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How remote sensing choices influence ecosystem services monitoring and evaluation results of ecological restoration interventions

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ABSTRACT

Large-scale ecological restorations are recognized worldwide as an effective strategy to combat environmental degradation and promote sustainability. Remote sensing (RS) imagery, such as obtained from Landsat and Sentinel-2 satellites, can provide spatial, spectral, and temporal information on ecosystem service supply to support monitoring and evaluation of restoration interventions. However, because of the abundance of satellite data and methodological analysis options, choices in data selection and processing options need to be made. This study explored the effect of RS choices on the evaluation of changes in ecosystem services as a result of ecological restoration interventions. Using the ecosystem service of forage provision for wildlife as an example, we used a before-after-control-impact (BACI) analysis to compare how the following choices affected restoration evaluation outcomes: a) different number of control pixels; b) different spatial distribution of control pixels; c) intra-annual image selection; and d) different reference periods. In addition, e) we evaluated the effect of using two different satellite sensor types, using the ecosystem service 'erosion prevention' as an example. We explored the effect of these five choices for restoration sites in the Baviaanskloof, South Africa. Results showed that the choice of intra-annual image selection, and the reference period describing the 'before state' had a strong effect on the outcomes, often leading to opposite BACI evaluation results. BACI results were less sensitive to choices related to the number of control points in the evaluation. The impact of methodological choices on the BACI outcomes was greater for the less degraded areas of our study site. Satellite sensor choice resulted in similar temporal trajectories of estimated supply. We demonstrated that RS choices have a strong effect on the evaluation results of restoration interventions. Therefore, we recommend that documenting the key RS choices results is essential when communicating restoration evaluation results in order to properly understand, manage and adapt restoration initiatives.

1. Introduction

Land degradation reduces ecological functions that support life (IPBES, 2018). In 2021 the UN Decade of Ecological Restoration started, aiming to halt and reverse ecosystem degradation worldwide. According to the Society for Ecological Restoration (SER), ecological restoration is 'the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed'. Since ecological restoration is defined as a process and not an outcome (Gann et al., 2019), in this study we consider all land that underwent a restoration intervention as 'restored' regardless of their level of recovery or degradation after the intervention occurred. Ecological restoration interventions can be costly and labor-intensive, but simultaneously can lead to economic and other benefits to people (Cornell et al., 2016; Stafford et al., 2017; Verdone and Seidl, 2017). For example, a recent study found that restoring and conserving nature frequently outweigh the overall profit that resource extraction generates (Bradbury et al., 2021). Investments in land restoration need to consider ecological and social costs and benefits carefully. Since ecosystem services represent the link between nature and human wellbeing, they can be used as a measure to evaluate restoration impact in an integrated way (Alexander et al., 2016; Carlucci et al., 2020; Matzek et al., 2019).

In the quest for addressing sustainability challenges, there is a growing recognition of the crucial role played by understanding the influence of interventions on ecosystem services (Costanza et al., 2017). The importance of the concept of ecological restoration has increased significantly with the declaration by the United Nations (UN) of the 'Decade on Ecosystem Restoration' for the period 2021–2030,

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recognizing the critical role of restoration on ecosystem health and human wellbeing. Ecological restoration projects also often fall within the scope of nature-based solutions, as they involve the recovery and rehabilitation of ecosystems to enhance their ecological functionality and provide multiple benefits to both nature and people (IUCN, 2020). Enabling evidence-based decisions regarding land restoration and nature conservation requires monitoring the spatial and temporal dimensions of ecosystem services, which has been recognized as essential by science-policy bodies such as the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) (IPBES, 2018). Accurate monitoring and evaluation are required to prioritize the allocation of resources, and to timely adjust restoration management (Buckingham et al., 2019). There is increasing research interest on how to assess restoration effectiveness in the field of restoration ecology (Ruiz-Jaen and Mitchell Aide, 2005; Wortley et al., 2013), which is particularly challenging at large scales (Aronson et al., 2020; Lindenmayer, 2020; Ockendon et al., 2018; Von Holle et al., 2020). Studies found that the effectiveness of restoration interventions on the supply of ecosystem services can vary even within a restoration site, especially in large heterogeneous landscapes (del Río-Mena et al., 2021) and can also change over time (del Río-Mena et al., 2020a; Willemen, 2020).

Several monitoring and evaluation frameworks and guidelines have been developed to guide restoration efforts; they are constituted of iterative steps that allow for adjustments and improvements to the original restoration intervention design (e.g. Machmer and Steeger, 2002; Muhar et al., 2018; NASEM, 2017; Nilsson et al., 2016; Pandit et al., 2018; Prach et al., 2019; Reed et al., 2011). The typical stages in these frameworks are: collecting baseline information on relevant indicators, restoration implementation, documentation of change (monitoring), analysis (evaluation), and reporting results (Gann et al., 2019). A literature review by Wortley et al. (2013) showed that 74% of the restoration evaluations that they reviewed included some form of reference or control site for comparison. From the remaining 26% that did not use a site for comparison, 68% (18 % of the total) monitored only the restoration site over-time without evaluating change against a control site. To explore the extent of a restoration intervention effect, information should best be collected before and after intervention on both control and restored sites (Gann et al., 2019). The Before-After-Control-Impact (BACI) design helps to separate the intervention effect from pre-existing differences between restored and non-restored sites, especially when multiple control sites are selected (Underwood, 1994). Because ecological and land surface changes may occur irrespective of whether an intervention took place, the use of several years for the comparison between control and restored areas in the BACI analysis can prevent the incorrect attribution of changes to restoration effects (Underwood, 1992).

Technological developments provide new opportunities for planning, managing, and monitoring restoration projects (de Almeida et al., 2020). Satellite RS has great potential to provide essential input for monitoring ecosystem services by offering freely-available, repeatable, standardized, and verifiable imagery (Cord et al., 2017; Pettorelli et al., 2018). This imagery can be used as an input to assess long-term trends and variability of multiple biophysical indicators (Pettorelli et al., 2014) for large and often remote areas at increasingly detailed spatial, temporal, and spectral resolutions (Tewkesbury et al., 2015). Models based on a combination of RS, field measurements, and other georeferenced variables (such as slope) can produce accurate estimates of ecosystem services supply for large areas (del Río-Mena et al., 2020b; Martínez-Harms et al., 2016).

Multispectral imaging from Landsat provides a relevant source of information to monitor long-term effects of restoration interventions, because the Landsat mission has provided 30 m resolution imagery since the 1980 s. Both Landsat and the more recent constellation of Sentinel-2 satellites, which acquires data down to 10 m spatial resolution and reduced global revisit times to five days, offer freely available images with short revisit times. These two satellites constellations offer a powerful tool to address large and landscapes (de Almeida et al., 2020). For example, Sacande et al., (2021) used a combination of Landsat and Sentinel-2 images to track changes of biomass in large arid to semiarid landscapes. A rich time series of multispectral observations at 10 to 30 m spatial resolution can be obtained by integrating these datasets, from which vegetation indices and other vegetation metrics can be estimated (de Almeida et al., 2020). Present and future attempts to extend and align images from Landsat and Sentinel-2 will deliver a new opportunity to monitor the Earth's surface at low cost to restoration practitioners for high temporal, medium spatial resolution monitoring (Claverie et al., 2018a). The images can be accessed and processed using open-source platforms, such as for example Google Earth Engine (GEE), which offers, among others, the full Landsat and Sentinel-2 archive (Gorelick et al., 2017). Although RS offers great potential to support the evaluation of ecological restoration actions, there is still a need to assess how this can best be done in way that leads to accurate and robust outcomes (Camarretta et al., 2020; de Almeida et al., 2020). To promote and expand the effective use of RS in monitoring and evaluation strategies of restoration initiatives, standardized guidelines are urgently needed that describe how RS should best be used to account for the complexity of interconnections between land management and biophysical changes over time and space.

Regardless of the method chosen to monitor and evaluate restoration interventions using satellite images, several RS related choices need to be made throughout the monitoring process related to the selection of the a) number and b) distribution of controls per impact site; c) intra-annual image selection; d) reference periods; and e) RS sensor type(s). To date, no studies exist that show how such choices may affect the evaluation of the effectiveness of a restoration intervention on ecosystem services. In this study, we demonstrate the impact of such choices using multiple restoration interventions that took place in Baviaanskloof, South Africa, during the past 30 years.

2. Materials and methods

2.1. Study area, assessed interventions and estimation of ecosystem services

To explore the implications of RS data and method choices on the evaluation of restoration interventions, we used the subtropical arid thickets and shrublands located in the central and eastern area of the Baviaanskloof Hartland Bawarea Conservancy, Eastern Cape in South Africa (Fig. 1). In this subtropical thicket biome, spekboom (*Portulacaria afra*) is one of the dominant and highly palatable species for wildlife and livestock (Vlok et al., 2003). The area is mainly composed of large private farmlands (between 500 and 7,600 ha in size) that have been mainly used for agriculture and tourism (Crane, 2006; Petz et al., 2014). This hilly area has been heavily degraded by unsustainable pastoralism, resulting in loss of vegetation that provides crucial ecosystem services in the area like forage provision and protection against soil erosion. In the valley-bottom, local communities, who represent the majority of the population of the area, share communal land (Petz et al., 2014). Inter- and intra-annual rainfall has been erratic, averaging 327 mm/year in the period from 1990 to 2018, ranging from 150 to 513 mm in the driest (2016) and wettest (1996) year respectively (WRC, 2018). Temperatures frequently reach 40 °C between December and February, while between June and August they can fall below 0 °C (Van Luijk et al., 2013).

A number of restoration and rehabilitation interventions have been implemented in an attempt to reverse severe degradation. These include revegetation, livestock exclusion, and a combination of both. For the illustration of long-term trajectories of ecosystem services (Section 2.2.2.) we focused on the revegetation intervention (red areas in Fig. 1), which took place between 2010 and 2015. During that period, 1,100 ha were planted with spekboom to help the rehabilitation of degraded thicket vegetation. For the evaluation of RS choices (Section 2.2.3) we only assessed restored areas that were planted in 2012.

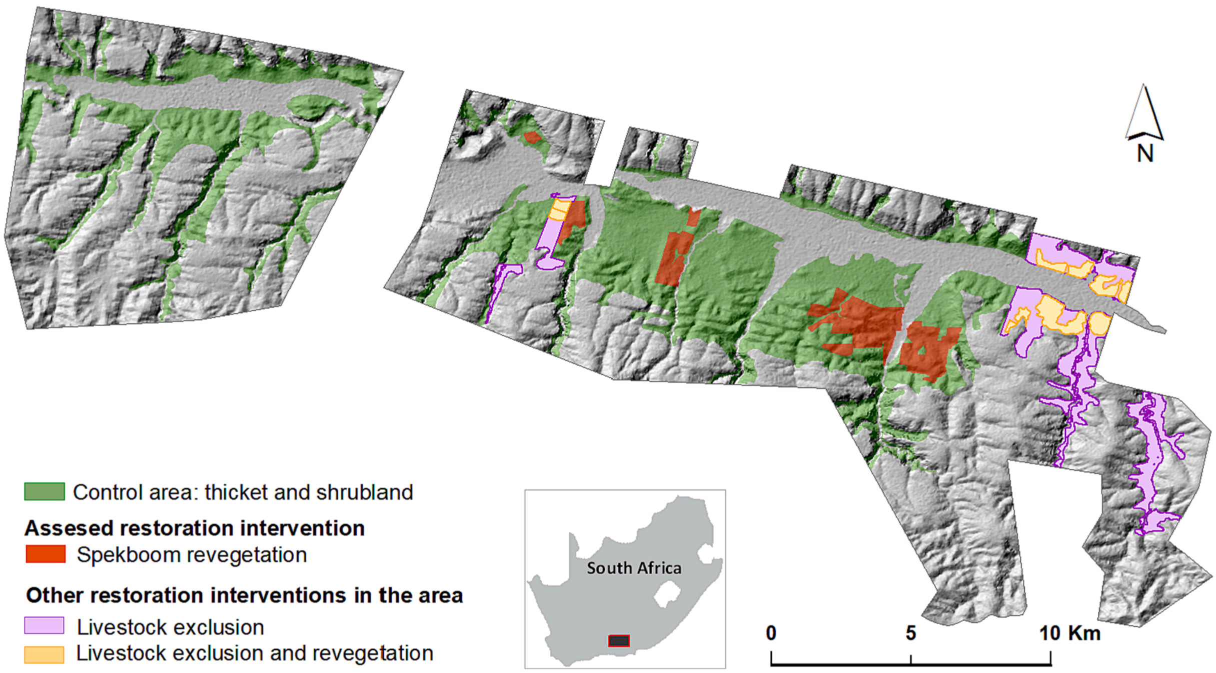

Fig. 1. Restoration intervention sites in the in the Baviaanskloof Hartland Bawarea Conservancy study area in South Africa. Shading indicates topographic relief.

These restoration interventions aimed to enhance five different ecosystem services in the area (Supplementary Materials, Table S1) (del Río-Mena et al., 2020b). We selected the two ecosystem services that could be quantified most accurately with satellite images for the detailed evaluation of RS choices: erosion prevention and forage provision for wild animals. We estimated these ecosystem services in the field using measurable indicators in 30 sample plots across the study area between May and July 2017. The erosion prevention service was expressed using the proxy of percentage of stratified vegetation cover (Str.VC) (Zhongming et al., 2010) and was calculated using field-measured vegetation cover of different vegetation strata (del Río-Mena et al., 2020b). To estimate forage provision, allometric equations (Flombaum and Sala, 2007) were used to estimate green biomass based on field measurements of canopy dimensions and canopy cover for grasses and shrubs.

Based on the field estimations of forage provision and erosion prevention, we previously selected RS based models for Sentinel-2 (del Río-Mena et al., 2020a,b) and Landsat images (del Río-Mena et al., 2021) showing the relation between field measurements and RS information (Table 1) (Data description Section 2.2.4). Using these RS models, we estimated the temporal variability in ecosystem service supply levels for areas with different vegetation densities within an intervention site.

The selection of the best models was based on the lowest Akaike Information Criterion (AIC) score (Table 2). The Landsat model for forage provision (R^2 of 0.71) was used to explore the effect of the number and distribution of controls, reference periods and intra-annual image selection. We did not use the Sentinel-2 model for forage provision since there were no images available for the reference periods (1989 – 1990; 2009 – 2011, Section 2.2.3) We used erosion prevention models to compare the Landsat-8 Operational Land Imager (OLI) and Sentinel-2 Multi Spectral Instrument (MSI) sensors. Erosion prevention was explained by the Landsat-8 and Sentinel-2 models with an R^2 of 0.85 and 0.81 respectively.

Table 1
Selected spectral indices and their equations using Landsat8 (LB) and Sentinel-2 (SB) bands.

Index	Index formula
Inverted Red-Edge Chlorophyll Index (IRECI)	(SB7 – SB4) / (SB5 + SB6)
Normalized Difference index (NDI45)	(SB5 – SB4) / (SB5 + SB4)
Bare Soil Index (BSI)	(LB6 + LB4) – (LB5 + LB2) / (LB6 + LB4) + (LB5 + LB2)
Normalized Burned Ratio (NBR)*	(LB5 – LB7) / (LB5 + LB7)

* NBR is sometimes named differently and has been applied for other purposes than detecting burned areas, i.e. Infra-Red 227 Index, Normalized Difference Infrared Index and Shortwave Vegetation Index (Ji et al., 2011).

Table 2
Selected ecosystem services models based on indices derived from Landsat-8, Sentinel-2 data and terrain variables. Str.VC: Stratified vegetation cover; GB: Green biomass; NRMSE: Normalized root-mean-square, i.e., RMSE divided by the mean.

Ecosystem service	Satellite	Function	R^2	NRMSE	df
Erosion prevention	Landsat-8	Str.VC (%) = 56.36 (BSI)^2 —36.66(BSI) + 6.06	0.85	0.19	30
	Sentinel-2	Str.VC (%) = 27.35 (IRECI) − 1.08	0.81	0.07	30
Forage provision	Landsat-8	GB (kg m-^{2}) = 47.62 (NBR)2 + 55.55(NBR) + 8.71	0.71	1.19	28

Source: (del Río-Mena et al., 2020b; del Río-Mena et al., 2021).

2.2. Remote sensing choices for monitoring and evaluation

2.2.1. Selection of monitoring and evaluation framework

To identify which and when RS related decisions need to be made within a monitoring and evaluation program, we selected the monitoring method developed by Herrick et al. (2006). This stepwise framework is helpful to identify where decisions related to RS need to be made in the monitoring and evaluation process. We selected this framework after reviewing available guidelines and approaches for restoration monitoring and evaluation. We found that the guidelines provided by organizations such as Society for Ecological Restoration (SER) (Gann et al., 2019); United Nations Convention to Combat Desertification (UNCCD) (Cowie et al., 2018); Intergovernmental

Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) (Kohler et al., 2018; Prince et al., 2018); and the Food and Agriculture Organization of the United Nations (FAO) (FAO, 2015) describe key monitoring and evaluation components with comparable steps. These included establishing sampling design, defining baselines, selecting and quantifying indicators, planning data analysis, documenting and archiving collected pre- and post-treatment data, verification and interpretation, communicating results, and adapting management strategies.

The monitoring framework by Herrick et al. (2006) describes three main phases (monitoring program development, short-term and long-term monitoring) and these phases have steps that are iteratively connected (Fig. 2). The framework highlights multiple choices that need to be made when setting up a monitoring program, irrespective of the monitoring technique used. We focus on RS-related choices, but many of the considerations are not restricted to RS only. In Fig. 2, the original steps of the framework are shown in grey boxes, to which we added RS and other spatial data (blue boxes), field data (green boxes), and intervention (yellow boxes) components. Roman numerals represent the following monitoring steps: i) the sample size and distribution of

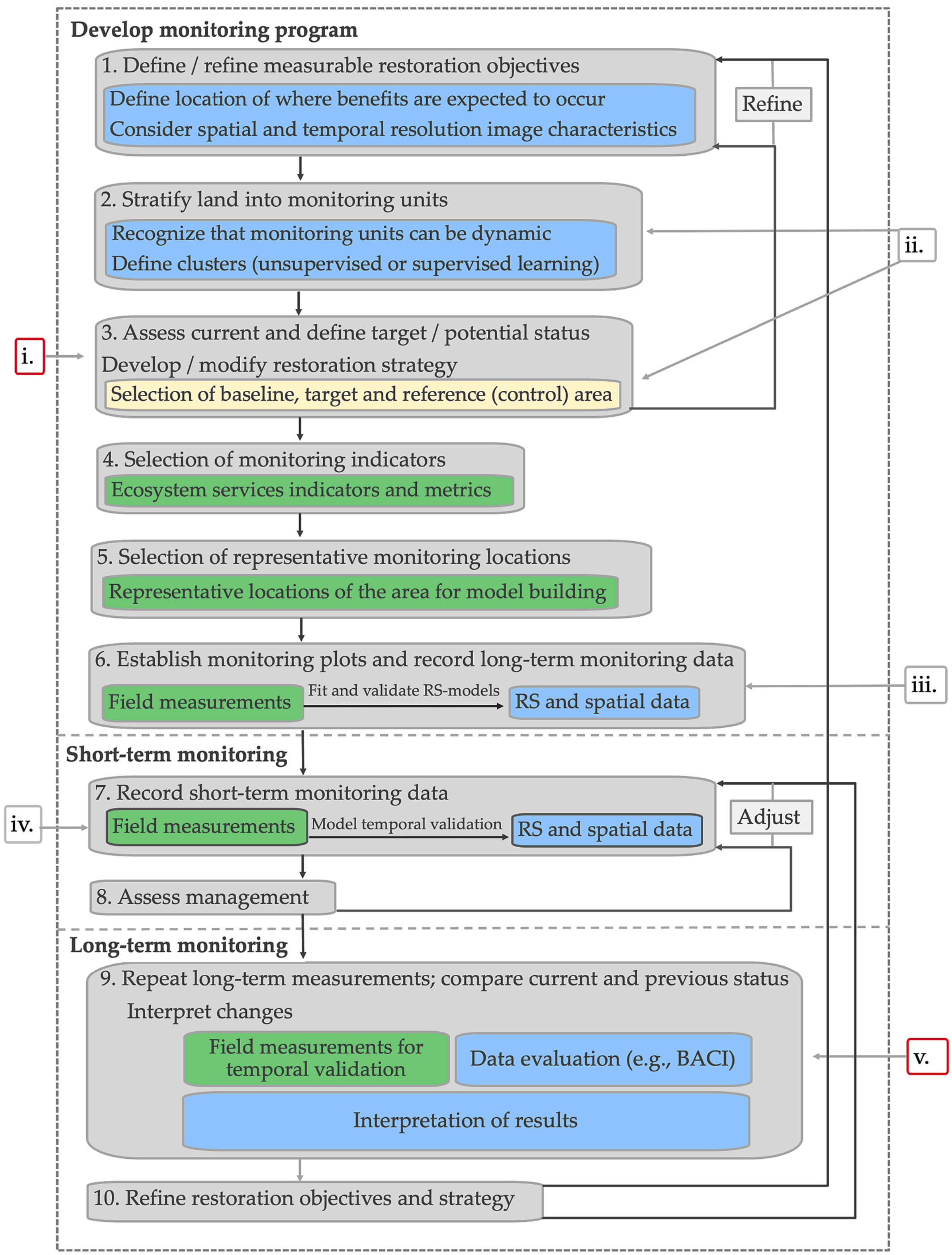

Fig. 2. Monitoring framework of restoration interventions using RS. Adapted from Herrick et al., (2006). Grey boxes indicate the ten original framework steps, blue indicates RS and spatial data, green indicates field data, and yellow relates to the intervention itself. Roman numerals represent the steps linked to RS related key choices within a monitoring program. Roman numerals in red boxes indicate the steps related to the key choices analyzed in this study. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

controls; ii) selection of comparable sites; iii) fitting field-based estimations of ecosystem services to RS models; and the selection of the best RS models; iv) the long-term variations of the relationships between field-based estimations of ecosystem services with RS models; and v) the selection of the reference period and image source. In this paper, we explore the impact of five key choices related to monitoring steps i and v (red boxes in Fig. 2): a) the number of controls and b) distribution of controls; c) intra-annual image selection and d) selection of the reference period, and e) image source (Landsat-8 or Sentinel-2) (Data description Section 2.2.4) on their effect on the evaluation outcomes. Choices a to d were assessed using the BACI analyses of forage availability, while choice e was explored by estimating the supply of erosion prevention (details in Section 2.2.3).

2.2.2. Illustration of long-term trajectories of ecosystem services

To account for differences in the initial vegetation and degradation state of the area, we identified clusters having similar ecosystem service levels before any intervention took place in the study area. We used the selected spectral vegetation indices from our previously developed ecosystem service models for Landsat images (Table 2). The clusters were obtained using an ISODATA unsupervised classification based on the time trajectory of the spectral index that represents the ecosystem service (De Oto et al., 2019); similar trajectories were grouped in a single cluster. The clustering was performed using 16 available cloud-free Landsat-5 Thematic Mapper (TM) images from 26/02/1989 to 27/10/1990 (Data description Section 2.2.4 for details). We refer to these reference clusters as 'vegetation clusters' in the remainder of this article. Based on the resulting clusters, we estimated the supply of erosion prevention and forage provision in revegetated areas between the years 1989 and 2020 to illustrate the variations of the supply of ecosystem services throughout the studied period.

2.2.3. Evaluating choices when using RS to monitor landscape restoration

We evaluated the effect of a set of RS choices (detailed below, letters a to e) on the outcome of the evaluation of revegetation. As in Meroni et al. (2017), in this study we evaluated the effect of ecological restoration by focusing on the differential change between impact and control sites compared before and after the intervention occurred. For choices a to d, we applied the BACI analysis using ecosystem service of forage provision as an example. The BACI contrast shows the differential change between impact and control pixels compared before and after the intervention (Eq. (1)). In addition, the BACI analysis provides the significance level of the BACI effect test. In this study only the resulting significant BACI contrast (p-value < 0.05) were considered.

BACI contrast $= (\mu CA - \mu CB) - (\mu IA - \mu IB)$ (1).

Where μ is the temporal (selected years) and spatial (controls) mean of the variables selected to represent the impact; the letters C and I stand for control and impact, respectively; and the letters B and A stand for the periods 'before' and 'after', respectively. A negative contrast indicates that the variable has increased more in the impact site with respect to the control sites during the time period ranging from before to after the implementation of the restoration project. Such as in this study, the above is valid when the relationship between the predictor variable (e.g. NBR index) and the ecosystem service (e.g. forage provision) is positive. The BACI contrast is expressed in the same units of the variable of interest, i.e. the spectral index NBR for forage provision, and consequently is unitless in our case. In this study we refer to the 'before' period as the reference period. There are different forms of BACI analysis including trend analyses (Wauchope et al., 2021). In this study we did not assume that a clear trend occurred in the before or after period. Therefore, we used the average of the periods before (details below, choice e) to estimate the BACI contrast for each year of the period after (del Río-Mena et al., 2021).

To calculate the BACI contrast, we selected 15 illustrative impact sites. Each site comprised four neighboring Landsat pixels. To select the 15 impact sites, we used a stratified randomization from areas that were revegetated within the same year (2012), with five impact sites in each of the most prevalent vegetation clusters (Clusters 3, 4 and 5). Each impact site was composed by four impact pixels. We took the average of the BACI contrast of four pixels to obtain more representative values of the area by reducing the effect of the spatial variability of BACI results within the same cluster, as previously observed in (del Río-Mena et al., 2021). The control pixels were selected from the non-restored sites belonging to the same vegetation cluster as the impact sites they were compared to. For each impact pixel, we calculated the BACI contrast for every year between 2012 and 2020 for forage provision to test RS choices a to d. We estimated the supply of erosion prevention to test RS choice e. Below we clarify in detail these six RS choices:

a**) Different number of controls** (related to monitoring step i, Fig. 2). The use of multiple controls minimizes the chances that the outcome of the BACI analysis is driven by the selection of one control (Meroni et al., 2017). Yet, the selection of independent control pixels for each impact pixel should also consider the total number of impact pixels, especially when the available control area is limited. Here, we compared the outcome of the BACI contrast between using the average of 20 control pixels with the average of 100 control pixels for each impact pixel within each restored site. For this comparison we used the minimum annual values and same set of control pixels for each of the four impact-pixels.

b**) Different distribution of control sites** (related to key choice i, Fig. 2). The provision of ecosystem services could vary even within the same vegetation cluster, especially for heterogeneous landscapes. Therefore, the result of the evaluation of an impact site could change when selecting a different group of control sites. To assess the effect of the selection of control site groups, we assessed the outcome of the BACI contrast when using the same set of 20 random control pixels, compared to using different sets of 20 control pixels for each impact pixel of every impact site. This evaluation was repeated for each vegetation cluster separately. For this comparison we used the minimum annual values.

c) Intra-annual image selection (related to key choice v, Fig. 2). To assess the effect of restoration evaluations during the estimated peak and nadir of forage supply, we compared the results of the BACI analyses between using the annual minimum or maximum annual supply of forage provision (determined by the NBR). By taking the maximum and minimum, we aim to describe the possible interannual range in ecosystem service provision. Although preferable, the selection of a multiple time observation of maximum and minimum values was not possible due to the low number of cloud-free available Landsat images for several years. For this RS choice we used 20 different control pixels for each impact pixel of each impact site. We did not consider filling missing values since the Landsat image availability from years before 2013 presented long periods of separation.

d) Combination of different distribution of control sites and intra-annual image selection (combination of RS choices b and c). This choice aimed to assess differences in the outcome of restoration evaluation when using the same or different distribution of controls together with different intra-annual image selection.

e**) Different reference periods** (key choice v, Fig. 2). For this RS choice we selected two different 'before' periods to explore differences on the evaluation outcome. This selection was arbitrary in order to compare the results between a reference period from 22 to 23 years prior to any intervention begun in the study area (1989–1990), and a short period immediately before the restoration intervention (2009–2011). We calculated the BACI contrast using input Landsat images from these two reference periods to calculate the corresponding ecosystem service spectral index (NBR for forage provision). We defined the period 'after' as every independent year after the ecological restoration took place (years 2012 to 2020). Each of the previous RS choices (a to d) was repeated for these two reference periods. Therefore, we compared this choice using different number and distribution of controls, and intra-annual image selection. The reference period refers to the selection of a temporal span that represents the initial conditions for comparison between impact and control sites, commonly known in BACI analyses as the 'before period'. This reference period does not necessarily relate to a reference ecosystem model (Gann et al., 2019), goal or condition (Science Task Force for the UN Decade on Ecosystem Restoration, 2021).

f) **Different sensors** as a last test we compared the RS derived ecosystem service of erosion prevention from 2015 to 2020 derived from Landsat-8 versus Sentinel-2 images (related to key choice v, Fig. 2). Because images of Sentinel-2 are only available since 2015, we did not perform a BACI analysis but compared the actual provision of erosion prevention for each year based on the pre-defined models. To compare evaluation results between these two satellites sensors we estimated the annual maximum and minimum erosion prevention using their respective equations (Table 2). For the Landsat image, we selected three of the previously analyzed restoration sites from vegetation Cluster 4 of erosion prevention. For each Landsat pixel, we selected four 10 m-resolution Sentinel-2 pixels that fit entirely within each selected Landsat pixel, i.e., we selected 16 Sentinel-2 pixels per restoration site.

To explore the impact of RS choices on the potential conclusions of BACI restoration evaluations, the resulting sign of the BACI contrasts using ten different combinations of the RS choices described above were also compared.

2.2.4. Data description

Different sets of satellite images were used for each step illustrated in the monitoring and evaluation framework in Fig. 2 (Table 3). The number of available Landsat images is detailed in Tables S4 and S5. In addition, we used the slope (degrees) extracted from a 12.5 m resolution DEM derived from ALOS PALSAR (Geophysical Institute of the University of Alaska Fairbanks, 2018), and a spatial dataset of the intervention sites for revegetation, including planting dates. Images from Landsat-5, 7 and 8 were acquired from path 172 and row 83, accessed and selected through Google Earth Engine. We used the Landsat Level-2 Surface Reflectance Science Product, courtesy of the U.S. Geological Survey (USGS, 2020), derived from the Landsat Collection 1 Tier 1 dataset. None of the selected images had cloud cover for the revegetated areas and small clouds were masked out for control areas. Pixels falling within the Landsat-7 Enhanced Thematic Mapper Plus (ETM +) Scan Line Corrector (SLC) off data (missing pixels due to satellite instrument malfunctioning) were excluded from calculations for that specific moment. The BACI contrast was not calculated if an impact or control pixel had missing data resulting either from cloud masking or SLC off data. The number of these 'no data' values was documented per analysis.

For the comparison between the estimated erosion prevention using Landsat-8 and Sentinel-2 as input images, we also used Google Earth Engine to select the dates of the spatial average in the analyzed area of the annual minimum and maximum values of ecosystem services supply. The selection of the relevant dates was based on the vegetation index values of Sentinel-2 (IRECI, see Table 2). The atmospherically-corrected Level-2a surface reflectance datasets were not readily available for the entire assessed period (2015–2020) on Google Earth Engine. To ensure a consistent processing, we therefore downloaded top-of-atmosphere Level-1c data (Copernicus, 2018) and used the ESA Sen2cor processor, available in the Sentinel Application Station (SNAP) version 8.0, to generate Level-2a images for the Sentinel-2 imagery (ESA, 2018). Prior to calculating a RS index, we used the 'super-resolving enhancement' method to resample the 20 m Sentinel-2 bands to 10 m (Brodu, 2018). For each retained image, we then extracted the relevant spectral indices. The dates of the selected images are listed in Supplementary Materials, Tables S2 and S3.

Table 3
Satellite images used in different steps of the monitoring and evaluation framework (Fig. 2).

Framework step	Satellite image description	RS variables
2. Monitoring units (vegetation clusters using ISODATA)	Landsat-5 (26/02/1989 to 27/10/1990)	Time series of spectral indices
6. Fit RS models	Landsat-8 from 14/05/2017 Sentinel-2A image from 24/06/2017	Spectral indices
9. Compare current and previous status	Landsat-5, 7 and 8 images periods 1989–1990; 2009–2020 Sentinel-2A images, period 2015–2020	Spectral index values: annual minimum and maximum

3. Results

3.1. Vegetation clusters and temporal trajectory of ecosystem services

The ISODATA classification of the satellite vegetation indices (BSI and NBR) acquired in the period before any intervention occurred, showed the distribution of five thicket vegetation clusters for a) forage provision and b) erosion prevention (Fig. 3). The cluster numbers were assigned based on their related level of ecosystem services supply over time, where Cluster 1 represents locations with high supply and Cluster 5 shows locations with low supply. The distribution of clusters of forage provision and erosion prevention showed a similar pattern. The smallest proportion of the area corresponded to Cluster 1 (4% for forage provision and 8% for erosion prevention). Most of the assessed land belonged to Clusters 3, 4 and 5 (79% for forage provision and 75% erosion prevention). These clusters were also predominant in the areas where revegetation interventions were carried out, which are depicted as black polygons in Fig. 3.

The temporal ecosystem service supply trajectories during the period 1989–2020 for forage provision (a) and erosion prevention (b) were based on the annual minimum and maximum of the RS-derived indicator (Fig. 4). Both graphs show the trajectory of the five vegetation clusters for the maximum annual values (light blue), and five clusters using the minimum annual values (light orange). Cluster 1 lines show the overall highest values of ecosystem service supply, while Cluster 5 presented the lowest overall values. The thick lines represent the average supply of all clusters. For the whole period, the restored area presented an average maximum annual forage provision ranging between 16.7 kg m^{-2} (2017) and 52.5 kg m^{-2} (1997) kg m^{-2} (SD = 7.1), and an average minimum between 4.8 kg m^{-2} in 2016 and 31.3 kg m^{-2} (1997) (SD = 4.8) of green biomass. The average maximum erosion prevention ranged from 2.43% (1991) and 8.51% (1997) (SD = 1.6) of stratified vegetation cover for the revegetated areas, whereas the average minimum values ranged from 0.43% (2016) to 5.97% (1997) (SD = 1). For both ecosystem services, the variability of the annual values was the smallest when using the minimum values and in clusters with less ecosystem service supply (clusters 4 and 5).

For forage provision and erosion prevention, the minimum values for 1997 and 2006 were particularly high and may not be representative of the real annual minimum value, but rather an artefact caused by the limited availability of cloud-free images for those years. Both years had six available images (on average 14 images per year for the period 1989–2020, SD = 9). Because the high values are likely an artefact of the limited amount of cloud-free data, these years were not considered for the BACI calculation when evaluating RS choices in Section 4.2.

3.2. Effect of RS choices on the evaluation outcomes of the revegetation intervention

The average trajectory of the BACI contrast (representing forage provision assessed after revegetation) of five impact sites in the most prevalent clusters in the intervention areas, was similar for clusters 3 and 4 for the evaluated RS choices (Fig. 5). Each graph shows the different RS analysis choices related to space and time (choices a to e in

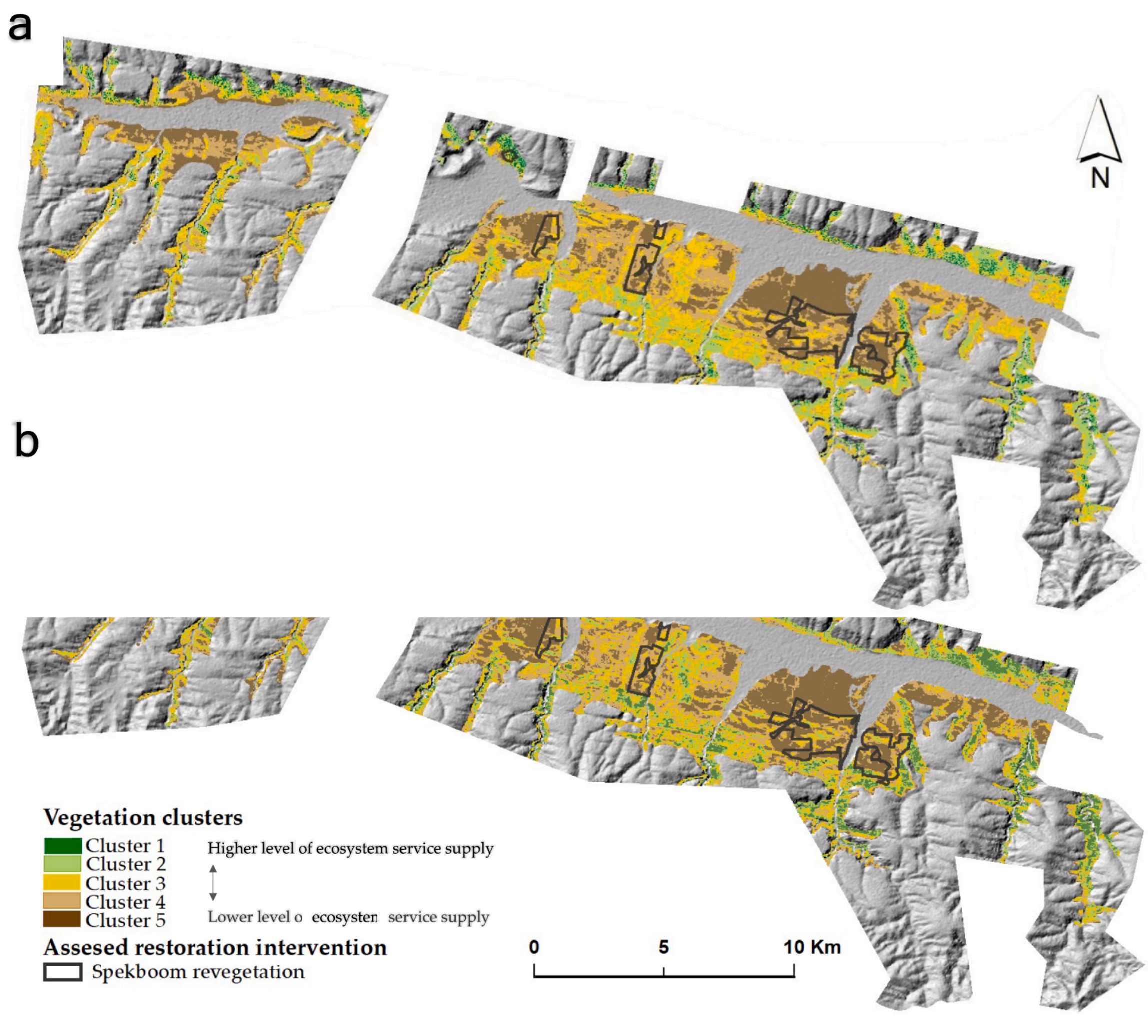

Fig. 3. Distribution of vegetation clusters from ISODATA classification for (a) forage provision and (b) erosion prevention.

Section 2.2.3) for two different reference periods (choice e). For each year, the average of all resulting BACI contrasts (originated from five impact sites per cluster, and each impact site composed of four Landsat-8 pixels), is shown together with their standard deviation. However, since the pixels within the SLC off data from Landsat-7 ETM + were excluded (see Section 2.2.4), the total number of impact pixels ranged from 10 to 20. Note that according to the BACI formula the negative values correspond to a positive effect of the restoration compared to the changes occurring in their respective control sites. The BACI contrasts for each impact site are presented in the Supplementary Materials (Fig. S1-S3).

Although yearly BACI contrasts varied based on the different RS choices, the average differences are more evident in clusters with the lowest ecosystem service supply, clusters 4 and 5 (Fig. 5b and 5c). In these clusters, we found a large difference in BACI results depending on the moment of the year that was used; the choice of the maximum annual values produced a relatively constant BACI contrast throughout the analyzed years, which denotes none or little effect of the intervention (BACI contrast close to zero). On the other hand, the use of the minimum annual values resulted in fluctuating results with 56% of the assessed years showing a negative BACI contrast (suggesting positive restoration effect) in Clusters 4 and 5. The presented results correspond to significant values (p-values < 0.05). Details on data composition of the resulting BACI significance and no-data values are presented in the Supplementary Materials (Tables S6-S8 and Fig. S4).

When comparing RS choices, we found that they often result in opposite BACI signs, suggesting different intervention evaluation conclusions. Table 4 shows the percentage of opposite BACI signs after contrasting ten combinations of RS choices described in the methods (Section 2.2.3), and the direction of the BACI sign change. Across all clusters, the largest discrepancies in BACI contrast result when using intra-annual image selection for comparison (maximum or minimum annual values, RS choices d and e), while the sample size resulted in few or no contrasting BACI results. The percentage of opposite BACI results for different intra-annual image selection ranged from 38 to 50% depending on the cluster and distribution of controls. Different reference periods resulted in a higher percentage of opposite BACI signs when using the maximum annual values, ranging from 29 to 38%. The use of maximum intra-annual values also increased the number non-significant BACI results in Cluster 4 and 5. Regarding the direction of the BACI sign change, most positive to negative BACI contrast sign changes (i.e. from negative to positive restoration effect) occurred when shifting from different to the same set of 20 controls (maximum annual values). Cluster 5 showed a consistent change of positive to negative BACI contrast when using the second reference (2009 – 2011) period instead of the first one (1989—–1990). The other comparisons did not show a clear pattern in relation to the direction of the change.

Regarding the effect of different satellite sensors (Section 2.2.3, RS choice e) we found that the trajectories of annual maximum and minimum values of erosion prevention (i.e., stratified vegetation cover) using Landsat-8 and Sentinel-2 and their respective ecosystem services models presented a similar overall shape (Fig. 6). However, the estimations of erosion prevention derived from the Sentinel-2-based model

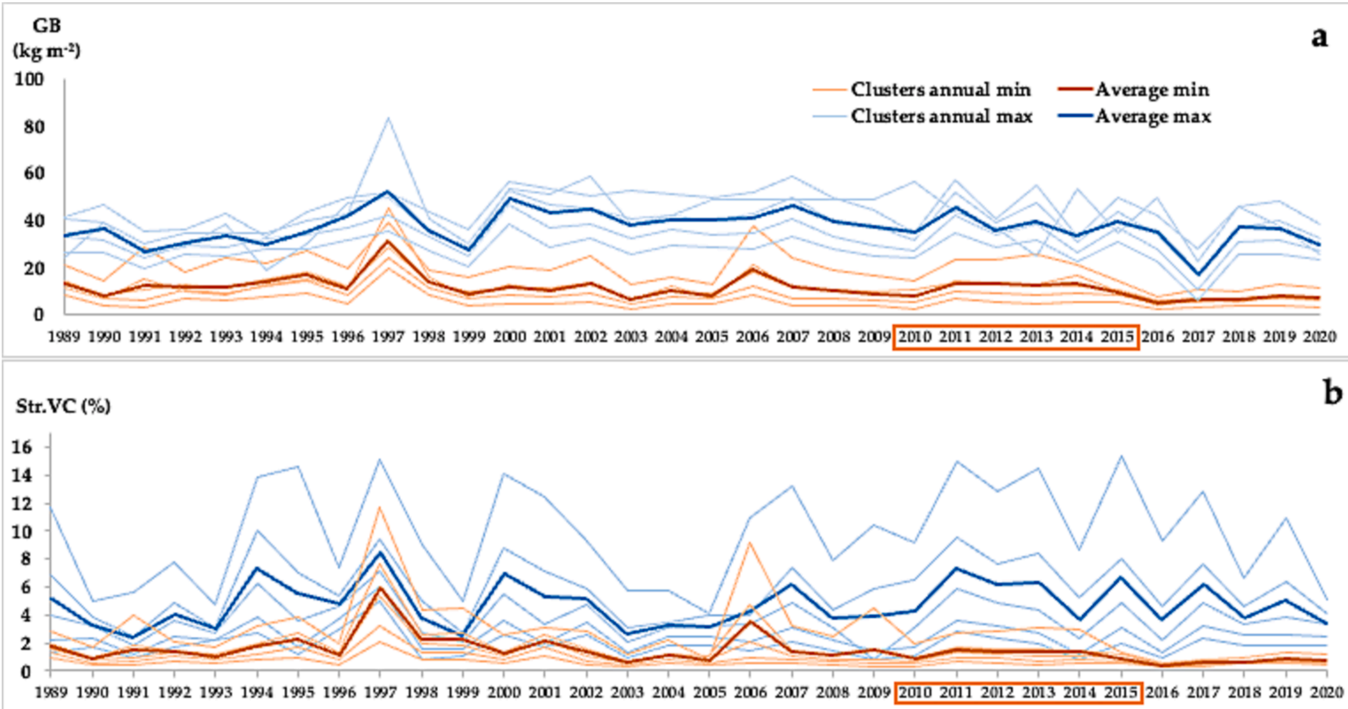

Fig. 4. Long-term trajectory of forage provision (a) and erosion prevention (b) for clusters in restored areas using the maximum and minimum annual ecosystem services values. GB: green biomass. Str.VC: Stratified vegetation cover. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

showed higher values than for Landsat-8. The 48 analyzed 10 × 10 m pixels from Sentinel-2 capture more variability in the area compared to the 12 pixels of 30 × 30 from Landsat-8.

4. Discussion

In this study, we systematically assessed the effect of RS choices on the monitoring and evaluation of restoration interventions based on changes in ecosystem services. Applying an existing restoration monitoring framework (Herrick et al., 2006)allowed us to summarize the most pertinent choices that need to be made, which are translated into RS choices for monitoring and evaluating restoration interventions. RS captured the temporal variation of restoration and the spatial differences even between pixels belonging to the same restoration site. We found that RS choices affected the resulting BACI results to varying degrees, potentially leading to different conclusions about the effectiveness of the restoration intervention. These different BACI results were particularly evident when analyzing the RS index at different moments during the year (minimum or maximum annual values), and secondly when using different reference periods. These results reflect the difference in intra- and inter-annual ecosystem service supply variation between control and impact sites. Understanding these variations may guide the selection of images and control sample selection in the BACI analyses, and helps to interpret the evaluation results. These methodological RS choices and their justifications must be included in restoration evaluation reports to ensure accurate claims of ecological restoration effects.

We found that restoration effects do not necessarily point towards success or failure, as this depends also on the time periods analyzed before and after the intervention. In this study, the compared ‘before’ reference periods were 20 years apart. Although BACI accounts for temporal variability caused by weather conditions (Underwood, 1994), local disturbances in the landscape (e.g. caused by pests, fires) could affect the relation between an impact site and its controls. These temporal changes could then be interpreted as a decrease (or an increase) in vulnerability to such disturbances caused by the intervention (Meroni et al., 2017). Therefore, to minimize the effect of short temporal disturbances, we recommend considering average conditions over a longer multi-year timeframe. Conversely, specific before and after periods could be useful for monitoring the response of restored sites to these disturbances by comparing restored and not-restored sites before and after a local disturbance (e.g. pest pressure).

The number of positive or negative significant BACI results does not indicate whether one choice is better than another. Instead, they provide different information, and one choice could be preferred over another depending on the context and monitoring and evaluation goals. For example, although the BACI contrasts remained similar when using different control sample sizes, the choice of sample size depends on whether the expected changes in the used ecosystem system indicators are substantial or more subtle. Nevertheless, it is crucial to consider and report the number of controls for each impact site when claiming BACI changes, because the use of a larger number of control samples could result in significant values for subtle changes that may otherwise go unnoticed. Using a larger control sample size also reduces the effect of local disturbances on the outcome of the evaluation. In this study, different locations of controls samples (i.e. choice b, Section 2.2.3.) did not significantly impact the evaluation results. However, it is advisable to repeat BACI analyses using different sets of controls to ensure that the identified outcome of the ecological restoration is robust and not a result of a potential (random) poor choice of control sites, for example because all controls are affected by a specific localized disturbance. It is worth noting that the evaluated RS choices in this study belong to a selection within several other decisions that need to be made in the monitoring process (Fig. 2), that could also impact the monitoring and evaluation outcomes. In addition, there are multiple options within the selected choices, such as the length of the reference period, the annual timing and number of images to estimate the control and impact values; or the number and distribution of impact sites.

The identification of vegetation clusters through the ISODATA unsupervised classification helped to identify appropriate control sites

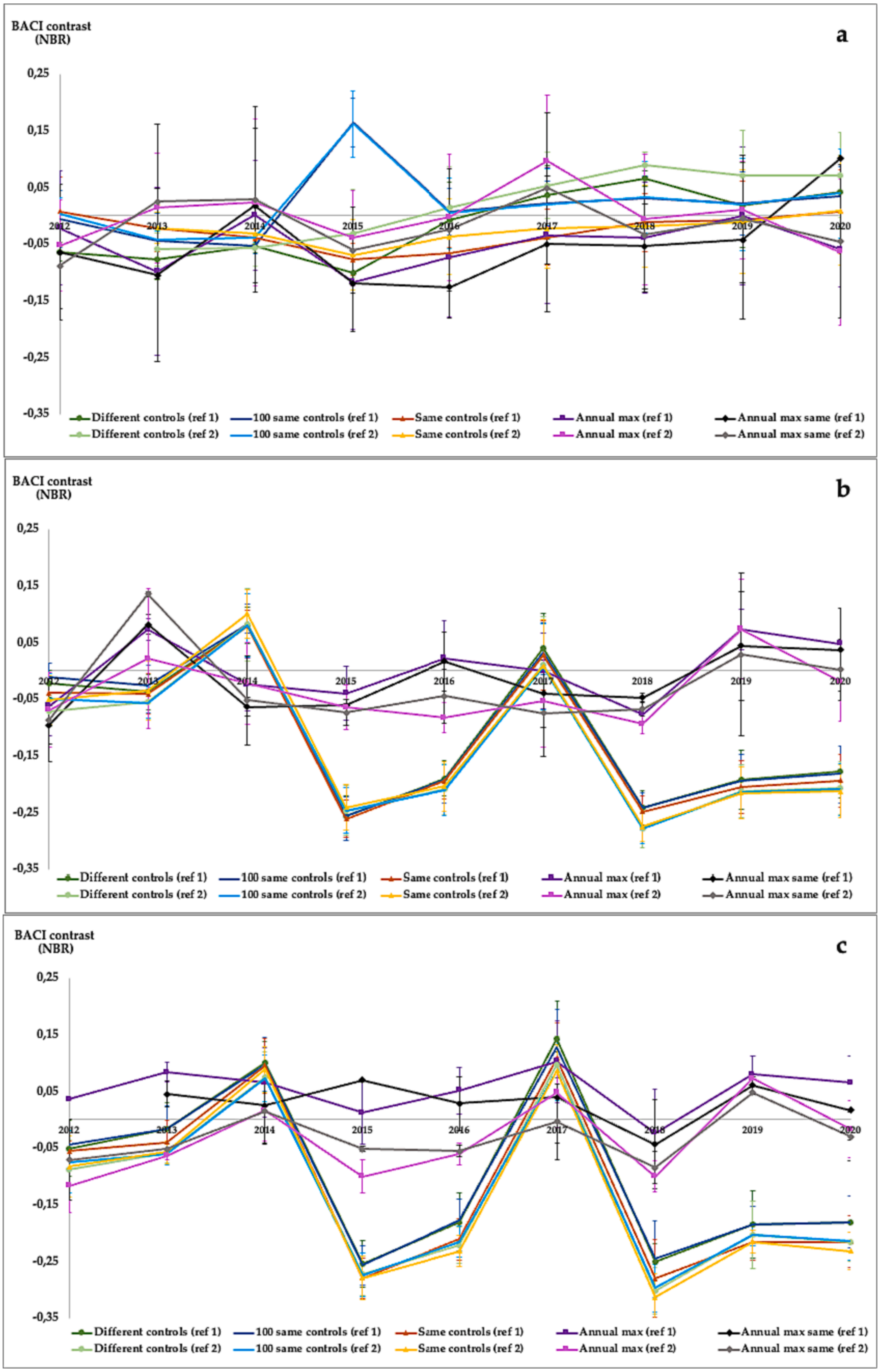

Fig. 5. BACI average contrast trajectory and standard deviation of forage provision for vegetation clusters 3 (a), 4 (b) and 5 (c) in five intervention sites using different RS choices. Each average is calculated using significant BACI contrast values from the four pixels per impact site and five impact sites per cluster. If not specified otherwise, the default RS choice is: annual minimal values and 20 different controls. All values are significant (p-values < 0.05). ref 1: Reference period 1 (1989 – 1990); ref 2: Reference period 2 (2009 – 2011). Different or same controls refers to choice b of Section 2.2.3. (distribution of control sites).

Table 4
Opposite BACI contrasts sign for each impact site throughout the period after (2012 to 2020) resulting from contrasting RS choices according to Section 2.2.3 for most abundant vegetation clusters 3 to 5, representing lowest level of ecosystem services prior to the interventions. 'n' represents the total number of compared impact sites calculated using only significant pixels (p-values < 0.05). Per RS choice we show the number of flips in the sign of the BACI contrast compared to the reference RS choices (on top number of changes to positive contrast value, bottom, the number of changes to negative contrast values). Different or same set of controls refers to choice b of Section 2.2.3. (distribution of control sites).

RS choice comparison	Opposite BACI results					
	Total	Sign flip direction to positive - to negative	Total	Sign flip direction to positive to negative	Total	Sign flip direction to positive to negative
a) From different to same set of 20 controls (minimum annual values)	38	0 4	65	2 0	67	0 1
b) From different to same set of 20 controls (maximum annual values)	49	2 16	45	1 3	39	0 6
c) From 20 to 100 controls (same set of controls, minimum values)	58	11 0	67	0 1	68	1 0
d) From minimum to maximum annual values (different set of 20 controls)	36	8 10	52	13 7	51	17 3
e) From minimum to maximum annual values (same set of 20 controls)	29	3 8	39	8 8	40	12 5
f) From first to second reference period (20 different controls)	18	2 1	34	0 1	32	0 1
g) From first to second reference period (20 same controls)	20	0 3	30	0 2	33	0 1
h) From first to second reference period (100 same controls)	34	3 5	37	0 4	34	0 1
i) From first to second reference period (20 different maximum annual values)	35	9 4	24	1 6	21	0 8
j) From first to second reference period (20 different controls)	21	4 2	18	1 1	15	0 5

based on the initial degradation state. The clustering illustrated the inter-annual fluctuation of ecosystem services supply in the landscape (Fig. 4), and the outcome of the restoration efforts (BACI contrast). In this study, the ISODATA clustering was calculated based on the images from years before any intervention occurred in the area, more than 20 years before the intervention. However, this period could change depending on the context of the restoration intervention and selected reference period. We found that the effect of the different RS choices on the BACI results depended on the evaluated cluster. Understanding the effect of the RS choices on different initial vegetation states can guide localized adaptive management of the restored landscape.

Although Table 4 shows a higher percentage of contrasting BACI results for different RS choices in Cluster 3, these BACI values are close to 0 (Fig. 5a), suggesting that there was little or no restoration effect in these areas. In addition, there was a high number of non-significant BACI results in Cluster 3 (Table S6). However, the BACI values are expressed in the same units as the spectral index selected to represent the ecosystem service variable (which is unitless). If the BACI values are converted to ecosystem supply using the models in Table 2, for Cluster 3 the estimated differences between choices can be of 4.9 kg m^{-2} (different reference period, year 2013) or 8.1 kg m^{-2} (different intra-annual image selection year 2020). Heavily degraded areas (Cluster 4 and 5) showed several years with negative BACI contrast (suggesting positive restoration effects). This result could be explained because the addition of a drought resistant species, such as spekboom, to a degraded area through revegetation would produce a positive change in greenness. Which moment/year is selected for BACI is of critical importance as evaluation outcomes may deviate substantially. This is highlighted in the comparison between the outcomes of the BACI analyses for each of the years 'after' the restoration intervention.

In our example of forage provision, we observed opposing BACI contrasts in up to 58% of the cases (across the evaluated sites and years), demonstrating the importance of reporting the selection of evaluation choices and their justification according to evaluation aims when determining restoration effects on ecosystem services supply. Because the BACI contrast is calculated using the difference (control – impact) of the differences (after – before), the resulting contrast values could be very small. However, a small difference in BACI results could still be significant. Evaluating interventions using the maximum spectral index values extracted from the satellite image time series resulted in a small negative BACI contrast, implying a slight positive change after the restoration intervention. The evaluated RS choices mostly affected the control values, their trend (Fig. S5) and therefore the BACI results. The trends suggest that the level of forage provision for all clusters have decreased overtime. The impact sites generally exhibited gentler slopes in their minimum values compared to the control sites (Fig. S5a to S5f). The steepness of the slope of maximum values (Fig. S5g to S5j) does not show mayor differences between impact and control sites.

Regarding the results of the BACI analyzes (Fig. 5), the percentage

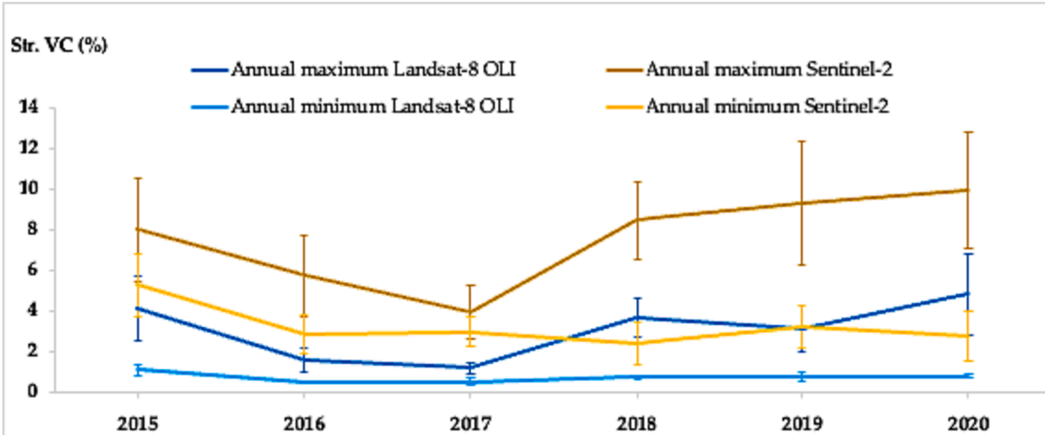

Fig. 6. Average trajectory and standard deviation of annual minimum and maximum values of erosion prevention in restored areas using Landsat-8 (12 pixels in total) and Sentinel-2 (48 pixels in total). Str.VC: Stratified vegetation cover.

BACI contrast suggesting improvements on forage provision after restoration when using the maximum values ranged between 63 (Cluster 3), 59 (Cluster 4) and 48% (Cluster 5). This is likely because the intervention involved revegetation with spekboom trees (which only propagates vegetatively), whereas herbaceous vegetation always grows in the area after rainy periods, regardless of whether the area was revegetated or not, and based on its larger areal extent thus leads to higher maximum values. In addition, the revegetated species, spekboom, is evergreen. Therefore, the effect of the tree vegetation is not apparent at times when there is a lot of green herbaceous vegetation (i.e. maximum values of vegetation indices). In contrast, the most degraded areas (Cluster 4 and 5), between 72 and 88% of the restored sites showed an improvement in forage provision for the moment of the year with lowest supply. A relative larger supply of forage in restored sites by spekboom vegetation (a highly palatable species) provides relief to animals during dry moments with otherwise little forage availability. However, a negative BACI contrast does not necessarily translate into a sufficient provision of the evaluated ecosystem service. It is also important to consider the difference between the actual supply of the ecosystem service and the time of the year when the assessed ecosystem service is mostly needed in the area (e.g. spring and summer months) (del Río-Mena et al., 2020a).

Our comparison of Landsat-8 and Sentinel-2 images to estimate the supply of erosion prevention, showed similar trajectories between both sensors, suggesting that the differences in erosion prevention estimation are consistent in time. The higher values from Sentinel-2 can be attributed to the use of different ecosystem service model and the higher image availability (average of 28 images for Sentinel-2 and 16 images for Landsat-8). From 2018 to 2020 the number of available images from Sentinel-2 were approximately double the number of Landsat-8 images. It is important to note that our estimation of ecosystem services using the sensor-based models provide insights on their supply and spatial–temporal variation to carry out restoration evaluation. However, the models do not intend to give absolute values of ecosystem service provision for one moment in time. The use of more than one sensor such as Landsat-8 OLI and Sentinel-2 MSI could help increase temporal, spatial and spectral information. However, it is important to harmonize the results obtained from each sensor and validate them with ground information. As such, the existing harmonized Landsat and Sentinel-2 dataset could be an important asset to use both sensors interchangeably (Claverie et al., 2018b). While Sentinel-2 offers higher revisit frequency, spatial and different spectral resolution, Landsat images, allows tracking longer-term trajectories that help to assess outcomes of past restoration activities. BACI offers opportunity of time continuity using older Landsat images and recent Sentinel-2 images since it looks for relative temporal differences between impacts and controls.

The monitoring and evaluation framework and the BACI approach presented in this study can be extended to other information technologies that provide quantifiable indicators of ecosystem services. Alternative optical RS sources, such as from other satellite sensors, airborne or unmanned aerial vehicles (UAV) can complement monitoring and evaluation adding different spatial, spectral and temporal information (Reif and Theel, 2017). The use of UAV provides an opportunity to bridge the gap between field observations and traditional air- and space-borne RS (Manfreda et al., 2018). Active sensors such as synthetic aperture radar (SAR) and LiDAR offer the advantages to overcome the cloud cover limitation while providing high resolution data (Nagendra et al., 2013). LiDAR sensors in combination with UAV have been used to capture parameters such as forest structure (Camarretta et al., 2020) or aboveground woody biomass (de Almeida et al., 2019), that could potentially be used for the estimation of other ecosystem services. While LiDAR is one of the best options to monitor canopy structural parameters that can be used as indicators for monitoring ecological restorations, SAR systems have the potential to accurately capture subtle changes in biomass (de Almeida et al., 2020). For many of these RS systems, it can be challenging however to acquire data from both before and after interventions, possibly limiting their applicability as input to a BACI framework. If such data can be obtained or collected from these systems, similar to our paper choices will need to be made in how to best utilize such alternative datasets for restoration evaluation.

This study is a first step towards the definition of guidelines that help to consistently evaluate restoration interventions using RS images that can capture ecosystem service supply. RS choices related to the other monitoring steps (Roman numerals in Fig. 2) have been addressed in earlier studies in which authors explored and described ways to operationalize ii) selection of comparable sites (del Río-Mena et al., 2020b; Meroni et al., 2017); and iii) fitting field-based estimations of ecosystem services to RS models; and the selection of the best RS models (Ayanu et al., 2012; del Río-Mena et al., 2020b; Martínez-Harms et al., 2016). To the best of our knowledge, decisions regarding iv) the long-term variations of the relationships between field-based estimations of ecosystem services with RS models, have not yet been studied through sensitivity analyzes. Unfortunately, we lack temporal field data from the studied period (1989–2016; 2016–2020) to evaluate this. We stress the need for robust models or indicators to monitor and evaluate changes of ecosystem services in restored areas. The use of RS for the BACI analysis should be based on fieldwork measurements to estimate and validate spatially and temporally representative RS models of ecosystem services.

Our work shows a need for transparency and documentation of data and methods used to evaluate restoration interventions to learn from the (lack of) success of previous restoration initiatives and better guide policies to meet sustainability targets and future restoration practices. Regardless of the input data and methods used for the evaluation of restoration interventions, RS choices need to be documented and consistent. Having a common language regarding the evaluation of restoration interventions could additionally promote better collaboration between multiple stakeholders across sectors (Dudley et al., 2021).

5. Conclusions

This study systematically describes and demonstrates different key choices on RS data use, and their effect on restoration evaluation outcomes for ecosystem service supply. While in our study satellite sensor selection and number of sampling sites show robust results, other methodological choices can lead to contradicting results about the recovery level of a site. Notably, the common choice to use the annual minimum or the maximum value for an ecosystem service can produce opposite statements of restoration success. We also found that areas with different levels of degradation respond differently to the analyzed RS choices. The most degraded areas show the most consistent outcomes across methodological choices. Restoration monitoring and evaluation is an important theme in this UN Decade on Ecological Restoration, as it underpins learning and accountability. Understanding the effect of RS related choices on the evaluation of restored sites can help to better interpret the evaluation outcomes. Our findings show that documentation and justification of the key RS choices in light of the evaluation objectives is crucial to effectively interpret, manage, and modify restoration initiatives. Finally, we highlight the call for transparent reporting and solid sensitivity analyses when using earth observation data to assess restoration impact.

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.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecoser.2023.101565.

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