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# Application of a health risk assessment model for cattle exposed to pesticides in contaminated drinking waters: A study case from the Pampas region, Argentina

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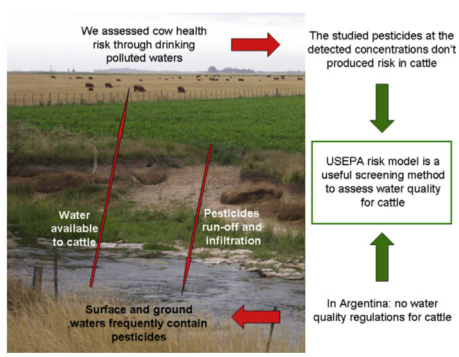
## HIGHLIGHTS

- Surface and ground waters in the Pampean plain frequently contain pesticides.
- We assessed the health risk for cows through drinking pesticides polluted waters.
- The studied pesticides at the detected concentrations don't produced risk in cows.
- USEPA risk model is a practical screening method to assess the cattle water quality.

## GRAPHICAL ABSTRACT

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## ABSTRACT

Using the USEPA methodology we estimated the probabilistic chronic risks for calves and adult cows due to pesticide exposure through oral intake of contaminated surface and ground waters in Tres Arroyos County (Argentina). Because published data on pesticide toxicity endpoints for cows are scarce, we used threshold levels based on interspecies extrapolation methods. The studied waters showed acceptable quality for cattle production since none of the pesticides were present at high-enough concentrations to potentially affect cow health. Moreover, ground waters had better quality than surface waters, with dieldrin and deltamethrin being the pesticides associated with the highest risk values in the former and the latter water compartments, respectively. Our study presents a novel use of the USEPA risk methodology proving it is useful for water quality evaluation in terms of pesticide toxicity for cattle production. This approach represents an alternative tool for water quality management in the absence of specific cattle pesticide regulatory limits.



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## 1. Introduction

Pesticide use in agriculture has become a common practice for preventing or reducing losses due to infectious plant diseases or plagues and improving the yield and quality of agricultural crops (Damalas and Eleftherohorinos, 2011; Bozdogan, 2014). The use of these chemicals can cause dispersion to multiple environmental compartments (e.g., soil, surface and ground waters, etc.) by means of drift, surface run-off and infiltration (Hildebrandt et al., 2008; Papadakis et al., 2015). Studies conducted in the last decade revealed the occurrence of pesticides in several groundwater aquifers worldwide (Hildebrandt et al., 2008; Melo et al., 2012; Shashavari et al., 2012) as well as in surface waters (Carriger and Rand, 2008; Centofanti et al., 2008; Loewy et al., 2011; Belenguer et al., 2014). In Argentina, several studies evidenced the presence of pesticides in the environment (Jergentz et al., 2005; Marino and Ronco, 2005; Silva et al., 2005; Peruzzo et al., 2008; Loewy et al., 2011; Bonansea et al., 2013; Miglioranza et al., 2013; De Gerónimo et al., 2014), including organochlorine pesticides which have been prohibited since 1990 (Di Marzio et al., 2010; Isla et al., 2010; Miglioranza et al., 2013; Ballesteros et al., 2014; Grondona et al., 2014). As pesticides residues in water and plants may be ingested by herbivores, cow meat and milk represent potential pesticide sources for humans, as reported in several studies (Ahmad et al., 2010; Bayat et al., 2010; Bulut et al., 2011; Fromberg et al., 2011; Nag and Raikwar, 2011; Avancini et al., 2013). Indeed, some studies conducted in Argentina revealed the occurrence of pesticides in agricultural or stockbreeding products (Villaamil Lepori et al., 2006, 2013; Ruíz et al., 2008) raising the awareness of what we consume.

Quality of surface and ground waters is an important aspect to consider for animal production to ensure good animal health and productivity however this aspect is often disregarded (Morgan, 2011). Among the established regulatory criteria promoted by the National Research Council (NRC) for assessing water quality for livestock production are odor, taste, pH, hardness, concentration of total dissolved solids, total dissolved oxygen, heavy metals, toxic minerals, organophosphates, hydrocarbons, nitrates, sodium, sulphates, iron and bacterial load (NASEM, 2016). In Argentina, water quality requirements for animal production have also been established (Bavera et al., 2001; Fernández Cirelli et al., 2010). However, the pyrethroids and organochlorine pesticides are not among the compounds for which regulations ensuring proper water quality for cattle have been set.

Studies conducted by Peluso et al. (2007) in Tres Arroyos County, Argentina, showed that surface and ground waters from a vast area of the county are polluted with organochlorine (α-Hexachlorocyclohexane −HCH−, γ-HCH, δ-HCH, aldrin, γ-chlordane, dichlorodiphenyldichloroethane −DDD−, dieldrin, endosulfan, endosulfan sulphate, heptachlor), and pyrethroid (deltamethrin and cypermethrin) pesticides. Because these waters are used for livestock production, the aim of this study was to apply a health risk model based on cattle chronic exposure to the pesticides through water intake. Thus, health risk assessment was performed for calves and adult cows destined for meat production using the USEPA risk model, in order to evaluate the suitability of this method as an alternative tool for water quality evaluation in cattle farming. The USEPA risk model allows characterizing the nature and magnitude of health risks due the exposure to stressors that may be present in the environment. This model is an estimate of the likelihood that a chemical agent of concern generates a toxicological effects in exposed people (USEPA, 1989). Although this model is widely used for human health risk estimation (Peluso et al., 2012, 2014; Chica-Olmo et al., 2017) there is a lack of studies on risk assessment applied on non-human animals. Thus, we consider that the cattle risk exposure to pesticides in drinking water is a novel use of the USEPA risk model with a potential utility in water quality management serving as an alternative tool to determine harmfulness when regulatory limits are absent.

## 2. Study area

Most of the province of Buenos Aires surface belongs to the Pampean plain region, including the study area. Major grain crops such as wheat, corn, soybean and sunflower are produced within this region; secondary crops are represented by sorghum, barley and linen. An important annual consumption of pesticides has been observed for the whole country, mainly related to the agricultural expansion in the last years: the amount (kg) of pesticides sales switched from 151.3 million in 2002 to 225 million in 2008 (Pórfido et al., 2014). Among the organochlorine pesticides mentioned in this study, HCH, aldrin, chlordane, DDD, dieldrin, and heptachlor were banned by Argentine law in 1990, and endosulfan and endosulfan sulphate are currently under a progressive elimination program (INTA, 2017a). Contrarily, pesticides based on deltamethrin and/or cypermethrin are freely commercialized.

The study area is located in the Tres Arroyos County (5962.88 $km^2$), southeast of Buenos Aires province (38°22′46″S − 60°16′38″W). This county plays an important role in the economy of the province due to the large extensions of land devoted to intensive agriculture and cattle-ranching (Carbone and Pícollo, 2002; Carbone, 2004). According to the 2013−2014 census carried out by the Ministry of Agriculture, Livestock and Fisheries of Argentina, 70.4% of the land in the county is used for agriculture whereas 23.1% is used for livestock production. The census showed that the main crops are soybean (237,170 ha), wheat (129,000 ha), sunflower (29,100 ha) and corn (26,700 ha) (SIIA, 2017). In the Pampean region, where our study took place, British cows and their crosses are the main cow breeds, with a predominance of Aberdeen Angus (INTA, 2007). In 2014, there were a total of 216,709 cattle heads in Tres Arroyos County, from which 91,240 were adult cows and 37,660 were calves (SENASA, 2017).

The Tres Arroyos basin has a surface area of 3017 $km^2$. Considering the 2000−2013 period, the average annual temperature is 14.5 °C, with an average annual minimum of 1.7 °C (in July) and an average annual maximum of 30 °C (in December); the average annual rain for this period is 765.6 mm (INTA, 2017b). The basin (Fig. 1) is formed by three shallow tributaries (first, second and third branches of the Tres Arroyos creek system) flowing through the city of Tres Arroyos, capital of the County. Downstream from the city, these three watercourses meet forming a single one: the Claromecó creek (A in Fig. 1). The latter runs throughout the rest of county and finally opens into the Argentine Sea. Two other watercourses flow in the same direction: the Quequén Salado River (B) and the Cristiano Muerto creek (C) (García Martínez et al., 2008). Due to the potential runoff of pollutants from the agricultural land into these watercourses, the water quality of the basin has been periodically monitored for pesticides (Peluso et al., 2011, 2014). Likewise, ground water wells (Figs. 2 and 3) were also tested for the presence of these substances (Othax et al., 2013). The Pampeano Aquifer System is the name of the ground water aquifer sitting below the Pampean plain from which water is extracted for drinking (Zabala et al., 2015).

## 3. Methodology

### 3.1. Cattle probabilistic risk assessment

To calculate the cow risk for the oral intake pathway, different pesticide daily exposure doses were estimated based on the USEPA

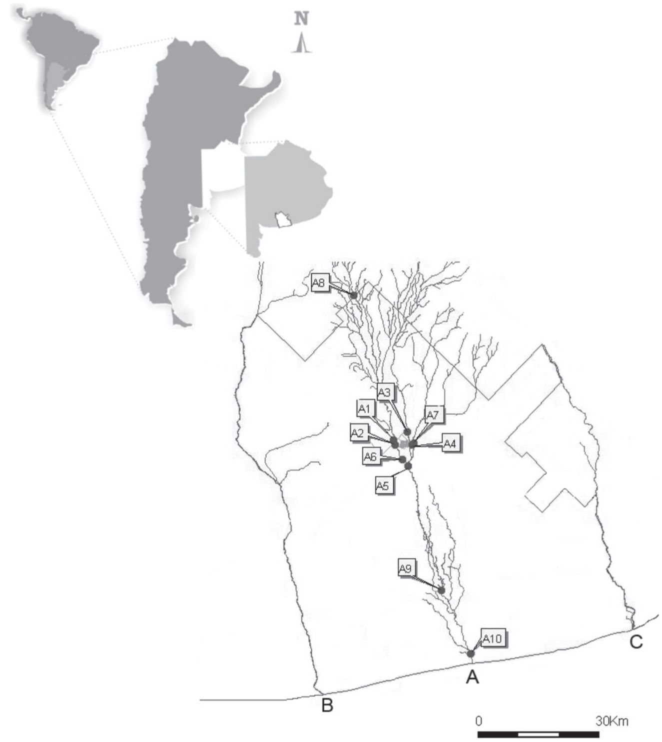


**Fig. 1.** Location of the study area, Tres Arroyos creek basin (Buenos Aires, Argentina). Sampling points in first, second and third branches of the Tres Arroyos creek system, and the Claromecó creek.

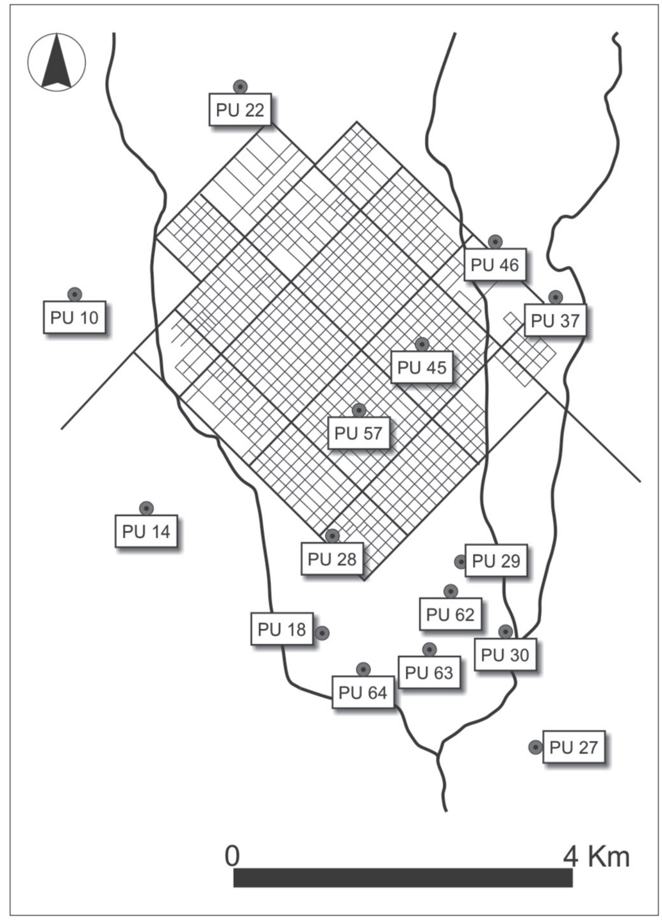


**Fig. 2.** Ground water wells within the urban and suburban area of the city of Tres Arroyos.

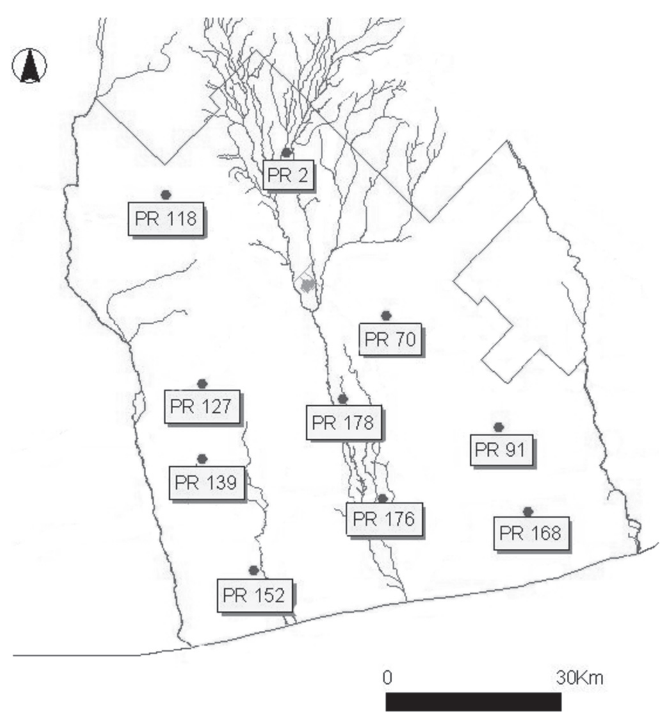


**Fig. 3.** Ground water wells within the rural area of the city of Tres Arroyos.

human risk methodology for chronic effects (USEPA, 1989, 1992a), following Eq. (1).

$$ADD = \frac{(C*IR)}{BWc} \tag{1}$$

where, *ADD* = Average daily dose of exposure due to water intake (mg of pesticide/kg of cow BW day); *C* = Concentration of the pesticide in water (mg/L); *IR* = Daily cow water intake rate (L d-1); *BWc* = Body weight of the exposed cattle (kg). Each ADD input variable is described in the next section.

The exposure doses were estimated for two different cattle age groups: calves (animals in between post lactation and nine months old) and mature cows (animals above three years old), both groups belonging to the Aberdeen Angus breed. An absorption ratio of 100% was assumed for all pesticide substances.

The risk (*ChR*), as shown below in Eq. (2), was calculated as the ratio between ADD and a threshold concentration acting as an oral reference dose (ORD), which is based on the non-observable adverse effects (NOAEL).

$$ChR = \frac{ADD}{ORDc} \tag{2}$$

Data on the toxicity of the studied chemicals for cattle are scarce therefore we needed to apply a NOAEL interspecies extrapolation methodology to overcome this problem. The cow NOAEL value was estimated applying a USEPA method derived from the ORD, which uses the body weight as the basis to extrapolate toxicity data obtained from assays performed in other animals (USEPA, 2011, 2014). The ORDc was calculated as shown in Eq. (3), based on USEPA (2011).

$$ORDc = \frac{[lta(ToxEnd)*BWlta]*\left(\frac{BWc}{BWlta}\right)^{0.75}}{BWc} \tag{3}$$

where, *ORDc* = *lta(ToxEnd)* = surrogate NOAEL based on laboratory animals (rats or dogs), *BWlta* = Body weight of the laboratory animal (in kg); *BWc* = Body weight of the exposed cattle (calf or mature cow; in kg).

The NOAEL values for rats and dogs were obtained from the Integrated Risk Information System (IRIS) database of USEPA (2015a, b) and WHO (2011), and are presented in Table 1.

*ChR* and *ORDc* values were calculated applying the Monte Carlo (MC) model through simple random sampling for 5000 trials based on the probabilistic distribution function (PDF) of C, IR and BWc using Crystal Ball 7.1 software (Decisioneering, 2007). The method consists in randomly selecting values, one value per trial from each input variable's PDF, and, with those values, generating the risk PDF as the final output. This new PDF contains the variability and uncertainty of the input variables.

Two types of risk were calculated: one considering the effect of each individual substance and the other considering the combined effect of all pesticides. For the latter, a cumulative risk index was employed which represents the risk caused by the simultaneous exposure to the whole group of substances. An additive model was used to estimate the cumulative risk. This risk type is calculated adding the MC risk value of each pesticide for each iteration. When the MC simulation is finished the resulting PDF represents the cumulative risk.

For each risk PDF obtained, the arithmetic mean, the standard deviation, the maximum, and the 95th percentile value were used as risk indicators. The analysis of the risk results was based on the 95th percentile of the distribution as a high-end risk indicator (semi-conservative approach). The *ChR*-limit value for considering the individual or cumulative risk as significant was one.

## 3.2. Description of model input variables

### 3.2.1. Concentration of pesticides in water

To obtain the concentration PDF for each pesticide (the C term from Eq. (1)), all concentrations measured from each sampling site were used as input values. The surface water samples were obtained from 10 sites along the three branches of Tres Arroyos creek system and the Claromecó Creek (A1 to A10) (Fig. 1). Sampling was conducted quarterly between June 2008 and December 2010 (n = 110), during dry periods, in the absence of rain at least 5 days before sampling (Peluso et al., 2012). Water samples were collected in amber glass bottles with internal teflon tops. The samples were taken in the middle of the watercourse at the subsurface level (30 cm below surface) and stored at 4 °C until analysed the next day. Samples were stirred before being processed.

The concentrations of organochlorine pesticides (α-Hexachlorocyclohexane −HCH−, γ-HCH, δ-HCH, aldrin, γ-chlordane, dichlorodiphenyldichloroethane −DDD−, dieldrin, endosulfan, endosulfan sulphate, heptachlor), and pyrethroid pesticides (deltamethrin and cypermethrin) were measured with a Hewlett-Packard 5890II gas chromatograph with ECD and NPD detectors, according to USEPA methods SW 846 M 8081 CG ECD, M 508-CG ECD (USEPA, 2007a) and SW 846 M 1699 CG NPD (USEPA, 2007b), respectively.

The ground water samples were obtained from 25 active wells, 15 within the urban and suburban area of the city of Tres Arroyos (PU10, 14, 18, 22, 27, 28, 29, 30, 37, 45, 46, 57, 62, 63 and 64 in Fig. 2) and 10 within the rural area (PR2, 70, 91, 118, 127, 139, 152, 168, 176, 178 in Fig. 3). All wells had approximately the same depth, with a mean value of 15 m. Ground water samples (n = 300) were taken quarterly in coincidence with the surface water sampling. Local water pumps were used to withdraw samples. The samples were stored at 4 °C and analysed the next day. Samples were stirred before being processed. Pesticide quantification was performed

**Table 1**
Pesticide concentration probability density functions (PDF) with their descriptive parameters grouped by water compartment (in mg/L) and toxicological endpoint values (in mg/kg of the laboratory animal).

| Pesticides | DL[l] | Surface water | | | | | | Ground water | | | | | | Tox. Endpoint | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | PDF | Min[a] | Max[b] | Mean[c] | SD[d] | P95[e] | PDF | Min | Max | Mean | SD | P95 | | |
| Aldrin | $2.00E^{-07}$ | Logn[f] | $2.00E^{-07}$ | $4.94E^{-06}$ | $1.55E^{-06}$ | $7.38E^{-07}$ | $3.05E^{-06}$ | Min E[g] | $2.09E^{-07}$ | $3.46E^{-07}$ | $2.52E^{-07}$ | $2.88E^{-08}$ | $3.04E^{-07}$ | 0.025 | NOAEL (rat) |
| α-Chlordane | $1.00E^{-07}$ | – | – | – | – | – | – | Logi[h] | $2.46E^{-07}$ | $1.08E^{-06}$ | $3.78E^{-07}$ | $1.14E^{-07}$ | $5.98E^{-07}$ | 0.050 | NOAEL (rat) |
| γ-Chlordane | $4.00E^{-07}$ | Min E | $4.00E^{-07}$ | $2.55E^{-06}$ | $9.74E^{-07}$ | $5.41E^{-07}$ | $1.82E^{-06}$ | Min E | $4.23E^{-07}$ | $7.25E^{-07}$ | $5.13E^{-07}$ | $5.74E^{-08}$ | $6.15E^{-07}$ | 0.050 | NOAEL (rat) |
| Dieldrin | $1.00E^{-06}$ | Beta[i] | $1.00E^{-06}$ | $1.60E^{-06}$ | $9.96E^{-07}$ | $3.56E^{-07}$ | $7.33E^{-05}$ | Logn | $2.29E^{-06}$ | $1.66E^{-05}$ | $4.20E^{-06}$ | $1.30E^{-06}$ | $6.34E^{-06}$ | 0.005 | NOAEL (rat) |
| Endosulfan I | $1.00E^{-07}$ | Min E | $1.00E^{-07}$ | $1.47E^{-06}$ | $4.85E^{-07}$ | $2.81E^{-07}$ | $9.17E^{-07}$ | Logi | $4.30E^{-07}$ | $1.71E^{-06}$ | $5.70E^{-07}$ | $1.19E^{-07}$ | $8.13E^{-07}$ | 0.600 | NOAEL (rat) |
| Endosulfan II | $9.00E^{-07}$ | – | – | – | – | – | – | Pare[j] | $9.00E^{-07}$ | $1.33E^{-06}$ | $9.52E^{-07}$ | $5.19E^{-08}$ | $1.06E^{-06}$ | 0.600 | NOAEL (rat) |
| Endosulfan Sulphate | $2.50E^{-06}$ | Min E | $2.50E^{-06}$ | $1.16E^{-05}$ | $5.84E^{-06}$ | $2.50E^{-06}$ | $9.16E^{-06}$ | Min E | $2.82E^{-06}$ | $7.32E^{-06}$ | $4.21E^{-06}$ | $8.75E^{-07}$ | $5.93E^{-06}$ | 0.600 | NOAEL (rat) |
| Endrin | $5.00E^{-07}$ | – | – | – | – | – | – | Min E | $5.51E^{-07}$ | $1.34E^{-06}$ | $8.05E^{-07}$ | $1.72E^{-07}$ | $1.14E^{-06}$ | 0.025 | NOAEL (dog) |
| α-HCH | $6.00E^{-07}$ | Max E[k] | $6.00E^{-07}$ | $2.78E^{-05}$ | $8.06E^{-06}$ | $4.83E^{-06}$ | $1.36E^{-05}$ | Min E | $3.21E^{-05}$ | $2.78E^{-04}$ | $1.09E^{-04}$ | $5.10E^{-05}$ | $1.98E^{-04}$ | 0.330 | NOAEL (rat) |
| β-HCH | $2.60E^{-06}$ | – | – | – | – | – | – | Min E | $5.00E^{-06}$ | $4.07E^{-05}$ | $1.68E^{-05}$ | $7.73E^{-06}$ | $3.17E^{-05}$ | 0.330 | NOAEL (rat) |
| δ-HCH | $4.00E^{-08}$ | Min E | $4.00E^{-08}$ | $6.23E^{-07}$ | $2.15E^{-07}$ | $1.26E^{-07}$ | $4.03E^{-07}$ | Min E | $1.83E^{-07}$ | $9.22E^{-07}$ | $4.06E^{-07}$ | $1.47E^{-07}$ | $6.75E^{-07}$ | 0.330 | NOAEL (rat) |
| γ-HCH | $5.00E^{-07}$ | Pare | $5.00E^{-07}$ | $2.79E^{-06}$ | $1.29E^{-06}$ | $4.12E^{-07}$ | $8.48E^{-07}$ | Min E | $6.91E^{-07}$ | $3.71E^{-06}$ | $1.64E^{-06}$ | $6.31E^{-07}$ | $2.86E^{-06}$ | 0.330 | NOAEL (rat) |
| Heptachlor | $4.50E^{-06}$ | Min E | $1.17E^{-05}$ | $2.81E^{-05}$ | $1.37E^{-05}$ | $5.46E^{-06}$ | $2.16E^{-05}$ | – | – | – | – | – | – | 0.025 | NOAEL (dog) |
| Deltamethrin | $2.00E^{-06}$ | Max E | $8.89E^{-06}$ | $8.91E^{-05}$ | $6.17E^{-05}$ | $1.71E^{-05}$ | $8.62E^{-05}$ | – | – | – | – | – | – | 1.000 | NOAEL (rat) |
| Cypermethrin | $2.00E^{-04}$ | Min E | $3.00E^{-02}$ | $6.94E^{-02}$ | $2.75E^{-02}$ | $1.50E^{-02}$ | $6.94E^{-02}$ | – | – | – | – | – | – | 7.500 | NOAEL (rat) |

References:
[a] Minimum.
[b] Maximum.
[c] Arithmetic mean.
[d] Standard deviation.
[e] 95th percentile.
[f] Lognormal.
[g] Minimum extreme.
[h] Logistic.
[i] Beta.
[j] Pareto.
[k] Maximum extreme.
[l] Detection limit.

following the same analytical method as for the surface water samples.

The concentration term for each pesticide in the exposure calculation model both for surface and ground water was based on the probability distributions obtained using all the concentrations measured in each water compartment (surface or ground water). Therefore, each substance concentration term at each water compartment is a synthetic integration of the monitored sites. The probability distribution model for each substance concentration was fitted with Crystal Ball 7.1 (Decisioneering, 2007); the software matches the concentration data with each of the continuous probability distributions (beta, exponential, gamma, normal, lognormal, logistic, Pareto, Weibull, etc.), selecting the best-fit curve after running Anderson Darling goodness-of-fit tests to detect the set of parameters of the distribution that best describes the characteristics of the data. The non-detect concentrations of each substance were replaced by the 95% upper confidence limit (95% UCL) of the arithmetic mean of the detected concentrations (Peluso et al., 2012, 2014). The 95% UCL of a mean is a value that, when calculated repeatedly for randomly drawn subsets of data from the same site, equals or exceeds the true mean value 95% of the times (USEPA, 1992b). The UCL estimation was performed using ProUCL software v.4.1 (USEPA, 2010), which carries out a number of parametric and non-parametric tests, and suggests the most appropriate UCL value to use based on the distribution of the data.

The PDF of each substance from each compartment was truncated at the limit of detection and at the maximum observed concentration in its corresponding water compartment, as left and right tails, respectively. The best-fit probability distribution model and descriptive parameters of each pesticide in surface and ground waters are presented in Table 1.

#### 3.2.2. Intake rate

Water quality requirements for cattle have been widely studied and several authors have established the relationship between cattle daily water consumption and temperature (Winchester and Morris, 1956; Lardner et al., 2005; Arias and Mader, 2011). In this study the daily water intake rate for cows (IR from Eq. (1)) was based on NASEM (2016). IR was probabilistically treated assuming a triangular distribution. The mode was set at 40.9 L d-1, which corresponds to the annual mean temperature of the study region. The minimum and the maximum IR for adult cows were set at 32.9 and 78 L d-1, based on the daily water intake rates for winter and summer in the study area, respectively. For calves, 15 and 36 L d-1 were the selected values for the lower and upper limits of the triangular IR PDF, with a mode of 18.9 L d-1.

#### 3.2.3. Weight

Cow body weight distribution (BWc from Eqs. (1) and (3)) followed a normal PDF. In the case of adult cows, the arithmetic mean and standard deviation were respectively set at 545 and 12 kg; for calves the values were 136 and 2 kg, respectively (Lardner et al., 2005). The PDFs were truncated at the 5th and 95th percentiles values, which constituted the lower and upper limits of the weight distribution, respectively.

## 4. Results and discussion

According to our analysis, oral intake of surface and ground waters from the study area would not produce chronic health effects neither in calves nor in adult cows since none of the risk values exceeded one ($ChR \leq 1$) even when considering the cumulative exposure (Table 2). These results are consistent with the absence of toxicological symptomatology associated with organochlorine and pyrethroid intoxication of cattle in the study area. Both organochlorine and pyrethroid pesticide intoxications can cause CNS (Central Nervous System) impairment signs: neuromuscular tremors, convulsions, behavioral changes and increased body temperature. Further, severe intoxication can cause death by respiratory failure (Merck, 2016). Poisoning in cattle can cause significant economic losses (García et al., 2017). None of these were observed or reported for this area.

Both risks (i.e., the risk due to exposure to a single pesticide and the one due to simultaneous exposure to all pesticides) were at least two or three orders of magnitude below the limit. We also observed that there was a higher risk associated with the oral intake of surface water than of ground water, roughly with a one-order-of-magnitude difference (i.e., $6.03\,E^{-02}$ - $9.33\,E^{-03}$ and $8.01\,E^{-02}$ - $1.81\,E^{-02}$ for adult cows and calves, respectively). Furthermore, risks were higher for calves than for adult cows. Adult cows consumes 0.07 L d-1 per kilogram of weight, and calves 0.14 L d-1 per kilogram of body weight. This increased water consumption by calves is what generates the increase in the potential risk in these animals. This highlights the importance of considering different age groups for the calculation of risk, since the rate of intake varies according to the weight of the exposed individual. This agrees with García et al. (2017) who observed that young cattle are more susceptible than adult cows.

Deltamethrin and dieldrin were the most risky substances in surface and ground waters, respectively. The ground water cumulative risk was mostly attributed to dieldrin (almost a 94% and a 96% contribution for adult cows and calves, respectively). In contrast, the surface waters cumulative risk was almost 99% due to the contribution of deltamethrin. Regarding the toxicity of deltamethrin and dieldrin, the former is less toxic than the latter. Because in surface waters deltamethrin has a much higher risk value than that for dieldrin (two-orders-of-magnitude difference), the measured concentration of deltamethrin was responsible for the highest risk attributed to this water compartment. Nonetheless, water could still be considered safe for cattle to drink. The analysis was performed under a conservative condition premise because the risk was calculated assuming a chronic exposure scenario, which is more conservative than an acute scenario. The results on pesticide exposure risk for cattle in the Tres Arroyos County complement the studies on pesticide exposure risk for humans performed by Othax et al. (2013) and Peluso et al. (2014) for the same geographical region.

Although information related to the quality of drinking water for animal production is available (Bavera et al., 2001; Fernández Cirelli et al., 2010), Argentine regulations do not address pyrethroid and organochlorine pesticides safety levels. Therefore, we consider that the USEPA health risk assessment could be a good surrogate tool to test pesticide polluted water and evaluate health hazard. To our awareness, no publications were available on health risk assessment modelling for cattle exposed to pesticides through oral intake of contaminated water until this study. Beyond the uncertainties discussed in the next paragraphs, the method that we applied could constitute a useful and novel screening tool for water quality evaluation in the scarcity of regulatory limits for pesticides in cattle.

Health risk assessments have operational advantages over the traditional procedure of comparing the measured concentrations of a compound against its safety limit (Peluso et al., 2012). One of these advantages is that, in the absence of regulatory toxicological safety limits, health risk assessment becomes practical and useful for management purposes in order to evaluate hazardous effects of contaminants through different exposure routes (water, air, food, skin contact) and to infer safety levels.

Despite its applicability for assessing risk, our study bears some uncertainties that are discussed as follows. Each pesticide concentration PDF for each water compartment was obtained by

**Table 2**
Adult cow and calf risk values for the oral intake of Tres Arroyos County surface and ground waters containing pesticides.

| Media | Pesticides | Adult Cow | | | | | Calf | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | | Min[a] | Max[b] | Mean | SD[c] | P95[d] | Min | Max | Mean | SD | P95 |
| SW[e] | Aldrin | 7.76E−06 | 1.46E−04 | 4.28E−05 | 2.23E−05 | 8.62E−05 | 9.15E−06 | 1.61E−04 | 5.54E−05 | 2.76E−05 | 1.07E−04 |
| | γ-Chlordane | 3.83E−06 | 4.30E−05 | 1.32E−05 | 5.82E−06 | 2.39E−05 | 5.15E−06 | 4.92E−05 | 1.72E−05 | 7.72E−06 | 3.23E−05 |
| | Dieldrin | 1.09E−04 | 2.54E−04 | 1.64E−04 | 3.24E−05 | 2.25E−04 | 1.40E−04 | 3.22E−04 | 2.14E−04 | 4.10E−05 | 2.99E−04 |
| | Endosulfan I | 7.65E−08 | 1.40E−06 | 5.04E−07 | 2.77E−07 | 1.03E−06 | 1.17E−07 | 2.14E−06 | 6.56E−07 | 3.58E−07 | 1.42E−06 |
| | Endosulfan Sulphate | 1.90E−06 | 1.36E−05 | 6.17E−06 | 2.30E−06 | 1.06E−05 | 2.56E−06 | 2.25E−05 | 8.06E−06 | 3.04E−06 | 1.37E−05 |
| | α-HCH | 1.07E−06 | 5.93E−05 | 1.33E−05 | 7.68E−06 | 2.79E−05 | 1.75E−06 | 8.91E−05 | 1.73E−05 | 9.97E−06 | 3.62E−05 |
| | γ-HCH | 1.07E−06 | 2.45E−06 | 1.62E−06 | 3.17E−07 | 2.21E−06 | 1.40E−06 | 3.20E−06 | 2.11E−06 | 4.03E−07 | 2.93E−06 |
| | δ-HCH | 5.40E−08 | 1.29E−06 | 4.06E−07 | 2.34E−07 | 8.50E−07 | 7.46E−08 | 1.56E−06 | 5.27E−07 | 2.94E−07 | 1.11E−06 |
| | Heptachlor | 7.89E−05 | 3.29E−04 | 1.58E−04 | 4.35E−05 | 2.48E−04 | 1.07E−04 | 4.46E−04 | 2.06E−04 | 5.46E−05 | 3.12E−04 |
| | Deltamethrin | 2.90E−02 | 6.63E−02 | 4.38E−02 | 8.58E−03 | 5.98E−02 | 3.78E−02 | 8.67E−02 | 5.71E−02 | 1.09E−02 | 7.94E−02 |
| | Cypermethrin | 8.20E−07 | 1.06E−05 | 5.20E−06 | 1.77E−06 | 8.16E−06 | 1.16E−02 | 1.39E−05 | 6.77E−06 | 2.31E−06 | 1.11E−05 |
| | Cumulative Risk | 2.92E−02 | 6.70E−02 | 4.43E−02 | 8.66E−03 | 6.03E−02 | 3.82E−02 | 8.77E−02 | 5.76E−02 | 1.10E−02 | 8.01E−02 |
| GW[f] | Aldrin | 3.63E−06 | 1.15E−05 | 6.36E−06 | 1.45E−06 | 9.13E−06 | 4.86E−06 | 1.50E−05 | 8.31E−06 | 1.94E−06 | 1.19E−05 |
| | α-Chlordane | 2.20E−06 | 1.72E−05 | 4.82E−06 | 1.75E−06 | 8.13E−06 | 2.74E−06 | 1.92E−05 | 6.27E−06 | 2.20E−06 | 1.04E−05 |
| | γ-Chlordane | 3.74E−06 | 1.17E−05 | 6.43E−06 | 1.43E−06 | 9.07E−06 | 4.82E−06 | 1.50E−05 | 8.41E−06 | 1.95E−06 | 1.21E−05 |
| | Dieldrin | 2.22E−03 | 1.70E−02 | 5.35E−03 | 1.88E−03 | 8.80E−03 | 2.73E−03 | 2.08E−02 | 6.97E−03 | 2.39E−03 | 1.13E−02 |
| | Endosulfan I | 3.09E−07 | 1.39E−06 | 6.06E−07 | 1.68E−07 | 9.10E−07 | 4.00E−07 | 1.82E−06 | 7.91E−07 | 2.24E−07 | 1.22E−06 |
| | Endosulfan II | 6.36E−07 | 1.67E−06 | 1.00E−06 | 1.98E−07 | 1.36E−06 | 8.18E−07 | 2.22E−06 | 1.31E−06 | 2.65E−07 | 1.79E−06 |
| | Endosulfan Sulphate | 2.11E−06 | 1.02E−05 | 4.43E−06 | 1.30E−06 | 6.82E−06 | 2.78E−06 | 1.35E−05 | 5.79E−06 | 1.73E−06 | 9.04E−06 |
| | Endrin | 3.81E−06 | 1.54E−05 | 7.73E−06 | 2.19E−06 | 1.17E−05 | 4.95E−06 | 2.20E−05 | 1.01E−05 | 2.93E−06 | 1.55E−05 |
| | α-HCH | 4.31E−05 | 6.61E−04 | 2.11E−04 | 1.07E−04 | 4.11E−04 | 5.75E−05 | 8.43E−04 | 2.75E−04 | 1.43E−04 | 5.35E−04 |
| | β-HCH | 7.63E−06 | 1.04E−04 | 3.27E−05 | 1.69E−05 | 6.34E−05 | 8.43E−06 | 1.44E−04 | 4.29E−05 | 2.25E−05 | 8.47E−05 |
| | δ-HCH | 2.53E−07 | 2.11E−06 | 7.77E−07 | 3.35E−07 | 1.43E−06 | 3.28E−07 | 2.74E−06 | 1.01E−06 | 4.24E−07 | 1.81E−06 |
| | γ-HCH | 1.02E−06 | 9.14E−06 | 3.13E−06 | 1.39E−06 | 5.84E−06 | 1.22E−06 | 1.28E−05 | 4.10E−06 | 1.86E−06 | 7.70E−06 |
| | Cumulative Risk | 2.36E−03 | 1.73E−02 | 5.63E−03 | 1.91E−03 | 9.33E−03 | 3.01E−03 | 2.61E−02 | 7.34E−03 | 2.44E−03 | 1.18E−02 |

References:
[a] Minimum.
[b] Maximum.
[c] Standard deviation.
[d] 95th percentile.
[e] Surface water.
[f] Ground water.

pooling all data from all the sampling sites and obtaining the best-fit regression curve. Not having sufficient site-specific data prevented us from studying the spatial distribution of pesticides within the study area. We are also aware that new data from new sampling events could change the pesticides' PDFs. However, although this could reduce the uncertainty, it does not influence the model structure. In other words, new data could change the risk values but the risk model remains the same.

The use of cattle as the target species instead of humans imply considering several significant differences with regard to the exposure parameters and the pesticides toxicological data. Concerning the exposure parameters, to reduce the uncertainty associated with considering a single life stage for cattle, we decided to carry out the assessment on two different life stages: calves and matures cows. This condition required that the estimation of some input variables (i.e., IR and BWc) were performed accounting for age. For that reason, the risk was estimated using a probabilistic approach and the above-mentioned input variables were considered as PDFs, and specific to each life stage. The water ingestion rate and the body weight used during the cow health risk assessment are main issues that need to be cleared. As previously mentioned, cattle water quality requirements have been widely studied. However, we were unable to obtain local information on daily variability in water ingestion for calves and mature cows to build accurate local IR PDFs. Therefore, for the estimations we used a triangular IR PDF, with mode, minimum and maximum values based on NASEM (2016) according to the weight of each of the considered life stages. In the absence of specific local studies on body weigh distribution for cattle, we took the minimum, maximum and mean BW for calves and adult cows from Lardner et al. (2005) to build the BWc PDFs.

Regarding pesticides toxicological data, bioassays have been the main source of information to identify chemicals capable of producing adverse effects in humans, therefore human health risk assessments rely on animal exposure data, mostly from rodents and dogs (Simons, 2014). This approach should be also valid in case of health risk assessment for cattle. However, there is a lack of specific toxicological data for cattle exposed to environmental pollutants (including the pesticides considered in this study) which makes more complex carrying out a health risk assessment. Although the rat NOAEL was used as a surrogate toxicity endpoint in an equine risk assessment carried out by Schleier III et al. (2008), we otherwise preferred to extrapolate the missing cattle toxicological data from rodents and dogs, as done by Riviere et al. (1997), applying the allometric method because of the following advantages. This method is based on the premise that many physiologic parameters are a function of the size (body weight) of the animal species (Ritschel et al., 1992). The use of a fractional power of body weight to derive toxicologically equivalent doses across species is an accepted practice supported by literature (Sharma and McNeill, 2009; Wang and Prueksaritanont, 2010; USEPA, 2011; Moyer et al., 2014). Although extrapolation provides an approach to overcome these data gaps, this method also contributes to reduce the level of uncertainty (Donohoe et al., 2000). Extrapolation of parameters values from one to another species and finding accurate toxicity endpoints are among the key areas of uncertainty analysis in both ecological and human health risk assessment (Raimondo et al., 2007).

Beyond the broad discussion about the allometric principles used for interspecies extrapolation in toxicological studies

(Schneider et al., 2004; Sharma and McNeill, 2009) the main application of the allometric approach relies in scaling up exposure estimates from laboratory animals to humans. Based on this premise, we took the same approach for cattle, which could be considered a body-mass-based approach for extrapolating the NOAEL from rats and dogs to cows. While this approach implies an additional contribution to uncertainty, we attempted to minimize it considering the toxicity endpoints as equal or interchangeable due to the fact that all are mammals. Accordingly, we still consider that the allometric approach is an appropriate method to make up for the missing toxicity data and that it is useful in cattle health risk assessment.

Further, particular aspects related to cow pesticide toxicology (water ingestion, chemical absorption, metabolization, excretion, bioaccumulation, etc.) seem to be disregarded but in fact they are already considered within the toxicological safe level (the cow oral reference dose) used for the USEPA risk model as a whole. However, and despite its simplicity, the USEPA human and ecological risk assessment guidances are adopted by many state environmental protection agencies in the U.S. as well as in other countries (Lester et al., 2007). Also, the adoption of the USEPA guidances improved food safety, human health and environmental protection through the risk assessment of pesticides (Damalas and Eleftherohorinos, 2011).

To us, the USEPA model applied in this study represents an appropriate screening method for assessing cow health risk despite its simplistic way of looking at the whole exposure process. We consider this is a novel application for pesticide exposure risk analysis in the absence of policies regulating water quality for cattle, which could contribute to livestock health protection.

## 5. Conclusions

The study showed that both surface and ground waters from Tres Arroyos County, Argentina, would not produce chronic health effects to cattle through drinking. None of the pesticides (aldrin, α and γ-chlordane, dieldrin, endosulfan I and II, endosulfan sulphate, endrin, α, β, δ, and γ-HCH, heptachlor, deltamethrin and cypermethrin) at the observed concentrations were toxic to cattle. There was a higher risk associated with the cow oral intake of surface than of ground waters and that the risk values were higher for calves than for adult cows. Deltamethrin and dieldrin were the main contributors to the cattle risk associated with surface and ground waters, respectively.

The methodology used in this study constitutes a practical screening tool for cattle health risk assessment based on a simplistic way of looking at the pesticide exposure process through water ingestion. In the absence of a more accurate pesticide evaluation tool for cattle farming, the USEPA risk model application, as presented here, could contribute to fill up this information gap.

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