Reintroduction of the adder *Vipera berus* to Nottinghamshire: a feasibility study

Final Report to
People's Trust for Endangered Species

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Executive summary

The adder has experienced drastic declines in Britain, driven largely by habitat loss, degradation and fragmentation, as well as disturbance and deliberate killing. There is an urgent need for proactive management of the species. This study sought to establish the status of the adder in Nottinghamshire and report on the feasibility of reintroductions.

Field surveys in and around the location of the last known adder population in Nottinghamshire found no evidence of adders. This, together with previous surveys and a lack of reliable adder sightings for over 10 years, suggests adders are indeed extinct in the county. The historic loss of adder habitat, particularly from the Sherwood area, is believed to have driven the species' long-term decline, while the last known population was likely extirpated by scrub encroachment on isolated clearings within a forestry plantation.

Following a review of adder translocations in Britain and the practice of reintroducing reptiles, a computer simulation model was used to help determine the feasibility of adder reintroduction and assess factors likely to influence its success. The model suggests that a relatively large number of adult adders (over 70) are required for long-term population viability. As well as the behavioural response to translocation and the suitability and extent of habitat, rates of juvenile mortality and female reproduction are likely to be critical influences on population establishment.

There is a clear case for adder reintroduction to Nottinghamshire, with potential to serve as a case study for adder conservation. Given that areas of suitable habitat exist to support a population, a potential strategy for reintroduction is discussed, with recommendations for further research and assessment.

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Acknowledgements

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1. Introduction and background

1.1 Rationale for the proposed adder reintroduction

The adders' distribution throughout Eurasia and its description as of "least concern" in the IUCN red list of threatened species belie its present status in many parts of its range. In Britain, the adder faces an uncertain future following centuries of persecution and the loss and fragmentation of favoured habitats. The continued management of these habitats, particularly plagioclimax systems of moorland, heathland and grassland, is also a major concern as the fine-scale environmental requirements of adders are rarely incorporated into modern landscape-scale management regimes. The adders' low vagility and high ecological specialisation likely impede its ability to respond to these pressures. In addition, remnant adder populations are often small and isolated (many in England comprise fewer than ten individuals; Baker *et al*., 2003) and are susceptible to stochastic and genetic effects that militate against their survival (Fagan & Holmes, 2006). Gleed-Owen and Langham (2012) suggest the adders' range in Britain has reduced by 39% in recent times, and the species is believed to be close to extinction in Buckinghamshire, Oxfordshire, Hertfordshire and Greater London, and already lost from Warwickshire and Nottinghamshire. There is widespread recognition of an urgent need for targeted management to improve the conservation status of the adder both locally and nationally.

Reintroduction of the adder to areas from which it has been recently extirpated is a logical strategy to consider, offering the possibility of reversing local declines and expanding the species' current range, thereby improving its status in Britain. Consequently, and in light of *Biodiversity 2020* (the government's strategy for biodiversity in England which encourages projects to enhance the status of species at risk of extinction; DEFRA, 2011) and the adder's listing in UK legislation as a species of principal importance for biodiversity (Natural Environment and Rural Communities Act 2006), Nottinghamshire Wildlife Trust has proposed a program of recovery with the overall aim of restoring a viable adder population to Nottinghamshire. The present study sought to investigate the feasibility of, and inform decision-making towards, achieving this aim.

1.2 IUCN Reintroduction Guidelines and the need for feasibility assessment

Referred to collectively as 'conservation translocations' by the International Union for the Conservation of Nature (IUCN), the deliberate movement of organisms to reinforce existing populations or establish new populations inside or outside a species' indigenous range is an important and popular form of conservation management (Fischer & Lindenmayer, 2000; Seddon *et al*., 2007). However, it has a mixed record of success and failure (indeed, most efforts fail to establish viable populations [Griffith et al., 1989; Dodd & Seigel, 1991; Beck et al., 1994; Wolf *et al*., 1996]), which some authors have suggested reflects deficiencies in the planning and decision-making process (Griffith *et al*., 1989; Robert *et al*., 2015). Consequently, the IUCN's Species Survival Commissions Reintroduction Specialist Group (IUCN RSG) was formed in 1998, and produced a set of guidelines for reintroductions which have recently been updated as *Guidelines for Reintroductions and other Translocations* (2013), herein referred to as the 'IUCN

Reintroduction Guidelines'. The guidelines define best practice on the justification, design and implementation of any conservation translocation. Consequently, they inform the objectives of the present study so that key sections of the guidelines (Annexes 3-9) are addressed in relation to the proposed adder reintroduction.

2. Study objectives

The four main objectives of this study form a logical process and are outlined below:

1. Report on the current status of the adder in Nottinghamshire, with reference to the factors that may have caused its decline.
2. Review previous adder translocations and reintroductions.
3. Identify key factors likely to influence the success of a reintroduction program.
4. Propose a potential strategy for adder reintroduction.

3. Status of the adder in Nottinghamshire

3.1 Historic records

Early attempts to map national adder distribution include accounts from Nottinghamshire (Leighton, 1901; Taylor, 1948, 1963), however little is known of the species' occurrence and abundance in the county pre-1980. Recent records of adders are also limited, although there are verified accounts at six main locations (Figure 1). In the south, individual adders were seen on two separate disused railway embankments in 1981 and 1987, and crossing a road in 1986. A pair was also seen close to a railway line in the north of the county in 1987, not far from a restored gravel extraction site at which there were multiple adder reports in the mid-1980s. However, there have been no confirmed sightings of adders at any of these sites since, and all are likely to have been small, isolated populations. Records exists of adders throughout Clipston Forest (Figure 1), a large, predominantly conifer plantation, and while juvenile and adult adders were seen there regularly in the 1980s and 90s, the most recent record is of a dead specimen found in a small clearing in 2005. Since then, there have been several reports of adders elsewhere in Nottinghamshire, but none of them verifiable and some of which are believed to be cases of grass snake *Natrix natrix* misidentification.

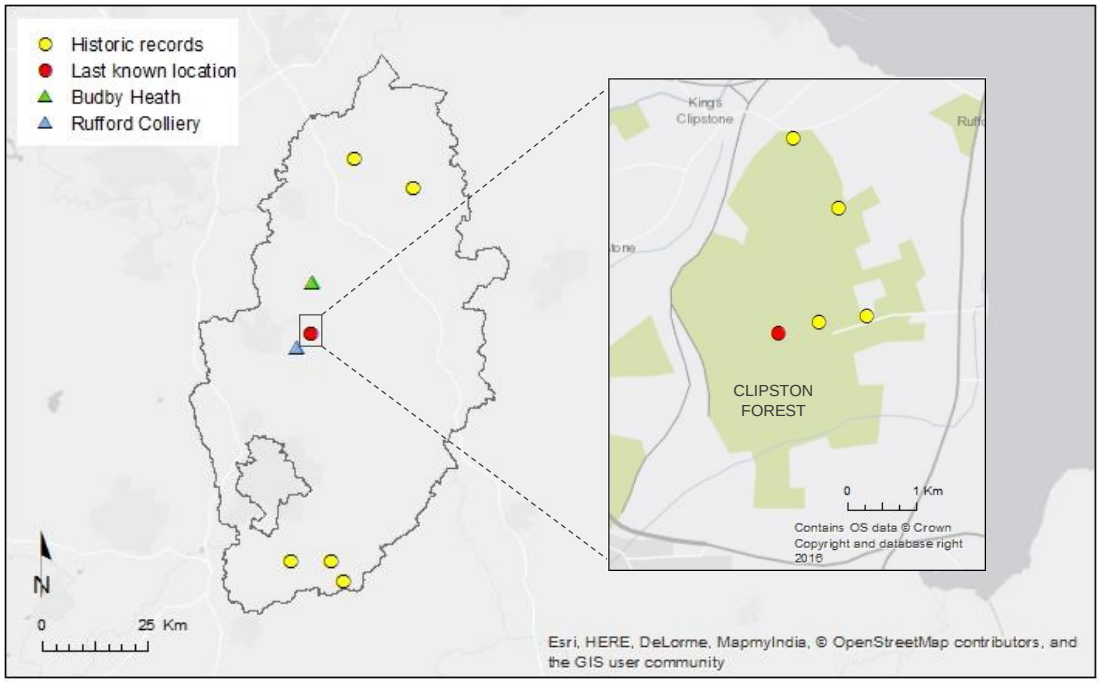

Figure 1. Locations of historic records of adders and the last known adder population in Nottinghamshire.

3.2 Field surveys

A key objective of this study was to establish the status of the adder in Nottinghamshire. Due to the limited historic records, and the absence of confirmed sightings for over ten years, we sought

initially to investigate whether an adder population is surviving in the county. Because this component of the study was restricted to a single survey season, and with limited resources, it was necessary to focus the survey effort. The most probable sites for a remnant adder population were identified as the disused railway embankments previously mentioned and the location of the last adder sighting, Clipston Forest. Each of the railway sites were surveyed by the county recorder for reptiles in 2012-14 (no adders or signs of adders were detected), and we were unable to obtain permission to access them within the limited timeframe. Therefore, survey effort was concentrated in and around Clipston Forest.

Survey area

There is a high level of human activity in the north of Clipston Forest due to a popular holiday village. Therefore, surveys were carried out in the quieter south of the forest and an adjacent disused coal mine, Rufford Colliery, that has been partially restored to heathland. The entire area covered approximately 650 ha, within which 13 survey sites deemed to be suitable for adders were established (Figure 2). These were composed of managed open heath areas, forest rides, and early growth (0-10 years old) plantation blocks (often used by adders [Jofré *et al*., 2015]). Open heath areas were characterised by a mosaic of dwarf shrubs, such as common heather *Calluna vulgaris*, dwarf gorse *Ulex minor* and bracken *Pteridium aquilinum*. A similar flora and vegetation structure was found in the early growth, or 'pre-thicket stage', plantation blocks, while forest rides were composed primarily of bracken and grasses with a complex sward structure.

Figure 2. Map of the survey area showing the 13 sites surveyed in the south of Clipston Forest (white) and adjacent Rufford Colliery (yellow).

Survey protocol

Consistent with recommendations for maximising the likelihood of detecting adders (Sewell *et al*., 2013), each of the sites was surveyed using a combination of refugia searching and directed visual transects. A total of 210 artificial refugia (roofing felt measuring 0.5m^2) were distributed in arrays over 0.2-0.5 ha with a spacing of 10 m apart. To increase their attractiveness for adders, refugia were positioned so as to receive the maximum amount of sunlight (i.e. unshaded and, where possible, south-facing) and in close proximity to vegetation cover. Simultaneously with the checking of refugia, directed visual transects involved an experienced surveyor (the author) and an assistant searching each site for >1hr. Surveys were typically carried out in the morning, with temperature 10-20°C. For presence/absence purposes, Sewell *et al*. (2012) showed that 4-5 survey visits are usually sufficient to detect adders at 95% of occupied sites. However, the number of visits should be increased where detectability may be low. Therefore, 7-8 survey visits were made, spaced at least one week apart.

Results

A total of 93 surveys in and around the location of the last known adder population in Nottinghamshire yielded no adders and no evidence of adders. Common lizards *Zootoca vivipara*, the only reptile species recorded, were found at all but one survey areas, and there are healthy lizard populations throughout Clipston Forest and in most of the restored heath at Rufford Colliery.

3.3 Conclusions

The results of field surveys carried out in this study and previously, as well as the lack of recent sightings, suggest the adder is indeed extinct in Nottinghamshire. Although it is believed that adders have always been scarce in the East Midlands (Taylor, 1963; Gleed-Owen & Langham, 2012), the Sherwood area may once have been a local stronghold for adders in Nottinghamshire; an extensive mosaic of oak and birch woodland interspersed with open sandy heath and rough grassland providing a relatively large area of accessible habitat (Clifton & Keymer, 2009). Due to agricultural intensification, commercial forestry and urbanisation, these open habitats have been reduced to less than 4% of their former extent during the last 200 years (Clifton & Keymer, 2009). Dry heath and acid grassland in Nottinghamshire now exists only as relatively small and isolated fragments (Figure 3). The severe loss of habitat and the degradation of remaining patches (Lambert, 2016) as a result of a lack of habitat management is believed to have driven the adders' long-term decline, while human disturbance and the encroachment of conifer (*Pinus spp*.) and silver birch (*Betula pendula*) in open rides within a forestry plantation likely caused the demise of the last known population in the county (Figure 4).

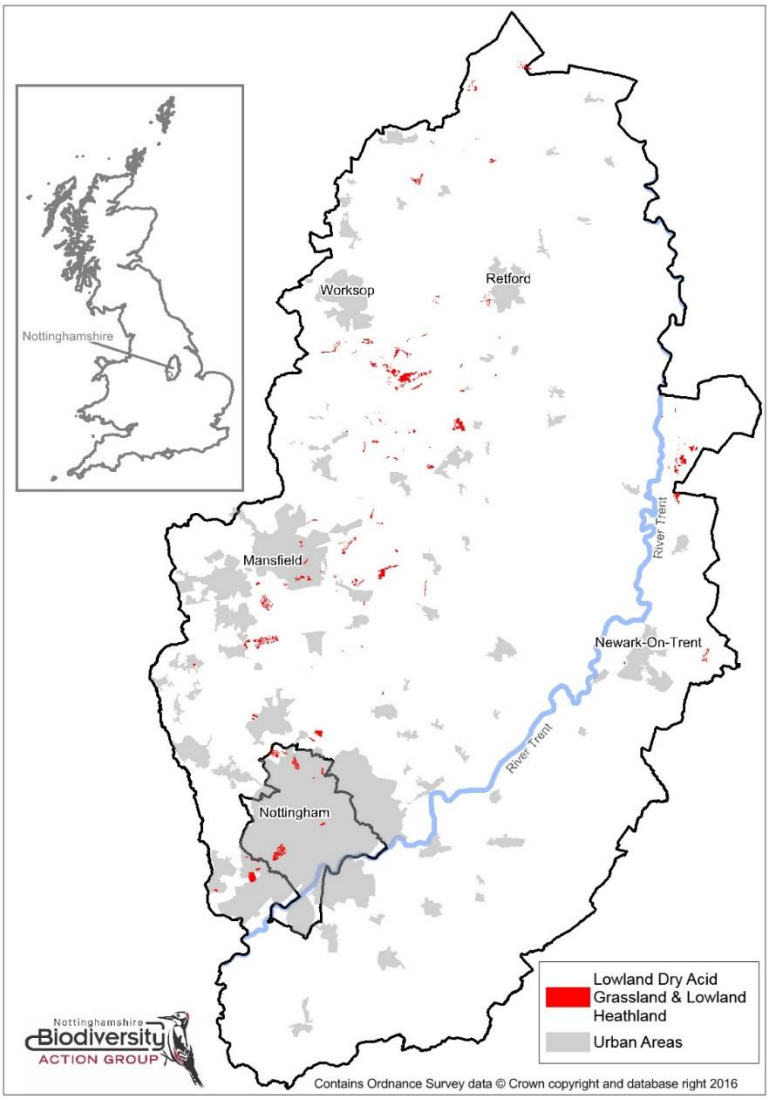

Figure 3. Present-day distribution of lowland dry acid grassland and lowland heathland, important habitats for adders, in Nottinghamshire. Map produced by Nottinghamshire Biodiversity Action Group using up to date Phase 1 survey data.

Figure 4. Encroachment of silver birch on a previously open heath area in Clipston Forest. While some tree cover is important for adders and other reptiles as it increases habitat structural complexity, the encroachment seen here has resulted in the area being over-shaded and unsuitable for adders.

4. Factors affecting the success of a reintroduction program

4.1 Lessons from previous translocations

Adder translocations have been undertaken in Britain for two main purposes. Firstly, the adder's legal protection against reckless killing and injury results in the frequent capture and translocation of populations away from development sites. This practice fulfils the legal obligation of the developer and is intended to avoid causing direct harm to adders. Secondly, a number of translocations have been carried out within the last decade explicitly for the purposes of reintroduction (i.e. adders were released into areas where they had been extirpated). This implies that a large body of knowledge exists from which clear lessons for future reintroductions can be drawn. In fact, most adder translocations, particularly those led by development, have included no feasibility assessment and little, if any, post-release monitoring. Consequently, they are in breach of IUCN Reintroduction Guidelines (2013). In addition, because success is often measured by short-term criteria, such as persistence into the following year or evidence of successful breeding (neither of which demonstrates that a population is self-sustaining), the long-term status and health of translocated adder populations is largely unknown. While there are too few documented cases of adder translocations to develop any common rules in terms of what regulates their success, several factors often lead to more successful translocations across taxa. These are addressed in relation to previous translocations of adders and reptiles in general.

Although intuitively obvious, a translocation must operate within the constraints of the biotic and abiotic requirement of the species. This depends to a large extent on habitat suitability (Griffith et al., 1989; Dodd & Seigel, 1991). If the release habitat is not of high quality, then the chances of a positive outcome are low even when all other factors are taken into consideration (Fischer & Lindenmayer, 2000). The habitat requirements of the adder are relatively well known (Frazer, 1983; House & Spellerberg, 1983; Reading & Jofré, 2015, 2016). They have a preference for an open, structurally complex habitat (Spellerberg & Phelps, 1977; Edgar *et al*., 2010; Reading & Jofré, 2009, 2015, 2016), such as lowland dry heath, that meets their thermal requirements, and offers foraging opportunities and shelter (Spellerberg & Phelps, 1977; Edgar *et al*., 2010). Habitat structure can be enhanced for adders through vegetation management, by providing features such as brash and log piles, and by constructing more substantial hibernacula (penetrable banks containing logs, rubble, reed bundles, etc.), such as those used for an adder translocation in Horsey, Norfolk (Whiting & Booth, 2012). In terms of climatic conditions and other geographic factors, the suitability of a release habitat is likely to be greatest within the historical range of the species, and when located as closely as possible to the source habitat (Griffith *et al*., 1989). The extent of suitable habitat must also be sufficient to support a translocated population and to allow its expansion. The carrying capacity of an area for adders can be difficult to assess, however, not least because adder populations tend to be patchily distributed in suitable habitat (Viitanen, 1967; Prestt 1971; Madsen & Shine, 1992). This "under-occupation" of habitats implies that adders have low ecological plasticity, although the individual

and population level implications for adder translocation (e.g. the size of receptor sites) are poorly understood.

Following translocation, adders often disperse over large distances, sometimes into habitat perceived to be less suitable (such as closed woodland). Experimental evidence suggests that translocated snakes move further and more erratically than resident snakes, possibly seeking familiarity in the landscape (Reinert & Rupert, 1999; Plummer & Mills, 2000; Butler *et al*., 2005). Adders translocated distances of almost 1 km for mitigation schemes in Essex returned to capture sites within 24 hours (Jon Cranfield, pers.comm.). As well as experiencing increased energetic expenditure, they may be less efficient at foraging or locating suitable refuges. Consequently, translocated snakes tend to exhibit reduced survivorship (Reinert & Rupert, 1999; Plummer & Mills, 2000). Homing behaviour and movement away from release sites is the most commonly reported cause of translocation failure among reptiles (Germano & Bishop, 2008). While a large area of suitable habitat will likely buffer against the behavioural response to translocation, a potential strategy is to temporarily restrict movement following release, so-called 'soft-release', to allow animals a period to acclimate to their new environment (Griffith *et al*. 1989). This approach was implemented at Horsey, where constructed hibernacula were enclosed within an area of grazing marsh. Following the translocation of 119 adders over a period of 2-3 months, fencing was removed to allow dispersal and a number of individuals used the hibernacula to over-winter and give birth the following year (Whiting & Booth, 2012). While the effect of soft-release on snakes is unstudied, a few cases show it can increase site fidelity and translocation success for reptiles (Tuberville *et al*., 2005; Alberts, 2007).

A number of reptile translocations have failed because of the release of insufficient numbers of animals (e.g. Cook, 2008). Small population size can result in susceptibility to stochastic factors (e.g. demographic and environmental stochasticity, genetic drift, and catastrophes), as well as problems associated with social behaviour, finding mates, and group living (Courchamp *et al*. 1999; Stephens & Sutherland 1999). A cross-taxa review reported that reintroductions appear to be more successful when at least 100 individuals were released (Fischer & Lindenmayer, 2000). Also important is the source of founding individuals, with wild animals being more successful than captive animals (Griffith *et al*. 1989; Fischer & Lindenmayer 2000). Behavioural changes that decrease survivorship of translocated snakes are accentuated for captive-bred individuals that may be less efficient at feeding or locating suitable refuges than wild-caught relocated animals (Plummer & Mills, 2000; Mathews *et al*., 2005). While all adders translocated for the purposes of development mitigation are wild-caught, those reintroduced to Maulden Wood, Bedfordshire, were captive-bred at the New Forest Reptile Centre in Hampshire. 63 juvenile (up to one year old) adders were translocated from the Reptile Centre to Maulden Wood over a period of five years (Sue Raven, pers.comm.). Some of these were photographed and several individuals were identified in subsequent years. Although surveys have been sporadic and there is evidence (two opportunistic sightings) of dispersal into adjacent conifer plantations, peak counts of adders in Maulden Wood have since declined to very low numbers (only one so far in 2016; Sue Raven, pers.comm).

The Maulden Wood reintroduction raises another important consideration, the developmental stage of released animals. Although there has been no research into the effects of age class on snake translocation outcomes and a review of reptile translocations found no difference in success rates (Germano & Bishop, 2009), certain age groups may be more appropriate than others. When dealing with species that show strong homing tendencies, it may be beneficial to release eggs or juvenile animals rather than adults that have had sufficient time to develop strong associations with a home site (Gill, 1979; Tuberville *et al*., 2005; Canessa *et al*., 2014; Gibbs *et al*., 2014). On the other hand, the greatest threats to individual survival come at younger life stages, when animals are more vulnerable to predation and other dangers in the wild (only 10-20% of adders survive to sexual maturity), so it may be better to release adults or sub-adults. In contrast to the Maulden Wood reintroduction, an adder population translocated from a development site to Hounslow Heath in Greater London comprised 42 adults (including gravid females). There is little chance of dispersal outside the 80 ha site and since the translocation in 2000, the population appears to have grown, with peak counts of around 50 adders in recent years (Chris Slack & Gareth Matthes, pers.comm.).

A critical factor in any reintroduction is detailed understanding and reduction, or preferably removal, of the initial causes of decline (Kleiman, 1989; Dodd & Seigel, 1991, IUCN, 2013). This was integral to the success (Daltry, 2006) or, where lacking, failure (Read *et al*., 2011) of well-documented snake reintroductions. The killing of adders by foresters at Maulden Wood and habitat degradation and disturbance at Hounslow Heath represent major causes of historic adder declines in Britain. Both factors were believed to no longer be acting at the point of adder reintroductions at these sites. However, it can be difficult to diagnose causes of population declines; multiple, interacting factors may be at play, emphasising the need for thorough investigation. This is especially important when animals are reintroduced to an area following an extended period of absence and may have new threats to contend with. At Maulden wood, for example, there are high densities of pheasants (known to kill young adders) due to increases in local game farming. Urban heaths support high densities of domestic and synanthropic predators (such as cats, dogs, corvids and foxes), as well as being subject to high levels of recreational activity. This does not appear to have been overly detrimental for reintroduced adders on Hounslow Heath, however. Visitors are reported to be remarkably accepting of its adder population, with no complaints or incidents of conflict between adders and the general public. Human presence may even deter predators of adders. The population has apparently also benefited from restricted public access on parts of the heath, together with educational notice boards and an encouraged sense of public responsibility for the adders. Bonnet *et al*. (2015) also demonstrate this to be an effective approach for the conservation of venomous snakes in densely population areas.

4.2 Simulating adder reintroduction using viability analysis

4.2.1 Methods

Population viability analysis (PVA) is a process of evaluating the full range of forces impinging on populations to make determinations about their risks of extinction, and chances of persistence,

within a given time frame (Gilpin & Soulé, 1986; Beissinger & McCullough, 2002). PVA has become an important tool in the field of conservation biology (Lindenmayer *et al*., 1993; Gerber & González-Suárez, 2010), although it has been little used on snakes. When reintroduction is an option, PVA can be used to help determine its feasibility, and to inform important aspects of reintroduction strategy. It is important to note, however, that this form of computer simulation modelling provides a prediction of the likely behaviour of populations in response to selected parameters (Lacy, 1993), rather than revealing what will happen. Various computer programs are available for PVA, and we used VORTEX (Version 10.1.0.0; Lacy & Pollak, 2014), an individual-based Monte Carlo simulation.

Baseline model and model assumptions

The life-history parameters used to model an adder population in this analysis are presented in Appendix 1, and are derived from detailed field studies of lowland adder populations from the UK (Prestt, 1971; Phelps, 2004; Sheppard, 2010) and Sweden (Madsen & Shine, 1992, 1993, 1994). Complimentary data from these studies enabled the construction of a baseline model using demographic parameters for two age classes: juvenile (0-3 years) and adult (>3 years). A single, isolated population was modelled, with no natural immigration. Deterministic factors that are known to influence the persistence of heathland reptiles such as environmental catastrophes (e.g. wildfire) and habitat degradation through scrub encroachment were not included because we assumed habitat management of receptor sites would be favourable for adders and to facilitate interpretation of the results.

Despite data being available for most demographic parameters, it was necessary to estimate certain variables and to make several assumptions in the analysis. Data are not available to allow the calculation of environmental variation and the associated fluctuations in birth and death rates of adders. It was assumed that low fluctuations in the environment would cause demographic rates to vary with a standard deviation across years of 10% of the mean value (e.g. the probability of an adult female producing young was 50% ± 5% SD). The effects of stochasticity on fecundity, survival, and carrying capacity (K) were assumed to be correlated within the population. K was set as double the initial population size, with a 'ceiling model' in which density-dependence in survival is invoked only if population abundance increases above K, at which point K is imposed by truncating each age class (Strong, 1986). Because true carrying capacities for adder are largely unknown, K was used in a broader sense in the analysis as a measure of habitat suitability and/or the extent of habitat.

Implementation and model output

The extinction threshold was set to one remaining adder with a time horizon of 50 years (approximately eight generations) to predict population persistence. Each scenario was subject to 1000 replications and the probability of extinction was estimated as the proportion of replications in which a population became extinct. The results were assessed using the deterministic growth rate (λ), stochastic growth rate (r) and mean values (±SD) of three statistics: time to extinction, final population size and expected heterozygosity.

Reintroduction scenario

A series of models was constructed to determine the viability of a reintroduced population according to variation in the following demographic parameters:

Population age structure

We specified age distribution to model the release of populations composed entirely of juveniles (age = 1) or adults (age = 3). An equal sex ratio was selected throughout.

Population size

In order to assess the influence of the number of animals released, initial (or founding) population size (N_i) was varied from 10 to 100 in increments of 10. Carrying capacity was adjusted as to keep a constant ratio ($K=2N_i$) for separate simulations.

Sensitivity analysis

To account for uncertainty in model parameter estimates and to identify those most greatly influencing growth rates and viability of simulated adder populations (McCarthy *et al*. 1995), sensitivity was evaluated by a conventional "manual perturbation" (Mills & Lindberg, 2002). This involved systematically varying the values of specific parameters, while holding all other parameters constant. Model parameters selected for sensitivity analysis were reproductive and mortality rates, and carrying capacity.

Reproductive and mortality rates

The percentage of females breeding and mortality of adders of different sex and age classes (i.e. juveniles, adults, adult males and adult females) were varied individually by ± 25% of their baseline value. Sensitivity of parameters was compared by calculating a standard sensitivity index using the formula $S=[(\lambda_{-25} - \lambda_{+25})/(0.5*\lambda_0)]$ where λ_0 is the annual rate of population growth calculated in the baseline model, and λ_{-25} and λ_{+25} are the outputs using the adjusted parameter values (Heppell *et al*., 2000).

Carrying capacity

K was set as double N_i in the baseline model. We assessed its influence using simulations where K relative to N_i was increased ($K=4N_i$) or decreased ($K=N_i$).

4.2.2 Results and discussion

Founding population age structure and size

The results of the PVA are shown in Appendix 2. Lower founding population sizes increased the risk of extinction in adders over the 50 year time frame, and founding populations composed of juveniles had a greater risk of extinction than those composed of adults (Figure 5).

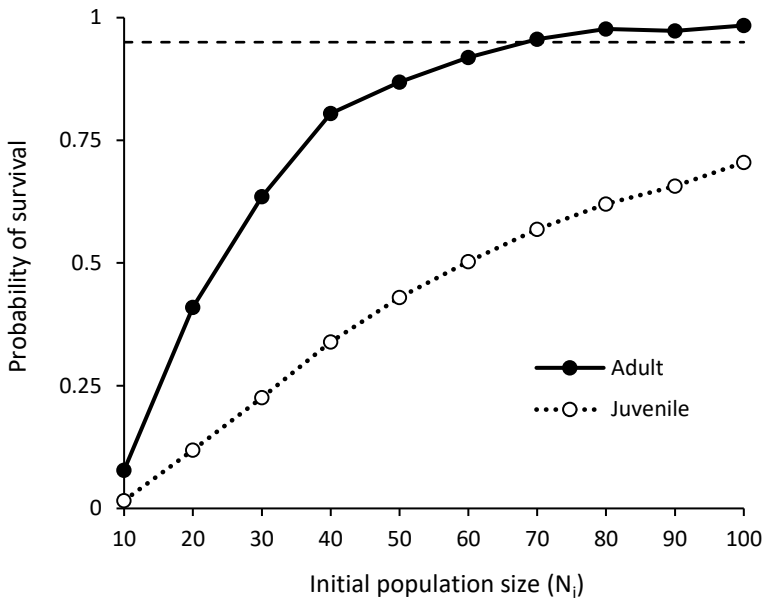

Figure 5. Changes in the probability of survival with increasing initial population size for founding populations composed of only juvenile or only adult individuals. The dashed horizontal lines marks a 95% chance of survival over a 50-year period, beyond which populations were considered to be viable in the long-term.

Founding populations of 10 juvenile adders became extinct within a single generation (<7 years), and populations of 50 juveniles faced a 57% chance of extinction over 50 years. The largest founding population of juveniles modelled, 100 individuals, declined by 3% per year, resulting in a mean final population of 72 individuals and a 30% chance of extinction. In founding populations of adult adders, the probability of extinction was reduced from around 90% with 10 individuals to around 20% with 40 individuals. 40 individuals were required for a positive stochastic growth rate, however this remained low (<1% per year) in all population sizes modelled. In the context of this study, populations were considered to be demographically stable and viable in the long-term when having a 5% or lower probability of extinction over a 50-year period. Constituting the minimum viable population size (MVP), this threshold was achieved only by founding populations composed of at least 70 adult adders, which, among extant populations, had a mean final population size of 74 individuals. However, populations of this size also experienced an average loss of around 20% of the expected heterozygosity. A 10% loss of genetic variation may lead to inbreeding depression (Ralls *et al*., 1988) and/or reduce the ability of a population to adapt to environmental changes, and has been recommended as the maximum acceptable loss in efforts to ensure the conservation of species (Soule *et al*., 1986). Larger founding population sizes reduced the loss of expected heterozygosity, and the present model suggests more than 100 adult adders are required to retain 90% genetic variation over 50 years. However, variation in parameter estimates, as well as the relatively simplified nature of population modelling and the suit of assumptions that underpin VORTEX, mean that the number of individual required for demographic and genetic stability may be greater than determined here.

Reproductive and mortality rates

Perturbation of vital rates showed that simulated adder populations were influenced most by juvenile mortality, followed by the percentage of females breeding (Figure 6). The effect of adult mortality on the model was accounted for almost entirely by female mortality, with very little influence of adult male mortality on simulated populations. This is logical for a polygynous species, in which females represent the breeding potential and therefore the ability of the population to grow and to recover from declines. The sensitivity of the model particularly to juvenile mortality, but also to adult female mortality and the proportion of females breeding, emphasises that these would be central factors against population establishment. Therefore, sufficient vegetative cover from predators, places to hibernate and an abundant prey base (i.e. a high level of habitat suitability) would strongly promote the viability of a translocated population.

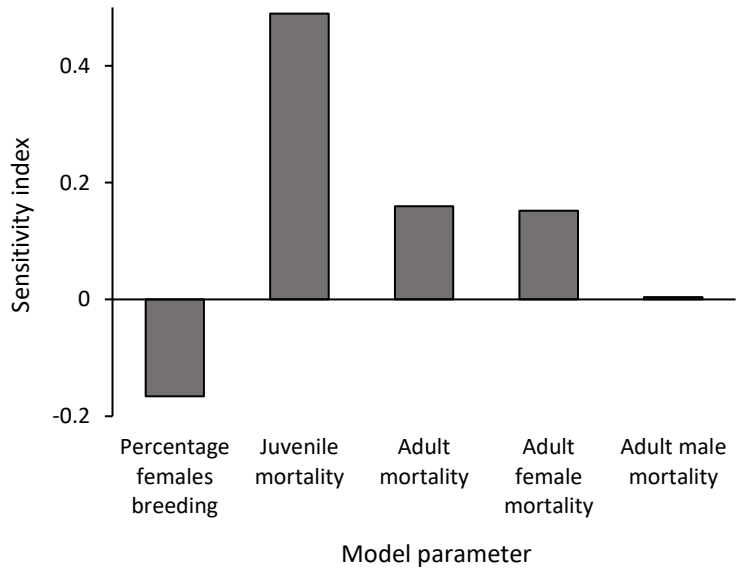

Figure 6. Sensitivity of population growth (λ) to adder vital rates in the PVA. The sensitivity index was calculated by varying each parameter ± 25% of its baseline value.

Carrying capacity

In the baseline model, carrying capacity was set as double the founding population size. Altering carrying capacity relative to a founding population of 70 adult adders led to changes in measures of viability. Intuitively, when carrying capacity was increased, there was an increase in the mean final population size and a decrease in the probability of extinction (Figure 7). A proportional reduction in carrying capacity resulted in a smaller mean final population size and an increase in extinction probability (Figure 6). Carrying capacity is interpreted here as habitat extent and/or quality; these factors directly relate to the number of individuals that an area can support. Therefore, to avoid limiting the growth of a translocated adder population and reducing its viability, it is important to maximise the amount and suitability of available habitat.

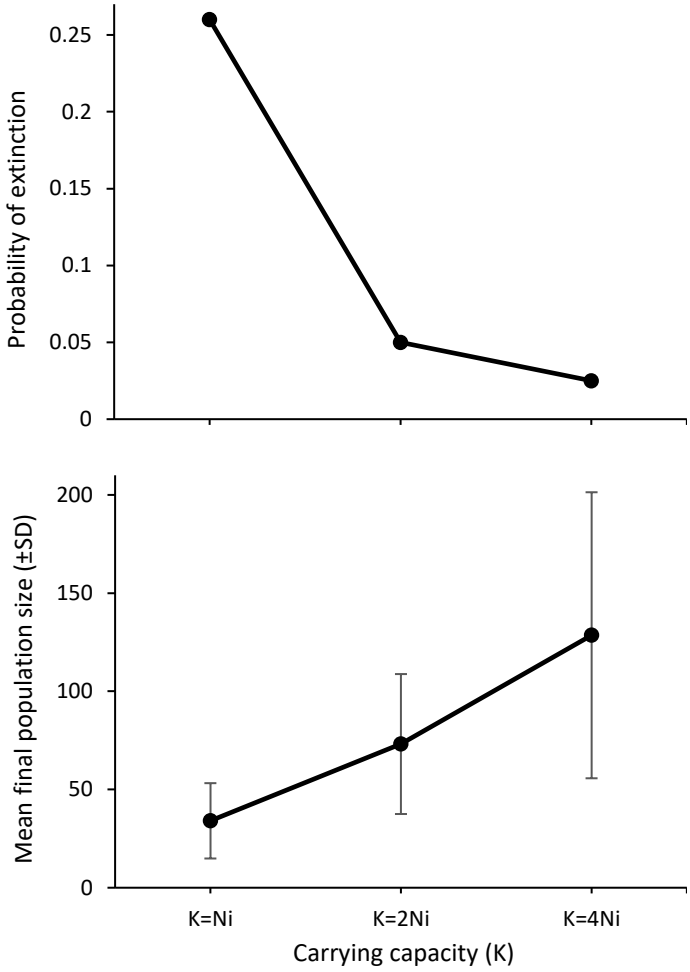

Figure 7. Effect of carrying capacity on mean population size (of those populations extant at the final time step) and the probability of extinction.

5. A potential strategy for adder reintroduction in Nottinghamshire

Source of adders for reintroduction

The PVA suggests a large number of adult adders (over 70) with a roughly equal sex ratio are required to found a viable population. Few existing adder populations in Britain are large enough to consider harvesting this number of individuals and there may be serious impacts on the donor population. Due to high costs of captive breeding, as well as associated risks and deleterious effects, the most logical source is a lowland adder population (or populations) subject to development-led translocation. However, increasing the founding population size and/or supplementing a reintroduced population using captive-bred juvenile adders from the New Forest Reptile Centre would likely increase viability.

Release strategy

Translocation is likely to occur in spring because adders are easiest to capture at this time of the year. The capture, transit and release of snakes should occur over a minimal time frame to avoid causing additional stress. The creation of hibernacula and a soft-release should be considered to ensure access to over-wintering sites and to discourage movement away from the release site in the early stages of the reintroduction. It may be beneficial to release juvenile adders with adults or within established home ranges because it is likely that they navigate to habitat features, such as hibernacula, by following the scent trails of adults.

Potential release sites

At present, the location of the last known adder population in Nottinghamshire, Clipston Forest, is not considered suitable for adder reintroduction because the perceived causes of decline and extirpation remain. Although this study does not include an assessment of available habitat in Nottinghamshire to identify locations for adder reintroduction, Rufford Colliery and Budby South Forest are suggested as potential sites for the following main reasons:

- They are within the wider Sherwood area previously occupied by adders, and in close proximity (<10 kms) to the location of the last known adder population (Figure 1).
- They represent large areas of suitable habitat: Rufford Colliery covers an area of approximately 250 ha and is in the process of being restored to dry heath/woodland mosaic, while Budby South Forest is the largest remaining patch (>150 ha) of continuous heathland in Nottinghamshire. Both sites scored as “good” or “excellent” in suitability for adders (Lambert, 2016) and are bordered by extensive plantation forest. Rufford Colliery also has good connectivity with Clipston Forest and two heathland reserves, offering the potential and natural (re)colonisation through dispersal.
- They are managed for conservation with scope for short- and long-term measures for a reintroduced adder population to be incorporated into site management.

While Rufford Colliery has excellent potential as a future reintroduction site for adders, Budby South Forest currently offers the most immediate potential in terms of the extent of suitable habitat. However, further site-level habitat assessments are needed to evaluate potential threats and prey availability, and to identify the most appropriate locations for release.

Post-release monitoring

Consistent with IUCN guidelines (2013), an adder translocation should be closely monitored to assess its progress, to check for any adverse impacts, to allow adaptive management if necessary and to evaluate success. It is necessary to design a full post-release monitoring program prior to any release taking place. This could include regular surveys immediately following release, and spring emergence and transects surveys to enable peak counts to be made and to identify hibernacula, basking sites and foraging grounds. This information can inform management activities to avoid causing disturbance and damage to habitat features and sensitive areas, although habitat suitability should also be monitored to ensure favourable conditions for adders over the long-term. Photographing, identifying and recording all released adders in a database would allow population and survival estimates (using annualised capture data in the capture-recapture programme MARK), as well as an assessment of dispersal. Using radio telemetry to monitor the behaviour of individuals, particularly movement and habitat use, would help to better understand adder response to translocation and identify further measures to improve survivorship. Survival into subsequent years, recruitment of juveniles into adults and breeding activity are positive signs, however population stability, or growth, measured over a longer timeframe (>10 years), and progressive colonisation throughout available habitat, are consistent with the establishment of a viable population. It can take a number of decades before reintroduction success can be reliably evaluated (Dodd & Seigel 1991; Nelson *et al*. 2002; Bell *et al*. 2004), and long-term monitoring is necessary to determine if further intervention is needed (Seddon, 1999). If feasible, designated site features, biodiversity and other species of conservation concern (such as ground nesting birds), should be monitored simultaneously to assess potential effects of a reintroduced adder population and/or associated management measures.

Communication and engagement

Consistent with IUCN guidelines (2013), a detailed account of all project stages should be maintained so that the reintroduction can be documented in the scientific and practitioner literature (e.g. the journal *Conservation Evidence*), noting methodological points of interest as well as the overall biological and conservation outcomes. Public awareness and the dissemination of information through press releases, presentations to local communities and, on open-access sites, educational notice boards is encouraged. However, while there may be important benefits of public engagement for reintroduction programs and for adder conservation specifically, the dissemination of information should be balanced against a number of risks, particularly disturbance and persecution. Safeguarding a reintroduced adder population should be prioritised, particularly during the early stages of establishment, and may include measures to limit or prevent public access to the release location.

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Appendices

Appendix 1. Values for parameters and life-history attributes input into VORTEX to build individual-based models for viability analysis of the adder *Vipera berus.*

Parameter	Value
Carrying capacity (K)	$2N_i$
Inbreeding depression	No
Mating system	Polygynous
Adult males in the breeding pool (%)	100
Breeding age (yrs)	3
Average litter size ± SD (1 litter per year)	8 ± 2
Sex ratio at birth	1:1
Maximum longevity (yrs)	30
Annual mortality of juveniles (%) (0-3 years old)	50
Annual mortality of adults (%) (>3 years old)	24
Percentage of adult females breeding	50
Environmental stochasticity	
Binomial variance for breeding success	10 % of the mean
Binomial variance for mortality rates	10 % of the mean
Poisson variance for carrying capacity	10 % of the mean
Deterministic population growth, lambda (resulting from above life table parameters)	1.0066
Calculated generation time (yrs)	6.11

Appendix 2. Results of the reintroduction scenario. Mean values ±SD of measures of viability are given for each initial population size (N_i) simulated. Time to extinction refers to the year of the first among those iterations that suffered extinctions, and population size and expected heterozygosity are calculated only from those iterations that persisted over the entire time frame.

	Probability of extinction (PE)		Mean time to extinction (years)		Stochastic population growth rate (r)		Mean population size (N)		Mean expected heterozygosity	
N_i	Juvenile	Adult	Juvenile	Adult	Juvenile	Adult	Juvenile	Adult	Juvenile	Adult
10	0.984	0.922	6.9 ± 8.52	18.8 ± 10.95	-0.1148 ± 0.57	0.000 ± 0.47	8.75 ± 3.42	10.55 ± 5.92	0.23 ± 0.22	0.28 ± 0.22
20	0.881	0.59	13.1 ± 11.66	28.8 ± 11.16	-0.0696 ± 0.5	-0.002 ± 0.36	21.19 ± 10.16	21.2 ± 11.48	0.43 ± 0.2	0.49 ± 0.17
30	0.774	0.365	16.6 ± 12.13	31.9 ± 10.74	-0.0584 ± 0.45	-0.002 ± 0.31	29.23 ± 16.65	30.09 ± 17.4	0.51 ± 0.19	0.63 ± 0.17
40	0.661	0.195	19.3 ± 12.11	35.6 ± 9.47	-0.0486 ± 0.42	0.002 ± 0.28	37.22 ± 23.81	39.61 ± 22.45	0.56 ± 0.19	0.69 ± 0.13
50	0.57	0.131	22.6 ± 12.4	35.9 ± 8.65	-0.0428 ± 0.4	0.003 ± 0.25	43.07 ± 28.19	50.12 ± 28.06	0.60 ± 0.18	0.74 ± 0.11
60	0.497	0.081	24.2 ± 11.45	39.5 ± 7.10	-0.0391 ± 0.37	0.007 ± 0.23	49.84 ± 33.17	63.29 ± 33.32	0.63 ± 0.19	0.78 ± 0.1
70	0.431	0.044	26 ± 11.54	40.1 ± 7.58	-0.0356 ± 0.36	0.008 ± 0.22	54.03 ± 39.99	74.18 ± 38.58	0.64 ± 0.18	0.81 ± 0.09

80	0.38	0.023	28.3 ± 12.23	39.7 ± 7.31	-0.0319 ± 0.34	0.009 ± 0.21	63.75 ± 46.5	86.63 ± 43.05	0.69 ± 0.16	0.84 ± 0.09
90	0.343	0.027	28.9 ± 11.03	41 ± 4.93	-0.031 ± 0.33	0.009 ± 0.2	66 ± 49.69	99.54 ± 47.86	0.69 ± 0.17	0.86 ± 0.06
100	0.295	0.016	29.9 ± 11.23	43.7 ± 6.25	-0.0296 ± 0.32	0.01 ± 0.19	71.85 ± 53.6	110.43 ± 53.63	0.7 ± 0.15	0.87 ± 0.06