

# The efficiency of two widely used commercial live-traps to develop monitoring protocols for small mammal biodiversity



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## ABSTRACT

Biodiversity monitoring programs have been implemented worldwide as a source of information on ecosystem functioning. However, controversy concerning the indicators that should be monitored, and the development of adequate monitoring protocols for multi-species communities still hamper such implementation, especially in the case of small mammals. We analyze differences in the efficiency of the two most widely used commercial traps (Longworth and Sherman) working simultaneously in eight different mountain habitats in Andorra country (NE Iberia) as a first step for establishing standardized sampling protocols for species-rich small mammal communities. From summer 2008 to fall 2010 (six sampling occasions) we captured a total of 728 small mammal individuals (1445 including recaptures) of 13 species (12 in Longworth and 11 in Sherman, 10 species shared). Despite some specific biases (underestimation of two large species by Longworth traps and underestimation of one small species by Sherman traps), estimates of community parameters and similarity indexes, sampling efficiency (number of small mammals trapped), detectability, mean weight, and sex-ratio of the most abundant species, were similar for both sampling methods. Our results suggested that both trap models could be used interchangeably – without relevant biases – in small mammal community assessments where large species are infrequent. Focussing monitoring programs on highly detectable small mammal species (common species) would allow the establishment of robust monitoring programs aimed at reducing the time invested and economic costs.

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## 1. Introduction

Biodiversity monitoring programs have been implemented worldwide as a source of information on ecosystems functioning, as well as to evaluate whether conservation policies are delivering their expected goals (EEA, 2010, 2012). Systems of indicators have been developed in parallel to policies (Bubb et al., 2010), and standardized robust sampling techniques and bioindicators of environmental change have been developed for a variety of focal groups (i.e., birds and butterflies, Herrando et al., 2015), following appropriate applied research (e.g. Voříšek et al., 2010 for birds).

Nevertheless, discrepancies among indicators that should be monitored and the development of monitoring protocols still exist (EEA, 2012; Diaz et al., 2015; Gao et al., 2015), especially for the focal groups most difficult to sample in field conditions (Overmars et al., 2014). This is usually the case of mammals in general

and small mammals in particular. Beyond any statutory obligation of monitoring population changes (Harris and Yalden, 2004), mammal monitoring allows the quantifying of impacts associated with anthropogenic environmental change (i.e. climate and landscape change), informing conservation and management priorities (Flowerdew et al., 2004; Wright et al., 2014). Besides, there is increasing evidence that landscape and climate change is affecting the composition of small mammal communities, especially in the Mediterranean region (Szpunar et al., 2008; Torre et al., 2015). In fact, long-term single-species monitoring of target small mammals such as lemmings *Lemus* spp. and voles *Microtus* spp., *Myodes* spp. have settled the ground for understanding how and why climate, food availability, habitat structure, predation and disease interact to regulate animal populations (see Stenseth, 1999; Ims et al., 2008 for reviews).

Monitoring small mammal biodiversity change in addition to population change is hampered by the lack of “universal” sampling protocols (and bioindicators) that preclude the application of standardized monitoring programs. For instance, most programs for small mammal monitoring established in the UK were short in time (<15 years, Flowerdew et al., 2004) and/or narrow in space (i.e.

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long-term monitoring but for single places, Newman et al., 2003), mostly due to methodological and logistic constraints. Further, all these surveys were based on live-trapping methods (Sibbald et al., 2006), using single types of live traps while aiming at monitoring several species of small mammals (Flowerdew et al., 2004). Longworth traps are recommended and used in many European cold and temperate sites (Flowerdew et al., 2004), whereas Sherman traps are the most commonly used traps in North America (Slade et al., 1993), being routinely used for long term single-species monitoring programs (Previtali et al., 2009). However, estimating small mammal community composition and structure by using single live-trapping methods can be unrealistic due to trap-specific differences in trappability among species (Anthony et al., 2005; Cáceres et al., 2011; Dos Santos-Filho et al., 2006; Lambert et al., 2005) and even among sexes, sizes or ages within species (Burger et al., 2009). Thus, several authors suggest using a combination of sampling techniques to account for trappability differences (Fonturbel, 2010), to restrict monitoring to the most trapped species (Solari et al., 2002), or to estimate detectability and correct for its effects (Mackenzie et al., 2002).

Recently, we established a small mammal monitoring program in Spain (SEguimiento de Micromamíferos Comunes de España, SEMICE, Torre et al., 2011) partially inspired by UK small mammal monitoring programs. We focused on common species easy to catch with commercial live traps. We here test differences in the efficiency of the two most widely used commercial traps (Longworth and Sherman) working simultaneously in eight different mountain habitats, as a first step for establishing standardized small mammal sampling protocols. Specifically, we test differences among species and within-species trappability, as well as trap-induced mortality of the most common species. In this way, we ascertained whether biases may be present when small mammal communities are sampled and estimated by both methods working altogether.

## 2. Materials and methods

### 2.1. Study area

Field work was carried out in the Principality of Andorra, located on the eastern side of the axial Pyrenees (Fig. 1). Despite its small size ( $468 \text{ km}^2$ ), Andorra is a country of strong contrasts due to its rugged terrain with an average height above 2000 m a.s.l. (840–2946 m). Three main vegetation belts can be delimited

depending on elevation and orientation (Folch, 1984): the montane zone (800–1700 m), the subalpine zone (1700–2400 m), and the alpine zone (beyond 2400 m). Forests cover 43% of the area, being Black pine (*Pinus uncinata*) and Scots pine (*Pinus sylvestris*) the most frequent tree species (Folch, 1984). Grasslands occupy 30% of the territory, and shrublands 8%, including siliceous alpenrose (*Rhododendron ferrugineum*) and juniper-dominated (*Juniperus communis*) scrub. Rock and scree areas occupy 15% of the territory, and synanthropic environments, such as farming and residential areas, less than 5%, although human influences are evident throughout the country.

### 2.2. Small mammal sampling

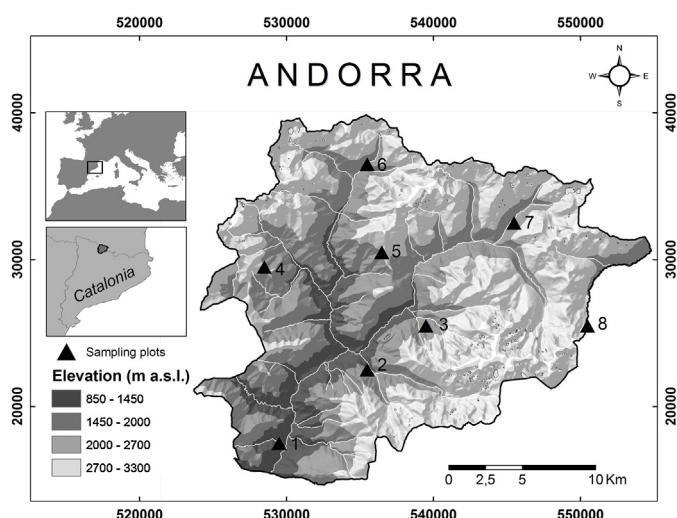
Sampling was performed in 2008 to fall 2010 on eight plots during six trapping sessions of three days each (two sessions per year, July and October), collectively lasting 18 days. Every plot was sampled by a  $6 \times 6$  trapping grid, consisting of 18 Sherman traps (Sherman folding small animal trap;  $23 \times 7.5 \times 9 \text{ cm}$ ; Sherman Co., USA) and 18 Longworth traps (Penlon Ltd., Oxford, UK), alternated in position (Cáceres et al., 2011; Nicolas and Colyn, 2006), spaced 15 m apart, and brought into operation for three consecutive nights. Traps were arranged singly rather than in pairs as no trap saturation was expected on the basis of the usual low density of small mammals in the study areas (authors unpub.). Traps were set in the evening of the first day, and checked on five occasions (the early morning of the first, second and third day, and the night of the second and third days). So, every trapping session (primary sampling occasion) consisted in five sampling occasions (secondary sampling occasions), collectively lasting 30 sampling occasions.

Traps were baited and re-baited when necessary (when the bait was eaten) with a piece of apple and a mixture of tuna, flour and oil. This bait was proven to be effective for rodents and shrews at least in Mediterranean mountain conditions (e.g. Torre et al., 2007; Torre et al., 2014a). Traps were insulated by including hydrophobic cotton for bedding. The small mammals caught were identified to species, sexed, marked (rodents with ear tags – National Band Co. USA – and shrews with fur clips), and released at the point of capture (Gurnell and Flowerdew, 2006). All the sampling procedures met the ASM care & use guidelines (Sikes et al., 2011). We used counts (e.g. the number of different individuals trapped within the three days, Morris, 1996) as an estimate of population size in each study plot, assuming that the unseen proportion of the population is constant and that counts and estimates would have yielded similar results (Slade and Blair, 2000).

The eight sampling stations were distributed along a strong elevation gradient (1060–2255 m a.s.l.) within the three vegetation/climatic domains. Three plots were situated on the montane domain (1 – *Quercus pubescens* and 2 – *Pinus sylvestris* forests, and 3 – a meadow site), four on the subalpine domain (4 – *Abies alba* and 5 – *Pinus uncinata* forests and 6 – a scree and 7 – a meadow site), and one in the alpine domain (8 – scree). Plots were established in sites as heterogeneous as possible (e.g. including clearing in forests and shrubs in meadows and scree) in order to capture all species present.

### 2.3. Data analysis

The variables for comparison among trap types were community parameters (species diversity and similarity), species' abundances and biomass, species' occupancy, detectability and mortality, and sex-related within-species differences. This wide range of relevant variables to compare among trap types required the development of multiple analytical strategies summarized in the Appendix. First, a matrix with the number of individuals of the small mammal species by sampling method was created, and



**Fig. 1.** Map showing the location of the study area and the sampling plots according to elevation. See methods for the meaning of plot numbers.

**Table 1**

Total number of individuals captured and recaptured (in brackets) and their relative frequencies (%) for the 14 small mammal species caught in Longworth and Sherman traps working simultaneously in eight contrasting sampling plots from 2008 to 2010.

Species	Longworth		Sherman		Total	
	N	%	N	%	N	%
<i>Neomys fodiens</i>	2(2)	0.55 (0.27)	0(0)	0.00 (0.00)	2(2)	0.27 (0.14)
<i>Sorex araneus</i>	26(28)	7.14 (3.83)	11(11)	3.02 (1.53)	37(39)	5.08 (2.69)
<i>Sorex minutus</i>	7(7)	1.92 (0.96)	9(10)	2.47 (1.39)	16(17)	2.20 (1.17)
<i>Crocidura russula</i>	26(41)	7.14 (5.60)	27(42)	4.40 (3.76)	42(68)	5.77 (4.69)
<i>Glis glis</i>	1(2)	0.00 (0.00)	1(4)	0.55 (0.56)	2(6)	0.27 (0.41)
<i>Eliomys quercinus</i>	12(34)	3.30 (4.64)	41(85)	7.97 (11.82)	41(119)	5.63 (8.20)
<i>Apodemus sylvaticus</i>	135(251)	37.09 (34.29)	152(259)	41.76 (36.02)	287(510)	39.42 (35.15)
<i>Mus musculus</i>	1(1)	0.27 (0.14)	0(0)	0.00 (0.00)	1(1)	0.14 (0.07)
<i>Arvicola sapidus</i>	0(0)	0.00 (0.00)	5(8)	1.37 (1.11)	5(8)	0.69 (0.55)
<i>Myodes glareolus</i>	130(329)	35.99 (45.77)	130(300)	35.71 (41.72)	261(635)	35.85 (43.76)
<i>Microtus agrestis</i>	9(12)	2.47 (1.64)	5(7)	1.37 (0.97)	14(19)	1.92 (1.31)
<i>Microtus arvalis</i>	5(6)	1.37 (0.82)	1(1)	0.27 (0.14)	6(7)	0.82 (0.48)
<i>Chionomys nivalis</i>	8(13)	2.20 (1.78)	4(7)	1.10 (0.97)	12(20)	1.65 (1.38)
<i>Mustela erminea</i>	2(2)	0.55 (0.27)	0(0)	0.00	2(2)	0.27 (0.14)
Total	364(732)	100(100.00)	364(719)	100(100.00)	728(1451)	100(100.00)

species accumulation curves were used to estimate species richness (Gotelli and Colwell, 2001). The expected richness function was calculated with EstimateS (version 9.1.0, Colwell, 2013), after 100 randomizations (default option) of the observed number of species as far as samples (secondary sampling occasions) accumulated. We also used richness estimators (Chao et al., 2009) to determine actual species richness, i.e. the observed plus the estimated unobserved species. To ascertain the completeness of the inventory by each sampling method, we used the Clench equation to adjust the species accumulation curves (Díaz-Francés and Soberón, 2005). To fit the Clench equation to the functions provided by EstimateS, we used the non-linear estimation module of Statistica v7.0 (Stat Soft Inc.) following the procedure outlined by Jiménez-Valverde and Hortal (2003). Since community parameters estimates can be affected by the number of individuals sampled (Gotelli and Colwell, 2001), they were calculated for every habitat and sampling method by means of rarefaction, that is, rarefying the number of species, Shannon diversity, PIE Hulbert's index, and dominance, to the same number of individuals sampled by both methods in the same habitat (Ecosim software, Gotelli and Entsminger, 2001). Similarity indexes (Chao-Jaccard and shared species) were computed for every possible comparison between plots (28 pairs) and were compared between trap models.

Second, the fitting of log-linear models to multifactor contingency tables was used to determine whether frequencies of occurrence of the small mammal species differed between sampling methods. Two general models were fitted to the observed frequencies: species × method and size × method. Also, partial (including the most frequent species) log-linear models were applied to test for sex biases and mortality biases between sampling methods (males vs females; shrews vs rodents, respectively). Standardized residuals were analyzed to ascertain oversampling and undersampling by both sampling methods (Anthony et al., 2005; Torre et al., 2004).

Third, site occupancy analysis (SOA) was used to determine whether estimated occupancy ( $\Psi$ ) and detection probabilities ( $p$ ) changed between species and sampling methods. The multi-method model approach used extends the single season model by allowing detection probabilities to vary for different methods of observation, correcting for the possibility of individual animals being detected by more than one method within a sampling occasion (Nichols et al., 2008). We fitted the same model for all the species analyzed: occupancy was left invariable between sampling periods, but with variable probability of detection (Otto and Roloff, 2011). Analyses were performed with Presence software (Mackenzie, 2012) only on species having high capture rates

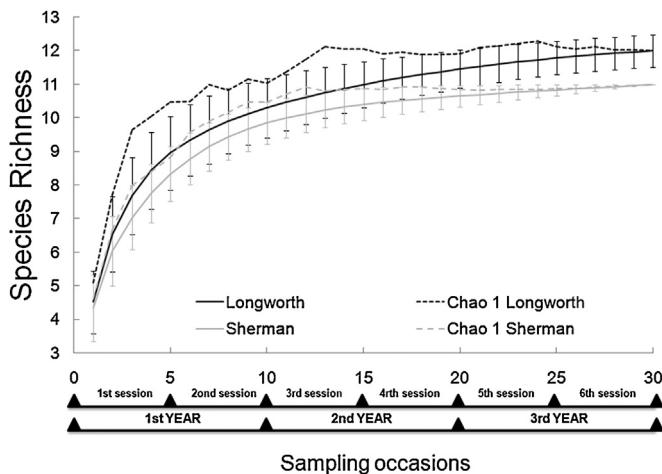
(Watkins et al., 2010). Density indexes (DI, i.e., the number of individuals trapped by season and plot corrected for sampling effort) were correlated with detection probabilities in order to ascertain whether the former can be considered as surrogates of the latter (Watkins et al., 2010).

Statistical comparisons of community parameters (diversity, similarity, and detectability) were performed by Wilcoxon Matched Pairs tests for dependent samples. Community parameters and similarity indexes were obtained with EstimateS (Colwell, 2013).

### 3. Results

We captured a total of 728 small mammal individuals (1445 including recaptures) of 13 species (Table 1) in the eight sampling stations corresponding to 8640 trap-nights (8.4% total trapping success). Average trapping success per sampling session was 9.18% (range 0.56–29.44%). Total estimated species richness (Chao1 estimator) was  $13 \pm 0.25$  (SD) species, so observed and estimated species curves converged. The Clench equation adjusted to the species accumulation curve fitted very well ( $r^2 = 0.996$ ), and the slope of 0.019 showed the proximity to an asymptote. In fact, the asymptotic species richness was 13.47 species, so 96.5% of the small mammal species would have been recorded during the present inventory.

Ten out of thirteen species (77%) were shared by both sampling methods (Table 1), but Longworth traps captured two exclusive species (*Neomys fodiens* and *Mus musculus*), whereas Sherman traps captured one exclusive species (*Arvicola sapidus*). As a curiosity, we captured two individual stoats *Mustela erminea* in Longworth traps, but this species was not considered in further analyses. Total estimated small mammal richness (Chao1 estimator) was  $12 \pm 0.17$  (SD) species for Longworth traps and  $11 \pm 0.48$  (SD) species for Sherman traps (Fig. 2). The Clench equation adjusted to the species accumulation curves fitted very well for both methods ( $r^2 = 0.9982$  and 0.9976 for Longworth and Sherman traps, respectively), and the slopes showed the proximity to an asymptote in both cases (0.024 and 0.021, respectively). In fact, the asymptotic species richness were 12.58 species for Longworth and 11.67 species for Sherman traps, so 95.3% and 94.2% of the small mammal species would have been recorded during the present inventory by both methods, respectively. Mean number of species detected per habitat was the same for both sampling methods (Wilcoxon Matched Pairs test:  $Z = 0.36$ ,  $p = 0.71$ ,  $n = 8$  for all tests). Longworth detected  $4.75 \pm 1.38$  species (range 3–7 species) and Sherman detected  $4.62 \pm 1.18$  species (range 3–6 species). Other community parameters were similar for both sampling methods:

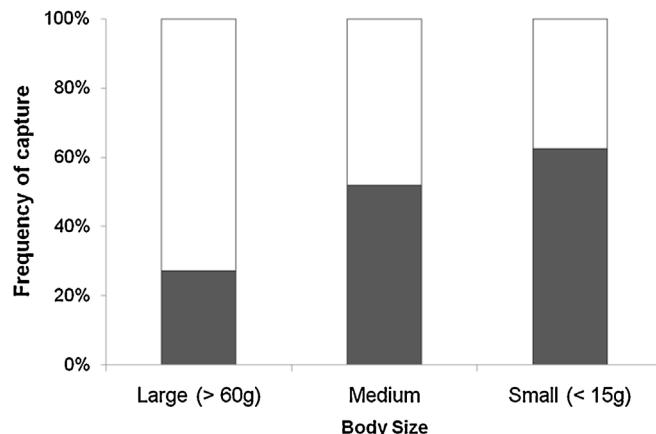


**Fig. 2.** Species accumulation curves ( $\pm$ SD) for either Longworth or Sherman traps along the 30 sampling occasions (six primary occasions with five secondary occasions each), and associated richness estimators (Chao1).

rarefied species richness ( $Z=0.33$ ,  $p=0.73$ ), Shannon diversity ( $Z=0.0$ ,  $p=1$ ), PIE Hulbert's index ( $Z=0.14$ ,  $p=0.88$ ), and dominance ( $Z=0.07$ ,  $p=0.94$ ). Similarity indexes between plots as estimated by every trap model were also similar (Chao-Jaccard:  $Z=0.31$ ,  $p=0.75$ ; shared species:  $Z=0.53$ ,  $p=0.59$ ,  $n=28$ ).

Two small mammal species were dominant throughout the study: the wood mouse (*Apodemus sylvaticus*, 287 individuals, 39.5%) and the bank vole (*Myodes glareolus*, 262 individuals, 36.1%), and the remaining 11 species showed frequencies lower than 10% (Table 1). Results of the fitting of log-linear models showed that total small mammal abundance significantly changed on a seasonal and interannual basis throughout the study period, with lower abundance in summer than in fall ( $G_1 = 141.31$ ,  $p < 0.0001$ ), and lower abundance in 2008 than in the following two years ( $G_2 = 132.03$ ,  $p < 0.0001$ ), without interactive effects ( $G_3 = 2.79$ ,  $p = 0.24$ ). The most frequent species, the wood mouse, showed the same pattern of abundance (season:  $G_1 = 6.73$ ,  $p = 0.01$ ; year:  $G_2 = 3.51$ ,  $p = 0.03$ ). *M. glareolus* and *Sorex araneus* only showed seasonal changes in abundance ( $F_{1,42} = 4.93$ ,  $p = 0.03$ ;  $F_{1,42} = 7.45$ ,  $p = 0.009$ , respectively), and *Eliomys quercinus* did not show any significant trend.

Both commercial live-traps captured exactly the same number of individuals over the three years (364 individuals each), and similar totals were evident when recaptures were included (Longworth: 728 catches; Sherman: 717 catches). Log-linear modeling showed that both traps were equally effective in catching small mammals ( $G_1^2 = 0.83$ ,  $p = 0.65$ ) and in detecting interannual variations in small mammal abundance (interaction year  $\times$  trap:  $G_2^2 = 0.83$ ,  $p = 0.65$ ). Nonetheless, significant trap biases were detected when species were considered in the analyses (interaction species  $\times$  trap:  $G_{12}^2 = 52.7$ ,  $p < 0.0001$ ). Three out of 13 species (23%) showed significant differences in trappability between methods (Table 2): *Eliomys quercinus* and *A. sapidus* were oversampled by Sherman and undersampled by Longworth traps, whereas *Sorex araneus* was oversampled by Longworth and undersampled by Sherman traps. The sampling biases observed in these three species resulted from conspicuous differences in body size between species. In fact, log-linear modeling showed that sampling efficiency by traps was related to small mammal size (interaction size  $\times$  trap:  $G_2^2 = 38.3$ ,  $p < 0.0001$ , Fig. 3). Large small mammals (adults weighting  $>60$  g) were oversampled by Sherman traps ( $G_2^2 = 30.08$ ,  $p < 0.001$ ), and small mammals (adults weighting  $<15$  g) were oversampled by Longworth traps ( $G_2^2 = 7.37$ ,  $p < 0.05$ ). Medium-sized small mammals were equally sampled by both methods ( $G_2^2 = 0.84$ ,  $p = 0.70$ ).



**Fig. 3.** Frequencies of occurrence of small mammals captured according to body size and trap type (closed bars: Longworth; open bars: Sherman traps).

There were no significant sex biases by both traps in the most frequent species: *A. sylvaticus* ( $G_1^2 = 0.08$ ,  $p = 0.76$ ), *M. glareolus* ( $G_1^2 = 0.94$ ,  $p = 0.33$ ), and *E. quercinus* ( $G_1^2 = 0.42$ ,  $p = 0.51$ ).

Despite harsh climatic conditions in some sampling sessions (especially in fall), trap mortality was relatively low and similar for both sampling methods (total 6.2%; 3.6% for Longworth and 2.6% for Sherman,  $G_1^2 = 0.42$ ,  $p = 0.51$ ). However, shrews took the bulk of mortality throughout the study period (24.7% for Longworth and 18.7% for Sherman:  $G_1^2 = 0.54$ ,  $p = 0.45$ ), being higher for shrews than for rodents ( $G_4^2 = 95.6$ ,  $p < 0.0001$ ).

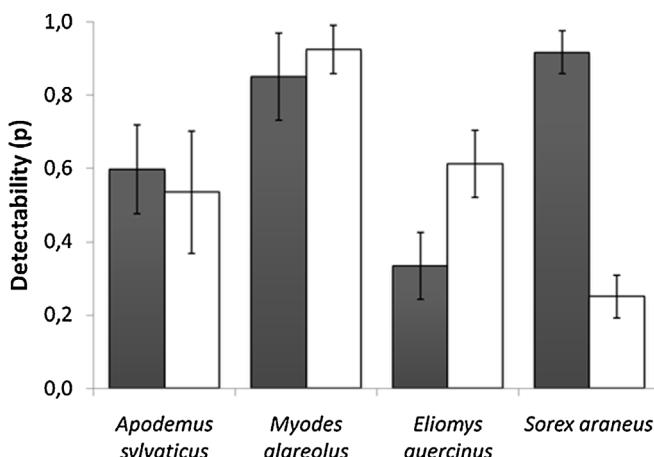
Average body mass of individuals trapped was larger in Sherman ( $30.03 \text{ g} \pm 17.13$  (SD), range 3–210 g) than in Longworth traps ( $25.31 \text{ g} \pm 11.54$ , range 3–80 g), being these differences highly significant ( $F_{1,1286} = 26.91$ ,  $p < 0.0001$ ). However, when considering the five more frequent species (95% of individuals trapped), most differences were not significant ( $F_{1,473} = 0.33$ ,  $p = 0.56$  for wood mice,  $F_{1,543} = 0.36$ ,  $p = 0.54$  for bank voles,  $F_{1,54} = 0.01$ ,  $p = 0.89$  for white-toothed shrews, and  $F_{1,31} = 0.21$ ,  $p = 0.64$  for the common shrews), except for garden dormice ( $F_{1,111} = 4.80$ ,  $p = 0.03$ ).

Site occupancy analysis (SOA) was performed with the four species more frequent and widely distributed (except for *C. russula* that was only present in one plot). The results showed marginal differences in seasonal detectability ( $p$ ) for *S. araneus* by both sampling methods (Wilcoxon Matched Pairs test:  $Z = 1.88$ ,  $p = 0.06$ ,  $n = 8$ ), but showed the same detectability for the most common small mammal species: *A. sylvaticus* ( $Z = 1.06$ ,  $p = 0.28$ ), *M. glareolus* ( $Z = 0.53$ ,  $p = 0.59$ ) and *E. quercinus* ( $Z = 1.06$ ,  $p = 0.25$ , Fig. 4). These differences rather affected occupancy estimates ( $\Psi$ ) that were slightly higher

**Table 2**

Likelihood-ratio Chi<sup>2</sup> tests (and associated  $P$  values) of frequencies of occurrence of 13 small mammal species sampled by two live-trapping methods after a two-way Log-linear model (interaction species  $\times$  trap model  $G_{12}^2 = 52.7$ ,  $p < 0.0001$ ). Significant  $P$ -values are marked in bold.

L-R Chi <sup>2</sup>	Longworth	Sherman	Total	P
<i>N. fodiens</i>	2.51	-1.09	1.42	<0.30
<i>S. araneus</i>	19.68	-12.52	7.16	<b>&lt;0.01</b>
<i>S. minutus</i>	-2.87	3.42	0.56	<0.30
<i>C. russula</i>	14.60	-11.98	2.62	<0.20
<i>G. glis</i>	-1.73	2.34	0.62	<0.30
<i>E. quercinus</i>	-38.79	62.09	23.30	<b>&lt;0.001</b>
<i>A. sylvaticus</i>	-12.38	12.69	0.31	<0.50
<i>M. musculus</i>	1.19	-0.68	0.51	<0.50
<i>A. sapidus</i>	-2.21	10.96	8.76	<b>&lt;0.01</b>
<i>M. glareolus</i>	30.01	-28.66	1.36	<0.30
<i>M. agrestis</i>	5.36	-4.18	1.18	<0.30
<i>M. arvalis</i>	6.20	-2.92	3.28	<0.10
<i>C. nivalis</i>	6.55	-4.91	1.63	<0.30



**Fig. 4.** Detectability ( $\pm$ SE) of the four small mammal species most frequently trapped according to trap type (closed bars: Longworth; open bars: Sherman traps).

than the observed (naïve) values for *A. sylvaticus* and *S. araneus*, and identical for *M. glareolus* and *E. quercinus*. Observed values were, on average, only  $3.2 \pm 3.7\%$  (SD) lower than occupancy estimates ( $\Psi$ ). Our results suggested that DI can be considered as a good surrogate of  $p$  only for *A. sylvaticus* ( $r = 0.76$ ,  $p = 0.02$ ,  $n = 12$ ).

#### 4. Discussion

Longworth and Sherman traps are the most used live-traps by investigators, being also recommended for monitoring programs (Flowerdew et al., 2004). However, there is still little information available on their efficiency working side by side for small mammal community assessments (Anthony et al., 2005; Kadjo et al., 2013; Morris, 1968), especially in mountain environments. Here we showed that both kinds of traps could be used interchangeably – without relevant biases – in small mammal community assessments where large species are infrequent. Estimates of community parameters and similarity indexes, sampling efficiency, detectability, mean weight, and sex-ratio of the most abundant species, were similar for both sampling methods.

The small mammal fauna of Andorra comprises, at least, 21 different species (Torre et al., 2014b), and in this study we detected only 13 (62%). Despite this low figure, we must stress that several species cannot be captured by the sampling methods used owing to the spatial distribution of the stations (i.e., placed on the ground in natural habitats) and to the variable size and behavior of small mammal species (Sibbald et al., 2006). Some species are too big to be trapped (i.e. marmots *Marmotta marmotta*, Pyrenean desman *Galemys pyrenaicus*), others are fossorial (mole *Talpa europaea*), some are arboreal (squirrel *Sciurus vulgaris*), and others live in human environments (rats *Rattus* spp.), so the number of species detected were in good agreement with expectations according to the habitats sampled and methods used. Furthermore, implementing monitoring programs for all the species present in an area can be impractical (Solari et al., 2002), and the selection of keystone/target species easy to monitor can be more effective and used as surrogates for ecosystem functioning (Solari et al., 2002). Furthermore, monitoring widespread species offers more information than monitoring rare species: population changes of common species can be measured with greater precision, allowing the detection of general changes over a large proportion of the landscape (Battersby and Greenwood, 2004).

We identified 728 small mammal individuals of 13 species, and this value was the same as the one expected by the richness estimator ( $13 \pm 0.25$  species). The asymptotic species richness

was 13.47, so 96.5% of the small mammal species actually present in the study area were recorded during the inventory. The local species list obtained from the analysis of live-trapping data can be considered complete (>90%; Moreno and Halffter, 2001). The only species missed from the inventory within the habitats sampled was the Pyrenean pine vole (*Microtus gerbei*), a semi-fossorial species potentially present at least in one of the studied plots (Torre et al., 2014b). In addition to effective species detection, results also showed that the trapping scheme used (0.56 ha grids with a 36 single-traps working during three consecutive nights) was able to capture all the individuals present within the area as competition for traps was unlikely due to the expected low overall trapping success. Despite other authors recommending higher trap densities and/or longer surveys (up to 8 nights, Conard et al., 2008) to improve precision and stability of the estimates, our findings showed that low density small grids will yield similar results when short surveys are repeated in time (i.e. for three consecutive years). These results can help to validate the method proposed for small mammal monitoring in Spain (SEMICE, Torre et al., 2011).

Changes in detectability (and derived occupancy estimates) of species can affect small mammal community assessments when detection probabilities are less than  $p < 0.15$  (Mackenzie et al., 2002). However, when detection probabilities are  $\geq 0.3$  the occupancy estimates ( $\Psi$ ) can be considered as unbiased when sampling was done in more than 5 occasions (Mackenzie et al., 2002), as was our case. All species analyzed (common species widely distributed in the area) showed mean values of detectability well above this threshold (average values of  $p$  ranging from 0.47 to 0.89), but one species (*S. araneus*) showed a significant decline of detectability when Sherman traps were used ( $p = 0.25$ ). However, this reduction did not affect occupancy estimates, evidenced that common shrews were undersampled by Sherman traps (28% of captures) and oversampled by Longworth traps (72% of captures). These results were in agreement with other authors who suggested highest efficiency of Longworth traps for caching shrews (Anthony et al., 2005).

In this study Longworth traps captured two species that were not captured by Sherman traps (*N. fodiens* and *M. musculus*), however, both species can in fact be captured by the latter (authors unpub.). This is not the case for the water vole (*A. sapidus*), which cannot be captured by Longworth traps due to their narrow tunnel entrance. Interestingly, two stoats were captured by Longworth traps, perhaps because they look like rodent burrow entrances. In one occasion, an adult stoat was found inside the trap with a field vole that was partially eaten, so the stoat was able to open the trap door that was closed by the previous entrance of the vole. A similar case of predation of small mammals within Longworth traps by weasels (*Mustela nivalis*) was reported (Lightfoot and Wallis, 1982).

Eight species (73%) showed a reduced number of captures over the three years ( $\leq 20$  individuals), accounting for 5.4% of total captures only, although seven out of these eight species can be equally sampled by both sampling methods. However, there was a high uncertainty in statistical comparisons of sampling efficiency when species reach these low levels of abundance, possibly due to stochastic processes. In sampling plots where species are represented by one or two individuals, captures may have been occurred in either Longworth or Sherman traps simply at random.

Longworth traps can be more suitable for small mammals than Sherman traps provided with a nesting place and water proofing, but the former had slightly higher (but non-significant) mortality rates for both rodents and shrews. These results were a priori unexpected, and somehow surprising, but were also confirmed by other authors (Anthony et al., 2005; but see Shonfield et al., 2013 for opposite results). Unfortunately, our results also confirmed the higher mortality rates for shrews than for rodents, but the mortality rate for shrews in our study (43.4%) was slightly lower than the average (47% in the review by Shonfield et al., 2013).

Summarizing, despite our results suggesting some specific biases (underestimation of two large species by Longworth traps and underestimation of one small species by Sherman traps), estimates of community parameters and similarity indexes, sampling efficiency, detectability, mean body mass, and sex-ratio of the most abundant species, were similar for both sampling methods. Overall, both kinds of traps could then be used interchangeably in small mammal community assessments without causing relevant biases where large species are infrequent, as happens in this study. Nonetheless, as far as communities hold larger species, the use of both commercial traps in combination, or the use of the largest trap (Sherman), are strongly recommended. Furthermore, high detectability of common small mammal species by the sampling methods used would allow the establishment of robust monitoring programs for the involved species, owing their important function in the terrestrial ecosystems as keystone prey, herbivores, or seed dispersers (Coppeto et al., 2006). Since detectability increases with survey duration (Kery and Schmidt, 2008), monitoring programs based on common species with high detectability will help to reduce the time invested and hence economic costs.

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## Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at <http://dx.doi.org/10.1016/j.ecolind.2016.02.017>.

## References

- Anthony, N.M., Ribic, C.A., Bautz, R., Garland Jr., T., 2005. Comparative effectiveness of Longworth and Sherman live traps. *Wildl. Soc. Bull.* 33, 1018–1026, [http://dx.doi.org/10.2193/0091-7648\(2005\)33\[1018:ceolas\]2.0.co;2](http://dx.doi.org/10.2193/0091-7648(2005)33[1018:ceolas]2.0.co;2).
- Battersby, J.E., Greenwood, J.J.D., 2004. Monitoring terrestrial mammals in the UK: past, present and future, using lessons from the bird world. *Mammal Rev.* 34 (1–2), 3–29.
- Bubb, P.J., Almond, R., Kapos, V., Stanwell-Smith, D., Jenkins, M., 2010. *Guidance for National Biodiversity Indicator Development and Use. UNEP-WCMC, Cambridge, UK*, pp. 120.
- Burger, J.R., Chesh, A.S., Castro, R.A., Ortiz Tolhuysen, L., Torre, I., Ebensperger, L.A., Hayes, L.D., 2009. The influence of trap type on evaluating population structure of the semifossorial and social rodent *Octodon degus*. *Acta Theriol. (Warsz.)* 54, 311–320, <http://dx.doi.org/10.4098/j.at.0001-7051.047.2008>.
- Cáceres, N.C., Napoli, R.P., Hannibal, W., 2011. Differential trapping success for small mammals using pitfall and standard cage traps in a woodland savannah region of southwestern Brazil. *Mammalia* 75, 45–52, <http://dx.doi.org/10.1515/mamm.2010.069>.
- Chao, A., Colwell, R.K., Lin, C.-W., Gotelli, N.J., 2009. Sufficient sampling for asymptotic minimum species richness estimators. *Ecology (Washington DC)* 90 (4), 1125–1133, <http://dx.doi.org/10.1890/07-2147.1>.
- Colwell, R.K., 2013. EstimateS: Statistical Estimation of Species Richness and Shared Species from Samples. Version 9. User's Guide and Application. <http://purl.oclc.org/estimates>.
- Conard, J.M., Baumgardt, J.A., Gipson, P.S., Althoff, D.P., 2008. The influence of trap density and sampling duration on the detection of small mammal species richness. *Acta Theriol. (Warsz.)* 53 (2), 1–14.
- Coppeto, S.A., Kelt, D.A., Van Vuren, D.H., Wilson, J.A., Bigelow, S., 2006. Habitat associations of small mammals at two spatial scales in the northern Sierra Nevada. *J. Mamm.* 87 (2), 402–413, <http://dx.doi.org/10.1644/05-mamm-a-086r1.1>.
- Díaz, M., Concepción, E.D., Alonso, C.L., 2015. Conservación de la biodiversidad en los sistemas forestales de Andalucía. In: Campos, P., Díaz, M. (Eds.), *Biodiversidad, usos del agua forestal y recolección de setas silvestres en los sistemas forestales de Andalucía. Memorias científicas de RECAMAN, vol. 2. Memoria 2.1. Editorial CSIC, Madrid*.
- Díaz-Francés, E., Soberón, J., 2005. Statistical estimation and model selection of species-accumulation functions. *Conserv. Biol.* 19 (2), 569–573, <http://dx.doi.org/10.1111/j.1523-1739.2005.00453.x>.
- Dos Santos-Filho, M., da Silva, D.J., Sanaotti, T.M., 2006. Efficiency of four trap types in sampling small mammals in forest fragments, Mato Grosso, Brazil. *Mastozool. Neotrop.* 13, 217–225.
- Environment Agency (EEA), 2010. *Assessing Biodiversity in Europe – The 2010 Report EEA, Copenhagen*, 45 pp.
- European Environment Agency (EEA), 2012. *Streamlining European Biodiversity Indicators 2020: Building a Future on Lessons Learnt from the SEBI 2010 Process. EEA, Copenhagen*, 50 pp.
- Folch, R., 1984. *El patrimoni natural d'Andorra. Els sistemes naturals andorrans i llur utilització*. Ketres editora, Barcelona, 448 pp.
- Flowerdew, J.R., Shore, R.F., Poulton, S.M.C., Sparks, T.H., 2004. Live trapping to monitor small mammals in Britain. *Mamm. Rev.* 34, 31–50, <http://dx.doi.org/10.1046/j.0305-1838.2003.00025.x>.
- Fonturbel, F.E., 2010. A methodological approach to assess the small mammal community diversity in the temperate rainforest of Patagonia. *Mamm. Biol.* 75 (4), 294–301, <http://dx.doi.org/10.1016/j.mambio.2009.03.012>.
- Gao, T., Nielsen, A.B., Hedblom, M., 2015. *Reviewing the strength of evidence of biodiversity indicators for forest ecosystems in Europe*. *Ecol. Indic.* 57, 420–434.
- Gotelli, N.J., Colwell, R.K., 2001. Quantifying biodiversity: procedures and pitfalls in the measurement and comparison of species richness. *Ecol. Lett.* 4, 379–391, <http://dx.doi.org/10.1046/j.1461-0248.2001.00230.x>.
- Gotelli, N.J., Entsminger, G.L., 2001. EcoSim: Null Models Software for Ecology. Version 7.0. Acquired Intelligence Inc. & Kesey-Bear <http://homepages.together.net/~gentmin/ecosim.htm>.
- Gurnell, J., Flowerdew, J.R., 2006. *Live Trapping Small Mammals: A Practical Guide. The Mammal Society, London*, 47 pp.
- Harris, S., Yalden, D.W., 2004. An integrated monitoring programme for terrestrial mammals in Britain. *Mamm. Rev.* 34 (1–2), 157–167, <http://dx.doi.org/10.1046/j.0305-1838.2003.00030.x>.
- Herrando, S., Brotons, L., Antón, M., Páramo, F., Villero, D., Titeux, N., Quesada, J., Stefanescu, C., 2016. Assessing impacts of land abandonment on Mediterranean biodiversity using indicators based on bird and butterfly monitoring data. *Environ. Conserv.*, <http://dx.doi.org/10.1017/S0376892915000260>.
- Ims, R.A., Henden, J.A., Killgreen, S.T., 2008. *Collapsing population cycles*. *Trends Ecol. Evol.* 23, 79–86.
- Jiménez-Valverde, A., Hortal, J., 2003. *Las curvas de acumulación de especies y la necesidad de evaluar la calidad de los inventarios biológicos*. *Revista Ibérica de Aracnología*.
- Kadjo, B., Kouadio, R.Y., Vogel, V., Dubey, S., Vogel, P., 2013. Assessment of terrestrial small mammals and a record of the critically endangered shrew *Crocidura wimmeri* in Banco National Park (Côte d'Ivoire). *Mammalia* 77, 439–446, <http://dx.doi.org/10.1515/mammalia-2012-0083>.
- Kery, M., Schmidt, B.R., 2008. *Imperfect detection and its consequences for monitoring for conservation*. *Community Ecol.* 9 (2), 207–216.
- Lambert, T.D., Malcolm, J.R., Zimmerman, B.L., 2005. Variation in small mammal species richness by trap height and trap type in southeastern Amazonia. *J. Mamm.* 86, 982–990, [http://dx.doi.org/10.1644/1545-1542\(2005\)86\[982:vismsr\]2.0.co;2](http://dx.doi.org/10.1644/1545-1542(2005)86[982:vismsr]2.0.co;2).
- Lightfoot, V., Wallis, S., 1982. *Predation of small mammals inside Longworth traps by a weasel*. *J. Zool.* 198, 521.
- Mackenzie, D.I., 2012. *PRESENCE User Manual. Proteus Wildlife Research Consultants*, 83 pp.
- Mackenzie, D.I., Nichols, J.D., Lachman, G.B., Droege, S., Royle, J.A., Langtimm, C.A., 2002. *Estimating site occupancy rates when detection probabilities are less than one*. *Ecology* 83, 2248–2255.
- Moreno, C.E., Halffter, G., 2001. On the measure of sampling effort used in species accumulation curves. *J. Appl. Ecol.* 38 (2), 487–490, <http://dx.doi.org/10.1046/j.1365-2664.2001.00590.x>.
- Morris, R.D., 1968. *A comparison of capture success between Sherman and Longworth live traps*. *Can. Field-Nat.* 82, 84–87.
- Morris, D.W., 1996. *Coeexistence of specialist and generalist rodents via habitat selection*. *Ecology* 77, 2352–2364.
- Newman, C., Buesching, C.D., Macdonald, D.W., 2003. Validating mammal monitoring methods and assessing the performance of volunteers in wildlife conservation – “Sed quis custodiet ipsos custodes?” *Biol. Conserv.* 113 (2), 189–197, [http://dx.doi.org/10.1016/S0006-3207\(02\)00374-9](http://dx.doi.org/10.1016/S0006-3207(02)00374-9).
- Nichols, J.D., Bailey, L.L., O'Connell, A.F., Talancy, N.W., Grant, E.H.C., Gilbert, A.T., Annand, E.M., Husband, T.P., Hines, J.E., 2008. *Multi-scale occupancy estimation and modelling using multiple detection methods*. *J. Appl. Ecol.* 45, 1321–1329.
- Nicolas, V., Colyn, M., 2006. *Relative efficiency of three types of small mammal traps in an African rainforest*. *Belg. J. Zool.* 136 (1), 107–111, Retrieved from: <Go to ISI>;</BIOSIS:PREV200600462017>.
- Otto, C.R.V., Roloff, G.J., 2011. Using multiple methods to assess detection probabilities of forest-floor wildlife. *J. Wildl. Manag.* 75, 423–431, <http://dx.doi.org/10.1002/jwmg.63>.
- Overmars, K.P., Schulp, C.J., Alkemade, R., Verburg, P.H., Temme, A.J., Omtzigt, N., Schaminée, J.H., 2014. *Developing a methodology for a species-based and spatially explicit indicator for biodiversity on agricultural land in the EU*. *Ecol. Indic.* 37, 186–198.
- Previtali, M.A., Lima, M., Meserve, P.L., Kelt, D.A., Gutierrez, J.R., 2009. Population dynamics of two sympatric rodents in a variable environment: rainfall, resource

- availability, and predation. *Ecology* 90, 1996–2006, <http://dx.doi.org/10.1890/08-0405.1>.
- Sibbald, S., Carter, P., Poulton, S., 2006. *Proposal for a National Monitoring Scheme for Small Mammals in the United Kingdom and the Republic of Ireland. The Mammal Society Research Report No. 6*, pp. 1–90.
- Shonfield, J., Do, R., Brooks, R.J., McAdam, A.G., 2013. Reducing accidental shrew mortality associated with small-mammal livetrapping I: an inter- and intrastudy analysis. *J. Mamm.* 94, 745–753, <http://dx.doi.org/10.1644/12-mamm-a-2711>.
- Sikes, R.S., Gannon, W.L., Animal Care and Use Committee, 2011. *Guidelines of the American Society of Mammalogists for the use of wild mammals in research. J. Mamm.* 92 (1), 235–253.
- Slade, N.A., Eifler, M.A., Gruenhagen, N.M., Davelos, A.L., 1993. Differential effectiveness of standard and long Sherman livetraps in capturing small mammals. *J. Mamm.* 74 (1), 156–161, <http://dx.doi.org/10.2307/1381915>.
- Slade, N.A., Blair, S.M., 2000. An empirical test of using counts of individuals captured as indices of population size. *J. Mamm.* 81, 1035–1045.
- Solari, S., Rodriguez, J.J., Vivar, E., Velazco, P.M., 2002. A framework for assessment and monitoring of small mammals in a lowland tropical forest. *Environ. Monit. Assess.* 76 (1), 89–104, <http://dx.doi.org/10.1023/a:1015272905263>.
- Stenseth, N.C., 1999. Population cycles in voles and lemmings, density dependence and phase dependence in a stochastic world. *Oikos* 87, 427–461.
- Szpunar, G., Aloise, G., Mazzotti, S., Nieder, L., Cristaldi, M., 2008. Effects of global climate change on terrestrial small mammal communities in Italy. *Fresenius Environ. Bull.* 17 (9B), 1526–1533.
- Torre, I., Arrizabalaga, A., Flaquer, C., 2004. Three methods for assessing richness and composition of small mammal communities. *J. Mamm.* 85, 524–530, <http://dx.doi.org/10.1644/bjk-112>.
- Torre, I., Diaz, M., Martinez-Padilla, J., Bonal, R., Vinuela, J., Fargallo, J.A., 2007. Cattle grazing, raptor abundance and small mammal communities in Mediterranean grasslands. *Basic Appl. Ecol.* 8, 565–575, <http://dx.doi.org/10.1016/j.baae.2006.09.016>.
- Torre, I., Arrizabalaga, A., Freixas, L., Pertierri, D., Raspall, A., 2011. *Primeros resultados del programa de seguimiento de micromamíferos comunes de España (SEMICE). Galemys* 23 (NE), 81–89.
- Torre, I., Diaz, M., Arrizabalaga, A., 2014a. Additive effects of climate and vegetation structure on the altitudinal distribution of greater white-toothed shrews *Crocidura russula* in a Mediterranean mountain range. *Acta Theriol. (Warsz.)* 59, 139–147, <http://dx.doi.org/10.1007/s13364-013-0128-y>.
- Torre, I., Freixas, L., Flaquer, C., Puig, X., Arrizabalaga, A., 2014b. *Petits mamífers i ratpenats d'Andorra. Monografies del CENMA. Govern d'Andorra*, 182 pp.
- Torre, I., Gracia-Quintas, L., Arrizabalaga, A., Baucells, J., Diaz, M., 2015. Are recent changes in the terrestrial small mammal communities related to land use change? A test using pellet analyses. *Ecol. Res.*, <http://dx.doi.org/10.1007/s11284-015-1279-x>.
- Voršek, P., Klvaňová, A., Wotton, S., Gregory, R.D., 2010. *A Best Practice Guide for Wild Bird Monitoring Schemes*. European Union, Brussels.
- Watkins, A.F., McWhirter, J.L., King, C.M., 2010. Variable detectability in long-term population surveys of small mammals. *Eur. J. Wildl. Res.* 56, 261–274.
- Wright, L.J., Newson, S.E., Noble, D.G., 2014. The value of a random sampling design for annual monitoring of national populations of larger British terrestrial mammals. *Eur. J. Wildl. Res.* 60 (2), 213–221.