Vincent Wildlife
Pine Marten Recovery Project

Feasibility Assessment for Reinforcing Pine Marten Numbers in England and Wales

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EXECUTIVE SUMMARY

Successful translocations of pine marten to recently and historically occupied, suitable habitat could be a major proactive step towards improving the conservation status and genetic diversity of pine marten in England and Wales. This report provides an initial assessment of the feasibility of undertaking translocations to reinforce existing populations that have failed to recover naturally.

The pine marten, *Martes martes*, was once widespread throughout Britain but during the 18th and 19th century pine martens declined dramatically due to habitat loss, compounded by increases in trapping and predator control associated with the rise in game shooting. By the beginning of the last century pine marten were only found in the far north west of Scotland, with small isolated pockets in some upland areas of northern England and Wales. Populations are now recovering well and expanding back into their former ranges in Scotland, but this is not the case elsewhere in the UK. Sightings reports and occasional unequivocal records suggest that some animals are still present in some parts of southern Britain but in such low numbers that population viability is highly vulnerable.

A landscape modelling approach was used to predict habitat suitability for pine marten across England and Wales at the 10km square resolution. The model identified a number of areas with high suitability values, these were in the north of England (Northumberland, the Lake District and in the Pennines), with some clustered in the West Country and around the Forest of Dean. There was a proportionately greater predicted area of high suitability in Wales, from Snowdonia in the north down the Cambrian Mountains through central Wales and in the Vale of Neath in the south.

Informed by the results of habitat modelling, and our database of reported pine marten sightings collected since 1996, six potential reinforcement regions (PRRs) were identified. These are areas of high predicted habitat suitability in regions where reports of recent sightings and other evidence suggest pine martens are still present in extremely low numbers. For each of these PRRs, initial desk-based analyses of variables that are likely to have an impact on establishment and spread were carried out. Roads are likely to be an important source of mortality affecting the viability of newly reinforced pine marten populations. The total length of roads in each PRR and the percentage of those within woodland, combined with annual volume of traffic were used to calculate the relative likelihood of marten mortality due to road traffic accidents. Even a low rate of additional mortality will increase extinction risk and jeopardise the establishment of a newly reinforced population. Therefore, translocations should first be to those PRRs identified here that have woodland blocks large enough to support relatively high marten numbers, and where likelihood of additional mortality is low.

The perception that the recovery of a native predator will have a negative effect on native prey species is a major concern for some stakeholders. A thorough risk assessment, evaluating the ecological roles of translocated animals in their new environment, and potential impacts on other species already present in release areas is an essential aspect of the feasibility study. The overlap between PRRs and the UK breeding distribution of rare bird species that might be predated by pine marten was assessed and the implications discussed, along with other wildlife that may be affected by an increase in pine marten numbers.

An integral part of the feasibility stage of any conservation translocation is gauging public attitudes towards the target species. A public opinion survey was carried out to assess public attitudes towards the potential re-stocking of the pine marten to parts of Wales. A total of 617 viable responses were achieved, of which 87.3% would support a restocking of pine marten in Wales. Respondents that worked in farming, game-keeping and estate management were most likely to oppose a re-stocking, while those that worked in leisure & tourism and wildlife conservation were most likely be supportive. The main reasons given in support of re-stocking pine martens were native status, a perceived positive effect on biodiversity, and its contribution to a balanced environment. The main reasons for opposition were predation of wildlife, prioritising the native population, and lack of suitable habitat/knowledge of the current population status.

Reinforcement of pine marten populations in England and Wales was assessed against criteria for conservation translocations published by the International Union for Conservation of Nature (IUCN). The work that has been carried out so far satisfies the criteria in annexes one to six. Detailed proposals for release and implementation will be produced and evaluated against criteria in Annex 7 of the IUCN guidelines during the second part of the feasibility study (to be completed in 2015).

Six potential reinforcement regions in Wales and England have been identified that contain sufficient areas with high predicted habitat suitability to support viable populations of pine martens. Field surveys will now be carried out to inform the final decision as to where the most optimal release sites will be. Detailed consultation with stakeholders, the general public, government and local agencies and other NGOs will be undertaken in potential reinforcement regions before proposals for pine marten translocations are finalised. The information gathered during the consultation process will inform the selection of release sites that have maximum local support.

CRYNODEB

Gallai trawsleoliad llwyddiannus o felaod coed i gynefinoedd addas lle roeddent unwaith yn bodoli – un ai'n hanesyddol neu'n fwy diweddar – fod yn gam sylweddol ymlaen o ran gwella statws cadwraethol ac amrywiaeth genetegol y rhywogaeth hon yn Lloegr a Chymru. Mae'r adroddiad hwn yn cynnig asesiad cychwynnol o ddichonoldeb ymgymryd â gwaith trawsleoli i atgyfnerthu poblogaethau sydd eisoes yn bodoli ac sydd wedi methu adfer yn naturiol.

Roedd bele'r coed, *Martes martes*, unwaith yn eang ei ddosbarthiad trwy Brydain. Gwelwyd cwymp anferth yn niferoedd y rhywogaeth hon yn ystod y 18fed a'r 19egG oherwydd diflaniad cynefinoedd, ynghyd â ffactorau eraill fel y gwaith trapio a rheoli ysglyfaethwyr a oedd yn gysylltiedig â'r cynnydd mewn saethu anifeiliaid helgig (gêm) ar yr adeg honno. Erbyn dechrau'r ganrif ddiwethaf roedd bele'r coed ond i'w ganfod ym mhellafoedd gogledd orllewin yr Alban, gyda phoblogaethau bychain ac ynysedig iawn mewn ambell ran o'r ucheldir yng ngogledd Lloegr a Chymru. Yn yr Alban mae poblogaethau bellach yn adfer yn llwyddiannus ac yn lledu nôl i'w hen ardaloedd dosbarthiad, ond nid yw hyn yn wir mewn rhannau eraill o Brydain. Mae arsylwadau ac ambell gofnod pendant achlysurol yn awgrymu bod ychydig o anifeiliaid yn dal i fod yn bresennol mewn rhai rhannau o dde Prydain, ond mewn niferoedd mor isel nes bod hyfywedd y poblogaethau yn fregus iawn.

Defnyddiwyd dull modelu tirwedd i ragfynegi addasrwydd cynefinoedd ar gyfer bele'r coed ar draws Cymru a Lloegr, gan ddefnyddio sgwariau 10km fel y raddfa ddadansoddi. Fe wnaeth y model adnabod nifer o ardaloedd a oedd yn addas iawn ar gyfer bele'r coed; roedd y rhain yng ngogledd Lloegr (Northumberland, Ardal y Llynnoedd a'r Pennines), gyda rhai ardaloedd wedi clystyru yn ne-orllewin Lloegr ac o gwmpas Fforest y Ddena. Yn gyfrannol, cafodd mwy o ardal addas iawn ei rhagfynegi yng Nghymru, o Eryri yn y gogledd i lawr drwy'r Mynyddoedd Cambriaidd ac yng nghwm Nedd yn y de.

Gyda'r wybodaeth o'r gwaith modelu cynefin, ynghyd â'r bas data sydd gennym o arsylwadau o felaod coed ers 1996 cafodd chwe rhanbarth atgyfnerthu posib (PRRs) eu hadnabod. Mae'r rhain yn ardaloedd lle cafodd lefel uchel o addasrwydd cynefin ei ragfynegi drwy'r gwaith modelu a lle mae adroddiadau o arsylwadau diweddar, ynghyd â thystiolaeth arall, yn awgrymu bod belaod coed yn dal i fod yn bresennol mewn niferoedd isel iawn. Ar gyfer bob un o'r PRRs hyn, gwnaed gwaith pen-desg cychwynnol i ddadansoddi newidion sy'n debygol o effeithio ar sefydlu a lledaeniad belaod coed. Mae ffyrdd yn debygol o fod yn achos marwolaethau a fyddai'n effeithio ar hyfywedd poblogaethau belaod coed sydd newydd gael eu hatgyfnerthu. Defnyddiwyd cyfanswm hyd ffyrdd ym mhob PRR a'r ganran o'r rheiny o fewn coedwigoedd, ynghyd â phwysau traffig blynyddol, i amcangyfrif pa mor debygol, yn gymharol, y byddai hi i farwolaethau belaod gael eu hachosi gan ddamweiniau gyda cherbydau ar ffyrdd. Byddai hyd yn oed cyfradd isel o farwolaethau ychwanegol yn cynyddu'r perygl o ddifodiant ac yn peryglu llwyddiant sefydlu poblogaeth sydd newydd gael ei hatgyfnerthu. Felly dylai trawsleoli ddigwydd yn gyntaf mewn PRRs sydd â blociau o goetir sy'n ddigon mawr i gynnal niferoedd cymharol uchel o felaod, heb fod yna debygolrwydd uchel o farwolaeth ychwanegol o achos ffyrdd.

Mae’r canfyddiad y byddai adferiad ym mhoblogaeth ysglyfaethwr brodorol yn cael effaith negyddol ar rywogaethau eraill sy’n ysglyfaeth i hwnnw yn bryder mawr i rai budd-ddeiliaid. Mae asesiad risg trwyadl, sy’n gwerthuso rôl ecolegol creaduriaid wedi eu trawsleoli yn eu hamgylchedd newydd, a’r effeithiau posib ar rywogaethau eraill sydd eisoes yn bresennol yn yr ardaloedd gollwng yn elfen allweddol o’r astudiaeth dichonoldeb. Aseswyd y gorgyffwrdd rhwng PRRs a dosbarthiad bridio’r adar prin hynny yn y DU a allai gael eu hysglyfaethu gan felaod coed a thrafodwyd y goblygiadau, ynghyd â bywyd gwyllt arall a allai gael eu heffeithio gan gynnydd mewn niferoedd belaod coed.

Elfen allweddol o’r cyfnod sefydlu dichonoldeb mewn unrhyw gynllun trawsleoli cadwraethol yw asesu agwedd y cyhoedd tuag at y rhywogaeth dan sylw. Cynhaliwyd arolwg o farn y cyhoedd er mwyn asesu agweddau’r cyhoedd tuag at y posibilrwydd o ail-stocio rhannau o Gymru gyda belaod coed. Cafwyd 617 o ymatebion y gellid eu defnyddio, ac o’r rhain roedd 87.3% yn cefnogi ail-stocio belaod coed yng Nghymru. Y rhai a oedd yn gweithio ym meysydd ffermio, cadw creaduriaid helgig a rheolaeth ystâd oedd yn fwyaf tebygol o wrthwynebu ail-stocio, tra bod y rhai a oedd yn gweithio mewn meysydd hamdden, twristiaeth a chadwraeth bywyd gwyllt yn fwy tebygol o fod yn gefnogol. Y prif resymau a roddwyd dros gefnogi ailstocio belaod coed oedd statws brodorol, canfyddiad y byddai effaith bositif ar fioamrywiaeth a’r cyfraniad y byddai’n gwneud at greu amgylchedd cytbwys. Y prif resymau dros wrthwynebu oedd hela bywyd gwyllt, angen blaenoriaethu’r boblogaeth gysefin a diffyg cynefin addas/ gwybodaeth am statws cyfredol y boblogaeth.

Aseswyd cynlluniau i atgyfnerthu poblogaethau belaod coed yn Lloegr a Chymru yn erbyn meini prawf ar gyfer trawsleoliadau cadwraeth a gyhoeddwyd gan yr International Union for Conservation of Nature (IUCN). Mae’r gwaith sydd wedi ei gynnal hyd yn hyn yn cwrdd â’r meini prawf yn atodiadau un i chwech. Bydd argymhellion manwl ar gyfer gollwng belaod a gweithredu cynlluniau yn cael eu cynhyrchu a’u gwerthuso yn erbyn meini prawf yn Atodiad 7 o ganllawiau IUCN yn ystod ail ran yr astudiaeth dichonoldeb (i’w gwblhau 2015).

Mae chwech o ranbarthau atgyfnerthu posib wedi cael eu hadnabod yng Nghymru a Lloegr sy’n cynnwys digon o ardaloedd gyda lefel uchel o gynefin addas i gynnal poblogaethau hyfyw o felaod coed. Bydd arolygon maes yn awr yn cael ei wneud er mwyn dylanwadu ar y penderfyniad terfynol ynglŷn â lleoliad y safleoedd gollwng gorau. Bydd gwaith ymgynghori manwl gyda budd-ddeiliaid, y cyhoedd, y llywodraeth a’i asiantaethau lleol, a mudiadau anllywodraethol eraill yn digwydd ym mhob rhanbarth atgyfnerthu posib cyn y bydd cynlluniau ar gyfer trawsleoli belaod coed yn cael eu cwblhau. Bydd y wybodaeth sy’n cael ei chasglu yn ystod y broses ymgynghori yn cael ei defnyddio i ddewis safleoedd gollwng sydd â chefnogaeth leol llwyr.

1. INTRODUCTION

Why consider pine marten translocations?

Pine martens, *Martes martes,* defend large territories, they have low reproductive rates and naturally occur at low densities (Balharry 1993b; Zalewski & Jędrzejewski 2006; Harris & Yalden 2008). As is the case for many carnivores, pine marten population declines in the UK resulted largely from conflicts with, and resulting persecution by, humans (Langley & Yalden 1977). This compounded the effects of habitat loss and fragmentation, with woodland cover at its lowest (less than 5%) by the beginning of the 20th century. The pine marten now has full legal protection in Britain and woodland cover has almost tripled over the past 100 years, so conditions have considerably improved for this species. Populations are now recovering well and expanding back into their former ranges in Scotland (Croose, Birks & Schofield 2013) and Ireland (O'Mahony, O'Reilly & Turner 2012), but large areas of Wales and southern England are relatively isolated from currently occupied range due to the distances involved and intervening areas of unsuitable habitat through which it is unlikely that martens would disperse. Therefore it is highly unlikely that pine marten populations in Scotland will expand beyond counties in the north of England.

The development of an increasingly holistic, ecosystem restoration based approach to biodiversity conservation has resulted in a growing interest in the ecological value of carnivores (Wilson 2004; Manning, Gordon & Ripple 2009; Taylor 2013; Ripple *et al*. 2014). Reintroduction and reinforcement through translocation is an increasingly common method employed to re-establish absent or endangered populations (Lewis, Powell & Zielinski 2012), and under international treaties such as the Bern Convention (1979) and the Rio Convention (1992), the UK is obliged to encourage the restoration of populations of its native species (Hetherington 2006).

Guidelines for the justification, design and implementation of conservation translocations have been published by the Reintroduction and Invasive Species Specialist Groups of the International Union for Conservation of Nature (IUCN) (IUCN 1995; IUCN 2013). These explicitly state that a thorough feasibility study should be conducted to determine if existing habitat, source populations, and the political and social environments are suitable for a successful translocation (IUCN 2013). The goal of this feasibility study is to determine if these criteria are met in England and Wales and, if they are, to identify potential release areas.

1.1. Objectives

The feasibility study will be completed by the end of 2015. The emphasis of the initial, desk-based study presented here is primarily on landscape scale habitat suitability and environmental factors that may affect establishment of viable pine marten populations. In other work, which will be undertaken from August 2014 to December 2015, the focus will be on the social feasibility of pine marten reinforcements in consultation with stakeholders and communities in potential release areas.

The objectives of the present study (phase I) are to:

- Determine if and where there is suitable habitat of sufficient size and connectivity to support viable, self-sustaining populations of pine martens in England and Wales
- Evaluate the risk of accidental mortality to pine martens in these areas
- Identify potential areas of conflict with economic interests and other species
- Shortlist a number of potential release areas for more detailed field surveys and fine scale habitat assessment
- Develop field survey protocols to determine if adequate prey and other food sources are present at these sites to support a pine marten population,
- Define methods for collecting baseline data at potential release sites to facilitate long term monitoring of potential impacts of increased pine marten numbers on other species present
- Make recommendations for further work to be carried out during phase II (August 2014 onwards)

A reinforcement will be considered *biologically* feasible in a potential release area if suitable foraging, denning and resting habitat exists in a forested landscape of sufficient quality and spatial configuration likely to support a self-sustaining pine marten population; and if an adequate number of animals can be taken from a genetically appropriate source population without affecting the viability of that source population. Social and economic factors will also be taken into account when prioritising potential release sites in order to maximise the likelihood of reinforcement being successful.

1.2. Background

The pine marten *(Martes martes)* is a medium-sized mustelid that occurs throughout mainland Europe, where it predominantly inhabits forested habitat (Mitchell-Jones *et al.* 1999). It is a generalist predator and in Britain its diet comprises small mammals (principally field voles *Microtus agrestis*), fruit and berries (principally bilberry *Vaccinium myrtillus* and rowan *Sorbus aucuparia*), small birds, invertebrates and carrion (Putman 2000; Paterson & Skipper 2008; Caryl *et al.* 2012). Pine martens preferentially den above ground in tree cavities, birds' nests and squirrel dreys, but will also den on the ground in cairns, burrows, tree roots and brash piles, and may den in both uninhabited and inhabited buildings (Balharry 1993a; Birks, Messenger & Halliwell 2005). Pine marten home range typically varies between <1 to 6km^2, although it can be as large as 32km^2 in upland habitat (see review in Caryl, Quine and Park 2012). Martens are solitary, with mating taking place in July and August and between one and five kits born the following spring.

The pine marten arrived in Britain post-glaciation and was historically widespread, with the species being the second commonest carnivore during the Mesolithic (Maroo & Yalden 2000). The population underwent a severe decline during the 18th and 19th centuries as a result of woodland clearance, intensive predator control to protect game birds and poultry and fur harvesting (Langley & Yalden 1977). By the early 20th century, the pine marten had become extinct in much of southern Britain and the population had become restricted to north-west Scotland and small pockets in northern England and Wales (Langley & Yalden 1977). From the mid-20th century onwards, the population in Scotland recovered and expanded its range south and eastwards, aided by a reduction in persecution pressure by legal protection afforded to the species under the Wildlife and Countryside Act (1981) and increased woodland cover through afforestation (Lockie 1964; Balharry *et al*. 1996; Croose, Birks & Schofield 2013; Croose *et al*. 2014). However, the situation in England and Wales is markedly different, as pine marten populations have failed to recover from their historical decline and the species remains very rare south of the Scottish border (Birks & Messenger 2010). Despite concerted detection efforts, only 16 unequivocal records of pine martens have been collected from England and Wales since 1990, along with many unverified sightings reported by naturalists, countryside professionals and members of the public (Birks & Messenger 2010; VWT, unpublished data). These records have been concentrated in the Lake District, the Cheviot Hills, the North York Moors and the Peak District in England, and in Snowdonia, Carmarthenshire and the Cambrian Mountains in Wales (Birks & Messenger 2010; VWT, unpublished data).

A recent study to compare the haplotype composition of historical and current populations in England, Scotland and Wales found no difference between the main haplotype of contemporary (post-1950) populations across the UK. There were historical differences between the haplotypes of museum specimens from Scotland and those from England and Wales (Jordan *et al*. 2012). This could suggest that the "relict" haplotype in England and Wales has been lost during the 20th century, either through genetic drift or by replacement with animals that have been released, escaped and/or translocated from elsewhere. However, it may also be an effect of sampling bias, as the very low number of samples available from extant populations in England and Wales, could result in non-detection of a less common haplotype.

There is now a general consensus that the pine marten population in England and Wales is so small as to be highly vulnerable to environmental, demographic and genetic factors which interact in a downward spiral or extinction vortex (Gilpin & Soulé 1986; Fagan & Holmes 2006); and natural recovery is unlikely without intervention (Jordan 2011). While it is probable that parts of northern England will be re-colonised by the pine marten population in Scotland spreading southwards (Croose *et al*. 2014), they are highly unlikely to expand their range to re-colonise central/southern England and Wales, due to the large conurbations in north-west and central England and a lack of suitable habitat in some of these areas. A strategy for restoring the pine marten to England and Wales was produced by The Vincent Wildlife Trust, in collaboration with a variety of stakeholders (Jordan 2011). This outlined actions and priorities for re-establishing pine marten populations in England and Wales and improving the conservation status of the species. It concluded that intervention via a reintroduction or reinforcement is required in order to restore viable self-sustaining populations of pine martens to their historical range in England and Wales.

Prior to a reintroduction or reinforcement of animals, a thorough feasibility study and background research will be completed in order to assess factors that could affect the success of the release, as detailed in this report.

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2. HABITAT ASSESSMENT: USING HABITAT MODELLING TO IDENTIFY PRIORITY AREAS IN ENGLAND AND WALES FOR PINE MARTEN TRANSLOCATIONS

2.1. Introduction

Reinforcement of existing populations of pine marten in England and Wales with releases of translocated animals is proposed if it is feasible to do so within current IUCN guidelines. Reviews of the outcomes of conservation translocations have often reported low rates of success (e.g. Fischer and Lindenmayer 2000; Wolf et al. 1996). One of the reasons frequently given for failure is low habitat suitability in the release area (Wolf *et al*. 1996; Armstrong *et al*. 2002; Cook, Morgan & Marshall 2010). Reintroductions of American martens, (*Martes americana*) have demonstrated that translocations into high quality habitats have a disproportionately high success rate (Slough 1994). The conclusion from this is that a large effort should be put into characterising suitable release areas before considering going ahead with translocations (Bright 2000). Release sites should not be chosen on the basis that the species used to be there or that the site looks right. It is suggested that a combination of detailed field knowledge and species distribution or habitat modelling should be used to assess a site's suitability for any proposed release (Osborne & Seddon 2012).

Understanding the distribution of suitable habitats for species is important, both in the conservation of those that are rare or declining and for the management of invasive non-native animals (Buckley & Lundy 2013). Species distribution models (SDMs) can provide useful information about current and future geographic distributions of species and can incorporate potential changes in climate and landscapes (Razgour, Hanmer & Jones 2011). Defining a model based on associations of species with landscape variables can be used to validate or quantify our understanding of the species' ecology (Starfield 1997; Gough & Rushton 2000). In addition to this, models can be used to target future survey effort by predicting distribution in un-surveyed areas, and to maximise the effectiveness of ecosystem restoration and species reintroduction programmes (Razgour, Hanmer & Jones 2011).

MaxEnt (Phillips, Anderson & Schapire 2006) uses a machine learning algorithm to characterise probability distributions from presence only data. It is widely used to model suitable habitat for various species (Phillips, Anderson & Schapire 2006; Gibson, Barrett & Burbidge 2007; Ward 2007; Hernandez *et al*. 2008; Stabach, Nadine & Olupot 2009) and has been shown to perform better than other presence only and presence-absence modelling techniques (Elith *et al*. 2006; Hernandez *et al*. 2006). MaxEnt requires relatively few (>30) presence locations to construct useful models, which is advantageous for rare or difficult to detect species when the amount of occurrence data is often limited (Wisz *et al*. 2008).

The aim of this study was to identify and prioritise search areas for potentially suitable sites for pine marten translocations in England and Wales by using the MaxEnt niche-based modelling technique combined with GIS analyses.

2.2. Methods

Habitat suitability was modelled at the 10km square resolution for the whole of the UK and Ireland. Presence locations were grid references where pine marten scats, confirmed by DNA testing, had been collected during recent surveys between 2005-2007 in Ireland, and 2012-2013 in Scotland (for details see O'Mahony, O'Reilly and Turner 2012 and Croose, Birks and Schofield 2013). Both surveys were based on 1-3 transects of 1-1.5km in length within 10km national grid squares. Experienced surveyors walked transects located along forest tracks or paths in wooded habitat searching for pine marten scats. Each transect was walked only once between May and September.

Land cover classes for Britain and Ireland were derived from the CORINE 2006 datasets (European Environment Agency 2007) and manipulated in ArcMap 10.2. Of the 44 pan-European land cover classes, 8 were not present within the UK or Ireland and the remaining 36 were reclassified into 16 land cover variables that were deemed biologically relevant for pine martens. The cumulative area of each re-classified land cover type was then summed onto a 10 x 10 km square grid. Two land cover types (bare ground and marsh) that were present in less than 20% of grid squares were excluded from the modelling, as it has been shown that including rare cover classes can increase the level of background noise in environmental data, leading to predictions that are biased towards local conditions. This can make it less acceptable to extrapolate model predictions to other areas (McCune, Grace & Urban 2002). Mean altitude per 10km square was derived from DIVA-GIS datasets (http://www.diva-gis.org/Data) for the UK and Ireland.

MaxEnt 3.3.3 (Phillips & Dudík 2008) software was used for model fitting. The random seed setting was chosen; this randomly selects 75% of presence locations for model training and uses the remaining 25% to test the model predictions each time the model is run. We used the 10th percentile as the threshold value for defining suitable habitats. This is the value above which the model correctly classifies 90% of the training locations, and is commonly used in species distribution modelling (Raes *et al*. 2009; Rebelo & Jones 2010; Razgour, Hanmer & Jones 2011). All other settings were left at default. The model was run for 20 replications; with model fit assessed using Receiver Operator Characteristic (ROC) curve analysis and the average Area Under the Curve (AUC) values across all of the replicated runs. AUC values greater than 0.9 are classed as very good, with 0.7-0.9 being good and an AUC of less than 0.7 classed as uninformative (Swets 1988).

Model performance was also evaluated using Cohen’s kappa values (Cohen 1960) and the True Skill Statistic (TSS) (Allouche, Tsoar & Kadmon 2006). The significance of each environmental variable in explaining the variance in the presence location data was evaluated from Jacknife plots. The software ENMTools (version 1.4.4) was used to test for correlation between variables.

2.3. Results

Pine marten presence was confirmed by DNA testing at 105 locations from across a surveyed area of 220 10km squares in Scotland, and at 136 locations out of the 228 10km squares surveyed throughout the Republic of Ireland. This gave a total of 241 presence locations used in the model.

When the model was run with mean altitude included as a predictor variable the discrimination ability was acceptable (mean AUC_{train}=0.919245, s.d.=0.004 ; mean AUC_{test}=0.859765, s.d.= 0.014533). However, this was improved by the removal of mean altitude. The final model used for predicting suitability of un-surveyed areas had good performance scores by all three evaluation criteria (mean AUC_{train}=0.950925, s.d.=0.002554; mean AUC_{test}=0.910035, s.d.= 0.013068; Kappa=0.53; TSS=0.6).

No two variables used in the final model were found to be highly correlated (all $R^2<0.7$). Although there were 14 potential predictor variables included in the final model, almost 70% of the contribution to model performance was from wooded land classes: transitional woodland scrub, broadleaf, conifer and mixed woodland. The largest single contribution (58.05%) was from transitional woodland scrub (Table 2.1).

Table 2.1 *Description of environmental variables and their contribution to the model*

Variable	Description	Contribution %
Arable	Summed area (ha) of 10km square with arable and crop cover	6.09
Pasture	Area (ha) of 10km square with pasture	10.09
Agri-mosaic	Area (ha) of 10km square with heterogeneous agricultural land	1.22
Broadleaf	Area (ha) of 10km square with broadleaf woodland	7.21
Conifer	Area (ha) of 10km square with conifer woodland	1.50
Mixed wood	Area (ha) of 10km square with mixed woodland	1.25
Natural grassland	Area (ha) of 10km square with natural grassland cover	3.30
Moor and heath	Area (ha) of 10km square with moors and heathland	2.65
Scrub	Area (ha) of 10km square with transitional woodland scrub cover	58.05
Bog	Area (ha) of 10km square with peat bog	0.67
Freshwater	Summed area (ha) of 10km square with inland water bodies and water courses	1.98
Coastal	Summed area (ha) of 10km square with all categories of coastal habitat	4.47
Urban	Summed area (ha) of 10km square with urban land cover	0.46
Urban green	Summed area (ha) of 10km square with urban green space and amenity land	1.06

Jacknife tests were run on training gain, test gain and AUC. Omitting each variable in turn did not result in a considerable decrease in training gain; therefore no single variable alone contained a significant amount of information that was not already contained in the other variables. However, the plot for AUC showed that woodland scrub was the most effective single variable for predicting the distribution of the occurrence data that was set aside for testing. Figure 2.1 shows the response curves from a single randomly selected MaxEnt replicate illustrating the contribution of each variable to prediction of pine marten occurrence. The value on the *y* axis is the predicted probability of suitable conditions (as given by the logistic output) with all other variables set to their average value over the set of presence locations.

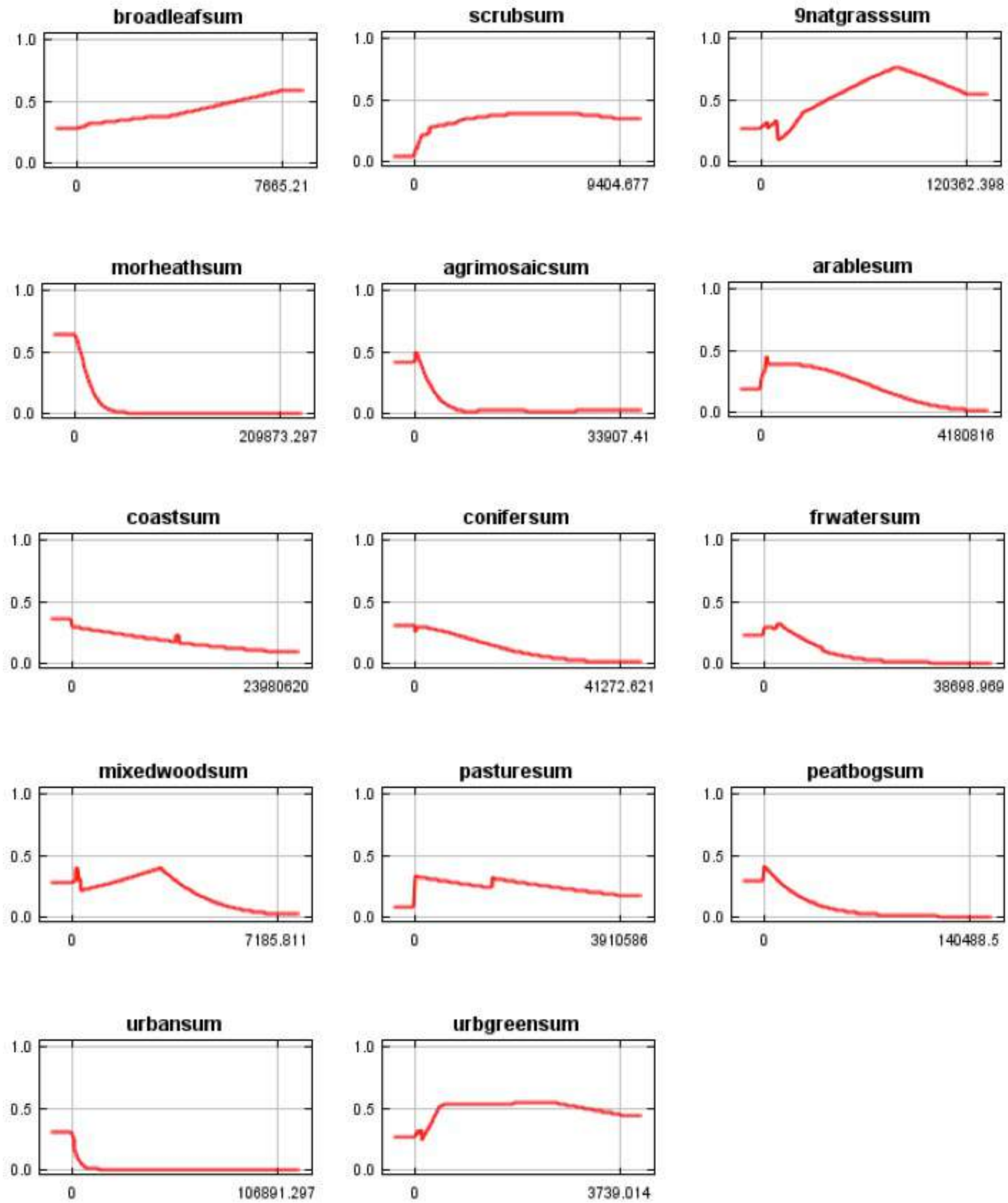

Figure 2.1 *Response curves from a single, randomly selected MaxEnt replicate, showing the contribution of each variable to prediction of pine marten occurrence*

This shows that the response is high for the smallest values of moor and heath and quickly drops to zero. There is also a negative association with increasing area of peat bog and urban land cover. Pine marten occurrence was positively associated with area of broadleaf woodland, woodland scrub and natural grassland.

Model predictions were plotted onto a 10km square grid map of the UK (figure 2.2) with values for suitability (model predicted probability of presence) ranging from zero to one. Average values were calculated for each region of the UK and Ireland. The highest mean values were for the Republic of Ireland (0.37) followed by Scotland (0.15), Wales (0.10), Northern Ireland (0.10) and England (0.03). The map highlights a number of areas within England and Wales with high suitability for pine martens.

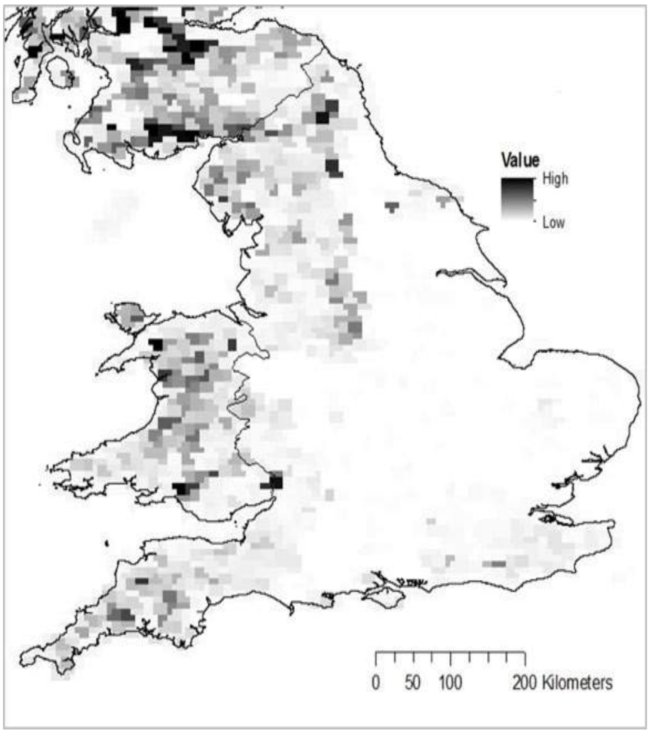

Figure 2.2 *Habitat suitability map for the pine marten in Britain based on MaxEnt model predictions. Darker shades of grey represent higher predicted habitat suitability*

In England, the 10km squares with the highest suitability values (dark grey to black in figure 2.2) are mainly in the north (Northumberland, the Lake District and in the Pennines), with some clustered in the west country. There is a proportionately greater predicted area of high suitability in Wales from Snowdonia in the north down the Cambrian Mountains through central Wales to the Vale of Neath in the south. The area around the Forest of Dean on the Welsh-English border is also highlighted. These areas will now be prioritised for further, more detailed analyses and field surveys.

2.4. Discussion

The aim of this study was to identify and prioritise areas at a landscape scale within which to focus the search for suitable pine marten release sites by using a niche-based modelling technique combined with GIS analyses. The model performed well, suggesting that the method is suitable for screening at this scale to identify optimal search areas for release sites. The advantage of the method is that it requires only presence data to develop the model. True absence data for elusive species with low detection probabilities such as the pine marten are impossible to verify. Up until very recently it was not possible to confirm presence of pine martens from field signs conclusively, as scats are morphologically very similar to other mustelid species as well as fox, *Vulpes vulpes,* and even surveyors with a high level of experience cannot always distinguish them (Davison *et al*. 2002). However, with advances in genetic techniques for confirming species identification from field signs such as hair and scats (O'Reilly *et al*. 2008; Ruiz-González *et al*. 2008), we can be confident that there are no false positives in our presence dataset. MaxEnt has also been shown to be more robust than other methods with respect to the number of presence locations needed (Elith *et al*. 2006; Hernandez *et al*. 2006; Phillips, Anderson & Schapire 2006), with as few as five locations required to develop a useful model in some cases (Hernandez *et al*. 2006; Pearson *et al*. 2007). One disadvantage of having few presence locations is that it can result in a model which is strongly influenced by biases in sampling effort. In this case the model output should be interpreted as showing areas which have similar environmental conditions to those where the species is known to occur, rather than defining probabilities of occurrence across a wider range, and perhaps used to target further survey effort (Wisz *et al*. 2008). Species distribution models are calibrated on a species' realised niche, relying on the assumption that location data used in the model are representative of its true requirements and that appropriate predictor variables have been used in the model. These are then used to predict areas that meet the requirements of the species' ecological niche and therefore its potential distribution. The assumption that location data are representative of a species' fundamental niche can be a problem if using data for a declining species. Some regions of its potential distribution may not be inhabited because it is being excluded by a competitor (e.g. red squirrel in the presence of grey squirrel), because it cannot disperse into an area because of geographical barriers to dispersal or because the species has been extirpated from an area for some reason.

Another factor to consider is source-sink dynamics which may result in a species being recorded as present in unsuitable (sink) habitats that do not provide the environmental conditions needed to support a viable population but which may be visited or frequently re-colonised by individuals from a nearby source habitat that does support a viable population. In this case, presence locations do not represent suitable habitat or the species fundamental niche (Pulliam 2000). This can also be an issue if remnant populations have been forced into refuges of sub-optimal habitat, in which case current ranges can be an unreliable indicator of habitat requirements. This may have been true during the pine marten's distributional nadir in Britain at the beginning of the 20th century when, it has been suggested, pine martens were selecting areas for absence of persecution rather than presence of particular habitat features; although this has been debated (Birks & Messenger 2010). It is certainly the case that where very few occurrence records are available these are unlikely to provide a sufficient sample to identify the range of environmental conditions occupied by the species. The number of presence locations used for the present study was sufficiently large (*n*=241) and collected from expanding populations across a wide enough geographical area and range of environmental conditions to support confidence in the predictions of habitat suitability from the final model. This is supported by the values for AUC, kappa and TSS which are all in the range of good to excellent.

Model extrapolation should be treated with caution when making predictions for areas with environmental values that are beyond the range of the data used to construct the model. Britain and Ireland are both islands off the north western coast of continental Europe with the same temperate maritime climate and a similar range of environmental conditions and land uses. So the areas within which presence data used for building the model were collected are similar enough to the area over which the model was extrapolated in this case to permit confidence in the model predictions.

The increase in model performance observed when altitude was not included as a predictor variable is consistent with that found across a number of studies using SDMs for mammal species (Hof, Jansson & Nilsson 2012). Pine marten distribution does not appear to be constrained by elevation within the range found in our study area, occurring in Europe up to 2000m above sea level (Balestrieri *et al*. 2010), however, it was thought that altitude provided a useful surrogate for other variables such as prey density and productivity which is why it was originally included.

The strong association with wooded land classes found in the model concurs with other studies (Buskirk & Powell 1994; Bissonette *et al*. 1997; Zalewski & Jędrzejewski 2006; Caryl 2008). The fact that survey transects were all located within wooded habitats has been accounted for in the way that environmental variables were used in the model. Summing the area of each land class within 10km grid squares should provide sufficient variation at both presence and background locations for the model to discriminate. In effect the model is not analysing the point locations at which pine martens were found, which is affected by where the surveyors looked, but at the characteristics of the 10km squares in which pine martens were detected. At this scale there is little effect of survey bias. At a smaller scale this would not have been the case. The high percentage contribution to the model by transitional woodland scrub is interesting. Most studies of pine marten habitat associations in Europe have been in the northern latitudes of its range where the species is associated primarily with well-structured, mature coniferous forests (Pulliainen 1981; Brainerd 1990; Storch, Lindstrom & De Jounge 1990; Brainerd *et al*. 1994; Brainerd & Rolstad 2002) but elsewhere in Europe it shows much more ecological adaptability, persisting in scrubland, coppices and patchily wooded areas (De Marinis & Massetti 1993; Pereboom *et al*. 2008; Manzo *et al*. 2012). A recent study by Caryl, Quine and Park (2012) reported that, while mature forest was the most preferred habitat, scrub and tussock grassland were also consistently selected by martens. It is suggested that this is probably because these habitats provide resources such as den sites and high densities of *Microtus* voles that are not widely available in commercially managed plantation forestry.

Nevertheless pine marten have been shown to strongly avoid open habitats (Storch, Lindstrom & De Jounge 1990; Brainerd & Rolstad 2002; Caryl 2008; Caryl, Quine & Park 2012), which is consistent with the negative association with moor and heath in the model presented here. This has implications for dispersal and shows that models of habitat suitability can also highlight how connected or isolated potentially suitable areas are, and provide an understanding of how landscape might influence the spread of species (Ovaskainen 2004; La Morgia *et al*. 2011). This will impact on the viability of increased pine marten populations as numbers increase and release sites approach carrying capacity.

The most recent guidelines for reintroductions and other conservation translocation from the IUCN (IUCN 2013) stress that matching habitat suitability and availability to the needs of candidate species is central to feasibility and design. They also state that habitat suitability should include assurance that the release of animals and their subsequent movements are compatible with permitted land uses in the affected area. Existing and adjacent land uses (such as agriculture and game shooting), fine scale habitat-structure and the attitudes of local communities will be a further indication of the suitability of these areas for releasing translocated animals. Suitable habitat will now be determined in areas highlighted by the present study as having high biological suitability, and evaluated for the level of potential conflicts with existing land uses, anthropogenic threats such as high risk of mortality from traffic and other potential causes.

Several priority areas have been identified in southern Britain (dark grey to black on the map in figure 2.2). These are located along the Cambrian mountains in mid Wales, Afan forest in south Wales, and the forest of Dean in Gloucestershire. Further analyses and field surveys of these priority areas are needed next to validate their suitability for pine marten re-stocking purposes.

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3. POTENTIAL PINE MARTEN REINFORCEMENT REGIONS IN ENGLAND AND WALES

3.1. Introduction

Historically, habitat fragmentation and loss coupled with predator control have caused population decreases and local extinctions of many *Martes* species worldwide. Legal protection has resulted in recovery of some populations, but this is not the case everywhere. In order to restore natural ecosystems of which these predators are recognised as important components, there have been a number of reintroductions or reinforcement of martens, sables and fishers, however, not all of these have succeeded. Powell *et al* (2012) found that the total number of animals released and the number of release sites were the two variables most associated with success. Reinforcement is defined as translocation into an existing population of conspecifics and the risks of unintended and undesirable impacts are likely to be lower in population reinforcements than in reintroductions (IUCN 2013).

Reinforcements of populations of European pine marten, American marten and fisher, *Martes pennanti*, have generally succeeded (Powell *et al*. 2012). With population reinforcement, as with reintroductions, there are two key phases: establishment and spread (Armstrong & Seddon 2008), with different limiting factors operating at each phase. During the initial phase, when the population is still very small, levels of mortality (and dispersal) will have most impact on establishment. Beyond the establishment phase factors, such as food availability, that affect breeding success will have the most influence on how the population increases and expands its distribution (spread). Therefore, optimal reinforcement regions will have low risk of mortality and high habitat suitability with sufficient prey.

The habitat modelling study described in the previous chapter suggested that there are a number of areas in England and Wales with high predicted habitat suitability for pine marten. Informed by these results, and The Vincent Wildlife Trust's database of reported pine marten sightings collected since 1996, we identified six potential reinforcement regions (PRRs), shown in figure 3.1. These are areas of high predicted habitat suitability in regions where reports of recent sightings and other evidence suggest pine martens are still present in extremely low numbers. For each of these PRRs we carried out initial desk-based analyses of variables that are likely to have an impact on establishment and spread.

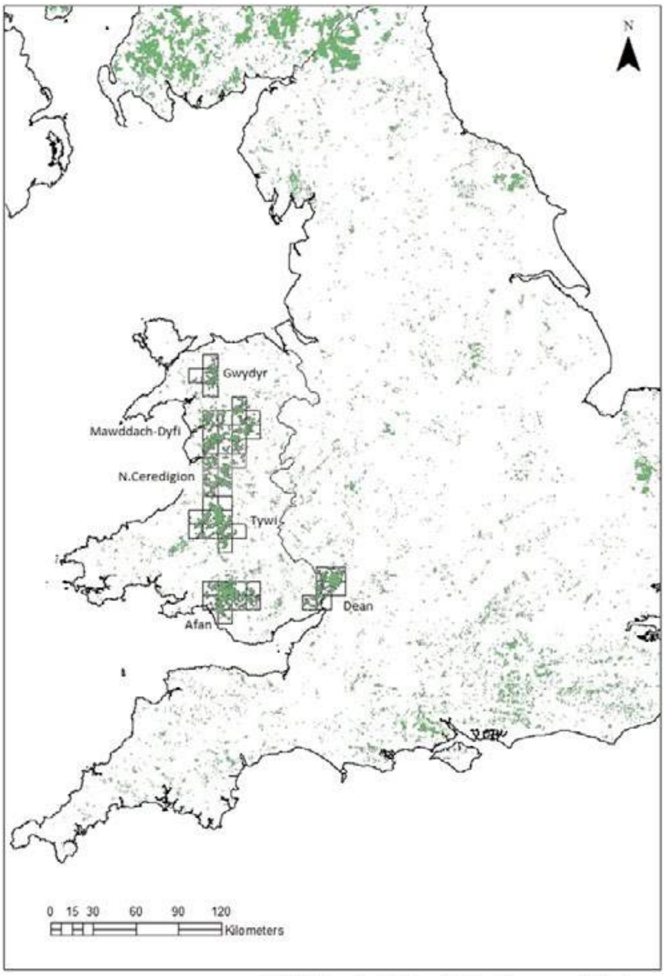

Figure 3.1 *Map of the UK showing the location of the six potential reinforcement regions (PRRs) evaluated here*

Pine martens are a species traditionally associated with mature deciduous and coniferous forests (Pulliainen 1981; Storch 1988; Storch, Lindstrom & De Jounge 1990). Recent studies suggest that they will utilise less forested areas (De Marinis & Massetti 1993; Pereboom *et al*. 2008; Balestrieri *et al*. 2010), and young forest stands (Brainerd *et al*. 1994), but forest cover is still necessary and pine martens strongly avoid open habitats (Storch, Lindstrom & De Jounge 1990; Brainerd & Rolstad 2002; Caryl, Quine & Park 2012). For a male marten, the minimum home range reported in the literature is 200ha (Zalewski & Jędrzejewski 2006; Balestrieri *et al*. 2010), so the size and connectivity of woodland will affect the number of animals that can be released into an area.

Bright and Harris (1994) formulated a model of pine marten population viability which predicted that even very low rates of additional mortality will result in a large reduction in population growth rate and increase in extinction probability. Extinction risk was greatly reduced when the number of animals released was increased above 30. This concurs with a study of American marten reintroductions which found that they succeeded in all cases where at least 30 animals had been released (Slough 1994). In a study of reintroductions, reinforcements and introductions of *Martes* species, the total number of animals released was also one of the variables associated with success along with the number of release sites (Powell *et al*. 2012). We therefore suggest that potential reinforcement regions should have sufficient woodland cover for 30-40 marten home ranges in total (minimum of 6000-8000 hectares of woodland), but that these could be across a number of well-connected woodland blocks or release sites within the region.

Pine marten are vulnerable to mortality as a result of road traffic accidents (Velander 1983), and animals exploring an unfamiliar landscape following release while establishing new home ranges are likely to be more so. Roads with high traffic volumes and those which pass through woodland probably represent the highest risk (Aaris-Sorensen 1995; Grilo, Bissonette & Santos-Reis 2008). So these factors should be quantified and taken into account when prioritising release sites.

Roads are not the only potential cause of mortality. Predation from foxes may also be important (Lindström *et al*. 1995), and the relative risks between PRRs to pine martens from intra-guild competition and predation by foxes will be evaluated during phase II following field surveys. Another risk is from accidental trapping or poisoning as a result of lawful pest control activities. Traditionally predator control and rodenticides have been used to protect reared game, particularly around pheasant pens, which may attract martens. In order to minimise the risk of this, the locations of commercial shoots were overlaid onto the map of potential release regions to identify where potential conflicts may arise.

The pine marten is a generalist predator, with a highly omnivorous diet. Small mammals are a major food item, but berries and fruit are also an important component of the diet, particularly in autumn (Putman 2000; Zalewski 2005; Lynch & McCann 2007; Rosellini, Barja & Pineiro 2008; Lanszki & Heltai 2011; Caryl *et al*. 2012). In Britain the pine marten has been shown to exhibit a preference for field voles (*Microtus agrestis*) (Caryl *et al*. 2012), so open areas within woodland and adjacent patches of rough grassland are an important resource as this is the preferred habitat of field voles (Hansson 1983; Petty 1999; Bogdziewicz & Zwolak 2014). A relatively low proportion of forest canopy cover is likely to be of benefit to pine martens because more light promotes a well-developed field layer with associated fruiting shrub species and a high density and diversity of small mammals.

The purpose of this chapter was to use available GIS data to compare some potential indicators of food availability and risks of mortality from road traffic accidents between potential release regions.

3.2. Methods

Potential Reinforcement Regions were defined as adjacent 10km-squares with at least 500 hectares of contiguous or non-contiguous woodland, that were also predicted from the MaxEnt model as being suitable or highly suitable habitat for pine martens (values between 0.5-1.0). Within each PRR, GIS data from the 2011 National Forest Inventory (NFI) (http://www.forestry.gov.uk/datadownload) were used to calculate the total area of woodland and the area within woodlands of each interpreted forest type (IFT). The National Forest Inventory woodland map is based upon Ordnance Survey 25cm resolution orthorectified colour aerial photography flown between 2002 and 2009 for England; and 40cm per pixel orthorectified digital imagery flown in 2006 for Wales. The NFI covers all forests and woodlands over 0.5 hectare in area and 20m wide, with a minimum of 20% canopy cover (or the potential to achieve it), including new planting, clear-felled sites and restocked sites. Field surveys are also carried out in 15000 randomly selected one hectare square sample plots, covering forests and woodlands in England, Scotland and Wales. The current NFI began in 2009 and will be completed in 2014.

Edge density (in m/ha) was calculated for all woodlands as a measurement of the complexity of the shapes of woodlands in each PRR. CORINE 2006 land cover data (EEA 2013) were used to sum the area of rough grassland within all 10-km squares at each PRR. The total length of all roads in PRRs were determined from Ordnance Survey Meridian 2 vector data (https://www.ordnancesurvey.co.uk/opendatadownload), which is supplied to a resolution of one metre (Ordnance survey, 2007). All GIS analyses were done in ArcMap 10.

Traffic flow was derived from 2012 annual volume of traffic data, which was the most recent data available, from the Department for Transport (http://www.dft.gov.uk/traffic-counts/download.php). This is defined as the annual traffic (also called volume of traffic) on each link of the major road network. Annual Average Daily Flow (AADF) is the number of vehicles estimated to pass a given Count Point (CP) on the road in a 24 hour period on an average day in the year. AADF figures are available for each junction to junction link on the major road network for every year. These are converted into traffic volume using equations 1 and 2.

$$(1)\ \ \text{Traffic}_{CP} = \text{AADF}_{CP} \times \text{Length}_{link} \times 365$$

$$(2)\ \ \text{Total traffic} = \Sigma\text{Traffic}_{CP}$$

(For further details on methods see https://www.gov.uk/government/uploads/system/uploads/attachment_data/file/230528/annual-methodology-note.pdf)

These figures were summed for all CPs within each PRR to obtain the total annual volume of traffic per PRR. The proportion of roads in each region that intersect woodland was used as a multiplier for the annual volume of traffic per kilometre of road to provide a relative index of traffic mortality risk for pine marten within each PRR.

An online search was carried out for commercial shoots, farms and syndicates that sold at least one day's shooting (all quarry) per year in Wales, Gloucestershire, Herefordshire and Worcestershire. The location of each shoot was mapped using ArcMap 10 and the distances measured from woodland edge within each PRR to the nearest shoot.

3.3. Results

The size of each PRR along with the area in hectares of each interpreted forest types within it are shown in table 3.1.

Table 3.1 *Woodland characteristics and amount of each interpreted forest type within each PRR.*

Characteristics	Dean	Gwydyr	Mawddach-Dyfi	N Ceredigion	Tywi	Afan
No. of contiguous 10km squares with >500ha woodland cover	6	4	12	7	10	8
Mean % woodland cover/10-km square (s.e.)	39(5.7)	28(11.2)	42(4.4)	36(5.2)	37(6.1)	56(7.7)
No. of contiguous 1km squares with ≥25% woodland cover	340	150	627	357	429	556
Mean Edge Density (m/ha)	7.17	8.43	8.19	6.45	7.69	6.44
NFI category (ha)						
Broadleaved	9,800	1,824	4,764	1,928	3,382	7,678
Conifer	6,953	4,911	20,793	9,402	14,968	16,993
Mixed- broadleaved	354	255	458	115	246	346
Mixed-conifer	529	163	428	160	160	304
Assumed woodland	115	114	405	227	156	411
Open area	243	366	719	262	721	927
Felled	350	480	3,585	3,360	3,060	2,728
Ground prepared for planting	99	7	380	188	840	681
Low density	8	18	38	116	8	126
Shrub land	50	28	63	10	39	39
Young trees	904	785	2,898	2,034	2,829	2,636
TOTAL all NFI woodland >0.5ha	19,420	8,952	34,532	17,803	26,408	32,869
Natural grassland in all PRR 10-km squares (ha)	0	9,571	36,754	29,282	95	15,071

The Forest of Dean has the highest proportion (50%) of broadleaved woodland of all the PRRS, with the others being predominantly coniferous. Mawddach-Dyfi is the largest of the potential reinforcement regions with woodland totalling 345 km^2. It is also the region containing the greatest area of rough grassland. How beneficial this would be for pine martens largely depends on its proximity to woodland edge which could not be quantified at this stage.

Field vole densities have been shown to increase for up to 15 years after clear cutting, as have wood mice, *Apodemus sylvaticus*, and shrews, *Sorex* spp. (Bogdziewicz & Zwolak 2014). For this reason all the interpreted forest types that are likely to support the highest numbers of small mammals (open, felled, ground prepared for planting, shrub land and young trees) were grouped together to compare the percentages of these habitats between PRRs. The results are shown in figure 3.2.

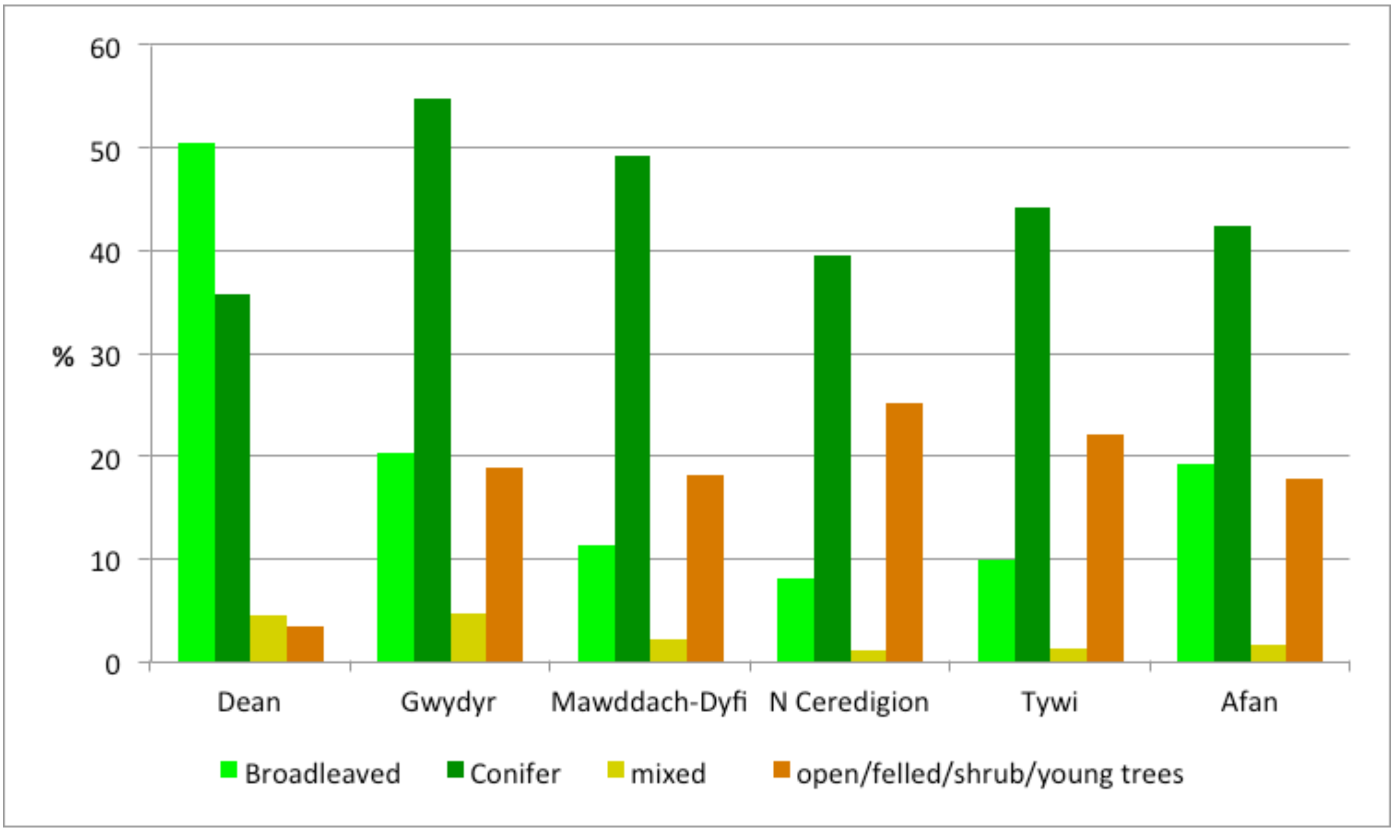

Figure 3.2 *The percentage of broadleaved, conifer, mixed and open IFTs within each PRR*

North Ceredigion and Tywi have the highest total percentages of open, felled, shrub and young trees, which are potentially good habitat for small mammals. However, these regions have less than 15% broadleaved and mixed woodland, so field surveys will need to determine if they have the quantity and diversity of fruiting trees that would be optimal for pine martens.

Risk of mortality from roads

Afan and Dean are the two PRRs containing the most length of roads of all types, including motorways. They also have the highest annual volume of traffic per kilometre of road. 20% of the roads in Dean intersect woodland which, combined with the high traffic volume, results in Dean having the highest index of road mortality risk for pine martens (see table 3.2).

Table 3.2 *Road lengths, traffic flow and index of road mortality risk for each PRR*

	Dean	Gwydyr	Mawddach-Dyfi	N Ceredigion	Tywi	Afan
Road lengths (km)						
Motorways	9.3	-	-	-	-	27.5
A roads	126	82	148	60	48	334
B roads	117	-	123	43	51	362
Minor roads	926	134	606	356	627	1,325
Km road/10-km square	196	54	73	66	63	229
2012 Traffic volume (10^3 vehicle kilometres)	507,619	47,477	187,643	40,148	36,987	2,023,437
% of roads within woodland	20	23	16	28	18	8
Index of road mortality risk	753	133	203	185	140	430

Gwydyr has the lowest index of road mortality risk, but this is also the smallest of all the PRRs which needs to be taken into consideration when prioritising reinforcement regions. There were 72 shoots identified within Wales and the southern border counties. The distances from each PRR to the nearest commercial shoot are shown in table 3.3.

Table 3.3 *Proximity of each PRR to commercial shoots*

	No. of commercial shoots within PRR	Distance from PRR (woodland edge) to nearest commercial shoot (km)
Gwydyr	1	0
Mawddach-Dyfi	7	0
N Ceredigion	0	1.5
Tywi	1	0
Afan	0	15
Dean	0	6.3

Individual maps of each potential reinforcement region are shown in figures 3.3 to 3.8 with further details of each PRR.

Afan PRR

Afan, shown in figure 3.3, has the lowest edge density; it is in a very populated part of Wales and has high recreational use including mountain bike trails, four wheel drive vehicle safaris and a visitor centre. Road density and traffic flow are the highest of all the PRRs, and, although few of the roads intersect woodland, it has the second highest index of road mortality.

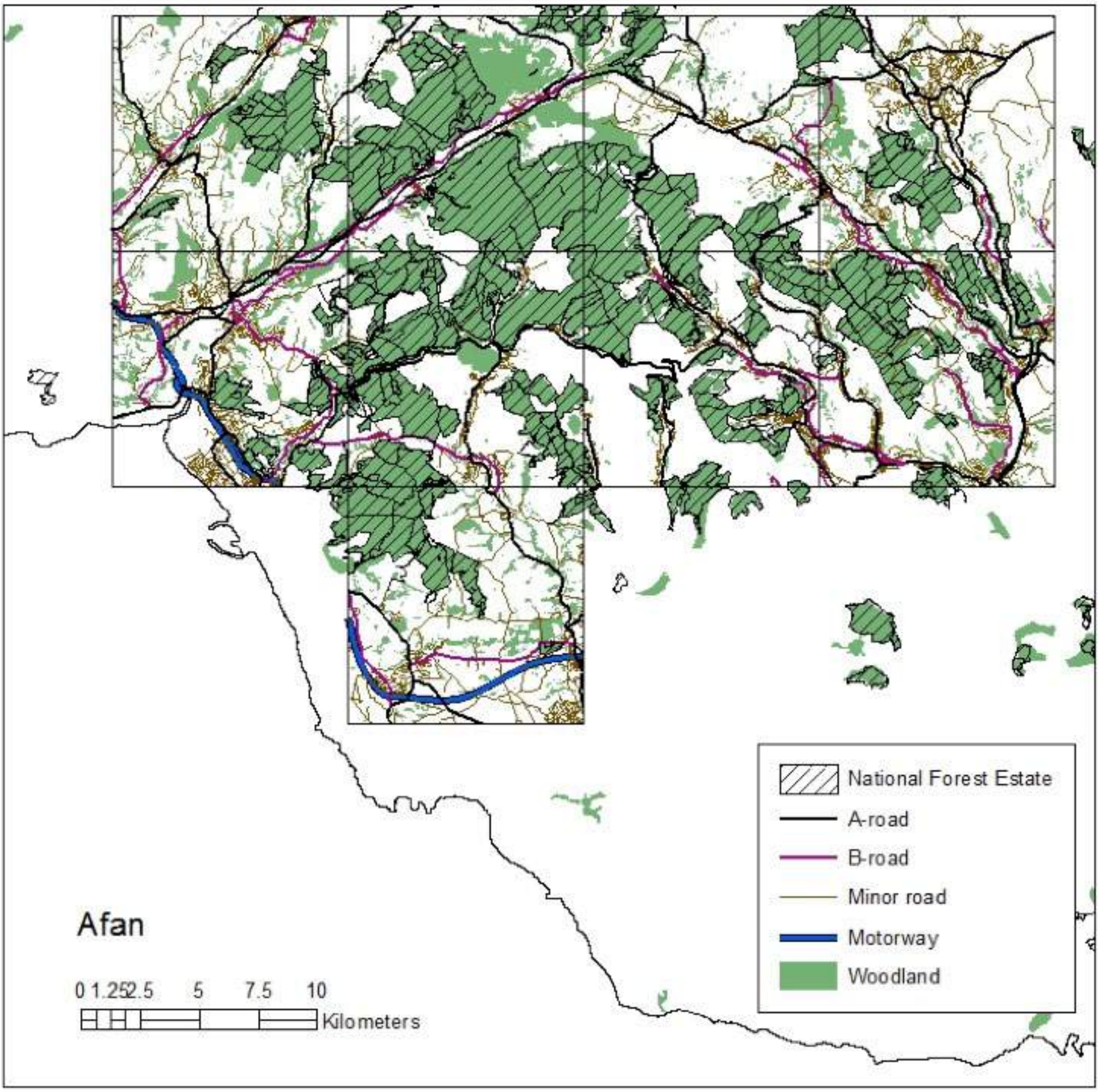

Figure 3.3 *Map of Afan PRR showing all motorway, A, B and minor class roads in relation to national forest estate and privately owned woodlands.*

Dean PRR

Dean is the region with the highest index of road mortality risk for pine martens. There is a high volume of traffic per kilometre of road in the region, and a fifth of all roads pass through woodland (see figure 3.4). Evidence from other mustelid species suggests that woodland roads are those where animals are at the highest risk of road traffic accidents (Aaris-Sorensen 1995). The high percentage of broadleaved woodland and the total area of woodland may make this region potentially very good habitat for pine martens. However, it has been previously suggested that translocations should first be to PRRs with the least risk of additional mortality, even if these have lower predicted density or suitability for spread (Bright & Smithson 2001).

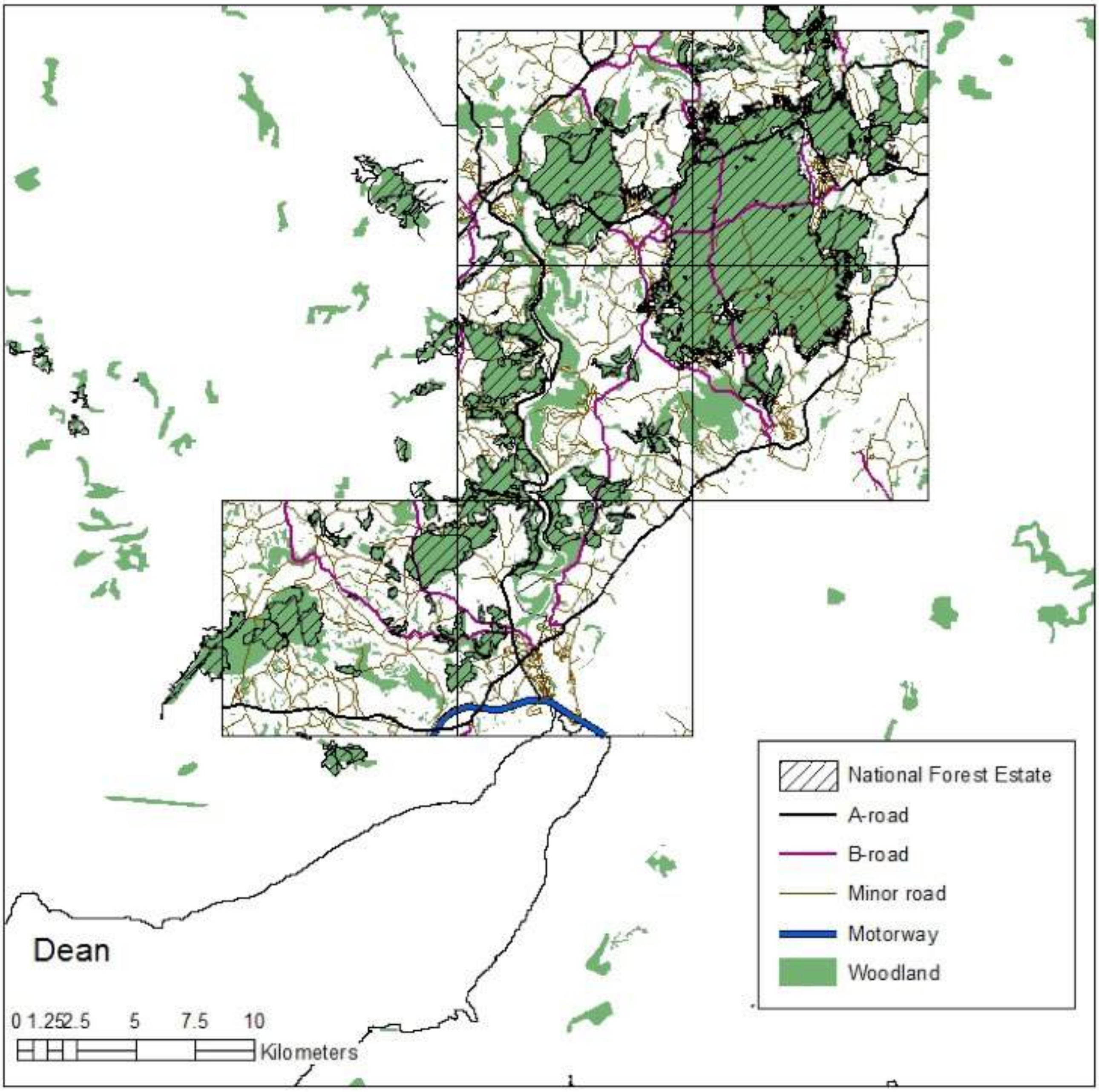

Figure 3.4 *Map of Dean PRR showing all motorway, A, B and minor class roads in relation to national forest estate and privately owned woodlands.*

Gwydyr PRR

Gwydyr (figure 3.5) has the lowest risk of additional mortality from roads, so pine martens released into this region may have a high probability of successful establishment. However, the total area of woodland is relatively small compared with some of the other potential release regions considered here, so the lower number of pine marten home ranges that could be accommodated may affect the viability of the population.

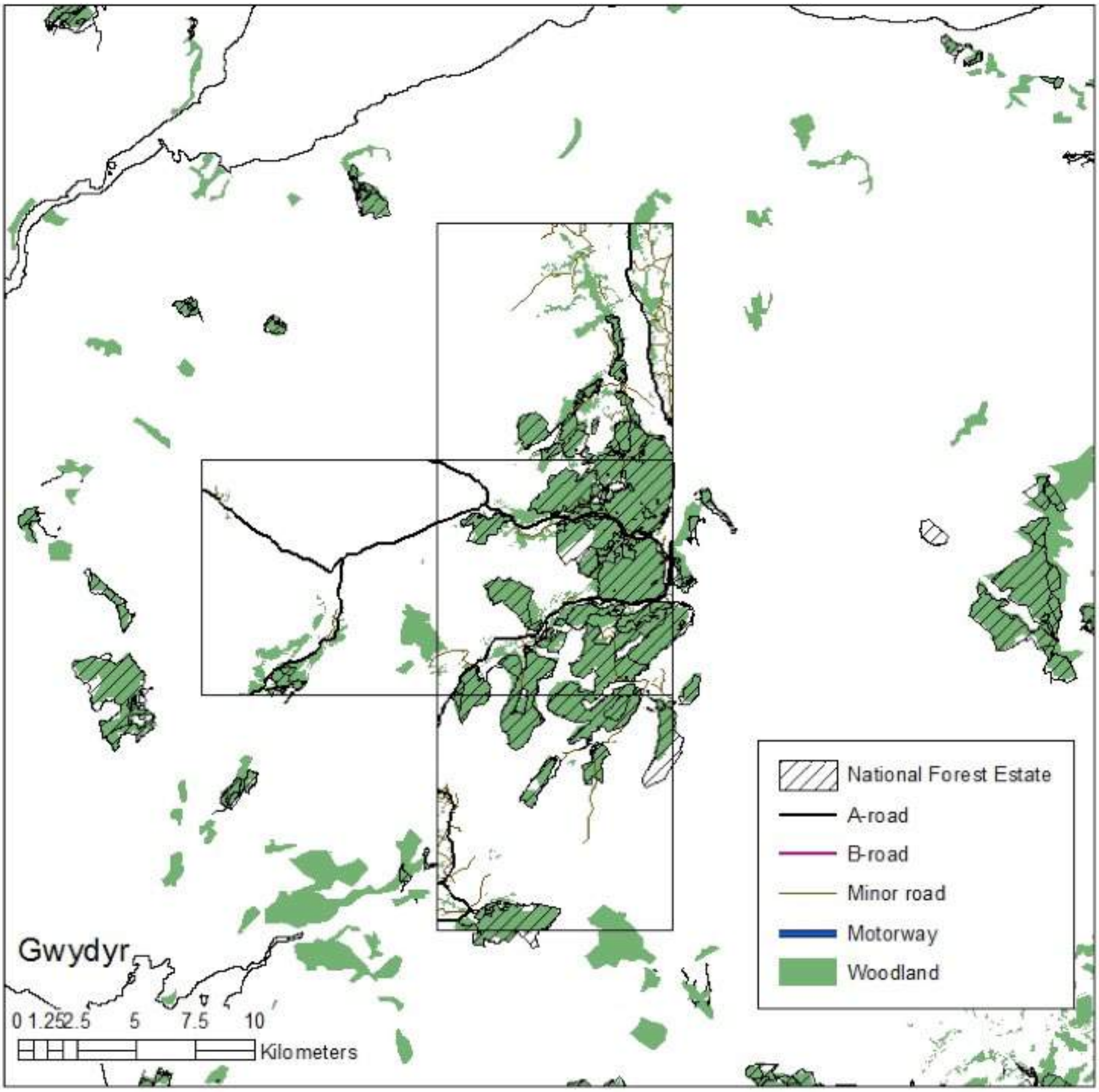

Figure 3.5 *Map of Gwydyr PRR showing all motorway, A, B and minor class roads in relation to national forest estate and privately owned woodlands.*

Mawddach-Dyfi PRR

Previous research into the feasibility of pine marten translocations to southern Britain suggested that the large expanse of contiguous woodland centred on the Cambrian Mountains in central Wales provides a habitat network with the potential to support about 350 pine martens (Bright & Harris, 1994). The study also concluded that the total area of woodland and low density of roads and gamekeepers in this region make central Wales ideal for conservation translocations of pine martens. For the present analyses, central Wales was separated into three potential reinforcement regions (Mawddach-Dyfi, North Ceredigion and Tywi). The rationale for this was that Mawddach-Dyfi (shown in figure 3.6) is separated from North Ceredigion (figure 3.7) by the Dyfi estuary, and two major roads (the A470 and A489), although the distance between woodland blocks large enough to support more than one marten territory is only approximately 4km. The most southerly large woodland block in the North Ceredigion PRR is approximately 6km from the nearest woodland in the north of Tywi. However, there is essentially a network of suitable “stepping stone” habitat along which colonization would likely occur, which connects all three. It is therefore suggested that translocations to the most optimal release sites within the three PRRs in central Wales would have a large capacity for population increase and expansion.

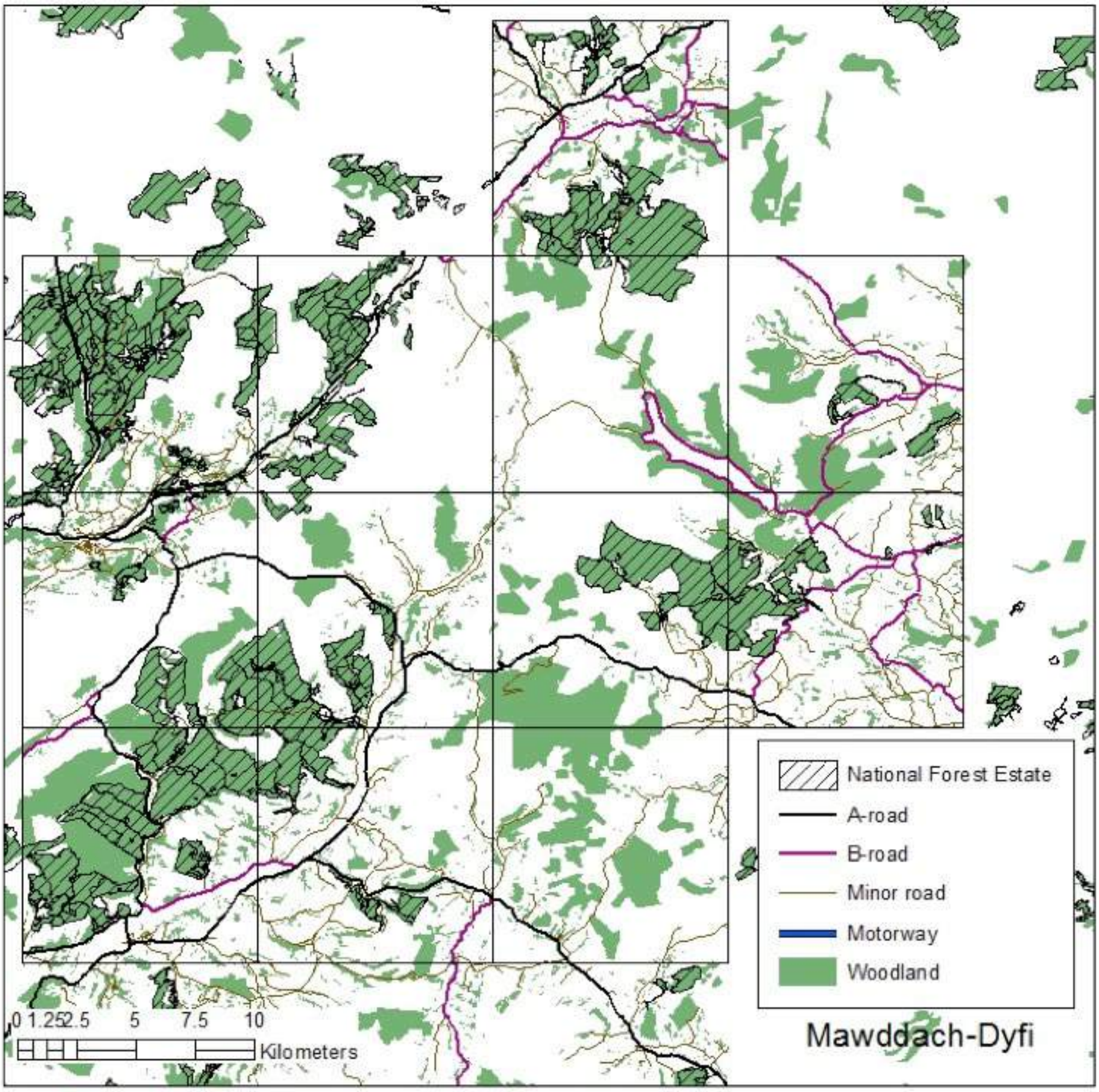

Figure 3.6 *Map of Mawddach-Dyfi PRR showing all motorway, A, B and minor class roads in relation to national forest estate and privately owned woodlands.*

North Ceredigion PRR

North Ceredigion has low traffic flow and few major roads, but many of the woodlands in this region are relatively small compared with those in Tywi and Mawddach-Dyfi. However, it does have good connectivity and some blocks of diverse mature broadleaved woodland. It would also provide potential for expansion and spread into both of the adjacent PRRs, once martens were established here.

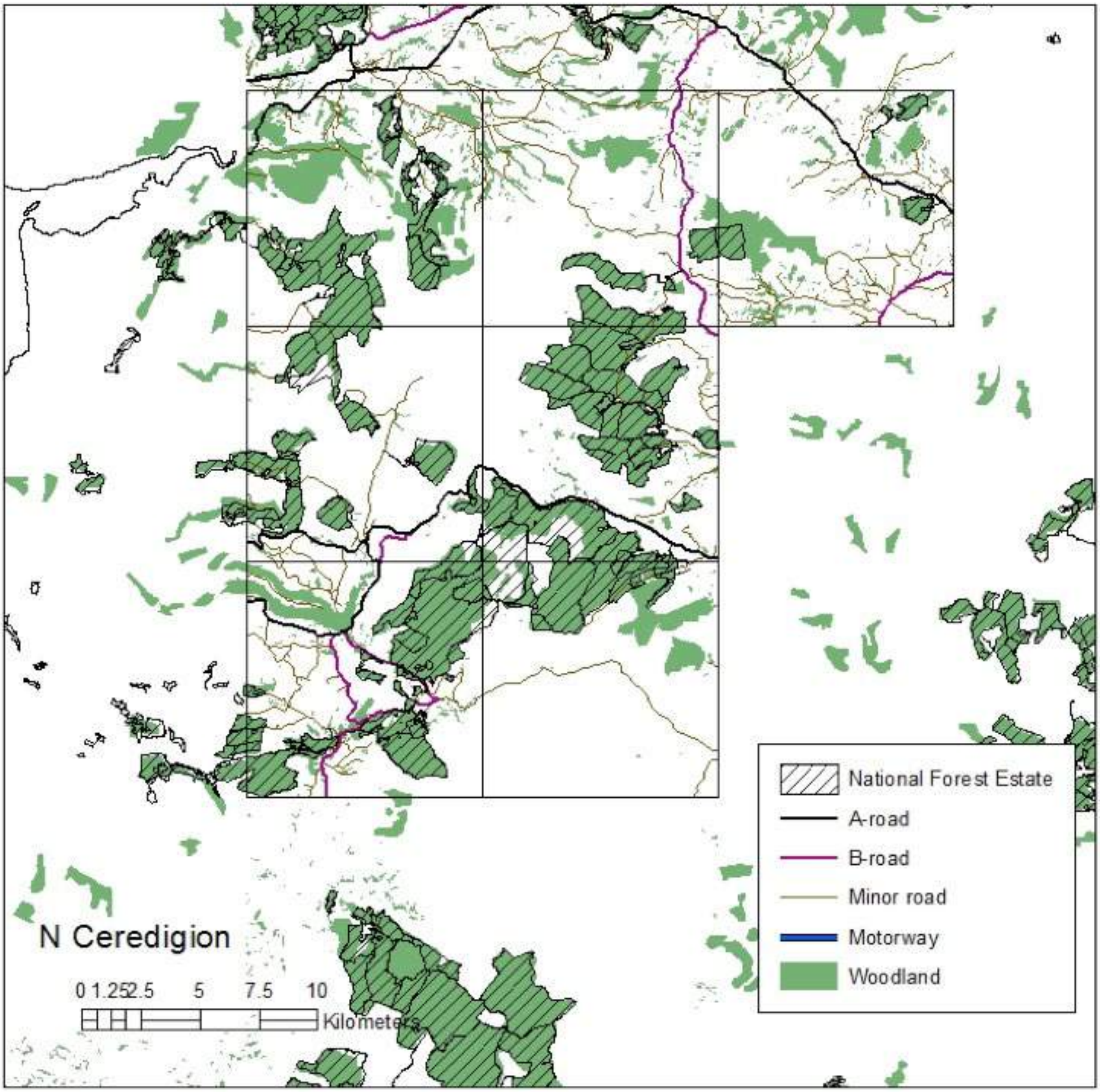

Figure 3.7 *Map of North Ceredigion PRR showing all motorway, A, B and minor class roads in relation to national forest estate and privately owned woodlands.*

Tywi PRR

The forests in this PRR (shown in figure 3.8) vary widely in character, from upland spruce plantations down to the smaller, mixed woodlands along the river valleys. The density of roads is low, as is the amount of natural grassland outside the woodlands. However, there is a relatively high proportion of potentially suitable habitat for small mammals within forest blocks, as shown in figure 3.2. One of only three significant populations of red squirrels, *Sciurus vulgaris*, remaining in Wales is found in the upland conifer blocks here, and the potential implications of this are discussed in chapter 4.

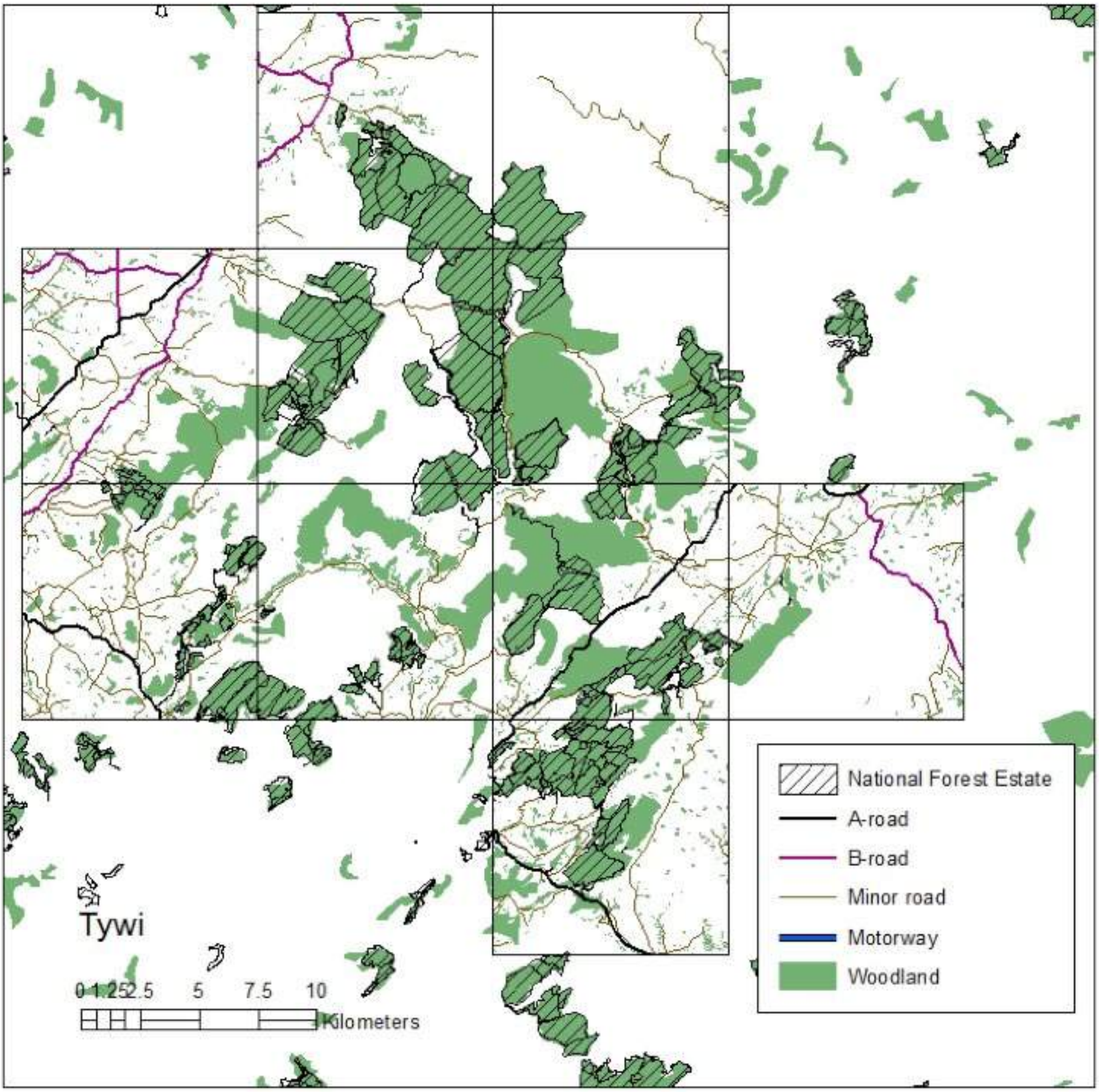

Figure 3.8 *Map of Tywi PRR showing all motorway, A, B and minor class roads in relation to national forest estate and privately owned woodlands.*

3.4. Discussion

Roads affect the survival of many carnivores (Kramer-Schadt *et al*. 2004; Reynolds-Hogland & Mitchell 2007) and are likely to be an important source of mortality affecting the viability of newly reinforced pine marten populations. The total length of roads in each PRR and the percentage of those within woodland, combined with annual volume of traffic were used to calculate the relative likelihood of marten mortality due to road traffic accidents. Even a low rate of additional mortality will significantly increase extinction risk and jeopardise the establishment of a newly reinforced marten population. It is therefore suggested that translocations should first be to regions with woodland blocks large enough to support relatively high marten numbers, and where likelihood of violent mortality would be low. Released animals should be closely monitored and the outcomes of these initial translocations be used to inform subsequent plans.

Road traffic accidents are only one of several potential causes of additional mortality, including predation. Pine marten has been recorded in the diet of red foxes in Poland (Hartová-Nentvichová *et al*. 2010; Kidawa & Kowalczyk 2011), although this could have been from carrion, however direct predation of marten by foxes has also been observed in Sweden (Lindström *et al*. 1995). In addition to this a decline in foxes in Scandinavia as a result of disease outbreak was correlated with a significant increase in pine marten density, suggesting that foxes have a strong negative effect on pine marten populations (Lindström *et al*. 1995). However, another study found no relationship between fox and pine marten abundance (Kurki *et al*. 1998), and pine marten killing of fox cubs has also recently been recorded (Brzeziński, Rodak & Zalewski). Interactions between fox and pine marten are clearly not straightforward but it seems that adults of both species will kill the other's young. The need for sufficient suitable, safe den sites for pine martens in potential release regions is therefore also an absolute requirement. Successful reproduction will depend upon the availability of natal den sites that will offer adequate insulation and protection from predators (Brainerd *et al*. 1995; Birks, Messenger & Halliwell 2005). Across continental Europe pine marten breeding dens are usually located 3-12 metres above ground in tree cavities, often in old green or black woodpecker holes (Brainerd *et al*. 1995; Kleef & Tydeman 2009; Van Den Berge & Gouwy 2011). Arboreal cavities are scarce in Britain (Birks, Messenger & Halliwell 2005), particularly in commercially managed forests because of relatively short felling rotations. This is another factor that will need to be investigated during field surveys.

Incidental trapping and poisoning as a result of lawful pest control activities is another potential threat, but one that is very difficult to quantify. The distance to commercial shoots is measured here as an indicator of potential risk. However, there is wide variation among shoots in the level and type of pest control that is carried out, so this is intended merely to flag up a potential conflict, and to highlight an important stakeholder group with which to engage at the outset. Before selecting release sites there will be a lengthy period of public consultation in shortlisted potential reinforcement regions, and the level of risk to martens arising from existing land uses will be assessed.

GIS data were used for these initial analyses of habitat but it is recognised that these data have limitations. It is not possible to evaluate structural or species diversity, or the extent of ground cover or denning sites. Within forest stands, vertical and horizontal structural diversity probably have more impact on marten habitat use than stand age or species composition. Although habitat structure can be associated with both of these, structural complexity can also be provided by a range of habitats not usually associated with pine marten use, such as young coniferous (Brainerd *et al*. 1994) or deciduous forests (Porter, St Clair & Vries 2005) and transitional scrub. These habitats may therefore provide the structural elements that martens require for foraging, denning and breeding. Many exotic conifer plantations will not provide open woodland habitat with good understory, with the exception of continuous cover areas and long term retentions. Upland conifer, especially, is likely to be more densely planted due to a high risk of windthrow (Kerr 1999). Therefore forest structure can only be properly assessed during field surveys. These will be carried out during the second phase of this project, before any final recommendations can made.

The data used for the analyses presented here were the most current available, however, due to the recent emergence of new diseases such as *Phythophthora ramorum* and *Chalara fraxinea*, there has been a large amount of additional felling of some tree species across the UK, in particular larch and ash. This may have altered the character of some of the woodland in the regions evaluated here, which will also require further investigation.

Open, felled and regenerating areas were assumed to be beneficial for martens because they are important habitat for small mammals, but risk of predation by foxes and raptors will also be greater in open areas. Several studies of the closely related American marten have shown that large clear cut and regenerating areas within commercial forests are of little use, particularly in winter (Hawley & Newby 1957; Soutiere 1979; Steventon & Major 1982; Hargis, Bissonette & Turner 1999). However, Storch, Lindstrom and De Jounge (1990) suggested that it was snow conditions rather than predation risk that limited the martens' access to the higher densities of *Microtus* voles in this habitat type in winter because, in Finland, pine marten avoidance of clear-cut areas did not change after a decline in red fox numbers. In south-west Scotland where prolonged periods of lying snow are rare, closed canopy and clearfell were both significantly preferred by pine martens over other habitat types in the study area (Bright & Smithson 1997), but this did not incorporate any distance measures of habitat use. Caryl *et al.* (2012) found significant intersexual differences in use of open space. Male martens showed less aversion to open habitats and travelled twice as far as females outside the forest edge. This may be because they are not as vulnerable to predation as a result of their much larger body size. It is likely then that the value of open habitat patches for pine martens will vary with the size and context of patches within the landscape. More detailed information from forest design plans and field survey will be needed to assess this. We present information on the amount of rough grassland in the landscape surrounding each PRR. This was highly associated with habitat suitability for pine martens in the model used for prioritising search regions. However, this takes no account of local variation in land use. Grazing pressure from domestic livestock may reduce vole densities on rough grassland at forest edges that would otherwise be very beneficial for pine martens. Again, this will need to be assessed with field surveys.

It is now recognised that diverse forests provide a range of benefits and are more resilient to changing environmental conditions (UK Forestry Commission 2011), and many commercial forest management plans now aim to increase structural and species diversity to promote sustainability of forest ecosystems. The pine marten can be considered a flagship woodland species and a viable population of pine martens will be a good indicator of more naturally structured and biodiverse forests.

Ultimately, the selection of reinforcement regions will need to take full account of local support and acceptability of pine martens and their potential impacts in the local area.

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4. POTENTIAL IMPACTS OF RECOVERING PINE MARTEN POPULATIONS ON OTHER SPECIES

4.1. Introduction

The perception that the recovery or restoration of a native predator will have a negative effect on native prey species is a major concern for some stakeholders. A thorough risk assessment is an integral part of the feasibility study and this includes evaluating the ecological roles of translocated animals in their new environment, and potential impacts on the conservation interests of other species already present in release areas (IUCN 2013).

Introduced, non-native predators can have a devastating effect on naïve prey populations, but it has been suggested that this is not the case with native predators (Salo *et al*. 2007). When predators and prey co-evolve over a long period of time, prey often develop morphological or behavioural adaptations to reduce the rate of encounters with predators or increase their prospects of escape if detected (Lima & Dill 1990). The question of whether predation limits prey populations has been discussed for many years (Valkama *et al*. 2005). When the removal of prey is offset by density dependent increases in the productivity of prey populations then predation is compensatory. In this case predation simply replaces other causes of mortality and there is no overall impact on the prey population. However, if there is no density dependent compensation then mortality from predation is additive and will result in a reduction in prey population size. Mortality may be compensatory in the case of native predators, but is likely to be additive as a result of non-native predators (Holt *et al*. 2008).

Bird populations have been extensively studied to answer the question of how predators affect the size of prey populations (e.g. Newton 1993). Some studies have found a negative correlation between predator numbers and prey populations (Tharme *et al*. 2001; Fletcher *et al*. 2010; Smith *et al*. 2010), however other studies have shown that presence or numbers of predators have no effect on prey numbers (Bolton *et al*. 2007; Holt *et al*. 2008).

In some circumstances predation is not an important factor in population dynamics because prey have spatial (MacPherson & Bright 2010) or temporal (Hebblewhite & Merrill 2009) refugia available or because predators choose more preferred or accessible prey. Landscapes are heterogeneous and predators have limitations, so there are areas and times when prey can escape predation risk (Heithaus *et al*. 2009). If predators are at low density and there is sufficient suitable habitat then prey can utilise areas that are temporarily predator free (Wirsing, Cameron & Heithaus 2010). Also if there is an alternative preferred prey or food available and this is abundant, then there may be little direct predation upon less preferred prey species (Tschanz, Bersier & Bacher 2007). This will particularly apply if predators are limited by density dependent effects, so that the abundance of one prey species does not result in increased predator density for other prey species.

However, where a prey species has already suffered significant declines as a result of habitat loss, fragmentation and other factors, its vulnerability to even slight changes in predation rate may be increased. This means that in some circumstances the recovery of a formerly very rare native predator could have a negative impact.

Populations of many wild birds in the UK have undergone steep declines over the past 40 years (Baillie *et al*. 2009), including woodland specialists and long-distance migrants (Gregory *et al*. 2007; Hewson & Noble 2009). Probably the most vulnerable life history stage for bird populations in altricial, semialtricial or semiprecocial species is the egg and nestling stage (Lima 2009). Adult birds can escape predation by flying, but they cannot move their eggs and so their ability to compensate for predation risk by avoidance is limited once they have committed to a nest site (Cresswell 2011). It has been suggested that increasing rates of nest predation could be a possible cause of the observed declines in UK bird populations (Fuller *et al*. 2005), although studies of individual species (Siriwardena 2004; Siriwardena 2006) and broad-scale surveys of changes in woodland birds (Amar et al. 2006, Newson et al. 2010b) do not support this.

The pine marten is an opportunistic, generalist predator with a diet that broadly reflects what is locally and seasonally abundant. This includes a wide range of mammals, fruits and berries, birds, invertebrates, amphibians and reptiles (Zalewski, Jedrzejewski & Jedrzejewska 1995; Baltrūnaitė 2002; Sidorovich, Krasko & Dyman 2005; Posłuszny *et al*. 2007; Caryl 2008). The pine marten shows some degree of specialisation on voles and has been shown to exhibit a type II functional response (Holling 1959) to the bank vole, *Myodes glareolus*, in Poland (Zalewski, Jedrzejewski & Jedrzejewska 1995). In the type II functional response, the rate of prey consumption by a predator rises as prey density increases, but eventually levels off at a plateau (or asymptote) at which it remains constant regardless of further increases in prey density. A review of pine marten diet across its geographical range found that small mammals (<150g), are the most important food for pine martens in the temperate zone, making up an average of 50% of the diet at 50-60° north (Zalewski 2005). The proportion of birds in the diet varies seasonally across the range. It is higher in spring and summer, but also increases with latitude (Zalewski 2005). In Britain pine martens show a preference for field voles, *Microtus agrestis*, (Balharry 1993; Halliwell 1997) independent of their relative abundance locally, but switch to fruits when they are available in the autumn, with birds and other supplementary prey being taken mainly in summer (Caryl *et al*. 2012).

Here we assess the overlap between potential reinforcement regions for pine marten and the UK breeding distribution of rare bird species that might be predated by pine marten. The implications are discussed along with other wildlife that may be affected by an increase in pine marten numbers, as well as recommendations for mitigation measures and future monitoring.

4.2. Methods

As with the previous study (Bright & Halliwell 1999), potential impacts on rare or declining bird species were assessed for six potential reinforcement regions (PRRs) in England and Wales that are currently being considered for pine marten translocations.

The red and amber lists of bird species of conservation concern (Eaton *et al*. 2009) were used to identify which species had a breeding distribution in the UK that overlapped with PRRs. Red listed species are those that are recognized as being Globally Threatened using IUCN criteria, have suffered a severe decline in the UK between 1800 and 1995, without substantial recent recovery, have undergone a severe long-term decline in the UK breeding or non-breeding population size or have shown a reduction of more than half in the number of 10km squares occupied by breeding birds in the UK over 25 years or more. Species on the amber list are those that are categorised as a Species of European Conservation Concern, were previously red but have more than doubled in the last 25 years or have undergone a moderate (more than 25% but less than 50%) decline in the breeding or non-breeding population or range. Other criteria are rarity, for species with a UK breeding population of less than 300 pairs, or non-breeding population of less than 900 individuals, localisation (at least 50% of the UK breeding or non-breeding population found in 10 or fewer sites) and international importance where at least 20% of the European breeding or non-breeding population is to be found in the UK.

Species from both red and amber categories that nest in woodland, woodland edge or habitats likely to be adjacent to woodland were considered as potentially vulnerable to impacts from marten predation. The breeding distribution of each of these species within PRRs was derived from data provided by the British Trust for Ornithology (BTO) from the 2007-2011 Atlas of Breeding Birds (Balmer *et al*. 2014). The number of 10km squares in which each species was recorded as breeding, and the proportion of each species' breeding distribution in mainland Britain that was within PRRs were calculated.

4.3. Results

Out of 178 (52 red and 126 amber list) species, 79 (38 red and 51 amber) are recorded as confirmed, probable or possible breeding in one or more 10km squares covered by the 6 PRRs. These are shown in table 4.1.

Table 4.1 *All red (n=28) and amber (n=51) listed birds recorded as breeding in PRRs (data from BTO breeding bird atlas 2007-2011)*

Red listed birds	Amber listed birds	
Hen harrier	Little Grebe	Dunnock
Black grouse	Little egret	Nightingale
Grey partridge	Greylag goose	Common redstart
Lapwing	Shelduck	Whinchat
Dunlin	Teal	Wheatear
Herring gull	Mallard	Mistle thrush
Cuckoo	Tufted duck	Dartford warbler
Nightjar	Honey buzzard	Common whitethroat
Lesser spotted woodpecker	Red kite	Willow warbler
Skylark	Osprey	Firecrest
Tree pipit	Kestrel	Pied flycatcher
Yellow wagtail	Merlin	Chough
Ring ouzel	Red grouse	Snipe
Song thrush	Quail	Woodcock
Grasshopper warbler	Oystercatcher	Curlew
Wood warbler	Ringed plover	Redshank
Spotted flycatcher	Golden plover	Common sandpiper
Marsh tit	Stock dove	Black headed gull
Willow tit	Barn owl	Lesser black backed gull
Starling	Short-eared owl	Great black backed gull
House sparrow	Swift	Bullfinch
Linnet	Kingfisher	Reed bunting
Twite	Green woodpecker	Woodlark
Lesser redpoll	Sand martin	
Hawfinch	Swallow	
Yellowhammer		
Turtle dove		
Wryneck		

Of these 79 species, the majority (48%) nest in woodland or woodland edge, 18% nest in habitats that could occur adjacent to woodland including grassland, moorland and heath. The others nest in habitats not likely to be utilised by pine marten such as coastal cliffs, shores and bare ground (see table 4.2). This is unsurprising as the PRRs were selected because of their high proportion of woodland and grassland habitat.

Table 4.2 *Nest habitat of red and amber list breeding birds recorded in PRRs.*

Nest habitat type	Red listed birds	Amber listed birds
Woodland	12 (43%)	14 (27%)
Woodland edge	7 (25%)	5 (10%)
Potentially adjacent (moorland, heath, grassland)	6 (21%)	8 (16%)
Non-woodland	3 (11%)	24 (47%)

The highest number of red and amber list species (43) is found in the Mawddach-Dyfi PRR. The number is broadly similar (35-39) in the rest of the PRRs (see table 4.3).

Table 4.3 *The number of potentially vulnerable red and amber listed bird species found in each PRR.*

PRR	Red listed birds	Amber listed birds
Gwydyr	20	18
Mawddach-Dyfi	22	21
N Ceredigion	18	17
Tywi	20	19
Afan	17	19
Dean	20	19

The percentage of each species' current British breeding distribution contained within PRRs is generally low (<1%). The means for each PRR are shown in Table 4.4. Only three species, Wryneck (*Jynx torquilla*), Redstart (*Phoenicurus phoenicurus*) and Hawfinch (*Coccothraustes coccothraustes*), have more than 5% of their British breeding distribution within PRRs. Wryneck is only present in one of the PRRs, Tywi, but this represents 7.69% of its breeding range in Britain. Redstart has a north and westerly distribution in the UK, with the greatest concentrations in Wales. This species breeds in all 6 PRRs and a relatively high percentage of its British breeding distribution is within Afan (6.02%), Tywi (7.52%) and Mawddach-Dyfi (9.02%). Hawfinches are now confined to just a few key breeding areas in the UK, one of which is within the Dean PRR, which contains 5.31% of the Hawfinch's British breeding range. Hawfinches also breed within one or more 10km squares covered by the Mawddach-Dyfi and Gwydyr PRRs, but these represent smaller percentages of the breeding distribution (2.65 and 0.88% respectively).

Table 4.4 *Percentage of the British breeding distribution of potentially vulnerable red and amber list bird species within PRRS (mean±SE).*

PRR	Red listed birds	Amber listed birds
Gwydyr	0.35 (0.06)	0.37 (0.16)
Mawddach-Dyfi	0.76 (0.12)	1.04 (0.41)
N Ceredigion	0.38 (0.05)	0.67 (0.30)
Tywi	0.99 (0.36)	0.87 (0.38)
Afan	0.56 (0.13)	0.73 (0.30)
Dean	0.71 (0.26)	0.71 (0.23)

4.4. Discussion

There is a small number of declining bird species present within PRRs that could be vulnerable to impacts from any additional predation and this must be taken into account when considering sites for pine marten releases.

The Redstart is associated with Atlantic oak woods but can also be present in well thinned upland conifers. It is a cavity nester but the cavities are probably too small for a pine marten to be able to access birds, eggs or chicks. Redstarts also prefer territories with sparse vegetation and bare ground to enable them to access the ground invertebrates on which they feed (Martinez *et al*. 2010). This is not the type of habitat preferred by pine martens, particularly when foraging, so there may be little overlap at the home range scale with these two species. The distribution of redstarts and pine martens overlaps across Europe, where redstart numbers have increased in recent years (PECBMS 2012).

Wrynecks are small, cryptically coloured woodpeckers found in open woodland and farmland. They are a secondary cavity breeder and also need access to sparsely vegetated or bare ground so that they can forage for ants and other invertebrates (Mermod *et al*. 2009). This coupled with their rarity means they are unlikely to be encountered by pine martens.

Hawfinches are declining and rare in Britain (Langston, Gregory & Adams 2002; Balmer *et al*. 2014). A number of potential reasons for this have been suggested including habitat loss, habitat change, (Langston, Gregory & Adams 2002) and predation, in particular from crows (*Corvus corvus*), jays (*Garrulus glandarius*) (Bijlsma 1998) and grey squirrels (*Sciurus carolinensis*) (Langston, Gregory & Adams 2002). However, elsewhere in Europe hawfinches show a fluctuating but consistently increasing trend despite the presence of other predators including the pine marten (Bijlsma 1998; Tomiałojć 2005). Low nesting losses for hawfinches have been observed to correlate with high rodent abundance which provides a more accessible prey for medium sized predators (Tomiałojć 2005).

Many generalist predators have a preferred prey species that maximises ease of capture and body size (Roth, Lima & Vetter 2006). In Britain the field vole is the predominant small mammal in the pine marten's diet (Lockie 1961; Velander 1983; Balharry 1993; Gurnell *et al*. 1994; Halliwell 1997; Coope 2007; Caryl *et al*. 2012). It is thought that field voles are profitable prey because of their size, clumped distributions and relative lack of anti-predator behaviours (Balharry 1993). There is evidence to suggest that, where vole population cycles have pronounced amplitudes, their proportion in marten diet varies more widely than where they are only weakly cyclic (Goszczynski 1986). One of the concerns in Britain is the alternative prey species that pine martens would switch to if they were affected by low vole densities. Field voles in Britain do undergo population cycles (Charles 1981; Lambin *et al*. 1998), although at much lower amplitudes than those in Fennoscandia (Lambin, Petty & Mackinnon 2000). Where field voles are at low density in Scotland, the alternative foods recorded as being taken have been predominantly invertebrates (Balharry 1993; Bright & Smithson 1997), but also passerine birds (Bright & Smithson 1997). However, even during rodent population crashes in Bialowieza, Poland, Zalewski, Jedrzejewski and Jedrzejewska (1995) found that alternative prey formed a much smaller proportion of the diet than rodents. They therefore considered it unlikely that significant declines of alternative prey would be observed, even during times of rodent scarcity.

If pine martens take rare prey species opportunistically as they encounter them, then prey vulnerability will be related to the amount of time pine martens spend in the same habitat as the prey. A species' vulnerability to predation by pine martens will also depend on a number of other factors including its breeding biology, population density, anti-predator strategies and the availability of alternative prey. In addition to this, there will be interactions with other predators, with which pine martens are competing for resources. When a range of predators is present, as well as interference competition between predators, there may also be intra-guild predation of the predators themselves (Polis, Myers & Holt

1989). The general perception is often that there will be additional mortality for prey species if pine marten numbers increase. However pine martens might have a negative impact on other nest predators, such as grey squirrels (Sheehy & Lawton 2014), and may consume prey that would otherwise have been eaten by other predators. Food webs are highly complex and predator impacts are rarely as simple as generally perceived.

It must be remembered that in order to maintain a stable population, on average each adult bird need only rear one chick to breeding age in its lifetime. Holt *et al.* (2008) found that predators had a larger impact on productivity than on breeding population and, from a conservation perspective, it is the breeding population that is important. Breeding bird populations can mitigate against increasing predation pressure by mechanisms such as compensatory reduction in mortality rates from reduced competition for resources, and the recruitment of surplus, non-breeding individuals (Newton 1993). While there is some evidence for predators having an impact on populations of passerines (Stoate & Szczur 2001; Stoate & Szczur 2006), ground nesting waders and game birds (Tapper, Potts & Brockless 1996), previous analyses of UK national bird monitoring data, focussing in each case on a single predator species, could not detect any marked effects (Gooch, Baillie & Birkhead 1991; Thomson *et al*. 1998; Summers *et al*. 2004; Chamberlain, Glue & Toms 2009).

Bright and Halliwell (1999) modelled the potential impacts of pine marten predation based on the proportion of birds in the pine marten's diet and the proportion of rare birds in communities. The model assumed a type I functional response by pine martens, where predation on rare birds would be directly proportional to bird abundance, so the proportional impact of predation was constant for low and high density of rare birds. The model was based on a high pine marten population density which would probably not be reached in PRRs (Bright & Smithson, 1997) and therefore represented a worst case scenario. The results of the model suggested that pine martens would be very unlikely to kill more than 0.8 individuals of a rare bird species per km^2 per year. This would probably not have a significant impact on rare bird populations, even those that were already declining for other reasons. If pine marten predation is in direct proportion to bird abundance then significant impacts are most likely on commoner species such as blackbird (*Turdus merula*). However the results of Bright & Halliwell's (1999) model showed that, for there to be any impact on blackbirds, birds as a whole would need to constitute more than 30% of pine marten diet, which is rarely the case. Even under these circumstances, pine martens would only predate two blackbirds per km^2 per year, not enough to have a significant impact on woodland blackbird populations which can number 60-100 per km^2 (Bellamy *et al*. 2000).

Anthropogenic habitat change is a factor that may interact with predation and its effects on prey. Reduced habitat heterogeneity may limit a prey's ability to evade predation (Trussell, Ewanchuk & Matassa 2006). There can be a high level of spatial variability in predation levels at the landscape scale as a result of heterogeneity in the physical characteristics of the landscape where predator and prey interact (Kauffman *et al*. 2007). It has been suggested that observed declines in farmland birds in the UK may be partly due to changes in habitat which have left bird species less able to effectively manage their risk of predation (Evans 2004; Whittingham & Evans 2004).

A lack of mature woodland in many places has resulted in reduced availability of cavities for hole-nesting birds (and other animals) which, in some cases, has promoted the use of nest boxes. Pine martens will predate natural nests of medium sized hole-nesting birds such as Tengmalm's owl, (*Aegolius funereus*) (Sonerud 1985) and black woodpecker (*Dryocopus martius*) (Nilsson, Johnsson & Tjernberg 1991), but they are unable to access natural nest sites of small, hole-nesting passerines. This is not the case with nest boxes, however, which are also usually placed at relatively high densities and are distinctive in appearance, increasing their detectability. Predators remember the location of nest boxes where they have found food, and predation by pine martens on great tit (*Parus major*) and blue tit nest (*Parus caeruleus*) boxes has been shown to significantly increase with the length of time boxes have been in place (Sorace, Petrassi & Consiglio 2004). There are, however, practical measures that can be put in place to militate against this which include placing nest boxes on less accessible branches or poles, fitting guards or making boxes from predator proof materials.

Pied flycatcher, *Ficedula hypoleuca,* is one of the amber list species present in all of the potential reinforcement regions. This species is a cavity nester but natural nest cavities are too small for a pine marten to access. However where there are nest box schemes for pied flycatcher these might need to be modified to prevent martens from accessing them. The same is true for hazel dormouse, *Muscardinus avellanarius*. This species is unlikely to be predated by pine martens in natural nest sites or while active. But dormice would be vulnerable to marten predation in a nest box, especially on cooler days when they go into torpor. Depending on the design used, some of these nest boxes would also need to be slightly modified to prevent pine marten opening them.

Measures can also be taken to protect game species from pine marten predation. An extensive study of pine marten diet in Scotland found that the number of pheasants (*Phasianus colchicus*) taken by pine marten (2.9/km^2) represented less than 1% of the birds released (Halliwell 1997). This is a small proportion in comparison to other predators, but this relates to free flying birds. Mammalian predators can cause considerable damage if they get into a pheasant release pen. However it has been shown that pens can be protected against pine martens and other predators with slight adjustments (Balharry 1998).

The main focus here is primarily on birds, a taxon for which reliable data were available from the BTO's national survey of breeding birds. This survey covered all 10-km squares in the UK and Ireland in both winter and the breeding season. Fieldwork was completed following the breeding season of 2011, and the data are now published in the 2007-2011 atlas of breeding birds (Balmer *et al*. 2014). However, other taxa must also be considered and local surveys carried out if recent data are unavailable.

One of only three remaining populations of red squirrel (*Sciurus vulgaris*) in Wales is found in the upland conifer plantations in the Tywi PRR. Red squirrels co-exist with pine martens across Europe and, during extended periods of snow cover when voles are inaccessible, it has been shown that pine martens switch to alternative prey including squirrels (Storch, Lindstrom & De Jounge 1990). However, studies of pine marten diet in areas of Britain where red squirrels occur have found no red squirrels (Caryl 2008; Paterson & Skipper 2008) or only small proportions (<1%) in the diet (Velander 1983; Halliwell 1997). The density of red squirrels in the Tywi forest is thought to be very low (Mid-Wales Squirrel Partnership 2009), so encounters with pine martens would probably happen only occasionally. Predation would be incidental and therefore rare, so is unlikely to have a significant impact.

Certain bat species may be at risk from over-predation by pine martens under particular circumstances. Some of the woodland bats select roost cavities with entrances smaller than the width of a marten's head (Ruczyński & Bogdanowicz 2005) and previous dietary analysis suggested that pine martens very rarely eat bats (Zalewski, Jedrzejewski & Jedrzejewska 1995). However bats have been found in the diet of stone martens (*Martes foina*) in Romania (Romanowski & Lesinski 1991), Hungary (Lanszki, Sardi & Széles 2009), and in the underground tunnels of the Nietoperek bat reserve in Poland there are examples of stone martens preying almost exclusively (80% of food biomass) on bats (Urbanczyk 1981; Lesinski & Romanowski 1988). Recently it has also been found that some individual pine martens repeatedly visit these tunnels (J.Power, unpubl. data), and there are examples in the literature of martens frequenting places that they will find bats like the Marl pits in Holland (Bekker 1988). It seems that, while the concentration of bats in colonies reduces the chance of martens finding them, once a colony has been discovered it provides a readily available food supply. Therefore it is important that up to date survey data are reviewed and any significant and accessible colonies of rare bats in PRRs should be taken into account before a final decision about release areas is made.

Pine marten predation is only likely to be a problem for species that are nationally rare but locally abundant, or colonial. In the majority of cases the rarity of a species would mean that martens would be relatively unlikely to encounter it. However there might be a particular risk relating to some rare species that occupy refuges in areas preferred by martens for foraging.

Black grouse, *Tetrao tetrix,* has been highlighted by some stakeholders as a species of potential concern that needs to be considered when selecting reinforcement regions. Black grouse is one of the fastest declining species in the UK and is now only found in the uplands of northern England, Scotland and north Wales. Black grouse need a fine-scale mosaic of habitats containing relatively small areas of woodland, moorland and grassland. The expansion and growth of conifer plantations, changes in agriculture on marginal land, and increases in deer and sheep numbers are all factors that may have affected black grouse habitat (Baines 1996). In Britain the species is now mostly associated with plantation and forest edges.

Grouse populations were not previously considered to be regulated by predation (Lindström 1994), although some effects at a local scale have been reported (Marcstrom, Kenward & Engren 1988; Valkama *et al*. 2005). In Scandinavia black grouse breeding success appears to be related to levels of predation on nests and chicks, which is linked to the population cycles of voles as alternative prey. In Britain there is no evidence for cyclic breeding success in response to predators switching between alternative prey. Despite this, a study by Summers *et al*. (2004) did find between year differences in black grouse breeding success which they thought may have been linked to predation. However, as it was unlikely that predator numbers had changed markedly between the two years of their study, it could have been because chicks are more susceptible to predation during and just after adverse weather conditions (Kastdalen & Wegge 1985). Summers *et al*. (2004) concluded that in years with good weather and reasonable numbers of insects the impact of predators was probably low and breeding success was similar, regardless of predator management.

Previous reviews of predator control and its effects on birds found that predator removal generally has a positive effect on productivity; however, effects on the breeding population are less clear (Côté & Sutherland 1995; Côté & Sutherland 1997; Newton 1998; Holt *et al*. 2008). This means that, while predator removal can fulfil the primary objective of game managers, which is to generate an artificial surplus of birds and, thereby, larger shooting bags, it may not be necessary to fulfil the rather different objective of conservation managers, which is to maximise stable breeding numbers.

The number of black grouse in strongholds in Dumfries and Galloway, Perthshire, Deeside and Speyside in Scotland has risen in recent years (RSPB 2012). These are all areas where pine martens are present. It is thought that good weather in 2010 resulted in a very productive breeding season also helped by major woodland initiatives and conservation efforts.

Capercaillie, *Tetrao urogallus*, another forest grouse, has also shown increases in Speyside (Forestry Commission Scotland 2010), despite reported rises in pine marten sightings since 1990 (Summers *et al*. 2004). This suggests that predation by pine marten is not a limiting factor for these species and concurs with findings from a Scotland wide study which showed that spatial variations in productivity of capercaillie were related to a combination of impacts from crows and red foxes, but not pine martens (Baines, Moss & Dugan 2004).

Across its range, including in Scotland and elsewhere in Europe, the pine marten coexists with many potentially vulnerable rare bird species. Pine martens are territorial, have large home ranges and live at low population densities, so their impacts on rare birds are likely to be lower than commoner predators such as stoats (*Mustela erminea*) and foxes. There are many reasons to suggest that recovering pine marten populations would not negatively impact other native species in PRRs. It is, however, important to evaluate specific potential risks in these areas. The species considered here were only those present within PRRs, however it is important to also evaluate potential impacts in areas likely to be colonized by pine martens in future, should translocations go ahead and populations subsequently increase and expand.

When pine marten reintroductions to England were considered in the 1990s, colonisation regions were defined as an area of 5000km^2 surrounding proposed release sites. However, habitat connectivity, in terms of the ability of a species to move between distinct habitat patches in a landscape, is highly species specific, therefore it is recommended that a GIS model be parameterised to predict the future spread of pine marten populations from PRRs, which incorporates differences in permeability of the landscape

(Watts *et al*. 2010). This will provide a clearer picture of any other specific issues that may need to be considered when evaluating potential reinforcement regions. If pine marten translocations are to go ahead in Wales and England then a long term programme will be put in place to monitor any impacts on other native fauna and game.

4.5. References

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5. ASSESSING PUBLIC ATTITUDES TOWARDS PINE MARTEN REINFORCEMENTS

5.1. Introduction

Guidelines for reintroductions and translocations produced by the *International Union for Conservation of Nature* state that an assessment of socio-economic circumstances; human community attitudes, values, motivations and expectations; and the anticipated costs and benefits of a reintroduction should be incorporated into planning for such programmes (IUCN 2013).

Translocations are by no means guaranteed to succeed, requiring careful planning and public engagement (Reading & Clark 1996), and can have the adverse effect of harming future investment of effort into species that are in need of support if they fail (Yalden 1993). Public attitude surveys have been conducted worldwide to inform the recovery efforts, reintroductions and conservation management of a number of carnivore species, including European lynx *(Lynx lynx)* in eastern Europe (Balčiauskas, Kazlauskas & Randveer 2010; Lescureux *et al*. 2011); black-footed ferrets *(Mustela nigripes)* in Montana (Reading & Kellert 1993); brown bears *(Ursus arctos)* in Slovenia (Kaczensky, Blazic & Gossow 2004); wolves *(Canis lupus)* in Scotland (Nilsen *et al*. 2007) and the USA (Kellert 1985; Shelley, Treves & Naughton 2011; Slagle, Bruskotter & Wilson 2012; Treves, Naughton-Treves & Shelley 2013); and jaguars (*Panthera onca)*, pumas (*Puma concolor)* and coyotes (*Canis latrans)* in El Salvador (Campbell & Alvarado 2011). Similarly, a survey to assess the public perception of pine martens was undertaken in Scotland, in the context of a recovering population (Balharry *et al*. 1996), and in the 1990s to assess attitudes towards a proposed reintroduction in England (Bright & Halliwell 1999). The status of the indigenous English and Welsh populations was uncertain at the time of the previous surveys, and it was deemed inappropriate to proceed with reintroductions under those circumstances. With increased knowledge of the status of the relict pine marten population in Wales and England reinforcement can now be considered.

5.2. Methods

A public opinion survey was conducted to investigate attitudes towards the potential recovery of the pine marten in Wales through active intervention, in order to inform how to best achieve the objective of restoring the pine marten to parts of its former range in southern Britain.

A questionnaire was designed to ascertain the attitudes of a representative sample of the Welsh public towards a potential restocking of pine martens in Wales. Questions were chosen to assess attitudes towards biodiversity, pine martens and the potential for restocking, including a willingness to pay (WTP) question; to establish where and when people had heard of pine martens, and to elicit more information about sightings, and to establish respondents' interests or work influences (see appendices for questionnaire).

A total of 7,500 bi-lingual (Welsh & English) questionnaire surveys, with a cover letter, information leaflet and freepost return envelope, were distributed to rural catchments in Wales. Eight rural catchments were identified by the highest concentration of recent pine marten records; these areas were classified as 'hot-spots' (shown in Figure 5.1). 500 surveys were mailed out, one per household, to each of the eight hotspot areas, totalling 4000 surveys. A further 3,500 surveys were mailed out across other areas in rural Wales approximating the population demographics of the hotspot areas, which were identified by externally contracted marketing specialists (CACI, www.caci.co.uk). The households selected in each catchment were representative of the demographics of those populations, determined from a geo-demographic segmentation tool (ACORN, CACI). The incentive of a weekend break for two at a prominent hotel in one of the hotspot areas was included in an attempt to reduce non-response bias.

Simultaneously, a bi-lingual online version of the survey went active, using the survey website SurveyMonkey (www.surveymonkey.net). A weblink to the survey was publicised via The Vincent Wildlife Trust's website, websites of other wildlife conservation and research organisations, in e-newsletters and via social media. Data from all survey responses were entered and standardised through SurveyMonkey, enabling comparative analysis across the different methods of response.

It was acknowledged that there would be an unavoidable element of selection bias in those respondents who chose to complete the survey, and that the bias would be strongest for those completing the survey via SurveyMonkey (hereafter referred to as 'Weblink'), which was advertised to a population already presumed to be sympathetic to wildlife conservation. The most objective gauge of opinion was likely to be obtained from mailing respondents, so figures are provided for 'mailing' and 'Weblink' separately where there is a dichotomous response, and presented inclusively where there is general agreement.

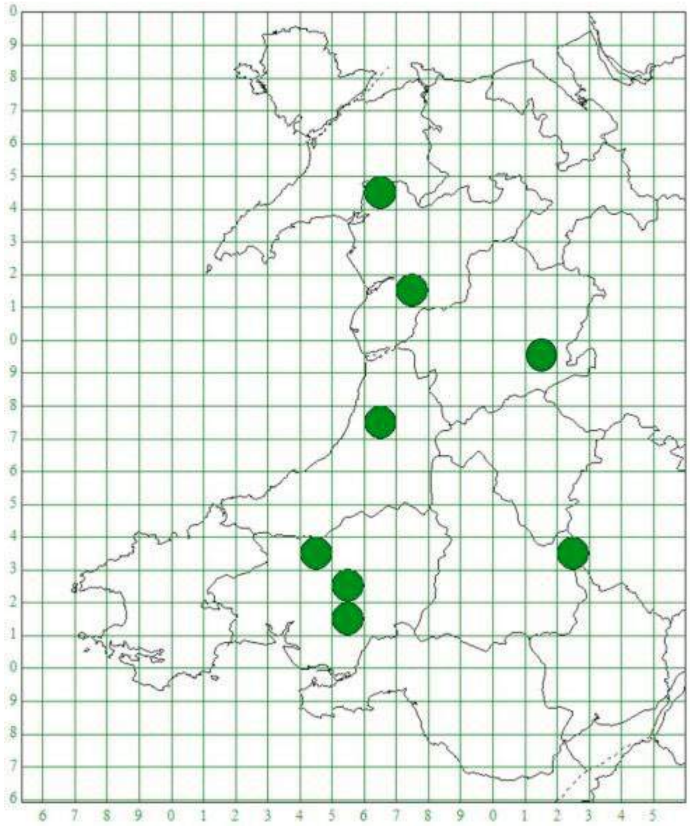

Figure 5.1 *The 8 hot-spot catchments in Wales, designated by the highest concentrations of recent pine marten sightings and records, collated by the Vincent Wildlife Trust since 1996.*

Willingness to pay (WTP) analysis was used to investigate public perception of the value of restocking pine martens in Wales. Contingent Valuation in the form of WTP is a survey-based economic valuation technique that uses a *stated* preference model, rather than a price-based *revealed* preference model for valuing non-market resources such as biodiversity and species conservation (Mitchell & Carson 1989). WTP was used during a survey assessing public perceptions of pine marten reintroductions to England (Bright & Halliwell 1999), and has been employed in other valuation studies for large carnivores in north America (Duffield & Neher 1996; Jorgensen, Wilson & Heberlein 2001), Canada (Martínez-Espiñeira 2006), and Sweden (Bostedt & Boman 1996; Boman & Bostedt 1999).

5.3. Results

Response rate

Of the 7,500 questionnaires mailed out, 140 were returned as undeliverable. 380 mailing responses were returned, though 8 were incomplete and could not be used, so the total number used for analysis was 372. An additional 315 responses were completed via Weblink, though 70 were incomplete, meaning a total of 617 viable responses were achieved from both methods. 26 and 10 respondents, respectively, replied to the mailing and Weblink survey in Welsh (7% & 4.1%), resulting in too small a sample size for comparison with English speaking respondents. Mailing response rates varied between geographical catchments from a low of 3.04% (n=15, Cefneithen) and high of 7.2% (n=36, Capel Bangor).

The overall response rate for the mailing survey was 4.96%, which was within the parameters expected from a public survey.

Attitudes and knowledge of biodiversity

51% (n=617) of all respondents had given a 'great deal' of thought to biodiversity loss (31% for mailing respondents, n=372), increasing to 78% (69.6% for mailing) when a 'fair amount' was included. 91% (n=556) of respondents (87.6% for mailing, n=326) had heard of pine martens, though more than one thought it was a bird. 47% (n=285) of all respondents cited television as the primary source of information, with no significant difference between mailing and Weblink respondents; education was cited equally second, and magazines and/or newsletters, internet and social media were cited equally third across both methods.

Attitudes to pine marten conservation/restocking

When asked how important it was to prevent pine martens becoming extinct in Wales, where 3=very important and 1=not important, the average response across both methods was 2.68 (2.56 for mailing respondents) with no significant difference between mailing & Weblink respondents.

87.3% (n=530) of respondents would support a restocking effort (84.7% mailing respondents, n=315). 91.5% (n=485) respondents gave reasons substantiating their support for restocking. This was an open question; the responses were separated into 10 categories based on frequent recurrence and ranked in order of number of responses (Table 5.1). 12.7% (n=77) of respondents would not support a restocking, of which 79.2% (n=61) gave reasons (Table 5.2). Weblink respondents were significantly more likely to support a restocking than mailing respondents (X^2=6.03, df=1, p=<0.05).

The WTP analysis showed the most frequently selected amount across both methods to be £10 (28.6%, n=143) followed by £5 (19%, n=95) (Figure 5.2). 16.4% (n=82) of respondents supported a restocking but chose £0 as a maximum WTP. 18 mailing respondents that indicated support for a re-stocking did not answer the WTP question, despite there being a £0 category available; this was also the case for 12 Weblink respondents, the reasoning for which is addressed in the discussion. Two people, one mailing and one Weblink respondent, selected '£100 or more'. The average willing to pay, discounting £0 and '£100 or more' bids to avoid distorting the analysis, was £14.15 for Weblink respondents, and £13.06 for mailing respondents. Sample sizes for occupation sub-groups were all lower than 30, with the exception of 'Farming' from mail respondents and 'Wildlife conservation' for Weblink respondents. When broken down to sub-group, farmers average WTP was £15.44 (n=43) for mailing respondents. When inflation is taken into account, this compares closely to the figure of £15.08 (inflation included) for farmers in Bright and Halliwell (1999). Six mailing respondents and two Weblink respondents were prepared to pay to prevent the restocking of pine martens; the maximum amount selected was £25.

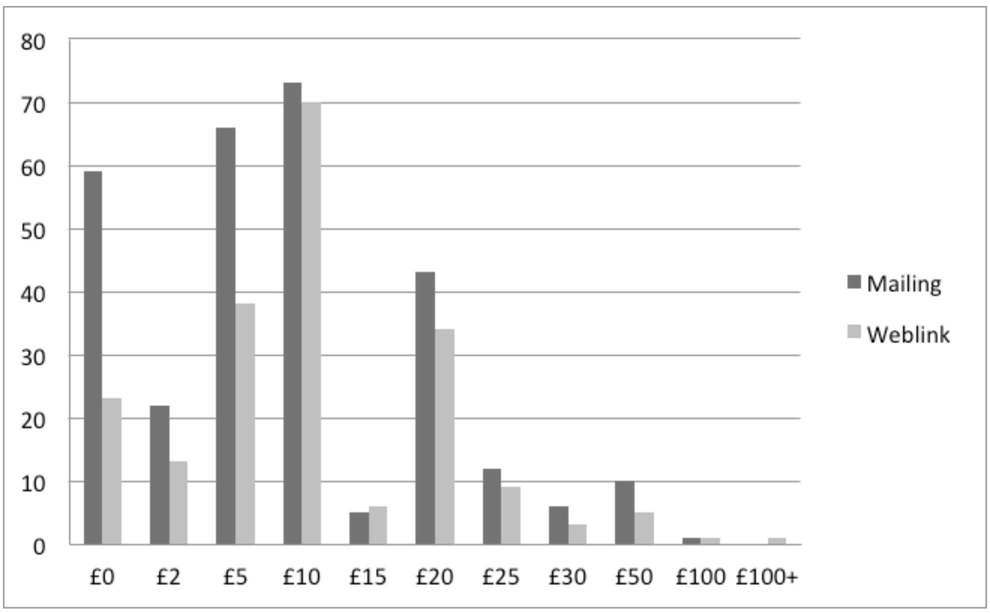

Figure 5.2
Frequency distribution of the amount mailing and weblink respondents were willing to pay to support a pine marten restocking.

Table 5.1 *Reasons respondents gave for supporting restocking, categorised and ranked based on number of responses (n=745).*

Reasons for restocking	n=745
1) Pine martens are a native species	168
2) Increase biodiversity	129
3) Prevent extinction	111
4) Restore natural balance	72
5) General support for wildlife/conservation	60
6) Duty/moral obligation	41
7) For the next generation	23
8) Grey squirrel/pest control	22
9) Wish to see them in the wild	20
10) Economic benefits	14
Other	85

Table 5.2 *Reasons respondents gave for being against restocking, categorised and ranked based on number of responses (n=57).*

Reason against re-stocking	n=57
1) Predation of wildlife	13
2) Encouragement of native stock	11
3) Lack of suitable habitat/fragmentation	7
4) Lack of knowledge on existing population	6
5) Economic	4
6) Let nature take its course	3
Other	13

Occupation and organisation membership of respondents

36% (n=220) of all respondents gave information about organisation membership (Figure 5.3). A much greater proportion of Weblink over mailing respondents who answered the question were members of conservation organisations (62%, n=73, were members of the Wildlife Trusts), and approximately 60% (n=132; 58 for mailing, 74 for Weblink) of all respondents were Royal Society for the Protection of Birds (RSPB) members; this was not surprising given that the Weblink survey was advertised through conservation friendly organisations (see acknowledgments). The sample sizes for the Game & Wildlife Conservation Trust (GWCT), Country Land & Business Association (CLBA) and British Association for Shooting & Conservation (BASC) were too small to allow meaningful inference of differences between response methods. Mailing respondents were significantly more likely to be members of the National Farmers Union (NFU) than Weblink respondents (X^2=15.9, df=1, p=<0.001); 23.2% (n=23) of mailing respondents were members of the NFU, compared to just 5% (n=6) of the Weblink respondents who answered the question.

Respondents who were members of conservation organisations (the Wildlife Trusts, RSPB and Woodland Trust, n=267), when grouped together, were significantly more likely to support a restocking over the collective grouping of members of organisations that supported land management, sport shooting and farming collectively (NFU, GWCT, CLAB & BASC, n=51, X^2=7.1, df=1, p=<0.001).

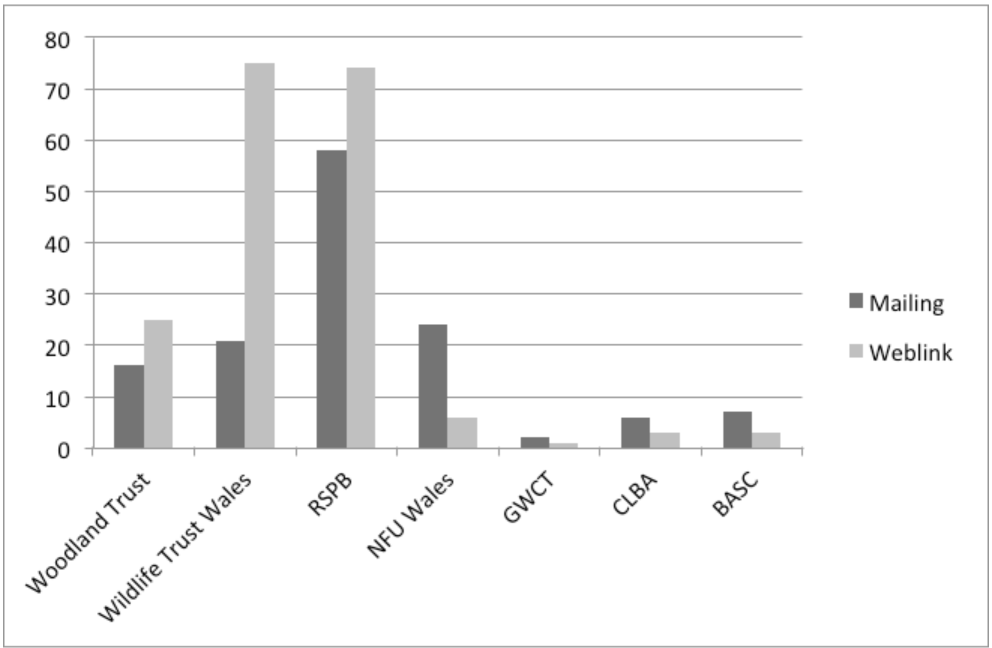

Figure 5.3
Frequency distribution of weblink (n=121) and mailing (n=99) respondents who gave information on organisation membership. Respondents were able to make more than one choice.

33.8% (n=205) of respondents gave information about their past or current occupation. A much greater proportion (65.8%, n=96) of Weblink respondents worked in wildlife conservation compared to mailing respondents (23.7%, n=32), shown in figure 5.4; this is likely to be the result of the online survey being publicised by wildlife conservation organisations. Weblink respondents were also more likely to work in countryside and estate management than mailing respondents, whilst a greater proportion of mailing respondents worked in farming (58.5%, n=79) compared to Weblink (26%, n=38). A similar proportion from both methods had worked in leisure & tourism. Chi squared tests to discern differences in willingness to support a restocking between occupations were not significant.

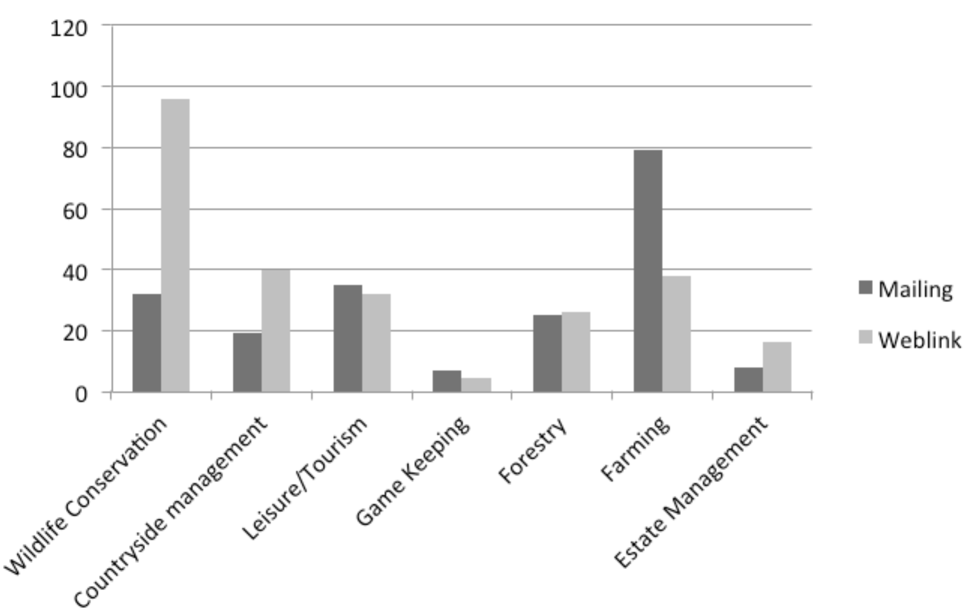

Figure 5.4
Frequency distribution of mailing (n=135) and weblink (n=146) respondents who gave information on past and current employment. Respondents were able to make more than one choice.

29% (n=176) of the mailing respondents were from rural areas, which approximated the conditions of hotspot areas, whilst 196 mailing respondents were from the hotspot areas. Respondents from rural areas were significantly more likely to support a restocking than those from hotspot areas (X^2-16.46, d.f=1, p=<0.001), and were willing to pay £13.55, 15% more than hotspot respondents, whose WTP was £11.77.

5.4. Discussion

Reestablishment of a lost or endangered species is only partly about science; socio-economics, politics and social acceptance are also important, the latter being indicated as a major factor effecting long-term success (Reading & Kellert 1993; O'Rourke 2014).

The majority of respondents would support a reinforcement (restocking) of pine marten in Wales, which concurs with the findings of a similar survey in England (Bright & Halliwell 1999), and were prepared to pay at least £13 (excluding £0 & £100+ bids) to support the effort. 13.5% (n=82) of respondents from both methods chose to support a restocking with £0 bids, which was interpreted in the main as willingness to support but economically unable to do so (Hanley & Spash 1993). This was corroborated by a number of additional comments associated with the question (i.e. respondents commented that they would support a restocking but were not financially able to contribute). This does not preclude a variety of other reasons however; Hanley and Spash (1993) highlight that respondents can feel that it is a governmental responsibility to cover the cost of maintaining natural heritage.

The mean WTP for this study compared closely with the findings of Bright and Halliwell (1999), suggesting a broadly acceptable amount that the public are willing to pay for pine marten conservation in England and Wales. This is encouraging when placed within the framework of an economy still recovering from the collapse in 2008, and might help to explain the high number of supportive £0 bids. There were some geographic and demographic differences; rural respondents were prepared to pay more than hotspot respondents, and farmers were willing to pay more than average. The latter point is interesting as farmers were one of the groups identified by the survey as being least likely to support a restocking; this highlights the diversity of attitudes within the catch-all group of 'farming'. Further segregation of farming by 'type', e.g. livestock, poultry or arable farming, farm size and whether farmers are involved in an environmental stewardship scheme or eco-tourism, would be very informative. The overall perception of importance for the work was supported by the majority of respondents who stated that prevention of pine marten extinction in Wales was either important or highly important.

Support for restocking was lower when just the mailing response was considered, but majority support for a restocking was still the case (84.7%). The mailing response was representative of the rural demographic in Wales, in contrast to the Weblink response, particularly for the farming community that owns or manages a significant proportion (82.4%) of the land area in Wales (Llywodraeth Cymru Welsh Government 2011). This highlights the critical importance of engaging the farming community as a major stakeholder, particularly as in this study farmers (identified by occupation & NFU membership), alongside members of sport shooting and land management organisations, were least likely to support a restocking compared to other land use occupations and organisation members.

Though the Weblink method was useful in spreading the reach of the survey and raising awareness, there was an associated responder bias arising from the nature of the organisations disseminating the online survey link that was apparent in the much greater proportion of respondents who either worked in wildlife conservation or were members of conservation organisations. This was reflected in the significantly greater proportion of Weblink respondents who had given a 'great deal of thought' to biodiversity loss and were prepared to support a restocking, compared to mailing respondents. There was value however in the information derived from the reasons given for support or non-support of a restocking.

The reason cited most frequently in support of a restocking by respondents was to restore a native species. This demonstrates the value that people attribute to indigenous species (Meuser, Harshaw & Mooers 2009), and could be used in future outreach work to create a sense of shared ownership and pride in the species, particularly in Wales where the majority of native species have Welsh names that can be traced back in historical documents, folklore and place names. Use of the species indigenous name, in this case *Bele'r Coed* for the pine marten in Wales, to highlight historical presence, could be a useful tool in engaging the public and increasing the species acceptance (Riley & Priston 2010; Jones 2011). This is important, as species can very quickly be lost from public consciousness when rare or absent for prolonged periods, even in rural areas and within conservation circles; a consequence of shifting baseline syndrome whereby consecutive generations perceive the state of the environment they are raised in as normal (Myers & Worm 2003; Baum & Myers 2004). Though a small proportion, (5%) of respondents cited moral obligation as a reason to restock pine martens, acknowledging the anthropogenic factors underpinning the species decline. 3% of respondents (n=22) cited 'pest control' as a reason to restock pine martens, specifically mentioning the invasive non-native grey squirrel (*Sciurus carolinensis*). This could be in response to the much publicised research from Ireland suggesting that the recovering pine marten population is causing a range contraction of grey squirrels, concurrently benefitting the native red squirrel (which is competitively inferior in mixed and broadleaf woodland, and highly susceptible to the fatal squirrel para-pox virus, for which the grey squirrel is an asymptomatic vector) (Lowe *et al*. 2000; Rushton *et al*. 2006; Sheehy & Lawton 2014). This example of the potentially positive role played by native predators in controlling invasive species, as has been suggested with the otter (*Lutra lutra)* in Britain (McDonald, O'Hara & Morrish 2007) and the dingo (*Canis lupus dingo*) in Australia (Johnson & VanDerWal 2009), can be used to ameliorate the reputation of predatory species and highlight the potential benefits for wider biodiversity, though caveats must be acknowledged. Two of the other most frequently cited reasons for supporting a restocking, to 'increase biodiversity' and to 'restore natural balance' are perhaps linked; both reflect a recognition by respondents that native predators can play a role in ecosystems that benefit other species and overall biodiversity, whilst also 'belonging' to that ecosystem. This, however, may be an artefact resulting from the conservation bias of Weblink respondents, who are likely to be better informed and possess a more advanced understanding of ecology than the mailing demographic.

Predation of wildlife, particularly the risk to small birds, nestlings, eggs and red squirrels, *Sciurus vulgaris,* was cited most frequently as a reason for not supporting a restocking. Concern for the red squirrel is perhaps attributable to misinformation; dietary studies in Britain and Ireland indicate that red squirrels comprise a relatively small proportion of pine marten diet (Halliwell 1997; Caryl *et al*. 2012; Sheehy *et al*. 2014). Interestingly, risk to livestock and domestic animals was not mentioned, despite real and perceived risk to livestock being a contentious issue in areas with recovering marten populations, such as Scotland (Balharry *et al*. 1996). Acceptance of the pine marten as a woodland predator needs to be achieved at the community level through education and awareness, in order to ensure the ultimate success of a translocation (Jacobson, McDuff & Monroe 2006; Espinosa & Jacobson 2012). Proactive rather than reactive community engagement is important in pre-empting conflict by inspiring tolerance through greater understanding of the species ecology, and appreciation of its aesthetic and cultural value.

The next three most prominent concerns, the need for 'encouragement of native stock', the 'lack of suitable habitat/fragmentation', and the 'lack of knowledge on existing population', relate to issues that have been investigated through research over the last two decades, and for which confident conclusions have been reached (Birks & Messenger 2010; Jordan *et al*. 2012). There is obviously scope to develop a more successful framework for disseminating and communicating this information to the public. Public consultation can be an effective approach towards increasing public participation, for which guidelines exist in Britain, and this will be carried out as part of the feasibility stage of a translocation.

Perhaps intuitively, the farming and gamekeeping community would be most likely to oppose a restocking, and those employed in leisure & tourism most likely to support it, arguably recognising the opportunities for revenue from wildlife tourism. This was also the finding of attitude surveys in England (Bright & Halliwell 1999) and Scotland (Balharry *et al*. 1996). Though this survey was specific to pine martens, it is probable that the same stakeholder groups will support and object to future work aiming to conserve or re-establish potentially controversial species, and it is recommended that future outreach and education work be targeted using this information. The media has the potential to play a significant role, with over half of respondents in this study citing television as the main source of their information on pine martens, followed by education, magazines and social media. Popular programs aired at peak time on mainstream channels could have the capacity to communicate conservation messages in a wide reaching and popular way, though caution must be exercised to avoid spreading false information, or failing to provide due coverage of contentious issues. It is also important to avoid 'dumbing down' the information available to the public; a number of respondents demonstrated a high level of understanding of some of the more complex issues underpinning a restocking effort such as population genetics, habitat fragmentation/isolation, and trophic interactions, citing them as either reasons not to, or reasons to support a restocking effort.

Support for restocking was geographically variable. Respondents in hotspot areas were less likely to support a restocking and were willing to pay less than respondents from rural areas. This may be attributed to perceived negative effects of living alongside pine martens, particularly the potential predation of other wildlife, notably birds. 65% of hotspot respondents were members of the RSPB, a conservation charity primarily involved in the conservation of bird species, and the hotspot areas are typically characterised by a greater proportion of designated, minimally managed, or unmanaged woodland, that harbours populations of charismatic woodland birds such as spotted flycatcher, lesser-spotted woodpecker, wood warbler and willow warbler, all of which have experienced large declines over the last three decades (Hewson *et al*. 2007; Balmer *et al*. 2014). Additionally, proportionally more hotspot respondents were members of the NFU than rural respondents, possibly contributing to the lower level of support. Support may be enhanced by raising awareness of the potential for wildlife tourism associated with the pine marten, as experienced in Scotland; 14 respondents cited economic benefits as a reason to restock pine martens, whilst also reinforcing that the pine marten is itself a native animal, and that predation is a natural occurrence, not something to be vilified.

The majority of respondents would support a restocking of pine martens in Wales to facilitate population recovery and prevent local extinction, though there are concerns, mainly relating to the potential impacts of a woodland predator on other wildlife. Nonetheless, even where the majority of responses to public consultations are favourable, this does not guarantee the success of a translocation programme, primarily because a small number of active opponents can have a disproportionate negative effect on public opinion (Stoskopf 2012).

Stakeholder engagement is fundamental to conservation practice, especially when managing potential conflicts (Treves, Wallace & White 2009; Redpath *et al*. 2013). Effective and sustainable conservation, particularly of carnivores, needs the full support of local people (Linnell *et al*. 2010), which requires that the opinions and needs of all groups are considered in a participatory process from the earliest stage. A consensus with local people will be easier to reach if they know that they will be continuously involved. Public consultations and community involvement will be a priority to gauge the level of local support for pine marten releases in all PRRs.

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6. ASSESSMENT OF PROPOSED PINE MARTEN REINFORCEMENT AGAINST IUCN GUIDELINES

The primary aim of conservation translocations is to create self-sustaining populations which will be resilient to stochastic events. Conservation Translocation is defined as "the intentional movement and release of a living organism where the primary objective is a conservation benefit". Reinforcement is "the intentional movement and release of an organism into an existing population of conspecifics", whereas reintroduction is "the intentional movement and release of an organism inside its indigenous range from which it has disappeared". The aims of reinforcement are to increase population size and genetic diversity in order to enhance population viability. However, whilst translocation is an important tool for species which are in imminent danger of, or have already undergone extinction in an area, it is accepted that this method should only be used if specific conditions can be met.

The Reintroduction and Invasive Species Specialist Groups of the International Union for Conservation of Nature (IUCN) have published guidelines designed to be applicable to all types of conservation translocations (IUCN 1995; IUCN 2013). They are based on principle rather than example and are intended to ensure that any translocation is justified because it will result in a quantifiable conservation benefit and will not cause adverse side effects of greater impact.

These can be distilled into a number of questions which need to be asked when planning and carrying out any conservation translocation (Seddon, Moehrenschlager & Ewen 2014). Here we assess translocation of pine marten to reinforce populations in England and Wales against these questions (numbers in parentheses are those of the IUCN annex to which each relates).

<u>Before resorting to translocations, have alternative courses of action been considered?</u> (3.3)

In recent years the population of pine marten in Scotland has been expanding and increasing its range southwards (Croose, Birks & Schofield 2013; Croose et al. 2014). There are now pine marten in some of the southern Scottish counties, with presence recently confirmed (by DNA) only 14km from the border with England. For this reason, translocation to suitable sites in the northernmost counties of England is not being considered, as natural recolonisation is likely to occur. However, further south in England and in Wales, the large distances involved, coupled with extensive intervening areas of unsuitable habitat, large conurbations, major roads and other infrastructure which are probable barriers to dispersing pine marten mean that natural recolonisation is unlikely to occur. This is why translocation is thought to be necessary in these areas.

<u>Can the past cause(s) of decline and extinction be identified and addressed?</u> (3.2)

The cause of the pine marten's decline in Britain is widely attributed to intensive predator control associated with the rise in sport shooting (Langley & Yalden 1977), along with loss and fragmentation of suitable woodland habitat (Birks & Messenger 2010). There has been a significant reduction in game keeping since the early 1900s (Tapper 1992) and pine marten now have full legal protection under the Wildlife and Countryside Act, 1981. Game keeping practice has also changed considerably over the last century, and lethal control is now closely targeted on a small number of pest species. Woodland cover in Britain is currently back to a similar level as in the 11th century (Watts 2006), so habitat availability in general for martens is at its highest for many years.

In Scotland the pine marten population suffered a significant decline but has been recovering well, probably because, it contracted to a single, contiguous population which, even at its nadir, was larger and occupied a wider area than those in England and Wales. It is not known precisely why those in England and Wales have failed to show similar recovery, but the most likely explanation is that they were reduced to such small, isolated populations that the effects of environmental and demographic stochasticity have prevented their expansion.

The IUCN guidelines acknowledge that the exact cause of decline or the relative importance of a range of limiting factors may not be known for certain. Where this is the case an experimental approach within the release programme can be justified, provided that all other criteria are met and the outcomes will be monitored and used to inform and refine subsequent plans.

Can potential future causes of decline/extinction be identified and addressed? (3.2.3; 5.3.2)

It is not sufficient to just address the causes of the original population decline, as new potential threats may have arisen since that would prevent establishment of a restored population. It is also necessary to anticipate future threats and to assess the potential impact of these.

There have been large changes in the landscape since pine martens were widespread and common. It is anticipated that there will continue to be changes in land use in future including further increases in urban expansion, roads and vehicle traffic particularly in the more densely populated parts of the UK. This has been accounted for, as far as possible, in the assessment of PRRs.

Is enough known about the biology and ecology of the species to inform the selection of release area(s)? (5.1)

Selection of suitable release sites within appropriate release regions is of paramount importance. It has been shown that release into unsuitable habitat is a major cause of failure in translocations (Magdalena Wolf, Garland Jr & Griffith 1998). There is a large body of published literature on the habitat ecology and biological needs of the pine marten (Pulliainen 1981; Storch, Lindstrom & De Jounge 1990; Brainerd *et al*. 1994; Putman 2000; Brainerd & Rolstad 2002; Porter, St Clair & Vries 2005; Zalewski & Jędrzejewski 2006; Caryl *et al*. 2012; Virgos *et al*. 2012). Because they are important elements of their native ecosystems, and because some are of economic importance as furbearers, there have also been large numbers of reintroductions, augmentations (reinforcements) and introductions of marten species (details in Powell *et al*. 2012). All the available literature has been reviewed and contact has been made with other researchers in Europe and North America who have been involved with marten translocations.

Are there sufficient areas of suitable habitat to sustain viable populations? (5.3)

Releases into areas with low chances of success are counter-productive and may also be considered an animal welfare issue (Harrington *et al*. 2013). The release region should have a large enough carrying capacity to accommodate an increase in population size and to sustain a viable population in the long-term. Tried and tested modelling methods have been used in the first instance to identify areas of potentially suitable habitat (chapter 2). The models were parameterised with data from expanding populations of pine martens in regions (Scotland and Ireland) that have similar environmental conditions to those in potential reinforcement regions. The most recent available GIS data have been used for preliminary analyses of the 6 prioritised potential reinforcement regions (chapter 3), and these analyses will be ground-truthed with detailed fieldwork over the coming 18 months before proceeding further. Further GIS modelling of habitat networks is planned to identify probable routes for expansion from each of the PRRs.

Are plans in place to formally integrate stakeholder concerns and social acceptability into all phases and a participatory process for all groups that might be involved in or affected by releases? (6.7)

The broad scale public opinion survey presented in chapter 5 gives an overview of the level of support for proposed reinforcement. However this does not include all groups that may be affected, nor does it target specific areas that have since been prioritised as potential reinforcement regions. Engaging with local communities in these areas and ensuring that all groups are involved is a high priority action for phase II of the feasibility study and work on this has already begun.

Has there been an evaluation of the risk that translocated animals may pose to the conservation interests of other species and habitats? (5; 6)

The guidelines state that the possible ecological roles of the focal species in any new environment should be carefully evaluated and that there should be an assessment of the risk that the translocated animals might pose to the conservation interests of other species or habitats in release areas. These are discussed in chapter 4, along with recommendations for further work. The risk of unintended and undesirable impacts is likely to be lowest in population reinforcements, which is why it is proposed that the first releases are in areas where there is recent evidence of pine marten presence.

In addition to the risks, the assessment of any translocation proposal should also identify potential benefits including ecological, social and economic impacts. The Trust commissioned a separate report earlier this year to explore socio-economic aspects of the programme, including landowner and other stakeholder interests and opinions, community perception and involvement, as well as possible wildlife tourism opportunities (Arwel Jones Associates 2014). The report identifies and outlines potential productive avenues of research and ways forward for the project in this area, which will be developed throughout phase II and beyond.

Can enough animals be taken from a genetically appropriate source population or populations without detrimental effects? (5.5)

Results of population viability analyses show that 30-40 pine martens would need to be released in an area to maximise the viability of pine marten populations (Bright & Harris 1994). It is recommended that, where sufficient stock is available, wild caught animals are used for reintroductions. These generally show higher survival and better adaptation to the new environments than captive bred animals (Jule, Leaver & Lea 2008).

Source populations should show characteristics based on genetic provenance, morphology, physiology and behaviour that are appropriate in comparison with remaining wild populations. Animals sourced from areas with similar prey species, competitors, predators and habitats may demonstrate higher rates of post-release survival and reproduction (Aber *et al*. 2013). A recent study which compared the haplotype composition of historical and current pine marten populations in England, Scotland and Wales found no differences between the main haplotype of contemporary (post-1950) populations across the UK (Jordan *et al*. 2012). Therefore, the increasing and expanding population of pine martens in Scotland would probably be a suitable source of animals for translocation, and this will be investigated and discussed further in phase II.

Pine marten populations can be susceptible to overharvest (Helldin 2000), therefore the effects of removing individuals from candidate source populations will be assessed, and source populations identified that are able to sustain the removal of individuals with no negative impact on viability. Bright and Halliwell (1999) used an age structured population model to examine the effects of removal on hypothetical source populations of 20 and 50 pine martens. This showed that, as the proportion of the population removed was increased from 10 to 25%, the probability of a population decline of at least 10% increased. However two years after 15% of adults or adults and sub-adult animals were removed there was more than an 80% probability that populations would have returned to their initial size. This supports the view that it will be possible to take enough animals from a genetically appropriate source population or populations without detrimental effects. The total number that can be translocated from each candidate source site will be carefully considered based on locally available data and more detailed population viability analyses over the next 18 months.

Are plans in place for pre- and post-release monitoring? (8.1; 8.2)

Monitoring the course of a translocation is essential and integral to this is collecting baseline information on any area before releases into it take place. Without this it will be impossible to determine if any observed changes after releases relate to the impacts of the released animals. Pre-release surveys will begin in summer 2014 and survey effort will focus on the species and ecological processes that may be affected by translocations.

Plans are also in place for post-release monitoring, the primary aim of which will be to assess and modify protocols if necessary to ensure the highest probability of survival and site fidelity for released individuals. The next step in determining the long term success of the recovery programme will consists of monitoring translocated animals for a multi-year period to estimate population trajectory, diet, habitat use, reproduction, recruitment and home range size in release areas. This information will all be used for an adaptive management programme and to inform future translocations.

Detailed proposals for release and implementation will be produced and evaluated against criteria in Annex 7 of the IUCN guidelines during the second part of the feasibility study (to be completed in 2015).

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7. CONCLUSIONS AND RECOMMENDATIONS

7.1. Conclusions

1. Reinforcements and reintroductions have the potential to contribute substantially to the recovery of the pine marten in Wales and England.
2. An assessment of the biological feasibility of reinforcing pine marten populations at selected sites was carried out in the initial phase, the results of which are detailed in this report. Six potential reinforcement regions in Wales and England have been identified that contain sufficient areas with high predicted habitat suitability to support viable populations of pine martens.
3. The three PRRs in central Wales are likely to be the most biologically suitable area for the first reinforcement because they are virtually contiguous and contain the largest amount of suitable habitat, and therefore have the highest potential pine marten carrying capacity. Because of the large extent of potentially suitable habitat in this region, there is scope to avoid certain areas if any local issues arise, and still maintain the integrity of the translocation programme. Central Wales is also the region with the lowest risk to pine martens of road mortality, which could be a factor during the establishment phase of translocations.
4. The results of a wide scale public opinion survey suggest that pine marten reinforcements in Wales would almost certainly have enthusiastic support from the majority of the community, but that the support of a significant minority would be crucial. For this reason it is essential to engage with local communities and stakeholders at the outset to gauge local levels of support for the project and identify potential conflict issues.


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7.2. Further Work & Recommendations

1. Field surveys will be carried out in phase II of the feasibility study (from August 2014), to ground-truth the desk based GIS analyses and address the questions of whether sufficient prey, foraging habitats and suitable den sites are available. This will inform the final decision as to whether translocations to these regions should proceed and where the most optimal release sites would be.

2. Baseline data will be collected through field surveys and in collaboration with other organisations to enable long term monitoring of potential impacts from pine marten releases on other species including grey squirrels.

3. Detailed consultation with stakeholders, the general public, government and local agencies and other NGOs will now be undertaken in potential reinforcement regions, before any proposals for pine marten translocations are taken forward. It is recommended that external experts be commissioned for this. The information gathered during the consultation process will inform the selection of the most acceptable of the suitable release sites.

4. In addition to this, it is recommended that opportunities be explored for maximising local socio-economic benefits with regard to potential for eco-tourism, education and general business opportunities.

5. A pine marten publicity and awareness campaign should also be initiated during the consultation period and continued during and after translocations. The campaign should include schools and other locally based community organisations and seek to actively engage with sections of the community who might perceive that they will be affected by an increased pine marten population.

6. An initial risk assessment suggests that pine marten reinforcement is not likely to have an adverse effect at the population level on other species of conservation concern currently present within the PRRs. However, further site specific analyses of potential impacts should be carried out in conjunction with local interested parties, e.g. wildlife groups, reserve managers and other NGOs. If and where translocations take place these will be closely monitored and mitigation measures resourced and put in place where there may be an issue, such as with nest box schemes.

7. Genetic information will now be reviewed and population viability analyses undertaken to identify if and where there is an appropriate source population or populations for translocations to Wales and England. PVA will be used to determine the number of individuals that can be removed without negatively affecting the source populations.

Provided that the further work outlined here confirms that there are sites where translocations are appropriate, and have the full support of local communities, a detailed release and monitoring plan will be produced in 2015. This will address all the relevant issues including final release sites; sources, capture, health surveillance and transportation of donor animals; pre- and post-release management and monitoring; and implementation timescale.

The Vincent Wildlife Trust

The Vincent Wildlife Trust has been involved in wildlife research and conservation since 1975. It has focused particularly on the needs of British mammals including the otter, pine marten, polecat, stoat, weasel, water vole, dormouse and the bats.

Currently the VWT's work is centred on the pine marten, polecat, and the rarer bats.

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