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Monitoring drylands: The MARAS system

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ABSTRACT

MARAS (Monitoring of Arid and Semiarid Regions) consists of 379 ground monitors in Patagonia, a 624.500 km^2 semiarid area of southern Argentina and Chile. The objective of this paper was to describe the system and analyze four variables of the initial data base. Floristic composition, diversity and cover were analyzed with intercept lines (500 points). Patches (resource-sinks areas) and Interpatches (areas that loose resources) were described using Gap intercept lines (50 m). Eleven Landscape Functional Analysis indicators were recorded in 10 interpatches: Soil stability, Infiltration and Nutrient cycling. Vegetation Cover was 43 ± 2%, Richness 15 ± 7 species/monitor, Interpatch Size 154 ± 134 cm and LFA Stability Index 46 ± 1%. Cover, Richness and Stability maps had bimodal distribution and maximum in S and NE areas, following rainfall gradients. Variability analysis shows that cover estimations are within 5% error at site and regional scales. Graphical analysis of single monitors shows observational biases in interpatch size and LFA Stability index. Richness estimations correlate significantly with α diversity (R^2 = 0.80). Analysis of 5-year change in 115 monitors shows significant reductions in cover and interpatch length, especially N of the region. These base line evaluations enable analysis of future changes that were not possible with multiple techniques and isolated data bases.

1. Introduction

Arid and semiarid lands are distributed over 60 of the world's nations comprising about 40% of the earth's land total area (MEA, 2005). Argentine ecorregions of Andes, Patagonia, Monte and Puna are arid or semiarid (Burkart et al., 2005) and occupy 43% of the territory. In Chile they include ecorregions of Absolute desert, Desertic shrubland, Short Desertic shrubland and Mediterranean and oriental Steppes and Grassland, conforming 29% of the land (Luebert and Pliscoff, 2006). People in these rural areas have livelihoods based on extensive grazing of rain-driven forage production of natural vegetation (Smith and Huigen, 2009) in lands subject to erosion and slow desertification processes. New climatic scenarios, land use conflicts with oil and mining industries and increased grazing due to expanding agriculture that displaces livestock to marginal areas are examples of the future external drivers of vegetation and soil change in arid lands (Viglizzo and Jobbágy, 2010).

Although remote sensing using multitemporal images has improved our monitoring capacity (Fensholt et al., 2012) land-based systems with standardized sampling protocols are still irreplaceable in order to track variations in vegetation cover, biodiversity, biological invasions, local extinctions and physical or chemical soil properties such as carbon content. Instead of tracking annual cycles of "fast" variables such as productivity these ground systems can focus in "slow" variables that determine status of desertification, such as cover, diversity or vegetation patch structure (Reynolds et al., 2007). A great number of ground monitoring initiatives are in place in scientific institutions of arid lands, but they currently use different sampling techniques suited for

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particular vegetation and soil conditions at small spatial scales. Scientific groups tend also to monitor at short time scales and have difficulties to share data, all of which undermines the ability to combine efforts and describe slow changes at wide spatial scales characteristic of desertification processes.

A unified system may enhance collaboration and produce data for different purposes. From the national point of view, it may help fulfill monitoring requirements of different conventions such as UN convention to Combat Desertification (UNCCD) and related conventions, United Nations Framework Convention on Climate Change (UNFCCC), and Convention on Biological Diversity (CBD). From the point of view of producers, they may be used to certify sustainable use of lands for grazing, mining or oil industries. And in scientific research area they may enable collaboration and analysis of large scale ecological processes that are out of reach of actual systems.

Monitoring in Patagonia is needed in view of substantial changes in vegetation and soil that followed domestic herbivore introduction in late XIXth century. Degradation and over-grazing signs were observed very early in this process (Willis and Moreno-Lacalle, 1914; Soriano, 1956a; b), and desertification maps produced in the 1990 decade showed that over 30% of the area was severely desertified (Del Valle et al., 1998). Grazing pressure, the main cause of degradation of arid lands (Soriano and Movia, 1986; Le Houérou, 1996a; DHV-SWEDFOREST, 1998) was high with over 20 M sheep from 1920 to 1980, but stocks fell thereafter, and only about a third of this number remains at present (Mendez Casariego, 2000; SENASA, 2017). This relaxation of stocking rates did not induce nevertheless an increment in productivity, as satellite images indicate a persistent negative trend in vegetation in the last decades (Gaitán et al., 2015; Oliva et al., 2016b). Vegetation recovery may be constrained by permanent changes in soil due to erosion, or by climate as rainfall diminished significantly since 1990–2010 in the NE and temperature increased across the region (Adler et al., 2003). Global climate models indicate further temperature increases of 4–5 °C and reductions of up to 15% of rainfall north-east of the region in the next decades (Stocker et al., 2013). Extreme rainfall events will be more frequent, enhancing superficial runoff and erosion. These changes will probably affect the distribution of plant and animal species by shifting the location of climates to which they are adapted (Meynecke, 2004; Penman et al., 2010; Svenning and Sandel, 2013). Reduced productivity, increased erosion and more biological invasions will also alter livelihoods of people that depend directly or indirectly on biodiversity such as the Patagonian sheep breeders, but up to date no regional, systematic monitoring system has been available to track these changes and interpret possible future trends.

Some examples of regional monitoring efforts are found in West Australia's WARMS system (Watson et al., 2007), AUS plots (White et al., 2012) and Jornada Monitoring system that is used by different agencies in US (Herrick et al., 2005). EPES or Biodesert systems (Maestre et al., 2012) are, on the other hand, examples of one-time large-scale sampling of vegetation and soil using common methodology.

MARAS (Spanish acronym for Environmental Monitoring of Arid and Semiarid Regions) is a monitoring system developed by INTA (National Institute of Agricultural Technology), of Argentina (Oliva et al. 2006, 2016a) between 2004 and 2008 and set up within GEF Patagonia PNUD ARG 07/G35 project between 2008 and 2015. It was also applied in southern Chile by the INIA (National Institute of Agricultural Research) between 2014 and 2015. It has applied a single protocol to sample a wide range of vegetation and soil types, a strategy that is not common: other monitoring systems such as WARMS (Watson, 1998) prescribe different techniques for shrublands and grasslands, and USDA Jornada's Experimental Range monitoring manual (Herrick et al., 2005) offers a suite of techniques to match different situations. Unified techniques allow work teams and institutions to share data, and variables may be mapped at regional scale without previous meta-analysis or standardization that are required in less standardized monitoring systems such as the Australian ACRIS (Bastin et al., 2009). As a drawback, a fixed suite of techniques may provide estimations with different errors when applied to different vegetation types. Given that monitoring teams are likely to change, they should minimize observer bias (Watson et al., 2007). They also should carefully balance sampling effort (number of sites, number of points or plots at site scale and visitation time) in relation to required precision in estimates, because of cost involved in placing trained technicians in the field.

Installation phase of MARAS finalized by 2017, and monitors reaching 5-year age are currently being reassessed. The objective of this paper is to present this system that may be useful in other arid and semiarid areas of the world and analyze four main attributes of an initial database in order to establish errors associated with the prescribed sampling effort at site scale and the minimum sampling effort that produce biozone estimations within an acceptable error. This analysis is the key to clarify errors associated to a fixed monitoring protocol at site scale, and to determine if the main biozones are adequately sampled or additional monitors are needed. We will also analyze regional changes in these variables using a subset of monitors that have been reassessed in this on-going effort and show the main changes that have been observed.

2. Material and methods

Study area was 624,500 km^2 of Argentine and Chile extra-Andean Patagonia, stratified in 11 biozones based on climate, vegetation and soils by Bran et al. (2005). Annual rainfall is mostly < 200 mm with an Aridity Index (rainfall/potential evapotranspiration relation) between 0.46 and 0.11 (Paruelo et al., 1998) that classifies mostly into arid climate (Le Houérou, 1996b).

379 monitors were installed in 8 of the 11 Biozones of the Bran et al. (2005) map with a higher density in most productive areas (Fig. 1). The area was further divided into 31 Great Landscape Units (López et al., 2005). Monitors were installed in two dominant Landscape Units of each biozone by five teams of 3–4 trained technicians. Sites were uniform areas with dominant vegetation type and representative management (in sheep farms they were typically ewe paddocks), distant at least 500 m from water sources and roads. Wetlands or other azonal vegetation types were not sampled. Basic procedure was described in detail in manuals (Oliva et al., 2011), included in Supplementary Material of this paper in Spanish and English. Layout was similar to WARMS system, developed in West Australia (Holm, 1998). A photographic pole (Fig. 2) was fixed and a 72-m central line was laid following the main resource flux direction (wind or water flow direction). Three poles were fixed at 8.5 m along this line, separated perpendicularly 2.5 m and two additional sets of three poles were set at 13.5 and 72 m separated by 6.5 m. Photos were obtained from photographic pole at 2 m height (Fig. 3) and positions of pole 1 and 9 were registered with GPS. Three 50-m tapes were placed, two for vegetation and one for patch structure sampling and Landscape Functional Analysis sampling.

Soil cover was estimated by point sampling (Daget and Poissonet, 1971) using 500 points at 20-cm intervals along lines 1 and 2. Plants were identified at least to genus. Non-vegetated points were classified in litter cover, bare soil, rock, cryptogams and standing dead. Patch structure was analyzed using gap-intercept method described in Herrick et al. (2005) along the soil line. Patches were defined as areas > 10 cm that retained resources. They were mostly live plants but occasionally consisted of standing dead or decomposing litter fixed to the soil. Interpatches were areas > 5 cm that lost resources, and mostly consisting of bare soil or desert pavement areas. A minimum of 25 and a maximum of 50 patch-interpatch pairs were recorded. Landscape Functional Analysis (LFA) was estimated in plots outlined over the first ten interpatches that exceeded 40 cm in length along the soil line. This was a modification of Tongway and Hindley's (2004) methodology, where plots were systematically placed at intervals along a line. Eleven

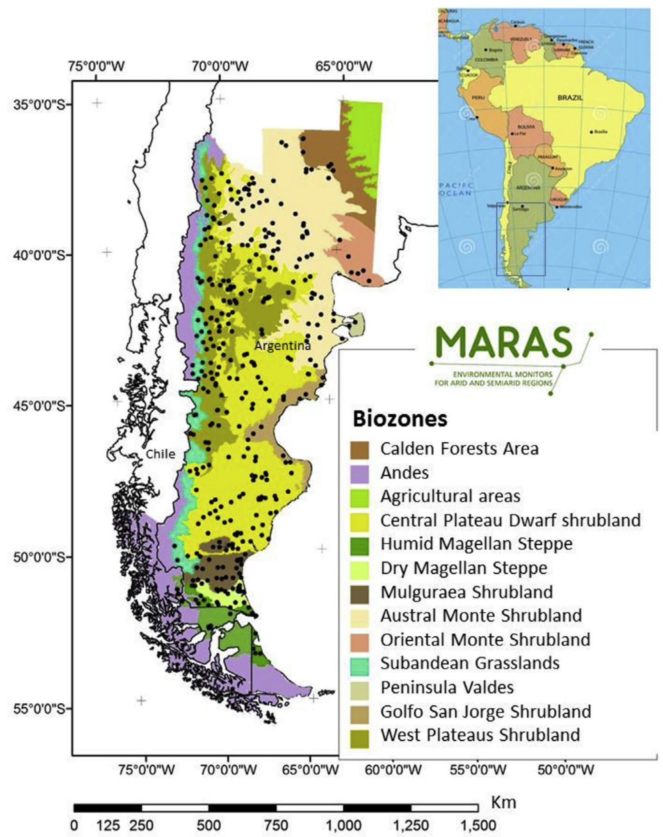

Fig. 1. Biozones of Patagonia (Bran et al., 2005) and Chile (Luebert and Pliscoff, 2006). Dots represent MARAS monitors.

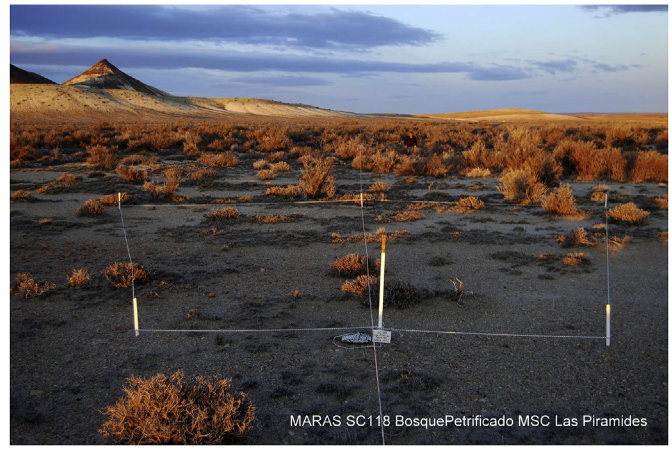

Fig. 3. MARAS monitor example in Santa Cruz (photograph obtained from pole 1).

indicators were visually rated using a scale proposed by Tongway (1994) that was modified to suit regional soil and vegetation. Three LFA indexes were estimated: Stability, Infiltration and Nutrient Cycling. In this paper we analyze stability index, an estimation of soil resistance to erosive forces that results from adding the following of indicators (1) Aerial cover for rain interception, (3) Litter cover, origin and degree of incorporation, (4) Cryptogram cover, (6) Deposited materials, (8) Soil crust type and degree to which it was disturbed, (9) Surface crust resistance and (10) Slake test, time that soil aggregates retained integrity in water. Sum of ratings for these indicators was divided by 30, the maximum possible sum and expressed as percentage. Composite superficial (0–10 cm depth) soil samples were obtained from patch and interpatch areas and tested for pH, conductivity, organic carbon, N, P and texture. Soil data was not further analyzed in this paper.

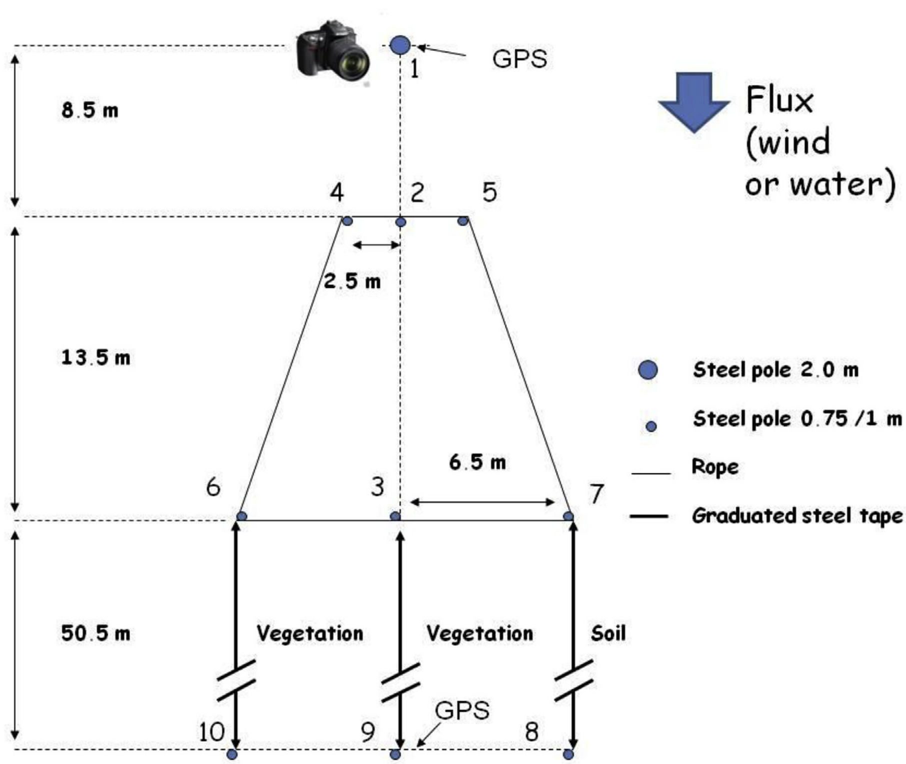

Fig. 2. Scheme of a MARAS monitor.

2.1. Data analysis

Four variables were analyzed: Vegetation cover (%), Richness (nº species detected in point-intercept lines), Interpatch length (cm) and Stability LFA Index (%). Differences between biozone means were tested using ANOVA with Duncan contrasts using Infostat (Di Rienzo et al., 2011). Interpatch length was normalized by log-transformation. Gradient maps were obtained for each variable using Arc GIs 10 interpolation tools.

In order to analyze influence of aridity, a main driver of cover, diversity and other attributes of arid and semi-arid lands (Noy-Meir, 1973), linear regressions of each variable with aridity index (AI), the precipitation/potential evapotranspiration were estimated. AI of each site was obtained from Global Potential Evapotranspiration database (Zomer et al., 2008), based on interpolations provided by WorldClim (Hijmans et al., 2005).

Point line intercept techniques do not reach easily a minimum area required to identify all species in a plot or α-diversity (MacArthur, 1965). In order to test the degree to which 500 point lines underestimated this parameter a subset of 160 monitors that had a reliable estimation of total number of species obtained by thorough examination of the 50 × 30 m plot was used to correlate MARAS's richness estimations with total species count.

Variability at different scales for each variable was analyzed in order to determine the number of monitors and associated error at two scales:

(1) Plot: The effect of increasing number of samples at a plot scale (intercept points, interpatches, or LFA plots) was studied graphically using diagrams of sequential sampling (Greig-Smith, 1983) of single monitors that were assessed yearly between 2012 and 2015. Means were drawn using increasing sampling efforts and lines were analyzed in order to determine when they smoothed out. A second, quantitative approach that incorporated different vegetation types was used to analyze vegetation cover based on a sample of monitors of each biozone using minimum sample size equation from Elzinga et al. (1998):

$$n = \frac{(Z_{\alpha/2})^2(\sigma)^2}{(E)^2} \tag{1}$$

Where.

n = number of samples
$Z_{\alpha/2}$ = False-change Type I error rate
σ Standard deviation
E = Error in absolute terms.

σ (intra plot standard deviation) was estimated in 5 monitors randomly selected for each Biozone. The standard deviation for cover estimation from point intercepts was obtained by dividing 50-m lines into 10 subsamples with 50 points each (n = 10). False-change Type I error rate $Z_{\alpha/2}$ was set at 1.96 (0.05 probability) and error in 5%.

(2) Regional: Inter-plot differences within Biozones arise due to climatic or soil heterogeneity and differences in management. Minimum sampling effort was estimated based on variability of vegetation cover in each Biozone using Equation (1). Number of monitors necessary to achieve a one-time estimation of a Biozone within 5% error was also estimated.

Analysis of change was performed in a subset of 115 monitors that had been reassessed after a period of 5–7 years by 2017. Cover, interpatch length, species richness and stability index differences were analyzed using paired t-tests of Infostat software (Di Rienzo et al., 2011).

3. Results

Mean vegetation cover was close to 43 ± 2% (Table 1) and ranged from 98 to 7%. Richness at site level was low, with a mean of 15 ± 7 species per plot, ranging from 40 to 2 species. A total number of 472 species were registered, and this represents a third of the species list for extra Andean Patagonia (Nicora, 1978), that includes wetlands and other special areas not sampled with MARAS system. Mean interpatch length was 154 ± 134 cm, but it ranged from 5 cm to 673 cm (Table 1). LFA Stability index was 46 ± 1% and ranged between 18 and 89%. Coefficients of variation of cover and richness were close to 40%, interpatch length was more variable with a CV of 87% and Stability index showed lowest CV with 29%.

Biozones (Map in Fig. 1) showed significant differences (Table 1): Tall *Larrea* shrublands of Austral Monte (Leon et al., 1998) had lowest vegetation cover and richness, with large interpatches exceeding 3 m long and low stability index. Other shrublands such as Golfo San Jorge, and dwarf *Nassauvia* semi deserts in Central District or mixed grass-shrub steppes of West Plateaus had smaller interpatches (1–1.5 m), low cover (< 50%) and richness (< 16 spp./plot). *Mulguraea* shrublands in the south were distinct because of their higher vegetation cover (60%) and richness (20 spp./plot) and smaller interpatches (42 cm). Communities dominated by *Festuca* tussock-grass in Magellan steppes showed the highest mean vegetation cover over 60% and richness > 30 spp., and interpatches smaller than 30 cm. LFA Stability index showed higher values in Humid Magellan Steppe mainly due to high cover and abundant litter in the soil surface and minimum values in Austral Monte shrubland and Central District.

Cover and Species richness correlate positively and significantly with Aridity Index: Cover = 26.7 + 59*AI R^2 = 0.16 $P < 0.01$ and Richness = 9.9 + 19*AI R^2 = 0.12 $P < 0.01$. Gradient maps (Fig. 4) show higher cover in the south and lower in the region's arid center, increasing towards the Andes and the NE. Gradients of richness are similar, as monitors in the Magellan Steppes detected 30 or more species, while those in Central Plateaus and northern Monte areas identified less than 10.

An inverse pattern was observed for Interpatch length that correlated negatively with Aridity Index: Interpatch = 258.7–407*AI R^2 = 0.12 $P < 0.01$. Long interpatches that exceeded 300 cm were registered in northern Monte shrublands, while small 24 cm ones were characteristic of southern grasslands (Fig. 4).

Stability LFA index correlation with Aridity Index was positive but not as strong: Stability = 39.6 + 25.7*AI R^2 = 0.08 $P < 0.01$. Main gradients in stability index maps are similar to those of cover map, as southern high cover locations are more stable.

3.1. Minimum number of samples at site level

Graphic analysis of sample size effect on precision of estimations for a single monitor in a dwarf shrubland of the Central District (Fig. 5) show that vegetation cover estimates smoothed off at approximately 400 intercept points, yielding 2% yearly differences that reflect high precision of the estimates. Interpatch length curves stabilized at approximately N = 15 interpatches (Fig. 5), but annual mean estimations in this case differed about 30%.

LFA estimations showed a different pattern in relation to sampling effort. Lines are smooth from the beginning of the plots and did not improve with increased sampling (Fig. 5). Inter-year differences in this case were 17%. Species Richness did not stabilize with increasing sampling effort, but rather increased in a linear manner as new points were sampled. Annual evaluations differed in a single species (7% of the mean).

Richness detected using line intercept points correlated closely (R^2 = 0.815 $P < 0.01$) with total number of species (Fig. 6).

A second approach to explore variability at Site level (Table 2) was to analyze standard deviations of successive subsets of intercept points and estimating minimum number of samples using Equation (1). This analysis shows that a sampling effort of 9 segments of 50 points (450 intercept points) was enough to estimate site Vegetation cover within

Table 1
Type of vegetation and mean values for vegetation cover, species richness, interpatch length and stability index in eight Biozones (EA) of Patagonia as shown in Fig. 1. Standard deviation is shown in parenthesis. Letters indicate significant differences for Duncan contrasts (P < 0.05).

Biozone	Vegetation	Number of monitors	Vegetation cover			Species richness			Interpatch length			Stability index		
			%			N°			*cm*			%		
Austral Monte Shrubland	Tall *Larrea* shrubland	70	32	a	*(13)*	12	A	*(5)*	305	e	*(146)*	44	a	*(10)*
Central Plateau Dwarf Shrubland	Dwarf *Nassauvia* shrubland	127	36	ab	*(12)*	13	A	*(4)*	148	cd	*(123)*	43	a	*(10)*
Dry Magellan Steppe	*Festuca* steppe	23	66	e	*(12)*	24	C	*(4)*	31	ab	*(10)*	55	bc	*(9)*
Golfo San Jorge Region	Tall mixed shrubland	11	42	abc	*(13)*	16	ab	*(4)*	117	cd	*(58)*	47	ab	*(9)*
Humid Magellan Steppe	*Festuca* steppe	22	81	f	*(18)*	30	D	*(8)*	24	a	*(17)*	61	c	*(16)*
Mulguraea Shrubland	Medium *Mulguraea* shrubland	30	60	de	*(9)*	20	B	*(5)*	42	b	*(12)*	49	ab	*(7)*
Subandean Grasslands	*Festuca* Grassland	24	47	bcd	*(14)*	15	A	*(7)*	105	c	*(63)*	49	ab	*(10)*
West Plateaus Shrubland	*Stipa - Mulinum mixed shrubland*	72	35	a	*(11)*	12	A	*(4)*	145	cd	*(75)*	43	a	*(9)*
Total		379	43		*(18)*	15		*(7)*	154		*(134)*	46		*(11)*

5% error. Analysis by biozone indicated nevertheless that segments of intercept lines differ markedly in shrublands such as the Austral Monte and the Mulguraea shrubland, where over 700 points would be necessary to reach this precision.

At a regional scale, sample size estimation indicates that the existing 379 monitors provide an estimation of vegetation cover within the 5% error target (Table 2). Based on the number of monitors already installed (Table 1), only biozones of Golfo San Jorge Shrublands and Subandean grasslands would need additional sampling.

3.2. Monitoring change

Monitors that had been reassessed show significant reductions in Vegetation Cover and Interpatch Length (Table 3). Vegetation Cover was lost mainly in Humid Magellan Steppe, and Central Plateau. Subandean Grassland showed a non-significant decrease. Austral Monte Shrubland, on the contrary, increased cover in this period. Interpatches were smaller by the end of 5-year reassessment periods in Austral Monte, Central Plateau and West Plateaus Shrublands. Species richness and LFA stability index do not vary significantly across the region.

4. Discussion

MARAS network covering 624,500 km^2 is one of the few examples of an ongoing monitoring effort with such a wide geographical distribution (Fig. 1) and different vegetation and ecological situations. Standardized methods and sampling efforts enabled us to draw base line maps at regional scale that will be key to analyze future changes (Fig. 4). As expected, maps reflect aridity as the main control of vegetation composition and structure: Western Pacific Ocean winds lose humidity when they go over the Andes and generate a strong west-east rainfall gradient. Rain shadow is not as marked in the south, where Andes are lower and allow more rainfall to reach southern Magellan Strait area. Precipitation also increases NE of the region due to mid-latitude rain systems (Paruelo et al., 1998). Center plateaus are arid areas dominated by dwarf shrubs of the Central District and *Larrea* shrublands in Austral Monte, with low vegetation cover and few species (Fig. 4). Large interpatches > 3 m probably generate marked source-sink dynamics of erosion and deposition that explain low LFA stability values. Vegetation covers almost completely southern tussock grasslands, where rainfall is slightly higher and temperature and evapotranspiration are lower (Faggi, 1985). In this area, highly diverse grasslands with extremely small interpatches are found. Soil surface is covered there with abundant litter and cryptogams that increase LFA Stability index. Local hotspots of diversity, probably related to topography are observed in this map (Fig. 4).

The cost of a monitoring system is defined to some extent by sampling effort in each site, and our initial data set allows for variability analysis in order to assess to sampling effort and relate it to required precision. MARAS protocol, with 500 point intercepts for cover and species richness, 50 interpatches along gap intercept line, and 10 LFA plots requires about 3 h of well-skilled, 4-technician group to do the task. Would it be possible to reduce this effort without compromising the precision of estimates? Variability at local scale clarifies this point:

- Vegetation cover estimations using 500 points converge within a 2% difference in subsequent readings in a low-change environment (Fig. 5). Using less than 300 intercept points, subsequent estimations vary widely, close to 14%. Results of extending this single-monitor analysis to a regional sample of monitors indicate that the number of intercept points needed to provide estimations within 5% error varies between 100 and 800 (Table 2). Overall mean of 450 points is close to the prescribed effort, but using a fixed protocol implies a higher error (close to 7%) when applied in shrublands with coarse grain of heterogeneity.
- A similar pattern of high variability with using less than the prescribed 50 interpatches is observed in Fig. 5. Estimations were initially not affected but they start to vary widely when a minimum sampling effort threshold of n = 15 interpatches is used.
- Estimations of LFA stability index using less than 10-plots did not to vary so widely, and sampling effort lines were smooth (Fig. 5). Three consecutive readings showed nevertheless wide differences, and given that cover and interpatch readings show minimum changes, they were likely explained by different appreciation of subjective LFA indicators. Using a smaller number of plots, and focusing future efforts in training and cross-standardization to reduce observational bias is an alternative in this technique.
- Species richness showed a different type of curve that did not stabilize but rather increased in a linear manner as new points were sampled. Vegetation communities show a few dominant species and a great number of rare ones. Increased point intercept sampling delivers higher numbers for richness until a minimum sample area is attained (Magurran, 2004). In this way Richness estimated as species count along lines underestimated plot α-diversity, but correlated significantly and linearly to total number of species estimated by examination of the whole plots (Fig. 6).

It is evident that a one-size-fits all strategy of MARAS protocol has some limitations, and that some sampling efforts should be increased and others could be relaxed in order to meet error targets. Nevertheless, changing techniques and sampling efforts at this time in order to improve site estimations would complicate future comparisons (Watson, 1998). Keeping to present protocols and performing training and cross standardization of observation teams would be best. Analysis also indicates that small changes will be harder to detect in the future, especially where shrubs generate coarse patch-interpatch structures.

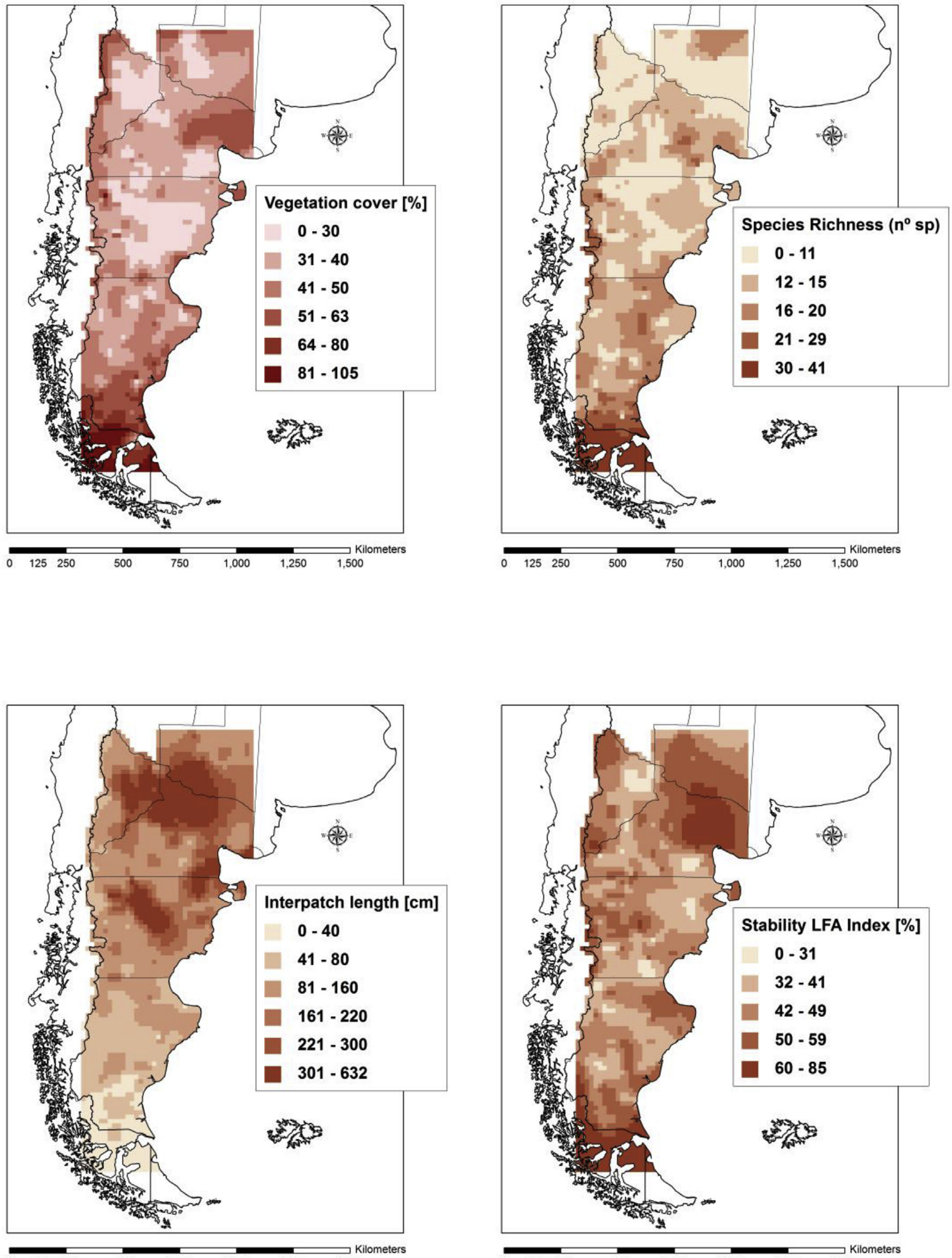

Fig. 4. Vegetation cover (%) Species Richness (n°species), Interpatch length (cm) and stability index (%) in Patagonia.

Number of monitors that have to be reassessed periodically is a second item that defines cost of any monitoring system. Should future efforts concentrate in installing more monitors or should they be focused on resampling the existing network? Minimum sampling effort analysis based on between-monitor variability that arises with gradients in soil, vegetation and climate within biozones, indicates that the actual density is enough to estimate Cover (Table 3) with a 5% error. Except in smaller and more variable biozones such as Golfo San Jorge Shrublands and Subandean grasslands, future efforts would be better directed to resampling.

Will future changes in cover be detectable? Vegetation of this cool semi desert, changes slowly in relation to other rangelands, probably because it is based almost exclusively in perennials with only 3% of ephemeral cover as a regional mean. An analysis of changes registered in about a third of monitors that had been reassessed after a 5–7 year period by 2017 detected significant changes at regional scale (Fig. 7):

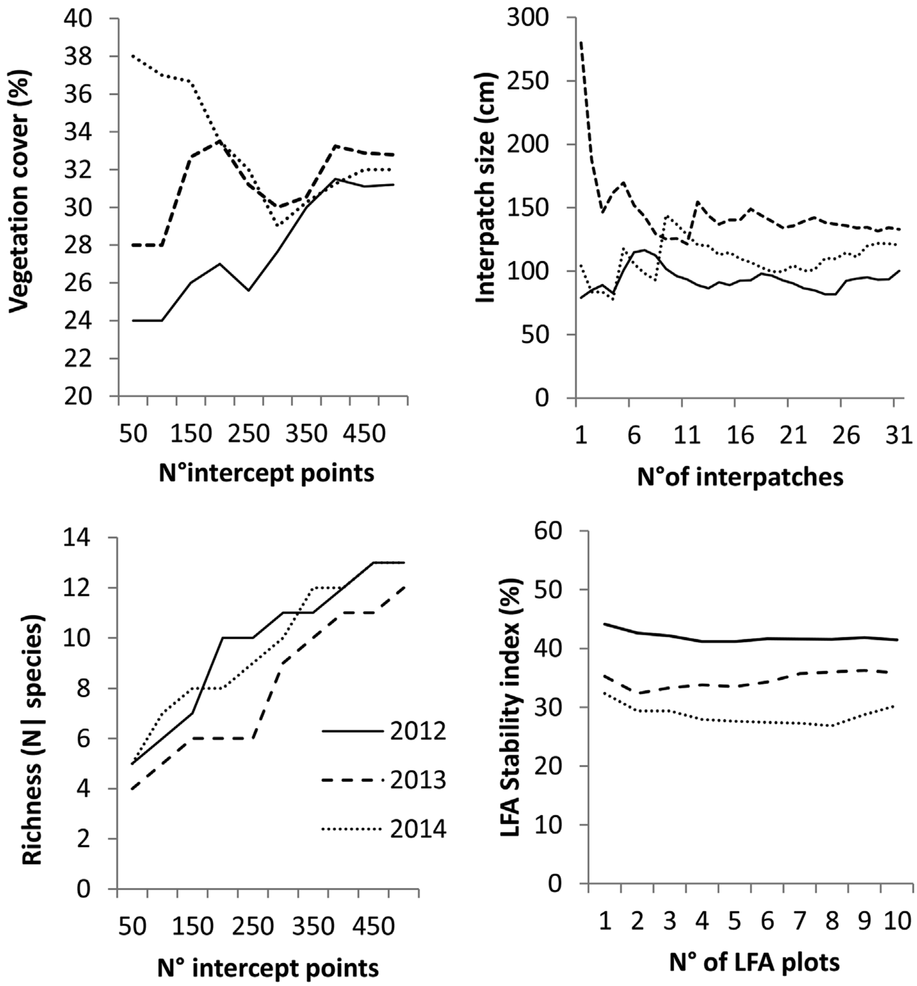

Fig. 5. Three successive estimations of cover, species richness, interpatch length and LFA stability index using increasing sampling effort (accumulated intercept points, interpatches or LFA plot) for the single monitor SC92 in a Dwarf Shrub Steppe of the Central District.

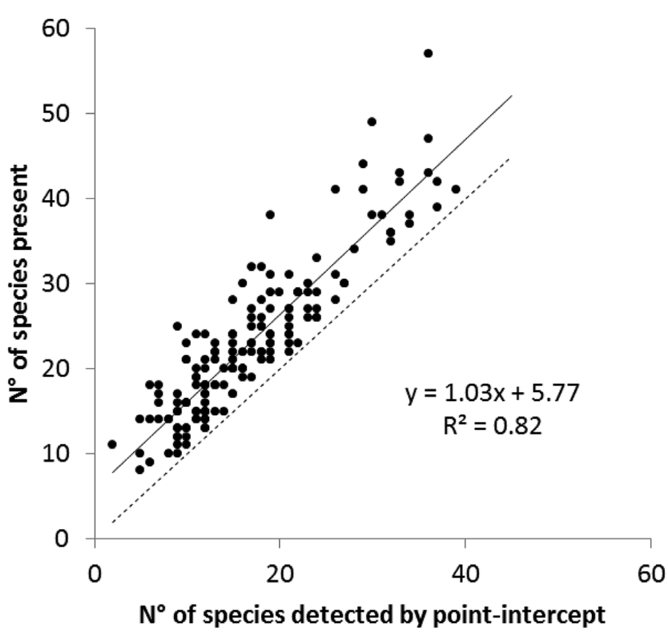

Fig. 6. Species detected by the 500-point line intercept method in relation to total number of species present in the plot for a sample of 160 MARAS monitors. The dotted line represents 1:1 relation.

Table 2
Minimum number of samples to estimate vegetation cover (error ± 5%), at site scale (5 monitors per biozone) and regional scale in the main Biozones of Patagonia using Equation (1).

Biozones	Local scale	Regional scale
	Total Nº intercept points	N° monitors per biozone
Austral Monte Shrubland	700	15
Central District	550	16
Dry Magellan Steppe	350	21
Golfo San Jorge shrubland	550	26
Humid Magellan Steppe	150	21
Mulguraea shrubland	800	12
Subandean grasslands	350	31
West Plateaus shrublands	100	12
Total general	450	220

Vegetation Cover diminishes and interpatches (mainly bare soil areas) grow smaller simultaneously. Change is concentrated in the North-west, where cover drops of 10% and 50-cm interpatch size reductions were observed. Both tendencies seem contradictory, but are probably explained by fragmentation of vegetation patches that give way to new, small interpatches. Persistent drought in 2000–2016 period (Garreaud, 2018), and increasing temperatures particularly in northern Patagonia may explain these preliminary observed changes. Increased statistical power and geographical coverage can be expected when the 379-monitor resampling is finished by 2019, and these tendencies will be

Table 3
Number of MARAS monitors that have been reassessed after 5-year period and differences in the values of total vegetation cover, species richness, interpatch length and LFA stability index. Asterisks indicate significant differences between dates in a paired T –test.

Biozone	Number of reassessed monitors	Vegetation cover		Species richness		Interpatch length		Stability index	
		%		N°		cm		%	
Austral Monte Shrubland	9	6.4	*	1.67	ns	−88.0	*	1.1	ns
Central Plateau	41	−2.7	*	−0.05	ns	−44.3	*	0.1	ns
Dry Magellan Steppe	6	−0.4	ns	−1.17	ns	−1.5	ns	−3.5	ns
Golfo San Jorge Region	9	−0.6	ns	0.33	ns	−13.3	ns	−2.6	ns
Humid Magellan Steppe	4	−7.5	*	2.00	ns	1.2	ns		
Mulguraea Shrubland	9	−1.4	ns	−1.11	ns	4.3	ns	−0.5	ns
Subandean Grasslands	11	−4.7	ns	−1.73	ns	21.1	ns	−0.8	ns
West Plateaus Shrubland	26	−2.9	ns	−0.16	ns	−5.9	ns	−3.8	*
Total	115	−2.0	*	−0.14	ns	−22.5	*	−1.3	ns

then fully analyzed in relation to climate and management.

Information available in MARAS's data base exceeds the small subset of variables analyzed in this paper. Future analysis may, for instance, investigate relationships between diversity and ecosystem function (Halloy and Barratt, 2007), alone or combined with remotely sensed data (Gaitán et al. 2013, 2014; Gaitan et al., 2014). Patterns of change of patch sizes may be are related to disturbances, as they deviate from typical distributions under grazing pressure (Kéfi et al., 2007; Liu et al., 2015). Soil samples will enable an analysis of carbon distribution and changes in carbon stocks that are of special interest for three United Nations Conventions (Cowie et al., 2011). In this way, MARAS offers a structure to perform sampling, to store and share information between scientists of different countries and institutions. Its data base structure and coding will be available for other research teams to use. The system seems to have potential to detect change of biophysical traits of arid lands with a detail and precision that was not possible with multiple techniques and isolated data bases.

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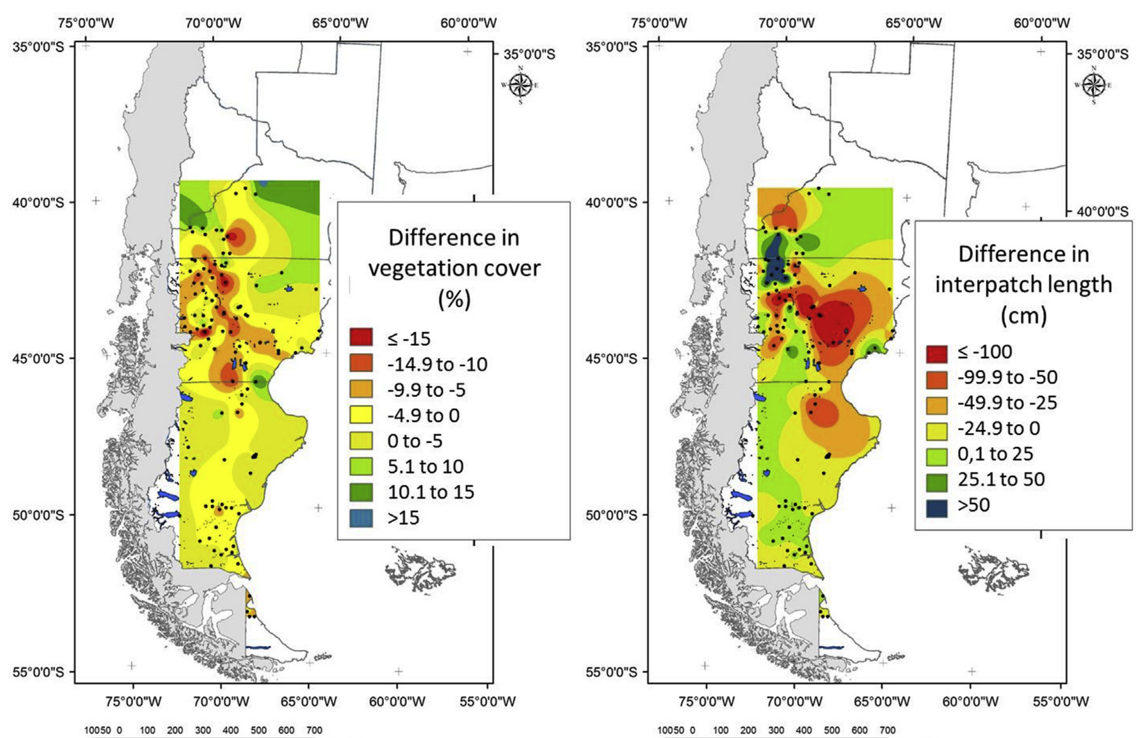

Fig. 7. Maps of change of vegetation cover and interpatch length based on a 115 monitor-sample reassessed after 5–7 year period.

properties, and many of them took part in installation process.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jaridenv.2018.10.004.

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