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Regional forage production assessment in arid and semi-arid rangelands – A step towards social–ecological analysis

M.H. Easdale[a,*], M.R. Aguiar[b]

[a] Instituto Nacional de Tecnología Agropecuaria (INTA) – Estación Experimental Agropecuaria Bariloche, Área de Desarrollo Rural, Grupo de Investigación en Sistemas de Producción y Territorios, Modesta Victoria 4450, (8400) San Carlos de Bariloche, Río Negro, Argentina
[b] Cátedra de Ecología – IFEVA, Facultad de Agronomía, Universidad de Buenos Aires – CONICET, Av. San Martín 4453, C1417DSE, Ciudad de Buenos Aires, Argentina

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A B S T R A C T

Many of the complex issues worldwide regarding environmental management and sustainable development require integrating the social and natural sciences. Nevertheless, while theoretical discussions have been increasingly developed, operative issues are still major barriers to integrated social–ecological analysis. The aim of this paper was to assess regional forage production in semi-arid rangelands as a key feature in social-ecological analysis, by using human organizational units (i.e. counties). We used these state-administrative units to explore demographic and farming indicators in order to address socio-productive implications of different regional forage production dynamics. We studied the forage spatial and temporal dynamics in two different large ecological regions: Monte and Patagonia, under a single administrative unit (i.e. province). Since forage production estimations in arid rangelands are not trivial, we tested two different methods. We found that inter-annual variability in forage production explained the main differences between regions. At a regional level, zones with higher temporal variability in forage production registered less rural residents and farm numbers, but inverse situations were registered at sub-regional scales. We found a non-linear relationship between forage production variability and rural population density. We proposed differentiated policy recommendations regarding rangeland management and animal husbandry, considering both the social and ecological contexts.

1. Introduction

Integrating the social and natural sciences is required in order to attain sustainable development in arid and semiarid lands (Reynolds et al., 2007). Nevertheless, while theoretical discussions in the fields of socio-ecological systems have been increasingly developed, operative issues, such as response variable selection, data resolution and unified methodological proposals are still major barriers to integrated understanding (Cooke et al., 2009; Folke, 2006). In general, applied ecological journals tackle ecological issues. Yet, interactions with the social component are marginally mentioned or considered. In contrast, theoretical or conceptual papers in ecological journals and books increasingly include and discuss both components (e.g. Aronson et al., 2007; Berkes and Folke, 1998; Chapin and Whiteman, 1998; Chapin et al., 2009; Gunderson and Holling, 2002; Reynolds et al., 2007). However, a frequent and concrete problem is that biophysical and socio-economic data, and analysis, have different temporal and spatial scales (Giampietro, 1999). To avoid scale-driven mismatches between biophysical and socio-economic data and to generate sound inferences, data must be assembled into a single and comparable scale (Cumming et al., 2006; Prince, 2002).

Extensive livestock production is the main agricultural activity in arid and semiarid regions of the world. Husbandry relies on primary production dynamics and therefore climate has an important influence on the structure and function of rangeland ecosystems, mainly through rainfall dynamics (Illius and O'Connor, 1999; Oesterheld et al., 1999; Paruelo et al., 1998). However, livestock production (either commercial or subsistence) is also controlled by socio-economic drivers (Reynolds and Stafford Smith, 2002; Walker and Salt, 2006). Forage production is a key ecosystem service, and determines the proportion of primary production that is potentially appropriated by human in the local ecosystem (i.e. through livestock production), with influence on the economic and therefore social conditions at a regional and farm level. Forage production also is very sensible to degradation induced by

* Corresponding author. Tel.: +54 2944 422731.
E-mail address: measdale@bariloche.inta.gov.ar (M.H. Easdale).

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inadequate management (Aguiar et al., 1996). In order to move forward to fully achieve the linkage between biophysical and socio-economic perspectives, we proposed that forage production estimations should be assessed with human organizational units (rather than purely ecological).

Two attributes of forage dynamics are particularly important for livestock management: i) average forage production and ii) temporal variability (either deterministic [such as seasonal] or stochastic). Average forage production (e.g. kg DM ha^{-1} yr^{-1} [DM: Dry Matter]), as a reference value, determines the first managerial decision with biophysical and socio-economic consequences: stocking rate (Holecheck, 1988). On the other hand, temporal variability promotes extreme and unpredictable results (Illius and O'Connor, 1999). There is a strong debate in applied ecology regarding equilibrium and non-equilibrium models, with different implications on rangeland management (Briske et al., 2003). In general, stochastic environments are associated to production risk and vulnerability (Cacho et al., 1999; Reynolds et al., 2007), and management might be more conservative than under deterministic conditions (Díaz-Solís et al., 2009; Higgins et al., 2007; Illius et al., 1998). However, reducing stocking rates creates a conflict since it affects farmers' income in a context of overall low-income, very common in rangeland-based husbandry (e.g. Easdale and Rosso, 2010). Hence, forage production (i.e. mean and temporal variation) set limits to policy design and technological development in arid and semiarid regions.

The aim of this paper was to assess regional forage production in arid and semiarid rangelands, by using spatially-defined human organizational units (i.e. counties). We used these state-administrative units to explore demographic and farming indicators as a step towards social-ecological integration, in order to address socio-productive implications of different regional forage production dynamics. We used county level information since: i) it is a scale for which biophysical and social variables can be obtained, and ii) these administrative units are used for political agency and intervention (e.g. agricultural emergencies), therefore this complementary data may bring closer scientific information to policy design. Since regional forage production estimations are not trivial in arid and semiarid rangelands, we applied two methods that use remote sensing data (i.e. spectral index), but differ in regional ecological knowledge needed beforehand. We worked with the animal husbandry system dominant in a territory that includes two different ecological regions but is under one single political and administrative unit equivalent to the State or Province, in order to reduce the effect of other drivers of the socio-economic domain (e.g. legal and regulatory norms, market issues). Particularly, we studied the spatial and temporal dynamics of forage production at a regional scale, which was complemented by selected key socio-economic variables that described the structure of the human system.

2. Methods

2.1. Study area

The study area is located in Río Negro province, Argentina (Fig. 1) and includes two ecological regions: Monte (84,198 km^2) and Patagonia (70,075 km^2) (Bran et al., 2000). The regions differ in many biophysical and agricultural features (Table 1).

2.2. Remote sensing data processing and forage production estimations

We estimated forage production from remote sensing data following previous works (Jobbágy et al., 2002; Paruelo and Lauenroth, 1995; Paruelo et al., 1998, 2004). We used the Normal Difference Vegetation Index (NDVI = (NIR − R)/(NIR + R), where R and NIR are the reflectance in the red and the near-infrared portions of the electromagnetic spectrum, respectively). We used a subset of the data collected by Fabricante et al. (2009). They

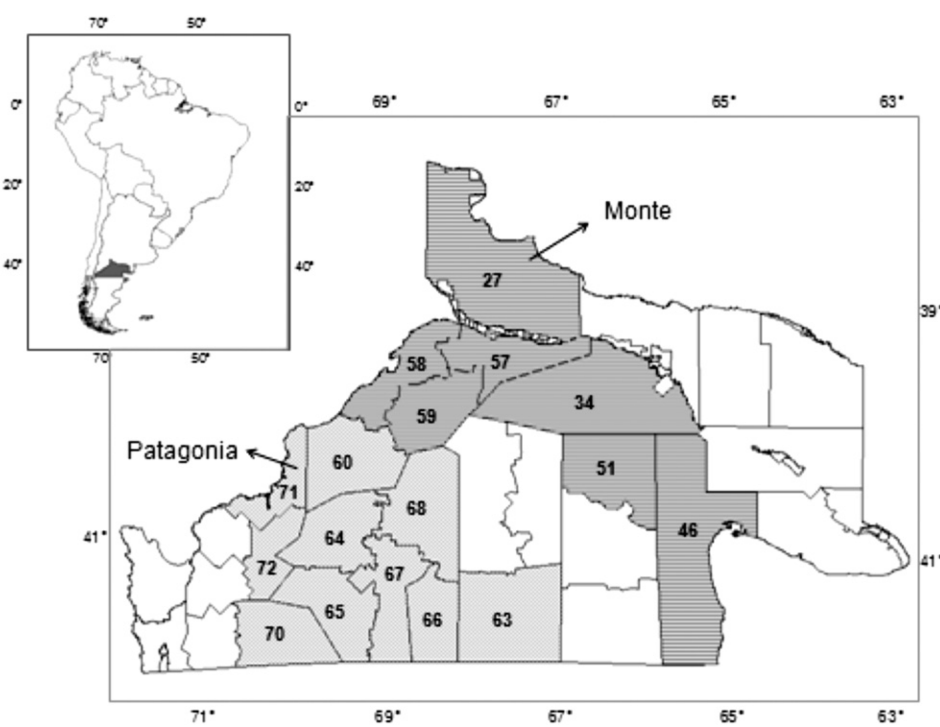

Fig. 1. Study area. Río Negro province, Northern Patagonia, Argentina. Selected counties for Patagonia and Monte ecological regions, respectively. Counties are identified with the numbers utilized by INDEC (2002).

Table 1
Biophysical and agricultural features of Monte and Patagonia ecological regions, Río Negro province, Argentina.

Features	Monte	Patagonia	References
Total area	84,198 km^2	70,075 km^2	Bran et al., 2000
Relief	Sedimentary plateaus and low plains	Hills and basaltic plateaus	Bran et al., 2000
Climate	Mean annual precipitation ~200 mm, distributed in the year. Mean annual temperature ~ 13 °C	Mediterranean. Mean annual precipitation ranges between 300 (W) to 150 mm (E), falling in autumn and winter. Mean annual temperature ranges from 8 (W) to 12 °C (E).	Bran et al., 2000; Labraga and Villalba, 2009; Paruelo et al., 1998; Prohaska, 1952, 1976
Soil	Aridisoles and Entisoles. Low organic matter content		Bran et al., 2000
Water availability	Mainly groundwater	Superficial and groundwater	Villagra and Giraudo, 2010
Dominant vegetation	Shrub steppe of *Larrea* spp. (*L. nitida, L. divaricata and L. cuneifolia*), associated with *Monttea aphylla, Prosopis alpataco* and *Atriplex lampa*. Grass cover is dominated by *Stipa tenuis*	Low shrub-grass steppes dominated by *Mulinum spinosum, Senecio* spp. and *Stipa speciosa*. Low shrub steepes dominated by *Nassauvia glomerulosa, Nassauvia axillaris* and *Chuquiraga avellanedae*. Medium shrub-grass steepes dominated by *Prosopis denudans* and *Lycium* spp., and *Stipa humilis*	Bran et al., 2000; Cabrera, 1971; León et al., 1998
Agrarian structure	Smallholders (63%), capitalized family-based farms (20%), commercial farms (6%) others (11%)	Smallholders (83%), capitalized family-based farms (6%), commercial farms (4%) others (7%)	Easdale et al., 2009
Livestock holding	Smallholders (30%), capitalized family-based farms (48%), commercial farms (16%), others (6%)	Smallholders (48%), commercial farms (26%), capitalized family-based farms (20%), others (6%)	Easdale et al., 2009
Mean farm area (ha)	Smallholders (5186), capitalized family-based farms (13,602), commercial farms (14,677), others (6754)	Smallholders (2549), capitalized family-based farms (10,375), commercial farms (21,533), others (3680)	Easdale et al., 2009
Dominant husbandry	Cattle, with increasing proportions of sheep in farms with lower capital	Sheep, increasingly mixed with goats in less capitalized farms, and with cattle in farms with meadows	Easdale et al., 2009; Easdale and Gaitán, 2010

collected daily data to compose images over a short period decadal- to generate nearly cloud-free images for 1981–2000. We excluded 1994 because there were no images available due to technical problems with AVHRR/NOAA sensor and anomalous values (e.g. negative). Decadal composites where used to generate monthly average values of NDVI for each pixel. Values corresponding to June and July (winter) of years 1984, 1988, 1993 and 2000 had a low resolution quality (probably due to cloudiness and low radiation at satellite passage time) and were also excluded from yearly composite to make them comparable. To include the growing season completely, each year started in August and finished in May of subsequent year (Fabricante et al., 2009). Monthly NDVI values where averaged to give an annual integrated value (NDVI-I).

In this paper the scale (i.e. grain and extent) of analysis was defined by administrative divisions, therefore additional processing was needed. We aggregated AVHRR/NOAA pixels (8 × 8 km) (grain of the biophysical data) included in the different counties (i.e. grain of socio-economic data) to represent each socio-economic unit (see description of socio-economic data). The National Institute of Statistics of Argentina calls this unit "segment" (i.e. county subdivision) and identifies them by a number (Fig. 1). Thereafter, we will refer to them as counties, keeping their respective number. To assure that NDVI values represent accurately the behavior of each county, pixels located within 8 km from boundaries were eliminated. We averaged pixels outside this range (i.e. boundary + 8 km) to obtain a unique NDVI-I value per county per year. The sum of all counties included in each ecological region (Monte and Patagonia) represented the extension of our analysis.

NDVI had been documented as lineally related to the fraction of absorbed incident photosynthetically active radiation (fAPAR, dimensionless) (Gamon et al., 1995; Ruimy et al., 1994). NDVI-I values where used to estimate fAPAR with a linear relation calculated for South America by Paruelo et al. (2004) (Eq. (1)). Absorbed photosynthetically active radiation (APAR, MJ m^{-2} yr^{-1}) is estimated by the product of fAPAR by incident photosynthetically active radiation (iPAR, MJ m^{-2} yr^{-1}). iPAR where obtained from FAO (1985) climatic statistics, calculated as half of the total incident radiation (Monteith and Unsworth, 1990). Incident radiation of five stations located in the study area where assigned to each county using a geographic distance criterion: Cipolleti (counties 27, 57, 58, 59), Choele Choel (county 34), San Antonio Oeste (counties 46, 51), Maquinchao (counties 60, 63, 64, 65, 66, 67, 68) and Bariloche (counties 70, 71, 72).

$$\mathrm{fAPAR} = \mathrm{NDVI-I}*1.1 - 0.055 \quad (1)$$

Aboveground net primary production (ANPP, g C m^{-2} yr^{-1}) can be estimated from absorbed radiation (APAR) and radiation use efficiency (ε, g C MJ^{-1}) (the amount of dry matter gained per unit APAR) (Kumar and Monteith, 1981; Monteith, 1977; Sinclair and Muchow, 1999) (Eq. (2)). Values of ε have not been determined for both regions under study, but even recognizing differences between vegetation types, Ruimy et al. (1994) suggest a constant radiation use efficiency value for regional analysis. Paruelo et al. (1997) recommended a value of $\varepsilon = 0.1$ g C MJ^{-1}, for North American rangelands with similar mean annual precipitation of Patagonian steppes, which was used in this study. NDVI has a greater importance in characterizing variability of ANPP at an annual temporal scale (Chapin et al., 2002), while APAR and ε are more important in intra-annual ANPP determinations. Thereafter, values of ANPP are presented in units of kgDM ha^{-2} yr^{-1} (DM: dry matter).

$$\mathrm{ANPP} = \mathrm{iPAR}*\mathrm{fAPAR}*\varepsilon \quad (2)$$

Forage Production (FP, kgDM ha^{-2} yr^{-1}) is a fraction of the ANPP. Because it varies spatially and temporally, estimation of the proportion consumable by herbivores (i.e. forage) in arid and semiarid regions is not a completely resolved problem. Therefore we used two models to estimate regional FP, in order to generate two comparable data sets.

Model 1 (Macro-ecological, FP1) estimated a harvest index (Hi, %) (Golluscio et al., 1999; Oesterheld et al., 1999, based on Oesterheld et al., 1992). Harvest index depends on the amount of ANPP (kgDM ha^{-2} yr^{-1}) (Eq. (3)). Annual forage production (FP1, kgDM ha^{-2} yr^{-1}) is estimated from ANPP and the harvest index (Eq. (4)).

$$\text{Hi} = -5.71 + 0.7154*(\text{ANPP})^{0.5} \quad (3)$$

$$\text{FP1} = \text{ANPP}*\text{Hi} \quad (4)$$

Model 2 (based on floristic information, FP2) estimated annual forage production (KgDM ha^{-2} yr^{-1}) as a proportion of ANPP (based on Hunt and Miyake, 2006) (Eq. (5)), where μ is the utilization rate of key species and φ is the fraction of not usable live biomass.

$$\text{FP2} = \mu*(1-\varphi)*\text{ANPP} \quad (5)$$

Average forage production usage (μ) in arid regions, during not very dry years ranges from 30% to 35% (Holechek et al., 1999). For this paper the forage usage rate was 35%. Whilst this method may be less sensible during extreme events such as droughts (in comparison with FP1 model, Eq. (4)), it is more robust for ANPP quality discrimination. Since for the study area there is not available information about the proportion of total biomass that is forage, we estimated $(1-\varphi)$ using the proportion of forage cover of dominant species in relation to total cover. We obtained from vegetation surveys (Bran et al., 1991; Godagnone and Bran, 2009) the different typical vegetation units included in the different counties of the study area. Aerial cover of each species included in the vegetation unit was transformed to usable cover by multiplying its cover by an index of forage aptitude: 1 = Forage, 0.5 = Intermediate and 0 = Non forage (Pelliza de Sbriller et al., 1984; Somlo et al., 1985; Somlo et al., 1994; UEP Río Negro, 2006) (see Table SM1, Supplementary Material). Then we added all forage cover of the species included in the vegetation unit and divided it by the total cover of the vegetation unit to obtain a surrogate of the usable biomass [$(1-\varphi)$ in Eq. (5)]. Finally, we estimated the index $(1-\varphi)$ for each county by calculating the weighted average of the vegetation units (i.e. occupied area) in each considered county (see Table SM2, Supplementary Material).

2.3. Forage production statistical analysis

In order to compare average values of temporal series from different counties, a mixed model statistical analysis was performed. The model considered both region and year as fixed factors. Degrees of freedom were calculated using the method proposed by Kenward and Roger (1997). A repetitiveness attribute was assigned for factor year, assigning a non equal distance in time within data series, because 1993–1994 and 1994–1995 periods were absent. For each county, a covariance matrix, a correlation matrix and least mean squares were calculated to assess statistical differences. The analyses were performed for both models (Eqs. (4) and (5)), assigning α value = 0.05. Regional average and coefficient of variation of inter-annual forage production were compared by performing ANOVA analysis ($\alpha = 0.05$) with InfoStat software (2008). In the case of coefficient of variation comparisons, data were previously transformed by using arcosin of square root of p value, because they were proportions.

We analyzed the two sets of forage estimations. We studied the correlation between temporal series of different counties by performing Pearson (r) analysis. Series of forage production in each county were characterized by: average series value (used as reference level), series coefficient of variation (CV), mean values of both three minimum and three maximum values, expressed as relative proportion of average value for each county. We used these county descriptors to perform Cluster Analysis (CA) and Principal Components Analysis (PCA), in order to obtain groups of counties with similar forage dynamics and main descriptors, respectively.

2.4. Socio-economic data processing

Demographic and farming indicators were obtained from the national population and housing census (INDEC, 2001) and the national agricultural census (INDEC, 2002), respectively. This data represent all the farms (i.e. economic) and rural residents (i.e. social) included in the study area, which means that they represent the universe of inquire, and not just a sampling. Both data subsets were aggregated at a county level. Since the county is the unit (grain) of analysis obtained from the agricultural census, a previous reorganization (i.e. aggregation) was needed for demographic data. The smallest demographic units were added in order to achieve the proposed scale. All counties could be reorganized, with the exception of counties 66 and 67 from Patagonia region (Fig. 1), and therefore both counties were considered together. Selected indicators were rural population, rural housing, number of farms and total livestock. Rural residents and housing (as a measure of households) were selected in order to explore the kind of sparseness of population with increasing variability and scarce resources (hypothesis proposed by Reynolds et al., 2007) at different scales, as measured by differences in forage production. Additionally, since rural households rely on livestock production, number of farms (i.e. indicating the main production unit) and total livestock were included as farming indicators. Cattle, sheep and goat were included in livestock calculations and were expressed in Sheep Livestock Units (SLU), using coefficients of equivalence based on the nutritional requirements of the three herbivores (based on Easdale et al., 2009). Finally, counties were first grouped and data were added according to the considered ecological regions: Monte or Patagonia (Table SM3, Supplementary Material). Regarding the aim of this paper, demographic and agricultural data were aggregated by regrouping counties according to the results obtained from regional forage production estimations (CA and PCA analysis). Finally, we explored our data sets with bivariate analyses between forage production (average and coefficient of variation) and stocking rates (total livestock [SLU]/county area [ha]), and rural population density (rural residents / county area [km^2]). Data was managed with SAS 9.0 (2009) and Infostat (2008) softwares.

3. Results

The main difference in regional forage production was explained by inter-annual variability (Fig. 2). The two models estimated higher forage production variability in Monte than in Patagonia, but did not discriminate statistical differences in series average values. The range of forage production was larger for the FP1 model (expedite-macroecological) than for the FP2 (laborious-floristic). Intra-regional comparison among annual average forage production did not discriminate different groups because of overlapping among counties (Table 2). According to the two models of forage production estimations, counties 64, 70 (Patagonia) and 58 (Monte) had the lowest average values in the study area ($p < 0.05$). On the other hand, counties 72, 66 and 71 from Patagonia presented the highest values in both models, while in the Monte region counties 46 and 34 had the highest values but only with the model one (FP1, Eq. (4)) (Table 2). Temporal series of different counties presented positive spatial correlations ($p < 0.01$) for all cases and both models used. Cumulative frequency was 99% for coefficients (r) over 0.8. This can be indicative that inter-annual oscillations of forage production from different counties had a common synchronic behavior in the same direction across each ecological region, and when analyzing both together.

Cluster analysis discriminated two main groups (i.e., larger mean Euclidean distance) (Fig. 3, cophenetic correlation coefficient = 0.72). Both groups corresponded to the ecological regions

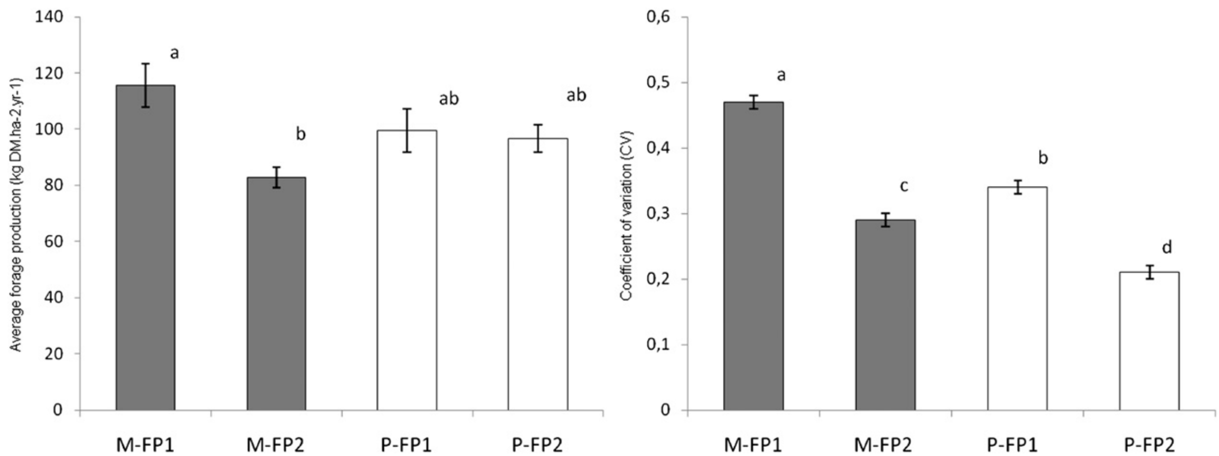

Fig. 2. Regional average and coefficient of variation (CV) of forage production, obtained from models FP1 (i.e. macro-ecological, Eq. (4)) and FP2 (i.e. floristic, Eq. (5)). Bars indicate standard error and different letters means statistical differences ($\alpha = 0.05$). Monte (M, grey) and Patagonia (P, white).

included in the analysis, Monte and Patagonia. An additional subdivision within each group, but with a shorter mean Euclidean distance, resulted in five end-groups (Fig. 3). Within Monte region, one group was conformed by counties 27, 57, 58 and 59 (hereafter denominated M1), and a second group was constituted by counties 34, 46 and 51 (hereafter denominated M2). Within Patagonia region three groups were obtained. One group included most of the counties (60, 63, 66, 68, 65 and 67) (hereafter denominated P1). The second included counties 64 and 70 (hereafter denominated P2), and the third group included counties 71 and 72 (hereafter denominated P3) (Fig. 3). We performed a cluster analysis for each model separately (FP1 and FP2) and results differed in the location of only one county from Patagonia (P68), which was included in group P2 for model FP1.

Principal Component Analysis (PCA), using the same variables as in cluster analysis resulted in the five groups obtained in the cluster analysis (Fig. 4). The first axes explained 73% of variability and was broadly related to the coefficient of variation of the series estimated with the two models (eigenvalue: +0.41). The second axes explained an additional 20% of variability and was related to average forage production, mainly from model FP1 (eigenvalue: +0.78). Geographically the groups were organized in different ways. Particularly in Patagonia the zone P2 appeared spatially fragmented (counties P64 and P71; Fig. 1).

Table 2
Annual average forage production (kg DM [Dry Matter] ha^{-2} $year^{-1}$) and inter-annual variability of forage production (%, between parentheses) for each county, obtained from models FP1 (Eq. (4)) and FP2 (Eq. (5)), respectively. Counties were ordered from high to low forage production estimated with the two models. Different letters mean significant statistical differences ($\alpha = 0.05$, least mean squares method). References: (M) Monte, (P) Patagonia.

County	FP1 model Forage production (kg DM ha^{-2} yr^{-1})	County	FP2 model Forage production (kg DM ha^{-2} yr^{-1})
M46	143.65 (42) a	P72	114.87 (16) a
M34	137.84 (44) a b	P60	110.89 (21) a b
P72	131.92 (26) a b c	P71	106.38 (17) a b c
M57	119.15 (50) a b c	P66	105.09 (23) a b c d
P66	116.26 (37) a b c	P63	103.92 (21) b c d
M51	115.50 (46) a b c	P67	96.99 (23) c d e
P71	113.03 (27) a b c	P65	96.19 (22) d e
P63	112.19 (36) b c	M46	95.71 (25) d e
P60	111.90 (35) b c	P68	92.88 (23) e f
M59	109.83 (47) b c d	M34	91.44 (27) e f g
M27	109.48 (52) b c d	M51	83.98 (27) f g h
P67	103.98 (37) c d	M57	83.75 (31) f g h
P65	101.81 (36) c d	M59	81.58 (29) g h j
P68	80.57 (38) d e	M27	79.09 (31) h j
M58	80.34 (47) d e	P70	72.70 (20) i j
P70	60.62 (34) e	P64	67.15 (23) i
P64	55.31 (36) e	M58	65.42 (30) i

The reorganization of demographic and agricultural indicators, according to the regional forage production groups previously obtained (Fig. 3), gave evidences of intra-regional socio-economic distinctions to be highlighted (Table 3). The Monte region had 61% of rural population and 56% of farms in zone M1, which registered less livestock units than the other zone (M2). The zone M1 had the most variable forage production of the whole study area (Fig. 4). In Patagonia region, the largest zone (P1) accumulated two thirds of livestock, almost 55% of rural population and 59% of farms. On the other hand, zone P2 had the lower proportion of farms and livestock units, but with more rural population than zone P3. While zone P2 had the least average forage production, the zone P3 presented the least forage variability of the study area (Fig. 4).

There was no significant relationship between stocking rate and either forage production average or forage production variability. However, there was a significant non-lineal relationship between rural population density and forage production variability (CV) estimated with both FP1 and FP2 (Fig. 5, c) Rural Population density $= 0.11 \quad FP1CV^3 - 7.10 \quad FP1CV^2 + 137.68 \quad FP1CV - 701.90$; $r^2 = 0.68$, $r^2adj = 0.60$, $F = 8.44$, $p = 0.0028$; d) Rural Population density $= 0.02 \quad FP2CV^3 - 2.08 \quad FP2CV^2 + 64.71 \quad FP2CV - 487.89$; $r^2 = 0.67$, $r^2adj = 0.59$, $F = 8.09$, $p = 0.0032$). The minimum was related to the ecological regions. In Patagonia, rural population density decreases with increasing forage production variability, whereas in Monte the inverse situation occurred.

4. Discussion

The arid and semiarid rangelands of northern Patagonia presented differences in the forage production dynamics both inter- and intra- ecological regions. In general, the temporal variability in forage production (Monte > Patagonia, Fig. 2) was the main difference between the two ecological regions. Albeit the spatial continuity in average forage production, it was possible to assemble groups of counties (i.e. socio-economic units) with similar inter-annual dynamics (Fig. 3). Grouping resulted mainly from differences in i) their temporal variability (i.e. M1 > M2 in Monte; P1 = P2 > P3 in Patagonia, Fig. 4), and ii) average forage production (i.e. P1 = P3 > P2 in Patagonia, Fig. 4). These results are consistent with previous studies of NDVI and ANPP (Fabricante et al., 2009;

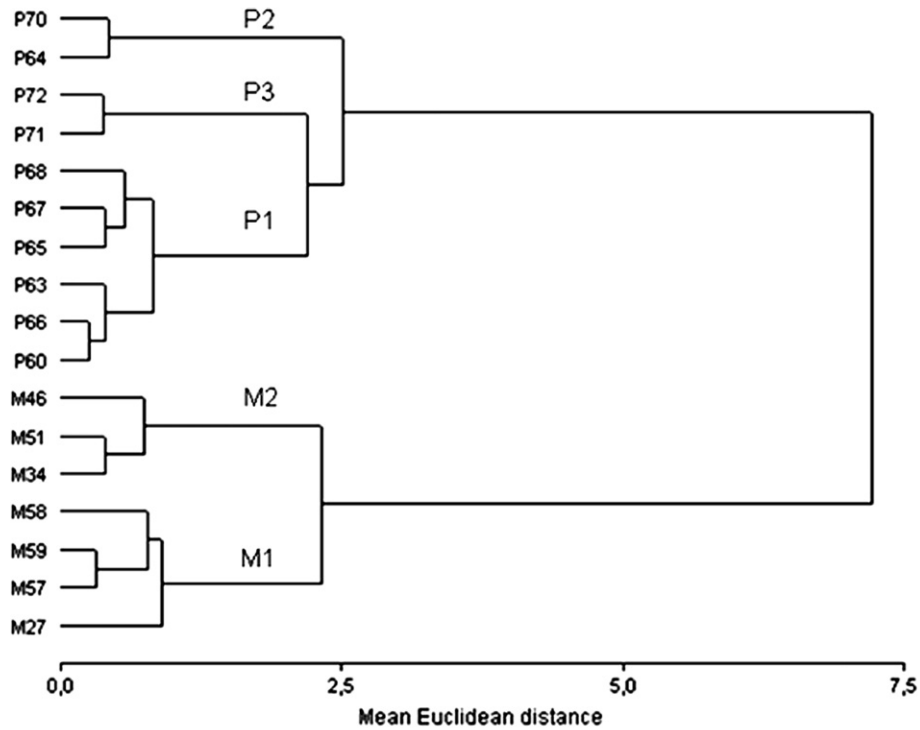

Fig. 3. Cluster analysis of forage production using counties as classificatory criterion (cophenetic correlation coefficient = 0.72). Variables included were descriptors of forage production for the studied period (1981–2000) estimated with the two models (FP1 [Eq. (4)] and FP2 [Eq. (5)]): average, inter-annual coefficient of variation of forage production (CV), minimum and maximum mean values. (M) Monte, (P) Patagonia.

Jobbágy et al., 2002). As in many other arid rangelands of the world, there is a strong climatic control on regional forage production (e.g. as measured in this study by the high levels of positive spatial correlations among series). In northern Patagonia, there is a strong climatic division (Prohaska, 1952) that approximately overlaps with regional ecological division studied. Patagonia region (influenced by the Pacific anticyclone) has a decreasing precipitation gradient and an increasing inter-annual variability from west to east towards the centre of the studied area (Jobbágy et al., 1995). In contrast, in Monte (influenced by the Atlantic anticyclone), precipitation increases again in direction north-east, but with greater atmospheric demand (Godagnone and Bran, 2009). This spatial heterogeneity is also evidenced in regional forage production dynamics (Fig. 4).

Complementary socio-economic data suggest differences across spatial scales. There were fewer rural residents and farm numbers at Monte, the region with higher forage production variability (Table 3), suggesting a hierarchical process of biophysical drivers

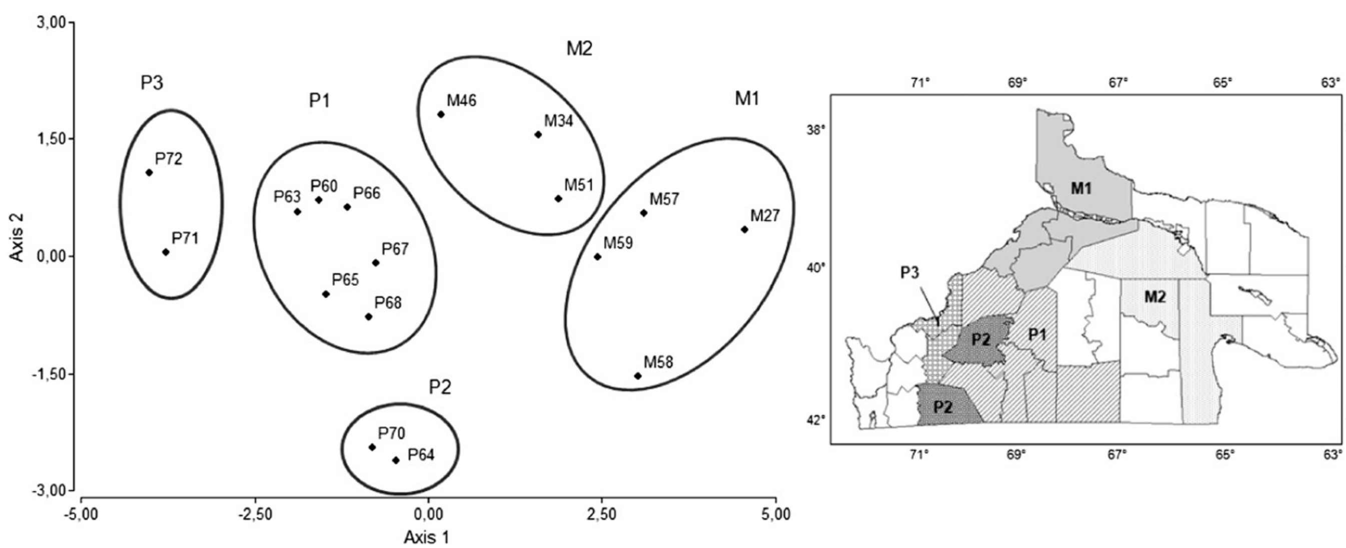

Fig. 4. First two axes of principal components analysis (PCA) performed with forage production descriptors. Variables and references are the same as for Cluster Analysis (Fig. 3). Axis 1 explained 73% of variability (highest eigenvalue (+0.41): coefficient of variation for both models). Axis 2 explained 20% of variability (highest eigenvalue (+0.78): average forage production for FP1 model). The obtained groups of counties (see Fig. 3) were located in the map of the study area.

Table 3
Demographic and agricultural indicators organized according to the regional forage production zones described in Fig. 3, expressed as relative proportions for each region. Average forage production (kg DM [Dry Matter] ha^{-2} yr^{-1}) and temporal coefficient of variation (%, between brackets) were estimated with models FP1 and FP2 for each zone, respectively (Eqs. (4) and (5)). Bold values indicate zones with potentially harmful perturbations due to biophysical drivers (M1 high inter-annual forage production variability, P2: low average forage production).

Forage production zones	Area (%, 10^3 km^2)	Rural residents (%, n)	Rural housing (%, n)	Farms (%, n)	Total livestock (%, SLU)	FP1 (kg DM ha^{-2} yr^{-1})	FP2 (kg DM ha^{-2} yr^{-1})
Monte							
M1	**46.7%**	**61.0%**	**58.1%**	**56.4%**	**44.5%**	**103.7 [49]**	**77.4 [30]**
M2	53.3%	39.0%	41.9%	43.6%	55.5%	131.2 [44]	90.3 [26]
Total	*63.2*	*2889*	*1423*	*527*	*507,834*		
Patagonia							
P1	68.7%	54.8%	57.5%	59.1%	75.9%	104.8 [37]	100.9 [22]
P2	**19.5%**	**26.5%**	**26.2%**	**19.6%**	**11.5%**	**59.6 [35]**	**70.1 [22]**
P3	11.8%	18.8%	16.2%	21.2%	12.6%	123.3 [27]	110.7 [17]
Total	*55.3*	*4742*	*2200*	*1059*	*667,441*		

influencing sparseness and remoteness (*sensu* Reynolds et al., 2007). Generally in Monte region, farms tend to have larger average areas (across different types of farms, Table 1), and has less permanent workers than in Patagonia (Easdale et al., 2009). Large farms provide spatial heterogeneity and may buffer against environmental variability, enhancing resilience (Walker and Abel, 2002), but at a regional scale it promotes less production units (i.e. farms), and hence less households and a sparser rural population (Table 3). However, further research is needed regarding differences within ecological regions. In particular, the increasing proportions of residents and farm numbers registered in zones with increasing variability and with the lowest forage production (M1 and P2, respectively in Table 3). These results suggest that human density might be more sensible than stocking rate with changes in forage production variability (Fig. 5). Future research is needed to understand such social-ecological heterogeneity at different scales, and the co-evolutionary role of different social drivers (e.g. historical and political contexts, culture, regional

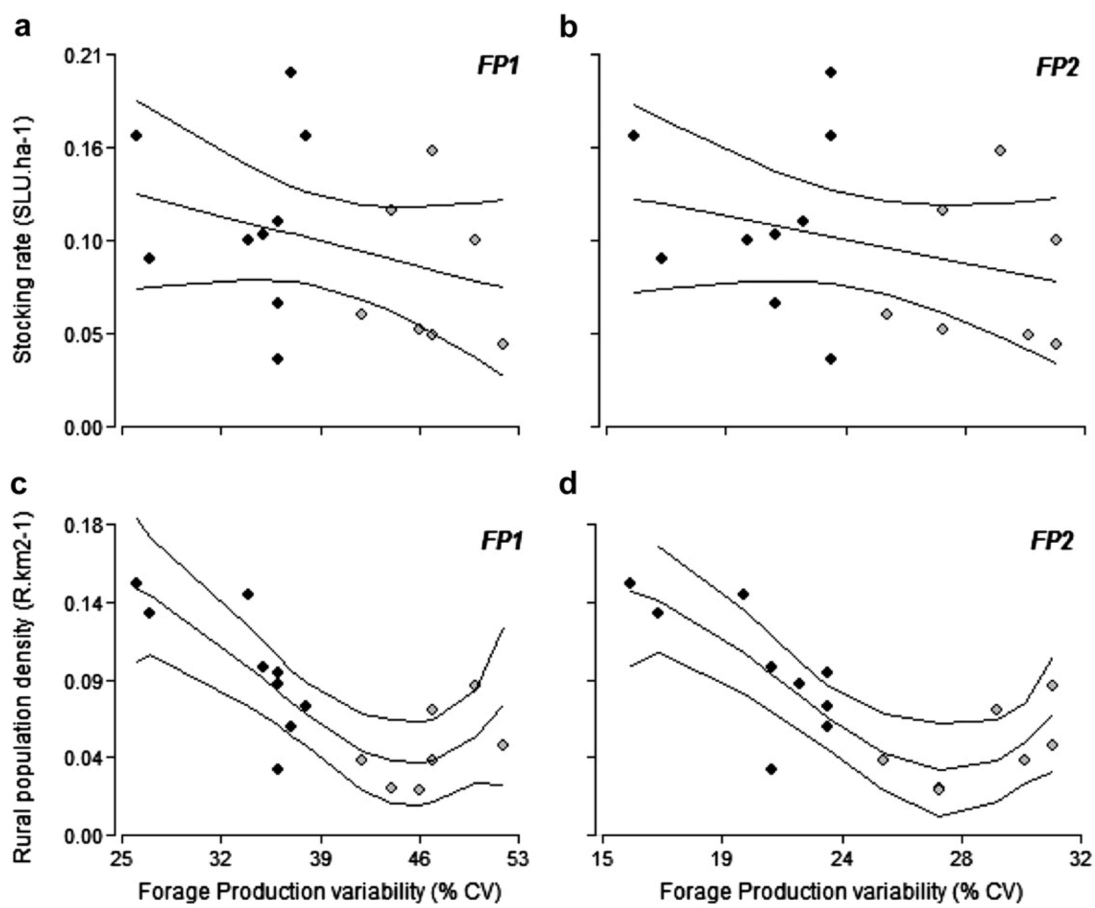

Fig. 5. Relationship between forage production and stocking rate (a-[FP1] and b-[FP2]) and rural population density (c-[FP1] and d-[FP2]). Dots represent the data and lines represent the adjusted model and its confidence interval. Note that the *X* axis is different for FP1 and FP2. Abbreviations: sheep livestock units (SLU), rural residents (R), coefficient of variation (CV). Patagonia (black dots), Monte (grey dots). Adjusted models a) Stocking rate = 0.18 * $-2e^{-03}$ FP1CV; $r^2 = 0.10$, r^2adj = 0.04 ($F = 1.60$, $p = 0.2267$). b) Stocking rate = 0.18 * $-3.1e^{-03}$ FP2CV; $r^2 = 0.09$, r^2adj = 0.02 ($F = 1.38$, $p = 0.2598$). c) Rural population density = 0.11 $FP1CV^3$ − 7.10 $FP1CV^2$ + 137.68 FP1CV − 701.90; $r^2 = 0.68$, r^2adj = 0.60 ($F = 8.44$, $p = 0.0028$). d) Rural population density = 0.02 $FP2CV^3$ − 2.08 $FP2CV^2$ + 64.71 FP2CV − 487.89; $r^2 = 0.67$, r^2adj = 0.59 ($F = 8.09$, $p = 0.0032$).

infrastructure, markets) or even biophysical ones (e.g. water availability) that may also be heterogeneous in space and time even within a political or biophysical unit. For example, land distribution among stakeholders during post-colonialism (Bandieri, 2005), may have influenced current structural socio-productive organization, with increasing proportions of smallholders in poorest lands. As an opposite hypothesis, a high initial human density could have promoted land degradation at a landscape scale, which resulted in a current positive relationship between human density and forage production variability in some areas.

Arid and semiarid rangelands worldwide are particularly vulnerable to low and unpredictable rainfall (Reynolds and Stafford Smith, 2002; Reynolds et al., 2007) that ultimately controls forage production. Zones with lower forage production or higher variability promote growing exposure to potentially harmful perturbations, suggesting increasing predisposition for socio-ecological vulnerable situations (Reynolds et al., 2007). In the context of this study, the zones M1 in Monte region and P2 in Patagonia might be exposed to relatively high forage variability (i.e. promoting high production risk) and to low forage production (i.e. promoting high levels of stress), respectively (Fig. 4). Further research is needed in these territories, with an inclusion of other disturbance factors, in order to describe the kind of exposure, sensitivity conditions and resilience in order to fully analyze vulnerability (Turner et al., 2003). Nonetheless and based on the kind of drivers described above, a combination of solutions for those zones might include both adaptive management and active political decisions (e.g. economic and forage subsidies).

The role of climatic variability in arid regions and its implications on secondary production have generated a strong debate in ecology, regarding equilibrium and non-equilibrium models (Briske et al., 2003; Vetter, 2005). However, both perspectives agree on the existence of increasing levels of stress in the system, with increasing levels of environmental variability (Vetter, 2005). Many scholars suggest that stocking rate management must be more conservative under stochastic than under deterministic conditions (Díaz-Solís et al., 2009; Higgins et al., 2007; Illius and O'Connor, 1999). The management aim would be keeping the system near the carrying capacity of drought years and only decoupling in wet periods (Illius and O'Connor, 1999). However, reducing stocking rate may not be economically (and therefore socially) viable in many situations (i.e. specially for smallholders). Consequently, any kind of economic or energetic subsidies may be coupled to environmental variability (e.g. zone M1), being episodically triggered when drought occurs. Recent evidences suggest that other survival strategies such as partnership agreements to achieve better prices and off-farm income are effective strategies to decouple the effect of drought on farm productivity from household income in semiarid rangelands (Easdale and Rosso, 2010).

Slow livestock recovery after drought-driven animal mortality, had been documented as one of the main livestock management problems in arid and semiarid regions (Angassa and Oba, 2007; Oba, 2001). Farmers usually rely on external acquisition for restocking when mortality surpasses birth rate (Texeira and Paruelo, 2006), decision that might not be economically sustainable in the long term (Ares, 2007). While livestock mortality during drought is a negative feedback that promotes rangeland recovery, restocking after drought (usually fostered by government subsidies and/or loans at low-rate) function as positive feedback increasing system entropy. In these contexts, aids to farmers during drought periods might be better oriented to fodder than to buying animals after the event (Ares, 2007), and an active promotion of flexible farm management. In the same direction, in zones with low forage production (e.g. P2, Fig. 4), the low carrying capacity might be restricting stocking rates, with similar socio-economic consequences as the ones described above. While further research is needed to explore the historical causes of current low forage production (i.e. natural or human induced), economic subsidies and promotion of novel rural employment opportunities might necessarily be permanent (instead of episodically triggered in drought periods) in order to enhance the overall household income.

Territories with relatively higher forage production or less interannual variability (e.g. in this study P1 and P3) also need specific policy design and management decisions directed to prevent a degradation-impoverishment cycle process (Reynolds and Stafford Smith, 2002), since they include the highest proportion of rural population of the study area (Table 3). In general, policy (i.e. loans and subsidies) must be oriented to promote knowledge and technology investment for sustainable management of natural capital and improvement of production efficiencies at finer scales (e.g. farm) (Easdale et al., 2009; Villagra and Giraudo, 2010), but without compromising long term social-ecological resilience.

5. Applicability of forage production estimation models

The two models differed slightly in both regional and intraregional estimations of forage production (Figs. 2 and 3). Since models differ in their conceptual origins and basic information needed to estimate forage production from ANPP, we propose that their applicability is rather defined by available information and scale. The harvest index (Hi, %) utilized in the model FP1 (Eq. (4)) is based on macro-ecological information for temperate rangelands from South America (Golluscio et al., 1999; Oesterheld et al., 1999, based on Oesterheld et al., 1992), and probably they can be applied in similar regions worldwide. On the other hand, model FP2 (Eq. (5)) is based on previous detailed knowledge on vegetation and species pastoral value to estimate the fraction of not usable live biomass (φ) (Hunt and Miyake, 2006), which implies time investment for field evaluations that increases as study area increases. We propose that FP1 is appropriate to regional or above-regional assessments (i.e. state or national), while FP2 is more appropriate for intra-regional (i.e. county) assessments.

6. Final thoughts

In this paper we assessed forage production for state-administrative designed units (i.e. counties), but hierarchically stratified by ecological differences (i.e. Monte and Patagonia ecological regions), with the purpose of moving forward in methodological issues regarding integrated socio-ecological analyses. At this early stage, we emphasize that the inclusion of complementary socio-economic data into the analysis gave an improved picture regarding rangeland ecology and socio-ecological heterogeneity at different scales, which provided additional insights for differentiated policy design. We consider this information useful for bridging policy-science and social-biophysical gaps (Bradshaw and Bekoff, 2001; Bradshaw and Borchers, 2000). This conceptual and operational aim is still one of the main scientific forthcoming challenges in a complex and changing world.

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Appendix. Supplementary data

Supplementary data associated with this article can be found, in the online version, at doi:10.1016/j.jaridenv.2012.03.002.

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