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Remote sensing for mapping ecosystem services to support evaluation of ecological restoration interventions in an arid landscape

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ABSTRACT

Considerable efforts and resources are being invested in integrated conservation and restoration interventions in rural arid areas. Empirical research for quantifying ecosystem services – nature's benefits to people – is essential for evaluating the range of benefits of ecological restoration and to support its use in natural resource management. Satellite remote sensing (RS) can be used to monitor interventions, especially in large and remote areas. In this study we used field measurements, RS-based information from Sentinel-2 imagery together with soil and terrain data, to estimate ecosystem service supply and evaluate integrated ecological restoration interventions. We based our research on the arid, rural landscape of the Baviaanskloof Hartland Bawarea Conservancy, South Africa, where several integrated interventions have been implemented in areas where decades of small livestock farming has led to extensive land degradation. Interventions included i) long term livestock exclusion, ii) revegetating of degraded areas, iii) a combination of these two, and iv) essential oil production as alternatives to goat and sheep farming. We assessed six ecosystem services linked to the objectives of the interventions: erosion prevention, climate regulation, regulation of water flows, provision of forage, biomass for essential oil production, and the sense of place through presence of native species. We first estimated the ecosystem service supply based on field measurements. Secondly, we explored the relationships between ecosystem services quantities derived from the field measurements with 13 Sentinel-2 indices and four soil and terrain variables. We then selected the best fitting model for each ecosystem service. Finally, we compared the supply of ecosystem services between intervened and non-intervened sites. Results showed that models based on Sentinel-2 indices, combined with slope information, can estimate ecosystem services supply in the study area even when the levels of field-based ecosystem services supplies are low. The RS-based models can assess ecosystem services more accurately when their indicators mainly depend on green vegetation, such as for erosion prevention and provision of forage. The agricultural fields presented high variability between plots on the provision of ecosystem services. The use of Sentinel-2 vegetation indices and terrain data to quantify ecosystem services is a first step towards improving the monitoring and assessment of restoration interventions. Our results showed that in the study area, livestock exclusion lead to a consistent increase in most ecosystem services.

1. Introduction

People living in rural landscapes strongly depend on ecosystems and their services for subsistence, income, and wellbeing (DeClerck et al., 2016; Power, 2010). However, ecosystems are rapidly degrading due to the expansion of crop and grazing lands into native vegetation and unsustainable agricultural practices, particularly in marginal agricultural areas (IPBES, 2018). Actions to avoid, reduce and reverse land degradation can increase food and water security and can contribute substantially to the adaptation and mitigation of climate change (McDonald et al., 2016).

One action to combat land degradation is ecological restoration which has been defined as "the process of assisting the recovery of ecosystems that have been damaged, degraded, or destroyed" (Clewell et al., 2004). Restoration interventions typically include increasing and/or improving vegetation cover by planting suitable species (e.g. Moreno-Calles and Casas, 2010; van der Vyver et al., 2013) or by improving water and land management (e.g. Chartzoulakis and Bertaki, 2015; Molden, 2013; Powlson et al., 2011). Here, we use the term "restoration intervention" for all activities aiming to stop, reduce or

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reverse the degradation of an ecosystem, so including rehabilitation practices.

Considerable efforts and resources have been invested in restoration interventions (Mills and Robson, 2017; Vermeulen et al., 2012) to achieve improved environmental and social integrity and resilience (e.g. Giller et al., 2009; Molden, 2013; Pretty et al., 2011). Understanding how interventions influence ecosystem services which underpin human wellbeing, has become increasingly important as part of the search for solutions for sustainability challenges (Costanza et al., 2017; Díaz et al., 2015). While rural development agencies, governments, financing institutes, the private sector, and civil society have all shown interest in investing in landscape improvements (e.g. OECD, 2016; UNEP-WCMC and IUCN, 2016; WBCSD, 2012), decision makers remain hesitant to commit resources due to the lack of robust and quantitative evidence on the positive impact of these rural interventions.

Restoration interventions often lack long-term monitoring schemes that would allow for an evaluation against the restoration objectives leading to biased or poorly informed statements of success (Nunes et al., 2016). The evaluation of restoration actions is often challenging because interventions may be located in large areas with difficult access. Additional challenges are the lack of affordable and standardized methodologies and the difficulty of obtaining long-term documentation on these projects, especially to monitor restoration interventions outside the timespan of a project (Meroni et al., 2017). Monitoring systems need to track interventions that often require a long time to start generating benefits (Alexander et al., 2016). Also, defining the appropriate location and time to monitor indicators is not an easy task; it requires the identification of the key landscape processes affected by interventions, and achieving acceptable levels of accuracy while considering financial, institutional, and human resource commitments (Heenan et al., 2016; Singh et al., 2014). A survey conducted in South Africa identified lack of knowledge, capacity constraints and lack of resources as the main obstacles for evaluating restoration practices (Ntshotsho et al., 2015). This lack of intervention evidence hampers the smart allocation of resources and represents a lost opportunity for improved decision making based on a critical reflection of lessons learnt (Nilsson et al., 2016). Standardized and effective systems for holistic evaluation and monitoring of restoration interventions are called for to inform decision makers if they are achieving the desired outcomes (Barral et al., 2015; Baylis et al., 2016; Mueller and Geist, 2016; Reed et al., 2016; Zucca et al., 2015).

Satellite images can provide synoptic information on vegetation characteristics (such as vegetation cover, biomass, carbon and crop yield) of large areas (Andrew et al., 2014; Pettorelli et al., 2016). Vegetation characteristics have been used as ecosystem services indicators to quantify multiple ecosystem services (De Araujo Barbosa et al., 2015; Frampton et al., 2013; Usha and Singh, 2013). The multispectral sensor on board the European Space Agency's constellation of Sentinel-2 satellites (Berger et al., 2012) has been successfully used for agricultural, forest and environmental monitoring applications (Pandit et al., 2018; Zheng et al., 2017). The two Sentinel-2 satellites, equipped with identical Multispectral Instruments (MSI), are capable of acquiring freely available images composed of 13 bands at resolutions ranging from 10 to 60 m (Mandanici and Bitelli, 2016). Sentinel-2 incorporates two dedicated spectral bands in the red-edge region allowing the acquisition of relevant information on vegetation spectral properties in this wavelength range at a higher spectral resolution than it has been previously possible with similar multispectral Earth observation satellites (Clevers and Gitelson, 2013; Frampton et al., 2013). The potential of RS observations in ecosystem services studies, has not yet been fully explored, but it shows great potential for monitoring processes in the context of worldwide sustainability challenges (Cord et al., 2017; Ramirez-Reyes et al., 2019; Vargas et al., 2019).

Ecosystem service maps can be used to monitor the impact of changes in the environment, and therefore support sustainable decision-making for targeting of investments and policies concerning natural resources (Daily et al., 2011). Approaches that combine RS-based information, soil and terrain GIS data and field measurements for mapping ecosystem services have been demonstrated to capture the spatial and temporal variation in the supply of ecosystem services (Choudhary et al., 2018; Martínez-Harms et al., 2016; Nizeyimana, 2016; Wood et al., 2010). Ecosystem services are often intangible and difficult to measure directly. Therefore, in order to map changes in ecosystem services supply, ecosystem services indicators are needed that are quantifiable, sensitive to changes, visible, scalable, and temporally and spatially explicit (Burkhard et al., 2012; Van Oudenhoven et al., 2012). For example, erosion prevention is difficult to visualize and quantify remotely, thus vegetation cover has been used as a proxy measure (e.g. Onyando et al., 2005; Vrieling et al., 2008). Similarly, above-ground woody biomass (AGB) has been used to quantify carbon storage in woody vegetation (Houghton, 2007; Wang et al., 2015).

This study aims to develop and apply a RS-based approach to evaluate a range of integrated ecological restoration interventions based on ecosystem service supply. For this we, (i) quantify ecosystem service supply based on field measurements; ii) use these field measurements to calibrate and test the ability of ecosystem service models based on Sentinel-2 and soil and terrain GIS data and (iii) compare ecosystem service supply from intervened and non-intervened sites for our study site in South Africa. We hypothesize that remotely sensed information can adequately capture field characteristics that determine ecosystem service supply, and as such can be used to improve monitoring and evaluation of restoration interventions in arid landscapes.

2. Study area and restorations activities

2.1. Study area description and background

Our research was based on the central and eastern areas of the Baviaanskloof Hartland Bawarea Conservancy (approx. 31,500 ha), Eastern Cape in South Africa (Fig. 1). This area comprises a mix of large, privately-owned farms (between 500 and 7600 ha) and communal rural lands. In the study area, local communities share communal land in the Baviaanskloof valley-bottom (Petz et al., 2014). These communities constitute the majority of the Baviaanskloof's population of about 2300 people (Living Lands, 2018), who are predominantly employed in seasonal farm work (Schramski, 2017). Agriculture and tourism are the main sources of income in the study area (Crane, 2006; Petz et al., 2014). The study area lies between the Baviaanskloof and the Kouga mountains and is surrounded by the Baviaanskloof Nature Reserve and the Beakosneck Private Nature Reserve. The region is also a biodiversity hotspot and is a World Heritage Site recognized for its beauty and biodiversity (Jansen, 2008). We focused our research in the subtropical thicket biome, an arid thicket form (Van der Vyver et al., 2013) where spekboom (*Portulacaria afra*) is a dominant and highly palatable species (Vlok et al., 2003).

The study area has an altitude range of 370 to 1250 m above sea level and average slope of 16.8 degrees (Supplementary Materials, Fig. S.1). The area has a mean annual rainfall of approximately 300 mm (Powell, 2009) with erratic patterns (Petz et al., 2014). Water is scarce and recurring droughts are often followed by flood events (Jansen, 2008). Our fieldwork took place in the middle of a dry period experienced in 2016 and 2017, when the Baviaanskloof River, that usually runs west to east, was dry (Fig. S.3). The average annual temperature in the area is 17 degreesC. Temperatures of up to 40 degreesC are frequently reported in mid to late summer, whereas in the valley bottoms, winter temperatures may occasionally fall below freezing (van Luijk et al., 2013).

Although thicket is relatively resilient to browsing by indigenous herbivores (Stuart-Hill, 1992), the area has been heavily degraded by unsustainable pastoralism (Havstad et al., 2000; Powell, 2009). Because spekboom is a succulent species that propagates vegetatively (Stuart-

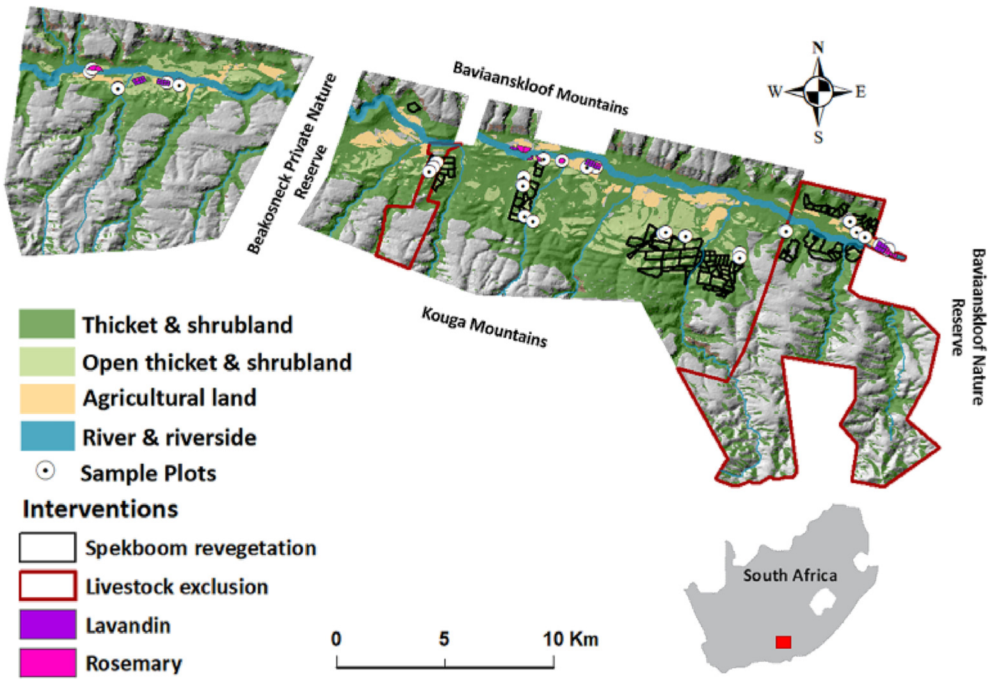

Fig. 1. Study area in the Baviaanskloof Hartland Bawarea Conservancy in South Africa with the boundaries of the assessed interventions. Shaded relief map based on a 12.5 m resolution ALOS PALSAR derived DEM from the Geophysical Institute of the University of Alaska Fairbanks.

Hill, 1992), spontaneous recovery does not occur in heavily degraded sites (Lechmere-Oertel et al., 2005b; Sigwela et al., 2009). To overcome spekboom depletion, the planting of spekboom cuttings has been practiced for over more than a decade as a practical restoration method (Mills et al., 2007; Mills and Robson, 2017; Powell, 2009; van der Vyver et al., 2013).

Land degradation has resulted in significant soil erosion (van Luijk et al., 2013). The reduction of natural vegetation, a common food source for extensive goat and sheep farming, has led to a drastic decrease in agricultural returns in recent years. In addition, it has been found that degradation of succulent thicket can affect water infiltration by decreasing the proportion of the soil surface covered with plant litter from 60 to 0.6% (Lechmere-Oertel et al., 2005a). Thicket degradation has led to the decline and replacement of (sometimes endemic) perennials by annuals plants, favoring the growth of alien (non-native) species and the loss of functional diversity (Kerley et al., 1995; Rutherford et al., 2014). For these reasons, there is an urgent need to take measures to change degradation trends and improve local livelihoods.

2.2. Interventions objectives and related ecosystem services

Four restoration interventions are implemented in the study area to address land degradation (Fig. 1):

1. Livestock exclusion: This intervention covers approximately 7,400 ha of farmlands where livestock has been removed and six small-sized fenced areas (approximately 4.6 ha in total) have been established to prevent access by livestock and wildlife. Livestock has been excluded from these areas for approximately 30 years to allow for natural revegetation that could restore the soil and water cycle.
2. Spekboom revegetation: Since the year 2004, around 1,100 ha have been planted with spekboom to reduce degradation trends and assist the recovery of the degraded thicket vegetation (Mills and Robson, 2017). The planting of spekboom truncheons was implemented through the national Department of Environmental Affairs, Natural Resource Management directorate, Expanded Public Works Program (EPWP). The planting of spekboom stopped in 2017.
3. Combination of revegetation and livestock exclusion: There are several locations with multiple interventions. Over time, spekboom was planted in most livestock exclusion areas, however, in some cases livestock was not effectively excluded. We considered the integration of these ecological restoration measures as an additional intervention to account for potential differences in ecosystem services supply. We estimated that there are 337 ha where revegetation was combined with livestock exclusion.
4. Organic essential oil production: The Baviaanskloof Development Company (Devco) was initiated through a collaboration of Commonland Foundation, Grounded, Living Lands and the Baviaanskloof Hartland Bawarea Conservancy to support the transition from small livestock farming to more environmentally sustainable and profitable agricultural businesses. Approximately 60 ha of rosemary and lavandin have been planted between December 2015 and July 2017 by four farmers. Devco and the farmers plan to increase the current planted area and to explore the production of other oil producing species in future.

Each of the four restoration interventions aimed to impact a set of ecosystem services. Table 1 shows the selected ecosystem services for evaluating the impact of each intervention. This selection includes provisioning, regulating and cultural ecosystem services and was based on the intervention objectives and critical local challenges related to land degradation in the study area. Even though erosion prevention was not an initial objective in the essential oil intervention, it was still measured in plots with essential oil fields and other agricultural lands, because these areas are prone to heavy wind erosion and soil deterioration when the finer particles are blown away (Lal, 2001). Since conserving and building soil quality is crucial in agricultural activities, we consider vegetation cover as a critical contributor for protecting the soil.

To quantify erosion prevention, we used the cover of different vegetation layers as the ecosystem service indicator. Different vegetation types differ in their capacity to prevent soil erosion; therefore, specific vegetation structures determine different levels of rainfall interception, storage, and runoff (De Jong and Jetten, 2007; Fu et al., 2005; Zheng et al., 2008). Vegetation also contributes to climate regulation by

Table 1
Ecosystem services and their indicators used to evaluate the interventions in the study area.

Ecosystem service	Ecosystem service indicator	Evaluated interventions
Erosion prevention	Stratified vegetation cover index (% Str.VC)	1,2,3,4
Climate regulation (Carbon storage)	Above ground carbon stocks (g m-^{2})	1,2,3,4
Regulation of water flows	Soil infiltration rate (cm hr^{-1})	1,2,3
Provision of forage	Green biomass (kg m^{-2})	1,2,3
Biomass for essential oil production	Fresh biomass (g m^{-2})	4
Presence native species	Spekboom cover (%)	1,2,3

storing carbon. Intact thicket in the Eastern Cape stores a relatively large amount of carbon for a semiarid ecosystem (Mills et al., 2003), and much of the biomass is comprised of spekboom (Vlok et al., 2003). Restoration with spekboom could potentially return large amounts of carbon to successfully transformed sites (Lechmere-Oertel et al., 2005a; Mills and Cowling, 2006). For the regulation of water flows, we used soil infiltration rates under different vegetation cover types. Infiltration rates are expected to increase over time in successfully revegetated landscapes due to higher soil macropores formed by invertebrates and colonizer organisms (Colloff et al., 2010). To quantify forage production, we estimated the green biomass of cover crops on essential oil fields and thicket green biomass. Thicket is an important forage source for animals since it generally has high palatability (Kerley et al., 2006). We used fresh plant biomass as an indicator for the production of essential oil derivatives. The presence of spekboom is a cultural ecosystem service in the area. Spekboom trees are a crucial component of the arid thicket that has been severely degraded in the study area. Spekboom increases biodiversity through the provision of a unique microclimate and a mulch-rich substrate for attracting other plant species (Lechmere-Oertel et al., 2008; Mills et al., 2005; Van der Vyver et al., 2013). The presence of spekboom not only contributes environmentally but also helps to conserve and promote the natural beauty and a sense of place which is important for the inhabitants (Jansen, 2008).

3. Methods

We used three main steps to compare the four selected interventions (Fig. 2). We first quantified six ecosystem service supplies based on field measurements. Secondly, we used these field measurements to calibrate and test ecosystem service models based on RS information from Sentinel-2 and soil and terrain GIS data. Finally, we compared the ecosystem service supply from intervened and non-intervened sites.

3.1. Field-based ecosystem services quantification

3.1.1. Experimental field data collection design

During a fieldwork period from 06/05 and 13/07 2017, we measured ecosystem services in 32 plots of 30 × 30 m that were distributed over the study area (Fig. 1). In this study we simplified the restoration impact assessment to a comparison of current ecosystem service supply between intervened and non-intervened sites. Plots were located in pairs, one in an intervened and one in a non-intervened location. Paired plots were located close to each other in order to avoid wide variations in soil and weather conditions. We also ensured that paired plots had similar slope angle and orientation. The paired plots were grouped according to their baseline management (employed before the intervention occurred) in order to distinguish the effect of different land management changes (Table 2). The baseline management for those plots located on farms where livestock has been removed, corresponded to abandoned crop fields that were taken out of production in the 1990s. For those plots located on farms that still had livestock, pasture

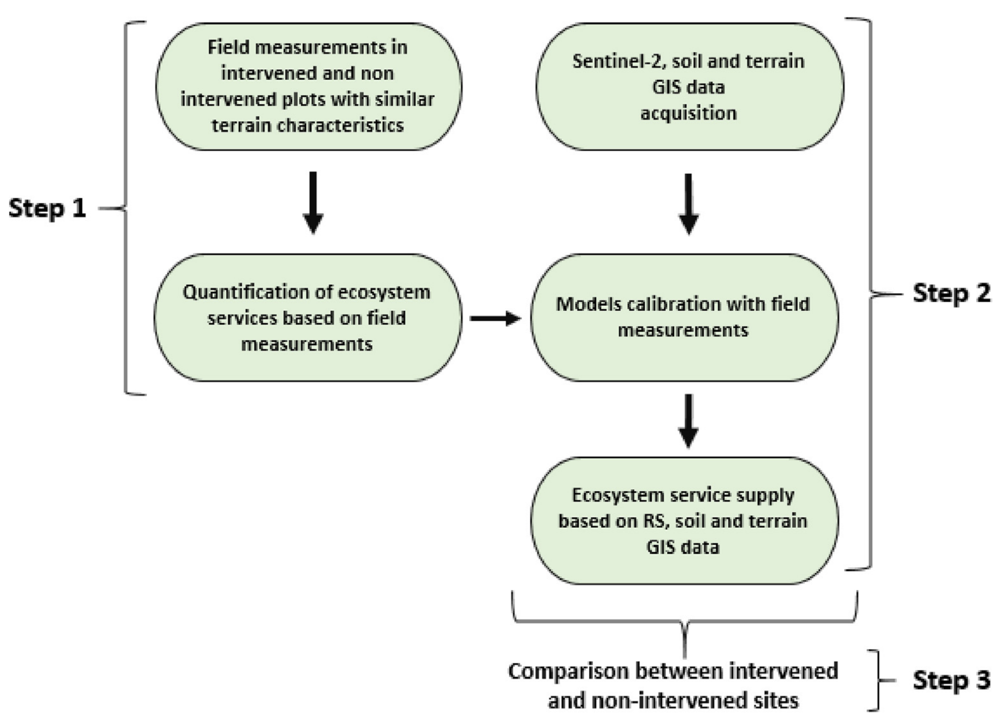

Fig. 2. General workflow.

Table 2
Evaluated interventions and their intervened and non-intervened components for each pair of plots.

Intervention (management difference)	Baseline management (non-intervened)	New management (intervened)	Number of paired plots
Livestock exclusion	Livestock + not revegetated	Livestock exclusion	1
	Spekboom revegetation	Livestock exclusion + spekboom revegetation	2
Spekboom revegetation	Livestock & not revegetated	Spekboom revegetation	3
	Abandoned fields + livestock exclusion	Livestock exclusion + spekboom revegetation	2
Livestock exclusion + spekboom revegetation	Livestock + not revegetated	Livestock exclusion + spekboom revegetation	2
Essential oil production	Abandoned fields + livestock exclusion	Essential oil production	2
	Pasture	Essential oil production	4

was assigned as the baseline management.

To minimize errors resulting from GPS inaccuracies, we measured corners, points in the middle of each side and center of the plot. Geolocations were obtained using a Garmin eTrex 30x. To mark a point we waited until obtaining at least 5 m horizontal accuracy indicated by the GPS, i.e. smaller than the Sentinel-2 pixel size. The study area mostly has open vegetation, so there is no interference of high trees. The overall geolocation uncertainty of Sentinel-2 level 1C data is less than 11 m for 95.5% of the cases (Clerc and MPC-Team, 2017), which is about the size of one Sentinel-2 pixel. To reduce the effect of Sentinel-2 geolocation uncertainty for the ecosystem service model calibration, we located each plot within a homogenous area, considering vegetation cover, slope angle and orientation, and land management. We allowed paired plots to be closer to each other than 16 m (the maximum expected geolocation error) in two cases where interventions were fenced. Here we recorded the geolocation of the fences and visually checked if the plot locations corresponded with the vegetation values on the image.

In each plot we measured the total number of trees, bushes and infiltration rates. Within each plot we placed four subplots of 2 × 2 m for measurements of canopy dimensions and herbaceous vegetation cover (details can be found in following sections). The subplots included representative vegetation cover and canopy dimensions from the plot (Supplementary Material, Figs. S.4. and S.5). The parameters measured in the four subplots were averaged and used as reference to estimate the ecosystem services in the plot. Between 18/09 and 05/10 2018 we measured 11 additional 30 × 30 m plots for establishing a relationship between biomass for essential oil production and climate regulation with RS variables (Table S.8). Following the same methodology as in 2017, we placed four subplots of 2 × 2 m inside each plot and performed the same measurements as in the previous year. We then compared the field based estimation of ecosystem services (Table S.1).

3.1.2. Quantification of erosion prevention

We used the stratified vegetation cover index (StrVC) as the indicator for erosion prevention. The StrVC integrates the cumulative effect of different vegetation layers with different capacities of controlling soil erosion (Zhongming et al., 2010). We measured the StrVC by counting all shrubs, aloes, or trees inside the plot and then multiplying their total number by the average canopy sizes and vegetation cover (%) measured in the subplots. Inside all the subplots, we measured the height, diameter and cumulative basal stem area (CBSA) for shrubs and the diameter at breast height (DBH) for trees. We used digital photos to separately calculate the vegetation coverage of each vertical layer following the approach of Zhongming et al. (2010). Pictures of trees were taken from below the canopy, looking upwards, using a Canon EF 15 mm f/2.8 Fisheye Camera. The coverage of shrubs, small cacti and herbaceous vegetation was estimated by taking pictures from above and looking downwards, using a smartphone (iPhone6). Pictures of herbaceous vegetation were taken in representative locations within each subplot using a frame of 25 cm^2 (internal area). Pictures were later standardized for same proportions and resolutions and all the values per plot were averaged. The green cover percentage was extracted using Canopeo (Fig. 3), software developed by the Plant and Soil Sciences department and the App Center at Oklahoma State University (http://www.canopeoapp.com).

We considered trees, shrubs and herbaceous cover as different vegetation strata and we disregarded the litter layer because it was mostly absent in the sampled areas. We first calculated the bush and tree strata (Eq. (1)). In Eq. (1), (C_c) is the canopy cover in the sampling unit for shrubs and trees; A_{su} is the area of every sampled unit;°$C_i$is the crown size of each tree or shrub;I is the number of trees or shrubs in the sampling unit, and; $C_{aver}$is the average fractional vegetation cover (FVC) of trees or shrubs.

$$C_c = C_{aver}\frac{1}{A_{su}}\sum_{i=1}^{n} C_i \quad (1)$$

The stratified vegetation cover for each plot was then calculated using Eq. (2), where a_i is the weighting coefficient of layer i for its contribution to soil conservation; i is the number of layers, or strata in the vegetation community, and; C_i is the measured coverage of the vertical layer i. We used the weighting coefficient (a_i) proposed for broadleaved vegetation types: 0.06 for trees and aloes, 0.55 for shrubs and plants for oil production and 0.28 for grass and herbaceous cover (Zhongming et al., 2010).

$$\text{StrVC} = \sum\nolimits_{i=1}^{n} \text{aiCi} \quad (2)$$

3.1.3. Quantification of climate regulation

To assess the amount of carbon stored in aboveground vegetation (g m^{-2}) we counted all trees and shrubs per plot (Supplementary Materials, Table S.3) and measured the height, diameter at breast height (DBH) for trees and cumulative basal stem area (CBSA) for shrubs in each subplot. All shrub and tree species inside the subplots were identified. Their aboveground carbon storage was calculated individually using species-specific allometric equations following the methods developed by Powell, (2009) (Table 3).

For species lacking allometric equations, carbon was estimated using the same equivalent species with similar growth patterns used by van der Vyver et al. (2013) for which regression equations existed (Supplementary Materials, Table S.6). We transformed the results of the aboveground carbon estimations for non-agricultural areas from kg to g for easier comparison with the carbon estimations in the essential oil fields.

To estimate climate regulation of lavandin and rosemary fields, we calculated the total dry aboveground biomass (dAGB) using our allometric equations based on 18 harvested rosemary plants and 12 harvested lavandin plants. We used the elliptical area (π × long radius × short radius) of the plant as the predicting variable of the total dry biomass (Table 4). We assumed 48% of the biomass to be carbon (Magnussen and Reed, 2004). The carbon stock of the herbaceous cover was considered negligible.

The carbon estimations of the subplots were then upscaled to plot level by multiplying the average measured value of one species by the total number of individuals of each species in the plot.

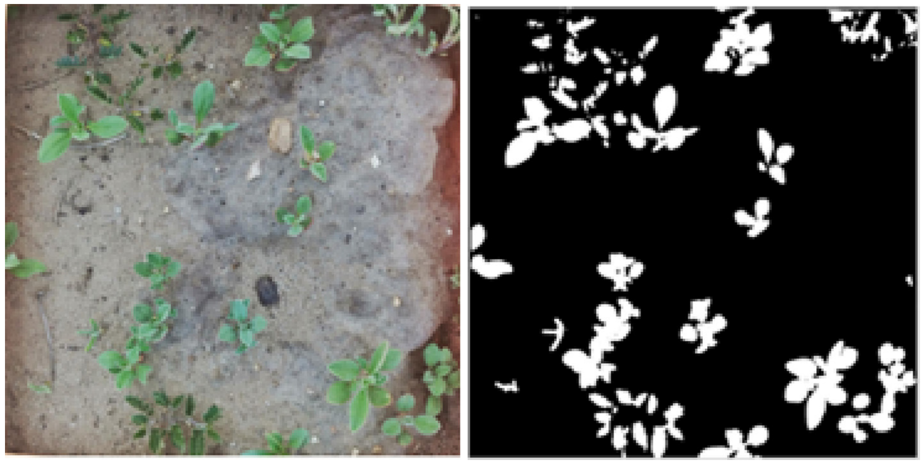

Fig. 3. Example of green vegetation cover (%) estimation using Canopeo. Left: original picture. Right: app generated picture used to calculate the percentage of the total area covered with green vegetation (white sections).

3.1.4. Quantification of regulation of water flows

The regulation of water flows was estimated by measuring the infiltration rate beneath different vegetation covers per plot. For each vegetation cover we considered between two and three repetitions per plot. In each repetition, the time taken for the soil to absorb 120 cm^3 of water poured into a 3.65 cm radius single-ring cylinder was measured twice consecutively. The second measurement was done immediately after the first measurement. The infiltration rates of different vegetation cover for dry and wet soil in non-irrigated fields were estimated based on the cylinder volume formula (volume = πr^2 *height). We estimated the height (2.87 cm) of the 120 cm^3 of water applied and measured the respective infiltration time (in seconds) on the field (Johnson, 1963). We only used the second infiltration rate measured under wet soil when calibrating the RS models.

The infiltration rate of one plot (in cm per hour) was calculated in four steps. First, we calculated the plot average infiltration rate for each vegetation type. Secondly, we calculated the total area covered by each vegetation type considering the average canopy areas of each vegetation type and their number of shrubs or trees, aloes and spekboom. The average cover percentage was used to estimate the total area covered by herbs. Then, we added all the areas covered by the different vegetation types. The difference between the plot area and the parts covered by vegetation was assumed to be bare soil. The estimation of the bare soil surface could have been underestimated where different vegetation covers overlapped. We finally calculated a single plot infiltration rate using a weighted average of the infiltration rates of different vegetation types and their respective cover ratios.

3.1.5. Quantification of provision of forage

To estimate provision of forage for wild animals, we used linear regressions between green biomass (GB) and vegetation cover (VC) for shrubs and grasses in arid environments (Flombaum and Sala, 2007) (Table 5). In these equations, green biomass represents all the green leaves for grasses and all the green leaves and current twigs for shrubs (secondary growth was not considered).

The vegetation cover was estimated as described in section 3.1.2. Small sized *Vachellia karroo* trees and aloes were considered as shrubs for fodder estimation (Breebaart et al., 2002). Although many trees in the area are palatable (e.g. *Pappea capensis*), animals would only browse up to their reach limit (Rutherford et al., 2014). Therefore, we did not include tree green biomass for provision of forage. In essential oil production fields, the green biomass of herbaceous strata (cover crops and weeds) was included in the forage estimates by using the general

Table 4
Dry aboveground biomass (dAGB) (g) allometric equations using the elliptical canopy area (EA) in (cm^2).

Crop	Equation	R^2	n
Lavandin	dAGB = 0.00243 $EA^{1.2824}$	0.93	12
Rosemary	dAGB = 0.0055 $EA^{1.4934}$	0.93	18

Table 5
Green biomass (GB) equations for shrubs and grasses using vegetation cover (VC) as predictive variable (Flombaum and Sala, 2007).

Vegetation type	Equation	R^2
Shrubs	GB (g m^{-2}) = 227.3 × VC (%)	0.64
Grasses	GB (g m^{-2}) = 78.1 × VC (%)	0.72

Table 3
Allometric equations used to predict aboveground carbon and biomass (kg) for studied species. CBSA is base stem area and DBH stands for diameter at breast height. n: number of shrubs or trees used by Powell (2009) to develop the equations.

Species	Equation	R^2	n
Ehretia rigida	$Log_{10}y$ (C) = 0.9623(Log_{10} CBSA(m^2)) − 2.485	0.63	24
Portulacaria afra	$Log_{10}y$ (C) = 1.1043(Log_{10} CBSA(m^2)) + 2.4464	0.97	5
Aloe ferox	$Log_{10}y$ (C) = 1.4306(Log_{10} CBSA(m^2)) + 3.6975	0.78	25
Grewia robusta	$Log_{10}y$ (C) = 1.0044(Log_{10} canopy area (m^2)) − 0.6259	0.85	37
Lycium ferocissimum	$Log_{10}y$ (C) = 0.8615(Log_{10} CBSA(m^2)) + 1.7706	0.77	35
Pteronia incana	$Log_{10}y$ (C) = 1.4032(Log_{10} canopy area (m^2)) − 0.4224	0.97	49
Putterlickia pyracantha	$Log_{10}y$ (C) = 1.0622(Log_{10} CBSA(m^2)) + 2.7834	0.78	46
Jatropha capensis	$Log_{10}y$ (C) = 0.9067(Log_{10} canopy area (m^2)) − 0.7349	0.57	21
Vachellia karroo	$Log_{10}y$ (C) = 2.034(Log_{10}canopy area (m^2)) − 1.20113	0.95	15
Rhus longispina	$Log_{10}y$ (C) = 1.1012(Log_{10}canopy area (m^2)) − 0.2938	0.51	24
Pappea capensis	$Log_{10}y$ (C) = 1.3355(Log_{10}canopy area (m^2)) − 0.2938	0.93	22

Table 6
Fresh aboveground biomass (fAGB) (g) allometric equations using the elliptical canopy Area (EA) (cm^2).

Crop	Equation	R^2	n
Lavandin	$fAGB = 0.0748\ EA^{1.2325}$	0.94	12
Rosemary	$fAGB = 0.0261\ EA^{1.4067}$	0.95	18

green biomass equations for grasses (Flombaum and Sala, 2007). To upscale the subplots values to a plot level, we multiplied the average herbaceous and shrub green biomass from the subplots with the total plot area.

3.1.6. Quantification of biomass for essential oil production

The total fresh aboveground biomass (fAGB) of rosemary and lavandin was considered to quantify the potential provision of essential oil production. In this study we present the ecosystem service as fresh biomass because the ratios between fresh and harvestable biomass can vary between different management practices. Similarly to the carbon estimations described in section 3.1.3., we used field-based allometric equations to link the canopy measured dimensions with the total fresh biomass (Table 6). The averaged fresh biomass from all measured plants was multiplied with the total number of plants and divided by the plot area to obtain one biomass value (g m^{-2}) for each plot.

3.1.7. Quantification of presence of native trees

The presence of spekboom was estimated by its total covered area (%). We counted the total number of spekboom per plot. We then measured all the spekboom canopy dimensions in the subplots and calculated the average spekboom canopy area. We finally obtained one value of spekboom cover for each plot by multiplying the total number of spekboom with the average canopy dimensions.

3.2. Calibration of the RS-based ecosystem service models

3.2.1. RS and GIS data acquisition

We downloaded a Sentinel-2A image from 24/06/2017 (acquired over tile 34HGH), corresponding to the middle of the field work period and another Sentinel-2A image from 7/10/2018 (for essential oil production estimates only). The ESA Sen2cor processor was used for the atmospheric and topographic correction of the Sentinel-2 Top-Of-Atmosphere Level 1C image (ESA, 2018). We used one of the two available DEM provided in Sen2Cor (90 m SRTM Digital Elevation Database from CGIAR-CSI) for terrain correction (Jarvis et al., 2008; Mueller-Wilm et al., 2019). Terrain correction is an important step in the preprocessing of data when land cover classification and quantitative analysis of multispectral data are carried out in mountainous surfaces, which lead to variations in reflectance of similar ground characteristics, leading to possible misclassifications (Twele et al., 2006).

The 'super-resolving enhancement' method (Brodu, 2018) was used to propagate the high-resolution spatial details to the lower-resolution bands while preserving their spectral content in order to calculate the Sentinel-2 indices that require the use of originally lower spatial resolution bands. Based on available literature and expert knowledge, we tested eleven vegetation indices, one soil (brightness index) and one water index (NDWI) for their ability to capture ecosystem service supply (Table 7). The formulas of these indices can be found in the Supplementary Materials (Table S.2).

Additionally to the Sentinel-2 data, we extracted slope (degrees), altitude (m) and aspect (north, east, south, west) from a 12.5 m resolution ALOS PALSAR derived DEM (Geophysical Institute of the University of Alaska Fairbanks, 2018). Information on soil types was derived from the SOTER-based soil parameter estimates (SOTWIS) for Southern Africa (Batjes, 2004) (Fig. S.2). To obtain models that were able to capture a representative ecosystem services supply of the larger areas from where the plots were located, we calculated a weighted average of the index values and terrain parameters using all pixels entirely or partly inside the sampled unit (Fig. 4).

All ecosystem service calculations for the field plots were done in the R (R Core Team, 2019a) and R-Studio software version R i386 3.4.3 (R Core Team, 2019b). In addition to field measurements, spatial datasets of spekboom revegetated areas were provided by the Gamtoos Irrigation Board (GIB), though Living Lands. Maps of fields planted with rosemary and lavandin were provided by the Baviaanskloof Development Company for essential oil production (Devco). Living Lands also provided property boundary maps, which originated from a cadastral map from the surveyor general of the South African government. All vegetation indices were calculated using the SNAP desktop implementation version 6.0.4 software and the Sentinel-2 Toolbox (S2TBX) version 6.0.2. (ESA, 2018). DEM-based calculations and weighted averaging was performed in ArcMap 10.5.1 (Esri, 2017).

3.2.2. Fitting RS and GIS variables to ecosystem service field measurements

We built linear regression models using the calculated based on the Sentinel-2 data and terrain variables derived from DEM and soil maps as predictors, to produce estimates of ecosystem services supply based on field data. The procedure to select the predicting variables was as follows. First, in line with the objective of this study of testing the use of RS to detect changes in ecosystem service provision, we prioritized the use of Sentinel-2 indices and subsequently added soil and terrain variables to potential models. Secondly, to avoid multi-collinearity between predicting variables, we only considered models having a Variance Inflation factor (VIF) lower than five (Kock and Lynn, 2012) using the R caret package (Kuhn, 2019).

There is general consensus that a VIF of 10 or higher, indicates serious multi-collinearity (O'Brien, 2007), although the use of more conservative thresholds can also be found in the literature (Hair et al., 2011; Kock and Lynn, 2012; Sheather, 2006). Then, we selected the five best models based on the lowest Akaike Information Criterion (AIC) (Akaike, 1974). Subsequently, we chose the best model using the Akaike weight of each model that helps to maximize both explanatory power and inclusion of the independent variables with the highest contribution to the explanatory power using the multi-model inference (MuMIn) R package (Bartón, 2018). Finally, to validate the models we used a five-fold cross-validation which we repeated 100 times (Efron and Gong, 1983) by using caret R package (Kuhn, 2019). We used cross validation due to our small sample size which did not allow us to use a held back data for independent validation of the models. When more than one predictor was used in the regression model, the partial R^2 was calculated using the asbio R package (Aho, 2019). The partial R^2 indicates the proportion of variation explained by a predictor variable which is not explained by the model if that variable is left out. Using the selected fitted RS-based models, we then calculated the ecosystem service supply for each plot.

3.3. Comparison of ecosystem services supplied by different interventions

To compare the ecosystem service supply between the intervened and non-intervened locations, we calculated the differences (in percentage) in ecosystem service supply between the baseline management (management before the intervention occurred) and the new management (intervened plot) for each paired set of plots. The new management could include one or a combination of interventions compared to the baseline management. To assess the impact of different interventions, we grouped paired plots having the same intervention and baseline management. For the evaluation of possible reasons behind the changes in ecosystem services supply, we also considered other drivers of change such as initial state of degradation and differences in management. To test the effect of the intervention lifespan, we ran simple linear regressions between the difference between intervened and non-intervened plots and the lifespan of livestock exclusion and spekboom

Table 7
Sentinel 2 indices tested in this study and references to their uses in previous studies to predict biophysical variables. LAI: leaf area index.

Index	Previous use as RS indicators for biophysical variables	Example studies
Normalized Difference Vegetation Index (NDVI)	Aboveground biomass; burn Severity; canopy chlorophyll content; LAI; Chlorophyll Content	Baloloy et al., 2018; Fernández-Manso et al., 2016; Frampton et al., 2013
Green Normalized Difference Vegetation Index (GNDVI)	Aboveground biomass; Burn Severity; Chlorophyll Content	Baloloy et al., 2018; Fernández-Manso et al., 2016
Normalized Difference Vegetation Index red-edge 2 (NDVIre2) or Red-edge Normalized Index 740 (NDVI740)	Burn Severity, Chlorophyll content; biomass	Fernández-Manso et al., 2016; Peng et al., 2017
Normalized Difference Vegetation Index red-edge 2 narrow (NDVIre2n)	Burn Severity	Fernández-Manso et al., 2016
Plant Senescence Reflectance Index (PSRI)	Burn Severity, leaf chlorophyll	Fernández-Manso et al., 2016; Stagakis et al., 2010
MERIS Terrestrial Chlorophyll Index (MTCI)	Leaf chlorophyll concentration; canopy chlorophyll content; LAI; chlorophyll content	Frampton et al., 2013; Peng et al., 2017
Inverted Red-Edge Chlorophyll Index (IRECI)	Leaf chlorophyll concentration; canopy chlorophyll content; LAI; aboveground biomass	Castillo et al., 2017; Frampton et al., 2013
Normalized Difference index (NDI45)	Aboveground biomass	Castillo et al., 2017
Soil Adjusted Vegetation Index (SAVI)	Aboveground biomass	Baloloy et al., 2018
Modified Soil Adjusted Vegetation Index (MSAVI)	Soil organic carbon, chlorophyll content	Gholizadeh et al., 2018; Raymond Hunt et al., 2011
Red-edge Normalized Index 705 (NDVI705)	Chlorophyll content; biomass	Peng et al., 2017; Viña and Gitelson, 2005
Normalized Difference Water Index (NDWI) (for vegetation water content)	Vegetation moisture content; pasture surface temperature; biomass; pasture degradation index	Dennison et al., 2005; Gao, 1996; Jackson et al., 2004; Serrano et al., 2019
Brightness Index (BI)	Soil organic carbon; texture (clay)	Gholizadeh et al., 2018

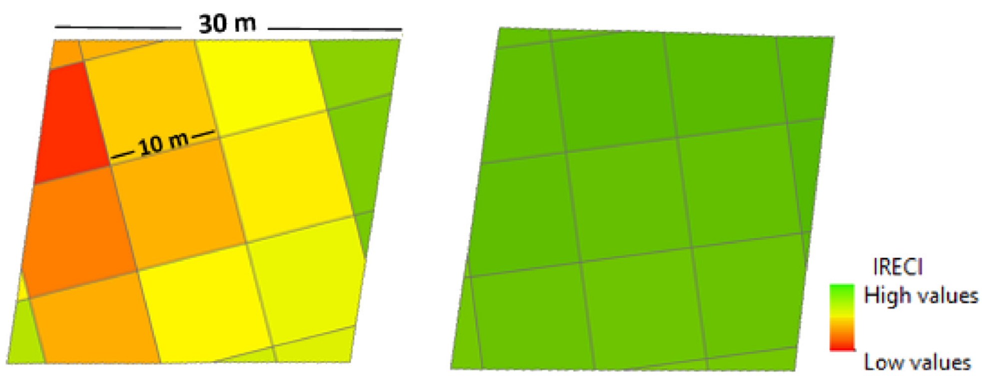

Fig. 4. Example of a pair of plots. In the example, one weighted average of IRECI vegetation index calculated for each plot. Values were calculated according to the value of each pixel and their percentage of areas in the plot.

revegetation.

4. Results

4.1. Field-based ecosystem service quantification

The supply of the six studied ecosystem services based on field measurements is presented in Table 8. The values for all six were positively skewed, i.e. most values were below the mean. The observed high variation in provision of ecosystem services is in agreement with the high variability of green cover between plots (Supplementary Materials, Table S.1. for details on field-based ecosystem services estimation at plot level). The stratified vegetation cover shows a mean of 1.58% across the plots. This value should not be interpreted as a vegetation cover, as it is calculated using weighting factors of vegetation strata according to their capacity to prevent erosion.

4.2. Fitting RS and GIS variables to ecosystem service field measurements

The mean R^2 obtained from the repeated cross validation of the best RS-based models ranged from 0.61 (regulation of water flows) to 0.89 (provision of forage). The contribution of the Sentinel indices in these

Table 8
Descriptive statistics of field-based ecosystem services estimation for n number of field plots. SbR: Spekboom revegetation; LE: livestock exclusion; Herb VC: herbaceous vegetation cover; fAGB: fresh aboveground biomass. *ES based on field data from 2018.

Ecosystem service	Indicator	Intervention	Mean	Median	Std. Deviation	n
Erosion prevention	Stratified vegetation cover (%)	All	1.6	0.7	2.0	32
Climate regulation	Aboveground carbon stock (g m^{-2})	SbR, LE, LE + SbR	1670.1	660.2	2130.0	22
Climate regulation*	Aboveground carbon stock (g m^{-2})	Rosemary	98.8	67.1	56.0	24
Climate regulation*	Aboveground carbon stock (g m^{-2})	Lavandin	42.8	40.8	27.9	20
Regulation of water flows	Infiltration rate (cm h^{-1})	SbR, LE, LE + SbR	0.8	0.70	0.4	20
Provision of forage	Green biomass (kg m^{-2})	All	9.1	4.6	9.6	32
Biomass for essential oil production*	Total fAGB (g m^{-2})	Rosemary	536.0	371.7	294.2	24
Biomass for essential oil production*	Total fAGB (g m^{-2})	Lavandin	199.3	188.1	127.1	20
Presence of native species	Spekboom cover (%)	SbR, LE, LE + SbR	0.9	7.70E-06	2.3	20

Table 9
The selected ecosystem service linear models based on indices derived from Sentinel 2 data and terrain variables. AIC: Akaike Information Criterion; Wi: Akaike weight; RMSE: root mean squared error; SbR: Spekboom revegetation; LE: Livestock exclusion; HerbVC: Herbaceous vegetation cover; AGC: Aboveground carbon; fAGB; fresh aboveground biomass.

Ecosystem service	Indicator	Intervention	AIC	Wi	R^2	Standardized RMSE	p-value	Explanatory Variable	β Estimate	Std. Error	p-value	Partial R^2	df
Erosion Prevention	Stratified vegetation cover (%)	All	68.8	0.59	0.81	0.07	< 0.001	Intercept	−1.08	0.207	< 0.001		30
								IRECI	27.35	1.744	< 0.001	0.81	
Climate Regulation	AGC (kg m^{-2})	SbR, LE, LE + SbR	82.8	0.52	0.62	0.16	< 0.001	Intercept	−2.68	0.844	< 0.001		20
								IRECI	58.18	10.565	< 0.001	0.62	
Climate Regulation	AGC (g m^{-2})		250.0		0.69	0.25	< 0.001	Intercept	−138.26	49.832	< 0.05		
		Rosemary		0.46				MTCI	70.12	19.467	0.002	0.39	21
								Slope	17.46	5.378	0.004	0.34	
Climate Regulation	AGC (g m^{-2})	Lavandin	171.0	0.9	0.76	0.17	< 0.001	Intercept	16.25	8.889	< 0.001		17
								NDVIre2n	844.05	123.919	< 0.001	0.73	
								HerbVC	−2.32	0.340	< 0.001	0.73	
Regulation of water flows	Infiltration rate (cm h^{-1})	SbR, LE, LE + SbR	10.7	0.48	0.61	0.24	0.002	Intercept	0.96	0.282	0.004		17
								NDWI	3.01	1.091	0.013	0.31	
								Slope	0.04	0.014	0.006	0.36	
Provision of forage		All	178.9	0.41	0.89	0.10	< 0.001	Intercept	2.91	4.392	0.51		28
	Green biomass (kg m^{-2})							NDI45	120.08	18.556	< 0.001	0.38	
								NDWI	43.16	10.487	< 0.001	0.60	
								Slope	0.24	0.105	0.030	0.16	
Biomass for oil production	Total fAGB (g m^{-2})		332.1			0.26	< 0.001	Intercept	−705.09	263.780	< 0.05		
		Rosemary		0.55	0.71			MTCI	368.10	103.050	0.002	0.38	21
								Slope	90.85	28.470	0.004	0.33	
Biomass for oil production	Total fAGB (g m^{-2})	Lavandin	230.6	0.9	0.77	0.16	< 0.001	Intercept	73.32	39.848	< 0.001		17
								NDVIre2n	3898.42	555.515	< 0.001	0.74	
								HerbVC	−10.56	1.524	< 0.001	0.74	
Presence of native trees		SbR, LE, LE + SbR	75.5	0.41	0.64	0.19	< 0.001	Intercept	−10.77	2.957	< 0.001		16
	Spekboom cover (%)							IRECI	56.63	10.585	< 0.001	0.64	
								Elevation	0.01	0.005	0.042	0.23	
								Slope	0.12	0.065	0.087	0.17	

models varied from 0.81 to 0.31 (expressed as the partial R^2) for erosion prevention (IRECI) and regulation of water flows (NDWI) respectively (Table 9). Ecosystem services indicators that are directly related to the presence of vegetation are predicted more accurately (higher R^2) than those depending strongly on other variables such as soil texture or presence of soil crust. The erosion prevention and provision of forage models presented the lowest residual variance (Standardized RMSE = 0.07 and 0.1 respectively). In contrast, the highest values of residual variance were present in the models for regulation of water flows (0.24), climate regulation and biomass of rosemary (0.25 and 0.26 respectively).

Of the tested Sentinel-2 indices, IRECI showed a best fit in models for predicting the erosion prevention, climate regulation services in non-agricultural areas and the presence of native species. More commonly used indices such as NDVI, SAVI, MSAVI and NDI45 were not selected as the best RS variables in any of the models (Table S.7). The NDWI showed to be the best predicting variable for the provision of forage, regulation of water flows, lavandin biomass for oil and climate regulation ecosystem service models

Although the spekboom cover percentage is linked to vegetation, the model did not capture this single species as accurately (64% of the spekboom cover variation) as models that include the overall presence of thicket vegetation information (e.g. in the stratified vegetation cover or provision of forage models). The selected model for predicting the regulation of water flows could only capture 61% of the infiltration variation.

Due to the differences in reflectance in rosemary and lavandin fields compared to non-agricultural areas, the Sentinel indices selected to estimate their aboveground carbon stocks also differed. Regarding the rosemary models, both selected models for predicting biomass for essential oil production and aboveground climate regulation were based on MTCI as the RS variable. The biomass and aboveground carbon of lavandin plots presented the best correlation with NDVIre2n. The models for lavandin biomass for essential oil production and climate regulation also contained herbaceous cover as a predicting variable with a negative coefficient.

Details on the Sentinel-2 indices per plot are shown in the Supplementary Materials Tables S.4. and S.5.

4.3. Comparison of ecosystem services supplied by different interventions

Using the selected RS-based models, we calculated the supply of ecosystem services in all the intervened and non-intervened plots. The supply of ecosystem services was grouped by intervention type (Tables 10 and 11). In Tables 10 and 11, each intervention is presented next to their baseline management and the lifespan of their respective intervention. The difference between the baseline management (non-intervened) and the intervened plot are presented in the same units as the ecosystem services. The green cells indicate a positive difference of the ecosystem service between the intervened plot compared with the baseline management, whereas the red and yellow cells indicate a negative and no difference respectively. In the case of erosion prevention, provision of forage and regulation of water flows, all the differences in ecosystem supply after livestock exclusion indicated an improvement in the ecosystem services supply. In comparison with the baseline management, the intervened plots with livestock exclusion showed on average 71% more erosion prevention, while the increase for livestock exclusion in combination with revegetation was 31%, on average. However, the spekboom revegetation intervention showed 10% less erosion prevention on average in comparison to baseline management. The provision of forage and regulation of water flows followed a similar pattern; only plots with interventions that included livestock exclusion showed on average a higher service supply compared to the baseline situation. While we expected the difference between baseline and intervention to become larger with an increased intervention lifespan, we did not find a significant correlation between ecosystem services supply

Table 10
Comparison of provision of erosion prevention, provision of forage and regulation of water flows for each paired plot calculated using RS-based models. The differences between the baseline (non-intervened) and the intervened plots are shown. Red and green cells indicate a higher and lower supply of ecosystem services provision respectively at intervention sites in comparison with the baseline management. StrVC: Stratified vegetation cover; VC: vegetation cover; NA: Not applicable. *Refers to planting dates of spekboom and oil producing plants.

Intervention (management difference)	Baseline management (non-intervened)	Intervention lifespan* (months)	Baseline erosion prevention (StrVC %)	Difference in erosion prevention (StrVC %)	Baseline provision of forage (kg m^{-2})	Difference in provision of forage (kg m^{-2})	Baseline regulation of water flows (cm h^{-1})	Difference in regulation of water flows (cm h^{-1})
Livestock exclusion	Presence livestock & Not revegetated	360	0.58	1.62	3.6	12.3	1.01	0.36
	Spekboom revegetated	110	0.85	2.03	6.88	12.27	1.61	0.37
		110	0.08	0.17	1.84	1.13	1.03	0.12
Spekboom revegetated	Presence livestock & Not revegetated	87	0.31	−0.15	14.75	−7.48	2.2	−0.27
		52	0.12	0.1	1.81	−0.43	1.22	−0.12
		48	0.45	−0.1	5.14	−2.83	1.39	−0.13
	Abandoned fields & livestock exclusion	70	1.1	0.11	9.98	1.31	2.17	0.07
		24	0.89	0.18	6.74	4.01	1.67	0.07
Livestock exclusion & Spekboom revegetated	Abandoned livestock & Not revegetated	15	1.32	0.3	9	0.8	1.04	0.03
		107	0.62	2.04	3.11	16.31	1.21	0.61
Rosemary	Abandoned fields & livestock exclusion	10	0.91	0.17	1.85	−0.2	NA	NA
	Pastures	14	8.48	−3.35	34.48	−2.89	NA	NA
		15	0.84	0.74	0.66	3.82	NA	NA
Lavandin	Abandoned fields & livestock exclusion	2	1.23	−0.74	4.09	−4.09	NA	NA
	Pastures	20	4.98	0.17	26.37	−7.42	NA	NA
		19	0.85	0.17	2.91	−0.82	NA	NA

and the time since the start of an intervention.

There was high variability in ecosystem service provision in plots with essential oil crops. The plots intervened with rosemary and lavandin presented a decrease in aboveground carbon stock of 95.9% and 99.7% respectively compared to the baseline situation in which field were abandoned. In addition, the plots with rosemary and lavandin had a lower erosion prevention values (−1 and −44% respectively on average). The provision of forage presented a slight increase for rosemary (0.24 kg m^{-2}) and a decrease of 4.11 kg m^{-2} for lavandin fields.

5. Discussion

5.1. Field measurements

Because of the low percentage of green vegetation cover due to land degradation in the study area, the average levels of the measured ecosystem services supply are low. In addition to land degradation, the study area was seriously affected by a drought period during the field measurement period in 2017. The drought restricted vegetation growth and dried out plants, thus lowering the values of our ecosystem services indicators. Moreover, our required field measurements for estimating ecosystem services could not be carried out correctly within dense thicket, this may have left-skewed our collected data.

Infiltration rates of soils under vegetation were significantly higher than those for bare soil ($p < 0.05$) (Supplementary Materials, Fig. S.6). However, there was a high variability of infiltration rates for vegetation types between plots. The above confirms that infiltration is strongly affected by other factors such as geology and topography (Dunne et al., 1991; Fox et al., 1997; Thompson et al., 2010) and supports our method of considering the average infiltration rates per vegetation type from within the measured plot only. This high variability of the measured levels of infiltration rates per vegetation type between plots could have hindered the ability of Sentinel-2 to detect vegetation covers that have different infiltration rates.

A likely source of error in the field-based estimation of ecosystem services can be found in the small sample size of the allometric equations used for the estimation of climate regulation and presence of native species. There was a degree of vegetation heterogeneity within plots that could have affected the mean values of ecosystem services supply of the plots. An improvement in the accuracy of the ecosystem service measurements could be achieved in future studies by also considering the plant distribution within the plots.

5.2. Capturing ecosystem services with RS data

Our RS-based models indicate that variables derived from Sentinel-2 images can help to estimate the provision of the studied ecosystem services measured in the field, showcasing the opportunity for continuous monitoring of ecosystem services in large areas. The Sentinel-2 indices that best capture the ecosystem services in the study area were IRECI, NDI45, NDWI, MTCI and NDVIre2n. These indices characterize the present vegetation and are based on the red (band 4); red edge (bands 5, 6, 7 and 8A); near-infrared (band 8) and short wave infrared wavelengths (band 11). IRECI showed the best correlations for ecosystem services related with the presence of thicket. Using IRECI as the RS variable for predicting the stratified vegetation cover we obtained an R^2 of 0.81. This outcome is in agreement with the results obtained from the original methodology using Landsat TM in China, where R^2 values of 0.794 (NDVI), 0.805 (MSAVI), 0.692 (DSVI), 0.819 (NDTI) and 0.828 (MSAVI and NDTI) were obtained (Zhongming et al., 2010). Even when several RS indices can predict the StrCV well, their level of accuracy is context specific since the reflectance will change according to the vegetation types, soil characteristics of the study area.

The ecosystem services that were directly connected to the presence of green vegetation (StrVC, fresh and green biomass) were best predicted by the RS-based models since spectral indices indirectly respond to the reflectance of green vegetation on the ground. On the other hand, ecosystem services that related to vegetation more indirectly such as regulation of water flows and climate regulation showed a lower accuracy and more variability in the prediction. Soil under greener vegetation does not necessarily have higher infiltration rates, obstructing the estimation of infiltration through Sentinel-2 indices. High variation in the results could be caused by differences in soil types, soil depths and topography as well as vegetation species, growth and health. Our best model selected to predict regulation of water flows used NDWI,

Table 11
Comparison of provision of climate regulation through aboveground carbon (AGC), presence of native species and available fresh biomass for oil for each paired of plots calculated using RS-based models. The difference between the baseline (non-intervened) and the intervened plots are shown. Red, green and yellow cells indicate decrease, increase and no change on the ES provision respectively in comparison with the baseline management. StrVC: Stratified vegetation cover; VC: vegetation cover; fAGB: fresh aboveground biomass; NA: Not applicable. *Refers to planting dates of spekboom and oil producing plants.

Intervention (management difference)	Baseline management (not intervened)	Intervention lifespan* (months)	Baseline AGC ($g\ m^{-2}$)	Difference in AGC ($g\ m^{-2}$)	Baseline Presence of native species (Spekboom VC %)	Difference in presence of native species (Spekboom VC %)	Baseline fAGB for oil ($g\ m^{-2}$)	Difference in fAGB for oil ($g\ m^{-2}$)
Livestock exclusion	Presence livestock & Not revegetated	360	438	2774	0	2.6	NA	NA
	Spekboom revegetated	110	41	9163	0.5	4.7	NA	NA
		110	1001	−396	0	0	NA	NA
Spekboom revegetated	Presence livestock & Not revegetated	87	383	5050	0.2	−0.2	NA	NA
		52	809	1998	0	0	NA	NA
		48	990	−385	0.3	−0.1	NA	NA
	Abandoned fields & livestock exclusion	70	199	−26	0.9	0.5	NA	NA
		24	502	41	1.9	−0.4	NA	NA
Livestock exclusion & Spekboom revegetated	Presence livestock & Not revegetated	15	2925	−564	0.3	1.1	NA	NA
		107	88	278	1.3	5.3	NA	NA
Rosemary	Abandoned fields & livestock exclusion	10	708	−679	NA	NA	0	171
	Pastures	14	0	10	NA	NA	0	72
		15	0	46	NA	NA	0	257
Lavandin	Abandoned fields & livestock exclusion	2	1822	−1817	NA	NA	0	29
	Pastures	20	0	139	NA	NA	0	639.9
		19	0	74	NA	NA	0	340

which has being used in the past to capture and map vegetation water content (Chen et al., 2005; Gao, 1996; Jackson et al., 2004).

By using a combination of vegetation indices (NDI45 and NDWI) and slope as predicting variables, our model predicted green biomass availability from herbaceous and shrubs cover with a R^2 of 0.84 and a standardized RMSE = 0.1. One challenge was to extract provision of forage within landscapes where not all species were edible but still affected the values of the Sentinel-2 vegetation indices (essential oil plants and trees). In agreement with Martínez-Harms et al. (2016), the integration of slope improved the performance of the RS-models for provision of forage, regulation of water flows, biomass and climate regulation of rosemary and the presence of native species. As shown in Table 9, slope information explained 16 to 36% of the variance not captured by Sentinel-2 indices for climate regulation and biomass of rosemary, provision of forage, regulation of water flows and presence of native trees, so considerably enhancing those models. Our results are in line with R^2 obtained in previous studies for the estimation of total aboveground biomass and aboveground carbon storages (Martínez-Harms et al., 2016); grass biomass (Sibanda et al., 2015) and shrubs and herbaceous biomass obtaining (Glenn et al., 2016).

It was more difficult to find RS models that could predict climate regulation biomass for essential oil production since the herbaceous cover of weeds and cover crops in between the oil crop rows had a stronger impact on the vegetation indices than the essential oil plants. However, we were able to estimate these ecosystem services as an indication of the crop performance. MTCI has been used to estimate gross primary production in wheat with a R^2 of 0.66 (Wu et al., 2009), similar to our model for rosemary, whereas in combination with slope we obtained an R^2 of 0.71. Regarding the lavandin models, to our knowledge there are no other studies that have used the NDVIre2n for aboveground biomass or carbon estimation.

Geolocation uncertainties caused by inaccurate GPS measurements and satellite images positional inaccuracy, and band resampling, cannot be entirely avoided and have potentially negative implications for the ecosystem service models. However, careful field sampling design, in which plots represent the surrounding areas, using multiple pixels to describe a field plot, and using GPS instruments with a high precision, could help to minimize these errors (Lunetta and Lyon, 2004). The spatial resolution of Sentinel-2 images allow to produce ecosystem services maps with a resolution 10 m. However, the current geolocation uncertainty could lead to a mismatch between field and RS locations as such affecting the accuracy of the map.

5.3. Comparison of ecosystem services supplied by different interventions

We found that interventions that included livestock exclusion presented a more consistent positive effect on ecosystem services than revegetation management alone, agreeing with field observations. Also, when the baseline included livestock exclusion, the initial levels and the increase of ecosystem supply after spekboom revegetation were higher. The ecosystem services supply found in plots intervened with a combination of livestock exclusion and revegetation increased with their intervention timespan. On the other hand, we believe that the effect of time was less perceptible in those plots intervened with revegetation alone because the presence of livestock would have a much greater and faster impact on the changes of ecosystem services supply.

The variability in ecosystem services supply among the seven spekboom revegetated pairs of plots can be explained by previous studies that attribute the success of spekboom restoration to a combination of several factors such as soil geology, soil depth, level of initial land degradation, topography, planting methods, rainfall and frost events after the intervention, microclimates, and association with other species (Duker et al., 2015; Mills et al., 2007; Vyver, 2011). In addition, we found that many revegetated areas were not located inside the thicket biome but in the transition area with the Nama-Karoo shrub land, that could have hampered the growth of spekboom mainly by climatic factors, most importantly winter temperatures (Becker et al., 2015; Duker et al., 2015). Regardless of the above, we found that only plots with livestock exclusion, in combination or not with revegetation, showed a positive difference with the baseline management for the studied ecosystem service supply.

It is important to consider the margin of error of the selected models, and interpret the results carefully especially when the differences in ecosystem services provision are very small. For example, in the case of the negative values in the difference of spekboom vegetation cover (%) would suggest very small to no effect rather than a negative impact of the intervention. Significant positive differences on ecosystem services supply after restoration actions in semiarid landscapes such as this study area occur slowly, suggesting that the protection of the still existing local vegetation is of crucial importance for ensuring future provision of ecosystem services. In addition, protection implies lower cost than restoration (Possingham et al., 2015).

Results for ecosystem services related to essential oil production are less conclusive in terms of finding clear differences between different baseline management, with the exception of estimations of carbon stocks, where abandoned agricultural fields as the baseline management have a higher amount of aboveground carbon compared to essential oil production fields. We recommend considering local management information such as planting dates, presence of cover crops, weed management, incidence of pests or diseases and the presence/type of irrigation system in each field to better understand the reasons behind the changes of interventions related to essential oil production. We expect that differences in ecosystem service supply in agricultural fields would be more plot-specific related to management of both the baseline and intervened sites. In agricultural production, management decisions are made daily, and these small changes could drastically affect the provision of ecosystem services. Therefore, conclusions on the effect of these interventions are not absolute since it depends on how they are implemented. Changes in vegetation cover are expected to be faster in oil production fields than in natural vegetation where substantial changes could need decades to occur.

The upscaling of ecosystem service supplies could be accomplished by using accurate land cover maps of the study area and data of the interfered sites. However, the developed models should be only applied in areas with similar topography and similar vegetation types. For example, in this study, models were developed for thicket and we do not recommended their use in fynbos vegetation types unless further field calibration is done. Also, our models did not consider slopes steeper than 18 degrees. Steep slopes generally produce shadow in satellite images, especially in the south and west facing hills (depending on the position of the sun and the time of satellite observation). The shaded areas in the image could lead to errors in the capture of vegetation reflectance and therefore the RS-based ecosystem service estimation would be incorrect. After spatially upscaling the ecosystem services supply of intervened and non-intervened areas, the temporal upscaling could provide insights about when the ecosystem services are provided (Carpenter et al., 2012; de Groot et al., 2010; Serna-Chavez et al., 2014).

6. Conclusions

This study aimed to empirically quantify ecosystem services through Sentinel-2 indices, complemented with soil and terrain data, to improve our understanding of the effects of restoration interventions on ecosystem services. The Sentinel-2 indices that best captured the assessed ecosystem services in the study area are based on bands 4 (red); 5, 6, 7, and 8A (red edge); 8 (near-infrared) and 11 (short wave infrared). The inclusion of slope information improved the RS-based ecosystem service models for provision of forage, regulation of water flows, presence of native trees, and aboveground biomass and carbon stocks of rosemary. The best performing RS based models are based on green vegetation indicators, such as stratified vegetation cover, fresh and green biomass. In contrast, weaker models of ecosystem services are obtained when links between ecosystem services indicators and green vegetation cover are indirect, such as water flow management, climate control, and native trees.

In addition to the effect of different restoration interventions, factors such as initial state of land degradation, planting methods, weather conditions during sensitive vegetation growth stages, soil characteristics, local topography and specific management also affect the degree and speeds of recovery of degraded landscapes. Therefore, the correct inference on whether a restoration is successful or not needs the consideration of these mentioned factors. Nevertheless, livestock exclusion appeared to have a consistent positive role on the levels of supply of the studied ecosystem services.

The presented approach to evaluate the effects of restoration interventions using RS complemented with soil and terrain spatial data, can be extended to wider range of ecosystem services in different contexts and objectives across different landscapes. The approach to quantify ecosystem services using Sentinel-2 vegetation indices is a first step to improve the monitoring and evaluation of restoration interventions. However, for our approach to be useful for long-term monitoring of interventions, the established relationship between ecosystem services and remote sensing indices need additional validation over time.

CRediT authorship contribution statement

Trinidad del Río-Mena: Conceptualization, Methodology, Formal analysis, Investigation, Data curation, Writing - original draft, Visualization. **Louise Willemen:** Conceptualization, Methodology, Supervision, Writing - review & editing, Visualization. **Ghirmay Tesfamariam:** Formal analysis, Investigation, Data curation. **Otto Beukes:** Writing - review & editing. **Andy Nelson:** Methodology, Supervision, 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.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecolind.2020.106182.

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