoregions can be found in the study area (Fig. 2), each representing a relatively homogeneous geographical unit with distinct environmental conditions and plant species composition (Olson et al., 2001). Overall, the area is considered to be a region of high biodiversity, with the high mountains being a hotspot (Körner, 2003; Zimmermann et al., 2013), and has been identified as one of the remaining regions with high potential for wilderness at the European scale (Kuiters et al., 2013).

2.2. Wilderness quality index

We operationalised the wilderness continuum concept by calculating a Wilderness Quality Index (WQI) for the entire Alpine Space area. The WQI is typically derived by using fuzzy GIS-based approaches, considering different spatial input data and their combination within a multi-criteria evaluation model such as Weighted Linear Combination (WLC) (e.g., Cao et al., 2019; Carver et al., 2012; Müller et al., 2015; Radford et al., 2019, 2019). In this study, we calculated the WQI largely following the fuzzy GIS-based approach proposed by Radford et al. (2019) and its modifications by Zoderer et al. (2020).

2.2.1. Data preparation

In order to maintain comparability between the individual countries and regions within the study area, data sets available at larger, sub-continental scales and collected using the same standardised methods were considered (Table 1). To map land use and land cover (LULC), we created a multi-source LULC map (i.e., LULC$_{ref}$) based on the integration of a number of different land use and LULC information. To this end, data from the Corine land-cover assessment 2018 with a spatial resolution of 100x100 m (EEA, 2020b) was complemented with high-resolution data from the European forest layer (EEA, 2017a), the Pan-European forest/non-forest map (Pekkarinen et al., 2009), and information on single houses and small villages extracted from the European soil sealing map (EEA, 2017b). In addition, all streams with an order > 4 were integrated from the European River Network (EU, 2020), as well as all highways, major roads, railways, and cycle lanes from the OpenStreetMap (OSM 2020, downloaded from https://www.openstreetmap.org/). Due to different spatial resolutions and accuracies, we combined them in a strictly hierarchical way: 1) Corine land cover 2018; 2) European forest layer, 3) Pan-European forest/non-forest map, 4) European soil sealing map, 5) river network, and 6) road, rail and cycle network. Overall, the calculation of the WQI and the subsequent mapping of priority areas for process-oriented conservation was carried out at a spatial resolution of 25x25 m. To avoid edge effects along the border of the study region, data outside of the Alpine Space area were also included within a 50 km buffer.

2.2.2. Mapping wilderness attributes

Calculations of the WQI were based on the quantification and

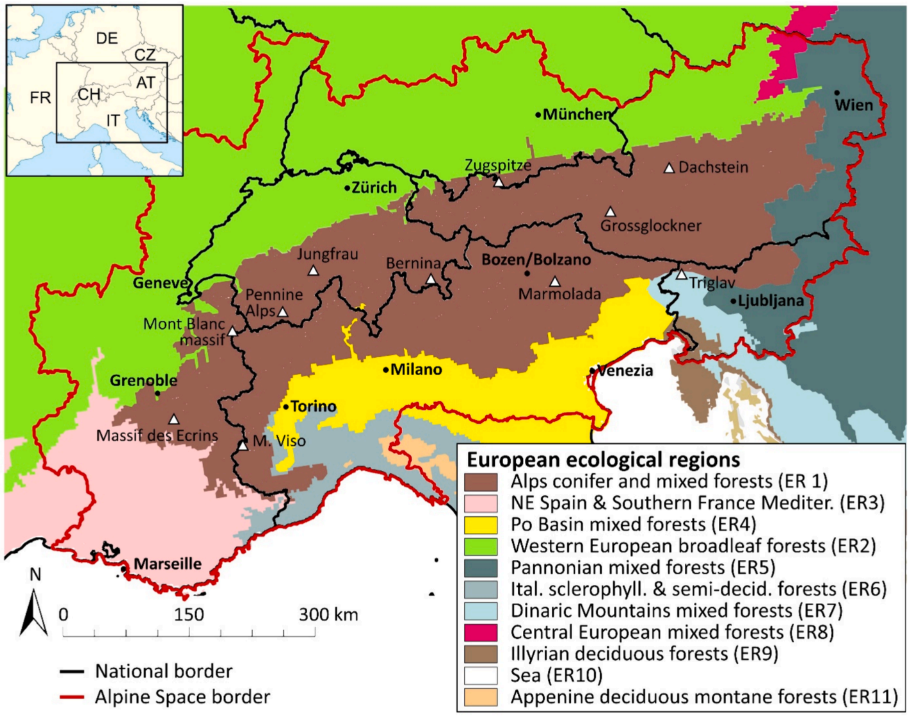

Fig. 2. The Alpine Space area with the European Ecological Regions (after EEA, 2015).

mapping of four wilderness attributes: 1) naturalness, 2) human impact, 3) remoteness, and 4) ruggedness (Carver et al., 2012; Radford et al., 2019; Zoderer et al., 2020). For each wilderness attribute, at least one spatially-explicit indicator was calculated and then combined by Weighted Linear Combination (WLC) into a wilderness quality layer.

(1) *Naturalness* is a commonly considered wilderness attribute for the operationalisation of the wilderness continuum concept (Carver et al., 2012; Kuiters et al., 2013; Lesslie, 2016; Müller et al., 2015; Plutzar et al., 2016; Radford et al., 2019; Zoderer et al., 2020). It reflects the extent to which an ecosystem has deviated from its original state due to human alterations associated with urbanisation, agriculture, and forestry use. To facilitate the mapping of naturalness at fairly large spatial scales, we used the *degree of naturalness* as a spatially-explicit indicator (Rüdisser et al., 2012). It is based on the hemeroby concept (cf. Ferrari et al., 2008), which mainly uses vegetation aspects to assess human influence, and extends it by additionally considering biodiversity-relevant anthropogenic interventions on plants, animals and ecosystems as a whole – in particular also considering soil disturbances (cf. Rüdisser et al., 2012). Given the status quo-oriented nature of the indicator, we determined the degree of naturalness of each pixel cell on the basis of our high-resolution $LULC_{ref}$. Using the seven staged naturalness scale proposed by Rüdisser et al. (2012), we assigned a naturalness value to each LULC type (SI, Table A1), ranging from 1 (natural systems with no or only minimal anthropogenic influence) to 7 (artificial systems or structures, soil sealing over 30%). To distinguish between semi-natural and intensively used grasslands in terms of their naturalness score, we assigned higher naturalness values to grasslands if they were located above the potential treeline of the European Alps (Pecher et al., 2011, SI, Table A1). Such grasslands are typically managed to a limited extent. Similarly, standing and running water were assigned higher naturalness scores when located above 1600 m a.s.l. (Zoderer et al., 2020).

(2) *Human impact*: Following the approach by Radford et al. (2019) and the modifications proposed by Zoderer et al. (2020), we considered nine spatial indicators to comprehensively quantify and map the impact of human activities on ecosystem structures and processes: *population density, light and noise pollution*, *infrastructure-free areas*, *landscape fragmentation, guesthouses, mountain huts*, and *hiking and skiing trails*. *Population centres* were mapped using the Global Human Settlement data at a resolution of 250x250 m (JRC, 2020). After resampling the data to 25x25 m, we classified them using the restrictive thresholds proposed by Radford et al. (2019): 1) 0 inhabitants, 2) 1–10 inhabitants, and 3) > 10 inhabitants per ha. While population density provides a first rough proxy for human disturbance in natural landscapes (Cao et al., 2019; Kuiters et al., 2013; Müller et al., 2015), disturbance associated with human settlements needs to be further detailed by considering light or noise pollution. *Light pollution* was considered by calculating the mean values of light pollution at night for the period 2011–2013 and 1992–1994 based on data from NOAA (see Table 1). Values were classified as 1) dark (brightness = 0) in both time periods, 2) light (brightness > 0) in 2011–2013 only, and 3) light (brightness > 0) in both time periods (see Radford et al., 2019). *Noise pollution* was considered by creating a harmonised noise layer for the study area. This was done by utilising national strategic noise maps from the Alpine countries (except Italy due to unavailability of geodata) (EIONET, 2021). In order to complete the dataset, missing noise information within a 1 km buffer around primary roads and highways in the study area was modelled using GIS-adapted noise propagation formulas from the Nordic Noise Prediction Model (Nielsen, 1997) with annual average daily traffic count data from 2010 to 2014. To ensure accurate predictions, the EU-DEM (EU, 2016) and Corine Land Cover 2018 (EEA, 2020b) were used for ground and vegetation corrections. The 24-hour average noise levels from road, rail, and air traffic, and from industrial sites were assessed. Anthropogenic average daily noise levels 10 dB above natural sound levels have been found to have adverse effects on ecosystems and wildlife (Buxton et al., 2017). The data were therefore categorised into three groups: 0 dB(A), 0–30 dB(A) and > 30 dB(A). *Infrastructure-free areas* were

Table 1
Overview of all spatial indicators used to map wilderness attributes and their associated data sources.

Wilderness attribute	Spatial indicator	Spatial resolution	Data source[1]
Naturalness	Degree of naturalness	100 m (Corine land cover) Vector (river network, OSM)	$LULC_{ref}$ (own calculations based on Corine land cover 2018 (EEA, 2020b), European forest layer 2010 (EEA, 2017a), Pan-European forest/non-forest map 2006 (Pekkarinen et al., 2009), European soil sealing map 2019 (EEA, 2017b), river network (EU, 2020), OSM roads, transport, traffic (OSM, 2020), potential treeline (Pecher et al., 2011)
Human impact	Population density	250 m	Global Human Settlement Map (JRC, 2020)
	Light pollution	800 m	National Oceanic and Atmospheric Administration (NOAA,2020)
	Noise pollution	Vector (EIONET)	END – National Strategic Noise Maps (EIONET, 2021)
	Infrastructure-free areas	25 m ($LULC_{ref}$) Vector (OSM)	OSM roads, transport, traffic, buildings, aerialway (OSM, 2020), OSM power (OSM, 2016) $LULC_{ref}$ (own calculations)
	Landscape fragmentation	25 m ($LULC_{ref}$) Vector (OSM)	$LULC_{ref}$ (own calculations), OSM roads, transport (OSM, 2020)
	Guesthouses	Vector (OSM)	OSM points of interest (OSM, 2020e)
	Mountain huts	Vector (OSM)	OSM points of interest (OSM, 2020e)
	Hiking trails	Vector (OSM)	OSM roads (OSM, 2020a)
	Skiing trails	Vector (OpenSnowMap)	OpenSnowMap (2020)
Remoteness	Remoteness from mechanised access	25 m ($LULC_{ref}$) Vector (OSM)	OSM roads, transport, aerialway (OSM, 2020), DEM (EU, 2016), $LULC_{ref}$ (own calculations)
Ruggedness	Ruggedness index	25 m (EU-DEM)	DEM (EU, 2016)

[1] Sources: EEA, 2020b. Corine Land Cover (CLC) 2018, Version 2020_20u1 Retrieved from https://land.copernicus.eu/pan-european/corine-land-cover/clc2018 (downloaded 11.2020); EIONET, 2021. EU Environmental Noise Directive DF4 and DF 8 National Strategic Noise Maps 2012–2017 Retrieved from https://cdr.eionet.europa.eu (downloaded 02.2021); EU, 2016. EU Copernicus programme. European Digital Elevation Model (EU-DEM), Version 1.1. Retrieved from https://land.copernicus.eu/imagery-in-situ/eu-dem/eu-dem-v1.1?tab = download (downloaded 11.2020); EU, 2020. EU Copernicus programme. Dataset: EU-Hydro – River Network Database, Version 1.3. Retrieved from https://land.copernicus.eu/imagery-in-situ/eu-hydro/eu-hydro-river-network-database?tab = download (downloaded 12.2021); JRC, 2020. Joint Research Center – European Commission – Global Human Settlement Layer dataset. Retrieved from https://ghsl.jrc.ec.europa.eu/download.php?ds = pop (downloaded 11.2020); OpenSnowMap, 2021. Ski slope data. Retrieved from https://www.opensnowmap.org/iframes/data.html (downloaded 11.2021); National Oceanic and Atmospheric Administration (NOAA), Version 4 DMSP-OLS Nighttime Lights Time Series. Retrieved from https://ngdc.noaa.gov/eog/dmsp/downloadV4composites.html (downloaded 11.2020); OpenStreetMap, 2020. Roads, transport, traffic, buildings, aerialway, points of interest. Retrieved from https://osm2shp.ru/ (downloaded 01.2020); OpenStreetMap, 2016. Power. Retrieved from https://osm2shp.ru/ (downloaded 03.2016).

identified by categorising areas of 500x500m resolution into three classes (Radford et al., 2019): 1) completely infrastructure-free (0% of the area covered), 2) minimal infrastructure (0–5% covered), and 3) considerable infrastructure (>5% covered). As done in Zoderer et al. (2020), the indicator was calculated by considering all linear transportation features (e.g., roads, railway lines, tracks and cycle paths, ski lifts), engineering structures (e.g., power pylons, dams), built features, and artificial surfaces (e.g., buildings, airports, ports, sports and leisure facilities). *Landscape fragmentation* was further calculated by considering all settlements, industrial and commercial areas, and all roads and railways as fragmenting infrastructure. Areas without fragmenting structures were reclassified based on area size: 1) > 50 km^2, 2) 10–50 km^2, and 3) 0–10 km^2 (Radford et al., 2019; Zoderer et al., 2020). To further consider impacts of recreational activities on ecosystems and wildlife, which are particularly influential in mountain areas (Sato et al., 2013), we considered the presence of *guesthouses* and *alpine huts* based on OSM data (OpenStreetMap, 2020) and a surrounding buffer of 200 m as suggested by Radford et al. (2019). In addition, the density of *hiking trails* was calculated per raster cell and then classified based on the length of trails per 25 m^2: 1) 0–1 m, 2) 1–5 m, and 3) > 5 m. The same was done for *ski trails*, considering all trails for downhill skiing, ski touring, cross-country skiing, and sledding downloaded from the OpenSnowMap (Table 1). Finally, the nine spatial indicators were normalised on a scale from 0 to 1 and combined using expert weights to derive one human impact layer (see SI, Table A3).

(3) R*emoteness* can be used as a proxy for human-induced disturbances to natural ecosystems and biodiversity (Cao et al., 2019; Carver et al., 2012; Ibisch et al., 2016; Kuiters et al., 2013; Lesslie, 2016; Plutzar et al., 2016). Especially in mountain areas, where disturbances are often caused by people's recreational activities such as hiking or ski touring (Gruas et al., 2023), it is crucial to consider not only accessibility by road or public transport, but also by foot. In this study, we calculated an adapted version of the *remoteness from mechanised access* indicator proposed by Carver et al. (2012), in particular by also considering cable car stations as starting points for exploration in both summer and winter. We estimated the walking time required to reach each pixel cell within the study area from the nearest road, railway, or cable car station, considering the effects of terrain, ground cover, and barrier features such as open water bodies and very steep slopes (>45°). Walking times were estimated by assuming an average walking speed of 4 km/h on flat terrain and an average time of 1 h for every 300 m ascent and 1 h for every 500 m descent as formulated by the German, Austrian, and Slovenian Alpine mountaineering clubs (DIN 33,466 standard). To account for the effect of ground cover on people's walking time, we created a cost-grid considering the extra time needed to walk through each LULC class (SI, Table A.2).

(4) *Ruggedness* refers to the topographic heterogeneity of a landscape (Carver et al., 2012). It is considered a key attribute for wilderness mapping, especially in mountainous areas (Carver et al., 2012; Radford et al., 2019; Zoderer et al., 2020), as it can facilitate the geographic isolation of species at higher elevations, for example by limiting gene flow between populations, and thus significantly contribute to species diversification (Tietje et al., 2022; Verboom et al., 2015). In this study, we calculated a ruggedness index by accounting for changes in terrain curvature based on a high-resolution digital elevation model (Table 1). Following the approach of Carver et al. (2012), we calculated the standard deviation of terrain curvature within a 250 m radius of each pixel cell.

2.2.3. Combining wilderness attributes using Weighted Linear Combination

The spatial indicator maps were subsequently combined using Weighted Linear Combination (WLC) to calculate an overall WQI (Carver et al., 2012; Zoderer et al., 2020). To account for differences in the contribution of each spatial indicator, indicator weights were derived from Radford et al. (2019), who conducted a survey with 22 international experts working in the fields of nature conservation. All attribute layers were normalised to a common relative 1–256 scale, where 1 indicates the lowest and 256 the highest attribute values (referred to 'min' and 'max' in subsequent figures). Normalisation was achieved in ArcGIS using equal interval classes to facilitate cross-comparison and guarantee that higher values of the individual attributes contribute to overall higher wildness and lower values to overall lower wildness. Finally, the four attribute layers were overlaid and weighted according to expert opinion (*naturalness*: 1.07, *human Impact*: 1.20, *remoteness*: 1.07, *ruggedness*: 0.67; see Radford et al., 2019) using the following weighted linear summation formula:

$$S_i = \sum_{j=i}^{n} W_{ij} X_{ij}$$

where n = 4, Si is the overall wilderness quality value of the i^{th} pixel cell, W the attribute weight, and X the standardised value of each attribute. The resulting map was again normalised to a scale of 1–256 using equal interval classes to derive a final wilderness quality map.

2.3. Selection of priority areas for process-oriented conservation

To identify priority areas for process-oriented conservation strategies, an *ecoregional* approach was adopted and compared to a *conventional* approach. The ecoregional approach involved the selection of the 10% wildest cells within each ecoregion of the study area. To this end, we first delineated all ecoregions contained within the administrative boundaries of the Alpine Space area by clipping the map of European ecological regions (thereafter named ecoregions ER) derived from the European Environment Agency (EEA, 2015) to the extent of the Alpine Space area. This resulted in a map displaying 11 distinct ER covering the Alpine Space area (Fig. 2). To identify ecoregion-specific priority areas for process-oriented conservation, the ER map was overlaid with the wilderness quality map and the 10% wildest areas of each ER were identified. The resulting ecoregion-based selection of priority areas both reflects differences in the proportion of area covered by each ecoregion in the study area (Fig. 2) and differences between the ecoregion-specific thresholds of wilderness quality value (Table 2). For the 'conventional approach', we identified the 10% wildest cells of the entire study region (as done by Kuiters et al., 2013). This corresponds to all cells with a wilderness quality value higher than a threshold of 157 (Table 2).

2.4. Characterising priority areas for process-oriented conservation

We characterised the identified priority areas according to their distribution across elevation classes, the ecosystem types they cover, their area size, and protection status. In total, 15 different ecosystems were considered for characterisation by reclassifying the $LULC_{ref}$ map, resulting in ten terrestrial, three freshwater, and two coastal ecosystems. We assessed the distribution of priority areas across different size categories (<500 ha, 500–1,000 ha, 1,000–3,000 ha, 3,000 – 10,000 ha, and > 10,000 ha). These correspond to different area-based thresholds (e.g., 500 ha, 1,000 ha, 3,000 ha) that have previously been discussed as critical requirements for the effective functioning of natural processes (Brackhane et al., 2019; Wild Europe, 2012). In addition, the protection status of the identified priority areas was assessed using data from the 2021 World Database on Protected Areas (WDPA; UNEP-WCMC and IUCN, 2022). The database contains spatial information, information on protection level, and year of designation. Following the International Union for Conservation of Nature (IUCN), the WDPA distinguishes between seven IUCN categories (Ia, Ib to VI), ranging from strictly protected areas (IUCN Ia, Ib, and II) to protected areas that allow certain human activities and sustainable resource extraction (IUCN III to VI). The WDPA further distinguishes these protected areas from all other protected areas that either do not qualify for IUCN classification ('not applicable'), weren't reported yet ('not reported'), or are still in the process of certification ('not assigned'). For the purpose of this analysis, we grouped the latter three categories under the category 'Non-IUCN' protected areas. Many of these are UNESCO World Heritage Sites or Natura2000 sites such as Special Protection Areas (SPA) or Sites of Community Importance (SPI), as well as regionally protected areas (e.g., wildlife tranquillity areas, nature parks). Before using the data for analysis, the database was checked for duplicates. In case of multiple IUCN categories per protected area, we considered the lower IUCN category representing the stricter protection status.

Table 2
Distribution of wilderness quality values (1–256) across the Alpine Space area (conventional approach) and across ecoregions within the Alpine Space area (ecoregional approach), with information on the cut-off value for the 10% wildest areas. Ecoregions are derived from the European Environment Agency (EEA, 2015).

	Min	Max	Mean	S.D.	10 % wildest cut-off
Conventional approach					
Alpine Space	1.0	256.0	110.7	37.6	157
Ecoregional approach					
Alps conifer and mixed forests (ER 1)	1.0	256.0	135.8	33.2	175
Western European broadleaf forests (ER2)	2.0	202.0	92.6	29.0	128
Northeastern Spain & Southern France Mediterranean (ER3)	8.0	212.0	112.9	32.4	150
Po Basin mixed forests (ER4)	6.0	193.0	75.8	23.5	108
Pannonian mixed forests (ER5)	8.0	174.0	93.5	29.4	129
Italian sclerophyllous and semi-deciduous forests (ER6)	6.0	180.0	105.0	29.7	139
Dinaric Mountains mixed forests (ER7)	8.0	215.0	131.6	25.2	159
Central European mixed forests (ER8)	8.0	161.0	96.6	26.2	127
Illyrian deciduous forests (ER9)	8.0	159.0	101.7	29.1	134
Sea (ER10)	5.0	174.0	120.0	36.3	146
Appenine deciduous montane forests (ER11)	21.0	171.0	135.5	16.6	154

3. Results

3.1. Wilderness quality map

The wilderness quality map (Fig. 3e) shows that areas with high wilderness quality are mainly located at higher elevations in the Central Alpine arc. The wildest areas occurred in the UNESCO World Heritage Site Jungfrau-Aletsch around the Aletsch glacier, the Mont Blanc massif, in the Pennine Alps on the Swiss border to Italy, and in the Massif des Ecrins range in the French Alps. These high mountain areas are among the least accessible places in the study area, as they are either still covered by vast extents of glaciers or are located in rugged and difficult to transverse terrain (Fig. 3c,d). In addition, they lack an extensive network of hiking trails as well as anthropogenic infrastructures such as skiing facilities or alpine huts. Areas with comparably lower wilderness quality, by contrast, are found on the intensively used, most densely populated valley bottoms and in the lowlands outside the Alpine arc, which are characterised by a high population densities, high densities of anthropogenic infrastructures, and intensive forms of agricultural use (see Fig. 3a,b).

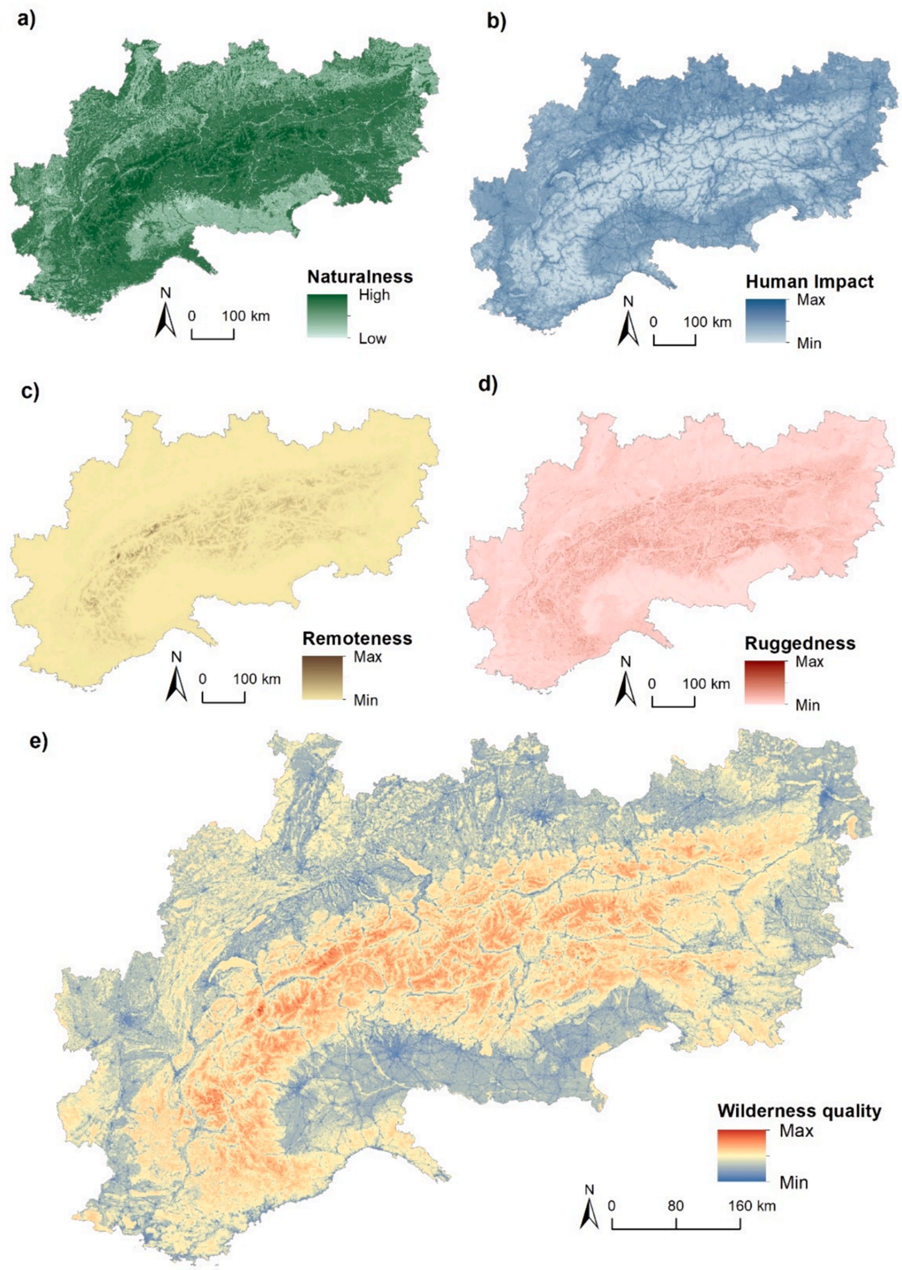

Fig. 3. Maps displaying the spatial variation in wilderness attributes (a-d) and wilderness quality (e) across the study region. All maps display normalised values, ranging from 1 (min) to 256 (max).

3.2. Priority areas for process-oriented conservation: Conventional approach

We first present the results of the conventional approach, which prioritised the 10% wildest areas of the entire Alpine Space. These areas predominantly lie within the Alpine arc, with 93% being located in ER1 (Alps conifer and mixed forests) and, to a lower extent, in ER3 (North-eastern Spain & Southern France Mediterranean) in the southwest (4.1%) and in ER7 (Dinaric Mountains mixed forests) on the Slovenian border to Austria and Italy (1.7%) (Fig. 4a, Table 3). Because of the dominance of high wilderness qualities in these ecoregions, no or only few small, scattered patches suitable for process-oriented conservation are found in other ecoregions. A comparison of priority areas by elevation class shows that more than 60% are located above 2,000 m a.s.l., with most being found between 2,000 and 2,500 m a.s.l. (Fig. 4c). This pattern is due to the significantly higher concentration of the most remote and least impacted areas in the subalpine-nivale zone. In line with the greater occurrence of priority areas at higher elevations, we find that areas suitable for process-oriented conservation according to this selection approach mainly comprise bare rocks (33.0%), sparsely vegetated areas (24.6%), and to a smaller extent coniferous forests (14.9%), broadleaf forests (5.0%), transitional woodlands such as alpine shrub vegetation (4.6%), and glaciers (4.4%) (Fig. 4d and Fig. 6c). In contrast, only few patches of freshwater and coastal ecosystems are found among the priority areas, as many of them are located in or near to areas with high population densities and intensive land use.

We find that the majority of identified priority areas comprise an area larger than 10,000 ha in size (n = 71 areas, 62.4% of all priority areas). Furthermore, 81 priority areas reach an area of at least 3,000 ha (Fig. 5a and Fig. 5c). Today, 52.6% of all priority areas identified using the conventional approach are protected (Fig. 6a, see conventional approach). While 13% are strictly protected by IUCN categories I or II

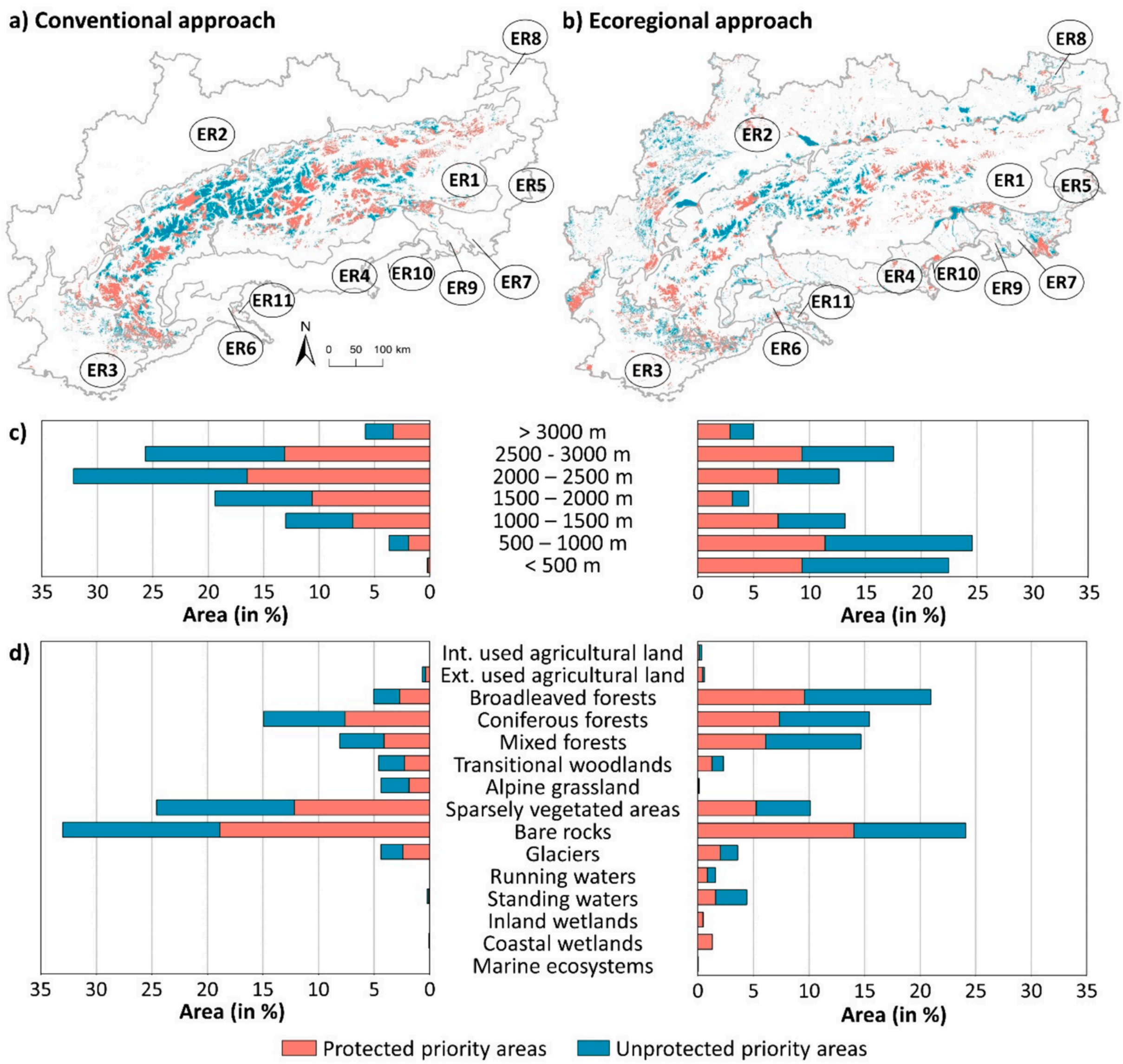

Fig. 4. Spatial distribution of priority areas for process-oriented conservation in the study region as identified by the conventional approach (a) and the ecoregional approach (b), and their distribution across elevation classes (c) and ecosystems (d).

(1.1% by IUCN Ia and Ib, 12.3% by IUCN II), the majority is under the protection of IUCN categories IV, V and other conservation schemes that currently lack an IUCN designation such as UNESCO World Heritage Sites or Natura2000 areas. The analysis further reveals that the largest protection gap is found in Switzerland, where a large part of the 10% wildest cells is located throughout the Alpine Space area. Whilst almost all of the few identified coastal and marine wetlands are currently protected, standing waters and naturally occurring alpine grasslands currently have the largest protection gaps (Fig. 4d).

3.3. Priority areas for process-oriented conservation: Ecoregional approach

The ecoregion-specific priority areas differ in location and characteristics from those identified by the conventional approach (Fig. 4b). In the ecoregion-based approach, priority areas were of lower wilderness quality due to the fact that wilderness quality cut-off values were lower in some ecoregions than the threshold derived for the entire study area (Table 2). This is particularly the case in ecoregions with significant human activities such as ER4 (Po Basin mixed forests), where areas of moderate wilderness quality constitute the last available areas for the restoration of natural processes. Conversely, in ecoregions such as ER1, where human impact is comparably lower, far fewer areas with high wilderness quality are prioritised for process-oriented conservation than in the conventional approach (Table 3). Compared to a concentration of priority areas in mountain areas at higher elevations in the conventional approach, the ecoregional approach results in a larger selection of priority areas at lower elevations (Fig. 4c), and correspondingly, a greater coverage of different ecosystems (Fig. 4d). Overall, significantly more broadleaf forests (21.0%), mixed forests (14.7%), coastal wetlands (1.3%), standing and running waters (4.4% and 1.6%, respectively) are prioritised. Depending on the ecoregion, however, the dominant ecosystems significantly differ (Fig. 6c). Whereas in ER 1 more than 80% of the areas selected are covered by bare rocks or sparsely vegetated areas, priority areas in ER2 to ER9 and ER11 are dominated by forest areas and in ER10 by coastal wetlands. Many of these priority areas, particularly those located in the lowland areas of ER 2, ER4 and ER5, are only small fragments with an area size of less than 500 ha (Fig. 5b). In comparison, 41 areas are larger than 10,000 ha in size and 152 areas larger than 3,000 ha (19.4% and 24.1% of the total identified area, respectively).

We find that 50.4% of the sites identified by the ecoregional approach are currently protected, with 8.2% being protected by IUCN I

Table 3
Distribution of priority areas identified by the conventional and ecoregional approach across the ecoregions of the study region.

Ecoregions (ER)	Conventional approach		Ecoregional approach	
	Priority areas (in km^2)	Priority areas (in %)	Priority areas (in km^2)	Priority areas (in %)
Alps conifer and mixed forests (ER 1)	36671.0	93.0	15068.4	37.7
Western European broadleaf forests (ER2)	154.2	0.4	13179.9	32.9
Northeastern Spain & Southern France Mediterranean (ER3)	1620.9	4.1	3349.0	8.4
Po Basin mixed forests (ER4)	81.4	0.2	3214.8	8.0
Pannonian mixed forests (ER5)	29.1	0.1	2738.7	6.8
Italian sclerophyllous and semi-deciduous forests (ER6)	131.9	0.3	1269.5	3.2
Dinaric Mountains mixed forests (ER7)	683.1	1.7	577.9	1.4
Central European mixed forests (ER8)	0.3	<0.1	243.1	0.6
Illyrian deciduous forests (ER9)	0.9	<0.1	206.4	0.5
Sea (ER10)	5.8	<0.1	70.5	0.2
Appenine deciduous montane forests (ER11)	52.1	0.1	86.5	0.2

or II (0.65% by IUCN Ia and Ib, 7.5% by IUCN II), 14.9% by IUCN V, and 17.3% by protected areas without IUCN designation (Fig. 6b, see aggregated results for ecoregional approach). Interestingly, this pattern significantly differs across ecoregions (Fig. 6a, see ER-specific results). While a larger share of priority areas is located within IUCN I or II protected areas in ER1, ER7 and ER10, none of the priority areas of ER2 and ER4 are currently protected by any of these stricter protected area types. Despite some of the identified areas being protected by IUCN category V or other non-IUCN protected areas in ER2 and ER4, they host the overall largest protection gap (57.3% and 58.7% unprotected priority areas, respectively). The protection gap is lowest in ER7, by contrast, as the majority of priority areas are located within the borders of the Slovenian Triglav National Park. Corresponding to the conventional approach, we find that more than 90% of all identified coastal or inland wetlands are currently protected, while only a third of the identified standing waters are localised within a protected area. Furthermore, we find that forests, despite their greater significance in the ecoregional approach, are less protected than other terrestrial ecosystems.

4. Discussion

4.1. Opportunities for wilderness protection

In line with previous studies (Kaissl, 2002; Kuiters et al., 2013; Plutzar et al., 2016; Radford et al., 2019; Zoderer et al., 2020), our results show that many of the mountainous regions of the European Alps still contain areas of high wilderness quality and are therefore likely to be covered by functionally intact, self-regulating ecosystems that are mainly governed by natural processes. In particular, mountain areas with high remoteness and ruggedness are characterised by high wilderness qualities and can be distinguished from mountain areas with more intensive forms of human activities and infrastructure development. Our study shows that these areas are often large, with more than half of them comprising areas larger than 3,000 ha, thus also meeting the area-based requirement for ‘wilderness’ designation proposed by Wild Europe (2012). Indeed, many of the identified wildest areas in the Alpine arc would constitute *de facto* wilderness areas and meet the requirements for strict protection proposed by IUCN under category Ia/b or the core zone of II (Dudley, 2008). Compared to *de jure* wilderness areas (i.e. protected by IUCN Ia/b or II), the proportion of *de facto* wilderness areas is significantly higher, highlighting the high potential associated with the designation of remaining wilderness areas in the European Alps (Kuiters et al., 2013). For instance, in the Alpine Arc less than a fifth of all identified wildest areas are currently strictly protected (see ecoregional approach, ER1), suggesting that there is a high potential for upgrading protected area status under the ‘10% target’ of the EU Biodiversity Strategy by 2030. As many of the identified wildest areas extend beyond national borders, transboundary protected areas would need to be established to realise the full potential of protecting these remaining large, contiguous natural habitats and their associated natural processes.

4.2. Potential of the ecoregion-based approach

The results of this study show significant differences in the location and characteristics of priority areas for process-oriented conservation depending on whether they were identified using an ecoregional or a conventional approach. In the conventional approach, the priority areas are mainly concentrated in one ecoregion (i.e., Alps coniferous and

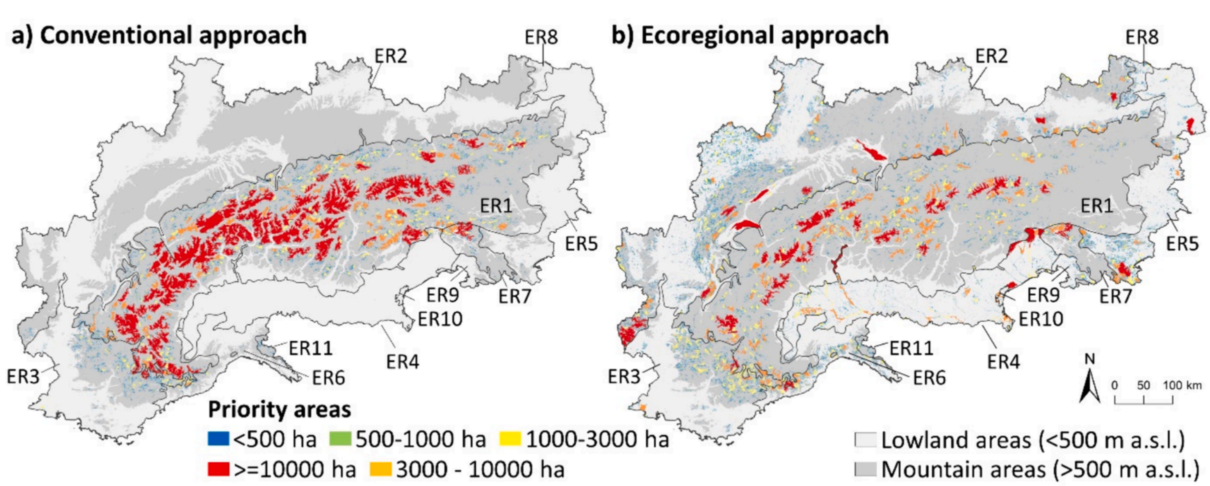

Fig. 5. Spatial distribution of priority areas for process-oriented conservation differentiated by area size and shown for both the conventional approach (a) and the ecoregional approach (b).

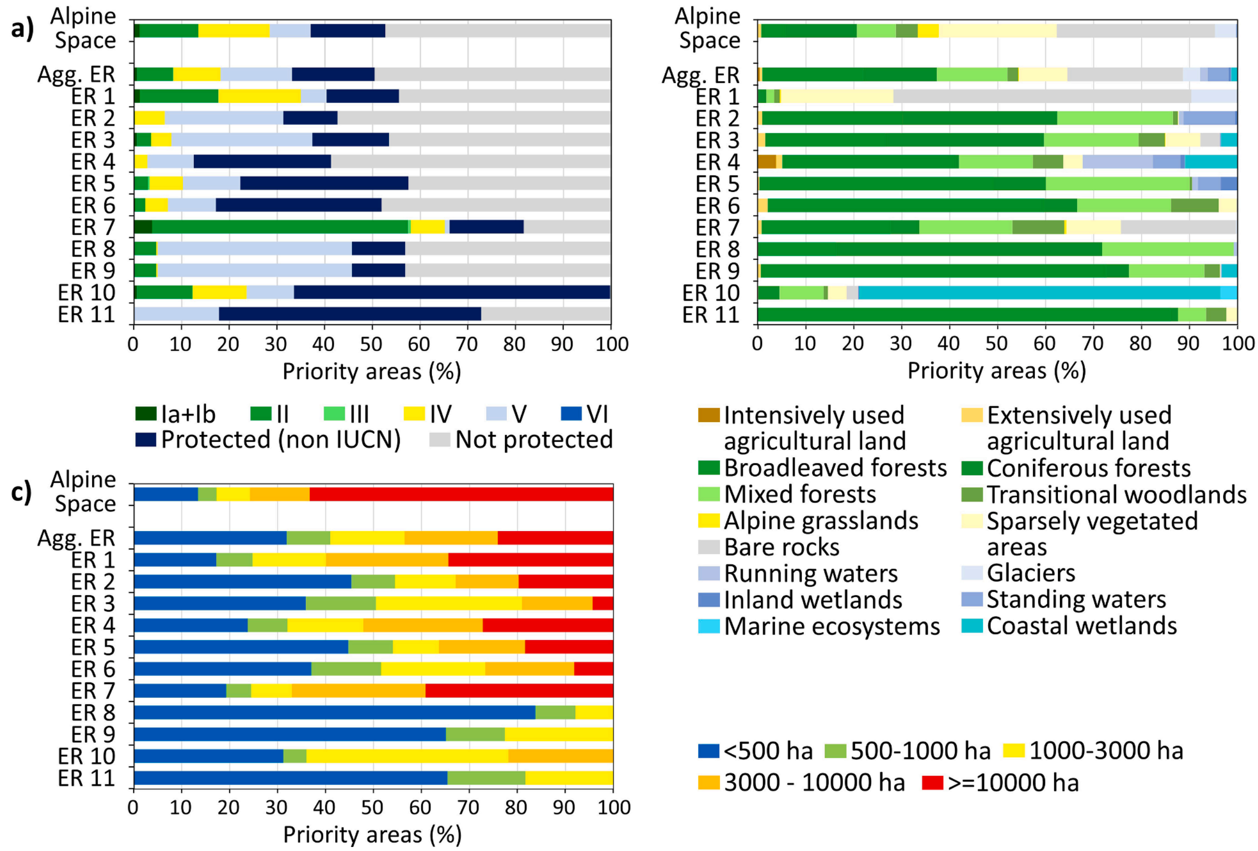

Fig. 6. Shares of priority areas across IUCN protected area categories (a), ecosystems (b), and area size classes (c). Results are shown for the conventional approach (i.e., Alpine Space area) and the ecoregional approach (i.e., aggregated across all ecoregions and for each individual ecoregion).

mixed forest) and predominantly target the above described *de facto* and *de jure* wilderness areas such as large-scale, contiguous mountain ecosystems at higher elevations. These areas represent valuable spaces of high connectivity value that are large and compact enough to allow natural processes to occur largely undisturbed and native species and communities to persist over longer periods without management interventions (Brackhane et al., 2019; Kuiters et al., 2013). While the conservation of these areas is of utmost importance for maintaining natural processes on large spatial scales, a sole focus on their conservation might risk overlooking the importance of securing natural processes also in other ecosystems with a different species-level diversity. Indeed, our study shows that the ecoregion-based selection of priority areas can result in a more balanced representation of ecosystems, including most of the large-scale, contiguous mountain ecosystems but also a greater diversity of smaller-scale, dispersed ecosystems at lower elevations. Thus, the results demonstrate that an ecoregion-based selection can promote a greater representation of ecosystem-diversity and associated species assemblages (see also Dinerstein et al., 2017; Smith et al., 2018). This is also in accordance with the central goals of the UN International Biodiversity Agreement (CBD/WG2020/2/4; https://www.cbd.int) and the EU Biodiversity Strategy to 2030 (EC, 2022), namely to comprehensively protect and restore the full spectrum of ecosystems and their species to halt biodiversity loss. As the latter is crucial for overcoming the biodiversity crisis (Díaz et al., 2020), we consider it necessary to complement a conventional approach to selecting suitable candidate sites for process-oriented conservation with an ecoregion-based selection in order to introduce process-oriented strategies where they are needed the most to protect and restore biodiversity-rich natural habitats.

Previous studies have also shown that a more nuanced selection of priority areas is needed to protect biodiversity as a whole, taking into account both the protection of remaining low-disturbance areas and more threatened areas in fragmented landscapes with greater human activity (Kennedy et al., 2019; Mokany et al., 2020; Pimm et al., 2018). The ecoregion-based selection proposed in this study would better reflect such a strategy in the context of process-oriented conservation, as it identifies both remaining large-scale wilderness areas in the most remote and least disturbed regions, as well as remaining small-scale intact habitat fragments embedded in highly modified parts of the study region. Conservation of remaining low-disturbance wilderness areas is particularly important for maintaining species with large home ranges, such as many carnivore species (Pimm et al., 2018; Ripple et al., 2014). However, these areas may not be among the most biodiverse areas in the study region and may not overlap with areas where endangered species are concentrated (Tasser et al., 2023).

Areas of high biodiversity value often lie outside mountain areas (Iannella et al., 2020) and are therefore likely to coincide with the small remnant natural and semi-natural habitats identified in the lowlands of the study region such as dry grasslands, steppe grasslands, and old-growth forests (Habel et al., 2013; Muys et al., 2022). Although of lower wilderness quality, the latter still represent the wildest areas in the ecoregion and may therefore provide refuges for species that may have lost much of their habitat elsewhere (Dinerstein et al., 2017; Mokany et al., 2020). Furthermore, our findings show that the small size, fragmented nature, and embeddedness of many of these areas in a highly modified landscape matrix make them particularly vulnerable to ongoing threats from the surrounding landscape (Kennedy et al., 2019). This suggests that priority actions are needed that focus not only on preserving and restoring these areas and their natural processes but also on actively managing ongoing threatening processes in the surrounding landscape (Mokany et al., 2020). A landscape-based conservation approach such as the 'Protected Area Centered Ecosystem' (cf. Belote et al., 2021) could be beneficial here. According to this approach, the area needed to sustain species and ecological processes in core protected areas would need to be much larger than their own extent, and be in the range of 4,000–40,000 km^2 (ibid.). Thus, applying a landscape-scale approach to conservation would require a landscape planning strategy that additionally promotes the sustainable use of land in the surrounding landscape to minimise ongoing threats and pressures.

4.3. Challenges of the ecoregion-based approach

Despite its benefits for biodiversity conservation, the ecoregion-based selection of priority areas for process-oriented conservation also comes with some challenges for conservation practice and local communities inhabiting or living close to the targeted areas. The following paragraphs will discuss these challenges in more detail.

4.3.1. The need for more restoration efforts

As the ecoregion-based approach also pinpoints suitable candidate sites in regions with considerable human activity, more restoration efforts will be required in addition to the protection of remaining wilderness areas and smaller intact ecosystems. Particularly in ecoregions where the wildest areas are small, scattered, and of relatively low wilderness quality, such as in ER2, ER4, ER8 or ER9, rewilding could be used as a process-oriented restoration strategy to restore natural processes, increase the self-sustaining capacity of the identified ecosystems as well as the connectivity of remaining intact areas (Carver et al., 2021; Perino et al., 2019). Our results show that in most ecoregions this will target forest ecosystems such as deciduous, coniferous, or mixed forests, and will need to be complemented by rewilding activities in other ecosystems such as aquatic ecosystems (e.g., ER2 and ER4) or even existing agricultural lands (e.g., ER3, ER4, ER6). Depending on the ecosystem, rewilding activities can range from transitioning forests to low or no intervention management regimes, to restoring natural river dynamics or reinstating natural grazing regimes by introducing large-bodied herbivores (Perino et al., 2019; Van Meerbeek et al., 2019). The latter could be considered as an option alongside more passive rewilding approaches, especially in ecoregions such as the Po Basin (ER4), where about 5% of the identified priority areas overlap with existing extensively and intensively used agricultural areas, as natural or semi-natural ecosystems are not sufficiently available (see also Müller et al., 2020). As the latter will lead to significant changes in ecosystem structure and function (Schulte to Bühne, Ross, et al., 2022), the suitability of an open-ended restoration approach will need to be carefully assessed in light of the local species composition.

For instance, in habitats with rare, disturbance-dependent species populations, management strategies that maintain semi-open conditions through continued human intervention may be more appropriate (Hughes et al., 2012; Van Meerbeek et al., 2019). In addition, the occurrence of neobiota and discussions about the appropriateness of their management can lead to conflicts between approaches of process-based conservation and species-based conservation (Westermann & Von Oheimb, 2021). While invasive neobiota are frequently not actively managed in areas where a process-oriented approach is applied (Brackhane et al., 2019; Schumacher et al., 2018), their colonisation of habitats with rare species may represent a significant threat. Thus, to avoid conflicts between process-oriented and more traditional conservation approaches, it is recommended to refrain from implementing process-oriented conservation strategies in areas already heavily colonised by neobiota (Brackhane et al., 2019). In addition, the implementation of preventive measures will also be necessary to reduce the necessity for control and eradication measures in areas where process-oriented strategies are about to be introduced.

Overall, our results suggest that different 'shades' of rewilding activities (cf. Jepson and Schepers, 2016) will be needed, depending on the ecological condition and wider socio-ecological context. While all efforts may increase the relative wildness of the existing ecosystem to some extent, rewilding will not always lead to *"the restoration of functioning native ecosystems containing the full range of species at all trophic levels while reducing human control and pressures"* (Carver et al., 2021, p. 1888). Where areas are small, depleted of native species, and threatened by the surrounding environment, restoration efforts are unlikely to produce these outcomes, but can still increase the 'autonomy' and spontaneity of individual natural processes and move these areas up the wilderness continuum (Corlett, 2016; Schulte to Bühne, Pettorelli, et al., 2022).

4.3.2. The need for more ambitious changes to the protected area network

While at an aggregated level both the ecoregional and conventional approach indicate that about half of the priority areas for process-oriented conservation lie outside existing protected areas, they differ in how they prioritise the expansion of existing protection efforts. The conventional approach places great focus on expanding and connecting already existing protected areas, including a higher share of strictly protected areas, in federal regions that have a high proportion of their territory located in ER 1. In contrast, the ecoregion-based approach distributes the priorities for protected area expansion more evenly across individual ecoregions, member states and/or federal regions. Furthermore, the comprehensive consideration of all ecoregions in the ecoregional approach makes evident where the protection gaps are largest and calls for more ambitious expansion of protected areas in ecoregions of significant human activity than in the conventional approach. Confirming previous studies (Müller et al., 2020, 2018), we find that the largest protection gap exists in the lowland plains of the Po basin, where less than 50% of the identified areas are protected. In this ecoregion, but also in ecoregions such as the Italian sclerophyllous and semi-deciduous forests (ER6) and Western European Broadleaved Forest (ER2), the expansion of the protected area network could particularly target a broad mix of ecosystems and associated species that are currently not protected. As their dispersed nature and often small size may pose particular challenges to such efforts, it will be crucial to identify opportunities to connect these areas with existing protected areas (Baumann et al., 2020). To this end, our analysis could be used as an input to wildland network planning that identifies core protected areas for process-oriented conservation and their linkages through corridors that are of relatively high ecological integrity or rewilding potential (Cao et al., 2020).

Compared to the conventional approach, the ecoregion-based selection of priority areas highlights a greater need to improve the management of existing protected areas if the '10% target' of the EU Biodiversity Strategy is to be achieved in these areas by 2030. According to the biodiversity strategy (EC, 2022), 10% of the EU's marine and terrestrial territory should be 'strictly protected' by 2030, leaving *'natural processes […] essentially undisturbed by human pressures and threats'* (EC, 2022, p. 19). While it was not our intention to precisely delineate areas to meet the '10 % target', our results can still provide indications of the potential of existing and newly established protected areas to contribute to this target. Most importantly, our results highlight that if 'strict protection' is interpreted as the implementation of process-oriented strategies only within IUCN Ia/b or II protected area categories, the majority of all ecoregion-specific priority areas located within existing protected areas would need to be upgraded.

However, this large gap and the fact that it is difficult to meet all the IUCN requirements for strict protection also suggest that in many cases 'strict protection' is unlikely to lead to IUCN Ia/b or II designation. Instead, future research needs to explore in more detail how process-oriented strategies can be implemented within existing protected area types (e.g., IUCN IV or V) or Natura2000 sites, identifying synergies but also potential conflicts with existing conservation objectives such as the protection of threatened species and habitats (but see European Commission et al., 2013). In addition, future work could explore the potential for implementing process-oriented strategies in smaller core areas of existing protected areas and use other zones of the existing protected area as buffer from surrounding human pressures and threats.

4.3.3. Greater impacts on local communities

As the ecoregion-based selection prioritises areas for process-oriented conservation in closer vicinity to where people live and use the land, it is likely to have a greater impact on people's livelihood and daily interaction with nature. We argue that this can present both challenges and opportunities that need to be carefully considered when planning and implementing process-oriented strategies in practice. Challenges may include an increase in human-wildlife conflicts, natural hazards, or uncertainties associated with the open-ended nature of process-oriented strategies. In particular, where process-oriented strategies are to be implemented in landscapes that have previously been used for agricultural purposes, as in ER 4, changes to the landscape composition with which people are familiar are likely to occur. As shown previously, such changes are often perceived by local communities as a threat to existing aesthetic, cultural, and other relational values associated with the landscape (Bauer et al., 2009; van der Zanden et al., 2018; Zoderer & Tasser, 2021). Such perceived risks may also vary between different social groups, with more affected groups likely to perceive more risks (Zoderer & Tasser, 2021). Despite these challenges, protecting and restoring self-regulating ecosystems in more densely populated areas can also increase opportunities for people to benefit from ecosystem services (Cao et al., 2022), experience 'wilder' nature, strengthen human-nature relationships, and thereby overcome the often cited 'shifting baseline syndrome' (Soga & Gaston, 2018).

4.4. Methodological considerations

The consideration of the wilderness continuum concept and its operationalisation through the wilderness quality index (WQI) has proven to be a valuable approach for screening potential areas for process-oriented conservation. First, the relativity of the WQI has allowed us to focus not only on the identification of remaining primary habitats, but also on the identification of areas that may be suitable for restoration efforts due to their relatively higher wildness compared to other areas. This better reflects the reality of the Anthropocene, where every parcel of land is likely to have already been altered by humans to some extent (Wohl, 2013); it also takes into account the fuzzy nature of the wilderness concept, which makes it difficult to draw clear boundaries where wild nature begins and ends (Vannini and Vannini, 2016; Zoderer et al., 2020). Second, using the WQI to identify priority areas follows a conservative approach that, even when combined with an ecoregion-based approach, still aims to minimise impacts on society, including avoiding areas of high population density and intensive agricultural land use (see also Schleicher et al., 2019).

Despite these advantages, our methodological approach comes with some limitations. First, our spatial analysis was impacted by some data limitations. For instance, spatially explicit data on forest management intensity were not available at the scale of the study region, nor were data on the use of large water bodies or the extent of natural river flow regulations. While we used expert estimates instead, including the consideration of the effect of elevation on use intensity, this may have introduced a bias in the mapping of the naturalness indicator. This may be particularly true for forests, water bodies, and rivers if they are located in lowland areas. Similarly, the wilderness quality of large lowland lakes may have been overestimated because disturbance from ferry routes and water-based accessibility were not considered. The latter was done in Carver et al. (2012), where a water-based remoteness model was additionally run to account for the fact that some sites are more easily accessible via ferry or water taxi routes. Second, the selection of the wildest areas per ecoregion was likely impacted by the administrative boundaries of the study region. For instance, while we identified the wildest areas of ER 7 to be almost entirely located in the Triglav National Park, these areas may not be among the wildest if we consider the entire ecoregion which extends far beyond our study region. Whilst the comparison of the ecoregional and conventional approach provided valuable insights into the overall pattern of the identified priority areas, we contend that an ecoregional approach is best suited for identifying priority areas at a continental scale (e.g., Müller et al., 2020, 2018). We therefore recommend that the EU uses an ecoregional approach at the scale of the entire EU territory to identify priority areas for (strict) protection as part of the EU Biodiversity Strategy to 2030.

Finally, interpretations of the suitability of priority areas for biodiversity conservation were based on the distribution of these areas across the 11 ecoregions of the study region. Previous studies have provided strong evidence that ecoregions can serve as a proxy for ecosystem-level, but also community and species-level biodiversity in the absence of high-resolution spatial data (Smith et al., 2018). Yet, our approach would benefit from comparison with biodiversity data collected through field surveys or habitat modelling to better assess how identified priority areas align with local patterns of species within each ecoregion, and how process-oriented strategies would affect the conservation of these species assemblages.

5. Conclusion

Systematic conservation planning can be an important tool for identifying priority areas for biodiversity conservation and for spatially assessing the suitability of different conservation strategies. This study provides a basis for prioritising potential areas for process-oriented strategies using the 'wilderness continuum' concept in combination with an ecoregion-based approach. Our results show that an ecoregion-based identification of priority areas can target a greater diversity of ecosystems and species but also highlight the need to move beyond a sole focus on protecting remaining, large-scale wilderness areas in remote mountain areas with little human activity. In particular, the findings suggest that if process-oriented approaches are to benefit biodiversity conservation, it will be essential to complement the protection of large-scale wilderness areas in less modified areas with the protection of intact, small-scale ecosystems, and the restoration of self-regulating ecosystems through rewilding in regions with greater human activity.

Given the relativity and scalability of the wilderness continuum concept, the approach adopted in this study is sensitive to the geographic extent and spatial characteristics of the study region. While our study shows large differences between an ecoregional and conventional approach in identifying priority areas for process-oriented conservation in the context of a mountain region, we encourage future research to explore how potential differences between the two approaches play out in lowland regions and at larger continental scales. In addition, future research can complement the geographical analysis carried out in this study with spatial data on species distribution, land ownership, or socio-economic costs to allow a more in-depth consideration of the suitability and feasibility of different process-oriented conservation strategies in practice. As an ecoregion-based approach to process-oriented conservation is likely to take place closer to where people live and use the land, complementing the geographic analysis presented here with participatory approaches that recognise the plural values, needs, and knowledge of different stakeholder groups will be crucial for successfully scaling up process-oriented conservation efforts in the future.

CRediT authorship contribution statement

Brenda Maria Zoderer: Conceptualization, Data curation, Formal analysis, Investigation, Methodology, Visualization, Writing – original draft. **Thomas Marsoner:** Data curation, Formal analysis, Writing – review & editing. **Erich Tasser:** Conceptualization, Formal analysis, Methodology, Visualization, Writing – review & editing.

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

Data will be made available on request.

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Appendix A. Supplementary material

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jnc.2024.126661.

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