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Chapter 8 - **Spatial dimensions of the risks of rodenticide use to non-target small mammals and applications in spatially explicit risk modeling**

Coeurdassier M.¹, Fritsch C.¹, Jacquot M.¹, van den Brink N.², Giraudoux P.^{1,3}

¹Laboratoire Chrono-environment, UMR 6249 CNRS/Université Bourgogne Franche-Comté, Usc INRA, 16 route de Gray, F-25030 Besançon cedex, FRANCE

Contact : M. Coeurdassier

Phone: +33 (0)381 665 741

E-mail: michael.coeurdassier@univ-fcomte.fr

²Wageningen University, Division of Toxicology, Box 8000 6700 EA, Wageningen, The Netherland.

³Institut Universitaire de France, Paris, FRANCE

Abstract – Both target and non-target small mammals are exposed to rodenticides (AR). A better understanding of the drivers controlling this exposure is critical for the conservation of threatened small mammal species but also because they may represent important pathways of poisoning for birds of prey and carnivore mammals. Here, we consider the spatial components involved in the process of small mammal exposure to ARs with the aim to address how these can be used in spatially explicit risk assessment. We present how various drivers operate on multiple spatial scales. On continental and/or regional scales, both biogeographical distribution of small mammals and other species of conservation value and international/national regulations of AR applications (indoor vs outdoor...) could be used to identify some countries or states where exposure is more likely. For application at the local scale (i.e. few km²), we reviewed published studies that analysed the spatial pattern of small mammal exposure to ARs according to species and distance to treatments. We evidence that most of the small mammals exposed to AR are found in the immediate vicinity of treatment areas, i.e., within 100 m. Over 100 m, exposed rodents are rare but can be found until 750 m distance from treatment areas. Species traits related to spatial dimension such as habitat preferences, home range size and mobility also influence exposure. Exposure is variable, in terms of proportion of contaminated individuals and levels of residues, for species showing small home-range size and a limited spatial mobility. The level of exposure depends on whether the main habitat of the given species is similar or not to the one of the target rodent. For instance, exposure of the common vole, a grassland species, is low when ARs are used indoor while it can be highly exposed when bromadiolone is applied outdoor to control the water vole, a sympatric species. For small mammals exhibiting a relatively large home-range size and a high spatial mobility such as the wood mouse and the bank vole, the exposure is commonly reported within a lower range than target species. Although this has not been studied in details, we also address how landscape and/or habitat features may modulate exposure, suggesting that landscape management may help to mitigate the risk of ARs to small mammals. Finally, we discuss both the advantages and disadvantages of statistical, analytical or simulation models to assess potential or actual exposure of NTSM to AR in a spatially explicit way. We conclude that in order to analyse global patterns in usage and exposure risks, large scale statistical modelling should be used while for detailed site specific assessments, simulation models may be more appropriate.

Keywords: small mammals, non-target, exposure, rodenticide, spatial, landscape, spatially explicit risk assessment

Introduction

Most of the literature dealing with the unintended effects of anticoagulant rodenticides (ARs) has focused on which species of wildlife were poisoned and where, by reporting the number of dead animals found, the proportion of exposed specimens in a population and/or the level of residues measured in tissues (see chapter 7 for a review). To date, AR exposure has been widely reported for birds (notably raptors), medium and large mammals (see chapters 6 and 7) and, to a lesser extent, invertebrates (Godfrey 1985; Spurr and Drew 1999; Albert et al. 2010; Sánchez-Barbudo et al. 2012). For some species such as the red kite (*Milvus milvus*), the fisher (*Martes pennanti*) or the red fox (*Vulpes vulpes*), the spatial pattern of intoxications have been described and related to ecological context (e.g., prey guild diversity, (Tosh et al. 2011a); diet specialization, (Coourdassier et al. 2012)) and/or risk prone practices (Berny and Gaillet 2008; Gabriel et al. 2012; Coourdassier et al. 2014). Although specific information on treatment characteristics (e.g. location, intensity, active ingredient, etc) is rarely provided, general mechanisms of AR transfer in the environment have been identified from the data provided by those monitoring studies. For instance, the high frequency of poisoning incidents involving predators and scavengers evidences the importance of secondary exposure due to the consumption of poisoned prey, possibly rodents and/or other animals. Surprisingly, data on non-target small mammals (NTSM) remain rare while, intuitively, we can assume that baits are also accessible to other rodents than the ones targeted, when distributed in the environment. Moreover, small mammals are occasional or main prey for numerous carnivorous vertebrates and thus, represent pathways for exposure to AR if they contain residues. To date, the few studies available have shown very variable patterns of exposure for NTSM. 3% (Elliott et al. 2014), 23% (Geduhn et al. 2014) and ~50% (Brakes and Smith 2005; Sage et al. 2008) of the specimens analyzed were exposed to ARs. A better understanding of the drivers of such variability would improve the assessment of both exposure and risk for small mammals themselves and also for their predators. Finally, many rodent and shrew species are threatened over the world (International Union for the Conservation of Nature 2015), and effects due to AR are of conservation concern if they contribute significantly to population decline.

Both field and laboratory experiments have been conducted to identify some of the drivers of wildlife exposure to AR more precisely (Cox and Smith 1992; Ntampakis and Carter 2005; Sage 2008; Montaz et al. 2014). However, Rattner et al. (2014) have pointed out that empirical field data detailing exposure pathways and compound transfer to wildlife remain limited. Several unsolved issues related to AR exposure assessment can be identified, among which some are related to the spatial dimensions of the transfer processes. When questioning pathways of ARs transfer to wildlife including NTSM, the features of AR applications are key determinants because exposure starts with bait placement (Rattner et al. 2014). Both primary and secondary poisonings only occur when the active ingredient is ingested, which implies the co-occurrence (potential exposure) and uptake (effective exposure) between the chemical, its consumers and their predators/scavengers (Smith et al. 2007). Spatial dimensions are inherent to ecological systems and processes and have become a growing interest in the field of ecotoxicology since the 1990s. It is now well-recognized that wildlife exposure to a wide range of chemicals varies because of, among other determinants, the spatial heterogeneity of the environmental contamination, the spatial distribution and activity patterns of the receptor species, and also landscape and habitat properties (Cairns 1993; Clifford et al. 1995; Marinussen and van der Zee 1996; Cairns and Niederlehner 1996; Smith et al. 2007; van den Brink et al. 2008). If the contribution of the spatial components to the determination of exposure likely differs

according to the chemical and the receptor, some studies evidenced that they may control a relevant part of the transfer processes to small mammals. For metals, for instance, using an individual-based exposure model, (Loos 2011) has shown that spatial variation of soil contamination is the major contribution (31-41%) to the cadmium exposure of the common vole *Microtus arvalis* compared to temporal and inter-individual variations. In a large smelter-impacted area, both soil contamination heterogeneity and landscape composition explained 46% and 50% of the Pb exposure of the bank vole *Myodes glareolus* and the greater white toothed shrew *Crocidura russula*, respectively (Fritsch et al. 2011). Thus, it is increasingly recognized that taking into account spatial components related to the chemical(s), the receptor(s) and the environment is a further and essential step in exposure and more generally risk assessment and would contribute to derive site specific risk mitigation approaches. When addressing AR exposure assessment, many questions dealing with spatial properties of ecological systems, treatments and species arise notably to characterize the circumstances making the contact between AR and NTSM effective. Where are the treatments achieved and located? Which species are present in the treated area and for what periods of time? Are the baits present in habitats exploited by non-target species, and are they accessible? Which biological traits of species modulate the contact with baits? How do landscape and habitat properties influence exposure to ARs?

To assess exposure of NTSM (and more generally wildlife) to ARs and the potential consequences on individuals and populations in a spatially explicit context, we must recognize that the main determinants may act at a variety of spatial scales, i.e., from several hundreds of thousands km² to some m². Moreover, ecological systems are nested within one another. Although recognized since long (Allen and Starr, 1982), hierarchical organization has been generally undervalued as a mean of understanding the connections between local processes and large-scale distribution patterns of transfer and risk assessment at various time and space scales (Giraudoux et al. 2013). This is consistent with the view of Johnson and Turner (Johnson and Turner 2010) who considered that, as natural phenomena and anthropogenic changes have characteristic spatial scales, it is critically important to incorporate explicitly scale issues into the design and implementation of ecological risk assessments. A conceptual way to address such issues lies in the “hierarchical patch dynamics paradigm” which conceives ecological systems as nested hierarchical systems (Landis 2003) and we believe that this framework is relevant to conceptualizing exposure assessment of ARs in a spatially explicit context. Indeed, some determinants of AR exposure occur on a large spatial scale: a national regulation may prohibit the outdoor application of some ARs, potentially mitigating exposure of non-commensal species. On the scale of a country, land use determines treatment locations, AR applications for plant protection being mainly done in grasslands or crops in rural areas for instance. Locally, habitat quality may influence home range and diet availability and thus, spatial activity pattern and bait consumption of a species. Therefore, as previously discussed for many other questions in ecology, both the understanding and prediction of AR fate and effects in terrestrial ecosystems rely on our ability to study patterns and processes on the appropriate scales and develop models that bridge across scales (Levin 1992; Johnson and Turner 2010; Chave 2013).

In this chapter, based on the literature review on primary exposure to AR presented in chapter 6, we discuss the spatial determinants that influence AR transfer to small mammals and how these can be integrated in models useful for AR exposure assessment, in relation to the spatial scale they operate. The spatial components of interest will be related to the chemical(s), the receptors, i.e., NTSM, and the environment where they live. Based on available literature and our own expertise, we propose a

conceptual scheme integrating the main determinants of AR transfer to NTSM that will be discussed further and their scale of expression (figure 1). We will consider 3 nested scales at which patterns of AR treatments, NTSM distribution and contact with baits are driven by ecological (related to both small mammals and landscape), regulatory and socio-economic factors. According to the scale dependency of processes modulating ARs transfer to wildlife we discuss above, the spatial scale at which drivers and processes act will be specified and we will bring a careful attention to illustrate and discuss links across spatial scales. Obviously, other characteristics of individuals and species control the accumulation of AR in tissues (toxicokinetic) and transfer (diet) but these will not be dealt in this chapter, except for those aspects linked to spatial dimension. Moreover, exposure assessment is only a step in risk assessment that furthermore calls for assessing the effects related to the exposure (dose-response assessment). In this chapter, “effect” will not be discussed because, to our knowledge, species specific data on the sensitivity of NTSM to AR are not available.

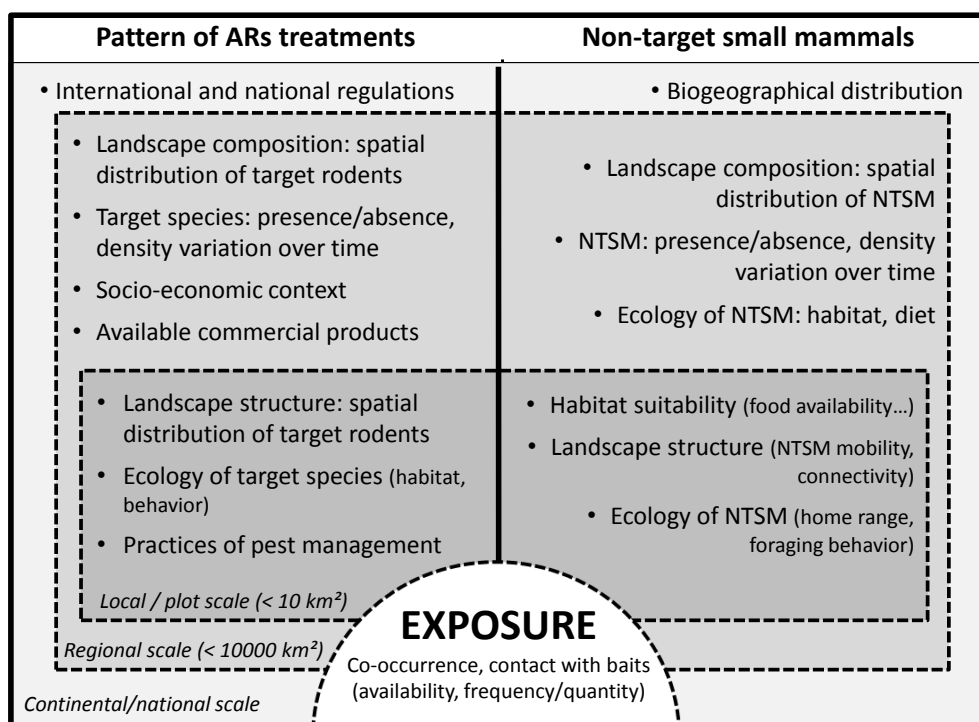


Figure 1. Conceptual framework about the main determinants and corresponding spatial scales usable in a spatially explicit assessment of exposure to anticoagulant rodenticides (AR) for non-target small mammals (NTSM). Hereafter, we use the term “continental/national scale” for areas of tens of thousands of km², “regional scale” for areas of hundreds to thousands km², “local scale” for areas of some km² and “plot scale” for areas of some thousands m² where AR are generally applied for small mammal control.

1/ Spatial patterns and drivers of AR applications

“Where are rodenticides applied?” is a key question when assessing risks they may represent for wildlife. AR is by far the most used control method against rodents worldwide (Capizzi et al. 2014) and, in most countries, application is strictly regulated according to control objectives. International/national regulations define which active ingredients are authorized for a given objective, what the target species are and the modalities of application (e.g. use of bait boxes, above- or underground disposal of baits, indoor/outdoor use). Thus, regulations are important drivers of

spatial distribution of AR in the environment and strongly contribute to modulate exposure and risk for non-target species. For this reason, to discuss spatial patterns and drivers of AR applications, it is more convenient to separate the 2 main types of AR usage commonly distinguished by regulations, i.e., biocidal and plant protection uses. Island eradication programs will also be discussed. Additionally, illegal use of AR may contribute significantly to non-target wildlife exposure (Bartos et al. 2012; Gabriel et al. 2012), however, as exploitable data are not available, it will be not considered below.

a. Biocidal use of AR

In USA and Western Europe, biocidal applications represent more than 90% of the total amount of AR use (U.S. EPA 2006; Jacquot 2013). To our knowledge, in other parts of the world, such data are not available. Currently, numerous rodenticide products with different active ingredients are available on the market of biocides. For instance, 8 AR are authorized in the European Union (regulation 528/2012) leading to the registration of several tens of commercial formulations mainly against house mouse (*Mus musculus*) and Norway rats (*Rattus norvegicus*)¹. Biocidal treatments are generally carried out by homeowners, farmers, local authorities and/or Pest Control Operators, depending on the national laws. Thus, a wide diversity of AR can be bought and used freely as biocide in most of the countries even although recent limitations have been adopted for instance in Canada and USA (see Elliot et al. 2014 and chapter 11 for details). Both qualitative and quantitative data on AR use as biocide are at the best global (national amount applied of total AR for instance (Dawson and Garthwaite 2004) in UK or U.S. EPA 2006 in USA). However, in most of the cases, they are unavailable because considered as “confidential business information” (Rattner et al. 2014) and because no (or only limited) traceability is required following control operations. Thus, although the intensity and spatial distribution of treatments represent key information in spatially explicit risk assessment, they can only be described on regional and/or national spatial scales in some cases (see for instance (Dawson and Garthwaite 2004; Brooks 2008; Tosh et al. 2011a; Hughes et al. 2013). On local scale, treatment features remain most often unknown because of the scarcity of site-specific and georeferenced monitoring on AR practices.

It is commonly reported that biocide AR are mainly applied in urban, suburban and rural areas, inside and/or at the vicinity of buildings, where food is available. That includes residences, sewers and landfills, agro-industry plants, farming facilities and warehouse. Thus, spatial patterns of treatments could be partly related to human population density and/or types of human activities. In Spain, AR occurrence in wildlife was positively correlated with human population density (López-Perea et al. 2015), which suggests that human population density could be a surrogate of treatment intensity at national and regional scales. Hughes et al. (Hugues et al. 2012) estimated that 129.8 tons of ARs baits were applied by farmers in 2012 in the entire Scotland by using relationships including the regions where the farms are located and the size of the holdings (in ha). The model coefficients were determined based on the assessment of the amounts of AR applied by 250 farms selected in 11 Scottish regions. As the coefficients depended on both the region and the farm size, that approach may allow estimating the spatial distribution of treatments at both national and regional scales, although Hugues et al. (2012) did not analyze their data in such a spatially explicit way. User behavior was also characterized for homeowners in USA (Morzillo and Mertig 2011a; Morzillo and Mertig

¹ In this chapter, *Apodemus* sp. will be considered as non-target species even if, in some cases, they can be targeted by biocidal treatments.

2011b; Bartos et al. 2012) and farmers in UK (Tosh et al. 2011b). Based on inquiries performed in 162 farms in Northern Ireland, Tosh et al. (2011b) have shown that SGAR and FGAR were used in 74 and 8.5% of the farms, respectively. Difenacoum was the most frequently used AR (33% of farms), followed by bromadiolone and flocoumafen. Baits were distributed within buildings by 68% of respondents and outside buildings by 48% (among which 10% applying baits away from buildings, in fields or hedges). Products that were licensed for indoor use only (flocoumafen, brodifacoum) were also used around the outside of buildings by 12% of farmers and in fields or hedges by 3% of respondents. Establishment of such quantitative relationships and qualitative characterization of user behavior in other contexts, would allow modeling the features of biocide AR treatments at regional scales using spatialized data (human population density, socio-economical activities, etc.) available in most countries. User behavior could be integrated as a probability for each potential user (farm, family, company, etc.) to use AR, to apply baits outside... according to inquiry results. This would help for instance to define areas where AR treatments are likely to be intensive and thus where wildlife exposure is likely to be high.

b. Use of ARs for plant protection

Some cases in Europe provide illustrative examples of spatial use-patterns of ARs as Plant Protection Products (PPPs)², how such patterns can be described at different scales and which drivers have been identified for spatially explicit modelling of exposure. Rodents, for instance the common vole, the field vole *Microtus agrestis* and the water vole *Arvicola scherman* (formerly *Arvicola terrestris*), may represent agricultural pests in many European countries (see the synthesis of (Jacob and Ktadlec 2010). Since 2011, three ARs are authorized in EU for use in PPP-applications, bromadiolone being by far the most widely used since the withdrawal of chlorophacinone in 2009 (regulation 1107/2009). Generally, AR campaigns are mainly managed by farmers, horticulturists and/or licensed operators... and, in some countries, regulated by institutions and/or technical institutes like in France where a traceability program of bait distribution and/or application exists. Thus, the level of knowledge on treatment characteristics is probably better than in case of biocidal use even if it remains highly heterogeneous, depending on national regulation. Huge AR field treatments have been reported in Germany, Spain, Switzerland and France during the last 20 years (Berny et al. 1997; Kupper et al. 2006; Olea et al. 2009; Jacob and Ktadlec 2010; Coeurdassier et al. 2014). For instance, since 1997, up to 1,440 tons of bromadiolone baits were applied on 60,000 ha per year in the Franche-Comté Region (Eastern France) against water voles (Coeurdassier et al. 2014). In Spain, several tens to hundreds of tons of baits have been used on areas up to 375,000 ha to control common voles in Castilla y Leon in 2007-2008, (Olea et al. 2009). To our knowledge, field applications are more limited in UK and Scandinavia while, for other countries we failed to find any accessible data in international literature. Since 2009, European regulations require the registration of both sales and use of PPPs for a period of 5 years by all EU member states (regulation 1185/2009). This has to be done in a harmonized way that allows comparisons between countries for the purpose of assessing the risk for both human health and the environment in a spatially explicit context (first period covered: 2009-2014). Such survey is also planned for sales of biocides but, to date, not for their uses. The gathering of these data would open new perspectives to assess both spatial and temporal trends of AR uses and to identify their key drivers (ecological, economical and sociological) at continental scale and

² Damages due to rodents and ARs application in forestry, tree plantations and orchards will be not discussed here.

may be at finer resolution depending on the data collected by authorities. In North America, the use profile of ARs is more restrictive (chapter 11 for details). The application of the second-generation compounds (SGARs; e.g., brodifacoum, difethialone, bromadiolone, difenacoum, flocoumafen) that are considered as more hazardous to wildlife than first generation ones (FGAR), is prohibited in agricultural fields (see chapter 11). Thus, only FGARs, notably chlorophacinone or diphacinone may be applied to control field rodents causing damages to crops and grazing land such as black-tailed prairie dogs (mainly *Cynomys ludovicianus*), Richardson's ground squirrels (*Urocitellus richardsonii*), pocket gophers (*Thomomys* sp. and *Geomys* sp.) or voles (e.g., *Microtus ochrogaster*, *Microtus pennsylvanicus*, *Microtus montanus*).

Within countries where AR are massively applied in fields, spatial pattern of treatments are driven by the target species, as defined by legislation, their densities and population dynamics and by the regional and local practices of pest management. In Castilla y Leon (Spain), common vole outbreaks followed a 5-year cycle since the 1970s and Luque-Larena et al. (2013) report "*a strong association between outbreaks and rodenticide campaigns*". In France, bromadiolone is the single AR registered since the 1980s to control cyclic peaks of water vole. Such outbreaks occurred every 5-6 years mainly in the grasslands of Franche-Comté and Auvergne (Centre France) regions, notably because of landscape changes related to agricultural activities (Delattre and Giraudoux 2009) (fig 2a). The most intensive applications of bromadiolone were mainly undertaken in those 2 regions during the last 3 decades (D. Truchetet, national expert on vertebrate pest management, Ministry of Agriculture, com. pers.). At the end of the 1990s, a detailed traceability of bromadiolone applications was established by authorities in Franche-Comté: for each treatment, the quantity of baits, precise location and date is registered each year. These data allow describing the spatial distribution of treatments at plot resolution (i.e., few hectares) in an area of 16000 km² and were used to evidence their impacts on red fox population at regional scale (over 5000 km², (Jacquot et al. 2013). During the last 10 years, water vole outbreaks occurred more often in France, with a subsequent geographical expansion of rodenticide campaigns at national and regional scales. Firstly, since 2005, water vole outbreaks of various extent and intensity have been reported in other regions of France than the "historically" known Auvergne and Franche-Comté, i.e., in Alps mountains, Burgundy and Champagne-Ardennes (Truchetet et al. 2014), fig 2b). To date, bromadiolone has been used seldom in these "new" areas but more intensive applications of AR are not excluded if needed. Secondly, while bromadiolone was only registered as PPP to control water voles in France, a recent regulation has extended its use to other rodent species among which the common voles (Journal Officiel 2014). This means that bromadiolone could now be used where agricultural damages due to common vole outbreaks may occur in grasslands and crops (e.g., Western France and other regions as proposed by (Quéré and Le Louarn 2011), fig. 2c). Based on spatialized information on both distribution and population dynamic of the target species, it is thus possible to identify the main areas where bromadiolone treatments are likely in France in the coming years. With the perspective to assess and prevent the risk of AR when applied in the field, such projections will serve to determine where the survey of wildlife poisoning by PPPs imposed by the EU (regulation 2009/128) and/or farmer information to mitigation practices will have to be implemented.

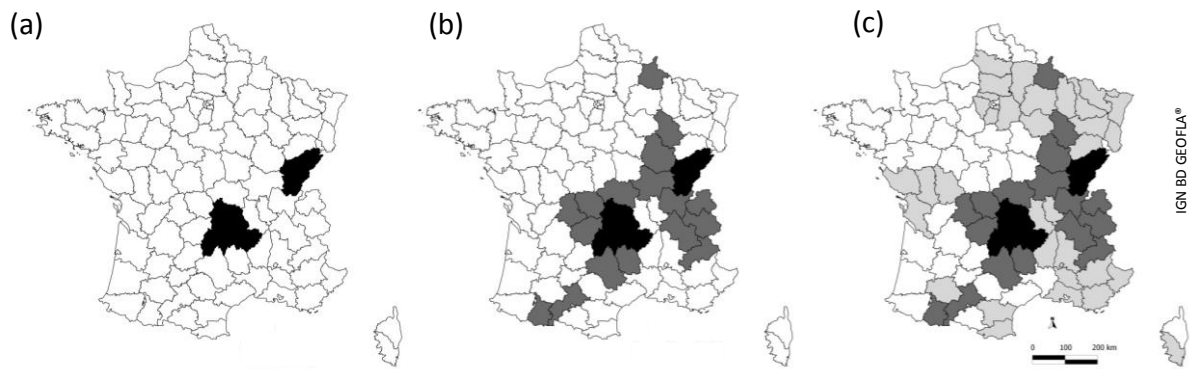


Figure 2. Main areas of current and possible future use of bromadiolone in farmland against voles in France. (a) Black patches indicate the areas where water vole outbreaks occur and where large rodenticides campaigns were implemented since the 1980s. (b) During the last decade, Water vole outbreaks have also been reported in new areas (dark grey patches). (c) Moreover, in 2014, national regulation has registered the use of bromadiolone to control water voles, common voles and Mediterranean pine voles when damages occurred in grassland, crops or orchards (light grey patches). These changes might extend the spatial distribution of bromadiolone applications in field from (a) to (c) and, as a consequence, the areas where a specific monitoring of wildlife poisoning is needed.

c. Island eradication

Small mammals may be introduced to islands, and establish populations. Many such populations have been described and generally attributed to human activities. Due to the general lack of endemic small mammals, eradication programs targeting introduced small mammals do not affect NTSM. However, many cases of poisoning of non-target non-mammalian species have been described including birds (Eason et al. 2002; Eason and Spurr 1995; Howald et al. 1999), reptiles, amphibians and even invertebrates (see Hoare and Hare (2006) for overview). Notwithstanding these non-target exposures, the general opinion is that eradication programs using AR are beneficial from a conservation point of view, the positive effects outcompeting the negative ones, although risks on non-target exposure or secondary poisoning should be mitigated as much as possible (Hoare and Hare 2006).

Various approaches have been developed to eradicate rodents from islands, each with specific spatial attributes. Many studies regarding island eradication programs originate from New Zealand, with a long history of rodent invasions of (small) islands and an early offset of use of AR for management of rodent pests. For small islands, the intention is often to eradicate the total population of the target species in a single attempt. For larger islands or areas, where complete eradication is deemed to be impossible, longer term and replicated use of AR are needed for management of rodent populations (Eason et al. 2010; Hoare and Hare 2006). Longer term application, however, has a higher potential for non-target exposure and secondary poisoning (Eason et al. 2010).

All approaches to eradicate rodents are intended to cover the area of interest as completely as possible. Early attempts applied rodenticides by hand on the ground, while later aerial application became more important, covering larger areas. Aerial application however, is generally less

controlled, and no direct mitigation measures, like for instance bait-stations, are being taken to minimise non-target exposure. This may result in higher prevalence of non-target exposure, while also a larger number of non-target species may be exposed (Innes and Barker 1999). It should however be noted that generally eradication programs, not limited to small mammals, are more successful if they carried out on small areas isolated (e.g. small islands) (Pluess et al. 2012).

One of the first recorded Island eradication in New Zealand was on Maria Island, a small (1 ha) island invaded by Norway rat which were eradicated by the use of rodenticide between 1960-1964, although not in a standardised planned way (Moors 1985). Later eradication programs became more standardised, and spatially explicit methods were applied in the program on Breaksea Island, in 1987 (Taylor and Thomas 1993). In that study, bait-stations with brodifacoum were placed in 25-60 m intervals, all over the island. Based on the study it was concluded that the spacing may have been unnecessarily small. Once rats were killed, neighbouring individuals were expected to increase their home range, so even if no bait was placed in each territory, all rats were expected to be exposed. The baits stations were designed to prevent non-target exposure, and although some non-target animals were found to tamper with stations, eat bait or even died (Eason and Spurr 1995), no effects on local populations of non-target species, mainly birds, could be established. This project was very successful in eradicating the rats from the island. In following ground-based programs the spacing of bait stations was tested in order to minimise the use of rodenticides. Programs using 100 meter spaced baits had variable success in eradicating rats, however, no indications of the bait-spacing on the exposure of non-targets are provided (Thomas and Taylor 2002). However, it was reported by Eason and Spurr (2010) that 80-90% of the population of Steward Island weka, *Gallirallus australis*, was eradicated by the use of brodifacoum (Talon®) bait at Ulva Island. The use of more bait shortened the period to extinction of the rats, and thus also the period that non-targets may be exposed, likely decreasing the probability of non-target exposure (Eason et al. 2010). No direct information is known to the authors linking bait-spacing to the risks of non-target exposure. Spacing of the stations was deemed to be more or less optimal at 50 meter intervals for Norway rats, since they tend to move towards the bait station due to “peer-pressured” feeding (following others) and also due to the fact that they tend to increase their home range following the disappearance of their neighbours.

Aerial programs are used to eradicate or control rats at a larger scale, or in places where ground-based programs are not feasible. Generally, non-target exposure and effects are larger in aerial eradication programs (Eason and Spurr 1995), likely due to the facts that mitigation measures are less applicable and also due to the fact that programs work on a larger spatio-temporal scale. However, no information is available to the authors on the effects of the spatial scale of application of AR on the risks for non-targets species.

2/ Patterns of exposure of non-target small mammals to AR

a. Distribution of small mammals at national and regional scales

In many areas in the world, biogeographical ranges of small mammals are generally well known at national or even regional scale (see for instance the data on geographic range and maps available on the IUCN or the European Mammal Society websites). That is at least the case in Europe and North America while the limits of range can be still imprecise at regional scale in many other parts of the

world (e.g. Central and Eastern Asia, etc.). In those areas where small mammal ranges is reasonably well known, the composition of the small mammal community potentially exposed to ARs can be defined by merging maps reporting presence/absence or density of small mammal species in a given area. Such maps would help to identify high biodiversity areas or presence of threatened species where ARs are applied in order to minimize the impact on NTSM. However, the co-occurrence between animals and treatments does not mean that exposure to ARs actually occurs. A more refined approach integrating the factors (including inter-, intra-species, habitat specificity and other environmental factors) that drive the contact between baits and individuals will have to be implemented to move from potential to actual exposure (see paragraphs 3a and 3b below for discussions of the drivers of exposure).

Another important issue is whether the diversity of the small mammal community modulates AR transfer to wildlife (Tosh et al. 2011a). In that study, the exposure to ARs of Red fox living in North Ireland (NI) was compared to Great-Britain (GB), two areas exhibiting differences in small mammal prey guilds with a higher diversity in GB. A greater prevalence of liver residues amongst foxes in NI relative to GB was shown. It was concluded that the restricted guild in NI, including species most likely to consume ARs (i.e., target commensal rodents and wood mouse, see chapter 6) might have enhanced secondary exposure of foxes. In GB, with a greater diversity of small mammal species, the additional species not likely to be exposed to ARs may have diluted the general exposure of predators to AR. To our knowledge, that is the only study reporting a possible link between prey biodiversity and transfer of rodenticides to predators. Thus, any attempt to generalize the findings of Tosh et al. would be hasty. However, we believe that type of approach could help to improve spatially explicit risk assessment of ARs for small mammals and their predators over large spatial scales.

b. Spatial patterns of small mammal exposure to AR

The review of literature dealing with the exposure of NTSM presented in the chapter 6 (table 6.1.) shows that all the studies addressed this issue on local scales. All studies included rodent species while only 3 shrew species were reported to be exposed. Most studies (8/12) were achieved in a context of biocidal usage only. Treatments were commonly characterized following questionnaire surveys, targeting users (Tosh et al. 2011b) or were controlled by the applicants carrying out experimental baiting programs in or around farms (Brakes and Smith 2005; Tosh et al. 2012; Elliott et al. 2014). Such methodologies are resource demanding; moreover, they rely on applicant trustability and/or on the design of baiting programs that mimic the true conditions of AR use after commercialization. Experimental baiting programs are difficult to conduct in villages, cities or larger areas because treatments can hardly be controlled with reasonable reliability. Thus, the great majority of the studies using these methodologies were performed in areas of a few hectares at less than 100 m of treatment areas. Only Jacquot (2013) has worked in areas of 3 km² (1 km distance from treatment area) including probably more of the spatial heterogeneity of landscape, environmental contamination by ARs and distribution of NTSM populations. In that study, small mammals were sampled in and in the vicinity of villages, where AR baiting was not controlled. Knowledges of treatment features (location, intensity, active ingredient used, etc.) were only limited, each building was considered, whatever its function, as a potential source of ARs. That method, although purely descriptive, was used to map resulting small mammal exposure and did not allow the study of transfer processes. Bait markers (e.g. food dye, radio-labelled active ingredient, etc.) were

rarely used (Brakes and Smith 2005) however they can help to analyze AR transfer processes to non-target species on local scale.

The limited number of studies does not allow a quantitative meta-analysis to identify the main factors controlling exposure of NTSM or biological traits of species determining their spatial activity pattern and the effective contact with baits. However, some common features can be pinpointed and may help to guide future research for a better understanding and the proposal of quantitative relationships usable in risk assessment modeling. Among the studies reviewed in the chapter 6, four have questioned the spatial distribution of the exposed NTSM, generally by relating the distance between baits and individuals trapped with the level of ARs measured in animals. This can be expressed as the probability for an individual to be exposed according to the distance to the AR source. In three cases, the authors have checked whether the distance to treatments influence residue concentrations in tissues (Tosh et al. 2012; Jacquot 2013; Gedhun et al. 2014).

For rodents, there is a clear spatial pattern with a higher frequency of occurrence of residues in animals within the immediate surroundings of treatments (< 100m) under both biocidal and PPP usages. Taking into account all the rodent species (i.e., both target and non-target ones), 90% of animals with residues were trapped within 50 m of bait stations (Gedhun et al. 2014) or 100 m of treatments (Tosh et al. 2012; Jacquot 2013). Within that range of 100 m, individuals of target species with residues are generally trapped very close to the treatments. All exposed house mice occurred within 30 m of farm building in the study of Tosh et al. (2012). Moreover, in Jacquot (2013), 100% of contaminated, and targeted rats and house mice were trapped within 30 m of bait placement and all water voles with residues were captured in plots with PPP-application. For non-target (NT) rodents, similar patterns to target species were observed. All wood mice with AR residues were trapped at least 30 m away from farm buildings and within 80–110 m from a baited area (Townsend et al. 1995; Tosh et al. 2012). In Gedhun et al. (2014), the proportion of NT rodents with residues was highest within 0-15m from bait stations for all species, while 25 to 50% of the wood mice and bank voles trapped at 15-45m were exposed to ARs, 87m being the furthest distance. In contrast to this, less than 10% of *Microtus* ssp. were exposed beyond 15 m of bait stations (Gedhun et al. 2014). In and around villages, the proportion of exposed NT rodents decreased with the distance to treatments with 11% of individuals exhibiting residues at 0-30 m, 7% at 30-60 m and 3% at 60-100 m (Jacquot 2013). Globally, within a radius of 100m from a village, 10% of individual wood mice exhibited exposure (11/113), with the bank vole at 8% (1/12) and the common vole at 4% (1/26) (Jacquot 2013). Only one study, detailed in Jacquot (2013), provides information on exposed animals further away from potential sources of treatments. In that work, among the 106 trapped rodents with residues, 10 were further than 100 m from a potential AR source, representing about 3.5% of all the rodents trapped. Under a biocidal usage, among 197 rodents trapped beyond 100 m from potential treatments, 2 *Apodemus* ssp., 1 common vole and 1 bank vole located between 220 and 750 m far from baits were exposed to AR. In PPP context, 5 *Apodemus* ssp. and 1 bank vole with residues were trapped between 100 and 360 m away from treatments (n = 89). Patterns linking concentration of residues in rodents and distance to AR source seem less consistent than that discussed above on their occurrence frequency in populations. Tosh et al. (2012) did not observe any apparent relationship between distance from farm buildings and residue concentration. The highest liver AR residues were detected in wood mice at 60 m (mean = 0.237µg/g, n=3) and 110 m (mean =0.317µg/g, n=2) from the nearest farm building. Conversely, according to Jacquot (2013) and Gedhun et al. (2014), tissue concentrations of ARs decreased considerably with increasing distance.

Indeed, under biocidal and PPP usages, 85% of the rodents with a body concentration $> 0.1 \mu\text{g.g}^{-1}$ (sums of ARs in the whole body, fresh weight) were target species, NT rodents being common voles (PPP use) and *Apodemus ssp.* (PPP and biocidal uses) trapped in treated places (Jacquot 2013). All the NT rodents with body concentrations $> 0.03 \mu\text{g.g}^{-1}$ were within 50 m from treatments, and those $> 0.01 \mu\text{g.g}^{-1}$ within 100m (Jacquot 2013). Considering only the individuals $> 0.01 \mu\text{g.g}^{-1}$, all were *Apodemus ssp.* in a biocide context, while they belong to 3 species under PPP usage (common vole $>$ bank vole = *Apodemus ssp.*). In Gedhun et al. (2014), most of the individuals with very high AR level ($> 1 \mu\text{g.g}^{-1}$ in liver, FW) belonged to 3 species, the wood mouse, the bank vole and *Microtus ssp.*, and were within 25 m of bait stations while concentrations greater than $0.1 \mu\text{g.g}^{-1}$ were not measured beyond 55 m.

For shrews, the pattern linking exposure of individuals to distance to bait stations at local scales is different than that of rodents. For both *C. russula* and *Sorex ssp.*, the occurrence of residues in populations was higher at 15-30 m than within 15 m from baits. Then, the proportion of exposed individuals decreased, although 10% of the *Sorex* trapped at more than 60 m still exhibited residues (Gedhun et al. 2014). For tissues concentrations, two patterns were observed: residues decrease with the distance (up to 60 m) to treatments in *C. russula* while they remain relatively stable in *Sorex* from 0 to 80 m to bait stations (Gedhun et al. 2014).

To conclude, most of the small mammals exposed to AR are found in the immediate vicinity of treatments, i.e., within 100 m. Spatial patterns of the distribution of exposed individuals are relatively consistent among studies with some slight variations among species. Both the frequency of occurrence of residues in populations and tissue concentrations tended to decrease with distance from treatments. Contaminated target animals were mainly trapped within 30 m, while NT ones, notably *Apodemus ssp.* and the bank vole in biocide context or the common voles in PPP, could be trapped within 100 m. Over 100 m, exposed rodents are rare but can be found up to 750 m from treatments. It is essential to keep in mind that most of the data presented here were obtained on local scale and that the patterns described cannot be extrapolated to larger scales without caution, as pointed out by Johnson and Turner (2010). Indeed, when trappings are carried out at the close proximity of the bait stations, the proportion of exposed individuals in both target and non-target species could be overestimated because, generally, the populations that do not live close to treated areas are not included in the sampling plan even though they are likely less or not exposed.

Despite the fact that both results and figures presented in some articles suggest that links between NTSM exposure and distance to bait exist, no study has explicitly reported quantitative relationships appropriate for data modeling, neither for residue occurrence nor for tissue concentrations. Quantification is a keystone of modeling and such relationships are indispensable to implement a spatially explicit risk assessment for ARs. Thus, we strongly recommend that future studies should describe spatial patterns of NTSM exposure to ARs and report the influence of different factors in a more quantitative way. Experiments and studies should focus on:

- assessing the spatial structures in exposure indicators (residue occurrence or concentrations measured in individuals, etc.) by using geostatistical models;
- establishing relationships between exposure indicators and spatialized features of treatments and environment (e.g. distance to baits, inside or outside baiting, habitat and landscape features);

- establishing relationships between exposure indicators and non-spatial factors (treatment intensity, species), which would attempt to include, if characterized, the spatial structure of the response variable in statistical models.

In all cases, both model formulae and parameters (including their uncertainty) need to be explicitly reported.

3/ Spatially explicit drivers of non-target small mammal exposure to ARs

a. How may the spatial activity patterns of NTSM influence their exposure to ARs?

The spatial activity pattern of an animal can be defined as the characterization of its movements over space during a given time. Several types of movements exist that may apply to local population processes in small mammals. These include dispersal, exploratory movements, and activity within home ranges (Wolton and Flowerdew 1985). These movements may correspond to different needs including foraging for food, searching for sexual partner, colonization of new habitats or juvenile dispersal. The spatial activity pattern of an individual is often considered as one of the drivers of its exposure to chemicals because it determines partly both the frequency and intensity of contact with the pollutant and thus the quantity that can be transferred into a receptor (Leyk et al. 2009; Loos 2011; Spurgeon et al. 2011). According to Loos (2011), the spatial variation in habitat quality (including food availability), the habitat patch connectivity, population density and foraging behavior govern the home range of an animal, the time it spends per unit area within its home range and also its diet in terrestrial ecosystems. As the spatial activity pattern of an animal is determined by the biological traits of species (mobility, diet...) and individual (age, gender), daily (foraging for food, rest site) and/or seasonal (breeding, migration) activities, the population and community characteristics and both the composition and the structure of the landscape, all these determinants can be considered as potential drivers of NTSM exposure to ARs.

b. Which specific traits determine the exposure of NTSM to ARs at local scale?

Here we will focus on NTSM species for which the amount of information allows to draw general conclusions, i.e. the four European species of rodents (wood mouse, bank vole, common vole and field vole) and the two of shrews (*Crocidura russula*, i.e., the greater white-toothed shrew and *Sorex* ssp. such as the common shrew or the crowned shrew) in the review presented in chapter 6 (table 6.1.). Their main ecological and biological traits linked with spatial activity patterns are summarized in table 1.

Wood mice and Bank voles are the most exposed NTSM to biocidal AR, although they exhibit a medium level of exposure to AR when compared to target species. That may be explained by their relatively large home range, mobility and ecological plasticity (Brakes and Smith 2005, Tosh et al. 2012, Gedhun et al. 2014, table 1), notably for the wood mouse that is highly mobile and opportunist, especially in agricultural habitats (Wolton and Flowerdew 1985; Brown et al. 1994; Macdonald et al. 2000). Wood mice may explore a wide range of habitats within its home range including inhabited areas (Dickman and Doncaster 1987) and may enter into buildings particularly in winter, making it one of the most commensal species among non-target rodents (Quééré and Le Louarn 2011). Such proximity with the baits can increase the probability of contact with AR, likely

leading to a relatively high occurrence of residues in wood mice. The mobility of the wood mouse may explain exposed individuals more than 100 m away from treatments.

Although the bank vole is less eurytopic and commensal than the wood mouse (Douglass et al. 1992), it also exhibits a relatively large home range compared to grassland species (e.g. *M. arvalis*, *A. scherman*, etc.) and has been shown to be much more mobile within their home range than previously presumed (table 1). Szacki and Liro (Szacki and Liro 1991) determined that 52% of the bank voles they monitored had moved > 400m in a few days with 22% of them travelling distance > 1000 m. Moreover, while bank vole is often considered a woodland species, the home range of some individuals covered the whole set of potential habitats, including the non-woody ones. Bank voles even used urban areas, including allotments and residential gardens (Baker et al. 2003). Thus, it is probable that, associated with the mobility and home range size, the degree of commensalism (i.e., % of time spent in and around buildings) of a species is a determinant of the intensity of exposure of NTSM to biocidal ARs.

Both common voles and field voles are globally less exposed than the wood mouse and the bank vole to biocidal AR, probably because they are more specialized in terms of both habitat preferences and diet (table 1). As they live more strictly in meadows, grasslands or crops, they are rarely found in buildings and their presence at the vicinity of human habitation is often linked to the presence of their habitat in the near surroundings of human settlements. In Gedhun et al. (2014), residues in *Microtus* species >1 µg/g occurred only when grassland bordered farm buildings. Moreover, as mentioned above, these voles also have smaller home ranges and lower mobility than other NTSM (van den Brink et al. 2011).

Available data strongly suggest that the traits preventing common voles from exposure to biocide-used ARs make them more vulnerable to PPP-used ARs. Indeed, when bromadiolone is used as PPP against water vole, the common vole exhibits both a high occurrence of residues and high bromadiolone concentrations when its populations co-exist with those of the target species in treated plots (Sage et al. 2008, Jacquot 2013). In such cases, the proximity of common voles with baits due to habitat preferences and stenotopy, their ecological similarities with the target species, notably in habitat and diet, their small home range and limited mobility, restricting individuals to the area of application, were likely to be responsible for their high exposure. Contrarily to what was evidenced for biocides, wood mice and bank voles are exposed to ARs to a lesser extent than common voles, likely because of their habitat preferences and high mobility. This last hypothesis is reinforced by results from a study checking whether the occurrence of bromadiolone residues was linked to the proportion of area treated with bromadiolone around the trapped rodents, within buffers of increasing radius (from 50 to 1000 m). The probability of residue occurrence in water voles was best explained by using a buffer radius of 50 m around individuals while the best buffer radii were 400 m and 500 m for the bank vole and *Apodemus* ssp., respectively (Jacquot 2013).

Table 1. Main ecological and biological traits of small mammals related to their spatial activity pattern (In (Brown 1956; Szacki et al. 1993; Peles et al. 1999; Sutherland et al. 2000; Borowski 2003; Wijnhoven et al. 2005; Fritsch et al. 2010; Loos 2011; Quéré and Le Louarn 2011; McDevitt et al. 2014).

	Habitat	Home range ¹ (m ²)	Dispersal distances ² (m)	Diet
Wood mouse <i>Apodemus sylvaticus</i>	Eurytopic/opportunist (woodlands, woodlots, grasslands, crops, shrublands, hedgerows...) Commensal in winter	2240 (230-21,800)	500 (< 4000)	Omnivorous: mainly granivorous/herbivorous. Animals up to 20% of the biomass ingested
Bank vole <i>Myodes glareolus</i>	Woodlands, woodlots, hedgerows, shrublands	1480 (200-10,000)	120-800 (>1500)	Granivorous/herbivorous Small amount of animal matter ingested
Common vole <i>Microtus arvalis</i>	Open agricultural lands, grazed pastures and meadows, crops	212 (30-1500)	30 (350)	Herbivorous
Field vole <i>Microtus agrestis</i>	Meadows, moist tall grass prairies, shrublands, forest clearings and edges	400 (31.2-1477)	50 (500)	Herbivorous
Fossorial Water vole <i>Arvicola scherman</i>	Open agricultural lands, grazed pastures and meadows, crops	(100-200)	30-60 (> 500)	Herbivorous
Greater white-toothed shrew <i>Crocidura russula</i>	Woodlots, hedgerows, grassland and crops; shrublands, around buildings	125 (50-395)	200 – 400 (1300)	Carnivorous (arthropods, molluscs, earthworms, etc.)
Common shrew Crowned shrew <i>Sorex ssp.</i>	Meadows, woodlands, woodlots, shrublands, hedgerows	500 (90-2800)	120 (869)	Carnivorous (arthropods, molluscs, earthworms, etc.)

¹ Median (min - max); ² maximum value from different sources reported in parentheses.

Shrews are more territorial than rodents although home-ranges may be similarly sized. However their exploratory movements and shifting of home range location are far more limited (Gliwicz and Taylor 2002). Greater white toothed shrew and *Sorex* shrews differ mainly in habitat preferences and home range size (table 1). However, exposure pathways are more variable for these insectivorous small mammals that may ingest ARs directly or due to secondary poisoning. Thus, the spatial distribution of exposed shrews cannot be discussed only in relation to the spatial distribution of baits but should also consider their potential preys. Generally, AR residues were higher in the population of the commensal *C. russula*. Larger home range could explain why tissue concentrations are spatially more homogenous in *Sorex* ssp. than in *C. russula* for which highly exposed individuals were mainly found close to the bait stations (Gedhun et al. 2014).

The facts mentioned earlier indeed strongly suggest that the probability of exposure, whatever the type of AR use, partly depends on habitat preferences, ability or not to adapt to a wide range of environments (i.e., eurytopic / stenotopic species), home range size and spatial mobility (movement intensity and distances intra- and inter-home range). We can question whether the overlap of NTSM species traits with those of the targeted species (notably for the traits cited above but also some traits not linked to the spatial dimension of exposure) could be used to predict risks of exposure. That could be investigated by first, compiling a 'trait-profile' of the targeted species and then comparing it with those of NTSM species with multi-variate techniques. Such an approach based on trait-profiling would be useful to achieve vulnerability analyses of non-target species.

Numerous factors related to individual or environment may change the spatial activity pattern of a small mammal. Its home range, mobility, dispersal, and/or diet may be influenced by the densities of the population itself and those of interacting species (see for instance (Yunger et al. 2002; Bujalska et al. 2009), seasons, notably during breeding and non-breeding periods (Quére and Le Louarn 2011) or gender and age (Borowski 2003; McDevitt et al. 2014). For instance, Sage et al. (2008) showed that female water voles had greater bromadiolone concentrations in tissues than males and assumed that this could be due to the lower mobility of females which would be thus more exposed to local baits. However, knowledge about whether such factors determine exposure of small mammals to ARs remains inconsistent and the potential influence of population density and interacting species has never been investigated. Thus, their integration in a spatially-explicit assessment of NTSM exposure to AR is not discussed further while the role of some environmental characteristics related to habitat and landscape features is addressed in more details in the following section.

c. Landscape as a driver of interactions between rodenticides and rodents?

The influence of landscape³ on chemical transfer to wildlife has known a growing interest since the 1990s. It is now well recognized that landscape features may affect pollutant transfer patterns in food webs and could modulate the effects of contaminants (Cairns 1993; Cairns and Niederlehner 1996; Schipper et al. 2009; Vermeulen 2009; Loos 2011; Fritsch et al. 2011; van den Brink et al. 2011; Fritsch et al. 2012). However, to date, this topic has only received a limited attention and how landscape influences the patterns of contaminant transfer and impact in food webs remains largely hypothetical. Spatial heterogeneity is thought as one of the fundamental factors influencing both

³Landscape is defined as a mosaic of different habitat patches containing more than one community-type and characterized by its composition, spatial arrangement and functions. Beyond this definition, we suggest to the readers to refer to Farina (2000) or Lidicker (2007) for a more complete view.

population and community processes, particularly in man-dominated landscapes (Kozakiewicz 1993), which is typically the case of urban and agricultural environments where AR are used. A huge number of studies has shown that landscape also determines numerous aspects of small mammal biology and ecology, among which their spatial distribution, population dynamic and movements within and outside the home range (see for instance Barrett and Peles 1999). Thus, a link between landscape features and small mammal exposure to AR seems highly probable but it has never been explicitly investigated yet. The potential role of landscape features in shaping the exposure of NTSM to AR is based on landscape influence on the community composition of small mammals, the distribution and dynamic of both target and NTSM populations and the spatial behavior of individuals. So far as landscape can be thought as a critical driver of exposure because it determines the spatial distributions of both treatments and small mammals, and may modulate contact between individuals and baits, this factor may be of great interest in assessment, management and mitigation of risks of AR for NTSM and wildlife.

C.1. How landscape determines the distribution and population dynamic of target rodents?

When addressing the question of how landscape controls small mammal exposure to ARs, a first issue we identified is related to the distribution of the target species because it determines where treatments occur and thus, where NTSM may be exposed. At regional and local scales, landscape composition and structure are among the main drivers of the spatial distribution and temporal dynamic of rodent populations as demonstrated for several agricultural pest species that may hugely fluctuate over time such as voles. This has been conceptualized for Arvicolid species as the Ratio of Optimal to Marginal Patch Area (ROMPA) hypothesis (Lidicker 1985; Delattre et al. 1992; Lidicker 1995), then included in the Trophic ROMPA integrated model (TRIM) (Lidicker 2000). Both represent a theoretical and operational framework to assess how landscape composition drives rodent dynamics over space and time. ROMPA is most frequently expressed as the proportion of a landscape composed of optimal habitat for the studied species. The main idea was that rodent population dynamics are influenced by the ROMPA, as a combined effect of dispersal and predation. At low ROMPA (i.e., scarcity of optimal habitats), rodent densities will be low and stable while at high ROMPA, densities may also be stable but relatively high and exhibit a seasonal cycle each year. Landscapes with intermediate ROMPA would favour multiannual cycles. Population dynamic patterns of vole pest species fit well with the ROMPA hypothesis, multiannual cycles of common voles or water voles occurred in areas of several km² where ROMPA (i.e., ratio of permanent grassland to farmland) was over 50% or 85% respectively (Delattre et al. 1992; Giraudoux et al. 1997; Raoul et al. 2001; Giraudoux et al. 2003). More locally (~1 km²), dense hedgerow networks and/or vicinity to forest may dampen the amplitude of the common vole (Delattre et al. 1999) and the water vole (Duhamel et al. 2000; Morilhat et al. 2008) fluctuations. In Spain, Jareño et al. (Jareño et al. 2015) showed that land use changes in Castilla-y-Leon, especially in irrigated crops and alfalfa, appear to be the main drivers behind the common vole range expansion, observed at regional scale. Thus, based on quantitative data on landscape composition in windows of some km², it is possible to identify areas where voles are likely to exhibit huge fluctuations and/or high densities responsible for damages to grasslands and thus where AR treatments may occur. This could also be used in a prospective way to assess and model whether different landscape management options may lead to promote or dampen rodent cycles and induce or reduce AR application in consequence (Foltête and Giraudoux 2012).

For urban rodents, the ability to predict the spatial distribution of target species, mainly rats, is less clear than for rural ones. Several studies have focused on the main factors related to landscape that describe the spatial distribution of rats, with the aim to optimize control operations. In some cities, rat occurrence has been related to the type of settlement and building features, the proximity of water ways, subway lines and/or public spaces, the presence of natural soil and food resources such as compost heaps or waste disposal (Langton et al. 2001; Traweger and Slotta-Bachmayr 2005; Traweger et al. 2006; Walsh 2014). It was suggested that rat distribution could be efficiently predicted using models that includes those landscape and habitat variables as shown in Salzburg (Traweger and Slotta-Bachmayr 2005). However, brown rats and mice are not limited to specific landscape variables, but are also dependent on the spatial and temporal distribution of point food resources (human wastes and trashes, etc.) and may adapt themselves to various types of habitats (Sacchi et al. 2008). Sacchi et al. (2008) have shown that, even if rats were irregularly scattered over an Italian city, their occurrence was more reliably predicted by spatial-interpolation (i.e., inverse square distance weighing) than by the use of landscape features as predictors of species occurrence. In an extensive survey in the UK, the prevalence of commensal rodents, *R. norvegicus* and *M. domesticus*, was related to: the occurrence of pets or livestock on the premises, whether houses were occupied, to areas with relatively low-density housing, the age of the houses and the occurrence of litter, the amount of unkept gardens and overall dereliction in a neighbourhood (Langton et al. 2001). Human densities have been related to the risk of birds of prey being exposed to ARs, likely related to the availability of contaminated prey (López-Perea et al. 2015). In another study, the percent of commensal rodents in the diet of barn owls was associated with the degree of rural landscapes in the territories of the owls (Hindmarch and Elliott 2014).

C.2. How landscape determines the community of non-target small mammals?

A complementary issue is related to how landscape features influence both structure (diversity, species richness) and composition of the small mammal community and thus the pool of potential NTSM exposed to ARs (Walz 2011). On continental/national scale, successful attempts have been conducted to address this. For instance, in temperate grasslands of South America, landscape variables explained 66.6% of variation in the composition of rodent assemblages in an area of 45,000 km² (Massa et al. 2014). However, at regional and site scales, consistent patterns remain to be established. Landscape composition partly determines the occurrence of species according to their habitat preferences, but small mammal communities were thought to be the result of complex interactions between landscape characteristics like habitat type and quality, size and shape of habitat patches, matrix quality, spatial arrangement and degree of connectivity between patches (Tews et al. 2004; Michel et al. 2007; Umetsu et al. 2008). Furthermore, multiannual variations of population densities of each species lead to large changes in community structure that may escape investigators if sampling is not carried out over a sufficient time span (Giraudoux et al. 2007). Moreover, the influence of habitat on small mammal communities operates at nested spatial scales, from landscape to micro-habitat (Burel et al. 2004; Michel et al. 2007) and the presence and dynamics of NTSM in a given habitat is affected by the surrounding mosaic of habitats. In agricultural areas for instance, Michel et al. (2006) showed that landscape features influenced small mammal communities living in hedgerows. It was shown that in most landscapes, the decrease of farmland fragmentation due to hedgerow removal and subsequent field size increase resulted in less diversity of small mammals,

although populations of some specific species may be more abundant. Moreover, while the wood mouse and the bank vole were predominant in the most opened landscape, *Sorex* shrews were more abundant in the more heterogeneous and connected ones.

C.3. How landscape modulates the spatial activity pattern of small mammals?

Landscape features (heterogeneity and patchiness, patch quality and connectivity, etc.) determine the spatial variety of resources, and the spatial configuration of patches drives their accessibility for small mammals as well as their movements on local scale (see for instance the review of Peles et al. (1999). It is commonly assumed that landscape may control small mammal exposure to chemicals by modulating their spatial activity pattern (Hope 2000; Carlsen et al. 2004; Vermeulen 2009). Concerning ARs, some studies have assumed that habitat and/or landscape characteristics can affect the occurrence of AR residues in NTSM (Brakes and Smith 2005; Gedhun et al. 2014) but these studies were not designed to address specifically this issue. Thus, identifying which landscape features drive AR transfer to NTSM remains an open and critical field of research. Hereinafter, we discuss in a prospective way how landscape may modify movement patterns of NTSM and their exposure to AR.

Several studies strongly suggest that distances travelled by small mammals in heterogeneous landscapes are longer than in homogeneous landscapes (see review and examples for different species in (Kozakiewicz 1993; Szacki et al. 1993; Diffendorfer et al. 1995). Small mammals preferentially move in habitat corridors rather than through unsuitable habitats and, in some cases, might even concentrate their activity into corridors (Wegner and Merriam 1979; Merriam and Lanoue 1990). Corridors and small suitable patches may facilitate small mammal movements, and, accordingly may structure the patterns of spatial activity (Marrotte et al. 2014). Some landscape features may act as barriers, preventing NTSM to encounter areas treated with ARs or, conversely, enhance the probability that NTSM home-range overlaps the treated area. The effects of barriers act differently at individual and population level. Habitat barriers are most often not absolute barriers: they can be effective inhibitors of movements acting at the level of individual home-range but do not stop dispersal or colonization since they can be crossed, although infrequently. Barrier characteristics are species-specific, meadows for instance can be considered as barriers for yellow-necked mice, while they are optimal for the common vole. Roads constitute a universal case of artificial habitat barrier for small mammals (McDonald and St. Clair 2004; Rico et al. 2007). Unsuitable habitats can either slow down dispersal and colonization or, contrarily, accelerate movement rate because individuals might cross them faster and are less tempted to settle their territory in such marginal or hostile habitats (Wijnhoven et al. 2006). By restricting the range and the direction of spatial movements or limiting the time spent in a given area by the individuals, habitat corridors or barriers might modulate the exposure of NTSM to AR. This sounds an important issue for risk assessment and may bring opportunities for risk mitigation.

Prospects

In order to improve our understanding of exposure patterns and processes and help in developing ecologically-relevant risk assessment and countermeasures, we propose some recommendations with regards to landscape ecology for further studies from literature cited and the hypotheses above:

- assess the overlap in traits of target and non-target species, and use a proxy for such overlap as an indication of risks that the specific non-targets may be exposed to a ARs used for a specific target species (trait profiling)
- the role of connectivity in various landscapes should be investigated, testing whether it may modulate exposure in case of PPP or biocidal uses.
- the relationships between landscape heterogeneity, small mammal diversity and main ecological traits of species in the community, notably those related to AR exposure, should be studied further in order to better characterize the features of potential NTSM pool in urban and rural areas.
- the impact of seasonal and long term multi-annual variations of food resources, for small mammal (e.g. fruits, seeds, human wastes, etc.) and of small mammal population densities for small mammal eaters (carnivores, birds of prey, etc.) which determines both space use and food/prey switches within a landscape and possibilities for exposure.
- it may be of great interest to assess how temporal variations in space use according to landscape feature, notably heterogeneity and connectivity, may shape exposure with regards to the temporal patterns of AR treatments.

4/ Modelling of spatially explicit risks of rodenticides for NTSM

In order to model exposure of NTSM to AR rodenticides in a spatially explicit way, it is essential to predict where rodenticides are being used, and where NTSM are and forage. When combining the modelled spatial distribution of the NTSM and the modelled potential area for AR-usage, areas of overlap may be delineated in which exposure potentially may occur. Such approach has for instance been applied to assess potential risks of fisheries on albatrosses (Copello et al. 2014). Proxies to quantify the extent of overlap (Fieberg and Kochanny 2005) may be useful for the final assessment of the potential of NTSM to be exposed to ARs (e.g. percentage of overlap in area of AR usage and optimal habitat, number of NTSM-territories within area of usage etc.). Whatever the spatial scale of interest, it should be noted that this approach only addresses the potential of NTSM to be exposed, it does not predict the actual occurrence of exposure, which is finally driven by the actual foraging of the NTSM on the bait. This may be driven by the status of the individuals, e.g. condition, fate reserves or other individual characteristics (Godsall et al. 2014) and measures taken to prevent NTSM exposure. This demands further quantification of the effects of those factors that drive relationships between potential and actual exposure.

There are different approaches to model more realistic exposure of NTSM to AR: 1) correlative/(geo)statistical models, 2) analytical models and 3) simulations models using Individual Based Models (IBM) or Agent Based Models (ABM). The choice of approach depends on the available data and information, the questions asked and the scale of interest.

Correlative/(geo)statistical models - ARs are being used as plant protection products and as biocides. As reported above in the chapter, both have their own spatially explicit usage patterns, e.g. as PPP generally in larger scale agricultural systems while biocidal use is generally in and around buildings. As PPP, rodenticides will be used in reaction to (local) outbreaks of commensal rodent populations. Depending on the geographic region and habitat suitability, such outbreaks may be related to for instance rainfall, increasing the primary production of the region (Stenseth et al. 2003; Pavey and

Nano 2013), to snow cover or temperature (Esther et al. 2014). Spatial distributions of small mammals have been correlated with habitat characteristics, species specific traits or other predictors (Gough and Rushton 2000). Based on the relationships between the occurrence of NTSM and (a)biotic factors, several statistical techniques may be used to predict rodent population outbreaks in a spatially explicit way. These techniques however focus on correlations, and as said, will not necessarily provide causal relationships. Esther et al. (2014) for instance used a Tree Regression approach to develop a model predicting vole densities using data of weather conditions. Such correlative approach, including Bayesian modelling, may be facilitated by remote sensing data providing information on habitat suitability, and GIS-tools allowing the spatially explicit modelling. In an approach to model the spatial distribution of bank voles across Europe, a so-called Genetic Algorithm for Rule-set Prediction (GARP) was applied (Amirpour Haredasht et al. 2013). In this algorithm, an initial set of rules is developed to relate spatially explicit information on e.g. climatic conditions and habitat characteristics to the abundances of bank voles. In further iterative steps the initial rules were changed and evaluated. It may be regarded as a stepwise regression approach. (Vaniscotte et al. 2009) used a mixture of regression techniques with discriminant analyses in order to predict the spatial occurrence of small mammal assemblages in China using habitat characteristics, derived from remote sensing data, as predictive variables. Multivariate approaches have also been developed to assess factors that drive spatio-temporal dynamics of rodent populations (Pavey and Nano, 2013). Classification and Regression Tree methods (CART) have been used to associate categorised data on outbreak risks with different predictors (Blank et al. 2011). Using CART methods, nodes can be defined at which a predicting factor splits the datasets with cases most relevant with regards to the categories. Each tree resulting from a node can be assessed again, creating a tree structure until criteria are met, terminating the calculations. With this method, categorized data on risks of (regional) outbreaks of voles were analysed to provide the most important factors underlying these risks in a spatio-temporal way. Such an approach may be useful in predicting the need for rodent control with e.g. ARs. Fairly simple correlative models have also been used to analyse outbreak patterns of target species in situations in which limited data was available (mainly rainfall) in order to further assess potential cost benefits of outbreak control (Davis et al. 2004).

Analytical models - Analytical models provide a definite solution to a given case, and are based upon a set of equations. Using matrix models, based on the most vital rates, e.g. growth, reproduction and mortality, the population dynamics of rodents may be projected. This has for instance been done for small rodents in Fennoscandia, as prey for the Arctic fox (Henden et al. 2008). A big advantage of analytical models is indeed that they provide a definitive closed solution to a modelling case. However, although analytical models are extremely useful for addressing temporal population dynamics, they are very limited for use in a spatially explicit context. This makes them less applicable for predicting potential risks that specific usage of AR may pose to NTSM in a spatially explicit way. In a general setting, potential effects of AR on rodent population dynamics can be quantified by the application of analytical tools, however without the spatial dimension.

Simulation approaches - Individual or Agent Based Models (IBM/ABM) may be used to simulate both the spatial (foraging) ecology (Gotteland et al. 2014) as well as the application of pesticides (Liu et al. 2013), likely also AR. In this approach, each rodent is modelled individually and given a set of properties and traits, e.g. age, sexual maturity, location, feeding status. Based upon these properties, each individual can make decisions, also in relation to environmental conditions. This may not only to change its position in the next step but also its activities and state. These decisions are based upon

the combination of the status of the properties and traits of the individual, with the potential for stochasticity. For the wood mouse a spatially explicit model has been developed, in which the spatial movement and behaviours of a local population of wood mice was coupled with the activities of farmers and the use of pesticides (Liu et al., 2013). Other models have been developed to simulate the spatially explicit interaction of rodents with cats, and the effects on the spread of toxoplasmosis (Gotteland et al., 2014). As mentioned before, simulation models do not provide a definite solution to the modelled case, but can be used to simulate the processes and test if hypotheses on the behaviours/processes may lead to the spatial patterns observed empirically. These models are quite flexible and have a wide envelope of applicability, using species specific information on the organisms involved and case specific info on e.g. the environment and habitat of the case.

Prospects for modelling

The prospect of modelling of the use of ARs and the potential risks for exposure to NSTM spatially explicit under relative scenarios are quite positive. From other fields of research different modelling approaches are available. Correlative and simulation approaches seem to be the most promising tools, due to their versatility in application (figure 3). In order to analyse patterns in usage and exposure risks, large scale statistical modelling exercises should be employed. The results of such pattern analyses will provide global overviews of usage patterns, for instance related to large scale outbreaks in case of PPP-use, and potential exposure both for biocidal as well as PPP usage. It should be noted that the design of correlative studies should enable extrapolation of results to other cases as much as possible.

For detailed site specific assessments, either as biocide or as PPP simulation IBM/ABM may be more appropriate, integrating species specific information on traits, behaviour and ecology with local conditions. Usage patterns may be derived based on statistical approaches, or also by simulation. It should be noted that use of ARs is dependent on the status of populations of target species, so, when applying simulation models it might also be useful to model not only the NTSM but also the target species. Analytical models are limited for modelling in a spatially explicit way, and have generally a limited scope for application. This limits their use in spatially explicit modelling of AR-exposure to NTSM.

As mentioned above, modelling frameworks have been developed in adjacent research areas, and can be adopted for spatially explicit AR-exposure modelling. In this light it is therefore more appropriate to focus research efforts on the derivation of parameter sets to feed the general models for application. A major issue that may need some attention is that exposure to ARs occurs as discrete events. Uptake of conventional environmental chemicals like POPs and metals is generally modelled as a continuous process, for instance in case of POPs using Bioaccumulation Factors. Event driven uptake of chemicals generally increases the influence of stochasticity, which is of prime importance for modelling through simulation using IBM/ABM.

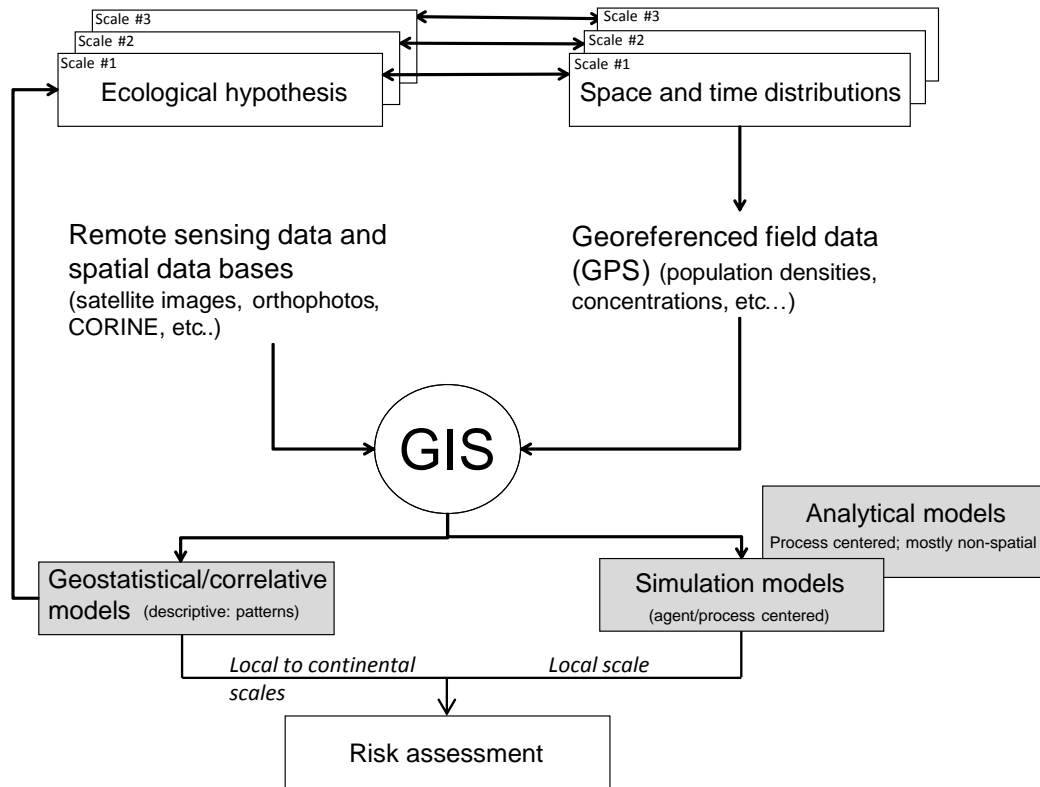


Figure 3. Synoptic scheme showing how different modelling approaches can be used to assess exposure of animals to AR in a spatially explicit way.

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